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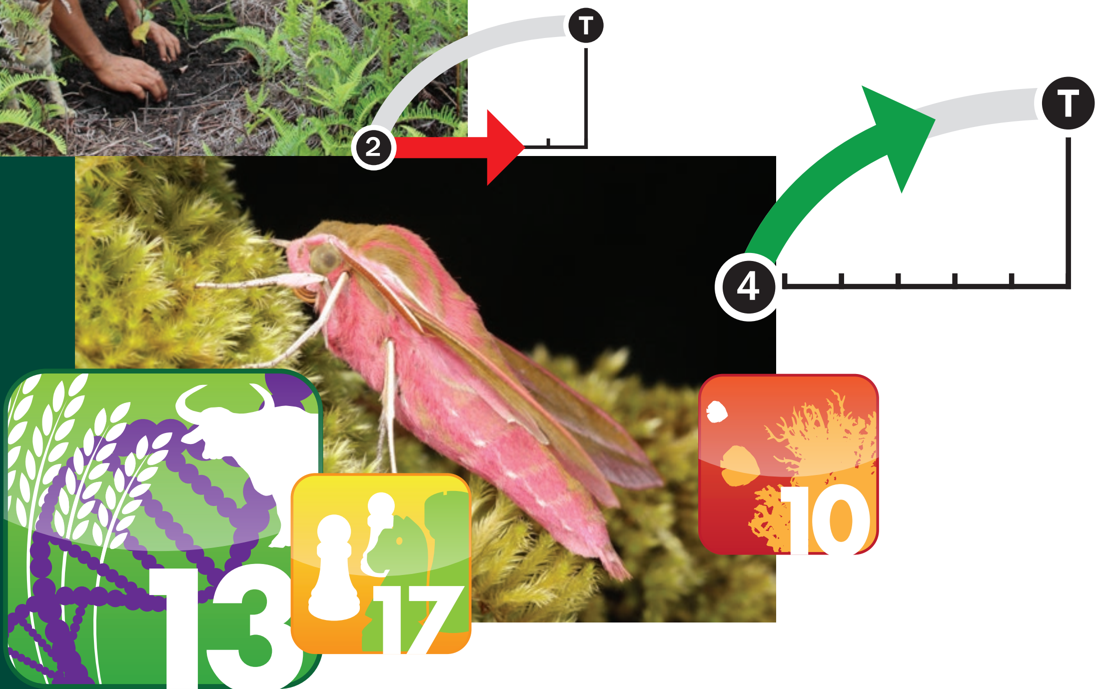
PROGRESS TOWARDS THE AICHI BIODIVERSITY TARGETS:

An assessment of biodiversity trends, policy scenarios and key actions



UNEP WCMC
Convention on Biological Diversity
Biodiversity Indicators Partnership
CE3C Centre for Ecology, Evolution and Environmental Changes
DIVERSITAS an international programme of biodiversity science
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Global Biodiversity Outlook 4 (GBO-4) Technical Report

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CONTENTS

FOREWORD

The fourth edition of the Global Biodiversity Outlook (GBO-4) provides a mid-term assessment of progress towards the implementation of the Strategic Plan for Biodiversity 2011-2020 and the Aichi Biodiversity Targets. This technical study, prepared over a two year period, represents the detailed scientific foundation on the basis of which GBO-4 has been prepared. It analyses the latest biodiversity information from a wide spectrum of sources and provides scientifically rigorous information on our progress towards the attainment of each of the Aichi Biodiversity Targets as well as reviewing, through statistical extrapolations and scenarios, the likelihood of achieving the Aichi Biodiversity Targets if current trends continue. On the basis of these lines of evidence the technical study recommends key actions for each target that would enable their achievement by 2020.

The assessment contained in this technical study represents the collective effort of more than fifty international experts drawn from various institutions from around the world as well as several rounds of peer review. It was overseen by the Advisory Group for GBO-4 and the Bureau of the Convention's Subsidiary Body on Scientific, Technical and Technological Advice. I am grateful to all the experts involved in this assessment for their contributions. It greatly advances our current understanding of the status and trends of biodiversity and provides information on the types of actions necessary to achieve the Targets which is vital for Parties to decide on how best they may contribute to the global attainment of the Aichi Biodiversity Targets. But it also demonstrates the challenges of covering all aspects of the Aichi Biodiversity Targets and the need to invest in systematic monitoring programmes that provide continuous status and trends information in areas needed for decision making.

The assessment contained in this study makes it clear that while progress is being made towards the attainment of most Aichi Biodiversity Targets, significant work remains before us; on our current trajectory we are unlikely to reach the majority of the Aichi Biodiversity Targets by their deadline. As we look towards the next six years of the Strategic Plan for Biodiversity 2011-2020, I am confident that the information contained in this study will help all stakeholders in charting the road ahead and in putting us on a path where biodiversity is valued, conserved and wisely used for the benefit of all people.

Braulio Ferreira de Souza Dias
Executive Secretary,
Convention on Biological Diversity

EXECUTIVE SUMMARY

For the executive summary, please refer to the GBO main report, *Global Biodiversity Outlook 4 – A mid-term assessment of progress toward the implementation of the Strategic Plan for Biodiversity 2011 – 2020* (SCBD, 2014). The summary is available in the six United Nations languages, and can be accessed on the CBD website (www.cbd.int/GBO4).

OVERVIEW OF THE GLOBAL BIODIVERSITY OUTLOOK 4 TECHNICAL REPORT

OBJECTIVES OF THE GBO-4 TECHNICAL REPORT

In 2010, the Parties to the Convention on Biological Diversity (CBD) adopted the Strategic Plan for Biodiversity 2011-2020 in Nagoya, Japan. This Strategic Plan includes a "shared vision, a mission, strategic goals and 20 ambitious yet achievable targets, collectively known as the Aichi Targets." The vision is that "By 2050, biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people." The Global Biodiversity Outlook 4 is an assessment of progress towards attaining the Strategic Plan roughly halfway between its adoption in 2010 and the deadline for achieving most of the Aichi Targets in 2020.

This Global Biodiversity Outlook 4 (GBO-4) Technical Report provides a detailed assessment of the evidence base underlying the conclusions in the main Global Biodiversity Outlook 4 report. The Technical Report examines this evidence base with the objective of providing policy-relevant answers to the following questions:

- Are we currently on a path to meet the Aichi 2020 Targets?
- What are the consequences of achieving or not achieving the Aichi Targets in terms of key indicators of biodiversity and ecosystem services?
- What actions would contribute to attaining the Aichi Targets, and what are the costs and benefits of these actions?
- For which plausible socioeconomic development pathways are the 2050 Vision attainable?
- To what extent would achieving the Aichi Targets help to reach the 2050 Vision?
- What are the trade-offs and synergies between the Aichi Targets?
- What is the contribution of meeting the Aichi Targets and the 2050 Vision with respect to human well-being and in particular the Millennium Development Goals and forthcoming Sustainable Development Goals?

The GBO-4 assessment is primarily based on research published in peer-reviewed scientific journals, as well as national and international assessments (e.g., IPCC, FAO, National ecosystem assessments). We have also relied on i) national reports for the CBD[1], ii) global indicators provided to the CBD by the Biodiversity Indictors Partnership (BIP)[2] and iii) analyses that were carried out specifically for the GBO-4 assessment. Where we have relied on unpublished research, we have carefully documented the methodology in appendices. The breadth of the assessment, the short time frame in which it was undertaken, and gaps in available evidence mean that it can never be entirely complete. Nevertheless it represents the most comprehensive synthesis of available knowledge from over 100 scientists across a huge number of institutions worldwide.

Our objectives are to provide clear input into policy, open the door to a stronger dialog with stakeholders concerning desirable endpoints, identify actions needed to reach these endpoints and examine a broad range of socioeconomic development pathways and their impacts on the environment. To achieve these objectives we have brought together analyses of key indicators of recent trends, current status, near term projections to 2020 and longer-term projections to 2050 for each of the Aichi Targets. For each Aichi Target, we have assessed progress towards the target, principal actions that would be required to meet the target, and the costs and benefits of doing so by building on the work of the High Level Panel on Global Assessment of Resources for Implementing the Strategic Plan for Biodiversity. We have also identified key uncertainties and knowledge gaps. In addition, we have examined the interactions between the Aichi Targets and used a variety of scenario-based studies to assess how various socioeconomic development pathways will lead us away from or towards the CBD 2050 Vision.

As in previous Global Biodiversity Outlooks, we use "biodiversity" in a broad sense as it is defined in the Convention on Biological Diversity; i.e. to mean the abundance and distributions of and interactions between genotypes, species, communities, ecosystems and biomes. This assessment has a strong focus on species as in previous reports, but includes a greater focus on drivers of biodiversity loss and on ecosystems than previous reports due the nature of the issues addressed in the Aichi targets. Genetic diversity is also addressed, but the lack of data and scenarios for genetic diversity has limited the assessment of diversity at this level.

Footnotes

[1] The GBO-4 report includes summaries of national biodiversity strategies and action plans (NBSAPs) that were available when preparing the report. The national reports used in the different chapters are referenced in the corresponding chapter.

[2] The Biodiversity Indicators Partnership (BIP) is an international consortium of organizations that was established in 2007 to provide a wide range of indicators that can be used to assess progress towards international biodiversity targets including those agreed by Parties to the CBD. For further information see: www.bipindicators.net/indicators.

ANALYSIS OF STATUS AND TRENDS

We have used a wide range of indicators to determine progress towards the Aichi 2020 targets including indicators developed by the Biodiversity Indicators Partnership (BIP) augmented by additional newly identified indicators that fill gaps in the coverage of Targets and sub-objectives. These indicators were chosen on the basis of the following criteria: i) from a credible source with well described, publically available methods, ii) pertinence to Aichi Targets and iii) broad geographic (preferably global) coverage.

The analysis has assessed trends to date and for a subset of indicators extrapolated these trends to 2020. Our additional criteria for including these indicators in the extrapolation were: iv) a start point before 2010 and end-point after 2010 where feasible, and where not feasible but the indicator was essential due to a lack of alternatives for the Target, a long series of data points ending as near to 2010 as possible; and (v) at least five annual data points in the time-series;

For the extrapolations to 2020 we determined trends using statistical fits to the data using a wide range of linear and non-linear models. These models were included in order to fit the range of possible shapes of curves from the time-series. The best-fitting statistical models were determined using a well-known metric that takes into account how well the model fits the data and the number of parameters in the model (Akaike Information Criterion, AIC). This metric is based on the assumption that the best model describes the data reasonably well with a small number of parameters. The best fitting models where then combined to provide a "mean" trend, weighted by their goodness-of-fit, as well as confidence bounds around the estimate of the mean trend. Further descriptions of the indicators used can be found in Annex 1 and 2 of chapter 21, as well as Tittensor *et al.*, (2014).

METHODS FOR FUTURE PROJECTIONS USED IN GBO-4

For projections to 2020 and 2050, we have taken a much broader approach to scenario analysis than in previous global assessments by complementing "storyline" approaches to socioeconomic scenarios (e.g., IPCC SRES scenarios, MA scenarios) with other types scenarios and extrapolations of current trends (see van Vuuren *et al.*, 2012 for a review).

Most global scenarios assessments for biodiversity and ecosystem services have been based on socioeconomic storyline approaches (e.g., MA, GEO, IPCC, and previous GBO reports, van Vuuren *et al.*, 2012, Figure 0.1). These are projections of socioeconomic development based on various plausible hypotheses about the future dynamics of key driving forces of global change such as population growth, per capita resource use, etc. In most cases, these scenarios of socio-economic development pathways have been coupled with quantitative models of their impacts on proximate drivers of change in biodiversity and ecosystem services (e.g., land use, fishing pressure, climate change) and models of the impacts of these proximate drivers on biodiversity and ecosystem services (Pereira *et al.*, 2010). These scenarios typically do not explore specific policy options, tend to explore a relatively small number of possible futures and focus on time scales of many decades (Leadley *et al.*, 2010, Pereira *et al.*, 2010, van Vuuren *et al.*, 2012). In this report, we have relied heavily on additional approaches to scenarios including extrapolation from current trends in drivers and in dynamics of biodiversity and ecosystem services and a broad range of types of scenarios of socioeconomic development pathways.

We have primarily, but not exclusively, relied on four main types of scenarios (van Vuuren *et al.*, 2012, Figure 0.1):

1. Extrapolations of current trends – statistical extrapolations of current trends are sometimes coupled with simple models of management or policy options. We have limited these extrapolations to the 2020 time period.
2. Socioeconomic storylines – plausible socioeconomic development scenarios are coupled with models of impacts; e.g., analyses based on MA, GEO, IPCC storylines.
3. Policy options – policy options are added to storylines of "business-as-usual" socio-economic development and then tested for impacts.
4. Backcasting or desirable endpoint analyses – desirable multi-criteria endpoints are set for the future and then plausible scenarios are developed that come as close as possible to reaching these end points.

Several other methods for exploring the possible future dynamics of social-ecological systems are widely used at national and sub-national scales including participatory approaches, econometric projections, bioeconomic viability analysis, and others. We have not relied heavily on these types of scenarios because the small spatial scale in most of these studies makes it difficult to scale up for a global assessment. More detailed explanations of the socioeconomic scenarios used in many of the studies that we examined for this assessment can be found in van Vuuren *et al.*, (2012) and references therein.

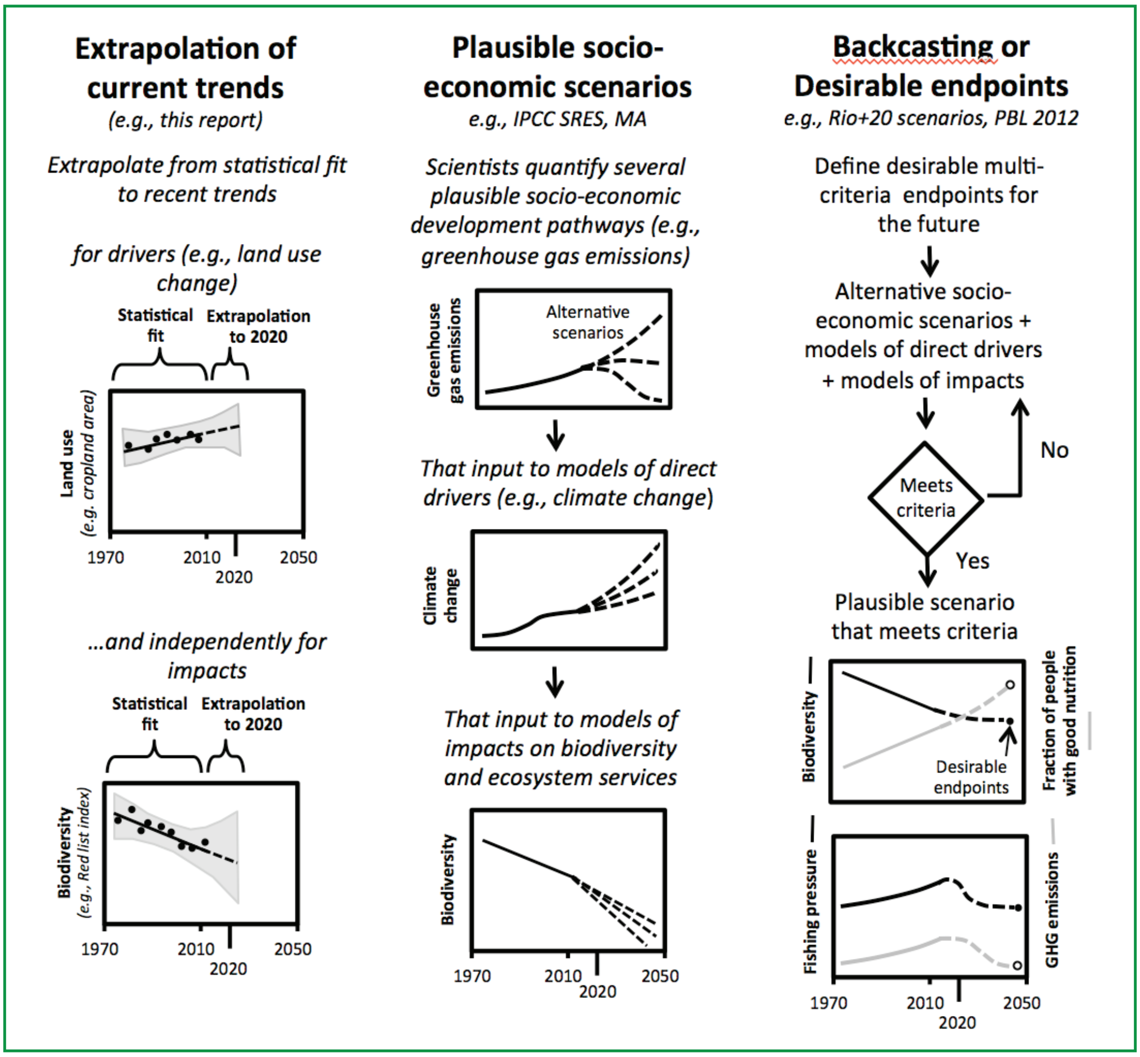

Figure 0.1. Typology of three different types of scenarios used in this report for projecting future trends. Note that policy scenarios (see text) are not shown. These are typically variants of the "plausible socioeconomic scenarios" analyzing the impacts of specific policy measures compared to business-as-usual scenarios (i.e., baseline). Current trends are indicated by solid lines and projections by dashed lines. The grey regions around the statistical extrapolations are the statistical confidence bounds around the trend line.

Each of these methods has strengths and weaknesses that are outlined in Table 0.1. As such, we have used a combination of these methods to provide a broad range of insights into plausible future trajectories.

Table 0.1. Strengths and weaknesses of various scenarios approaches.

Model Type	Strengths	Weaknesses	Examples
Statistical Extrapolation	These analyses are simple to understand. They rely on straightforward analyses requiring very modest computing power. They generally accurately describe current trends.	These analyses are limited to extrapolation only for short term and with the assumption that underlying processes follow current trends. They do not identify key drivers; they only fit trends. These analyses are difficult to carry out for many indicators due to lack of high quality time series.	Nicolson *et al*. (2012), Extrapolations in this report (see above and Appendix 1, Chapter 21)
Socioeconomic scenarios	Some scenarios are very widely used which facilitates comparison between studies and analysis of uncertainty. A limited number of scenarios (typically four) simplifies comparisons across studies The time frame of scenarios is typically several decades, so they are useful for exploring long-term dynamics	Most current scenarios focus too heavily on climate change criteria. Current scenarios do not include positive outcomes across a wide range of criteria. Policy options are difficult to extract from scenarios.	IPCC SRES, Millennium Assessment, Global Environmental Outlook 4
Policy options	Policy options are explicitly accounted for this these analyses. Options are more easily understood by policy makers and stakeholders than complex scenarios.	These analyses can create a large number of scenarios to be evaluated. They are not yet widely used.	“Rethinking” analysis (PBL 2010), OECD Second Environment Outlook
Backcasting or Desirable endpoint analysis	Backcasting encourages the exploration of positive outcomes and pathways of how to achieve desired end-points. They open the door to incorporating stakeholder or policy input when defining desirable outcomes. They can help determine short-term priorities as consequence of long term (normative) analysis.	Few institutions are capable of carrying out these types of analysis. Large investments in human and computing resources are required. They are not yet widely used.	Rio+20 analysis (PBL 2012) (see description of scenarios in Chapter 22).

STRUCTURE OF THE REPORT

Analyses of each of the individual Aichi 2020 Targets are structured to respond to the questions outlined in the *Objectives of the GBO4 Technical Report* section above. The structure of the chapters addressing the individual Aichi 2020 Targets is as follows:

Preface
1. Are we on track to achieve the Target?
  a. Status and trends
  b. Projecting forward to 2020
2. What needs to be done to reach the Target?
  a. Actions
  b. Costs and Cost-benefit analysis
3. What are the implications for biodiversity in 2020?
4. What do scenarios suggest for 2050 and what are the implications for biodiversity?
5. Uncertainties and data requirements
6. "Dashboard" - Progress towards Target
7. References

It should be noted that some of the chapters do not include a section 3 (projecting forward to 2020), or section 4 (outlook to 2050), as either targets did not lend themselves to a scenario analysis, or no information was available on scenarios and projections for these targets.

Interactions between the Aichi Targets are then evaluated with a specific focus on the strengths of interactions between targets and an analysis of key synergies and trade-offs among targets. We also analyzed how achieving the Aichi Targets can contribute to the longer term goals embodied in the CBD 2050 Vision and considered how biodiversity and the Aichi Targets could be incorporated into the proposed Sustainable Development Goals that are expected to supercede the Millennium Development Goals which mature in 2015.

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TARGET 1: AWARENESS

By 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably.

PREFACE

Addressing the direct and underlying drivers of biodiversity loss will ultimately require behavioural change by individuals, organizations and governments. While the exact relationship needs to be better understood, it is clear that awareness and appreciation of the diverse values of biodiversity, underpin the willingness of individuals to make the necessary changes and take the necessary actions to address these drivers. Awareness is also important to the creation of a public that is ready to act politically and which in turn can help to generate the "political will" for governments to act.

Meeting this target requires that for a variety of target groups, and different contexts, people are aware of the values of biodiversity and of the actions they can take to conserve and sustainably use it. Attainment of most, if not all of the Aichi Biodiversity Targets, will be greatly facilitated if there is a greater awareness of the value of biodiversity and of the actions required to reach each target.

1.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGETS?

1.1.1 Status and trends

Meeting Aichi Biodiversity Target 1 requires that people are aware of the values of biodiversity as well as of the actions they can take to conserve and sustainably use it. Though the amount of information regarding people's awareness of biodiversity is increasing, comprehensive data remains limited at the global level. While some national and regional data sets are available, there are significant geographic gaps. Further, in general, information on biodiversity awareness is limited for mega diverse countries, as well as for African and Asian countries in general.

National data on environmental awareness, sustainability or biodiversity-specific questions have been collected by ministries of environment and national parks services in a number of countries. At least 80 biodiversity awareness surveys, commissioned by governmental departments, non-governmental organizations, corporations, and academic institutions, are known to exist (Secretariat of the Convention on Biological Diversity; SCBD 2013). These surveys have sought to capture information related to a variety of issues, including individuals' awareness and understanding of the term biodiversity, the value of biodiversity and nature to individuals and practices, and consumption patterns related to biodiversity and sustainability among other things. Surveys such as these provide a good basis for assessing trends in awareness of biodiversity in the countries concerned. However, because of different methodologies used, it is difficult to aggregate these results to generate a global picture of biodiversity awareness.

Box 1.1: Volunteer time spent on conservation

One example of national information related to individuals' engagement with biodiversity is the United Kingdom's Index of volunteer time spent in selected UK conservation organizations. The index is calculated using information provided by a range of non-governmental organizations operating in the United Kingdom, which work with volunteers. The index (see Figure 1.1) illustrates that between 2000 and 2012 the amount of time that individuals volunteered with conservation organizations increased by 27%. However there has been a decrease of 6% between 2007 and 2012. Indicators such as this one illustrate one method of assessing people's actions in support of biodiversity. In this sense this indicator provides information which is relevant to the second part of Aichi Biodiversity Target 2 related to individuals' awareness of the types of measures they can take to conserve biodiversity.

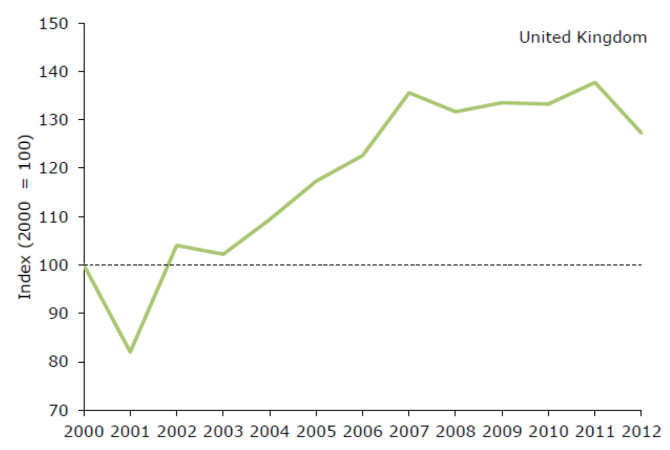

Figure 1.1. Index of volunteer time spent in selected UK conservation organisations, 2000 to 2012. Source: Department for Environment, Food and Rural Affairs 2013.

The Eurobarometer study represents one example of where data on biodiversity awareness is collected at national level and are then aggregated for a region. Three Eurobarometer surveys have been conducted on biodiversity awareness across the European Union (EU), collecting data in 2007, 2010 and 2013. The 2013 survey collected information from more than 25,000 respondents (European Commission; EC, 2013). The surveys looked at the familiarity of Europeans with the term biodiversity, their awareness of biodiversity loss and their understanding of the consequences of this, and their involvement with actions to protect biodiversity. Data is aggregated at the regional level, as well as reported for national samples.

The results from the latest issue of the Eurobarometer show that familiarity with the term biodiversity has increased in 18 member states since 2010. In 2013 fewer than half of Europeans (44%) reported having heard of the term biodiversity (as compared to 35% in 2007), and some 30% had heard of the term and knew what it meant. The survey also found that slightly more than a quarter of respondents (26%) had never heard of the term biodiversity, a decrease from 35% in 2007 (EC, 2013).

The results from the Eurobarometer surveys also show that in some countries there has been a substantial increase in the proportion of respondents who feel informed about biodiversity loss. Overall 24 out of the 26 countries surveyed show increases in the number of people familiar with biodiversity (EC, 2013). However, the survey also found that in Europe there was a declining sense that biodiversity loss is a serious problem in people's own country. In 2007, 43% of respondents felt that biodiversity loss was a problem in their own country. In 2013, this number had fallen to 35%. Nevertheless, 88% of respondents felt that biodiversity loss in Europe in general was a problem, and 66% felt that biodiversity loss at a global level was a very serious problem. Biodiversity was also seen as important for human wellbeing. 55% of respondents felt that it was important to halt biodiversity loss because it is indispensable for the production of food, fuel and medicine. Further 85% agreed that biodiversity is essential in tackling climate change. Over three quarters of Europeans felt that it was important to halt biodiversity loss because it was a moral obligation (EC, 2013).

Data on public awareness has also been collected by associations and other organizations. One example of this sort of study is the Union for Ethical Bio Trade's (UEBT) Biodiversity Barometer. The UEBT Biodiversity Barometer provides insights on how biodiversity awareness among consumers and how the beauty industry reports on biodiversity is evolving over time. Each year the Biodiversity Barometer spreads the scope of the research by adding new countries. It also intends to understand trends of 'biodiversity awareness' by periodic recurrent research in the countries involved. Since the first edition of Biodiversity Barometer in 2009, the global research organization IPSOS, on behalf of UEBT, has interviewed 31,000 consumers in 11 countries (Brazil, China, France, Germany, India, Japan, Peru, South Korea, Switzerland, the United Kingdom and the United States of America, UEBT, 2013). The combined population of the countries surveyed represents almost half of the world's total population; however, Africa is not represented.

The results from the Biodiversity Barometer surveys suggest that between 2009 and 2013 there has been a steady increase in the number of people that can provide correct and partially correct definitions of the term biodiversity. Of the 11,000 individuals surveyed in 2013, 67% had heard of the term biodiversity. Overall the results of the survey demonstrate that for the most part, there is an increase in the level of awareness of consumers regarding not only the term biodiversity, but also what it means (UEBT, 2013). These results also indicate that there are large variations in the number of people that have heard the term biodiversity and can correctly or partially define it.

The World Association of Zoos and Aquariums (WAZA), as part of a global campaign for raising awareness about biodiversity, conducted an international evaluation of biodiversity understanding and knowledge of actions to help protect biodiversity among zoo and aquarium visitors worldwide (Moss *et al.*, 2014). More than 6,000 visitors to 30 zoos and aquariums participated in this pre- and post-visit repeated-measures survey. Prior to their zoo or aquarium visit, 69.8% of respondents had at least a reasonable understanding of biodiversity, that is they could provide a reasonably correct definition of biodiversity. Further, 50.5% of respondents could identify an individual action beneficial for biodiversity. Following their zoo or aquarium visit there was a statistically significant increase in both of these variables. Biodiversity understanding increased from 69.8% to 75.1% while knowledge of actions to help protect biodiversity increased from 50.5% to 58.8% (Moss *et al.*, 2014).

A further source of information on progress towards this target is trends in internet searches for biodiversity information. Information from Google trends, which reflects the number of Google searches for a given term relative to the total number of searches done, shows that searches for the term "biodiversity" has declined since 2004 (see Figure 1.2). Further there have been fewer searches for the term "biodiversity" and "ecosystem" in comparison to "climate change". While this indicator does not measure awareness of the term biodiversity or if individuals are aware of its value, it does provide an indication of interest in biodiversity. It suggests that interest in biodiversity has remained relatively low over the last 10 years, particularly when compared to climate change, and that it may be declining. It is important to note, however, that while information from these types of Google search data they are used in numerous fields and have been shown to be a clear proxy for underlying trends (McCallum & Bury, 2013), it has also been noted that this type of data should be interpreted with caution (Ficetola, 2013).

The results from the various surveys that have been undertaken suggest that there has been a general, but modest, increase in the number of people aware of the term biodiversity. However this awareness has not necessarily been translated into an understanding of what the term biodiversity means or of the actions that can be taken to protect it.

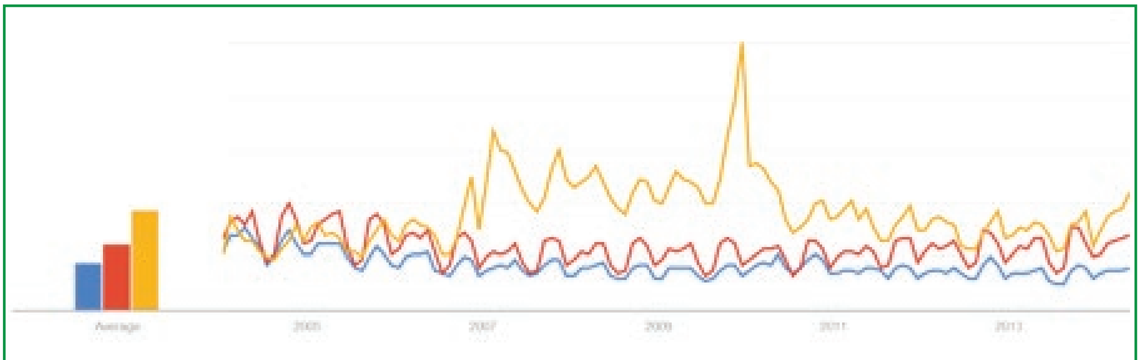

Figure 1.2. Information derived from Google Trends for the search terms "biodiversity" (blue), "ecosystem" (red) and "climate change" (yellow) between January 2004 and May 2014. Source: http://www.google.ca/trends/explore#q=biodiversity%2C%20 climate%20change%2C%20ecosystem&cmpt=q , accessed August 2014.

1.1.2 Projecting forward to 2020

Extrapolation of the information from the Biodiversity Barometer shows that the ability of respondents to provide a correct definition of biodiversity remains low, with fewer than one third of the survey respondents able to define biodiversity correctly in 2013, which is not predicted to increase markedly by 2020 (Figure 1.3A). More encouragingly, approximately two thirds of the survey respondents had heard of biodiversity in 2013 and this is projected to increase steadily to 2020 (Figure 1.3B). These figures are remarkably similar to the WAZA figure of some 69.8% of pre-visit survey respondents demonstrating awareness of biodiversity. While comparisons made between the two will require further qualification, the similarity does support the projections (Moss *et al.*, 2014). As additional data both from the next iterations of the WAZA survey and the Biodiversity Barometer are available, the level of confidence in the projections is likely to increase.

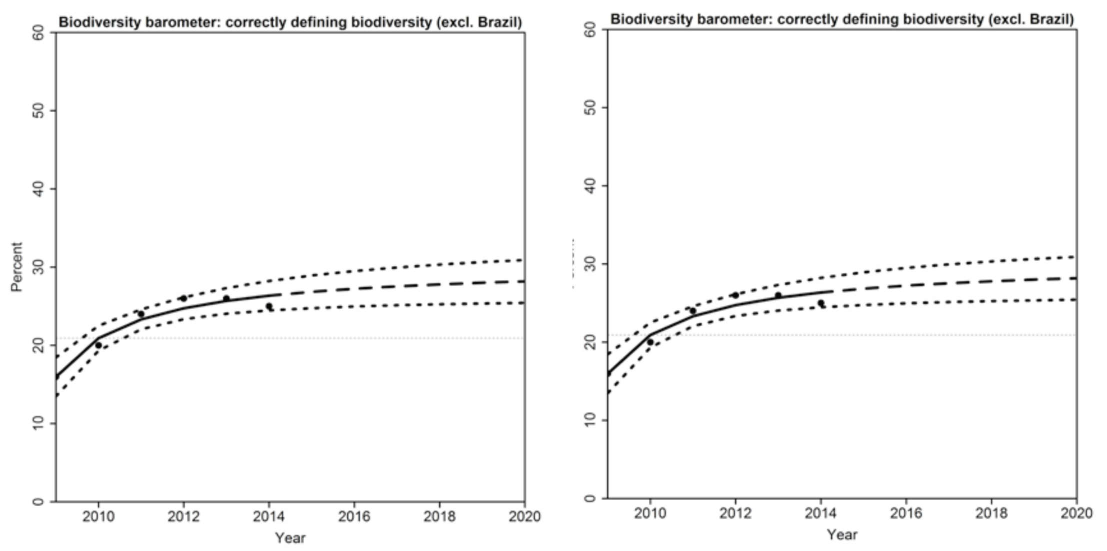

Figure 1.3. Modelled trend in the Biodiversity barometer from 2009-2013 and statistical extrapolations from 2013 to 2020. A) the percentage of respondents giving correct definition of biodiversity B) the percentage of respondents that had heard of the term biodiversity. Both A) and B) show a significant increase in the underlying trend between 2010 and 2020. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant and because of methodological reasons are only based on data from Germany, France, the United Kingdom and the United States of America. Source: based on BIP indicator, UEBT & IPSOS (2014).

Extrapolations of the number of searches for biodiversity-related subjects on the Google search engine shows that online interest in biodiversity has in general decreased since 2004 and is projected to continue to decrease to 2020 albeit at a slower rate (Figure 1.4).

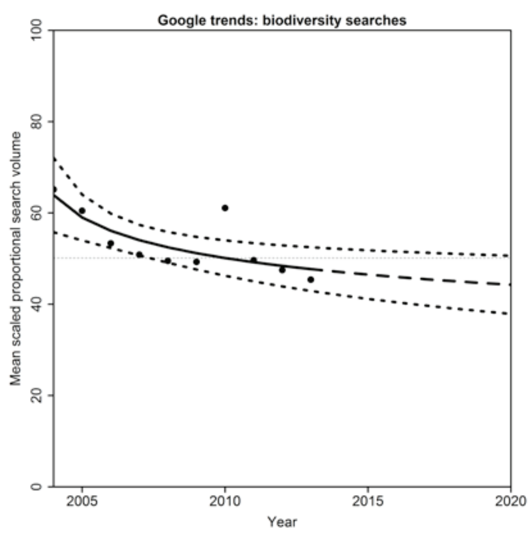

Figure 1.4. Modelled trend in online interest in biodiversity from 2004-2013 and statistical extrapolations from 2013 to 2020. The trend indicates a non-significant decline in the underlying trend between 2010 and 2020. Solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant. Source: extrapolation based on google trend data http://www.google.ca/trends/explore#q=biodiversity%2C%20climate%20change%2C%20ecosystem&cmpt=q

AidData is an organisation that collects data on global funds and assigns funds to specific uses and sectors. The data is gathered from a wide range of funding sources including: the World Bank; the United Nations (UN); the Global Environment Facility (GEF); the Multilateral Fund for the Implementation of the Montreal Protocol; nation states; and multilateral donors such as the African Development Bank; the Caribbean Development Bank; the OPEC Fund for International Development; the Nordic Development Fund; the Asian Development Bank; and the Inter-American Development Bank. Information from AidData on investments in environmental education provides an indication of the global commitment to increase awareness of environmental issues. The data covers education projects varying from increasing capacity at schools for environmental education, to practical training in environmentally-friendly practices, to increasing awareness in the public through local facilities and campaigns. Investment in environmental education has shown a general though non-significant decline in the last decade and this is extrapolated to continue to 2020, though the difference between 2010 and 2020 is not calculated as significant and the confidence in the projections is quite low (see Figure 1.5).

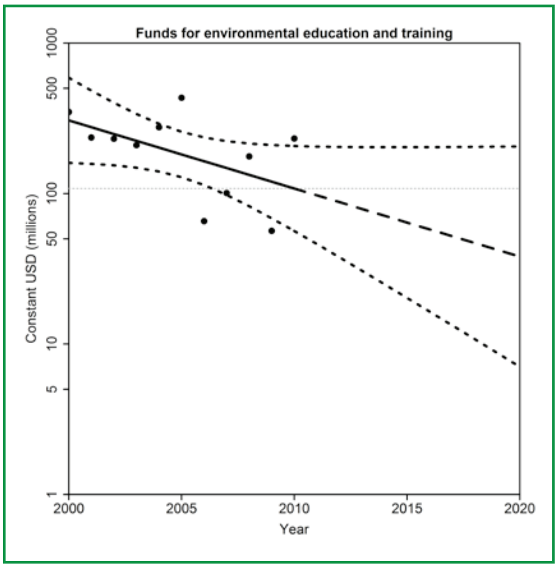

Figure 1.5. Modelled trend in investment in environmental education 2000-2011 and statistical extrapolation from 2011 to 2020. Data is presented in constant US dollars (set at 2009 levels). As data collected pre-2000 and post-2011 was considered to be incomplete, trends were based upon funds committed from 2000-2011 only. The trend suggests a declining but non-significant trend between 2010 and 2020. Note that the y-axis is log-scaled. Solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant. Source: Tittensor et al. (2014).

Awareness of biodiversity is likely to continue to increase as a result of advances in education and communications. However a great deal of additional work will be needed if Aichi Biodiversity Target 1 is to be achieved by 2020. The few survey results indicate that while awareness is increasing in general, the rate is not particularly high, and moreover, it is not consistent from country to country. If current trends continue, by 2020, we will not have a sufficiently high level of awareness to be able to support a claim that Aichi Biodiversity Target 1 has been achieved.

Research also points to the need to gather more data on changes in the level of awareness of the actions that people could take to save biodiversity. In this regard, the WAZA survey needs to be repeated. Further, data on consumer actions, gathered through the Biodiversity Barometer, needs to be highlighted and compared with the biodiversity-relevant findings of other sustainable consumption surveys such as Greendex[1].

1.1.3 Country actions and commitments[2]

Almost all of the National Biodiversity Strategies and Action Plans (NBSAPs) examined contain targets or similar commitments related to increasing public awareness. The majority of these targets are in line with the general scope and aim of Aichi Biodiversity Target 1. Most targets refer to increasing awareness of the importance of biodiversity generally. Comparatively fewer targets explicitly refer to raising peoples' awareness of the types of actions they can take to conserve biodiversity. One example, which is counter to this general trend, is Australia which has set a target of having a 25% increase in the number of Australians and public and private organisations who participate in biodiversity conservation activities.

Almost half of the targets contained in the NBSAPS assessed identify associated supporting or priority actions. For example, Ireland has identified a number of actions related to increasing appreciation of the values of biodiversity including working with relevant departments and stakeholders to include biodiversity and ecosystem goods and services in relevant courses in secondary and third level education and developing and implementing a communications campaign in support of full implementation of its Biodiversity Action Plan.

Footnotes

[1] *http://environment.nationalgeographic.com/environment/greendex/*

[2] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil.*

A few countries have also set quantitative targets. For example, Malta has established a target of having 55% of its citizens being aware of the term biodiversity and of the steps they can take to conserver and use it sustainably. Some commitments have also been made with regards to specific groups. For example, Suriname has set an objective of raising awareness of the importance of biodiversity in the agricultural and fisheries sector. Similarly some countries have set targets which focus on different avenues for awareness raising. For example, in its NBSAP, Belarus has committed to creating a network of "green schools" and to establishing 15 ecological centers.

Communication, education and public awareness (CEPA) has also been reflected in NBSAPs to include general communications campaigns, awareness-raising within the national government, across ministries, educational initiatives, and actions targeted at particular sectors for which Aichi Biodiversity Targets exist, such as agriculture. 23 NBSAPs submitted to the Secretariat were analyzed for inclusion of a CEPA strategy. 10 of these NBSAPs included an explicit strategy and 8 included a relevant target and plans to develop a strategy. In most cases, the integration of CEPA into the strategies included plans to mainstream biodiversity into education, whether in the formal system, or in other informal contexts of learning.

1.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

1.2.1 Actions

In order to progress towards this target, Parties will need to develop and implement coherent, strategic and sustained communication, education and public awareness initiatives, alone and in partnership with other actors. There are a variety of communication and outreach vehicles which can be used to attain this target. Different types of education and public awareness activities or campaigns, including social marketing techniques, will be needed to reach different audiences, as activities and messages which are effective for one group may not be for others. Such activities will help to mainstream biodiversity across society. There are multiple avenues for increasing awareness of biodiversity. These include formal learning through schools and universities, informal learning both at home and at the local community level, and non-formal learning at museums, botanical gardens, zoos, aquariums, and parks, as well as awareness that is generated from exposure to material on biodiversity featured on television and radio, in print publications, and on the internet and social media. Awareness and learning also occurs through participation in events (see Box 1.2 and Box 1.3) and other opportunities for information exchange between stakeholders.

Box 1.2: Actions for Biodiversity

In support of the International Year of Biodiversity, the Royal Belgian Institute of Natural Sciences worked to compile a list of 366 actions that citizens could take to protect biodiversity. With the support of the European Commission the list was later condensed to 52 actions – one a week, and published as a booklet in the 6 United Nations Languages as well as in Dutch. The actions are presented one per page, along with a colour cartoon, and an explanation of how the action can make a difference for biodiversity. The actions contained in the booklet cover issues related to over consumption (water wastage, excess energy consumption), over exploitation (careful choices of seafood), awareness (nature walks and urban biodiversity) and invasive alien species for example.

Similarly the Ministry of Environment of Benin, with the support of the Royal Belgian Institute of Natural Sciences, produced their "12 gestes pour la biodiversite." The 12 actions, one for each month, are linked to a theme of biodiversity protection, and include a set of actions that can be carried out each month. The project presented the information in a wall calendar format as well as a booklet that also showed some of the important international days that could be celebrated during a given month. The product was used in schools and linked to capacity development activities.

The World Association of Zoos and Aquariums (WAZA), representing a community that attracts more than 700 million visits every year, has designed a global campaign for raising awareness about biodiversity. This is in support of the United Nations Decade on Biodiversity, especially towards achieving Aichi Biodiversity Target 1. In addition to posters and films, campaign materials include modern technologies aimed at global youths, such as social media and a smart phone/tablet application in five languages. This application is aimed at inspiring a new generation by raising awareness about the values of biodiversity. For each species, the application includes a short profile, IUCN Red List status and distribution map, as well as having a strong focus on simple actions everyone can take to conserve and use biodiversity sustainably (https://www.biodiversityisus.org/).

Box 1.3: The International Day for Biological Diversity

The International Day for Biological Diversity (IDB) is a major tool that is increasingly being used by Parties to raise awareness and to focus their communications efforts. From the year 2003, when information on IDB celebrations held by Parties and organizations was first collected by the SCBD to the present, there has been an increase in the number of celebrations and activities reported, particularly after the year 2010. Table 1.1 provides data on activities reported to the Secretariat of the Convention on Biological Diversity (CBD) on IDB celebrations between 2003 and 2014. As this data is self-reported, there is a significant possibility that the actual number of celebrations is higher.

Table 1.1. Reporting on Celebrations for the International Day for Biological Diversity.

Year	Theme	Participating countries	Participating organisations
2014	Island Biodiversity (reported as of 28 May 2014)	71	11
2013	Water and Biodiversity	51	17
2012	Marine Biodiversity	71	12
2011	Forest Biodiversity	41	6
2010	Biodiversity, Development and Poverty Alleviation	62	9
2009	Invasive Alien Species	36	22
2008	Biodiversity and Agriculture	35	3
2007	Biodiversity and Climate Change	67	19
2006	Protect Biodiversity in Drylands	34	2
2005	Biodiversity: Life Insurance for our Changing World	19	3
2004	Biodiversity: Food, Water and Health for All	17	2
2003	Biodiversity and poverty alleviation - challenges for sustainable development	11	0
2002	Dedicated to forest biodiversity	N/A	N/A

Source: SCBD (www.cbd.int/idb)

Generally a better understanding of the relationship between awareness and action is needed. Researchers have been paying increasing attention to the relationship between awareness, values and behaviour change in order to better understand more effective ways to implement Aichi Biodiversity Target 1. Researchers are bringing together findings from the domains of social psychology, behavioural psychology and biodiversity conservation in order to better understand the different ways in which social-psychological and material factors interact with economic factors to shape behaviours that have an impact on the conservation and sustainable use of biodiversity. Recent literature from the behavioral sciences, point to a complex decision-making process (Schultz, 2011)[3] whereby individuals do not always act as rational economic agents looking for optimum solutions but are motivated by factors such as ego, emotion, culture and religion, among others (Duraiappah *et al.*, 2013). Identifying and understanding the factors that motivate and constrain individual, collective, and organizational behaviours can help contextualize and facilitate efforts to promote sustainable biodiversity management.

Footnote

[3] Some of the other writings in this field include: Kahneman, Daniel, 2011, Thinking Fast and Slow Farrar, Straus and Giroux, USA, McKenzie-Mohr, D. Lee, N.R. Schultz, P.W. Kotler, P. 2012, Social Marketing to protect the environment What works, Sage publications, Los Angeles USA, Prager, K. Schultz "Understanding Behavioural Change: How to apply theories of behaviour change to SEWeb and related public engagement activities," Life10 ENV-UK-000182. Much of this previous literature has been brought together with work on institutions by: Anantha Kumar Duraiappah, Stanley Tanyi Asah, Eduardo S. Brondizio, Nicolas Kosoy, Patrick J O'Farrellm Anne-Helene Prieur-Richard, Suneetha M Subramanian and Kazuhiko Takeuchi, "Managing the mismatches to provide ecosystem services for human well-being: a conceptual framework for understanding the New Commons," Current Opinion in Environmental Sustainability 2014.

The research suggests that there are a great variety of tools and approaches to promote "pro-biodiversity" behaviours. The research also shows that these mechanisms could be excellent complements to mechanisms that use formal control and the enforcement of sanctions. Employing strategies that use motivations and social and moral, as well as economic incentives as the mechanisms for promoting behaviour change can not only bring about such changes, but can also more effectively empower people to sustainably manage biodiversity, which is an important factor. One emerging conclusion is that while education and information regarding the value of biodiversity to society is important, the impact is limited when learning tools are developed and delivered by external experts through a non-participatory process.This research has been applied in the work of organizations such as Rare Conservation, which has carried out 265 campaigns in 56 countries to date[4]. PCI-Media Impact has employed the ideas of this research in their communication campaigns, currently running in 50 countries around the world[5].

It is clear that more research is needed to understand how social-psychological and material factors interact with economic factors to shape pertinent responses and behaviours with an impact on biodiversity management.

There are many indigenous and local communities and civil society groups which have undertaken public awareness activities related to biodiversity and Parties to the Convention could encourage or support these, as appropriate, as a means of attaining this target. Similarly facilitating and encouraging the engagement of citizens on biodiversity issues, including through activities to monitor biodiversity and to promote its conservation and sustainable use, would help to make progress towards this target.

There is a need to move away from general catch-all awareness raising strategies. Awareness raising activities need to be more targeted interventions. Key demographic groups that set consumption trends or make important decisions affecting biodiversity could be a focus of such interventions. Similarly the integration of biodiversity issues into national educational curricula, taking into account approaches related to Education for Sustainable Development (ESD) would also facilitate progress towards this target.

Policy design needs to link awareness with specific goals, rather than just a general need for greater awareness. Governments at all levels need to connect policy goals and behaviour change goals with awareness raising strategies. This would imply that awareness-raising becomes a key part of policy implementation, rather than a stand alone goal. These efforts could be integrated into NBSAPs, and fully combined with any mainstreaming initiatives. In general, awareness-raising activities need to be more targeted interventions. Based on national-level priorities under the NBSAPs, key demographic groups can be identified for such actions. For example, those groups that set consumption trends or make important decisions affecting biodiversity for strategic biomes or areas of action could be a focus of such interventions.

Better coordination in the collection and compilation of existing information combined with enhanced efforts to assess trends in awareness of biodiversity would increase our ability to assess the effectiveness of the types of actions taken. Periodic monitoring of awareness of the values of biodiversity would also allow for baselines and trends in awareness to be assessed.

1.2.2. Costs and Cost-benefit analysis

There is relatively little information available on the costs and benefits associated with the attainment of this target. The first report of the High-Level Panel on Global Assessment of Resources for Implementing the Strategic Plan for Biodiversity 2011-2020 estimated that the cost of meeting Aichi Biodiversity Target 1 would require a global investment of US$0.05 billion between 2013 and 2015, followed by annual recurrent expenditures of between US$0.44 and US$1.41 billion between 2015 and 2020 (High Level Panel 2012). However these costs are highly speculative given the various actions that could be undertaken to reach this target and that to be effective awareness raising actions would need to be tailored not only to national circumstances but also to effectively reach different national stakeholder groups. There is no global estimate of the potential benefit of reaching this target. However given that its attainment would help to address the underlying causes of biodiversity it has the potential to play a catalytic role in bringing about the wider societal changes that are needed to attain the other Aichi Biodiversity Target and to fulfill the mission and vision of the Strategic Plan more broadly. It would therefore seem prudent to invest in the achievement of Aichi Biodiversity Target 1 as a means to support implementation of the Strategic Plan for Biodiversity 2011-2020 at large.

Footnotes

[4] *www.rare.org*

[5] *http://mediaimpact.org*

1.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

Ultimately halting the loss of biodiversity at the global level will require that the underlying drivers of biodiversity loss are reduced. Addressing the drivers of biodiversity loss requires behavioural change by individuals (e.g., to reduce waste or unsustainable consumption) and by governments (e.g., to change regulations or incentives). Understanding, awareness and appreciation of the diverse values of biodiversity, are necessary to underpin the ability and willingness of individuals to make such changes and to create the "political will" for governments to act.

However increasing awareness and bringing about behaviour change on the scale required to have an effect on the underlying drivers of biodiversity loss are likely long term endeavours. This emphasizes the need to implement all the Aichi Biodiversity Targets in parallel as actions taken to raise awareness and bring about behaviour change will require time to take hold and affect the underlying causes of biodiversity loss.

1.4 UNCERTAINTIES

The main uncertainty associated with this assessment is the limited information on biodiversity awareness that is available. Globally coherent data on peoples' awareness of biodiversity is not available. This not only reflects challenges in aggregating national and regional measures at a global level, but also the paucity of resources devoted to data collection more generally.

In some settings collecting information on awareness of biodiversity can be challenging as awareness of biodiversity is deeply embedded in national and local contexts rather than in abstract concepts of biodiversity.

Individuals who have not heard of the term biodiversity or who do not have a set definition for it may nonetheless know what it is. Therefore while measuring awareness of the term biodiversity may be the easiest way to develop an indicator it might obscure important instances where actors understand biodiversity and the values it provides to their lives, but do not refer to it using this term. Therefore, measuring understanding of the term biodiversity may obscure important advances in awareness of biodiversity and lead to more pessimistic interpretations of biodiversity awareness.

1.5 DASHBOARD – PROGRESS TOWARDS TARGET

Element	Current Status	Comments	Confidence
People are aware of the values of biodiversity		Limited geographical coverage of indicators. Strong regional differences	Low
People are aware of the steps they can take to conserve and sustainably use biodiversity		Evidence suggests a growing knowledge of actions available, but limited understanding of which will have positive impacts	Low

Author: Secretariat of the Convention on Biological Diversity
Extrapolations: Derek Tittensor
NBSAPs and national reports: Kieran Noonan-Mooney
Dashboard: Tim Hirsch

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TARGET 2: INTEGRATION OF BIODIVERSITY VALUES

By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.

PREFACE

The Convention on Biological Diversity (CBD) acknowledges the 'intrinsic, ecological, genetic, social, economic, scientific, educational, recreational and aesthetic values of biological diversity and its components'. Typically, these values are not reflected in decision-making resulting in negligence and/or overexploitation of biodiversity and ecosystems resources that if retained will endanger the well-being of future generations (Millennium Ecosystem Assessment; MA, 2005). Aichi Biodiversity Target 2 aims at addressing the underlying causes of biodiversity loss and ecosystem degradation by bringing them into decision-making. This requires knowledge on all values of biodiversity, but also keeping track of the stocks and flows of these resources through appropriate accounting and reporting mechanisms.

This chapter aims at analysing the progress towards Aichi Target 2. Section 2.1 provides an assessment of the current status towards the achievement of the Target. This section begins with a short introduction to the status of biodiversity and ecosystems valuation, it continues with an analysis on the integration of biodiversity values in national and local development strategies, namely how biodiversity is considered in poverty reduction strategies and environmental impact assessments and strategic environmental assessments. An analysis of environmental national accounting status is also presented. Next, it is analysed how the CBD is promoting biodiversity mainstreaming at the national level and the countries actions and commitments presented in the recent National Biodiversity Strategic Action Plans (NBSAPs) regarding mainstreaming biodiversity and ecosystems values. Section 2.1 ends by exploring the progress towards the achievement of Target 2 until 2020. Section 2.2 presents some of the actions that need to be taken into account to achieve the target. Section 2.3 explores the implications of Target 2 to biodiversity. Section 2.4 highlights some of the uncertainties associated with Target 2. Finally, Section 2.5 provides a graphical summary of the progress towards Target 2.

The CBD suggests several possible indicators to measure progress towards Target 2. However, until now there are no globally harmonised datasets that fulfil the data requirements to monitor this Target (Group on Earth Observations Biodiversity Observation Network; GEO BON, 2011).

Scenarios to 2050 and beyond are not presented because this analysis does not lend itself to long-term projections.

2.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

2.1.1 Status and trends

2.1.1.i Valuation

Integrating biodiversity values into policy and decision making requires the knowledge on what these values are. Valuing something implies that there is an importance within the object being valued, regarding biodiversity this importance can be attributed based on a number of reasons, for example, economic, moral, spiritual or aesthetic, and it differs from individual to individual (Dietz *et al.*, 2005; Environment Protection Agency, 2009; The Economics of Ecosystems and Biodiversity; TEEB, 2010). As a result, putting a value on biodiversity is a complex and multi-dimensional task, that should take into account several and different types of values (Office for Economic Cooperation and Development; OECD, 2002; TEEB, 2010; Turner *et al.*, 2003).

Biodiversity, as a regulator of fundamental processes that deliver ecosystem services, as a final ecosystem service itself or as good (Mace *et al.*, 2012) is certainly important to economic processes and human well-being (OECD, 2002; TEEB, 2010; UK National Ecosystem Assessment; UKNEA, 2011). This importance is normally referred to in terms of ecosystem services and can be expressed in monetary units under the Total Economic Value

(TEV) framework (Figure 2.1). Ecosystem services are defined as the direct and indirect contributions of ecosystems to human well-being (TEEB, 2010). Several classifications for the different types of services provided by ecosystems exist (Haines-Young and Potschin, 2012; MA, 2005; TEEB, 2010). Services can be classified as provisioning (the products derived from ecosystems), regulating and supporting (the services that regulate ecosystems and enable the production of all other ecosystem services) and cultural (the non-material benefits derived from ecosystems). Depending on the classification used supporting services can also be named habitat services, or can be joined with regulating services.

Components of TEV usually are represented using a value-taxonomy. The main distinction made is between use and non-use values (Pearce & Turner, 1990). Use values are composed by direct use, indirect use, and option values. Direct use values refer to the benefits that can be taken directly from the ecosystem, indirect use values refer to societal or functional benefits derived from ecosystems, option values to potential future direct and indirect use values. Non-use values are composed by existence and bequest values. Existence values refer to the value put on knowing that future generations will still benefit from biodiversity and ecosystems (TEEB, 2010; UKNEA, 2011). Bequest values concern the values associated with knowing that species and ecosystems will continue to exist (TEEB, 2010, UKNEA, 2011).

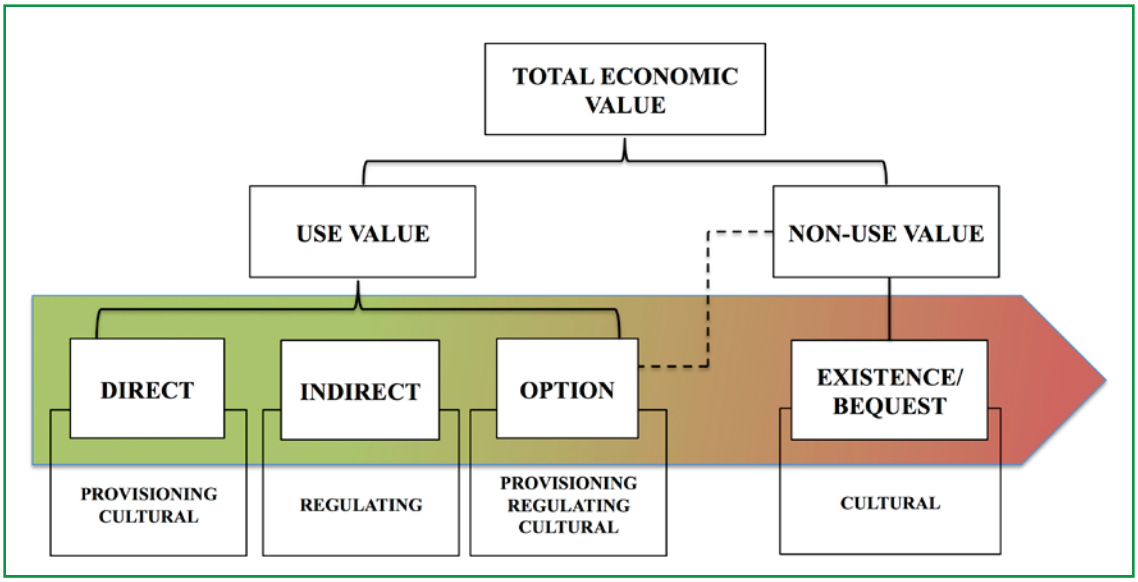


Figure 2.1. The Total Economic Value Framework (TEV) and the different types of values of different ecosystem services categories. The gradient arrow indicates the increasing difficulty (from green to red) in providing a monetary estimate for the different types of values. Source: Produced by the author of this chapter.

Economic valuation of ecosystem services is useful in several ways; it can help to communicate the value of nature to people by using a common unit known by most, it allows to assess the trade-off between different policy options by expressing the effect of a marginal change in ecosystem services supply due to a policy choice against the alternatives and it allows to assess the cost and benefits of conservation policies (Costanza *et al.*, 1997; Liu *et al.*, 2010; Millennium Ecosystem Assessment, 2005; OECD, 2002; Turner *et al.*, 2003). While important, economic valuation should not be seen as the only way of understanding the value of biodiversity and ecosystems. There are several aspects of biodiversity that cannot be measured in monetary units, for example its spiritual importance or aesthetical value.

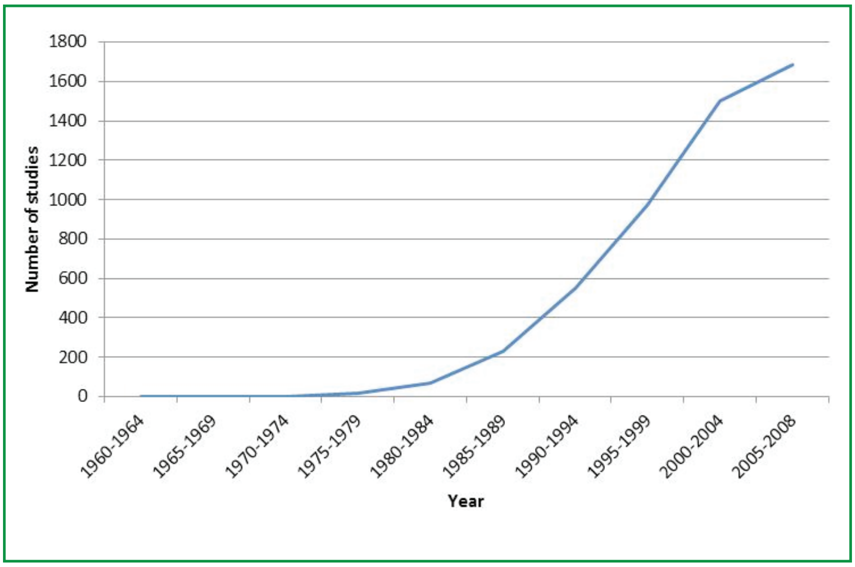
The United Nations Environment Programme's (UNEP) report "Cultural and Spiritual Values of Biodiversity" (UNEP, 1999) presents examples of biodiversity values that cannot be monetised, for example by demonstrating the importance of biodiversity to culture through the exploration of the links between language diversity and human observation of nature and natural cycles, and how language encodes indigenous local knowledge essential to effective conservation of natural resources. Gregory and Trousdale (2009) show how market based approaches, amongst other conventional practices, are inadequate to measure culture losses experienced by North American Aboriginal Communities due to loss of ecosystem services. In a literature survey exploring the intangible links between nature and human well-being, Russell *et al.* (2013) concluded that the positive effects of nature on physiological and mental health have been unequivocally documented and that the strong positive effects of nature on identity and spirituality have been strongly documented for indigenous groups but lack of knowledge exists for other cultures. They pointed out that the inclusion of these values in decision-making has been hindered by the existence of a vast and heterogeneous body of knowledge regarding the non-economic values for biodiversity. Nature also holds a value in itself that is independent from human experience and valuation (intrinsic value) (UNEP, 2013).

In the remainder of the section we explore the several initiatives on economic valuation of ecosystem services and their integration into decision-making.

The idea of putting a monetary value on nature has a long history (see Liu *et al*. (2010) for an historical perspective on ecosystem services valuation research). But this approach only gained wider attention in 1997, when Costanza *et al*. (1997) provided a value for the services provided by ecosystems globally, with an increasing number of studies addressing ecosystem services valuation (Figure 2.2).

benefit-transfer methods, of the value of ecosystem services per 10 biome types. Values found range from 490 int$per year per hectare of open oceans to almost 350,000 int$per year per hectare of coral reefs. More importantly, the analysis showed that it is not possible yet to cover all the biomes and services with the same detail, with consequences for the results obtained. For example, for open oceans and rivers and lakes only 14 and 15 studies where used covering 5 and 4 of the 22 services analysed. To determine the value of coastal wetland and inland wetlands, 139 and 168 studies where used, covering 13 and 17 services. Costanza *et al*. (2014) provide an update estimate, from the 1997 estimate (Costanza *et al*., 1997) of the global value of ecosystem services, based on ESVD and de Groot *et al*. (2012). In 1997, the global value of ecosystems was estimated at

Figure 2.2. Cumulative total of ecosystem services valuation studies sourced from the Environmental Valuation Reference Inventory (www.evri.ca), from 1960 to 2008. Source: From de Groot *et al*. (2012).



The MA (2005) has surveyed the contribution of ecosystems to human well-being, and was a keystone in shifting the perspective from ecosystem functions to services. Recently, TEEB (2010) has provided the most comprehensive review on biodiversity and ecosystem services valuation. A part of this study consisted of an estimation of monetary values of ecosystem services per 10 biomes types, at a global level. Approximately 1300 studies were reviewed and information systematised in an open access database, the Ecosystem Services Valuation Database (ESVD, van der Ploeg & de Groot, 2010). De Groot *et al*. (2012) analysed 685 studies from the ESVD to provide global monetary estimates (in 2007 international dollars[1] per hectare per year), through

US$46 trillion/yr in 2007, in 2011 the value of ecosystem services was estimated to be US$125 trillion/yr in 2007. The loss of ecosystem services due to land use change from 1997 to 2011 was estimated at US$4.3-20.2 trillion/yr.

Other global databases exist; Canada hosts the Environmental Valuation Reference Inventory (www.evri.ca)[2] with over 3600 entries that can be used, as ESVD, to estimate economic values through benefit-transfer[3] methodologies. Its geographical coverage is greater for North America (1773 studies) and Europe (1066 studies). Earth Economics developed the Ecosystem Valuation Toolkit (www.esvaluation.org), which provides a library for the research community with a collection of ecosystem services valuation studies,

Footnotes

[1] An international dollar is a hypothetical unit of currency that has the same purchasing power parity (PPP) that the US dollar had in the United States in the reference year.

[2] EVRI's open access is limited to residents of Australia, Canada, France, New Zealand, United Kingdom and the United States.

[3] Benefit transfer is the procedure of estimating the value of an ecosystem service by transferring an existing valuation estimate from a similar ecosystem, it is its approach that aims at overcoming the lack of site-specific information in an expedite and inexpensive way (TEEB, 2010).

resources for education, policy and best practices and also a tool (named SERVES) for the estimation of the value of ecosystem services in specific areas. The Natural Capital Project (www.naturalcapitalproject.org) developed InVEST, a software tool, designed to support decision making in assessing the trade-offs of different management decisions by mapping and valuating ecosystem services (Daily *et al.*, 2009). Some interesting case studies on the usefulness of InVEST in supporting policy decisions are reported in McKenzie and Rosenthal (2012). Artificial Intelligence for Ecosystem Services (ARIES) is another tool developed to provide a rapid ecosystem services assessment and valuation (www.ariesonline.org). It aims to quantify ecosystem services in a manner that acknowledges dynamic complexity and its consequences. An ensemble of models exists within ARIES that allows modelling at a specific spatial scale and ecological and socioeconomic context (Villa *et al.*, 2014). The Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services (IPBES) has been developing a catalogue of assessments on biodiversity and ecosystem services (www.catalog.ipbes.net/) that includes monetary valuation and non-monetary valuation exercises.

At the national level, there are several initiatives aiming at providing economic estimates of a country's biodiversity and ecosystem services values. Following up the TEEB study, several countries have initiated TEEB-like studies. Currently, 19[4] initiatives are listed in TEEB's Country Profiles ranging in geographical scope from local to national. Recently, five more countries, Bhutan, Ecuador, Liberia, Philippines and Tanzania, expressed interest in taking TEEB country studies under a project financed by the European Commission (Reflecting the value of ecosystems and biodiversity in policy-making). At the European level, the European Biodiversity Strategy for 2020 requires that all Member States map and assess the state of ecosystems and their services by 2014 and that by 2020, the economic value of such ecosystem services is assessed and integrated into accounting and reporting systems (Target 2 – Action 5, EC, 2011). An overview of the ecosystem services valuation initiatives by Member States revealed that many assessments at the European level are inspired in TEEB[4] and the majority is still in an early development phase; in all assessments only a small subset of ecosystem services was or is going to be valued (Brouwer *et al.*, 2013). The choice of valuation methods is still not clear but it is shown to be dependent on data availability. In Germany, Norway and Netherlands non-market valuation is seen as a non-acceptable procedure due to the lack of data necessary for this type of valuations and the uncertainties with it associated (Brouwer *et al.*, 2013). Also in relation with the achievement of Target 2 of the European Biodiversity Strategy for 2020, a working group on mapping and assessment on Ecosystems and their services (MAES) was established. Until now it has developed an analytical framework for biophysical assessments (European Commission, 2013, 2014) that is consistent with standard economic definitions by applying the Common International Classification of Ecosystem Services (Haines-Young & Potschin, 2012).

The UKNEA (2011) is probably the most comprehensive assessment available at the present date. It provided conceptual advances, for example how biodiversity was addressed (Mace *et al.*, 2012) and how to avoid double counting in ecosystem services valuation (UKNEA, 2011, Box 2.1). For example, national assessments are very resource intensive, therefore, it is unlikely that all countries can carry out such a task. The UKNEA also found that it is difficult to value all ecosystem services using the same methodology rendering difficulties in comparisons (Brouwer *et al.*, 2013; UKNEA, 2011). The assessment also highlighted that certain social values, like the spiritual value of the environment, cannot be easily measured with currently available economic valuation methods. Martín-López *et al.* (2014) also found that different methods used to value ecosystem services actually revealed different information, supporting previous works, which state that valuation methods are not neutral and that the choice of methods may be as important as the valuation result itself (Gómez-Baggethun & Ruiz-Pérez, 2011).

Footnotes

[4] *Armenia (local), Southeast Asia (regional), Belgium (national and regional), Brazil (national), Czech Republic (national), Georgia (national), Germany (national), Heart of Borneo (regional), India (national), Japan (national), Netherlands (national), Nordic Countries (regional), Norway (national), Poland (local), Portugal (national), Republic of Korea (national), Slovakia (national and local), South Africa (national) and United Kingdom (national).*

Box 2.1: Case Study: Biodiversity values are shown to be enhanced by ecosystem services policies in the UK National Ecosystem Assessment (UKNEA)

Between 2009 and 2011 a UK government-led assessment considered the state of the UK's ecosystems and their contributions to people's welfare over coming decades (UKNEA, 2011). The work was based on the ecosystem services concepts developed by the MA (2005). Biodiversity was however considered somewhat independently of ecosystem services. The assessment recognised that while many ecosystem services are underpinned by biodiversity, at the same time there are significant biodiversity values that are themselves dependent upon ecosystem management (Mace *et al.*, 2012). In particular, species and habitat conservation priorities are often outcomes of ecosystem management rather than inputs to it. One key policy question then concerns the extent to which management of the landscape for ecosystem services is consistent with management for biodiversity conservation.

The work in the UK NEA was designed to accommodate economic valuation of ecosystem services, in an explicitly spatial manner. Those ecosystem services for which economic values could be estimated (even in the absence of markets) were considered in an analysis that compares the economic values of alternative approaches to land use management between 2010 and 2060. There are four ecosystem services for which economic values could be reliably estimated across the UK (agricultural production, recreational values, economic benefits from urban green space, and greenhouse gas fluxes). The changes in net economic benefits from each of these were compared to the situation in 2010 under two different land use policy scenarios. Under the World Markets scenario, environmental regulation and policy are weak and land use is therefore primarily driven by market forces which will favour agricultural production. In contrast, under the Nature at Work scenario, land use decisions are strongly driven by environmental policy and planning processes to enhance multifunctional landscapes and ecosystem function; land use is therefore driven by wider social values. Unsurprisingly, the World Markets scenario leads to increased economic benefits from agriculture in 2060, but all other ecosystem service values fall. In contrast, under Nature at Work-based policies, all ecosystem services other than agriculture have markedly higher overall value to the UK economy (see Figure 2.3). In this analysis a biodiversity index was also measured based on modeled wild bird species richness. Under World Markets policies, the biodiversity index declined especially in areas of high agricultural value. In contrast, under Nature at Work policies, the biodiversity index was mostly improved relative to the current situation. The findings of this work demonstrate the greater social values that come from adopting a broader ecosystem services approach to land use decisions, and demonstrate that this also has benefits for biodiversity conservation (Bateman *et al.*, 2013).

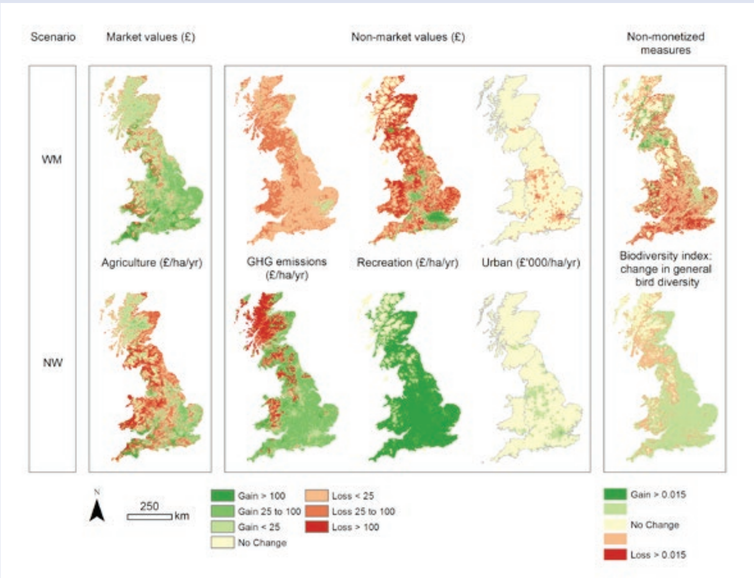


Figure 2.3. Spatial distribution of the changes in market and nonmarket ecosystem service economic values and nonmonetary wild species–diversity assessments. The biodiversity index was measured as changes in Simpson's Diversity Index induced by moving from the year 2010 baseline to the WM and NW scenarios for 2060 [all analyses assume high GHG emission climate-change projections]. Source: From Bateman *et al*. (2013).

2.1.1.ii. Poverty Reduction Strategy Papers (PRSPs) and biodiversity

Poverty Reduction Strategy Papers (PRSPs) are required by the International Monetary Fund (IMF) and the World Bank (WB) as a basis for debt relief or monetary aid to low income countries. In these documents, countries detail their strategy to promote growth and reduce poverty. More recently, PRSPs also provide the basis for monetary aid concerning the achievement of the Millennium Development Goals.

Analysing the extent to which biodiversity and ecosystem services are contemplated in PRSPs provides insights about their integration into development and poverty reduction strategies (Bojo & Reddy, 2003). Roe (2010a) analysed 54 PRSPs (35 from Africa, 7 from America Latina, 7 from South Asia and 5 from South East Asia) to examine how countries integrate biodiversity in their development strategies (Figure 2.4). A scoring system from 0 to 3 was used, 0 meaning that countries do not acknowledge the importance of biodiversity in their development and 3 meaning that not only the importance is acknowledged but also the links between poverty and biodiversity loss, and biodiversity and poverty reduction are recognised. Although the scoring system for each criterion can be somewhat subjective, it allows for a comparison between the countries.

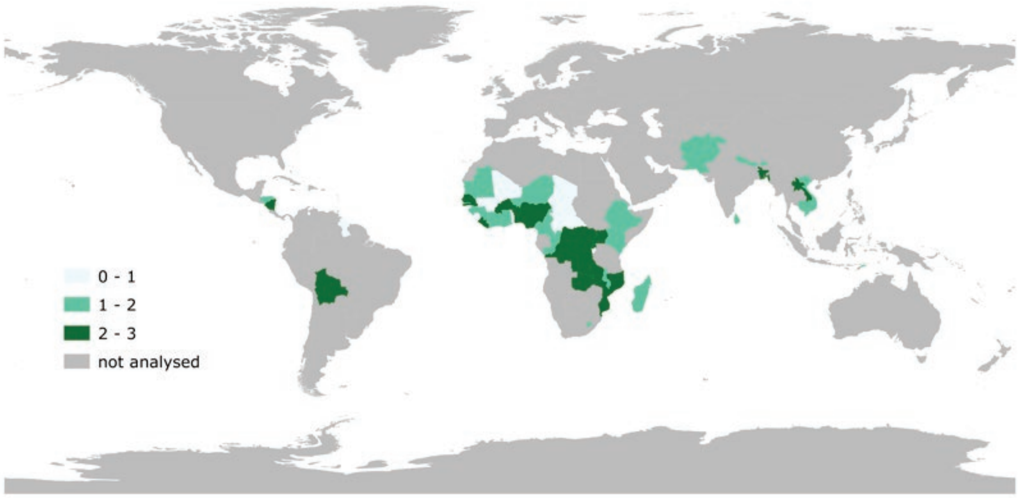


Figure 2.4. Integration of biodiversity in PRSPs of selected countries, scored from 0 to 3. Source: Data from Roe (2010a).

From the countries analysed, 25 % (15) have scored more than 2 in the assessment (Figure 2.1), which means a high level of recognition of the importance of biodiversity in development strategies. The author of the study concluded that overall, the role of biodiversity in contributing to poverty alleviation is acknowledged by the countries but the perspectives of each country can vary (Roe, 2010a). One perspective is that biodiversity is fundamental for poverty alleviation because it provides livelihood for the poorest; another perspective is that biodiversity can contribute to national Gross Domestic Product (GDP). An example of the former can be found in the Democratic Republic of Congo PRSP: *"Some 40 million of the poorest Congolese depend upon the forest for their food, materials, energy, and medicine"*. Example of the latter can be found, for example, in Bolivia's PRSP: *"Preliminary studies indicate that within a period of approximately 15 years the contribution of biodiversity could come to represent an increase of about 10 per cent in GBP"*; in Sri Lanka's PRSP it is highlighted that the unsustainable use of natural resources can cost to the national economy around 2.5% of GDP. Roe (2010a) highlights PRSP's from Bangladesh, Bolivia, Kenya, Lao PDR, Liberia, Rwanda, Sri Lanka and Tanzania as best practice examples. Box 2.2 presents a case study on the recognition of the value of forests and the ecosystem services they provide in Kenya, also on the importance of environmental accounts (see also Section 2.1.1.6) to keep track of these values, and some policy recommendations from the Kenyan Government to preserve forests and their value.

Recognising the importance of biodiversity and ecosystem services for poverty reduction is the first step for consideration of conservation in development strategies. However, linking conservation and development is not straightforward (Adams *et al*., 2004; Roe, 2010b; Sanderson & Redford, 2003). Much of the costs of conservation in poor countries are borne by the poorest and the benefits arising from it are not equitably distributed (Roe & Elliott, 2004). Designing effective policies that benefit both biodiversity and poverty alleviation requires a deeper understanding of which attributes of biodiversity have positive influences on poverty (Roe *et al*., 2014), and which conservation mechanisms have positive influence on poverty reduction (Roe, 2010b). Daw *et al*. (2011) emphasised that disaggregating human well-being is important to understand the contribution of biodiversity and ecosystem services to poverty alleviation. For example, food production (provisioning services) can contribute to poverty alleviation by providing nutritional requirements or by providing income. A literature review on biodiversity conservation as a mechanism for poverty reduction identified nature-based tourism as the mechanism where biodiversity conservation had greater impact on poverty reduction (Table 2.1; Roe, 2010b).

Box 2.2: Case Study: Good practices in Poverty Reduction Strategies Papers – Kenya's Forest Accounts

The Government of Kenya strategic development aspirations and visions are captured in a document entitled 'Vision 2030' (Republic of Kenya, 2007). Among the six sectors identified as priorities within the Medium Term Plan 2008-2012 of Vision 2030, at least four (agriculture, tourism, wholesale and retail trade), which make up the largest part of Kenya's GDP, have linkages either directly or indirectly to montane forests and the crucial services they provide. Measuring and understanding the economic value of forests is important for decision-making processes including planning and budgetary allocations.

Evidence of the Value of Forest and Deforestation on Kenya's Economy - UNEP's technical report *Kenya Integrated Forest Services* (UNEP, 2012a) shows evidence of the value of these forests and the effects of deforestation. In the 10-year period, 2000-2010, deforestation in Kenya's Water Towers amounted to an estimated 50,000 hectares (ha). By 2010 such deforestation of montane forests yielded a timber and fuelwood volume of 250 m^3/ha, with a cash value of 272,000 KSh[7]/ha. At an estimated deforestation rate of 5,000 ha/yr by 2010, this was equivalent to a revenue of approximately KSh 1.362 million in 2010. It is these types of revenue streams that provide an incentive for illegal deforestation activities. However, this cash revenue comes at a large cost to the national economy, through losses in regulating services.

Whereas the cash value of forest products has a once-off value, the benefits of regulating services in preceding years continue to be felt in the economy in every subsequent year that the national asset, the Water Towers, is degraded. By 2010, the cumulative negative effect of deforestation on the economy through reduction in regulating services was an estimated KSh 3.652 million/yr, more than 2.8 times the cash revenue of deforestation. The largest component of this was attributable to changes in river flows resulting from a reduction in dry-season river flows, which reduced the assurance of water supply to irrigation agriculture. This reduced agricultural output by KSh 2.626 million in 2010. The benefits of the forests have an economy-wide effect with a considerable multiplier effect. An industry that directly depends on regulating services generates demand upstream (for intermediates from other industries) and also supplies inputs to other industries downstream. Taking into account these interdependencies between sectors, the decrease of regulating services due to deforestation caused a total impact of KSh 5.8 billion in 2010. This means that the cost of limiting regulating ecosystem services as a production factor for the economy was all in all 4.2 times higher than the actual cash revenue of KSh 1.3 billion (UNEP, 2012b).

Internalizing the benefits of sustainable management of forests: the role of a forestry account for Kenya - Kenya's initiative to build a forestry account had as its main objective to capture the value addition to forest products through the manufacturing sector; the provision of goods (timber and non-timber) to the subsistence economy (also referred to as the non-monetary economy); and the supply of a set of cultural services to residents of and visitors to Kenya; and the supply of a set of ecosystem services that regulate ecological processes.

Preliminary assessment concludes that the value of the Forestry sector value chain to the economy of Kenya is at least three times larger than currently estimated by Kenya National Bureau of Statistics (KNBS). KNBS estimates a contribution of forestry to GDP of Ksh 15.333 million in 2005. The sector provides the non-monetary economy with at least Ksh 6.988 million per year worth of raw materials. This transaction is not accounted for in the national accounts. Similarly, the charcoal manufacturing sector, attributing and estimated Ksh 12.460 million per year to GDP, is not accounted for in the national accounts (Mutimba, 2005). The national GDP of Kenya is therefore understated by approximately 1.4%.

The preliminary estimate of the partial contribution of forestry in Kenya to the economy of Kenya, is 3.6% per year (this value is most likely underestimated as, for example, the tourism sector and carbon sequestration service were not considered).

Footnote

[7] *Kenya shilling*

Box 2.2: Case Study: Good practices in Poverty Reduction Strategies Papers – Kenya's Forest Accounts *continued*

Policy Implications and Recommendations - Some key policy recommendations were made including:

- Reducing the loss of regulating ecosystem services as the cost of not doing so is 4.2 times higher than the actual cash revenue of KShs 1.3 billion from deforestation.
- Ensuring that Kenya has in place a fully functioning forest resource account in order to fully capture the various benefits provided by the forest.
- Encouraging investment in the forestry sector in order to increase efficiency in production, especially in sawn timber and charcoal production. The increased use of micro-credit schemes from the government, for instance, would decrease the size of the informal sector, slow down unsustainable resource depletion and would create job opportunities, particularly in rural areas.
- Adequate regeneration after harvest and an increased forest plantation growth in the long term, together with better coordination of regulating institutions, producers and consumers of forest products (Sedjo, 2005).
- Mainstreaming the use of instruments and incentives such as payment for ecosystem services, trading and insurance schemes.

Table 2.1. Summary of poverty reduction evidence for conservation mechanisms. Adapted from Roe (2010b).

Mechanism	Poverty Reduction Benefits	Biodiversity or Biomass Important for Poverty Reduction
Non-timber forest products	Low	Biomass
Community timber enterprises	Medium	Biomass
Payment for Ecosystem Services	Low	Biomass
Nature-based tourism	High	Biodiversity
Fish Spillover	High	Biomass
Mangroves	Medium	Biomass
Protected area jobs	Low	Biodiversity
Agroforestry	Medium	Biomass
Grasslands	Low	Both
Agrobiodiversity	Medium	Biodiversity

The Iwokrama Canopy Walkway (www.iwokramacanopy walkway.com), in Guyana, is a good example of eco-tourism project, where the private sector, a conservation Non-Governmental Organisation (NGO) and the local communities form a financially successful partnership that share the benefits. This projects shows how ecotourism can provide tangible benefits to local communities (SCBD, 2010).

2.1.1.iii Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA)

The CBD requires Parties to apply Environmental Impact Assessment (EIA) to projects and Strategic Environmental Assessment (SEA) to programmes, plans and policies with potential adverse impacts on biodiversity. Concurrently, national (and usually regional/state/provincial) legal requirements for both EIA and SEA require the environmental assessment of positive and negative impacts on biodiversity.

Despite legal requirements, practice is still insufficient. Gontier *et al.* (2006) reviewed 38 EIAs published between 1999 and 2003 addressing road and railway projects from 4 European countries to assess the state of integration of biodiversity issues and the use of prediction methods to quantify the impacts of the project on biodiversity. The term biodiversity was seldom used in the assessments, only in seven reports the term biodiversity was found in the section dealing with the aims of the assessment, but it was not found anywhere else in the report. The assessments were restricted to protected species and protected areas and rarely considered the ecosystem level. Most of the assessments were descriptive in nature and did not consider quantification and methods for predictions of impacts.

More recently, Seebun *et al.* (2011) reviewed 50 environmental assessment reports (EIAs and SEAs) from both developed and developing countries, and from infrastructure, land use, mining, tourism, transport and energy, to assess their effectiveness in analysing the impacts of the proposed developments on biodiversity and ecosystem services as well as to determine if the assessment had impacts on the subsequent decision making and development planning process (Seebun *et al*., 2011). The reports were selected to cover two time periods in a similar way, 2002-2007 and 2008-2011. The analysis concluded that the majority of the assessments include considerations on biodiversity, and that the trigger for its inclusion was legislation requirements. About one third of the EIAs and SEAs have been able to influence the decision and development in order to minimise the impact on biodiversity and ecosystem services. The majority of the reports applied an ecosystems perspective and not just impacts on flora and fauna. Also the analysis showed that in the more recent reports biodiversity was considered to a greater extent than in older ones probably due to the release of guidelines by international agencies, as well as the guidance published by the CBD.

Similarly, Monteiro and Partidário (2013) shared results of an international review, on the consideration and incorporation of biodiversity and ecosystem services in SEA based on 14 environmental reports of SEA carried out in several countries, and 44 environmental reports of SEA carried out in Portugal alone. All cases collected were published between 2005 and 2011 and represent different typologies (protected areas, spatial plans, energy, waste, water, coastal zones and transport plans). For each case, the analysis was conducted according to: the (1) approach to biodiversity and ecosystem services (as a critical decision factor, assessment criteria or indicator for analysis); (2) consideration of biodiversity and ecosystem services in the plan's objectives; (3) Mitigation measures; and (4) Monitoring guidelines and respective indicators. Results achieved show that biodiversity and natural heritage issues were vastly considered (in 84% of reports internationally and in 77% of environmental reports in Portugal), however the benefits of ecosystem services were only addressed in 8% of international environmental reports and in 27% of environmental reports in Portugal. Practice shows that although both EIAs and SEAs were able to incorporate biodiversity aspects in the development process, SEAs by their longer-term nature can better address the cumulative impact usually associated with biodiversity and ecosystems (Partidario & Gomes, 2013; Seebun *et al*., 2011).

2.1.1.iv Environmental-Economic Accounts

The objective of establishing environmental-economic accounting is to obtain "a better measurement of the crucial role of the environment as a source of natural capital and as a sink of by-products generated during the production of man-made capital and other human activities" (UN, 1992a, 1992b).

In 2007, the United Nations Statistics Division (UNSD) carried out a global assessment of the implementation of environmental statistics and environmental-economic accounting (UNSD, 2007). From the 100 respondent countries (52% of total), 90% had an environmental statistics programme and 50% had an environmental-economic accounting programme. From the countries that did not have an environmental-economic accounting programme in place, approximately half said that they are planning in the near future to start compiling these accounts. Europe is the region with higher implementation levels (70% of the respondent countries), whereas Latin America and the Caribbean and Western Asia are the regions with lower implementation levels of environmental-economic accounts (22% and 30%, respectively) (Figure 2.5).

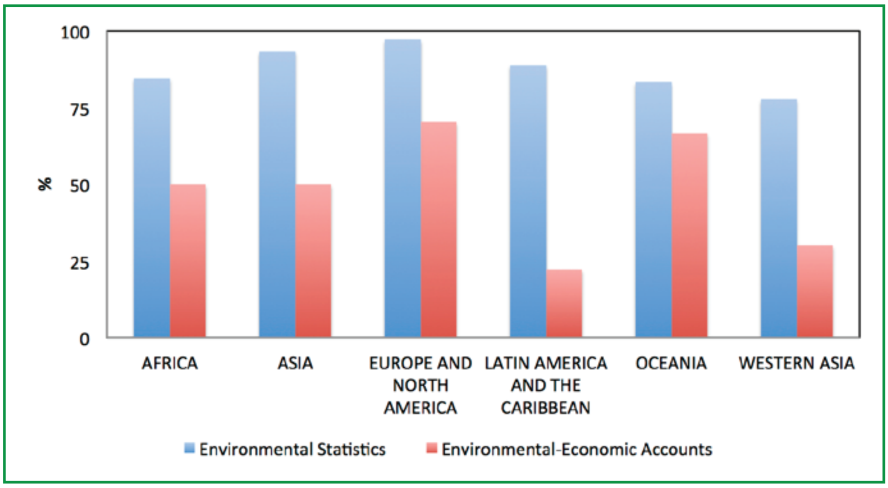


Figure 2.5. Geographical breakdown of countries with environmental statistics and environmental-economic accounts programmes in place, in 2006 (in %).
Source: UNSD (2007)

Developed and developing countries exhibit different accounting priorities, probably due to different environmental policies and concerns (Edens *et al.*, 2011). Most developed countries compile energy and emissions accounts, environmental protection expenditure accounts and material flow/waste accounts. The emphasis given to energy and emissions accounts reflects specific environmental legislation. For example, in European countries the existence of legislation on environmental accounting makes obligatory for Member States to report air emissions, material flows and environmental taxes (Pasquier *et al.*, 2007). Most developing countries compile water accounts, energy and emissions accounts, mineral assets accounts and forest accounts.

When questioned about future expansion of environmental accounts, 88% of the respondent countries were planning to do it; for developed countries the priority for expansion were the material flow/waste accounts (68% of the countries), whereas for developing countries the priority were the energy and emissions accounts. Interestingly, only 23% of developed countries planned to expand their programmes to account for land and ecosystems, while 64% of the developing countries plan to do so.

In 2012, the United Nations Statistical Commission (the apex entity of the global statistical system) adopted the System of Environmental-Economic Accounting – Central Framework (SEEA-CF) (UN, 2012) as an international statistical standard for environmental-economic accounting. SEEA-CF organises and reconciles basic economic and environmental statistics to obtain time-series of comparable and coherent datasets, applying the accounting concepts, structures, rules and principles of the System of National Accounts (SNA). SEEA-CF includes asset accounts for mineral and energy resources, land, soil resources, timber resources, aquatic resources and water resources; in terms of ecosystem services it focus on the provisioning services for which a market price exist (Brouwer *et al.*, 2013).

Acknowledging the relevance of the linkages between ecosystems and the economic activity, the United Nations Statistical Commission supported the development of the SEEA-Experimental Ecosystem Accounting (SEEA-EEA) (UN, 2013). Ecosystem accounting measures not only the contribution of ecosystem to the economy, but also the role of ecosystems in the supply of other benefits to humans (that are commonly not accounted for; UN, 2013). In SEEA-CF, environmental assets are measured from an individual perspective, for example, timber resources or water resources, in SEEA-EEA environmental assets are measured from an ecosystems perspective, assessing how *"different individual environmental assets interact as part of natural processes (…) to provide a range of services for economic and other human activity"* (UN, 2013).

The development of environmental-economic accounting is deeply related with economic valuation and the several initiatives mentioned in Section 1.a.i. will also contribute to the development of such accounts. Other initiatives have a focus on developing environmental-economic accounts. In 2010, the World Bank initiated the WAVES partnership (Wealth Accounting and the Valuation of Ecosystem Services), whose main objective is *"to promote sustainable development by ensuring that natural resources are mainstreamed in development planning and national economic accounts"* (WAVES, 2013). To achieve this, WAVES not only helps countries to adopt and implement environmental-economic accounts, but also intends to develop an ecosystem accounting methodology. Currently, WAVES is supporting eight countries to implement natural capital accounts. Botswana, Colombia, Costa Rica, Madagascar, and the Philippines were the first countries under the WAVES partnership (Table 2.2). In 2013, Guatemala, Indonesia and Rwanda have also joined. WAVES Policy and Technical Experts Committee (PTEC) aims to develop internationally agreed guidelines for ecosystem accounting by building on experiences, PTEC is currently working on the development of guidelines for coastal and marine ecosystem accounting, specially focusing on the inclusion of regulating services.

Table 2.2. Accounts implemented by WAVES partners and progress.

COUNTRY	ACCOUNTS	PROGRESS
Botswana	Water, land and ecosystems, mineral and energy and macroeconomic indicators of sustainable development	Detailed water accounts for 2010-11 and 2011-12.
Colombia	Water and forests	Water and forest accounts developed.
Costa Rica	Water and forests	Established technical working groups for both the water and forest accounts
Guatemala	No information	Water, forests, energy and emissions, waste, fisheries, subsoil, environmental costs accounts.
Indonesia	No information	-
Madagascar	Mining, water and forests/protected areas and coastal	-
Philippines	Water, mineral, mangroves, land and ecosystem (at two identified sites) and macroeconomic Indicators of Sustainable Development.	Land cover change matrixes (for the two identified sites). Water use supply and use table.
Rwanda	No information	-

At European level, the EU Regulation 691/2011 requires member states to develop environmental-economic accounts (European Parliament, 2011), air emissions, material flow and environmental accounts are mandatory, ecosystem services accounts are listed among the potential new accounts to be added in next revision processes. The development of ecosystem accounts by member states can build on the experimental ecosystem accounts developed by the European Environmental Agency (EEA, 2011).

In the UK, the commitment to fully include natural capital into national accounts was stated in the Natural Environment White Paper (UK Government, 2011). In response to this, the UK Office for National Statistics in collaboration with the Department for Environment, Food and Rural Affairs were tasked to develop experimental ecosystem accounts. A pilot study on forestry accounts, followed by land use and cover accounts is planned (Khan, 2011).

Other countries with concrete developments in ecosystem accounts are Canada and Australia. In Canada, the Measuring Ecosystem Goods and Services (MEGS) project, initiated in 2011, aims at creating pilot ecosystem accounts. Progress has already been achieved in the biophysical characterisation of ecosystems. The MEGS geo-database is gathering relevant information for ecosystem accounting, for example land cover and land uses changes and changes in the provision of ecosystem services. MEGS also applied and developed new ecosystem accounting concepts, it refined land cover ecosystem unit described in SEEA-EEA by adding the dimensions of ruggedness and terrain elevation (Statistics Canada, 2013).

In Australia, land cover or land use accounts in monetary and physical terms have been produced for three regions (the Great Barrier Reef region, the Murray-Darling Basin and the state of Victoria), the accounts for ecosystem condition are still in early developments but progressing (ABS, 2013). Experimental biodiversity accounts have been developed for the terrestrial environment adjacent to the Great Barrier Reef, although not all the species known to occur in the region were accounted for, this is a great development (Bond *et al*., 2013). These accounts include species grouped under animals, plants, fungi and Protista it also includes some insects; and distinguish by whether they are introduced or native, rare or endangered, protected or not by state laws (Bond *et al*., 2013). In Victoria, biodiversity accounts were also developed and they consisted of a threat status accounts for birds detailing changes that have occurred in two different points in time (Bond *et al*., 2013).

2.1.1.v National Reports to the Convention on Biological Diversity

The CBD requests, through Article 26, that all parties *"present to the Conference of the Parties, reports on measures which it has taken for the implementation of the provisions of this Convention and their effectiveness in meeting the objectives of this Convention"*. These national reports are important monitoring tools.

Parties have submitted their fourth national report and are in process of delivering the fifth. The assessment of these reports enables an understanding of the state of biodiversity mainstreaming. Mainstreaming biodiversity refers to the extent to which biodiversity is integrated into national policies and strategies (for example, national development or poverty eradication), the extent to which it is taken into account in several economic sectors (like, agriculture, tourism, education, etc.), and the extent to which it is taken into account in local planning.

Of the 193 CBD Parties, 91% have delivered the fourth national report. An analysis of these reports (SCBD, 2010) revealed that 86% of the Parties are taking concrete measures towards biodiversity mainstreaming, and 80% of the Parties recognise the value of biodiversity, as they indicate that biodiversity is important for the human well-being of their country. The integration of biodiversity in national-level, sectoral and cross-sectoral strategies, plans and programmes has been reported by 72% of the Parties, particularly into poverty reduction and sustainable development strategies. The main sectors were actions are in place towards mainstreaming are forestry and agriculture. Only 30% of the Parties report the integration of biodiversity into sub-national or local plans. Nevertheless, 91% of the Parties have mechanisms in place for environmental impact assessment, and 38% strategic environmental impact assessment. Despite this progress, the majority of the Parties (77%) acknowledge that they still have limited biodiversity mainstreaming, which impairs concerted national actions to meet the objectives of the Convention. The main reasons identified were fragmented decision making and limited communication between all stakeholders. Regarding the obstacles to the mobilization of resources towards efforts to promote the conservation of biodiversity and its sustainable use 61% of the Parties identified the lack of economic valuations of biodiversity.

In their fourth national reports almost no details were found regarding the effects of the implemented changes in national, local or sectoral policies in biodiversity.

2.1.1.vi Country actions and commitments[5]

Generally, most countries have established national targets, or equivalent instruments, in their NBSAPs which correspond to Aichi Biodiversity Target 2 (high). These targets are generally aligned with the main direction of the Aichi Biodiversity Target (high). If these targets are fulfilled they will make a significant contribution towards the attainment of this Aichi Biodiversity Target by moving biodiversity issues from the periphery more towards the centre of decision making.

Among those that have provided updated NBSAPs to date, few countries note in their targets that biodiversity values will be integrated into national development or poverty reduction strategies (high). An example, which is counter to this trend, is Brazil that has set a national target addressing the integration of biodiversity into poverty eradication and inequality reduction strategies.

The targets contained in the NBSAPs tend to focus on issues related to the integration of biodiversity in planning processes and/or government policies (medium) and in several cases are supported by a range of enabling actions. For example, Malta has set a target to recognise and integrate the values of biodiversity and ecosystem services in planning processes and decision making while Belgium has established an objective to improve the integration of biodiversity into relevant sectoral policies.

Overall, there appears to be less of an emphasis on integrating the values of biodiversity into national accounting and reporting systems with only a few countries explicitly addressing this issue (high).

A number of countries have also set targets that address issues, which though not directly addressed in the Aichi Biodiversity Target, would make significant contributions towards it. For example several countries, including Australia, Ireland, Myanmar and Tuvalu, have established targets, which relate to reforming legislation and policies in order to facilitate the conservation and sustainable use of biodiversity. Similarly, Spain, in its national biodiversity strategy and action plan, has set an objective related to reducing the negative impact of public procurement on biodiversity.

2.1.2 Projecting forward to 2020

In an effort to understand how current status and trends will possibly evolve until 2020, and what will that mean to the achievement of Target 2, we focus on three aspects. First we try to understand the evolution of biodiversity and ecosystem services valuation exercises, then the evolution of ecosystem accounts; finally we discuss the integration of biodiversity into national and local development strategies.

The number of economic valuation studies on ecosystem services has greatly increased and this tendency is likely to continue (Figure 2.6). The number of initiatives dealing the valuation of biodiversity and ecosystem services is also increasing. The recommendations of the integration of biodiversity and ecosystem services values into NBSAPs may contribute to a continuing increase of the interest on the economic valuation of biodiversity and ecosystem services.

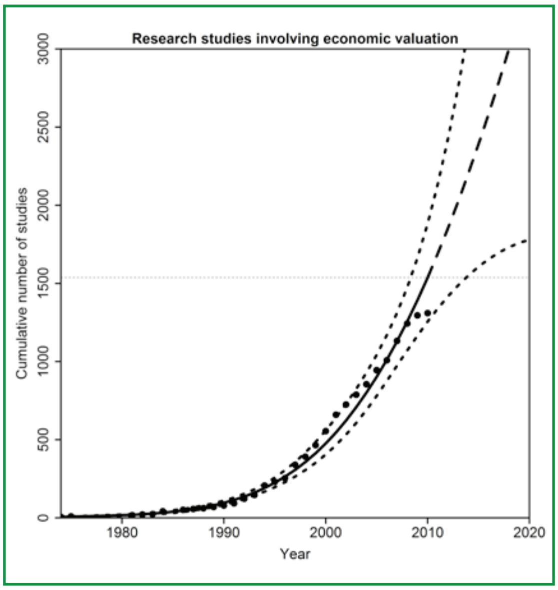


Figure 2.6. Statistical extrapolation of the number of research studies involving economic valuation to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Source: Tittensor *et al.*, (2014).

Footnote

[5]This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.

However, the number of studies on the non-economic values of biodiversity and ecosystem services is not as high (Martín-López *et al*., 2012; Nieto-Romero *et al*., 2014). In order to fully understand the contribution of biodiversity and ecosystems to human well-being it is important that knowledge on the non-economic value of biodiversity develops (Abson *et al*., 2014; Martín-López *et al*., 2012).

The adoption of the SEEA-CF as an international statistical standard was an important milestone in environmental accounting by providing common concepts and guidelines. As seen before, different countries seek to extract different types of information from environmental-economic accounts. Taking this into account, the United Nations Committee of Experts on Environmental-Economic Accounting (UNCEEA) developed a strategy for SEEA implementation that follows a flexible and modular approach. Countries do not have to implement all accounts at the same time, instead implementation of SEEA is expected to occur incrementally and following mainly policy demands and national requirements (UNCEEA, 2013). Also, countries can and should leverage on existing environmental statistics, hence minimising the effort of implementing environmental-economic accounts (UNCEEA, 2013; WAVES, 2012). Countries should analyse the quality of already existing data, as well as data needs and their sources. With this, countries are able to draft integrated action plans for the implementation of environmental-economic accounts. It is likely that this modular and flexible approach shortens the implementation time of SEEA, and that the majority of countries will have some kind of environmental-economic accounts by 2020.

Progress is expected to occur until 2020 in both understanding the values of biodiversity and ecosystem services and compiling them into harmonised and consistent datasets. Notwithstanding, technical challenges persist that need to be addressed. The development of agreed common classifications, for example, for land use or ecosystem services, is mandatory to further develop the area of valuation and accounting (UNCEEA, 2013; UNSD, 2013; see also Section 4). Another challenge concerns the intrinsic multidisciplinarity of biodiversity and ecosystem services valuation exercises and how the several disciplines interact (or should interact) to provide the best approximation possible (for example, EPA, 2009; Liu *et al*., 2010; Reyers *et al*., 2013; Ring *et al*., 2010; Russell *et al*., 2013; TEEB, 2010).

Mainstreaming biodiversity and ecosystem services values in least developed countries (LDCs) requires determining the values of biodiversity and ecosystems in these countries. Christie *et al.* (2012) reviewed, using the EVRI database and the ISI Web of Knowledge (WoK), the extent to which monetary and non-monetary valuation techniques are currently used in LDCs and the main challenges associated. There results from EVRI showed that only 11.6% of the studies were conducted in LDCs. The results from WoK allowed the identification of main valuation techniques used. From the 284 papers analysed, 183 used monetary techniques, the others used non-monetary techniques. To assess if expertise exists on LDCs countries to perform valuation exercises, Christie *et al.* (2012) analysed the authorship of the studies. They concluded that while some expertise exists, the majority of studies were led by authors from developed countries. The main challenges identified in applying valuation exercises in LDCs were manifold (Christie *et al*., 2012). For example, the low level of literacy by local people and the lack of ability to express complex scenarios in local languages, and the fact that people in LDCs have a very deep but personal understanding of their natural environment. Another challenge is the prevalence of subsistence economies. This does not make the understanding of market prices straightforward, as money is not embedded in the daily routine. The lack of scientific knowledge, the lack of local research capacity to undertake valuation exercises is another challenge, as is the lack of guidelines that take into account the differences between developed and developing countries are also challenges to be taken into consideration. Two essential steps are necessary to overcome the challenges. First, it is necessary to build local capacity, second it is necessary to incorporate participatory, deliberative and action research methods in LDCs to improve the robustness of the valuation exercise (Christie *et al*., 2012).

Evidence shows an increasing concern and incorporation of biodiversity and ecosystem services issues into SEA and EIA. Much is due not only to the legal requirements and guidance released through the CBD but also to TEEB initiative and multiple resulting projects. The initiative of the World Business Council on Sustainable Development (WBCSD) on biodiversity and ecosystem services and the guidance released addressing corporations, is also encouraging. There are many more discussions and initiatives than 10 years ago, when the MA results were released, thus, the issue is definitely on the agenda. But what has been happening so far is still limited when looking at the 2020 target. The task is demanding. There is a need for action, beyond discourses. And action does not happen only with legal requirements and guidance, even though these are essential ingredients. In order to meet the 2020 target we need changes of focus, changes of priorities by the several sectors of the economy and society. That target will not be achieved only by control and pressure imposed by legal requirements. The decrease in funds for environmental impact assessments raises concerns with the applicability of such tools in the short-term future (Figure 2.7).

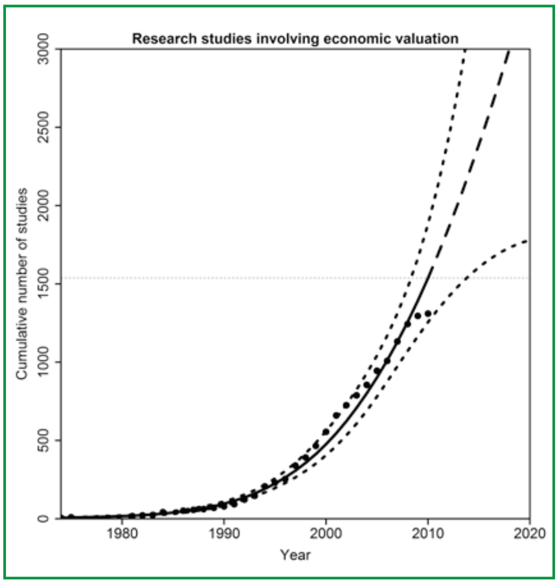


Figure 2.7. Statistical extrapolation of funds for environmental impact assessment to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Source: Bateman *et al.*, (2013).

The integration of biodiversity in EIA and SEA has been evolving from conservation to an integrated approach. For Slootweg *et al.* (2006), biodiversity should be seen as a provider of goods and services set through ecosystem services in EIA and SEA contexts. The concept, the purpose and the rationale of biodiversity and ecosystem services need to be built into the driving forces that push economic development, so that both EIA and SEA, but in particular the latter, given its earlier intervention and wider scope, can play its role of encouraging better practices that make good environmental, social and economic sense. It is necessary to change minds in relation to the role played by EIA and SEA and move beyond the typically control role of both EIA and SEA that result in limited mitigation measures. The final outcome of a control role, rather than changing courses of practice, is a contribution to continuous small-scale resources depletion, through apparently insignificant, but deleterious, cumulative processes.

2.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

2.2.1 Actions

To reach the Aichi Biodiversity Target, it is necessary that governments are aware of the value of biodiversity and ecosystem services and that this value is effectively reflected in decisions and policies. A first step concerns the assessment at a national level of biodiversity and ecosystems and the services they provide (Brouwer *et al.*, 2013; Rode *et al.*, 2012). Such an assessment is resource intensive and probably not all countries are able to be perform it (Brouwer *et al.*, 2013), nevertheless it can be done incrementally starting with ecosystems that are more important or more easy to assess (Rode *et al.*, 2012). The development of environmental and ecosystem accounts can start by building on already available statistics and by promoting collaboration between national agencies, government ministries and statistical offices (Brouwer *et al.*, 2013). Another important aspect that needs urgent attention is the development of an agreement upon valuation techniques that is applied in the context of environmental accounts (Hein, 2012; UN, 2012; UNSD, 2013).

These exercises provide the basis for mainstreaming biodiversity in development strategies and planning processes. However, effective mainstreaming is only possible if the values of biodiversity and ecosystem services are embedded in institutions and in people, this requires change (Daily *et al.*, 2009). A successful way of bringing institutional change is by linking research to policy via a pilot project that includes incentives for the protection of biodiversity and ecosystem services by explicitly recognising their values (Daily *et al.*, 2009). To promote social change it is necessary that a broad discussion between stakeholders and understanding their perception of nature, and promote participatory management that reconciles traditional knowledge and environmental management (Daily *et al.*, 2009; Reed, 2008).

At the global level, it is important that developed countries assist developing countries in building the technical knowledge necessary for biodiversity and ecosystem services valuation and accounting (Christie *et al.*, 2012; SCBD, 2012). Global initiatives like WAVES are examples of actions that can be taken to address this challenge.

The High-Level Panel on Global Assessment of Resources for implementing the Strategic Plan for Biodiversity 2011-2020, prepared a report identifying the costs of carrying out TEEB-like studies to assess biodiversity values, promoting actions to raise awareness of the importance of biodiversity amongst policy makers and developing national natural capital accounts (SCBD, 2012). These actions would represent, for each country, a total investment between US$450 and US$600 million during the 2013-2015 period (SCBD, 2012). The maintenance of the progresses achieved would represent an annual expenditure of US$70 to US$130 million.

The total resources needed to achieve Target 2 by 2020 are estimated to be between US$800 million and US$1.3 billion over the 2013–2020 period. Despite being amongst the least financial resource intensive Targets, it should be noted that the achievement of Target 2 will have positive consequences on all other Targets, and therefore investing in achieving it can help reduce the costs of achieving other Targets.

There is a great potential in EIA and SEA to assist better environmental and sustainable decision-making. And that means not only in project development decision-making but also in thinking through the best strategic ways to encourage, and conduct, development. EIA has been described as the process of identifying, predicting, evaluating and mitigating the biophysical, social and other relevant effects of development proposals prior to major decisions being taken and commitments made (IAIA/EIA, 1999). On the other hand SEA emerged as it became obvious that before development could be materialised in actual projects, concepts, intentions and directions for development would be considered, thus affecting subsequent projects. With SEA emphasis is placed in influencing the strategic development concept ahead of plans, programmes or policies formulation to enable the integration of relevant environment and sustainability issues before options are closed and commitments are laid out in plans or programmes (Partidário, 2012).

From am EIA and SEA perspective, fresh points of view on the potential use of EIA and SEA, and the benefits they can bring, are needed. How quickly and efficiently can that change happen is one of the major uncertainties in this process. EIA and SEA need to be seen, and used, to create opportunities. Ecosystem approaches and ecosystem services offer good reasons to enable constructive approaches towards adaptive management. Reconciling the views of public authorities, including the environmental administration, stakeholders and developers is crucial to that purpose. However these groups have learned to adopt opposite positions and got used to use conflicting arguments, even where constructive approaches could be an obvious solution. Overcoming this tension is urgent but also uncertain in relation to its timing and process for success. Biodiversity long-term objectives and principles should act as a driver in the effort towards fair and equitable sharing of biodiversity benefits for human beings, encompassing both the commercial use of natural resources, as well as the fair and legitimate traditionally access to resources. This harmonisation, and how this can happen, is also the source of major uncertainty.

Safeguarding livelihoods must be a major driver in the application of EIA and SEA as instruments to safeguard biodiversity and ecosystem services. Partidário and Slootweg (2012) showed that EIA and SEA are key impact assessment instruments not yet used to their full potential. Both have a major role in bridging economic, social and biophysical dimensions to assess future development opportunities. But for these instruments to be more useful in this regard it is urgent to change gears, and perspectives, on how biodiversity and ecosystem services are being considered in the policy arena. Partidário and Slootweg (2012) proposed that EIA and SEA could proactively identify the impacts of human actions on ecosystems and biodiversity, and advance the necessary measures in order to avoid, or mitigate, the expected negative consequences. According to the same authors, EIA and SEA can help at local and regional levels in three ways: 1) improve projects quality by adapting it to existing biodiversity values to favor projects quality and attractiveness; 2) improve the attractiveness of lands because of existing natural assets and enable compatible uses avoiding conflicts; 3) identify development opportunities created by existing biodiversity values and ecosystem services.

Finally, in order to integrate the diversity of value-types in decision making processes at local scale, place-based valuation (economic and non-economic) research is essential. Place-based research about the values of biodiversity and ecosystem services is actually needed in order to show the diversity of human-nature relationships (Russell *et al.*, 2013). However, many challenges arise in place-based valuation research, such as: (i) to consider the broad spectrum of values (besides economic value) given by different types of stakeholders (see, Martín-López *et al.*, 2012); (ii) to respect and to include different knowledge systems because while scientific and technical knowledge is related to economic values, local or traditional ecological knowledge (see chapter 18) is basically related to cultural and spiritual values; (iii) to design a methodological framework able to integrate (but not reduce) the different value-types attached to biodiversity and ecosystem services by different stakeholders.

2.2.2 Costs and benefits

Valuation allows, among other things, to understand the contribution of biodiversity and ecosystem services to human well-being, making easier to understand what is at risk. For example, economic valuation allowed to determine that the minimum cumulative losses of not meeting the 2010 biodiversity target globally were equivalent to 7% of the global GDP in 2050 (Bakkes *et al.*, 2008).

Economic valuation also allows understanding the trade-offs, costs and benefits of different decisions and policies targeted to biodiversity and ecosystem services (see Box 2.2). For example, in the UK National Ecosystem Assessment (UKNEA, 2011), the provision of ecosystems and their monetary value was analysed under 6 different future land-use scenarios, up until 2060 (each analysed for high and low emissions scenarios). The results show that, if only market values are taking into consideration for policy making (in the UK NEA, agricultural output), great monetary losses are likely to occur (in respect to other ecosystem services and biodiversity; Bateman *et al.*, 2013, see Box 2.1).

A recent study analysed the benefits of biodiversity for poverty alleviation, at the global scale, by assessing the flows of ecosystem services provided to people by priority habitats for terrestrial conservation (Turner *et al.*, 2012). The benefits consisted of direct benefits but also payments for ecosystem services to those stewarding natural habitats. The aggregate benefits are three times higher than the estimated opportunity costs, and exceed US$1 per person per day for 331 million of the world's poorest populations. The top 25% of conservation priority were estimated to provide approximately 57% of these benefits. The results from this study show that win-win synergies between conservation and poverty alleviation are possible.

Another example is a recent study in the EU that estimated the benefits of the Natura 2000 network of protected areas (Ten Brink *et al.*, 2011). Per year, the Natura 2000 Network provides benefits worth between €200 and €300 billion (2% to 3% of EU's GDP), in comparison the annual costs of the implementation and management of the network are less than €6 billion.

2.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

According to Target 2, the values of biodiversity should have been integrated into national accounting and reporting systems by 2020. Effective accounting and reporting systems will allow decision makers to know about the loss or degradation of biodiversity components is known in time for actions to be taken to conserve or restore them, or perhaps to compensate for their loss. Therefore implementing such systems at national scale is especially important in order for governments to be able to develop and enact appropriate policies. In England, a Natural Capital Committee was formed in 2012 to advise the Government on natural capital[6]. One specific task was to provide advice on when, where and how natural assets were being used unsustainably, to the extent that the benefits to people are at risk. The Natural Capital Committee approach to this was to develop a preliminary risk register for natural capital in England, based on the following steps:

1. **Define a set of natural capital *assets*** (e.g. species, ecological communities, water, clean air etc.)
2. **Define a set of *benefits* that people derive from these assets collectively or individually** (e.g. food, energy, clean water, hazard protection, wildlife conservation etc.)
3. **Identify a *target* level that is a 'required' level of benefits from natural capital assets** (e.g. the Good Ecological Status required by the EU Water Framework Directive, the species and habitat targets that the UK is committed to under the EU Wild Birds and Habitats Directive; the greenhouse gas budgets in the UK Climate Change Act, etc.)
4. **Identify the status and trend of the current level of benefits relative to the target and use this to gauge the level of risk associated with each benefit.** Very high risk is identified where the benefits are below target and the trend is deteriorating. High risk occurs when the status is close to target and the trend is stable or close to it. Low risk occurs when the status is at or above target level and there is little indication of deterioration.

Each benefit was assessed in each of the eight major habitats mapped in the UK according to the UKNEA (2011). The results, recorded in a preliminary risk register for natural capital led to the following benefits being considered to be high risk because of deterioration in natural capital assets (NCA, 2014):

- Clean water from mountains, moors and heaths, due to the quality of those habitats;
- Clean water from the current extent and projected growth of urban areas leading to a deterioration in freshwater, soils and natural water purification processes;
- Wildlife in many land use categories (semi-natural grasslands, enclosed farmland and freshwaters) due to poor quality habitats and unfavourable spatial configurations; and,
- Equable climate, essentially England's contribution to carbon storage, is at risk from the degraded condition of mountains, moors and heaths which have the potential for much greater carbon storage.

Footnote

[6] *Natural capital describes all the elements of nature that directly or indirectly produce value for people.*

The risk register is preliminary because much information does not exist in a form that can be used for this kind of analysis. However, the committee concluded that taking a risk register approach could provide an efficient mean to focus biodiversity and environmental data gathering efforts, and provide information relevant to decision-making. Restoration, recovery or maintenance of natural assets can sometimes be achieved relatively easily and at low cost with measurable benefits (e.g. restoring urban green space). Some kinds of degradation are very difficult, costly or slow to restore (e.g. rebuilding marine fisheries, restoring ancient woodlands, reversing atmospheric pollution). However even in these difficult cases the benefits may be great (NCA, 2014). Decision makers need to understand the risks of natural capital degradation in order to make better decisions about when and where to direct resources to maintain or restore critical assets.

If the values of biodiversity and ecosystem services are not taken into account in decision and policy making current trends are likely to prevail, which will probably mean extinction of more species and degradation of ecosystems. Achieving Target 2 provides the information necessary to develop for example, risk register and effectively integrate biodiversity and ecosystems in development plans both at the local and at the national level.

2.4 UNCERTAINTIES

In the integration of biodiversity values in decision making there are several sources of uncertainty steaming from both the ecological and economic sciences that should be taken into account (EPA, 2009; TEEB, 2010).

Despite the progresses made in establishing the link between biodiversity and supply of ecosystems services (Cardinale *et al*., 2012; TEEB, 2010), there is still no comprehensive understanding of these relationships. Two ecological sources of uncertainty in economic valuation exercises are the potential trade-offs and linkages among different ecosystem services and the other one concerns the stochastic and random nature of ecosystem responses to change (EPA, 2009; Ring *et al*., 2010).

Most studies focused on the benefits arising from the supply of one ecosystem service. However, ecosystems function as a whole and different services are bundled together. Addressing only one in valuation exercises and decision-making can have detrimental consequences for other important services. For example, the European Common Agricultural Policy promoted the increase food production (provisioning services), often achieved through increase in fertilisers use, which will have a negative impact on supply of clean water (regulating services), a goal of the European Water Framework Directive (Hauck *et al*., 2013). Not only trade-offs exist, for example, positive relationships may exist between the maintenance of soil quality and primary production that will then positively affect climate regulation. There is still not sufficient knowledge on the relationships between different ecosystems as well as their feedback mechanisms (Carpenter *et al*., 2009). The non-consideration of these relationships adds great uncertainty in economic valuation exercises.

Most economic valuation studies are based on marginal changes on the provision of ecosystem services assuming that ecosystems are in a stable condition (Limburg *et al*., 2002; TEEB, 2010). However, little is known about the stability of ecosystems and their response to change. A very disturbed ecosystem can reach a critical threshold that triggers a structural change (Barnosky *et al*., 2012; Leadley *et al*., 2010). At that point the marginality assumption no longer holds and the estimation of reliable economic values very difficult (Ring *et al*., 2010; TEEB, 2010).

The uncertainties arising from economic sciences to valuation of biodiversity and ecosystem services are manifold. TEEB (2010) provides a comprehensive overview of the several sources of uncertainty in the different methods used for valuation of biodiversity and ecosystems services (supply uncertainty, preference uncertainty and technical uncertainty) and provide best practices solutions to deal with them.

Benefit-transfer is the procedure of estimating the value of an ecosystem service by transferring an existing valuation estimate from a similar ecosystem (TEEB, 2010), it is normally seen as an approach to overcome the lack of specific information in a timely and inexpensive manner. There are several challenges associated with this approach that add errors and uncertainties to valuation exercises and should be considered when decisions are made using this type of approach. Errors can steam from the primary valuation estimates, but also from differences (in population characteristics or environmental characteristics) between the policy site (site where the first value was obtained) to the study site, these are normally referred to as generalisation errors, also the existence of publication selection bias can be a source of error when applying benefit-transfer (Rosenberger & Stanley, 2006; TEEB, 2010). Another important issue concerns the different spatial scales at which ecosystem services are supplied and demand both in the policy and study site (TEEB, 2010), such differences bear consequences regarding estimated values.

In the sustainable development debate, a controversial issue concerns the choice of discount rates. A discount rate is used to inform how is it worth investing today in conservation of the environment considering future benefits. The discount rate reflects the responsibility of the present generation to the future one. It has been suggested that negative discount rates should be used in valuing natural capital for future time periods (Blignaut & Aronson, 2008; Dasgupta & Maskin, 2005; Kumar *et al.*, 2013). The choice of the discount rate will have a high influence in the final outcome. For example, a four per cent discount rate implies that biodiversity loss 50 years from now will be valued at only one-seventh of the same amount of biodiversity loss today (Kumar *et al.*, 2013). High discount rates typically lead to long-term degradation of biodiversity and ecosystems. Therefore, the discount rate is *per se* a major source of uncertainty. Moreover, there are no purely economic guidelines for choosing a discount rate. The choice is a matter of ethics, and our responsibility of preserving biodiversity and ecosystems to provide well-being to future generations (Kumar *et al.*, 2013).

Challenges in addressing biodiversity in EIA and SEA are related with the long-term and cumulative nature of the effects, the complexity of the cause-effect relationships and uncertainties (EU, 2013a; 2013b). Suggestions to overcome these challenges are for example, to avoid static analysis and consider the trends on biodiversity with and without the proposed project/policy, consider environmental limits, work with worst-case and best-case scenarios, base recommendation on the precautionary principle and prepare for adaptive management (EU, 2013a; 2013b).

2.5 DASHBOARD – PROGRESS TOWARDS TARGET

Element	Current Status	Comments	Confidence
Biodiversity values integrated into national and local development and poverty reduction strategies		Differences between regions. Evidence largely based on poverty reduction strategies	Medium
Biodiversity values integrated into national and local planning processes		The evidence shows regional variation and it is not clear if biodiversity is actually taken into consideration	Medium
Biodiversity values incorporated into national accounting, as appropriate		Initiatives such as WAVES show growing trend towards such incorporation	High
Biodiversity values incorporated into reporting systems		Improved accounting implies improvement in reporting	High

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TARGET 3: INCENTIVES

By 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the Convention and other relevant international obligations, taking into account national socio-economic conditions.

PREFACE

This analysis focuses on which way incentives, including certain subsidies, can be harmful to biodiversity and examines how these can be modified or eliminated to reduce impacts on biodiversity. It should be noted that fish stocks and forests are used herein as examples of biodiversity. The chapter also examines which incentives exist that are beneficial to biodiversity, and how incentives can be designed to support the sustainable use of natural resources. The likelihood that this target will be met will be explored using extrapolations of indicator trends, taking into account commitments of countries as set out in their National Biodiversity Strategies and Action Plans (NBSAPs).

Economic (and other) incentives that are potentially harmful to biodiversity are implemented in a number of sectors, and include subsidies for fisheries, agriculture, mining, transport (e.g. fossil fuel subsidies, road building) and water use. They can negatively impact on biodiversity (and sustainable use of resources by either increasing production, or under-pricing the use of natural resources (Convention on Biological Diversity; CBD, 2011).

In fisheries, subsidies are important incentives that can affect biodiversity conservation, and sustainable use of marine resources. As shown below, some fisheries subsidies are argued to be particularly harmful; hence, we choose to focus on these in this analysis.

In agriculture, subsidies are used to achieve a wide range of objectives, the aim of improving yield being the most important. Subsidies to this effect have been initiated by a wide range of countries (both developing and developed), and include subsidies for fertilisers, pesticides and water for irrigation (CBD, 2011). Other agricultural subsidies encourage deforestation, thereby affecting biodiversity. The Common Agricultural Policy (CAP) of the European Union was launched in 1962 to improve agricultural productivity, ensure a steady food supply and guarantee income for farmers. However, many countries have phased out direct subsidies of pesticides and fertilisers over the last years due to their negative impact on the environment.

Bioenergy crops are promoted through subsidies for a number of reasons, among them reducing dependence on fossil fuels, and achieving climate mitigation through the lowering of GHG emissions and carbon sequestration (e.g. Dauber *et al.*, 2010). Concerns for biodiversity arise mainly from direct and indirect land use change, e.g. through the displacement of agriculture.

Examples for incentives that are considered beneficial for biodiversity, and support the sustainable use of natural resources are instruments such as carbon finance, payment of ecosystem services schemes (PES), Reducing Emissions from Deforestation and Forest Degradation (REDD+) programmes, biodiversity offsets, markets for green products and a variety of taxes, fees and charges added to sustainably produced products (Office for Economic Cooperation and Development; OECD, 2013). Within CAP, a number of Agri-Environmental Schemes (AES) have also been implemented to maintain and protect biodiversity in agricultural landscapes.

In PES schemes, resource owners and communities providing certain ecosystem services are compensated for the provision of those services, while those benefiting from these services pay for the availability and use of this resource (Brouwer *et al.*, 2011). More broadly, PES cover a range of agreements, in which local communities, farmers and managers are being paid for activities and practices that provide ecosystems services, such as biodiversity conservation, carbon sequestration or water protection and provision (Brouwer *et al.*, 2011). Biodiversity offsets offer a range of options that address the effects of infrastructure projects on biological diversity, and allow leveraging additional funding for conservation of biodiversity and natural resources (Quintero & Mathur, 2011). However, the practice of ecosystem credit stacking, i.e. the sale of credits representing different ecosystem functions or services of a single site or area to compensate for an impact (Gardner & Fox, 2013), might lead to net environmental losses (Robertson *et al.*, 2014).

The United Nations Framework Convention on Climate Change (UNFCCC) REDD-mechanism has been formally recognised as a climate change mitigation option through the reduction of emissions caused by deforestation (Strassburg *et al*., 2012). Through UNFCCC Decision 1/CP.16, Annex I (UNFCCC, 2011), which outlines REDD+ safeguards, including safeguards on biodiversity, requires countries to take conservation of biodiversity and sustainable use thereof into consideration. Thus, REDD+ programmes thus transcend solely deforestation and forest degradation programmes, and include the conservation of forests, sustainable forest management and forest carbon stock enhancement. These initiatives thus have potentially positive effects on the conservation of (forest) biodiversity, as well as the sustainable use of forest resources (Karousakis, 2009).

3.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

3.1.1 Status and trends

3.1.1.i Fisheries

Some fisheries subsidies are of concern because they can contribute directly or indirectly to overcapacity and overfishing, and result in distortions in the trade of fisheries products. The reason for the latter is that countries or regions that receive subsidies can undercut regions that do not by selling their fish at lower prices (OECD, 2000). As such, they not only impact ecological sustainability, but also socioeconomic development (APEC, 2000). The Food and Agricultural Organisation of the United Nations (FAO) initiated the global subsidy debate in the early 1990s, when it argued that subsidies were a major causal factor in the creation and perpetuation of excess fishing capacity (FAO, 1992). Although overfishing results from multiple causes, it is widely acknowledged that overcapitalisation is a major contributor to the depletion of fishery resources (Hatcher & Robinson, 1999; Munro & Sumaila, 2002). Subsidies that inflate revenues or reduce fishing costs leads to an 'artificial' increase in profit, providing in turn incentives to increase fishing effort (Sumaila, 2003; Worm *et al*., 2009). The resulting overcapacity leads to detrimental effects on fish stocks. Heymans *et al*. (2011) demonstrated through a modelling exercise that the negative impact that subsidies can have on both the biomass of important fish species and the possible profit from fisheries. Also, Sumaila *et al*. (2014), through the analysis of extensive fisheries data on the Western Central Pacific Ocean revealed that many of the tuna fishing operations in this important tuna region of the world would not be viable without subsidies.

There is no single criterion for classifying fishery subsidies; the various categories (Milazzo, 1998; OECD, 2000; APEC, 2000) mostly overlap depending on the nature of the subsidy and the purpose of classification. The complexity of this issue is based on the fact that there is no single agreement on what a subsidy is or how its effect can be measured. Subsidies, support programmes, financial support, economic assistance, and government financial transfers are just five of the most commonly used names for payments that governments provide to the fisheries sector. The following guidelines are useful in identifying and assessing fisheries subsidies in this paper: (i) policy objective of the subsidy; (ii) the subsidy programme descriptions; (iii) scope, coverage and duration; (iv) annual US$ amounts; (v) sources of funding; (vi) administering authority; (vii) subsidy recipients; and (viii) the mechanisms of transfer (FAO, 2003; Westlund, 2004)[1].

It is worth noting that not all subsidies are bad in terms of their potential effects on overcapacity and overfishing. Hence, Khan *et al*. (2006) classified subsidies into three broad categories based on their potential impact on the sustainability of the fishery resource. These included: i) beneficial or good; ii) capacity enhancing or bad; and iii) ambiguous subsidies. Beneficial subsidies are programmes that enhance the growth of fish stocks to achieve maximum long-term sustainable net benefits. Capacity enhancing subsidies are programmes that facilitate development of fishing capacity to a point where resource overexploitation makes it impossible to achieve maximum sustainable long-term benefits. Ambiguous subsidies are programmes whose impacts on fishing capacity and the long-term sustainability of fisheries resources are undetermined (Table 3.1).

Footnote

[1] *These guidelines for identifying and assessing fisheries subsidies are for the purpose of the assessment of progress towards Aichi Target 3 only, and do in no way preclude or prejudge the WTO negotiations.*

Table 3.1. Classification of subsidy programmes according to Khan et al. (2006)

Category	Programme Types
Beneficial	Fisheries management and services
	Fishery research and development
Capacity Enhancing	Tax exemption
	Foreign access agreements*
	Boat construction renewal and modernisation
	Fishing port construction and renovation
	Fishery development projects and support services**
	Marketing support, processing and storage infrastructure***
	Fuel subsidies
Ambiguous	Fisher assistance
	Vessel buyback
	Rural fishers' community development

**When these agreements are partly or fully paid by governments then the payments are considered as subsidies.*

***This is likely to attract labour from, for e.g., the agriculture sector thereby increasing capacity.*

****With this type of subsidy, wholesalers and processor are more likely to transfer the benefit they get in this manner to fishers by paying higher prices thereby stimulating capacity.*

Previous global estimates of fishery subsidies ranged from US$14-20 billion (Milazzo, 1998) to US$54 billion (FAO, 1992). Sumaila & Pauly (2006) estimated global subsidies in the range of US$30-34 billion for the year 2000, while a recent re-estimation of that study amounted to US$27 billion annually for 2003 (Sumaila *et al.* 2010; Figure 3.1). Of this, 68% was provided in developed countries, and fuel subsidies accounted for the largest portion (about 23%) of the global total. Asia provided the largest amount of fisheries subsidies, about US$15.7 billion, while Africa had the least amount of subsidies, at around US$780 million. These differences in regional subsidies payments distort international fish trade to the detriment of regions such as Africa, Asia and South America. Capacity enhancing subsidies accounted for the largest portion of subsidies (about 60%), followed by beneficial and ambiguous subsidies (Figure 3.2). Other regional estimates include US$12 billion for the Asia Pacific Rim (APEC, 2000) and about US$2.5 billion for the North Atlantic (Munro & Sumaila, 2002).

3.1.1.ii Agriculture

Subsidies

Agricultural production is heavily subsidised, in particular in developed countries. In order to reform trade, and to make policies more market-oriented, the World Trade Organisation (WTO) Agreement on Agriculture was established in 1995. The agreement was also intended to improve predictability and security for importing and exporting countries alike. The agreement rests on three pillars, market access, export subsidies and domestic support, which has been classified into different "boxes". Subsidies falling into the amber box (i.e. those that are distorting production and trade) were to be reduced in the period 2000 – 2005 (2010 for all developing countries), while those in the blue box (subsidies designed to limit production but still distort trade) and green box (subsidies not distorting trade and not targeted at specific products, providing direct income to farmers, environmental protection and regional development programmes) could remain. Green box subsidies encompass domestic support measures, including environmental protection measures. Based on a 2013 proposal by the G-33, these now also cover measures for land rehabilitation, soil conservation and resource management, as well as drought management and flood control. Subsidies in this group are thus expected to be the least harmful, or even beneficial, to biodiversity while allowing the financial development of developing countries. However, environmental protection and related measures are only one of the many support measures included in this category, and many examples of "green box" subsidies exist that are in fact harmful or damaging to the environment (Brunner & Huyton, 2010).

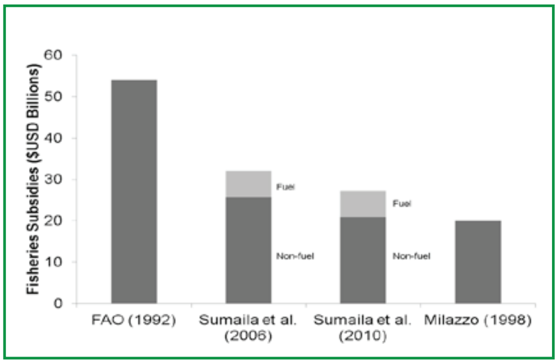


Figure 3.1. Comparison of global fishery subsidies estimates. Source: Table produced by authors

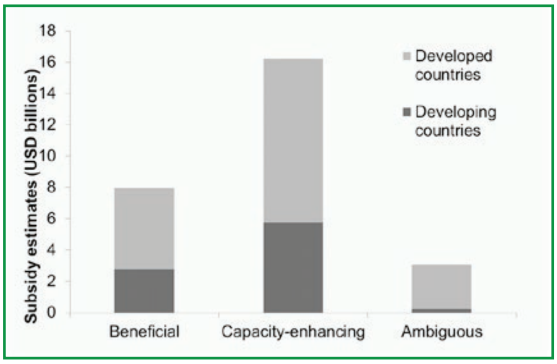


Figure 3.2. Global fisheries subsidy estimates by categories for 2003. Source: Sumaila *et al.* 2010

While spending on domestic ("green box") subsidies has decreased since 1996 in Japan, it has increased considerably both in the European Union (EU) and United States (US). Domestic support subsidies are of less importance in developing countries, for example, Brazil, South Africa and India spent considerably less (in % GDP) on agricultural subsidies than the EU, US, or Japan. In China, however, expenditure on agricultural subsidies, in particular green box subsidies, has risen considerably since the year 2000, with a part of them intended for environmental measures. For example, China has provided funding for pollution control, ecological conservation, returning farmland to forest, and natural forest conservation. Nevertheless, payments that promote actions that benefit the environment (and thus, biodiversity conservation and sustainable use of resources), are only a relatively small part of domestic ("green box") subsidies spending. In this sector, the EU spends considerably more on this than the US or Japan. In developing countries, spending on environmental measures is negligible.

In a joint exercise, a number of European Commission "directorates-generals" (Agriculture and Rural Development, Environment, Eurostat and Joint Research) and the European Environment Agency (EEA) developed a set of agri-environmental indicators (dubbed IRENA[2]) to monitor the integration of environmental concerns into the common agricultural policy (CAP), (EEA 2005). One of the indicators that directly link to EU policies, the share of utilised agricultural area under agri-environment schemes (IRENA 1) has been increasing steadily between 1998 and 2002 (EEA, 2006; Figure 3.3). However, it is difficult to assess the functioning and significance of agri-environment schemes (Kleijn & Sutherland, 2003), and data is currently not available for this indicator (EEA, 2014). It has been integrated, together with the Indicators *"High nature value farmland area"* and *"Area under organic farming"* into a new indicator: *"Agriculture: area under management practices potentially supporting biodiversity"* (EEA, 2014). Area under organic agriculture has equally been increasing steadily in both the EU 15 and EU 27 (EEA, 2014; Figure 3.4) over the last decade.

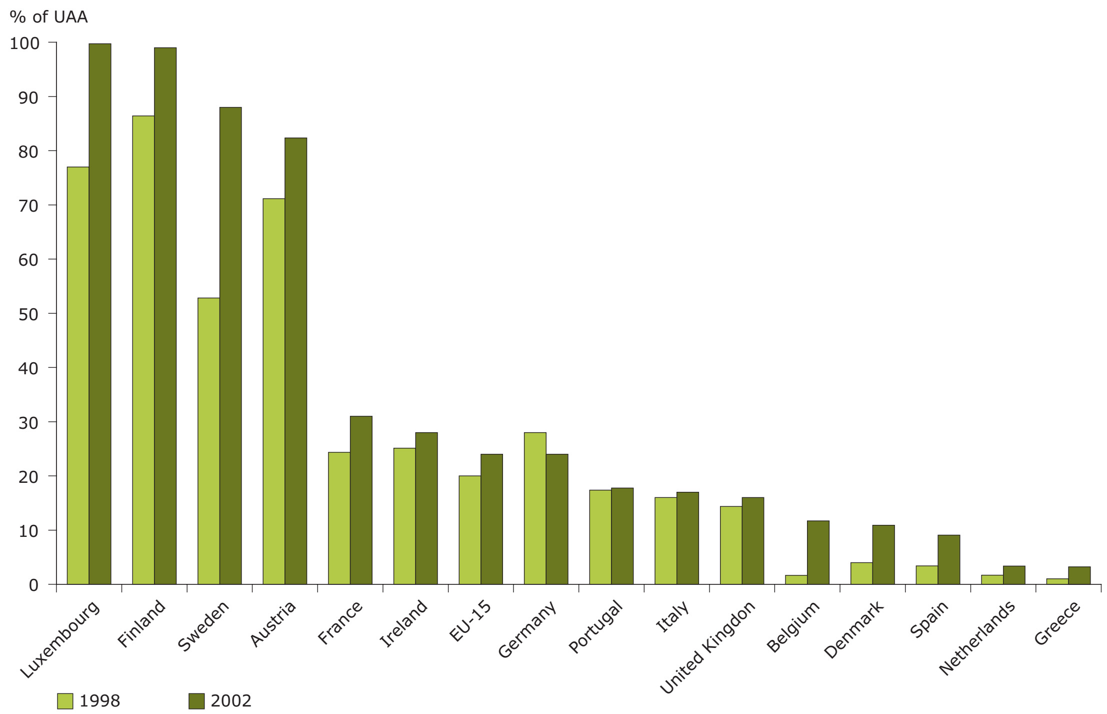


Figure 3.3. Share of utilised agricultural area (UAA) enrolled in agri-environment schemes in the EU 15. Source: EEA, 2006.

Footnote

[2] *Indicator Reporting on the Integration of Environmental Concerns into Agriculture Policy.*

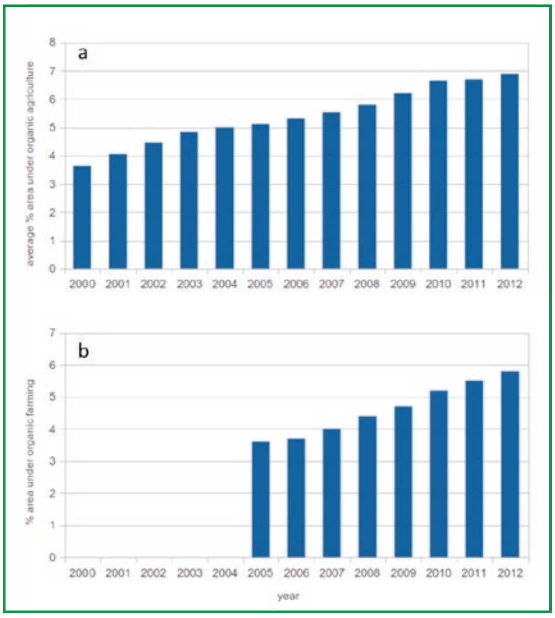


Figure 3.4. Average % area under organic farming for (a) EU 15 between 2000 and 2012, and (b) EU 27 between 2000 and 2012. Source: EUROSTAT, 2014.

According to the OECD (2013a), the Producer Support Estimate (PSE, a measure of direct support to agricultural producers individually) in OECD countries has been steadily decreasing in the period 1997 – 2005. Nevertheless, the use of less distortive forms of policies, which include incentives for improved environmental action, and adaptation and mitigation measures for climate change, has been increasing. In contrast, support to farmers in Indonesia and China has been increasing over the last decade, likely linked to self-sufficiency targets. To reach these, production-enhancing policies are necessary, which potentially have a negative impact on the environment, e.g. through increased fertiliser or pesticide use.

Payment for Ecosystem Services

Payment for Ecosystem Services (PES) Schemes are a tool to bridge trade-offs between conservation of biodiversity and ecosystem services provision with the exploitation of resources, and represent a more direct way to promote conservation by offering monetary or other compensation. The OECD describes PES as *"a flexible, incentive-based mechanism"* that can make use of (existing) policy and incentive instruments *"to promote the conservation and sustainable use of biodiversity and ecosystem services"*, and have the potential to *"make more efficient use of available finance in existing biodiversity programmes"* (OECD, 2010a).

In agriculture, incentive payment schemes, including direct payments for conservation and ecosystem services, are a common strategy to conserve biodiversity and to enhance the supply of ecosystem services (Armsworth *et al.*, 2012; Vickery *et al.*, 2004). For example, the EU and its member states allocate US$7.2 billion in incentives to safeguard environmental benefits, including biodiversity, in the US, US$1.7 billion per annum are being paid out in the form of annual rent fees to encourage management that benefits the environment (Armsworth *et al.*, 2012). Many countries around the world have initiated payment for ecosystem services schemes (PES) that are aimed at safeguarding the provision of ecosystem services and maintain biodiversity in landscapes under agricultural use. In Switzerland, PES has been initiated to counteract climatic and socio-economic changes in mountain regions that could lead to the loss of historically biodiversity-rich pastures (Huber *et al.*, 2013a). In Tanzania, a PES agreement between tourism operators and local villages ensures that habitat crucial for wildlife migration is maintained (Nelson *et al.*, 2010), and in Cambodia, a PES scheme was successfully used to protect the nest of endangered bird species, the programme, however it did not contribute to halting land clearance, the original cause of biodiversity loss (Clements, 2013).

The effectiveness of such programmes depends on a number of factors, among them the equitability of payment distribution, the acceptance of the service paid for, legitimacy of institutions involved, and the complementarity of the scheme to other enforcement methods (Gross-Camp *et al.*, 2012). They are most (cost-effective) when areas with highest biodiversity benefits, highest threat of loss or highest possibility of ecosystem service enhancement are targeted, as well as in areas where opportunity costs are lowest (OECD, 2010a). However, it has been suggested that programmes can be most cost-effective if payments for ecosystem services are made directly (Ferraro & Simpson, 2002), have mandatory involvement of communities, and involve cash payments (Brouwer *et al.*, 2011). The success of PES can further be increased when biodiversity is bundled with other services such as water provision or carbon sequestration, and are integrated with existing conservation efforts (Wendland *et al.*, 2010). Such integrated approaches are, for example, used in Madagascar, where sites high in biodiversity and standing carbon where selected (Wendland *et al.*, 2010), and Bolivia, where biodiversity conservation, watershed protection and water supply are linked in a PES (Asquith *et al.*, 2010). In Costa Rica's PSA programme, mitigation for GHG emissions, hydrological services, biodiversity and scenic beauty, all ecosystem services provided by forest ecosystems, are integrated (Pagiola, 2008). The biggest obstacles in effective implementation of PES include demand-side limitations and lack of supply-side know-how regarding implementation. Also, decisions on who the beneficiaries of payments are need to be carefully considered, and property rights need to be ascertained (Clements *et al.*, 2013), to make the PES socially acceptable. According to Wunder (2007), PES is not a silver bullet, as "*like other economic incentives, PES makes most sense at the margin of profitability, when small payments to landowners can tip the balance in favour of the desired land use*".

Biofuel Directives

To reduce dependence on fossil fuels, and to mitigate climate through lowering of GHG emissions and increasing carbon sequestration, a number of countries / regions have put in place targets for the production and use of biofuels, mainly in the transport sector (e.g. Dauber *et al.*, 2010; Steenblik, 2007). In order to reach the targets set, biofuels are heavily subsidised (Doornbosch & Steenblik, 2007; Eickhout *et al.*, 2008; Ravindranath *et al.*, 2009), and it has been suggested that biofuel production would be considerably reduced if subsidies are removed (e.g. FAPRI 2009; Searchinger *et al.*, 2009). Countries such as Indonesia and Malaysia, are investing in biofuel production, as there is demand for them in other regions of the world, mainly the EU (Campbell & Doswald, 2009).

Brazil introduced the Proàlcool Programme in 1975, as a response to the 1970 oil crisis, with sugar cane as main feedstock for the production of bioethanol (Webb & Coates, 2012). More recently, a subsidised biodiesel programme, with an increasing blending target of 5% from 2013 onwards, has been introduced. To increase energy independence, and to assist the transition to alternative fuels in the transport sector, the US created the Renewable Fuel Standard (RFS) programme in 2005, and expanded it in 2007 to include diesel in additional to gasoline. It was also hoped that the increased use of renewable fuels like ethanol and biodiesel would provide an expanded market for corn and soybeans, and contribute to reductions in carbon dioxide emissions (EPA, 2007). The new programme called for a blending of 36 billion gallons (136 billion litres) of renewable fuels by 2022. The EU renewable energy directive (RES-D) calls for a minimum of 20% of transport fuels to be composed of renewable sources, with 5.75% coming from biofuels (Banse *et al.*, 2013), and a 6% reduction in GHG emissions, compared to fossil fuels (EU, 2009). In 2012, the EC proposed an amendment to the EU Renewable Energies Directive from 2009 that is to include, among others, a limit on the contribution from biofuels / bioliquids derived from food crops, coupled with an enhanced incentive scheme to promote sustainable and advanced biofuels from feedstocks that do not create an additional demand for land and requires reporting estimated emissions from carbon stock changes caused by indirect land-use change (EC, 2012).

As a result of subsidy implementation to reach the fuel targets, as well as market demand, production of bioethanol has risen sharply since 2000, with a four-fold increase in production in the last 10 years. An even sharper rise in biodiesel production has occurred from 2005, with production increasing 10-fold in the last 8 years (Figure 3.5). An expansion in agricultural land to produce biofuels may lead to a decline biodiversity through habitat loss (Banse *et al.*, 2008; 2011; Hertel, 2010), or through increased fertiliser use (Sturmer *et al.*, 2013).

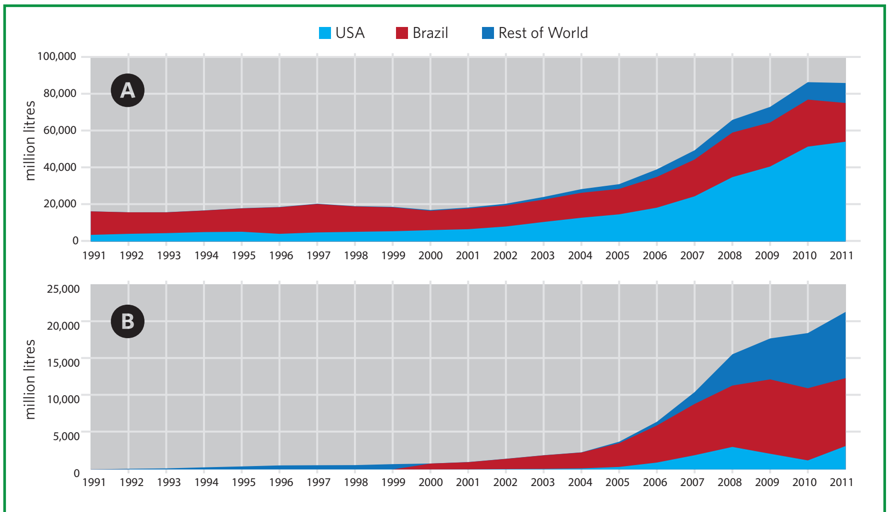


Figure 3.5. Development of (A) bioethanol production; and (B) biodiesel production between 1991 and 2011. Source: Earth Policy Institute, 2012.

However, in a report to the European Commission, Hamelinck *et al.* (2012) concluded that, for most European countries, the actual deployment of transport biofuels is lower than the target.

In Canada, direct subsidies supporting federal biofuels have been slowly lapsing with the expiration of the Biofuels Opportunities for Producers Initiative (BOPI), and all direct subsidies associated with the Canadian Renewable Fuels Strategy are scheduled to expire in 2017.

3.1.1.iii Carbon credits and mitigation

The UNFCCC's REDD mechanism, launched at their Conference of the Parties (COP) 13 in Bali in 2007, is aimed at mitigating climate change through the reduction of GHG emissions (UNFCCC, 2007). The scope of REDD includes the reduction of emissions from deforestation, reductions of emissions from forest degradation, the conservation of forest carbon stocks, the sustainable management of forests, and the enhancement of forest carbon stocks (Gardner, 2012). At the end of 2011, total support to countries implementing UN-REDD programmes totalled US$108 million (UN-REDD 2012). In 2014, 18 countries are partners of UN-REDD, receiving support to implement National Programmes, and a further 31 countries that do not have such programmes are receiving support (Figure 3.6). For the first two years of their National REDD+ Action Framework Programme (2011-2015), UN-REDD estimated a budget of more than US$50 million (UN-REDD 2010; Figure 3.7). For the period 2011-2015, the UN-REDD Programme has a goal of supporting countries in the development and implementation of their REDD+ strategies in order to speed up their REDD+ readiness (UN-REDD, 2010). The UN-REDD programme is supported by the World Bank Forest Carbon Partnership[3], which supports, through the Carbon Fund, countries that have demonstrated their REDD+ readiness. In November 2013, the BioCarbon Fund Initiative for Sustainable Forest Landscapes was launched at the UNFCCC COP19 in Warsaw, with funding pledges from Norway, the United Kingdom, the United States and Germany. Funding for the first year of this initiative will exceed US$280 million. A number of countries, for example Norway and Indonesia, have entered bilateral agreements with significant funding and mitigation potential.

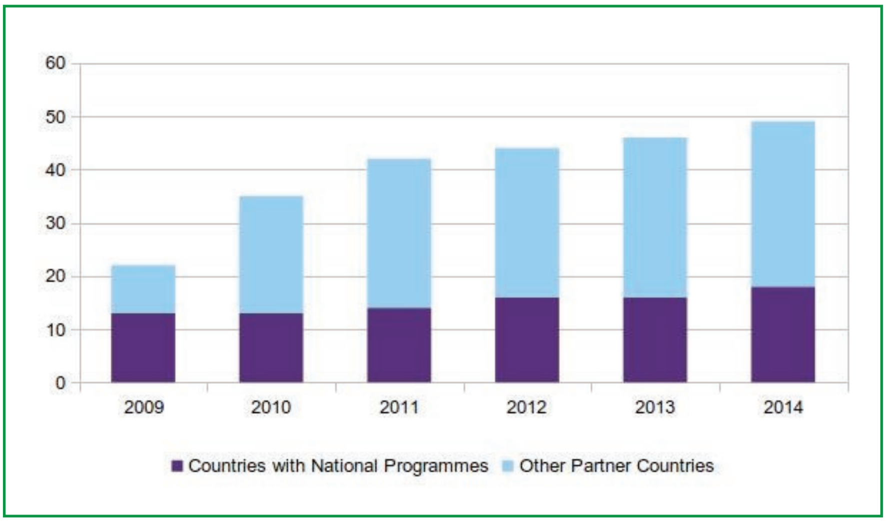


Figure 3.6. Number of countries with national REDD+ programmes, and number of other partner countries in the UN-REDD Programme in the period 2009 – 2014.
Source: UNDP REDD Programme 2014.

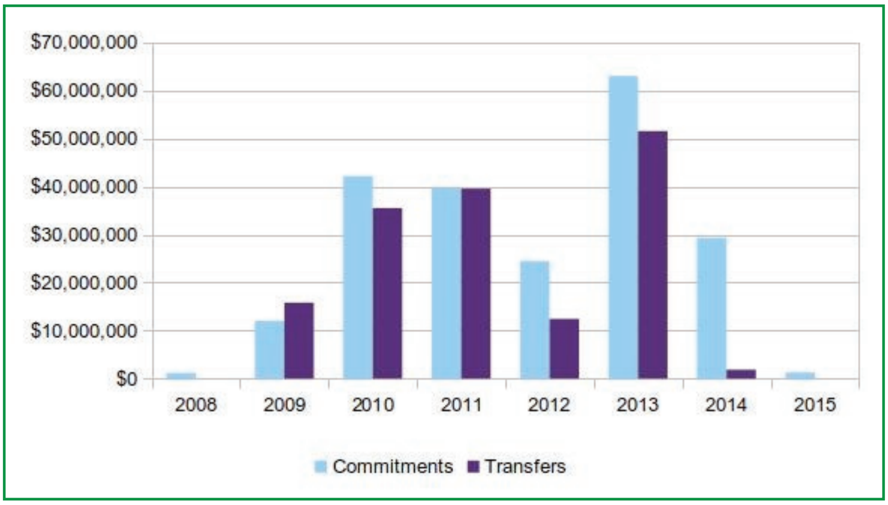


Figure 3.7. Funding commitments and funds disbursed in the UN-REDD Programme Fund. Note: Numbers for commitments and transfers for 2014/2015 are preliminary.
Source: UNDP Multi-Partner Trust Fund Office.

Footnote

[3] *https://www.forestcarbonpartnership.org/*

Although REDD+ contributes to the achievement of Aichi Biodiversity Target 5 (as well as Target 3), the implementation of REDD+ might, in some instances entail some risks for biodiversity as well as opportunities (Miles *et al.*, 2013). However, the safeguards implemented through UNFCCC Decision 1/CP.16, Annex I (UNFCCC, 2011), include safeguards on biodiversity, and require countries to take conservation of biodiversity and sustainable use thereof into consideration.

3.1.2 Projecting forward to 2020

3.1.2.i Fisheries Subsidies

Past estimates of fisheries subsidies that contribute to overcapacity and overfishing show no clear trends. A significant decrease will probably require successful completion of the Doha round of multilateral trade negotiations under the WTO.

Extrapolations of current trends suggest an increase in expenditure supporting sustainable fisheries is predicted to increase over the next few years (Figure 3.8).

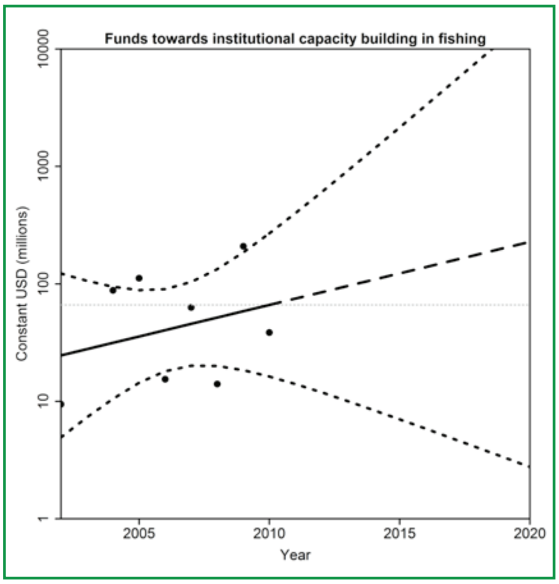


Figure 3.8. Statistical extrapolation of Funds towards institutional capacity building in fishing to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Source: Data from AidData (www.aiddata.org).

3.1.2.ii. Agriculture

Subsidies

"Amber" and "blue" box subsidies are likely to be phased out over the long-term, while global spending on "green box" subsidies is expected to increase leading up to 2020. This is consistent with extrapolations from current trends (Figure 3.9).

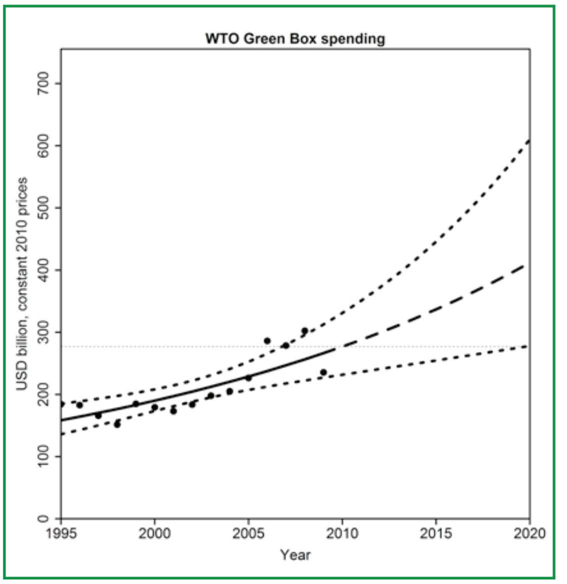


Figure 3.9. Statistical extrapolation of WTO green box spending to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Source: Data from WTO.

Biofuel Directives

The International Energy Agency (IEA) predicts that biofuel production and consumption is likely to rise three- to eight-fold, depending on scenario, up to 2035 (IEA 2010; Table 3.2). Under the "New Policies Scenario", subsidies could rise to US$67 billion per year.

Table 3.2. Predicted global consumption of biofuels up to 2035. Source: IEA 2010.

	Year	Global Consumption (MB/D)	Source
Actual	2000	0.19	IEA Data
	2009	1.11	IEA Data
Projections	2035	4.38	New policies scenarios
	2035	3.50	Current policies scenarios
	2035	8.11	450 scenarios

Extrapolations predict a ten-fold increase in biofuel uptake between 2010 and 2020. Despite the expectation that the EU will not be able to meet their biofuel target by 2020 (Hamelinck *et al*., 2012), Laborde (2011) predicts that EU biofuel production will increase to 17.8 – 20.9 Mtoe. Furthermore, global ethanol and biodiesel production are expected to expand, driven by demand promoting policies (OECD-FAO, 2013), with production reaching 168 billion litres for ethanol, and 41 billion litres for biodiesel by 2022. OECD-FAO estimated that this requires 12%, 29% and 15% of world coarse grains, sugar cane and vegetable oil production respectively. The three major producers of bioethanol are expected to remain the United States, Brazil and the European Union, who will also be the main producer of biodiesel. Argentina, the United States and Brazil, Thailand and Indonesia will be other significant producers. Production and consumption are driven by policies, and the EU will likely be forced to import and increase their imports of biodiesel and bioethanol grains considerably (Laborde, 2011), mainly from Brazil, countries of the Commonwealth of Independent Countries, and sub-Saharan Africa. The proposal of the European Commission to limit first generation biofuels that count towards the 10% target (EC, 2012) will have very little influence on these projections (OECD-FAO, 2013).

Fischer *et al*.(2010) estimate that, depending on scenario considered, between 44 – 72 million hectares of land could be freed up from agriculture and used for production of 1st and 2nd generation biofuel crops. A large proportion of the land will become available after yield gaps are being closed in Eastern Europe and the Ukraine (Figure 3.10).

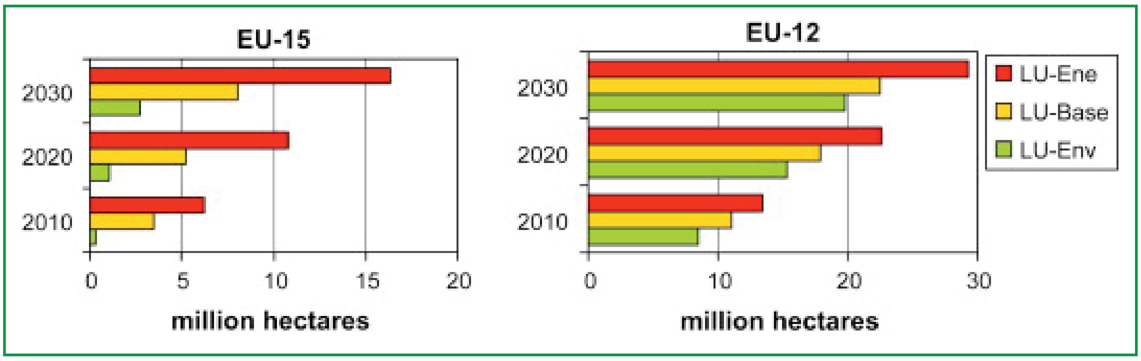


Figure 3.10. Agricultural land potentially available for biofuel crop production in EU 15 and EU 12 (acceded to EU after 1.1.2004) for three land conversion scenarios. LU-Ene: energy oriented scenario with substantial conversion, including conversion of pastures, LU-Base: reflects current policy in terms of biofuel and agricultural production, LU-Env: environment oriented scenario with emphasis on sustainable production. Source: Fischer *et al*., (2010).

3.1.2.iii. Carbon credits and mitigation

At the UNFCC COP19 in Warsaw in 2013, parties adopted the Warsaw Framework for REDD+, which builds on the Cancun agreement on REDD+. Included in the framework is a results-based finance for REDD+, the Green Climate Fund, designated in UNFCCC Decision 1/CP.16, as an instrument to channel the new multilateral funding for adaptation to countries (UNFCCC 2011). Decision 1/CP.16 further commits [developed] countries to jointly mobilise US$100 billion per year for this fund. In conjunction with this, the World Bank announced the US$280 million BioCarbon Fund, and Norway pledged an additional US$40 million to the UN-REDD programme. Despite these pledges, new and additional funding will be required to ensure successful implementation of REDD+. Without a comprehensive agreement on climate change with a significant price for carbon, REDD+ is unlikely to be implemented at scale.

3.1.3 Country actions and commitments[4]

Few countries have set national targets, or established similar elements, related to Aichi Biodiversity Target 3 (high). Based on an overall assessment of the available NBSAPs it appears that national efforts will need to be significantly scaled up and accelerated if Aichi Biodiversity Target 3 is to be achieved by 2020 (high).

Where targets have been set they have tended to focus on the promotion of positive incentives for biodiversity (medium) and on reforming or reducing harmful incentives (medium). For example one of the identified actions in Ireland's NBSAP is to develop positive incentives while Finland has set as a national target related to the identification and reform of incentives and subsidies harmful to biodiversity. There is comparatively less emphasis on eliminating harmful incentives altogether (low). An example counter to this general trend is Belgium, which has set the elimination, phasing out or reform of incentives harmful to biodiversity as an objective in its NBSAP. A number of countries have also established targets or similar commitments related to the identification of harmful incentives. For example the Dominican Republic has set a target of having an updated analysis of incentives, including subsidies harmful to biodiversity, as well as an action plan for their reduction, reform or elimination by 2016.

3.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

3.2.1 Actions

3.2.1.i Fisheries Subsidies

Removing subsidies that contribute to overcapacity and overfishing is one sure way of reducing overfishing (OECD, 2006; Milazzo, 1998). For instance, Sumaila *et al.* (2012) assumed that rebuilding global fisheries necessitated either cutting all harmful capacity-enhancing and ambiguous subsidies immediately, or turning them into beneficial subsidies by investing them in managing the rebuilding process. Heymans *et al.* (2011) provide support to this assumption by showing that while removing subsidies might reduce the total catch and revenue, it increases the overall profitability and the total biomass of commercially important species of North Sea fish. For example, cod, haddock, herring and plaice biomass increased over the simulation period when optimising for profit, and when optimising for ecological stability, the biomass for cod, plaice and sole also increased.

Current total global subsidies are equivalent to about 40% of global fisheries landed value of US$87.7 billion. Eliminating all subsidies that contribute to overcapacity and overfishing would reduce global subsidies to 17% of global landed value (Table 3.3), while eliminating these subsidies from the top fishing nations alone would reduce this to 27% of landed value. The effect of reducing harmful subsidies from other country groups is less.

Table 3.3. Breakdown of global harmful subsidies by country/political entity groups

Country group	Harmful subsidies (US$billion)	% of total global harmful subsidies	% of total global subsidies
Top Fishing*	11.8	59	33
Top 10 Developing**	4.5	22	13
Top 10 Developed***	9.6	48	27
Friends of Fish****	1.3	6	4
All countries	20.1	100	57

* *That is, the EU, Japan, China, Republic of Korea, Indonesia, USA, Thailand, India, Chile and Peru.*

** *This group consists of the Top 10 Developing country providers of harmful subsidies: Russia, Philippines, Indonesia, Thailand, India, Viet Nam, Malaysia, Brazil, Peru, and Argentina.*

*** *This group consists of the Top 10 Developed country providers of harmful subsidies: Japan, China, Korea Republic, Spain, USA, Poland, France, Greenland, UK, and Taiwan/Chinese Taipei.*

**** *Friends of Fish coalition include Argentina, Australia, Chile, Colombia, Ecuador, Iceland, New Zealand, Norway, Pakistan, and Peru.*

Footnote

[4] *This assessment is based on an examination of the NBSAPs from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPS and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

For some fisheries, removal of subsidies requires international cooperation because unilateral action puts the acting country at a trade disadvantage, and is further complicated by the transboundary nature of fish and mobile fishing fleets. Thus, to effectively reduce subsidies that contribute to overcapacity and overfishing necessitates multilateral action that involves all fishing nations reducing or eliminating subsidies under similar rules (Sumaila *et al.*, 2007). Some (e.g. Roger Martini, pers comm.) believe that unilateral reform may still be optimal for an individual country in this situation (see discussion below). The World Trade Organisation (WTO) is the only global international institution that has mechanisms in place to enforce its agreements. But so far, WTO negotiations have not yielded an agreement. Forging the way towards eliminating fishery subsidies will require countries to put aside their short-term self-interests in future WTO negotiations. Sumaila (2011) argued that a way to circumvent national interests is to split the world's fisheries into domestic and international fisheries, as incentives to eliminate subsidies that contribute to overcapacity and overfishing differ according to whether a fishery is domestic or international. Preliminary analysis shows that about 50% of the global catch of marine fisheries is from shared stocks (Sumaila *et al.*, Report Submitted to the Global Ocean Commission). Greater transparency and improved reporting of government transfers to the fishing industry is also needed to obtain a better understanding of how government fisheries subsidies are distributed globally.

3.2.1.ii Agriculture

Subsidies

"Amber box" subsidies are generally considered to be supporting "intensive agriculture", thus, phasing these out can be considered as positive for the environment and biodiversity. In contrast, "green box" subsidies are considered to be more environmentally friendly; however, there are many examples of "green box" subsidies that are harmful or damaging to the environment (Brunner & Hayton, 2010). The phasing out of these subsidies can have both negative and positive effects (Steenblik & Tsai, 2010), depending on whether: 1) new activities replacing previously subsidised activities are less damaging; 2) the intensiveness of practices will decrease or increase as a result; and 3) increased agricultural activities in new areas to compensate for decreases in previously subsidised areas are less environmentally damaging. Furthermore, intensification of agriculture, which concentrates production, can free up land for conservation purposes, while an "extensification" of agriculture makes it more land-intensive, potentially contributing to habitat loss. The trade-off between intensive and extensive agriculture, and how much (and what types) of biodiversity the two types can support, needs to be carefully considered (but see also chapters 5 and 7 for this discussion).

The EU has recently conducted a mid-term review of its Common Agricultural Policy (CAP), (Overmars *et al.*, 2013; Chiron *et al.*, 2013), with the aim of increasing biodiversity in agricultural landscapes. So far, the EU-Agri-Environmental Scheme has only achieved moderate biodiversity gains (Whittingham *et al.*, 2011), as the effectiveness, results and impacts of these schemes depend on taxa considered, scale and landscape context, and on landscape heterogeneity (e.g. Armsworth *et al.*, 2012; Whittingham *et al.*, 2011). However, by bringing the core insights of ecosystem services research, e.g. by considering ecosystem services in bundles at appropriate scales, the CAP reform could present a unique opportunity to direct policies towards sustainability (Plieninger *et al.*, 2012). With its CAP reform 2014 – 2020, the EU agricultural policy maintains a two-pillared approach, but with better integration between the two (EC, 2013). The "new" CAP now contains a "pillar 1" policy instrument directly aimed at the provision of environmental public goods. This is achieved by compensating farmers for the provision of services such as farmland biodiversity and climate stability. Pillar 2 support includes Natura 2000, agri-environment-climate as well as organic production. Overall, through better integration of the various policy instruments, it is hoped that more sustainable production can be achieved.

In the United States, the large scale at which agriculture is practiced has considerable negative effects on the environment (Earley, 2010), impacting soil, air and water quality. Following for example expansion of corn and soy cultivation, marginal lands are often used for agriculture, leading to habitat destruction and land degradation, as well increased erosion risk (Wright & Wimberley, 2013). The majority of "green box" subsidies are paid out to large, industrial-scale farms (Earley, 2010), potentially contributing to increased nutrient pollution. Increased insecticide and fertiliser applications have been attributed to expansion and intensification of agriculture (Mehan *et al.*, 2012). Large-scale farms are considered to be more efficient in fertiliser and pesticide use, and are more likely to use the latest farming methods compared to small-scale farms, and might thus have a smaller negative impact on the environment (Earley, 2010). Nevertheless, smaller farmers in general use fewer agro-chemicals, and their practices are, overall, less soil-damaging and polluting. Direct payments to farmers, intended for conservation measures such as wetland conservation or erosion protection, are restricted to producers of annual crops (grains, oil seeds), and include bio-energy crops, while producers of perennial crops are excluded from payments. The resulting corn ethanol rush lead to conservation reserve land being put back into production (Earley, 2010).

Effective agri-environment schemes (AES) need to be designed in such a way that they provide the maximum possible benefit with a fixed budget (Armsworth *et al.*, 2012), and policies need to be optimised through e.g. prioritisation of regions for conservation investment (Armsworth *et al.*, 2012). AES can further be improved by including adaptive management in the implementation of guidelines, and strategic spatial location, as there are more beneficial in heterogeneous landscapes and areas with higher levels of biodiversity (Whittingham *et al.*, 2011). To halt the biodiversity decline in agricultural areas, a two-tier approach to AES is required. As a first step, sufficient habitat needs to be secured, in a second step, interventions targeted of specific species are required (Vickery *et al.*, 2004).

Box 3.1: Incentives measures for the conservation and sustainable use of biodiversity in China

Over the last decade, China has implemented a number of incentive measures to promote conservation and sustainable use of biodiversity in agriculture and forestry. These measures include:

(1) Eliminating subsidies unfavourable to biodiversity. To avoid negative impacts on biodiversity and the environment, China eliminated in 2007 export tax rebates of 553 products of high energy consumption, pollution and resources consumption, including endangered animals and plants and their products, leather products, some wood products and disposal wooden products.

(2) Subsidising households that return cultivated land to forests. Since 1999, the central government has been subsidising those households that have returned their cultivated land to forests according to the actual areas returned and verified. These households also have the ownership of forests that grow on returned land, with contract period for owning and using returned land being as long as 70 years, while enjoying preferential tax incentives for benefits from use of returned land. In 2007, the State Council issued a notice on improving the policy of returning cultivated land to forests, with a view to increase the subsidies to related households. According to this notice, households living in the Yangtse River Basin and South China can be subsidised in cash by 1,575 Yuan RMB per hectare of land annually, while households living in the Yellow River Basin and North China can get a cash subsidy of 1,050 Yuan RMB per hectare of land. Farmers that return land to forests with ecological functions can be compensated for eight years, while those that return land to forests with economic functions can be compensated for five years. From 2008 to 2011, the central government provided specialised grants totalling 46.2 billion Yuan RMB. By the end of 2012, the central government has invested cumulatively 324.7 billion Yuan RMB, and 124 million farmers in 2,279 counties directly benefited from this investment, with per household being subsidised 7,000 Yuan RMB on the average.

(3) Subsidising the projects on natural forest protection. Natural forest resources protection projects were initiated in 17 provinces in 2000. The central government subsidised forest management and conservation as well as seedling cultivation and reforestation. The central government also provided subsidies by covering pension insurances for forest enterprise employees and social expenditures of forest enterprises, and providing basic life guarantees for laid-off forest workers. The total investment for the first phase of this project went up to 118.6 billion Yuan RMB. At the end of 2010, the State Council decided to implement a second phase of this project from 2011 to 2020, with 11 more counties (cities, districts) to be included in the project. The subsidy provided for reforestation will be 4,500 Yuan per hectare, and those for enclosing mountains for forest conservation and aerial seeding will be 1,050 Yuan RMB per hectare and 1,800 Yuan RMB per hectare respectively. Education subsidy is 30,000 Yuan RMB per person per year. Sanitation subsidy for forest areas in the upper reaches of the Yangtse River, the upper and middle reaches of the Yellow River and Inner Mongolia is 15,000 Yuan RMB per year and 10,000 Yuan RMB per year respectively. For state-owned forests, the central government provides 75 Yuan per hectare annually as forest conservation fee. For those collectively-owned forests that also belong to national-level public benefits forests, during 2011-2012, the central government provided 150 Yuan RMB per hectare annually as part of the funds for ecological compensation. Since 2013, this rate has been increased to 225 Yuan RMB per year. For local benefits forests the compensation funds are provided mainly from local government budgets, while the central government also provides 45 Yuan RMB per hectare per year as forest conservation fee. The total investment of the second phase of this project will be around 224 billion Yuan RMB.

(4) Subsidising projects of returning grazing land to grasslands. Since 2003 such projects have been implemented in eight provinces such as Inner Mongolia, Sichuan, Qinghai and Xinjiang. The central government has been subsidising the construction of fences and the provision of forages. In 2011 the central government raised the subsidy standards and percentages. 300 Yuan RMB per hectare is provided to fence building in Qinghai-Tibet Plateau while 240 Yuan RMB per hectare to other regions. A subsidy of 300 Yuan RMB per hectare is provided to reseeding grass; 2,400 Yuan RMB per hectare to artificial forage farming and 3,000 Yuan RMB per household for building feeding stables and rings. The central government invested cumulatively a total of 17.57 billion Yuan RMB in this project during the period 2003-2012, with projects having benefited 174 counties, more than 900,000 farm households and more than 4.5 million farmers and herdsmen.

Bioenergy Directives

Biofuel production would be considerably reduced if subsides are removed (Searchinger, 2009). Subsidies encouraging low levels of bioenergy deployment could have net positive effects on economics and human wellbeing, without large negative impacts on biodiversity and the environment. In contrast, subsidies encouraging high levels of bioenergy deployment are likely to have negative impacts not only on biodiversity and the environment, but also human wellbeing (Keeney & Nanninga, 2008), e.g. through competing land use with food production.

Protection of biodiversity could be improved by taking into account the full Greenhouse Gas (GHG) balance of bioenergy sources, include (indirect) land use change impacts (Keeney & Nanninga, 2008). Although biofuels may increase GHG emissions in comparison with fossil fuels when direct and indirect land use change as well as nitrous oxide emissions are accounted for (e.g. Gibbs *et al.*, 2008; Hertel *et al.*, 2010a; Plevin *et al.*, 2010; Searchinger *et al.*, 2008), a multitude of factors, among them feedstock, production region and method, and calculation methodology, influence the GHG emissions of biofuels. Furthermore, no consensus has been reached yet on how to appropriately consider ILUC in GHG emissions calculations. As a consequence, the Government of Canada has excluded Indirect land use change (ILUC) impacts in its Renewable Fuels Regulations. The US has revised the statutory requirements for renewable fuels, requiring the application lifecycle greenhouse gas performance threshold standards. Thus, the RFS2 programme lays the foundation for achieving significant reductions of GHG from the use of renewable fuels (EPA, 2013). To data, Canada is the only country that could demonstrate to the EPA the renewable nature of its biofuel crops and crop residues, using their aggregate compliance approach and is thus the only country to meet the EPA's definition of "renewable biomass".

Using marginal/degraded lands for the production of bioenergy crops has been suggested to further relieve pressure on natural habitats (Eickhout *et al.*, 2008). However, greater benefits for biodiversity could be achieved by allocating these so-called marginal and degraded lands for conservation and restoration (Dauber *et al*, 2012; Edwards *et al.*, 2011; Plieninger & Gaertner, 2011). To lessen impacts of bioenergy crop production on biodiversity, Doornbosch & Steenblik (2008) suggest applying the lessons learnt from forestry certification. However, a harmonisation of standards is necessary (Webb & Coates, 2012), with a development of a comprehensive framework to mitigate GHG emissions from agriculture, land use and land use change for all types of fuels.

Webb and Coates (2012) further suggest an assessment of biofuel benefits and impacts against all energy sources (not only against fossil fuels, as is currently the practice), and point out the need for a holistic life cycle analysis (LCA) that is not only focussed on GHG emissions, but also includes direct and indirect land use change, as well as alternative land uses, and the delay in carbon uptake.

3.2.1.iii Carbon credits and mitigation

At its 19th COP, the UNFCCC "Reaffirms the importance of addressing drivers of deforestation and forest degradation in the context of the development and implementation of national strategies and action plans by developing country Parties" and "Encourages Parties, organizations and the private sector to take action to reduce the drivers of deforestation and forest degradation" (Decision 15/CP.19, http://unfccc.int/resource/docs/2013/cop19/eng/10a01.pdf#page=43). In this meeting, countries adopted the seven decisions of the "Warsaw Framework of REDD+", which includes[5]:

1. A work programme on results-based finance to progress the full implementation of the mitigation actions (Decision 9/CP.19)
2. Coordination of support for the implementation of activities in relation to mitigation actions in the forest sector by developing countries, including institutional arrangements (Decision 10/CP.19)
3. Modalities for national forest monitoring systems (Decision 11/CP.19)
4. Timing and the frequency of presentations of the summary of information on how all the safeguards referred to in decision 1/CP.16, appendix I, are being addressed and respected (Decision 12/CP.19)
5. Guidelines and procedures for the technical assessment of submissions from Parties on proposed forest reference emission levels and/or forest reference levels (Decision 13/CP.19)
6. Modalities for measuring, reporting and verifying (Decision 14/CP.19)
7. Addressing the drivers of deforestation and forest degradation (Decision 15/CP.19)

Footnote

[5] *http://unfccc.int/methods/redd/items/8180.php*

In addition, countries are to promote and support the REDD+ safeguards, and provide information on how these safeguards are being addressed and respected during the implementation of REDD+ activities (Peskett & Todd, 2012). In particular, countries are to promote and support the *"conservation of natural forests and biological diversity and enhancement of other social and environmental benefits"*, and are to be consistent in regard to *"objectives of national forest programmes and relevant international conventions and agreements"*. If these safeguards are taken into consideration, and REDD+ activities are implemented accordingly, REDD can become both a climate mitigation tool and biodiversity conservation tool.

Nevertheless, there are a number of environmental concerns around REDD+ mechanisms and activities. Activities should ensure that the long-term ecological integrity of forests is maintained, and functional significance of biodiversity beyond carbon storage should be recognised (Gardner *et al.*, 2012). If the focus of REDD+ activities is placed mainly on maximising carbon storage, there is a potential risk of forests being replaced with plantations, the displacement of deforestation to areas of low carbon value (and high biodiversity value), increased pressure on non-forest systems with high biodiversity value, and afforestation of areas with high biodiversity value (CBD, 2011).

Lindenmayer *et al.*, (2012) even warn of three potential "bio-perversities" associated with REDD+ activities: 1) clearing of native vegetation to establish plantations; 2) planting of trees that may become invasive; and 3) plantations negatively affecting key processes, e.g. fire and hydrology. Many forest "improvement" treatments aimed at increasing carbon stocks have net negative impacts biodiversity (Putz, 2009). Carbon enhancements may be positive for biodiversity if natural regeneration of forests is taken into consideration, a diversity of native species is planted, management of forests is improved, and if afforestation contributes to habitat connectivity (Harvey *et al.*, 2010). To further avoid pitfalls in the implementation of REDD+, a thorough risk assessment is required before implementing REDD+ actions, including full carbon accounting and examination of incentives (Lindenmayer *et al.*, 2012). A number of options and approaches to integrate biodiversity conservation in REDD+ mechanisms are shown in Table 3.4 (Harvey *et al.*, 2010).

Table 3.4. Options and approaches to integrating biodiversity conservation into REDD+ mechanisms. Source: Adapted from Harvey *et al.*, 2010.

Approach	Design Features	Implementation Features
1. REDD options that contribute **both** to climate mitigation and biodiversity conservation	a. Promote REDD+ (deforestation, degradation, forest conservation, carbon stock enhancement and sustainable management of forests) b. Ensure REDD includes countries with high forest cover and low deforestation rates (HFLD countries) c. Design REDD framework to minimize international and intranational leakage d. Carefully select appropriate reference levels to ensure real and measurable emissions reductions e. Increase amount, sustainability and availability of finance for REDD readiness and implementation f. Develop appropriate definition of "forest"	a. Use context appropriate strategies to reduce deforestation and degradation b. Ensure active engagement and appropriate compensation of forest stakeholders to ensure long-term forest conservation c. Within a given country, prioritise the reduction of deforestation (over the reduction of forest degradation and forest carbon stock enhancement)
2. REDD options that deliver more biodiversity conservation, without compromising mitigation benefits	a. Establish a financing mechanism that gives countries access to additional (non-REDD) finance in cases where they deliver biodiversity benefits in addition to emission reductions	a. Within forests of identical carbon stock, prioritise REDD implementation in those of greatest biodiversity value b. Within forests of identical carbon stock, prioritise forests that contribute most to landscape connectivity

Approach	Design Features	Implementation Features
3. REDD options that deliver more biodiversity conservation, but could weaken mitigation benefits	a. Implement biodiversity safeguards within REDD architecture to prevent potential negative impacts on biodiversity b. Prioritise REDD activities in areas of high biodiversity value c. Ensure REDD areas contribute to landscape connectivity, promoting biodiversity conservation d. Provide premiums for REDD credits that arise from the conservation of forests of high biodiversity value e. Create international certification standards for REDD that ensure positive impacts on biodiversity conservation f. Link the UNFCCC to the CBD, the Ramsar Convention on Wetlands, and the Convention to Combat Desertification (UNCCD), requiring that REDD contribute to the biodiversity goals of these conventions	a. Within countries, prioritise REDD in areas of high biodiversity value b. Require Environmental and Social Impact Assessments (EAIAS) for REDD programmes

3.2.2 Costs and Cost-benefit analysis

Overall, most studies of the economic effects of subsidies, in particular, harmful subsidies to the environmental and natural resources sector, is that they are negative because such payments do not add value to society (e.g., Milazzo, 1998; World Bank, 2009). That is, for society as a whole, the net present value of a policy that eliminates harmful subsidies is positive over time, implying that the discounted benefits of doing so far outweigh the discounted cost over time. However, it should be noted that there are social and political costs to be incurred because in the short term, the removable of subsidies would almost surely result in losses of jobs, incomes and livelihoods even though these would be compensated for many times over in the medium to long term. Also, for the removal of subsidies to be helpful in reducing over capacity, it needs to be coupled with better and effective management of the fishery.

Benefits of meeting this target include the protection of biodiversity, the maintenance of ecosystem services, the improvement of economic efficiency through improved allocation of resources, and budgetary savings through abolishment of subsidies harmful to biodiversity.

3.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

3.3.1 Fisheries

If fishery subsidies are not reduced, it is expected that current levels of fishing overcapacity will continue, leading to continued decline in marine fisheries populations and ecosystems (see Target 6). On the other hand, Sumaila *et al.* (2012) estimated that eliminating all fishery subsidies that contribute to overcapacity and overfishing, along with reducing current excess fishing capacity through effective management, could in the longer term potentially increase global marine fisheries catch by 3-19%, and catch value by 5-25%, relative to current levels. Follow up analysis by Halpern *et al.* (2012), Costello *et al.* (2013) and Srinivasan *et al.* (2013) suggest that these percentages are low estimates. In October 2013, European Union law makers decided that for the period 2014-2020, annual fisheries subsidies of 1 billion euros would not be allocated to building new vessels. This is a promising sign that policy makers are starting to make some progress towards reducing subsidies that contribute to fishing overcapacity.

3.3.2 Agriculture

This section combines the discussion of effects of agricultural subsidies and incentives, and biofuel policies, as (first generation) biofuel crops are a subset of agricultural crops, and management of biofuel crop production will be similar to that of feedstock and food crops. The effects of agricultural production on biodiversity and the environment are thus similar. The reader is referred to chapter 7 (sustainable agriculture), chapter 8 (pollution) and chapter 5 (habitat loss) for further discussion on the impacts of agricultural production on biodiversity.

Biological diversity on farmland is beneficial, as this increases the provision of ecosystem services, and buffers agricultural land against future environmental change (Whittingham *et al.*, 2011). Despite the investments in AES across Europe, farmland birds are still declining (Whittingham *et al.*, 2011), and endemic species are threatened. This is likely linked to continued agricultural intensification, which contributes to the decline of a number of different taxa (Rusch, 2013; Lomba *et al.*, 2013), among them birds (Vickery *et al.*, 2004), butterflies (Luetolf *et al.*, 2009), and vascular plants (Dietschi *et al.*, 2007; Zechmeister *et al.*, 2003). These declines are driven by changes in quantity of suitable habitat (Brambilla *et al.*, 2010), as well as quality of nesting sites (Wright *et al.*, 2013) and forage / resource availability and quality (Butler *et al.*, 2010).

The CAP reform, which now includes direct payments for environmental measures, namely maintaining permanent grassland, diversification of crops, and maintaining an "ecological focus" area of at least 5% (EC, 2013a) has the potential to create an agricultural landscape that provides a diversity of different habitats. Properly implemented, this CAP reform may slow the decline (Chiron *et al.*, 2013), or even lead to a slight increase in farmland bird diversity (Overmars *et al.*, 2013; Figure 3.11). By creating and maintaining "ecological focus areas", i.e. field rows, hedges, or buffer strips, as well as permanent grasslands, the agricultural matrix is slowly restored. This reduces habitat fragmentation, connects populations and may improve ecosystem function and ecosystem processes (Donald & Evans, 2006), for example by allowing movement of pollinator or seed dispersers through the landscape. Buffer strips also have the potential to capture pesticide and fertiliser run-off (Vickery *et al.*, 2004), and may aid in erosion control.

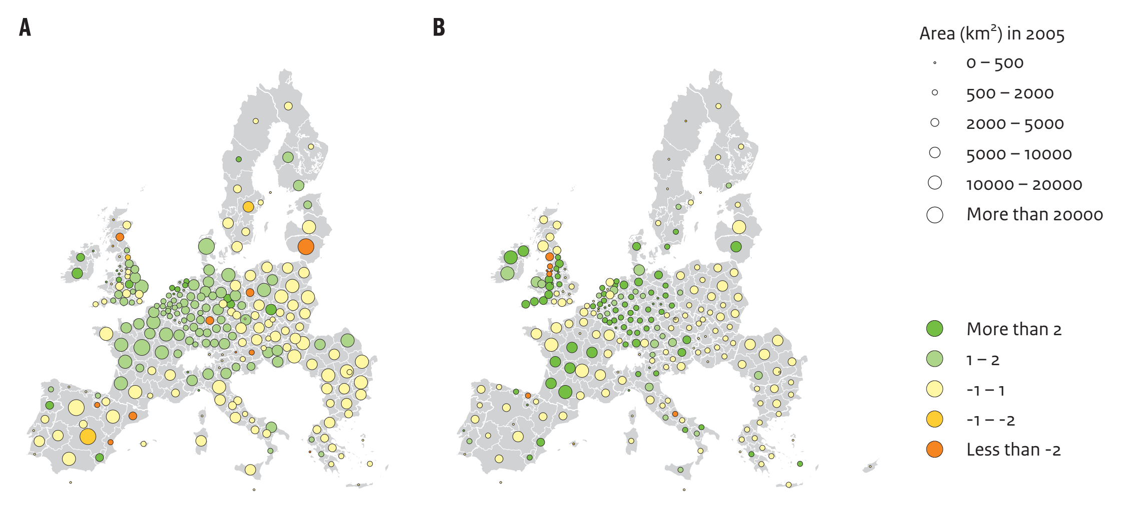


Figure 3.11. Changes in bird species richness relative to baseline in grassland (A) and arable land (B). Source: Overmars *et al.*, 2013.

Appropriate mitigation measures, such as policies that promote agricultural extensification may also have positive effects on functional composition and community structure (Rusch *et al.*, 2013; Prince *et al.*, 2013). However, as stated above, "extensification" of agriculture makes it more land-intensive, potentially contributing to habitat loss, and trade-offs between intensive and extensive agriculture, and how much (and what types) of biodiversity the two types can support, need to be carefully considered.

In the European Union, the widespread abandonment of low-intensity and semi-natural farmland has considerable conservation implications (Guilherme *et al.*, 2013), and multi-faceted impacts that are highly context-dependent (Keenleyside & Tucker, 2010). The land abandonment threatens mainly species of conservation concern that are associated with semi-natural habitats (Brambilla *et al.*, 2010) and endemic species (Lomba *et al.*, 2013). However, many species are able to adapt to these changes (Guilherme *et al.*, 2013), and policies that promote multifunctional rural landscapes (Turpin *et al.*, 2009), resulting in high landscape heterogeneity may buffer this local species loss (Lomba *et al.*, 2013).

Klink and Machado (2005) estimate that in Brazil, 55% of the Cerrado has been cleared for human use, including pastures and soybean cultivation for feed for livestock and, to a minor extent, biofuels. Although the impact of biofuel crop production on land transformation is very small, it is expected to become a major driver in the future due to further growth in the market. Increasing global demands for grains used for feed and biofuels are expected to contribute to the expansion of agriculture in to the Amazonian Forest (Nepstad *et al.*, 2008). Positive feedbacks with fire regime and drought might lead to further forest-dieback, which has far-reaching consequences for the local, regional and global climate. In the short to medium term, negative effects on biodiversity arising from land use change outweigh any benefits from climate mitigation (Eickhout *et al.*, 2008).

It has further been recognised that, especially when natural areas are being transformed for biofuel production, emissions from many first generation biofuel crops are actually higher than those for fossil fuel production (Campbell & Doswald, 2009; Searchinger *et al.*, 2008; Gallagher, 2008; Howarth, 2009; Ravindranath, 2009).

Investigating the EU renewable fuels directive, Eggers *et al.* (2009) found that a doubling of the biofuel target will lead to more species suffering from habitat losses rather than benefiting, while abolishing the biofuel target would mainly have positive effects on species distributions (Figure 3.12). However, the possible impacts showed spatial variation and depended on the biofuel crop choice. Woody biofuel crops may capture pollutants and provide some biomass (Christen & Dalgaard, 2013). However, they compare unfavourably to natural habitats for birds and mammals (Riffell *et al.*, 2011), and GHG benefits are lower than previously thought (Holtsmark, 2013). At field scale, woody bioenergy crops compare favourably with arable fields for species richness of nearly all taxa (Dauber *et al.*, 2010); however, they have lower species richness compared to woodland habitats. For birds, species composition in these fields is very different from forest habitats, and rather similar to open farmland or transitional shrubland (implying that that forest specialist species are lost).

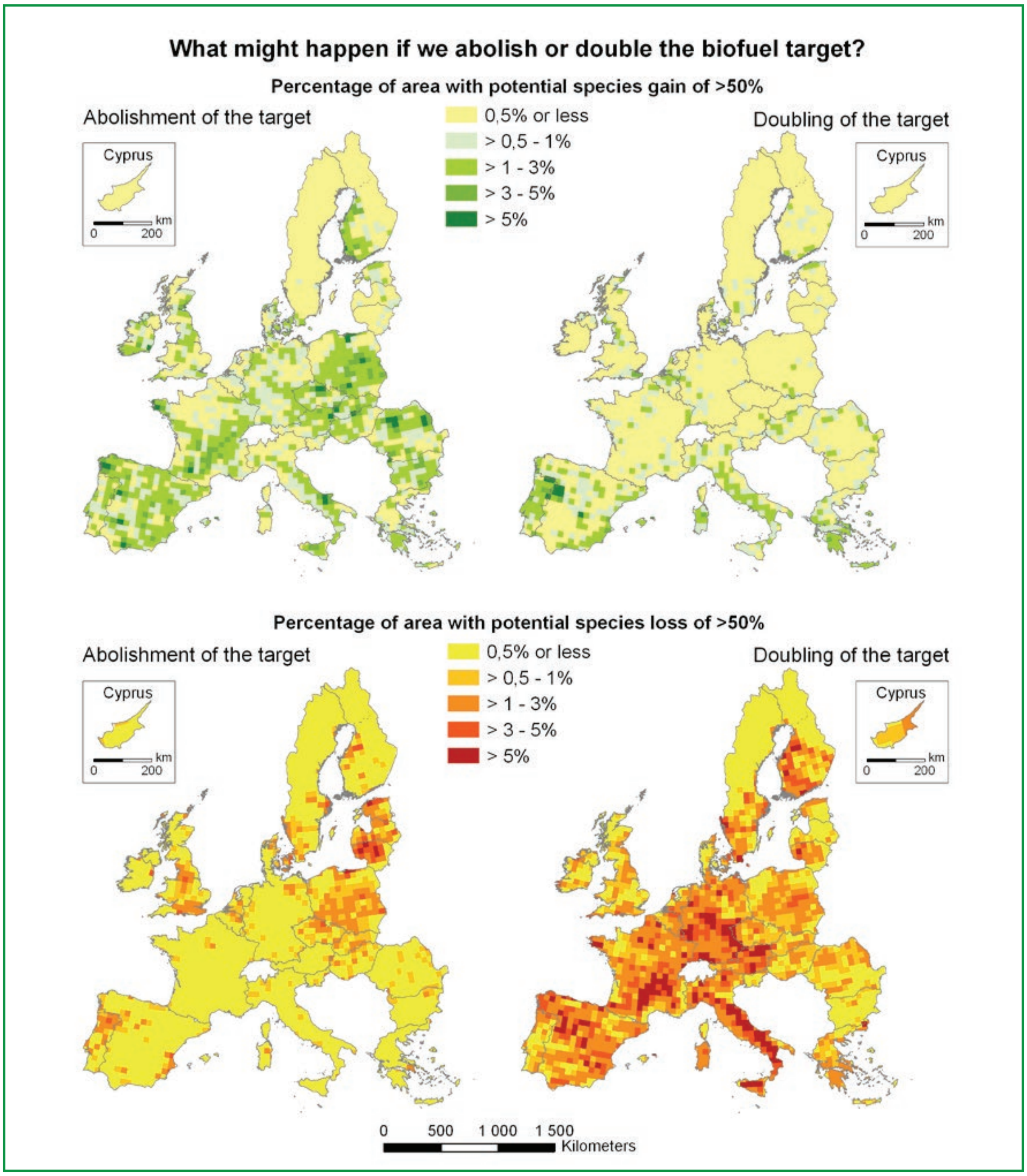


Figure 3.12. Effects of doubling or abolishing of the current EU biofuels target on the distribution of 313 mammal, bird, reptile and amphibian species. [Percentage of area of 50 km x 50 km cells where 50% of the species would gain or lose their potential habitat when abolishing or doubling the current biofuel target for arable biofuel crops is considered. Relative species shares are based on the total number of species in policy option e2 (reference policy option, reflecting the current biofuel target of 5.75%). Figures are summarised for birds, mammals, reptiles and amphibians]. Source: Eggers *et al.*, 2009.

To counteract potential habitat conversion and habitat loss from biofuel crop production, the revised EU renewable fuels directive imposes "safeguards", i.e. sustainability criteria aimed at preventing the conservation of land that has high biodiversity and high carbon stock (EC, 2012). It is estimated the EU can only produce 50% of biofuels domestically by 2020 (EC, 2012), and the increased demand for biofuel crops in the EU is expected to have considerable impact on agricultural production on European and global level (Banse *et al.*, 2011). To meet demand, the EU will need to increase imports from Argentina and the US (soy biodiesel), South East Asia (palm oil) and Brazil (bioethanol) to meet demand. Although it is predicted that more than 80% of land conversion for biofuel crops will take place in pastures and managed forests, savannas and grasslands (16%) and primary forest (3%) are also affected (Banse *et al*, 2011; Laborde *et al.*, 2011). This conversion is associated with considerable losses in biodiversity in all converted habitats (EC, 2012). A number of policy options can be implemented to increase the effectiveness of the directive in reducing indirect land use change, thus mitigating biodiversity loss. However, it has been suggested that the revised renewable fuel directives in the US and the EU have a slight positive impact on the environment (Lankoski & Ollikainen, 2010).

In a meta-analysis of biofuel crops used, or considered for use, in the US, Fletcher *et al.* (2010) found that bird and mammal diversity are lower in biofuel crop habitats than in non-crop habitats, with birds of conservation concern particularly affected. However, the conversion of row-crops (annual biofuel crops) to grasslands dedicated to biofuel crop production could improve local diversity and abundance of birds. For example, in North American Midwest, a projected expansion of corn and soy biofuel crop production onto marginal land will lead to declines in bird species richness (Meehan *et al.*, 2010; Figure 3.13), while a switch from annual to perennial bioenergy crops, e.g. grasses, is expected to lead to increases in species diversity, and may even contribute to the recovery of species of conservation concern (Meehan *et al.*, 2010).

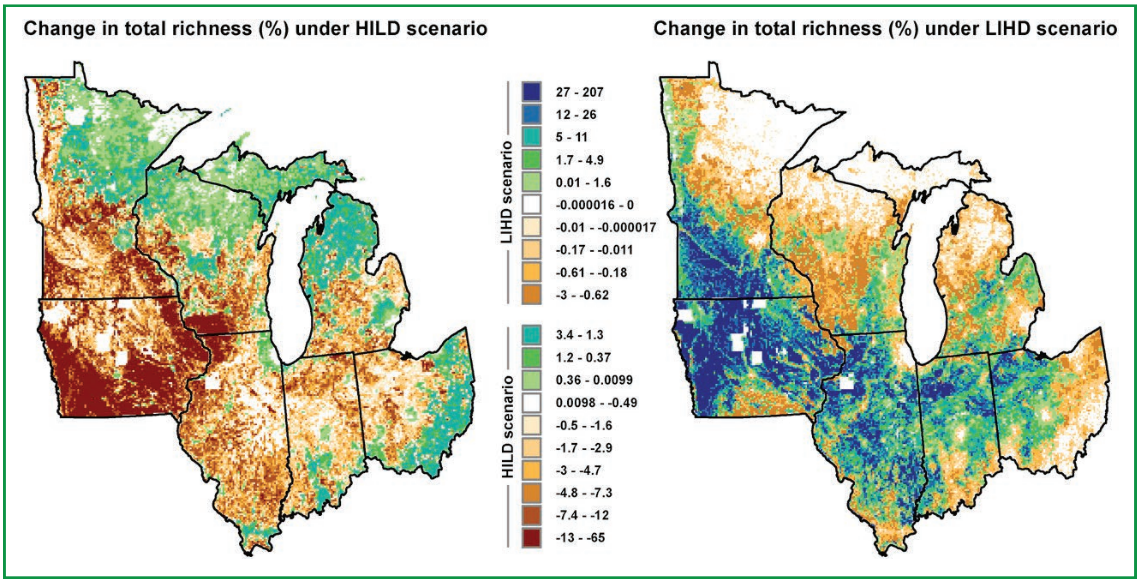


Figure 3.13. Percentage change in predicted bird species richness predicted for 25km2 landscape blocks under two biofuel crop type scenarios, HILD (left), and LIHD (right). Each colour shade corresponds with 10% of the distribution of percent change values.
HILD: High-Input Low-Diversity crops, corn and soybean. 9.5 million ha of marginal land that currently contain LIHD habitats were converted to HILD biofuel crops. LIHD: Low-Input-High-Diversity crops, native perennial grasses and forbs. 8.3 million ha of marginal land that currently contain HILD crops were converted to LIHD habitats. Source: Meehan *et al.*, (2010)

In California, shifts in agricultural activity to increase biofuel production for transport fuels, part of a strategy to reduce GHG emissions and dependence on foreign oil, can be expected over the next decade (Stoms, 2010). The authors compared potential biofuel crops (sugar beet, Bermuda grass and canola) in their impact on transport cost, wildlife, land and water use, and total energy produced. While sugar beets required the least land area, and canola the least water, impacts on biodiversity were greatest for canola. Bermuda grass had the least impact on biodiversity, and even resulted in a habitat improvement for a number of species.

3.3.3 Carbon credits and mitigation

The elements of REDD+ that are most effective for climate mitigation, i.e. greater financing combined with broad country participation, (Figure 3.14) are also most effective for biodiversity conservation and sustainable use (Busch *et al.*, 2011). However, the extent of these benefits depends on the interactions between pattern of deforestation, species distribution, and forest carbon stocks (Strassburg *et al.*, 2012). In particular, an adequately funded REDD programme may reduce species losses considerably (Strassburg *et al.*, 2012; Grainger *et al.*, 2009), but effectiveness depends on carbon pricing. Nevertheless, even at US$10 / tonne of CO_2, nearly 70% of species extinctions could be avoided (Fig 3.15). At US$25 / tonne of CO_2, species extinctions could be reduced by 92%, at the same time, up to 3 Gt of CO_2 emissions could be avoided (Strassburg *et al.*, 2012).

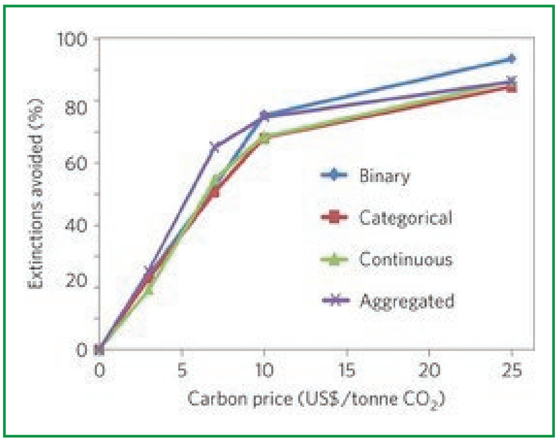


Figure 3.15. Relationship between carbon pricing and reduction of extinctions. Source: Strassburg *et al.*, 2012.

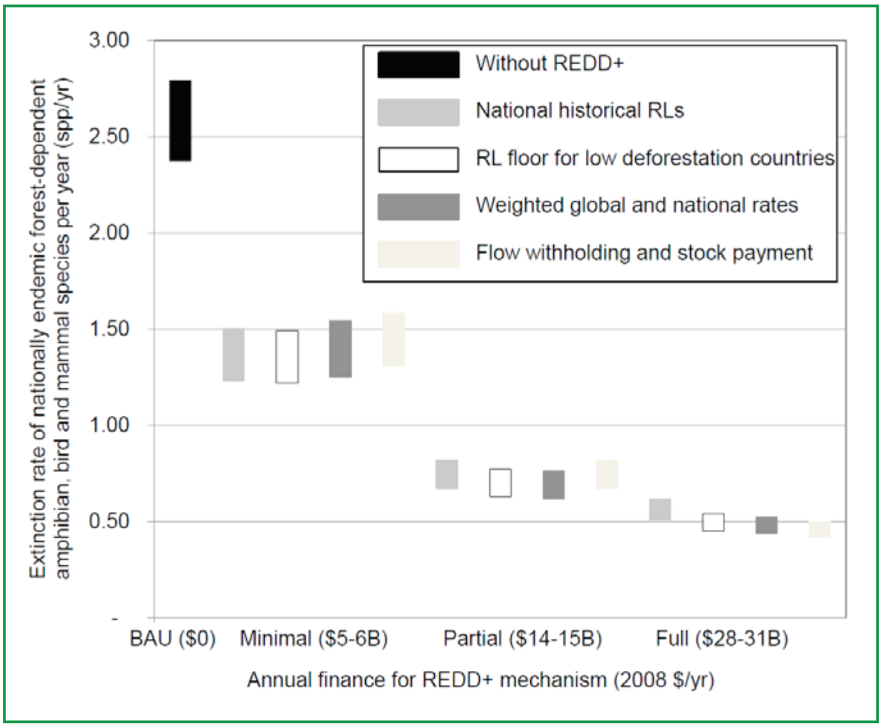


Figure 3.14. Annual extinctions of national endemic, forest-dependent amphibian, bird and mammal species across 85 tropical forest countries under alternative Reference Levels (Rls) and levels of finance for REDD+ for the period 2005-2010. Source: Busch *et al.*, 2011.

The relationship between carbon and biodiversity-based conservation are however unevenly distributed (Strassburg *et al.*, 2010), with the effects of deforestation highest in biodiversity hotspots in the tropics (Strassburg *et al.*, 2012). Many biodiverse regions that are carbon poor, such as savannas, grasslands or mediterranean-type shrublands, could come under increasing pressure (Strassburg *et al.*, 2010; Putz, 2009). These biodiversity hotspots also have high conservation costs (Grainger *et al.*, 2009).

The safeguards integrated into REDD+ request countries to conserve natural forests, and forest biodiversity, as well as the sustainable use of these resources. The conservation of biodiversity can for example be achieved by taking biodiversity pattern into account, and by prioritising areas that are high in biodiversity for implementation (Strassburg *et al.*, 2010). To target non-forest high biodiversity areas, with the aim to improve the conservation status of carbon-poor regions, a premium for emission from biodiversity-rich areas could be implemented into REDD, and a fraction of REDD financing could be set aside for targeting biodiversity-rich areas outside carbon-rich areas (Strassburg *et al.*, 2010). To achieve effective conservation of areas neglected by REDD, an active collaboration between biodiversity actors and REDD implementers in a region are required (Venter, 2009; Venter, 2013), and best results are achieved when biodiversity conservation complements carbon-based conservation (Venter *et al.*, 2013). Harnessing additional funds for conservation, and complementing

REDD+ actions is the best chance to stop forest loss and secure a future for biodiversity (Strassburg *et al*., 2012; Venter & Koh, 2012), as well as including forest conservation as an option in REDD+ (Harvey *et al*., 2010). Venter *et al*. (2009) suggest that the most effective way of reducing GHG emissions and at the same time, protecting biodiversity is to protect mature forests from destruction.

The collaboration does not stop at local and regional actors, though. To achieve appropriate biodiversity conservation in the context of REDD+, biodiversity needs to be given adequate consideration throughout the REDD+ process, and a framework to incorporate biodiversity into REDD+ is required (Gardner *et al*., 2012). There is a need for greater coordination between UNFCCC and CBD, as well as a range of stakeholders to ensure that biodiversity safeguards are fully adopted and implemented (Gardner *et al*., 2012).

Nepstad *et al*. (2011) illustrate a systematic approach where conservation of the Amazon Basin is achieved by coupling a REDD+ programme with cap and trade policies, and carbon offsets bought from offsets to lower deforestation rates. This, however, requires commitment from number of (decision-making) institutions to align policies, infrastructure investment, law enforcement, and institutional structures around carbon payments.

3.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

3.4.1 Fisheries subsidies

It is difficult to predict the future trajectory of fisheries subsidies, as it is driven by international policies and agreements that are influenced by trade and other economic factors. The application of Rio +20 Pathways[6] to rebuilding sustainable fisheries by 2050 is described in Target 6. It is assumed that as part of the fisheries rebuilding process, all fishery subsidies that contribute to overcapacity and overfishing have to be eliminated or redirected, while beneficial subsidies remain constant. Under these circumstances, it is expected that the decentralised solutions, Pathway will have the most positive impact on biological yields, as it provides the highest maximum catch potential by 2050, whereas the Global Technology Pathway provides the lowest catch potential (see Target 6 for details). Under all Pathways, harmful subsidies can be re-directed to monitoring and research to improve management, or for providing alternative livelihood opportunities to fishing communities. So there is potential for a win-win situation with respect to fishing productivity, alternative employment and biodiversity conservation.

3.4.2 Agriculture

To avoid negative effects on biodiversity and ecosystem services, a number of positive incentive mechanisms, like PES and AES schemes aimed at conservation of biodiversity and sustainable use of resources, will need to be put in place (OECD, 2010a). Harmful subsidies, like fuel subsidies, as well as water and fertiliser subsidies will need to be improved. Furthermore, biodiversity safeguards will need to be implemented into the various renewable energy directives, and 2nd and 3rd generation of biofuels will need to be explored. Radeloff *et al*. (2012) explore potential land use changes in the US for 2051 under four scenarios - business-as-usual, afforestation, removal of agricultural subsidies, and increased urban rents. Land use changes are greater in the business-as-usual scenario; however, differences between scenarios are relatively small (Figure 3.16). The authors conclude that although land use policies affect trends, the most main factors shaping land use changes in the US are of economic and demographic nature. Even rather large policy changes, as take place in the afforestation scenario, only produce only moderate deviations from the baseline scenario.

Footnote

[6] *Ref to report section that describes Pathways*

Figure 3.16. Projected land use changes in the United States by 2051 under four different policy scenarios.
Source: Radeloff *et al.*, 2012.

3.4.3 Carbon credits and mitigation

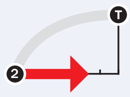
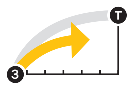
If implemented with biodiversity safeguards, and in close collaboration of carbon and biodiversity-conservation stakeholders and actors on all levels, and biodiversity safeguards are put in place on international and national levels, the synergies between climate mitigation actions and biodiversity conservation can be harnessed (Pistorius *et al.*, 2011). It is critical that biodiversity is considered in the development and implementation of carbon credits and climate mitigation measures (CBD, 2011). Opportunities for climate mitigation and biodiversity conservation synergies not only benefit biodiversity, but also forest ecosystem services, such as the provision of stable carbon storage by resilient forests (CBD, 2011).

3.5 UNCERTAINTIES AND DATA REQUIREMENTS

As always, uncertainties are present in the subsidies debate. There is, for example, uncertainty regarding subsidy estimates because not all governments report the amount of subsidies they give to their fishing sector. And, even those who do so may not provide a comprehensive coverage. It is therefore left to researchers to find ways to fill the knowledge gaps using different statistical approaches. A sure way to reduce the uncertainty here is for governments to be more transparent in their reporting of subsidies data.

A further uncertainty lies in the willingness of countries for international cooperation. Removal of harmful subsidies and implementation of beneficial incentives requires a global effort.

3.6 DASHBOARD – PROGRESS TOWARDS THE TARGET

Target Elements	Status	Comment	Confidence
Incentives, including subsidies, harmful to biodiversity, eliminated, phased out or reformed in order to minimize or avoid negative impacts		No significant overall progress, some advances but some backward movement. Increasing recognition of harmful subsidies but little action	High
Positive incentives for conservation and sustainable use of biodiversity developed and applied		Good progress but better targeting needed. Too small and still outweighed by perverse incentives	High

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TARGET 4: SUSTAINABLE CONSUMPTION AND PRODUCTION AND USE OF NATURAL RESOURCES

By 2020, at the latest, governments, business and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits.

PREFACE

The use of natural resources takes place at the society-nature interface. Society extracts natural resources from ecosystems that enter production processes to satisfy consumption. But society's dependence on nature goes beyond provisioning services provided by ecosystems. Society also relies on the regulating and supporting capacities of ecosystems. For example, ecosystems can work as a sink for pollutants generated as by-products of the economic process, but also can be an important element of cultural identity or human well-being. Using an ecosystem within its ecological limits requires, at least, that the rate at which society extracts resources and generates pollution does not exceed its regeneration time and absorption capacity (UNEP, 2010).

Aichi Target 4 aims at contributing to the target established by the Johannesburg Plan of Implementation of changing unsustainable patterns of consumption and production by requiring that by 2020 the Parties have taken, or planning to take measures to promote sustainable consumption and production (SCP). In this chapter, we aim to understand which steps have been taken towards the achievement of Aichi Biodiversity Target 4. We first provide a brief overview of internationally agreed goals towards SCP, together with an overview of current SCP initiatives worldwide. After, we explore some of the indicators suggested by the Convention on Biological Diversity (CBD) to assess progress towards the target (CBD, 2012). These include the Ecological Footprint (EF) and related indicators as well as trends in population and extinction risk of utilised species. We analyse the EF, the Human Appropriation of Net Primary Production (HANPP), the Water Footprint (WF) and the Primary Production Required to sustain marine fisheries (PPR).

The EF measures the amount of biologically productive land needed to fulfil human consumption of renewable resources and to absorb anthropogenic carbon dioxide (CO_2) emissions (Wackernagel and Rees, 1996). The WF measures the freshwater required for the production or consumption of a country and the freshwater required to assimilate the load of pollutants (Hoekstra and Hung, 2002). The rationale behind the choice of the indicators was to cover pressures from consumption and production on terrestrial, freshwater and also marine ecosystems. For terrestrial ecosystems, EF and HANPP were used. The EF provides information on the effects of production and consumption patterns and, by considering trade, it also allows to understand how a country appropriates domestic and international biocapacity. The HANPP is spatially explicit and provides an indication of the intensity of land use, in comparison with EF, it allows a better understanding of the effects of production on the ecosystems (Haberl *et al.*, 2004a). These are different metrics that provide complementary information. We also explore the trends in the extinction status of utilised species, through the Red List Index (RLI). Section 4.1 ends with short term (2020) projections of the main indicators. Section 4.2 focuses on the main actions required to achieve Target 4, as well a cost-benefit analysis. Section 4.3 highlights the implications for biodiversity of not achieving the Target. Section 4 .4 provides a long-term vision (2050) of the evolution of the indicators and achievement of the Target. Section 4.5 discusses the uncertainties. Finally, Section 4.6 provides a summary table of the progress towards the achievement of the Target.

4.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

4.1.1 Status and trends

There is widespread recognition that production and consumption patterns are the main cause of environmental degradation (UN, 1992; UN, 2002; UN, 2012). Several initiatives, at different geographical scales, have been put in place to address society's unsustainable patterns of consumption and production.

At the international level, the adoption in 2012 of the 10-Year Framework of Programmes on Sustainable Consumption and Production (10YFP) developed by the Marrakesh Process was the latest effort towards SCP. The 10YFP main objectives are the support of regional and national policies and initiatives to accelerate the shift towards SCP, provide technical assistance and capacity building to developing countries and serve as a platform for international knowledge sharing (UN, 2012). Two main means of implementation are a trust fund, to support SCP programmes and the global SCP clearinghouse, to support knowledge sharing (UN, 2012).

Substantial progress on SCP has been achieved through multilateral environmental agreements (MEAs). For example, the Montreal Protocol on Substances that Deplete the Ozone Layer phased out 98% of the ozone-depleting substances (Ozone Secretariat, 2012). Currently, there are six biodiversity related MEAs, all highlight the importance of sustainable use of biodiversity. The CBD objectives are *"the conservation of biological diversity, the sustainable use of its components and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources, including by appropriate access to genetic resources and by appropriate transfer of relevant technologies, taking into account all rights over those resources and to technologies, and by appropriate funding"* (UN, 1992b). The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) aims to ensure that international trade in specimens of wild animals and plants does not threaten their survival. In more than 40 years of existence, CITES has been able to achieve some success, fifty-seven taxa were transferred from Appendix I (species threatened with extinctions) to Appendix II (species whose trade must be controlled in order to avoid utilisation incompatible with their survival). Controls on international trade and national and regional management can be an effective way of improving species conservation status, and promote its sustainable use (UNEP-WCMC, 2013). Nevertheless, a case by case analysis is needed to measure real success (UNEP-WCMC, 2013). The Convention on the Conservation of Migratory Species of Wild Animals (CMS) aims to conserve terrestrial, marine and avian migratory species throughout their range. The main objectives of the International Treaty on Plant Genetic Resources for Food and Agriculture are the conservation and sustainable use of plant genetic resources for food and agriculture and the fair and equitable sharing of the benefits arising out of their use, for sustainable agriculture and food security. The Ramsar Convention on Wetlands main mission is *"the conservation and wise use of all wetlands through local and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world"*. Currently, approximately 2200 sites are listed as wetlands of international importance and cover 208 million hectares. Finally, the World Heritage Convention (WHC) aims to conserve the world's natural and cultural heritage. Under this convention approximately 197[1] sites are conserved due to its natural importance.

Regional and national scales are very important for SCP policies; as it is at this scale that most of the voluntary, information-based, regulatory and economic tools can operate (UNEP, 2012a). Numerous initiatives at both scales exist (UNEP, 2012a). For example, under the African 10YFP, a project for the development of an African Eco-labelling scheme is underway. Certification initiatives under different schemes are already in place for several sectors, fisheries, forestry, tourism, leather and textiles, agriculture and energy (Janisch, 2007). The development of an African Ecolabelling scheme would help this region to expand market access of African products, enhance the progress towards the Millennium Development Goals and would demonstrate Africa's engagement in SCP (Janisch, 2007). In Europe, the Europe 2020 strategy presents the growth strategy for the coming decade; one of its pillars is sustainable growth, which aims at promoting a more resource efficient, greener and more competitive economy (EC, 2010). Linked to this strategy is the European Biodiversity Strategy for 2020 (EC, 2011). Through the SWITCH programmes (SWITCH-Med and SWITCH-ASIA), the European Union is fostering cooperation to promote SCP in other regions. The SWITCH-Med is supporting nine countries (Morocco, Algeria, Tunisia, Libya, Egypt, Jordan, Israel, Palestinian Occupied Territories and Lebanon) in the development of National SCP Action Plans (EC/UNEP, 2012). The SWITCH-Asia programme aims to promote sustainable products, processes, services and consumption patterns in Asia by improving cooperation with European retailers, consumer organisations and the public sector and by providing funds to projects that will contribute to this goal (www.switch-asia.eu).

Footnote

[1] *According to the website http://whc.unesco.org, accessed in July 2014.*

At the national level one of the key tools to achieve SCP practices is through sustainable public procurement (SPP)[2], one of the seven Marrakesh process task forces (Brammer and Walker, 2011; UNEP DTIE, 2012; UNEP, 2011). The potential of SPP to influence SCP is related not only with the great volume of money involved, but also with variety of sectors involved. Public procurement accounts to 13 to 20% of GDP in industrialised nations, and more in developing nations, for example, in Brazil it accounts for 47% of GDP (IISD, 2012). Public procurement can foster innovation at the supply chain level, but also promote change both at the production and consumption side (Marty, 2012; UNEP, 2012a) (see Box 4.1). The other Marrakesh process task forces are sustainable products, sustainable buildings and construction, sustainable tourism, sustainable lifestyles, education for sustainable consumption and cooperation with Africa.

Box 4.1: Case Study: Sustainable Timber Action

Approximately 18% of all wood and related products imported by the European Union (EU) every year are from illegally logged timber (WWF, 2008a). European public authorities buy approximately 15% of the total timber and paper sold (STA, 2013). As of March 2013, the EU Timber Regulation (EUTR) renders illegal the import of illegally harvested timber in the EU. The establishment of procurement policies, requiring governments to purchase only legal timber, can be an effective way of excluding illegal timber from segments of a consumer country's market (Brack and Buckrell, 2011). The goal of Sustainable Timber Action (STA) is to use public procurement to increase awareness in Europe about the human and environmental issues caused by deforestation and forest degradation in developing countries, and about the impact of unsustainable consumption and production of forest products on climate change, biodiversity and people dependent on forests. STA's work has developed a toolkit for sustainable timber procurement, and has enabled the establishment of the European Sustainable Tropical Timber Coalition, a coalition of European local governments whose aim is to use public procurement to boost the market for sustainable tropical timber.

Businesses and civil society organisations have a key role in moving towards SCP (UNEP, 2012a). For example, the number of business adhering to voluntary sustainability reporting initiatives and product certification schemes has been increasing (for more information on certification schemes see Target 6 and 7; UNEP, 2012a). Recent developments in the Life Cycle Analysis field may open a door to product certification based on quantified impacts to biodiversity (Baan *et al.*, 2013; Curran *et al.*, 2011; Koellner *et al.*, 2013).

The International Union for Conservation of Nature (IUCN) with World Business Council for Sustainable Development (WBCSD) developed a guide to help business addressing biodiversity in their operations (IUCN, 2014). In this guide it is shown how four knowledge products[3]; the IUCN Red List of Threatened Species, Protected Planet, Key Biodiversity Areas and the Red List of Ecosystems; are relevant for businesses to manage the risks and opportunities associated with their impact on biodiversity (Table 4.1).

Table 4.1. Relevance of four knowledge products to businesses

IUCN Red List of Threatened Species	Red List of Ecosystems	Key Biodiversity Areas	Protected Planet
Identification of sensitive areas during screening process and baseline surveys.			
Supporting conservation actions			
Compliance with environmental standards, certification schemes and biodiversity safeguard policies.			
Valuation of ecosystem services.			
Reporting a company's environmental footprint			
Application of the mitigation hierarchy.		Design of offsets	
		Minimisation of impacts on biodiversity	
		Rehabilitation and restoration programmes	

Footnotes

[2] *The definition of Sustainable Procurement according to UNEP is the following: "process whereby organisations meet their needs for goods, services, works and utilities in a way that achieves value for money on a whole life basis in terms of generating benefits not only to the organisation, but also to society and the economy, whilst minimising damage to the environment.". Sustainable Public Procurement (SPP) is the sustainable procurement of governments.*

[3] *Knowledge products are platforms or baskets of knowledge that comprise assessments of authoritative biodiversity information supported by standards, guidelines, data, tools, capacity-building and tangible products (IUCN, 2014).*

As highlighted in Table 4.1, biodiversity offsets are mechanisms that can be used to compensate for significant biodiversity impacts arising from a project, after prevention and mitigation measures have been taken (BBOP, 2009). A recent assessment of offset and compensatory programmes worldwide accounted 45 active programmes, 27 in development (Madsen *et al*., 2011). The size of the market associated with these programmes was assessed to be US$2.4-4.0 billion per year, with at least 18 7000 ha of land covered by some sort of conservation or protection (Madsen *et al*., 2011). North America is the world region with more active programmes (15) and where more area is covered (Madsen *et al*., 2011).

UNEP's SCP Clearinghouse (www.scpclearinghouse.org) hosts a worldwide database on SCP initiatives. Currently, the database contains 680 entries, 91% correspond to on-going initiatives. There is a fairly good distribution of SCP initiatives around the globe; Europe takes the lead with approximately 22% of the initiatives. In all regions, the role of the UN or other international organisations in promoting the implementation of SCP initiatives is relevant, comprising 42% of the initiatives. In Europe, North America and Asia/Pacific this relevance is shared with governments and other public institutions. Interestingly, the role of the business sector as a promoter of SCP is higher in North America. The number of SCP initiatives allocated to energy, water, agri-food and waste (major themes of SCP) totalised 461 entries[4]. Amongst these, 30% addressed the agri-food sector, 26% energy and waste sectors and 19% the water sector.

4.1.1.i Ecological Footprint

The Ecological Footprint (EF) measures the amount of biologically productive land and sea area needed to produce the renewable resources required to fulfil human consumption and to absorb the anthropogenic carbon dioxide (CO_2) emissions (the so called carbon footprint; Borucke *et al*., 2013; Galli *et al*., 2012; Wackernagel and Rees, 1996). The comparison of the EF (of production or consumption) with the biocapacity (a measure of the amount of biologically productive land and sea available to provide these services), allows understanding at some extent the (un)sustainability of currents patterns of both production and consumption (Kitzes *et al*., 2009a, 2009b; Monfreda *et al*., 2004; Wackernagel *et al*., 1999). Since 1961, the EF has always increased (Figure 4.10A). In the early 70's, consumption by humans exceeded biocapacity, meaning that since then Humanity is in an overshoot situation (WWF, 2012). Overshoot occurs when ecosystems services are demanded at a pace faster than they can be renewed (Catton, 1982). Increasing anthropogenic CO_2 emissions have been the main driver of overshoot (Moore *et al*., 2012; WWF, 2008b). In 2007, the global EF was approximately 18 billion global hectares (gha); i.e., humanity used the equivalent of 1.5 Earths to support its consumption and absorption of CO_2 emissions. Through time EF's more significant components have changed. For example, cropland represented approximately 50% of the total footprint in the 60's, and 20% in 2008; carbon uptake area represented approximately 10% of the total footprint in 60's, and 50% in 2008 (Figure 4.1 -A).

In absolute terms, the highest EF is found in Middle Income countries, whereas the biggest ecological deficit[5] is found in High Income countries and is mainly caused by a high carbon footprint (Figure 4.1B). As proposed by Ehrlich and Holdren (1971), the human impact on the environment can be determined by three main variables, population, affluence (consumption) and technology[6]. Galli *et al*. (2012) analysed a time-series of EF and Biocapacity to understand the contribution of the different variables[7] in the environmental impact generated by economic growth (High, Middle and Low income countries). From 1965 to 2005, the global per capita footprint remained stable (Galli *et al*., 2012), however global population has more than doubled representing a huge increase in global EF (Figure 4.10A). The analysis of per capita trends of EF and population showed that High Income countries were the only ones where per capita EF increased more than the population; in Low income countries there was a reduction in the per capita EF (Galli *et al*., 2012). Such results enable to understand the main drivers of environmental impact in the different countries' groups. In High Income countries, the increase in total EF was mainly driven by increased consumption patterns not accompanied by similar efficiency gains. In Middle Income countries the increase in total EF was due to population growth, and to a lesser extent improvements in life style, in Low income countries the increase in total EF was mainly driven by population growth, since per capita EF has decreased.

Footnotes

[4] *Accessed on 2nd January 2014. Note that in this total double counting exists, as an initiave can be accounted under several diferente sectors. Nevertheless, this analysis sheds some light on the relative importance given to SCP in the diferente sectors.*

[5] *Ecological deficit refers to the situation in which a country's Ecological Footprint of consumption is higher than that country's biocapacity.*

[6] *Under the well known I(mpact)=P(opulation) x A(ffluence) x T(echnology) formula.*

[7] *In this study EF is intended as the product of per capita consumption (A) and level of technology (T).*

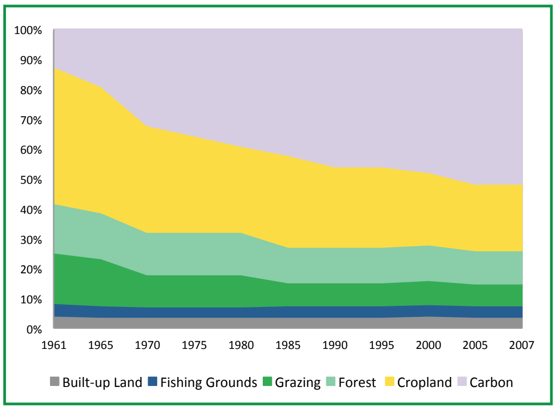


Figure 4.1. A) Evolution of the Ecological Footprint by different EF components, B) Ecological Footprint and biocapacity for High, Middle and Low income countries[8], in 2007. Source: Global Footprint Network (2012).

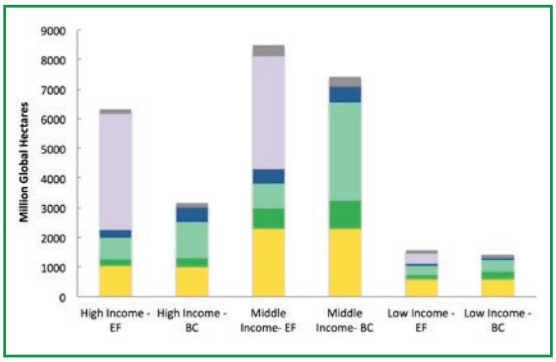


Figure 4.2. Per capita EF and % composition by different components, for High, Middle and Low Income countries, in 1965, 1985 and 2005. Source: Galli *et al*. (2012).

The reasons behind changes in per capita EF vary between the different countries' groups (Figure 4.2). In High income countries, the carbon footprint component grew from 31% in 1965 to 63% in 2005, and the cropland footprint component decreased from 37% in 1965 to 18% in 2005. This might be interpreted as a sign of structural change of the economy from agricultural dominated society to an industrial dominated society (Galli *et al*., 2012b; Krausmann *et al*., 2013). In Middle Income countries, the same patterns as in High Income countries were found. The carbon footprint component increased from 16% in 1965 to 46% in 2005 and the cropland footprint component decreased from 51% in 1965 to 28% in 2005. Low Income countries exhibit the more pronounced increase in the carbon footprint, more than 100% in 2005 when compared to 1965 levels, the cropland footprint component decreased from 62% in 1965 to 44% in 2005. (Galli *et al*., 2012b) point out that these results suggest that Middle and Low income countries are following the same development pattern as High Income countries. As it is in Middle and Low income countries that the highest economic and population growth will occur in the coming decades, decisions made in such countries will largely influence global sustainability (see Box 4.2). Also, reducing consumption levels in High Income countries is essential to global sustainability. Increases in EF will result in greater pressures on terrestrial ecosystems in these countries. The EF of production refers to the demand placed on local ecosystems, if compared to biocapacity it provides insights on the extent of overexploitation (Figure 4.3).

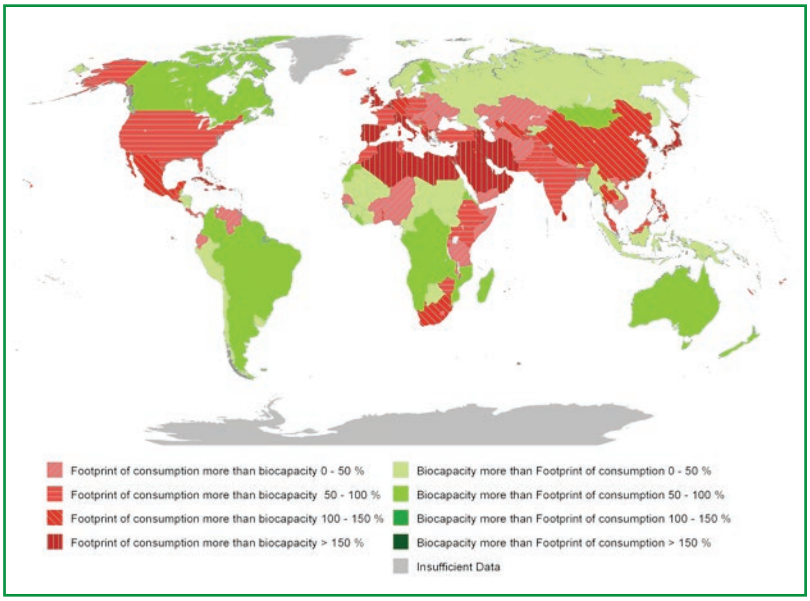


Figure 4.3. Ecological Footprint of Production vs. Biocapacity. Source: Galli *et al*. (in press).

Footnote

[8] *The definition of the different income categories can be found in Galli et al. (2012).*

There is a lot of controversy around the EF. On the one hand, it is one of the best communication tools when it comes to inform society about their impact on the planet. On the other hand, it has received several criticisms ranging from its methodology to the policy messages it conveys (see van den Bergh and Grazi, 2014; Blomqvist *et al*., 2013; Giampietro and Saltelli, in press; Rees and Wackernagel, 2013; Wackernagel, 2014, for the latest criticisms and responses). One of the criticisms relevant for this Target concerns the fact that the EF should not be considered as a sole indicator to assess (un)sustainability of ecosystems exploitation. It only considers CO_2 emissions as source of pollution, whilst other pollution sources, like nitrogen or phosphorous deposition (see Aichi Target 8) might have more detrimental impacts on ecosystems and biodiversity (Butchart *et al*., 2010). Another criticism concerns the fact that the EF cannot provide insights about the sustainability of the exploitation of the ecosystems (van den Bergh and Verbruggen, 1999; Fiala, 2008). Van den Bergh and Verbruggen (1999) point out that EF accounting does not provide information on the environmental consequences of fertiliser and pesticides use, groundwater control and irrigation. Fiala (2008) analysed the correlations between the EF and land degradation, and the results show that the footprint captures almost no effect on land degradation. When it captures it, it does so in an opposite direction.

Probably the most widespread critique of the EF is that it is mainly a measure of carbon footprint. Recently, Blomqvist *et al.* (2013) showed that overshoot occurs only in the carbon footprint component, and that none of the other land use categories have an ecological deficit suggesting that no depletion is occuring at the global level. In their reply, Rees and Wackernagel (2013) acknowledge this limitation, that stems from current accounting methodologies, since cropland, built-up land and grazing land footprints can only be less than or equal to the respective biocapacity, correcting the footprint in respect to the issues raised will lead to higher footprint estimates.

The EF does not directly measure biodiversity; however, it can be used as an indicator of the drivers or pressures that cause biodiversity loss (Galli *et al*., 2014; Kitzes and Wackernagel, 2009; WWF, 2008b). The main consumption categories assessed within the EF are energy (fossil fuels), food products (cropland and grazing land, fishing grounds) and forest-related products (forest land). All of these can be related to anthropogenic threats to biodiversity. The carbon footprint component measures the unbalance between the rate of anthropogenic CO_2 emissions and that of CO_2 sequestration by the biosphere. The higher the footprint, the higher is the accumulation of CO_2 in the atmosphere, which represents increasing threats to species through shifts in habitat ranges due to climate change. The footprints of food and forest-related products are deeply related with two of the main drivers of biodiversity loss: habitat loss (land use change, habitat destruction and fragmentations) and overexploitation. The EF can provide insights on the drivers of land use change. Weinzettel *et al*. (2013) analysed how world regions displace land use to other regions, and found that 24% of the global EF (without the carbon footprint component) was displaced through international trade. They showed that richer countries displace land use to lower income countries. For example, Europe's consumption requires land from Central and Latin America and Asia. Similarly, Galli *et al.* (2014) concluded that Switzerland's stabilisation of its EF of production, mainly cropland footprint, during the 1974-2008 period, reflects a stabilisation of the pressures on ecosystems which lead to an improvement of Switzerland's biodiversity status. Nevertheless, the EF footprint of consumption did decrease as the products (and biocapacity) were imported from other countries. This means that Switzerland reduced the pressures on its ecosystems, but instead exerted indirect pressures on the ecosystems in the countries from which imported products.

4.1.1.ii Human Appropriation of Net Primary Production (HANPP)

Net Primary Production (NPP) is the net carbon assimilated by terrestrial vegetation in a given period. NPP is the energetic basis of all ecosystems as it determines the amount of energy available for transfer from plants to other trophic levels. The Human Appropriation of Net Primary Production (HANPP) tries to capture the aggregate impact of land use on biomass available, in each year, in ecosystems (Haberl *et al*., 2007a). It is measured as follows (Erb *et al*., 2009; Haberl *et al*., 2007b; Krausmann *et al*., 2013). NPP_{pot} is the potential NPP or the NPP that would be produced by the vegetation in the absence of human interference; NPP_{eco} is the NPP that remains in the ecosystems after harvest. In NPP_{eco} computation, NPP_{act} is the NPP of the actual vegetation, and $HANPP_{harv}$ the NPP harvested by humans. $HANPP_{luc}$ represents the differences between the NPP of the vegetation under actual land uses and the NPP if it had remained native vegetation not influenced by management practices. Normally, HANPP is expressed as a percentage of potential NPP.

The HANPP has been increasing through time, more than doubling in the last century (Krausmann *et al.*, 2013). This increase occurs as a result of land use change and biomass harvest mainly due to agriculture (Haberl *et al.*, 2007a; Krausmann *et al.*, 2013; Vitousek *et al.*, 1986). Globally, the HANPP is approximately between 30% and 40% (Haberl *et al.*, 2007a; Imhoff *et al.*, 2004; Krausmann *et al.*, 2013; Vitousek *et al.*, 1986), However, from the remaining 60% of NPP, 53% is not harvestable since it comprises growth in root systems, preserved land and wilderness areas with no logistical infrastructure for harvest (Smith *et al.*, 2012). As a result, 30% of HANPP indicates a high level of anthropogenic dominance of the biosphere.

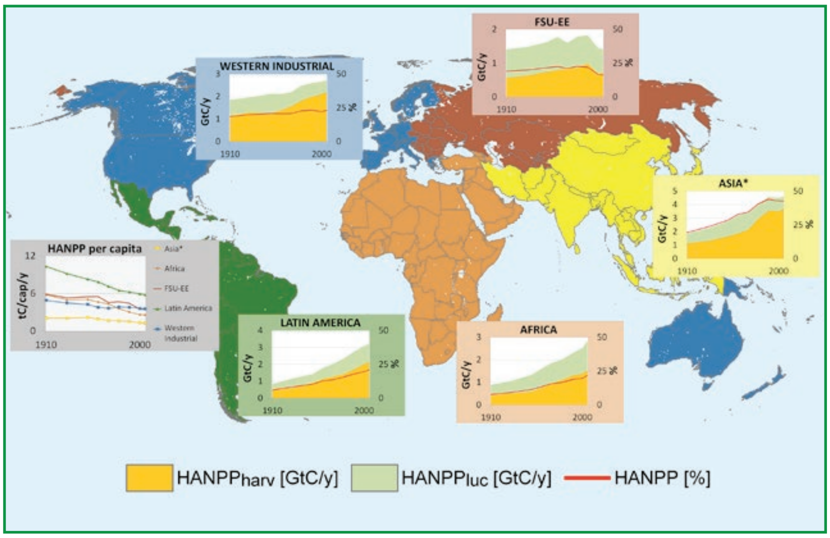


Figure 4.4. HANPP and HANPP per capita from 1910 to 2005 from five world regions (FSU-EE = Former Soviet Union and Eastern Europe; Western Industrial = North America, Europe and Oceania). $HANPP_{harv}$ is the NPP harvested by humans, and $HANPP_{luc}$ represents the differences between the NPP of the vegetation under actual land uses and the NPP if it had remained native vegetation not influenced by management practices. Source: Krausmann *et al.* (2013).

Asia is the world region with the highest increase and the highest values of HANPP in 2005, with approximately 47% of NPP_0 being appropriated by humans (Figure 4.4). In all other world regions, the increase in HANPP throughout the last century was not as fast, in the former Soviet Union and Eastern Europe the percentage of NPP_0 appropriated by humans has decreased since 1990, reflecting the disintegration of the agricultural production system after the collapse of the Soviet Union (Krausmann *et al.*, 2013). In all regions (except Asia), HANPP was below 25% in 2005. Despite the general increase in the rate of HANPP, per capita HANPP has been decreasing in all world regions since 1910 (Figure 4.4). Since biomass consumption is almost in perfect correlation with population growth (Krausmann *et al.*, 2013), the global decrease in per capita HANPP indicates an increase in the efficiency of biomass production in relation to NPP_{pot}, whereas regional differences reflect the amount and type of biomass consumed and the net balance of biomass imports and exports. Although it is clear that HANPP provides an idea of the human domination of the biosphere a decrease in HANPP does not necessarily reflect moving towards a more sustainable situation. For example, increases in NPP_{act} can be reached due to use of fertilisers, resulting in a lower HANPP.

The HANPP has been suggested as an indicator to measure the pressures of the socio-economic activity in biodiversity (Haberl *et al.*, 2007b), as it keeps track of one of the most important drivers of biodiversity loss: land-use change. Although some studies have explored the relationship between biodiversity loss and HANPP, this still needs to be better understood (Haines-Young, 2009). Haberl *et al.* (2004b) analysed the relationship between HANPP and the species diversity of autotrophs and heterotrophs in agricultural landscapes in Austria; they found that HANPP (%) was inversely correlated with the species diversity. Haberl *et al.* (2005) analysed NPP_{act}, NPP_{eco}, $HANPP_{harv}$ and HANPP (%) as potential determinants of bird species diversity. NPP_{act} revealed to be the best predictor of bird species richness and was inversely correlated with the percentage of endangered species. Recently, Dullinger *et al.* (2013) studied Europe's extinction debt and showed that historical data of HANPP (and other socioeconomic indicators) are better correlated with the proportion of threatened species than the current values of the same indicators. Despite these indications on the relationship between HANPP and biodiversity, it is important to mention that HANPP does not account for the qualities of the primary productivity appropriated (Smil, 2011). For example, harvesting food crops on land that has been cultivated for centuries is clearly a different appropriation from cutting down a forest stand in a biodiversity hotspot.

4.1.1.iii Water footprint

The water footprint (WF) was developed by Hoekstra and Hung (2002). It measures all the freshwater required for the production or consumption of a country and the freshwater required to assimilate the pollution load. Since 1995, water consumption has increased approximately 30%, but the relative shares of the water footprint's components have remained the same (Arto *et al.*, 2012). Hoekstra and Mekonnen (2012) determined that the global annual average WF, for the period 1996-2005 was 9087 billion m^3. Of this total, 74% concerned the consumption of green water (rainwater stored in the soil, mainly important for agriculture), 11% blue water (surface and ground water), and finally 15% grey water (for the assimilation of pollutants). Sector wise, agriculture alone appropriated 92% of global WF, industrial production 4.4% and domestic water supply 3.6%. Globally, irrigation for agriculture was responsible for 90% of the consumption of blue water, also irrigation is responsible for 70% of global water withdrawals (Siebert and Döll, 2010).

China, India and the United States are the regions with higher WFs of production (Figure 4.5A), being responsible for 38% of global WF. However, the WFs of consumption (Figure 4.5B) of China and India, in per capita terms, are amongst the lower values. Niger, Bolivia and Mongolia have the higher WFs of consumption mainly due to the green WF (Hoekstra and Mekonnen, 2012).

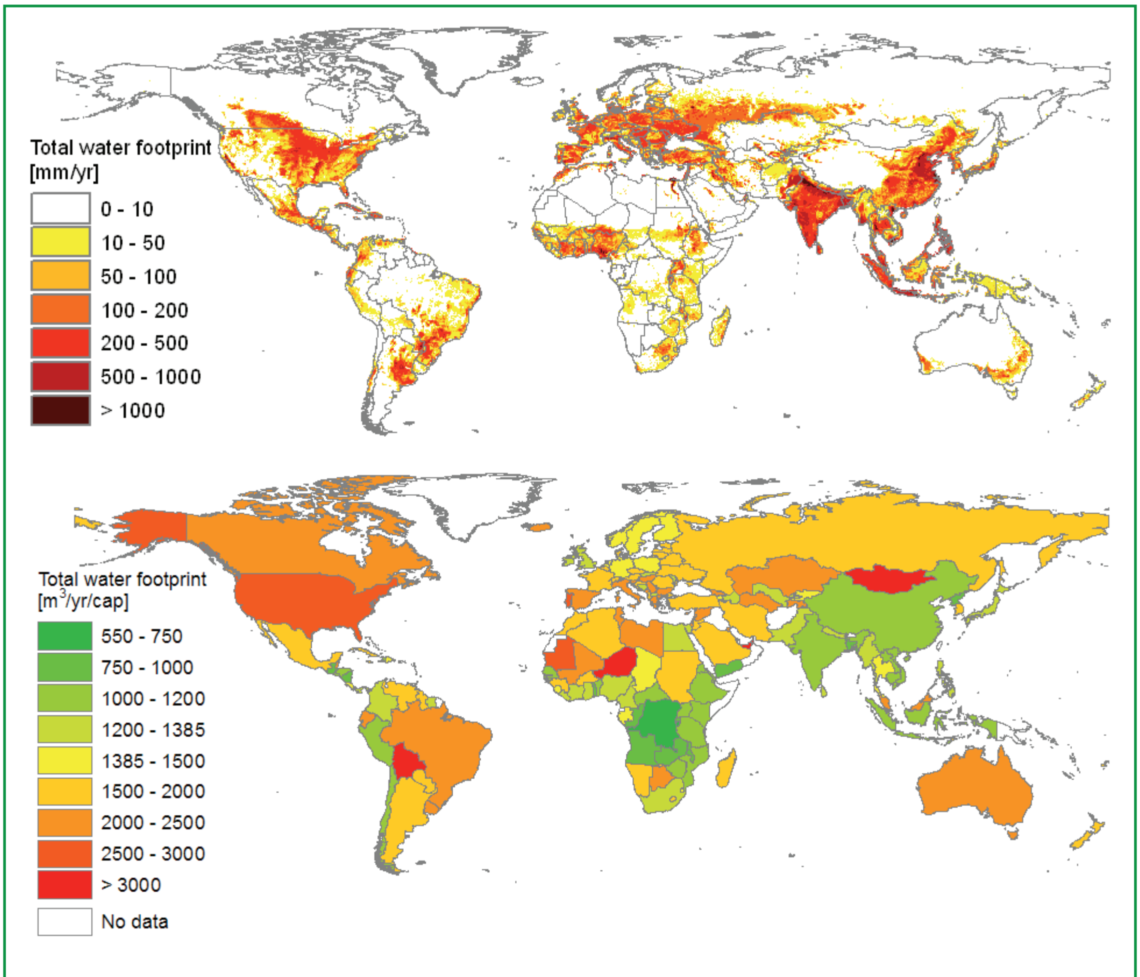


Figure 4.5. A) Global water footprint of production (mm/y on a 5'X5' grid), B) Global per capita water footprint of consumption (m^3/y/cap). Source: Hoekstra and Mekonnen (2012).

Freshwater ecosystems are amongst the most altered, because of water flow modifications and water consumption (Millennium Ecosystem Assessment, 2005), and are probably the ecosystems where more competition for resources, between humans and the rest of the biosphere, exists. Such alterations and competition have obvious consequences for freshwater biodiversity. The Freshwater Living Planet Index shows the greatest decline amongst all the biomes analysed, 37% between 1970 and 2008 (WWF, 2012). The main threats to freshwater biodiversity are overexploitation, water pollution, flow modification, destruction or degradation of habitat and invasion by exotic species (Dudgeon *et al.*, 2006).

Vörösmarty *et al.* (2010) performed a global analysis of threats to freshwater, taking into consideration both the water security and biodiversity dimensions. According to their results, 80% of the world's population lives in areas where incident biodiversity and human water security threats exceed the 75th percentile. Figure 4.6 shows the geographical distribution of both threats and unravels their close relation. In regions like the United States, Europe, China and India, the high threat incidence is also geographically related with high water footprints (Figure 4.5A). In areas of high incident threats to human water security and biodiversity, the main drivers for both are the same: water resource development and pollution and watershed disturbance and biotic factors, the latter are more relevant for biodiversity (Vörösmarty *et al.*, 2010).

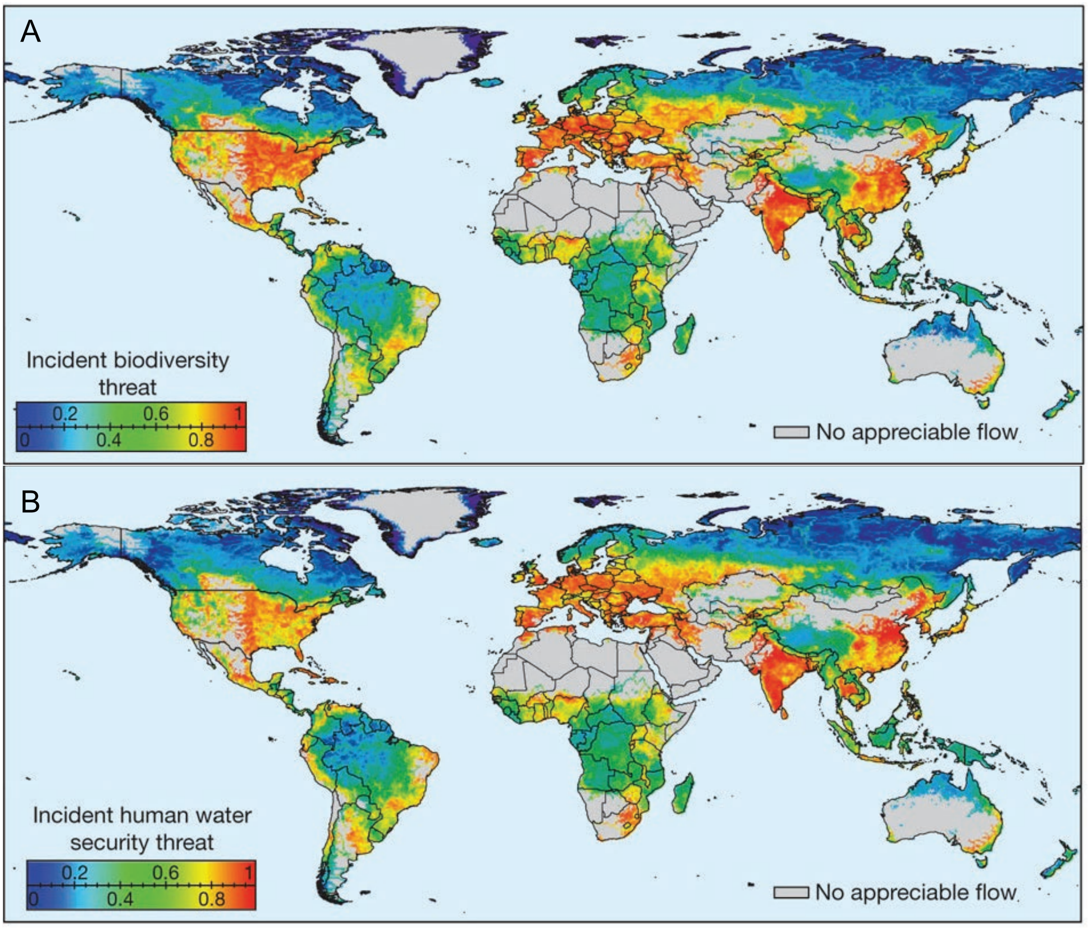


Figure 4.6. Global geography of incident threat to A) biodiversity and B) human water security. Source: Vörösmarty *et al.* (2010).

4.1.1.iv Primary Production Required to sustain the marine fisheries catch (PPR)

The Primary Production Required to sustain the fisheries catch (PPR) is obtained by back-calculating the fisheries flows, and is expressed in primary production and detritus equivalents for all pathways from the exploited species captured in the catch down to the primary producers and detritus (Pauly and Christensen, 1995). This index can be expressed per unit of fishery catch relative to primary production and detritus of the ecosystem required to sustain that catch (%PPR). Total catch has increased exponentially since 1950s to 1990s and since then it has been fluctuating (Coll *et al*., 2008; Pauly *et al*., 2002). The %PPR has also shown a geographic expansion from the northern countries to the global ocean and also in depth (Swartz *et al*., 2010, Figure 4.7).

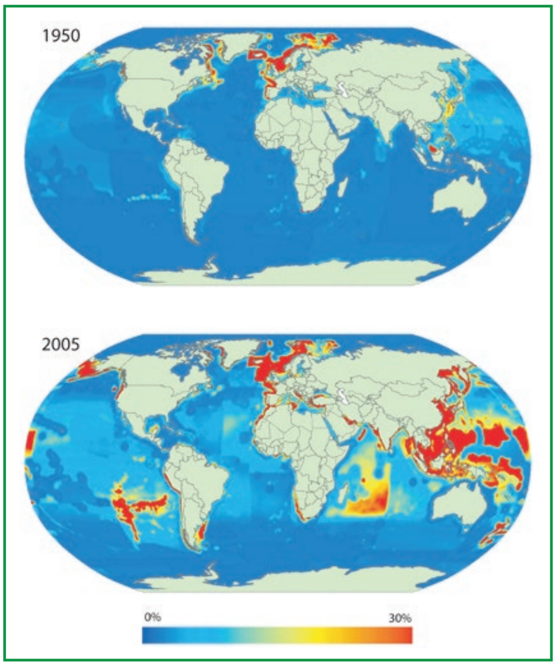


Figure 4.7. Primary Production Required (PPR) to sustain global marine fisheries landings, expressed as percentage of local primary production for A) 1950 and B) 2005.
Source: Swartz *et al.* (2010).

Associated to PPR calculations, a measure of the sustainability of fishing activities linked with the production lost from the oceans when fishing showed that fishing has gone less sustainable with time (Coll *et al*., 2008). In 1950, the probability of a fish catch being occurring sustainably in a certain geographical area was higher than 75% (Figure 4.8). Through time, this probability has decreased considerably. In 1990, in certain areas of the large marine ecosystem, especially in the Eastern China coast, the probability of sustainable exploitation was close to 0% (Figure 4.8). Globally, in 2004 the probability of having areas of sustainable exploitation of the oceans was lower than 60% (Coll *et al*., 2008).

Current and future trends of exploitation of marine resources poses also important impacts to marine biodiversity worldwide (Costello *et al*., 2010; Worm *et al*., 2006) (Chapter 6 provides an in depth analysis of fisheries and sustainability). This is especially relevant for larger organisms (Christensen *et al*., 2003; Lotze and Worm, 2009), but also to smaller organisms (Anderson *et al*., 2011). The ecosystem-based approach to fisheries process (Christensen and Maclean, 2011; Link, 2011; Pikitch *et al*., 2004) argues for a change in the way fishing is developed. It requires a change of vision to optimise and reconcile the exploitation of marine resources and conservation of diversity and ecosystem services. This approach is still in its infancy globally, but regional case studies are promising. For example, when countries work towards sustainability of fishing by following the Code of Conduct for Responsible Fisheries developed in 1995 by the Food and Agriculture Organization (FAO) of the United Nations, which includes a set of recommendations for reducing the negative impacts of fishing activities on marine ecosystems, the sustainability of fishing activities increases (Coll *et al*., 2013). This can have a positive impact on the quantity of fishing products and on exploited marine ecosystems.

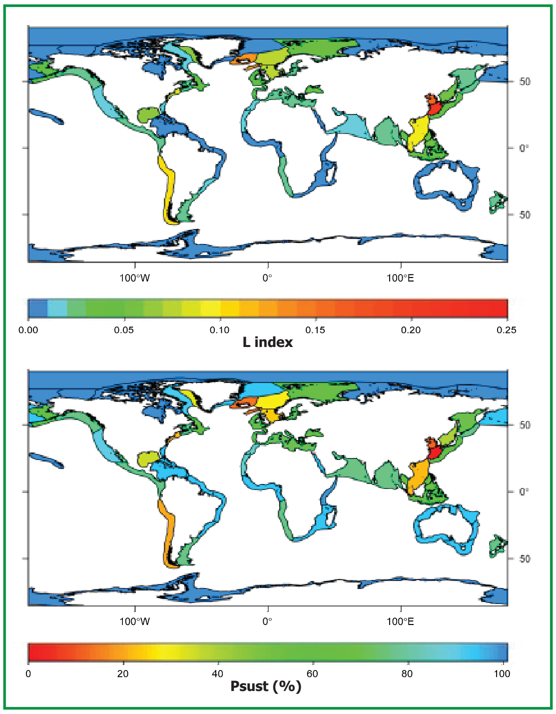


Figure 4.8. Historical ecosystem overfishing assessment for large marine ecosystems. Psust (%) is the probability of having sustainable fishing occurring in a certain area during the A) 1950s and B) 1990s. Source: Adapted from Coll *et al*. (2008).

4.1.1.v Impacts of utilisation on the status of species

People depend upon biodiversity and use wildlife in a variety of ways. Birds, mammals and amphibians are hunted, trapped and collected for food, sport, pets, medicine, materials (e.g. fur and feathers) and other purposes. The Red List Index for birds, mammals and amphibians showing trends in survival probability driven by utilisation illustrates the changing status of these species groups, owing to the balance between negative trends driven by unsustainable exploitation, and positive trends driven by measures to reduce overexploitation. It excludes changes in status driven by other factors (such as habitat loss or climate change).

For all three groups, there is a small decline in RLI (Figure 4.9). The RLI reveals the trends in the overall extinction risk of species, a decreasing RLI means that the rate of extinction is expected to increase, whereas a flat RLI means that the rate of extinctions is expected to remain relatively unchanged. Hence, a small decline in RLI indicates that overall levels of utilisation are unsustainable. Many species are now threatened with extinction owing to over-exploitation. It is likely that the results for these groups will be mirrored for other wildlife groups once data become available.

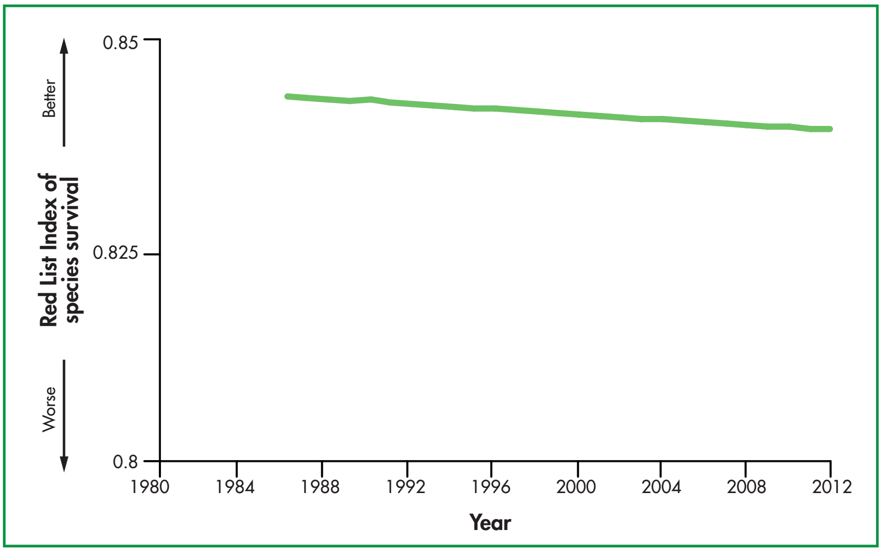


Figure 4.9. Red List Index for all birds (9861 species), mammals (4350 species) and amphibians (4554 species) showing trends in status driven by utilisation or its control; declines indicate that unsustainable exploitation is driving these species ever-faster towards extinction. Source: BirdLife International and IUCN.

Lenzen *et al.* (2012) provided an insight on the implications of consumption patterns to threats to biodiversity. They found that approximately 30% of species threats were driven by international trade, and mainly exerted by consumption patterns of developed economies in developing economies (Table 4.2).

Table 4.2. Five top-ranking internationally traded commodities (causes), by country of production and final consumption. Source: Adapted from Lenzen *et al.* (2012).

Threat Suffered due to Production in	Driven by Consumption in	Commodities/Threat Causes
papua New Guinea	Japan	Agricultural products
Malaysia	Singapore	Biological resource use, pollution
China	USA	Pollution from manufacturing
Mexico	USA	Coffee and tea
Canada	USA	Forestry, agriculture, grazing, pollution from manufacturing and mining

4.1.2 Projecting forward to 2020

Until 2020, it is expected that economic development and population growth continue to drive the increasing consumption of natural resources (Figure 4.10A). As income rises, the demand for processed food, meat, dairy and fish increases, exerting additional pressure in food producing systems and more specifically on land (Godfray *et al.*, 2010).

In per capita terms, the Ecological Footprint (EF, without Carbon Footprint; CF) and HANPP show a decreasing trend that is likely to continue until 2020 (Figure 4.10B). This result indicates a decrease in the appropriation of land by each person, which could be a result of efficiency gains in the agricultural sector. However, Krausmann *et al.* (2013) show that the decrease in per capita HANPP is mainly a result of a decrease in the consumption of biomass for energy purposes. While the EF without the CF has decreased, the EF with the CF remains stable. This comparison highlights the fact that decreased demand for cropland due to agricultural activities has only been possible due to industrialisation of agricultural practices, hence contributing to a higher carbon footprint (more energy inputs and fertilizers)[9].

The intensity of resource use (quantity of resource per monetary unit) has been decreasing, and short term projections indicate that it will keep decreasing, except for water, whose intensity seems to have stabilised (Figure 4.10C). Until 2020, efficiency improvements are likely to be over-compensated by economic and population growth indicating that the main goal of sustainable production and consumption, which is decoupling economic growth from resource use, has not been achieved.

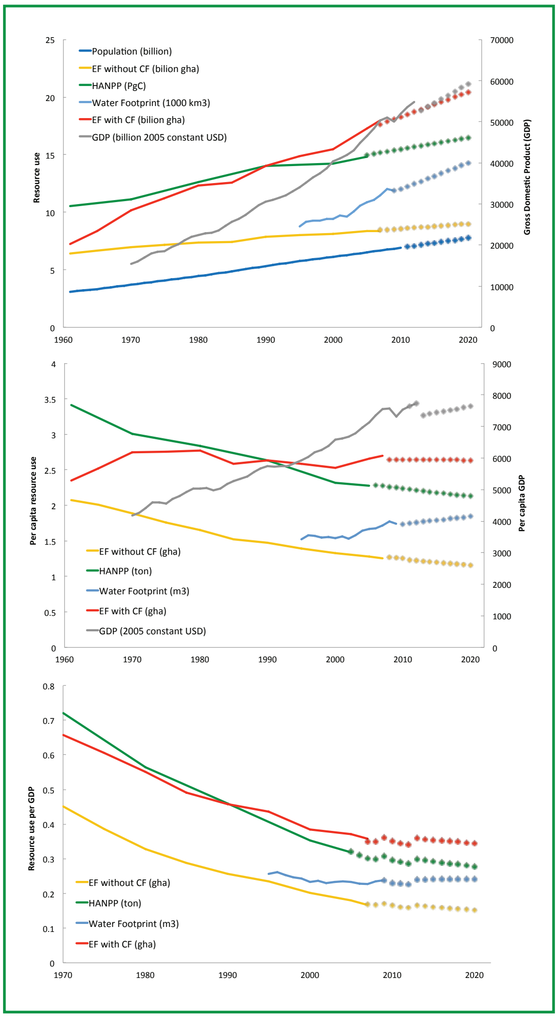


Figure 4.10. A) Extrapolations of currents trends of population, Gross Domestic Product (GDP) (secondary axis), Ecological Footprint (with and without the Carbon Footprint component), Water Footprint and Human Appropriation of Net Primary Production; B) per capita extrapolations of current trends of GDP (secondary axis), Ecological Footprint (with and without the Carbon Footprint component), Water Footprint and Human Appropriation of Net Primary Production, C) extrapolations of currents trends of intensity resource use of Ecological Footprint (with and without the Carbon Footprint component), Water Footprint and Human Appropriation of Net Primary Production intensities (resource use per unit GDP).

Source: UN (2013a), for Population; UN (2013b), for GDP; Global Footprint Network (2012), for Ecological Footprint Data; Krausmann *et al*. (2013), for Human Appropriation of Net Primary Productivity; Arto *et al*. (2012), for Water Footprint. Extrapolations were performed applying linear regressions on current trends.

Footnote

[9] *Other emissions then the ones associated with agricultural practices are included in the carbon footprint component of the ecological footprint.*

Box 4.2: Case Study: Ecological Footprint of China

China is the most populated country in the world and one of the countries with the highest growth rates. China's economic growth has induced structural changes in the economy, has improved the quality of life of their population and has been inducing lifestyles changes (Chen *et al.*, 2007; Galli *et al.*, 2012b; Gaodi *et al.*, 2012; Hubacek *et al.*, 2009). Such progresses have unavoidable repercussions on the environment; the per capita EF of China increased from 0.95 global hectares (gha) in 1961 to 2.1 gha in 2005 (still below the world average of 2.6 gha per capita). The main changes occurred at the carbon footprint level; in 1961 it represented 7% of the global per capita footprint whereas in 2005 it represented 54% (Chen *et al.*, 2007; Galli *et al.*, 2012b; Gaodi *et al.*, 2012).

Hubacek *et al.* (2009) estimated the EF China for 2020 taking into account population, income growth, urbanisation and lifestyle changes as well as structural economic changes (increased efficiencies[10] in agricultural, industrial, transportation and communications and in the services sectors), their projections indicated that in 2020 the national per capita EF would be 2.1 gha. This value has been reached in 2005, between 2000 and 2005 the increase in the chinese per capita EF was higher than in any other period (Galli *et al.*, 2012b; Gaodi *et al.*, 2012). In 2020, and considering that China seems to be following a development pattern similar to developed countries with high resource use (Galli *et al.*, 2012b) it is likely that the per capita EF will be much higher. China's per capita footprint is well related with urbanisation rates (Gaodi *et al.*, 2012). In Beijing, it was estimated that the per capita EF would almost double 2005 values reaching 9.93 gha in 2020 (Hubacek *et al.*, 2009). These results highlight the urgency of implementing measures to change consumption and production patterns in China. These actions should include raising public awareness for the need of sustainable consumption patterns, specially in urbanised areas; promote sustainable construction and efficient buildings, expansion and improvement of the public transportation system and optimisation of the industrial structure (Chen and Lin, 2008; Gaodi *et al.*, 2012; Hubacek *et al.*, 2009).

In an effort to assess the sustainability of the levels of consumption in 2020, we compared the projected values of the indicators analysed with indicative ecological safe limits established for each of the indicators boundaries (Table 4.3). By 2020, it is projected that humanity will have reached critical boundaries for at least two indicators. The EF exceeds biocapacity by 8 billion gha, due to increasing anthropogenic carbon emissions that cannot be assimilated by the sinking capacity of forests. Other studies also show that carbon sinks cannot assimilate the total of carbon emissions generated due to human activities (Canadell *et al.*, 2007; Le Quéré *et al.*, 2009). A similar perspective is shown for the levels of nitrogen and phosphorous in major rivers around the world; water requirements for pollution assimilation may compromise the water availability for human consumption (Canfield *et al.*, 2010; Liu *et al.*, 2012).

Table 4.3. Comparison between levels of consumption in 2020 and the ecological safe limits established for each of the indicators.
Source: [a]Global Footprint Network (2012) and WWF (2012); [b]Gerten *et al.* (2013) [c]Running (2012) and Smith *et al.* (2012).

INDICATOR	USE BY 2020	AVAILABILITY
Ecological Footprint (without CF)	9 (billion gha)	12 (billion gha)[a]
Ecological Footprint (with CF)	20 (billion gha)	12 (billion gha)[a]
Blue Water Footprint (not shown)	2300 (km^3 y^{-1})*	1100 - 4500 ($km^3 y^{-1}$)[b]
HANPP	17 GtC **	25 GtC[c]

** Projected values might be under estimated, in UNEP (2012b) current consumption of blue water is shown as 2600 km^3.*

*** Projected values might be underestimated, in Running (2012) current values of HANPP reached 20 GtC.*

Footnote

[10] Efficiencies in the resource use refer to the use of resource per unit of total economic output. Improved efficiencies mean that more economic output is produced using fewer resources.

By 2020, the Blue Water Footprint is expected to reach by 2300 km3y-1 (data not shown in Figure 4.10). Such value is likely to be greatly exceeded as some studies present the current freshwater consumption level at 2600 km3y-1 (Rockström *et al.*, 2009). Taking into consideration the interval of freshwater consumption's safe ecological limit is 1100-4500 km3y-1, it is possible that by 2020 this has been exceeded globally. Smakhtin *et al.* (2004) analysed water stress at the global level, taking into consideration not only water availability and total use but also the environmental water requirements. They showed that basins where water use was in conflict with environmental water requirements covered 15% of world land surface. More recently, Hoekstra *et al.* (2012) analysed the water scarcity in 405 river basins taking into consideration the environmental water requirements (Figure 4.11). Twelve of the river basins analysed are continuously under water scarcity. But the majority of the basins, face water scarcity between 9 and 2 months per year.

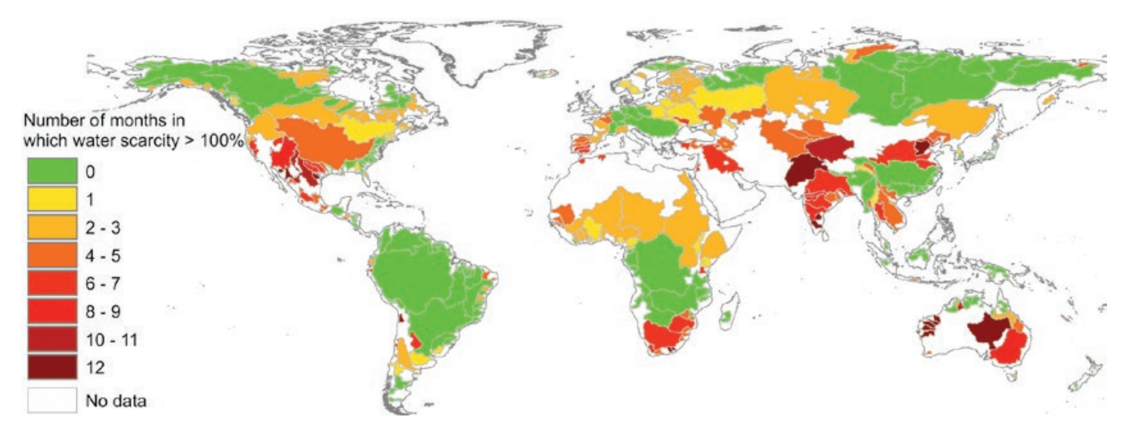


Figure 4.11. Number of months of water scarcity in world's major river basins, based on the period of 1996-2005. Water scarcity occurs when blue water footprint exceeds blue water availability. Source: Hoekstra *et al*. (2012).

By 2020, HANPP is projected to reach 17 GtC (Figure 4.10). However some studies place current HANPP at 20-24 GtC (Bishop *et al.*, 2010; Running, 2012). Having these differences into consideration, it is likely that by 2020 HANPP has exceeded the critical boundary of 25GtC. Bishop *et al.* (2010) estimated that, in order to minimise the changes to natural biomes, humans must reduce their HANPP to 9.72 GtC per year. This would imply not only reductions in consumption, but also changes in the ways human appropriate natural productivity. Apart from strict dematerialisation changes can include extension of product life, re-using and recycling (Bishop *et al.*, 2010).

Irrespective of the ecological limits considered, the effect of current and future values of consumption on nature are difficult to determine. Tipping points (critical thresholds) are still difficult to determine and identify. Reaching a tipping point might mean an irreversible regime shift with potentially dramatic consequences for biodiversity, ecosystems services and human well-being (Barnosky *et al.*, 2012; Leadley *et al.*, 2010; Scheffer *et al.*, 2012; Steffen *et al.*, 2011).

In the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), species are grouped according to their degree of protection. Appendix 1 includes species threatened with extinction; trade is permitted only in exceptional circumstances. In CITES, countries are classified according to the alignment of their national legislation to meet the requirements for implementation of the Convention. Category 1 countries are those who legislation is believed to meet the requirements for implementation of CITES.

The projected increase in the number of countries classified as Category 1 in CITES shows a continually improving commitment from the international community to ensuring that international trade in specimens of wild animals and plants does not threaten their survival (Fig 4.12 – left). By 2020 it is projected that approximately two thirds of the Parties of CITES will have introduced legislation that will meet the requirements for implementation of CITES. Appendix 1 listings are expected to increase, which indicates that more species that are internationally trade will be threatened by extinction. However, the rate of increase seems to be decelerating (Figure 4.12 – right).

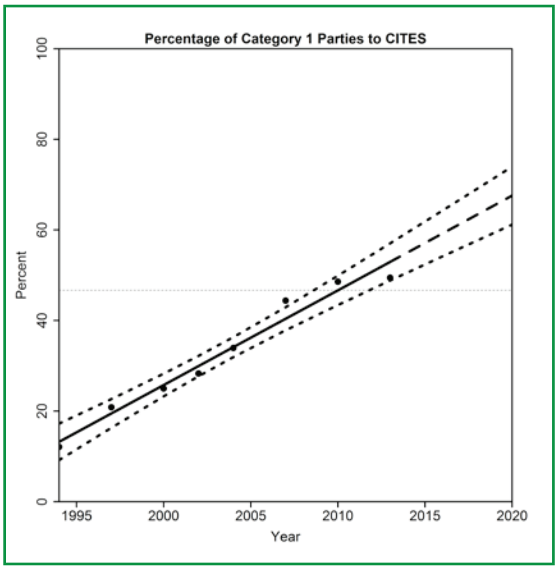


Figure 4.12. Statistical extrapolation of the number of percentage of Category 1 nations in CITES (left) and CITES appendix 1 listings (right). Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant.

4.1.3 Country actions and commitments[11]

Most countries have established targets or similar measures related to the promotion of sustainable consumption and production (high). These targets are generally inline with the Aichi Biodiversity Target (high) and if achieved would make a significant contribution towards the achievement of this target.

The national targets that have been set tend to focus on supporting sustainable production (medium). For example Serbia, in its National Biodiversity Strategies and Action Plan (NBSAP), has set an objective related to developing and strengthening mechanism to ensure the sustainable use of biodiversity. There has been comparatively less emphasis on issues related to sustainable consumption. Two examples, which are counter to this general trend, are Brazil and Finland, which have both established targets, which directly refer to promoting sustainable consumption.

A number of countries have addressed specific issues that are not directly referred to in Aichi Biodiversity Target 4 but would none the less make contributions towards its attainment. Some countries have chosen to focus their attention on the impacts of specific sectors. For example Suriname has established an objective related to the promotion of responsible tourism and Japan has established a key action goal on the development of policies specifically related to sustainable business activities.

4.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

4.2.1 Actions

Promoting sustainable consumption and production (SCP) is a complex task and requires actions a combination of supporting policies, technological innovations and lifestyle changes (UNEP, 2012a). Lebel and Lorek (2008) point out some challenges associated with moving towards sustainable production and consumption. For example, efforts to tackle environmental problems in one place might shift them somewhere else; this may also represent an increase of the environmental problem if technologies available in the displacing country are more efficient (an example is the carbon leakage phenomena). Efforts to reduce environmental impacts of making products can be overcompensated by a net growth in the demand for those products, or if their end of life generates more environmental impacts. Also changing lifestyles towards sustainable development may be difficult when marketing urges people to consume.

Footnote

[11] This assessment is based on an examination of the NBSAPs from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPS and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.

Several actions on SCP already exist, however the lack of integration into coherent policy frameworks and strategic plans might impair their effectiveness in contributing towards sustainable development. Thus, it is recommend that at the country and international level, all stakeholders work together towards an integrated SCP action plan (UNEP, 2012a). Also, countries should seek to gather more data and establish harmonised sets of indicators to measure effectiveness and track progress of SCP policies (UNEP, 2012a). Government cooperation between businesses and industries can harness market forces to drive the shift to SCP. Governments should develop incentives to encourage business investments in SCP (UNEP, 2012a). Such incentives can have the form of loans or financial assistance, or sustainable public procurement policies (see Box 4.1; OECD, 2008; UNEP, 2012a). Another action towards the achievement of SCP regards the establishment of national standards or mandatory labels to limit environmental damages of a certain production process or product, fostering innovation and promoting greener supply chains (Lebel and Lorek, 2008; OECD, 2008; UNEP, 2012a). Actions should also focus not only on the production side, but also on the demand side raising awareness on people about their impact on the environment (Lebel and Lorek, 2008; UNEP, 2012a). One of the most pressing issues in SCP is waste; solid waste production has risen tenfold in the past century, and it is likely that it continues to increase in the next century (Hoornweg *et al*., 2013).

Reducing or minimising the direct and indirect pressures on biodiversity and ecosystem services is the first step to promote their sustainable use (SCBD, 2011). This is the goal of the Strategic Goal B of the Aichi Targets (see Targets 5, 6, 7, 8, 9 and 10). Table 4.4 presents a summary of actions to decrease the pressures on biodiversity and ecosystems.

Table 4.4. Example of actions needed to achieve Strategic Goal B.

AICHI TARGET 5	
By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.	Increase productivity of converted lands [through culturally and ecologically appropriate means] and restrict further [industrial] agricultural expansion. Restrict infrastructure expansion. Change consumption patterns (less demand of lad-related products). Involve local communities in management. Stop illegal timber harvest and trade.
AICHI TARGET 6	
By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits.	Eliminate harmful subsidies. Stop Illegal, Unreported, and Unregulated Fishing. Eliminate destructive fishing gears and practices. Precautionary approach to prevent overfishing. Support Indigenous peoples' and local communities' conserved territories and areas in river, coastal and marine ecosystems, including customary and subsistence-based fishing practices (such as satoumi in Japan and tagal in Malaysia)".
AICHI TARGET 7	
By 2020 areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.	Production certification. Reduce post-harvest losses and minimise food waste. Change consumption patterns, promoting sustainable diets. Support Indigenous peoples' and local communities' conserved territories and areas, particularly low-impact and subsistence-based livelihood strategies.
AICHI TARGET 8	
By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.	Uncrease nutrient efficiency. Decrease manure production. Reduce nutrient loss.

AICHI TARGET 9	
By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.	Develop indicators of invasions. Standardisation of terminology. Control pathways of introduction (for example, ornamental plants, pets, ballast water).
AICHI TARGET 10	
By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning.	Reduce fishing effort. Mitigation of pollution and land-based activities on coral reefs. Development of marine ecotourism. Support Indigenous peoples' and local communities' conserved territories and areas in coastal and marine and other vulnerable ecosystems, including customary and subsistence-based fishing practices.

4.2.2 Costs and Cost-benefit analysis

A green economy is a resource efficient economy, that invests in natural capital and promotes human well-being (UNEP, 2011). Promoting SCP is the first step for the establishment of green economies. Investing in greening key sectors, like agriculture, fisheries, water and forestry requires a small cost when compared to long-term benefits (UNEP, 2011). The aggregate global cost required for the transition towards green agriculture was estimated to be US$198 billion per year (between 2011-2050), and represents an yearly increase in value added of about 9%, and additional 47 million additional jobs in comparison with the business as usual scenarios (UNEP, 2011). Greening the fisheries sector could increase resource rents from negative US$26 to positive US$45 billion a year, under such scenario fisheries value added was estimated to be US$67 billion a year (UNEP, 2011). Regarding the water sector, an annual investment of US$198 billion per year (between 2011-2050) water use can become more efficient and consequently increase agricultural and industrial production. Under such scenario, by 2050 7% less people would live in water stressed regions in comparison with the business as usual scenario (UNEP, 2011). Investing US$40 billion in reforestation per year (between 2011-2050) could raise the value added in forest by 20%, and the carbon stored by 28%, in comparison with business as usual (UNEP, 2011).

At the national level, achieving Target 4 will require domestic studies focusing on key impacts of consumption and production patterns on biodiversity, in order to identify priorities for action and the potential role of different actors in the public and private sectors. These national studies are expected to require up to US$19.5 million (SCBD, 2012), globally. Finally, national action plans should be developed to ensure that national production and consumption is kept within safe ecological limits; in this regards the High-level Panel stress the role of governments in developing public procurement strategies in line with SCP objectives. The approximate initial investment required for these actions is expected to be between US$44 and 85 million, with recurrent yearly expenditures of US$8 to 15 million (SCBD, 2012).

4.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

Biodiversity status has been declining (Butchart *et al.*, 2010). The main identified drivers of this decline are habitat loss and degradation driven by unsustainable agriculture, infrastructure development and other factors, overexploitation, pollution, biotic change, invasive alien species and climate change. All these drivers result from the human domination and exploitation of the biosphere for satisfying consumption needs. The projected increases in the indicators analysed are in line with other projections from the literature, and suggest that, despite of efficiency gains, decoupling economic growth from environmental pressure as not yet occurred.

As a result, pressures on ecosystems and biodiversity will continue. As "*…it is only possible to reduce or halt the loss of biodiversity if the drivers and pressures on biodiversity are themselves reduced or eliminated*" (SCBD, 2011), barring major transformations in patterns of production and consumption, *it is likely that we will* continue to exacerbate the decline of biodiversity status until or beyond 2020 for the majority of ecosystems (see Targets 5, 6, 7, 8, 9, 10, 11, 12, 13 and 14).

4.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

Scenarios for 2050 of the Ecological Footprint (EF), Human Appropriation of Net Primary Production (HANPP) and Water Footprint (WF) are available. All scenarios project an increase in the human dominance of the biosphere in 2050, unless strong measures are put in place (Ercin and Hoekstra, 2014; Krausmann *et al*., 2013; Moore *et al*., 2012).

Moore *et al*. (2012) indicate that, following a business as usual path, humanity's Ecological Footprint (EF) is projected to increase to over 31 billion gha (3.4 gha per capita) by 2050. Alternative scenarios showing a reduction of the Footprint until 2050 require a stabilisation of the emissions at 50% of 2005 levels by 2050 (IEA's BLUE map scenario), a low population growth (UN, 2008) and a change in dietary patterns towards less meat (Moore *et al*., 2012). For this scenario, humanity would require 0.9 Earths to meet their consumption and assimilation of emissions (Moore *et al*., 2012). These scenarios suggest that in order to meet a minimum sustainability (EF equal or smaller than biocapacity) aggressive goals are required for each of the drivers. However, if only the environmental goal of emissions stabilisation at 50% of 2005 levels is kept projections show a stabilisation of the EF at about 1.5 Earths (Figure 4.13).

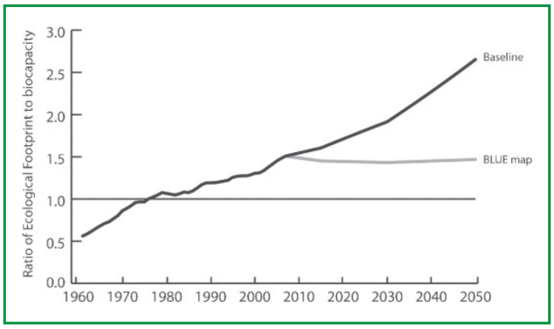


Figure 4.13. Baseline and BLUE map Ecological Footprint scenarios until 2050. Source: Moore *et al*. (2012).

In HANPP projections for 2050 all scenarios show an increase in total HANPP (Krausmann *et al*., 2013, Figure 4.14). The scenario with a lower increase in HANPP (Scenario C), the assumption underlying this scenario are a constant global average biomass consumption of 0.3tC/cap/yr; a decrease in average HANPP per unit of final biomass consumption of 26%, reflecting gains in efficiency (Krausmann *et al*., 2013). Scenarios with higher HANPP values for 2050 reflect different choices of bioenergy scenarios (Scenarios D and E). Scenario D considers that additional 50 EJ/y over the present level of 50EJ/y will come from bioenergy, Scenario E considers additional 250Ej/y. For the both scenarios 60% of the additional bioenergy will be supplied by agriculture and 40% by forest, also in both scenarios increases in production efficiencies are taken into account.

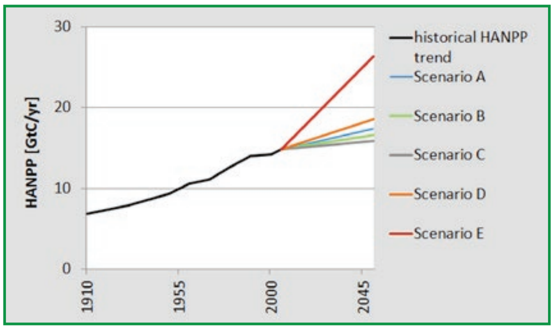


Figure 4.14. Scenarios for the development on HANPP until 2050, under 5 different scenarios. Source: Krausmann *et al*. (2013).

Biofuels are seen as one of the most promising alternatives for fossil fuels, in fact the supply of energy from biofuels was amongst all other renewable energy sources the one that grew the most (IPCC, 2011). However, many biofuels actually increase GHG emissions in comparison to fossil fuels when direct and indirect land use change as well as nitrous oxide emissions are accounted for (Gibbs *et al*., 2008; Hertel *et al*., 2010; Plevin *et al*., 2010; Searchinger *et al*., 2008). The increased demand in bioenergy, especially due to increased biofuel consumption, has raised concerns about how much land can be used to produce both food and energy purposes for an increasing population with greater energy demands.

Bioenergy potentials have great ranges. Dornburg *et al*. (2010) performed a sensitivity analysis based on water limitations; protected areas and food demand and established a range between 200-500 EJ/y. This would require the conversion of natural areas to croplands or a great increase in yields, or both (Haberl *et al*., 2013). However, current yield trends will be insufficient to meet food demand in areas currently designated to food production (Ray *et al*., 2013), as a result land conversion for bioenergy production would compete with land conversion for food production.

Recently, Haberl *et al*. (2013) estimated that the maximum physical potential of the world's area outside croplands, infrastructure, wilderness and denser forest to deliver bioenergy would be at approximately 190 Ej/y. Another study examined the consequences of different conservation scenarios for the global potential of bioenergy (Erb *et al*., 2012). Under a less restrictive scenario, wilderness areas with two or more wilderness parameters were excluded for bioenergy production, areas with 2 or more biodiversity hotspots were partially excluded, and the area availability for protected areas was reduced by 80%. This resulted in a 9% reduction of the global bioenergy potential. Under a more restrictive scenario, wilderness areas were excluded for biooenergy production, all land areas suffer a decrease in available energy crop area by 10%, areas with biodiversity hotspots were partially excluded and protected areas excluded completely. This resulted in a 32% reduction of the global bioenergy potential.

In Water Footprint (WF) projections for 2050, all scenarios show an increase in total WF (Ercin and Hoekstra, 2014; Figure 4.15). The scenario with a lowest increase in WF is based in IPCC's storyline B1, a low population growth, change in dietary patterns towards less meat, a biofuel expansion, technology and efficiency improvements regarding water usage and trade liberalisation. In this scenario the green share of the WF decreases, whereas blue and grey shares increase. Under this scenario the WF associated with agricultural products as the smallest increase when compared to 2000 levels (18%). However, water footprints are very sensitive to each driver and the projections associated with each scenario vary significantly.

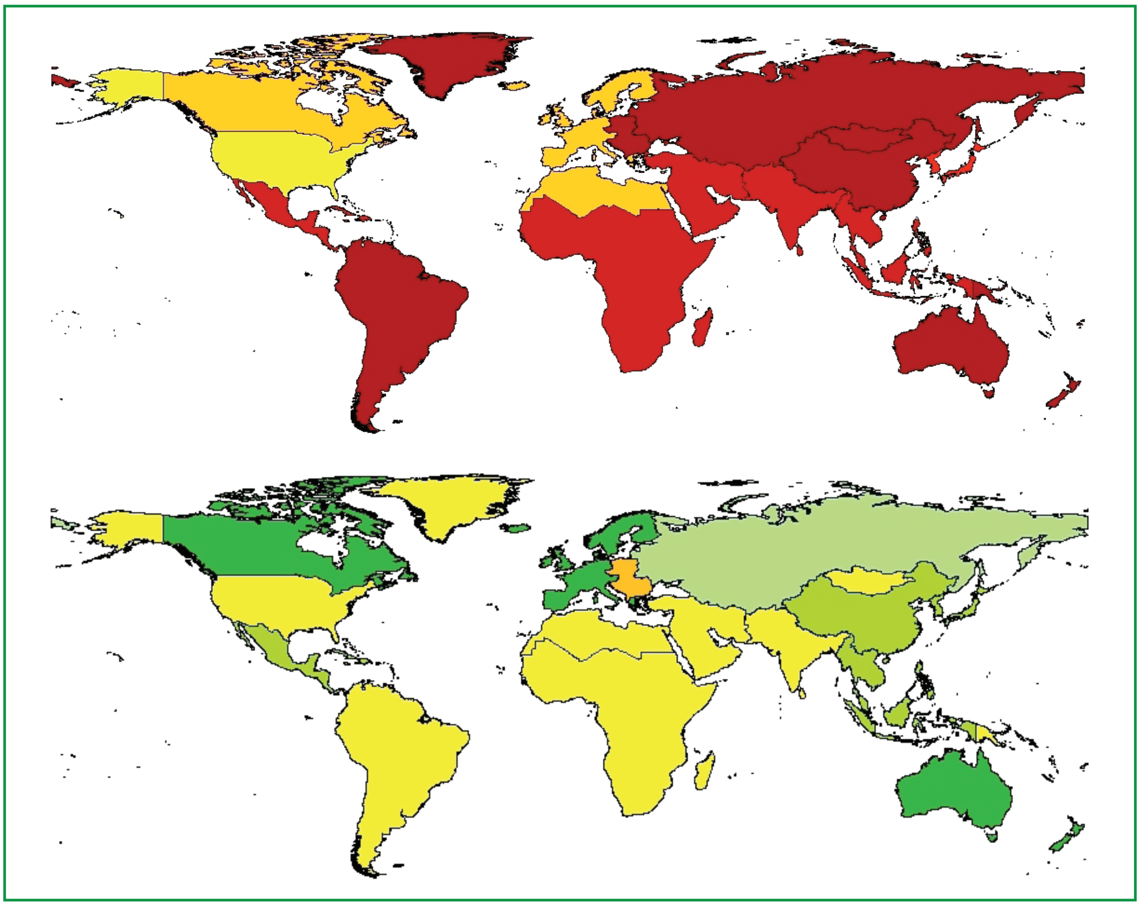


Figure 4.15. Percentage change of the water footprint of consumption per capita relative to 2000. S1 is the highest increase scenario; inspired by IPCC's A1 storyline it assumes low population growth, diets with high meat content and high fossil fuels usage. S3 is the lowest increase scenario; inspired by IPCC's storyline B1 it assumes low population growth, diets with low meat content, biofuel usage and technical and efficiency improvements. Source: Ercin and Hoekstra (2014).

In the Sustainable Development Scenarios for Rio+20, the PBL (Netherlands Environmental Agency) integrated assessment model is the one which best considers the achievement of biodiversity goals together with other development and environmental goals (Roehrl, 2012). In their approach three different scenarios provide three different pathways to achieve the desired goal (PBL, 2012; Figure 4.16). In the global technology pathway, averting the decline in mean species abundance (MSA) will be achieved mainly by an increase in agricultural productivity and climate change mitigation. In the decentralised solutions pathway, emphasis is still given to the increase in the agricultural productivity, but in a lesser extent than in the global technology pathway. Other important contributions for the protection of biodiversity are the reduction of infrastructure expansion and expansion of protected areas. The consumption change pathway is built upon several important assumptions for biodiversity, namely dietary changes towards less meat, great reductions of food wastes and lower rates of energy end-use increases due to changes in lifestyles. As a result, this is a more balanced pathway. Important contributions for the protection of biodiversity are reduction of consumption and waste, increases in agricultural productivity are still important, but much less than any of the other pathways and climate change mitigation.

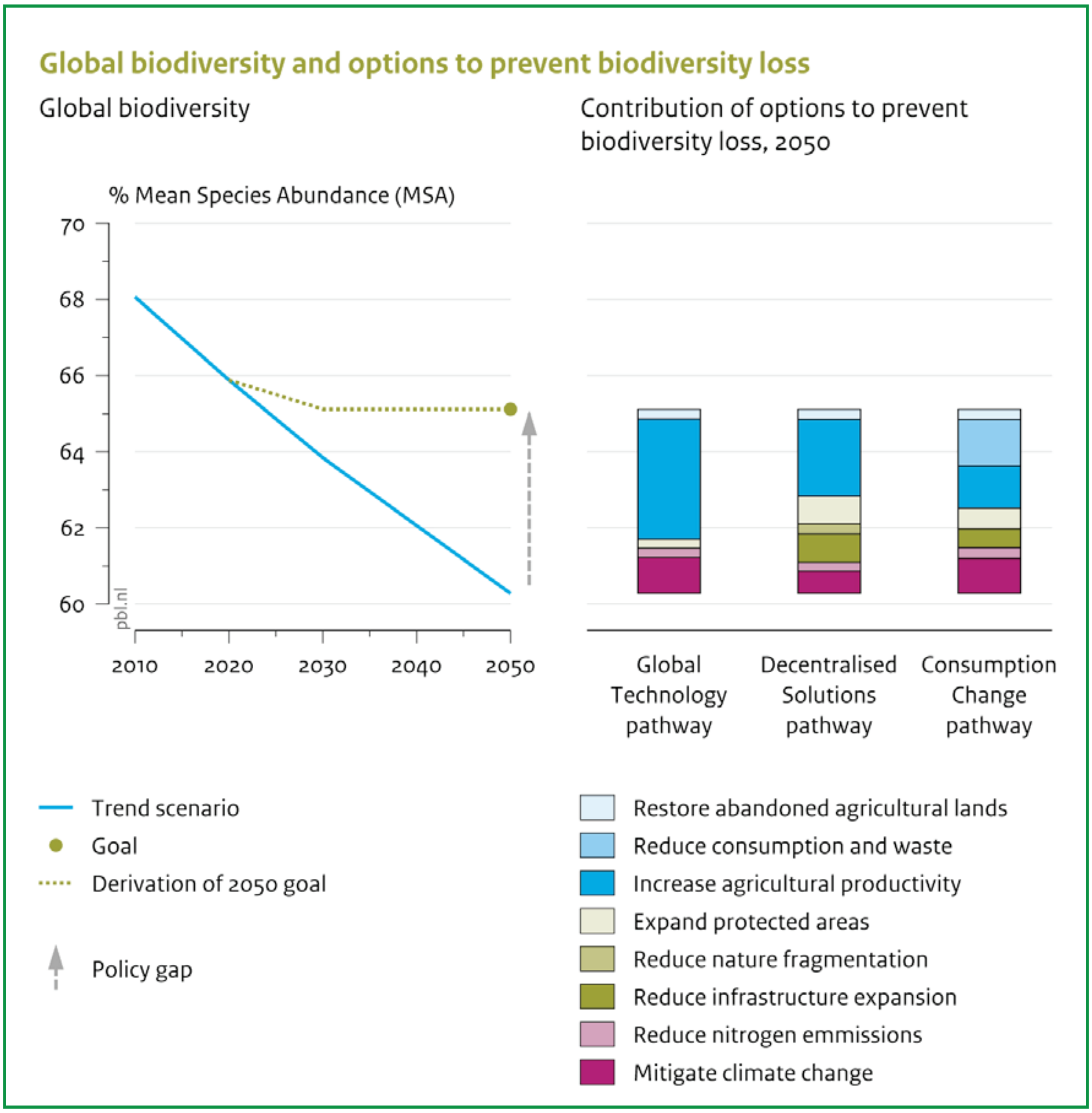


Figure 4.16. Global biodiversity and options to prevent global biodiversity loss, by 2050. Source: PBL (2012).

4.5 UNCERTAINTIES

Measuring sustainable consumption and production is multidisciplinary task that can and should be tackled by different approaches. In this chapter, we opted to use a suite of indicators that reflect anthropogenic pressures on terrestrial, freshwater and marine ecosystems. Besides the indicators chosen several others could be used, nevertheless associated with all indicators a level of uncertainty exists.

The Ecological Footprint (EF) calculations require extensive input data that have associated several sources of uncertainty (Borucke *et al.*, 2013). The quality, reliability and validity of EF calculations depend on the level of accuracy and availability of the used datasets. However, many of them have incomplete coverage and do not specify confidence limits (Borucke *et al.*, 2013; Kitzes *et al.*, 2009a).

The EF provides a systemic approach to combine information on pressures that are usually assessed independently. To translate material extraction and waste emissions into units of productive area key constants are used, these constants have a high influence on the final result and introduce a source of uncertainty (Kitzes *et al.*, 2009a). The EF and biocapacity are expressed in global hectares (gha), to convert regular ha to gha equivalence factors are used. Currently, equivalence factors are based on estimates of achievable crop yields as compared to maximum potential crop yields from the Global Agro-Ecological Zones (GAEZ) assessment (Borucke *et al.*, 2013). The choice of different equivalence factors would certainly have consequences for EF results (Giljum *et al.*, 2007). Yield factors normalise differing levels of productivity for particular land use types. Haberl *et al.* (2001) showed that different assumptions concerning yield factors could change results by a factor of 2. The Carbon Footprint component of the Ecological Footprint, as previously discussed, is one of its major sources of criticisms and also a source of uncertainty. Blomqvist *et al.* (2013) discuss the uncertainty introduced by the carbon sequestration factor used in the calculations of the carbon footprint. In order to address the major sources of uncertainty in footprint calculations and understand their impact in the final results sensitivity analysis should be performed (Giljum *et al.*, 2007; Kitzes *et al.*, 2009a).

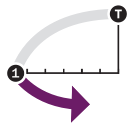
Krausmann *et al.* (2013) identifies the major uncertainties related with calculations of HANPP as uncertainties related with the underlying data, underlying assumptions of the estimation procedure and uncertainties related with the global vegetation and water balance models used. Their sensitivity analysis took into consideration uncertainties of data on biomass harvest and uncertainties associated with the assumptions underlying the calculation of $HANPP_{luc}$, and proved the robustness of their results.

Smil (2011) points out the fact that five different global quantifications of HANPP show a mean value of 25%, with ranges from 4% to 55%, and highlights that there has been no uniform approach for calculating HANPP. Smil (2011) also highlights the choice of the denominator chosen to calculate the appropriation ratio, namely the choice between above-ground and below-ground NPP and the uncertainties associated with computing NPP. In a recent meta-analysis of NPP estimates a mean of 56.4 GtC per year were estimated with uncertainty of about ± 15% (± 8-9 GtC) (Ito, 2011). This would represent differences in HANPP extreme shares of 26% less and 34% more of the mean 25% value (Smil, 2011).

The determination of the Water Footprint (WF) is a data intensive exercise (Hoekstra *et al.*, 2011), all the data sources carry their uncertainties, that normally are not well documented, and might influence the results (Hoekstra & Mekonnen, 2012; Hoekstra *et al.*, 2011). Hoekstra & Mekonnen (2012) identify basic sources of uncertainty on the global precipitation, temperature, crop, and irrigation maps that they have used and on the yield, production, consumption, trade, and wastewater treatment statistics. The assumptions underlying the WF that also add uncertainty to its calculations are, for example planting and harvesting dates per crop per region and feed composition per farm animal type per country and production system and that WFs of industrial production and domestic water supply are geographically spread according to population densities (Hoekstra & Mekonnen, 2012).

Lastly, it is important to refer the uncertainties associated with defining a safe ecological limit for natural resource use. In Rockström *et al.* (2009) three systems had already exceeded their safe operating space, rate of biodiversity loss, climate change and interference with nitrogen cycle. Global freshwater use was amongst the systems with a higher margin. However, a recent reassessment of this limit has placed current freshwater use near its limit (Gerten *et al.*, 2013). Humanity is now leaving the stable Holocene to enter a new geological period, the Anthropocene, determined by our own impact on Earth systems and characterised by uncertain conditions (Crutzen, 2002; Steffen *et al.*, 2011). Keeping the Earth system in a stable Holocene-like state requires adopting the precautionary principle when addressing planet boundaries.

4.6 DASHBOARD – PROGRESS TOWARDS TARGET

Target Elements	Status	Comment	Confidence
Governments, business and stakeholders at all levels have taken steps to achieve, or have implemented, plans for sustainable production and consumption...		Many plans for sustainable production and consumption are in place, but they are still limited in scale	High
... and have kept the impacts of use of natural resources well within safe ecological limits		All measures show an increase in natural resource use	High

Authors: Alexandra Marques and Henrique Pereira, with contributions from Marta Coll
Extrapolations: Derek Tittensor
NBSAPs and national reports: Kieran Noonan-Mooney
Dashboard: Tim Hirsch

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TARGET 5: HABITAT LOSS AND DEGRADATION

By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.

PREFACE

Habitat loss is the main driver of biodiversity change in terrestrial and inland water systems. In particular, the conversion of natural systems including forests, woodlands and grasslands to agricultural areas has diminished the area of natural systems and has often reduced species richness. Various meta-analyses have shown that species abundance and species richness declines after conversion in many cases, but not all species from the natural system disappear and other species may colonize converted habitats (Gibson *et al.*, 2011).

Conversion of land started from the establishment of agriculture about 10,000 years ago and continues to modern times. The extent of land conversion and other human alterations of the environment have led to the notion that earth has entered the era of the Anthropocene (Ellis *et al.*, 2010; Zalasiewicz *et al.*, 2011). Increasing human population and growing wealth, leading to a rising demand of food, bioenergy, wood and fibre are the primary drivers of land conversion and thus habitat loss. It is projected that more land will be needed to achieve increases in production of agricultural and forestry products in the future. However, increases in productivity per unit land can potentially provide a large increase in global production.

The main focus of this analysis is on habitat loss in terrestrial systems, especially focusing on forests. The main forest areas occur in the high northern latitudes (boreal forests), the temperate zone and the tropics. Forest definitions typically depend on various combinations of thresholds of tree canopy closure, tree height and types of land-use. Commonly used thresholds for canopy closure are "closed forests" with a tree canopy density greater than 40% or 45%, "open forests and woodlands" with a tree canopy density ranging between 20-45% and "non-forest ecosystems" with a tree canopy density ranging between 10-25% (Potapov *et al.,* 2008; Laestadius *et al.*, 2012). The "non-forest ecosystems" include savannas, grasslands and mountain ecosystems (Potapov *et al.*, 2008) and are treated in the assessment of grassland ecosystems. Differences in definitions of forest types can lead to substantial variation in estimates of forest cover dynamics. Closed forests cover about 18% and open forests and woodlands cover about 9% of the Earth's total land area (Potapov *et al.*, 2008).

Changes in forest cover are assessed in several ways. This chapter focuses on gross forest cover loss (defined as forest cover loss due to natural and human-induced disturbances), gains in forest cover (due to forest regrowth or human driven reforestation and afforestation) and net forest cover change. Gross forest loss is a particularly important indicator in tropical forests because many are primary forests that contain high biodiversity that is only very partially recovered during reforestation (Gibson *et al.*, 2011). The primary methods for determining forest cover change at large spatial scales include remote sensing (e.g., Potapov *et al.*, 2011; Hansen *et al.,* 2013) and national reports (e.g., FA0, 2010). Remote sensing provides uniform regional and global evaluation of gross loss, gain and net change forest cover, but has difficulty in distinguishing the causes of this loss. This can be due to deforestation which is a change in land-use vs. logging which does not necessarily alter land-use classification as forest vs. natural factors such as hurricanes or fire. National reports and ground-based studies can be used to estimate different types of forest loss, but suffer from heterogeneity in reporting.

Trends in grasslands are also described since they cover about 40% of the Earth's surface (excluding Greenland and Antarctica) and have high biodiversity values (White *et al.*, 2000). In Europe, for example, about 50% of the endemic plant species are dependent on grassland biotopes (Veen *et al.*, 2009). Grasslands can be defined as ecosystems dominated by herbaceous and shrub vegetation and maintained by fires, drought, grazing and/or low temperatures (White *et al.*, 2000). Non-forest ecosystems, such as savannas, woodlands, shrublands and tundra, are also included in grassland ecosystems. Grasslands are found on all continents; the largest amount is located in sub-Saharan Africa and Asia, while the Middle East and Central America have the least grassland ecosystems (White *et al.*, 2000). In sub-Saharan Africa, grasslands are mostly savanna systems, while in Oceania and Asia grasslands are often shrubland, in Asia mostly non-woody grasslands and in Europe tundra ecosystems (White *et al.*, 2000). However, these grasslands are increasingly modified due to human activities, such as cultivation, urbanization, desertification, fire, livestock grazing, fragmentation and introduction of invasive species (White *et al.*, 2000). Nevertheless, uncertainties exist due to the use of various grassland definitions and difficulty in monitoring by remote sensing (White *et al.*, 2000; Verburg *et al.*, 2011). Therefore the change in grassland extent is not as thoroughly described as forest cover change.

Trends in aquatic habitat types, such as freshwater and coastal systems are less extensively described in this chapter. Coastal systems and low-lying areas include all areas near mean sea level, comprising a diversity of ecological systems including rocky coasts, beaches, barriers and sand dunes, estuaries and lagoons, deltas, river mouths, wetlands and coral reefs (IPCC, 2014). Generally, there is no single definition for the coast and the coastal area. In relation to exposure to potential sea level rise, the LECZ (low-elevation coastal zone) has been used in recent years with reference to specific area, ecosystems and population up to 10 m elevation (Vafeidis *et al.*, 2011). As of 2000, the LECZ constitutes 2% of the world's land area but contains 10% of world's human population (600 million; McGranahan *et al.*, 2007). In addition, approximately 65% of the world's cities with populations of over 5 million are located in the LECZ (McGranahan *et al.*, 2007). The extent of intact coastal ecosystems is an important indicator as these systems provide a wide variety of regulating, provisioning, supporting and cultural services (MA, 2005). However, they have been heavily altered and influenced by human activities, resulting in tightly coupled social-ecological systems (Berkes & Folke, 1998; Hopkins *et al.*, 2012; IPCC, 2014; Vörösmarty *et al.*, 2010). Key drivers of coastal habitat loss and degradation continue to be increasing human population and land-use (including pollution), sea level rise (coastal ecosystem flooding and erosion) and ocean temperature change (IPCC, 2014). Given the diversity of ecological systems that comprise coastal systems, there is a paucity of information available for many of these systems. As many existing studies as possible were used, however explicit numbers on the extent, loss or degradation are not available for all ecosystems on a global scale. Therefore only broad categories of ecosystems are distinguished; changes in specific vulnerable ecosystems are described in the chapter on Target 10.

Freshwater ecosystems most commonly refer to lakes, wetlands, rivers and streams, and groundwater. These systems occupy less than 1% of the Earth's surface (Strayer and Dudgeon, 2010). The global extent of freshwater wetlands has been estimated at 9.2 – 12.8 million km^2 at the end of the 20th century (Finlayson, 2006; Lehner & Döll, 2004; MA, 2005). Despite this, freshwater ecosystems support more than 10% of all known species including around a third of all vertebrates (Strayer & Dudgeon, 2010). Pollution and exploitation of these systems for food, energy, transport, and water supply (Vörösmarty *et al.*, 2010), together with the emerging threat from climate change (Woodward *et al.*, 2010), has led to freshwater ecosystems suffering more strongly from human activities than marine and terrestrial ecosystems (Darwall *et al.*, 2008; Dudgeon *et al.*, 2006; Keenleyside & Tucker 2010, Ricciardi & Rasmussen, 1999). Similar to coastal ecosystems, information and data on the extent of freshwater ecosystem fragmentation at the global scale are limited.

5.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

5.1.1 Status and trends

At global level, the extent of all natural ecosystems, terrestrial and aquatic, are declining (Figure 5.1), however, large regional differences exist. The causes of decline for forests, grassland, coastal and freshwater systems are described below.

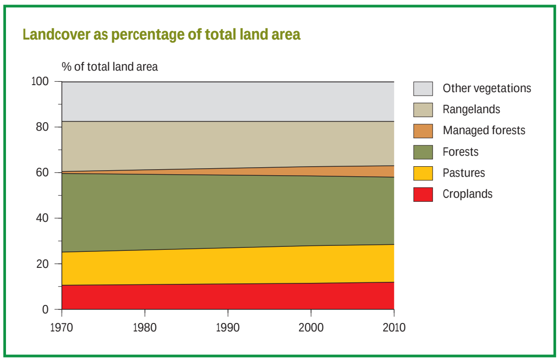


Figure 5.1. Change in land cover types from 1970 – 2010. Derived from analysis using the IMAGE model. Source: Bouwman *et al.*, (2006) and based on data from FAO (2014).

5.1.1.i Forests

The most recent estimates of global forest cover change based on high-resolution satellite imagery indicate substantial forest loss (2.3 million square kilometres) and gain (0.8 million square kilometres) over the period 2000 to 2012 (Figure 5.2; Hansen *et al.*, 2013). Gross forest cover loss is high in all forested biomes, but differs greatly among regions. Rates of loss in terms of total area are particularly high in boreal forests and the humid tropics (Figure 5.2; Margono *et al.*, 2012). There are no temporal trends in rates of gross loss except for an increasing trend for tropical forests (Hansen *et al.*, 2013). These recent remote sensing interpretations provide a good overview of the trends and spatial distribution of deforestation. Interpretation of the causes and underlying drivers of deforestation relies largely on local case-studies that have studied the processes leading to deforestation decisions in more detail. Meta-analysis of such case studies has provided insight in the generalities and context-specificities of the underlying driving factors (Geist and Lambin, 2002; Magliocca *et al.*, 2014).

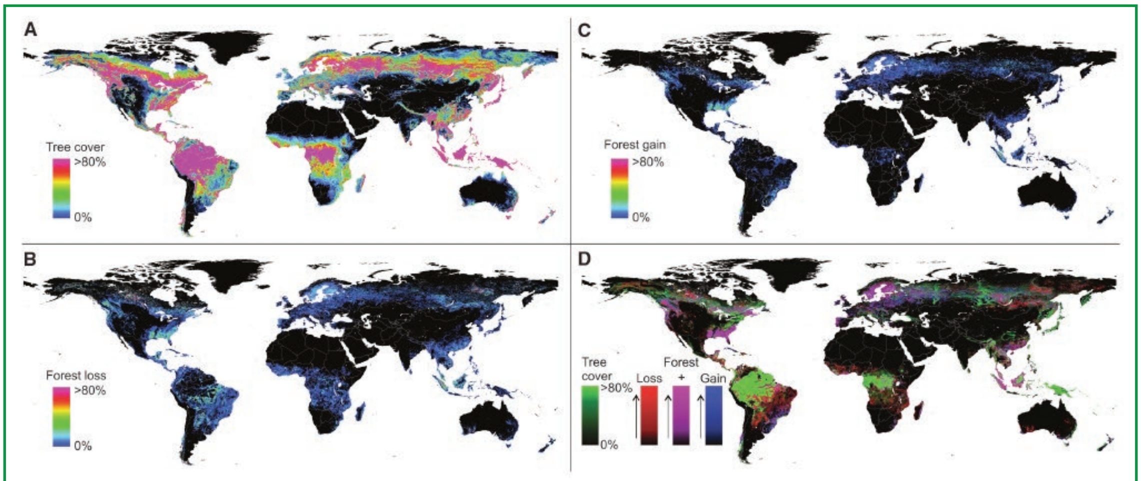


Figure 5.2. (A) tree cover, (B) gross forest loss and (C) forest gain from Hansen *et al.* (2013). A colour composite of tree cover in green, forest loss in red, forest gain in blue and forest loss and gain in magenta is shown in (D) with loss and gain enhanced for improved visualisation. All maps layers have been resampled for display purposes from the 30-m observation scale to a 0.05° geographic grid. Source: Hansen *et al.*, (2013).

Based on the study of Hansen *et al.* (2013), gross loss in the tropic rainforest accounts for 32% of global forest loss over the period 2000 to 2012. Loss in South American rainforests accounts for about half of gross cover loss in tropical rainforests over this time period. In some tropical regions, for example Brazil, gross forest loss and reported deforestation have declined markedly over the last decade (Figure 5.3; FAO, 2010; Lambin & Meyfroidt, 2011; FAO & JRC, 2012; Malingreau *et al.*, 2012; Hansen *et al.*, 2013; see also Box 5.3). Deforestation in the Brazilian Amazon has declined due to numerous factors including improved agricultural management reducing the need for expansion of pasture and crop areas (Lambin & Meyfroidt, 2011; Malingreau *et al.*, 2012), improved legislation and control (Malingreau *et al.*, 2012), extension of protected areas (Soares-Filho *et al.*, 2010) and intensive, publically available monitoring (Hansen *et al.*, 2013). However, there are concerns that recent changes in the forest code in Brazil may lead to a resurgence in deforestation (Soares-Filho *et al.*, 2014; see Box 5.3). While considerable attention has been given to deforestation in the Brazilian Amazon, substantial deforestation is occurring in other parts of Latin America (Aide *et al.*, 2013).

Overall, gross loss increased across all types of tropical forest by about 2100 square kilometres per year over the period 2000 to 2012, but there are large regional differences. The greatest increase in gross loss over time has been in Eurasian tropical rainforest (as shown for Indonesia in Figure 5.3), followed by increasing rates of loss in dry tropical forests of South America, Africa and Eurasia. In these regions, the high ratio of gross loss to gain in remotely sensed forest cover indicates that deforestation is responsible for most of this loss (Figure 5.2). This is coherent with high levels of reported deforestation in these regions, although the temporal dynamics are not always coherent between remotely sensed and report based estimates (FAO, 2010). In Southeast Asia, hotspots of forest area loss have been in large part attributed to the establishment of large-scale agro-industries, especially oil palm plantations (Hansen *et al.*, 2008; 2010; Koh *et al.*, 2011; Lee *et al.*, 2014).

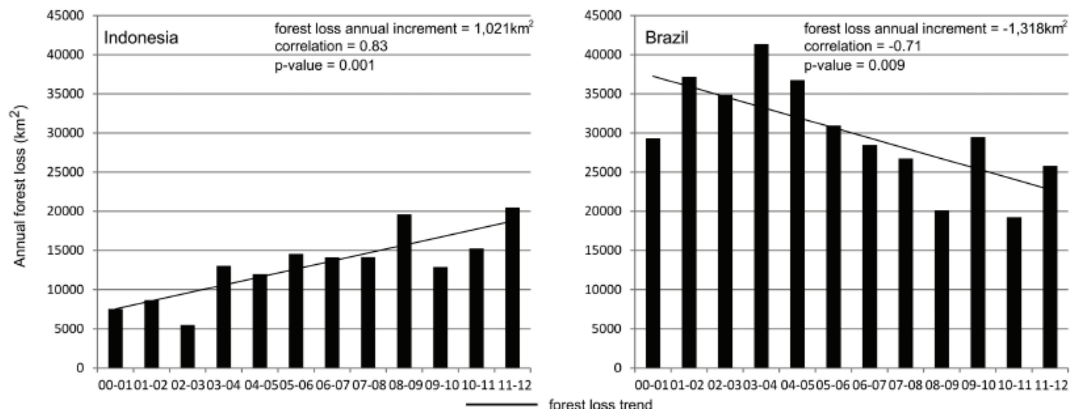


Figure 5.3. Annual gross forest cover loss for Brazil and Indonesia from 2000 to 2012 based on remote sensing. The forest loss annual increment is the slope of the estimated trend line of change in annual forest loss. Source: Hansen *et al.*, 2013.

Reported rates of deforestation in tropical Africa have been lower than in other tropical regions because some of the most common driving forces of deforestation present in other parts of the world, such as governmental land settlement schemes and industrialised agriculture have largely been absent (Rudel, 2013). For instance, gross deforestation rates in the Democratic Republic of the Congo have been estimated at 2.3% of the forest area for the 2000-2010 period, while deforestation rates in Cameroon are twice as high (Potapov *et al.*, 2012). Large differences in deforestation rates between African countries are attributed to differences in forest type (deforestation rates in dry forest have been highest) and differences between countries in the contribution of the mineral industry to the country economies (Rudel, 2013), but low rates of reporting create high uncertainty in estimates (FAO, 2010).

Logging without a change in land-use (i.e., forests are allowed to regenerate or are replanted) is the primary driver of forest dynamics in many regions. Europe dominates the area of forest designated for production and there has been a slight global decline in forest area designated for production (FAO, 2010). Remote sensing data and national reports indicate that gross forest cover loss and gain dynamics in northwest United States, temperate Canada, Portugal and Russia is heavily influenced by intensive logging (Hansen *et al.*, 2013). Dynamics of subtropical forests in all regions are also dominated by forestry activities (FAO, 2010; Hansen *et al.*, 2013).

Natural causes are the dominant drivers of forest loss in several regions. High gross forest loss in boreal forests is primarily driven by fire with significant additional contributions from logging and mortality due to insect pest damage in some regions (Edburg *et al.*, 2012). The mountain pine beetle has affected more than 11 million hectares of forest in Canada and western United States since the late 1990s (FAO, 2010). In Australia, severe drought and forest fires have caused an increase in gross cover loss since 2000 (FAO, 2010; Hansen *et al.*, 2013).

Reported gain in forest area is particularly high in East Asia and parts of Southeast Asia, especially China and Vietnam (Box 5.1; FAO, 2010), due to large-scale afforestation (FAO, 2010). However, remotely sensed forest cover shows much smaller gains, perhaps due to the time needed for planted trees to establish sufficient cover (Figure 5.2; Hansen *et al.*, 2013). Globally, 7% of the forests are planted (FAO, 2010). In many cases forest plantations may have low biodiversity values, but this depends on the context and previous land-use (see Target 7; Bremer & Farley, 2010). National reports also indicate a long-term trend of increasing area of temperate and boreal forests, primarily due to abandonment of agriculture in some regions (FAO, 2010). For example, gains of forest cover have been substantial in Eurasian, the Ukraine and other former Soviet republics due to agricultural abandonment and forest recovery after fires (Kuemmerle *et al.*, 2011; Hansen *et al.*, 2013).

Box 5.1: Conserving and Restoring Habitats in China

In recent years, the government of China has reinforced its efforts in biodiversity conservation, through measures such as restoring degraded ecosystems and afforestation. A number of key ecological projects continue to be implemented, such as natural forest protection, returning cultivated lands to forests and forest belt construction in north, northeast and northwest China as well as in the Yangtse River basin and coastal areas. Forest resources in China have been increasing recently, with forest areas increased by 23%, forest coverage rate by 4% and forest reserves by 22% compared with those of a decade ago. The implementation of key ecological projects has enhanced recovery of degraded ecosystems and habitats for wild species, thus effectively conserving biodiversity.

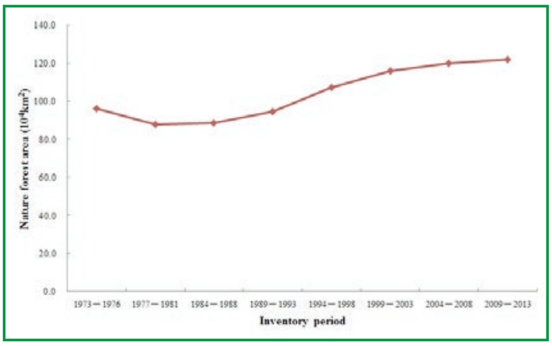


Figure 5.4. Change in forest resources in Areas of Natural Forests in China. Source: China 5th National Report to the CBD, 2014.

The expansion of agricultural production to fulfil an increasing demand for food, feed, fibre and fuel is an important determinant of forest loss, and habitat loss in general. Increases in agricultural production can be accomplished by increases of agricultural area and by agricultural intensification (Lambin, 2012; Meyfroidt *et al.*, 2013). Measures to stimulate agricultural intensification and measures aimed at protecting forest and other natural habitats can both lead to an intensification of agriculture and a so-called 'land sparing' effect (Butsic *et al.*, 2012; Grau *et al.*, 2013). Forest protection in general shows positive results at the location of protection (Robinson *et al.*, 2013), but it does not necessarily reduce global forest losses due to displacement processes (Andam *et al.*, 2008; Lambin & Meyfroidt, 2011; Grau *et al.*, 2013). Such displacement, also referred to as "leakage", may occur through the migration of the agents of deforestation to neighbouring locations or through trade in timber or agricultural products (Meyfroidt & Lambin, 2011). Lambin & Meyfroidt (2011) also identified rebound effects as an important process that should be accounted for when considering policies to reduce forest loss. Rebound effects refer to a response of agents, or of the economic system, to new technologies or other measures introduced to reduce land-use. As an example, a more efficient agriculture is likely to be more profitable and could, therefore, lead to an expansion of the cultivated area and increased consumption (Angelsen & Kaimowitz, 1999). Rebound effects can, therefore, lead to ineffective biodiversity policies (Angelsen & Kaimowitz, 1999; Maestre Andrés *et al.*, 2012).

The driving factors of agricultural intensification and associated displacement and rebound effects are considered to be context specific and are not sufficiently well understood (Angelsen & Kaimowitz, 1999; Keys & McConnell, 2005; Rudel *et al.*, 2009; Magliocca *et al.*, 2013). Displacement processes and rebound effects are also important to consider when analysing the countries that show strongly reduced deforestation or even forest recovery. This phenomenon is referred to as the forest transition and associated with increasing affluence, urbanization and agricultural intensification (Rudel *et al.*, 2005; Lambin and Meyfroidt, 2011). However, recent analyses show that forest transition is often offset by increased imports of agricultural commodities and wood from abroad, leading to leakage or displacement of deforestation (Rudel *et al.*, 2005; Meyfroidt & Lambin, 2009; 2011; Lambin & Meyfroidt, 2011). Even though the positive trends in deforestation are partly offset by such leakage, there has been a substantial overall reduction in the loss of global forest areas (Meyfroidt & Lambin, 2011). Other positive net forest changes are associated with increased forest plantations, while losses of natural and semi-natural forest continue (FAO & JRC, 2012). A focus on net forest area changes can, therefore, obscure the loss of primary forest with important biodiversity consequences (Brown & Zarin, 2013).

In Europe and parts of the USA, positive trends in forest cover are strongly related to the abandonment of marginal agriculture. In regions with currently fragmented landscapes, this abandonment can lead to the recovery of larger, continuous 'wild' areas favouring biodiversity (Navarro & Pereira, 2012). Nevertheless, many species are associated with extensive farmland and there are concerns about how those species will fare under rewilding (see Box 4 in Target 15, and Queiroz *et al.*, 2014). In some cases, increasing pressure on the abandonment farmland has led to afforestation and intensive forest management leading to fewer benefits for biodiversity (Cramer *et al.*, 2008).

While deforestation in some countries has shown a decreasing trend, this does not necessarily hold for forest degradation. Most deforestation estimates are based on satellite images that only reveal forest changes based on clear cuts, or on reports that only account for deforestation when a minimum area is deforested, thereby underrepresenting forest fragmentation and edge effects (Laurance *et al.*, 2011). Forest degradation is not included in many inventories and may possibly offset some of the positive trends (Malingreau *et al.*, 2012). For example it is estimated that only about 24% of the global forests can be considered as intact forest landscape (Potapov *et al.*, 2008). The extent of forest degradation has been estimated globally using a combination of FAO data and satellite images, which provides forest degradation at macro-regional and local scale (Figure 5.4; Laestadius *et al.*, 2012). In addition, expansion of agriculture at the forest fringes or in fragmented forests often leads to a gradual decline in the forest that is difficult to detect in satellite images. Gradual changes in mosaic landscapes are often not accounted for in deforestation statistics. These losses in smaller forest fragments not only lead to a direct reduction of habitat, but also reduce the connectivity between remaining larger forest areas. Increasing road construction reduces connectivity and accelerates consequent land-use change (Laurance *et al.*, 2009).

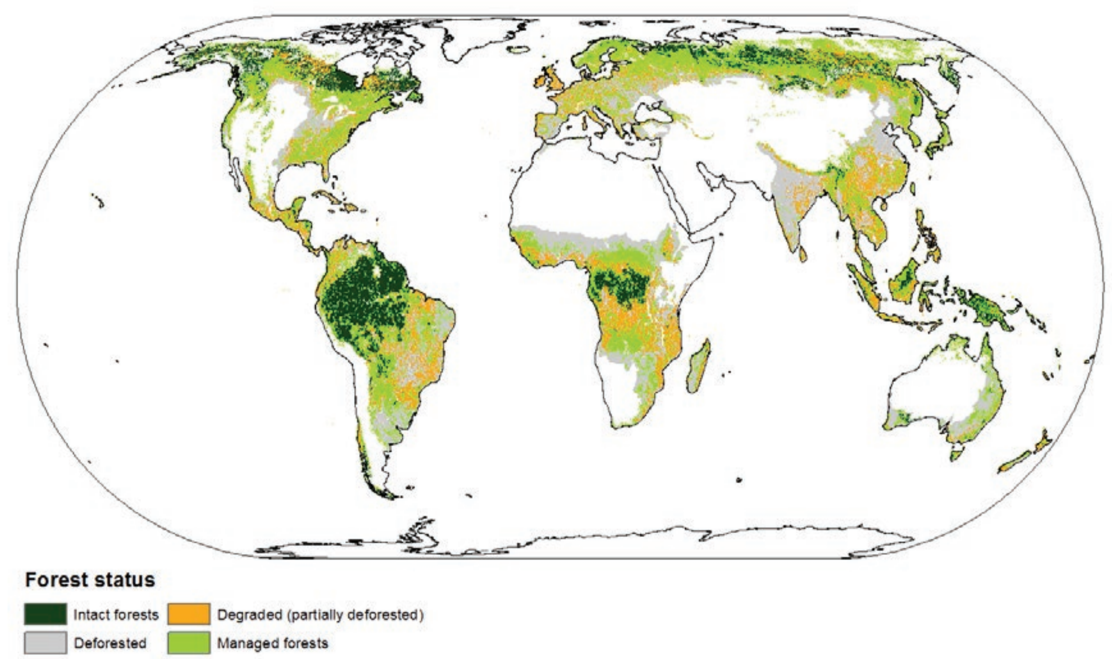


Figure 5.4. The extent of deforestation and forest degradation. Source: Laestadius *et al.* (2012).

5.1.1.ii Grasslands

The main causes of grassland habitat change are cultivation and livestock grazing (White *et al.*, 2000; FAO, 2005). Globally, grassland extent is declining, and by 2000 nearly 50% of all grasslands were lightly to moderately degraded, and more than 5% were strongly degraded (White *et al.*, 2000). However, uncertainties exist about the location and extent of grassland conversion due to the use of various grassland definitions and difficulty in monitoring by remote sensing (Verburg *et al.*, 2011; White *et al.*, 2000).

It has been predominantly temperate grasslands, savannas and shrublands that are converted to agriculture to meet growing food demand. In North America, the recent increase in biofuel demand has also resulted in increased grassland conversion (White *et al.*, 2000; FAO, 2005; Brink & Eva, 2009; Estes *et al.*, 2012; Wright and Wimberly, 2013).

Increasing numbers of livestock and the disruption of migration routes in Central Asia, Africa, South America and Patagonia has led to overgrazing causing degradation, desertification and erosion of grassland ecosystems (Romero-Ruiz *et al.*, 2012; FAO, 2005; 2006; Rada, 2013). In China overgrazing has resulted in the degradation of 32% of grasslands (FAO, 2005). Large-scale land acquisitions by national and international companies are also contributing to the conversion and intensification of rangeland habitats, as illustrated by the rapid conversion of flooded grasslands in Cambodia (Packman *et al.*, 2013). In addition, increasing numbers of livestock and grassland degradation also result in deforestation to provide more rangelands for livestock (FAO, 2006).

However in some regions, grasslands are increasing in extent. Improved pasture technology has led to good conditions for grasslands, for example on the South American Campos (FAO, 2005). In Mongolia, pastoral areas are increasing, caused by cropland abandonment (FAO, 2005). Nevertheless, this effect is halted due to increased grazing activities leading to local degradation (FAO, 2005).

In addition to grassland habitat loss and degradation, grasslands are often fragmented. This is mainly due to the increased fragmentation by roads leading to isolation, decreased recolonization, increased edge effects and increase number of invasive species (White *et al.*, 2000; Beale *et al.*, 2013). Especially in the United States and Botswana, roads cause grassland fragmentation (White *et al.*, 2000).

5.1.1.iii Marine and coastal habitats

In the tropics, mangroves are experiencing substantial changes in area (Duke *et al.*, 2007; FAO, 2010), with associated changes in biodiversity (Polidoro *et al.*, 2010) and ecosystem functions (e.g. Donato *et al.*, 2011). Major threats to mangroves include aquaculture, land reclamation and urban development. There are varying estimates of change in mangrove extent; indeed, the 95% confidence intervals in a recent study by Fries and Webb (2013) were so broad that projections of future trends of mangrove cover were difficult. Due to the variability between available estimates of mangrove area (FAO, 2007; 2010), it may be difficult to discern any convincing trend of deforestation (Friess & Webb, 2013; Grainger, 2008). For example, data variability makes it difficult to ascertain whether Indonesia – the country with the world's largest area of mangroves – is experiencing a loss or gain of mangroves. Estimates of the rate of change of mangrove area can vary from–1.62% yr^{-1} to +0.15% yr^{-1} (Friess and Webb, 2013). In addition, estimates of global mangrove cover are based on previous or current extents and do not yet account for smaller scale observations of pole-ward expansion of mangrove in North America and New Zealand due to climatic warming (Stokes & Healy, 2010; Comeaux *et al.*, 2011; Raabe *et al.*, 2012; IPCC, 2014). With this in mind, it remains difficult to effectively scale-up biophysical and ecological data, and to implement informed conservation policies, due to the paucity of accurate historical information and future projections regarding mangrove area (Friess & Webb, 2013).

Vegetated coastal habitats are declining globally (Duarte *et al.*, 2005), rendering shorelines more vulnerable to erosion due to increased sea level rise and increased wave action (e.g. Alongi, 2008). Coastal wetlands experience coastal squeeze in urbanized coastlines (e.g., Pauchard *et al.*, 2006) with no opportunity to migrate inland with rising sea levels (IPCC, 2014). Kelp forests have been reported to decline in temperate areas in both hemispheres, a loss involving climate change (Johnson *et al.*, 2011; Wernberg *et al.*, 2011; Fernández, 2011; IPCC, 2014). In Europe, for example, it is estimated that more than 50% of original coastal wetlands and seagrasses have been lost since 1960 (Airoldi & Beck, 2007) and this rate is accelerating (EEA, 2013). Decline in kelp populations attributed to ocean warming has been reported in southern Australia (Johnson *et al.*, 2011; Wernberg *et al.*, 2011b) and the North Coast of Spain (Fernández, 2011). A global analysis of human impacts on marine ecosystems showed that less than 4% of the earth's oceans have very low human impact, and that over 40% of oceans and coasts worldwide are heavily affected by human activities such as destructive fishing practices, poor land-use practices, pollution, and coastal development (Halpern *et al.*, 2008). Among marine ecosystems, most vulnerable to habitat destruction and loss are seagrass meadows, mangroves, and coral reefs, due to their proximity to dense human populations (Waycott *et al.*, 2009). For example, approximately 75% of the global continental shelf is subject to trawling and dredging for fisheries (Kaiser *et al.*, 2002), which can have destructive effects on seafloor communities and habitats (Hixon & Tissot, 2007; Thrush & Dayton, 2002). Trawling also removes organisms that create structures, such as crabs and scallops. Subsequent loss in habitat complexity from trawling, can affect predation and recruitment dynamics of fish populations (Auster *et al.*, 1996). Further, trawling can remove keystone species that control ecosystem dynamics (e.g., algal-grazing urchins) and thereby impacts habitats indirectly (Kaiser *et al.*, 2002). The destructive effect of trawling is especially serious for vulnerable and sensitive habitats like seagrass and mangroves (Waycott *et al.*, 2009).

5.1.1.iv Freshwater habitats

The majority of freshwater habitats have been altered in some way by humans, resulting from human dependence on freshwater resources (Rockström *et al.*, 2010), combined with localised and distant disturbances from upstream drainage networks (such as pollution from agriculture and industry; more information see Target 8; Vörösmarty *et al.*, 2010). The alteration of flow regimes is claimed to be the most serious and a continuing threat to ecological sustainability of rivers and their associated floodplain wetlands (Lundqvist, 1998; Bunn & Arthington, 2002). Agricultural and urban development are major drivers of altered flow regimes; water abstraction, diversion and modification of natural water bodies results in disturbing natural timing of river flows and subsequent habitat degradation, fragmentation and species decline (Van Asselen *et al.*, 2013). Between 1970 and 2000, populations of freshwater species included in the Living Planet Index declined on average by 50%, compared to 30% for marine and also for terrestrial species (MA, 2005).

Large dams occur in the majority of, if not all, countries and freshwater ecoregions in the world. There are about 50,000 large dams (higher than 15 m), of which nearly 7,000 are mapped in the GRanD database (Lehner *et al.*, 2011), and hundreds of thousands of small and medium-sized dams are distributed over the majority of small and large rivers (e.g., Lehner *et al.*, 2011; Liermann *et al.*, 2012; Januchowski-Hartley *et al.*, 2013). At the global scale, dams, large and small remain poorly mapped (Lehner *et al.*, 2011). As do the occurrences of road-stream intersections where roads and railroads cross over streams, potentially impacting the movement of materials, nutrients and organisms through culverts (e.g., Januchowski-Hartley *et al.*, 2013; O'Hanley *et al.*, 2013). Further, there remains a paucity of information on the extent to which different riverine ecosystem types have been degraded and fragmented by humans. However, local and regional scale studies have demonstrated that natural flow regime and the longitudinal and lateral connectivity of freshwaters, essential for maintaining important ecosystem services, and sustaining biophysical and ecological processes are disrupted by levees, embankments, dams and weirs and other infrastructure such as roads and railroads (Benda *et al.*, 2004; Januchowski-Hartley *et al.*, 2013; Nilsson *et al.*, 2005; Ziv *et al.*, 2012). The degradation and fragmentation from these infrastructure is not only limited to rivers, but extends to all types of freshwater ecosystems including lakes and wetlands, impacting the spatial-temporal habitat heterogeneity, connectivity among habitat patches and temporal fluctuations of nutrients, organism population abundances and diversity (Arlettaz *et al.*, 2011; Januchowski-Hartley *et al.*, 2013; McCluney *et al.*, 2014).

With an increasing awareness of the negative impacts of dams and road crossings, many industrialized nations, especially parts of Europe and North America, are moving towards the removal of non-functional and aging dams, and remediating road culverts (Arlettaz *et al.*, 2011; Doyle & Gavlick, 2009; Januchowski-Hartley *et al.*, 2013). Where available, studies have shown long-term positive effects from removal and remediation efforts (Stanley & Doyle, 2003; Arlettaz *et al.*, 2011). In other areas, such as Australia and the southwest United States, timed watering events are being used as an alternative way of returning flows to rivers that have otherwise begun to run dry over the last century as a consequence of damming and diversions for agriculture and urban populations (e.g. Colorado River, USA and Murray-Darling Rivers, Australia). The use of environmental flows is a relatively recent advancement; consequently the environmental and ecological returns from these efforts remains poorly studied (Poff & Matthews, 2013). Despite the trend to remove aging infrastructure and facilitate environmental flows back to highly degraded riverine systems, rates of new dam construction in South America, Asia, and Africa are increasing rapidly (Finer & Jenkins, 2012; Ziv *et al.*, 2012), large dams having strikingly poor performance records in terms of economy, social and environmental impact, and public support (World Commission on Dams, 2000; Singh, 2002; Scudder, 2005; IEA, 2006; Sovacool and Bulan, 2011; Ansar *et al.*, 2014; McCluney *et al.*, 2014). The potential implications of expanding dams into both already fragmented and non-fragmented river systems are further discussed below.

As discussed above, there is limited knowledge of coastal and freshwater wetland extent change at the global scale. The existing wetland literature is scattered and uneven in regards to coverage of wetland types and world regions. Current global estimates of natural wetland conversion are based on satellite images and are very uncertain, as shrimp ponds, man-made reservoirs and rice fields are difficult to distinguish from natural wetlands. Reliable estimates of historic and current losses of freshwater habitats, including wetlands, are only available for a few countries (Moser *et al.*, 1998; Finlayson & Davidson, 1999; MA, 2005). Therefore there is currently no agreed global map of these wetland ecosystems. However, the majority of studies that have measured wetland extent change suggest high rates of global wetland area decreases. An average global loss of wetland area of 6% in the period 1993-2007 was observed (Prigent *et al.*, 2012), with estimates up to 1.5% decrease each year (Talberth & Gray, 2012; Hansen *et al.*, 2008). The largest changes occur in densely populated areas (Hansen *et al.*, 2008; Prigent *et al.*, 2012).

An agreed global map of wetlands would allow tracking of progress towards reaching Aichi Biodiversity Target 5. Therefore it is essential that work be undertaken to estimate the global baseline rate of decline. The Wetland Extent Index is the result of recent work to try and gather the existing wetland extent change literature to provide a first indication of the status of this habitat globally. It is a new method to estimate the average rate of wetland extent change with incomplete data, establishing a baseline for the status of wetlands globally. The analysis uses a variation of the Living Planet Index methodology (Loh *et al.,* 2005; Collen *et al.*, 2009) to combine wetland extent time-series data from the published scientific literature and the latest analysis uses over 1,000 wetland extent time-series gathered from 170 different source references. It can be disaggregated to the three main wetland types as defined by Ramsar Convention on Wetlands: marine/coastal, inland and human-made, and into six regions: Africa, Asia, Europe, Neotropics, North America and Oceania (Ramsar, 2012; Ramsar, 2014). The methodology was developed to account for the irregular and uneven coverage of the wetland extent literature both geographically and thematically; *i.e.,* there are more studies on wetlands in North America than in the Neotropics and more extensive datasets for mangroves than for lagoons.

Using time-series data, the Wetland Extent Index shows that both marine/coastal and inland wetland extent have declined across the world, although regional differences exist (Figs. 5.5 & 5.6). In contrast, human-made wetlands have increased over the 38-year period, especially in southern Asia due to conversion of natural wetlands into rice paddies.

Both Figure 5.6 and the literature indicate a declining wetland extent in North America. In the USA, approximately 53% of wetlands were lost from the 1780s to 1980s (Dahl & Johnson, 1991; Dahl, 2000). Also in southern and coastal Canada high rates of wetland loss have been experienced, and some detailed estimates exist for populated regions. It is estimated that 65–80% of wetlands have been lost in coastal marshes in the Atlantic and Pacific regions respectively, 71% of wetlands in the lower Great Lakes – St. Lawrence river region, and 71% of wetlands in the prairie pothole region of Canada (Mitsch and Hernandex, 2013). More recently Carroll *et al.* (2011) has shown a net reduction of more than 6,700 km^2 in the surface area of water in Arctic lakes across Canada between 2000 and 2009. In Africa and South America, historic losses of wetlands have been limited, but the rate of loss has increased since the end of the 20th century (Van Dam *et al.*, 2014). China has lost a net 50,000 km^2 wetlands (nearly 30% of its natural wetlands) between 1990-2000 (Cyranoski, 2009; Gong *et al.*, 2010; Wendland *et al.*, 2011).

On the other hand, the Ramsar Convention on Wetlands, since its start in 1971, has succeeded in gradually expanding its 'list of wetlands of international importance' to 2.1 million km^2 (nearly 2,200 sites in 144 countries) in 2013 (Ramsar, 2014). Therefore the Ramsar Convention is one of the most successful conservation organizations worldwide. The sites are however unevenly distributed over the world, and many of the wetlands that are on the list are still threatened by external pressures. Their target is a further increase to 2.5 million km^2 by the next CoP in 2015 (Ramsar).

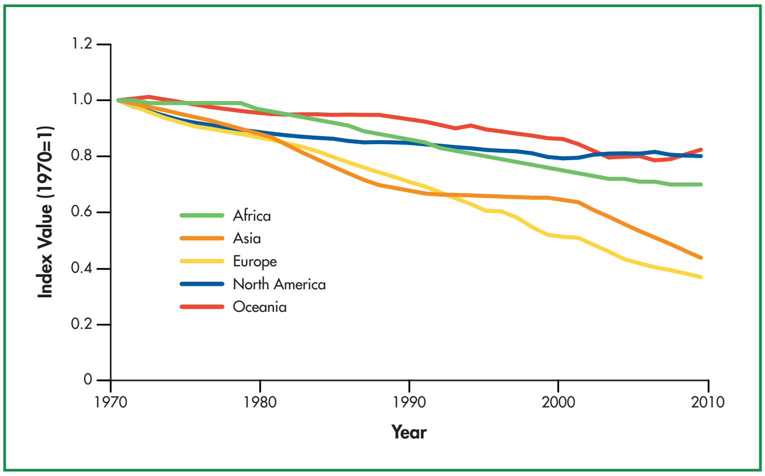


Figure 5.5. The global average marine/coastal, inland and human-made wetland extent trends relative to extent in 1970 and up to 2008 as estimated by the Wetland Extent Index. A decrease in the index means that wetland extent has declined on average while a constant index represents no overall change in wetland extent or that gains and declines cancel each other out. Source: Collen *et al.*, (2009); Loh *et al.*, (2005).

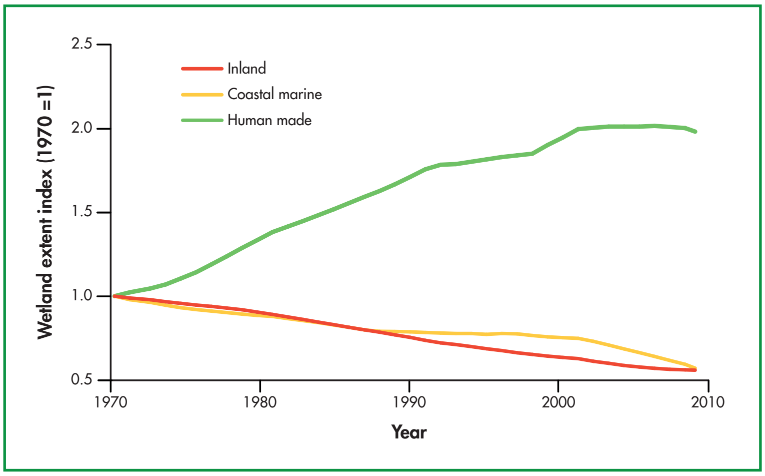


Figure 5.6. The average trends in natural wetland extent, which is the aggregation of equally weighted marine/coastal and inland wetland trends, relative to 1970 and up to 2008 by region as estimated by the Wetland Extent Index. A decrease in the index means that wetland extent has declined on average while a constant index represents no overall change in wetland extent or that gains and declines cancel each other out. An accurate trend for the Neotropics could not be calculated due to insufficient data. Source: Collen *et al.*, 2009; Loh *et al.*, 2005.

5.1.2 Projecting forward to 2020

Extrapolations of recent trends show continued loss of natural habitat at the global level (Figure 5.7). The fraction of natural habitat appears to have stabilized over the last two decades (Figure 5.7), however this levelling off is not statistically significant. Up to 2030, agriculture is projected to expand as world population and economic growth leads to an increase in wood and food demand, a higher consumption level and an increased consumption of meat and other livestock products (IAASTD, 2009; Conforti, 2011; IFPRI, 2013). There are, however, some notable exceptions to this. Europe's farmland area is expected to keep decreasing, in part due to the aging rural population in remote areas (Keenleyside & Tucker, 2010). Some scenarios project a further decrease of up to 15% the total agricultural area of the EU27 by 2030 (Verburg & Overmars, 2009), consistent with projections of up to 20% loss in the area used by the main food crops in developed countries by 2050 (Balmford *et al.*, 2005). Together with restoration, this could lead to an increase in natural area in these regions (see chapter on Target 15).

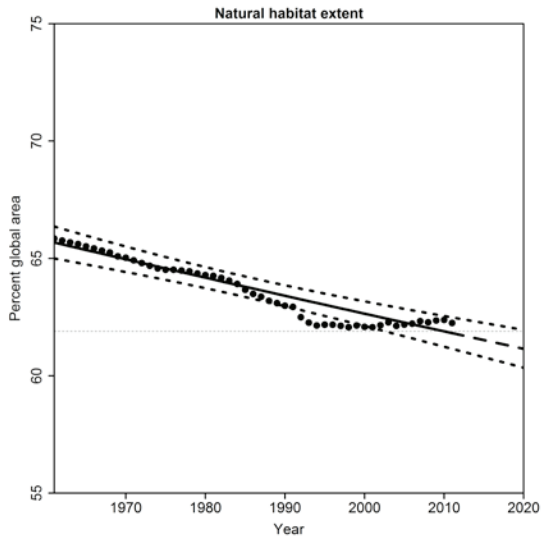


Figure 5.7. Statistical extrapolation of the percentage of natural and semi-natural areas, including forests, rangelands and other systems to 2020 at the global scale. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Source: Based on FAO (2014).

Due to population growth, the demand for forest products is projected to increase (PBL, 2012). Optimistic projections suggest that this demand can be met by plantations and natural regeneration of already managed forests (Wise *et al.*, 2009). Improved productivity per land area and landscape planning could also substantially reduce the rate for deforestation over the next several years. Pessimistic projections suggest that demand for cropland, pastures and land for bioenergy could lead to a resurgence of deforestation over the next two decades. Scenario analyses of future land cover also suggest that national policy initiatives will strongly affect forests and other vegetation types, and these land cover shifts are likely to have important impacts on conservation of biodiversity (Martinuzzi *et al.*, 2013).

The wide range of representation and assumptions on the deforestation processes in response to food, feed, fuel and fibre demands and the role of land management in the models used to explore future deforestation indicates that there is high uncertainty in scenarios of biodiversity loss over the next decades (Hertel, 2011; Verburg *et al.*, 2013). A trend that is projected in all scenarios is a substantial increase in urban and built-up areas leading to a loss of both agricultural and natural areas (Seto *et al.*, 2012). There are very large regional and country-to-country differences for other projected land-use transitions. Some scenarios have been developed at regional scales for the next two decades (Figure 5.8). This overview in Figure 5.8 is illustrative and not exhaustive. These scenarios are not based on the same hypotheses about underlying drivers and are therefore not directly comparable. These scenarios should be viewed as providing insight into mechanisms and not as predictions of future land-use change.

As indicated above, most projections indicate that croplands and pastures will decrease in area in the United States and Europe over the next several decades, with the largest increases in land cover occurring for urban areas and regenerating forests (USA - Alig *et al.*, 2010; Europe – Verburg *et al.*, 2010). Differences in socioeconomic scenarios or policies are not projected to substantially alter these trajectories (Verburg *et al.*, 2010; Radeloff *et al.*, 2011). Losses of natural and semi-natural systems are projected to be substantially larger in the Brazilian Amazon and India in business-as-usual scenarios (Figure 5.8), but projected losses are substantially lower under assumptions of improved governance and increased agricultural efficiency (Lapola *et al.*, 2011; 2014). Cover by natural vegetation in India is already very low, and projected losses of natural vegetation are related to an increase in urban and crop area (Schaldach *et al.*, 2011). Scenarios for montane regions of Southeast Asia show much smaller land-use changes, with the primary driver of loss forests being increased crop areas (Fox *et al.*, 2012). In the Brazilian Amazon, past and projected losses of forest and other natural vegetation are primarily related to increases in pasture area (Lapola *et al.*, 2010). The Brazilian Amazon has been particularly well studied in terms of land-use scenarios, including scenarios that take into account climate change impacts. Natural vegetation cover is relatively high and scenarios range from very extensive deforestation, to strong reductions deforestation rates with deforestation rates that may depend heavily on the impacts of climate change on forests and on crop productivity (e.g., Nepstad *et al.*, 2009; Lapola *et al.*, 2010; 2011; 2014).

Total urban area is expected to triple between 2000 and 2030 (CBD, 2013). This trend is projected to be especially pronounced in Mediterranean habitat types, Guinean forest of West Africa, tropical Andes, Western Ghats in India and Sri Lanka. This will result in habitat loss (CBD, 2013). Many of the world's cities are located in biodiversity rich areas such as floodplains, estuaries and coastlines. Increased urbanization in these regions may lead to rapid habitat loss, for example in the Atlantic Forest Region in Brazil, the Cape of South Africa and coastal Central America (CBD, 2013). Therefore sustainable urban planning can help to preserve natural habitats.

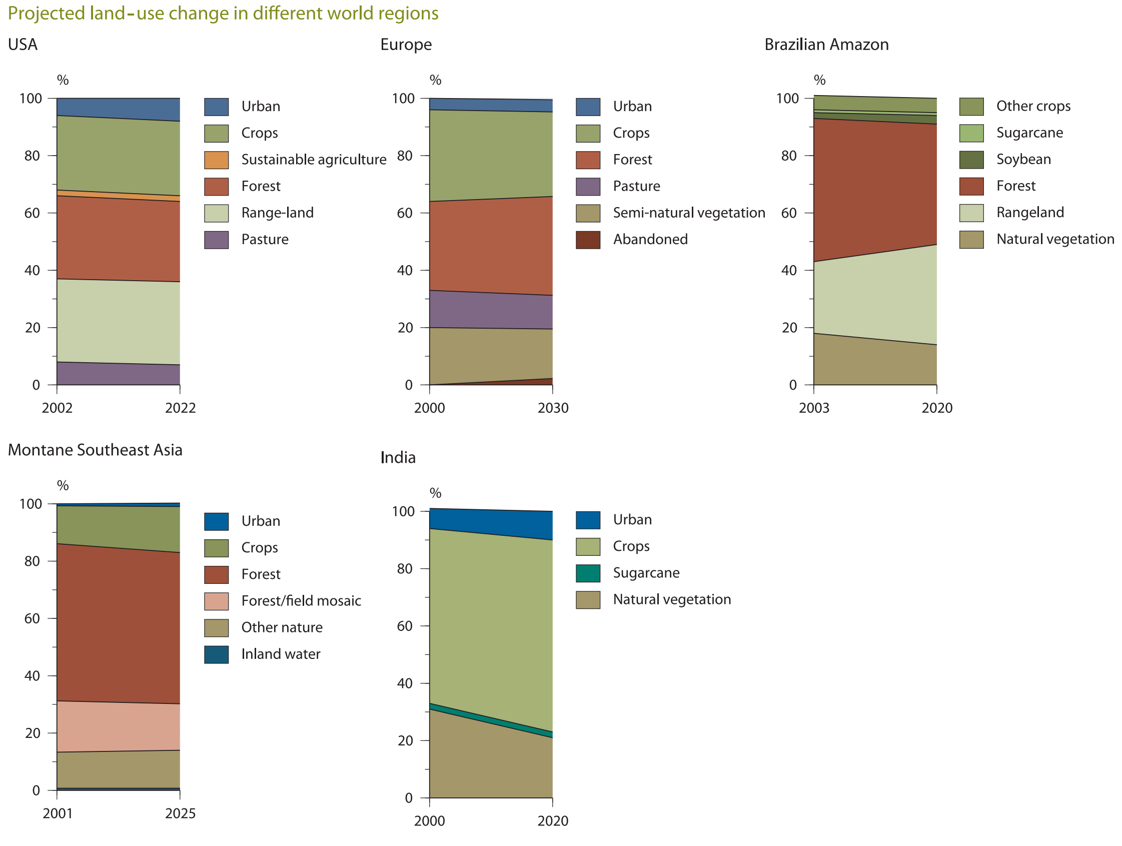


Figure 5.8. Examples of projected land-use change in different world regions A: Projected land-use change in the USA from 2002 – 2022. Data from Alig et al. (2010). **B: Projected land-use change in Europe from 2000-2030.** Data from Verburg et al. (2010) **C: Projected land-use in montane Southeast Asia from 2001-2025.** Data from Fox et al. (2012). **Land use categories in Fox et al. (2012) are summed. Land use category "crop" is composed of "crops" and "irrigated crops". Category "Forest" is composed of "evergreen needle leaf", "deciduous broad leaf", "evergreen broad leaf" and "mixed forest". Category "other nature" is composed of "shortgrass", "tallgrass", "sparse vegetation", "evergreen shrubs" and "deciduous shrubs". Category "inland water" is composed of "inland water" and "bogs and marshes". D: Projected land-use change from 2000 - 2020 in India.** Data from Schaldach et al. (2011). **E: Projected land-use change from 2003-2020 in Brazilian Amazon.** Data from Lapola *et al*. (2010).

Given global projected increases in agricultural and urban land cover, the trend in conversion of wetlands and fragmentation and degradation of freshwater habitats is also likely to increase up to 2020 (Martinuzzi *et al.*, 2013). In certain regions, increases in human land-uses have been tied to further projected declines in freshwater species diversity (Mantyka-Pringle *et al.*, 2014). In addition, population growth and resource use are also driving the expansion of dams for hydropower in regions such as South and Central America, parts of Africa, and Southeast Asia (Kareiva, 2012; Ziv *et al.*, 2012; Grumbine & Pandit, 2013). In South America for example, there are 2,215 planned hydropower projects, which entail adding dams to 673 rivers that are currently free of dams, and adding dams to 388 rivers that are already dammed (Kareiva, 2012; The World Bank, 2013). In addition to these proposals, there are an estimated 100,000 kilometres of roads crisscrossing the Amazon Basin – a pattern rapidly being observed across other growing regions, like the Congo Basin, Borneo and Siberia (Laurance & Balmford, 2013). Both dams and roads provide invaluable resources, such as energy and irrigation for agriculture, and also open up opportunities for improved movement of people and goods and services. This could provide food security for millions of people (Reidy Liermann *et al.*, 2012; Januchowski-Hartley *et al.,* 2013). However, these benefits do not come without a cost, since dams and roads negatively influence aquatic biodiversity (Collen *et al.,* 2013). In the Mekong Basin, construction of all planned 78 tributary dams, in the absence of main stem dams, would cumulatively have more impact and produce less energy than the six upper-most mainstream dams (the proposed Pakbeng, Luang Prabang, Xayaburi, Paklay, Sanakham, and Pakchom dams). Recent reviews of dam projects in India have recommended reductions in dam numbers even without including analysis of sediment load changes, road construction, climate change, and livelihood impacts (Grumbine & Pandit, 2013).

While there is growing awareness about the need for cost-benefit analyses prior to the placement of dams in order to minimize negative environmental effects and economic costs (e.g., Ziv *et al.*, 2012), these calls are going unheard and unactioned when it comes to policy and on-ground advancement of dam projects across the globe (Grumbine & Pandit, 2013). However, given that freshwaters, and the biodiversity that they support, are already one of the most threatened systems on the planet it is imperative that more soundly based decisions be made with regards to future impacts to these systems.

5.1.3 Country actions and commitments[1]

While most countries have targets related to habitat loss, few targets have been established which cover all elements of the Aichi Biodiversity Target. Many of the countries have established targets that refer to reducing the rate of habitat loss. A number of these have established national targets which exceed the Aichi Biodiversity Target. For example Finland has established a target to halt the loss of habitats. However, few of the targets from the remaining countries specify the magnitude of the reduction being sought.

Few targets explicitly address the issues of habitat fragmentation and degradation and few targets explicitly refer to habitat loss in aquatic environments.

Box 5.2: Federal Act on the Protection of Water in Switzerland

On 1 January 2011, the amended Federal Act on the Protection of Water came into force. It specifies that rivers and lakes in Switzerland must be close to nature, and defines measures and responsibilities, including the necessary delineation of adequate spaces and the strategic planning of revitalisation by the cantons. A study aiming at increasing the understanding of the added value of natural watercourses (flowing waters) was conducted. Based on the method of discrete choice experiments, the willingness to pay for restoration projects was explored for four specific rivers in Switzerland: the Dunnem, Some, Broye and Glatt rivers. The study revealed that:

- Rivers and streams are important elements in an attractive landscape for about 90% of the population.
- A large majority of the respondents (66% to 87%) find that the watercourses in their area are in a satisfactory condition, yet 73% to 80% consider that remodelling of watercourses would be worthwhile.
- The willingness to pay for a restoration project on the Dunnem river is 149 CHF per person. But for the Glatt river there is no significant willingness to pay for a restoration project. Willingness to pay (to the amount of 52 CHF) only emerges with an additional enlargement of the riverbank, that would provide, among others things, benefits for leisure.

Source: Switzerland 5th National Report to the CBD, 2014

Footnote

[1] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

A number of countries have set targets, or similar instruments, which refer to reducing the rate of habitat loss for specific types of habitats. For example, Malta has established a target related to preventing the loss of habitats that are of conservation value while Ireland has established a target related to the effective management of hedgerows and scrubland. Some countries have also established targets related to specific types of pressures on habitats. For example Suriname has an objective in its NBSAP related to reducing the loss of biodiversity resulting from mining pressure. In addition some countries have established targets related to promoting sustainable management, establishing frameworks or otherwise putting in place the institutional infrastructure needed to prevent habitat loss. For example, an objective of Belgium's NBSAP is to define the framework and conditions necessary to ensure no net loss of biodiversity or ecosystem services.

The national targets related to habitat loss, if implemented, would bring the world community closer to achieving Aichi Biodiversity Target 5. However as many of targets established to date do not specify the extent to which habitat loss is to be reduced by it is difficult to assess how close these commitments will bring us to the attainment of the target.

5.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

5.2.1 Actions

As illustrated above, deforestation in many tropical areas of the world is still increasing, and habitats of all types, including grasslands, wetlands and river systems, continue to be fragmented and degraded. Even though many countries have enforced and updated their forest policies and legislation to protect forests and other habitats in recent decades, additional actions should be taken to accelerate progress towards this target.

Forest and habitat protection from clearance and/or further degradation from fire and the overharvesting of timber and non-timber resources can be among the most effective instruments for conserving forests and other habitats (Beresford *et al.*, 2012). However, further efforts to improve education, monitoring systems, capacity and support from the judicial system are needed for National Forest Programmes to be more effective (Hardcastle & Hagelberg, 2012). Restoration and reforestation of degraded of converted forests have varying impacts on biodiversity, depending on location, scale, initial conditions, forest type and wider landscape context (Parrotta *et al.*, 2012).

Forest management by local populations is often advocated as a potential win-win for local people and biodiversity. There is evidence, that indigenous land in the Amazon is highly effective at inhibiting land conversion (Soares-Filho *et al.*, 2010). In addition, a study of community conserved areas in India found that they conserve biological values more effectively than open access areas (Shahabuddin & Rao, 2010). However, local populations may often have priorities other than biodiversity conservation, and evidence for the general effectiveness of community forest management is limited (Bowler *et al.*, 2010).

In addition to national objectives for land-use, spatial planning and protected areas, ***incentives for forest conservation*** such as Payments for Ecosystem Services (PES) can be effective. A PES program in Costa Rica has protected over 800,000 ha of forest since 1997 (Porras *et al.*, 2013). The objective of the UN REDD+ program has been to scale up this approach and implement it worldwide. Despite many challenges to wider application, over 40 countries are developing national REDD+ strategies and implementing REDD+ projects and supporting policies. This has resulted in extensive experience as to what works and what does not in different contexts and forest types (Angelsen *et al.*, 2009, Parrotta *et al.*, 2012).

Protecting forests and other habitats is unlikely to be effective unless the drivers of habitat loss are addressed. Therefore the main drivers affecting a habitat should be identified to be able to apply the correct actions. The main proximate causes of deforestation are agricultural expansion, infrastructure development and logging (Geist & Lambin, 2002). ***Limiting illegal logging*** could protects forest habitats, since 40-61% of timber production in Indonesia and 70% of the harvested timber in Gabon is believed to stem from illegal logging (Lawson, 2010). Another promising way of addressing these drivers is to meet demand for agricultural products and timber by ***increasing productivity*** on lands that have already been converted (the notion of sustainable intensification, to minimise other drivers of biodiversity loss, is discussed in Chapter 7). Improving infrastructure in productive areas close to population centres, while restricting infrastructure expansion into more intact, remote areas, could support such a strategy (Rudel, 2009).

The effectiveness of increasing the productivity of agricultural lands and managed forests to help reduce habitat loss – so-called land sparing (Phalan *et al.,* 2011) – depends greatly on local and landscape context. In some cases land sharing, where natural elements are integrated in a managed landscape, can be a more effective means of protecting biodiversity. It has been suggested that the relative utility of *land sparing vs. sharing* depends on the shape of the relationship between biodiversity and yield (Phalan *et al.,* 2011). However, this dichotomy may not be entirely appropriate in many circumstances because it glosses over the complexity of mechanisms maintaining biodiversity at multiple scales (Fischer *et al.,* 2014). Land sparing is an attractive concept, but poses two key risks. Typically, yield increases have negative impacts on biodiversity and some ecosystem services (the notion of sustainable intensification, to minimise these impacts, is discussed in the chapter on Target 7). A second risk is that higher profits from more productive land-use may incentivise rather than discourage further land conversion. This risk can be addressed by joint policies that simultaneously promote productivity and restrict agricultural expansion – this could be achieved through land-use planning, legal instruments, habitat banking, conditional PES schemes or other means.

A reduction of the demand for land can also be assisted by *changing consumption patterns* away from land-demanding products such as meat (especially beef), reducing waste and inequitable distribution to make better use of the food that is produced, and removing incentives for land-demanding biofuels (Stehfest *et al.*, 2009; Meier *et al.*, 2014; see also Target 7).

Box 5.3: Pathways for reductions in habitat loss: Brazil case study

This case study illustrates the factors contributing to the rapid reduction in deforestation in the Brazilian Amazon and Atlantic Forest over the past decade. These biogeographical regions had very high and rapidly rising deforestation rates at the end of the 20th century and up until 2004. This case study shows that the successful campaign to reduce deforestation has depended on a broad range of actions corresponding to Aichi Biodiversity Targets and Strategic Goals, but that significant challenges remain (Lapola *et al.*, 2014; Soares-Filho *et al.*, 2014).

Land-use and cover change (LUCC) in Brazil has become a national and global concern because natural habitats provide a wide range of ecosystem services including the maintenance of biodiversity, pollination, pest control, soil conservation, nutrient cycling, regulation of regional rainfall and hydrology, and carbon sequestration and storage (MA 2005; Oliveira *et al.*, 2013). Native vegetation still covers 62% of Brazil, an area totalling 530 million ha (Mha). These ecosystems are home to 17% of world's flora and 13% of vertebrate species (Raven, 1988; Myers *et al.*, 2000; BMMA 2006).

- The Brazilian Amazon lost about 20% of its forests between 1970 and 2012 (INPE, 2013). As a result, net emissions from land-use changes in Brazil from pre-colonial times to the present amount to 88±44 $GtCO_2e$ (Leite *et al.*, 2012), the equivalent of 12 years of worldwide emissions from land-use (Houghton *et al.*, 2008).
- The Cerrado is the second largest biogeographical region in South America. Conversion of vegetation has occurred over 50% of its area (LPIG, 2013; Figure 5.9). Its high level of endemic species, extensive arable lands, and 40 Mha of native vegetation that can be legally deforested (Soares-Filho *et al.*, 2014) make it one of Earth's 25 hotspots for biodiversity conservation (Myers *et al.*, 2000).
- Once the second largest forest in the Neotropics, the Atlantic Forest is heavily fragmented and only 12-21% of the original Atlantic Forest remains (BMMA, 2007). As 40% of its species are endemic, it is also a biodiversity conservation hotspot (Myers *et al.*, 2000).
- Caatinga, Pantanal, and Pampa, the other major biogeographical regions host large tracts of native vegetation that total 63 Mha.

Deforestation has declined rapidly since 2004 in the Amazon and Atlantic Forest - Recent efforts have reduced Amazon deforestation in 2013 by 70% below the historical 1996-2005 baseline of 19,600 $km^2\ year^{-1}$ (Figure 5.9). This reported reduction in deforestation rate is fully coherent with the most recent high-resolution global analyses of deforestation and "to date, only Brazil produces and shares spatially-explicit information on annual forest extent and change" (see section 1 of this chapter, Hansen *et al.*, 2013). Deforestation has also steadily declined in the Atlantic Forest despite a slight increase in 2013 (Figure 5.9). Current rates of deforestation of ≈200 $km^2\ year^{-1}$ in Atlantic Forest are in the same scale of estimated regrowth rates of 280 $km^2\ year^{-1}$ (Baptista & Rudel 2006; but see Teixeira *et al.*, 2009 reporting land-use intensification around large cities).

Box 5.3: Pathways for reductions in habitat loss: Brazil case study *continued*

Unlike the Amazon and Atlantic forests, deforestation rates in the Cerrado remain high. Cerrado deforestation rates have fluctuated between ≈3,000 to 9,000 km² year⁻¹ over the last decade (Figure 5.9), primarily due to expansion of agricultural lands, some of which may be driven by leakage from Amazon deforestation towards the Cerrado (Lapola *et al.*, 2014).

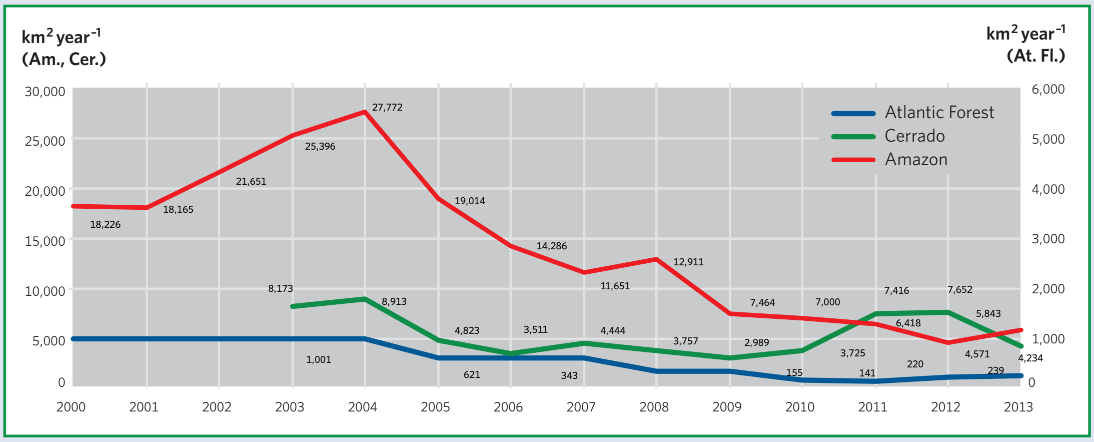


Figure 5.9. Deforestation trajectories in Brazil's major biogeographical regions. Source: Soares-Filho *et al.* (2014).

The rapid decline in deforestation in the Brazilian Amazon and Atlantic Forest are the result of a wide range of interrelated public and private policy initiatives. In 2004, Brazil launched the Action Plan for the Prevention and Control of Deforestation in the Amazon (BMMA, 2013a). This action plan included more efficient satellite-driven enforcement campaigns by Brazil's environmental agency (Börner *et al.*, 2011) to crack down on illegal deforestation and logging, as well as creation of protected areas (PA), including demarcation of indigenous lands (Soares-Filho *et al.*, 2010). Thus, moving towards the objective of strong reductions in habitat loss embodied in Aichi Biodiversity Target 5 has depended on many actions that correspond to a broad range of Aichi Biodiversity Targets and Goals as outlined below.

People have become aware of the values of biodiversity, incentive structures have been changed, and governments, businesses and stakeholders have implemented plans to reduce deforestation (Targets 1-4). Environmental sustainability has become the sixth top concern in Brazilian society (BMMA, 2012b) and Brazilians have among the highest awareness of biodiversity of all countries included in the "Biodiversity Barometer" survey (see Chapter 1). Government actions to reduce deforestation have included embargo of credit for rural landowners in municipalities among top Amazon deforesters ("black list"), reinforced investigation and enforcement by federal police and public prosecutors, and exclusion of illegal deforesters from supply chains. Private initiatives have included moratoria on soy grown on recently-cleared lands and bans on buying cattle from properties not on the rural environmental registry. These actions have created awareness among landowners that avoiding illegal deforestation is in their best interests.

Major efforts have been made to increase coverage of protected areas (PAs, Target 11). Approximately 40% of natural vegetation is legally protected by PAs; 7% is located on undesignated public lands and 53% on private properties (Figure 5.9, bottom inset). From 2002 to 2009, the Brazilian Amazon PA network expanded by 60%. A large part of these news areas were created in regions of intense land conflict to act as green barriers against deforestation, an important shift in PA paradigms (Soares-Filho, 2010).

Protecting and enhancing ecosystem services has been one of the main factors mediating reductions in deforestation (Targets 2 and 14). In spite of continued expansion of agricultural production, Brazil is strongly committed to achieving GHG (Greenhouse Gas) reductions from LUCC. Declining deforestation represents a reduction in GHG of 2.7 $GtCO_2e$ from the baseline. Overall, ecosystems in PAs store 117±22 $GtCO_2e$ (billion tons of CO_2 equivalents), while native forests and savannahs on private properties store 105±21 $GtCO_2e$ (Soares-Filho *et al.*, 2014). Thus, sound management of PAs and private landscapes in Brazil will be necessary globally to curb climate change and conserve biodiversity. In Atlantic forest, the myriad of forest remnants provide a wide range of ecosystem services including water supply and hydroelectric power for more than 70% of the Brazilian population.

Box 5.3: Pathways for reductions in habitat loss: Brazil case study *continued*

Challenges for 2020 and beyond - The goals of expanding agricultural production and enforcing forest conservation created intense political pressure that resulted in revision of the Law of Native Vegetation Protection (LNVP) – the key piece of legislation regulating environmental conservation on private properties and previously known as Forest Code. The LNVP prescribes that Brazilian landowners have a 21±1 Mha of forest debt – *i.e.,* illegally deforested areas – of which 78% encompasses Legal Reserve (LR - native vegetation set aside areas) and 22% Areas of Permanent Preservation (APP – native vegetation set aside areas around water bodies and on steep slopes). APP deficits must be restored and LR debts may be overcome by restoration or compensation using tradable environmental certificates (CRA) from properties with more vegetation than the legal requirements or by land tenure regularisation in PAs. An important concern is that both the old and new laws allow for an additional 88±6 Mha of legal deforestation on private properties (Soares-Filho *et al.*, 2014).

The LNVP introduces new mechanisms to address fire management, forest carbon, and payments for ecosystem services (PES), which could bring environmental benefits (Targets 2 and 3). It creates a mark*et al*lowing landowners to trade CRA and an online rural environmental registry system (SICAR) that streamlines registration of property boundaries and environmental information (http://www.car.gov.br/#/). This monitoring and documentation system of Brazil's 5.4 million rural properties could become an effective way to enforce conservation on private lands and help meet the target of restoring at least 10 Mha of native vegetation within the next 20 years (Target 15).

Additional conservation initiatives must also focus on consolidating PAs in the Amazon through programs such as ARPA – Amazon Region Protected Areas Program (Soares- *et al.*, 2010), as well as expanding the PA network outside of the Amazon (Target 11). Whereas PAs cover 46% of the Brazilian Amazon, PAs in the other regions are still below the 17% recommended in Aichi Biodiversity Target 11 and the Brazilian National Biodiversity Targets (BMMA, 2013b). PAs currently cover only 8.3% of Cerrado and 9.3% of Atlantic Forest (BMMA, 2014). In addition, it is necessary to upgrade extractive production chains in sustainable use reserves (Nunes *et al.*, 2012) and support the supply of certified timber by reducing the unfair competition by illegal logging (Target 7; Merry *et al.*, 2009). Additionally, the 39 Mha of undesignated public land, mostly located in the Amazonas states, represent an important opportunity for furthering a sustainable forestry policy, including concessions (Target 7).

Brazil must also continue to invest in its monitoring and enforcement capabilities (Targets 19 and 20). The widely acclaimed satellite-based deforestation monitoring systems maintained by the National Institute for Space Research (INPE) needs to be expanded to map land-use change outside the Amazon (Coutinho *et al.*, 2013). These monitoring systems must also be tied to land tenure certification, economic incentives (Soares-Filho *et al.*, 2014), productivity increases and PES, which will be critical to offset the high costs of forest restoration and the opportunity costs of forgoing agriculture rents.

Private initiatives are also essential for reconciling conservation with increased agricultural production (Targets 4, 5 and 7). This includes transparent and certified supply-chains, fire prevention and suppression, and boycotts of agricultural products grown in recently-deforested or high-priority conservation areas. Voluntary commitments to improving social and environmental performance and certification schemes are expected to improve farmer's access to PES, special markets or green investments. Overall, there is a need to guide the responsible expansion of agriculture while redoubling investments in environmental conservation, thereby transforming apparently divergent goals into complementary strategies (Targets 5 and 7). This effort is bolstered by Brazil's low-carbon agriculture program (ABC, agricultura.gov.br/desenvolvimento-sustentavel/plano-abc), which emphasises sustainable intensification of cattle ranching as one way to reduce pressure on forests and spare land for crop production, especially in the face of expanding domestic and international markets for agricultural products. One key concern in these land-use change polices is how to avoid rebound effects caused by financial gains related to productivity increase.

5.2.2 Costs and cost-benefit analysis

Deforestation provides benefits due to production of timber, the possibility to produce other goods and crops and financial benefits. For example, deforestation of montane forest in Kenya between 2000-2010 resulted in a benefit of US$16 million per year. However, by 2010, the cumulative negative effect of deforestation on the economy (through reduction in ecosystem services) was an estimated US$42.5 million per year, more than 2.8 times the cash revenue from deforestation (HLP2). Therefore, halting deforestation can reduce environmental costs, but to implement these actions several investments have to be made. For example, to effectively protect forest areas monitoring systems should be implemented, professionals should be educated and law enforcement should be enhanced. To set up a national monitoring system costs about US$0.5 million to US$2.0 million and annually US$400,000 are needed to implement the monitoring system (Eliasch review, 2008). To educate professional officers to enforce forest protection US$200,000 is needed. To enforce forest-relevant laws US$3 million annually is needed (Hardcastle & Hagelberg, 2012).

Table 5.1. Estimated Net Present Values in US$/yr/ha. Source: Eliasch review (2008).

Brazil	Indonesia	Cameroon
Soybeans: 3,275 Beef cattle (medium/large scale): 413 One-off timber harvesting: 251 Beef cattle (small scale): 3	Large scale palm oil: 3,340 One-off timber harvesting: 1,099 Smallholder rubber: 72 Rice fallow: 28	Cocoa with marketed fruit: 1,448 Annual food crop, short fallow: 821 Annual food crop, long fallow: 367

Halting deforestation may result in missed benefits. These opportunity costs of forest conservation vary widely, according to the returns from alternative land-uses (Table 5.1; Grieg-Gran, 2008).

However the benefits obtained by reduced deforestation outweigh the costs. For example reducing deforestation rates has been estimated to result in an annual benefit of US$183 billion, due to the high values of ecosystem services provided by forests (HLP2). For the Amazon, WWF estimated the benefits obtained by ecosystem service per hectare of forest (Table 5.2; WWF, 2009). One of those ecosystem services is carbon storage, on a carbon market this could potentially lead to an economic value of US$750 –10,000/ha. Also, non-timber forest products are an important contribution of household incomes in many Asian countries. In Southern Asia, the economic benefits of non-timber forest products are estimated at US$1,000 - 6,000/ha/yr. For forest dependent communities these benefits generally constitute 50-80% of average annual household income (HLP2). Notably, indigenous peoples and other forest-dependent communities gain a range of other non-economic benefits from forests, including food security, health security through access to traditional medicines, non-timber forest products for subsistence, fodder and building materials, among others.

Wetlands provide a wide range of ecosystem services including flood control, recreational and commercial fisheries, wildlife watching, hunting, amenities, habitat and storm protection. The economic value of these ecosystem services could be expected to range between US$125 and US$2,156 per hectare per year and enhance policy objectives related to coastal zone management, water quality, water infrastructure, climate and recreation (HLP2). Protection of wetlands could involve annual savings in expenditures on dams of US$5.7 billion and in other public water infrastructure of US$11.4 billion globally (HLP2).

Table 5.2. Overview of ecosystem services and associated economic values provided by the Amazon. Source: HLP2.

Ecosystem services	Economic value (US$)
Production of non-timber forest products	50-100/ha/year
Production of timber, net present value of Reduced Impact Logging (not necessarily sustainable production)	419-615/ha
Erosion prevention	238/ha/year
Fire protection	6/ha/year
Pollination of coffee plantations from forest (Ecuador)	49/ha/year
Disease protection	Unknown
Carbon storage - damage avoided due to CO_2 emissions avoided	70-100/ha/year
Carbon storage - value of total carbon stored in intact forest	750–10,000/ha
Maintenance of biodiversity	Unknown
Cultural and spiritual aspects of the forest	Unknown
Existence value	10-26/ha/year
Recreational and ecotourism use	3-7/ha/year

5.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

Habitat conversion and degradation are the primary drivers of biodiversity loss in terrestrial and inland water ecosystems (MEA, 2005). Therefore, the expected increase in habitat loss will also lead to an expected increase of biodiversity loss. Species that are habitat specialists can be good indicators of general health of the environment (BIP, 2014). Among the best-studied habitat specialists are common birds in North America and Europe. Long-term bird population indices are currently only available from North America (from 1968) and Europe (from 1980). Figure 5.10, which combines the wild bird index for North America and Europe, shows that specialist birds have declined by more than 20% since 1980 (BIP, 2014; Sauers *et al.*, 2014; EBCC/RSPB/BirdLife/Statistics Netherlands). The largest population declines have occurred in grasslands and arid lands in North America and in farmed lands in Europe (Figure 5.10, see also Target 7) indicating large biodiversity losses in those habitat types. However, widespread specialists of forest habitats show fluctuating, but stable trends in both North America and Europe.

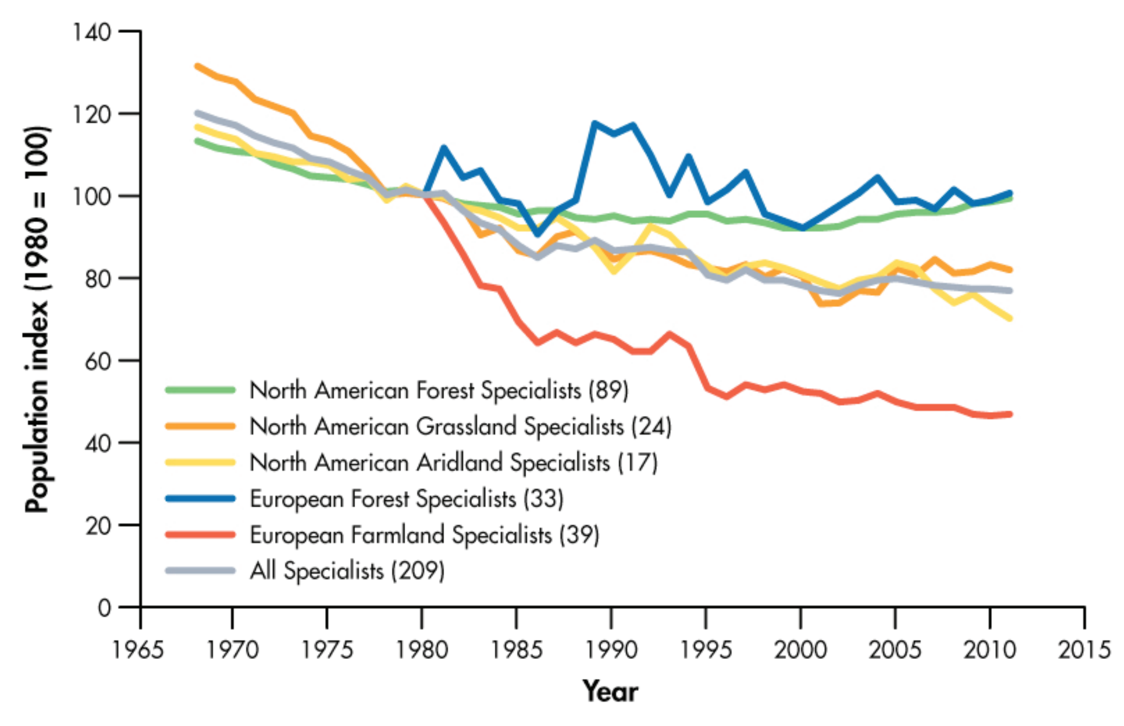


Figure 5.10. The Wild Bird Index for 209 habitat-specialist bird species in Europe and North America, showing the average population trends based on continental-scale systematic surveys and monitoring schemes. Sources: EBCC/RSPB/BirdLife/Statistics Netherlands; Sauers *et al.* (2014).

Farmland birds in Europe are ecologically similar to the grassland birds in North America. Both are dependent on semi-natural grasslands and other non-cultivated habitats in agricultural landscapes. These habitats are rapidly declining due to increase of intensive agriculture and forestry, and this is one of the key factors driving the decline in the Farmland Bird Index (Birdlife international, 2004; 2013). However, there are some suggestions that North American forest and grassland specialists and European forest specialists have been recovering in recent years, but it is uncertain whether this trend will continue (Figure 5.11; BIP, 2014). This is coherent with the net forest cover gain in these regions over the same period.

Overall, extrapolations from current trends suggest that the composite wild bird index across all habitat specialists in North America and Europe is projected to continue to decline, but level off by 2020 (Figure 5.11). In the future, these analyses may extended to other regions, which is critical because trends in bird populations are likely to vary substantial across regions due to highly contrasted trends in land-use change. Efforts are underway by several organizations to do so; for example, the Global Wild Bird Index project collates bird monitoring information and encourages the establishment of breeding bird surveys in countries and regions where none exist. National schemes have recently been successfully established in several African countries and Australia (BIP, 2014).

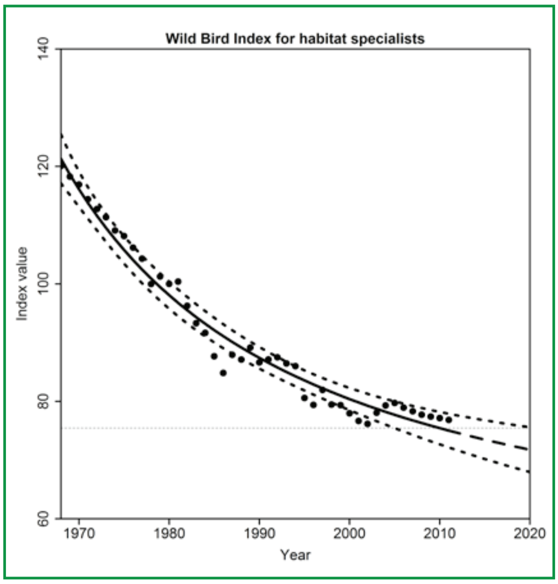


Figure 5.11. Statistical extrapolation of Wild bird index for all habitat specialists to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. The data are the average of habitat specific bird populations in North America and Europe (see black line in Fig 5.11. Source: BIP, 2014, Sauers et al., (2014), EBCC/RSPB/BirdLife/Statistics Netherlands.

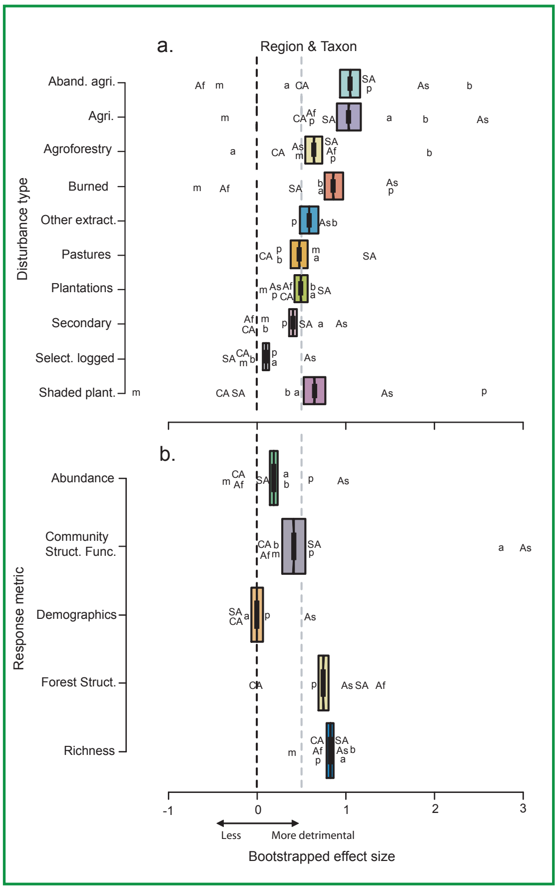


Figure 5.12. Impact on local species richness of conversion of primary tropical forests to other land-uses. A value of zero indicates that species richness is not different from primary forest and larger values indicate greater reductions in species richness. Source: Gibson *et al.* (2011).

All taxonomic groups, not just birds, are impacted by habitat degradation. Birds, however, are a very sensitive group, while mammals are generally less sensitive to disturbances (Gibson *et al.*, 2011). Small mammals and bats have a high tolerance to degraded forest and forest edges (Gibson *et al.*, 2011). However, the extent of the impact of land-use change depends largely on the type of conversion. Converting forests to arable land results in large impacts on local species richness and species composition because the habitat is completely changed. For example, converting primary tropical forests to cultivated agricultural systems has much larger impacts on species richness and composition than conversion to agroforestry or plantations (Figure 5.12; Gibson *et al.*, 2011). The effect of forest plantations on biodiversity is discussed in more detail in Target 7.

In addition to habitat degradation and conversion, fragmentation of the remaining natural habitats also influences biodiversity and species composition (Ewers & Didham, 2007; Tscharntke *et al.*, 2012). Fragmentation is a process in which a large habitat is transformed into a number of smaller patches of smaller total area, isolated from each other by a matrix of habitats unlike the original (Wilcove *et al.*, 1986; Fahrig, 2003). The expected increase in habitat loss, degradation and conversion might result in more fragmentation. Also the increase of infrastructure, leading to the isolation of the fragmented habitats, is expected to negatively influence biodiversity.

Fragmentation results in small patches of habitats, these remaining small patches contain fewer species than large habitats, since individual species have a minimum patch size requirement (Fahrig, 2003). Fragmentation might also lead to a declined reproductive success, reduced biotic interaction and increased local extinction rate (Fahrig, 2003; Aguilar *et al.*, 2006; Laurance *et al.*, 2011; Tscharntke *et al.*, 2012). Especially when corridors and stepping stones are absent, this prevents species to cross the matrix and to migrate (Watling *et al.*, 2011). In particular leaf bryophytes, tree seedlings, palms, birds, primates and larger herbivorous mammals are sensitive to the remaining patch size (Prugh *et al.*, 2008; Laurance *et al.*, 2011). A decrease of stepping stones and corridors for pollinators and seed dispersers may also affect plant communities (Aguilar *et al.*, 2006; Laurance *et al.*, 2011).

Efforts are made to protect the remaining fragments for further degradation and conversion, for example in the Amazon (Laurance *et al.*, 2011). However the shape of the remaining habitat also determines biodiversity and species composition (Ewers & Didham, 2007; Tscharntke *et al.*, 2012). The conservation of fragments with complex shapes will benefit edge-dwelling species, while large fragments will benefit more core-dwelling species. Habitats with complex shapes have higher perimeter-to-area ratio, increased amounts of edge-affected habitat and reduced core area (Ewers & Didham, 2007). Therefore habitats with more edge habitat benefit edge-dwelling species and increase the turnover rate and demographic variability (Ewers & Didham, 2007; Tscharntke *et al.*, 2012). Also for larger habitat fragments, the shape of the remaining habitat is important. Large habitat fragments often provide more core habitat and can support bigger populations; however, the populations in these cores could be spatially discontinuous. For example, habitats with convoluted shapes have large edge-penetration distances which can divide the core habitats into multiple cores. This can reduce habitat availability for core-dwelling species (Ewers & Didham, 2007).

Fragmentation also influences the landscape biodiversity, besides the biodiversity in the fragments (Tscharntke *et al.*, 2012). In a mosaic landscape, where the matrix connects the fragmented patches with stepping stones and corridors, many small fragmented patches spread over a landscape with a high environmental heterogeneity, will result in a higher landscape biodiversity than one large habitat. The small patches can cover a greater environmental heterogeneity, thereby maximizing the landscape-wide biodiversity (Fahrig, 2003; Tscharntke *et al.*, 2012). However when the matrix is hostile and does not connect the fragmented landscape with stepping stones and corridors the fragmented patches are isolated (Aguilar *et al.*, 2006; Watling *et al.*, 2011). This does not support the landscape biodiversity and results in declined reproductive success, reduced biotic interaction and increased local extinction rate (Aguilar *et al.*, 2006; Prugh *et al.*, 2008).

5.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

As illustrated above human land-use influences the extent of natural systems. However, climate change may also influence natural habitat extent by 2050. Therefore this section first describes the land-use change by 2050 as projected by socioeconomic scenarios. After which the expected effect of climate change on natural habitat extent is discussed.

4.1 Land-use scenarios for 2050 and beyond

A wide range of socioeconomic scenarios has been used to project land-use and land cover changes up to 2050 and beyond (Table 5.3). Pessimistic scenarios suggest that land-use and high levels of greenhouse gas emissions could lead to substantial loss of natural systems by mid-century. New, optimistic scenarios suggest that habitat loss and greenhouse gas emissions can be reduced based on several key changes in socioeconomic development pathways. These pathways depend on several factors including improved agricultural productivity, reduction of waste in food systems, changes in eating habits (e.g., healthy levels of consumption of calories and meat) and an increase in protected areas.

Global scale scenarios vary in their assumptions about the underlying socioeconomic development pathways that affect land-use change such as population growth, technological development and per capita consumption (see Chapter 0 for further details). Storyline approaches are based on scenarios where development continues in "business-as-usual" fashion (OECD, 2012) or scenarios that describe the future in the light of a range of plausible socioeconomic development pathways (e.g., GEO4 scenarios and MEA scenarios; MA, 2005; UNEP, 2012). This section also focuses on policy option scenarios that have been designed to test policy relevant options to reduce conversion of natural land. Examples of reduced conversion scenarios are high productivity scenarios (IAASTD, 2009; Wise *et al.*, 2009; PBL, 2012), protected area scenarios (PBL, 2010) and consumption change scenarios (Stehfest *et al.*, 2009; PBL, 2010). There are also a number of scenarios in which land conversion is increased above "business-as-usual" and these include climate mitigation scenarios (Wise *et al.*, 2009; PBL, 2010) and limited production increase scenarios (IAASTD, 2009).

Table 5.3. Examples of different types of land-use and land cover change scenarios. See Chapter 0 for a description of model types. The "Plausible socioeconomic development pathway" scenarios have been synthesised as a group of "Trend" scenarios in Figure 5.15 & 5.16. "Reduced land-use" scenarios are scenarios illustrating the effect of different policy options on land-use. "Climate mitigation" and "Limited agricultural productivity" scenarios rely on greatly increased land conversion compared to "business-as-usual" scenarios. Scenarios in italics have been singled out in Figure 5.15 because they represent extreme scenarios. The "Trend" scenarios also include several forestry options scenarios. Source: PBL, 2010.

Plausible socioeconomic development pathways = "Trend" scenarios	Reduced land-use scenarios	Limited agricultural productivity & climate change mitigation scenarios
"RIO+20 Trend" scenario (PBL, 2012)	Consumption change options: • Healthy calorie and meat consumption (PBL, 2010; Stehfest *et al.*, 2009) • No meat diet (PBL, 2010; Stehfest *et al.*, 2009) • Reduce food consumption and post-harvest losses as part of the "Rio+20 Consumption Change" scenario (PBL, 2012)	Agriculture and food system options: No-AKST scenario (agricultural knowledge, science, and technology; Alkemade *et al.*, 2013)
OECD "baseline" scenario (OECD, 2012)	Climate mitigation options: Low biofuel deployment (OECD, 2012; PBL, 2010)	Climate mitigation options with high biofuel deployment (OECD, 2012; PBL, 2010)
GEO4 scenarios (market first, policy first, security first and sustainability first; UNEP, 2004)	Agriculture and food system options: • High increase in agricultural productivity (PBL, 2010) • Reduction in post-harvest losses (PBL, 2010) • Broad increase in food system efficiency as part of the "Rio+20 Global Technology" scenario (PBL, 2012)	
MA scenarios (adaptive mosaic, global orchestration, order from strength and techno-garden; MA, 2005)	"Rio+20 Decentralized solutions" scenario (PBL, 2012)	

As highlighted in the Global Biodiversity Outlook 3 report and in Pereira *et al.* (2010), most storyline-based socioeconomic scenarios that have been developed to date result in relatively pessimistic views of land-use change over the coming century. The newest land-use scenarios for the IPCC AR5 report are no exception (Figure 5.13; Hurtt *et al.*, 2011). Very high rates of loss of primary habitats in the IPCC scenarios are associated with the low greenhouse emissions scenario (RCP2.6) as a result of massive deployment of bioenergy as a means of climate change mitigation. Or by an absence of pro-active measures to control land cover change (RCP8.5; Chapter 4; IPCC 2014). Somewhat lower reduction of primary vegetation rates occur in the RCP6.0 scenario (IPCC, 2014). This scenario is projected to lead to global warming of between 2.0°C to 3.7°C above pre-industrial temperatures by the end of the century (IPCC, 2014). This level of global temperature rise is projected to result in substantial displacement of species and biomes by the end of the century (IPCC, 2014, see discussion below). No option is foreseen by in the IPCC scenarios in which low impacts on biodiversity and strong climate change mitigation targets are met simultaneously.

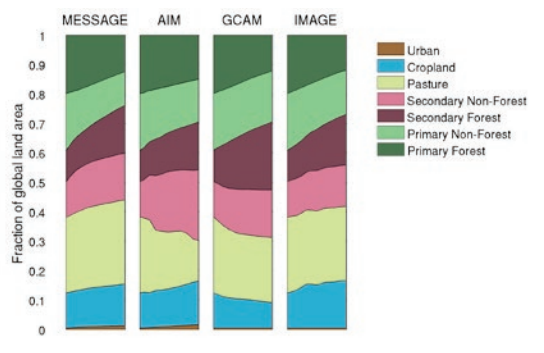


Figure 5.13. Land-use scenarios associated with the IPCC AR5 greenhouse gas emissions pathways (RCP). Source: Hurtt *et al.* (2011).

In most "Trend" scenarios (Table 5.3) rapid natural habitat loss is projected to continue up to 2050 (Figure 5.14; PBL, 2010; 2012; IFPRI, 2013). Agricultural expansion is projected to be especially fast in Southeast Asia and sub-Saharan Africa (PBL, 2010). World population and economic growth is anticipated to drive this rapid agriculture expansion up to 2030 (IAASTD, 2009; Conforti, 2011; IFPRI, 2013). After 2030 the pressure of agricultural land on natural habitat is anticipated to slow down, but not stop, due to declining population growth rates and increases in yield improvement (Alexandrator and Bruinsma, 2012; Conforti, 2011).

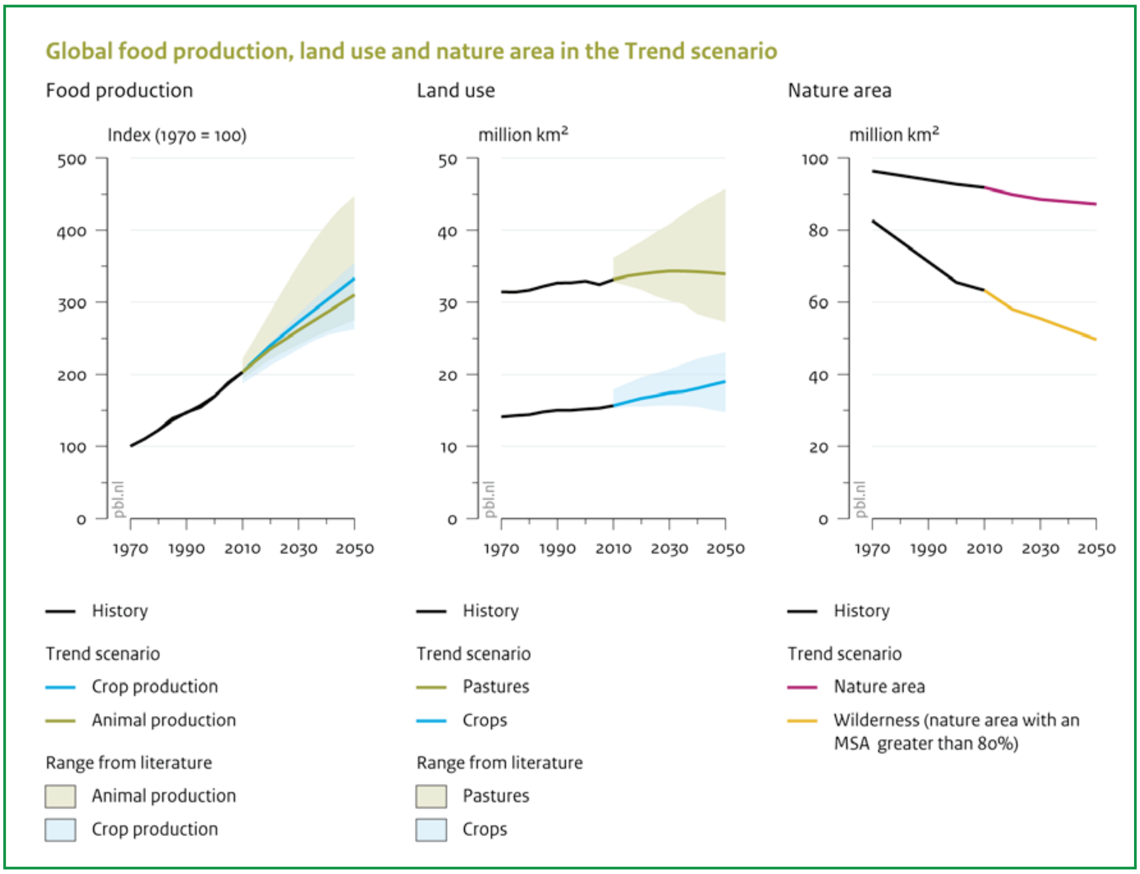


Figure 5.14. Global food production, land-use for pastures and crops and natural area for the "business-as-usual" scenario as described in (PBL, 2012) **from 1970-2050. The blue lines indicate the amount of crop production and cropland. The green lines indicate the amount of animal production and pastures. The blue and green areas indicate the range of other "business-as-usual" projections in the literature. The red line indicates the expected trend in the amount of natural area and the yellow line indicates the expected trend in the amount of wilderness area for the "business-as-usual" scenario as described in** PBL (2012)**.**

Several scenarios, as indicated in Table 5.3, provide insights into plausible means of reducing pressure on natural habitats through reductions in the rate of agriculture expansion (Figure 5.15). One of the potential keys is fast technological development in agriculture. An increase of 40% in crop productivity by 2050 could reduce agricultural area expansion by 6 million km² compared to the "Trend" scenarios, leading to a significant decrease in land conversion (IFPRI, 2013; PBL, 2010). This degree of improvement in productivity is high compared to many estimates, but is plausible given the productivity gains that are feasible in areas with high "yield gaps"; i.e., areas where there are large differences between current productivity and what is technically reasonable given environmental constraints (Mueller *et al.*, 2012; IFPRI, 2013).

Technological improvements in harvest efficiency, harvest storage and transport, and the reduction of wastes in the entire food chain, could reduce loss of food from producer to consumer. Current post-harvest losses are estimated to be about 30% of the total production (PBL, 2010). Reducing losses would thus lead a lower land conversion compared to baseline assumptions, as found in the "Reduction in post-harvest losses" scenario (PBL, 2010). In the "Healthy diet" and "No meat" scenarios less consumption of meat and dairy products result in a lower demand for grazing area and feed production compared to the baseline (PBL, 2010). This is projected to not only stop conversion of natural area into agricultural land, but also could result in recovery of abandoned agricultural area to their natural state. In the "Rio+20 Consumption change" scenario, combined reductions in postharvest losses and changes food consumption patterns result in substantially reduced agricultural area demand (Figure 5.15; PBL, 2012; IFPRI, 2013). However, if technology develops more slowly than foreseen, the projected demand for agricultural land increases as well as the pressure on natural habitat. This is illustrated by the "No AKST" scenario, where natural area drastically declines (PBL, 2010).

The expansion of agricultural land is not only influenced by the food and feed demand, but also by bioenergy demand (Alexandrator & Bruinsma, 2012; IFPRI, 2013). Bioenergy use can lead to lower CO_2 emissions; however, current biofuels increase demand for agricultural area at the expense of grasslands and forests (OECD, 2012). The scenarios which mitigate climate change and meet the 2°C target use bioenergy ambitiously, since it dampens the mitigation costs (Figure 5.15). In the "High biofuels" scenario, 25% of the energy demand is, for example, delivered by bioenergy (PBL, 2010). This drives land-use change from grassland into agriculture, especially in sub-Saharan Africa (Figure 5.15; OECD, 2012). Scenarios using more renewable energy and nuclear power or pursuing a less ambitious target reduces the pressure on grassland as occurs in the "Low biofuels" scenarios of OECD and PBL compared to "High biofuels" scenarios (PBL, 2010; OECD, 2012).

In addition to agriculture expansion, deforestation rates are also influenced by increasing demand for wood. Scenarios with increased production intensity in forest plantations can reduce deforestation rates. Due to the increased wood production from planted forests, the pressure on natural forests can be reduced, as illustrated in the "Improved forest management – high ambition" scenario in which 40% of the global wood demand is delivered by plantations (Figure 5.15; PBL, 2010). However in this scenario possible rebound effects were not included. As noted above, increasing production efficiency could also lead to rebound effects, and result in the expansion of cultivated areas and increased consumption (Maestre Andrés *et al.*, 2012).

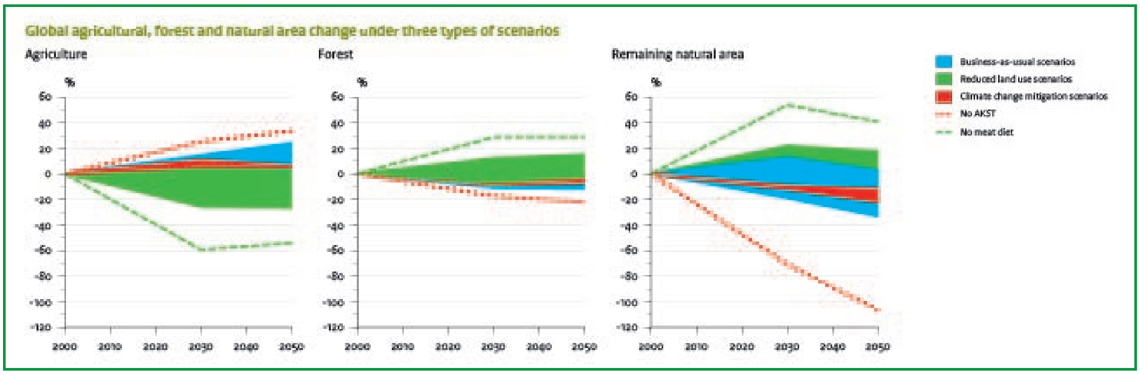


Figure 5.15. Global agricultural (including extensive grasslands and biofuels), natural and forest area under three types of scenarios from 2000- 2050. The blue area indicates the range of projections in "Trend" scenarios. The green area indicates the range of projections in reduced land-use scenarios. The red area indicates the range of projections in mitigating climate change scenarios. See Table 5.3 for an overview of the scenarios used in this analysis. The red dashed line illustrates a scenario with no improvements agricultural productivity. The green line indicates a scenario without meat consumption. Source: PBL, (2012).

Scenarios reducing agricultural expansion result in higher biodiversity in 2050, as calculated by Means Species Abundance (MSA) compared to the "Trend" (Figure 5.16). Scenarios with increased agricultural expansion compared to the baseline result in lower biodiversity values, due to slower technology development or increased biofuel plantations (Figure 5.16; Visconti *et al.*, 2011).

Fragmentation caused by agriculture also affects biodiversity, as described above (PBL, 2012; Powell and Lenton, 2013). Large, highly simplified agricultural landscapes harbour few corridors and stepping-stones for species to use as refuges. Therefore the remaining natural habitats are isolated which reduces the biodiversity (Figure 5.16). In mosaic landscapes, where agriculture and nature are interwoven, corridors and stepping-stones are present. This leads to an increase of MSA in agricultural areas and populations in natural habitats are no longer isolated.

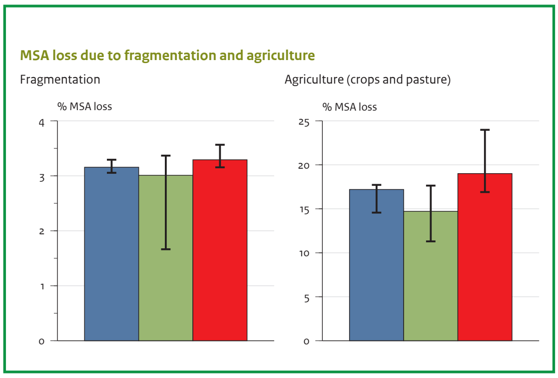


Figure 5.16. Biodiversity loss expressed as Mean Species Abundance (in % MSA loss) in 2050 compared to 2000 due (a) fragmentation and (b) agriculture for three types of scenario. The blue bars indicate the average % MSA loss in trend scenarios. The green bars indicate the average % MSA loss in reduced land-use scenarios. The red bars indicate the average % MSA loss in mitigating climate change scenarios. The error bars illustrate the range of projections from literature. See Table 5.3 for an overview of the scenarios. Source: PBL.

5.4.2 Climate change

In addition to the impacts of climate change mitigation efforts on natural habitats outlined above, climate change is projected to have large direct impacts on species and ecosystems (Allen *et al.*, 2010; IPCC, 2014). Mediterranean type ecosystems are especially vulnerable to climate change due to increasing temperature, rainfall change, increased droughts and increased fire frequency (IPCC, 2014). Forest ecosystems around the world are responding to climate change effects illustrated by indications of increased tree mortality across the globe due to increasing temperatures (warming) and decreasing precipitation (drought; Allen *et al.*, 2010). Heat and drought stress increase tree mortality and reduce reproductive success (Morin *et al.*, 2008; Frelich & Reich, 2010), while greater frequency of wild fires (Frelich & Reich, 2010; Gonzalez *et al.*, 2010), as well as an increase in forest-leveling windstorms (Frelich & Reich, 2010), contribute to tree mortality. In addition, warming may also be associated with a higher prevalence of pests and pathogens (Frelich & Reich, 2010; McDowell *et al.*, 2011), to which heat and drought-stressed trees show a higher susceptibility (Kurz *et al.*, 2008; Raffa *et al.*, 2008; McDowell *et al.*, 2011). Increased frequency of wild fires and tree mortality may lead to a shift from forest habitat to savanna ecosystem (IPCC, 2014). Next to this, on elevation gradients, increased CO_2 levels could lead to savanna boundaries moving into grasslands (IPCC, 2014).

Changes in climate may also alter competitiveness of tree species, resulting in changing co-occurrence pattern and forest communities (Meier *et al.*, 2011). These climate induced forest changes are predicted to occur mainly in temperate mixed forests and boreal forests of Eurasia and North America (Scholze *et al.*, 2006; Gonzalez *et al.*, 2010; Meier *et al.*, 2011), tropical forests of Central America and Amazonia (Salazar and Nobre, 2010; Scholze *et al.*, 2006), and the Mediterranean Basin (Hickler *et al.*, 2012). However, some of the heat- and drought-induced tree mortality might be offset by an increase in productivity due to CO_2 fertilization (Salazar & Nobre, 2010; Keenan *et al.*, 2011; IPCC, 2014).

In some regions, a warming climate may result in forest expansion, such as the expansion of boreal forest into tundra and taiga ecosystems (IPCC, 2014; Scholze *et al.*, 2006), a greening of semi-arid savannas and upward shifting of the tree line in Alpine ecosystems (Scholze *et al.*, 2006; Heubes *et al.*, 2011; IPCC, 2014). In temperate, arctic and alpine regions, these range expansions are driven by longer growing seasons and warmer winters (Hickler *et al.*, 2012), which are associated with a higher probability of fruit production and ripening and flower frost survival (Morin *et al.*, 2008). In agricultural areas increasing temperatures and elevated tropospheric ozone may reduce crop production, both in tropical and temperate areas (IPCC, 2014). While increasing temperature in high latitude regions positively influence agriculture (IPCC, 2014).

5.5 UNCERTAINTIES

Target 5 calls for loss of "all natural habitats" to be halved, and degradation and fragmentation to be "significantly reduced". While remote sensing data are useful for quantifying the rate of clearance of forest and some other habitats, they are less useful for quantifying habitat degradation and its causes.

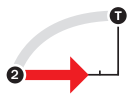
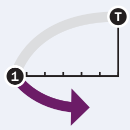
Since various canopy closure thresholds are used to define forest cover, differences in forest cover change between studies can arise (Hansen *et al.*, 2010). Therefore the forest loss rates between studies are hard to compare. For example regenerating forests of < 5meter tall are excluded in some studies, and large forest areas in the boreal region which are regenerating from fires and harvesting are thereby excluded. Also by using satellite images oil palm plantations can be marked as forest, while these agro-industries areas cause deforestation.

In addition, literature on tropical habitat conversion is regionally biased to Asia and South America. This implies that the findings on habitat conversion in tropical regions might be more generalised to Asia and South America. Therefore there is an urgent need for more research in tropical Africa (Gibson *et al.*, 2011).

The interplay of driving factors of agricultural intensification is considered to be context specific and not well understood (Keys & McConnell, 2005; Rudel *et al.*, 2009; Magliocca *et al.*, 2013). The wide range of representation and assumptions on the processes governing deforestation processes in response to food, feed, fuel and fibber demands and the role of land management in the models used to explore future deforestation provide a major uncertainty to scenario outcomes of biodiversity loss during the next decades (Hertel, 2011; Verburg *et al.*, 2013). Scenarios have not accounted for novel drivers of deforestation and habitat loss, e.g., tar sands in Canada and sea-level rise globally.

Birds are useful indicators of environmental health. They occur in all habitats, can reflect trends in other animals and plants, and can be sensitive to environmental change. Next to this population trend indices are available based on long-term systematic monitoring and robust sampling (BIP, 2014). However, long-term population trend indices are only available for two temperate developed regions. This means that data coverage is currently patchy and the wild bird index is not presently applicable at a global scale (BIP, 2014).

5.6 DASHBOARD – PROGRESS TOWARDS TARGET

Element	Status	Comments	Confidence
The rate of loss of forests is at least halved and where feasible brought close to zero		Deforestation significantly slowed in some tropical areas, although still great regional variation	Medium
The loss of all habitats is at least halved and where feasible brought close to zero		Varies among habitat types, data scarce for some biomes	Medium
Degradation and fragmentation are significantly reduced		Habitats of all types, including forests, grasslands, wetlands and river systems, continue to be fragmented and degraded.	Medium

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Box 5.3: Britaldo Silveira Soares-Filho and Carlos Alberto de Mattos Scaramuzz.
Extrapolations: Derek Tittensor
NBSAPs and National Reports: Kieran Mooney/CBD Secretariat
Dashboard: Tim Hirsch

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TARGET 6: SUSTAINABLE FISHERIES

By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits.

PREFACE

This chapter focuses on the present state of global marine and freshwater fisheries, the impact of fisheries on marine biodiversity and habitats, and the factors that impact upon fisheries sustainability. We first review the current status and trends of fisheries sustainability to determine whether the Aichi Biodiversity Target can be met by 2020. The second part of the analysis involves scenario modelling for marine fisheries, in which we forecast biological and socio-economic outcomes of fisheries in the short (2020) and long term (2050). Trends in marine fisheries to 2050 are assessed based on the Rio+20 Pathways and other analyses.

6.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

6.1.1 Status and trends

Recent studies agree that marine fisheries are, in general, overexploited. There is, however, disagreement on the extent of overexploitation, and the status and trends of global marine fisheries. For instance, Worm *et al.* (2009) showed that 63% of 166 assessed fish stocks (the majority of which were well managed, developed country fisheries) have lower biomass levels than required to obtain maximum sustainable yield (MSY). While rebuilding had not yet led to overall biomass recovery, nor reversed the general trend of increasing depletion of many individual stocks, these assessed stocks were found to have the potential to recover where low exploitation rates were maintained (Worm *et al.*, 2009). This has since been demonstrated for the Northeast Atlantic, where exploitation of the major fish stocks has declined significantly during the last decade and the biomasses of the stocks are rebounding (Fernandes & Cook, 2013). On the other hand, Branch *et al.* (2011) reported that 28-33% of assessed stocks are overexploited, and 7-13% are collapsed. They also stated that the proportion of fished stocks that are overexploited or collapsed has remained stable in recent years, and that rebuilding efforts for these fisheries have reduced exploitation rates.

There has been considerable debate on the use of marine catch data as an indicator of stock status (Pauly *et al.*, 2013). At the global level, catch trend analysis shows a less optimistic situation compared to stock assessments (Pauly, 2008; Froese *et al.*, 2012). According to the Food and Agricultural Organisation of the United Nations (FAO), 57% of assessed marine fish stocks are considered fully exploited (i.e., at or near the maximum sustainable yield), 30% are overexploited, and the remaining 13% are non fully-exploited (FAO, 2012). The percentage of overexploited stocks has remained in the 25-30% range for the past 20 years. The most recent State of the World Fisheries and Aquaculture report (SOFIA) indicates that in 2011, 28.8% of assessed marine fish stocks were considered to be fished at a biologically unsustainable level. Fully fished stocks accounted for 61.3% of assessed stock, while underfished stocks accounted for 9.9% (Figure 6.1; FAO, 2014).

Unlike the trend from stock assessments, the continuous declining trend from catch data does not stabilise (Worm & Branch 2012). Rather, Froese *et al.*, (2012) found that the percentage of non fully-exploited (under and moderately exploited) stocks has decreased gradually through time, whereas the percentage of overexploited and depleted stocks has increased (Figure 6.2).

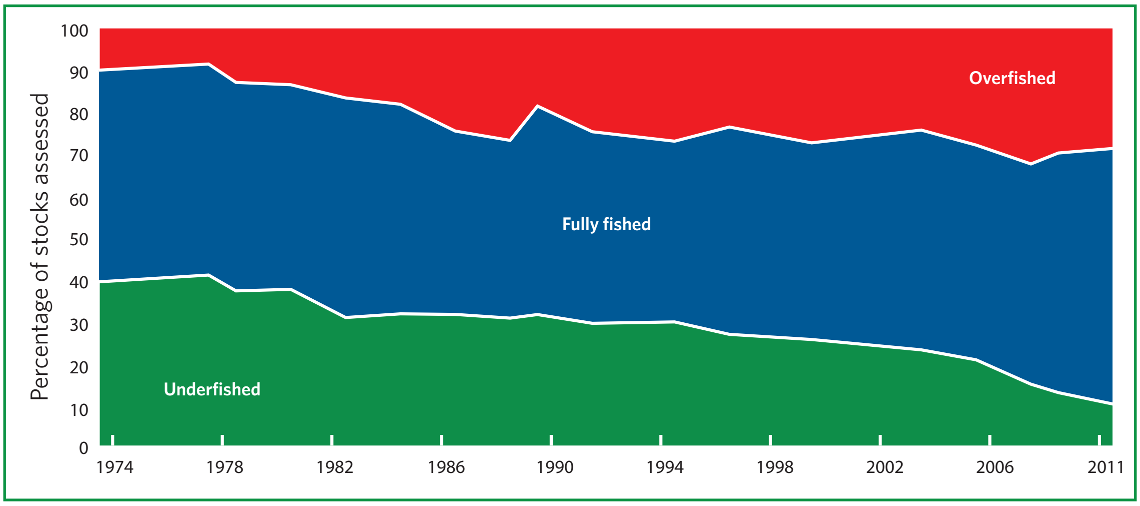


Figure 6.1. Global temporal trend showing the status of assessed world marine fish stocks. Source: FAO (2014).

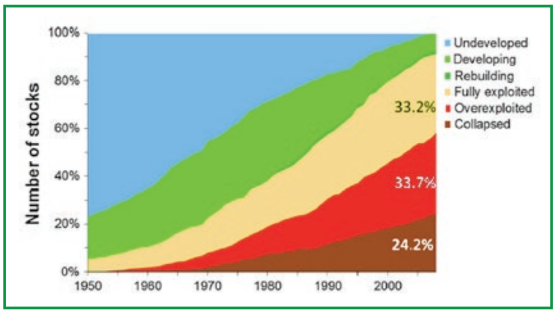


Figure 6.2. Historical trend on the status of fish stocks (1950-2008), showing an increase in the proportion of overexploited and collapsed fish stocks over time. Data is from the FAO catch database, and revised by Froese et al. 2012. Source: Froese *et al*. 2012.

An analysis of 8 indicators of fishing pressure, state, benefits, and responses of fisheries indicated an overall decline in global marine fisheries and long-term fisheries benefits (Ye *et al*., 2013). Furthermore, despite management and policy actions taken by coastal states, pressures on fisheries are increasing (Ye *et al*., 2013). In 2000 alone, overfishing resulted in potential catch losses that amounted to 7-36% of actual landed tonnage that year (Srinivasan *et al*., 2010). A subsequent reestimate of these numbers using an updated method by Costello *et al*. (2013) revealed that catch losses of 7-36% were actually low. Analysis of catch and primary production data also shows an increasing trend of ecosystem overfishing (i.e., overfishing that leads to an alteration in ecosystem diversity, productivity, variability, and species composition) from 1950-2000 (Coll *et al*., 2008).

There has been a global expansion of marine fisheries (Swartz *et al*., 2010) over the last 60 years of monitoring. Fishing effort measured in total kilowatt days shows an increasing trend, with nominal effort more than doubled from 1950 to 2010, suggesting a global decline in catch per unit effort (Watson *et al*., 2012) (Figure 6.3 a,b). The number of fishers also showed a temporal increase from 1970 to 2010 (Ye *et al*., 2013).

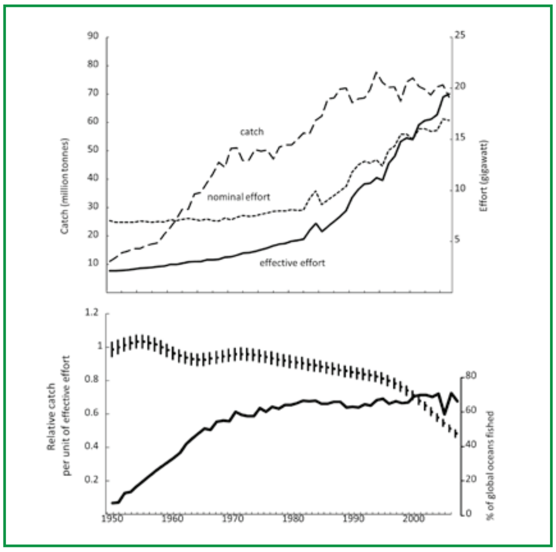


Figure 6.3 (a) Global trends in estimated fisheries catch and fishing effort (nominal and effective) 1950-2006. (b) Global trends in changes in standardised catch-per-unit-effort (CPUE) of global fisheries and percentage of global oceans that is exploited by fishing. The broken line represents CPUE and solid line represents % of ocean fished. Source: Watson *et al*. (2012).

Further, international fish trade has shown a growing trend in recent decades. Fish and fishery products account for about 10% of global agricultural product exports. The proportion of fishery production that is exported grew from 25% in 1976 to almost 40% in 2010, and decreased slightly to 37% in 2012 (Figure 6.4; FAO, 2014). This growth has been driven and facilitated by the globalisation of markets, sustained demand, technological innovations, trade liberalisation, and globalisation of food systems (FAO, 2012). Global trade in fish and fishery products has increased by 4.1% in real terms for the period 1976-2012 (FAO, 2014).

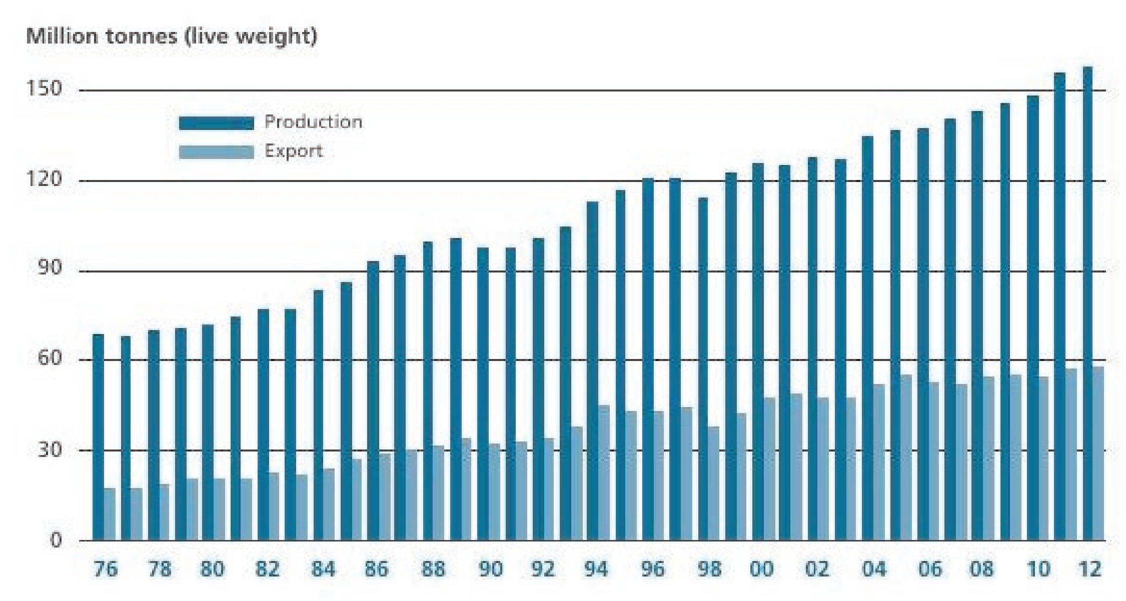


Figure 6.4. Temporal trend showing the annual quantity (million tonnes) of fisheries production destined for export.
Source: FAO (2014).

In a recent study of over 1793 unassessed fisheries, Costello *et al.* (2012) found that 64% of these fisheries had lower stock biomass than required to support MSY, and that 18% of unassessed stocks were collapsed. While all unassessed stocks were on a declining trend, 64% of these stocks could potentially increase sustainable harvest if they were rebuilt.

The persistence of overfishing in the world's oceans will continue to negatively affect marine biodiversity and ecosystems. A global assessment of 207 marine fish population trends indicated that the assessed marine fishes declined 38% between 1970 and 2007 (Hutchings *et al.*, 2010), and an analysis based on more than 200 ecosystems showed a global decline of 52% between 1970 and 2010 for predatory fish biomass (i.e. fish with a trophic level of 3.5 or more (Christensen *et al.*, submitted). In certain cases, fishing has driven population levels to such low levels that it results in collapse and local extinction of marine species (Dulvy *et al.*, 2003; Baum *et al.*, 2003). Currently, over 550 species of marine fishes and invertebrates are listed on the IUCN Red List as Critically Endangered, and Vulnerable. This may be an underestimate in itself, due to insufficient data to assess the conservation status of many marine organisms. In particular, many deep sea fishes and other large bodied, slow growing fishes are especially vulnerable to over exploitation (Cheung *et al.*, 2005; Norse *et al.*, 2012). 'Fishing down marine food webs' occurs when higher trophic level fish are progressively depleted, and replaced with lower trophic fish – a process that has been documented in many ecosystems (Pauly *et al.*, 1998; Pauly & Palomares, 2005; Stergiou & Christensen, 2011), albeit debated (Essington *et al.*, 2006).

Destructive fishing practices directly damage or modify habitat structure and heterogeneity, with resulting impacts on both target and non-target species (Turner *et al.*, 1999). The use of bottom trawls has increased globally in marine ecosystems (Watson *et al.*, 2006) (Figure 6.5). Bottom trawls directly impact benthic habitats, and can reduce overall biomass and shift the benthic composition towards small opportunistic species. The use of destructive fishing gears is of particular concern for vulnerable habitats such as coral reefs, seagrasses, cold water corals and sponge grounds, which are declining at accelerating rates worldwide (Waycott *et al.*, 2009; Burke *et al.*, 2011). Destructive fishing such as dynamite and poison fishing threatens over 55% of coral reefs (Burke *et al.*, 2011), and also contribute directly to seagrass loss, which is occurring at a rate of 7% per year since 1990 (Waycott *et al.*, 2009).

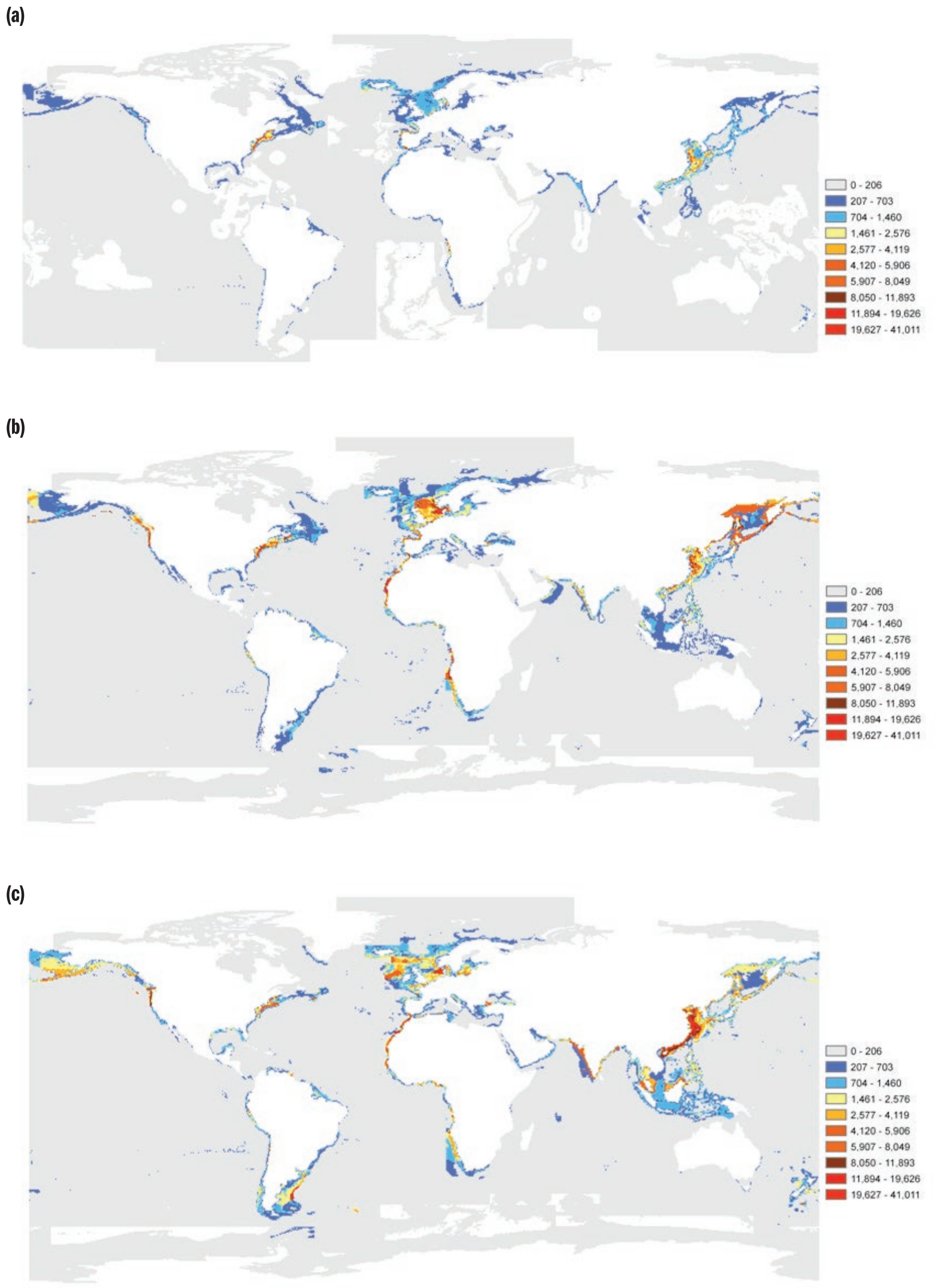


Figure 6.5. Global expansion of bottom trawling. Maps show global distribution of catches from trawling at different time periods (a) 1950-1960; (b) 1970-1980; (c) post-2000 (units are tonnes of catch). Source: Based on the database of Watson *et al*. (2012)

Unselective gears capture large quantities of bycatch, and is considered a primary driver of population declines in some species of marine megafauna (Wallace *et al.*, 2010). Bycatch here is defined as the unintended catch of non-target fish during fishing, whether discarded or not. Read *et al.* (2006) estimated that over 600,000 marine mammals were caught globally as bycatch every year. Turtle bycatch alone is estimated to be around 85,000 turtles a year (Wallace *et al.*, 2010). This has serious consequences for the conservation of marine turtles, as 6 out of 7 species of marine turtles are categorised as Vulnerable, Endangered, or Critically Endangered on the IUCN Red List. However, recent modelling analyses also suggest that non-selective fisheries may have lower impact on fish biomass and fishery sustainability than intensive selective fisheries (Garcia *et al.*, 2012).

The most recent estimate of marine fisheries bycatch worldwide is 38.5 million tonnes a year, representing about 40% of annual global marine catch (Davies *et al.*, 2009). Unutilised bycatch generates significant wastage - from 1992 to 2001, an average of 7.3 million tonnes of fish, or about 8% of the world's catch, was discarded annually (Kelleher, 2005). There has been a reduction in discards in recent years due partly to increased demand for previously discarded species or sizes, and from improved gear selectivity reducing catch rates of unwanted catch (Zeller & Pauly, 2005; Gilman *et al.*, 2013). Nevertheless, discards and bycatch are still not managed adequately at the regional level (Gilman *et al.*, 2013).

The impact of fisheries on biodiversity is further exacerbated by factors that directly affect fish populations, the physical marine environment, and ecosystems. Marine pollution has caused changes in the structure and function of phytoplankton, zooplankton, benthic and fish communities, and also caused interruptions in the life cycle and physiological development of marine organisms (Islam & Tanaka, 2004; Hutchinson *et al.*, 2013). Land reclamation, eutrophication, disease, and direct exploitation has led to the loss of wetlands, seagrasses, and other submerged aquatic vegetation (Lotze *et al.*, 2006). It appears that 41% of 20 assessed marine ecosystems worldwide are strongly affected by multiple anthropogenic drivers, and not one area is unaffected by human activities (Halpern *et al.*, 2008). The biological and ecological impacts of fishing also affect fisheries participants. In particular, industrial scale fishing has negatively affected the societal well-being of small-scale artisanal communities, including their livelihoods, subsistence economies, and culture.

Illegal, Unreported, and Unregulated (IUU) marine fishing threatens fisheries sustainability by distorting the accuracy of fisheries monitoring and assessments (Sumaila *et al.*, 2006). IUU fishing is estimated to take at least 35% of global catches (Agnew *et al.*, 2010), and has led to depletions of certain fish stocks (e.g., toothfish stocks, Osterblom *et al.*, 2010). On top of all this, climate change is expected to affect fish stocks, marine ecosystems, and biodiversity. Direct and indirect effects of climate change on marine ecosystems include changes in primary productivity, oceanographic conditions, shifts in the abundance and distribution of targeted fish species, and change in marine habitat quantity and quality (IPCC 2007; Cheung *et al.*, 2011; Sumaila *et al.*, 2011).

A key issue for improving the status of marine fisheries is the quality of fishery management. A positive trend is that the number of countries ratifying the UNCLOS (United Nations Convention on the Law of the Seas) increased annually since 1982, reaching 161 countries in 2010 (Ye *et al.*, 2013). However, an assessment of 53 countries that landed 95% of world fish catch showed that their overall compliance with the FAO Code of Conduct for Responsible Fisheries was low, with over 60% of countries failing and none obtaining an overall 'good' grade (Pitcher *et al.*, 2008). Although the FAO Code of Conduct is voluntary and states are encouraged to apply the codes they deem relevant, the overall poor performance demonstrates a low priority placed on fisheries management. Similarly, an evaluation of ecosystem based fisheries management found that out of 33 countries that landed 90% of world fish catch, over half failed, none received a 'good' rating, and only 4 were 'adequate' (Pitcher *et al.*, 2009). Further, a global assessment of overall management effectiveness found that only 5% of all Exclusive Economic Zones (EEZs) were in the top quarter of the scoring scale (Mora *et al.*, 2009). Importantly, high income EEZs had significantly better overall management than low income EEZs. Factors that contributed to low management effectiveness in high income EEZs were subsidies and excess fishing capacity, whereas deficient scientific, political, and enforcement capacity contributed to low effectiveness in low income EEZs (Mora *et al.*, 2009). On a more positive note, there are examples of successful fisheries management in rebuilding fish stocks (National Research Council, 2013). In addition, implementation of co-management models at the community level was found to be associated with successful fisheries (Gutiérrez *et al.*, 2011).

The Marine Stewardship Council (MSC) certification requires that for each fishery, target stocks are maintained at MSY or above, and that fishery impacts on ecosystems are minimised. As such, the number of MSC certified fisheries can be used as an indicator of progress towards achieving Aichi biodiversity targets. Since 2008, the number of Marine Stewardship Council (MSC) certified fisheries has increased by over 420% (Figure 6.6). MSC certified fish now represent around 9% of the global wild-capture (FAO, 2012), suggesting that at least 9% of catch is extracted within sustainable limits and with minimising impacts on marine ecosystems. The percentage could potentially be higher due to the presence of fisheries that are not certified by MSC but which are sustainably managed.

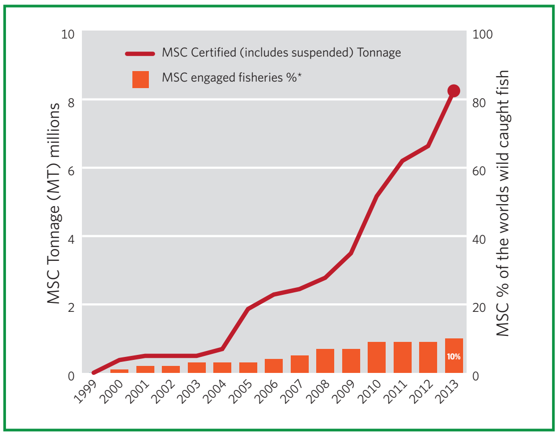


Figure 6.6. Trend in number of MSC certified fisheries. Source: MSC.

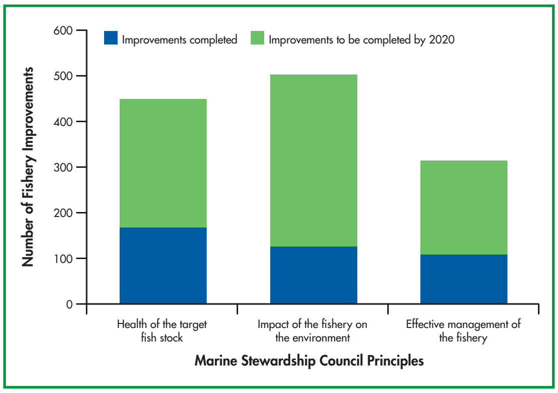


Figure 6.7. Number of fishery improvements completed and to be completed by Marine Stewardship Council certified fisheries by 2020. Source: MSC.

So far over 400 fishery improvements have been completed across the three MSC principles: (1) Health of the target fish stock; (2) Impact of the fishery on the environment; and (3) Effective management of the fishery (Figure 6.7). A higher number of improvements are expected to be completed by 2020. This increase in the number of MSC certified fisheries and improvements made highlights the continued commitment from fishers, seafood companies, scientists, conservation groups and the public to promote fisheries best practices through the MSC certification programme and seafood eco-label. At the same time however, it has been argued that MSC's sustainable fishing principles are too lenient and discretionary, suggesting that MSC certification may be misleading in some cases (Jacquet *et al.*, 2010; Christian *et al.*, 2013). On the other hand, other researchers have argued that the MSC certification process is credible due to its third party system, high compliance with FAO Guidelines for ecolabelling, and ability to raise objections to certifications (Gutiérrez & Agnew, 2013; Gulbrandsen, 2014). An outstanding issue is that currently, most MSC certified fisheries are industrial fisheries in developed countries. The application of MSC certification for small-scale fisheries, particularly those in the tropics, is a large challenge. The MSC is seeking to improve this and has identified developing country certification as a priority programme of work (http://www.msc.org/documents/developing-world).

Freshwater fisheries have received much less attention than marine fisheries, but may have even worse prospects for long-term sustainability. In contrast to marine catches, yields of freshwater fishes have increased continuously over the last few decades (FAO, 2012; 2014), even as the condition of rivers and lakes around the world continues to be degraded (Vörösmarty *et al.*, 2010; Carpenter *et al.*, 2011). Freshwater catches are notoriously under-reported due to their low-technology and geographically diffuse nature (Welcomme *et al.*, 2010; World Bank, 2010). Data on inland fisheries resources are extremely limited and poor in quality, and a few assessments rely on catch and landing data (Beard *et al.*, 2011). Yet, overfishing of inland waters may be going on unnoticed due to weak reporting (Allan *et al.*, 2005). Case studies suggest that actual global catches are 2-3 x higher than those reported to FAO (UNEP, 2010; World Bank, 2010). Pooling all reported commercial, artisanal, and subsistence fisheries together, global capture from freshwater ecosystems amounts to less than 10% of marine catches (FAO, 2012), and accounts for less than 13% of total global capture production (FAO, 2014).

The economic value of freshwater catches is lower per unit mass than that of ocean fishes. However, river and lake fisheries provide an accessible, low-cost source of animal protein for hundreds of millions of people in developing nations where alternative nutritional resources and employment opportunities are unavailable (UNEP, 2010; World Bank, 2010). There are no comprehensive, quantitative syntheses of the status of freshwater fisheries, but case studies from around the world generally indicate overfishing, including fishing-down of food webs (Allan *et al.*, 2005; UNEP, 2010). At the same time, freshwater fisheries are jeopardised by loss of native species, reconfiguration of food webs (Carpenter *et al.*, 2011), spread of exotic species (LePrieur *et al.*, 2008), physical and chemical degradation of freshwater habitats (Vörösmarty *et al.*, 2010), blockage of migrations by dams (Reidy Liermann *et al.*, 2012), and rapid shifts in water temperature and river flow due to climate change (Schneider & Hook 2010; van Vliet *et al.*, 2013). Together, these numerous indirect stressors are likely to undercut fishery sustainability at least as much as ongoing increases in fishing pressure, but quantitative analyses are lacking. It is also worthwhile to note that freshwater fisheries involve nearly no bycatch because fishes of all sizes and species are consumed for subsistence, as is also typical of small-scale marine fisheries (UNEP, 2010, World Bank, 2010).

Box 6.1: Case study: UK Fisheries

Current status and trends

The fisheries around the British Isles were already severely overexploited by the late 1900s. This situation is changing, however, throughout the Northeast Atlantic, including around the UK, where the proportion of fish stocks that are being harvested sustainably and are at full reproductive capacity has shown an increasing trend since 1990 (Figure 6.8). This sustainability indicator reached a maximum in 2011, at 47% of the 15 stocks for which accurate time series are obtainable from stock assessment reports (www.jncc.defra.gov.uk). Advice from the International Council for Exploration of the Sea (ICES) in 2012 has also indicated that many of these indicator stocks are being fished at or below the rate that will provide long-term maximum sustainable yield (MSY). The benefits of a push towards sustainability can be seen in stocks for which long-term management plans based on the MSY principle have been applied. In the North Sea, for example, haddock, herring and Norway lobster are currently being fished with increased landings and incomes for fishermen and coastal communities (www.ec.europa.eu). The proportion of fish stocks being harvested sustainably in UK waters may further increase following reforms to the Common Fisheries Policy (CFP). These reforms came to effect on 1st January 2014 and introduce a legally binding commitment to fish at sustainable levels, achieving MSY where possible, by 2020. CFP reforms also focus on allowing countries to work together regionally to implement measures appropriate to their own fisheries and on banning discarding.

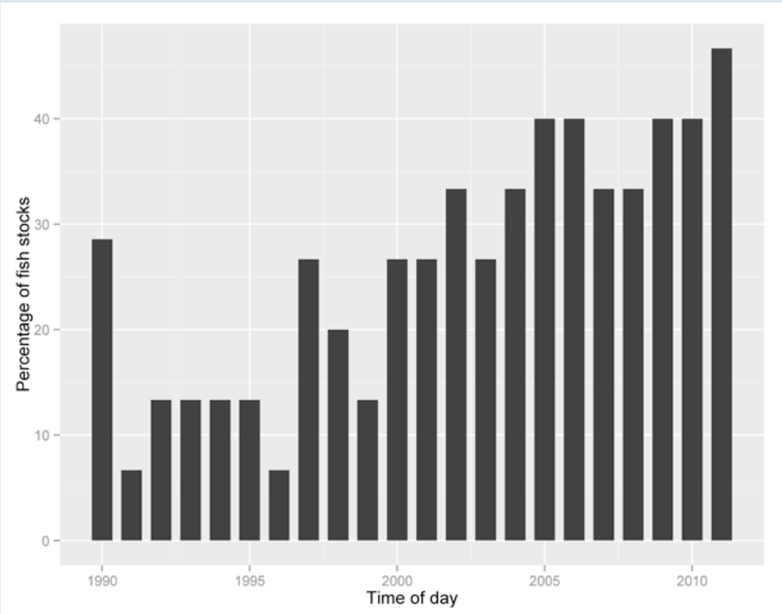


Figure 6.8. The percentage of fish stocks harvested sustainably and at full reproductive capacity, 1990 to 2011. Source: ICES Advisory Committee on Fisheries Management Reports; CEFAS (www.jncc.defra.gov.uk).

However, many marine species are also at risk of overexploitation though accidental catch, or bycatch. Species of elasmobranchs (sharks, skates and rays), for example, frequently suffer from overfishing and dangerously low population numbers. The UK government has thus expressed its commitment to ensuring that all elasmobranch fisheries are sustainable and that any endangered species are afforded adequate protection (www.gov.uk). In addition, the UK does not support the practice of removing shark fins before landing, and has called for mandatory 'fin on' landings to be introduced and properly enforced across the EU fleet through changes to European Council Regulation 1185/2003 (www.gov.uk).

Box 6.1: Case study: UK Fisheries *continued*

Projected climate change impacts

The moves towards sustainability described above may also help buffer adverse impacts of climate change and promote resilience within the marine ecosystem and fishing sector. In the European continental shelf, for example, a response to warming has been demonstrated in the abundances of 72% of the 50 most common species in UK waters (Simpson *et al*., 2011), while in the North Sea marine species were observed to move polewards by 22 km per decade (Perry *et al*., 2005) and deepened by 3.6m per decade (Dulvy *et al*., 2008). Immigrant fish such as sailfin dory have also recently been recorded around the southern coast of the UK for the first time, correlated with temperature data for the North Atlantic. Further, it is predicted that the majority of 31 key commercially targeted species will experience a decrease in environmental suitability by 2050 within the UK Exclusive Economic Zone (EEZ), resulting in shifts in species distributions under SRES A2 scenario (Jones *et al*., 2014). Potential distribution shifts, combined with projected deceases in primary production within the region are predicted to have a negative impact on the catch value obtainable within the UK EEZ, assuming fishing location does not change (Figure 6.9). Thus, climate change is predicted to result in a median 10% decrease from current levels of profitability, which were calculated at 36.2% over a 45-year time period (2005 - 2050). Outlying projections from alternative modelling procedures present a best (3% decrease) and worst (19% decrease) case scenario of predicted change. If fuel price increases according to observed trends, profitability is projected to decrease further. When catches reflect the rebuilding of stocks to their maximum level, a large increase in profitability is observed, to 61.7%. The impact of climate change on species distribution causes this profitability values to decrease to 59.4%.

Understanding the potential impact of climate change on fisheries is particularly important to achieve Aichi Target 6 in the longer term, through effective implementation of the EU Marine Strategy Framework Directive (2008) and Fisheries 2027 (DEFRA, 2007). If the relationship between the environment and fisheries subject to climate change is not properly understood, indicators such as those for individual species may either not be achieved or will provide a misleading assessment of a population's status.

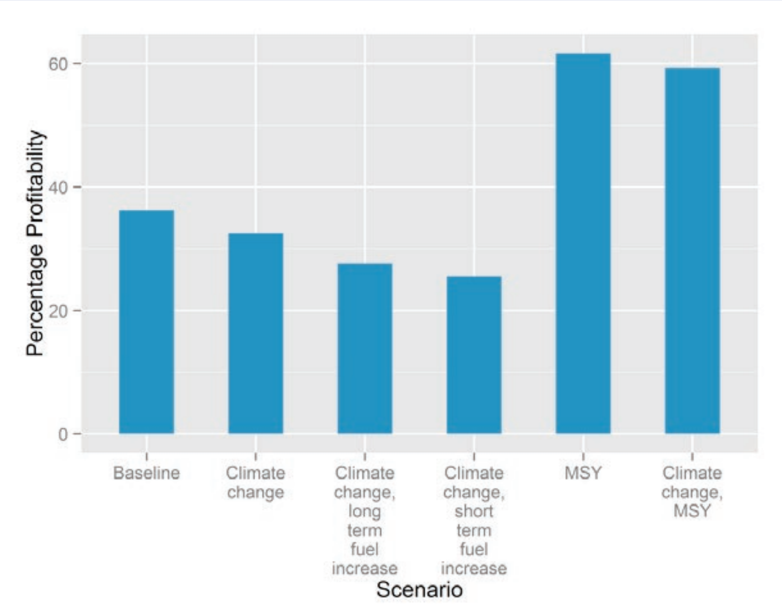


Figure 6.9. Percentage profitability of UK fisheries within the UK EEZ over the period 2005-2050 under different scenarios. These describe continued fishing at current levels (Baseline), the impacts of climate change, increases in fuel price and the introduction of management objectives to return stocks to levels approximating their Maximum Sustainable Yield (MSY) Source: Jones *et al*., 2014.

6.1.2 Projecting forward to 2020

Demand for fish is expected to grow (Delgado *et al*., 2003; Garcia & Rosenberg 2010, World Bank, 2013), as fish and fishery products will continue to be highly traded, with 36% of world fish production projected to be exported in 2022 (OECD, 2013). Combined with the continued spread of human impacts from the coastal zone to deep sea, it is expected that past trajectories of biodiversity loss and reduced ecosystem resilience will forecast future changes in the ocean if no measures are put in place to stop the trend (Lotze *et al*., 2006).

It is possible for marine ecosystems to recover if exploitation rates are substantially reduced. Despite the extended time needed for marine species and ecosystems to recover (e.g., fish stock recovery requires 4-26 years, while ecosystem recovery ranges from 10–42 years; Lotze *et al*., 2011), recent progress has been made in well developed and managed fisheries in North America, New Zealand, and Europe where current exploitation rates are predicted to achieve a conservation target of less than 10% collapsed stocks (e.g., Hilborn, 2007; Murawski *et al*., 2007; Worm *et al*., 2009; Branch, 2011). Therefore, although it is unlikely that all overexploited stocks will be restored to a level that can produce MSY by 2015 (Ye *et al*., 2013; FAO, 2012), innovative rebuilding policies and legislation can potentially shift the trend towards achieving the Target.

Given that the highest annual fisheries catch levels (80-100 million tonnes) has already been reached, it is unlikely that global fisheries catch will change significantly in the next 20-30 years (Garcia & Grainger ,2005, World Bank, 2013), unless substantial improvements in fisheries policy occurs. Projected trends of effective trawling effort show an increase to 2020 (Figure 6.10), and the proportion of fish stocks that are within safe biological limits is projected to decline (Figure 6.11). Thus, despite divergent views about the current status of global fisheries, it appears that having all fish stocks exploited at, or rebuilt to, safe biological levels (defined conceptually as biomass above biomass-at-maximum sustainable yield) by 2020 is unlikely, unless attaining the MSY objective is relaxed (Hilborn, 2010). Overall, notwithstanding several positive rebuilding results in developed country fisheries, the projection forward to 2020 will most likely reflect past trends – i.e., increasing exploitation rates in most of the world's fisheries (except for several developed countries and where market drivers make it uneconomic to fish), accompanied by declining catch rates and biomass of exploited species. Moreover, there is an urgent need to drastically reduce exploitation rates of vulnerable marine animals. A case in point is sharks, as a recent study indicated that global exploitation rates of sharks exceeded the potential for populations to rebound (Worm *et al*., 2013). Further, an example about the worsening status of species susceptible to bycatch is the projected decline in the IUCN Red List Index for seabirds to 2020 (Figure 6.12).

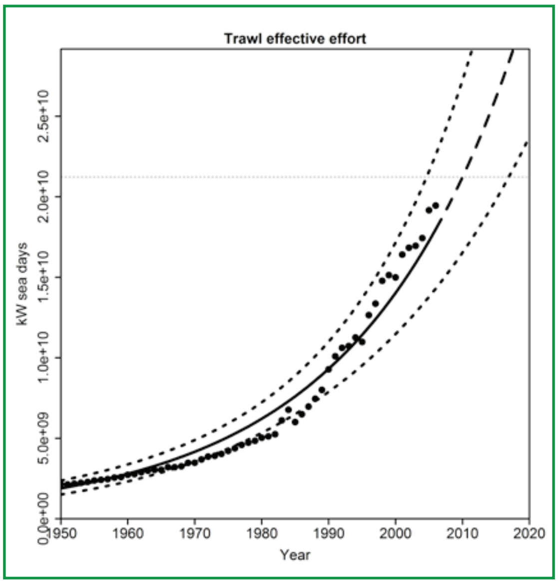


Figure 6.10. Projected trends in effective trawling effort to 2020. Data from recent trends are indicated by points, continuous lines indicate the fit to data, dashed lines are extrapolations to 2020 and dotted lines indicate the 95% confidence intervals. Source: Data is from the *Sea Around Us Project* database. Extrapolations are based on the assumption that underlying mechanisms continue to follow trends. Methods for model fitting are described in the introductory chapter.

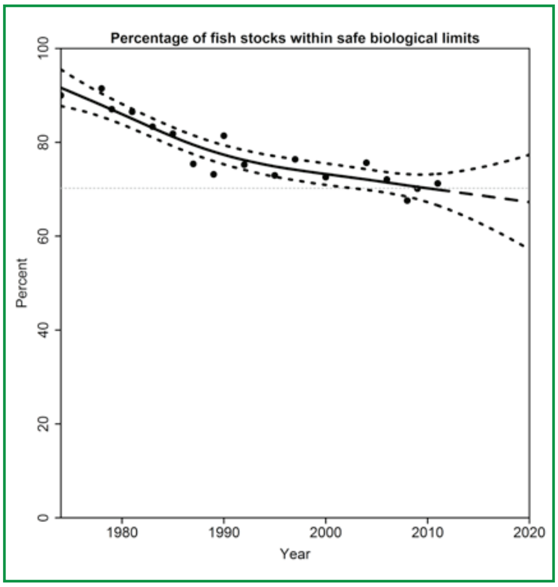


Figure 6.11. Projected proportion of fish stocks within safe biological limits by 2020. Safe biological limits is defined as the percentage of fish stocks exploited within their level of maximum biological productivity. Data from recent trends are indicated by points, continuous lines indicate the fit to data, dashed lines are extrapolations to 2020 and dotted lines indicate the 95% confidence intervals. Source: Data is from the *Sea Around Us Project* database. Extrapolations are based on the assumption that underlying mechanisms continue to follow trends. Methods for model fitting are described in the introductory chapter.

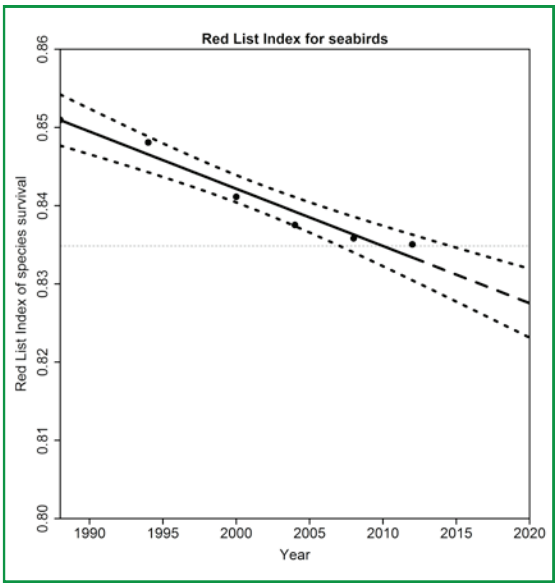


Figure 6.12. Projected Red List Index value for seabirds by 2020. Data from recent trends are indicated by points, continuous lines indicate the fit to data, dashed lines are extrapolations to 2020 and dotted lines indicate the 95% confidence intervals. Source: BirdLife International www.birdlife.org/datazone/sowb/indicators. Extrapolations are based on the assumption that underlying mechanisms continue to follow trends. Methods for model fitting are described in the introductory chapter.

The number of fisheries that has been under Marine Stewardship Council (MSC) certification increased steadily during 2000-2010, declined by 12% from 2010 to 2011 and by 22% from 2011 to 2012. With this decline in mind, the number of MSC certified fisheries in 2020 was projected based on (1) the assumption that the number of new certifications will correspond to the average for 2010 to 2012, and (2-4) will decrease with 10%, 20% and 30% per year, respectively. Based on this, the number of MSC certified fisheries is projected to reach 566, 458, 391, 342 by 2020, respectively, up from the 198 in 2012 (Figure 6.13a). An increasing trend is also projected for the tonnage of MSC engaged fisheries (Figure 6.13b). While the number of MSC certified fisheries is projected to approximately double by 2020, it must be noted that the certification is strongly centred on developed countries with approximately 90% of all fisheries (certified, in assessment, or suspended) coming from OECD countries. This raises the question if MSC certification, similar to forestry certifications (see Chapter 7) has reached a saturation point in the developed part of the world. In addition, it is clear that the biodiversity impact of certification is very limited in the developing part of the world, although this is expected to improve in the future as the MSC's Developing World Programme progresses.

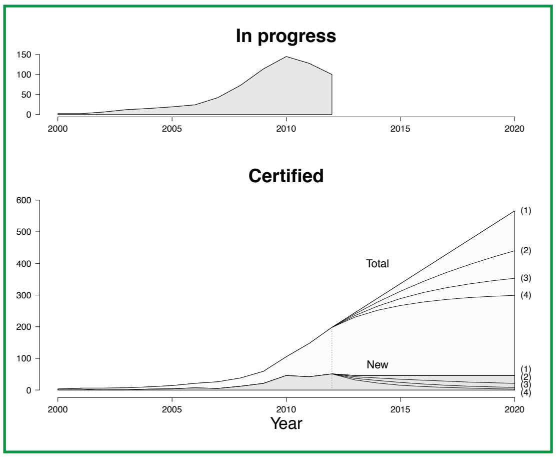


Figure 6.13a. Projected number of MSC certified fisheries by 2020. Four different projections are illustrated based on (1) average number of certification 2010-2012 will continue to 2020, (2-4) will decrease with 10%, 20% and 30% per year, respectively.

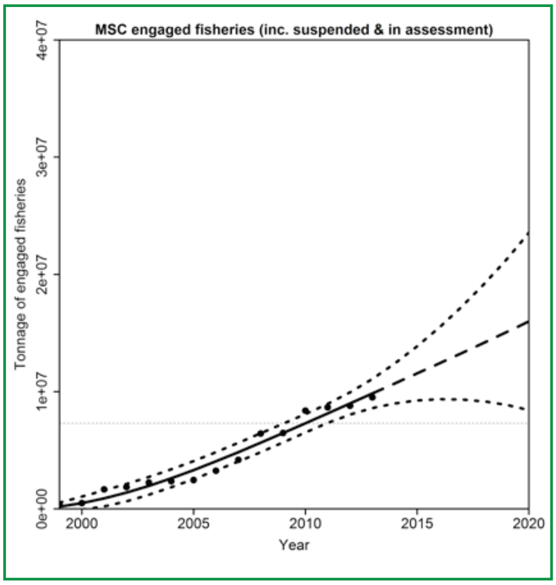


Figure 6.13b. Projected tonnage of MSC certified fisheries to 2020. Data from recent trends are indicated by points, continuous lines indicate the fit to data, dashed lines are extrapolations to 2020 and dotted lines indicate the 95% confidence intervals. Source: Data is from the Marine Stewardship Council. Extrapolations are based on the assumption that underlying mechanisms continue to follow trends. Methods for model fitting are described in the introductory chapter.

6.1.3 Country actions and commitments[1]

Slightly more than half of the national reports examined contained targets, or similar elements, which are directly relevant to Aichi Biodiversity Target 6. The national targets that have been established are generally in line with the global target however few address all of the different elements of the global target.

Several of the National Biodiversity Strategies and Action Plans (NBSAPs) identify specific priority actions which would create more sustainable fisheries. For example an objective of Belgium's NBSAP is to promote the implementation of good fishing practices in the North Sea while a key action goal of Japan's NBSAP is to promote initiatives that seek a balance between sustainable fisheries and the conservation of biodiversity. Further the Dominican Republic refers to the Code of Conduct for Responsible Fisheries in their targets.

Based on the NBSAPS considered it is likely that commitments will need to be scaled up if Aichi Biodiversity Target 6 is to be achieved by 2020.

6.2 WHAT NEEDS TO BE DONE TO ACHIEVE THE AICHI TARGET?

6.2.1 Actions

It is important to recognise that there is a limit to the amount of fish that the ocean can support. Therefore, rebuilding overfished stocks is crucial in order to achieve sustainable fisheries that deliver benefits through time for current and future generations. To do so, current excess fishing capacity has to be drastically reduced. This encompasses eliminating or diverting subsidies that contribute to overcapacity and overfishing, and stopping IUU fishing, which contribute to excess fishing capacity (Agnew *et al*., 2010; Sumaila *et al*., 2010a)[2]. Further, eliminating destructive fishing gears that damage marine habitat and have high bycatch is essential for minimising biodiversity and ecosystem impacts, as is adopting 'greener' fishing technology that minimises greenhouse gas emissions. Common management tools used to reduce exploitation rates include gear restrictions, creating marine protected areas, and the use of economic incentives (e.g., vessel buybacks, individual transferable quotas (ITQs)) to encourage reducing fishing effort (Worm *et al*., 2009). Social and economic assistance programmes that provide retraining and business or financial assistance have been used in some countries such as Canada, Norway, and Australia, and are important for helping displaced fishers transition to other employment.

ITQ systems are increasingly popular and their use is expanding worldwide – currently, ITQs are used by at least 18 countries for managing over 200 fish species (Chu, 2009). However, their use remains controversial. While it has been shown that ITQs prevented declining fisheries catch trends in certain well managed fisheries (Costello *et al*., 2008), a recent assessment of 20 ITQ managed stocks found that stock biomass did not respond positively in all cases after implementation (Chu, 2009). Further, as with other rebuilding tools, socio-economic costs associated with ITQs have to be weighed against their benefits, (e.g., Pinkerton & Edwards, 2009). Hence, ITQ programmes need to be designed carefully where they are appropriate (Sumaila, 2010). Importantly, ITQs highlight the need for a shift towards institutional change which emphasises the use of incentives in fisheries management (Hilborn *et al*., 2005; Pascoe *et al*., 2010). As an alternative to ITQs, systems of marine tenure in Chile allocate user rights and responsibilities to fisher collectives. This has improved the sustainability of interconnected social-ecological systems, and provides a potential model for improved governance of marine resources around the world (Gelcich *et al*., 2010).

Fisheries regulations have to be viewed as legitimate by stakeholders in order to gain their support and compliance. Devolution of governance to indigenous people and local communities, shared governance, and co-management arrangements are a means to attain this legitimacy, and have contributed to successful fisheries management outcomes (Gutiérrez *et al*., 2011), especially in small-scale fisheries in developing countries (Cinner *et al*., 2012). For example, coastal communities have demonstrated the ability to responsibly steward and manage marine ecosystems through a network of several hundred Locally Managed Marine Areas (LMMAs) in the South Pacific, as well as similar initiatives in Madagascar, Kenya, Spain, and Japan, among others. Given that the majority of the world's fishers are engaged in small-scale fishing (Béné, 2005), the use of shared governance and co-management arrangements is a promising action that can lead towards sustainable fisheries, bearing in mind that co-management can also overlook crucial dimensions of governance (Béne & Neiland 2006), and lead to undesirable social and ecological outcomes (Béné *et al*., 2009).

Footnotes

[1] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

[2] *It is also possible that in some cases the direction of causality is that excess capacity may lead to IUU fishing.*

At the same time, other authors caution against using a single management system, instead recommending that ecosystem-based management approaches perform best (Pitcher & Cheung, 2013). One example of ecosystem based management (EBM) being adopted on a regional basis is the Northwest Atlantic Fisheries Organization (NAFO) Roadmap for Developing Ecosystem Approach to Fisheries (http://www.nafo.int/science/advice/2013/13.pdf). Among the steps taken by NAFO to progress towards the implementation of an ecosystem approach include undertaking institutional reform, development of multi-species models, defining spatial management units, and creating joint managers and scientist working groups.

Market-based mechanisms such as certification, and individual conservation-led actions such as fishery improvement projects, have considerable potential to be effective tools. As noted above, there is a steady rise in the number and volume of MSC certified fisheries, delivering biodiversity targets. In addition, many actors are now working to incentivise fisheries to improve so that they can achieve certification. While the projections show that the entire world fisheries catch is unlikely to be certified by 2020, and progress in the developing world is slow, this is nonetheless a solution to be considered.

It should be noted that progress towards sustainable fisheries management is mainly being made in well developed fisheries in Europe and North America. For example, the United States Magnuson Stevens Act mandates that overfished fish stocks have to be rebuilt within 10 years. As of 2013, 21 of 44 fish stocks requiring rebuilding were considered to be rebuilt, while 7 had made significant rebuilding progress (Sewell *et al*., 2013). The European Union Common Fisheries Policy regulates all fishing activities in European waters. One recent study suggests that the Policy resulted in improved status for commercially exploited stocks in the North East Atlantic, North Sea, and Baltic Sea (Cardinale *et al*., 2013), although others have been less positive about the Policy's effectiveness (Froese *et al*., 2010). In 2013, the EU took action to move towards sustainable management of all fish stocks by 2020 by reforming the EU Common Fisheries Policy. Among the new measures are the use of fisheries based multi-annual management plans, as opposed to single stock plans; banning of discards; decentralised governance, and developing sustainable aquaculture. On a global scale, the international frameworks of Regional Fisheries Management Organisations (RFMO) provide a well-organised platform to appropriately manage fisheries resource toward 2020. At the same time however, some authors have indicated that the performance of RFMOs have been inadequate for managing fisheries in the high seas (Cullis-Suzuki & Pauly, 2010). Overall, however, rebuilding often only takes place once the fishery has experienced drastic overexploitation. In order to progress towards Target 6, fisheries managers have to put more emphasis on taking a precautionary approach to prevent overfishing, rather than reacting to it (Sumaila *et al*., 2011). At the same time, the need for feasible alternative livelihood or income options should be considered in conjunction with rebuilding measures. This is especially pertinent for developing country small-scale fisheries, where it has been argued that management interventions based on rent or wealth based models are inappropriate (Béné *et al*., 2010). Instead, the authors emphasise the need for policies that invest in areas such as fishers' health and education, governance improvement, and addressing justice and security. Importantly, rebuilding strategies have to take into the account the prevailing socio-political context in identifying feasible management options (Martinet *et al*., 2010). Further, rebuilding efforts will be hampered if actions are not taken to address global issues such as IUU fishing, the provision of fisheries subsidies (see Target 3), and management of shared and high seas fish stocks (Munro, 1979; Sumaila, 2013).

6.2.2 Costs and Cost-benefit analysis

It is estimated that excess fishing capacity costs approximately US$50 billion a year in net economic losses (FAO, 2009). Sumaila *et al.* (2012) estimated that fishing effort needs to be reduced by between 40 and 60% in order to generate resource rent from global fisheries. Resource rent in the study was defined as the surplus leftover after deducting fishing costs and subsidies from fishing revenue. This was equivalent to 2.6 million boats, and implies having to move 15–22 million fishers to other livelihood activities worldwide. To do so, it was estimated that governments have to invest between US$130-292 billion in present value for restructuring policies such as vessel buybacks. The implications on food security from removing so much fishing effort would need to be addressed, but this was not directly accounted for in Sumaila *et al*.'s (2012) study. Globally, a net gain of US$600–1400 billion in present value using a discount rate of 3% can be achieved over 50 years after rebuilding. Rebuilt world fisheries could increase current resource rent (net of subsidies) from US$13 billion to more than US$54 billion per year. However, benefits will only outweigh costs 12 years after rebuilding begins.

Another analysis estimated that in order to attain the WSSD (World Summit on Sustainable Development) target of restoring overexploited fish stocks to MSY levels by 2015, global fishing capacity had to be cut by 36-43% from the 2008 level (Ye *et al*., 2013). This would create unemployment for 12-15 million fishers and cost US$96-358 billion for implementing buybacks.[3] Achieving the WSSD target would increase annual fishery production by 16.5 million t (20% increase), and increase fisheries rent from negative to US$32 billion annually (Ye *et al*., 2013).

Footnote

[3] *Fisheries buybacks can be funded by government, industry, or cost shared between governments, NGOs, and industry.*

6.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

If Aichi Biodiversity Target 6 is not achieved, trends in marine species loss and degradation of marine ecosystems will likely continue, resulting in: i) decline in abundance of targeted and non-targeted species; ii) loss of keystone species and top predators from marine ecosystems, potentially causing shifts to alternate ecosystem states (e.g., coral-algal systems); iii) unsustainable levels of bycatch and discards. Bycatch of marine megafauna is of particular conservation concern as it threatens populations of many threatened and vulnerable species; iv) decrease in marine trophic index, impacting fish community structure and food webs; v) loss or degradation of marine habitats.

If Target 6 is achieved, there is potential for depleted marine species and ecosystems to recover. A recent review indicated that despite long histories of exploitation, recovery is possible for marine species populations and coastal habitats with the onset of protection (Worm *et al.*, 2006; Lotze *et al.*, 2011). Furthermore, rebuilding efforts has resulted in the recovery of some fish populations in well managed fisheries in developed and developing countries (Worm *et al.*, 2009, Fernandes & Cook, 2013). However, recovery may not be common among all marine animals - long term trends in coastal and estuarine ecosystems indicate that only 14% of depleted species (inclusive of large marine animals and fish) showed some recovery in the 20th century. Most of the recovered species were birds, pinnipeds, and sea otters, while other species continued to decline or remained in low abundance (Lotze *et al.*, 2006). Among fish, Hutchings *et al.* (2010) found that only 12% of 232 fish stocks had fully recovered 15 years after collapse, whereas 40% showed no recovery. It is suggested that recovery only occurs in 10 to 50% of species or ecosystems, thus indicating the need for much improved management and conservation (Lotze *et al.*, 2011). In addition, shift in ecosystems to an alternative stage and long-term changes in environmental conditions add to the uncertainties on the possibility and rate of recovery.

6.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

Current trends suggest that the high-end global greenhouse gas emissions scenario as described by the Intergovernmental Panel on Climate Change Fifth Assessment Report (IPCC, 2013) will continue in the next few decades, and will prevail by 2050. Climate change will directly and indirectly impact fisheries through physiological and behavioural effects of fishes, and physical and chemical changes in the environment (Brander, 2007; Sumaila *et al.*, 2011). Distributions of commercially important species are shifting as a result of temperature changes, mainly towards higher latitude and deeper water (Pinksy *et al.*, 2012; Perry *et al.*, 2005; Dulvy *et al.*, 2007). Changes in species composition of fisheries catches are shown to be partly attributed to long-term ocean temperature changes (Cheung *et al.*, 2013a). These changes are expected to continue into the next few decades.

Species distribution is projected to shift in the future, resulting in increased diversity and fisheries potential in high latitude regions, while the opposite will occur in the tropics (Cheung *et al.*, 2010). It is expected that northward movement of species will lead to more temperate species in northern European seas, while subtropical species move northward to temperate regions (Phillpart *et al.*, 2011). This is expected to affect endemic species, whose niches may be filled by species originating from adjacent waters (Sherman *et al.*, 2009). In certain areas such as the North Sea, distribution shifts will likely increase the risk to critically endangered species such as the common skate and angelshark (Jones *et al.*, 2013). Further, endangered marine megafauna such as turtles are deemed vulnerable to climate change; the combined pressures of human stressors and current rates of climate change are expected to have both positive and negative effects on turtle populations (Poloczanska *et al.*, 2009).

Fish population dynamics will be affected through effects on recruitment (Perry *et al.*, 2010; Jennings & Brander, 2010), while losses or gains in species driven by climate change is expected to change the composition of fish assemblages in the Mediterranean Sea within the 21st century (Albouy *et al.*, 2012). Maximum body size and growth of fishes are projected to decrease by 14-24% by 2050 (relative to current time), due to decreased aerobic scope as fishes live in increasingly warmer waters under the SRES A1 scenario (Cheung *et al.*, 2013b), affecting the yield-per-recruit of fisheries (Baudron *et al.*, 2014). In addition, ocean acidification and deoxygenation are expected to reduce habitats for exploited marine organisms and fisheries yield in some regions (Cheung *et al.*, 2011; AMAP Assessment, 2013). Therefore, there is a need to incorporate climate change and ocean acidification effects on food webs when carrying out stock assessments and management plans for the future (IGBP; IOC; SCOR, 2013).

There is a projected reduction of 2-13% in primary production by 2100, relative to 1860 (Steinacher *et al.*, 2010), with consequent effects on marine food webs and ecosystems under the SRES A1 scenario. For instance, climate induced changes in feeding conditions across European seas increased fisheries yields in northern seas while decreasing yields in southern and enclosed seas (Sherman *et al.*, 2009). Based on future net primary production changes, Merino *et al.* (2012) predicted that by 2050, marine fisheries for "large" fish species may increase by 6% in 69 EEZs, while those for "small" fish increase by 3.6% in the top fishmeal producing countries. Blanchard *et al.* (2012) project a decline of 30-60% in potential fish production across some tropical shelf and upwelling seas by 2050, while the production of pelagic predators is projected to increase by 29-89% in some high latitude shelf areas. Similarly, Cheung *et al.* (2010) projected a 30–70% increase in the fisheries yield of high-latitude regions but a drop of 40%- 60% in the tropics by 2055 relative to 2005 under the SRES A1B scenario. As well, Barange *et al.* (2014) predicted increased fish productivity at high latitudes and lowered productivity at mid to low latitudes.

Climate change is expected to make fisheries management more challenging as many commercially important fish stocks are likely to be affected by ocean warming, with resulting changes in fisheries catch potential (Sumaila *et al.*, 2011; Cheung *et al.*, 2012). At the same time, Merino *et al.* (2012) conclude that it is possible for marine ecosystems to sustain per capita fish consumption rates through to 2050 if effective fisheries management policies are implemented and technological improvements are made.

In the following section we examine future scenarios of fisheries for 2050 using two approaches:

6.4.1 Future fishing effort scenarios (FFES)

The effect of future fishing effort levels on marine biodiversity, fisheries, and aquaculture by 2050 were modelled by Wilting *et al.* (submitted), using a spatially stratified model (EcoOcean). They assessed three scenarios - increasing, constant, and reduced fishing effort levels:

- Increasing fishing effort following historical trends from the period 1970-2010 – this leads to rapidly declining fish stocks and catches. By 2050 fish stocks will be reduced by half, with some functional groups approaching total depletion by 2050 (Figure 6.14). To compensate for the loss in capture fisheries, aquaculture production is projected to increase from 67 million tonnes in 2010 up to 157 million tonnes in 2050.
- Fishing effort maintained at a constant 2010 effort level as estimated by Anticamara *et al.* (2011) for all marine fishing fleets to 2050 – stocks continue to be depleted while catches are reduced. The regions under highest threat include the Atlantic Ocean, Mediterranean Sea, and Black Sea, while the most affected species include commercially important species such as tuna, cod, salmon, haddock, and halibut. Aquaculture production is expected to increase to 148 million tonnes by 2050.
- Gradual reduction in fishing effort over a 10-year period starting in 2010, after which the level is kept constant at the reduced level to 2050 – stocks of most species are restored by 28%, and there is an improvement in biodiversity. Catches can increase to higher levels relative to current levels, within a period of 10 years. The increase in marine catches results in a lower demand for aquaculture production.

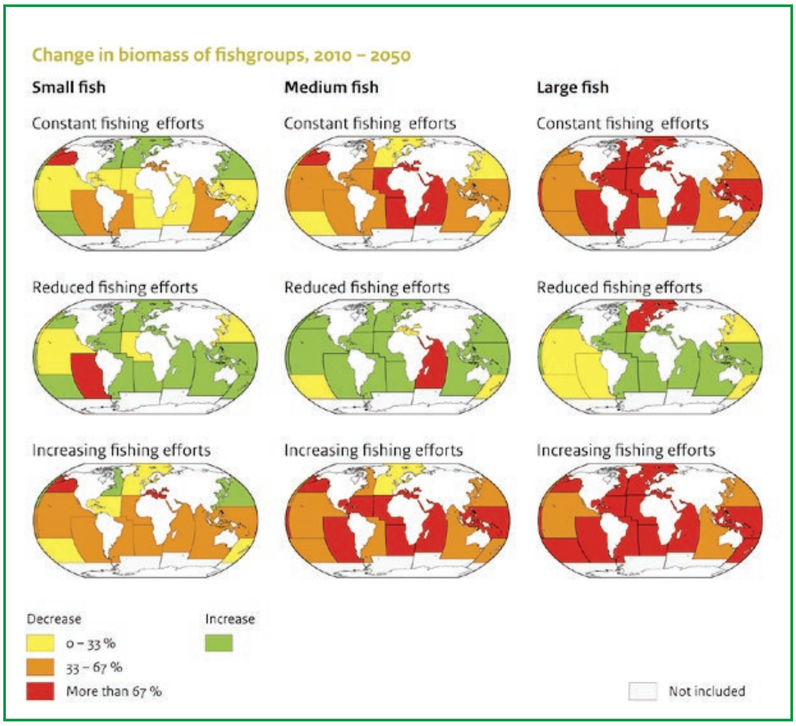


Figure 6.14. Relative change in biomass of small, medium, and large sized fish species under 3 fishing effort scenarios Source: Wilting *et al.*, submitted

The study notably concludes that if fishing effort continues to increase this will lead to depletion of fish species of all sizes throughout the world (Figure 6.14). With fishing effort kept constant to 2050, it is predicted that small fishes may increase, while medium and larger-sized fish populations will continue to decrease.

6.4.2 Rio+20 Pathways

(i) Method

Current (2011) reported global marine catch is 78.9 million tonnes (FAO, 2012), but annual sustainable catches of 82.7 million tonnes may be achieved by 2023 if rebuilding efforts take place immediately. We analysed global fisheries catch data from the Sea Around Us project (www.seaaroundus.org) with a population dynamics model developed by Martell and Froese (2012). In this analysis, we defined fishery stocks by species and FAO statistical area. We only included stocks with catch data reported at the species level. This resulted in a total of 1343 stocks within Exclusive Economic Zones (EEZs) and 537 stocks in the high seas, inclusive of fishes and invertebrates.

We applied the Catch-MSY method to each fish stock to simulate changes in biomass and exploitation rate. The Catch-MSY method is a biomass dynamic model that is run with time-series of catch removal (based on catch data) and a Monte-Carlo simulation (N = 100,000) of stock biomass with random values of the intrinsic population growth rate (r) and carrying capacity (K) (Martell and Froese 2012). The ranges of r and K values from which the Monte-Carlo simulations were drawn were based on the "resilience" categorisation of FishBase (www.fishbase.org) and the maximum catch of the time series (see Martell and Froese 2012 for details). Only those simulation runs that generated reasonable results were accepted ("reasonable results" were defined as having never collapsed the stock or exceeded carrying capacity, and the resulting biomass falling within the assumed range of depletion) (Martell and Froese 2012). The simulation time-frame was from 1950 to 2006. In addition, exploitation rate (E^t) was predicted for each simulation run from simulated annual biomass (B^t) and catch (C^t) at year (t) where $E \approx C/B$. We also estimated the maximum sustainable yield ($MSY = K*r/4$) of each major fish stock and the fishing effort required to achieve MSY (r/2).

Subsequently, we used the accepted Catch-MSY model to project future changes in stock size under different scenarios of fishing and climate change. We ran the Catch-MSY model for each accepted simulation from year 2007 to 2050. Scenarios of fishing were incorporated through assumed changes in exploitation rate over time. For climate change, we projected future changes in habitat suitability under climate change for each species using three different species distribution models: MAXENT, Aquamap and Dynamic Bioclimate Envelope Model (see Jones *et al*., 2013; Jones and Cheung, submitted for details). The climate change scenario that we considered was the Representative Concentration Pathway (RCP) 8.5 scenario (Moss *et al*., 2009) that is projected to lead to +4 °C increase in average global surface temperature by 2100 (IPCC 2013). We then calculated the projected changes in average habitat suitability for each stock (species-FAO area) for each year from 2007 to 2050. The estimated changes in habitat suitability were then used to drive the Catch-MSY model for future projections by assuming that changes in carrying capacity (K) of each stock in the future is directly proportional to changes in habitat suitability. We assumed that fisheries management does not adapt to climate change; thus reference points of biomass and exploitation rate at MSY remain constant over time for each stock under climate change.

Based on the simulation results from the Catch-MSY model, we calculated the proportion of stocks that have a high risk of overexploitation. We used a threshold of 40% of initial carrying capacity (K) without climate change effect as a criterion to define overexploitation; i.e., when stock biomass is below 0.4K, we considered the stock to be overexploited. For each stock, we counted the number of accepted runs generated by the Monte-Carlo simulation that were overexploited. If it was more often than not (i.e., probability >50%, binomial test $p < 0.05$) that the stock would be overexploited, we considered the stock to have high risk of overexploitation. This was calculated for each stock in year 2006 and in year 2050 under a combination of fishing and climate scenarios.

In addition, we applied the method described in Cheung *et al.* (2010; 2011) to predict the maximum catch potential (a proxy of MSY) of exploited fish stocks in the world ocean under scenarios of climate change. Firstly, the distribution of 850 species of exploited marine fishes and invertebrates was predicted for the current (average of 1991–2010) and future (average of 2041–2060) period, using the Dynamic Bioclimate Envelope Model (Cheung *et al*., 2009). We then used an empirical equation to predict maximum catch potential of each species based on the range area of the species and total net primary production within the range area.

(ii) Scenarios

Based on the Rio+20 Pathways, it is assumed that marine fisheries will be managed to rebuild overexploited or deplete stocks to MSY levels by 2050. Further, stock rebuilding, phasing out of bottom-impacting fishing gears, and lowered fishing effort will lower the overall impact on marine biodiversity. Potential scenarios for rebuilding marine fisheries to MSY levels in 2050 are as follows:

- ***Decentralised solutions:*** Under this scenario, fisheries management focuses on local and participatory, community based solutions. There is increased use of Individual Transferable Quotas (ITQ) and co-management initiatives. Seafood is caught and used locally, resulting in a shift towards targeting high value, inshore species and subsequent reduction in bycatch. An increase in fish prices can be expected due to increased targeting of high value species. Improved governance will lead to reductions in subsidies that contribute to overcapacity, which may raise fishing costs. The reduction in subsidies will also decrease distant water fishing (Sumaila *et al.*, 2010b), leading to a decrease in high seas catches while the coastal fishery increases (but within the limits of MSY). Overall, it is assumed that fishing effort will be reduced to sustainable levels, such that global fisheries catches are at a maximum sustainable yield level (100% MSY); the fishing mortality rate reduces from current levels to the MSY level (FMSY) by 2020. Increased attention on incentive based approaches such as ITQs may potentially decrease the tendency for non-compliant behaviour such as misreporting catches or illegal fishing.
- ***Global Technology:*** The need for intensive production means that pond based aquaculture of piscivores will likely increase, with a subsequent rise in demand for forage fish. It is therefore assumed that coastal (i.e., within EEZ) non-tuna purse-seine fishing effort will increase by 2% per year until 2020 to meet the increased demand for fishmeal and oil. In addition, improved fishing technology will enable fisheries production to shift to harder-to-reach resources, e.g., high seas fisheries. Overall, global fisheries catches and exploitation rates by 2050 are assumed to remain the same as the status quo (i.e., catches at 95% MSY). Due to the larger scale operations involved (e.g., deep sea trawling) and further travelling distances, fishing costs will likely increase, while fish prices remain fairly constant. Increased fishing activity in the high seas may deter IUU fishing in some locations as monitoring and surveillance technology become more advanced.
- ***Consumption Change:*** Seafood demand is expected to increase as consumers change from a meat based to fish based diet. Due to the use of 'greener' fishing technology, it is assumed that aquaculture will focus on more herbivores, and fishmeal and oil will be produced from recycling waste. Fishing effort of coastal non-tuna purse-seines is therefore assumed to decrease by 1% per year to 2020, while the FMSY level is reached for all other species by 2020. 'Greener' fishing technology also signals a reduction in bycatch and less energy consumption by fishing vessels. To encourage the shifts towards greener fishing technology, subsidies that reduce overcapacity and overfishing replace subsidies that encourage it. As such, fishing costs are expected to decrease in the long run, although there may be high capital costs associated with switching technology now. Overall there is a reduction in large-scale fishing such as deep water fisheries. Coastal fishing is maintained, and there is a reduction in bycatch. Therefore, it is assumed that populations of targeted high seas stocks are rebuilt by 2050, and high sea catches are at MSY, while coastal fisheries catch is maintained at the status quo (95% of MSY). The frequency of IUU fishing is assumed to be same.

These socio-economic scenarios were then combined with projections of climate change impacts on fish distribution and biomass.

(iii) Results

Under current conditions, the Northwest Atlantic had the largest proportion of EEZ stocks that was predicted to be overexploited, while the Arctic Sea had the lowest proportion. For high seas fisheries, the Northeast Pacific and Arctic Sea had the highest and lowest proportion of stocks predicted to be overexploited, respectively (Table 6.1).

Table 6.1. The proportion of stocks that are overexploited (i.e., probability of overfishing > 50%) in each FAO region in 2006.
Source: Produced by the authors of this chapter.

FAO Area	FAO Area name	Proportion of Overexploited Stocks EEZ	High Seas
18	Arctic Sea	0.36	0.25
21	Atlantic NW	0.80	0.75
27	Atlantic NE	0.59	0.86
31	Atlantic WC	0.62	0.60
34	Atlantic EC	0.66	0.79
37	Mediterranean and Black Sea	0.49	-
41	Atlantic SW	0.55	0.66
47	Atlantic SE	0.78	0.73
51	Indian Ocean W	0.49	0.33
57	Indian Ocean E	0.40	0.43
61	Pacific NW	0.71	0.74
67	Pacific NE	0.68	0.88
71	Pacific WC	0.47	0.32
77	Pacific EC	0.85	0.77
81	Pacific SW	0.62	0.51
87	Pacific SE	0.73	0.74

The status quo picture changes under the 2050 projections for stocks with >50% probability of being overfished (referred to as pof stocks hereafter) (Table 6.2):

- ***Decentralised solutions:*** Overall, this scenario resulted in lower pof stocks in both EEZ and high seas relative to current conditions. The Western Central Pacific and Southeast Atlantic (SW Africa) had the lowest and highest proportion of pof stocks in EEZs, respectively. The Southwest Pacific (SE Australia & NZ) had the highest proportion of pof stocks in the high seas, while no high seas pof stocks were projected in the Arctic Sea.
- ***Global Technology:*** Among the 3 scenarios, Global Technology projected the highest proportion of pof stocks in EEZs. In general, EEZ pof stocks were lower than the status quo, except in the Arctic Sea and Eastern Indian Ocean; the Eastern Indian Ocean had the highest proportion of pof stocks, while the lowest proportion was projected in the Northwest Pacific. Similar to current conditions, the Arctic Sea and Northeast Atlantic had the lowest and highest proportion of high seas pof stocks, respectively.
- ***Consumption Change:*** The proportion of pof stocks in both EEZs and high seas were lower relative to current conditions, except for EEZ stocks in the Arctic Sea. In EEZs, the Northwest Pacific had the lowest proportion of pof stocks, while the Eastern Indian Ocean had the highest proportion, followed closely by the Eastern Central Atlantic (W Africa), Southeast Atlantic (SW Africa), and Southwest Pacific (SE Australia & NZ). As with the other two scenarios, the lowest and highest proportion of high seas pof stocks occurred in the Arctic Sea and Northeast Atlantic, respectively.

Without consideration of climate change, all scenarios resulted in a considerable reduction in pof stocks by 2050 in both EEZs and high seas (Table 6.2). This is because in all scenarios, exploitation rates for most species are assumed to be set at a sustainable level (required to achieve MSY). The pof stocks are higher in the Global Technology and Consumption Change scenarios because fishing for small pelagic species are assumed to be intensified in these two scenarios. There are residual risks of overexploitation even under the Decentralised Solution scenario because a small proportion of slow growth and low productivity stocks that are currently overexploited may take more than 40 years to fully recover without a full fishing closure.

Table 6.2. The proportion of stocks with probability of overfishing > 50% by 2050 under the three Rio +20 scenarios. The values are reported as EEZ/High Seas. Source: Produced by the authors of this chapter.

		Proportion of Over-fished Stocks (pof stocks)		
FAO Area	**FAO Area name**	Decentralised Solutions	Global Technology	Consumption Change
18	Arctic Sea	0.29/0.00	0.50/0.00	0.43/0.00
21	Atlantic NW	0.32/0.33	0.46/0.33	0.44/0.33
27	Atlantic NE	0.31/0.42	0.43/0.42	0.39/0.42
31	Atlantic WC	0.19/0.28	0.44/0.28	0.42/0.28
34	Atlantic EC	0.30/0.33	0.50/0.33	0.49/0.33
37	Mediterranean and Black Sea	0.24/-	0.41/-	0.39/-
41	Atlantic SW	0.26/0.29	0.48/0.29	0.46/0.29
47	Atlantic SE	0.33/0.24	0.51/0.24	0.49/0.24
51	Indian Ocean W	0.16/0.10	0.48/0.10	0.48/0.10
57	Indian Ocean E	0.14/0.30	0.55/0.30	0.50/0.30
61	Pacific NW	0.13/0.26	0.35/0.26	0.30/0.26
67	Pacific NE	0.27/0.25	0.49/0.25	0.37/0.25
71	Pacific WC	0.11/0.16	0.52/0.16	0.40/0.16
81	Pacific EC	0.29/0.33	0.47/0.33	0.43/0.33
87	Pacific SW	0.29/0.41	0.51/0.41	0.49/0.41
Global (no CC)		0.24/0.30	0.31/0.30	0.28/0.3
Global (RCP 8.5)		0.42/0.67	0.49/0.67	0.46/0.67

Under the high climate change scenario (high emissions, RCP 8.5), the model projected substantial increase in risk of overexploitation for both EEZ and highseas stocks. For the Decentralised Solution scenario, pof stocks almost doubled under climate change (RCP 8.5), while the increase in fishing effort for pelagic fishes under the Global Technology scenario further increases the pof stocks. Although all scenarios result in a decrease in pof stocks from the current level, climate change is expected to substantially increase the risk of not achieving Target 6.

Rio+20 combined with climate change scenarios - Climate change will affect the distribution in future catch potential of marine fisheries. The change in maximum catch potential in 2050 under two climate change scenarios (high and low emissions) is estimated using the approach by Cheung *et al.* (2010).

At a global level, the predicted combined impacts of climate change and Rio+20 scenarios on maximum catch potential (i.e., the % change in MSY) by 2050 is not large. Compared to the current global fisheries catch level, which is at around 95% MSY, the highest sustainable catch level (97% MSY) is expected to occur in the Decentralised Solution pathway under a low emission scenario. The lowest sustainable catch (91% of MSY) is expected under Global Technology change in a high emission scenario (Table 6.3).

Table 6.3. Predicted maximum catch potential (%MSY) under combined Rio +20 Pathways and climate change (CC) scenarios by 2050. Source: Produced by the authors of this chapter.

	2050 Scenario		
	Status quo (no CC)	Low CC	High CC
% change in max catch potential RIO +20 PATHWAY (% of MSY relative to no-climate change)	1	-3	-3.9
Decentralised solutions	100	97	96.1
Global technology	95	92.2	91.3
Consumption change	96	92.9	92.1

Larger differences in catch potential occur at the regional (FAO fishing area) level, and vary between coastal EEZ and high seas catches (Table 6.4).

Table 6.4. Change in maximum catch potential in EEZs and high seas under low and high climate change scenarios. Source: Produced by the authors of this chapter.

	Low Climate Change		High Climate Change	
FAO Area	EEZ	High Seas	EEZ	High Seas
18	35.36	276.21	75.08	473.28
21	-12.98	-1.00	-14.24	-3.66
27	-3.09	-10.30	-6.11	6.57
31	-6.60	8.29	0.41	46.16
34	-9.95	21.24	-2.79	115.80
37	6.99	NA	16.34	NA
41	16.17	53.51	25.81	93.05
47	4.73	13.49	11.99	27.51
48	39.48	63.19	55.42	42.74
51	-10.46	-8.52	-18.52	-17.66
57	-6.70	-4.79	-14.75	-15.99
58	5.03	96.37	6.78	105.90
61	10.58	28.44	18.64	35.69
67	-15.35	66.41	-29.59	48.24
71	-22.45	-45.57	-39.08	-58.91
77	-9.13	-1.11	-20.28	-16.42
81	45.51	13.58	22.57	-7.14
87	-2.91	-5.45	-3.22	-7.45
88	NA	85.11	NA	31.53

Under both climate change scenarios, the largest increase in catch potential is expected in Arctic and Antarctic (Area 18 and 58) high seas catches, while largest losses are expected in both the coastal EEZ and high seas catches of the Pacific Western Central Ocean (Area 71) (Table 6.4). This brings up concerns about food security and livelihood for many of the fish dependent small island developing states located in the Western Pacific.

Rio+20 scenarios and relationship to economic variables - The potential trajectory of economic variables for fisheries by 2050 under each Rio+20 Pathway is assessed (Table 6.5) based on similar assumptions made by Sumaila *et al.* (2012): i) rebuilding takes 10 years (i.e., MSY is reached by 2023); ii) fishing effort has to be reduced by a minimum of 40% to 60%; iii) harmful and ambiguous subsidies are eliminated upon rebuilding, while beneficial subsidies remain constant; and iv) rebuilding is assumed to reduce fishing effort by a minimum of 40%, and the rate in fishing effort change is directly proportional to catch landed.

Table 6.5. Assumed trajectory of economic variables by 2050, based on Rio + 20 pathways. Fmsy = fishing mortality required to achieve MSY. This was obtained from the relationship Fmsy=R/2, where the intrinsic rate of population increase, R, of each stock was estimated.
Source: Produced by the authors of this chapter.

	Pathway Scenario		
Trend by 2023	Decentralised Solutions	Global Technology	Consumption Change
Fishing Effort Within EEZ	F^{msy} by 2020	+2% per year (pelagic) F^{msy} by 2020 (demersal)	+1% per year (pelagic) F^{msy} by 2020
High seas	F^{msy} by 2020	F^{msy} by 2020	F^{msy} by 2020
Fishing cost	+	+	-
Price of fish	+	Constant	Constant
Subsidies			
- Harmful	-100%	-100%	-100%
- Others	Constant	Constant	Constant

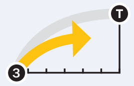
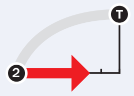
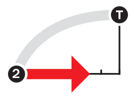
Overall, the combined climate change and Rio +20 Pathway scenarios suggest that the same policy pathways will have substantially different regional outcomes. Therefore, fisheries policies have to be adapted to regional fisheries context and management frameworks. This is especially pertinent for priority areas such as the Western Central Pacific, which is predicted to be highly affected, but where coastal populations tend to be most reliant on fisheries resources for livelihood and food security.

6.5 UNCERTAINTIES AND DATA REQUIRED

The effect of climate change on marine ecosystems, species, and biodiversity is one of the major uncertainties about the future trajectory of global marine fisheries. This arises because there is a general lack of knowledge about the current state of marine systems, and how systems are structured and function. Uncertainties arise because of unpredictability due to the natural variability in marine biological and biophysical systems over time and spatial scales. This impedes understanding about how marine species and ecosystems will respond to future stressors, including climate and human driven changes. Besides biophysical and ecological uncertainties, there is also uncertainty about future economic, societal, and political trajectories, and the consequent effects on human systems associated with marine ecosystems and fisheries.

Further, while fisheries scientists for decades have pointed to the "simple" solution to managing fisheries: reduce fishing effort, this is easier said than done. While there has been progress in managed fisheries in developing countries, there is no or little management in developing countries in general. Reduction in fishing effort, as assumed in the sustainable 2050 projections, calls for major changes in how fisheries are conducted, including ending harmful subsidies, and it remains uncertain that this may happen.

6.6 DASHBOARD – PROGRESS TOWARDS TARGET[4]

Element	Current Status	Comments	Confidence
All fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches		Great regional variation, positive for some countries but data limited for many developing countries	High
Recovery plans and measures are in place for all depleted species		Variable progress in some regions	Medium
Fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems		Some progress e.g. on long-lining used in tuna fisheries, but practices still impacting vulnerable ecosystems	Medium
The impacts of fisheries on stocks, species and ecosystems are within safe ecological limits, i.e. overfishing avoided		Overexploitation remains an issue globally, but with regional variation	Medium

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6.7 REFERENCES

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Footnote

[4] *This provides a current assessment of progress towards the Aichi Biodiversity Target based on the material presented in this chapter and the expert judgement of the authors of the GBO-4 Technical Report. It is subject to change as additional material becomes available, including information from national reports, NBSAPs and the BIP partnership.*

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TARGET 7: AGRICULTURE, AQUACULTURE AND FORESTRY

By 2020, areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.

PREFACE

High levels of habitat loss, pollution, land degradation and declining biodiversity associated with agriculture, aquaculture and forestry indicate that practices in these sectors are not sustainable when evaluated at regional and global scales. Achieving sustainability in these sectors is therefore one of the most important keys to protecting biodiversity in terrestrial, inland water and coastal systems.

Sustainability in agriculture, aquaculture and forestry has social, economic and environmental components. Sustainable management in these sectors should address six main criteria (Gold, 1999; Pearce *et al*., 2003; Pretty, 2008; Soni & Salokhe, 2009; FSC, 2012; CEFAS, 2013; PEFC, 2013; SARE, 2013):

- Produce sufficient high quality, nutritious and culturally appropriate food, as well as wood and fiber, to fulfill current and future human demand and support overall human wellbeing. This includes efforts to reduce demand.
- Efficiently use land, water, energy and other production inputs such as nutrients, pesticides and herbicides. Post-harvest losses should be reduced.
- Avoid the degradation of land and water resources.
- Avoid negatively affecting, and where possible enhance, environmental quality including ecosystem services and biodiversity, and the preservation of its genetic diversity of the species harvested (crops, livestock, trees, fish, etc.).
- Be economically viable, ensuring fair pricing for farmers, traders and consumers, now and for the future.
- Respect workers' rights and provide sufficient livelihood and quality of life for workers, now and for the future.

This wide range of main criteria makes assessment of sustainability in these sectors complex. For example, the OECD uses 15 broad themes to assess the sustainability of agriculture, and most of these themes include several indicators. For forestry, FAO uses seven broad themes and 18 indicators to assess sustainable management (FAO, 2003; 2010b). Due to data availability and to limit the scope of this assessment to environmental issues, an overview of a range of environmental indicators is provided, and a small fraction of these indicators are assessed in depth. Several other indicators are covered in more depth in other chapters including indicators of habitat loss (Target 5), pollution (Target 8), genetic diversity of crops and livestock species (Target 13) and degradation (Target 15). Indicators for social and economic sustainability are beyond the scope of this chapter.

Sustainability in agriculture, aquaculture and forestry is highly scale dependent (Verburg *et al*., 2013). It needs to be assessed at site, landscape, regional and global levels, because sustainability at one scale does not necessarily imply sustainability at all scales. For example, agricultural practices can become more locally sustainable in terms of pollution if chemical inputs are reduced, but if this reduces productivity at the local level it may increase the land exploited elsewhere at regional or global levels. On the other hand, intensification of agriculture at the site level often increases pollutants that have negative effects on the environment including degradation of biodiversity at multiple scales (see Target 8).

Sustainable management of agriculture, aquaculture and forestry are influenced by consumption, waste and equitable distribution (Gustavsson *et al*., 2011; Hardcastle & Hagelberg, 2012; Beveridge *et al*., 2013). Increasing global population and changing consumption patterns are foreseen to put substantial pressure on these systems to produce increased amounts food and fiber over the coming decades. This pressure to produce more while reducing negative impacts is one of the greatest challenges for sustainability, so wise consumption and reduction of waste are considered to be an essential component of meeting this challenge.

Based on the considerations above, this chapter focuses on indicators of sustainable management that are most closely tied to impacts on biodiversity and other Aichi Biodiversity Targets. Those indices that are covered in detail in this chapter or other targets are in *italic*.

- Inputs needed for food, wood, fiber and energy production: *land, fertilizers, pesticides, water*, feed for livestock and aquaculture, energy and labor (see Target 8 for details on fertilizers and pesticides). The input indicators are dominated by agriculture.

- Environmental impacts from production: *Nitrogen (N) and Phosphorus (P) pollution in air and water, pesticides* in water, greenhouse gas emissions, and *genetic pollution of wild populations* (see Targets 8 and 13 for details).
- Efficiency: Much of the effort to improve sustainability in agriculture, aquaculture and forestry focuses on increasing efficiency at the site level (i.e., reducing inputs compared to outputs).
  - Land use efficiency: changes in productivity per unit area have important impacts on *land area for agriculture, land area for plantation forests*, land and inland water/coastal area for aquaculture, and *natural habitat lost* due to activities in these sectors (see Target 5 for details).
  - Nitrogen use efficiency (=outputs/inputs): low efficiency indicates that excessive nitrogen fertilizer is being used and therefore lost to the environment, while very high efficiency indicates that nitrogen stocks in soils are being depleted.
  - Water use efficiency: changes in water use efficiency impact the total water use on agricultural land and per capita and can be illustrated by the water footprint (see Target 4 for details).
  - Energy efficiency: changes in energy used per unit.
- Degradation: soil organic material in agricultural systems, *soil erosion, forest degradation* (see Target 15 for details).
- Biodiversity: impacts on biodiversity on production sites have been measured in a variety of studies, but it is difficult to find indicators at large regional levels that specifically indicate biodiversity in agriculture, aquaculture and forestry areas. *Farmland and grassland birds and grassland butterflies* have been widely used as a measure of the impact of agriculture on biodiversity on agricultural area because they have reasonable specificity, but high quality time series data is only available for Europe (farmland birds), Canada and the United State (grassland birds). *Diversity of crop plants and livestock* is also a key indicator (see Target 13 for details).

Management is considered sustainable if the environmental impact of production is low, the efficiency of using inputs is high, degradation of resources is avoided, and biodiversity and ecosystem services are maintained. At regional and global levels, sustainability also implies that sufficient food, wood and fibers are being produced to meet human demand.

To ensure and verify whether food, wood and fibers are being produced sustainably, various labelling and certification systems have been developed. Most of these systems do not cover all criteria for sustainable management, the adoption of these systems however indicate movement towards sustainable production.

Environmental labelling and sustainable use programmes: this chapter examines *organic farming, the Forest Stewardship Council (FSC) and the Program for the Endorsement of Forest Certification (PEFC), and aquaculture labels*, as well as the *conservation agriculture community of practice*. A wide range of commodity specific labelling and certification programmes exists (e.g. Milder *et al*., 2012), but are not included in this assessment. Many of these labels and certification programmes are used as a measure of sustainability, although the pertinence of these as measures of sustainability varies considerably (Banerjee & Solomon, 2003; Horne, 2009).

This chapter does not cover bioenergy, which is covered in Target 3. Future trajectories of bioenergy consumption will be important for sustainability in agriculture and forestry. Moderate deployment of bioenergy may have net benefits; for example, when on-farm bioenergy production can replace fossil fuels used for agricultural production. However, although massive deployment of bioenergy foreseen in some scenarios might have benefits for global climate change, it also adds pressure to intensify production and therefore seriously compromises sustainability of agriculture and forestry (see detailed discussions in Targets 3, 5, 8 and Chapter 22; Howarth & Bringezu, 2012).

7.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

7.1.1 Status and trends

7. 1.1.i Agriculture

Environmental impacts

Environmental impacts of nutrients and pesticides used in agriculture are covered in detail in Target 8. In summary: i) in areas of high fertilizer inputs, particularly in parts of the USA, Europe and Asia, large nitrogen and phosphorus losses to the environment have negative impacts on biodiversity and ecosystem services, ii) nitrogen and phosphorus losses to the environment are projected to stabilize or slightly decline in developed countries and continue to increase rapidly in many developing countries over the next two decades (Sutton & Bleeker, 2013), and iii) the amount of pesticides used is still increasing globally (De *et al*., 2014), however current legislation might lead to a decline in pesticide use in developed countries.

Land use for agriculture is covered in detail in Target 5. In summary: i) agriculture is currently the primary driver of land conversion and of declining biodiversity globally, ii) in some regions the rate of loss of natural habitats as a result of the expansion of agricultural lands is declining (e.g., USA, Europe, Brazil), but in other regions the rate of loss is increasing over time, and iii) there is evidence that land sparing, i.e., increasing agricultural efficiency in terms of production per unit land area, has played a role in reducing loss of natural habitats in some countries (Lambin & Meyfroidt, 2011; Malingreau *et al*., 2012).

Agricultural water use represents approximately 6% of global internal renewable water resources and accounts for 70% of all water withdrawals (FAO, 2013a). There are substantial local and regional variations in agricultural water use, especially in the Mid-East and Asia, where a high water demand may cause water shortages in many areas. Water withdrawals for agriculture have generally stabilized or are slightly declining in developed countries, but irrigation of agricultural area is growing rapidly in many other regions, especially Asia (FAO, 2013a).

Energy use for agricultural production has increased steadily and represents slightly more than 2% of total global energy use. As a fraction to total global energy use, this fraction has been decreasing over the last two decades (FAO, 2013a). On-farm bioenergy production can replace fossil fuels use for agricultural production.

Efficiency

The main non-labor resources for agricultural production are land, water, nutrients, energy and pesticides. The efficiency of the use of these resources is expressed as the amount of production divided by the resources used. We provide a brief overview of land, nitrogen and water use efficiency.

Global land use efficiency, or productivity per unit area, has nearly tripled over the last five decades, but large regional differences exist and the rate of increase is showing signs of slowing (FAO, 2013a). Productivity per unit area has stagnated at low levels in sub-Saharan Africa, while in many Asian countries productivity increased rapidly in the 1980s and 1990s but is currently leveling off. Productivity per unit area in Europe and North America is high but the rate of increase has slowed recently. The productivity per unit area has mainly increased by applying more nutrients and pesticides, increasing mechanization, employing more efficient and secure water supply and using high yielding crop and livestock varieties.

To indicate sustainable land use, the efficient use of land should be combined with an efficient use of nutrients. Crop Nitrogen and Phosphorus Use Efficiency (NUE and PUE) (Sutton & Bleeker, 2013) are the amounts of N and P harvested divided by the total amount of nutrients added at field level. For NUE inputs included fertilizers, biological N-fixation and nutrient imports (e.g., atmospheric N-deposition). NUE values of 70-80% indicate optimal efficiency of nitrogen use, while low NUE values indicate a risk of high nitrogen losses to the environment and high NUE values indicate that nitrogen is being extracted non-sustainably from soils. Very high NUEs found in much of sub-Saharan Africa indicate that more nutrients are taken out of the system than are being added, and reflect degradation of soil fertility in these areas (Sutton *et al*., 2013). Low values of NUE are found in many parts of Southeast Asia due to very high N fertilizer inputs. Europe is approaching the range of optimal NUE, which illustrates the difficulty of using efficiency as a measure of sustainability since N pollution remains a significant environmental problem in Europe due to the large amount of N fertilizer used at regional scales.

Degradation

The main factors driving degradation are erosion, depletion of nutrients and carbon from soils. Agricultural practices like soil treatments, plowing and insufficient inputs compared to harvests can result in degradation. In addition, reductions in vegetation cover drives degradation, especially in mountainous areas, croplands and grazed areas. Long-term livestock grazing negatively affects vegetation cover if more biomass is removed by livestock than has been produced (Schuman *et al*., 1999; Jones, 2000; Amezaga *et al*., 2004). Reduced vegetation cover may increase erosion risk (Asner *et al*., 2004; Reynolds *et al*., 2007). Arid areas with relatively low productivity and intensive livestock grazing are prone to overgrazing and have the highest erosion risks, examples can be found in North America, eastern Africa, Mongolia and large parts of South America.

Indicators of land degradation are discussed in more detail in Target 15. Land degradation currently covers roughly 19 million km² of areas with reduced soil productivity, and 43 million km² with moderately to severely degraded areas with losses in soil quality, water retention and/or biodiversity. This is respectively 15% and 33% of the global terrestrial surface (Conforti, 2011).

Biodiversity

Agriculture intensification is recognized as one of the main driving forces behind the decline of biodiversity in agricultural landscapes (Gregory *et al*., 2005; Haberl *et al*., 2005; Flohre *et al*., 2011). (The impacts of agriculture on biodiversity outside areas of agricultural production are discussed in Targets 5, 8, 13 and 15). To determine the status biodiversity in agricultural landscapes, habitat specialists can be good indicators of general health of the environment (BIP, 2014). Among the best-studied habitat specialists are common farmland birds in Europe. Hence farmland birds are useful indicators of a broader biodiversity trend in agricultural landscapes that cover about half of the land area of Europe. However, this indicator must be interpreted with caution as it provides a relatively narrow view of agricultural impacts on biodiversity because it covers only Europe and focuses on the dynamics of a single species group. Many characteristic species of European farmlands are in decline, and very few have stable or increasing trends, resulting in a decline of European farmland bird diversity of 52% over the period 1980-2010 (Figure 7.1; Butler *et al*., 2010). This is caused by a reduction of foraging and nesting sites due to a decrease in agricultural grasslands, semi-natural habitats in agricultural landscapes (e.g., hedgerows) and fallows, as well as heavy use of pesticides (Butler *et al.*, 2010; PECBMS, 2012). Grassland, arid land and rangeland birds in North America are also declining, but trends are not available for agricultural landscapes in other regions.

Although patchy, information from other taxa, e.g. insects, butterflies, plants and mammals, suggest that these groups are also declining in Europe owing to agricultural intensification (Haberl *et al.*, 2005; Flohre *et al.*, 2011; EEA, 2013;). For example, butterfly populations in Europe have also declined by 50% since 1990 (EEA, 2013).

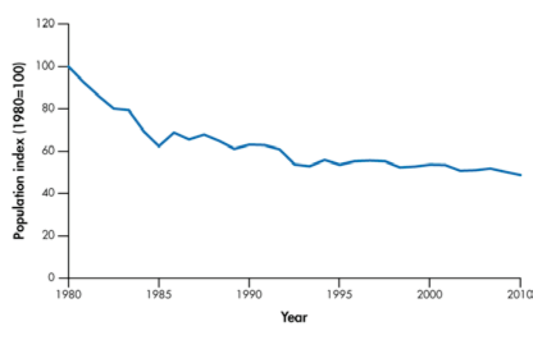

Figure 7.1. Wild Bird Index for European farmland birds, showing mean population trends across 27 countries for 39 species characteristic of agricultural landscapes. Source: EBCC/RSPB/BirdLife/Statistics Netherlands (2010).

Certification and sustainable agriculture programmes
Several certifications and programmes have been developed to promote sustainable agriculture. However no agricultural environmental label covers all social, economic and environmental criteria for sustainability and many focus on site scale sustainability. Organic agriculture labels are one of the best-known certifications and focus on input and environmental effect indicators by eliminating the use of synthetic fertilizers, synthetic pesticides and GMOs. The goals of organic agriculture are generally expressed in terms of broad sustainability (IFOAM, 2013). However, organic agriculture certification typically does not include criteria such as nutrient pollution, soil erosion, crop diversity, land use efficiency and displacement or economic sustainability and may not always lead to improvements in these criteria (e.g., Gattinger *et al.*, 2012).

There has been an increase of cropland area under organic agriculture during the last decades, reaching a total of 37 million hectares in 2011 (comprising about 2% of the total global cropland area; Figure 7.2; IFOAM, 2013). This eliminates many important agricultural pollutants and may reduce others; for example, nitrogen leaching per unit area from organic agricultural areas is on average 31% lower than conventional agricultural areas (Tuomisto *et al.*, 2012). Generally, organic agriculture has a positive effect on landscape species diversity (see section 4.3; (Tuck *et al.*, 2014).

Land use efficiency expressed as productivity per unit area is generally 20-30% lower in organic agriculture than in conventional agriculture (Tuomisto *et al.*, 2012; Seufert *et al.*, 2012; De Ponti *et al.*, 2012; Badgley *et al.*, 2007). However these differences are highly contextual, depending on environmental conditions and cropping systems (Seufert *et al.*, 2012; De Ponti *et al.*, 2012). In general, for legumes and perennial crops the difference is minor (ca. 5%), whereas for annuals the difference is substantial (more than 25%). However if the best organic practices are performed these differences can be reduced (ca. 15% for annuals; Seufert *et al.*, 2012). These reductions in productivity in organic agriculture could lead to increased land use for agricultural production at larger scales. *In fine*, the sustainability of organic farming compared to conventional farming depends on the how they are practiced and the choice of indicator.

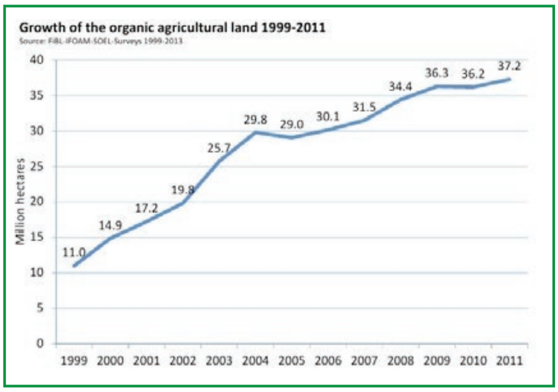

Figure 7.2. Area certified as organic farming has increased by 26.2 million hectares since 1999. Source: IFOAM (2013).

The organic agriculture label is widely known, but there are a number of other important labels or practices that are widespread and have sustainability aims. For example, Conservation Agriculture is a community of practice that focuses on no or limited tillage, permanent plant cover and crop diversity to reduce environmental impacts and enhance the status of biodiversity in agricultural landscapes (FAO, 2014). Conservation agriculture practices have increased substantially to 127 million hectares in 2011 or roughly 7% of total global cropland area (Figure 7.3; FAO, 2013a; 2014). This production system strives to maintain or increase profitability together with high and sustained production levels while concurrently conserving the environment with a strong focus on soil health (FAO, 2013a; 2014). An important aspect of conservation agriculture is the use of low or no-tillage systems that generally keep soils intact, improve soil diversity, reduce soil erosion, reduce

CO_2 emissions from machinery and may improve soil carbon sequestration (Derpsch *et al.* 2010; Ogle *et al.*, 2012; Soane *et al.* 2012; Scopel *et al.*, 2013). However, conservation agriculture does not explicitly set limits on inputs and frequently relies on herbicide resistant GMOs and inputs of herbicides to control weeds (Soane *et al.*, 2012; Scopel *et al.*, 2013). The effect of no-tillage on land use efficiency varies substantially, but no clear tendency has been detected (Giller *et al.*, 2009; Lahmar, 2010; Ogle *et al.*, 2012; Ndlovu *et al.*, 2014).

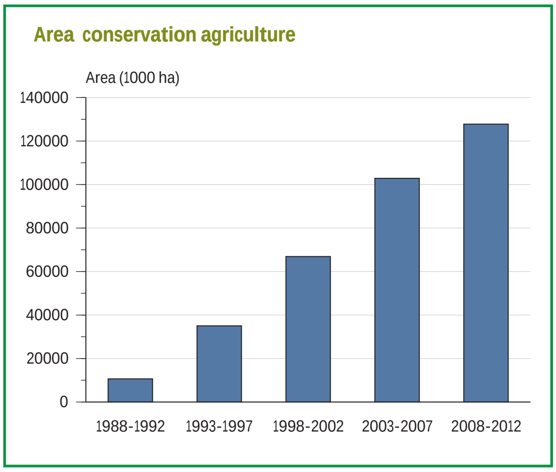

Figure 7.3. Global area using practices of Conservation Agriculture (in 1000 ha). Source: FAO (2013a).

In addition to the organic agriculture label and the Conservation Agriculture programmes, there are many national and international certification schemes and communities of practice promoting sustainable agricultural practices and "fair trade" labels. Many countries have also adopted agri-environmental programmes to promote various agricultural techniques that reduce environmental impacts (Kleijn & Sutherland, 2003; Batary *et al.*, 2013).

7.1.1.ii Forestry

Sustainable forest management (SFM) has been defined as, "the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems", (MCPFE, 2014). SFM is a concept that embraces and reconciles the wide interests, use and benefits of forests, while ensuring that the forests are managed in such a way that similar benefits can be obtained in the future. Through a wide range of criteria and indicators SFM has been developed to provide the relevant information for forest policy development and evaluation, national forest policies, plans and programmes (Rametsteiner & Simula, 2003; MCPFE, 2014). These SFM indicators are developed from a policy point of view and include indicators for carbon cycles, biodiversity, forest health and vitality, productivity, conservation status and socioeconomic conditions (Rametsteiner & Simula, 2003; MCPFE, 2014).

As illustrated above the environmental goals of sustainable forestry are to produce sufficient wood and fiber, while preserving the production capacity and biodiversity, avoiding the degradation of soil and water and enhancing the capability of carbon sequestration (European Forest Institute, 2013). FAO assessed progress towards sustainable forestry from 1990 to 2010 and found that most indicators showed no trend or were positive (Table 7.1; FAO, 2010b). We highlight and update some of these indicators below.

Biodiversity

Several studies indicate that sustainable forest management minimizes impacts on biodiversity more than conventional logging methods (the effect of forest logging, degradation and fragmentation are discussed in more detail in Target 5; see also Kuijk *et al.*, 2009). By implementing SFM practices (which are often also part of the certification schemes discussed below) the impact on biodiversity could be reduced. For example by leaving retention trees in clear cuts some of the original habitat is maintained, which could provide benefits to many species. The availability of dead wood from different tree species positively influences insect and fungal biodiversity (Johansson *et al.*, 2013). Leaving corridors in logging areas can also provide shelter to many species and reduce fragmentation effects (Kuijk *et al.*, 2009). These practices have a positive effect on biodiversity when compared to conventional logging, but not when compared to undisturbed forests (Kuijk *et al.*, 2009). Knowledge of the long-term impacts of the SFM practices is limited (Johansson *et al.*, 2013). For example the retention trees can provide a lifeboat function for some species, but the long-term survival of these populations is uncertain (Johansson *et al.*, 2013). In addition, the implementation of the certification scheme on the ground, national legislation and neighboring forestry areas also influence the impact on biodiversity (Elbakidze *et al.*, 2011).

Forest plantations have both positive and negative effects on biodiversity; therefore, the contribution of plantations to sustainability is unclear and depends on the criteria used. As noted above, plantations have a high land use efficiency, which can contribute to sustainability if this reduces pressure on other forests. However, compared to primary or naturally regenerating forests, biodiversity values in plantations, especially mono-specific stands, are generally low (Brockerhoff *et al.*, 2008; Gibson *et al.*, 2011). These monocultures do not provide a wide variety of habitats and support little biodiversity. In addition, when exotic tree species are used in plantations they often invade native forest ecosystems (Pawson *et al.*, 2010; for details on the effect of invasive species see Target 9). On the other hand, many native flora and fauna exist in plantation forests (Brockerhoff *et al.*, 2008; Seaton *et al.*, 2009). Even uncommon and threatened species are increasingly recorded in plantations, for example some raptor species have been found in higher densities in plantations than in natural forests (Brockerhoff *et al.*, 2008; Seaton *et al.*, 2009). Forest plantations can also serve as ecological buffers from adjacent non-forest land uses and connect indigenous forest remnants (Pawson *et al.*, 2010; Johansson *et al.*, 2013).

Environmental impacts
Inputs (other than land), outputs and pollution related to forestry are generally low compared to the agricultural sector. However, inputs can be substantial in some intensive short-rotation, coppice forestry systems where nutrient inputs and losses to the environment may approach those in agricultural systems.

Efficiency
Land use efficiency in forests can be increased through the establishment of highly productive wood plantations. Fast growing, industrial forest plantations account for about 1.5% of the world's forests (FAO, 2010b). The global area of plantations is estimated as 54 million ha in 2010 and provides about 22% of global round wood supply, with the most rapid increases in North and Central America and in Asia (FAO, 2010b; INDUFOR programmes 2012). Forest plantations are highly productive and can reduce the pressure on natural forests; however, plantations often have negative impacts on biodiversity and other environmental indicators as highlighted below.

Degradation
Soil erosion and degradation are generally very low for forests, but can be very high following burning and logging, especially clear-cutting in mountain systems and tropical wet forests (West *et al.*, 2014). Erosion risk and depletion of nutrient and water resources in forest plantations can also be high due to the use of exotic species and the water and nutrient requirements of fast growing trees (FAO, 2010a). On the other hand, planting forests is a frequently used in restoration measures to reduce erosion and rehabilitate land (see Target 15).

An important contribution to sustainable forest management is reduced impact logging (RIL). These techniques reduce soil erosion, collateral damage within forests and greenhouse gas emissions as well as enhancing above-ground biomass recovery (West *et al.*, 2014). Reduced impact logging includes harvesting techniques that reduce impacts on the remaining forest stand and simultaneously enhance forest re-growth and carbon storage (Putz *et al.*, 2008). Reduced impact logging techniques include pre-harvest mapping of crop trees, secure planning and design of infrastructure, liana cutting and felling techniques such as directional felling and cutting stumps low to the ground to avoid waste (Putz *et al.*, 2008).

Certification
Forest certification is dominated by two international systems: the Forest Stewardship Council (FSC) and the Program for the Endorsement of Forest Certification (PEFC) (FAO, 2010a; PEFC, 2013). FSC accounts for about one-third of the total global area of certified forest (180 million hectares; FSC, 2013), while the PEFC covers about two-thirds (253.8 million hectares; FAO, 2010a; PEFC, 2013) with relatively little overlap between the two certification programs. Figure 7.4 shows the development of the area of forests managed with a FSC or PEFC certificate. Following an abrupt increase between 2004 and 2005, the rise in area has been steady.

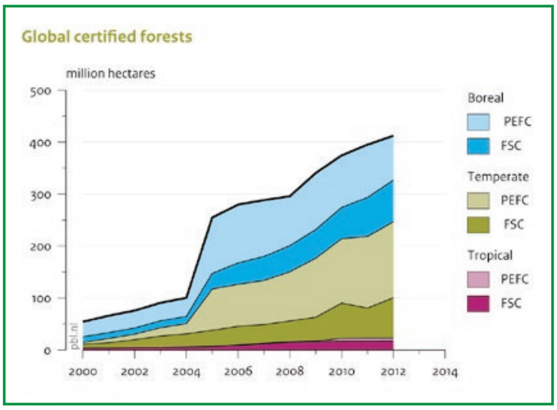

Figure 7.4 Total Area of forestry under FSC and PEFC Certification in boreal, temperate and tropical regions. Source: FSC (2013); PEFC (2013).

While the criteria of PEFC and FSC encompass the main criteria of sustainability, including those prescribed in SFM, there is also criticism of these certification schemes (Moog *et al.*, 2014; SSNC, 2010). For example, certification of plantations has been criticized, since it is debated whether or not plantations contribute to sustainability (see above; Lee, 2009). Certification for logging in native and high-conservation-value forest and lack of attention to degradation of watercourses have also been criticized (Greenpeace, 2013; Harkki, 2004). The benefits of forest certification for biodiversity are not clear due to a lack evidence-based management schemes and monitoring (Wintle & Lindenmayer, 2008; Zagt *et al.*, 2010).

Approximately 9% of global forest area has been certified under a variety of schemes (FAO, 2010a; PEFC, 2013). A recent estimate indicates that approximately one quarter of global industrial round wood now comes from certified forests (FAO, 2010a). Most of these advances have occurred outside the tropics: less than 3% of forest area in African, Asian and tropical American forests are certified (Figure 7.5 and 7.6; PEFC, 2013). A few countries, mostly in Eastern Europe, have more than 50% of their total forestlands certified. However, other countries with extensive forest areas comprise the main locations for FSC-certified forests. Russia, for instance,

Table 7.1. Progress towards sustainable forestry.
Source: FAO (2010b).

H = High (reporting countries represent 75–100% of total forest area)
M = Medium (reporting countries represent 50–74% of total forest area)
L = Low (reporting countries represent 25–49% of total forest area)

● = Positive change (greater than 0.50%)
● = No major change (between -0.50 and 0.50%)
● = Negative change (less than -0.50%)
– = Insufficient data to determine trend

Progress towards sustainable forest management at the global level, 1990–2010

Thematic element	FRA 2010 variables	Data availability	Annual change rate (%)		Annual change		
			1990–2000	2000–2010	1990–2000	2000–2010	Unit
Extent of forest resources	Area of forest	H	-0.20	-0.13	-8 323	-5 211	1 000 ha
	Growing stock of forests	H	0.13	0.14	n.s.	n.s.	m^3/ha
	Forest carbon stock in living biomass	H	-0.18	-0.17	-538	-502	million tonnes
Forest biological diversity	Area of primary forest	M	-0.40	-0.37	-4 666	-4 188	1 000 ha
	Area of forest designated primarily for conservation of biodiversity	H	1.14	1.92	3 250	6 334	1 000 ha
	Area of forest within protected areas	H	1.09	1.97	3 040	6 384	1 000 ha
Forest health and vitality	Area of forest affected by fire	M	-1.89	-2.15	-345	-338	1 000 ha
	Area of forest affected by insects	L	-1.88	-0.70	-699	-231	1 000 ha
Productive functions of forest resources	Area of forest designated primarily for production	H	-0.18	-0.25	-2 125	-2 911	1 000 ha
	Area of planted forest	H	1.90	2.09	3 688	4 925	1 000 ha
	Total wood removals	H	-0.50	1.08	-15 616	33 701	1 000 m^3
Protective functions of forest resources	Area of forest designated primarily for protection of soil and water	H	1.23	0.97	3 127	2 768	1 000 ha
Socio-economic functions of forests	Area of forest under private ownership	H	0.75	2.56	3 958	14 718	1 000 ha
	Value of total wood removals	M	-0.32	5.77	-241	4 713	million US$
	Employment in primary production of goods	M	-1.20	-0.11	-126	-10	1 000 FTE
Legal, policy and institutional framework	Forest area with management plan	M	0.51	1.07	6 964	15 716	1 000 ha
	Human resources in public forest institutions	L	-1.94	0.07	-23 568	830	total staff
	Number of students graduating in forestry	L	15.67	8.83	4 384	4 081	number of students

has 29.7 million hectares certified, though this only represents 3.6% of Russia's forests (Figure 7.5). Canada's 60.6 million hectares of FSC-certified and 119.8 million hectares of PEFC-certified land constitute respectively. 32% of the global FSC and 47% of the global PEFC of certified forests (Figure 7.5 and 7.6; FSC, 2014). Increasing the extent of certification in the tropics remains a goal for many organizations (Zagt *et al.*, 2010).

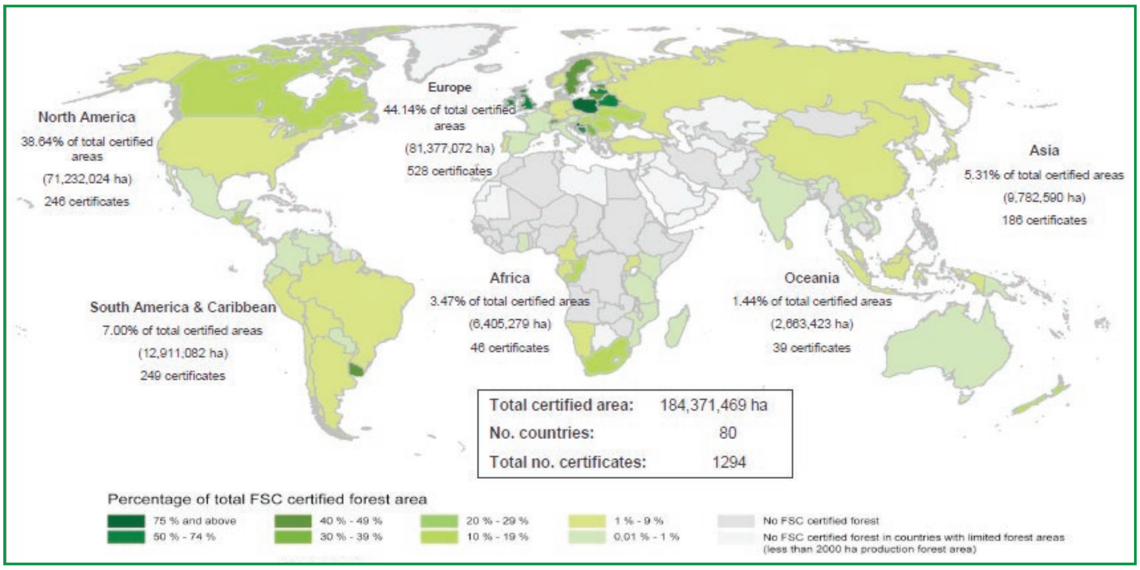

Figure 7.5. Global distribution of FSC certified forest (FSC, 2013). The darkest green regions indicate >75% FSC certified forest area, while dark grey indicates no FSC certified forest in countries with significant forest cover. Source: FSC (2013).

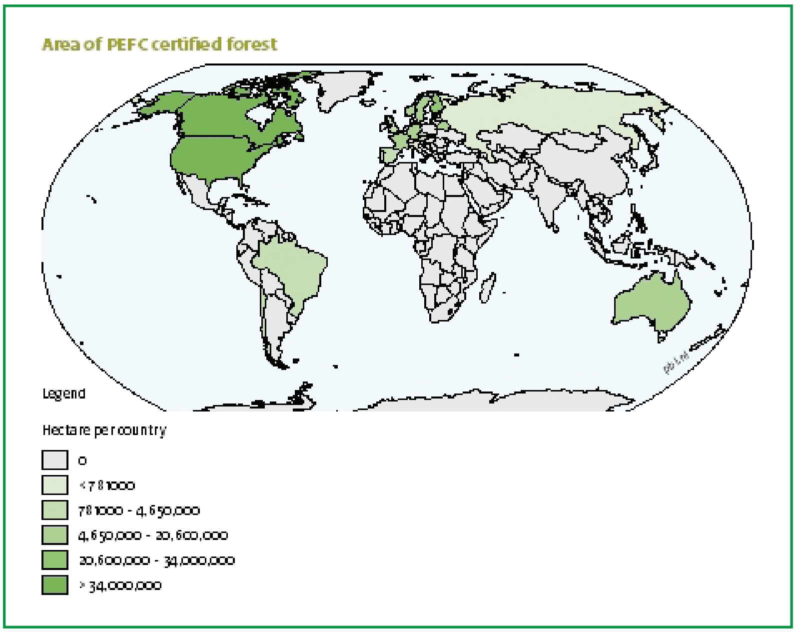

Figure 7.6. Global distribution of PEFC certified forest (PEFC, 2014). The darkest green regions indicate up to 120,000,000 ha of PEFC certified forest, while the grey areas indicate no PEFC certified forest. Source: PEFC (2013).

7.1.1.iii Aquaculture

Aquaculture, the aquatic counterpart of agriculture, has grown rapidly in recent decades, and today more than 40% of the total seafood production comes from aquaculture (Figure 7.7; Beveridge *et al*., 2013). Its rapid expansion has led to substantial provision of aquatic protein and other products for humanity. At the same time there is growing evidence of negative side effects arising from chemical and biological pollution produced by aquaculture facilities that include food, fecal and urinary products, and unabsorbed chemicals, as well as microorganisms, parasites, and feral animals that may be introduced or propagated in the natural environment (Beveridge, 2004, Hargrave, 2005). The release of fecal and urinary wastes and uneaten food may lead to eutrophication and oxygen depletion (for more detail see Target 8), while pathogens and naturalized animals can dramatically affect native species. Eutrophication can damage not only the local surrounding environment (ponds, lake or coastal area for example) but also reduce aquaculture production itself (Beveridge *et al*., 2013).

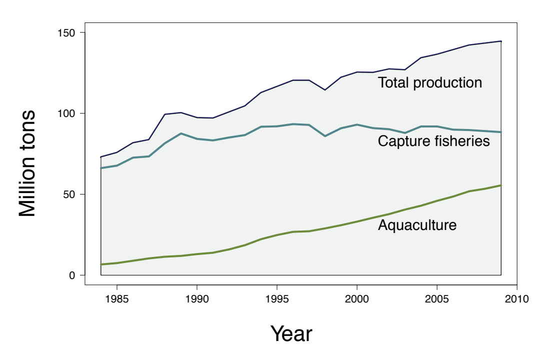

Figure 7.7. Total fisheries and aquaculture production (biomass) 1984-2010. Source: The World Bank (2013).

Though aquaculture is growing rapidly (Tacon & Metian, 2013), pollution arising from aquaculture remains localized and modest compared to other forms of agriculture or urban areas. Nevertheless, the aggregated effects of many farms may have regional impact on biodiversity and ecosystem functions (Brummett *et al.*, 2013). Most water pollution problems arise from poor management (Troell *et al.*, 2013), hence, the sector is working to increase efficiency, reducing preventative treatments with chemical medications (Beveridge *et al.*, 2010), and minimizing physical disturbance. Certain practices (e.g., semi-intensive, extensive, traditional, polyculture, and integrated systems) also enable substantial waste assimilation internally (Beveridge, 2004; Edwards, 2009; Troell, 2009; Troell *et al.*, 2013), thereby reducing contaminant losses to surrounding ecosystems. Such movement toward 'closed' production practices is essential for reducing the ecological impact of aquaculture but might also come at the price of greater energy consumption during the production process (Henriksson *et al.*, 2012).

Certification

As aquaculture expands to help meeting the rising demand of fish protein, a range of environmental and social problems must be addressed (Bush *et al.*, 2013). The aquaculture sector has only recently started to certify a fraction of its production, building on the model provided by forestry and organic farming certifications. The first generation of certification methods have raised some concerns (Bush *et al.*, 2013; Jonell *et al.*, 2013). Indeed, the potential of eco-certification to reduce the negative environmental impacts of aquaculture appears uncertain since: (a) certification schemes currently focus on species predominantly consumed in the EU and US, with limited coverage of Asian markets; (b) the share of certified products in the market as currently projected is low; (c) there is an inequitable and non-uniform applicability of certification across the sector; (d) mechanisms or incentives for improvement among the worst performers are lacking; and (e) there is incomplete coverage of environmental impacts, with biophysical sustainability and ecosystem perspectives generally lacking (Jonell *et al.*, 2013). Certification of aquaculture products currently represents 4.2% of the global production (Figure 7.8, aquatic plants excluded, data from 2010-2011. For more details see the review from Jonell *et al.*, 2013).

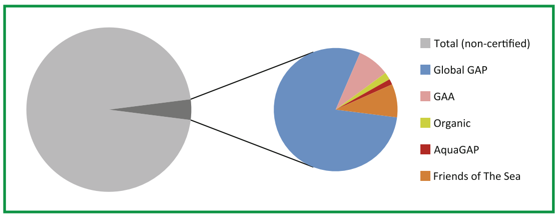

Figure 7.8. Current eco-certified share of total aquaculture production and coverage of individual eco-certification schemes Source: Jonell et al. (2013).

7.1.2 Projecting forward to 2020

7.1.2.i Agriculture

Statistical projections for sustainable production to 2020 include area under conservation agriculture (Figure 7.9) and European farmland birds (Figure 7.10). These provide a contrasting view of agricultural impacts on sustainability. It must be kept in mind that these indicators cover a small fraction of the agricultural area globally and are not regionally balanced. For example, the European farmland bird index only measures a component of biodiversity (Gregory & van Stien, 2010) and Conservation Agriculture does not consider all sustainability indicators, as discussed above.

Adoption of conservation agriculture has been strong in several major agricultural producers including Brazil, Argentina, Australia and the USA. Adoption of Conservation Agriculture has been slow in Europe, Asia and Africa for a variety of reasons. These trends are expected to continue up to 2020 (Figure 7.9). Conservation agriculture can substantially increase production per area and farmer income in some cases. But there are many situations in which these practices are not adopted or not viable due to knowledge, environmental or material constraints. Therefore these constrains could slow broader adoption in the future (Giller *et al.*, 2009; Lahmar, 2010; Ndlovu *et al.*, 2014; Ogle *et al.*, 2012).

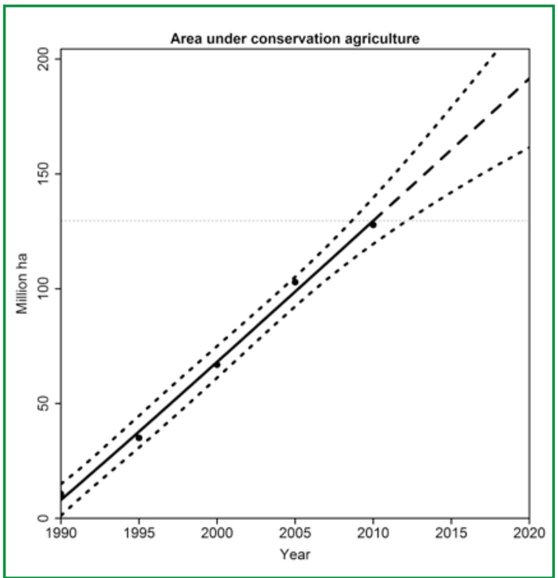

Figure 7.9. Statistical extrapolation of the area under conservation agriculture to 2020. Points are data, the solid line is the fit to data, long dashes represent extrapolation period and the dotted lines represent 95% confidence bounds. The horizontal dotted grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying mechanisms follow past trends. Source: FAO (2014, accessed 23/1/2014).

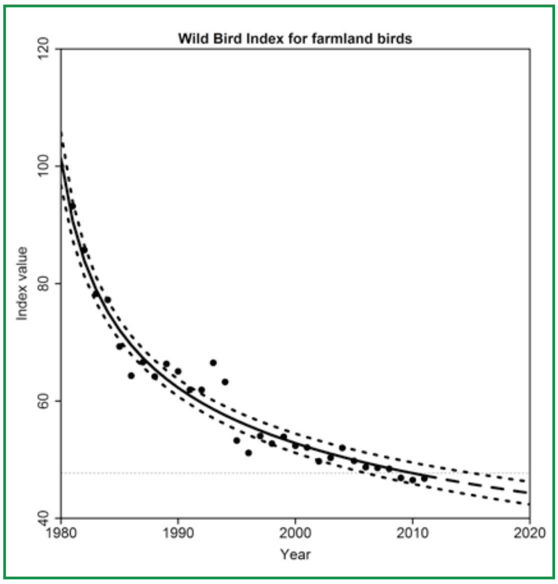

Figure 7.10. Statistical extrapolation of the Wild Bird Index for European farmland birds to 2020. Points are data, the solid line is the fit to data, long dashes represent extrapolation period and the dotted lines represent 95% confidence bounds. The horizontal dotted grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying mechanisms follow past trends. Source: Data is from (Sheehan *et al.*, 2010) and illustrates the mean population trends for 39 species characteristic of agricultural landscapes.

The European farmland bird index projects a slowing decline by 2020 (Figure 7.10). The most rapid decline in these habitat specialists occurred before the 1990s due to the loss of vast areas of semi-natural grasslands and destruction of other habitats such as hedgerows in many areas in Europe. The continuation of habitat loss may lead to a decline in farmland birds, which is for example projected for in "business-as-usual" scenarios (See box 7.1) (Chiron *et al*. 2013; Butler *et al*., 2010; Mouysset *et al*., 2012). Also the observed acceleration of agricultural intensification combined with abandonment in Eastern Europe is expected to lead to a decline in farmland birds (Butler *et al*. 2010, Gregory *et al*., 2010). Abandonment and re-wilding of agricultural land is believed to positively impact biodiversity, it negatively affects farmland bird diversity (Butler *et al*., 2010).

Agri-environmental policies to enhance grasslands have proven to be effective in improving the status of farmland birds (Pywell *et al*., 2012; Baker *et al*., 2012) and could potentially lead to a positive trend in this indicator by 2020 (See box 7.1; Mouysset *et al*., 2012). Increasing the area of arable land without pesticide application would be a major step towards increasing food resources availability for farmland bird species and would most likely influence the indicator positively (Ewald *et al*., 2002; Marshall *et al*., 2003; Marshall *et al*., 2001; Taylor *et al*., 2006; Geiger, 2011). In addition, the protection and creation of landscape elements like hedgerows, woodlots, buffer strips, field margins and wetlands increases biodiversity in agricultural landscapes and can have benefits for agriculture (Wezel *et al*., 2013; Taylor *et al*., 2006; Altieri, 1999).

The statistical projections of Conservation Agriculture and the Wild Bird Index provide a very partial view of the trajectory of sustainability in agricultural systems, but reflect reasonably well what is expected for other indicators. The trajectory of other indicators will include both positive and negative trends, and will differ between regions.

Some indicators are expected to have a positive trajectory. For example sustainable certification and labels, such as organic farming, low- and no-tillage, Conservation Agriculture and integrated pest management, are expected to increase due to mounting pressure to reduce environmental impacts of agriculture and rising costs of inputs, e.g. fuel, machinery, mineral fertilizer, water and energy (e.g., Soane *et al*., 2012). For example the application of agri-environmental schemes in Europe is expected to increase with the new measures in the Common Agricultural Policy (e.g. Aebischer *et al*., 2000). However, the impact of such certification, sustainable practices and agri-environmental policies on biodiversity is highly debated (see section 7.3) (Westhoek *et al*., 2012). In addition, restoration of farmlands is projected to increase in coming decades and to cover large areas in some countries e.g., China (see Target 15).

On the negative side, agricultural intensification and excessive nitrogen and phosphorus use is projected to increase in many regions, and these have large and widespread negative impacts on biodiversity and ecosystem services (see Target 8; Butler *et al*., 2010). In addition, degradation of cropland and rangelands in many parts of the world is projected to continue (see Target 15) and due to continuing intensification, biodiversity in agricultural landscapes are projected to further decrease (see section 7.4).

Box 7.1: Scenarios for European farmland birds.

Several mechanistic scenarios taking into account land use change, climate change, economics, agricultural policy and bird responses to environment have recently been developed to explore the future dynamics of farmland bird populations in Europe. These projections suggest that strong agricultural policies could result in recovery of farmland bird populations, but others foresee substantial declines in all scenarios. For example, Mouysset *et al.* (2012) found that several environmental policy-relevant actions are projected to lead to recovery of farmland birds over the next several decades compared to "business-as-usual" scenarios in France (Figure 7.11). The interactions of land use with climate change have also been explored for the UK (Bateman *et al*., 2013). Projections for the UK indicate that strong environmental policy, both in terms of climate mitigation and habitat conservation, could lead to small gains or no change for bird diversity (all wild birds), but with substantial decreases in profitability of agriculture. Weak policy for climate mitigation and habitat conservation is projected to lead to large declines in bird diversity over much of the UK (Bateman *et al*., 2013). Butler *et al.* (2010) explored scenarios based on current trends and several changes in EU agricultural policies (all assuming plausible reductions in grassland protection) and found that farmland bird populations are projected to substantially decline in all scenarios by 2020, particularly in Eastern Europe due to intensification of agriculture on fertile soils and abandonment of extensive agriculture on infertile soils.

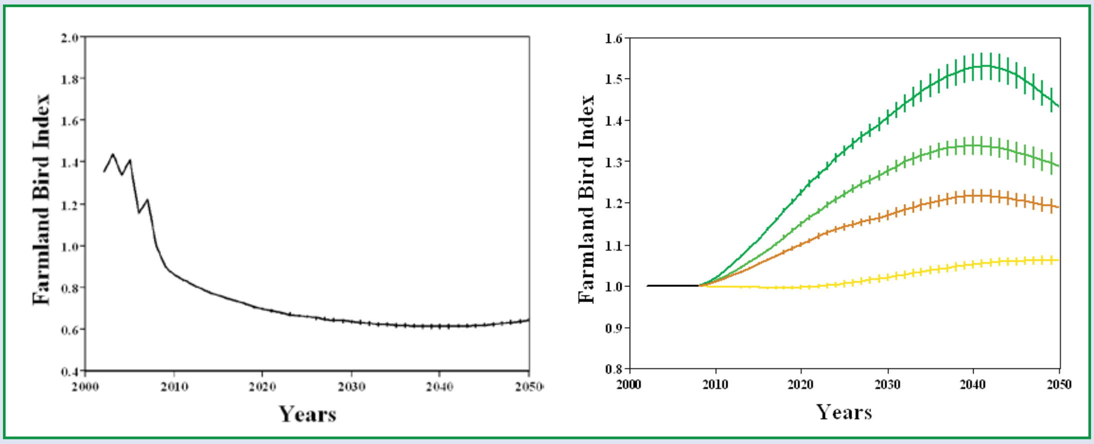

Figure 7.11. Dynamics of farmland birds in France as projected for business-as-usual (A) and alternative scenarios (B) based on plausible changes in European agricultural policy. Source: Mouysset *et al*. (2012).

7.1.2.ii Forestry

Total area under FSC and PEFC certification is used to illustrate the projected increase in sustainably managed forests, since these certification schemes include multiple sustainability indicators. The statistical extrapolation of the sum of FSC and PEFC certification (Figure 7.12) indicates an increasing area under FSC and PEFC management out to 2020. However, certification is more difficult in tropical areas of developing countries due to the time and expense (Carlson, 2012; Marx & Cuypers, 2010). Therefore this trend may remain weak for tropical forests and may slow down globally. It must be kept in mind that even with the growth foreseen by 2020, certified forest products will represent a modest fraction of total forest products globally. Internationally agreed standards on sustainable forest management however exist (International Tropical Timber Agreement, ITTA). A global legally binding agreement on all types of forests is presently being discussed under the United Nations Forum on Forests (UNFF).

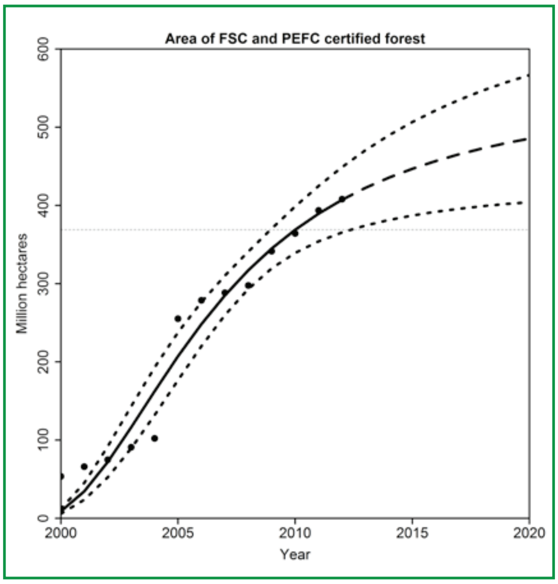

Figure 7.12. Statistical extrapolation of the sum of the area of FSC and PEFC certified forests to 2020. Points are data, the solid line is the fit to data, long dashes represent extrapolation period and the dotted lines represent 95% confidence bounds. The horizontal dotted grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying mechanisms follow past trends. Source: Data is from the FSC certificate database (FSC, 2012; FSC, 2013) and from PEFC (2013).

A recent pact between Indonesia and Norway has, in principal, led to a suspension of clearing or logging of peat and old-growth forest, providing hope that trends in a country with increasing and large scale tropical deforestation can be turned around (Edwards *et al.*, 2012). Planted forests are projected to rise from about 260 to 300 million ha by 2020 (FAO, 2010b), but there are concerns that the tradeoffs associated with many planted forests may mean that the land-sparing effects of planted forests are partially offset by other negative impacts on the environment including biodiversity (Brockerhoff *et al.*, 2008).

7.1.2.iii Aquaculture

Notwithstanding the historic tendency to underestimate the rise of aquaculture, several projections of future production are available (summarized in Hall *et al.*, 2011) and by 2020, the production is expected to range between 65 and 85 million tons (excluding seaweeds; Hall *et al.*, 2011). Global distribution of production will remain similar with 90% of the global production located in Asia. Production of freshwater fish, such as carp and tilapia, will likely see large increases up to 2020 (IFPRI, 2003; FAO, 2012a), which could be beneficial for the sustainability given their low trophic level and thus the supposedly independence to fish meal and oils compared to carnivorous fish such as salmon. However, a slowing in aquaculture growth is anticipated, from 5.8% to 2.3% growth by 2020 (FAO, 2012a; The World Bank, 2013).

As the aquaculture sector grows, there are important questions about how greater yields can be sustainably supported. Ideally, the expansion of aquaculture production will be enabled by increased yields per unit area with attention paid to minimizing environmental impacts, rather than expanding the area under cultivation. This would avoid competition for land from terrestrial agriculture and for aquatic habitat from wild-capture fisheries (IFPRI, 2003). Yield increases might be achieved either from increased inputs of feed and environmental control, or through more efficient conversion of current inputs into biomass. It is likely that in the next several decades, aquaculture production will take advantage of both strategies to support growing yields (IFPRI, 2003). In the long term, sustainable intensification will be required, and could be encouraged through appropriate certification schemes. In addition, there is substantial potential for aquaculture to expand in unproductive marine waters where competition with wild-capture fisheries and other users is limited. Most aquaculture production is currently from inland waters including rivers, lakes, and ponds (viz. 70% in 2010, excluding macro algae), but mariculture has great promise if a range of economic and ecological challenges can be overcome.

7.1.3 Country actions and commitments[1]

The majority of national biodiversity strategies and action plans examined contain targets or similar commitments related to sustainable management. These targets are broadly in line with Aichi Biodiversity Target 7 though some only refer to improving sustainable management generally. Few of the targets that have been established are quantitative.

Overall the national commitments in the NBSAPs tend to focus on agriculture and forestry. There has been relatively less emphasis on the sustainable management of aquaculture.

Footnote

[1] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

Approximately half of the national commitments examined are accompanied by supporting actions. Further, a number of countries have developed commitments related to specific issues related to sustainable management. For example one of the operation objectives in Belgium's NBSAP is to promote agricultural diversification while Japan has set a key action goal of implementing initiatives to create Satoumi areas.

Aichi Biodiversity Target 7 is particularly ambitious in scope. While the national targets and other commitments which have been set in relation to Target 7 will make a significant contribution towards its attainment, additional commitments will be required if this target it to be achieved globally by 2020.

Box 7.2: Whole-of-paddock rehabilitation, New South Wales and Western Australia.
Australian Government funding supported Greening Australia to work with farmers, catchment management authorities and NRM groups in central-western New South Wales and southwest Western Australia to deliver rehabilitation of enclosed grazing areas over three years. Greening Australia engaged farmers to temporarily volunteer a paddock of at least 10 hectares to be planted with native trees and shrubs, with the aim of returning around 25% of the paddock to deep-rooted perennial vegetation. The vegetated paddocks are withdrawn from production for five years and farmers receive stewardship payments to offset some of their production loss. Livestock are permitted to be reintroduced after five years under a rotational grazing system after the plantings have established. This whole-of-paddock rehabilitation project was a practical, cost-effective way of integrating conservation and production goals. Key benefits included increased biodiversity, carbon sequestration, return of ground cover and productive native perennial pastures and shrubs, and salinity and erosion control with improved grazing productivity of paddocks. These outcomes will have long-lasting impacts on the environment and agricultural production. Re-establishing connectivity and restoring landscape biodiversity will help mitigate the effects of climate change and help contain pests and diseases as well as providing shelter and shade for livestock and improving soil condition.

7.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

7.2.1 Actions

As indicated above sustainable production is measured by social, economic and environmental criteria. Most of the sustainable actions focus on relatively few criteria. Additionally some actions apply to agriculture, aquaculture and forestry sectors, while others are more sector specific. Therefore a combination of actions is needed to achieve overall sustainable production (see Box 7.4). For some of these actions, there is broad agreement, but for others there is considerable dissent leading to conflicts and inaction. In this section, several general actions to move towards sustainable production are described, and then some specific actions for each production systems are given.

Reduce losses

Losses in agriculture, aquaculture and forestry are often high, with considerable potential for improvements. Losses in the entire food supply chain add up about one third (Gustavsson *et al.*, 2011) to half (Foley *et al.*, 2011) of total production. In developed countries large losses occur at the retail and consumer end of the chain which could be improved through changes in behavior. Large losses in developing countries occur at the post-harvest and processing end of the chain which could be improved by better infrastructure (Gustavsson *et al.*, 2011). It has been suggested that restructuring agricultural markets would help reduce post-harvest losses (Moran *et al.*, 2012).

Reduction of losses in forestry can decrease pressure on natural forests, since less material has to be harvested. In many tropical forests, processing industry is very inefficient and volume recovery from felled trees is often less than 20% (Hardcastle & Hagelberg, 2012). Modernizing processing and competition may help to improve efficiency. For example, competition on Rwanda's charcoal markets has led to more efficient conversion and reduced the volume of material harvested (Hardcastle & Hagelberg, 2012). In boreal and temperate regions close to 100% of the harvested volume is utilized.

Alter consumption patterns

Increasing consumption of food in some developing countries is vital for achieving human development goals. However, in many developed countries calorie intake is well above levels considered to be healthy, so reductions could contribute to improved health and reduce pressure on agriculture and aquaculture (See section 4; Stehfest *et al.*, 2009; Foley *et al.*, 2011; Erb *et al.* 2009). Reducing the consumption of animal products in many developed countries would yield considerable human health benefits and reduce the area necessary to produce sufficient food for a growing world population (UNEP, 2014). Pressure on forestry would be reduced by increased use of renewable materials and recycling.

Additionally, alternative protein sources could be included in diets such as insects, seaweed and fungi (FAO, 2013b). These alternative protein sources can be produced more sustainably than current protein sources. For example insect production uses less feed, irrigation water and production area and results in less greenhouse gas emissions and nutrient pollution compared to conventional livestock production (FAO, 2013b). In addition, many alternative protein sources can be reared on organic wastes (e.g. manure and compost), thereby recycling agricultural and forestry wastes (FAO, 2013b).

Expand and improve certification
Many certification schemes and round tables have been developed to indicate whether products have been sustainably produced (FAO, 2010a). Different certification schemes put emphasis on one or more of the aspects of sustainability (Soni & Salokhe, 2009). For example, in addition to the certification schemes reviewed in section 4.1.1, certification schemes exist for a number of specific agricultural commodities such as palm oil, soy, coffee and cacao. Some other schemes focus on agricultural practices regardless of the crops, e.g., the Rainforest Alliance and the Sustainable Agriculture initiative. The lack of consistent criteria makes it difficult for consumers to identify products that meet a broad range of sustainability criteria and opens the door for product labels which appear "green" but do not yield measurable sustainability benefits. It has been suggested that it would be helpful to promote practices that could be labelled "biodiversity intensive sustainable agriculture" (BISA), e.g., that would include practices such as limited tillage, reduced external inputs, crop rotation and polyculture, effective livestock management, and efficient irrigation (Moran *et al*., 2012).

Forestry certification schemes not only focus on the provision of timber but also on a wide range of other forest ecosystem services. Currently certifications are a pre-requisite for selling tropical and subtropical timber products on North American and European markets (Hardcastle & Hagelberg, 2012). Many other markets do not require certifications, so many exporting countries, e.g., India, do not invest in expensive certification processes, as the return on investment is low. In addition, many developing countries lack access to North American and European markets (ATIBT *et al*., 2013). Therefore more action is needed to increase certification in southern-hemisphere countries to produce timber sustainably (Hardcastle & Hagelberg, 2012; ATIBT *et al*., 2013). For example legislation ad law enforcement may stimulate the investments in certification, as illustrated by the Mexican environmental impact assessments of forestry practices which help to improve sustainable forest management.

Examples of sustainable aquaculture practices are integrated multi-trophic aquaculture (IMTA). IMTA is a practice that combines the cultivation of aquatic food with organic extractive aquaculture species and inorganic extractive aquaculture species, thereby diminishing use and losses of nutrients (Harding *et al*., 2012).

Appropriately recognize and support Indigenous peoples' and community conserved territories and areas (ICCAs)
Many ICCAs include areas of primary economic production based on agriculture, aquaculture, fisheries, and/or forestry and they are governed and managed for sustainable use and conservation of the resources upon which they depend. An objective of many ICCAs focusing on production landscapes is the maintenance or revival of agricultural biodiversity. Securing these models of sustainable practices requires appropriate recognition of the rights of local communities of farmers, forest-dwellers and fisher folk in order to, among others, support culturally and ecologically appropriate modes of production and consumption and to protect against large-scale land and resource acquisitions by industrial agriculture, aquaculture/fisheries, and forestry (Kothari, 2012).

Improve the cost-benefit ratio in sustainable vs. non-sustainable management
This would involve getting rid of harmful subsidies and enhance positive subsidies, while ensuring equal access to affordable food, fair pricing for farmers, traders and consumers and provide sufficient livelihood and quality of life for workers (Target 3). It would also mean taking into account externalities in agriculture, aquaculture and forestry and could be implemented in a variety of ways including polluter pays, taxes on inputs and payments for ecosystem services. For example, implementing sustainable production forest conservation should also provide marketable goods and services in addition to wood and fibre products (Hardcastle & Hagelberg, 2012).

Education
Technical guidance from governmental rural extension services should be increased, to support sustainable practices. Also the general understanding of farmers on how agricultural production relies on biodiversity and ecosystem services should be enhanced (see also Target 1). Education can play a vital role in this process of improving the sustainability of agriculture, forestry and aquaculture. Several countries have endorsed and/or implemented educational programs targeting different groups of stakeholders such as young people, consumers, farmers, foresters and fishermen. The educational schemes and their impacts on sustainability differ widely between countries. Improving and broadening existing educational programs will remain one of the most important challenges for the next decades.

Box 7.3: Satoyama initiative

The Satoyama initiative promotes a combination of activities to achieve sustainable production from environmental and socioeconomic perspectives. The initiative focuses on "socioecological production landscapes and seascapes" (SEPLS), which are mosaic production landscapes and seascapes that have been shaped through long-term harmonious interactions between humans and nature (IPSI, 2014). SEPLS have continued to provide nature's bounty to local communities around the world for many years and are therefore recognized as a potentially tool to benefit biodiversity and human wellbeing. They have, however, been increasingly threatened in recent years by trends toward urbanization, abandonment and industrialization. Pressures on biodiversity from large-scale development can be avoided through holistic and participatory management that considers a wide set of ecosystem services at the landscape or seascape level and their links to human well-being. Therefore the aim of the Satoyama initiative is to attain societies in harmony with nature, comprising human communities where the maintenance and development of socio-economic activities (including agriculture and forestry) align with natural processes (IGES, 2013; UNU-IAS, 2013; IPSI, 2014).

The Satoyama initiative does not use a defined set of actions or measurement to maintain and rebuild the socioecological production landscapes, because every site requires a unique approach. Instead, the Satoyama initiative provides a guide to understand and develop resilience-strengthening strategies that encourage local innovation, ecosystem protection and beneficial interactions between different landscape components. This guide includes five main ecological and socioeconomic perspectives (UNU-IAS, 2010):

- Resource use within the carrying capacity and resilience of the environment
- Cyclic use of natural resources
- Recognition of the value and importance of local traditions and cultures
- Multi-stakeholder participation and collaboration in sustainable and multi-functional management of natural resources and ecosystem services
- Contributions to sustainable socioeconomies including poverty reduction, food security, sustainable livelihood, and local community empowerment.

At the local level, these guidelines are intended to result in the adoption of low-impact production methods that contribute to mitigation of soil erosion and chemical run-off, as well as retention of soil nutrients. Examples of socioecological production landscapes include the natural pastures in the Huascaran National Park in Peru, sustainable use of forest resources through community forestry in Nepal and sustainable use of wetlands in the Lower Songkhram Basin in Thailand (IPSI, 2014).

7.2.1.i Agriculture

A number of well-understood actions can improve efficiency and to reduce losses to the environment in agriculture; however, there is little agreement on the best ways to improve agricultural efficiency, or even which criteria to use for measuring efficiency. Several solutions are summarized in Table 7.2. These solutions have to be dealt with at site, landscape and regional scales. There is no single best solution, so the objective should be to find appropriate mixes for each individual site (e.g., Pareto optimization). For example sustainable agricultural management can have different implications in different parts of the world. In intensive agriculture with excessive use of fertilizers, pesticides and energy, sustainable management involves moving towards systems with less input, more efficient use of these resources and reduction of emissions of pollutants. This can often be attained without lowering productivity, although modest losses of productivity may occur in some cases. In low input agriculture, sustainable management implies an improvement of yield, either by technical means or by efficient use of ecological processes, or both (e.g., Tittonell & Giller, 2013). Increased nutrient inputs are often needed in regions with low inputs to move out of poverty traps and avoid soil degradation (Tittonell & Giller, 2013).

Table 7.2. Options to improve efficiency and to reduce losses to the environment. Source: Produced by the authors of this chapter.

Option	Practices
Optimization of existing practices	Avoid surplus fertilizer inputs and reduce fertilization to levels strictly necessary for optimal plant growth Limit livestock production to levels compatible with sustainable land use with regard to greenhouse gas-emissions and N-output to soil and atmosphere
Greater use of biodiversity and ecological functions to replace inputs	Ecological intensification Nitrogen fertilization with legumes Biological pest control On-farm bioenergy production
Technical solutions	New crop varieties including GMOs Precision agriculture More selective pesticides Agricultural robots (Moran *et al.*, 2012)

In agroecological practices, biological and ecological processes are integrated to sustainably managed agriculture (MEA, 2005; Wezel *et al.*, 2013). These agro-ecological practices include: integrated pest management (IPM); pollination management; integrated nutrient management; conservation tillage and no tillage systems; agroforestry; aquaculture integrated into farm systems; water harvesting in dry lands; and integrated crop-livestock farming systems (see literature cited in Pretty, 2008). Advanced technologies that increase the efficient use of resources, while avoiding external impacts can be combined with the more ecologically oriented technologies, leading to sustainable intensification (Brussaard *et al.*, 2010).

Box 7.4: Common Agricultural Policy in the EU

The Common Agricultural policy (CAP) is a funding instrument at the European Union level that can support biodiversity associated with agriculture, by influencing land management practices (Poláková *et al.*, 2011). On semi-natural habitats biodiversity is dependent on beneficial agricultural practices, such as extensive grazing, traditional haymaking and traditional agroforestry. This biodiversity may be lost due to intensification or abandonment. By promoting agri-environment measures the CAP tries to encourage farmers to adopt management practise that are beneficial to biodiversity. These agri-environmental measures are flexible and can be developed to suit the local conditions. For example there are highly targeted and tailored schemes for the conservation of threatened habitats and species, but there are also measures to encourage the maintenance of low intensity management on "high nature value farmland" in the wider countryside.

Even though the overall evidence is variable and the effectiveness is debated (Westhoek *et al.*, 2012; Pe'er *et al.*, 2014), several studies have confirmed that several measures of biodiversity on agricultural sites subjected to agri-environment measures are significantly better than they would have been without such measures. The benefits are mainly found on intensive croplands where the agri-environmental measures such as fallow patches, over-wintered stubbles, reduced pesticide use, field margins with seed-rich plants and diverse crops help reduced impacts on biodiversity and other environmental indicators.

Agri-environmental schemes have been successful in maintaining cultural landscapes, which often harbour particular biodiversity. Some examples of success stories of CAP are the maintenance of the semi-natural wooded pasture habitats (Sweden, Estonia), hay-meadows and mountain pastures (Slovakia, Romania), pastures (Bulgaria), moorland grazing (UK) and traditional agro-forestry systems (Spanish 'dehesas').

However these success stories depend heavily on the local implementation. Ideally regionally coordinated, permanent green infrastructures should be implemented to support biodiversity. However the effectiveness of the greening measures in the CAP 2014-2020 is highly debated (Westhoek *et al.*, 2012; Pe'er *et al.*, 2014). In these measures individual farmers have a large degree of freedom and several loopholes, which might result in annual changes of ecological focus areas. This could lead to ineffective agri-environment and little effect on biodiversity (Westhoek *et al.*, 2012; Pe'er *et al.*, 2014).

7.2.1.ii Forestry

To increase efficiency in forestry, plantations could be put in place (but see section 4.1.1.ii). Many plantations are based on short-rotation monocultures with low biodiversity values, but sustainability plantations can be enhanced by planting in small blocks, leaving natural forest along watercourses and using mixtures of indigenous species (Hardcastle & Hagelberg, 2012). An increase in rotation time generally increases the presence of indigenous species (Brockerhoff *et al.*, 2008). Plantations are also very vulnerable to fires; for example, weed growth and trails due to harvesting in moist and wet forests can cause fires to penetrate deeply into residual forest (Hardcastle & Hagelberg, 2012). This could lead to more frequent and more harmful fires and as such, fire management actions should be taken to prevent and reduce forest degradation. Other actions that are beneficial for biodiversity include avoidance of harvesting in areas of high conservation value (e.g., Putz *et al.*, 2001), or the sustainable exploitation of semi-natural forests. In many developing countries, however, the governance structure of the forest sector is poor. To avoid illegal and unsustainable practices forest ownership and regulation of logging is required.

7.2.1.iii Aquaculture

The FAO Code of Conduct for Responsible Fisheries and Ecosystem Approach to Aquaculture and the CBD Ad Hoc Technical Expert group on Mariculture have put forward specific actions to develop aquaculture that is more sustainable (CBD, 2004; FAO, 1995). The main actions are discussed below.

Give priority to farming lower trophic level and native species.

The choice of which species are farmed is driven by market forces and national policy decisions (Harding *et al.*, 2012). Therefore actions to change the market, for example through consumption change and regulations, are needed to meet this target.

Minimize biological, chemical and organic pollution.

Pollution can be minimized by improving management practices (Diana *et al.*, 2013). In particular, feeding methods can be improved by reducing overfeeding and egestion (CBD, 2004; Diana *et al.*, 2013). For example, in multitrophic aquacultures fish could receive half of their consumption from natural production, which reduces feed demand and pollution. The production of seaweed for food, feed and pharmaceuticals can result in a win-win situation, when they are produced in a marine polyculture (CBD, 2004). The waste of one species can be converted to protein by another species. For example the feed that is not consumed by finfish and can be taken up by seaweed (CBD, 2004). This can reduce nutrient concentrations by 20%-94% in open seas (Diana *et al.*, 2013). Minimizing pollution can also be achieved through enclosed systems and better waste treatment (See Target 8 for more details), for example though recirculating aquaculture systems (CBD, 2004; Diana *et al.*, 2013, Harding *et al.*, 2012). Using photoperiod management in salmon aquaculture can mitigate the use of hormones and reduces pollution (CBD, 2004). Bio filters, closed containment systems and sterility techniques can minimize harmful pollution (Diana *et al.*, 2013; Harding *et al.*, 2012). Minimizing habitat modification, especially in mangroves, contributes not only to maintaining essential ecosystem services that are provided by mangroves, but also provides nursery habitat for many finfish and shellfish species (Naylor *et al.*, 2005).

7.2.2 Costs and cost-benefit analysis

Unsustainable practices in agriculture, aquaculture and forestry result in high costs for biodiversity and society. For example, unsustainable aquaculture has resulted in widespread degradation of mangrove forests. Restoring these forests costs US$225/ha - US$216,000/ha (Harding *et al.*, 2012). Large costs of unsustainable practices included effects of soil degradation on agricultural productivity, water pollution due to agricultural practices, sedimentation of waterways, decreased water supply due to forest clearing, increased emissions of greenhouse gases, etc. (e.g. Power *et al.*, 2010, Yaron *et al.*, 2011). For example reduced insect pollination due to unsustainable agriculture could have an economic impact of €153 billion (Power, 2010).

Implementing actions to produce sustainably is also associated with costs (Table 7.3). For example Bangladesh identified several sustainability programmes they want to implement, such as Sustainable Ecosystem Management Programme, Community-based Fisheries Management, Coastal & Wetland Biodiversity Management Project, and Coastal Afforestation Programme. To implement these programmes it is estimated that they need a total of US$360 million for a period of 2010-2020 (HLP2, 2014). Nepal too has identified sustainable activities for which an estimated US$86 million will be required (HLP2 2014).

Also to implement certifications several costs have to be made. First, there are direct costs associated with auditing fees and other charges required to obtain or maintain a programme's certificate. Second, there are indirect costs associated with preparing to undertake an audit. Finally there are costs associated with the changes an operation makes to comply with the certification standards (RESOLVE, 2012).

However implementing agro-ecological practices could also save money. For example biological control of pest insects reduce the populations of pest insects and weeds and the amount of pesticides needed (Power, 2010). In the USA the natural pest control is estimated to save US$13.6 million per year (Power, 2010).

Table 7.3 Upfront and yearly costs of actions needed to make forestry more sustainable Source: Hardcastle & Hagelberg (2012).

Actions	Upfront Costs (US$)	On-going costs /year (US$)
Efficiency in processing	12,000,000,000	4,000,000,000
Fire management in vulnerable ecosystems	200,000,000	200,000,000
Product creation (tourism, PES, carbon, ABS)	2,000,000,000	5,000,000,000

The development and running of sustainable polycultures in aquaculture is more expensive than monocultures. For example a shrimp IMTA has US$829 extra investment costs per hectare and US$179 extra running cost per hectare (Harding *et al.*, 2012). For integrated salmon-mussels production system these costs are respectively US$468 and US$1076 per ton salmon. However, the net present value for salmon exceeds those investment costs (Whitmarsh *et al.*, 2006). IMTA shrimp farms provide less short-term economic benefit for farmers than unsustainable shrimp farms, but this does not account for important externalities. For example, sustainable shrimp farms protect mangrove forests, which provide at least US$1.6 billion per year in ecosystem services worldwide, in part because nearly 80% of global fish catches are directly or indirectly dependent on mangroves (Harding *et al.*, 2012).

7.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

7.3.1 Agriculture

Based on the trends presented in section 4.1 and the expected changes in the near future, there may be increased emissions of nutrient and pesticides, increased water use and continuous conversion of natural lands including forests. This implies ongoing biodiversity loss in areas under agriculture. The increased application of sustainable management practices may have a counter effect, as the application of these practices often favour biodiversity and ecosystem services. For example biodiversity in agricultural landscapes is often enhanced where organic farming practices are put in place (Balmford *et al.*, 2012; Bengtsson *et al.*, 2005) (Tuck *et al.*, 2014), since it eliminates the use of synthetic fertilizers, synthetic pesticides and inorganic fertilizers, and often with a more diverse crop rotation. It generally increases biodiversity richness and evenness (Crowder *et al.*, 2010), however effects differ between organism groups and landscapes (Bengtsson *et al.*, 2005) (Tuck *et al.*, 2014). Especially bird, insect and plant biodiversity benefit from organic farming and are on average 50% more abundant (Tuck *et al.*, 2014; Bengtsson *et al.*, 2005). Insect-pollinated plants benefit more from organic management than non-insect pollinated plants (Batary *et al.*, 2013). On the other hand, organic farming does not address all sustainability indicators. For example yields in organic agriculture in developed countries are 20-30% lower than in conventional agriculture, which might result in an increase of agricultural area (Balmford *et al.*, 2012). This may have large impacts on biodiversity, since currently agriculture is responsible for 60% of the world's deforestation. However the net impacts of agricultural expansion on biodiversity will depend on the type of habitats converted, the percentage of arable fields and the positive role that organic agriculture can play in improving landscape scale biodiversity (for more details see Target 5; Balmford *et al.*, 2012). Impacts are particularly high in tropical regions where tropical forests may support as much as 70% of the planet's plant and animal species (for more details see Target 5; Donald, 2004; Gibson *et al.*, 2011).

7.3.2 Forestry

Sustainable forestry contributes to halting deforestation and biodiversity loss. Rigorous forest certification programs can contribute to improved biodiversity conservation within managed forests (Zagt *et al.*, 2010). Sustainable forestry also includes provisions for retaining and restoring plant community diversity, limiting conversion of natural forests, protecting areas of high conservation value (HCV) and carrying out ecologically oriented silviculture. This can have a positive impact on biodiversity (Zagt *et al.*, 2010). Human impacts on tropical forest biodiversity vary greatly between regions and taxa (Gibson *et al.*, 2011). Impacts were greatest in Asia, where the most sensitive species are impacted on widespread expansion of palm oil plantations (Gibson *et al.*, 2011). Although all taxa are negatively impacted by forest degradation, in particular birds were the group most sensitive to human disturbances (Gibson *et al.*, 2011).

7.3.3 Aquaculture

Negative impacts of aquaculture on biodiversity arise from the consumption of resources, such as land (or aquatic habitat), water, feed, and the subsequent release into the environment of greenhouse gases and wastes from uneaten food, fecal, urinary products, pharmaceutical chemicals, pathogens and feral animals (for more details see Target 8). These effects may be direct or indirect through loss of habitat and niche space and changes in food webs (Jonell *et al.*, 2013). Compared to livestock and poultry production, aquaculture remains a very efficient means of producing animal protein for human consumption, resulting in minimized pollution levels. Further improvement is still possible and changes in development strategies, toward more sustainable production have already begun to influence aquaculture developments. Consequently, improvements in sustainable practices have a decreasing negative influence on biodiversity. Direct and or indirect benefits of aquaculture on biodiversity conservation exist and are often neglected (see details in De Silva, 2012).

7. 4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

7.4.1 Agriculture

Few scenarios exist that specifically focus on sustainable agricultural production. Scenarios from the International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD) and AgriMonde assume sustainable intensification, but do not include an analysis of sustainability characteristics (IAASTD, 2009; INRA & CIRAD, 2010). Three backcasting scenarios developed for the Rio+20 conference explicitly account for sustainable production (PBL, 2012). The sustainability assumptions are analysed below, based on their impact on sustainability indicators such as land use, pollution, erosion risk, water scarcity and biodiversity (PBL, 2012).

Sustainable intensification of agricultural production is interpreted differently between the three scenarios. Large-scale precision and low-emission agriculture was applied in the Global Technology and Consumption Change scenarios. Smaller-scale mixed agriculture, interwoven with natural elements, was applied in the Decentralised Solutions scenario. Figure 7.13 illustrates the effect of these different agricultural practices on the sustainability indicators.

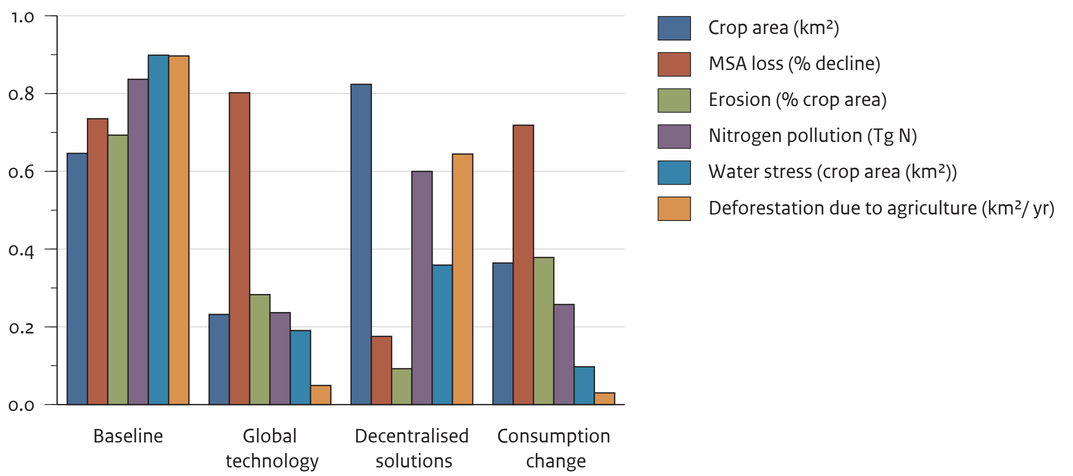

Figure 7.13. Indicators for sustainable agriculture for the baseline (="business-as-usual") and three alternative policy scenarios in 2050 using the "Rio+20" backcasting scenarios (PBL, 2012). Each indicator is set to an index ranging from 0-1. The 0-1 index for "crop area " ranges from 5,000,000 - 21,000,000 km^2 crop area (based on data from PBL, 2012). The 0-1 index for "MSA loss" ranges from 80 – 90% MSA loss on crop area (based on data from PBL, 2012). The 0-1 index for "erosion" ranges from 10% – 20% of the cropland with erosion risk. The 0-1 index for "nitrogen pollution" ranges from 120 – 190 Tg N pollution per year (based on data from PBL, 2012). The 0-1 index for "water stress" ranges between 7,500,000 - 4,500,000 km^2 crop land experiencing water stress (based on data from PBL, 2012). The 0-1 index for "deforestation due to agriculture" ranges from 7000 - 25000 km^2/yr. Source: based on data from PBL (2012).

Inputs, outputs and environmental impacts - Agricultural land use is expected to increase up to 2050 due to expected population increase. Based on current trends, without increases in sustainable production, agricultural area might need to grow up to 530 million km^2 by 2050 to meet food demand (Figure 7.13; IAASTD, 2009; PBL, 2012). This would result in environmental impacts such as increased deforestation (See Target 5 for more details). The expansion of agricultural area can be largely avoided by, for example, adopting wise consumption patterns (as in the "Consumption Change" scenario). By reducing the meat consumption by 50% in countries with high meat consumption, the amount of feed needed globally almost halves, reducing the need for forage grown on arable land by 90% (Westhoek *et al*., 2014). Agricultural area expansion can also be prevented by increasing productivity per unit land in high technology scenarios (as in the "Global Technology" scenario; Figure 7.13; INRA & CIRAD, 2010; PBL, 2010; 2012). The assumption of land sparing in the Global Technology scenario is plausible under some circumstances; however, there is evidence that intensification can lead to increased land use change due to a variety of positive feedbacks including immigration and increased financial resources (Angelsen & Kaimowitz, 1999; Walker *et al*., 2009).

One of the main external effects of agricultural production is nutrient loss. Nutrient losses can reduce the quality of surrounding terrestrial and aquatic ecosystems, as discussed in detail in Target 8. Currently, phosphorus and nitrogen nutrients are lost into the surrounding ecosystems due to unsustainable fertilizer use. In the baseline scenario, nutrient losses increase up to 2050 (Figure 7.13). This can be halted in several ways. Large-scale production sites have, for example, less external effects than small production sites. Since large-scale production areas have less natural area neighbouring the production site, this limits the external effects like nutrient leaching (as in the "Global Technology" scenario; PBL, 2012). On the other hand large-scale production practices may have larger impacts on groundwater and biodiversity at the site level. In addition to large-scale production areas, similar results can also be reached by increasing the resource efficiency, for example through increased resource efficiency of livestock (PBL, 2012), or by reducing the rate of increase in consumption level of livestock products (as in the "Consumption Change" scenario; Figure 7.13; PBL, 2012).

Efficiency - The trend of water demand for agriculture up to 2050 is projected to slightly increase or even, according to some studies, decrease (OECD, 2012; PBL, 2012; Van den Berg *et al*., 2012). However, water demand for other sources (domestic use, manufacturing and electricity production) is projected to strongly increase (OECD, 2012; PBL, 2012). This may result in increased competition for water use and might increase the agricultural area under water stress (OECD, 2012; PBL, 2012). This leads to depletion of ground water, on which irrigated agriculture heavily depends on (OECD, 2012). Sustainable water-use in agricultural production, for example by increased irrigation efficiency, increased knowledge and technology, could reduce the agricultural areas under water stress (Figure 7.13). Taking action now is important, because Steward *et al*. (2013) project that shifting to sustainable water use now will provide substantial future benefits. This occurs, in part, because technological progress and better practices are assumed to increase water use efficiency over time. This is an underlying hypothesis in the three 'Rio+20' scenarios and results in lower water scarcity compared to the baseline (Figure 7.13).

Degradation - Erosion risk is projected to increase in the baseline scenario due to the increase agricultural land needed to meet the growing food demand. Widespread adoption of sustainable agricultural practices, such as conservation tillage practices and appropriate fertilizer use, might reduce the risk of erosion (as in the "Global Technology" and "Consumption Change" scenarios). For example, increased use of fertilizers can prevent nutrient depletion and erosion risk in areas that have low inputs (PBL, 2012). In addition, natural vegetation on cropland is assumed to reduce erosion risks, therefore the erosion risk is diminished by the mosaic landscape in the "Decentralized Solutions" scenario (Figure 7.13).

Biodiversity - The overall global response of biodiversity, as measured by the mean abundance of species relative to their abundance in original, pristine ecosystems (MSA); (Alkemade *et al*., 2009), is identical in all of these scenarios because the MSA was set as a desirable target. Biodiversity loss stabilizes by 2030 at levels much higher than those in the baseline scenario. However, the means of reaching this goal are very different and have important differences in biodiversity at site and landscape scales. For example biodiversity values are low on large scale, intensively used agricultural landscapes (Alkemade *et al*., 2009; Flohre *et al*., 2011; Gibson *et al*., 2011). Because of this, biodiversity declines up to 2050 in the baseline scenario. Very efficient and technologically optimized agricultural practices are projected to reduce the MSA values on agricultural areas even more, as illustrated by the "Global Technology" scenario (Figure 7.13; PBL, 2012). By creating mosaic landscapes higher biodiversity values on the production sites are expected, as natural elements provide refuges and corridors for species (as in the "Decentralized Solutions" scenario; Figure 7.13; Liebman *et al.*, 2013, PBL, 2012, Sarukhán *et al.* 2012). However more agricultural land is needed to meet the human food demand, which might result in natural habitat conversion.

As illustrated above, sustainable agriculture has many facets. Current visions on sustainable agriculture are not able to have a positive effect on all those facets. For example sustainable agriculture scenarios with large-scale agriculture (as the "global technology" scenario) reduce the external effects, like nutrient leaching and deforestation. But biodiversity on those large-scale mono-functional agricultural landscapes is low, indicating low environmental quality. On the other hand scenarios with mixed agricultural systems and natural elements (like the "decentralized solutions" scenario) have higher biodiversity values and lower erosion risk, but more agricultural land is needed (INRA. & CIRAD, 2010).

Several other scenarios exist projecting agriculture practices by 2050. In general, most scenarios assume an increase in productivity per unit area, but this increase in production efficiency does not match consumption. In some scenarios, crop production is expected to globally increase with 70%, while an increase of 110% is needed to meet projected consumption (Tilman *et al.*, 2011). In these scenarios, food is produced on a larger area in order to match consumption. This might result in an accelerated environmental pollution, loss of forests and biodiversity, and degradation of land and other natural resources. In particular, sub-Saharan Africa degradation of agricultural land increases further in this scenario (for more details see Target 5).

The overall message from these scenarios is that a combination of consumption change, reduced food losses and sustainable agricultural intensification is needed to sustainably increase food availability and utilization. These transformations are needed in all regions, although priorities and implementation differ (see section 4.2). A simplistic, universal sustainability blueprint is not considered to be viable (UNSDSN, 2013).

7.4.2 Forestry

Sustainable forestry should conserve natural resources, should enhance environmental quality and minimize external effects (SARE, 2013). In sustainable forestry natural resources loss can be prevented by sufficient recovery time after logging, which prevents soil degradation (Egoh *et al.*, 2012). However this may require development and application of new technologies and sustainable practices.

As in agriculture, forestry production areas have lower biodiversity compared to undisturbed state (CBD, 2010). However different types of forestry techniques have different effects on biodiversity and other aspects of environmental quality. By applying pre- and post-harvest techniques the logging impact on the remaining forest stand is reduced, and simultaneously forest re-growth is enhanced (Pinard *et al.*, 2000; Rockwell *et al.*, 2007). It is therefore projected that biodiversity is higher using these techniques compared to conventionally logged forest (Kuijk *et al.*, 2009; PBL, 2010; 2012). Following the trend line forestry is projected to exert increasing pressure on biodiversity, since area under conventional logging is expected to increase (for more details also see Target 5; OECD, 2012). In scenarios with increased sustainable forestry, in contrast, the biodiversity is expected to increase (Figure 7.14).

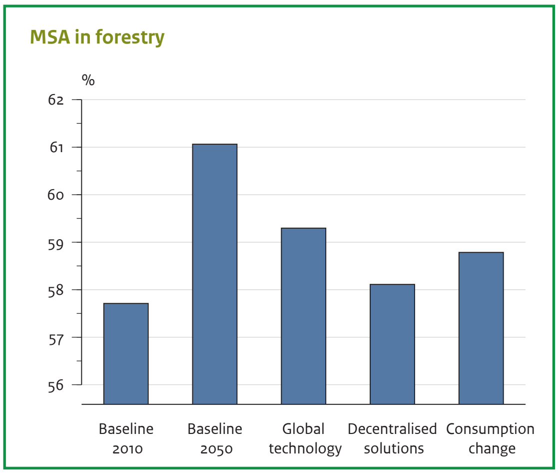

Figure 7.14. Biodiversity in forestry areas in 2050 for each RIO+20 scenario Source: based on data from PBL, 2012.

7.4.3 Aquaculture

The average projected growth of aquaculture is 110 million ton/yr by 2030, however different projections exist (Hall *et al.*, 2011). For example the IMPACT projections, based on demand-supply relationships, are more conservative and point to a production of 93 million ton/yr in 2030 and 119 million ton/yr in 2050 (The World Bank, 2013). However the expected global warming may benefit the production of freshwater species such as tilapia, carp and milkfish (IPCC, 2014). On the other hand, coastal enterprises are expected to encounter problems, for example the production of calcifying organisms (molluscs) is projected to experience loss of suitable habitats through ocean acidification (IPCC, 2014). Projections also suggest that the current high growth rate in aquaculture production will slow, but that production will be substantial and exceed capture fisheries production within a few years.

A doubling of aquaculture production from the current level by 2050 will place increasingly high demands on land, water and feed. While aquaculture overall has relatively low impact on biodiversity, there is very limited information available about where the doubling of aquaculture production will take place and what the consequences of this are for biodiversity (Figure 7.15).

The goal for eco-certified production is set at least 58 million tons by 2030 (Jonell *et al.*, 2013). This represents a major challenge that will require improvement from all actors in the aquaculture sector (public and private engagement), willingness of consumers to shift their purchasing toward more sustainable products, and implementation of incentives by government in response to environmental threats (Bush *et al.*, 2013).

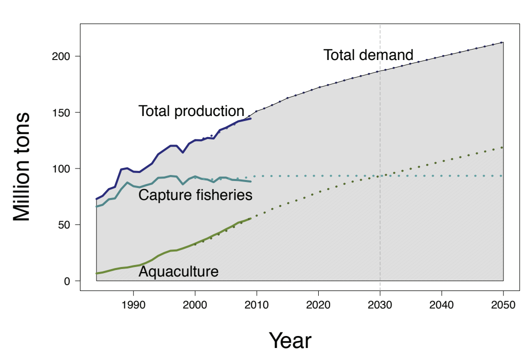

Figure 7.15. Projected global fish production in biomass from capture fisheries and aquaculture Source: Bank (2013).

7.5 UNCERTAINTIES

Three of the greatest difficulties in assessing progress towards sustainability in agriculture, aquaculture and forestry are the high diversity of indicators, lack of global coverage for indicators and questions concerning the pertinance of these indictors for sustainability. For exmaple, we have used the European farmland bird index as an indicator of environmental health in agricultural systems. Clearly, caution is needed when interpreting this indicator, since they only measure a component of biodiversity (Gregory & van Stien, 2010). In addition, the indicator not only declines due to increasing agricultural intensification, but also due to increased agricultural land abandonment (Butler *et al.*, 2010). Due to these difficulties, it is difficult to assign more than medium confidence to progress towards Aichi Biodiversity Target 7, even though many of the indicators that we have examined suggest that signficant progress is being made towards this goal. Substantial efforts by the scientific community, policy makers and other stakeholders are essential for developing better indicators of sustainability.

7.6 DASHBOARD – PROGRESS TOWARDS TARGET

Target Elements	Status	Comment	Confidence
Areas under agriculture are managed sustainably, ensuring conservation of biodiversity		Increasing area under sustainable management, based on organic certification and conservation agriculture. Nutrient use flattening globally. No-till techniques expanding	Medium
Areas under aquaculture are managed sustainably, ensuring conservation of biodiversity		Progress with sustainability standards being introduced, but in the context of very rapid expansion. Questions about sustainability of expansion of freshwater aquaculture	Medium
Areas under forestry are managed sustainably, ensuring conservation of biodiversity		Increasing forest certification and criterion indicators. Certified forestry mostly in northern countries, much slower in tropical countries	Medium

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TARGET 8: POLLUTION

By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.

PREFACE

Biodiversity is affected by numerous pollutants including heavy metals emitted by mining, industry and transport, pesticides used in agricultural practices, oil spills and excess of nutrients especially nitrogen (N) and phosphorus (P). Given the high importance of N and P impacts on biodiversity and ecosystem services at local to global scales (Fowler *et al.*, 2013; MEA, 2005), the main focus of this chapter is on the pollution from N and P. Other pollutants, such as oil spills, plastic debris and pesticide use, are treated briefly in the section on status and trends and 2020 projections, to keep this chapter within page limits. Nevertheless the impact of these pollutants on biodiversity should not be underestimated. It should also be stressed that we discuss the effects of single pollutants separately, while in reality areas might suffer from mixed toxicity from multiple pollutants.

Nitrogen – Nitrogen pollution refers to the impacts of reactive Nitrogen on the environment through chemical, physical and biological processes, threatening the quality of air, soil and water. Nitrogen in the form N_2, abundantly present in the Earth's atmosphere, has no environmental impacts. (Sutton *et al.*, 2013).

There are many natural processes that generate N inputs into ecosystems, but industry, transport and agriculture have greatly increased N inputs (Fowler *et al.*, 2013; Sutton *et al.*, 2013). The amount of N not taken by crops (N surplus) exceeding natural levels is referred to as N pollution. In non-agricultural terrestrial ecosystems, N pollution primarily arises from wet and dry deposition of N that has been emitted into the air by industry, transport and agriculture (direct effects of N fertilizer addition to agricultural and aquaculture systems are treated in Target 7). In aquatic ecosystems, N pollution comes primarily from runoff and leaching of fertilizers from agricultural and to a lesser extent from sewage and wet deposition.

Nitrogen pollution impacts ecosystems through a variety of mechanisms: the two most important being eutrophication and acidification (Bobbink *et al.*, 2010; Fowler *et al.*, 2013). Eutrophication is the rise and accumulation of the nutrient load and potential subsequent increase of productivity in ecosystems, especially by nitrogen and/or phosphorus, leading to an undesirable disturbance of the balance of organisms in the ecosystem, affecting both terrestrial and aquatic biodiversity (Sutton *et al.*, 2011). Acidification of ecosystems is caused by for example oxidation of NO_x and NH_3 in air, soil and water (Pardo *et al.*, 2011). In soils nitrogen oxides, ammonia, nitrate and ammonium ions can enhance the rate of acidification, where nutrient bases (calcium, magnesium, and potassium) are replaced by acidic elements (hydrogen and aluminum; Sutton *et al.*, 2011). This may lead to nutrient disorders and to toxic effects in plants (Sutton & Bleeker, 2013). Because N pollution is transported in the air as gases or aerosols, atmospheric N pollution can cause eutrophication and acidification in terrestrial and aquatic ecosystems at substantial distances from sources of emissions.

Nitrogen accumulation is a significant driver of species composition changes since many species-rich terrestrial ecosystems are adapted to conditions of low N availability. In these ecosystems excess of nitrogen change competitive relations, and can make conditions unfavorable for some species (Stevens *et al.*, 2004; Bobbink *et al.*, 2010). Other effects (e.g., direct toxicity of nitrogen gases and aerosols, long-term negative effects of increased ammonium and ammonia availability, soil-mediated effects of acidification, secondary stress and disturbances) are more ecosystem- and site-specific and often play a supporting role (Bobbink *et al.*, 2010). N pollution may also increase the dominance of invasive alien plants in terrestrial ecosystems and decrease the diversity of plant communities worldwide (IAASTD, 2009). In addition, N emissions make an important contribution to greenhouse gas balance (Butterbach-Bahl & Dannenmann, 2011). NO_X emissions are a major contributor to the formation of tropospheric ozone, while N_2O emissions cause depletion of stratospheric ozone. Tropospheric ozone has negative effects on plant productivity and human health.

Phosphorus – The main causes of P pollution are the runoff from agriculture fields, sewage water from households containing human excreta and detergents, and wastewater from industries. Phosphorus has no significant atmospheric component, except for some transport of P attached to soil particles. P pollution primarily impacts aquatic ecosystems through eutrophication, since P is transported more often in water and much less so in air. Traditionally N pollution is thought to primarily impact non-tropical terrestrial ecosystems and coastal zones, while P pollution is traditionally thought to have the largest impacts on freshwater ecosystems. However, recent evidence suggests aquatic ecosystems are often affected by both N and P pollution (Conley *et al*., 2009b; Elser *et al*., 2007). For example, eutrophication in lakes, rivers and coastal areas caused by N and P pollution can lead to the generation of oxygen depleted areas (<2 ml O_2/L, hypoxia and for coastal systems also known as "dead zones"), the formation of harmful algal blooms (Anderson *et al*., 2002), and uncontrolled population growth of certain weedy plants or gelatinous zooplankton.

To regulate N and P pollution national or trans-boundary regulations have been developed. To regulate N pollution these regulations are often based on the concept of "critical loads" (UBA, 2004). Critical loads are thresholds above which N pollution has negative effects on biodiversity, ecosystem function or ecosystem services. For each ecosystem type, different levels of N are detrimental, and critical loads may therefore differ between countries. For example, critical loads for atmospheric N deposition in the European Union are currently set at 5-40 kg N/ha/yr depending on ecosystem type (Bobbink *et al*., 2010), while in the United States the critical load ranges from 1 kg N/ha/yr to nearly 40 kg N/ha/yr (Pardo *et al*., 2011) and in China from 10 kg N/ha/yr – 300 kg N/ha/yr (Duan *et al*., 2010). These thresholds are based on experiments, observations and models. The use of critical loads has been questioned because it is difficult to precisely determine critical loads, since they are highly system specific and some systems appear to be affected even at very low loads (Payne *et al*., 2012).

Pesticides - pesticides cover a wide range of compounds including insecticides, fungicides, herbicides, rodenticides, molluscicides and nematicides. Ideally a pesticide is lethal to the targeted pests, but not to non-target species. Unfortunately, pesticides can be toxic to a host of other organisms, including birds, fish, beneficial insects, and non-target plants and can be lethal or may cause, among others, neurological and behavior disorders (Mitra *et al*., 2011). Pesticides can contaminate soil, water, turf, and other vegetation and can therefore be transported over great distances via air and water. Pesticides are persistent and can persist in soils, waterways and plants for more than a year (van der Sluijs *et al*., 2014). This can lead to the accumulation of pesticide in an animal or within the food chain which can lead to risks to non-target organisms (Aktar *et al*., 2009, Mitra *et al*., 2011). Pesticides can also cause soil degradation when they are toxic to microorganisms (Aktar *et al*., 2009).

Other pollutants - Plastics, pharmaceuticals, personal care products (PCPs), steroids, hormones, surfactants, perfluorinated compounds (PFCs), various persistent organic pollutants (POPs), nanomaterials and swimming pool disinfection by-products (Farre *et al*., 2008) are derived from diverse industrial activities, fossil fuel combustion, waste disposal and many other sources (Travis & Hester, 1991). Often they enter the ocean and rivers through waste-water treatment plants (Farre *et al*., 2008). Some toxic pollutants are natural elements that become widely dispersed by human activities, as in the case of radioisotopes and petrochemicals (Pirrone *et al*., 2010). Others are xenobiotic compounds synthesized to serve as pesticides, solvents and lubricants. Those micro pollutants can be acutely or chronically toxic to individual organisms or accumulate in food chains and become concentrated in top predators (Persson *et al*., 2013). Next to this, most of these compounds are long-lived once released into the environment and they are able to circulate widely in the atmosphere and oceans, allowing long-distance transport to sites that are distant from any known source, resulting in high concentrations of many pollutants in polar animal species (e.g. Braune *et al*., 2005).

Heavy metals – metals (e.g., lead, cadmium and copper) are emitted via industry, agriculture, households and other human activities. This results in local and diffuse environmental concentrations of metals that vary in space and time. Consequently, this leads to a variety of contaminated areas; some with diffuse concentrations of metals while hot spots have substantial local soil contamination. This leads to a highly variable exposure of, amongst others, soil biota. Thereby affecting soil biodiversity, soil functioning and soil processes such as the breakdown and recycling of organic materials, nutrient cycling, soil fertility, as well as the breakdown of organic contaminants (ecosystem services).

Through exposure of soil biota (acting in local soil food webs) to different local composition and concentrations of the contaminant mixture biodiversity is affected. Ecotoxicity effects are expected to affect ecosystem function and soil biodiversity at different impact levels. For large diffusely contaminated areas, the impact on soil biota originates from relatively low concentrations of rather complex mixtures of contaminants over wide areas. Hotspots with contamination have high enough concentrations of the mixture of contaminants to impact large parts of the local soil biota in the soil food web. These contaminated hot spots often haven been the result of unintended spills and other local discharges.

8.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

8.1.1 Status and trends

8.1.1.i N emissions

Nitrogen losses to the environment per capita are more than twice the world average in North America (Figure 8.1). The largest fraction of this N pollution comes from agro-food systems, including biofuels. N emissions into the atmosphere from fossil energy use in industry and transport make substantial contributions to N pollution in North America and Europe, but only small contributions in Africa.

Atmospheric N deposition varies greatly across regions (Figure 8.2). Currently, the most heavily polluted areas are the eastern half of North America, Europe and Eastern and Southern Asia (Lamarque *et al*., 2013). N deposition increased rapidly in Europe and North America over much of the 20th century, but has stabilized or decreased since the 1980s, in part due to regulation of sources of emissions (Sutton *et al*., 2013). The decrease in N deposition has been particularly strong in some European regions. Nitrogen deposition has increased rapidly in Eastern and Southern Asia over the last several decades due to increased emissions from industry, transport and agriculture (Liu *et al*., 2013; Figure 8.2).

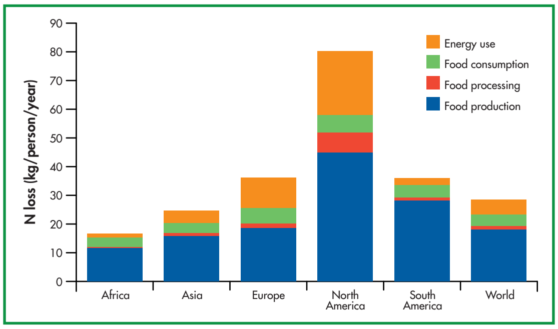


Figure 8.1. Average loss of reactive nitrogen to the environment per inhabitant per continent in 2008. The highest nitrogen losses to the environment per capita are highest in North America. Source: INI (2014).

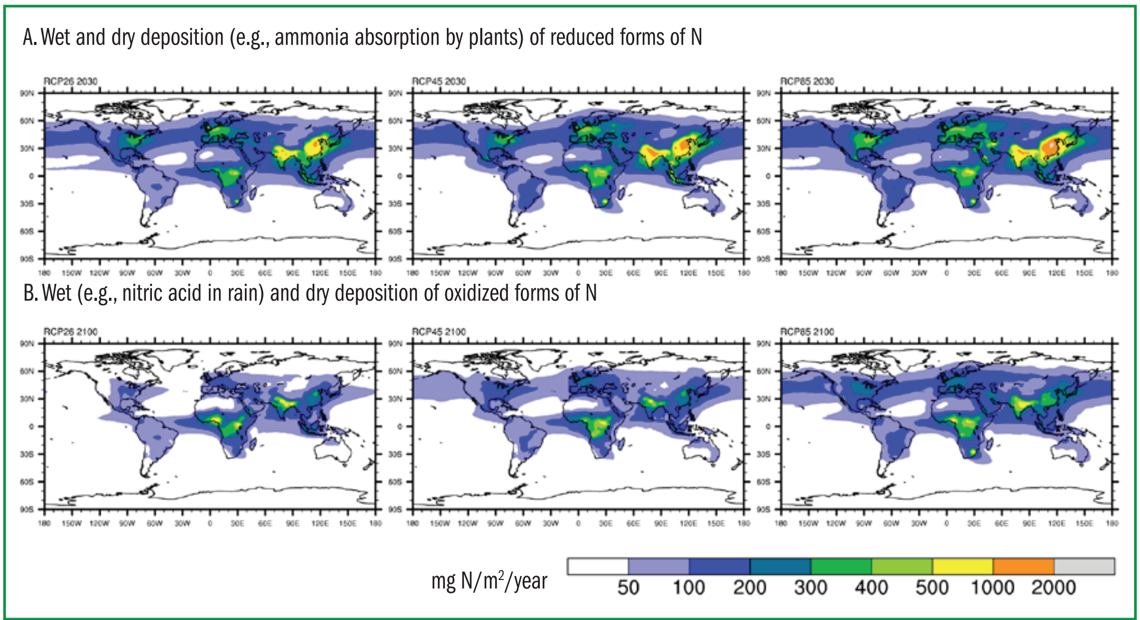


Figure 8.2. Multi-model estimates of atmospheric N deposition in 1850, 1980 and 2000 (Lamarque et al., 2013). Eutrophication of terrestrial systems from N pollution depends on total N deposition which is the sum of deposition in A and B. Source: Adapted from Lamarque *et al*. (2013).

The widespread emissions of N have increased N deposition beyond critical loads in large areas (Figure 8.3; Bobbink *et al*., 2010; PBL, 2012; Posch *et al*., 2011). Nitrogen deposition currently exceeds estimates of critical loads for large parts of Europe, eastern North America, Eastern and Southern Asia, as well as parts of central Africa and South America.

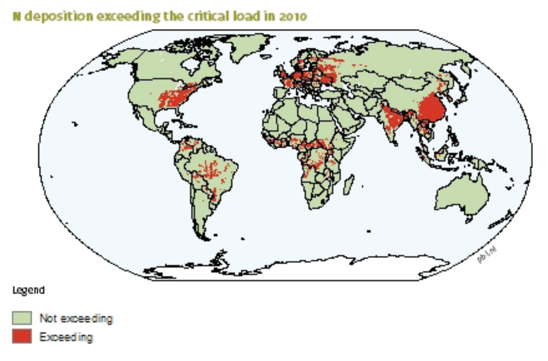


Figure 8.3. Locations where nitrogen deposition exceeds critical loads. Especially in Europe, eastern USA, India and China nitrogen deposition exceeds the critical load. Source: Figure based on data from (OECD, 2012; PBL, 2012) and critical loads on Bobbink *et al*. (2010), Bouwman *et al*. (2002).

8.1.1.ii P emissions

The current P losses to the environment from urban wastewater discharge differ considerably between countries. In Northern and Western Europe, 80-100% of all laundry detergents are P free, while in Eastern European only 15-20%, and in many developing countries even smaller proportions of laundry detergents are P free (van Drecht *et al*., 2009). Combined with the increasing use of P-free detergents in Europe, North America and Oceania, much of the nutrients in wastewater are removed in treatment plants. As a consequence, P release to surface water has stabilized in recent decades (van Drecht *et al*., 2009). However, with the use of P-based detergents and lacking wastewater treatment, combined with the rapid population growth and sewage connections in South Asia and Africa, the P pollution has been growing rapidly (211% resp. 300%) in these regions (van Drecht *et al*., 2009). In Central and South America, the population growth, sewage connection, use of P-based detergents, and lagging wastewater treatments, has resulted in a 220% increase of P pollution (van Drecht *et al*., 2009).

Due to growing fertilizer use in agriculture, the global P flow to the biosphere has increased by at least 75% the last half-century (UNEP, 2011) and has tripled compared to preindustrial levels (Macdonald *et al*., 2011). This has lead on 71% of the croplands to a P surplus. However 29% of the global crop area has limited availability of P fertilizers, which often leads to soil degradation caused by P depletion (Figure 8.4). The largest deficits were found in South America, northern USA and Eastern Europe (Figure 8.4; Macdonald *et al*., 2011). The largest P surpluses can be found in East Asia, Western and Southern Europe, coastal USA and Southern Brazil (Macdonald *et al*., 2011). In these regions, residual P has accumulated in the soil, leading to increased availability of soil P to plants (Sattari *et al*., 2012). As a consequence, P fertilizer application rates have gone down, and resulted in many countries in an application rate close to the plant uptake.

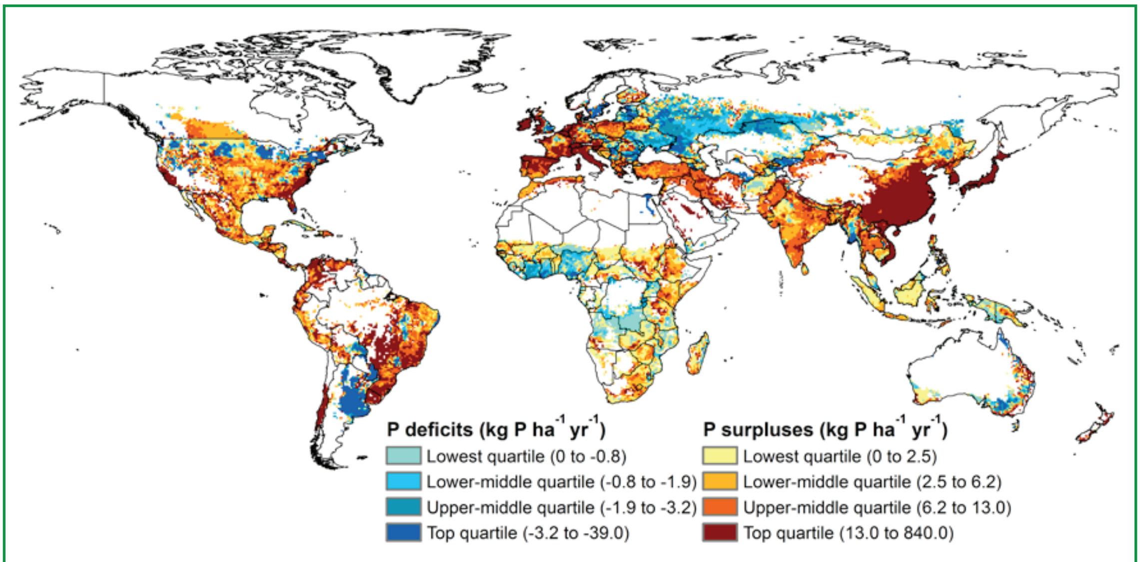


Figure 8.4. Global map of agronomic P imbalances for the year 2000 expressed per unit of cropland area in each 0.5° grid cell. The P surpluses and deficits are classified according to quartiles globally (0–25th, 25–50th, 50–75th, and 75–100th percentiles) Source: Macdonald *et al*. (2011).

8.1.1.iii N and P pollution in freshwater ecosystems

For more than 60 years, enrichment of fresh waters (e.g., rivers, lakes, wetlands and the catchment areas draining to them) with N and P from fertilizers and sewage discharges, has been a major causes of water quality impairment (Carpenter *et al.*, 1998). Nutrient loading now affects virtually all rivers on earth (Vörösmarty *et al.*, 2010) and can result in eutrophication and hypoxia causing algal and bacterial blooms and loss of light and oxygen from the water (BIP, 2014). Due to the different dynamics and origins of N and P pollution, the control of these emissions is not equally successful. For example, in Europe P emissions to fresh waters have progressively been controlled during the last two decades, while N emissions remains high (Grizzetti *et al.*, 2012; Romero *et al.*, 2013). This situation is producing imbalances in both nutrients, and in Silica. When imbalance between Silica and N or P occurs, the probability of a harmful algal bloom increases (Garnier *et al.*, 2010). Next to this, the declining water quality can cause loss of species and shifts from pollution-sensitive towards pollution-tolerant organisms (BIP, 2014). Increase of sediment loads can also interfere with fish respiration, smother bottom-living organism and cover spawning areas (BIP, 2014).

To determine the effect of water quality on freshwater biodiversity, the Water Quality Index for Biodiversity (WQIB) is used. The WQIB is based on a comprehensive global-scale dataset on water temperature, dissolved oxygen, pH, electrical conductivity (salinity), nitrogen and phosphorus (UNEP-GEMS, 2008). Based on this index, water quality is fair or poor in more than half of all freshwater ecosystems in the Americas, Europe and Africa (Figure 8.5). General declines in the percentage of stations classified as good or excellent were detectable in the Americas and Europe dating back to the 1970s and 1980s, but have stabilized in recent decades (Figure 8.5). Water quality in Oceania appears to have increased in the last decade or two, as the proportion of stations classified as excellent or good has increased. The water quality in Asia and Africa is declining, but regional differences exist. For example the N and P pollution in parts of Indian rivers has significantly declined (Evans *et al.*, 2012). N and P pollution are the dominant factors that underlie low water quality, with approximately 60% of all stations having fair or low water quality based on N and P pollution criteria (UNEP-GEMS, 2008).

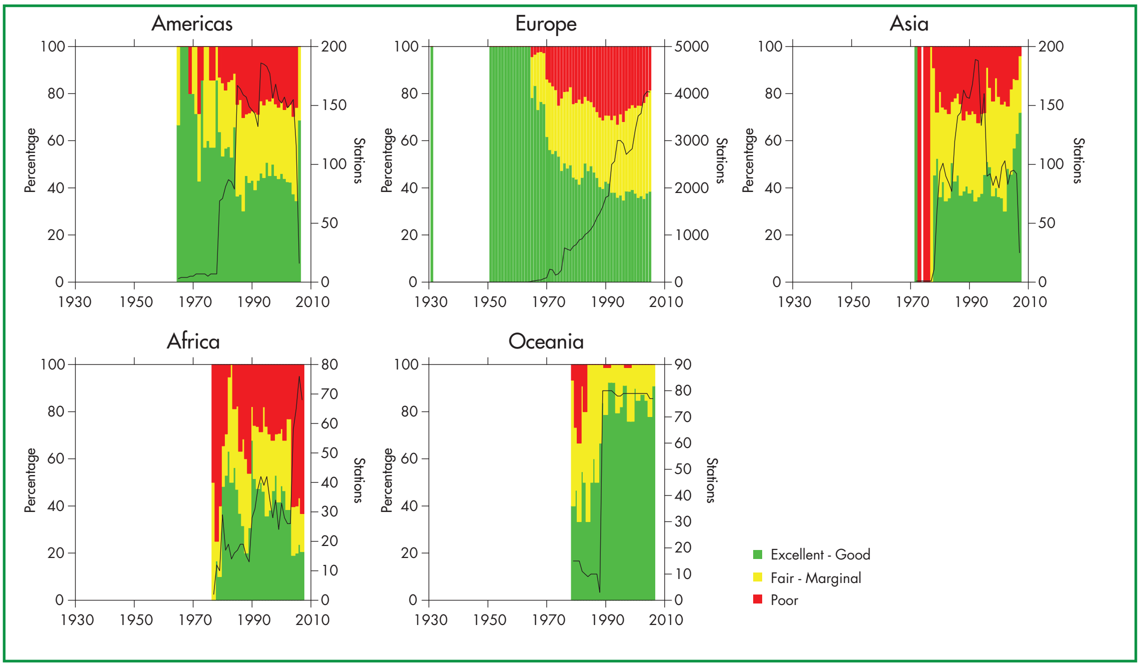


Figure 8.5. Regional Water Quality Index for Biodiversity (WQIB) scores for monitored freshwater systems. Excellent to good scores are indicated in green, fair to marginal and poor scores are indicated in yellow and red respectively. Solid black line indicates number of stations reporting in any given year. Source: UNEP-GEMS (2008).

The most severe manifestation of N and P pollution is the appearance of "dead zones" where oxygen levels in the water drop to such low levels that many aquatic organisms are killed. Prior to the 1960s, eutrophication-induced hypoxia was mostly associated with rivers, estuaries, and bays. By the end of the 1980s the so called "dead zones" had developed and expanded in continental seas, such as the Baltic Sea, Kattegat, Black Sea, Gulf of Mexico, and East China Sea (Diaz & Rosenberg, 2008). In inland waters, such dead zones have been recorded in North America's Lake Erie (Conroy *et al.*, 2011) and Africa's Lake Victoria (Verschuren *et al.*, 2002). Since the 1960s, the number of hypoxic systems has about doubled every decade and by 2010 hypoxia had become a major worldwide environmental problem with about 650 systems with reports of hypoxia (Figure 8.6). About 10% of these hypoxic systems have improved from management of nutrient and organic matter discharges. Additionally, over 400 coastal sites globally were identified as areas of concern that currently exhibit signs of eutrophication and are at risk of developing hypoxia (Figure 8.6; Conley *et al.*, 2011; Diaz *et al.*, 2010).

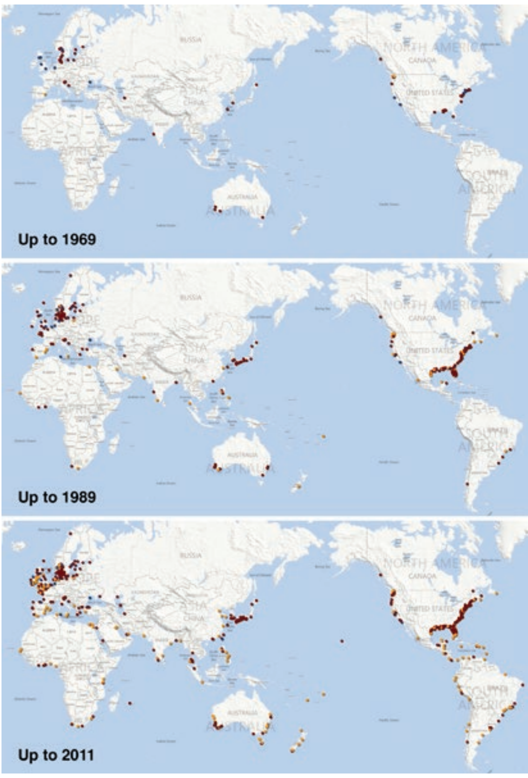


Figure 8.6. The global pattern of development of coastal eutrophication (yellow dots) and hypoxia (red dots). Each dot represents a documented case related to human activities. The number of hypoxic sites is cumulative through time (from Diaz *et al.*, 2013). An interactive version of this figure can be found at: http://www.wri.org/our-work/project/eutrophication-and-hypoxia/interactive-map-eutrophication-hypoxia

8.1.1.iv Other pollutants

Micro pollutants - There is a wide range of emerging environmental contaminants or micro pollutants that are not covered by current regulations, but that may pose risks to aquatic species. However, the threat posed by many of these compounds to marine species is poorly studied and often unknown. These include plastic, pharmaceuticals, personal care products (PCPs), steroids, hormones, surfactants, perfluorinated compounds (PFCs), various persistent organic pollutants (POPs), nanomaterials and swimming pool disinfection by-products (Farre *et al.*, 2008). For example plastic debris on the ocean surface has been estimated up to 3,520,000 items km^2 (Barnes *et al.*, 2009) and up to 100,000 items m_2 on some shores (Yamashita & Tanimura, 2007). Plastic debris may act as a vector for invasive species (Gregory, 2009) and may entangle or be ingested by marine species. More than 260 species have been reported as being affected by plastic debris.

Heavy metals – Industry, agriculture, households and other human activities have emitted various heavy metals and it is expected that global output of metals will increase with 85% by 2020 compared to 1995 (HELI, 2004). These heavy metals have resulted in many contaminated sites across the globe; chemicals have contaminated soil, water and air especially in developing countries (Blacksmith Institute, 2012). Many developing countries suffer from heavy metal pollution. In China, for example, the heavy metal pollution is worsening; it has been estimated that about one-tenth of China's cultivatable land has been contaminated, mostly in economically developed areas (Qi, 2007; Chen *et al.*, 2014). Below we discuss lead, chromium and mercury pollution as examples of heavy metal pollution.

Mercury pollution can be found worldwide and is released into the environment through mine tailings or by evaporation from gold-mercury amalgams to recover the metallic gold. Biodiversity is heavily affected by this pollutant since it is bio accumulative and persist in the food chain (Blacksmith Institute, 2012). For example in the Arctic, which is a major sink of mercury, this has led to an increasing mercury concentrations in marine animals (CAFF, 2013).

Chromium pollution is caused by tanneries and mining in various African, South American and North Asian countries. The majority of the chromium-contaminated sites can be found in Pakistan and India (Blacksmith Institute, 2012).

Lead can result in air and water pollution and can even contaminate food (Norton *et al.*, 2014). The global production of lead has increased by 9% in 2011, due to increases in China, India and Mexico (Blacksmith Institute, 2012). Most contaminated sites can be found in Africa, South America, South and Southeast Asia (Blacksmith Institute, 2012). To halt the lead pollution, increasing quantities are recycled, however recycling often occurs at poorly controlled facilities making lead processing itself a source of pollution.

Heavy metal pollution persists also in developed countries. In Canada, for example, there are more than 10,000 contaminated sites (McKie, 2014) and in Australia more than 150,000 contaminated sites (Carbonell, 2013; Ruehl, 2013). At the beginning of 2014, the superfund programme of the USA identified 1,319 contaminated sites in the USA (EPA, 2014), however there is some discussion whether this programme includes all contaminated sites (Scorecard, 2011). In Europe on average 4.2 potentially contaminated sites are reported per 1,000 inhabitants and about 5.7 contaminated sites per 10,000 inhabitants (van Liederkerke *et al.*, 2014). The distribution of the different contaminants is similar for both liquid and solid matrices. Next to heavy metal pollution also mineral oils are frequent contaminants. Contamination by mineral oil is especially dominant in Belgium (solid matrix: 50%) and Lithuania (solid matrix: 60%), while for Austria (solid matrix: 60%) and the FYROM (solid matrix: 89%) heavy metals predominate. Also indirect pollution through mining activities occurs by the release of acid mine drainage, causing acidification of aquatic ecosystems (Janssens De Bisthoven *et al.*, 2005). Generally, phenols and cyanides make a negligible overall contribution to total contamination.

To delineate the spatially and temporally variability of the heavy metal threats to soil ecosystems the *mixture toxic pressure of a site* can be determined. It is expressed as multi-substance Potentially Affected Fraction of Species (msPAF; Posthuma *et al.*, 2002). The msPAF can be interpreted as potential loss of species, signifying hazard differences amongst areas. As example, a spatial analysis of emission, fate, exposure and mixture assessment data has been compiled in a map that shows spatial differences in toxic pressures of two metals in Europe (Figure 8.8, Cd and Pb). The results demonstrate that there is variation in the net toxic pressure in top soils across Europe (De Zwart *et al.*, 2010; van der Voet *et al.*, 2013). Assuming a 1:1 association between toxic pressure and species loss, a loss of biodiversity from Cd- and Pb-deposition is expected to occur only in certain areas in Southern and Eastern Europe (De Zwart *et al.*, 2010; van der Voet *et al.*, 2013). Throughout Europe, the predicted loss of species due the expected metal (Cd and Pb) loads at steady state soil concentrations with current depositions is lower than 1% (De Zwart *et al.*, 2010; van der Voet *et al.*, 2013). However note that large uncertainties are involved in biodiversity loss estimation, up to a factor 1000 (Unsworth *et al.*, 2006). Also other metals like copper and zinc should be taken into account.

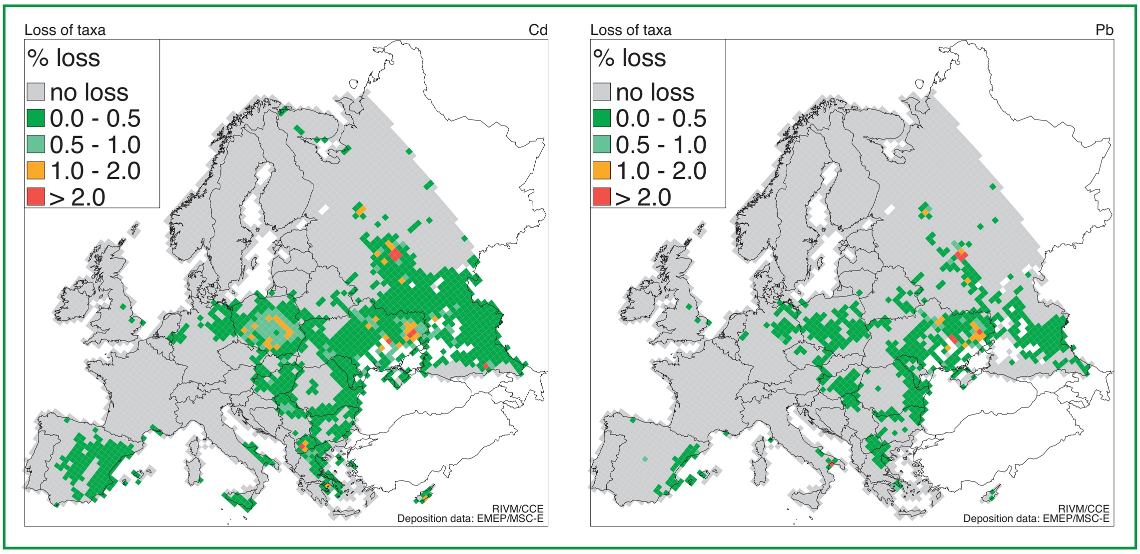


Figure 8.7. The 99th percentile of loss of species at steady state for cadmium (left) and lead (right) Source: De Zwart *et al.* (2010).

Oil spills - In the open ocean, small quantities of oil are readily dispersed and degraded (GESAMP, 2007). However, catastrophic releases of oil in coastal waters can be a threat to marine ecosystems and species. Regulation that has followed catastrophic oil spills has been partially successful in reducing the occurrence of such events. Oil spills from tankers have decreased dramatically from an average of 314,000t in the 1970s to 21,000t in the early 2000s (Jernelöv, 2010). This has partially arisen as a result of redesign of oil tankers as well as improvements in the management of shipping traffic and navigational technology (Jernelöv, 2010). Operational discharges of oil are still substantial (in the order of 100,000t) but again the trend is downwards (Jernelöv, 2010). However oil spills from pipelines, mainly on land, have increased largely as a result of ageing infrastructure, poor maintenance and lack of prompt remedial action especially in former Soviet Union and West Africa (Jernelöv, 2010). One trend for oil exploration and production has been the increasing depth of oil wells (Caineng *et al*., 2010) and also movement into more extreme environments such as the Arctic (CAFF, 2013). The Deepwater Horizon disaster of 2010 has shown that impacts from catastrophic oil spills can extend into deep-sea ecosystems and negatively impact species such as cold-water corals (White *et al*., 2012) and other components of benthic communities (Montagna *et al*., 2013). It may take ecosystems to recover from such disaster as little as 3-4 years (for rocky shores; Kingston, 2002). Although in others, recovery can take as long as 10-15 years, as in the case of the Torrey Canyon oil spill on the shores of southwestern England (Hawkins *et al*., 2000), and oil may be detectable on shores up to 25 years or more (Kingston, 2002).

Pesticides – Leaching of pesticides is one of the main mechanisms leading to groundwater and surface water pollution (Van Dijk *et al*., 2013). Globally approximately 9,000 species of insects and mites, 50,000 species of plant pathogens, and 8,000 species of weeds damage crops. Therefore, about one-third of the agricultural products are produced using pesticides (Zhang *et al*., 2011). This has resulted in an increase of pesticide use, since 1990 pesticide use has been increasing up to two million tons per year (De *et al*., 2014). 45% is consumed by Europe and 25% by the USA (De *et al*., 2014). Most frequently used pesticides are in the form of herbicides (Figure 8.7), however regional differences exist. For example in India 80% of the used pesticides is in the form of insecticides (De *et al*., 2014).

Even though it is believed that pesticide use is necessary to ensure food safety, pesticides are also highly toxic to humans and the environment (Zhang *et al*., 2011). For example, one of the current debates concerning biodiversity is about the potentially harmful impact of neonicotinoids on pollinating insects like honeybees and bumblebees. This type of insecticide has become one of the most widely used insecticides over the last two decades and accounts for one third of the world insecticide market, however there is evidence that the use of this type of systemic pesticide is contributing to large-scale die-off of bee populations, although many other factors appear to play important roles (van der Sluijs *et al*., 2013; Simon-Delso *et al*., 2014). Therefore neonicotinoid use as a systemic pesticide in crops was recently suspended in the European Union due its suspected negative impacts on pollinators. However due to the high persistency of this pesticides, the impact on biodiversity will not be reduced immediately (van der Sluijs *et al*., 2014).

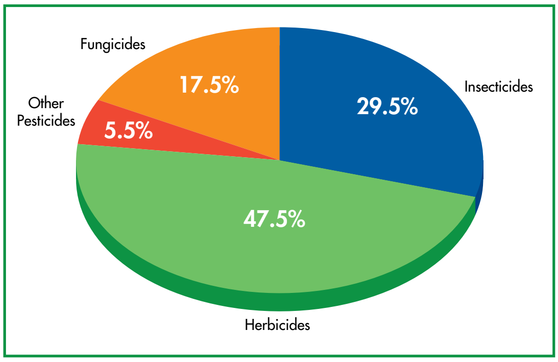


Figure 8.8. Worldwide consumption of pesticides Source: De *et al*. (2014).

8.1.2 Projecting forward to 2020

A large number of scenarios of future N and P pollution are available. A substantial fraction of these scenarios cover the 2030, 2050 and 2100 time horizons. Scenarios for 2030 have been used in this section when scenarios for 2020 are not available. 2020 pollution is based on interpolating from "current" (often for 2000) N and P emissions to 2030 estimates.

8.1.2.i N and P emissions

It is foreseen that the N and P losses to the environment will increase in the coming decades in a variety of business-as-usual scenarios (Bodirsky *et al*., 2014; Bouwman *et al*., 2005; Dentener *et al*., 2006; Seitzinger *et al*., 2010; PBL, 2012). In several regions N losses to the environment are expected to stabilized, but these are counterbalanced by large increases in Asia, Central and South America and Sub-Saharan Africa (Figure 8.9; Dentener *et al*., 2006; PBL, 2012). Similar trends are expected for P pollution; the stabilizing of P emissions in some regions is counterbalanced by the increase of P pollution in others, e.g. sub-Saharan Africa (Figure 8.9).

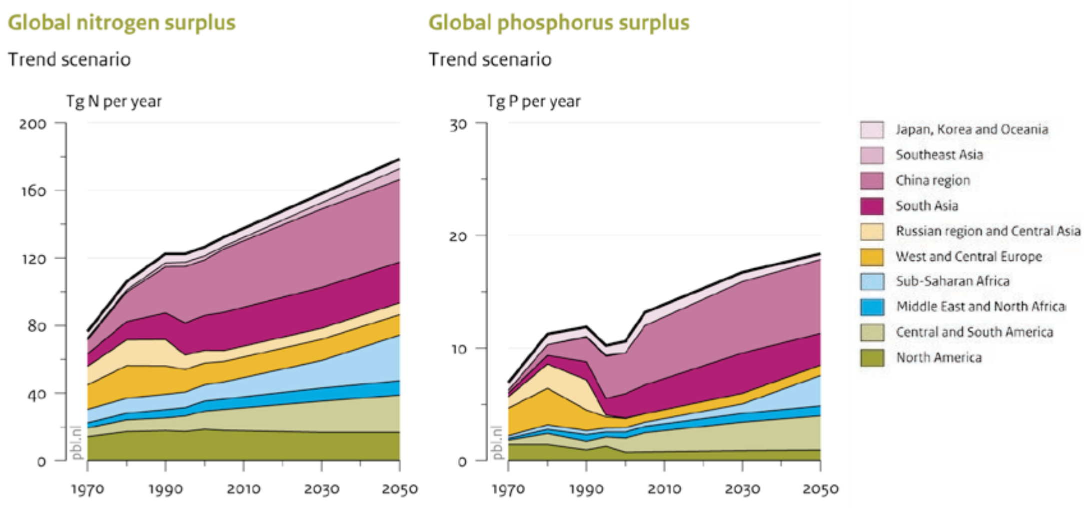


Figure 8.9. Global nitrogen and phosphorus surpluses in agriculture per region following the Rio+20 "baseline" scenario up to 2050. It is expected that the N and P losses from agricultural fields to the environment will increase in the coming decades. Europe, North America, China and India may have declining N and P surpluses, but these may be counterbalanced by large increases in fertilizer use in parts of Asia, Central and South America and sub-Saharan Africa. Source: PBL (2012).

8.1.2.ii N pollution in terrestrial ecosystems

The total N fertilizer input per hectare in developing and emerging countries is expected to exceed those of industrialized countries by 2030, due to high cropping intensities that prevail in many tropical countries (Sutton & Bleeker, 2013). In all scenarios developed for the IPCC this leads to a substantial increase of global N deposition, with especially high increases in Southern and Eastern Asia (up to 40-100%; Figure 8.10a, compare with Figure 8.2a; Dentener *et al.*, 2006; Lamarque *et al.*, 2013; Paulot *et al.*, 2013). Regulations on transport and industrial N emissions are assumed to continue to reduce emissions of oxidised forms of N in developed countries, but rising transport and industrial emissions are assumed to increase deposition substantially in Asia (up to 50-100% increase; Figure 8.10b, compare with Figure 8.2B; Dentener *et al.*, 2006).

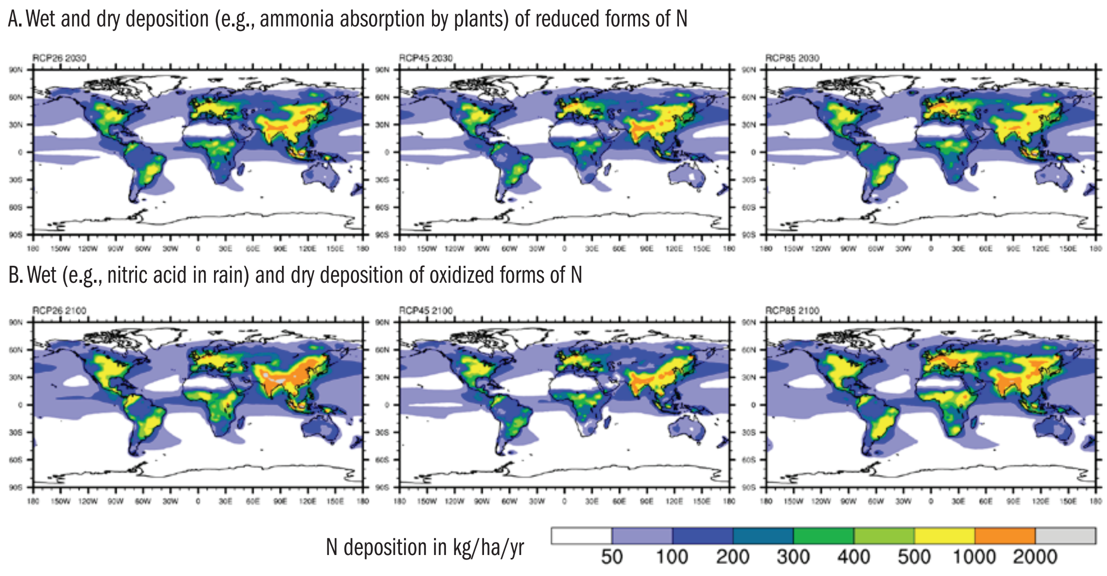


Figure 8.10. Scenarios of atmospheric N deposition in 2030 for three IPCC scenarios for strong (RCP2.6), moderate (RCP 4.5) and weak (RCP 8.5) climate mitigation scenarios. Deposition of reduced forms of N is primarily due to emissions from agricultural systems (panel A). Deposition of oxidized forms of N principally arises from industrial and transport emissions (panel B). Source: Adapted from Lamarque *et al.* (2013).

The extent to which future N deposition may exceed critical loads in terrestrial ecosystems has been explored for the 2020 to 2030 time horizon in several studies. Bobbink *et al.* (2010) estimated N deposition for the WWF Global 200 (G200) ecoregions of high conservation value. There are very large differences between scenarios even at this time scale. Assuming an average critical load of 10 kg N/ha/yr (Bobbink *et al.*, 2010), the percent of G200 ecoregions above this limit is similar to current levels in the most optimistic scenario (10 -12%; Figure 8.11, inset MFD scenario), but more than double the current levels in the most pessimistic scenario (25%; Figure 8.11, inset A2 scenario). Ecoregions in Southern and Eastern Asia are projected to be the most heavily impacted.

Also Paulot *et al.* (2013), using the RCP IPCC scenarios, project N deposition to exceed critical loads for some ecosystems in protected areas. Even though pollution from oxidized forms of N is projected to stabilize (China) or decline (US), reduced forms increase. For example in the Rocky Mountain area N deposition will exceed the critical loads, since this area is very sensitive to N deposition and has low critical loads.

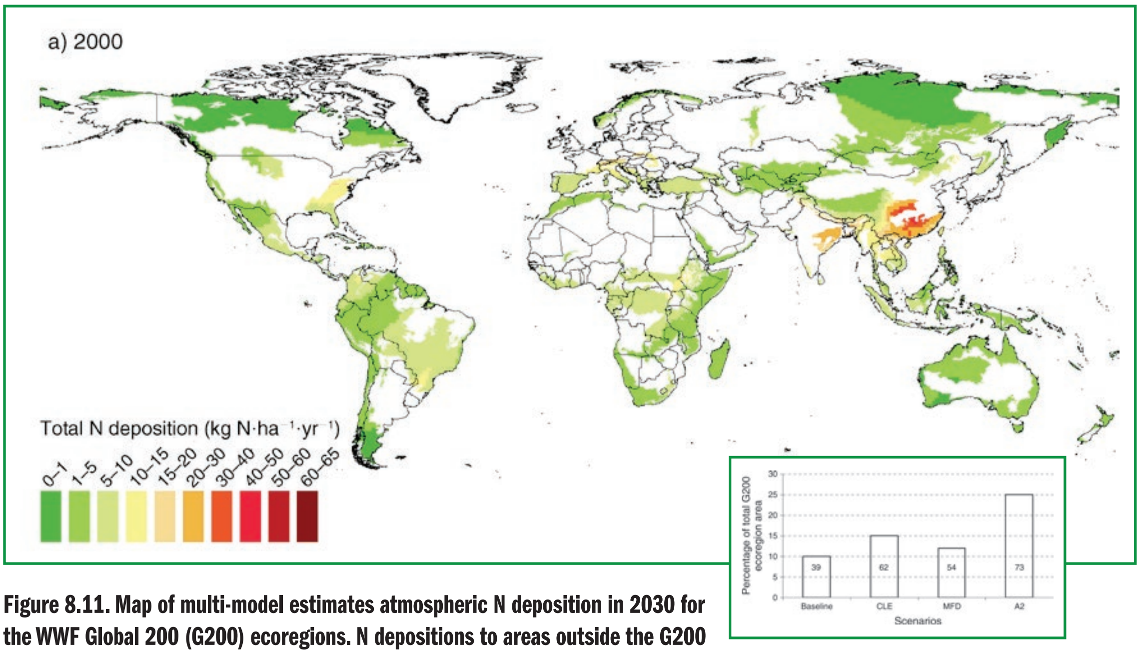


Figure 8.11. Map of multi-model estimates atmospheric N deposition in 2030 for the WWF Global 200 (G200) ecoregions. N depositions to areas outside the G200 ecoregions are not shown. The map is based the IPCC A2 emissions scenario which assumes rapid population growth with little attention paid to environmentally friendly actions. Inset: Percent of G200 ecoregions having N deposition exceeding 10 kg N/ha/yr which is a rough average of critical loads for terrestrial ecosystems. CLE = scenario assuming current legislation is enforced, MFD = maximum feasible reduction in deposition given foreseeable technologies and A2 = IPCC A2 scenario (corresponds to map). The numbers inside the bars show the number of G200 ecoregions in the area affected by each scenario. Note that projected deposition rates are low in Madagascar, Australia and New Zealand. Source: Bobbink *et al*. (2010); Dentener *et al*. (2006).

Currently, only 20% of the sites in Europe are protected from N eutrophication. Projections suggest that this will increase to about one third by 2020 based on N deposition reductions under current legislation. A "maximum technically feasible emission reductions scenario" would protect only half of sites by 2020, thus not fully protect ecosystems across Europe from N eutrophication (Holmberg *et al*., 2013).

The potential impact of N deposition on protected areas with high conservation value is estimated by using a "current legislation scenario" for N deposition (Bleeker *et al*., 2011; Dentener *et al*., 2006; Dentener *et al*., 2005). Protected areas in tundra, Mediterranean and mangrove systems are least exposed, but more than half of temperate forests and a third of temperate grasslands and tropical and subtropical broadleaf forests in protected areas are projected to be exposed to N deposition levels that could have negative impacts on biodiversity (>10 kg N/ha/yr; Figure 8.12). However, relatively few protected areas are projected to have N deposition levels exceeding 20 kg N/ha/yr; the largest fractions are in tropical, subtropical and temperate forests.

Optimistic scenarios suggest that it is possible to stabilize or achieve reductions in N pollution of terrestrial ecosystems over the 2020-2030 time frame in most regions. However, these levels of pollution are high enough in many regions to remain "detrimental to ecosystem functioning and biodiversity". To meet or exceed these optimistic scenarios, efforts need to be made in many countries to control agricultural N emissions. In addition, to control transport and industrial emissions advanced control techniques could reduce N emissions by 50%, however further efforts would need to be made (see section 8.2).

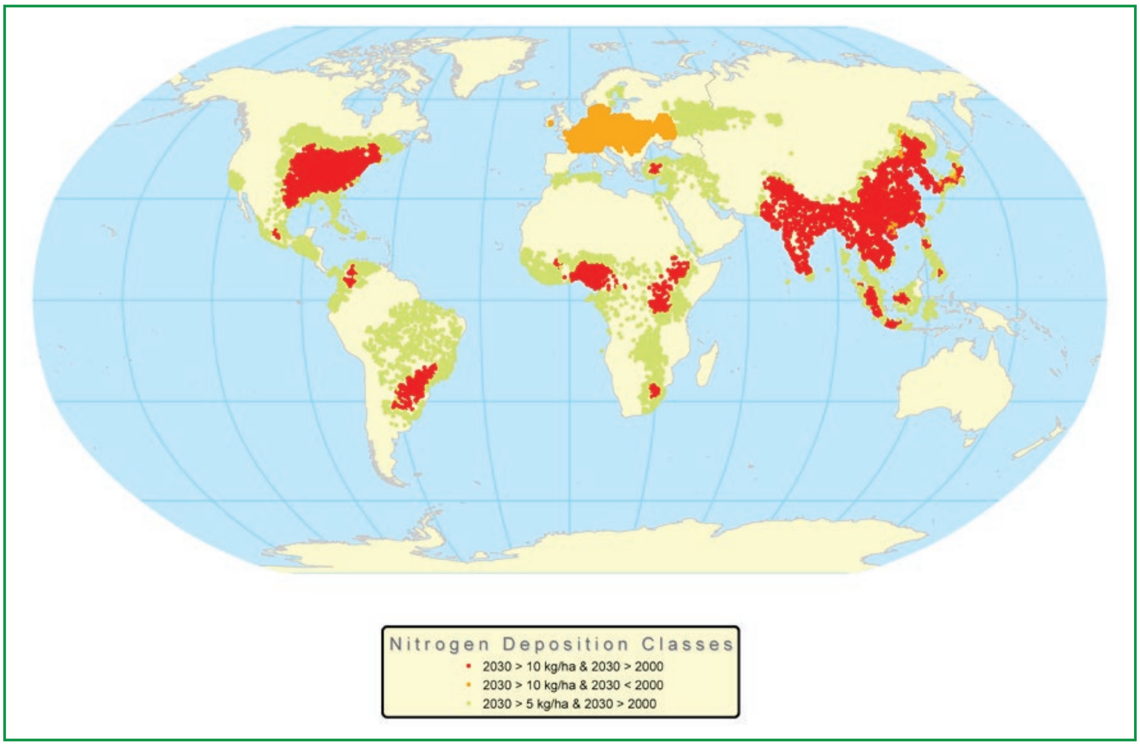


Figure 8.12. The distribution of deposition classes in protected areas. Red indicates areas where the N depostion in 2030 exceeds 10 kg N/ha/y and where the deposition in 2030 is higher than in 2000. Orange indicates areas where the N depostion in 2030 exceeds 10 kg N/ha/y and where the deposition in 2030 lower than in 2000. Green indicates areas where the N depostion in 2030 exceeds 5 kg N/ha/y and where the deposition in 2030 lower than in 2000 Source: Bleeker *et al*. (2011).

8.1.2.iii N and P pollution in aquatic ecosystems

Recent model estimates suggest that N and P loading in rivers by 2030 will differ from current levels substantially depending on socioeconomic development pathways (Figure 8.13; Qu & Kroeze, 2012; Seitzinger *et al*., 2010; Strokal & Kroeze, 2013; Suwarno *et al*., 2013). In general, scenarios suggest that agriculture will remain the key driver of N loading to rivers through 2030, while the future of riverine P will depend more heavily on sewage treatment, detergent P content and retention of particulate P in reservoirs (Seitzinger *et al*., 2010). Estimated increases over the last thirty years and projections for the next thirty years for both N and P loading are most striking in Southern Asia (Figure 8.13). In scenarios with high increases of agricultural production and consumption of meat, current trajectories of increasing export of N and P in rivers are predicted to continue or worsen in most regions especially in developing and emerging regions (Figure 8.13; GO 2030; Seitzinger *et al*., 2010). In scenarios with moderate increases in agricultural production and meat consumption, N export in rivers generally declines and P export generally increases, but only modestly (Figure 8.13; AM 2030). Models project that dissolved inorganic forms of N and P will increase, dissolved organic forms of N and P will remain relatively stable and particulate N and P will decline.

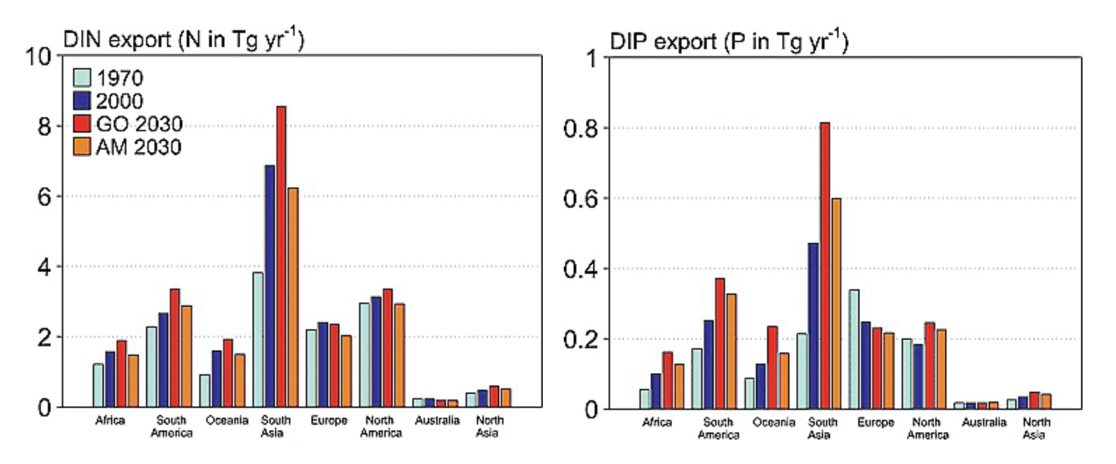


Figure 8.13. River export at continent/regional scale for (A) Dissolved inorganic nitrogen (DIN) and (b) dissolved inorganic phosphorus (DIP) in 1970 (light blue), 2000 (dark blue), and 2030 for the Global Orchestration (GO; the least favorable) and Adapting Mosaic (AM; the most favorable) scenarios of four the Millennium Assessment socio-economic scenarios (MEA, 2005). Source: Seitzinger *et al*. (2010).

Methods similar to Seitzinger *et al*. (2010) have been applied at national and regional scales including watersheds feeding the Baltic Sea (Strokal & Kroeze, 2013), Chinese coastal zones (Qu & Kroeze, 2012) and Indonesian coastal zones (Suwarno *et al*., 2013). These studies broadly confirm the global analysis of Seitzinger *et al*. (2010), but they have also explored a number of variants of scenarios that explore plausible reductions in factors mediating N and P loading to rivers. Large, but plausible reductions in organic and inorganic fertilizer use accompanied by improved removal of N and P from sewage are projected to substantially reduce N and P loading in rivers and coastal zones below "business-as-usual" scenarios and often result in substantial reductions compared current loadings by 2020 (see example for China; Figure 8.14). Despite these improvements, projected N and P pollution remain above limits that would avoid eutrophication (Strokal & Kroeze, 2013).

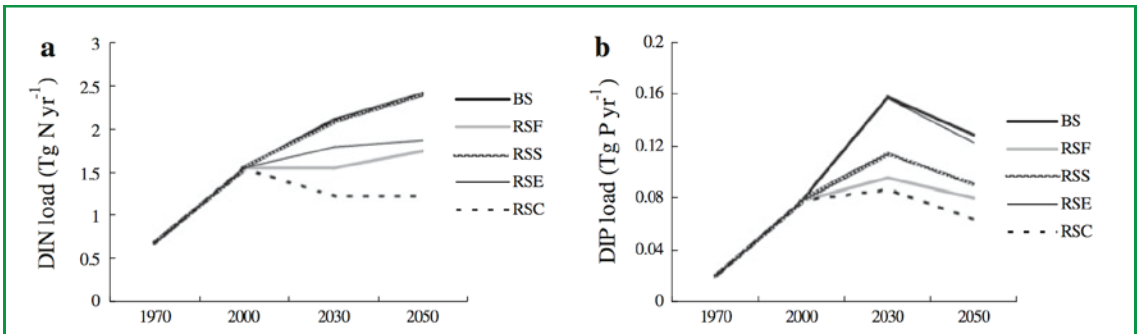


Figure 8.14. Trends and scenarios of A) dissolved inorganic N (DIN) and B) dissolved inorganic P (DIP) loading in rivers feeding Chinese coastal areas. BS = baseline scenario, RSF = reduction in fertilizer scenario, RSS = improved sewage treatment scenario, RSE = reduction of atmospheric N emissions, RSC = combination of RSF, RSS an RSE. Source: Qu & Kroeze (2012).

8.1.2.iii Other pollutants

Pesticides - Total pesticide used (in tonnes per year) is increasing globally. Next to this the potency of many newer pesticides and diversity of pesticides is increasing, as well as the use per unit area. For example in the USA, an increase in pesticide application of 5-14% is estimated (Koleva & Schneider, 2009). However, the use of some pesticides is expected to decline or stabilize; botanical carbamate, neonicotinoid, inorganic isoxazolidinon and triazine (Koleva & Schneider, 2009). The increase of pesticide application is expected to differ between crops; the lowest average increase is expected in fruit and vegetables: 5% in 2030, while the biggest average increase is expected in cereals: 10% (Koleva & Schneider, 2009). For the EU, the median predicated level of application for the 2090 scenario is expected to be 2,4-fold that of 1990 (Kattewinkel *et al.*, 2011). Therefore a minimum increase in the level of application of 22% is predicted (Kattewinkel *et al.*, 2011).

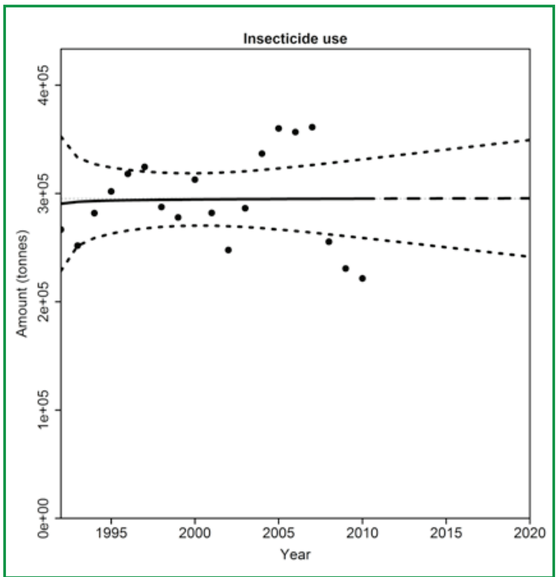


Figure 8.15. Statistical extrapolation of insecticide use to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant.
Source: FAO (2014).

Despite development of newer pesticide, the most common used pesticides still remain the choice of small farmers because they are cost-effective, easily available, and display a wide spectrum of bioactivity (De *et al.*, 2014). Next to this the toxic effect of pesticides will not disappear directly from the environment after reducing the use. For example, if we stop all DDT emissions now, only in the tropical regions the environmental levels are significant positive affected (Schenker *et al.*, 2008). In the temperate regions, the environment is still negatively affected for up to 20 years and in the Arctic for about 50 years (Schenker *et al.*, 2008).

Oil spills – The current decreasing trends of oil spills in the ocean are expected to continue (Figure 8.16). This includes spills from accidents involving tankers and accidental loss of oil during operations (GESAMP, 2007). On land there is an expected increase in loss of oil from pipelines and installations (Caineng *et al.*, 2010; Jernelöv, 2010). The trend in large-scale losses of oil to the environment from oil exploration and production at the frontiers of what is currently technically feasible is less easy to predict (Caineng *et al.*, 2010; Jernelöv, 2010)

Box 8.1: Pollutants in Arctic biodiversity

International agreements on toxic substances have made significant contributions in some pollutant reductions, as certain legacy chemicals have diminished in some Arctic wildlife populations. The Stockholm Convention on Persistent Organic Pollutants is often credited as a driving force behind lower levels of legacy persistent organic pollutants (POPs) in species, but levels can still remain high enough in some species, such as polar bear and some seabirds, to affect wildlife and human health.

Continued use of existing pollutants and emerging new ones pose complex problems for species in the Arctic, an area of the world where ocean and atmospheric currents result in a high deposit and accumulation of substances. A variety of recently emerging, but poorly studied, contaminants, such as polybrominated diphenyl ethers (PBDEs), are increasing. In addition, mercury concentrations are increasing in parts of the Arctic, including areas in Canada and Greenland, and remain a concern, especially for top predator species. Further complicating the issue is the unpredictable interaction between contaminants and climate change, and the largely unknown sensitivities of Arctic species to contaminants.

Source: Arctic Biodiversity Assessment (CAFF, 2013)

8.1.3 Country actions and commitments[1]

Most countries have established national targets, or similar objectives, related to Aichi Biodiversity Target 8 in their NBSAPs. These targets are generally in line with the Aichi Biodiversity Target. Further, several countries, such as Ireland, have identified actions to assist with the implementation of their target. The majority of targets which have been developed refer to reducing pollution generally. Few targets refer to reducing excess nutrients specifically. Overall, if implemented, the national targets and other mechanisms contained in the NBSAP would take a significant contribution towards the attainment of the Aichi Biodiversity Target.

Box 8.2: Clean and Green programme in Nauru

Poor waste management and uncontrolled pollution can exacerbate the degradation and hamper the restoration of both inland and coastal ecosystems on Nauru. Very few efforts have been carried out by local communities to address the issues of waste and pollution. Although a national waste collection system exists a huge percentage of wastes in Nauru does not make its way into the Public Dumpsite and ends up around homes in the coastal areas and inshore reefs. The programme Clean and Green, which is government funded and community driven, has proven highly successful at educating the public and facilitating the proper management and disposal of household wastes. The Clean and Green Programme engage and train a contingent of about 140 of young workers who are fully employed selected from the 14 districts to help promote awareness and education on waste management and provide support services to facilitate the effective collection and disposal of household wastes by district communities.

8.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

In this section we only describe actions and costs to reduce N and P pollution, however there are many actions possible to reduce the impacts of pollutions from other sources.

8.2.1 Actions

As described above, the main source of N and P pollution is fertilizer use in food production systems, however, fossil fuels and sewages are also a source of N and P pollution. Since fertilizer is essential for increasing food production (Seitzinger *et al*., 2010) and all scenarios illustrate an increasing of per capita food consumption and GDP, N and P pollution are expected to increase. Thus the challenge is to identify actions that decouple N and P pollution from human development (PBL, 2012). Given that N and P pollution also have many negative impacts on human health, there are substantial opportunities for net benefits by controlling N and P pollution for biodiversity and human well-being (Grinsven *et al*., 2013)

There are a number of well-studied actions that can reduce N and P pollution of terrestrial and aquatic ecosystems (Galloway *et al*., 2008; Seitzinger *et al*., 2010). Some of the actions are primarily technical, while others require behaviour change. Some actions should result in net gains for nearly all actors (better matching of N and P fertilization with crop demands), while others will require considerable investments for the sectors concerned (e.g., P removal in sewage plants). These actions can intervene in different parts of the N cycle. Some actions focus on the increase of nutrient efficiency use to reduce nutrient loss from agricultural areas. Others aim to change consumer behavior or recycling to reduce nutrient loss. The major actions are outlined below in Table 8.1 and 8.2.

Footnote

[1] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

Table 8.1. Actions to reduce nitrogen pollution Source: Sutton *et al*. (2013).

Sector	Action
Transport	Improve fuel efficiency and reduce the amount of emissions per combustion to reduce vehicle NOx emissions per km. Reduce fossil fuel use by minimizing vehicle use and promoting and increase the use of alternative energy sources.
Industry	Reduce NOx emissions, especially from power plants using a variety of proven methods (Frost *et al*., 2006).
Agriculture	Increasing nutrient efficiency by adapting N fertilizer application to needs of crops and climate. This will not only reduce nutrient loss but also increase crop production which will help meeting target 7. Reduce nutrient loss from agricultural sites by erosion control measures, maintain or restore wetlands and vegetation along rivers and streams, increase the use of legume/non-legume mixtures, cover crops, tillage management, buffer strips and minimize use of high N demanding crops such as maize. Minimize emissions from animal housing, manure storage and handling. Couple crop and livestock systems by better integration of animal manure in crop system.
Sewage and recycling	Expand and improve sewage treatment plants Recycle organic waste containing N by sewage treatment plants.
Culture	Decrease manure production by reducing meat and milk production and consumption where appropriate. Recycle organic waste containing N by sewage treatment plants. Recycle and reduce waste by improving the overall food supply chain and change consumption patterns.

Table 8.2. Actions to reduce phosphorus pollution Source: Sutton *et al*. (2013).

Sector	Action
Industry	Increase efficiency of phosphorus mining and processing.
Agriculture	Reduce nutrient loss from agricultural sites by erosion control measures, maintain or restore wetlands and vegetation along rivers and streams, increase the use of legume/non-legume mixtures, cover crops, tillage management and buffer strips. Minimize emissions from animal housing, manure storage and handling. Couple crop and livestock systems by better integration of animal manure in crop system.
Sewage and recycling	Eliminate phosphates from laundry and dishwasher detergents. Recycle organic waste containing P by treating sewage and wastewater before it reaches ground or surface waters and by improving P removal at sewage treatment plants.

Some of the solutions to N and P pollution are primarily socio-technical. Many countries have regulations concerning sewage treatment, and this is projected to become more systematic in scenarios with strong socioeconomic development. Next to this many developed and emerging countries have air quality regulations that aim to reduce NOx emissions from vehicles and industry. These regulations are generally getting more stringent over time, but need to be uniformly enforced. Efforts must be made to get older vehicles out of circulation and to minimize vehicle traffic (Seitzinger *et al*., 2010).

Ecosystem services, such as water purification, can reduce N pollution. For example wetlands, including constructed wetlands, play a major role in sequestering nutrients by retaining sediment particles (which often have adsorbed phosphorus) and facilitating denitrification of reactive to non-reactive N. However it should be noted that denitrification could also result in the formation of N_2O, which can increase climate change. Next to this, wetlands continue to be lost worldwide through drainage and channelization (Carpenter *et al*., 2011). Protecting (Targets 5 and 11) or restoring (Targets 14 and 15) wetlands could therefore make an important contribution to achieving Target 8.

However, the most important projected changes in N and P pollution over the coming decade will arise from agriculture. Sustainable agricultural practices are the most important key to reducing N and P pollution. By achieving Target 7, through increasing nutrient use efficiency, while simultaneously increasing agricultural production would make a major contribution to achieving Target 8. The key for balanced fertilization is efficient recycling of N from animal manures. Bouwman *et al.* (2013) estimate a potential reduction of the gross nitrogen balance in 2050 of 25% when N in manure is recycled. Balanced fertilization (defined as a balance between N fertilization input and plant demand) is a promising measure. This measure can reduce N leaching by 39% in 2020 relative to 2000 in Europe (Table 8.3; Oenema *et al.*, 2009). Smaller effects can be achieved with low-protein animal feeding, reconnection of crops and livestock farming, reduced meat consumption diet, reducing waste and post-harvest losses and ammonia emissions abatement measures (Oenema *et al.*, 2009; Grizzetti *et al.*, 2013; Westhoek *et al.*, 2014).

However, some policies can also cause increasing nutrient pollution, for example biofuel policies. Increased demand for biofuels could increase global fertilizer use to satisfy additional production needs. For example with a 50% increase in the biofuel mandate, N_2O emissions from fertilizer use could rise by 1.4% (Mosnier *et al.*, 2012).

Nevertheless, due to the wide range of N and P sources and the many ways to improve their management there is no single solution. A package of measures is needed to address nitrogen and phosphorus systems as a whole (See box 8.2; Sutton & Bleeker, 2013). For example in the USA the package of legislations to control nitrogen pollution consists of efforts to decrease sewage pollution, smog, acid rain, nitrogen inputs in coastal systems and runoff from crop and animal production (EPA, 2011). The legislations in the EU combine balanced fertilization, low-protein animal feeding and ammonia emissions abatement measures. This could potentially reduce N leaching by 41% by 2020 in Europe (Table 8.3; Oenema *et al.*, 2009). Nevertheless the N leaching levels remain generally above critical loads levels, illustrating that big efforts are needed to be able to reduce nutrient pollution below the critical loads. Next to this there can be unintended consequences when focusing on one pollutant only, since not all practices are effective for both N and P pollution reduction. Therefore an integrated strategy is needed to reduce both the effect of N and P pollution on biodiversity (EPA, 2011).

Table 8.3. Percentage of N emissions reduction in Europe by 2020 when different policy options are enforced Source: Oenema *et al.* (2009).

		NH_3 loss	N-leaching
2000	Reference year (million kg)	2873	2782
2020	Baseline scenario	-10%	-10%
2020	Balanced fertilizer in designated nitrate vulnerable zones (NVZ)	-14%	-31%
2020	Balanced fertilizer in whole EU-27	-18%	-39%
2020	Balanced fertilizer in NVZ and low-protein animal feeding	-20%	-34%
2020	Balanced fertilizer in whole EU-27 and low-protein animal feeding and NH_3 emissions abatement	-31%	-41%
2020	NH_3 emissions abatement	-21%	-9%

Box 8.3: European nitrogen legislation

The EU legislation to reduce nitrogen loading consists of actions to reduce atmospheric deposition and leaching. The three most important pieces of EU legislation for reducing the nitrogen loading to ecosystems are:

1. The Nitrates Directive (1991/676/EEC) which caps the total application of N from animal manures to 170 kg N/ha and restricts application of manure and inorganic fertilizer in situations with high risk of N loss.
2. The National Emissions Ceilings Directive (NECD 2001/81/EC) caps emissions of NH_3 and NO_x at national levels to reduce acidification and eutrophication. This directive also defines best management practices to reduce ammonia losses.
3. The Urban waste water treatment directive (1991/271/EEC) sets targets for N-removal efficiencies.

Due to these and other regulations, ammonia emissions in the EU27 declined 30% between 1980-2011 (EMEP, 2013). On average, the gross nitrogen balance (i.e., an indicator of losses to the environment) decreased in the EU27 between 1980-2005 by 36% (Bouwman *et al.*, 2013). Before 2000, when European directives were not yet implemented, the Gothenborg protocol (regulations for reducing acid rain and air pollution) resulted in major reductions in N emissions. Emissions reduction effects of the NECD and Nitrates Directive after the year 2000 were small (Velthof *et al.*, 2014). However, individual EU member states with strict national nitrate and ammonia policies, e.g. Denmark, Belgium and The Netherlands, achieved higher reduction of ecosystem loadings, although levels generally remain well above those that cause ecological damage (Grinsven *et al.*, 2012). Nevertheless, total N loads to rivers in EU 15 have remained relatively high and stable since 1990, although for some rivers such as the Rhine there has been substantial improvement (Bouraoui & Grizzetti, 2011).

8.2.2 Costs and cost-benefit analysis

Reducing N and P pollution results in several costs, however not reducing N and P pollution also leads to high costs (Table 8.4 and 8.5). For example, the damage costs caused by harmful algal blooms due to N and P pollution might range between US$37 million and US$72 million for the fishery industry in the month following a fish kill (Compton *et al.*, 2011). When including the damage for recreation, waterfront real estate and recovery of species, eutrophication costs US$2.2 billion per year in the USA.

The costs of nitrogen pollution in Europe are estimated at €75–€485 billion per year (Grinsven *et al.*, 2013). Globally the environmental costs only range between €13 - €65 billion per year (Erisman *et al.*, 2013). According to the US acid rain programme, acidification and damage to material due to N oxides cost US$133 million annually (Compton *et al.*, 2011). In the UK it is estimated that the cost of ecosystem services loss due to nitrogen runoff is €0.3 per kg N (Brink & Grinsven, 2011). This has been estimated that the costs to restore the occurrence of disappeared target species are €2.5 per kilo NO_x, €2.3 per kilo NH_3 (Brink & Grinsven, 2011).

As illustrated in Table 8.1 and Table 8.2, P and N pollution can be reduced by several actions, however implementing these actions also incurs substantial expenses. For example, actions to reduce nitrogen runoff from agricultural sources have estimated costs of US$26.80 per kg of nitrogen runoff avoided (Talberth & Gray, 2012). The costs include the implementation of cover crops, riparian forest buffers, grass buffers, water control structures and animal waste management create.

Next to this, to reduce N and P pollution in water and to reduce the number of dead zones, wetlands should be maintained or restored, upstream wastewater treatments plants should be increased in regions that currently lack treatment or sewage and wastewater can be recycled. Increasing wastewater treatments plants has different costs for N and P pollution. For N pollution it has been estimated that this could cost between US$6.13 - US$9.33 per person per year, resulting in US$0.27-US$0.41 billion per year (Talberth & Gray, 2012). Wastewater treatment plants for P pollution could cost between US$8000-15000 per tonne of phosphorus (koppelaar & weikard, 2013). This is 30-40 times the more than the cost of phosphate rock production (Koppelaar & Weikard, 2013). However, recent technological developments have reduced the cost of wastewater treatments to US$5300-US$8200 tonne phosphorus (koppelaar & weikard, 2013). Also after P has been removed from the water, incineration could cost up to US$7280 (Koppelaar & Weikard, 2013). Besides technical solutions, wetlands can also remove P from wastewater. The capital expenditures necessary to implement a constructed wetland are competitive with those for alternative technologies (Kadlec, 2006). The costs of the land, earthmoving and water conveyance and control are to be compared to these for tanks, pumps, chemicals and sophisticated controls (Kadlec, 2006). Wetlands also need maintenance (inspections, vigor and health of vegetation, damage from muskrats and beavers), however these activities are a factor ten less than required for mechanical treatment (Kadlec, 2006).

However, addressing N and P pollution also leads to several benefits. Decreased N and P pollution reduces the costs of treating water, increases human health, increases recreational opportunities, improves fish habitat and health, increases property values, avoids costs associated with dredging and finding water supply substitutes, and increases aesthetic and existence values for biodiversity (Talberth & Gray, 2012; HLP 2014). For example reduced sludge handling and eutrophication could save up to US$6.61 per lb. P removed (Seymour, 2009). In the UK the decrease of reactive nitrogen in the atmosphere by 25% has resulted in a net benefit (Equivalent Annual Value) of £65 m (£5 m to £123 m, 95% CI; Jones *et al*., 2013). In the USA it was estimated that the commercial timber industry would benefit about US$800 million annually by reduced nitrogen oxides, because nitrogen oxides contribute to ozone formation, which can reduce forest and crop production. This also results in an estimated US$700 million benefit of grain crops (Compton *et al*., 2011) and US$650 prevented agricultural productivity loss per year (Koppelaar & Weikard, 2013).

Improving sanitation improves human health, increases productivity and prevents deaths. Therefore every dollar spent on improving sanitation results in an average economic benefit of US$8.5 (Hutton & Haller, 2004). Even though increasing sanitation could cost up to US$0.41 billion per year, it would result in a benefit of US$1.8-US$2.87 billion per year (Talberth & Gray, 2012). Overall environmental benefits resulting from treatment of wastewater can result into €50,035,000 per year or €0.245 per m3 wastewater treated (Hernández-Sancho *et al*., 2010).

In addition to these benefits, the eliminated agricultural subsidies leading to excess nutrients (Target 3) could be used to fund the technical and behavioural changes needed to meet this target (US$350 billion per year; Talberth & Gray, 2012).

Table 8.4. Costs of nitrogen pollution and the costs and benefits of reducing nitrogen pollution Source: Talberth & Gray (2012).

Cost of nitrogen pollution	Cost
European nitrogen pollution cost	€75 - 485 billion per year
European environmental costs	€13-65 billion per year
Algal blooms for fisheries industry	US$37 - US$72 million per algal bloom
Recreation	US$2.2 billion annually
Acid rain programme	US$133 million annually
Ecosystem services loss	€0.3 per kg N
Restore species	€2.5 per kilo NO_x
Restore species	€2.3 per kilo NH_3
Costs of action	**Cost**
Reducing dead zones by sanitation development	US$0.27-$0.41 billion per year
Reducing dead zones by reducing N runoff from agriculture	US$26.80 per kg nitrogen runoff reduced
Benefits from reduced nitrogen pollution	**Benefit**
Reduced ozone formation: forestry	US$800 million annually
Reduced ozone formation: grain crops	US$700 million annually
Reduced reactive nitrogen in atmosphere	£65 m Equivalent Annual Value
Improved health by improving sanitation	US$1.8-US$2.87 billion annually
Reduced agricultural subsides	US$350 billion annually

Table 8.5. Cost of phosphorus recycling Source: Hernández-Sancho *et al*. (2010); Koppelaar & Weikard, (2013); Molinos-Senante *et al*. (2011); Seymour (2009).

Cost of action	Cost
Non-food recycling	US$7,030 per tonne P
Wastewater recycling	US$6,750 per tonne P €2.1 million per year
No-till farming	US$1,030 per tonne P
Capital costs wetlands	US$50 per tonne P
Incineration of nutrient retrieved from wetland	US$7,280 per tonne P
Benefits from reduced P pollution	Benefits
Prevented cost (sludge handling/ eutrophication)	US$6.61 per lb. P removed
Overall benefits from wastewater treatment in Spain	€50 million per year €0.245 per m^3 water treated
Overall benefits from wastewater treatment in Valencia	€907,000 per year €1.022 per m^3 wastewater treated
Prevented agricultural productivity loss	US$650 million per year

8.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

8.3.1 Terrestrial biodiversity

The N and P pollution is not expected to decrease by 2020, as well as their impact on biodiversity. Especially (sub)tropical moist broadleaf forest temperate forest and ecosystems in northern Europe and North America are affected (Azevedo *et al*., 2013b). For example N deposition can cause soil acidification and foliar damage, especially in mosses and lichen, resulting in degradation of plant diversity (Figure 8.16b; Bobbink *et al*., 2010; Sutton *et al*., 2013; Sutton *et al*., 2011; Fenn *et al*., 2008; Guo *et al*., 2010). High concentrations of atmospheric ammonia result the replacement of local and regional lichen biodiversity with a few species able to profit from these conditions (Sutton *et al*., 2013). Due to acidification, oligotrophic species can be out-competed by more nitrogen-loving or acid-tolerant plants. Chronically elevated nitrogen deposition can also enhance susceptibility to stress such as frost damage, herbivory or disease (Dise *et al*., 2011).

Also the decline of species richness in grassland is expected to continue due to continued N deposition increase. Over the past 70 years, species richness in Great Britain, Germany and the Netherlands have declined significantly due to N deposition (Figure 8.16a; Stevens *et al*., 2010). Especially in Southern and Eastern Asia, where high increases of N deposition are expected, species richness in grasslands are expeced to decline.

The consequences of N deposition in arid zones are rather unclear, although some indications suggest increased invasions of exotic species (Target 9). This can be expected in several Mediterranean ecosystems (UNEP, 2004). Most arid zones are low N-deposition regions, however this does not mean that there are at low risk. The effect of N deposition on species richness seems to be curvilinear (Stevens *et al*., 2010). This implies that small increases in N deposition will have a larger impact on species richness when background deposition levels are low than when initial deposition levels are higher (above 20kg N ha-1yr-1; Stevens *et al*., 2010). Because at high N deposition rates many of the nitrogen-sensitive species have already declined, leaving mainly the less nitrogen-sensitive species (Stevens *et al*., 2010). Therefore areas which are undammed by N deposition have a higher potential for species loss (Stevens *et al*., 2010).

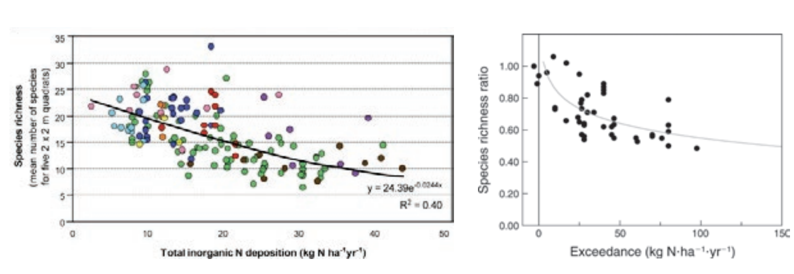


Figure 8.17. A) Correlation between observed levels of N deposition in Europe and species richness in acidic grasslands. Source: Stevens et al., 2010. **B) Species richness ratio (values less than one indicate species loss) in experiments with N enrichment.** Source: Bobbink *et al*., 2010.

Next to nutrient pollution, pesticide application is also influencing species richness. Many insects, e.g., pollinators and pest control species, birds, worms and aquatic species, are affected by pesticides (van der Sluijs *et al*., 2014). Pesticides can, for example, be held responsible for the global collapse in the bee population. A decline in biodiversity also affects many ecosystem services, for example services influencing the food production e.g., nutrient cycling, soil respiration and pollination (Chagnon *et al*., 2014).

Also birds, as non-target species, are highly impacted by pesticides. Insect-eating birds can be impacted via two pathways. Firstly due to trophic accumulation of pesticides trough consumption of contaminated invertebrates and crops (Gibbons *et al*., 2014). Also other herbivorous and insectivores mammals can be affected this way (Goulson, 2013). Secondly by a decrease in food due to a decline in insect population in agricultural areas (Mitra *et al*., 2011; Gibbons *et al*., 2014). Insecticides affects different groups of insects since insecticides can be washed from soils into waterways and affect aquatic insects, fish and amphibians (Hallmann *et al*., 2014; van der Sluijs *et al*., 2014). Insecticides can also be lost as toxic dust and affect flying insects (Goulson, 2013; van der Sluijs *et al*., 2014). Or they can be absorbed by crops and affect herbivorous insects (Goulson, 2014; van der Sluijs *et al*., 2014). Particularly herbicides impact food, nesting and shelter availability for specialist farmland bird species (Chiron *et al*., 2014; Geiger *et al*., 2010). However reduced competition with specialist bird species may result in an increase of generalist species and might therefore lead to increased species richness (Chiron *et al*., 2014).

The Red List Index can be used to track trends in the net impacts of pollution and attempts to manage or control these (BirdLife International, 2013). The Red List Index (RLI) shows trends in the survival probability of sets of species. It is based on data from The IUCN Red List – the number of species in each IUCN Red List category of extinction risk, and the number of moving categories between assessments owing to genuine improvement or deterioration in status (Butchart *et al*., 2004; 2005; 2007).

The RLI can be disaggregated to show trends driven by particular types of threats, showing the net balance between the number of species deteriorating in status owing to negative impacts, and the number improving in status owing to successful attempts to control or reduce pollution impacts. The RLI for birds showing trends driven by pollution has showed shallow declines since 1988, indicating that more species are moving closer to extinction owing to negative impact of pollution (Figure 8.18). Trends for other taxonomic groups are likely to be similar. However, other factors such as unsustainable agriculture, logging and hunting/trapping are much more significant drivers of declines than pollution in those terrestrial groups for which we have trend data at a global scale: birds, mammals and amphibians (see Target 12).

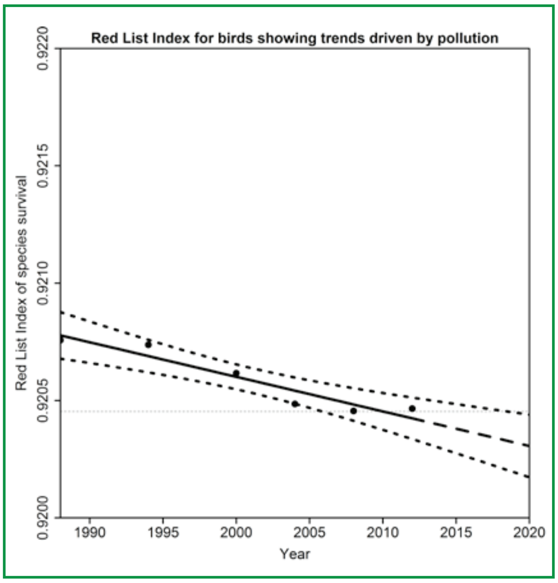


Figure 8.18. Statistical extrapolation of Red List Index for birds showing trends driven by the impacts of pollution or its control to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Source: PBL.

8.3.2 Aquatic biodiversity

Vörösmarty *et al*., (2010) estimated that pollution was one of the greatest threats of biodiversity in rivers, on par with water resource development. However the effect of P on aquatic species richness differs between types of water bodies. Rivers can withstand higher P levels before species richness is affected than lakes (Azevedo *et al*., 2013a). Phytoplankton species are less sensitive to P change than macrophytes (Azevedo *et al*., 2013a). Species richness in cold-region lakes and rivers has the highest sensitivity to P levels, while temperate streams and xeric lakes have the lowest sensitivity to P increase (Azevedo *et al*., 2013a). This analysis illustrates the very large negative impacts of pollution on freshwater biodiversity globally and that reductions in pollution levels are urgently needed.

The expected increase of pollution in aquatic ecosystems can impact on biodiversity through at least three distinct pathways (see Carpenter *et al*., 1998; 2011; Vörösmarty *et al*., 2010). First via habitat alteration, the addition of nutrients and toxic chemicals can lead to inhospitable conditions for many species, for example through direct toxicity, eutrophication or the development of "dead zones". In the worst cases, where such conditions last for several years and nutrients continue to accumulate in the system, the hypoxic zone will expand and anoxia may be established accompanied by release of H2S by microbial communities (Conley *et al*., 2009a; Diaz & Rosenberg, 2008). As well as direct mass mortality of organisms, hypoxia or anoxia reduce the available habitat for marine organisms (habitat compression), potentially affecting the life cycle, local movement and even large-scale migration of affected species (Craig & Crowder, 2005).

Secondly by extinction; a minority of native species is likely to thrive under polluted conditions. For instance, populations of certain aquatic plants often explode in lakes and rivers as nutrient availability is increased. Though these species may be a natural part of the system, the change in conditions can enable them to dominate the community. An example are harmful algal blooms (HABs), which occur as a result of eutrophication (Andam *et al*., 2008; Beveridge *et al*., 2013). Their impacts vary depending on the species of algae or ciliate, the nutritional or physiological status of the organism, its stage of life history and its concentration in the water. Effects include acute toxicity causing mass mortality of marine life, including invertebrates, fish, and birds but also other organisms such as cetaceans, pinnipeds and sirenians (Landsberg, 2002; Bouwman *et al*., 2005; Silvagni *et al*., 2005).

Finally, nutrient loading and other stressful chemical shifts can enhance the likelihood that aquatic ecosystems will be invaded by exotic species. Such invasions can directly or indirectly harm native species, leading to further losses of biodiversity (more detailed information can be found in the chapter about Target 9).

8.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

For many pollutants no projections for 2050 are available. However, a large number of scenarios exist for N and P use and emissions (Bouwman *et al*., 2005; van Drecht *et al*., 2009; PBL, 2012). Similar to the projections up to 2020 the 2050, business-as-usual scenarios project that N and P surpluses at the global scale will not decline in the coming decades (Bouwman *et al*., 2005; Seitzinger *et al*., 2010; PBL, 2012).

The expected global increase in N deposition by 2020 is expected to continue, leading to an expected increase of, on average, 35% by 2050 (Balmford *et al*., 2005; Bouwman *et al*., 2009; van Vuuren *et al*., 2010; Hardcastle & Hagelberg, 2012; PBL, 2012; Bodirsky *et al*., 2014). The expected variation of nutrient pollution in world regions by 2020 is expected to be maintained up to 2050. Decreasing trends are expected in North America, Western Europe and Japan, while in India stabilization is expected, due to a decrease in fertilizer inputs and animal manure production (Bouwman *et al*., 2005). In most developing countries an increase of N and P emissions is expected due to continued population and economic growth, urbanization, development of sewer systems, lacking wastewater treatment, increasing fertilizer use and increasing food (especially meat) consumption (Bodirsky *et al*., 2014; Bouwman *et al*., 2005).

If the nitrogen use efficiency is kept at a constant level, the use of nitrogen fertilizers is projected to grow from 82 million tonnes in 2000 to around 120-140 million tonnes in 2050 (see "Constant Nitrogen Efficiency" scenario from IAASTD, 2009). However, the projected increase of fertilizer use differs between studies: Galloway *et al*. (2004) projected that N fertilizer use will range from 140 Tg N/yr to 200 Tg N/yr, while Bouwman *et al*., (2009) calculated that N fertilizer use will range from 157 Tg/yr to 231 Tg/yr by 2050. This difference in projected fertilizer use is related to enhanced productivity, expansion of agricultural land and growing demand for agricultural products, especially in sub-Saharan Africa, Southeast Asia and Latin-America (Bouwman *et al*., 2005; IAASTD, 2009; van Vuuren *et al*., 2010; PBL, 2012,). Increased fertilizer use does not always have a negative influence on the environment, as agriculture without fertilizer use on nitrogen deficient soils can lead to soil degradation. Increased fertilizer use at at N deficient locations may therefore prevent land degradation.

Similar to the 2020 projections the transport and industrial emission are expected to rise up to 52 Tg N yr^{-1} (Galloway *et al.*, 2004). However these nitrogen emissions will be lower in scenarios with emphasis on climate change mitigation than in baseline scenarios, due to a shift from fossil fuels to renewable fuels (PBL, 2012).

The P use increase is expected to slow down after 2020 due to an expected stabilizing global population, higher efficiency rates in fertilizer use and improved agricultural management (van Vuuren *et al.*, 2010; van den Berg *et al.*, 2011). Nevertheless, the P surplus is expected to range from 18 Tg a^{-1} to 35 Tg a^{-1} by 2050 (Bouwman *et al.*, 2009). Similar to 2020 the increase of global P use is primarily attributed to fertilizer use (40% increase between 2010-2050), which projected to increase in developing regions and decrease in Europe and Northeast Asia (an Vuuren *et al.*, 2010; van den Berg *et al.*, 2011). Also P emissions from detergents and in livestock increase in all scenarios in developing countries, due to an increase in laundry and dishwasher detergents in Asia, South and Central America (an Vuuren *et al.*, 2010; van den Berg *et al.*, 2011).

Up to 2050 wastewater treatment will keep influencing N and P emissions. With the expected population growth in developing countries, sewage N and P discharge to surface water is projected to increase substantially between 2000 and 2050, especially in southern Asia, where increases of N and P emissions up to a factor 4–5 are foreseen. This results from a combined effect of increasing numbers of people, urbanization, and enhanced sewerage connectivity. Also North America (with high levels of N and P removal), rapid population and economic growth may also lead to increasing sewage N and P discharge to surface water (van Drecht *et al.*, 2009). Similar to the 2020 scenarios the optimistic scenarios for the development of wastewater treatment systems expect N and P effluents to remain above limits that would avoid eutrophication.

As outlined in section 8.2, several actions can reduce N and P pollution. Several scenarios have explored the effect of the key actions on the N and P pollution in 2050. These scenarios can be divided into three types of scenarios: i) increasing resource efficiency of crops and livestock ii) reducing post-harvest losses and promoting dietary changes and iii) recycling of fertilizer and human excreta.

In the first type of scenarios, integrating animal manure better into agricultural systems can reduce P fertilizer use by 10% in the USA and by 12% in Europe (Van den Berg *et al.*, 2011). Technological developments might result in even higher nutrient efficiency, which can decrease the N and P surplus, as shown by the "Rio+20 Global technology" scenario (Figure 8.19 and 8.20; PBL, 2012). However, due to increased food demand the IAASTD (2009) indicates that the fertilizer use is still expected to increase up to 110-120 million tonnes in 2050.

In the second type of scenarios, a decrease in meat consumption lowers the demand for fertilizer, since reduced meat demand in human diets also leads to decreased requirements for animal feed and associated fertilizer use (Bodirsky *et al.*, 2014; van Vuuren *et al.*, 2010). In the PBL "Rio+20 Consumption Change" scenario it is projected that the reduction of meat consumption reduces fertilizer use in industrialized countries. Global nitrogen use is reduced by 10% and phosphorus use by 11% (Figure 8.19 and 8.20); however in many regions the reduction is bigger (e.g. for phosphorus 16% in North America and 22% in western and central Europe; Bouwman *et al.*, 2009; PBL, 2012). According to Westhoek *et al.* (Westhoek *et al.*, 2014) a 50% reduction of meat and dairy consumption could result in 40% less N emissions in Europe.

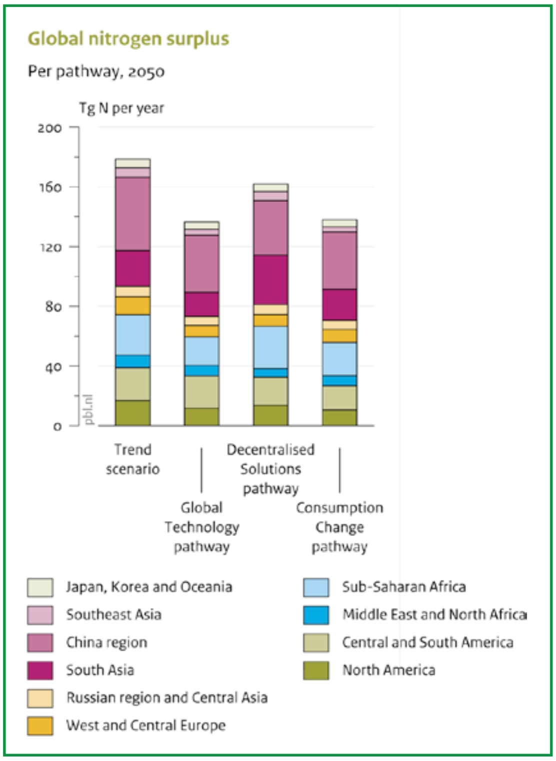


Figure 8.19. The regional N surplus in 2050 based on the Rio+20 "baseline" and three alternative socio-economic scenarios (PBL, 2012). Due to higher nutrient efficiency (scenario "global technology") and by reducing human excreta and fertilizer surplus (scenario "decentralized solution") nitrogen surpluses can be reduced. Reducing meat consumption (scenario "consumption change") leads to a reduction of global nitrogen surpluses of 10%. Source: PBL.

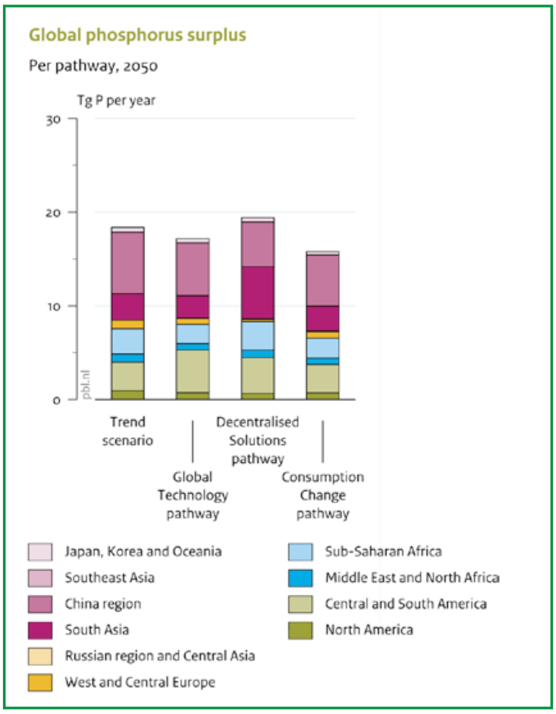


Figure 8.20. The regional P surplus in 2050 based on the Rio+20 "baseline" and three alternative policy scenarios (PBL, 2012). Due to higher nutrient efficiency (scenario "global technology") phosphorus surpluses can be reduced. Reducing human excreta and fertilizer surplus (scenario "decentralized solution") could lead to global reduction of P surpluses by 15%. Reducing meat consumption (scenario "consumption change") allows global P surpluses to be reduced by 11%. Source: PBL.

In the third type of scenarios, recycling of human N and P from households results in a lower N and P increase compared to the "business-as-usual" scenarios (Bouwman *et al.*, 2005; van Vuuren *et al.*, 2010; Bodirsky *et al.*, 2014). About 30% (3.1 million tonnes) of the P requirement for crop production can be delivered by animal manure and human excreta (van Vuuren *et al.*, 2010). This could lead to a reduction of global P fertilizer use of 15% (van den Berg *et al.*, 2011). In developing countries with nutrient deficit soils, better integration of animal manure is considered difficult because inorganic fertilizer use is minimal and animal manure already plays an important role in sustaining crop production.

All three types of scenarios reduce the N pollution, and thereby the effect on biodiversity (Figure 8.21). The PBL "Rio+20 Consumption Change" scenario indicates that the impact of nitrogen deposition on biodiversity (as measured by Mean Species Abundance (MSA)) can be reduced by 50% due to the recycling of N (PBL, 2012). Nevertheless the impact of N deposition on biodiversity at the global level is limited (about 0.4 – 0.7% reduced MSA loss).

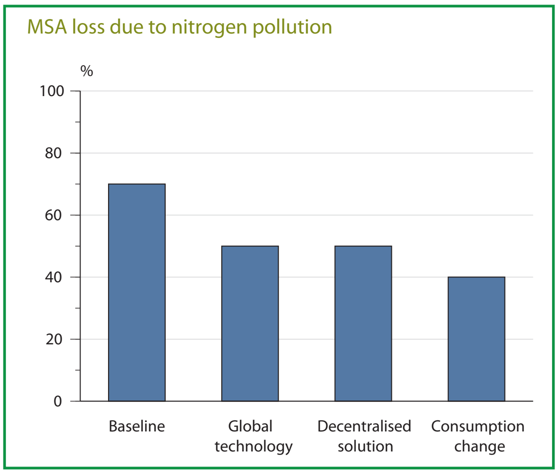


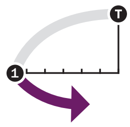
Figure 8.21. Mean Species Abundance (MSA) loss due to nitrogen pollution in 2050 in the Rio+20 scenarios. Source: PBL.

8.5 UNCERTAINTIES

The trend in nutrient deposition values are well studied in Europe and the USA. However, more routine measurement in South America and Africa are needed to reduce uncertainties (Lamarque *et al.*, 2013). The model estimations for the nutrient deposition also include uncertainties. Dentener *et al.*, (2006) illustrated that the wet deposition of nitrate is in relative good accordance with the measurements, with 60–70% of the model calculated deposition agreeing within ±50% with measurements. However in South America the models have a low spatial correlation and in India the nitrate deposition is with a factor 2 underestimated (Dentener *et al.*, 2006). The NHx deposition estimates show a small bias in all continents. However in East Asia the measurements are 50% higher than the model estimates, while in India the model overestimates the measurements with a factor 2 (Dentener *et al.*, 2006). This is also reflected in the determination of the effect of nitrogen deposition on biodiversity. The effect of nitrogen deposition has well been studied in temperate ecosystems. However more research is needed to establish the sensitivity of tropical and sub-tropical ecosystems (Bobbink *et al.*, 2010).

Also a major concern in determining the global water quality is ensuring good geographic representation of monitoring stations and temporal coverage of the same water quality parameters (UNEP-GEMS, 2008). For the WQIB data of approximately 100 countries have been included, however the reporting of data is inconsistent. Not all countries provide yearly or do not provide all quality parameters (UNEP-GEMS, 2008). In addition, some countries only supply data from few monitoring stations, or only from impacted sites with very little data from non-impacted or baseline sites (UNEP-GEMS, 2008).

8.6 DASHBOARD – PROGRESS TOWARDS TARGET

Element	Current Status	Comments	Confidence
Pollutants (of all types) have been brought to levels that are not detrimental to ecosystem function and biodiversity	No clear evaluation	Highly variable between pollutants	--
Pollution from excess nutrients has been brought to levels that are not detrimental to ecosystem function and biodiversity		Nutrient use leveling off in some regions, e.g. Europe and North America, but at levels that are still detrimental to biodiversity. Still rising in other regions. Very high regional variation	High

Authors: Jennifer van Kolck and Rob Alkemade, with contributions from Paul Leadley, Peter McIntyre, Marc Metian, Robert Diaz, Alex Rogers, Hans van Grinsven, Stephanie Januchowski-Hartley, Michiel Rutgers, Leo Posthuma, Kees Versluijs and Mark Huijbregts.
Extrapolations: Derek Tittensor
NBSAPs and National Reports: Kieran Mooney / CBD Secretariat
Dashboard: Tim Hirsch

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TARGET 9: INVASIVE ALIEN SPECIES

By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment

PREFACE

This chapter focuses on invasive alien species pressure, state and response from current, short-term and long-term perspective. "Alien species" refers to a species, subspecies or lower taxon, introduced outside its natural past or present distribution; it includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce. "Invasive alien species" means an alien species whose introduction, establishment and spread threatens biological diversity. This chapter also highlights past, current and future drivers of biological invasions around the world, and explores actions and policies that should be implemented to achieve the Aichi Biodiversity Target 9 in 2020.

9.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

9.1.1 Status and trends

9.1.1.i Trends in introduction of invasive alien species

Globally, there is no improvement regarding the pressure of invasive alien species since the previous Global Biodiversity Outlook report (i.e., GBO3, 2010). Biological invasions are continuing at an unprecedented rate (Figure 9.1). The number of alien species in Europe (i.e., metazoans in the Mediterranean, freshwater animals, and mammals) increased by +76% between 1970-2007 (Waage *et al*., 2008; Hulme 2009; Butchart *et al*., 2010). The numbers of alien species have also been increasing in China (Xu *et al*., 2012), and North America (Levine & D'Antonio 2003; Aukema *et al*., 2010).

Overall, considerable efforts have been made to define, characterise and identify species at different stages of the introduction-naturalisation-invasion continuum (Catford *et al*., 2012). More specifically, work is on-going in the development of global indicators determining trends in the numbers of introduced and invasive alien species to assess progress made towards the achievement of Target 9 (Pagad *et al*., 2014). The indicator measures the trends of invasive alien species (IAS) of 21 countries, which were selected for having at least 30 records of species with known invasion date (Figure 9.1). The indicator based on 3,914 invasive alien species shows that in 2012, countries (N=21) hosted on average 1.7 times more invasive alien species than they did in 1970 (Figure 9.1; Pagad *et al*., 2014). The average annual increase declined from 1.72 during 1970-1990 to 0.82 during 1990-2012. However, this less pronounced recent increase might reflect incomplete data coverage for the most recent years. Islands showed a threefold increase of introduced species since 1900, while continental countries showed a sevenfold increase. This trend might be the result of islands having been exposed to large numbers of alien species introductions before 1900, and thus recent increases in alien species may be less pronounced than in continental countries. However, the number of invasive alien species may be significantly underestimated, as lack of knowledge on impacts of invasive alien species is widespread (Scalera *et al*., 2012). For example, Norway has listed more than 217 invasive alien species that have a severe or high impact listed (Gederaas, *et al*., 2012). In the USA, UK, Australia, South-Africa, India and Brazil, there are approximately 120,000 alien species (Pimentel *et al*., 2005). It is inherently difficult to know how many of these alien species will become invasive in the future, and when, because there is generally a time lag of several decades between introduction and establishment of most invasive alien species (Essl *et al*., 2011). Half of the alien vertebrate species introduced to Europe and North America did establish themselves (Jeschke & Strayer 2005). The rate of successful invasions for plants is usually considered lower, as the "tens rule" holds that approximately 10% of introduced alien species will become established, and approximately 10% of these species will become invasive (Williamson, 1996).

Since the publication of GBO3, the responses to control invasive alien species have significantly increased. The global trend in policy response has been positive for the last few decades. As reported in 2010, 55% of the countries signatory to the Convention on Biological Diversity have enacted invasive alien species relevant national legislation, and 82% of these countries have signed multinational agreements (international conventions, organization agreements and organization guidelines) relevant to preventing the spread and promoting the control/eradication of invasive alien species (McGeoch *et al*., 2010a). Among these countries, 8% are signatory to all ten international agreements (McGeoch *et al.*, 2010 and Figure 9.1). For example, the Council of Europe has been developing and adopting codes of conduct addressing key pathways of introduction of invasive alien species (e.g., horticulture, botanic gardens, zoos, hunting, or fishing). Once the European regulation on invasive alien species is fully adopted, it will have major implications for neighbouring countries, and also at the global scale since the European institution is a major partner for global trade. The recent implementation of regulations regarding plant imports might also offer an effective control of alien species introductions (NAPPRA: Not Authorized Pending Pest Risk Analysis, 2012). However, it is not yet possible to assess the success of such measures.

The current increase of policy tools has not yet led to a significant reduction in the number of alien species introductions (Butchart *et al.*, 2010; McGeoch *et al.*, 2010a). This indicates inadequate implementation of adopted policies, lack of coherence between policies, or lack of adapted policies with regards to biological invasions. For example, about 28% of infestations (pest invasions) over the United States were detected under standard inspection procedures (Liebhold *et al.*, 2012). This study highlights that present regulations need to be more stringently implemented. Overall, policy responses cannot be equated with management effectiveness and no studies have evaluated at a global scale the effectiveness of the different policies measures.

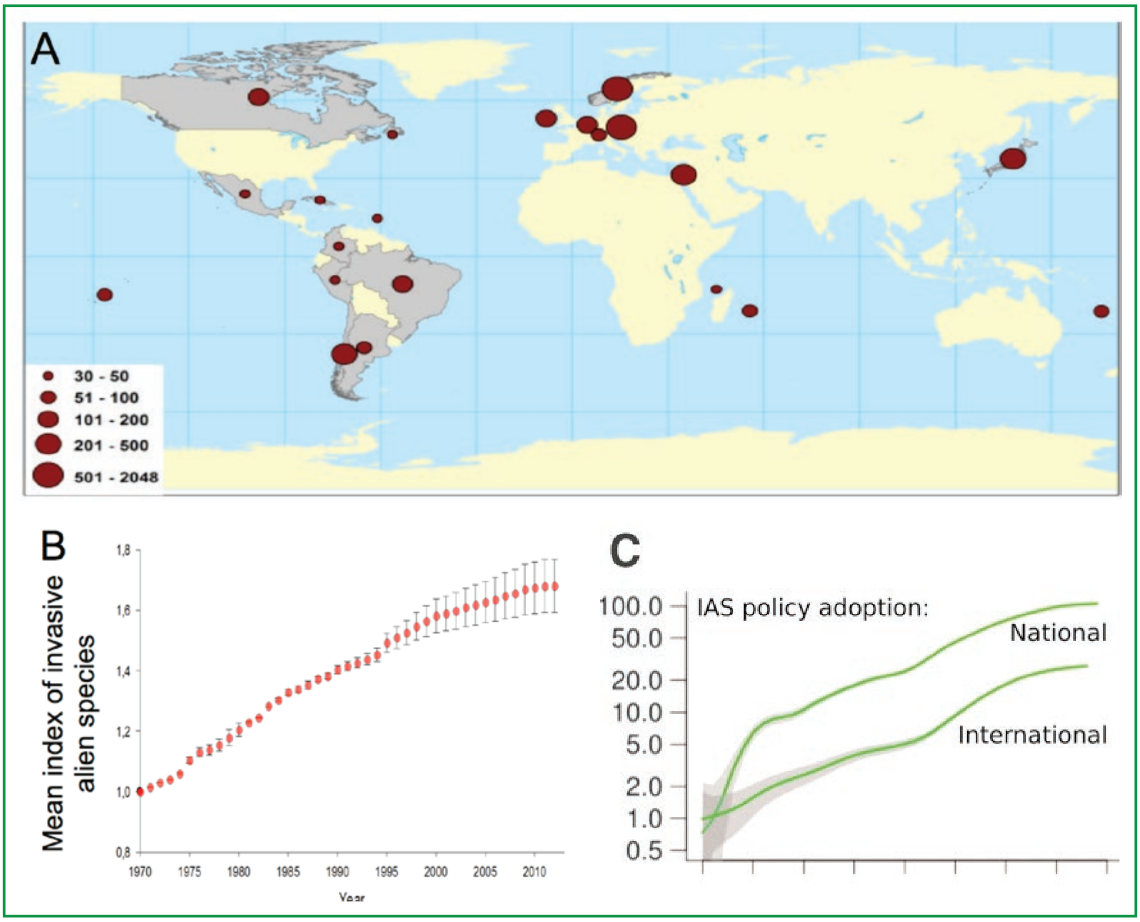


Figure 9.1. (A) Trend indicator showing the number of invasive alien species across 21 selected countries with known introduction date. The indicator was based on 3,914 invasive alien species and 4,903 species-country records. While all taxonomic groups were considered, the majority of the records are plants, invertebrates, fishes, mammals, and birds. (B) Trend indicator showing the geometric mean of the cumulative number of invasive alien species across 21 selected countries. Data scaled to 1 in 1970. (C) Indicators trends for invasive alien species policy adoption by countries responses to address its loss. Data scaled to 1 in 1970; modelled and plotted on a logarithmic ordinate axis. Shading shows 95% confidence intervals. Source: Butchart *et al.* (2010a)

9.1.1.ii Consequences for biodiversity

The Millennium Ecosystem Assessment (2005) recognizes invasive alien species as the second major driver of biodiversity loss after habitat loss. Invasive alien species through several mechanisms such as predation, hybridisation, competition, disease transmission cause negative impacts for biodiversity at gene, species, and ecosystems levels. Of the 170 animal extinctions for which we know the causes of extinction (there are 680 known animal extinctions), 54% included the impacts of invasive alien species, and for 20% of extinctions, invasive alien species were the only cited cause (Clavero & García-Berthou, 2005). These species extinctions have mainly occurred in insular ecosystems where native species are not adapted to new predators or competitors (Whittaker & Fernandez-Palacios, 2007). Invasive rats and cats have been a primary cause of species extinction over the last 500 years especially in islands ecosystems (Donlan & Wilcox, 2008). Furthermore, of 395 European native species listed as "Critically Endangered" by The IUCN Red List of Threatened Species, 100 are in danger because of invasive alien species (IUCN, 2011; Scalera *et al.*, 2012). Invasive alien species are a major threat to biodiversity, with, for example, 33%, 6% and 11% of threatened birds, mammals and amphibians respectively being threatened by invasive alien species. The IUCN Red List Index for birds considering trends driven by invasive alien species shows that, although some species have been down-listed to lower categories of extinction risk owing to successful measures to control or eradicate invasive alien species, these are outweighed by the number of species being up-listed to higher categories of extinction risk owing to increasing threats from invasive alien species (Figure 9.2). For amphibians, invasive alien species are by far the most important driver of extinctions (McGeoch *et al.*, 2010a). For example, the invasive alien Chytrid fungus (*Batrachochytrium dendrobatidis*) continues to spread, and threatens amphibians globally (Olson *et al.*, 2013). On the contrary, successful management of invasive alien species during the last several years has benefited a small subset of species that consequently have been down-listed to a lower category of threat of The IUCN Red List. Although less well documented, current impacts of invasive alien species also occur at ecosystem or gene levels (see Pejchar & Mooney, 2009 for examples). One such example is the Golden apple snail (*Pomacea canaliculata*) that has transformed wetlands across Southeast Asia from a clear water purification system to a turbid, algae-dominated state (Carlsson *et al.*, 2004), or the hybridization of the European honeybee with the far more aggressive Africanised honeybee in Latin America that is moving northward. Forests in North America have also been seriously impacted by invasive alien species (i.e., chestnut blight, Dutch elm disease, gypsy moth, emerald ash borer, etc …) to an extent that major elements of the forest biome have now disappeared or have been drastically reduced from their historical level of ecosystem presence and function (Poland & Mccullough, 2005). The Japanese ecosystems are also prominently impacted by the effects of invasive alien species, especially in recent years. In general, the impacts of invasive alien species on biodiversity are influenced and exacerbated by biological resource use (harvesting), human disturbance, and habitat loss (Berglund *et al.*, 2012).

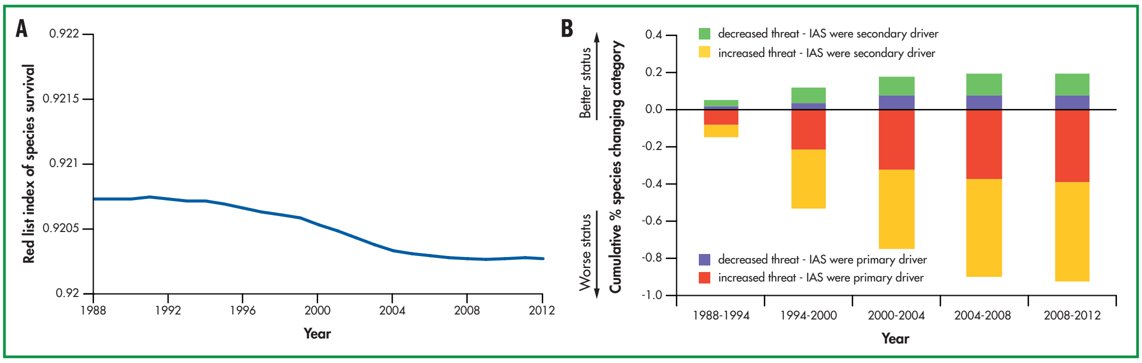


Figure 9.2. A) IUCN Red List Index for the world's birds (9,869 species) showing trends in status driven by the impacts of invasive alien species or their control; declines indicate that despite some notably successful local eradication and control efforts, invasive alien species are driving birds ever closer towards extinction. B) Percentage of 9,869 bird species that changed IUCN Red List category between 1988 and 2012 owing to impacts of invasive alien species or their control, distinguishing those for which invasive alien species were the primary or secondary driver of the change in status. Source: Figures from McGeoch *et al*. (2010a) and Butchart *et al*. (2010b), both updated by S. Butchart January 2014.

9.1.1.iii Current situation of actions against invasive alien species

The guiding principles on invasive alien species adopted by the Conferences of Parties to the Convention on Biological Diversity (2002) clearly indicate that prevention is the priority response; early detection, rapid response and possible eradication should follow when prevention fails. This is because prevention and rapid responses are generally more effective than long-term management (Simberloff *et al*., 2012).

There has been progress in the development of guidance relevant to invasive alien species. For instance, International Plant Protection Convention (IPPC) has included invasive aquatic plant species in the setting of international standards for phytosanitary measures. The World Organisation for Animal Health (OIE) is developing guidelines for assessing the risk of non-native animals becoming invasive. The Food and Agriculture Organization of the United Nations (FAO) is developing guidance on the implementation of phytosanitary standards in forestry as well as applying risk analysis in aquaculture. Regional Initiatives have also been encouraging (e.g., 22 Pacific countries have developed biosecurity legislation within a common biosecurity authority). The European Plant Protection Organization has also implemented standards and recommendations for risk assessment of pests and invasive alien plants in Europe that are followed by its members and the Mediterranean region (EPPO 2014).

National programmes of preventive measures have been adopted to implement a number of fundamental tools; e.g., New Zealand with a biosecurity system, Australia with risk analysis, or Brazil with national survey and funding mechanisms (see for example Australian Government, 2011). Among the different tools to prevent invasions, the Australian Weed Risk Assessment system (A-WRA) is one of the most effective risk assessment tools used. Overall, the WRA rejects an average of 80% of weeds (Weber *et al*., 2009), and 90% of major invaders are correctly identified. Another example in Florida showed that about 92% of test species that have been documented to be invasive were correctly rejected using the WRA system (Gordon *et al*., 2008). The WRA system is also successfully used in New Zealand and Japan (Nishida *et al*., 2008). Consequently, the number of introduced alien mammals in New Zealand has stabilized during the last few years (Box 9.2). The WRA have also been modified to be used for assessing the potential invasion of other taxa such as freshwater fish, marine fish, freshwater invertebrates, marine invertebrates, amphibians (Copp *et al*., 2005), vertebrates (Bomford 2003; 2008), and aquatic invertebrates (Tricarico *et al*., 2010). However, important gaps remain regarding risk assessments (see Kumschick *et al*., 2013 for details). Fewer risk assessments have been developed for vertebrates or invertebrates (except for ants see Ward *et al*., 2008) as compared plants (Kumschick & Richardson, 2013), and these models were mainly developed for Australia and New Zealand (Bomford 2003, 2008; Massam *et al*., 2010). However, several plans are currently under development within North America (e.g., within the U.S. Fish and Wildlife Service and USDA Wildlife Services, Canada's Centre of Expertise for Aquatic Risk Assessment) and for terrestrial vertebrates in Europe (e.g., generic impact-scoring system to alien mammals, see Nentwig *et al*., 2010). For example, Great Britain has implemented a framework strategy against invasive non-native species (GB Non-native Species Secretariat 2008). The World Animal Health Organisation has also developed guidelines to assess the risk of non-native animals becoming invasive (OIE 2013). Recently, Blackburn and colleagues (2014) have proposed a new method to evaluate, compare, and eventually predict the magnitudes of the different impact of alien species. Ultimately, this should help existing practices and future policies of risk assessments in many regions (Blackburn *et al*., 2014).

There are significant improvements to document the major pathways of invasions since GBO3. Civil aviation and shipping pathways are starting to be studied at a global scale (Drake & Lodge, 2004; Tatem & Hay, 2007; Seebens *et al*., 2013). However, a global scale pathway management tool does not yet exist, although the International Civil Aviation Organization and the International Plant Protection Convention have attempted to develop international guidance for these pathways. Seebens *et al*. (2013) identify high-risk global marine invasion routes, offering an interesting perspective for the development of effective, and targeted bio-invasion management strategies. More specifically, air-travel has been used to study the connectivity that exists across the malaria-endemic world, to provide a first assessment of the infection risks resulting from movement of infections (Huang & Tatem, 2013). Identification of important pathways such as horticulture and ornamental plants for specific regions (i.e., Europe) has also been achieved (e.g., Hulme 2009; DAISIE, 2009).

Beyond the identification of major pathways of invasions, there are a number of past and on-going efforts to incorporate them in future policies (e.g., the pet trade). The European Union has also adopted a target in Biodiversity Strategy (EC 2011) and its coming legislation on invasive alien species (EC 2014) that states that its members have to identify and prioritize pathways of introduction and to adopt action plants to manage these pathways.

Eradication campaigns events, especially for vertebrates, have resulted in many important successes, particularly in island ecosystems. Over 1,600 alien vertebrate species eradications have been undertaken on islands worldwide, with 1,128 confirmed as successful compared to 173 failures (an 87% success rate), (DIISE, 2014; Figure 9.3). Historically, the number of eradication campaigns has increased over time, although the last 10 years revealed a significant decrease of the eradication events (Fig. 9.3). Overall, there have been very few successful eradication programmes on mainland (Baker, 2010). However, New Zealand eradication practitioners have demonstrated significant innovation for the eradication of invasive vertebrates from islands, with only very small islands being undertaken 25 years ago. For example, growing to rat eradication from the relatively large 113 km^2 Campbell Island in 2002 (Broome, 2009), eight species simultaneously removed from Rangitoto and Motutapu near the city of Auckland in 2011 (Griffiths, 2011; see also Glen *et al*., 2013), and the eradication of invasive animals has been assessed from the human inhabited 1,746 km^2 Stewart Island (Bevan, 2008). Human inhabited islands represent a new challenge and frontier for eradicating invasive animals (Oppel *et al*., 2011; Glen *et al*., 2013). In parallel, costs of eradication has decreased over the years (Carrion *et al*., 2011): over 160,000 goats were removed from over 500,000 ha in less than five years for only US$18 per ha (Donlan *et al*., submitted; Carrion *et al*., 2011).

Overall, design of eradication programmes has significantly improved, with a growing number of multispecies programmes, and prevention of non-target effects such as primary or secondary poisoning of non-target species or predators (Zavaleta *et al*., 2001; Jolley *et al*., 2012). Moreover, successful eradication programmes have allowed the recovery of native biodiversity in many cases (e.g., Courchamp *et al*., 2011; Kessler & Service, 2011; Whitworth *et al*., 2013). For example, Fukasawa *et al*. (2013) and Watari *et al*. (2013) showed remarkable recovery of various native vertebrates during eradication of invasive predator (small Asian mongoose) in Amami Island (712km^2), Japan. Local communities have also undertaken efforts to control and eradicate invasive alien species though traditional stewardship and management practices (Kothari *et al*., 2012). For example, in India's Biligiri Rangaswamy Temple Sanctuary and Tiger Reserve, Soliga people have reclaimed community rights to the forests and are preparing a management plan that includes traditional and new methods of controlling invasive alien species such as *Lantana*, which the conventional governmental management has failed to control. Thanks to all these eradication programmes, the overall risk of extinction has been substantially reduced for 11 bird species, five mammal species, and one amphibian. However, the number of species whose conservation status has improved is out-weighted by species with deteriorating status due to the pressure posed by invasive alien species (McGeoch *et al*., 2010b). Nevertheless, progress has allowed control or eradication of species from ecosystems that were until recently deemed too large or too complex (Simberloff *et al*., 2012). This suggests that eradications of species or populations that are currently considered unfeasible might become feasible in the near future (Carrion *et al*., 2011).

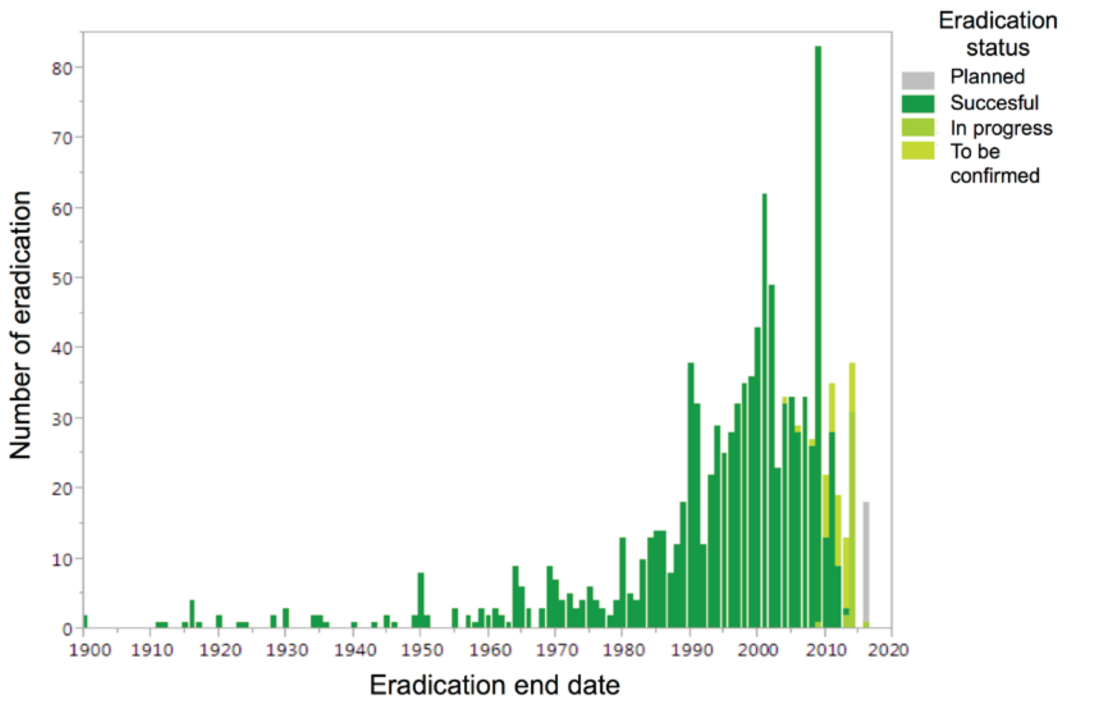


Figure 9.3. Number of eradications of invasive vertebrates that are planned, successful, in progress and to be confirmed on islands since 1900. Excludes reinvasions, failures and eradications with unknown end dates Source: Figure from DIISE (2014).

A few years ago, the vast majority of eradication projects were not initially based on a systematic prioritization; however, eradication has been starting to take a more regional and national approach, calling for prioritisation schemes. The application of quantitative method to evaluate eradication feasibility is also a significant improvement in the design of eradication programmes (Ramsey *et al*., 2009; Fukasawa *et al*., 2013a). Consequently, invasive alien species eradications on islands have been prioritized according to the native biodiversity threatened, eradication feasibility, economic cost, and reinvasion potential (e.g., Brooke *et al*., 2007; Donlan & Wilcoxon, 2009; Capizzi *et al*., 2010; Harris *et al*., 2011). Prioritization of eradication programmes is currently a dynamic field of research. Adopting a return on economic investment approach to guide mammal eradication is only beginning (e.g., Donlan *et al*., submitted), and integration of climate change issues into prioritization of eradication programs have also been discussed recently (Runting *et al*., 2013; Courchamp *et al*., 2014).

In parallel to this progress on invasive species management, the recent literature has been critical about invasion biology as a scientific discipline and about the relevance of management of invasive alien species. For example, there are some recent claims that the native/non-native dichotomy, and human vs. non-human transport origin has only modest scientific value (Davis *et al*., 2011; Valéry *et al*., 2013 but see Richardson & Ricciardi, 2013; Blondel *et al*., 2013; Simberloff & Vitule, 2013; Shah & Uma Shaanker, 2014). These arguments may become more prominent as the impact of climate change on native and invasive alien species increases resulting in a redefinition of invasive alien species (Engel *et al*., 2011). It has also been suggested that most introductions are benign and thus do not merit management that are costly (Hasselman *et al*., 2012; Thomas, 2013). However, in practice, as managers are limited by availability of resources, they already tend to prioritize the most problematic invasive alien species (Richardson & Ricciardi, 2013).

9.1.2 Projecting forward to 2020

In the short-term, the threat from plants and mammals invasions to mammals, birds, and amphibians, is unlikely to diminish in most parts of the world (for Europe, see Hulme *et al*., 2009) as well as the damage caused per species to biodiversity and society. In addition, the pressure caused by invasive alien species is likely to increase over the next decade if significant actions are not implemented rapidly (Figure 9.4). Extrapolations of cumulative introduction events over Europe suggest that the number of invasive species will continue to increase by 2020 if there is no significant change in the key drivers of invasion (Figure 9.4). This increasing trend is likely to be accentuated in the near future at a global scale, as trade between climatically and environmentally similar regions are predicted to increase and habitat continues to be disturbed. Although there are examples where the number of introduced non-native mammals has been stabilized or slightly decreased over the last decades thanks to the strong and innovative policies against invasive alien species (see Figure 9.2 for further details), there is no indicator of the effectiveness of such policies to fight biological invasions.

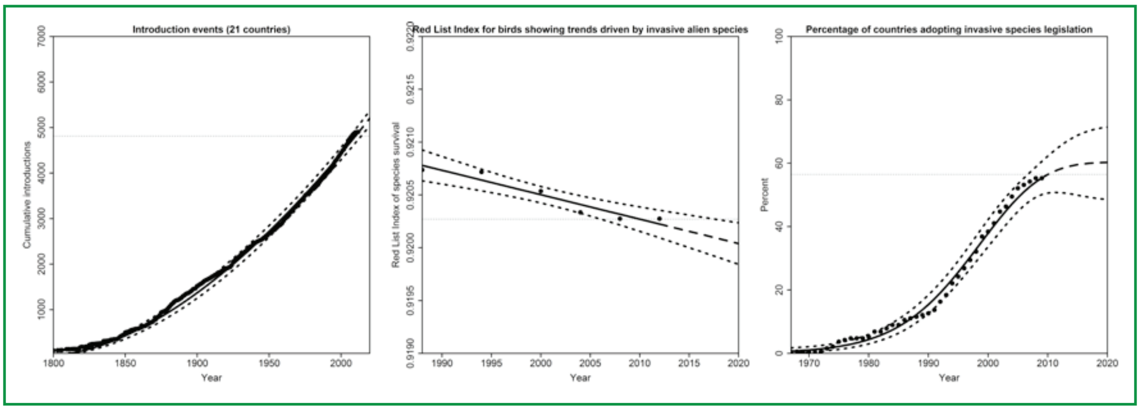


Figure 9.4. Statistical extrapolation of A) cumulative introduction events in 21 selected countries, B) Trends in the Red List Index for birds that can be attributed to invasive species and C) Percentage of countries adopting invasive alien species legislation to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Source: Data for A) from from Pagad *et al*. (2014) and the countries are indicated in Fig. 9.1; data for B) based on Butchart *et al*. (2010a), updated by Butchart in 2014. Data for C) from Butchart *et al*. (2010a).

The extrapolation of the current trend of adoption of national and international policies against invasive alien species shows a promising short-term perspective (Figure 9.4). This extrapolation is consistent with countries commitments for which NBSAPs reports are available. For example, the European Biodiversity Strategy's Target 9 on Invasive Alien Species provides the focus for the EU countries in their work in identifying and prioritizing priority invasive alien species and managing their pathways of introduction. With the adoption of a new EU regulation on invasive alien species in 2015, the 28 member states of the European Union will commit themselves to preventing the introduction and spread of the worst invasive alien species and cooperating in managing pathways of introduction to minimize introductions of these species and prevent damage to biological diversity (EC, 2011, EC, 2014). A number of European countries such as Belgium, Norway, Finland, Sweden, Denmark, Austria, and Switzerland, have identified a suite of actions, such as conducting comprehensive and widely accepted risk assessment procedures, developing actions for addressing main introduction pathways, and establishing early detection and control mechanisms, in order to implement their operational objective related to invasive alien species. If these policies are adequately and promptly enforced, there is a reasonable confidence that it will significantly help to achieve the Target for these countries. Ultimately, the effect that the Target will have will mainly depend on the extent to which actions are taken to implement them (Section 9.1.3 provides additional information on countries commitments in their NBSAPs).

The successful establishment of invasive alien species as a short-term perspective will depend on many factors. Overall, characteristics that define invasive potential include both intrinsic factors (e.g., species traits) and extrinsic factors such as international trade and habitat degradation. Identifying species traits that could be related to the likelihood of invasions has been an important field of invasion biology for over two decades now, with still relatively few general results; plants and pests of agricultural concern have been studied to determine the likelihood of establishment, spread and negative effects on production and biological diversity. For example, establishment of invasive plant species is often related to a large native range size, presence of clonal organs, vigorous vegetation growth, early and extended flowering, occupation of disturbed habitats, early time of introduction and attractiveness to humans (Cadotte *et al*., 2006; Pysek & Richardson, 2007). The relative importance of different traits is environment-dependant and changes over time. In addition, factors determining success concerning establishment and invasion also differ between taxa and are context-specific (Richardson & Pyšek, 2012; Ricciardi *et al*., 2013). Some invasive plant species traits (e.g., life form, stature or pollination syndrome) could also help to predict impact (Pyšek *et al*., 2012a).

Overall, human-related processes (the number of alien species introduced and human population size, land use, and infrastructure) drive the diversity of introduced species (e.g., exotic birds in Europe, Chiron *et al*., 2009, or marine species, Gallardo & Aldridge, 2013). In the Czech Republic, Chytrý *et al.* (2008) have also shown that the major determinants of the level of invasion by alien plant species were related primarily to habitat properties, followed by climate, and propagule pressure (Chytrý *et al*., 2008). Propagule pressure is a key element mediating establishment success for some species such as insects, some plants, or plankton, but less so for other large species such as mammalian predators. Consequently, increases in the number of introductions and magnitude of spread of alien species is also strongly associated with substantial increases in the extent and volume of trade and transport, particularly over the last 25 years (Levine & D'Antonio, 2003; Hulme, 2009). European maritime transport is predicted to rise from 3.8 billion tonnes in 2006 to 5.3 billion tonnes in 2018 (Scalera *et al*., 2012). The risk of biological invasions in marine environment caused by global shipping with discharge of ballast waters and hull fouling organisms is also increasing globally (Seebens *et al*., 2013). In the short-term, faster modes of transport may also increase the opportunities for an organism to survive in transit and establish in new environments (Ruiz & Carlton, 2003; Burgiel *et al*., 2006). The development of new trade routes has already led to the introduction of new alien species either deliberately or accidentally, while the growth in the volume or trade along those routes has increased the frequency with which introductions are repeated. An important short-term trend is also the development of regional trade agreements that may increase the risks of biological invasions if they are not implemented with strong policies against invasive alien species. About 420 regional trade agreements have been signed by the end of 2008 (Perrings *et al*., 2010). The Treaty on Free Trade between Colombia, Venezuela and Mexico has for example facilitated the introduction of exotic fish into Mexico (Mendoza Alfaro *et al*., 2010). Consequently, if regional trade agreements are not implemented with transparency and sharing of information and prevention measures, they might increase the risk of biological invasions.

There is also evidence that increasing establishment rates of invasive alien species can be attributed to an increase in degraded habitats at least for some types of habitats (Johnson *et al*., 2008; Spear *et al*., 2013). Habitat degradation will therefore be among the most important drivers related to invasions in the short term. For example, the highest levels of alien plant invasions were projected for arable land, urban areas, and abandoned land in Europe (Chytrý *et al*., 2012). In parallel, increases in human population density will also be an important driver of biological invasions by adding pressure on natural areas as more land will be needed for food production, but also by generating more intentional releases of exotic plants and pets. At finer spatial scales, historic and contemporary land use, as well as economic benefits of the species play also a major role in the dispersal, distribution and establishment of invasive alien species (Mattingly & Orrock, 2013).

Fast economic development (i.e., Gross Domestic Production) has also been demonstrated to accelerate biological invasions in China (Lin *et al*., 2007), which might mean there are similar trends in other economically emerging countries. For example, higher income could lead to buy pets or ornamental plants that increase the potential risk of invasions. Between 1970 and 2004, the ratio of world trade to global Gross Domestic Production has increased from around 13 to 29%.

Climate change has also started to affect the survival, establishment, spread, distribution and impact of alien species throughout the world (See Walther *et al*., 2009 for many examples). One such example is the pine processionary moth, *Thaumetopoea pityocampa*, a major forest pest from the Mediterranean Basin that is rapidly expanding its range towards higher latitudes and altitudes in response to climate change (Battisti, 2005, 2006). The Lessepsian migration, the dispersal of at least three hundred species from the Red Sea into the Mediterranean Sea following the opening of the Suez Canal, is one such example of invasion that has been exacerbated by climate change (Raitsos *et al*., 2010). This new context calls for a better integration of climate change and its interaction with the existence of major infrastructures into alien species predictive studies (see below).

Box 9.1: Are we on track to achieve Target 9 by 2020?

Invasive alien species are identified – Many alien species have been introduced globally. Among them, many are inoffensive for biological diversity or have little economic impact, and some have a large economic or social value in their areas of introduction (*e.g*., domesticated plants; aquaculture species). However, a significant proportion of alien species become invasive, and those need to be identified and characterised. Considerable efforts are on-going to identify and characterise invasive alien species (see GRIS initiative; Blackburn *et al*., 2014). There is also an urgent need to standardise terminology related to invasive alien species. Some issues about time lag between introduction of alien species and impacts require more attention. We are on track to achieve the target on this sub-objective, although important issues remain on a standardized definition and on gaps in the current coverage, which are mainly developed countries and terrestrial species.

Invasion pathways are identified – The control of individual invasive alien species is time and resource consuming, and cannot be successful for the control of all invasives because there is a huge number of invasive alien species that can be introduced anywhere. It is now widely accepted that it is far more effective to identify invasion pathways and implement measures to manage them (e.g., control of ballast water or hull fouling organisms, inspection of horticultural products, regulation of the nursery, aquarium, and pet trade to prohibit the trade of invasive alien plants/pets). There are good examples showing how identification of pathways could improve efficacy of prevention at a global scale (e.g., see Briski *et al*., 2012, Katsanevakis *et al*., 2013 or Seebens *et al*., 2013 for global shipping or Tatem & Hay, 2007 for airline transportation network). Working at the invasion level can also exclude entire guilds of invasive alien species (e.g., wood boring insects or marine macro-invertebrates). However, the link between pathways and invasive alien species success and impact remain unknown.

Invasive alien species and pathways are prioritized – Given the sheer number of alien species, and the relatively large proportion of them that can become invasive (sometimes after a time lag of several decades), prioritizations are usually made on invasive alien species known to have a large impact elsewhere, or on invasion pathways that are known to be important sources of invasive alien species. There are significant improvements to document the major pathways of invasions and to consider them into future policies. National programmes of preventive measures have been increasingly adopted (e.g., Australia, New Zealand, Norway, Belgium, Ireland, and increasingly in Caribbean and Pacific Island nations). However, the high numbers of invasive alien species and the cost of implementing stringent biosecurity measures have hindered efficient legislation in many countries, especially in countries that are not efficiently protected by physical barriers to plant and animal dispersion. In addition, there is an absence of spatial prioritization of the border control that could be based on sampling efforts (e.g., in Europe; Bacon *et al*., 2012).

Box 9.1: Are we on track to achieve Target 9 by 2020? *continued*

Priority species are controlled or eradicated - Control options including eradication to manage invasive alien species has significantly grown over time including for species and ecosystems that were still until recently deemed impossible to eradicate (see Simberloff *et al*., 2012 for examples). Novel criteria to control or eradicate species are also considered in prioritization schemes, such as conservation value, feasibility, durability, and cost effectiveness (Brooke *et al*., 2007; Donlan & Wilcoxon, 2009; Capizzi *et al*., 2010; Harris *et al*., 2011), but other factors such as climate change exposure need also to be included (Courchamp *et al*., 2014). There are now numerous examples of conservation successes resulting from control or eradication of invasive alien species (McGeoch *et al*., 2010). However, the increase in the number of new alien species introductions out-weighted the number of eradications, pointing towards a need for further efforts in measures to prevent introduction and establishment. In addition, the number of completed eradication programmes seems to have decreased in the last ten years, compared to an exponential increase in the previous decades. This might in part be due to a tendency to tackle logistically more challenging ecosystems and/or multi-species eradications in single projects. The recent success of the Macquarie Island Pest Eradication project is a good illustration of this. Consequently, it is likely given current trends that this sub-objective of the target will be missed, although some significant progresses have been made to control and eradicate invasive alien species.

Measures are in place to prevent their introduction and establishment. - The number of established invasive alien species has significantly increased in all taxonomic groups, with no signs of slowing down (mainly developed countries for which data are available). The increasing establishment rates of invasive alien species are widely attributed to increased rate of species introductions due to increasing international trade and human density. To date, there is an encouraging increase in the adoption of national and international conventions and agreements, regulations and codes of conduct to prevent introduction, establishment, and spread of invasive alien species. Yet, there still exists a gap between international agreements, regulations and measures that are implemented at the national levels. Consequently, current adopted policies and their implementation remain insufficient to achieve this sub-objective of the target.

Although notable progresses have been performed in some areas, achievement of all of the sub-objectives of Target 9 is not likely given current trends.

9.1.3 Country actions and commitments[1]

Almost all countries have identified targets or actions related to invasive alien species, however few Parties have developed quantitative targets (high confidence). Generally the targets or actions that have been established address the main elements of Aichi Biodiversity Target 9 (high confidence). For example Suriname has set a sub-objective of limiting the spread of dangerous organisms. Finland and Japan have also a set of targets to identify invasive alien species and their pathways and to prioritize these.

The targets and actions that have been set have an emphasis on controlling introduction pathways. Communication and raising awareness on invasive alien species is also another key issue that has been reflected in a number of new initiatives implemented by several countries. Among the actions, generally there appears to be less of an emphasis on controlling invasive alien species or identifying species and pathways (moderate confidence). The ultimate effect of the targets will depend on the extent to which actions are taken to implement them. In this light, several countries have identified priority actions to implement their targets, and are building on existing legislation or programmes or have plans to develop new legislation (moderate confidence). For example the Dominican Republic plans to further strengthen its Program for Control of Invasive Alien Species. Similarly England has identified, as a priority action, the continued implementation of the Invasive Non-Native Species Framework Strategy for Great Britain. Belgium has also identified a suite of actions, such as conducting comprehensive and widely accepted risk assessment procedures, developing actions for addressing main introduction pathways, and establishing early detection and control mechanisms. Many countries, for example East Timor and Malta, have also identified or noted indicators in their NBSAPs, which can be used to monitor progress towards their targets or actions or have identified desired outcomes (moderate confidence).

Footnote

[1] This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.

The importance of minimizing the impact of invasive alien species to make progress towards other national priorities, in particular the reduction of extinction, is noted in many NBSAPs, such as Switzerland (moderate). Several countries have also noted the link between the identification of invasive alien species and monitoring systems more generally (low).

Finally, if actions, which have been identified by Parties in their NBSAPs, are fully implemented they would necessarily bring the world community closer to attaining the Aichi Biodiversity Target 9. The importance of minimizing the impact of invasive alien species to make progress towards other national priorities, in particular the reduction of extinction species, is also acknowledged by many countries.

9.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

9.2.1 Actions

To achieve Target 9 - by 2020, *invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment* – multiple measures should be implemented.

There is a need to develop indicators of invasions that track any progress towards achieving the targets (Rabitsch *et al.*, 2012). These indicators can be based on a range of taxa, cover large spatial scales, assess temporal trends in invasions, or consider impacts of invasive species as well as develop large dataset about invasive alien species (but see McGeoch *et al.*, 2010; Nentwig *et al.*, 2010; Pyšek *et al.*, 2012b). For example, the levels of biological invasion across ecosystems could be calculated through relative alien species richness and relative alien species abundance (Catford *et al.*, 2012). Indicators have already been developed such as the number of invasive alien species per country but the available data does not have global coverage, in particular in developing countries (see section: "Uncertainty and data requirements"). In this context, citizen participatory monitoring can provide high quality data for assessment of invasion (Kadoya *et al.*, 2009). Data on aquatic invasive species is also currently lacking for current trends of invasions in inland waters, which limits extrapolation (but see Ricciardi, 2006). Moreover, non-standardised terminology of invasive species is still a significant limit to the development of pertinent indicators. An internationally standardised procedure as for The IUCN Red List might be an option to solve this issue (see for example Blackburn *et al.*, 2014). Although adoption of national and international policies against invasive alien species is an important indicator of responses, this indicator fails to inform about the efficiency of such responses. Additional indicators such as the economic costs of invasive species, their impacts on ecosystem services or human health should also be developed to assess the achievement of this target (Genovesi *et al.*, 2013). Hence, success of management is critically dependent on adequate information and on understanding of the pathway, size and nature of biological invasion (McGeoch *et al.*, 2010), but also on the knowledge of adequate control measures that have proven to be successful. Further facilitation of global information sharing and centralising that assist recognition of the risk of biological invasions and analysis of these risks, is needed. In order to collate and centralise the data, a number of new initiatives have been initiated to provide scientists, environmental managers, policy-makers and others with information databases and discussion forums. For example, the Global Invasive Alien Species Information Partnership (GIASI Partnership) has come together in order to assist Parties to the Convention on Biological Diversity, and others, implement Article 8(h) and Target 9 of the Aichi Biodiversity Targets, building upon databases developed by the IUCN SSS Invasive Species Specialist Group (IUCN-SSG), CABI Invasive Species Compendium, DAISIE among others.

The multiple pathways of introduction and the huge volume of trades call for prioritization of prevention efforts that should be focused on key pathways. The data collected so far —mainly for developed countries — can already permit to identify some key activities and vectors responsible for the past introduction of invasive species; for example, the commercial trade in ornamental plants has been identified as a major and often the primary pathway for the introduction and dissemination of terrestrial invasive alien plants and invertebrate pests; shipping through the release of ballast water is also the primary pathway for introductions of aquatic organisms (mainly invertebrates); pet trade is a key pathway of introduction of alien terrestrial vertebrates (Bacon *et al.*, 2013). The ballast water management convention has been adopted by 38 countries representing 30% of the world's merchant fleet in 2013. Japanese legislation on invasive species enacted in 2005, introduced a ban on the import of selected high risk invasive alien species and an authorisation process for a broader number of key invasive alien species, managed to significantly reduce the number of introductions of several taxonomic groups (Goka *et al.*, 2010). Ballast-water treatments have also been shown to reduce freshwater zooplankton concentration in ship tanks by 99% (Gray *et al.*, 2007). The introduction of invasive alien mammals has now been halted in New Zealand, through an effective biosecurity policy (Simberloff *et al*., 2012). Development of promising tool to quantify the volume of agricultural trade that should be inspected has also been recently developed for insects in European countries (Bacon *et al*., 2012). Furthermore, detailed country level information is still needed, especially for underdeveloped countries. Furthermore, expansion, standardization, and interoperability of databases is required. International collaboration is essential at this scale, especially because pathways prioritization should be defined and identified at the regional/national level.

Data on trends exist for alien species in Europe, but these are not for "priority" or "invasive" species *per se* and impacts of invasive alien species on extinction risk are only available for mammals, birds, fish, and amphibians. However, the European Parliament will be voting on a bill to draw up a blacklist to fight invasive alien species. In addition, the European Alien Species Information Network (EASIN) aims to facilitate the exploration of existing alien species information in Europe from distributed sources, and to assist the implementation of European policies on biological invasions. The need for an integrated approach in policy development at both the regional, national, and international levels was emphasised as some countries have policy against invasive alien species that only included alien species that are unwanted for the country (e.g., Solofa, 2009).

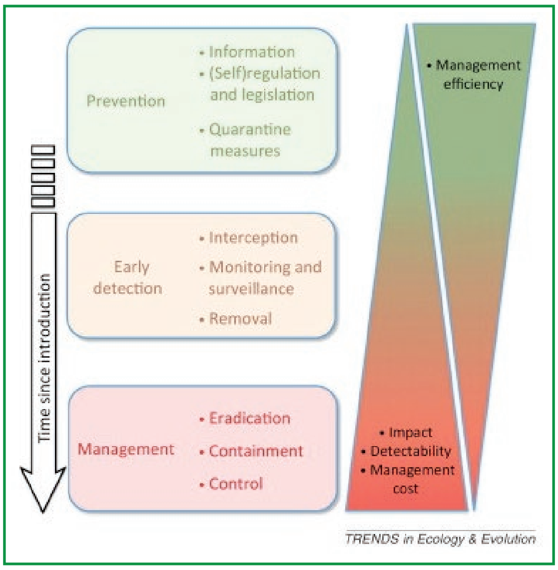


Figure 9.5. Management strategy against invasive species. The optimal strategy evolves with time-since introduction, with management efficiency decreasing and management costs increasing with time since introduction. Source: Simberloff *et al*. (2012).

The development of early detection and rapid response policies is by far the most cost-effective intervention in some cases reducing costs of intervention by over 40 times (Simberloff *et al*., 2012 and Fig. 9.5). For example, (Heikkilä & Peltola, 2004) determined the prevention costs of €350,000 and eradication costs of €946,000 for the Colorado potato beetle in Finland. However, when prevention measures have failed, decision support tools need to be in place for an efficient application of control measures at global, regional and national scales. The importance of the size of the infestation for eradication success suggests that eradication measures should concentrate on the early phase of the invasion when infestations are still relatively small (Pluess *et al*., 2012), as well as on the invasion front.

Finally, policies concerning invasive alien species have been increasingly adopted globally; however, there is a need to bridge the gap between growing scientific understanding of biological invasions, policies adaptation, and management action to ensure efficient measures against invasive alien species. For example, South Africa's national-scale strategy has failed to effectively control the extent of invasions, demonstrating the need to prioritize both the species and the areas (van Wilgen *et al*., 2012). In addition, setting policy frameworks such as eradication programmes that fail to consider climate change and sea-level rise issue can mean missing out on vital issues that are required to succeed as a long-term perspective (Mainka & Howard, 2010; Courchamp *et al*., 2014).

Box 9.2: Invasive species management in New Zealand

New Zealand is one of the most invaded countries in the world, primarily by virtue of the very high propagule pressure exerted upon it by intentional species introductions (Allen *et al*., 2006). This legacy of introductions and transformation by European colonists was intended to recreate a familiar landscape and lifestyle. Today, New Zealand is a country whose primary industry depends on alien species, but which has leveraged its isolation, as both an island nation and one very distant from major trading partners, to turn the tide on unwanted species invasions (Kriticos *et al*., 2005). New Zealand's strong policy of border protectionism originated from agriculture, both from a desire to prevent deleterious invasions of disease and other organisms, to protect local markets, and also to promote export of products from New Zealand that are considered as highly valuable regarding sanitary and phytosanitary concerns (Trampush, in press). New Zealand is also a country rich in endemic biodiversity, and the agricultural border protection measures put in place translated readily to conservation border protection, when the biodiversity impacts of invasive species were recognized.

Despite these excellent border protections, many alien species have been and continue to be introduced, and some of these become invasive. Recognizing the impacts of these invasive species on agricultural and biodiversity values, New Zealand has developed tools to respond to species invasion post-border (Wotton *et al*., 2004). The successful implementation of these tools in New Zealand to combat invasive species spreading has benefitted from the small size and comparatively horizontal governance structure of New Zealand agencies tasked with pest control. Two strong legal frameworks have been implemented in New Zealand: the Hazardous Substances and New Organisms Act and the Biosecurity Act.

For protection of biodiversity from invasive species impacts, New Zealand has focused on using islands as arks where threatened species can be reintroduced (McLean & Armstrong, 1995). New Zealand has also pioneered the development of methods to eradicate pests as invasive alien species particularly introduced mammals, from islands to increase the amount of pest-free land area (Towns *et al*., 2013). New Zealand has eradicated introduced mammals from over 100 islands.

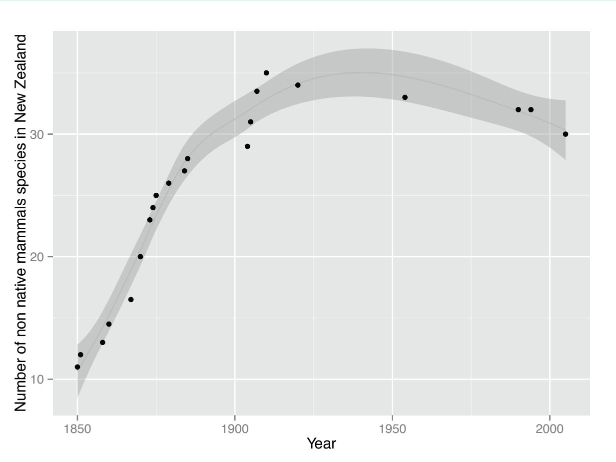


Figure 9.6. Number of non-native mammal species in New Zealand between 1876 and 2005 (points), Grey line represents the fit with Loess fit, and the shade represents the 95% confidence interval.
Source: Data compiled by M. Clout, P. Genovesi from Simberloff *et al*. (2012), updated by J. Russell.

Following its success on smaller islands, New Zealand has developed "mainland islands", which allow the technologies developed for invasive species eradications on islands to be applied in a larger landscape context. "Mainland islands" can utilize either novel barrier technologies, specifically mammal-proof fences, to create fenced enclosures within larger landscapes, or can use sustained pest control methods to maintain pest density at close to zero for agricultural or biodiversity benefits (Innes *et al*., 2012). Through spontaneous community driven processes there are currently over 25 fenced, and 100 unfenced "mainland islands", across New Zealand. By increasing pest control connectivity among these sites and expanding their "halo" of influence, it is predicted that pest control may scale to the entire country with appropriate governance guidance (Glen *et al*., 2013).

Box 9.2: Invasive species management in New Zealand *continued*

New Zealand has demonstrated that expediency in invasive species control (eradications) and prevention (implementation of biosecurity measures) is critical for successful invasive species management (Jay, 2003). Unnecessary delays in action due to uncertainty decrease the likelihood of successful invasive species management outcomes. As well as rolling back pest species distributions at key sites, New Zealand has developed comprehensive biosecurity protocols for surveillance and detection of incursions both pre-border and post-border at key sites. Implementation of these methods has demonstrated that the more rapid the intervention on new incursions, the greater the likelihood of success. Red imported fire ant have been detected at New Zealand ports on three occasions and successfully eradicated. The painted apple moth was detected in an inner city suburb and eradicated through an orchestrated eradication campaign. Reinvasion of islands previously cleared of invasive rats is close to zero where biosecurity surveillance occurs.

The success of invasive species prevention and management in New Zealand has depended on the buy-in of diverse groups (e.g., agricultural and tourism industries) and a strong awareness of invasive species in the wider population (Russell, in press). This awareness and buy-in has been crucial to overcome obstacles to successful invasive management, particularly issues relating to the controversial methods of invasive species control and eradication (e.g. toxin use). This assessment is also made through a strong legal framework for the introduction of hazardous species and new organisms.

New Zealand is still in the early stages of many species invasions, most invasive plants are only beginning their invasion, and even if no more invasive species colonised it, New Zealand will still have a legacy of invasion to combat for many decades to come. Source: Essl *et al.* (2011).

9.2.2 Costs and cost-benefit analysis

There are several benefits of investment in reducing the pressure of invasive alien species. As the cost of invasions is likely to rise as more species arrive each year and more species that are already present become invasive or more widespread, cost-benefit analyses in order to control invasive species is paramount. Meeting Target 9 would substantially reduce the total economic cost of damage caused by invasive alien species, which is roughly estimated at 2-5% of world Gross Domestic Product (GDP), or approximately US$2.6 to 6.5 trillion per annum (Pimentel *et al.*, 2005; High Level Panel Report). A more conservative estimate of damages is about 1.5% of GDP (High Level Panel Report). Current costs of invasive alien species include direct use costs in terms of extraction of resources from the ecosystem; indirect use costs (e.g., disruption of ecosystem services) that for example encompass the effects on pollination, fertilisation, seed dispersal or flood attenuation. More specifically, the estimated annual cost of alien invasive species has been estimated to be US$336 billion per year for USA, UK, Australia, South-Africa, India and Brazil, CA$29.2 billion for Canada and US$17.3 billion in Europe (Pimentel *et al.*, 2005; High Panel Level Report, EC 2011b +, Table 9.1). For example, exotic pest species caused annual losses of US$12.0 billion in Brazil (Oliveira *et al.*, 2013). It should be noted that alien species can be positive for agriculture as most food crops are deliberately introduced alien species; yet other invasive alien species can reduce crop yields by billions of dollars annually (Pejchar & Mooney, 2009). Moreover, climate change could increase the cost of invasive alien species. For example, Kriticos *et al.* (2013) showed that the pine processionary moth could reduce New Zealand's merchantable and total pine stem volume production by 30%, resulting in a total loss between NZ$1,550 M to NZ$2,560 M if left untreated following climate change (Kriticos *et al.*, 2013). In Europe, invasive alien species control programme would require to create between 520 and 2,520 employments (Jurado *et al.*, 2012).

Nowadays a number of studies are modelling the cost effectiveness of different management measures (Keller *et al.*, 2007; Lehrer *et al.*, 2011). For instance, Keller *et al.* (2007) showed that risk assessment produces positive net economic benefits in the Australian plant quarantine programme. The economic cost incurred by the Emerald Ash Borer could reach around US$12.5 billion with no programme to mitigate spread in 2020 (Kovacs *et al.*, 2011). However, appropriate management measures could decrease the cost to US$0.1 to 0.7 billion (Kovacs *et al.*, 2011). Further, Wilson *et al.* (2007) applied a new model to prioritize biodiversity actions based on cost effectiveness, showing that 24% of the funds should be allocated to invasive plant control in the 17 Mediterranean eco-regions as this yielded to the greatest marginal return on investment for biodiversity. It should be noted that there are different stakeholders paying costs and profiting benefits, therefore a cost benefit analysis is dependant of who is paying and who is benefiting. Besides, the cost benefit analyses may vary according to the part of the biodiversity targeted.

Overall, the first High Level Panel Report estimated that between US$34,100 to 43,900 million of investment with an additionally recurrent expenditure per annum of about US$21,005 to 50,100 are needed to achieve the Target 9 in 2020.

Table 9.1. Annual economic impact of terrestrial invasive species on a national scale. Source: From Barlow & Goldson (2002); Bergman *et al*. (1999); Bomford & Hart (2002); Clout (2002); Colautti *et al*. (2006); Gren *et al*. (2009); Kettunen *et al* (2008); McLeod (2004); Oliveira *et al*. (2013); Pimentel *et al*. (2005); Reinhard *et al*. (2003); Sinden *et al*. (2004); Singh & Kaur (2005); White & Harris (2002); Williams & Timmins (2002); Williams *et al*. (2010); Wi *et al*. (2006).

	Type of invasive		
Country	**Plant**	**Animal**	**Microbial**
Australia (in AU$)	4 billion	491.5 million (9 vertebrates) 703.9 million (10 vertebrates)	
Brazil (US$)		1.6 billion (24 pest)	
Canada (CAN$)	38.21 million	101.3 million (3 invertebrates) 14-16 million (emerald ash borer)	1.5 million (Dutch elm disease) 73.34 million (potato wart fungus) 1,000,000 (BSE)
China (US$)		14,450 million	
Europe (27 countries; US$) (EUR/year)	17.3 billion 5,985 million		
England (£)	1,291,461		
England (US$)		239 million (vertebrates)	
Germany (Euros)	103 million (8 species)	60.2 million (6 species)	5 million (dutch elm disease)
India (Rs)	1.68 billion (Fungal, bacterial, viral & nematode pathogens)		
New Zealand (NZ$)	200 million (weeds)	270 million (vertebrates) 2 billion (invertebrates)	
United States (US$)	34.5 billion	59.4 billion 28 billion (birds;1990-1997) 14 billion (mammals; 1990-1997) 1,806,787 (black rats) 400,000 (reptile; 1990-1997)	39.7 billion
Scotland (£)	244,736,000		
Sweden (SEK)	1620-5080 (13 invasive)		
Wales (£)	125,118,000		

9.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

The successful achievement of Target 9 is likely to guarantee a positive outcome for biodiversity conservation at the gene, species and ecosystem levels. Moreover, this will also contribute to achieve Targets 5, 10, 12, 14, and 15. For example, 49 populations of 12 invasive mammals were eradicated from 30 Mexican islands to prevent extinctions. These actions resulted in the protection of 202 endemic taxa with recolonization of some seabirds in several islands and new recruitment of endemic tree species (Aguirre-Muñoz *et al*., 2008) contributing to achieve Target 12. However, recovery of native biodiversity is often uncertain, as invasions may be indicators of more fundamental environmental change.

Furthermore, in order to provide updated information about the consequences of invasive alien species on endangered species during the next years, the IUCN SSC Invasive Species Specialist Group is working in cooperation with The IUCN Red List Unit at ensuring a full interoperability between The IUCN Red List, and the ISSG Global Invasive Species Database (GISD). The work is almost completed, and a beta version of GISD interlinked to the IUCN Red List is planned for release in the second half of 2014. To interlink the two IUCN products, for each IUCN Red list assessed species, all relevant information (scientific and common name of the alien species posing a threat, impact mechanism and outcomes, level of impact, etc.) have been analyzed and integrated in the GISD. For impact mechanisms and outcomes, a revised classification that has been developed with the support of leading scientists on the topic, has been produced and integrated into GISD. Also information on the impact level has been included in the database, based on the information provided in the assessment (high, medium, low, no/negligible/unknown/future/past). The final integrated information system will allow users to i) identify invasive species affecting each IUCN Red List species and ii) identify threatened species affected by each GISD invasive species. The integration of the two products will thus provide a valuable tool of prioritization of invasive alien species for mitigating the consequences for biodiversity in 2020.

9.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

Increased human population, global movement of people and goods, and land-use changes will remain the major driver of biological invasions in the future. They are all expected to increase and therefore to continue accelerating the current rate of invasions. In addition, the effects of climate change are expected to increase in importance over time for some species and some regions. Thus, climate change, global land use change and increased global trade will facilitate opportunities for invasive alien species to arrive and establish in new places at a long-term horizon. On the contrary, some invasive alien species may suffer from climate change, creating new opportunities for ecosystem restoration (Bradley *et al.*, 2009; Bellard *et al.*, 2013).

9.4.1 Propagule pressure

Increases in human population density will lead to greater disruption and degradation of habitats. Increasing global trade and movement of people will also favour propagule pressure, leading to new invasions by 2050 (Seebens *et al.*, 2013). For example, more exotic plant species are expected to become invasive on islands over the next century (Sax *et al.*, 2008).

Future climate change will also facilitate unintentional introductions through higher intensity and/or frequency of extreme events (Walther *et al.*, 2009; Figure 9.7). The Formosan subterranean termite had invaded nine southern states of the United States before Hurricane Katrina in 2006. Following Hurricane Katrina, millions of tons of wood debris, including debris infested with Formosan Termites have been under quarantine (Mainka & Howard, 2010) to limit the invasion in the United States. Climate change could also provide new routes that were not available previously such as the Northwest Passage in the northwest Atlantic, as the arctic ice cover is reduced and the ice-free season is extended, which offers a seasonal trading route through the northern ocean (Reid *et al.*, 2007).

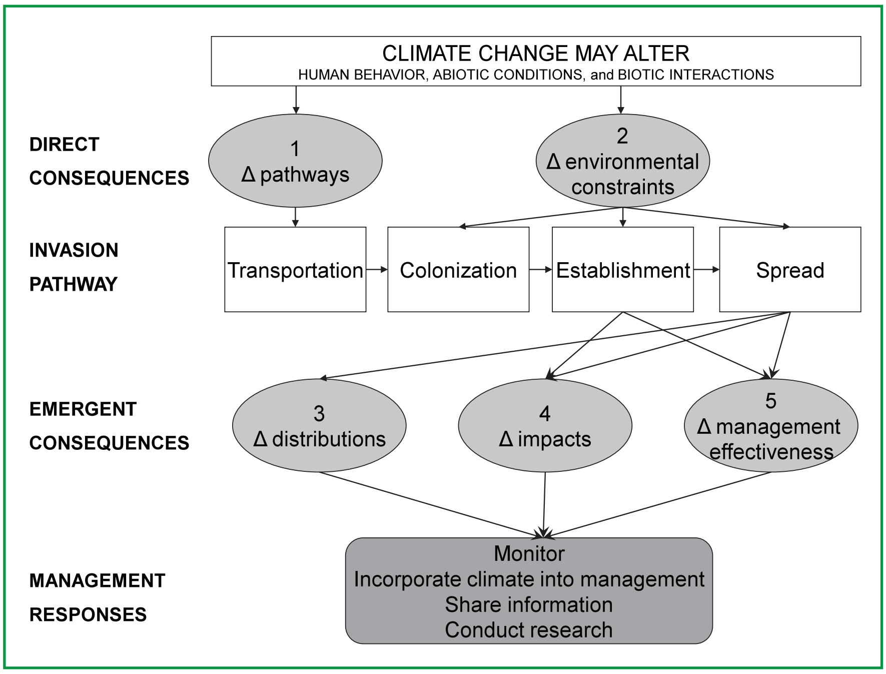


Figure 9.7. Potential consequences of climate change on the invasion pathway. Source: Extracted from Hellman *et al.* (2008).

9.4.2 Alteration of spread

Climate, global trade and land-use changes will alter the success of invasive alien species to invade new areas by creating disturbance events, which decrease the resilience of natural communities to invasion (Roura-Pascual *et al.*, 2011). Climate change is also expected to alter species survival and species reproduction, particularly for some plant groups (Peñuelas *et al.*, 2013) and zooplankton (Panov *et al.*, 2007). Climate warming can also result in an increase of dispersal performances for some invasive alien species, allowing for range expansion and invasions into new areas. For example, the mountain pine beetle can now complete a life cycle in one season, due to increased temperatures at higher latitudes and altitude (Logan & Powell, 2001).

Future changes of invasive alien species distributions are uncertain, but several generalizations can be made from species distribution models analyses. Currently, species distributions models are generally performed on a species by species basis. Little attention has been devoted to address multiple invasive alien species (but see Peterson *et al.*, 2008; Chytrý *et al.*; 2012; Bellard *et al.*; 2013). Peterson *et al.* (2008) showed highly nonlinear and contrasting projected changes in suitable areas of the European plants distribution. Plant species with expanding potential on one continent often had contracting potential on others. These changes suggest important community reorganization. Chytrý *et al.* (2012) also showed that the strongest increases of invasive alien species following land-use changes were projected for areas of north-western and northern Europe where current levels of invasion are low or average. In contrast, some areas such as Eastern Europe and some parts of southern Europe may experience no increase or even decrease in the level of invasion (Chytrý *et al.*, 2012).

At a global scale, the distributions of some invasive alien species will change with poleward migrations and movement to zones of higher altitude as regions experience elevated temperature. In the long-term (i.e., 2041-2060 period), some "high risk" regions to invasive alien species (i.e., the list of the "*100 among the worst invasive alien species*") are predicted to occur in Europe, United States, Southern Australia, Argentina, and Pacific and Caribbean islands due to climate and land use changes (Bellard *et al.*, 2013). Moreover, some regions will offer higher suitable environmental conditions such as eastern part of United States, northern part of Europe, Argentina, south of China and India (Bellard *et al.*, 2013, Figure 9.8). Some regions could lose a significant number of invasive alien species (e.g., Central America and Australia). In fact, areas of suitable habitat showed contrasting results according to the region and taxa considered, some taxonomic group will suffer from a consistent shrinking of their suitable area while other group species are projected to substantially increase (see Xu *et al.*, 2013; Bertelsmeier *et al.*, submitted). Thus, species distribution and climate analogue analyses could be a useful tool to prioritize regions and target species to monitor because close climate matching is generally a fundamental requirement for invasive success (but see Broennimann *et al.*, 2007; Gallagher *et al.*, 2010). Overall, climate change will also lead to novel climates across the world (Williams *et al.*, 2007), this will increase uncertainties to predict the presence of invasive alien species, in particular at low latitudes where the rates of novel climate will be important.

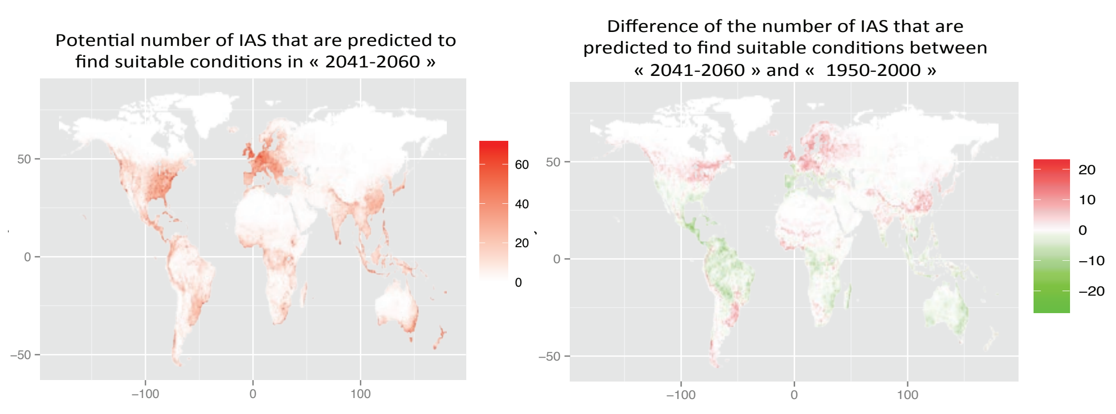


Figure 9.8. On the left, predicted number of invasive alien species of the list of the "100 among the worst invasive alien species" around the world that could find suitable environmental conditions in "2041-2060" period. On the right, difference of the number of predicted invasive alien species that will find suitable conditions between "future" and "current" period. Source: Bellard *et al.* (2013).

Although the effect of climate change on invasion will be important in the future, the probability that species from specific ecosystems will be introduced to climatically similar but geographically distant ecosystems will mainly depend on global trade patterns and volumes, both of which are predictable over a reasonable period.

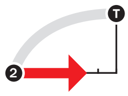
9.4.3 Implications for biodiversity

In the long term, the composition of biological communities is projected to change substantially in particular because of the population responses to climate change and high number of species predicted to be at higher risk of extinction (see Target 12). Potential shifts of some invasive alien species strongly suggest that biological communities will undergo dramatic reorganisation in coming decades (Peterson *et al.*, 2008). For example, some biomes including cool coniferous forest, temperate deciduous forest, temperature mixed forest might be more suitable to invasive alien species in the future, while tropical forest and tropical woodland will be less favourable to invasive alien species in 2080 (Bellard *et al.*, 2013). It is also projected that many more invasive alien species will become naturalised on islands in the future (Sax *et al.*, 2008) threatening insular biodiversity. Finally, although it may be known that changes in the composition and volume of trade and land use will both affect the likelihood of species introductions, establishment, and spread, there is still insufficient data to predict with high confidence whether the direction and magnitude of changes will continue to increase at the current rate.

9.5 UNCERTAINTIES AND DATA REQUIREMENTS

The literature about invasive alien species has important gaps in areas of theoretical and practical importance. In fact, most of the current research is focused on the causes of biological invasions (58% of publications), while 32% of publications is focused on the impacts of invasions (Lowry *et al.*, 2012). Among them, most of the studies focused on the consequences of invasive alien species at the species level and do not include impacts at the gene or ecosystem levels (Pejchar & Mooney, 2009). In addition, most studies are concerned with terrestrial invasions, and in particular of plant species (Lowry *et al.*, 2012). Studies are also geographically biased positively for North America, Western Europe, Eastern Australia and New Zealand, and there is a dramatic lack of studies in the tropics and on other areas of interests, such as the boreal tundra and taiga. Consequently, there are high uncertainties about the current and future pressure of invasive alien species in these regions. With regards to projections, the potential distribution of invasive alien species assumes that species will have a constant impact over the world, while the intensity and multiplicity of impacts are highly context dependent (Simberloff *et al.*, 2012; Ricciardi *et al.*, 2013). Moreover, there is a lack of information in the listing of invasive alien species (but see Global Register of Introduced and Invasive Species), in particular regarding impacts (Kulhanek *et al.*, 2011). Observed impacts often fail to translate to ecosystem services or evidence of environment degradation (Hulme *et al.*, 2013). However, the recent collaboration between the IUCN SSC Invasive Species Specialist Group and The IUCN Red List Unit will facilitate integration and sharing information on the impact of invasive alien species on threatened native species. Furthermore, less than 11% of countries are considered to have adequate data on invasive alien species (Genovesi *et al.*, 2013). Finally, while a majority of countries have identified targets or actions related to invasive alien species, few parties have developed quantitative targets. This has made difficult to evaluate and assess progress towards achieving Target 9. The development of new indicators to monitor the achievement of the target is thus required. With regards to practical management, the exchange of knowledge of successful management measures but also measures that have failed, and that should not longer be applied should also be a future challenge.

9.6 DASHBOARD – PROGRESS TOWARDS TARGET

Element	Current Status	Comment	Confidence
Invasive alien species identified and prioritized		Measures taken in many countries to develop lists of invasive alien species	High
Pathways identified and prioritized		Major pathways are identified, but not efficiently controlled at a global scale	High
Priority species controlled or eradicated		Some control and eradication, but data limited	Low
Introduction and establishment of IAS prevented		Some measures in place, but not sufficient to prevent continuing large increase in IAS	Medium

Authors: Céline Bellard and Franck Courchamp with contributions from Piero Genovesi and Shyama Pagad.
Box 9.2 text contributed by James Russell

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TARGET 10: VULNERABLE ECOSYSTEMS (CORAL REEFS)

By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning.

PREFACE

This analysis evaluates trends in indicators of coral reef health and the implications of these trends for biodiversity in 2020 and 2050. It discusses progress towards increasing the area of coral reefs under full protection, but also the challenges involved in curbing the effects of rising human populations and climate change. Although this chapter focuses on shallow water corals, it is recognized that reef ecosystems are part of an interconnected coastal seascape, and that efforts to protect reefs have to encompass considerations of these connected coastal ecosystems. Deep water corals are also highly threatened by anthropogenic activities and climate change (Roberts and Cairns, 2014), and may be even more impacted by ocean acidification and the rise of aragonite. Due to the specific impacts and threats identified and forecasted, and to the large differences in geographic distribution and ecosystem dynamics compared to shallow water corals, cold water corals will not be directly dealt with in this chapter.

The level of human dependence on coral reefs is high. Approximately 850 million people live within 100 km of coral reefs and are dependent on reefs either for food, livelihood, coastal protection, or amenity. Of these, 275 million people live in the direct vicinity of coral reefs (Burke *et al.*, 2011). People living on small-island states tend to be the most reef-dependent, in part because of paucity of alternative livelihoods. More than 94 countries and territories provide reef-based tourism, which accounts for more than 15% of gross domestic product in 23 countries.

Most of the ecosystem functions of reefs, such as the provision of productive fisheries, tourism appeal, and coastal protection from storms, are founded on having a complex reef structure that keeps accreting (growing). A structurally complex reef provides habitat (and hiding places) to support high levels of biodiversity (Gratwicke and Speight, 2005), which span a diversity of fishes and invertebrates, many of which remain poorly documented. If a reef is to continue functioning then it must at least have net growth – i.e., that the deposition of a carbonate skeleton by corals and calcareous algae must exceed the rate at which the skeleton is removed by physical damage and the erosion caused by a host of taxa including burrowing algae, sponges, and worms. The balance of reef construction and erosion is known as a carbonate budget (Stearn *et al.*, 1977). Perhaps that greatest threat to coral reef biodiversity is the long-term loss of reef habitat that could occur if carbonate budgets become persistently negative (erosive).

10.1 ARE WE ON TRACK TO ACHIEVE THE 2015 TARGET?

10.1.1 Status and trends

10.1.1.i Local threats

Vulnerable habitats like tropical coral reefs are threatened by both local and global stressors. These threats affect not only the corals, but also coral reef associated communities that form the reef ecosystem. Local stressors include over-harvesting of fisheries (McManus, 1997), destructive fishing methods (e.g., explosives, cyanide), marine-based pollution and damage (e.g., oil and gas installations, shipping and anchor damage), watershed-based pollution (e.g., nutrients and fertilizer runoff, Richmond *et al.*, 2007), and coastal development (e.g., sewage discharge, dredging), and marine recreation (e.g., diving and boating). Global stressors are principally rising sea temperatures, which reduce coral calcification and can elicit coral bleaching events, and ocean acidification, which has a variety of deleterious impacts on reef systems (Hoegh-Guldberg *et al.*, 2007). Superimposed upon these threats are natural perturbations such as cyclones (Rogers, 1993). According to the Reefs at Risk Revisited report, the percentage of reef area rated as threatened increased by 30% in the decade from 1997 to 2007 (Burke *et al.*, 2011). Threat levels were estimated by integrating indicators of local and global threats within a geographic information system (GIS). Much of the elevated increase (80%) was driven by rising threats from fishing in the Indian and Pacific Oceans, largely because of elevated density of coastal populations (Burke *et al.*, 2011). The main conclusions about current local anthropogenic impacts include: 1) More than 60% of the world's coral reefs are under immediate and direct threat from one or more local stressors; 2) Of local stressors, fishing is the most pervasive threat, affecting more than 55% of reefs. Coastal development and watershed-based pollution each threaten about 25% of reefs. Marine-based pollution threatens about 10% of reefs; 3) Local pressures are most severe in Southeast Asia, where nearly 95% of reefs are threatened and 50% are in the 'high' or 'very high' threat category (Figure 10.1, Table 10.1). Indonesia has the largest area of threatened reef followed by the Philippines (Burke *et al.*, 2011). Although much of this threat stems from fishing, it should be noted that land based activities also impact heavily upon the reef (e.g., Brodie *et al.*, 2012, see Box 10.2), and these cumulative impacts need to be addressed through sound coastal zone management.

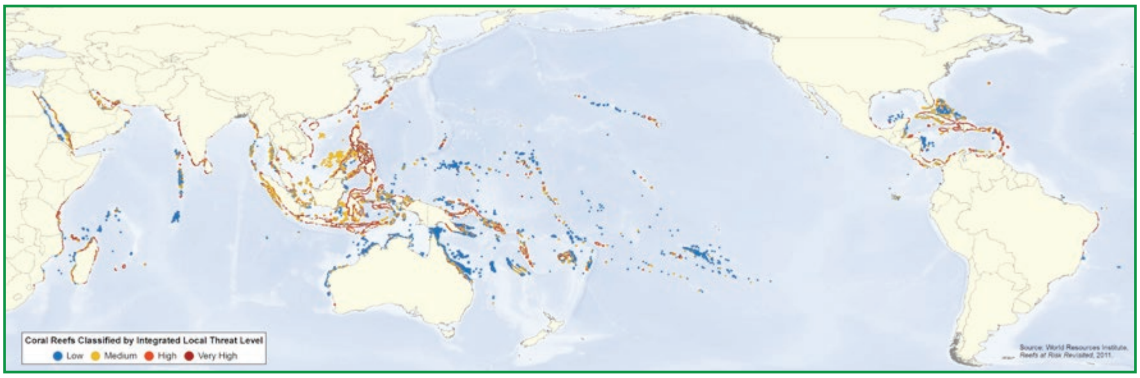


Figure 10.1. Distribution of coral reefs classified by human threat level. Red represents very high; orange – high; yellow – medium; and blue – low threat. Source: Burke *et al*. (2011).

Table 10.1. Geographic trends in reef threat from Reefs at Risk Revisited

Region	Source of threat	Percentage of reefs Threatened*	By threat category High / Very high
Southeast Asia	Main threat is from overfishing and destructive fishing	95%	50%
Atlantic Ocean	Multiple threats, Bahamas have largest reef area at low threat	75%	30%
Indian Ocean	Fishing most widespread threat	65%	35%
Middle East	Multiple threats. Exceptions include Chagos Archipelago, Maldives, Seychelles	65%	20%
Wider Pacific	French Polynesia, the Federated States of Micronesia, Hawaii and the Marshall Islands have some of the lowest sources of local stress.	50%	20%
Australia	Least threatened globally**	14%	1%

** Includes 4 local threats (coastal development, watershed-based pollution, marine-based pollution and damage – such as oil exploration and shipping – and fishing impacts) and 1 global threat (historical coral bleaching events in last 10 years).*

*** Although Australian reefs were considered to be the least threatened globally in 2011, new analyses of long-term monitoring and survey data have revealed that coral cover has decreased dramatically from a mean of 28% to 14% between 1985 and 2012 (De'ath et al., 2012). The causes of such decline were attributed to cyclones (48%), coral predation by crown-of-thorns starfish (42%), and coral bleaching (10%). Of these, only coral predation is likely to be caused by local human impacts (principally high nutrient runoff).*

In addition, international market demand for reef resources, such as aquarium fish and corals, directly affect the integrity of reef ecosystems through the removal of reef organisms and modifying habitat. This reinforces the need for national as well as international legislation that not only focuses on reducing fishing pressure, but also regulates trade in reef organisms. In the past decade, there has been an overall declining trend in the trade of wild corals (Figure 10.2); however, it should be noted that while coral rock and dead coral trade is declining, the trade in live coral is increasing (Wood *et al.*, 2012).

10.1.1.ii Global threats

Reefs are principally impacted by two processes at the global scale (Hoegh-Guldberg *et al.*, 2007, Pandolfi *et al.*, 2011, Frieler *et al.*, 2012). The first is rising sea surface temperatures, that are currently increasing at rates of 0.2°C decade^{-1} in SE Asia (Penaflor *et al.*, 2009) and up to 0.5°C decade^{-1} in the Caribbean (Chollett *et al.*, 2012). Long-term rising sea temperatures can be a chronic stressor that reduces the calcification rate of corals (Carricart-Ganivet *et al.*, 2012). Higher temperature also predisposes corals to 'coral bleaching' which is a serious disruption of the symbiosis between the coral host and the dinoflagellate algae that live within its tissues. Recent ENSO (El niño-Southern Oscillation) events have caused massive coral bleaching, often followed by extensive mortality, at regional and global scales. The most severe global event occurred in 1998, but other events have occurred at regional scales in 2005 and 2010 (Eakin *et al.*, 2010) and their frequency is expected to increase under global warming (Frieler *et al.*, 2012). A global representation of bleaching events and severe thermal stress is given in Figure 10.3.

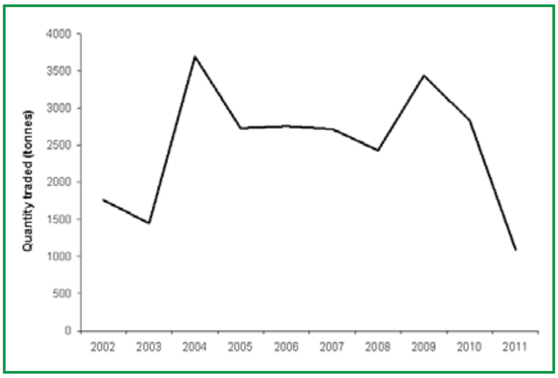


Figure 10.2. Trend in the trade of wild corals 2002-2011, as reported by exporter countries. Source: CITES trade data.

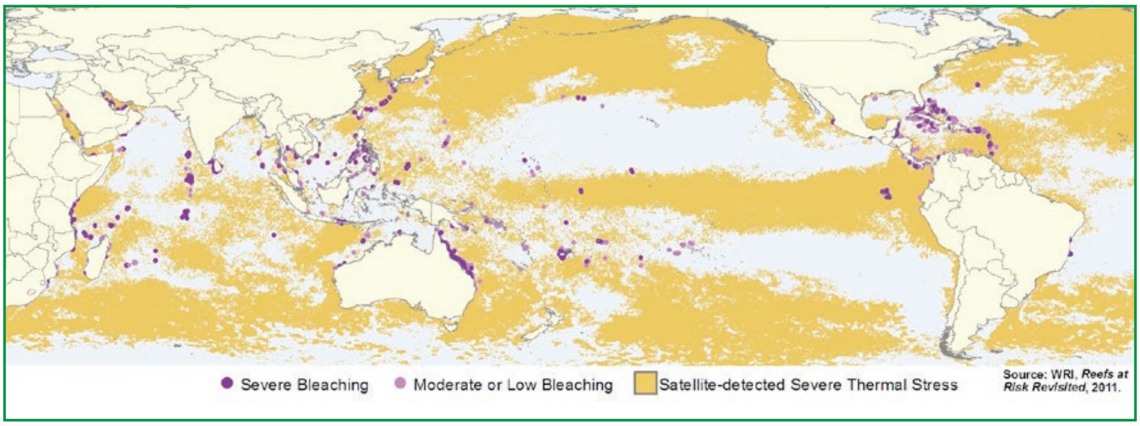


Figure 10.3. Thermal stress on coral reefs (1998-2007) Source: WRI, *Reefs at Risk Revisited* (2011).

The second major stressor is ocean acidification (OA) which occurs as atmospheric carbon dioxide continues to be absorbed by the ocean. OA will continue to affect the biology and ecology of reef ecosystems, though the precise consequences are difficult to estimate. OA can impede the rate of calcification and interfere with a range of biological processes, including the sensory capabilities of fish (Munday *et al.*, 2009) and corals (Doropoulos *et al.*, 2012), and the competitive interactions between algae and coral (Diaz-Pulido *et al.*, 2011). Although these two global stressors are unavoidable and cannot be directly mitigated in the near term, it is still essential to take international actions to diminish climate change impacts, even if results from these actions may only manifest in the long term. At the same time, actions to mitigate global carbon emissions will be positive for society in general by improving energy use and other climate policies.

Management action is hampered by a lack of understanding about whether the effect of multiple global and local stressors is synergistic or not (e.g., Dunne, 2010; Gurney *et al.*, 2013). It is likely that multiple stressors can act synergistically, though the outcomes will vary according to the stressors involved. For instance, studies of fishing and bleaching impacts on Kenyan corals found no evidence of synergisms (Darling *et al.*, 2010) whereas the dual impacts of rising sea temperature – chronic reductions in coral calcification and more frequent bleaching – are predicted to have synergistic impacts on coral reef resilience (Bozec and Mumby, 2014).

10.1.1.iii Trends in reef health

Overall trends of reef health, measured by the cover of living coral, are generally strongly negative. For example, average Caribbean coral cover has declined, on average, from around 50% in the 1970s to approximately 10% by 2000 (Figure 10.4; Gardner *et al*., 2003). In particular, the loss of formerly dominant *Acropora* species in the Caribbean was attributed to the emergence of the aggressive white band disease, first observed in the early 1970s (Aronson and Precht, 2001). Bruno and Selig (2007) analysed coral cover from various data sets across the Indian and Pacific Oceans and found an overall average loss of 0.72% y^{-1}. However, both studies show marked intra-regional variability in these overall trends. For example, the Caribbean still includes countries where coral cover can exceed 50%, such as Bonaire. Much of the net loss appears to be attributable to the severe global bleaching event of 1998.

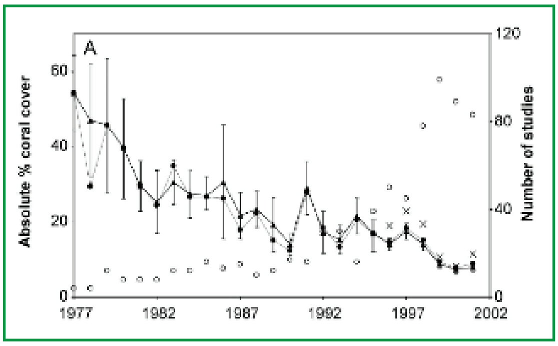


Figure 10.4. Absolute percent coral cover in the Caribbean from 1977-2001. Triangles (▲) and circles (●) show weighted and unweighted mean coral cover estimates for each year, respectively. X shows the unweighted mean coral cover without one study location (Florida Keys Coral Monitoring Project, 1996-2001). Open circles show the sample size per year. The temporal trend shows a fall in coral cover from around 50% in the 1970s to 10% in 2002. Source: Gardner *et al*. (2003).

It is insightful to examine cases where corals still have the resilience required to bounce back after disturbance. In 1997, Connell (1997) found marked evidence of recovery in the Indo-Pacific but little in the Caribbean. Revisiting this question, Roff and Mumby (2012) found only a single recovery trajectory in the Caribbean, whereas striking coral recovery is still reported from many sites in the Indo-Pacific (e.g., Adjeroud *et al*., 2009, Halford and Caley, 2009). It seems that reef resilience is particularly impaired in the Caribbean, in part because of few fast-growing coral species, relatively few herbivorous fish species, and a predisposition of seaweed to bloom (Roff and Mumby, 2012). But even relatively well-managed systems in the Pacific can show a net "ratcheting down" of coral cover as they experience repeated disturbance over a short period of time. A recent report concluded that the average state of the Great Barrier Reef has declined by around 50% of living coral over the last 25 years (De'ath *et al*., 2012).

Here, we attempted to estimate the degree to which global change (rising sea temperature and OA) may have affected coral communities (Wolff *et al*., in review). This study suggests there have been increasing levels of climate stress from the 1970s to 2000 (Figure 10.5). Present levels of stress vary markedly around the world, being relatively low in Australia, the South Pacific, North Asia, and the tropical Western Atlantic.

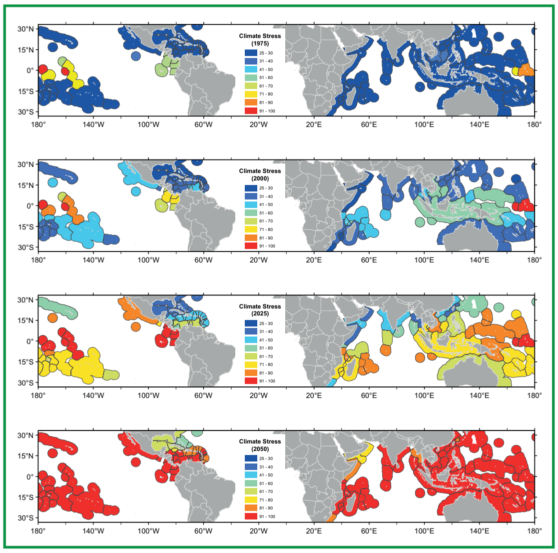


Figure 10.5. Models of global climate-related stress upon corals in 2000, 2025, and 2050 based on AR4 SRES A2, from which AR5 RCP8.5 was derived. Source: Wolff *et al*. (in review).

10.1.1.iv What has been done to protect reefs?

Marine protected areas (MPAs), if well enforced and integrated with coastal watershed management, can mitigate the direct effects arising from multiple local anthropogenic stressors on coral reefs. MPAs cannot directly address global threats such as ocean acidification; nonetheless, well managed MPAs can increase the resilience of reefs, thereby making them less susceptible to environmental and climate stressors such as coral bleaching. For instance, no-take MPAs have been used effectively to rebuild fish stocks (Russ *et al*., 2008) on coral reefs and even help corals recover after bleaching (Mumby and Harborne, 2010). Approximately 27% of the world's reefs are located inside MPAs. However, many MPAs turn out to be 'paper parks' due to lack of enforcement capacity; an analysis of effectiveness concluded that only about 15% of reserves have reduced the threat from fishing (Burke *et al*., 2011). Nevertheless, many new coral reef MPAs are now being planned (Fig. 10.6), although it should be noted that the coverage of many large MPAs was found to be strongly biased away from areas of greatest threat.

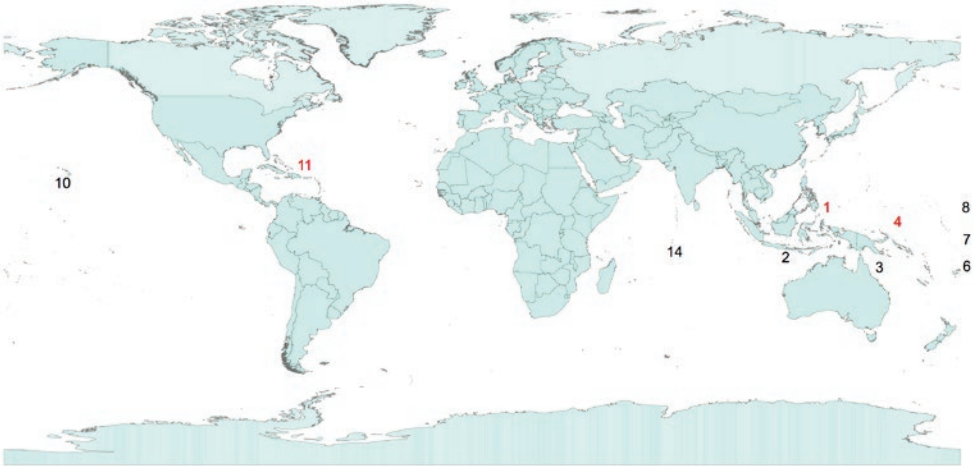


Figure 10.6. Map of recently established large-scale MPAs or MPA initiatives. Red numbers indicate regional MPA initiatives involving more than one country; bold numbers indicate MPAs that are pending establishment; * indicates MPAs that are less than 500,000 km^2 in size; ** indicates MPAs that are larger than 500,000 km^2 in size; and ^ indicates regional initiatives for which MPA area is not specified. See Appendix 10.1 for more details. Note that locations of MPAs are indicative of general area only, and not precise. Source: map created by L. Teh.

10.1.2 Projecting forward to 2020

Target 10 is one of the few Aichi 2020 targets with a goal for 2015 rather than 2020. While it is possible that non-linear responses of coral reefs to stress and disturbance might cause sudden flips to degraded states (Mumby *et al.*, 2013), it is difficult to see how current trends as described in section 1.a. will substantially change by 2015 compared to our analysis of current trends; therefore, we focus this analysis on trends out to 2020. It is important to note that the response of animals and plants to conditions expected in 2020 and beyond are difficult to evaluate, as many physiological studies of climate change response use treatments that simulate environments far into the future.

Human population in coral-reef countries is expected to increase by nearly 15% between 2010 and 2020 (UN Department of Economic and Social Affairs, 2004). Thus, while it is difficult to project future changes in local threats, recent trends and the expanding human population suggest that threat levels will continue to increase. Sectors which impact upon coral reefs, such as forestry, marine tourism and agricultural development, also show increasing trends. For instance, the area of industrial oil palm plantations in Southeast Asia has grown continuously since 1990, with greatest growth since 2007, and it is expected that this growth trend will continue to 2030 (Miettinen *et al.*, 2012). Further, global tourism is projected to grow at an average rate of 4.1% to 2020 (UNWTO, 2001). This has implications for reef biodiversity hotspots, such as the Coral Triangle, where coastal and marine tourism growth has been strong (Crabtree, 2007). While marine ecotourism can potentially be a sustainable alternative to fishing for coastal communities (Fabinyi, 2010), mass tourism that is not within the physical limits of the environment may further degrade reef and coastal habitats (Teh and Cabanban, 2007), potentially resulting in decreased fisheries and other ecosystem services. International trade is another factor that can likely impact coral reef biodiversity, with trade in wild corals projected to increase to 2020 (Figure 10.7).

Studies of recent rises in the sea temperature in coral reef areas (post 1985) find striking regional variability. In Southeast Asia, for example, mean rates of warming are 0.2°C/decade but some areas in southern Asia have actually net cooling (Penaflor *et al.*, 2009). Similarly, average rates of recent warming are 0.29°C/decade in the Caribbean, but warming is most intensive in the south and net cooling occurred around parts of Florida (Chollett *et al.*, 2012). Temporal trends in ocean acidification during this period are generally unclear, though intensification has been predicted for the northern Caribbean, where data are available (Gledhill *et al.*, 2008).

Despite the anticipated increase in threat and ineffectiveness of many MPAs, the designation of protected area status continues to increase. Some countries, including the United States, Kiribati, Australia, and the United Kingdom have declared massive MPAs over coral reef regions (Figure 10.6). In addition, several regions, including the Caribbean and Micronesia, have signed up to ambitious targets to protect large areas of coast within the next decade or two (Figure 10.6). Not all areas are intended to become no-take marine reserves, but the increased level of protection should increase the efficacy of management and reduce the probability of destructive activities, thereby potentially increasing the resiliency of protected reefs.

10.1.3 Country actions and commitments[1]

Relatively few countries have established national targets, or similar elements, related to this Aichi Biodiversity Target. (Note, however that a number of National Biodiversity Strategies and Action Plans (NBSAPs) examined are from countries that do not have coral reefs). Many countries note the growing role of climate change as a main driver of biodiversity loss in their NBSAP. Those national targets that have been established are generally in line with the Aichi Biodiversity Target. However there tends to be a general emphasis on building resiliency to climate change.

Few targets explicitly refer to reducing anthropogenic pressures on coral reefs. Similarly, few targets explicitly refer to reducing anthropogenic pressures on ecosystems which are vulnerable to climate change. Three examples which are counter to this trend are Finland, Brazil, and Japan, which have established national targets that refer to reducing anthropogenic pressures on vulnerable ecosystems. For instance, Japan's national target promotes initiatives geared towards minimising human-induced pressures that cause ecosystems that are vulnerable to climate change (coral reefs, seagrass beds, tidal flats, islands, and subalpine and alpine zones) to deteriorate by 2015.

10.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

10.2.1 Actions

A recent study modelled the trajectories of Caribbean coral reefs under various management scenarios on greenhouse gas (GHG) emission levels (business as usual and low carbon economy), and evaluated their expected carbonate budgets (Kennedy *et al.*, 2013). The study found that positive carbonate budgets could be maintained at least towards the end of this century but only if compelling action is taken to reduce GHG emissions (to the most optimistic scenarios being considered by the IPCC) and if local threats including overfishing and water quality are managed (Figure 10.8). Thus, the achievement of Target 10 requires a strong global commitment to reducing GHG emissions, ideally following Representative Concentration Pathway 2.6.

Footnote

[1] This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.

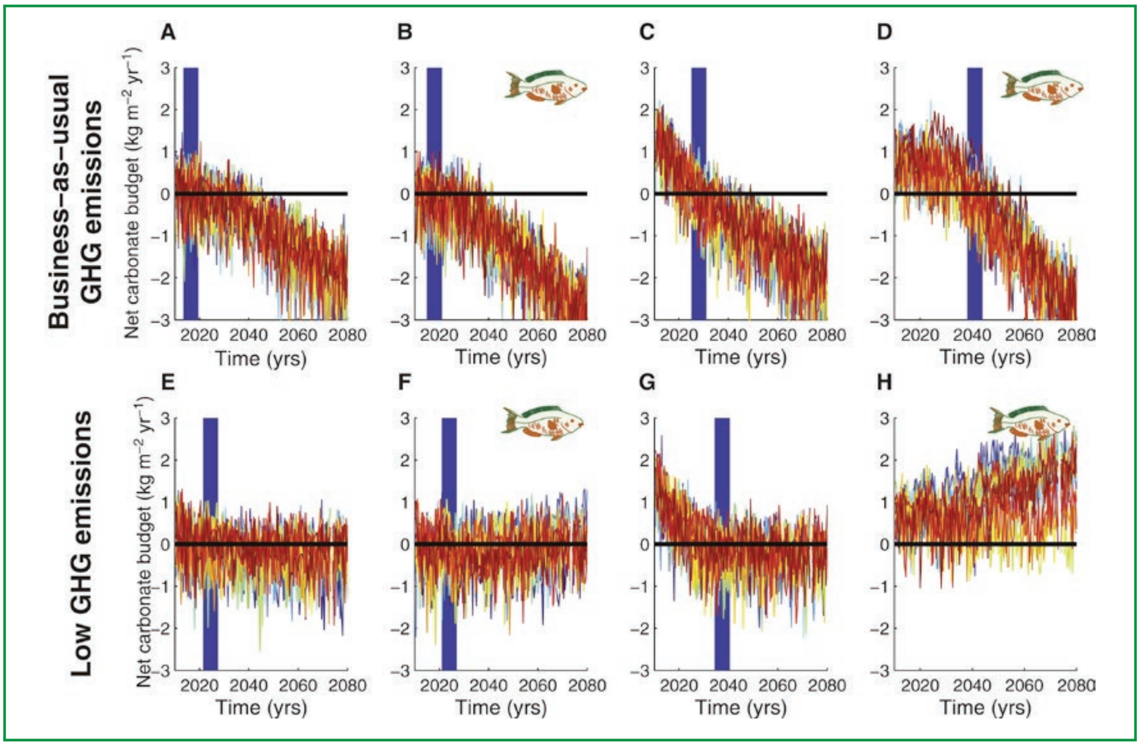


Figure 10.8. Projected carbonate budgets of a Caribbean reef under climate change and ocean acidification with and without local protection of herbivores under scenarios of realistic GHG emissions (top) and aggressive reduction (bottom). Initial conditions of reefs are either degraded with 10% coral cover (A, B, E, and F) or healthier with 20% coral (C, D, G, and H). Herbivorous fish are either overfished or fully protected (denoted with parrotfish symbols). Each plot displays 20 simulations, with outputs generated at 6 month intervals and run for years 2010–2080. Vertical blue bars indicate point at which the projected budget becomes negative (<–0.1 kg for >5 years). Source: Kennedy *et al*. (2013).

The exploitation of herbivorous fish can lead to an increase in fleshy seaweeds (macroalgae) that pre-empt space for coral settlement and then compete with those corals that do manage to settle (Williams and Polunin, 2000; Hughes *et al*., 2007; Mumby *et al*., 2007). Herbivores are usually protected inside no-take marine reserves but, while useful, this step fails to address concerns over the loss of reef habitat quality in areas subjected to exploitation. Seascape-wide management of herbivory requires fishery regulations including reductions in the use of fish traps (Hawkins *et al*., 2007) and species-level catch limits or bans. For example, herbivore fisheries have been prohibited in Belize, Bermuda and Bonaire.

Box 10.1: Reducing local threats through private coral reef management

Local anthropogenic threats pose the greatest risk to coral reefs in Southeast Asia. However, reef management in the region is often limited by lack of funds and resources. One approach for overcoming this challenge is the use of private sector resources for coral reef conservation. The establishment of the Sugud Islands Marine Conservation Area (SIMCA) in Sabah, Malaysia was initiated by owners of the sole dive resort situated within SIMCA, in collaboration with the Sabah Wildlife Department, for the purpose of protecting the area's coral reefs and marine environment. The SIMCA was officially declared an IUCN category II conservation area in 2001. Reef Guardian, a conservation organization, manages conservation activities to reduce local threats to the coral reefs within SIMCA. These include enforcement patrols to regulate illegal fishing, turtle monitoring and conservation, coral reef and environmental monitoring, sewage and wastewater treatment, removal of coral predators (crown of thorns), and conducting education programmes for school children to raise awareness about marine conservation. Reef Guardian's conservation work is funded by conservation fees charged to visitors to the dive resort, donations, and grants. This private management approach has helped to mitigate the impacts of tourism and fishing on SIMCA's reefs, resulting in improved biodiversity conditions. For instance, coral cover and fish abundance is greater within SIMCA compared to fished areas, and the number of turtle nestings shows an increasing trend through time. See www.reef-guardian.org.

Source: Teh *et al*. (2008).

MPAs are now being planned with multiple stressors in mind. Designation of an MPA will not reduce impacts of climate change or OA, but it is possible to identify regions of the ocean that have a more benign physical environment (West and Salm, 2003; McLeod *et al.*, 2012). For example, Mumby *et al.* (2011) used a climatology of satellite-derived sea surface temperature (SST) measurements to identify those areas where corals are likely to be best acclimated to stress and subjected to relatively mild acute bleaching events (Fig. 10.9). In principle, locating MPAs in the most benign areas allows for a targeted reduction in biological stresses (e.g., restored food webs and less competition from algae) in areas that have relatively low physical stress.

The local stressors modelled by Kennedy *et al.* (2013) included the exploitation of herbivorous fish and nitrification of watersheds. Nutrient and sediment runoff can be reduced by cutting back on the use of fertilizer, stabilizing soils by keeping riparian watersheds forested, and maintaining estuarine habitats including mangroves and marshlands.

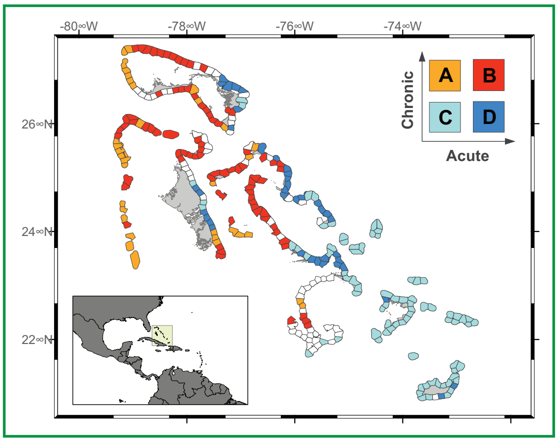


Figure 10.9. Stratification of coral reefs of the Bahamas based on their thermal characteristics. Two aspects of thermal stress are recognized - chronic stress that represents usual summer temperatures - and acute stress that occurs during episodic bleaching events. Ordering all four environments from the most benign to stressful gives A, C, B, and D respectively.
Source: Mumby *et al*. (2011).

Box 10.2: Reducing land based threats to coral reefs

Agricultural run-off has increased levels of sediments, nutrients, and pesticides on the Great Barrier Reef (GBR), leading to degraded water quality. This has detrimental long term impacts on reef health, as pollutants change the environmental conditions for reef species and ecosystems. In response, the Australian and Queensland governments have introduced several policy initiatives and regulations. The Reef Plan, introduced in 2003, aims to stop and reverse the decline in GBR water quality by reducing source pollutant loads and rehabilitating reef catchment areas that play a role in removing water borne pollutants. A component of the Reef Plan was Reef Rescue, an A$200 million investment by the Australian Government to fund voluntary programmes for on the ground work, monitoring, and research. Land management activities funded by Reef Rescue mainly targeted the sugarcane and grazing industries; projects included introduction of new farming practices, fencing along streams for cattle management, modification to fertilizer and pesticide application gears, and modifying cultivation and tillage equipment and practices. In 2009, the Queensland Government introduced the Great Barrier Reef Protection Amendment Act, which provides for the implementation of fertilizer, pesticide and erosion management regulations for the GBR. Despite these useful management actions, it is expected that improvements to water quality and reef health may not be detectable for several decades. Source: Brodie *et al.*, 2012. For more information see www.gbrmpa.gov.au/outlook-for-the-reef/declining-water-quality

10.2.2 Costs and cost-benefit analysis

Mitigating anthropogenic threats to coral reefs require cost-effective land and marine based conservation actions. A study by Klein *et al.* (2010) calculated the rate of return on investing in two conservation actions to mitigate high impact threats for the 16 ecoregions in the Coral Triangle. The analysis involved estimating the cost of effectively managing coral reefs and terrestrial protected areas, based on the assumption that protection linearly reduced the threat in each ecoregion. Depending on ecoregion, estimated annual management costs for coral reefs ranged from US$15,300/km^2 to US$383,500/km^2. An earlier study modelled the change in coral cover in Montego Bay, Jamaica, arising from different interventions (e.g., sediment traps, building a large scale waste treatment facility, solid waste collection, Ruitenbeek *et al.*, 1999). Costs associated with mitigating actions that increased coral abundance by up to 20% had a present value of US$153 million over 25 years. Achieving a 10% increase in coral abundance had a present value cost of US$12 million. The study showed that the optimal intervention may depend on targeted coral quality levels, such that the lowest cost management intervention may not be the most optimal. While these studies focused on the cost-effectiveness of protecting coral reefs, it is also important to consider that conservation actions which reduce fishing effort will impose substantial costs on fishers' livelihoods and food security in the short term. The Higher Level Panel estimates that achieving Target 10 will require an initial investment of US$600 million to US$900 million, with average annual expenditures of US$80 million to US$130 million for the period 2013 - 2020, and recurrent expenditures of US$6 million to US$10 million. Despite an immediate cost, conservation makes economic sense because of the potential future benefits it generates. For instance, reef conservation in the Caribbean was estimated to avert potential annual services losses ranging between US$350 million and US$ 870 million (Burke *et al.*, 2008).

Economic valuation of coral reef goods and services in different countries have provided a wide range of estimates, mainly due to the different services and time frames that are used in the valuation process (Table 10.2). Nevertheless, based on these studies, it is reasonable to conclude that the global net present value of coral reefs reaches billions of US$. While the economic values presented here are unlikely to reflect potential non-monetary costs arising from the cascading effects of reef habitat and ecosystem loss, they are nonetheless important for demonstrating the magnitude of economic benefits derived from coral reef ecosystems.

Table 10.2. Economic valuation of coral reef goods and services in different countries. Compiled by the Higher Level Panel report.

Country	Value	Services/Value	Source
Sri Lanka	Net Present Value (NPV) of US$14 – 750 million/ ha over 20 years	Multiple services, especially tourism and erosion control	Berg *et al.* (1998)
Phi Phi Islands, Thailand	US$497 million / year, including US$205 million recreational values	Use and non use values	Seenprachawong (2003)
Great Barrier Reef, Australia	NPV of US$53 billion (100 yrs, 2.65% discount rate)	Range of values, especially tourism, non-use and coastal protection	Oxford Economics (2009)
Belize	US$268 - 370 / km^2/ year	Tourism, fisheries, shoreline protection	Burke *et al.* (2008)
Northern Marianas	US$0.8 million / km^2	Multiple goods and services	van Beukering *et al.* (2006)
Pacific Islands	US$506 – 17,873 / ha/ year	Tourism, coastal protection, fisheries and other services	Various
Caribbean	US$3.1 billion - 4.6 billion p.a.	Shoreline protection, tourism and fisheries	Burke *et al.* (2008)

10.3 WHAT ARE THE IMPLICATIONS OF NOT REACHING THE TARGET FOR BIODIVERSITY IN 2015?

The direct impact of anthropogenic impacts tends to be either a loss of coral species or a shift in species composition to one able to tolerate more stressful conditions. Coral bleaching commonly leads to an immediate loss of the most thermally-sensitive corals (Edwards *et al.*, 2001). A long-term study found that a major bleaching event led to an immediate loss of coral species richness (van Woesik *et al.*, 2011). However, while total richness recovered after 10 years, the assemblage of corals changed. Ascertaining exactly how coral assemblages will change in the long-term is challenging but a shift towards thermally-tolerant species and those that are able to recover from partial mortality is likely (van Woesik and Jordan-Garza, 2011), as is an overall reduction in coral species richness and abundance.

Other forms of local impact, such as sedimentation, have a fairly predictable impact on coral assemblages as it tends to favour a particular subset of species, such as *Turbinaria mesentaria* (Anthony, 2006) that can tolerate higher sediment and make greater use of heterotrophy. Nevertheless, much of the impact of sedimentation on reefs ultimately depends on other environmental factors, such as oceanography and physiography. Further, it has been suggested that hypoxia derived mortality is likely to increase in the future as global temperature rises and other coastal stresses increase the sensitivity of benthic organisms to oxygen depletion (Vaquer-Sunyer and Duarte, 2008).

The impacts of climate change on non-coral invertebrates have been under-studied (Przeslawski *et al.*, 2008), making it difficult to make predictions at this stage. The impacts of a loss of living coral and habitat complexity are fairly profound on coral reef fish, particularly those of smaller size that are more vulnerable to predation (Wilson *et al.*, 2006). A bleaching event and plague of crown-of-thorns starfish in Papua New Guinea led to a vast loss of coral from around 65% to 20% over eight years (Jones *et al.*, 2004). Over 75% of fish species declined in abundance with 50% declined to less than half their original abundance (Figure 10.10). Similar events were also reported from French Polynesia (Kayal *et al.*, 2012). Elsewhere, many studies have quantified a strong negative relationship between the structural complexity of reefs and fish species richness (Luckhurst and Luckhurst, 1978; Graham *et al.*, 2006).

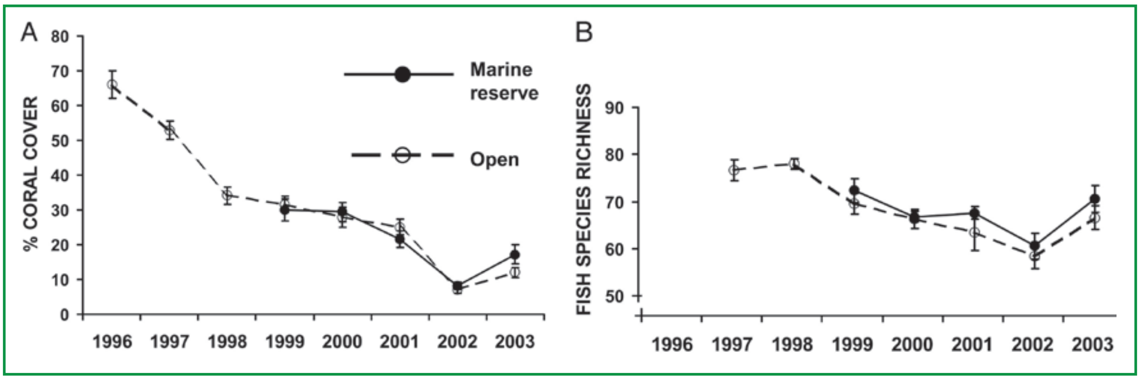


Figure 10.10. Loss of coral cover and reef fish species richness from Kimbe Bay. Species richness calculated for the fish families Acanthuridae, Chaetodonidae, Labridae, and Pomacentridae. Source: Jones *et al*. (2004).

10.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

In coral reef countries, total human population is set to increase by 27% between 2010 and 2050 (United Nations Department of Economic and Social Affairs, 2004). It is likely that anthropogenic impacts will at least increase from current baselines, perhaps strongly given the increased pressure on coastal resources. Rising sea temperatures are expected to cause more intense and frequent bleaching. Using a consortium of global circulation models, Frieler *et al*. (2012) estimate that damaging bleaching conditions would affect approximately >90% of the world's reefs under RCP8.5 and 60% of reefs under the more optimistic GHG emissions scenario, RCP3PD. Ocean acidification will continue and affect the biology and ecology of reef ecosystems, though the precise consequences are difficult to estimate for 2050 because many experimental studies simulate environments further into the future.

The consequences for coral reef biodiversity are, of course, uncertain by 2050 but are likely to be highly influenced by the trajectory taken for GHG emissions. It is even more difficult to estimate the potential consequences on other non-coral vulnerable ecosystems which are relatively less documented. Maintenance of positive carbonate budgets is possible with aggressive action on emissions and effective local management (Kennedy *et al*., 2013). Of significant concern is how changes in reef state and ecological processes will affect functions like fisheries productivity and coastal protection (Pratchett *et al*., 2011). At this point, few quantitative predictions have been made for future coral reef ecosystem function. A recent study predicted that a loss of coral reef habitat complexity (i.e., flat reefs), which would occur if carbonate budgets remained strongly negative, would reduce the productivity of Caribbean reef fisheries by at least 3-fold (Rogers *et al*., 2014).

If climate change and ocean acidification continue to follow current trajectories then the outlook for coral reefs is poor (e.g. Figure 10.8). Local conservation actions to manage fisheries and water quality will remain fundamentally important in reducing rates of reef decline and helping build recovery capacity. Even reefs of low resilience, such as those of the Caribbean, are projected to fare better if managed locally, possibly buying a few decades for more assertive action to reduce greenhouse gas emissions (Edwards *et al*., 2011; Kennedy *et al*., 2013). However, if coral cover continues to decline, particularly under frequent coral bleaching, then there is a very real risk that carbonate budgets might become negative. Recent evidence from the Caribbean suggests that at least 10% coral cover is needed to maintain positive reef growth (Perry *et al*., 2013). The longer-term outcome of this is a flattening of reef structures and a decline in key ecosystem services (Pratchett *et al*., 2014). The first services to be affected will be reef fisheries production and biodiversity because both respond rapidly to a loss of habitat structure (Graham *et al*., 2006). In the longer term, the ability of reefs to provide shoreline protection from storms will also decline.

Finally, while shallow-water coral reefs have been chosen as the vulnerable ecosystem of focus in this report, the threats and actions described in this chapter are equally relevant for other vulnerable ecosystems, particularly of those that are relatively less well documented. A case study on identifying vulnerable seamounts for conservation is provided in Box 10.3.

Box 10.3: Identifying vulnerable marine ecosystems for conservation

Conservation of Vulnerable Marine Ecosystems of the deep sea, including cold-water coral (CWC) aggregations and sponge fields, is a global priority (FAO, 2009). Cold-water coral ecosystems have been identified as biodiversity hotspots (Watling *et al*., 2011), and are of significant ecological and economic value (Foley *et al*., 2010). In contrast to tropical reefs, the cold temperatures and inconstant food supply found on the deep-sea floor implies that most of its sessile inhabitants have reduced growth rates, long reproductive cycles and low rates of recruitment. Such life history characteristics imply that cold-water ecosystems have a reduced capacity to recover from any disturbance events (Williams *et al*., 2010). Anthropogenic activities such as bottom fishing and hydrocarbon drilling are among the major threats to these fragile ecosystems (Davies *et al*., 2007; Roberts *et al*., 2009). Cold water reefs or gardens are found mainly on seamounts slopes (Genin *et al*., 1986). To conserve these vulnerable ecosystems, a framework for locating potential ecologically or biologically significant seamount areas based on the best information currently available was developed (Taranto *et al*., 2012; Figure 10.11). This framework combines the likelihood of a seamount constituting an ecologically or biologically significant area (EBSA) and its level of human impact, and can be used to locate priority areas for seamount conservation at global, regional and local scales. This framework will also allow the identification of ecologically or biologically significant seamount areas with high data uncertainty and is thus in urgent need of research. If this knowledge is translated into policy action, the methodology may constitute an important step forward in the implementation of conservation measures in deep sea habitats and open ocean waters and help to fulfill the international commitments signed under the Convention on Biological Diversity.

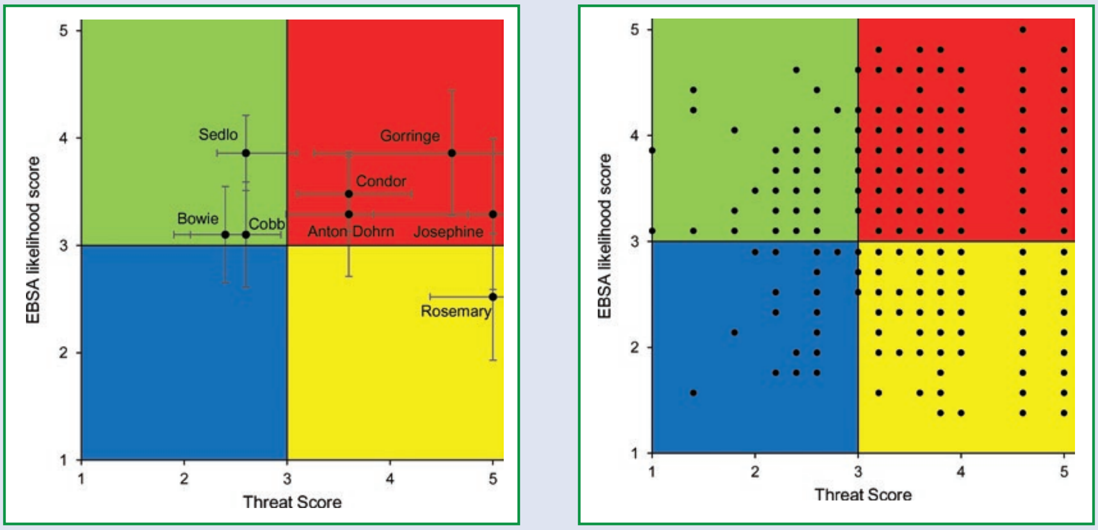


Fig. 10.11. Seamount EBSA portfolio plot based on EBSA likelihood scores and threat scores for eight case studies (left) and for the whole dataset (right). The upper right seamounts (red) are very high and high EBSAs with very high and high threats, while the upper left seamounts (green) are very high and high EBSAs with medium and low threats. In the bottom are the medium and low EBSAs with low (blue) and high (yellow) threats. Error bars represent the data uncertainty index (see methods) proportional to data availability and quality. Source: Text and figure provided by Telmo Morato (see Taranto *et al*., 2012)

10.5 UNCERTAINTIES AND DATA REQUIREMENTS/GAPS

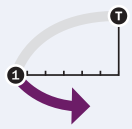
There are many uncertainties in projecting the future biodiversity and functioning of coral reefs (Mumby and Van Woesik, 2014). First, there are uncertainties in the level of stress experienced by reefs. For example, global circulation models (GCMs) need to be downscaled and there remains significant uncertainty over how ocean chemistry is modified in coastal environments. In particular, the metabolism of reef organisms can modify the biogeochemical environment substantially (Kleypas *et al*., 2011; Anthony *et al*., 2013). In other words, natural daily fluctuations in sea water chemistry might be equivalent to the projected impacts of climate change on mean oceanic conditions over many decades. While a process like ocean acidification will slowly reduce the mean oceanic saturation states of aragonite and calcite, it is not clear how such changes influence the dynamic environment experienced by most corals.

A second source of uncertainty is the response of reefs and organisms to a changing environment (Pandolfi *et al*., 2011). The response of individual species to stress varies dramatically (Comeau *et al*., 2013) and while there may be an overall negative trend (Chan and Connolly, 2013), the outcome can be highly variable from one ecological community to another. Moreover, in addition to there being no theoretical impediment to evolution of corals to bleaching (Day *et al*., 2008), some have argued that the high rate of somatic mutation in organisms like corals provides a greater opportunity for genetic adaption than has been previously considered (van Oppen *et al*., 2011).

Third, many physiological studies of climate change response use treatments that simulate environments far into the future. Thus, the response of animals and plants to conditions expected in 2020 and 2050 are difficult to evaluate. One reason for this is the existence of trans-generational plasticity (Franks and Hoffmann, 2012). Most studies of coral reef organismal response to climate change have been undertaken for a single generation. Yet, multi-generational studies are finding considerable scope for altered fitness, possibly through epigenetic effects (Miller *et al*., 2012).

Fourth, although it seems likely that some reefs will slip into net erosive structures (negative carbonate budgets), a loss of reef structure and complexity is not instantaneous. Yet, there has been virtually no attempt to predict the actual rate at which reef structures will erode, particularly in environments with more erosive chemistry (OA). Moreover, changes in reef structure and the emergence of novel reef communities (Yakob and Mumby, 2011) could have surprising ecological outcomes. For example, habitat structure can mediate predator-prey interactions of reef fish (Hixon and Beets, 1993; Syms and Jones, 2000). But habitat complexity could change in surprising ways through a combination of net reef erosion and a shift in coral species composition. How such uncertain shifts might then influence the multitude of fish species' interactions is almost impossible to predict at this stage. Moreover, some coral reef communities are predicted to experience alternate attractors, such that a small increase in stress (e.g., bleaching mortality, nutrients, fishing) could result in a drastic and undesirable change in community structure, including a loss of species, that resists management efforts to reverse the decline (Mumby *et al*., 2013).

10.6 DASHBOARD – PROGRESS TOWARDS TARGET

Element	Current Status	Comments	Confidence
Multiple anthropogenic pressures on coral reefs are minimized, so as to maintain their integrity and functioning		Pressures such as land-based pollution, uncontrolled tourism still increasing, although new marine protected areas may ease overfishing in some reef regions	High
Multiple anthropogenic pressures on other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning	Not evaluated	Insufficient information was available to evaluate the target for other vulnerable ecosystems including seagrass habitats, mangroves and mountains	--

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Dashboard: Tim Hirsch

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APPENDIX

Table 10. Recent initiatives to increase the area of coral reefs under protection.

Initiative	Time Frame	Description/Goal	Area Protected	Outcome Of Initiative	Type Of Initiative
Micronesia Challenge (Established)	2006 – 2020	Effectively conserve at least 30% of near shore marine resources	Not specified	MPAs and regional trust fund for providing sustainable revenue stream	Regional – FSM, Marshall Islands, Palau, Guam, CNMI
Caribbean Challenge (Established)	2007-2020	Protect at least 20% of the near-shore marine and coastal habitats by establishing a network of 20 million acres of marine parks across the territorial waters of at least 10 countries.	20 million acres (80,397 km^2)	National systems of protected areas and sustainable financing tools	Regional- The Bahamas, Dominican Republic, Jamaica, Saint Vincent and the Grenadines, Saint Lucia, Grenada, Antigua and Barbuda as well as Saint Kitts and Nevis. Endorsed by 5
Coral Triangle Initiative (Established)	2009-2020	Place 20% of each major marine and coastal habitat under protected status by 2020	Not specified	Effectively managed linked network of multiple use MPAs, sustainable financing plan	Regional – Philippines, Indonesia, Malaysia, Timor-Leste, Solomon Islands, Papua New Guinea
Western Indian Ocean Coastal Challenge (Pending)	Declared in 2012, Target 2032	Commit to island conservation and sustainable livelihoods, including responding to climate change threats	Not yet determined	Not yet determined	Regional - Comoros, Reunion, Kenya, Madagascar, Mauritius, Mozambique, Seychelles, Zanzibar
Fiji (Established)	2005-2020	Effectively manage and finance at least 30% of inshore areas by 2020	30% of ~ 30,000 km^2 (high water mark to outer barrier reef)	Locally managed marine areas (multiple use)	National - Fiji
Papahanaumokuakea Marine National Monument (Established)	2006	Ecosystem protection, and preserve ecosystem function, key processes, recover resources where necessary	362,075 km^2	World Heritage Site – no take marine reserve	National - USA
Phoenix Islands Protected Area (Established)	2008	Biodiversity conservation while allowing for sustainable economic opportunities	408,250 km^2	UNESCO World Heritage Site	National - Kiribati

Initiative	Time Frame	Description/Goal	Area Protected	Outcome Of Initiative	Type Of Initiative
Marine National Monuments - Pacific Remote Islands, Marianas Trench, Rose Atoll (Established)	2009	Protect and conserve biodiversity	Pacific Remote Islands (199,480 km^2), Marianas Trench (250,487 km^2), Rose Atoll (34,800 km^2)	No-take marine reserve	National - USA
Chagos Marine Reserve (Established)	2010	Biodiversity conservation	640,000 km^2	No-take marine reserve	National - UK
Australian Commonwealth Marine Reserves (Established)	2012	Add to Australia's existing system of marine reserves managed primarily for biodiversity conservation and sustainable use in some areas	2.3 million km^2 – Reserves with coral habitats: Coral Sea (989,482 km^2), North (157,483 km^2), and North-west (335,437 km^2)	Marine reserve with multiple use zones	National - Australia
Indonesia (Established)	2012-2020	Establish and effectively manage 20 million ha of MPAs by 2020	20 million ha (200,000 km^2)	MPA network (multiple use)	National - Indonesia
Primeiras and Segundas Marine Protected Area (Pending)	Declared 2012	Protect marine biodiversity and manage marine resources for sustainable future	10,409 km^2	MPA (multiple use?)	National - Mozambique
Cook Islands Marine Protected Area (Pending)	Proposed 2012	Proposal to protect around 50% of country's EEZ as a multiple use marine park	1,065,000 km^2	Multiple use marine park and establishment of trust fund	National - Cook Islands
New Caledonia Marine Protected Area (Pending)	Declared 2012	Commitment to create MPA under Pacific Oceanscape Programme	1.4 million km^2	Marine Protected Area (multiple use?)	National – New Caledonia
Maldives (Pending)	2013-2017	Pledge for entire country and EEZ to be a Biosphere Reserve by 2017	Size of Maldives EEZ is ~90,000 km^2 (coral reefs and lagoons ~21,300km^2)	UNESCO Biosphere Reserve	National - Maldives

TARGET 11: PROTECTED AREAS AND OTHER EFFECTIVE AREA-BASED MEASURES

By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes.

PREFACE

Aichi Biodiversity Target 11 covers both protected areas and other area-based conservation measures, but we focus primarily on the former in this report. We consider protected area coverage, both geographically and in terms of ecological representation (using WWF ecoregions). The use of ecoregions to assess ecological representativeness of protected areas ignores the considerable ecological variation within these regions, but addressing this shortcoming was beyond the scope of this work. To analyse coverage by protected areas of areas of importance for biodiversity, we focused on Alliance for Zero Extinction sites (AZEs) and Important Bird and Biodiversity Areas (IBAs), because there were good data with which to assess coverage. We also explore protected area effectiveness, in terms of management inputs and biodiversity outcomes, taking into account climate change-induced changes in protected areas in longer term scenarios. Preliminary analyses are also presented on equitable management. Freshwater environments are accorded a relatively large degree of attention given their areal coverage. This is because freshwater environments are poorly represented in terms of data, assessments and protection, thus necessitating a more qualitative treatment, and because of the added complexities of these systems given their inherent connectedness.

11.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

11.1.1 Status and trends

Globally, protected area coverage has increased rapidly in recent years on land and in the sea (Figure 11.1 A, B). Rates have slowed somewhat in recent years (Figure 11.1 A, B), but this is probably an artefact of lags in reporting of new protected areas.

In 2011, 10.9% of global land area was covered by protected areas. In January 2011, 49 of the parties to the Convention on Biological Diversity (23%) had exceeded the target of protecting 17% of terrestrial areas.

Figure 11.1 (overleaf). Recent trends and extrapolations to 2020 in the cumulative percentage of global terrestrial (A) or marine (B) area covered by terrestrial and marine protected areas; in the percentage of terrestrial (C), freshwater (D) and marine (E) ecoregions that meet a threshold level of protection (17% for terrestrial; 10% for marine and freshwater); in the coverage of the distributions of bird, mammal and amphibian species by protected areas (F); in the global cumulative number of protected area management effectiveness assessments (G); and in funding for protected areas (H). Data from recent trends are indicated by points, continuous lines indicate the fit to data, dashed lines are extrapolations to 2020 and dotted lines indicate 95% confidence intervals. Horizontal grey line represents model-estimated 2010 value for indicator. Source: World Database on Protected Areas (WDPA ; http://protectedplanet.net/) (A-B); S. H. M. Butchart *et al*. (unpublished data) (C-F); J. Geldmann *et al*. (unpublished data) (G); and AidData (http://aiddata.org/) (H). Extrapolations are based on the assumption that underlying mechanisms continue to follow trends. Methods for model fitting are described in the introductory chapter.

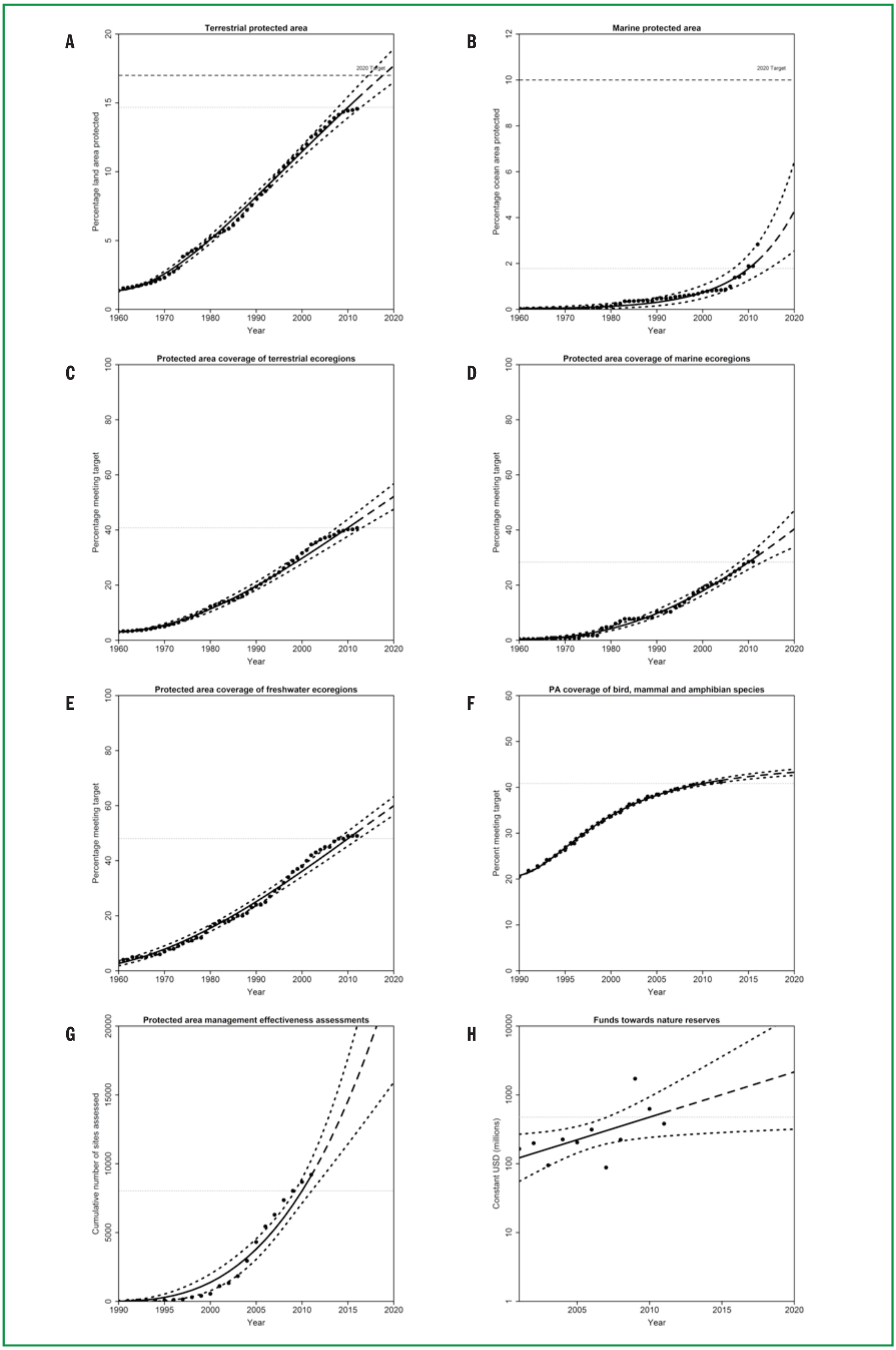

A
Terrestrial protected area
2020 Target
Percentage land area protected
Year
B
Marine protected area
2020 Target
Percentage ocean area protected
Year
C
Protected area coverage of terrestrial ecoregions
Percentage meeting target
Year
D
Protected area coverage of marine ecoregions
Percentage meeting target
Year
E
Protected area coverage of freshwater ecoregions
Percentage meeting target
Year
F
PA coverage of bird, mammal and amphibian species
Percent meeting target
Year
G
Protected area management effectiveness assessments
Cumulative number of sites assessed
Year
H
Funds towards nature reserves
Constant USD (millions)
Year

Figure 11.1

In 2011, 2.3% of the global marine surface area was represented by protected areas. Since 2010, the number of countries and territories (out of 193 with some marine jurisdiction) with 10% or more of their marine jurisdictional area incorporated into marine protected areas increased from 12 to 28 (Spalding *et al*., 2013). On the other hand, 111 have less than 1% coverage by marine protected areas (MPAs) (Spalding *et al*., 2013). It should be noted that only a small number of MPAs are responsible for most of the existing global coverage (Devillers *et al*., 2014). Furthermore, conservation progress may not be as great as it appears because many MPAs are placed where they minimise conflict with stakeholders, rather than where biodiversity is most threatened (Devillers *et al*., 2014). The majority of MPAs are situated within jurisdictional waters, and coverage of the high seas remains low (Spalding *et al*., 2013).

Establishment of high seas MPAs is limited because the international legal framework currently has inadequate enforcement mechanisms for ensuring compliance with conservation and management regulations in areas beyond national jurisdiction (Kimball, 2005). Extensive protection of the high seas only began in 2010, with the declaration of the South Orkney Islands Southern Shelf MPA and six OSPAR (Convention for the Protection of the Marine Environment of the North-East Atlantic) MPAs in the North Atlantic (Spalding *et al*., 2013). The need for conservation of biodiversity in the high seas was recognised at the 2012 United Nations Conference on Sustainable Development, at which government leaders considered the possible development of a new legal instrument under the United Nations Convention on the Law of the Sea (Ban *et al*., 2014). While there is, as yet, no global agreement to establish MPAs in areas beyond national jurisdiction (Kimball, 2005), the United Nations General Assembly has called for the protection of vulnerable marine ecosystems in the high seas[1]. Importantly, some authors have noted the need for more ecologically representative systems of MPAs in areas beyond national jurisdiction (Ban *et al*., 2014; Freestone, 2012). For instance, the Global Open Oceans and Deep Seabed Biogeographic Classification system classifies open oceans and deep sea habitats within and beyond the continental shelf (UNESCO, 2009).

In 2010, 17% of the world's total river length was protected. The evaluation of protection afforded to inland waters is more complicated than simply summing the total area protected. Given the longitudinal nature of rivers and streams, and their interconnections, it is important to consider not only the total area or length of inland waters protected, but to also quantify the amount of river or stream protected upstream (Abell *et al*., 2007; Linke *et al*., 2007; Nel *et al*., 2007; Januchowski-Hartley *et al*., 2011). Reporting on the protection of inland waters has been hampered by this complexity, and to the best of our knowledge no comprehensive assessment of national level protection of inland waters exists. Globally, 69% of rivers have no protected areas in their upstream catchment, and only South and Central America have greater than 10% of total upstream catchment area protected (with 26% in South America and 12% in Central America; Lehner, B. *et al*., unpublished data). Regions with the lowest percentage of river length protected include Asia and North America (11 and 12% protected, respectively), while the poorest protection of upstream catchment area is in Europe and the Middle East, and in North America (less than 7% protected; Lehner, B. *et al*., unpublished data).

Protected area coverage has also represented a growing number of the world's ecoregions: currently 55% of terrestrial ecoregions and 32% of marine ecoregions have at least 10% coverage (Figure 11.1 C, D; Figure 11.2), and 7% of terrestrial and 7% of marine ecoregions have at least 75% coverage (S. H. M. Butchart *et al*., unpublished data). On the other hand, 7% of terrestrial and 28% of marine ecoregions have less than 1% coverage by protected areas (S. H. M. Butchart *et al*., unpublished data); 49% of freshwater ecoregions have at least 10% coverage (Figure 11.1 E), but 8% of freshwater ecoregions have less than 1% coverage (Januchowski-Hartley, unpublished data). Many of the poorly protected freshwater ecoregions occur in areas of North America, islands in the Pacific Ocean, and in xeric or endorheic basins where inland waters are often temporary. Protected area coverage varies widely across ecoregions (Figure 11.2).

Footnote

[1] *Protection of VMEs was first called for in Res 59/25 and subsequently reaffirmed by additional resolutions, most notably Resolutions 61/105 and 64/72: UNGA Resolution 59/25 (paragraphs 66 – 69) http://daccess-dds-ny.un.org/doc/UNDOC/GEN/N04/477/70/PDF/N0447770.pdf?OpenElement; UNGA Resolution 61/105 (paragraphs 10, 80-83, 88-90) http://daccess-dds-ny.un.org/doc/UNDOC/GEN/N06/500/73/PDF/N0650073.pdf?OpenElement; UNGA Resolution 64/72 (para 77, 113-117, 119-123, 124, 126). http://daccess-dds-ny.un.org/doc/UNDOC/GEN/N09/466/15/PDF/N0946615.pdf?OpenElement.*

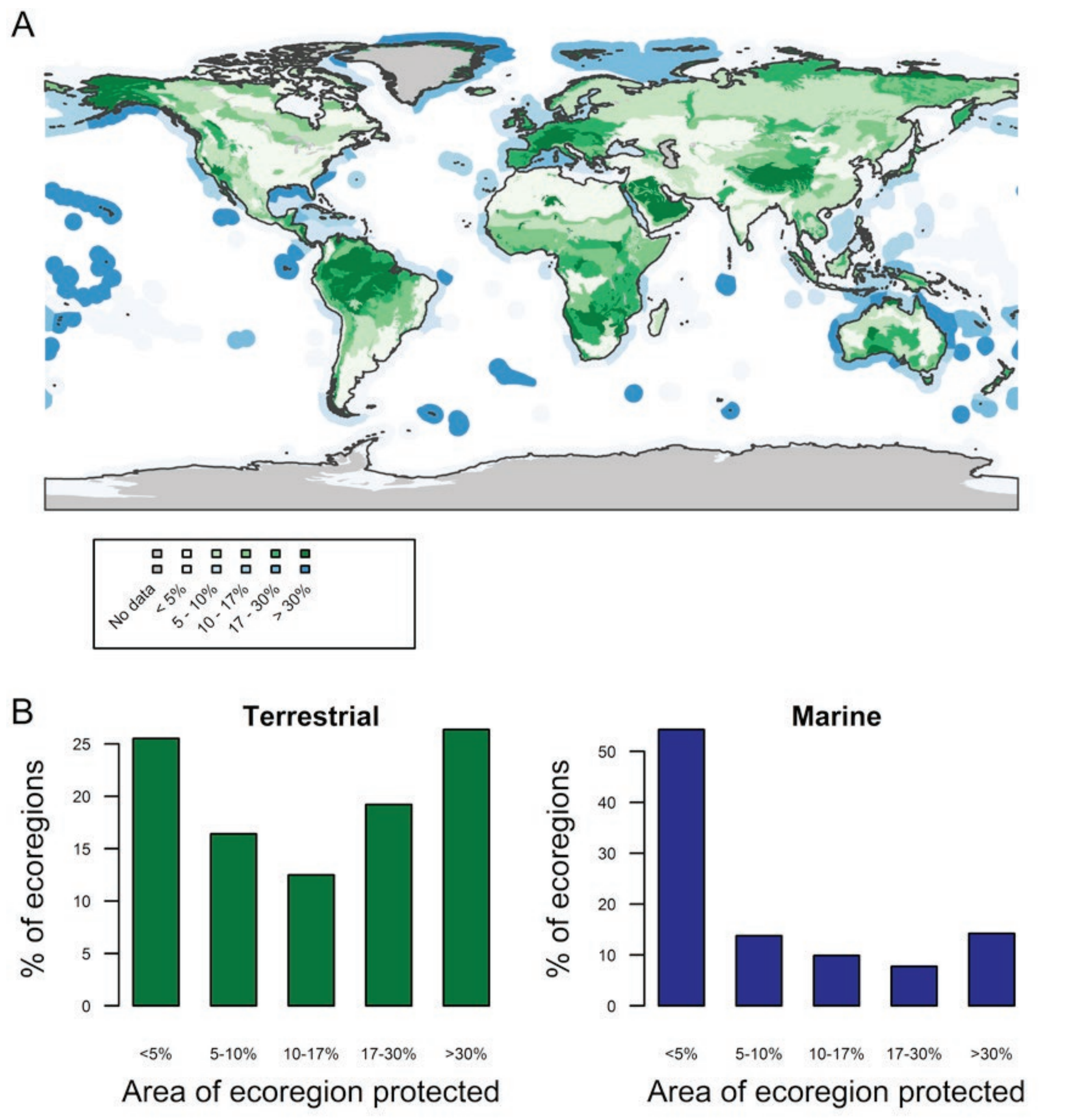


Figure 11.2. A) Percentage coverage by protected areas of marine and terrestrial ecoregions; B) percentage of ecoregions within different bands of areal coverage. Source: S.H.M. Butchart (unpublished data) and terresitrial and marine ecoregions (WWF; http://maps.tnc.org/gis_data.html)

Areas of particular importance for biodiversity or Key Biodiversity Areas (Langhammer *et al.*, 2007) have been increasingly well represented over the last 100 years. We focus here on Alliance for Zero Extinction sites (AZEs), which are sites holding the last remaining population of one or more highly threatened species, and Important Bird and Biodiversity Areas (IBAs), which are sites of significance for the persistence of bird (and other) diversity. Globally, 23% of AZEs and 22% of IBAs fall entirely within protected areas (see cross-reference to Target 12; Butchart *et al.*, 2012; updated by S. H. M. Butchart, unpublished data). Sites of importance for biodiversity are often used to inform plans for designating protected areas (e.g., many European countries have explicitly based new protected area designations on published inventories of Important Bird and Biodiversity Areas; BirdLife International 2013). However, this is not always the case and the rate at which such sites are protected is falling relative to the overall rate of growth of protected areas (Butchart *et al.*, 2012; Cantú-Salazar *et al.*, 2013). The coverage of the distributions of bird, mammal and amphibian species by protected areas has increased over the last two decades; currently 37.5% of species meet target levels of protection (Butchart *et al.*, 2012; Figure 11.1 F). For freshwater environments, at least 25% of the Amazon River and its associated tributary length are protected in the current network of protected areas (Lehner, B. *et al.*, unpublished data). The protection afforded in the Amazon Basin is important for the security of freshwater biodiversity as the river and its tributaries supports a higher number of freshwater-dependent species than any other larger river system in the world (Collen *et al.*, 2013). However, basins in the Southeast United States and Southeast Asia also support high levels of freshwater biodiversity (Collen *et al.*, 2013), but have less than 10% of total river length protected, and in a number of cases (e.g., coastal basins along the Gulf of Mexico) have less than 5% of river length protected. In addition, many of these basins with high species richness and low protection are subject to high levels of human impact (e.g., Vörösmarty *et al.*, 2010), suggesting the need for further protection and conservation actions to mitigate these stressors (see Targets 5 and 8).

Box 11.1: Global coverage of IUCN protected areas

The IUCN classification of protected areas ranges from the strictest levels of protection, categories I (Wilderness area and Strict nature reserve) and II (National Park), to categories where some human intervention and resource use are allowed, categories III to VI (Natural monument, Habitat management area, Protected landscape, Managed resource protected area). The World Database of Protected Areas reports the IUCN classification level for the majority of protected areas. However, there are almost 38% of protected areas for which no IUCN level is reported; even for those areas for which the category is reported, the accuracy is limited. Nevertheless, an examination of the distribution of the different protected area categories reveals different socioeconomic contexts, opportunity costs and historical perspectives across the world.

North America's pioneering of national parks is still visible in the high coverage of protected areas in category I and II, particularly in the Western part of the continent (Figure 11.3A). Conversely, Europe has focused in the last decades on protected areas managed for specific species or habitats (European Council, 1979, 1992). Additionally, cultural aspects related to rural lifestyles are emphasized in the management plans of many European protected areas. This has led to a higher coverage of protected areas in categories III-VI than in categories I-II, and in several ecoregions the protection of wilderness is lower than 2.5% (Figure 11.3).

Some world ecoregions with high categories I and II coverage coincide with low human population densities such as the most northern latitudes of North America and much of Australia. However, in South America, sub-Saharan Africa and Southeast Asia, where both conservation efforts and human population pressures are high (McKee et al., 2004; Brooks et al., 2006), we encounter relatively high area coverage of protected areas in all categories (Figure 11.3).

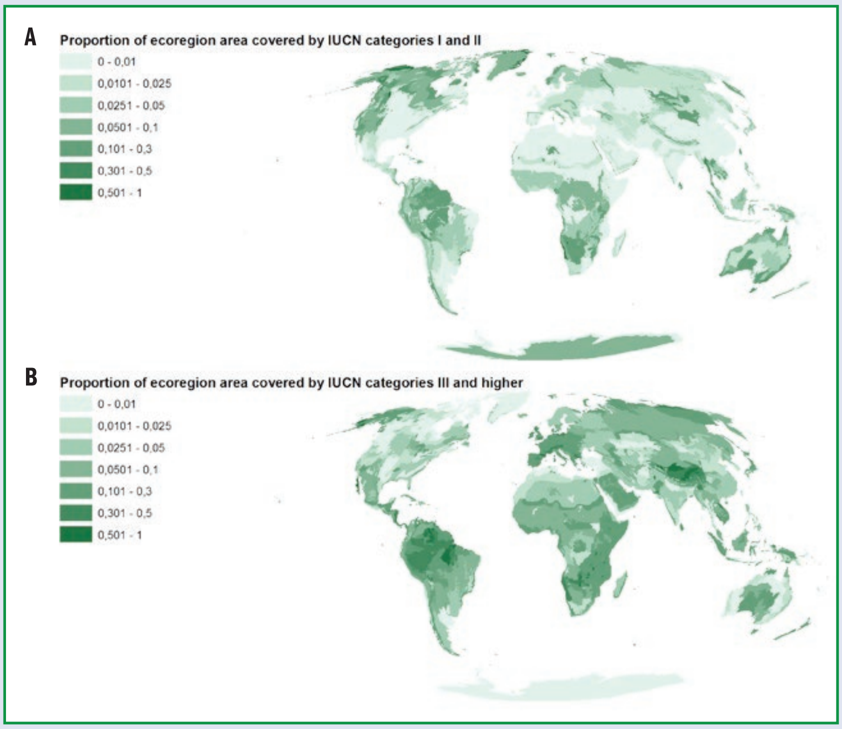


Figure 11.3. The distribution across world ecoregions of coverage of protected areas in IUCN categories I and II (A) and in all other categories (III, IV, V, VI) and without information on category (B). Colours represent the proportion of each ecoregion's land surface covered by protected areas. Source: World Database of Protected Areas (UNEP-WCMC; http://protectedplanet.net/) and Terrestrial Ecoregions (WWF; http://maps.tnc.org/gis_data.html).

It is also important to know the coverage of areas of importance for ecosystem services by protected areas. However, insufficient information exists at present to assess this.

Available evidence suggests that there has been a rapid increase in the last 20 years in private, community-based and co-managed (communities with some combination of national or subnational government and/or private company) approaches (Blomley *et al*., 2008; Bowler *et al*., 2010; Bertzky *et al*., 2012; Weeks *et al*., 2010; Stolton *et al*., 2014); although data on these governance types are not comprehensive globally and often not reported by national protected area authorities (Stolton *et al*., 2014). On average, community-managed forests have been shown to more effectively reduce rates of deforestation than the large protected areas officially recognized by IUCN (Porter-Bolland *et al*., 2012). In the marine realm, Locally Managed Marine Protected Areas (LMMAs) contribute much of the protection afforded to coral reefs, mangroves and sea grasses (Visconti *et al*., 2013). In Fiji, LMMAs protected 40% of fringing reefs, non-fringing reefs, mangroves, intertidal zones and other benthic substrata (Mills *et al*., 2011); they are also important in the Philippines, Japan (Makino *et al*., 2009) and elsewhere in Southeast Asia. Locally managed freshwater protected areas are common across areas of Southeast Asia and parts of South America, but there are no reliable statistics because these areas are highly underreported.

Protected areas ought to be more effective if they are better managed and more effectively enforced. Assessments of management inputs and actions, as measured using various management effectiveness tools (Leverington *et al*., 2010), have increased dramatically over the past decade (Figure 11.1 G), with over 8,000 sites now assessed and hundreds being added each year, particularly in regions where the Global Environment Facility is actively supporting protected-area projects. The apparent levelling of this trend is almost certainly an artefact of lags in reporting. Results from different protected areas show a very wide range of scores, and a recent assessment of 4,100 protected areas designated 13% as having 'clearly inadequate' management, 62% as having 'basic management' and 24% as having 'sound management' (Leverington *et al*., 2010). However, repeat assessments suggest that management effectiveness scores are generally increasing over time (Leverington *et al*., 2010). There is no global assessment of marine protected area (MPA) effectiveness. Many MPAs are less effective than intended due to management problems or poor spatial selection or design (Spalding *et al*., 2013), and a recent assessment of 1,147 coral reef MPAs worldwide found that almost half (47%) were ineffective, while only 15% were considered fully effective, and 38% were partially effective (Burke *et al*., 2011).

Effective management of protected areas relies, at least in part, on adequate funding. There has been no clear recent trend in funding allocated to protected areas (Figure 11.1 H).

Effectivenness is likely to be increased if species are able to move among protected areas, especially in the face of climate change. For mammals, the scale of connectivity in networks of protected areas differs among species groups, because large species move across wide areas and can reach protected areas that are far apart (Figure 11.4). Nonetheless large mammals are more at risk of extinction than small ones (Schipper *et al*., 2008), suggesting that higher connectivity is not sufficient to balance the disproportionate effect of threatening processes on these species. Connectivity is also uneven across continents, with North and South America having the most connected networks. Europe's protected areas – although at a high density – are small on average, and connectivity is lower than it would be if the protected areas were larger. The protected area network in Asia is poorly connected for all mammals, including the highly threatened ungulates and primates.

In recognition of a general lack of connectivity, there are a large number of initiatives around the word that are aiming to develop corridors between protected areas to allow movement of animals (and plants). For example, recent work in South Africa has identified that corridor networks that allow long-distance movement of large mammals are important for conserving plant species distributions and long-distance inter-population seed dispersal (Potts *et al*., 2013). Similarly, there are several 'flyway' initiatives attempting to ensure protection for the migratory routes of birds (e.g., http://www.eaaflyway.net/).

Optimizing connectivity is also a crucial consideration in the design of marine protected area (MPA) networks because it ensures larval exchange and the replenishment of biodiversity in areas affected by natural or anthropogenic disturbances. In 2008, 18% of MPAs (54% by area) were estimated to be part of a network (Wood *et al*., 2008). One key challenge in assessing the connectivity of MPAs is understanding patterns of larval dispersal (Almany *et al*., 2007; McLeod *et al*., 2009; Burgess *et al*., 2014). To accommodate larval dispersal and movement of mobile species, it has been recommended that entire ecological units be included in MPA network design (Salm *et al*., 2001). Using a system-wide approach that recognizes the connectivity between ecosystems (e.g., mangroves, seagrass beds, and reefs) can also help to maintain ecosystem function and resilience (Mumby *et al*., 2006).

Connectivity between reserves is of particular importance for protecting and maintaining populations of freshwater-dependent species (Pringle, 2001; Fausch *et al.*, 2002; Fullerton *et al.*, 2010; Hermoso *et al.* 2012; Simaika *et al.* 2013).

In general, more data and analysis is needed on how protected areas and other site-specific areas are integrated to wider landscapes and seascapes, including socioecological production landscapes and seascapes in all regions.

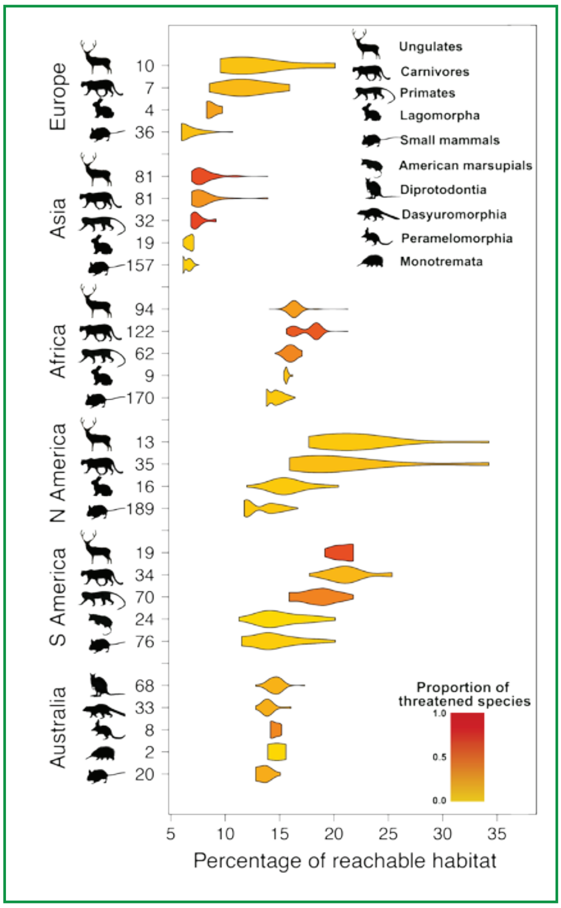


Figure 11.4. Connectivity of protected areas for different mammal groups in each continent, measured as percentage of suitable habitat that species can reach within and across protected areas. Numbers of species per continent are reported next to each animal picture; bar thickness represent the proportion of species. Colour shading represents the percentage of threatened species from 0 (yellow) to 100% (red) (see floating bar for colour reference). Source: Santini *et al.*, unpublished data.

11.1.2 Projecting forward to 2020

Extrapolations of recent trends in protected areas establishment do not reach the target of 10% of the total marine area protected by 2020 (Figure 11.1 B). However, extrapolations for the terrestrial area and a consideration of countries stated commitments on establishment of terrestrial protected areas (UNEP/CBD/WG---RI/4/INF/5) both suggest that the target of having 17% of the total terrestrial area protected by 2020 will be met (Figure 11.1 A). It is unlikely that all ecoregions will meet the sub-target of 10% coverage by 2020 (Figure 11.1 C-E). The coverage of the distributions of bird, mammal and amphibian species by protected areas is not likely to increase substantially by 2020 (Figure 11.1). More than 80% of Alliance for Zero Extinction sites (459 sites) and 70% of Important Bird and Biodiversity Areas (8,106 sites) require additional protection if they are to be fully included in the protected area estate (Butchart *et al.*, 2012; see also chapter 12). There is currently no complementary data for inland waters, which limits extrapolation of protection for these systems.

Several socioeconomic scenarios have been developed to meet the 2020 target for the terrestrial realm. In these scenarios, the coverage of protected areas is set to meet the target of at least 17% of land surface by 2020 within known socioeconomic constraints, showing that achieving the target is realistic. For example, in the Rio+20 scenarios (see

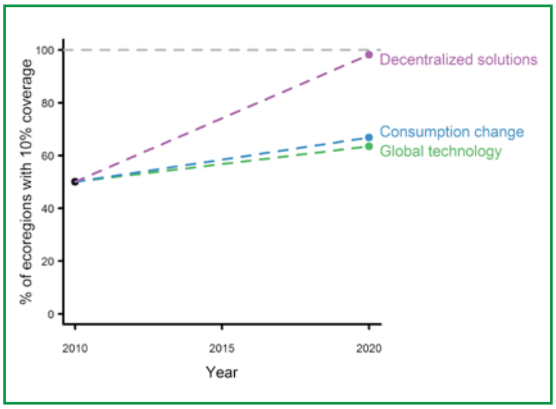


Figure 11.5. Predicted percentage of terrestrial ecoregions having more than 10% coverage by protected areas under the Rio+20 scenarios. Source: The Netherlands Environmental Assessment Agency (PBL)

introductory chapter), the best representation of ecoregions is achieved in the 'Decentralised Solutions' scenario (Figure 11.5), which is designed to protect all ecoregions. However, some of the ecoregions will not achieve 10% protection because conversion from agricultural areas to protected areas is assumed to be unrealistic and ecoregions in desert or ice biomes are assumed not to need explicit protection. The assumption that deserts do not need protection is unjustified given the uniqueness of the flora and fauna they contain and the threats they face (Durant *et al.*, 2014). A less geographically balanced effort to increase protected area coverage (Rio+20 'Global Technology' scenario, focusing on protecting 17% of biomes) results in percentages of ecoregions meeting the target that are essentially equal to the current status (Figure 11.5). Note that these scenarios assume effective management of protected areas, and are based on a different baseline value for 2010 than the status and trends work.

It is far harder to project how management effectiveness will change between now and 2020 owing to a shortage of effectiveness assessments, and our limited understanding of what makes a protected area effective.

1.c. Country actions and commitments[2,3]

Almost all of the national biodiversity strategies and action plans (NBSAPs) examined contain targets, or similar elements, related to protected areas. These targets are largely in line with Aichi Biodiversity Target 11. Generally the emphasis of the targets is on increasing the size of protected area systems. A few countries, for example Belgium, Japan and Finland have set targets which call for increases to the size of protected areas. However, most countries have not specified a specific quantitative target related to protected area coverage. Further, there appears to be a general focus on terrestrial environments.

A number of countries, such as Myanmar and Suriname, have chosen to focus on improving the management or effectiveness of their existing protected areas estate. However, there appears to be relatively less attention to this issue in most countries. Similarly, few targets explicitly address the connection or integration of protected areas into wider landscapes and seascapes. However, Colombia is linking the further development and consolidation of its system of protected areas with wider land-use planning in order to promote ecological connectivity. Australia has set a target of establishing four collaborative continental-scale linkages to improve ecological connectivity by 2015.

Few targets explicitly address issues related to ecological representativeness. Similarly, relatively few targets explicitly refer to protecting areas which are particularly important for biodiversity. One example that is counter to this general trend is Brazil, which among other things has committed to protecting 30% of the Amazon.

In addition to NBSAPs, many countries have developed plans to address gaps in their protected-area systems. In fact, 72 countries have identified 197 priority actions within Protected Area Action Plans formally submitted to the Secretariat relating to PoWPA goal 1.1: "To establish and strengthen national and regional systems of protected areas integrated into a global network as a contribution to globally agreed goals"[3]. Examples of countries with such plans include but are not limited to: South Africa, Mexico, Peru, Colombia, Argentina, Costa Rica, Croatia, Yemen, Guatemala, Brazil, Cook Islands, Kiribati, India, Burundi, and Palau.

Organizations, networks and initiatives other than national governments are also playing an important role in achieving Aichi Biodiversity Target 11. For example, the Satoyama Initiative was established by Decision X/32 of CBD COP10 for the conservation of biodiversity in and around protected areas focusing on socioecological production landscapes and seascapes (SEPLS). These may exist both inside and outside of protected areas, and often serve as vital buffer zones and corridors to integrate protected areas into the wider landscape or seascape. These can serve to complement protected areas.

Overall these national targets or similar commitments will make a substantial contribution towards the attainment of Aichi Biodiversity Target 11 (medium). The diversity of the formulation of national targets is likely a reflection of different national circumstances and the different elements contained in the global target. In general it appears that a greater attention to management effectiveness and ecological representativeness may be needed if this target is to be met by 2020.

Footnotes

[2] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

[3] *PoWPA action plans can be accessed at http://www.cbd.int/protected/implementation/actionplans/*

11.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

11.2.1 Actions

To achieve the target of protecting 17% of terrestrial areas will require coverage to be increased by 5.5 million km^2; to do so in an ecologically representative way will require 10.8 million km^2 (Ervin and Gidda, 2012). To cover 10% of all marine areas will require 27.8 million km^2 of additional area, but only 2.9 million km^2 or 425 000 km^2 to achieve the target in waters up to 200 and 12 nautical miles of shorelines, respectively (L. Teh, unpublished data). While there are limited data to identify what it would take to effectively achieve the 17% target for inland waters, the additional complexity of protecting upstream areas for inland waters suggests that it could require greater (or at least different areas) than are needed to meet the target for terrestrial environments (Abell *et al.*, 2007; B. Lehner *et al.*, unpublished data).

It is also necessary that protected areas are effectively managed. In order to achieve this will require more effort to assess the effectiveness of protected areas and to ensure that appropriate management practices are put in place.

11.2.2 Costs and cost-benefit analysis

The High Level Panel report (Ervin and Gidda, 2012) estimated that to achieve the target will cost by 2020 a total of between US$73.8 billion and US$679.9 billion (US$9.2 billion to US$85.0 billion annually), through: (a) creating new protected areas (US$44.2 billion to US$278.6 billion); (b) establishing connectivity corridors (US$21.3 billion to US$344.8 billion); (c) effectively managing new and existing protected areas (US$7.7 billion to US$53.5 billion); (d) strengthening protected area enabling environments and sustainable finance (US$0.5 billion to US$2.9 billion); and (e) conducting key protected areas assessments (US$25 million to US$78 million). Balmford *et al.* (2002) suggest a lower figure of US$45 billion for an effective network of marine and terrestrial protected areas. On the other hand estimates assuming an ecologically representative network arrive at estimates toward the upper end of the estimates from the High Level Panel report: to represent and effectively manage areas of importance for biodiversity (specifically Key Biodiversity Areas; Langhammer *et al.*, 2007) is estimated to cost US$76.1 billion annually (McCarthy *et al.*, 2012). Larger protected areas are likely to be more cost-effective in terms of both establishment (McRae-Strub *et al.*, 2011) and effective management (Ervin and Gidda, 2012).

There are several benefits of investment in protected areas apart from biodiversity conservation, including water security, food security, hazard mitigation, health and climate-change mitigation (Balmford *et al.*, 2002; Scharlemann *et al.*, 2010; Meyerhoff *et al.*, 2012). The return on investment in terrestrial protected areas has been estimated at between 7:1 and 100:1 (Balmford *et al.*, 2002; Ervin and Gidda, 2012; Meyerhoff *et al.*, 2012; High Level Panel, 2013). There has been no comprehensive cost-benefit analysis for marine protected areas owing to the difficulty of predicting and estimating the economic benefits of future marine protected areas. However, regional studies suggest that investments will yield positive economic outcomes, with estimated returns on investment between 1.8:1 and 41.5:1 (van Beukring and Ceasar, 2004; Pham *et al.*, 2005; Pascal, 2011; High Level Panel, 2013). Furthermore, it has been shown that economic benefits from fisheries and tourism are greater after reserve establishment than before (Sala *et al.*, 2012).

11.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

The successful achievement of area-based targets for protected areas does not guarantee a desirable outcome in terms of biodiversity conservation. Recent estimates confirm that the current global network of terrestrial protected areas still falls short of adequately representing biodiversity (Butchart *et al.*, 2012; Cantú-Salazar *et al.*, 2013). Overall, evidence suggests that existing protected areas tend to have a positive effect on natural land cover, although results vary widely across different reserves (Joppa and Pfaff, 2011; Geldmann *et al.* 2013; Kadoya *et al.*, 2014). In terms of conserving species diversity, results have been much more mixed, with the majority of protected areas seeing ongoing declines in plant and animal populations, although at lower rates than in surrounding areas (Craigie *et al.*, 2010; Laurance *et al.*, 2012; Geldmann *et al.* 2013; Kadoya *et al.*, 2014). Other approaches have shown that extinction risk was lower and increased more slowly for species for which most or all important sites were protected compared to those for which fewer or no sites were protected (Butchart *et al.*, 2012).

It is expected that the effective management of protected areas leads to improvements in the status of biodiversity within them. Although there is little reported evidence of the relationships between management interventions and conservation outcomes for terrestrial protected areas, one recent review of 35 studies did reveal that targeted interventions (anti-poaching etc.) had a positive effect in over 80% of cases (Geldmann *et al.*, 2013).

In the marine realm, poor design and management of many marine protected areas (MPAs) means that they currently have a minimal effect on achieving marine biodiversity conservation (Carey *et al.*, 2000). However, there is strong evidence that well-managed marine protected areas can have positive effects on biodiversity: recent studies show that several measures of biodiversity are substantially improved compared either with the same area before the establishment of the reserve or with unprotected areas nearby (Lester *et al.*, 2009; Babcock *et al.*, 2010). A recent review suggested that MPAs that are large, old, isolated, well-enforced and have no fishing permitted are more effective at conserving biodiversity than other MPAs (Edgar *et al.*, 2014). Locally Managed Marine Protected Areas have been shown to have effective outcomes for benthic habitats (Mills *et al.*, 2011) and in securing the abundance of fish species (Almany *et al.*, 2013).

Inland waters are likely to be the least effectively managed environments because there are few targeted protected areas for inland waters, and in many cases where protection does exist (e.g., Ramsar sites) upstream areas are not protected or managed in a way that will effectively abate threats (Abell *et al.* 2007; Januchowski-Hartley *et al.*, 2011; Chessman, 2013). Furthermore, the pervasiveness of in-stream barriers can prevent fish movement into and out of protected areas (Januchowski-Hartley *et al.*, 2011; 2013). Regional-scale assessments of the coverage and effectiveness of protected areas have shown that freshwaters are not only under-protected, but that the placement of protected areas is ineffective for conserving freshwater habitats and species (Herbert *et al.*, 2010; Januchowski-Hartley *et al.*, 2011; Chessman, 2013).

Results of modelling analyses suggest that expansion of the world's protected areas network will have a positive effect on biodiversity (Figure 11.6; see also Target 12). Expanding protected areas to 20% of land surface area could lead to a net reduction in biodiversity loss by 2030 compared to a baseline 'business-as-usual' scenario (Figure 11.6, bottom bar). This net effect is comprised of a positive effect owing to reduction of habitat modification inside protected areas compared to the baseline scenario (Figure 11.6, top bar), and an indirect negative effects primarily related to the displacement of agricultural activity from newly protected areas (Figure 11.6 middle bar).

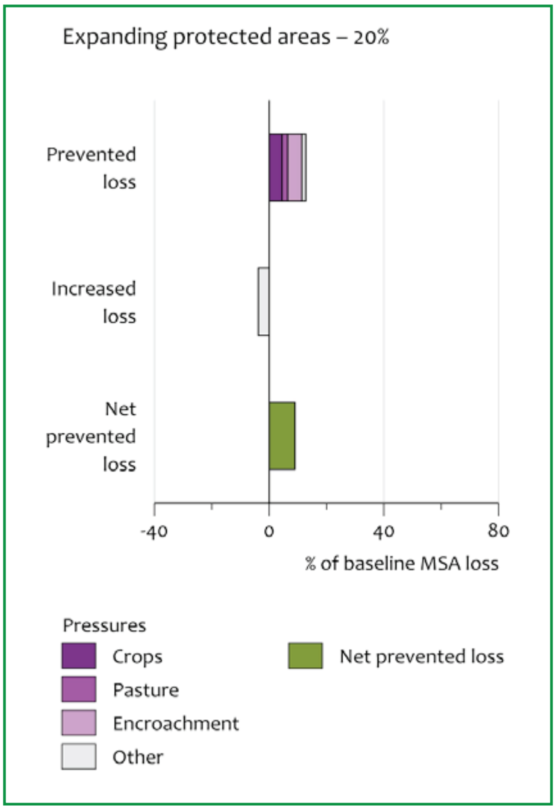


Figure 11.6. Consequences for projected biodiversity (measured as Mean Species Abundance; MSA) in 2030 of expanding the terrestrial protected area coverage to 20% of the terrestrial surface, compared to a baseline scenario where the existing network of protected areas is unchanged. Increased loss is caused by transfer of agricultural activity to non-protected areas. Source: Netherlands Environmental Assessment Agency (2010).

11.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

In all scenarios, habitat loss and fragmentation, pollution and existing roads continue to negatively affect biodiversity in terrestrial protected areas until 2050, but climate change will become an increasingly important threat.

Terrestrial scenarios that include reductions of these pressures in addition to increasing protected areas (the Rio+20 scenarios; see introductory chapter) are much more efficient in reducing biodiversity loss than scenarios that focus on protected areas alone (see Target 12). Comparisons of several development options suggest that increasing the coverage of protected areas to 20% has modest but important effects on reducing biodiversity loss that are similar in magnitude to reducing deforestation to low levels or strongly limiting the use of biofuels, but are smaller than the effects of changing dietary consumption patterns or reducing agricultural waste (see chapter 21).

Towards the middle of the century, species are expected to respond to climate change through changes in their physiology, phenology and distribution (Bellard *et al.*, 2012), leading to species range shifts, changes in community composition, vegetation structure and ecosystem function (e.g. Thuiller *et al.*, 2005; Araujo *et al.*, 2006; Araujo *et al.*, 2011; Schloss *et al.*, 2011; Hickler *et al.*, 2012). There is now strong observational evidence that mobile species such as insects and birds have responded to climate warming over the last several decades by moving at rates of approximately 17 km/decade towards the poles (Chen *et al.*, 2011).

This might make the existing network of protected areas less representative of biodiversity (Hole *et al.*, 2011). Future conservation efforts, including the designation of protected areas, which currently often take a static view of biodiversity, need to account for these changes (Araujo *et al.*, 2011; Strange *et al.*, 2011). By 2080, some models suggest that suitable climate area will be lost for about 50% of species in protected areas in Europe, and for nearly two-thirds of species currently protected in Natura 2000 areas (Araujo *et al.*, 2011). Considerable regional differences can be observed, with alpine and sub-arctic species particularly strongly affected. Similarly in Asia (Figure 11.7), it is predicted that suitable climate area in Important Bird and Biodiversity Areas (IBAs) will decrease for nearly half of bird species of current conservation concern by 2085, although the IBA network as a whole will remain critically important for representing the distributions of these species (Bagchi *et al.*, 2013). In sub-Saharan Africa, although suitable climate will persist until 2085 for most species within the IBA network as a whole, a considerable turnover of species (> 75%) is predicted for nearly half of IBAs. Considerable regional differences in species turnover are shown, with priority species mainly affected in the wet savanna (Miombo) regions of East and Southern Africa (Hole *et al.*, 2009). Climate change is projected to reduce the overall effectiveness of the IBA network in Southern Africa (Coetzee *et al.*, 2009).

Protected areas might play an important role as stepping stones and establishment centres for species spreading to new habitats (Hiley *et al.*, 2013; Lawrence *et al.*, 2011), provided that dispersal across these landscapes is not prevented by other changes such as habitat loss and fragmentation (Beaumont and Duursma, 2012; Hamilton *et al.*, 2013).

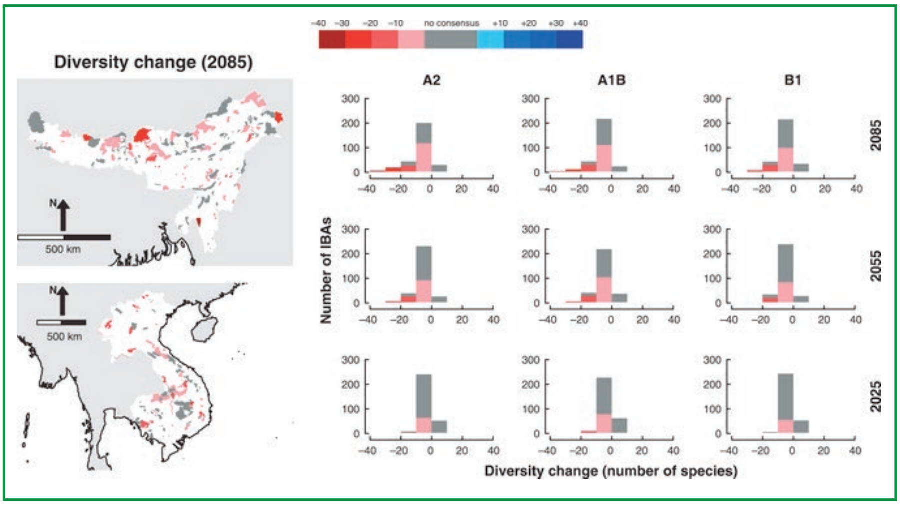


Figure 11.7. (a) Projected changes in number of species of conservation concern and (b) percentage species turnover by 2085 in Important Bird Areas of the Eastern Himalayas (top) and lower Mekong (bottom). Projections are based on a strong greenhouse gas emissions scenario (IPCC SRES A2). Source: Bagchi *et al.* (2013), Global Change Biology.

To minimize the impacts of climate change on the effectiveness of terrestrial protected areas, a number of measures have been suggested (Hannah *et al.*, 2007; Hannah, 2010; Araujo *et al.*, 2011; Carvalho *et al.*, 2011; Hole *et al.*, 2011; Lemieux *et al.*, 2011; Kingsford, 2011; Beaumont, 2012; Bagchi *et al.*, 2013; IGES, 2013). These include:

- Designation of protected areas to include regions where species of special concern are projected to occur in future. This will require regional and continental scale cooperation.
- Maximizing representation of environments in a given region, for example by including altitudinal or latitudinal gradients within protected areas or protected-area networks.
- Implementation of mechanisms for integrated landscape management to facilitate movement of species between conservation areas.
- Climate adaptation strategies on conservation sites.
- Restoration of critical habitats.
- Reduction of non-climate pressures.
- Retention of natural vegetation within human-inhabited landscapes to secure ecological networks.

Gillson *et al*. (2013) suggest that conservation strategies should not only be based on predicted climate-driven range shifts, and proposed a conservation and management prioritization framework based on landscape conservation capacity attributes in addition to species vulnerability to climate change.

For inland waters, climate change could exacerbate the negative effects of drying conditions that are currently a natural feature of many temporary river systems (Hermoso *et al*., 2012). Coupled with existing and growing threats from dams and water extraction, this could affect the distribution and movement of freshwater biodiversity (Bates *et al*., 2008; Morrongiello *et al*., 2011). Therefore, it will be essential to protect refugia to maintain individuals that can repopulate a wider range of habitats when more favourable conditions are restored after seasonal or prolonged droughts (Larned *et al*., 2010). Minimizing and managing upstream and downstream threats from changes in human land use, expansions of dams (e.g., Lehner *et al*., 2008; Vörösmarty *et al*., 2010) and water extraction will also be critical for protected areas to be effective for inland waters and the species that they support.

The ability of marine protected areas to meet their established conservation objectives may be compromised by the following climate change impacts: (1) changes in quality and distribution of critical habitats such as coral reefs; (2) changes in the distribution of marine biodiversity (but see Jones *et al*., 2013); (3) changes in protected area connectivity; (4) changes in ecosystem structure and productivity; and (5) changes in human activities, such as spatial fishing patterns (Soto, 2001; McLeod *et al*., 2008). Climate change is projected to cause shifts in geographic ranges of marine organisms, affecting the distribution of marine biodiversity (Cheung *et al*., 2009; see also Target 12). Projections using species distribution models suggest a generally poleward shift in exploited marine fishes and invertebrates, with high rates of local extinction in the tropics and semi-enclosed seas and high rates of invasion in the Arctic. Trophic interactions in marine food webs are also projected to be affected (Ainsworth *et al*., 2011; Fulton 2011; Fernandes *et al*., 2013).

Oceans also face the threat of acidification, which occurs when carbon dioxide emissions absorbed into the ocean decrease the ocean's pH. Ocean acidification can weaken the shells of certain marine organisms, such as molluscs and crustaceans, and can cause coral bleaching (Mora *et al*., 2013).

The Study Group on Designing Marine Protected Area Networks in a Changing Climate developed the following guidelines to improve the resilience of MPAs in the face of climate change (Brock *et al*., 2012):

- Protect species and habitats with crucial ecosystem roles, or those of special conservation concern.
- Protect potential carbon sinks.
- Protect ecological linkages and connectivity pathways for a wide range of species.
- Protect the full range of biodiversity present in the target biogeographic area.

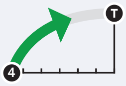
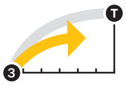
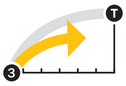
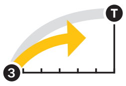
11.5 UNCERTAINTIES

Target 11 can be split into a number of separate components: 1) the total coverage of protected areas and other effective area-based conservation measures; 2) the degree to which biodiversity is represented; 3) management effectiveness and equitability; and 4) connectivity and integration into wider landscapes/seascapes. While data exist for the assessment of the first two components, those for the third and fourth are less well-developed. The IUCN Green List of Protected Areas is one framework that can fill this gap, to some extent. Ten countries, including China, Kenya, France and Colombia, are assisting IUCN, the World Commission on Protected Areas (WCPA) and other partners to develop a Global Green List Protected Area Standards, and a robust yet simple and context-specific assurance model. The IUCN Green List of Protected Areas will evaluate existing evidence to recognize success in achieving equitable governance and effective management that delivers conservation outcomes. Along with the 'Conservation Assured/Tiger Standards' Initiative, which adds species-specific standards onto the Green List criteria, the IUCN Green List of Protected Areas will be launched at the IUCN World Parks Congress in November 2014.

In the terrestrial Rio+20 scenarios explored here, a protected area is defined as an area free from agricultural land use, infrastructure development, and hunting and gathering. The effect of protected areas on biodiversity is therefore also based on this definition. However in reality (and in accordance with the IUCN definition of some categories of protected area) the protected areas might not be free from human use, and therefore the effect of protected areas on biodiversity might be dampened. Key assumptions made by the socioeconomic scenarios include: that bare areas cannot be turned into protected areas, so deserts are excluded; that grid cells close to agriculture areas are preferred for new protected areas; and that agricultural land cannot be transformed into natural habitat as this would be too expensive. The assumption that deserts do not need protection is not justified, given the threats these environments face and the unique flora and fauna they contain (Durant *et al*., 2014). Comparable scenarios of protected areas expansion are not available for the marine environment.

Future distributions of species depend on a range of drivers including abiotic conditions, biotic interactions, human-induced environmental changes and species-specific dispersal, establishment and demographic processes. These processes are also likely to change over time, but are ignored by most correlative models used to predict species distribution (Anderson, 2013; Dormann, 2007; Cheaib *et al.*, 2012; Pagel and Schurr, 2012). Furthermore, only very few models take into consideration the potential for species to adapt to new conditions through phenotypic plasticity and local adaptation (Bocedi *et al.*, 2013, Morin and Thuiller 2009). Uncertainties and errors in model prediction may also arise from the quality of initial data sets used to parameterise and validate the models (e.g., Lintz *et al.*, 2013; Buisson *et al.*, 2010), and mismatches between scales of data and modelling (Wiens *et al*, 2009).

11.6 DASHBOARD – PROGRESS TOWARDS TARGET

Target Elements	Status	Comment	Confidence
At least 17% of terrestrial and inland water areas are conserved		Extrapolations and existing commitments suggest the target will be met. Inland water protection has distinct issues.	High
At least 10% of coastal and marine areas are conserved		Marine protected areas are accelerating but extrapolations suggest we are not on track to meet the target. With existing commitments, the target would be met for territorial waters but not for exclusive economic zones or high seas	High
Areas of particular importance for biodiversity and ecosystem services conserved		Progress for protected Key Biodiversity Areas, but still important gaps. No separate measure for ecosystem services	High
Conserved areas are ecologically representative		Progress, and possible to meet this target for terrestrial ecosystems if additional protected areas are representative. Progress with marine and freshwater areas, but much further to go	High (for terrestrial and marine), Low (for inland waters).
Conserved areas are effectively and equitably managed		Reasonable evidence of improved effectiveness, but small sample size. Increasing trend towards community involvement in protection. Very dependent on region and location	Low
Conserved areas are well connected and integrated into the wider landscape and seascape		Initiatives exist to develop corridors and transboundary parks, but there is still not sufficient connection. Freshwater protected areas remain very disconnected	Low

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Dashboard: Tim Hirsch

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TARGET 12: PREVENTING EXTINCTIONS AND IMPROVING SPECIES CONSERVATION STATUS

By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained.

PREFACE

The report on modelling and scenarios for GBO3 summarized predicted biodiversity loss under future scenarios based on four key measures: (i) species extinctions, (ii) changes in species abundance, (iii) habitat loss, and (iv) changes in the distribution of species, functional groups or biomes (Leadley *et al*., 2010; Pereira *et al*., 2010). In this report, habitat loss is considered under the targets that comprise Strategic Goal B ('Reduce the direct pressures on biodiversity'). Here, we consider extinction risk and population trends. For extinction risk, we use measures based on the threat status of species, such as the IUCN Red List Index (Butchart *et al*., 2004). We also consider changes in the species diversity and composition of ecological communities, because these accumulate to cause global loss of species, including of known threatened species. We do not consider genetic diversity here because of the focus of the target on conserving species. Genetic diversity is considered under Chapter 13, although only for cultivated plants, domesticated animals and their wild relatives. The long-term conservation of species and of ecosystems more broadly will depend on conserving genetic diversity (e.g., Forest *et al*., 2007), and thus genetic diversity should be considered in future reports and target-setting. Microbial diversity was also beyond the scope of this report. However, microorganisms are an essential component of biodiversity and should also be considered in future reports. Freshwater environments are lacking much of the information required to present an equivalent assessment to those for the terrestrial and marine environments, necessitating a more qualitative treatment.

12.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

12.1.1 Status and trends to date

Observed extinctions have generally increased over the last 200 years (Barnosky *et al*., 2011). For birds and mammals, there has been a slowing in the apparent rate of extinction in the last 50 years (Figure 12.1A), although uncertainty about whether and when a species became extinct is large (see 'Uncertainties' below), and there is generally a long time-lag between a species becoming extinct and being recorded as such. For freshwater fish species on the other hand, numbers of observed extinctions have increased unabated over the last 100 years (Figure 12.1B). Consistent data on extinctions for other taxonomic groups are not available, but the number of extinctions since 1500 is very high. For example, at least 34 amphibians became extinct during this period, and 165 possibly extinct (Stuart *et al*., 2008). At least 310 of the approximately 6,800 species of molluscs assessed by IUCN have become extinct since 1500, nearly 5% of the total assessed (www.iucnredlist.org). In order to prevent extinctions of known threatened species in the near future (the main aim of Target 12) will require the protection of the sites containing the last populations of critically endangered species (Alliance for Zero Extinction sites): coverage of these by protected areas has increased steadily over the past 25 years (Figure 12.2 D; Butchart *et al*., 2012). Protection of Important Bird and Biodiversity Areas (IBAs) has also increased rapidly in the same time period, although from a much lower starting point (Figure 12.2 E). All measures to protect known threatened species will require significant investments; there have been no clear trends in funding for the protection of species in the last 15 years (Figure 12.2 F).

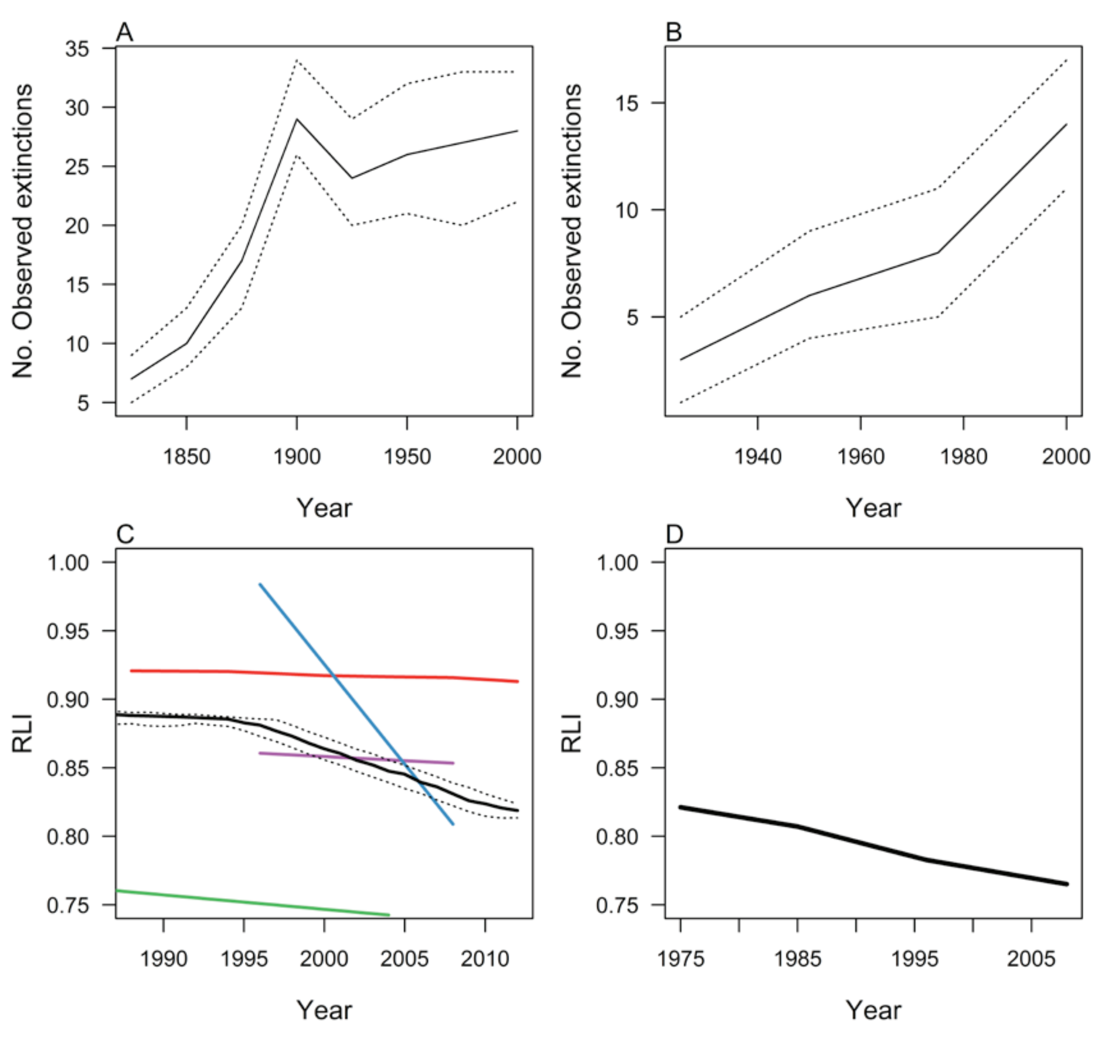


Figure 12.1. Median and 95% confidence intervals of the number of extinctions in 25-year intervals from 1800 to the present of (A) mammal and bird (data source: BirdLife International, 2014) species and (B) freshwater fish species (in total, 47 mammal species, 90 bird species and 32 fish species were declared extinct, and 27 mammals and 13 birds presumed extinct since 1800); (C) IUCN Red List Index of species survival (RLI) for the world's birds (red, 9,869 species), mammals (purple, 4,556), amphibians (green, 4,355) and corals (blue, 704); and an aggregate estimate (black); Source: Butchart *et al*., 2010; updated by S. H. M. Butchart *et al*., unpublished data**; declines correspond to increases in extinction risk; D) reconstructed trend in extinction risk for carnivores (284 species) and ungulates (262 species) in the last 40 years. For estimating observed extinction rates (A & B), when an exact extinction date was not given, an interval during which a declared extinction is thought to have occurred was used; otherwise, the last sighting of the species was considered as the earliest date and the latest date was assumed to be 30 years after the last sighting. For the 43 species presumed extinct, we considered the last sighting as the earliest date of extinction and added the average time between last sighting and the declaration of extinction of other species to derive the latest date. The RLI (C) ranges from 0 (all species are extinct) to 1 (all species are considered "Least Concern"). Confidence intervals around the aggregate RLI trend represent 95% confidence intervals, and were based on multiple sources of underlying uncertainty – see Butchart et al., 2010. To estimate the past trend in the IUCN Red List status of carnivores, the current IUCN criteria were applied to information on past population and geographic range size, structure, and trend, as well as habitat loss and other threats, available from historical IUCN Red Data Books and Action Plans** Source: Di Marco et al. in press; doi: 10.1111/cobi.12249**.**

The average extinction risk of assessed species – measured as the IUCN Red List Index – increased steadily over the past 40 years with no signs of slowing (Figure 12.1C, D), although increased attention and investment towards threatened species has prevented some critically endangered species from going extinct (Butchart *et al*., 2006; Hoffmann *et al*., 2010). Among terrestrial species, amphibians have a high level of threat and are increasing in extinction risk strongly (Figure 12.1C), with 32% of species threatened and 40% declining according to IUCN (www.iucnredlist.org; Stuart *et al*., 2008). For plants, comprehensive assessments of current extinction risk are only available for gymnosperms, for which 41% of species are considered threatened (www.iucnredlist.org). There are no reported trends in extinction risk of plants at present. For flowering plants, only 6% of the approximately 270,000 known species have been assessed (including all cacti); however, of these 56% (31% of cacti) are considered threatened. The extinction risk of terrestrial invertebrate species and fungi is poorly known at a global scale, but assessments are available for certain regions suggesting that significant proportions of species are threatened with extinction: 9% of European butterflies (van Swaay *et al*., 2010), 15% of European dragonflies (Kalkman *et al*., 2010), 25% of Japanese dragonflies (Kadoya *et al*., 2009), 11% of European saproxylic beetles (Nieto and Alexander, 2009) and 16% of British fungi (Evans *et al*., 2006). Freshwater species are also showing strong declines, with 32% of freshwater vertebrates and 32% of decapods at risk of extinction (Collen *et al*., 2014). In the marine realm, over 550 species of marine fishes and invertebrates are listed on The IUCN Red List as Critically Endangered, Endangered, and Vulnerable (www.iucnredlist.org). This is an underestimate, owing to insufficient data with which to assess the extinction risk of many marine organisms. As with trends in the extinction risk of species, population trends – as measured by the Living Planet Index (Figure 12.2 C), the Wild Bird Index of habitat specialist bird species (Target 5) and the Wildlife Picture Index (O'Brien *et al*., 2010) – continue to decline.

For all terrestrial vertebrate groups, habitat loss because of agriculture, aquaculture and logging is responsible for the decline of the greatest number of species (Hoffmann *et al*., 2010). Reptiles and amphibians are particularly sensitive to habitat degradation because of their comparatively low dispersal ability, relatively small home ranges and thermoregulatory constraints (Kearney *et al*., 2009).

For mammals and birds, hunting is the greatest threat after habitat loss (Hoffmann *et al*., 2010). For amphibians, invasive species and disease (in particular the chytrid fungal pathogen *Batrachochytrium dendrobatidis*; Cheng *et al*., 2011) are the main drivers of decline after habitat loss, although the interaction of these threats with climate change is likely to exacerbate amphibian decline in the near future (Hof *et al*., 2011; Pounds *et al*., 2006). Invasive alien species are also a major threat to birds, particularly those on oceanic islands (BirdLife International, 2014). Disease is also an important threat for certain other taxonomic groups (for example, white-nose syndrome in bats). The relative threat posed to other animal taxonomic groups by different human activities is less well known. For invertebrates – bees in particular – pesticides appear to be a serious threat (e.g., Gill *et al*., 2012).

Plant species are mainly affected by loss, degradation and increased fragmentation of habitats, and alien invasive species (Bilz *et al*., 2011). In the future, the threat from climate change is predicted to grow (Bilz *et al*., 2011; Giam *et al*., 2010). Genetic erosion and extinction have been identified as important threats to the crop wild relatives and to plant populations that occur on islands (Bilz *et al*., 2011). Aquatic plants are affected by ecosystem modification and loss caused by the transformation of wetland habitats, and the intensification of agricultural activities accompanied by eutrophication and pollution (Bilz *et al*., 2011). In some countries the collection of wild plant species (for medicines, food, aesthetic value, or value for collectors) is causing a loss of species and a reduction in reproductive success (Bilz *et al*., 2011).

For marine species, data are available on the threats driving species loss for several well studied groups. The chondrichthyans (sharks and rays) are overexploited through targeted fisheries as well as incidental by-catch (Dulvy *et al*., 2014). In addition, half of the 69 high-value sharks and rays in the global fin trade are threatened (53.6% of 37 species), while low-value fins often enter trade as well, even if meat demand is the main fishery driver (Dulvy *et al*., 2014). Similarly, for parrotfishes and surgeonfishes, species that play critical roles in coral reef ecosystems, 40% (73 species) of known species are impacted by small and large scale fisheries and 6% are recorded to be affected by habitat modification, 3% by pollution and 1% by by-catch (Comeros-Raynal *et al*., 2012). For groupers and wrasses, 12% of the 163 species assessed are considered to be at risk of extinction (classified by IUCN as Critically Endangered, Endangered or Vulnerable) if current trends continue, with a further 13% considered Near-Threatened. However, 30% of species could not be assessed owing to insufficient data. The major driver of extinction risk in this group is overfishing, with poor or no fishery management (Sadovy de Mitcheson *et al*., 2012). For seabirds, particularly albatrosses and large petrels, longline and gillnet fisheries present a severe threat (Croxall *et al*., 2012).

At the global scale, limited data prevent comparable reporting for freshwater species, especially freshwater fishes and invertebrates. However, based on current data, the main threat to freshwater vertebrates and decapods is habitat loss and degradation, affecting 80% of species. This is followed by pollution (50% of threatened species) and exploitation (40% of threatened species) (Collen *et al.*, 2014). The combined effects of overexploitation and habitat degradation are acute for freshwater-dependent chondrichthyans, with over one-third (36%) of the 90 obligate and euryhaline freshwater chondrichthyans considered threatened (Dulvy *et al.*, 2014). The degradation of coastal, estuarine and riverine habitats threatens 14% of sharks and rays: through residential and commercial development (twenty-two species, including river sharks *Glyphis* spp.); mangrove destruction for shrimp farming in Southeast Asia (four species, including Bleeker's variegated stingray *Himantura undulata*); dam construction and water control (eight species, including the Mekong freshwater stingray *Dasyatis laosensis*); and pollution (twenty species) (Dulvy *et al.*, 2014).

Importantly, within broad groups of species, the effect of threats will not fall evenly on different species. In terrestrial environments, large-bodied, slow-breeding species with strict habitat and dietary requirements have been shown to be more adversely impacted by habitat loss (e.g., Vetter *et al.*, 2011; Newbold *et al.*, 2013), and to be at greater risk of extinction (Cardillo *et al.*, 2005; Davidson *et al.*, 2009) than other species. Similarly, among marine fishes, turtles and mammals, large-bodied, late maturing species have been shown to be more sensitive to fishing and pollution than other species (Reynolds *et al.*, 2005; Cheung *et al.*, 2005; 2007; Davidson *et al.*, 2012; Norse *et al.*, 2012; Maxwell *et al.*, 2013).

12.1.2 Projecting forward to 2020

Concerted conservation action has been shown to be effective in reducing the extinction risk of vertebrate species (Butchart *et al.*, 2006; Hoffmann *et al.*, 2010), and further action might prevent some extinctions that would otherwise occur by 2020. However, extrapolations suggest that it is very unlikely that all extinctions of known threatened (bird and mammal) species will be prevented by 2020 (Figure 12.2; A). Indeed, many species are at high risk of imminent extinction (e.g. Wake, 2012) and the level of resourcing required to prevent extinctions of known threatened species is an order of magnitude greater than current investment (McCarthy *et al.*, 2012). Furthermore, many undescribed species have already, or will by 2020, become extinct without our knowledge (Mora *et al.*, 2011). The global rate of extinctions might be slowing (Figure 12.2; A); however, at least for birds the rate of extinctions in continental areas may be accelerating (Szabo *et al.*, 2012) and lags in reporting might lead to an underestimate of recent extinctions. In any case, the rate of extinction of freshwater fish species is likely to continue increasing (Figure 12.1B); however, as noted elsewhere, data for freshwater species are very limited.

Extinction risk – as measured by the IUCN Red List Index for birds, mammals and amphibians – is predicted to continue to increase (Figure 12.2B), while population trends, as measured by the Living Planet Index (Figure 12.2C; Collen *et al.*, 2009) and Wild Bird Index (cross-reference to Target 5), are predicted to continue to decrease. On the other hand, coverage of Alliance for Zero Extinction sites by protected areas is predicted to increase, although based on the current rate of increase it is unlikely that 25% of sites will be protected before 2020 (Figure 12.2D; Butchart *et al.*, 2012). Coverage of Important Bird and Biodiversity Areas is predicted to increase, but still leaving 75% of sites inadequately protected in 2020 (Figure 12.2E). The incomplete coverage of assessments of marine species and the short time series of existing data preclude a numerical extrapolation of marine species' trends to 2020. Future trends in funding for species protection are difficult to predict (Figure 12.2F).

Figure 12.2 (opposite). Recent trends and extrapolations to 2020 of six key measures of the extinction, extinction risk and population trends of species: observed extinction rates of birds and mammals (A); the aggregate Red List Index of birds, mammals, amphibians and corals (B); the Living Planet Index (C); the coverage by protected areas of the entirety of sites whose protection could avert the extinction of known threatened species – Alliance for Zero Extinction (AZE) sites (D) and Important Bird and Biodiversity Areas (IBAs) (E); and funds for the protection of species (F). Data from recent trends are indicated by points, continuous lines indicate the fit to data, dashed lines are extrapolations to 2020 and dotted lines indicate 95% confidence intervals. Horizontal grey line represents model-estimated 2010 value for indicator. Source: Visconti *et al*. ,(unpublished data and Birdlife International, 2014) (A); Butchart *et al*., 2010 (unpublished data) (B). Collen *et al*. (unpublished data) and described in Collen *et al*. (2009) (C); Butchart *et al*. (unpublished data) and described in Butchart *et al*. (2012) (D-E); AidData (http://aiddata.org/) (F). Extrapolations are based on the assumption that underlying mechanisms continue to follow trends. Methods for model fitting are described in the introductory chapter.

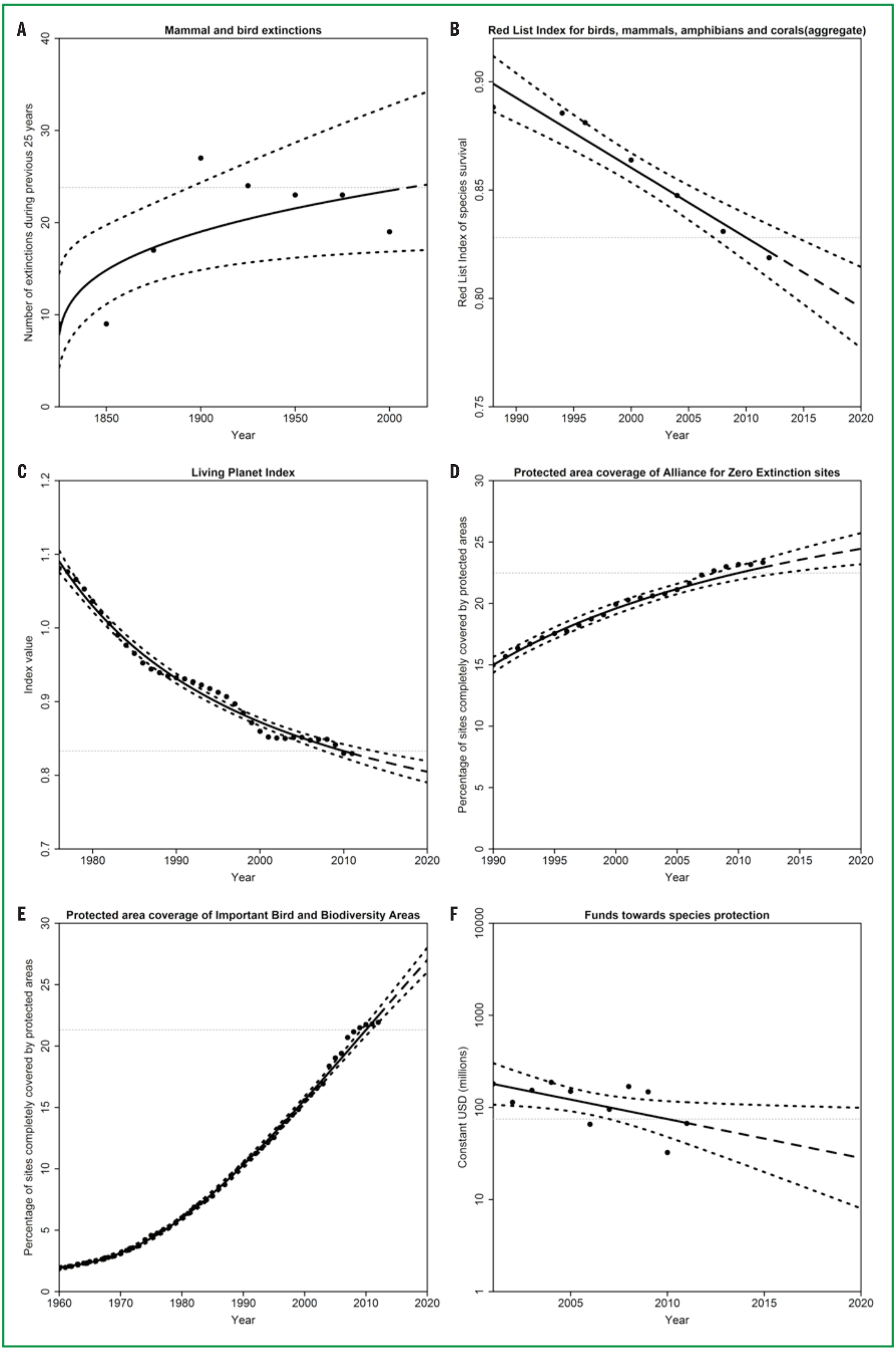

A
Mammal and bird extinctions
Number of extinctions during previous 25 years
Year
B
Red List Index for birds, mammals, amphibians and corals(aggregate)
Red List Index of species survival
Year
C
Living Planet Index
Index value
Year
D
Protected area coverage of Alliance for Zero Extinction sites
Percentage of sites completely covered by protected areas
Year
E
Protected area coverage of Important Bird and Biodiversity Areas
Percentage of sites completely covered by protected areas
Year
F
Funds towards species protection
Constant USD (millions)
Year

Figure 12.2

In terrestrial, marine and freshwater environments, habitat destruction, fragmentation and degradation (hereafter "habitat loss") are likely to remain major stresses on biodiversity until 2020 and beyond (Green *et al*., 2005; Jetz *et al*., 2007; Alkemade *et al*., 2009; Martinuzzi *et al*., 2013a,b). In addition, for both marine and freshwater species, overexploitation is and will remain a major threat (Pitcher and Cheung, 2013; see Target 6). Many studies have predicted the impact that habitat loss will have in the future on the ranges (e.g., Jetz *et al*., 2007; Cheung *et al*., 2009), population trends (e.g., WWF, 2012) and extinction risk (Bird *et al*., 2011) of species, and on the diversity of ecological communities (Gibson *et al*., 2011; Allan, 2004; Cheung *et al*., 2009; Newbold *et al*., in review). Moreover, emerging threats such as deep-sea mining may further increase the extinction risk associated with habitat changes (Boschen *et al*., 2013).

Short-term future projections of the extinction risk of species as a result of projected habitat loss generally predict a worsening situation. However, improvements can be seen under some scenarios. Under business-as-usual scenarios, species within ecological communities are projected to continue declining in abundance on average (Figure 12.3A; Alkemade *et al*., 2009), and the number of species within communities is also projected to decrease (Figure 12.3B; Newbold *et al*., in review). Global populations of carnivore and ungulate species are projected to continue decreasing steeply (Figure 12.3C), and these species will likely lose substantial proportions of their ranges (Visconti *et al*., 2011). This leads to a predicted increase in species' extinction risk (Figure 12.3D; Di Marco *et al*., in press; Visconti *et al*., in review; see also Millennium Ecosystem Assessment, 2005). Regional model predictions under business-as-usual scenarios mirror these results: for example, in the Brazilian Amazon approximately 2% of vertebrate species, on average, are predicted to become locally extinct as a result of habitat loss, with a further 12% committed to extinction (Wearn *et al*., 2012), while the percentage of threatened bird species in the same region is predicted to increase from 3% to between 8% and 11% (Bird *et al*., 2011).

Under the Rio+20 scenarios (Netherlands Environmental Assessment Agency, 2010; Chapter 0), designed to mitigate biodiversity losses, losses of within-community diversity (abundances and numbers of species) are slowed to some extent, but not prevented (Figure 12.3; A; B). Population size and extinction risk trends are reversed under these mitigation scenarios in the short term, caused by a net gain in natural habitat in Africa and Southeast Asia, which are hotspots of carnivore and ungulate richness (Figure 12.3C; D). The scenarios assume that the natural habitat gained is biotically equivalent to primary natural habitat; relaxing this assumption would lessen the modelled effectiveness of mitigation. In the Brazilian Amazon, the local extinction of vertebrate species is predicted to decrease by one- to two-thirds under scenarios that assume a reduced rate of deforestation (Wearn *et al*., 2012).

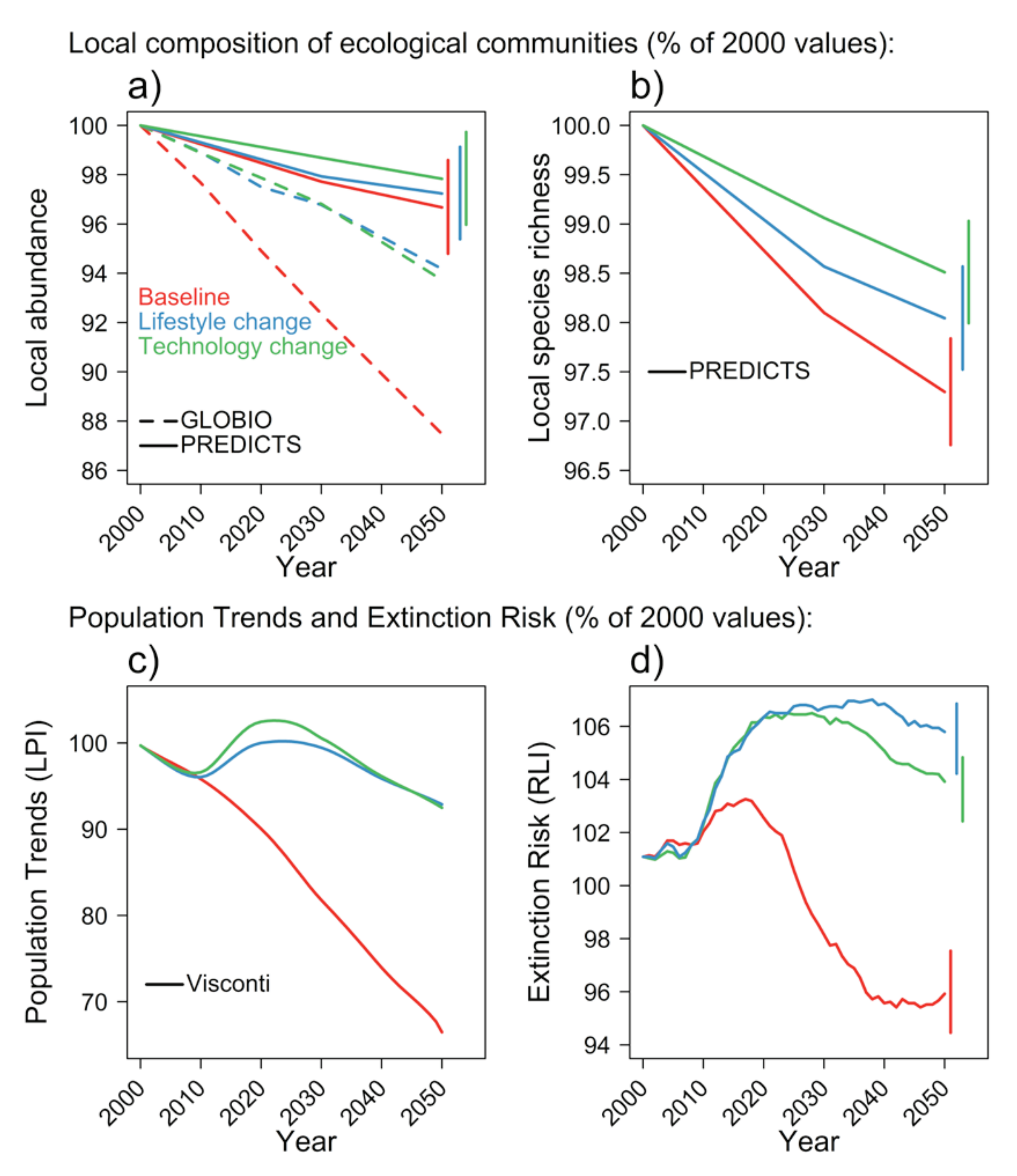


Figure 12.3. Predicted changes in the global average of local abundance (A) and species richness of ecological communities (B), and of average population trends (C); calculated following the Living Planet Index (LPI) methods, (see Collen et al., 2009) **and extinction risk (D); IUCN Red List Index (RLI) of carnivores and ungulates from 2000 to 2050, under three of the Rio+20 scenarios** (Netherlands Environmental Assessment Agency; Chapter 0)**. Values are scaled to equal 100% in 2000. Projections of composition of local ecological communities are from the PREDICTS** (Newbold *et al*., in review) **and GLOBIO** (Alkemade *et al*., 2009) **models. The population trends and extinction risk projections are the 'maximum physiological dispersal' projections of terrestrial carnivore and ungulate species from Visconti et al. (in review). The modelled indicators measure slightly different aspects of conservation status, hence the differences in projected trends. GLOBIO projects mean species abundance (MSA) across all taxonomic groups relative to pristine conditions as a response to multiple pressures including habitat loss, climate change and human disturbance. MSA does not allow for increases in species abundance beyond their original levels. Local total abundance (PREDICTS model), global population trends of species and extinction risk (RLI) do not have this constraint. PREDICTS and GLOBIO measure local losses, which are aggregated spatially across grid cells, while RLI and LPI trends are measures of global species decline, which are aggregated across species. Vertical bars show uncertainty in 2050 (shown only for 2050 for clarity); uncertainty estimates were not available for the GLOBIO or Visconti LPI projections.**

While positive changes to land-use policies (e.g., investing in forest restoration or modifying investments in crop production) can affect trends, there are limits to what can be achieved. In the United States, for example, the basic economic and demographic factors shaping land-use change are powerful, and even fairly dramatic policy changes have been shown to lead to only moderate deviations from a business-as-usual scenario (Radeloff *et al.*, 2012). However, some policy tools will be easier to enact, highlighting that opportunities exist for exploring different policy options that can have more substantial impacts on habitat loss.

12.1.3 Country actions and commitments[1]

The majority of the 22 National Biodiversity Strategies and Action Plans (NBSAPs) examined contain targets, or other commitments, which explicitly refer to the threat of extinction. The targets which have been established tend to focus on reducing the risk of extinction of species generally. However a few countries, such as Belarus, Myanmar and Switzerland have tailored their targets to focus on either specific species or on priority species. A number of countries have also developed targets aimed at reducing certain pressures on species. For example Venezuela's plan includes the prevention and management of the illegal trafficking of species. Furthermore, a few countries in their NBSAPs have noted the link between this and other targets, such as those related to invasive alien species, habitat loss and protected areas.

Fewer targets explicitly refer to preventing final extinction, and few countries have set corresponding quantitative targets.

The targets or similar commitments which have been set out in the NBSAPs examined will make a significant contribution towards this target if implemented. However, given the ambitious goal of preventing the extinction of all known threatened species, additional efforts will likely be required if this target is to be achieved by 2020.

12.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

12.2.1 Actions

To prevent further extinctions of known threatened species, substantial conservation investment is needed across terrestrial, freshwater and marine ecosystems: it is estimated that investment needs to increase by an order of magnitude in order to reduce the extinction risk of known threatened species (McCarthy *et al.*, 2012). It is essential to continue monitoring the status of species that have already been assessed, and to obtain information on the distribution and extinction risk of less well-studied species to make assessments and future projections of the status of these species. Assessments for species in all environments, but especially for freshwater and marine species, are a work in progress (see Carrizzo *et al.*, 2013 and http://sci.odu.edu/gmsa/index.html), and many critical regions have not been evaluated. Investments in completing these assessments are crucial for conservation decision makers to be able to adequately represent, evaluate and conserve less well-known taxa.

Actions aimed at the conservation of threatened species can be broadly categorised as species-level, which are aimed directly at threatened populations (e.g., legislation on hunting and trade, vaccinations, captive breeding and reintroductions), or site, ecosystem, or landscape-level, which are directed at species' habitat (e.g., protected areas, invasive species control, forest management) (Boyd *et al.*, 2008). Species-level actions are generally targeted at the protection of a single species, although the protection of habitat thus achieved might also protect other species (Skaala *et al.*, 2014), while broader site-, ecosystem-, and landscape-level actions are likely to benefit multiple species (Boyd *et al.*, 2008). In the last three decades, species-level conservation actions have been a major driver in reducing the extinction risk of nearly 30 species of vertebrates (Hoffmann *et al.*, 2010). Habitat-level actions have had a positive effect on 36 species (Hoffmann *et al.*, 2010). Further such actions will help to avoid extinction and improve the conservation status of species in the future.

Footnote

[1] This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor L'este, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.

Among threatened terrestrial vertebrates, 20% can have their extinction-risk status improved in the next decade with conservation actions in single sites, because the entire global population is restricted to – and assumed able to be conserved at – sites managed as a single unit (Boyd *et al*., 2008). Approximately 60% will benefit most through interventions in a network of sites (Boyd *et al*., 2008). This suggests that the maintenance of effectively managed protected areas (or networks of protected areas) aimed at protecting the remaining populations of threatened species is the most important action to achieve Target 12. In order to effectively protect freshwater biodiversity, it will be necessary to target protected areas towards freshwater habitats (see Target 11). Similarly, effectively managed Marine Protected Areas have been shown to allow recovery of fish biomass, particularly of predatory species (Edgar *et al*., 2014), although the contribution of marine protected areas to reduction in extinction risk has not been quantified. Protection of Alliance for Zero Extinction sites (AZEs) and Important Bird and Biodiversity Areas has increased over time, although progress has slowed recently (Figure 11.2D; Butchart *et al*., 2012). Increasing efforts to protect these sites will help to avert extinctions in the near future.

Species-level conservation needs to be complemented by landscape- or ecosystem-scale policy measures aimed at reducing habitat loss, overexploitation, pollution, and the impact of invasive species and pathogens. In terrestrial and inland-water environments, habitat loss is by far the greatest threat to animal and plant species (Hoffmann *et al*., 2010). Therefore, actions aimed at stopping habitat loss, mitigating fragmentation (see chapter 5), and actively restoring degraded habitat (see chapter 15) will be critical for the persistence of many terrestrial and inland water species. The highest density of endangered terrestrial species is in Southeast Asia, owing to deforestation and consequent conversion to cropland and wood plantations, and to direct exploitation of plant and animal species (Orme *et al*., 2005; Sodhi *et al*., 2010). For terrestrial species, this region, together with other regions of high endemicity, such as global plant biodiversity hotspots (Myers *et al*., 2000) and tropical islands, require immediate attention.

Given the stress placed on native freshwater and marine fishes and invertebrates by unsustainable harvesting, there is an urgent need to reduce such harvesting and to develop more sustainable harvesting methods and improve livelihoods for humans (Pauly *et al*., 2002; see also Targets 6, 7 and 14). Ecosystem-based fisheries management, which considers fisheries management in the context of global and local environmental changes and other human impacts, should be implemented (Pikitch *et al*., 2004; Marasco *et al*., 2007; Ruckelshaus *et al*., 2008).

Invasive species are also a major threat: for example, for declining terrestrial invertebrates, invasive species are listed as a major threat for 15% of species (Hoffmann *et al*., 2010). Invasive species are a particularly strong threat for island endemics: 64% of IUCN-listed extinctions have occurred on islands, including about 95% of bird, 90% of reptile and 70% of mammal extinctions (Keitt *et al*., 2011); most of these species were impacted by invasive species (Blackburn *et al*., 2004; Keitt *et al*., 2011; Turvey *et al*., 2009; Turvey *et al*., 2011). Therefore, targeted efforts at eradicating invasive species, especially cats and rats on islands, are urgently required to prevent imminent extinctions (Genovesi, 2011; Keitt *et al*., 2011; see also Target 9). Although the contribution of marine invasive species to extinction has not been quantified, marine invasion should be prevented through a number of measures. These measures include effective control of ballast water discharge, improved public education, and monitoring and removal actions to eliminate or suppress invasive species (Molnar *et al*., 2008; Williams and Grosholz, 2008).

For more than 300 amphibian species affected by the chytrid fungus *Batrachochytrium dendrobatidis* (Vredenburg *et al*., 2010), and several other critically endangered species with very small and declining populations, captive breeding will be required until the causes of decline are removed or mitigated sufficiently to permit reintroduction (Boyd *et al*., 2008; Stuart *et al*., 2008).

Habitats and species are rarely affected by single pressures, and therefore multiple coordinated actions are required (e.g., Rondinini *et al*., 2011). Species-level management is therefore best coordinated through action plans. These have been produced and updated for several taxonomic groups by IUCN (https://www.iucn.org/about/work/programmes/species/publications/species_actions_plans/), other NGOs, and regional and local government authorities worldwide. A notable example is the Global Strategy for Plant Conservation (GSPC), established by the Convention on Biological Diversity and updated for the period 2011-2020 http://www.cbd.int/gspc/. The incorporation of GSPC targets and species actions plans into NBSAPs (see also Target 17) and their timely implementation will be critical to prevent the extinction of many known threatened species.

12.2.2 Costs and cost-benefit analysis

Global estimates of the costs of meeting Target 12 suggest that US$3.4 billion to US$4.8 billion will be required per year (McCarthy *et al.*, 2012; High Level Panel, 2014). This estimate was based on extrapolation of the estimated cost of actions needed to reduce the extinction risk (IUCN Red List status) of a sample of 211 threatened bird species, combined with data on the relative costs of conservation actions for birds and a wide range of other taxa (McCarthy *et al.*, 2012). Assuming that conservation actions undertaken for each species are entirely independent of one another, it is estimated that improving the status of all bird species will cost US$1.23 billion per year (McCarthy *et al.*, 2012). Recognizing that some conservation actions will benefit species other than the target species, total costs are estimated at US$0.88 billion per year (McCarthy *et al.*, 2012). Extrapolating these costs from the 1,115 globally threatened bird species to the 13,452 other known threatened species, it is estimated that improving the status of all known threatened species will cost between US$3.41 billion and US$4.76 billion per year (McCarthy *et al.*, 2012). Current funding is only 12% of that required (McCarthy *et al.*, 2012).

Quantifying the total value of the benefits provided by biodiversity to human society, and thus the economic benefits of preventing extinctions and meeting Target 12, is impossible. However, almost all analyses that have been carried out have suggested that the benefits of conservation actions outweigh the costs. For example, pollination services provided by insect species have been estimated to be worth US$19 billion to US$21 billion per year in the European Union alone (High Level Panel, 2014). Furthermore, it has been estimated that 2.5-16% of all jobs in the European Union depend on the environment to some degree and that 5.8% of jobs in sub-Saharan Africa depend on tourism, much of which is nature-based (High Level Panel, 2014). It has been estimated that a network of protected areas that adequately conserved biodiversity would achieve a benefit-to-cost ratio of 100:1 (Balmford *et al.*, 2002).

12.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

Extinction of species, both local and global, will have profound effects on ecological communities more broadly and on the functioning of ecosystems. The non-random loss of species from ecological communities leads to those communities becoming homogenised and dominated by certain functional types of species (Newbold *et al.*, 2014). The loss of key species from communities can lead to altered interactions among species and ultimately to trophic cascades (Estes *et al.*, 2011). Finally, the local extinction of species from ecological communities will impair the functioning of ecosystems: recent meta-analyses have shown that more diverse communities function more resiliently over space and time, in the face of environmental changes (Isbell *et al.*, 2011).

12.4 WHAT DO SCENARIOS SUGGEST FOR 2050?

Land-use change and overexploitation will remain the major drivers of terrestrial species loss until 2050, but with climate change increasing in importance over time (Alkemade *et al.*, 2009; Collen *et al.*, 2014). Most of the Rio+20 scenarios (Netherlands Environmental Assessment Agency, 2010) predict further declines in population trends of terrestrial species and in the average local diversity of terrestrial ecological communities, and further increases in species' extinction risk, although these changes are slowed or in some cases reversed under scenarios that assume efforts at mitigation (consumption or technology changes) (Figure 12.3).

Climate change may affect species directly, through their physiological tolerance, or indirectly through changes in vegetation (Powell and Lenton, 2013). Marine species are also threatened by ocean acidification and hypoxia (Vaquer-Sunyer and Duarte, 2008; Godbold and Calosi, 2012). The combined effects of these stressors may further exacerbate the effects of climate change on marine biodiversity (Mora *et al.*, 2013). The frequency and intensity of extreme climate events are also likely to have a major impact on future fisheries production in both inland and marine systems. Shifts in the migration phenology of many species important for commercial and recreational fisheries have been attributed to climate change, including: Pacific salmon (Quinn and Adams, 1996), Atlantic salmon (Juanes *et al.*, 2004) and smelt (Ahas and Aasa, 2006). There are strong interactions between the effects of fishing and the effects of climate because fishing reduces the age, size, and geographic diversity of populations and the biodiversity of marine ecosystems (Brander, 2007), which makes species more vulnerable to the potential effects of climate change. Inland (freshwater) fisheries are additionally threatened by changes in precipitation and water management (Palmer *et al.*, 2008; Strayer and Dudgeon, 2010).

In all environments, synergistic effects of multiple drivers could further increase biodiversity loss. For example, the impact of habitat loss on species has been shown to be worsened by climate change (Mantyka-Pringle *et al.*, 2012), and the invasion and spread of exotic plants has been shown to be more likely given higher rates of land-use change (Chytrý *et al.*, 2012). These results suggest that future biodiversity assessments should consider the interacting effects of multiple threats to biodiversity loss, rather than treating the effects of drivers as being additive. Distribution shifts driven by climate change will alter biodiversity patterns (Lawler *et al.*, 2009) and may affect trophic interactions, although the implications of the latter for extinction risk are not yet clear.

Biodiversity in some marine habitats, such as coral reefs (cross-reference to Target 10), is particularly sensitive to projected climate change and ocean acidification. At a global scale, the potential impact of climate change on freshwater biodiversity remains poorly understood, but is projected to present a growing challenge to the integrity and function of freshwater systems (Dudgeon *et al.*, 2006).

In the technical report on modelling and scenarios for the Global Biodiversity Outlook 3 (Leadley *et al.*, 2010), models predicting future changes (to 2050) in extinction rates, average abundance of species within ecological communities, and species distributions were reviewed. In summary: projected extinction rates ranged from values similar to current ones (for models of projected species-specific habitat loss) to two orders of magnitude larger (for models based on the species-area relationship); models of projected species abundance (all based on the GLOBIO model; Alkemade *et al.*, 2009) predicted a mean decline of 9-17% in abundance by 2050; for both species loss and decrease in abundance, the socio-economic scenarios reviewed only made small differences in the predicted outcomes; all studies of changes in species distributions (mostly based on niche models or global vegetation models) predicted distributional shifts that would result in changes of biotic communities and potentially the creation of new communities. Since the publication of the Global Biodiversity Outlook 3, several studies have advanced our understanding of biodiversity scenarios for 2050.

There is a consensus that there will be widespread local extinctions of species in both marine and terrestrial environments driven by climate change (Figure 12.4; 12.5; Cheung *et al.*, 2009; Bellard *et al.*, 2012), which are likely to trigger cascade effects through co-extinctions of dependent species (Brook *et al.*, 2008), possibly resulting in loss of ecosystem services (Hooper *et al.*, 2012; Tilman *et al.*, 2012).

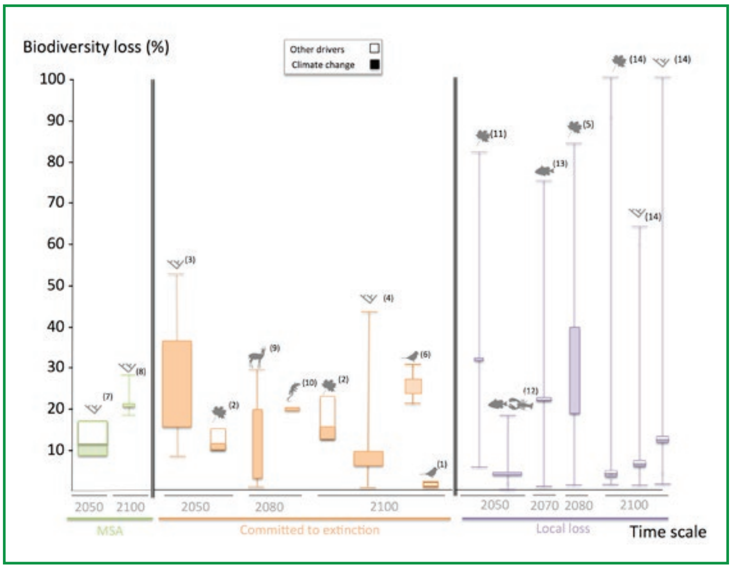


Figure 12.4. Projections of biodiversity loss owing to climate change (and other drivers). The width of the box indicates the generality of the predictions with respect to spatial scale and taxonomic breadth. The box is delimited by the upper and lower boundaries of the intermediate scenario, while the whiskers indicate the highest and lowest biodiversity losses across all scenarios. The highest estimates of local losses are obtained when considering direct effects of climate on species by projecting their bioclimatic envelope (e.g., Thomas *et al.*, 2004; Thuiller *et al.*, 2005) and at the lowest end when considering only indirect effects through changes in land cover (Jetz *et al.*, 2007). Source: Reproduced from Bellard *et al.* (2013).

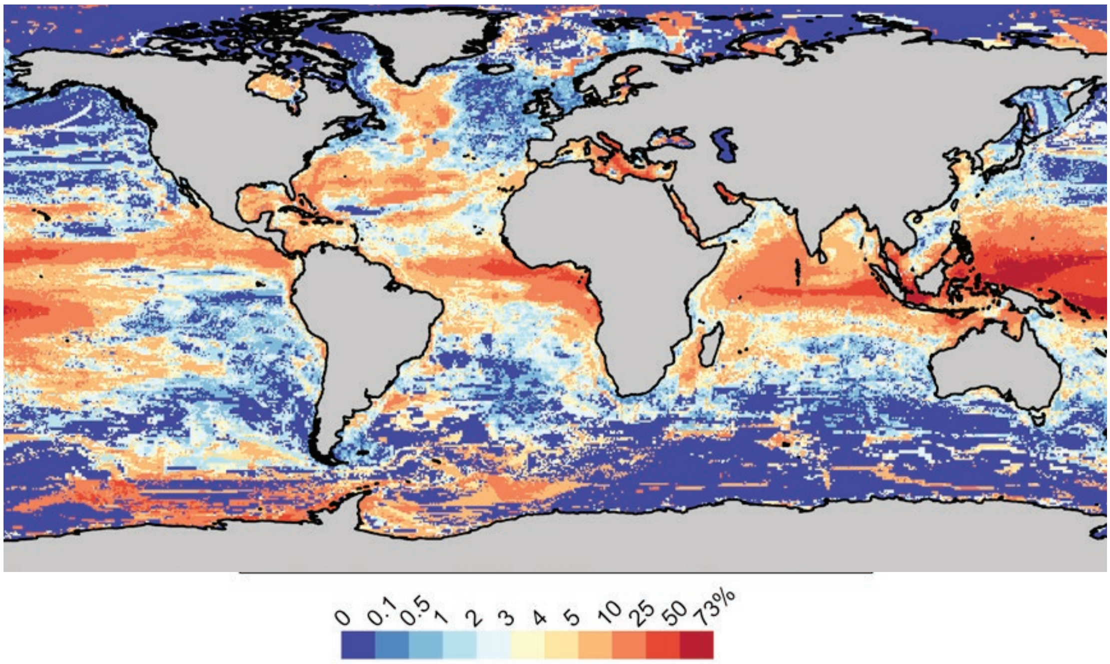


Figure 12.5. Percentage of exploited marine fishes and invertebrates (out of a total of 802 species) predicted to become locally extinct given predicted climate change under the Intergovernmental Panel on Climate Change's Representative Concentration Pathways RCP8.5 scenario by 2050, relative to 2000. Source: Based on data from Jones *et al*. (in press).

Species can survive climate change by shifting their ranges or by adapting, either through evolutionary change (i.e., changes in behavioural, physiological or ecological traits) or through phenotypic plasticity (i.e., the species already possess the required traits to survive under new climatic conditions and these traits are selected for within the existing pool). For terrestrial species, the velocity of climate change (Loarie *et al*., 2009) is expected to outpace the dispersal ability of most species, across several studied taxa (Bertrand *et al*., 2011; Devictor *et al*., 2012; Schloss *et al*., 2012). Species with narrow altitudinal ranges and low thermal tolerance, especially those inhabiting high mountains, are predicted to incur local extinctions in several regions of the world (Laurance *et al*., 2011; Dullinger *et al*., 2012). Furthermore, for terrestrial species to adapt evolutionarily to climate change would require rates of niche evolution that are more than 10,000 times faster than those typically observed (Quintero and Wiens, 2013). However, a recent study revised upwards many previous estimates of species ability to shift their range (Chen *et al*., 2011), and several marine and freshwater groups appear able to keep pace with climate change (Kinlan and Gaines, 2003; Kappes and Haase, 2012).

Projected changes vary substantially in different parts of the world owing to variation in the different drivers of biodiversity change. Under business-as-usual scenarios, particularly strong declines are predicted in Africa, because of expanding agriculture, livestock production and forestry (Jetz *et al*., 2007; Visconti *et al*., 2011; Figure 12.6). Large declines of terrestrial species are also predicted in the Amazon, a region with very low spatial climatic gradients that is predicted to experience no-analog future climates (Williams *et al*., 2007; Figure 12.6) and that contains a rich fauna of vertebrate species with high intrinsic vulnerability to climate change (Foden *et al*., 2013). Finally, large declines and turnover rates are predicted in areas rich in elevational specialists (Laurance *et al*., 2011), such as the Andes for mammals (Lawler *et al*., 2009; Schloss *et al*., 2012) and the Himalayas for birds (Jetz *et al*., 2007). For marine fish and invertebrate species the areas with highest expected local extinctions by 2050 are sub-polar regions, the tropics and semi-enclosed seas (e.g., the Mediterranean and Red Seas; Fig. 12.5; Cheung *et al*., 2009; Jones *et al*., in press); while the areas with highest number of expected invasions are the Arctic and Southern Ocean (Cheung *et al*., 2009; Jones *et al*., in press). Inland waters remain one of the most highly threatened ecosystems (Vörösmarty *et al*., 2010) and, regardless of the scenario of land-use change considered, the biodiverse freshwater catchments of the southeast United States are expected to see dramatic urban expansion (Martinuzzi *et al*., 2013b). Under some scenarios the southeast United States is also expected to see crop expansion that would further fragment and pollute critical freshwater habitats (Martinuzzi *et al*., 2013b).

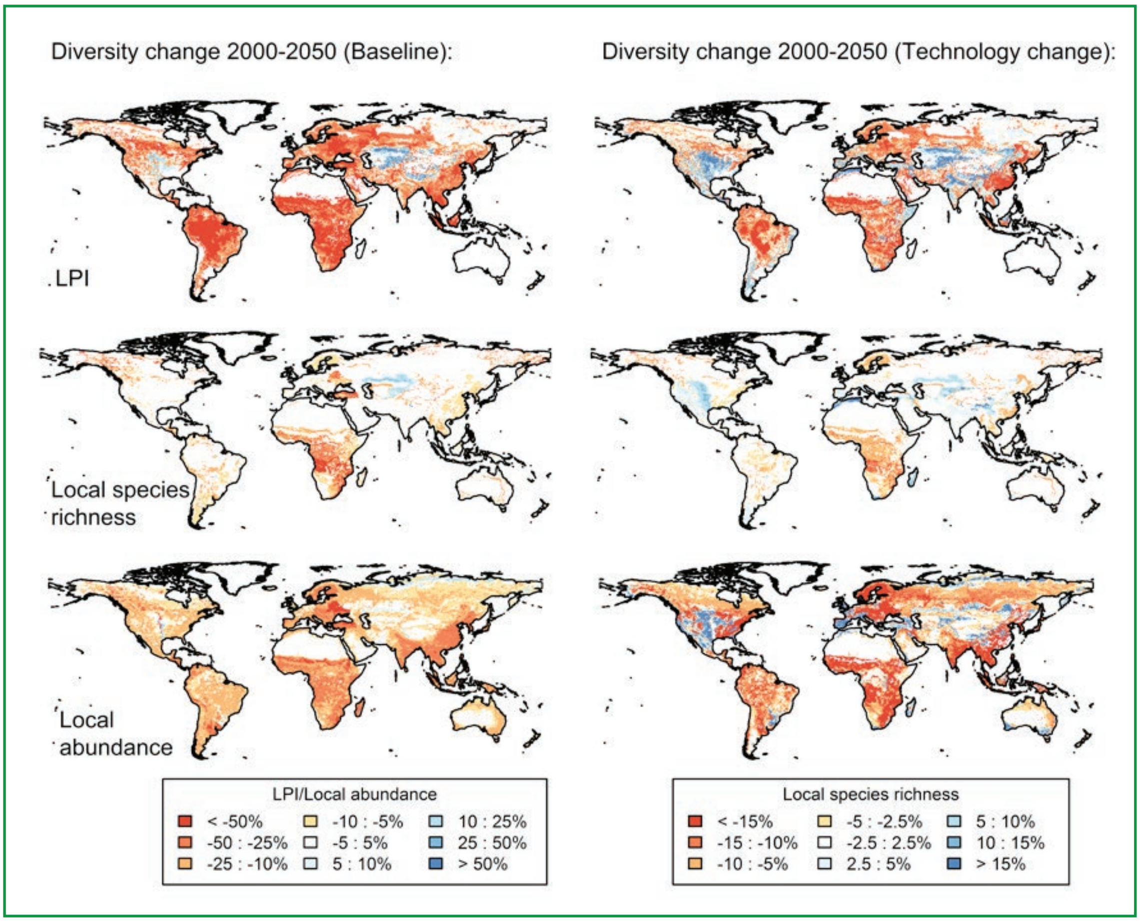


Figure 12.6. Modelled change in several measures of biodiversity – average population trends of carnivore and ungulate species, calculated following the methods used by the Living Planet Index (LPI) (Visconti et al., in review); average local species richness (Newbold et al., in review); and average local abundance of species (Alkemade et al., 2009) between 2000 and 2050 under a baseline scenario (left panels) and the 'technology change' Rio+20 scenario (right panels) and in Netherlands Environmental Assessment Agency, 2010). Local abundance and LPI measures included both direct effects of climate change and indirect effects (through land cover change); local species richness change accounted only for the indirect effects. LPI was calculated for each grid cell by aggregating population trends in projected population size within the cell for all carnivore and ungulate species. This contrasts with the LPI calculations in Figure 12.3 where the trend in population size for each species were calculated globally and aggregated across all species. In LPI calculations, the local extinctions where replaced with 1% of the maximum population size to avoid calculating a geometric mean with a zero. Because changes in LPI are sensitive to species richness, grid cells with <10 species of carnivore and ungulate species where removed from the LPI analyses.

A number of studies have focused on particular regions. Range shifts and contractions are predicted by 2050 for two-thirds of European breeding birds (Barbet-Massin *et al.*, 2012), for tree species in France (Cheaib *et al.*, 2012) and for Alpine plants (with an almost 50% average reduction in range size by 2100; Dullinger *et al.*, 2012). Contractions and shifts in the distributions of European plants, birds and mammals are expected to be similar across taxonomic groups, because sensitivity to climate change is not strongly correlated with phylogeny (Thuiller *et al.*, 2011). In Australia, 67% of Australian savanna bird species are predicted to suffer a contraction in their ranges. However, migratory and tropical-endemic birds are predicted to benefit from climate change with increasing distributional area. Richness hotspots of tropical savanna birds are also expected to move, increasing in southern savannas and southward along the east coast of Australia, but decreasing in the arid zone (Reside *et al.*, 2012).

Long-term projections of land use change under the Rio+20 scenarios indicate that habitat loss will continue to pose a threat to biodiversity under the business-as-usual scenario (Figure 12.3; A-D), particularly in Africa and Central Asia (Figure 12.6; Visconti *et al.*, in review). Regional models that predict the impacts of land-use change on species extinction similarly predict a worsening situation under business-as-usual scenarios: in the Brazilian Amazon 10% of species, on average, are predicted to become locally extinct as a result of forest loss, with a further 27% committed to extinction (Wearn *et al.*, 2012). In the mitigation scenarios, short-term trends in the diversity of ecological communities (see 'Projecting forward to 2020') are continued (Figure 12.3; A; D; Newbold *et al.*, in review), driven by continued land-use change. For population trends and extinction risk of carnivores and ungulates, the short-term gains begin to be reversed by 2050 as climate change overcomes the beneficial effects of reduced land-use change (Figure 12.3; C; D; Visconti *et al.*, 2011). Similarly, scenarios for the Brazilian Amazon that predict reduced rates of deforestation lead to reduced local extinctions of species (by 37-57% for actual extinctions and 61-82% for extinction debt, depending on the scenario adopted; Wearn *et al.*, 2012).

12.5 UNCERTAINTIES

There are, unavoidably, many uncertainties in the various methods used to make predictions about the future of biodiversity. However, in all of the studies reviewed here there is a high degree of confidence in the estimates of ongoing declines in biodiversity, and in predictions of future declines, at least under business-as-usual.

Status and trends

Estimates of past extinction rates are uncertain for several reasons. First, the number of field biologists is small relative to the number of species, and therefore extinction rates can be estimated only for a few well-studied and possibly atypical groups (mainly vertebrates), while extinctions can go undetected in species-rich but poorly studied groups (Balmford *et al.*, 2003). Second, being confident that a species is actually extinct requires levels of survey effort that very often exceed available resources even for very well-studied groups (Butchart *et al.*, 2006; Scott *et al.*, 2008). Finally, species do not immediately respond to human pressures, and extinction can be delayed for centuries (Tilman *et al.*, 1994). Even for species that are almost certainly extinct, knowing exactly when extinction occurred is difficult, and most known extinctions are accompanied by a range of likely dates of extinction.

All of the measures used to assess status and trends were biased toward vertebrate species in terrestrial environments, and therefore our knowledge of recent changes in the status of invertebrate species, and of all species in freshwater and marine environments, is much more limited.

Projections

There are fundamental differences between extrapolating the past trend of an indicator into the future, and modelling future trends based on scenarios of the underlying pressures, which leads to large differences in projected outcomes. The statistical extrapolations used to project trends to 2020 assume that the underlying processes continue on current trends, while the other models use scenarios of how the underlying pressures will change. For example in the case of the IUCN Red List Index, the extrapolations assumed a constant trend in the indicator, and therefore a further increase in extinction risk, while the more process-driven models assumed a constant trend in the pressures, a slower short-term decline in population size and therefore a reduction in extinction risk.

For scenario-based modelling, there is great variation in projected future extinction rates both within and between studies, with three factors explaining much of this variation. First and foremost, the degree of land use and climate change predicted by different scenarios: for example, Thomas *et al.* (2004) projected vertebrate extinctions of 11-34% for 0.8°C to 1.7°C global warming versus 33-58% for >2.0°C warming (the magnitude of these predicted losses has been disputed since, and they are based on species distribution models, which are subject to numerous sources of uncertainty – see e.g., Thuiller *et al.*, 2004; Araújo and Guisan, 2006). Second, an important contribution to the broad range of projections within studies is different assumptions about species life-history traits, especially with regard to dispersal ability (for example, projected extinction rates can range from 38% with unlimited dispersal ability to 58% with no migration; Thomas *et al.*, 2004) and habitat specificity (extinction rates from 7% with broad habitat requirements to 43% with narrow habitat requirements; Malcolm *et al.*, 2006). This emphasizes the need for research on these fundamental aspects of species ecology and their incorporation into global models (Foden *et al.*, 2013; Thuiller *et al.*, 2008). Third, there is a substantial degree of uncertainty in the climate and land-use change models themselves especially at the fine spatial scales often used in biodiversity modelling (Stock *et al.*, 2011), which is not generally quantified.

Species response models

A large fraction of the variation in predicted outcomes for biodiversity among studies arises from differences between modelling approaches. For example, Sekercioglu *et al.* (2008), using a logistic model of extinction risk as a function of range size, predicted ten times more birds extinction than Jetz *et al.* (2007), using a linear, mechanistic model of extinction as a function of habitat suitability. The assumed linear relationships between habitat and population decline, which underlie many predictions of global extinctions (e.g., Jetz *et al.*, 2007; Visconti *et al.*, in review), may lead to underestimates of species global extinction risk (Di Fonzo *et al.*, 2013). On the other hand, other uncertainties may lead to overestimates of extinction risk.

Studies using bioclimatic envelope models to predict climate change impacts tend to project larger range contractions and increases in extinction risk than other approaches. This is likely in part due to the largely untested assumption that species will not survive climatic conditions never experienced before, whereas species might adapt to climate change through phenotypic plasticity or micro-evolution (Charmantier *et al.*, 2008; Boutin and Lane, 2014). However, there are a number of other known limitations of species distribution models, which could contribute to the uncertainty (Araújo and Guisan, 2006).

Models based on the species-area relationship also tend to give estimates of high extinction risk (Pimm and Raven 2000; Thomas *et al.*, 2004) because they are based on the accumulation of species expected with sampling an expanding area, which may not accurately reflect the scaling of species extinction with reduced habitat area (Lewis, 2006; He and Hubbell, 2011; but see e.g., Pereira *et al.*, 2012; Pimm and Brooks, 2013). These models have in at least one case (Thomas *et al.*, 2004) been criticized for misapplying the IUCN Red List criteria (Akçakaya *et al.*, 2006). Species-area relationships also measure species committed to extinction. However, the lag time between being committed to extinction and actually going extinct may range from decades to centuries (Stork, 2010; Wearn *et al.*, 2012).

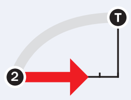
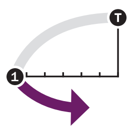
Spatially-explicit metapopulation models probably make more conservative and robust estimates of extinction risk by avoiding several of the assumptions described above (Pearson *et al.*, 2014), but are computationally intensive and require data available for only a fraction of species.

Global change models

Additional uncertainty in model projections arises from their coverage of threats affecting species. The Living Planet Index and IUCN Red List Index projections for large mammals (Figure 12.3; 12.6) only accounted for land use and climate change despite direct persecution being an important threat for these species. The PREDICTS model projections presented here (Figure 12.3; 12.6) were based on land-use change and indirect impacts of climate change through biome shifts. None of the terrestrial models reviewed here accounted for direct harvesting of species, and the Rio+20 scenarios did not include future projections of human population density, which could act as a proxy for pressure from direct harvesting in terrestrial environments. The qualitative differences in biodiversity outcomes predicted here under the Rio+20 scenarios would unlikely be affected by the inclusion of direct harvest, because several factors (low food security, poor access to food markets and a high proportion of people living in rural areas) mean that direct harvest of species is likely to be greatest in the business-as-usual scenario.

For marine species, the projections (Figure 12.5) focused on climate change as a driver. Addition of other threats, particularly fishing and habitat loss might modify the rate of local extinction.

12.6 DASHBOARD – PROGRESS TOWARDS TARGET

Target Elements	Status	Comment	Confidence
Extinction of known threatened species has been prevented		Further extinctions likely by 2020, e.g., for amphibians and fish. For bird and mammal species some evidence measures have prevented extinctions	Low
The conservation status of those species most in decline has been improved and sustained		IUCN Red List Index still declining, no sign overall of reduced risk of extinction across groups of species. Very large regional differences	High

Authors: Tim Newbold, Piero Visconti, Carlo Rondinini, William Cheung and Stephanie Januchowski-Hartley, with contributions from Andy Purvis, Stuart Butchart and Miranda Jones

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TARGET 13: GENETIC DIVERSITY

By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socioeconomically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.

PREFACE

The conservation and maintenance of genetic diversity (i.e., the variety of genes and genetic characteristics) as well as genetic resources (i.e., the genetic material of actual or potential value) fulfils a number of different objectives (Boettcher *et al.*, 2010). Genetic diversity in both plant and animal resources allows for the sustained ability of a breed or population to respond to selection to increase productivity, and allows breeds or populations to adapt to changing environmental conditions, like changes in climate, markets, management or husbandry practices, as well as changes in disease prevalence. Genetic diversity thus contributes to ensuring long-term food security. Furthermore, conservation of genetic diversity contributes to the preservation of particular cultural or historical values, and sustains the "bequest value" of livestock. Last but not least, the conservation of genetic resources also fulfils the right of an existing genetic resource to continue to exist (Boettcher *et al.*, 2010).

A loss of genetic diversity, including the loss of individual genes, as well as particular gene combinations (e.g., in locally adapted land races or breeds), is termed ***genetic erosion*** (FAO, 1997). The main causes of genetic erosion are the replacement of local varieties, habitat loss and overexploitation of species (FAO, 1997). Genetic erosion can be measured as the proportion of genetic diversity lost in current populations compared to earlier populations (Brown, 2008), and should be focussed on genes or genotypes of specific concern within regions or production systems.

Genetic vulnerability is the susceptibility of a crop (or a breed) to pests, pathogens or environmental hazards based on its genetic make-up (FAO, 1997), and is inversely related to locally present genetic diversity (Brown, 2008). The main reason for genetic vulnerability is the widespread replacement of genetically diverse traditional or farmer's varieties by homogenous modern varieties (FAO, 1997).

The assessment is mainly focused on the genetic diversity of domesticated animals and plants, with some information provided on genetic diversity of crop wild relatives (CWR), as well as forest genetic resources. The genetic diversity of wild marine and freshwater species, as well as species used for aquaculture is not considered in this assessment. For more information on marine and freshwater genetic resources, the reader is encouraged to consult the FAO report on the State of World Fisheries and Aquaculture (FAO, 2010), as well as the report on Status and Trends in Aquatic Genetic Resources (FAO, 2007). The genetic diversity of microrganisms and invertebrates, responsible for many ecosystem functions, for example nutrient cycling or pollination, is also not treated.

13.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

13.1.1 Status and trends

A number of different proxies or measures are used to determine genetic diversity or genetic variation. Differences between individuals can be characterised by, for example, comparing alleles at specific loci on the chromosome, or determining variation in enzymes or phenotypes. Variation between populations, breeds or land races can then be determined by comparing the distribution of alleles or phenotypes within the different populations, breeds or land races. However, the term 'breed' usually refers to a sociocultural concept, and not necessarily a distinct physical or genetic entity (Hoffmann *et al.*, 2013). Despite this, the variation among breeds may be summarised as a phylogenetic tree (e.g., Rowshan *et al.*, 2011; Pertoldi *et al.*, 2010; Martin-Burriel *et al.*, 2007). Other practical measures of genetic diversity include the number and frequency of species, land races or breeds, and the area occupied *in situ*, as well as the number of species adequately sampled in gene banks, and seed samples (accessions) within gene banks *ex situ* (Brown, 2008).

Measures of genetic diversity should account for both richness (number of different variants) and evenness (similarity of frequencies of different variants) (Brown, 2008). A low evenness indicates a dominance of only a few variants (Jarvis *et al.*, 2011).

13.1.1.i Animal Genetic Resources (AnGR)

As of 1 June 2012, a total of 8,262 livestock breeds (mammals and birds, existent and extinct breeds) were recorded in the Domestic Animal Diversity Information System DAD-IS (http://www.fao.org/dad-is), hosted by FAO (DAD-IS, 2014). In June 2012, 1 881 (~ 23%) of all breeds were classified as being at risk (critical or endangered), and 653 (~8%) were classified as being extinct. For about a third of the breeds (2,777, 33%), the risk status was unknown, a further 2,976 species (36%) were classified as being not at risk (Table 13.1).

Table 13.1: Risk status of Mammalian and Avian livestock breeds per region, based on data reported by National Coordinators for the Management of Animal Genetic Resources (AnGR) to DAD-IS by June 2012. Source: FAO (2013a).

	critical/ critical-maintained	endangered/ endangered-maintained	extinct	not at risk	Unknown	Total
All breeds	**693**	**1188**	**624**	**2976**	**2777**	**8262**
Avian	**261**	**466**	**64**	**580**	**930**	**2301**
Africa	8	12	2	69	132	**223**
Asia	16	26	5	206	228	**481**
Europe & Caucasus	197	384	56	164	334	**1135**
Latin America & Caribbean	3	8	0	14	126	**151**
Near & Middle East	33	6	0	15	33	**54**
North America	27	8	1	4	2	**43**
Southwest Pacific	0	4	0	7	42	**53**
International Transboundary breeds	10	18	0	101	32	**161**
Mammalian	**432**	**722**	**564**	**2396**	**1847**	**5961**
Africa	14	30	32	220	388	**684**
Asia	28	53	42	780	445	**1348**
Europe & Caucasus	349	535	446	861	383	**2574**
Latin America & Caribbean	14	28	21	93	341	**497**
Near & Middle East	0	5	5	84	109	**203**
North America	9	35	11	13	52	**120**
Southwest Pacific	14	16	6	17	93	**146**
International Transboundary breeds	4	20	1	328	36	**389**

To avoid the potentially misleading consequences of including breeds for which no updates of population data have occurred for many years, a 10 year cut-off point, after which breeds revert to the "unknown" risk-status category if population figures were not updated, was introduced, and applied on the data reported to FAO as of January 2014. This calculation method leads to a more realistic picture: about 16% of the approximately 8,200 breeds that have been reported to FAO as of January 2014 are classified as being at risk of extinction based on the most recently available population figures – 8% are already extinct. For another 54%, no population data are available and therefore risk status is unknown (FAO, 2014, pers. comm.)

Assessing the geographical distribution of threat to breeds is complicated by uneven data coverage, and differing recording histories. Although existing data show that breeds are particularly threatened in North America, Europe and the Caucasus (FAO, 2009; FAO, 2012; FAO, 2013a), and these regions also have the highest number of extinct breeds (FAO, 2012; FAO, 2013a), this is very likely a reflection of the long history of breed recording that has taken place. These regions have by far the highest number of breeds registered in the data base. Latin America, the Pacific and Africa have the highest proportion of breeds with unknown risk status (FAO, 2012; FAO, 2013a), making it difficult to assess the real risk and conservation needs for species and breeds in these regions. This if further exacerbated by the environmental changes these regions in particular are experiencing and will experience in future (IPCC, 2014). It is also likely that many of the animal breeds in use in these regions are not yet captured in the data base.

The mammalian species having the highest proportions of breeds at risks are rabbits (114 breeds out of 285), followed by, horses (202 of 831) and pigs (164 of 723). Sheep have the highest number of breeds at risk (242 of 1694), followed by cattle (257 of 1392), (FAO, 2012). Since documentation started in 1993, 184 cattle, 160 sheep, 111 pig and 89 horse breeds have been reported as extinct (DAD-IS, 2014). For many species and breeds, in particular deer, asses and dromedaries, no information on the risk status is available, making it difficult to put in place appropriate conservation measures (FAO, 2013a).

For avian species, chickens have the highest number of breeds at risk (481 breeds out of 1,480, corresponding to 36%), while ostrich (seven breeds out of 16), geese (71 of 192), pigeons (26 of 71) and turkeys (38 of 113) have the highest proportion of breeds at risk. For 924 avian breeds, the risk status is not known, in particular for partridge, pheasant and guinea fowl. To date, 62 chicken breeds have been reported as being extinct, but only very few cases for other species (ducks, geese, guinea fowl and turkey). About a quarter (659) of avian breeds has been classified as being not at risk (DAD-IS 2014).

Indicator: Genetic diversity of terrestrial domesticated animals

Currently, only the proportion of breeds classified as at risk, not at risk and unknown are included in the indicator. An increase in the percentage of breeds reported to the FAO categorized as at risk or extinct and a decrease in the percentage categorized as not at risk indicate a decline in livestock diversity. However, in interpreting this, it is important to bear in mind that breed diversity does not fully reflect genetic diversity, because it does not account for within-breed diversity or for how closely breeds are related to each other. Furthermore, breeds often are cultural concepts rather than physical entities, and these concepts differ from country to country, as does breed pedigree history. There is no clear definition of genetic level what constitutes a breed. Genetic diversity between and within breeds is thus highly variable, and quantitative characterisation at the genetic level is rather difficult (Boettcher *et al.*, 2010).

Data updates are insufficiently regular at present to allow for an accurate assessment of trends. However, many individual breeds continue to decline in numbers, and there has been an increase in the percentage of species at risk. The percentage of livestock breeds classified as being "at risk", i.e., classified as critical or endangered, shows an increasing trend (Figure 13.3).

According to a survey conducted by FAO (2009), the three main threats to livestock breeds and reasons for eroding AnGR are 1) economic and market drivers, 2) poor livestock sector policies and 3) poor conservation strategies for breeds at risk of extinction (FAO, 2009). Other factors include sociopolitical instability, lack of functional institutions, disease and lacking disease control and loss of labour force. The importance of these threats, however, differs between regions and breeds (FAO, 2009), as well as production system and intensity, and market levels (Hoffmann, 2011). Another factor impacting on livestock production is the encroachment of invasive species, be it (often noxious) weeds and grasses, as for example in the US (DiTomaso, 2000; Duncan *et al.*, 2004) or woody plants in African savannas (e.g., Dalle *et al.*, 2006).

In the coming years, economic and market drivers, and drivers for increased resource efficiency, will remain the main threats to livestock diversity (Hoffmann, 2011). Climate change has little direct influence on animal genetic resources to date, but might become more important in the future (FAO, 2009). Changes in local conditions, such as changes in temperature and rainfall regime, and increasing frequency of extreme events like droughts or floods, as well as disturbances (Hoffmann, 2010) may have negative effects on the animals directly, for example through increased heat stress (e.g., Sherwood and Huber, 2010; Hoffmann, 2010), or exposure to parasites (Mas-Coma *et al.*, 2008) or diseases (Hoffmann, 2010; Thornton *et al.*, 2009), or indirectly, through changes in availability of food resources (Hoffmann, 2010; Thornton *et al.*, 2009), water (Thornton *et al.*, 2009), or spread of invasive species (see Chapter 9), which may negatively impact on pastures. These indirect effects might be mitigated by food preservation or migration (FAO, 2009).

Furthermore, changes in livestock production systems towards intensification of production systems (including the internationalization of markets, and a shift from subsistence to commercial farming), coupled with increased productivity, have resulted in the livestock sector being dominated by a few high-producing breeds (FAO, 2009). As a result, the gene-pool of mammalian and avian species and breeds has narrowed, leading to genetic erosion. The reduction in genetic variability reduces the ability of a species or breed to respond to selection pressures, and to adapt to changing environmental conditions, and increases genetic vulnerability. This trend is, however, counteracted by various efforts to maintain rare and unusual breeds, be it through a focus on heritage breeds, to adapt breeds to changing ecological or sociopolitical conditions (e.g., Boutrais, 2007) or through efforts to improve or develop livestock breeds, e.g., cattle breeds that emit less methane (Shafer *et al.*, 2011). Traditional and heritage breeds (and their surrounding farming systems) often need to be maintained through active conservation efforts on national and subnational level. Many traditional breeds have characteristics that are becoming desirable, for example, breeds with lower fat meat, or specific wool or coat characteristics. Although traditional and heritage breeds are adapted to (current) local environmental and socioeconomic conditions, they contain the genetic variety to adapt to changing conditions and to create new breeds.

1.a.ii Plant genetic resources

Many factors contribute to the loss of plant genetic diversity in agricultural landscapes, affecting both crops and their wild relatives. One of the main threats to crop genetic diversity is a change in agricultural practices, in particular a shift to intensive production practices (FAO, 2010a), which rely on few varieties, breeds or species. Other factors leading to a decrease in crop genetic diversity include the replacement of local varieties, overexploitation, inappropriate legislation and policy, as well as pests, diseases and weeds (Akhalkatsi, 2012; FAO, 2010a). Crop wild relatives (CWR), are mainly threatened by land clearing, overgrazing, environmental degradation agricultural practices shifting towards intensification (Akhalkatsi, 2012; FAO, 2010b), as well as climate change (FAO & PAR, 2011).

However, traditional and subsistence farming systems still rely on a variety of diverse foods with a high level of genetic variation, and this genetic variety is also an important buffer against disease and environmental change (FAO, 2010a). Traditional varieties within the production systems also increase the farmer's option values, as it enables the crop populations to better adapt and evolve to changing environmental and economic conditions (Jarvis *et al*., 2008a). Maintaining genetic diversity of crops also contributes to conserving traditional local knowledge, and vice versa, as a loss of traditional knowledge (including language) has been linked to a loss in genetic diversity of indigenous crops (Kai *et al*., 2014; FAO, 2010a, see also Target 18).

A number of studies have been carried out to determine large-scale genetic diversity for important crops and their wild relatives (rice: McNally *et al*., 2009; Xu *et al*., 2011; soy bean: Lam *et al*., 2010; sorghum: Morris *et al*., 2013; maize: Hufford *et al*., 2013), determining the breeding history of the crops, as well as the relationship of genetic diversity between crops and their wild relatives. Mace *et al*. (2013) could demonstrate that, compared to genetic diversity in wild lines, very little genetic diversity has been captured in sorghum. This observation is a general phenomenon in crops, and has been observed for many different crops species (e.g., Harlan, 1992).

Reported genetic erosion in traditional crops and their wild relatives is greatest in cereals, followed by vegetables, fruits and nut and food legumes (FAO, 2010a). For example, many (indigenous) rice varieties are no longer cultivated in India (Rana *et al*., 2009); a number of coffee species as well as numerous coffee clones have gone extinct in the last decades (Labouisse *et al*., 2008); and the genetic variation within and among European maize varieties decreased significantly over the last fifty years (Reif *et al*., 2005). In Madagascar, a number of varieties of traditional rice are disappearing, genetic erosion of coffee has been observed, and a variety of manioc is extinct. In Madagascar, over a period of 20 years, 100 out of 256 crop varieties, and five crop species, have become extinct (Andriamahazo *et al*., 2009). *Ex situ* collections may also be vulnerable to genetic erosion. Over the years there has been a significant loss of diversity from a unique coffee collection covering a wide diversity of endemic species in Madagascar, with 46% of the accessions having been lost. In addition, the entire coffee collection at Ilaka Est was lost in a severe tropical cyclone and was completely abandoned in the early 1990s due to lack of budget for its rehabilitation and maintenance (Dulloo *et al*., 2009).

A farmer's decisions on which crop varieties to use are influenced by a multitude of socioeconomic factors, among them yield, pest resistance, market demand and nutritional value (Balemie and Singh, 2012), but also sociocultural preferences and suitability to local climate (Rana *et al*., 2009). One of the main factors leading to disappearance of traditional varieties and land races is the introduction of modern (often hybrid) crop varieties (Balemie and Singh, 2012), and a shift to intensified agriculture (Rana *et al*., 2009). Other factors include the loss of knowledge of traditional farming methods (Velásquez-Milla *et al*., 2011), or the loss of cultural identity (Perales *et al*., 2005). The loss of varieties increases the genetic uniformity in farmer's fields, contributing to genetic vulnerability of the crops.

Despite the reports of decreasing crop genetic diversity, other studies conclude that genetic diversity of crops is maintained (e.g., for maize and in peas in France, le Clerc *et al*., 2006; millet and sorghum in Niger, Bezançon *et al*., 2008). Two meta-analyses of 27 and eight crop species, respectively, found that, overall, crop diversity was maintained, if not increasing (van de Wouw *et al*., 2010). However, these studies only show the tendencies for cultivars over the 20th century, while the greatest losses in genetic diversity occurred in industrialised nations before 1900. There are also genetic shifts in the varieties being used, and adoption of "modern" varieties does not mean that traditional varieties are not maintained, rather, many small-scale farmers continue to plant traditional varieties alongside the modern varieties, thus increasing in-field diversity (FAO, 2010a).

Diversity trends in "commercial" (released) crops are also not consistent. Some studies report increasing genetic diversity in crops, while others report an initial decrease, followed by an increase. Overall, there seems to be no significant reduction in crop genetic diversity, or narrowing of the genetic base of varieties used (FAO, 2010a). Nevertheless, there is consensus regarding the occurrence of genetic erosion as result from a shift from traditional to modern production systems (FAO, 2010a). Although indicators tracking genetic erosion and vulnerability have been agreed upon in the Global Plan of Action for Plant Genetic Resources for Food and Agriculture (FAO, 1996; FAO, 2011), they are not uniformly or rigorously applied yet by national governments. However, given the current state of technology, regular assessment of between and within a varieties' genetic diversity can give an indication of how the genetic diversity changes over time (see e.g., Chakanda *et al*., 2012; Gonzalez Castro *et al*., 2013 and Hagenblad, 2013).

All crop species were developed, through selective breeding by humans, from wild species. Most crops thus still have many (closely) related species in the wild (Crop Wild Relatives, CWR). Maxted *et al*., (2006) propose the following definition for a CWR: "*A crop wild relative is a wild plant taxon that has an indirect use derived from its relatively close genetic relationship to a crop; this relationship is defined in terms of the CWR belonging to Gene Pools 1 or 2, or taxon groups 1 to 4 of the crop*". In tomatoes, however, genepool 3, for example, proved useful in breeding. In many cases, CWR have been (and are used) to improve the resistance and resilience and yield of crops (Honnay *et al*., 2012; Maxted *et al*., 2012). Despite their value and importance as a "*critical resource to sustain global food security*" (Maxsted *et al*., 2012), their conservation is, in many cases, not a priority, and very little information is available on their conservation and population biology (Honnay *et al*., 2012). Maxted *et al*., (2012) estimate that there are about approximately 50,000 CWR species worldwide, 800 of which are of highest conservation concern. This high number of CWR species, coupled with "*the fact that the responsibility for their conservation often falls in a void between the food and agriculture community (whose focus is primarily on crops and farming systems) and the nature conservation community (whose focus is primarily on habitat and rare species conservation)*" (Maxted *et al*., 2012), makes the conservation and management of the CWR even more difficult.

CWR are increasingly under threat from habitat loss, habitat fragmentation and degradation, and changing climatic and environmental conditions, and associated changes in disturbance regimes (e.g., Maxted and Kell, 2009), and they also are subject to genetic erosion (Maxted *et al*., 2012). Some progress has been made in the conservation of CWR in protected areas, but only relatively few countries actively conserve these species (FAO, 2010a), with recent efforts being undertaken in Armenia, Bolivia, Madagascar, Sri Lanka and Uzbekistan (Lane, 2005). A considerable number of CWR grow outside protected areas, and even as weeds in agricultural areas. These crop relatives are threatened by widening of cropping rows, the removal of hedgerows, overgrazing, and the use of herbicides or other weed control regimes (FAO 2010a, Hunter *et al*., 2012).

A further important threat to CWRs (and to land races) is gene flow from domesticated crops and the resulting introgression of alleles from related domesticated crops into the CWR gene pool. This introgression influences genetic diversity and evolutionary processes of CWR species (Ellstrand, 2003; Lu, 2013). A considerable number of studies have provided evidence of introgression from crops into their CWRs, for example, in Zea Mays (Bitocchi *et al*., 2009), Sorghum bicolor (Morrell *et al*., 2005), Coffea arabica (Aerts *et al*., 2013) and Vitis vinifera (De Andres *et al*., 2012). However, this introgression only lasts when domestic genes bring some selective advantage to the wild population. In turn, introgression of wild genes to the benefit of some crop species, for example in pearl millet for pest resistance or adaptation to abiotic stress has been observed (e.g., Sanoussi *et al*., 2011).

In a recent assessment of 572 native European CWR of high priority human and animal food crops, more than a quarter were classified as at risk, and nearly a third as data deficient. Although more than half of the species were regarded as being of least concern on European level, a third of those are threatened at national level (Kell *et al*., 2012). In Bolivia, the first country to publish a Red List dedicated to CWR, 45 out of 152 species were classified as threatened (Hunter and Dulloo, 2009).

As the awareness about the risks faced by plant genetic resources has increased together with a growing capacity derived from improved biotechnologies for using the existing diversity, *ex situ* conservation efforts have been boosted across the world. This is reflected by the increase in the number of gene banks storing plant genetic samples (+33%), as well as genera (+81%) and distinct/unique accessions (+27%) conserved in gene banks over the period 1995-2009 (FAO, 1997; FAO, 2010a; FAO, 2014). There are now more than 1, 750 individual gene banks worldwide, maintaining about 7.4 million accessions. The *ex situ* crop collections enrichment index, measuring the bio- and geographical diversity contained within *ex situ* collections, has been steadily increasing over the last 60 years (Figure 13.1).

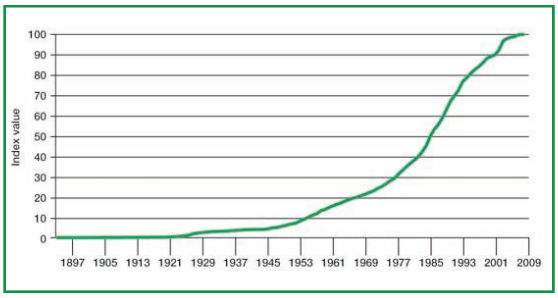


Figure 13.1: *Ex Situ* Crop Collections Enrichment index, measuring the bio- and geographical diversity contained within *ex situ* collections. Source: Dataset pooled from EURISCO (European National Inventories), USDA-GRIN, ICRISAT, CIAT and SINGER (excluding ICRISAT and CIAT) data.

To safeguard the genetic diversity of crops for future generations, countries have initiated the establishment of national seed banks. These include the seed bank of the N.I. Vavilov institute for Plant Industry (VIR), housing more than 322,238 accessions (2010 report of Russian Federation to FAO), the United States Department of Agriculture's National Plant Germplasm system (NPGS), hosting currently over 53,600 accessions (United States country report 2010 to FAO), and Navdanya, the Indian network of seed keepers and producers that connects 111 community seed banks (Navdanya, 2014). In addition, international seed storage units were created. The Norwegian Ministry for Agriculture and Food, in partnership with the Nordic Genetic Resource Center and the Global Crop Diversity Trust, has established the Svalbard Global Seed Vault. To date, there are 824 625 seed samples (accessions) of 470 botanical species are stored in the vault (Nordigen, 2014). The Millennium Seed Bank Partnership currently stores nearly 2 billion seeds of 33 491 species, 12 of which are globally extinct (Kew 2014). However, these storage units rely on active gene banks with *ex situ* collections to obtain seed material, and have very little activity in regeneration of material.

While these efforts are noteworthy for conserving species of particular importance, be it crops, or species of particular conservation concern, the *ex situ* conservation of species has nevertheless drawbacks (Schoen and Brown, 2001). In particular, species conserved in seed banks are not exposed to genetic change, or experience any evolutionary pressures through the environment or biotic interactions. As a result, species may be maladapted to any environmental shifts that might have occurred since seed storage should they be brought back in the wild (Schoen and Brown, 2001). Also, seeds of large forest trees and fruit species, especially those of the old world tropics, are recalcitrant, and their regeneration causes some technical challenges. It should also be noted that, as a rule, the genetic diversity of CWRs that can be preserved *ex situ* is rarely representative of the full genetic diversity of such species. It is thus important that such species are primarily preserved *in situ*, especially in natural areas.

For agricultural crop varieties (and their relatives), seed banks of agricultural crop varieties (and their relatives) are a very useful tool, as they provide sources of genetic diversity for crop improvement, and replacement of seeds for local varieties that were lost due to catastrophes (Schoen and Brown, 2001). For wild plant species, *ex situ* conservation is also very valuable, not only as last resort for species at the brink of extinction. As CWR are of interest for breeding efforts, their *ex situ* conservation makes them available to the scientific and breeding community.

Despite all collection efforts, the gene bank CWR collections of the world's major crops are considered incomplete (Crop Wild Relatives and Climate Change, 2013). Of 1,089 CWR species analysed, only 51 (5%) required no further collection, while 763 (70%) were considered High Priority Species, i.e., should be the focus of collection and storage (Crop Wild Relatives and Climate Change, 2013). Among these species are banana and plantain, sorghum and cassava, all important food sources for developing countries, but also apple and sunflower. Regions for priorities for collection include the BRIC countries, the Austral-Asia region, as well as part of the Mediterranean and North America (Crop Wild Relatives and Climate Change, 2013). To aid *in situ* and *ex situ* conservation planning of CWR at global, regional and national level, the Harlan and de Wet CWR inventory (http://www.cwrdiversity.org/checklist/) was established (Vincent *et al*., 2013). This inventory contains 1 667 priority taxa in 173 crop complexes, covering all regions of the world.

Another very important factor to consider when regarding plant genetic diversity is plant-microbe relationships that can improve the resistance of host plants to a wide variety of stresses and stressors, like disease, drought, salinity, nutrient shortages, and extreme temperature. Microorganisms can thus play beneficial roles towards plant preservation and plant genetic diversity conservation. Arbuscular myccorhizal fungi (AMF), for example, live with plant roots, from which they send out filaments that collect critical nutrients for their host plants. Bacteria with a gene encoding a certain enzyme can protect host plants against a variety of stresses, including drought flooding, heavy metals, high salinity and pathogens (Reid, 2011).

13.1.1.iv Forest genetic resources

Forest genetic resources (FGR) include the variation within forest tree species, and woody perennial shrubs. This genetic diversity underpins the vast majority of terrestrial biodiversity and is fundamental to ecosystem services and human livelihoods. The genetic diversity within forest species is vulnerable to habitat fragmentation, overexploitation and associated reduction in population size. Fragmentation, loss of seed dispersal agents and disruption of plant pollinator interactions, can all compromise FGR's (Kettle, 2012; McConkey *et al.*, 2012). These factors coupled with climate change and reduced species distribution, unsustainable logging, increase in the threat status of some 8000 tree species (Oldfield *et al.*, 1998) and increasing invasion of alien plant species, all lead to increasing threats to FGR's. Yet the scales of fragmentation and loss of forest habitat continue at alarming rates. Nearly 50% of temperate broadleaf and mixed forest biomes and nearly 25% of tropical rainforest biomes fragmented, degraded or converted (Wade *et al.*, 2003). Tropical and Boreal deforestation continue at alarming rates (Hansen *et al.*, 2010). Reductions in the distribution and abundance of tree species represent the most practical operational indicator of loss of FGR's.

The FAO have assumed the responsibility for development of indicators for FGR's and this recently lead to the publication of the State of the Worlds Forest Genetic Resources (SoWFGR's). At the European Scale there is a considerable effort to both establish operational indicators and detailed databases of FGR's through networks such as the EUFORGEN[1] and EUFGIS. The dynamic conservation of FGR's currently includes some 86 species, and 1967 units over 33 European countries (Lefevre *et al.*, 2012). Over this range Mediterranean and Boreal forest biomes are the least well represented, but only < 2% of conserved population are considered at risk. At the Global scale our knowledge is much poorer, not least due to the orders of magnitude larger numbers of forest tree species. The MAPFORGEN[2] network for Latin America currently includes a database of 100 tree species, but lacks information on extent or vulnerability of genetic diversity in these species. Other regional networks such as SAFORGEN for South Africa and APFORGEN[3] for Asia Pacific have been useful in generating regional action but have yet to establish clear mechanisms or indicators for status of FGR's. Monitoring of FGR's is still significantly lacking, but should be a priority group for advancing the monitoring of genetic resources, better positioned than numerous other taxonomic groupings e.g., soil organisms.

Footnotes

[1] http://www.euforgen.org/

[2] http://www.mapforgen.org/

[3] http://www.apforgen.org/

13.1.2 Projecting forward to 2020

13.1.2.i Animal genetic resources

The projection for the proportion of livestock breeds classified as being "at risk", i.e., classified as critical or endangered shows an increase with a slowing rate. (Figure 13.2).

This indicates that the threats identified in section 13.1.1.i continue to negatively impact on breed persistence in the near future. A growing demand of products of animal origin further shifts production systems toward intensification, using a limited number of (highly) productive breeds. This shift displaces traditional and locally adapted animal breeds. Whether this trend can be stopped or reversed depends on the efficiency and efficacy of strategies put in place to combat threats, and measures promoting traditional livestock breeds.

However, the slowing of the rate can be interpreted as positive impact of implementing the Global Plan of Action for Animal Genetic Resources (FAO, 2007). A report on the status of implementation of the Global Plan (FAO, 2012a) revealed gaps in policies, institutions and capacity building related to genetic diversity of terrestrial domesticated animals and severe problems in funding the implementation of relevant actions in most of the reporting countries. Therefore lack of policies considering animal genetic diversity, lack of institutions, capacity building and funding can be seen as the main hindering factors for achieving Aichi Biodiversity Target 13.

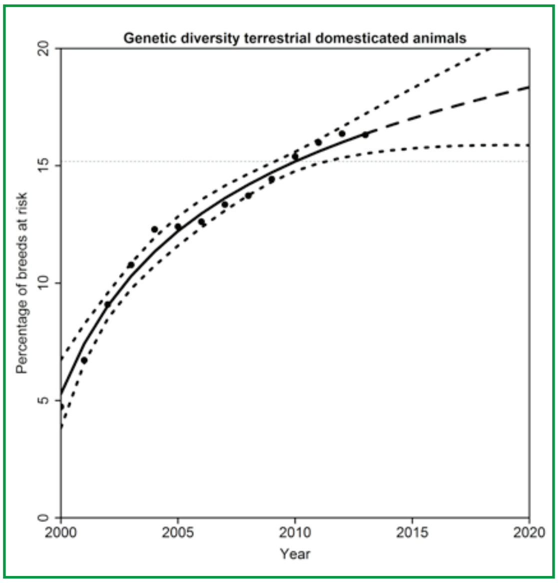


Figure 13.2: Statistical extrapolation of percentage of terrestrial domesticated breeds classified as at risk to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Source: FAO (2014).

13.1.2.ii Plant genetic resources

As for animal genetic resources, based on growing demand for food, a continued shift to intensification of agriculture will continue to reduce genetic diversity of (traditional) crop species. The loss of knowledge of traditional farming systems further exacerbates the impact, increasing the genetic vulnerability of species.

Crop wild relatives continue to be threatened by habitat loss and degradation (Maxted *et al*., 2013; Hunter *et al*., 2012), changes in land use, agricultural intensification and invasive species, as well as gene flow and intragression of alleles from crops. Climate change is likely to exacerbate the vulnerability of the species to these factors (Crop Wild Relatives and Climate Change, 2013). Reduction or shifts in genetic diversity in CWR are the result of human-induced environmental changes (Maxted *et al*., 2013; Vincent *et al*., 2013). Another important threat to plant genetic resources is the lack of awareness of the importance of these CWR species, as well as local (traditional) varieties and landraces of crops (Akhalkatsi *et al*., 2012).

13.1.2.iii Forest genetic resources

Despite the inclusion of FGR's in many sustainable forest management plans for example through the ITTO and the Montreal Process, consideration of genetic factors or conservation of FGR's remain poorly implemented or lack clear operational guidelines in forest management (Janolene *et al*., 2014). The Global Plan of Action for FGR's was agreed upon at the 14th Regular Session of the Commission on Genetics Resources for Food and Agriculture in 2013. Although priority areas for action were identified including - access to information on FGR's, *in situ* and *ex situ* conservation and sustainable use of FGR's, clear mechanisms for integrating such action into forest management are lacking.

13.1.3 Country Actions and commitments[4]

Slightly more than half of the NBSAPs examined contained targets or similar commitments related to genetic diversity. For the most part these targets generally relate to the protection or sustainable use of genetic diversity. A number of countries have set targets which address both extinction risk and genetic diversity. Further a number of countries, such as Belarus and Suriname, have established targets or similar mechanisms which are related to preventing biotechnology or genetically modified organisms from negatively affecting genetic diversity.

Overall the targets that have been established focus on cultivated plants and on domesticated or farmed animals. There is comparatively less emphasis on conserving the genetic diversity of wild relatives and socio economically or culturally valuable species. Further few countries have set targets related to the development or implementation of strategies to minimise genetic erosion or to safeguard genetic diversity.

13.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

13.2.1 Actions

13.2.1.i Animal genetic resources

To reach the Aichi Biodiversity Target, it is necessary to develop appropriate conservation strategies for breeds that are at risk. Options include *in situ* and *ex situ* in vitro conservation, i.e., the conservation of breeds in their production environment (where they can adapt to changing climatic conditions) and in gene banks, where they can be accessed to reconstitute a breed (FAO, 2009). In-vitro conservation measures are most appropriate to maintain breeds that occur at low population sizes (small number of individuals, FAO, 2009).

Measures to maintain indigenous breeds, or improve the conservation status of breeds also need to be implemented at governmental level. There is currently a lack of incentives and public policy to maintain indigenous breeds, and appropriate policy tools will need to be developed on national and international levels. Ideally, these conservation policies should be cost-effective to ensure broad implementation (FAO, 2009). Furthermore, there is also need for market-based or regulatory tools to protect indigenous breeds (FAO, 2009), consistent with relevant international obligations. Providing support and incentives to local communities to maintain traditional farming systems will ensure the persistence of traditional breeds and landraces and their genetic diversity.

Footnote

[4] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Democratic People's Republic of Korea, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Suriname, Switzerland, Timor Leste, and Tuvalu. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

To combat the erosion of animal genetic diversity, and to promote the sustainable use of animal genetic resources (AnGR), the FAO developed a Global Plan of Action for Animal Genetic Resources (FAO, 2007a), which was adopted by the international community in 2007. Intended to contribute significantly to achieving Millennium Development Goals 1 and 7, the indicators and targets for animal genetic resources developed also support achieving Aichi Biodiversity Targets 4 and 13 (FAO, 2013a). The plan includes 23 strategic priorities for action that are grouped in four areas: 1) characterization and monitoring; 2) sustainable use and development; 3) conservation; and 4) policies, institutions and capacity-building (FAO, 2007a).

Based on this strategic plan, many countries are either preparing National Action Plans, or are taking steps to improve the management of AnGR (FAO, 2013c). Nevertheless, despite the significant impact of the Global Plan of Action, the task of improving the management of the world's AnGR remains far from complete. The reason for this lies mainly in a lack of sufficient financial resources (especially in developing countries, FAO, 2013a), but also in low levels of collaboration between countries, a lack of established policies and legal frameworks, and a lack of strong institutional and human capacity for planning in the livestock sector (FAO, 2013c).

13.2.1.ii Plant genetic resources

The Second Global Plan of Action (GPA) for Plant Genetic Resources for Food and Agriculture (PGRFA) was adopted by the FAO council in November 2011 (FAO, 2011a). Updating the Global Plan of Action for the Conservation and Sustainable Use of Plant Genetic Resources for Food and Agriculture from 1996, it reaffirms the commitment of governments to the promotion of plant genetic resources as an essential component for food security through sustainable agriculture in the context of climate change. It includes 18 priority activities, grouped under four areas: 1) *In situ* conservation and management; 2) *ex situ* conservation; 3) sustainable use; and 4) building sustainable institutional and human capacities. Following the adoption of the Second GPA, the Commission adopted three targets for PGRFA (conservation, sustainable use and capacity), which contribute to Aichi Biodiversity Target 13, and a set of indicators for monitoring the implementation of the 18 priorities of the Second GPA (FAO, 2013d).

The genetic diversity maintained by farmers *in situ* and by gene banks *ex situ*, is fundamental in achieving global food security (Jarvis *et al*., 2011). Sustainable use of plant diversity includes breeding to develop new crop varieties adapted to the changing needs of farmers. For example, the FAO coordinates the Global Initiative on Plant Breeding, and various CGIAR research programmes also include activities on plant breeding and pre-breeding.

A number of actions to safeguard PGRFA have been proposed by the Commission on Genetic Resources for Food and Agriculture. These include (FAO, 2010b): 1) the strengthening of plant breeding capacity and expansion of plant breeding programmes to develop varieties with traits that can cope with climate change; 2) raising awareness of the importance of PGRFA diversity (plant genetic resources for food and agriculture) and their contribution to local food security (this links also to Target 1); 3) securing the diversity of crop wild relatives and underused species relevant for food and agriculture; and 4) the development and adoption of national programmes, laws and regulations governing PGRFA management (this action also contributes to achieving Target 3).

Jarvis *et al*., (2011) propose a framework to support the conservation and use of traditional crop varieties within agricultural production systems. This framework rests on four pillars, namely 1) on-farm diversity assessment, 2) access to diversity, 3) improving use through better information, materials and management, and 4) benefiting from the use of local crop genetic diversity. Actions taken within these pillars will take place on different scales and levels, and will have to be adapted to the specific circumstances of each farm(er). Implementation of actions depends on the farmers (and farming community) having the knowledge in order to be able to evaluate the benefits of a certain action, which in turn requires actions on national, regional and international level to strengthen local institutions. This then ultimately enables farmers to take a greater role in the management of their resources (Jarvis *et al*., 2011).

On farm level, genetic resources will need to be managed accordingly, e.g., through participatory plant breeding programmes, or "informal" exchanges of seeds (FAO, 2010a). This, however, is only possible, where Intellectual Property (IP) is not an issue. In Bangladesh, Ecuador, Morocco, Nepal and Tunisia, for example, national legislation supports the conservation and use of traditional crop varieties (FAO, 2010a). Participatory breeding programmes are particularly prevalent in Africa and Latin America and growing in developed countries (e.g., Pautasso *et al*., 2013). Examples from India (Witcombe *et al*., 2003; Virk *et al*., 2003), North Africa and the Middle East (Ceccarelli *et al*., 2001) and Mexico and Honduras (Smith *et al*., 2001) show that participatory breeding programmes contribute to the development of varieties that not only are adapted to local conditions, but also have yields that are higher than those of traditional land races. Farmers also benefited from the crop improvement. In Uruguay, the National Native and Local Seed Network (La Red Nacional de Semillas Nativas y Criollos) has been established to rescue and revive native or traditional plant varieties, to increase the availability of seed whether for consumption or to supply local markets, in the context of strengthening food security. Empowerment of local communities and farmers to sustainably manage agricultural diversity, as well as using traditional knowledge systems can make a substantial contribution to maintaining crop diversity (FAO, 2010a).

In Sri Lanka, efforts for *in situ* conservation of plant genetic resources include the establishment of an atlas of 46 crop species (Muthukuda and Wijerathana, 2007), supported by on-farm conservation programmes and 1.3 million home gardens in different agri-ecological zones. Furthermore, seeds of rice, grain legume, vegetable and oil crop varieties are stored in a number of gene banks. Uzbekistan has established a national *ex situ* and *in situ* conservation and management programme for crops and CWR (Abdukarimov *et al*., 2004), and is supporting farmers in maintaining traditional crop varieties. Land races are best preserved in *ex situ* genebanks or by farmers and gardeners that have the resources and the legal frame to work with such cultivated material.

To safeguard genetic diversity of CWR, the establishment of a global network for the conservation of CWR has been recommended (Maxted and Kell, 2009, FAO, 2010a), that covers both *in situ* and *ex situ* conservation. In addition, systematic CWR conservation at international, national and local protected level needs to be implemented (Heywood *et al*., 2007; Maxted *et al*., 2009; FAO, 2010a), and integrated with existing plant genetic resource programmes (Heywood, 2007). Prerequisite for this is a consensus on what constitutes a CWR, the development of an inventory of candidate species, as well as a prioritisation of species identified (Heywood *et al*., 2007). Maxted and Kell (2009) provide such a list of priority CWR for 14 crop species across the globe. The implementation of CWR protection requires collaboration between countries, as well as the agriculture and environment sector across all levels of governance. Some countries, like Australia and South Africa, require permits and licenses to collect plants, and plant genetic resources can only be accessed under agreements. Successful conservation of CWR also requires raising awareness of the importance of CWR and their conservation for agriculture and future food security across sectors (Heywood *et al*., 2007). It has further been suggested to devise and implement an early warning system for genetic erosion for all plant genetic resources (FAO, 2010a).

Nevertheless, CWR are not adequately protected in conservation areas (Hunter *et al*., 2012), as monitoring of and appropriate management practices for CWR are not considered in protected area management plans (Phillips *et al*., 2014). Very few examples exist of *in situ* conservation of CWR in the tropics (Hunter *et al*., 2012) or in non-tropical areas. One of the earliest examples is the protection of wild relatives of wheat *in situ* in northern Israel. India established a gene sanctuary for citrus species; in Vietnam, a reserve for the conservation of wild relatives and landraces of rice, litchi, citrus and tea was created; and in Mexico, a Biosphere Reserve for a wild relative of maize was established (http://manantlan.conanp.gob.mx/). More recently, CWR management plants were developed and implemented in Armenia, Bolivia, Madagascar, Sri Lanka and Uzbekistan (Hunter *et al*., 2012), or Cyprus (Phillips *et al*., 2014). On pan-European level, following on the Convention on Biological Diversity Global Strategy for Plant Conservation, CWR related targets were included in the European Strategy for Plant Conservation 2008 – 2014 (Maxsted *et al*., 2013). Despite these national and regional efforts, there is still a need for a global approach to conservation priority and threatened species of CWR (Hunter *et al*, 2012).

Ex situ conservation is equally (or even more) important for many obligatory weeds that are extinct in modern agroecosystems, for example *Agrostemma githago*, the common corncockle. Many of these species have ingredients that are of economic or medicinal value, are adapted to cultivation and are at the verge of becoming crops.

Box 13.1: *Ex situ* conservation of Hungarian vascular wild plants

To preserve the seeds of the wild vascular flora of the Pannonian biogeographical region of Hungary over the long-term, a five-year project *"Establishment of the Pannon Seed Bank for the long-term ex situ conservation of Hungarian vascular wild plants"* was initiated in 2010 with co-financing from the Hungarian Ministry of Rural Development and the EU Life+ Fund. The project is coordinated by the Centre for Plant Diversity, which is the largest seed bank in Hungary implemented together with the Ecological and Botanical Institute (Vácrátót) of the Hungarian Academy of Sciences and the Aggtelek National Park Directorate.

The project aims to achieve this goal through expanding the current functions of Hungary's main agricultural gene bank, the Centre for Plant Diversity. The establishment of a joint seed bank for the agricultural and wild flora will conserve the genetic diversity of the Pannonian biogeographical region's entire flora, including the wild flora as well as crop and vegetable plants serving human nutrition.

By the end of 2014, at least 800 species (around 4,000 accessions) – of the species of the wild native flora will be collected. A Base Collection will serve as a long-term conservation facility of reserve samples, while an Active Collection will contribute to facilitating research and the distribution of research material. Duplicate stores of both collections ensure the safety of the collections.

A model reintroduction of certain species of the sand steppe community typical to the Pannonian biogeographical region is carried out at Natura 2000 priority habitats (Pannonic sand steppes and inland dunes) of the Kiskunság National Park in Central-Hungary. This project illustrates the value of the preservation of genetic material, and how it can be utilized in nature.

Source: Hungary 5th National Report to the CBD, 2014

13.2.1.iii Forest genetic resources

Better protection of forest reserves, better integration of genetic knowledge in to forest management, collection of material for *in situ* and *ex situ* conservation are all priority action areas. Focusing on highly vulnerable and valuable forest species should be a priority.

A lack of understanding within the forest sector of the importance of genetic diversity for natural forest regeneration and seed production, as well as adaptive potential for environmental change is a major barrier in some regions. Especially in many tropical regions there is an urgent need for transfer of scientific understanding about genetic erosion in forest trees. This needs to be converted into relevant and operational guidelines for forest managers to ensure sustainable forest management. Current efforts for conservation of European FGR's are likely to be powerful and effective. However at the global scale trends look much bleaker. One major constraint it that *ex situ* conservation in seed banks are on the whole impractical for large and recalcitrant tropical tree species. General flowering and mast fruiting also place limitation of the ability to conserve FGR's with a need to improve infrastructure to effectively respond when such reproductive event occur. Clear guidelines on the management of FGR's for restoration and conservation of threatened tree species are needed. This includes ensuring that forest reserves conserve large enough populations of rare species to conserve genetic diversity, or maintain these in agroforestry landscapes.

13.2.2 Costs and cost-benefit analysis

The decreasing number of crops, as well as the increasing homogenization of crop varieties around the globe is considered as one of the threats to global food security (Khoury *et al.*, 2014). Maintaining genetic diversity of plant and animal genetic resources in agriculture is one important factor in maintaining food security for coming generations. Traditional and locally adapted breeds and crop varieties may contain the genetic material to adapt to changing environmental conditions, and to improve existing commercial breeds and crops. Traditional crops also act as a buffer and provide food security when modern crops are failing due to environmental changes (Shava *et al.*, 2009). Indeed, recent studies suggest that one of the responses of poor rural communities to climate change is to increase the use of traditional materials in their production systems (Jarvis *et al.*, 2011). In addition, a diversity of foods, including traditional varieties and wild species, provide important nutrients (Grivetti and Ogle, 2000; Ebert, 2014; Khoury *et al.*, 2014). Traditional crops, if providing adequate yields, may also provide a significant source of income to rural communities, contributes to their sustainability (Dan-Azumi, 2010). Plant and animal genetic resources also support provisioning, regulating and supporting services. These include, for example, the regulation and control of pest and diseases, the maintenance of pollinator diversity, and the support of below-ground biodiversity and soil health (Jarvis *et al.*, 2011). The provision of these ecosystem services contributions to the reductions of financial and health risks associated with high levels of agricultural inputs, such as fertilizer and pesticides to farmers and the environment.

Local livestock breeds provide various ecosystem services, mostly *provisioning services* (typical, quality products), *supporting/regulating services* (habitat) and *cultural services*. This can be regarded as an opportunity to support the conservation and sustainable use of local livestock breeds and associated agroecosystems, through adding value. Products of local breeds can be valorized in food value chains putting emphasis on traditional know-how, specific tastes, or the image and cultural identity of local communities and associated agroecosystems. Local breeds can also be the basis for products with a regional identity for which there is increased consumer interest.

Other incentives to maintain livestock and crop diversity may include payment of ecosystem services schemes for carbon sequestration, rangeland rehabilitation, or breed conservation. While some incentives may address the public good nature of the ecosystem service in question and will require public funds, there are also opportunities for market driven incentives, for example through ecotourism.

CWR are important to maintain food security under changing climatic and environmental conditions. They contain the genetic material to improve the adaptation of crops to new environmental conditions, or to develop new crop varieties. In Nepal, individual farmers were willing to pay US$4.18 for *in situ*, and US$2.20 for *ex situ* conservation per year for rice landraces (Poudel and Johnsen, 2009), with willingness to pay correlated with education and food sufficiency level. CWR have an estimated economic value of US$115 billion per year worldwide to improving food production (Pimentel *et al.*, 1997), underlining their importance to the global economy and the need for their effective conservation (Phillips *et al.*, 2005).

The value of (plant) genetic resources is composed of use value and non-use value (Smale and Koo, 2003). Use value includes the current and future direct use value, indirect use value and option value, while non-use value includes the existence value and bequest value of genetic resources. The direct use value is derived from food, fibre and medicinal products to which these genetic resources contribute. Indirect use value includes the contribution of genetic resources to surrounding habitats and ecosystems, while the option value provides the flexibility to respond to future demands (Smale and Koo, 2003).

The HLP estimated that US$550 million to US$1,400 million in investments, US$15 million to US$17 million in annual recurrent costs, and between US$80 million to US$190 million in annual expenditure are required to achieve the target. The *ex situ* conservation of wheat and maize, stored in perpetuity in the CIMMYT seed bank has been estimated to cost US$8.86 million to US$3.87 million for 123,000 wheat accessions and US$4.99 million for the 17,000 maize samples (Pardey *et al.*, 2001). Providing these accessions free of charge would cost an additional US$5.28 million in perpetuity. Achieving the target will also contribute to achieving Aichi Biodiversity Targets 2, 7, and 12.

13.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

13.3.1 Animal genetic resources

If economic and market drivers and poor livestock sector policies are not counteracted by appropriate incentives and regulations at national and international levels, the number of breeds at risk will continue to rise, and, more breeds might become extinct.

Hoffmann (2013) found a spatial overlap between breeds locally adapted to extreme climates and scarce and coarse feed resources, and rangeland and mountain ecosystems, most of which are ecologically vulnerable and inhabited by poor, marginalized peoples. Targeting such areas in particular, would address poverty reduction, and at the same time, conserve wild biodiversity and contribute to sustainable use of locally adapted breeds. Local breeds can be an important tool in managing landscapes and habitats (e.g., Fraser *et al.*, 2014; Auffret *et al.*, 2012), and, at appropriate grazing levels, they can maintain or even enhance vegetation diversity (e.g., Petersen *et al.*, 2014; Garcia *et al.*, 2013).

13.3.2 Plant genetic resources

Despite the efforts to maintain the genetic diversity of crops, and the importance of traditional crops and their wild relatives for food security, plant genetic diversity is likely to decline, as many efforts to conserve this diversity are insufficient, be it a lack of financial resources, or economic, social and political constraints.

Maintaining genetic diversity of crops also helps conserve traditional local knowledge (FAO, 2010a), while maintaining traditional knowledge and indigenous languages will contribute to halting the loss in genetic diversity of indigenous crops (Perales *et al.*, 2005, FAO, 2010a). Here, a clear link with Target 18 exists. Appropriate on-farm management of indigenous crops, e.g., through agro-biodiversity reserves, can contribute to the conservation of cultivated diversity and associated agricultural practices and knowledge systems. This will need to be done in such a way as to preserve the freedom of the farmer to choose crops and production systems that serve their needs.

A number of options have been suggested for the conservation of diversity in agricultural systems (FAO, 2010a), including adding value to crops by characterizing local genetic material (e.g., in the Czech Republic, through financial support to farmers), improving local genetic material through breeding (e.g., in a number of European countries, among them Spain, Italy, Germany and the UK), increasing consumer demand for local varieties through market incentives and public awareness (in a number of developing countries in Africa, Asia and Latin America), and providing improved access to information and material (through pilot project in a number of countries across the globe).

Many countries have established effective *ex situ* conservation programmes for CWR (FAO, 2010a). In Armenia, as part of the countries network of protected areas, some protected areas were established with the particular aim of protecting CWR (Akhalkatsi *et al*., 2012). Erebuni State Reserve, for example, was established to protect more than 100 varieties of wild rice, and their habitats. The establishment of protected areas is coupled with specific conservation, management and monitoring measures for these species. In addition, efforts are underway for the *in situ* management of CWR outside protected areas in forests, pastures and grasslands (Akhalkatsi *et al*., 2012). Unfortunately, inventorying and monitoring efforts often fail due to a shortage of resources, absence of proper coordination, non-application of monitoring standards and failure to use proper methodology (Akhalkatsi *et al*., 2012). Although many of the reserves are managed with local communities, on-farm management activities are limited due to a lack of awareness (Akhalkatsi *et al*., 2012).

Ex situ efforts in Armenia include accessions collections and other initiatives in collaboration with Bioversity International (Akhalkatsi *et al*., 2012), among the establishment of a national monitoring and information system. Madagascar has equally established programmes for the *ex situ* conservation of crop varieties, mainly through gene banks (Andriamahazo *et al*., 2009).

13.3.3 Forest genetic resources

Much of the efforts mentioned in 13.3.2 are applicable to FGR's. Aichi Biodiversity Target 15 offers huge potential for contributing to the conservation of FGR's through well planned and coordinated restoration, for example, for tropical forest biomes. FGR's have important implications for ecosystem function, numerous ecosystem services and directly impact on higher level biodiversity (Davies *et al*., 2014). The erosion of FGR's is thus expected to have far reaching consequences. Forest degradation and conversion of forest to alternative land use are expected to continue to over the come decades thus further threatening FGR's.

13.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

The main drivers impacting biodiversity in the second half of the century include climate change, human population growth (coupled with increasing food demand), and water scarcity (see chapter 22). Responses to the needs of a growing human population, the increasing food demand, and potential shifts towards a more Western-style diet include increases in agricultural areas, as well as further intensification of agriculture employing high-performing livestock breeds, and highly productive crop varieties.

A changing climate brings with it changes in species distributions, population changes and selection pressures that have further impacts on genetic diversity and gene flow. Changes in genetic diversity, in turn, may impact on populations, species and communities through, for example, increased inbreeding depression and constrained evolvability (Neaves *et al*., 2013).

To enable farmers to cope with ecological and socioeconomic changes, access to both new and traditional varieties of crops and breeds is necessary. New crops, varieties and livestock breeds are used to meet changed production conditions, while traditional agricultural varieties remain an essential part of adaptation strategies (Mijatovic *et al*., 2011).

13.4.1 Animal genetic resources

According to Hoffmann *et al*., (2013), a high genetic diversity in livestock populations provides society with the opportunities to meet the challenges set out above. Genetically diverse (livestock) populations allow farmers to select breeds with specific traits or to breed new varieties that are better adapted to a changing environment, like different climatic conditions, emerging or recurrent diseases, changing human dietary requirements, as well as novel socioeconomic conditions. To which extent, however, a farmer has a choice in selecting specific breeds will be dictated by market demand and availability of breeds.

In the longer-term future, climate change will pose considerable threats to animal husbandry, through direct and indirect effects (Pilling and Hoffmann, 2011). Depending on the regions, breeds will have to cope with heat stress, increased disease prevalence and parasite load (e.g., Singh *et al*., 2011; Wall *et al*., 2011), as well as changed access to water and food resources. Suitable climatic and environmental conditions for breeds and species are likely to shift, and breeds vary in adaptability to climate change. This leads to mismatches of livestock genetic diversity and the production environment, and the need for breed and species substitution that can cope with the new environmental conditions. However, locally adapted breeds may potentially provide the potential traits to selectively breed new varieties that are able to cope with new conditions (Hoffmann *et al*., 2013).

13.4.2 Plant genetic resources

Climate change will have major consequences for the geographical distribution of crops, individual varieties, and CWR (FAO, 2010a). For example, Jarvis *et al*., (2008b) estimate that about 16-22% of CWR relatives of three important crop genera (*Arachis* (peanuts), *Solanum* (potato, tomato, eggplant) and *Vigna* (beans)) will become extinct (Figure 13.3) by 2055. Most of the species considered are predicted to lose over 50% (some even nearly all) of their climatic range. In addition, for many species, suitable areas are predicted to become highly fragmented (Jarvis *et al*., 2008b).

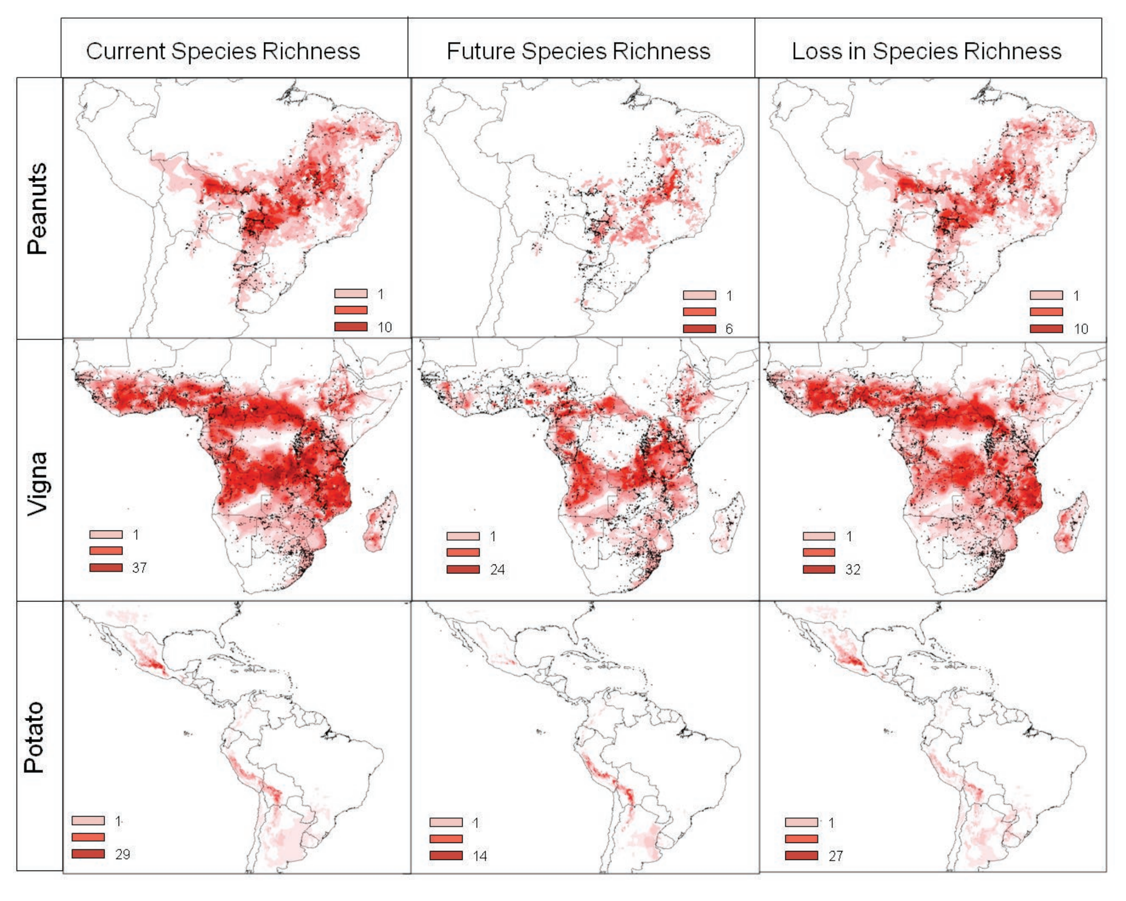


Figure 13.3. Modelled changes in species richness and area occupied for peanut (Arachis), bean (Vigna) and potato (solanum) wild relative species under two different migration (dispersal) scenarios. Red: losses, blue: gains; number indicates number of species lost or gained compared to current species richness. Source: Jarvis *et al*. (2008b).

Climate change will likely not only have an impact on the ranges of CWR, but will also lead to local extinctions and reduction in genetic variation (Jarvis *et al*., 2008b; Maxted and Kell, 2009). Changes in local and regional climate regimes will also alter environmental conditions under which crops grow (Maxted and Kell, 2009), making it necessary to develop new crop varieties that are better suited to these changed conditions. The ability to develop these new varieties, however, depends on the breadth of genetic variability available in CWR, currently used crop varieties as well as abandoned varieties and land races.

Climate change will also likely affect the distribution of crops, such as maize and its relatives in Mexico (Ureta *et al*., 2012), or various cereal crops in Ethiopia (Evangelista *et al*., 2013). Most species will have considerably reduced ranges; however, there are few species or races that might be able to extend their range (Ureta *et al*., 2012; Figure 13.4).

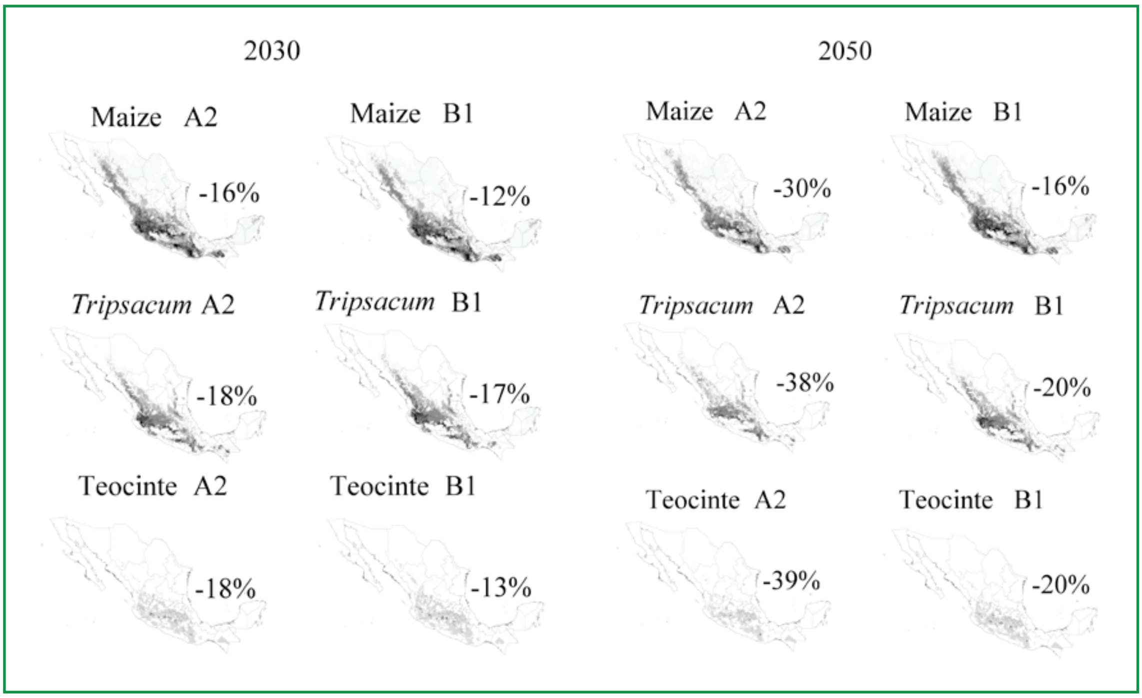


Figure 13.4. Potential distribution of maize and related species in Mexico in 2030 and 2050 under two emission scenarios, percentages indicate loss of suitable area. A2 and B1 refer to different IPCC SRES greenhouse gas emissions scenarios. A2 is a high emissions scenario and B1 is a low emissions scenario. Teocinte is considered to be the ancestor of modern maize (corn) varieties and *Tripsacum* is a close wild relative of maize. Source: Ureta *et al*. (2012).

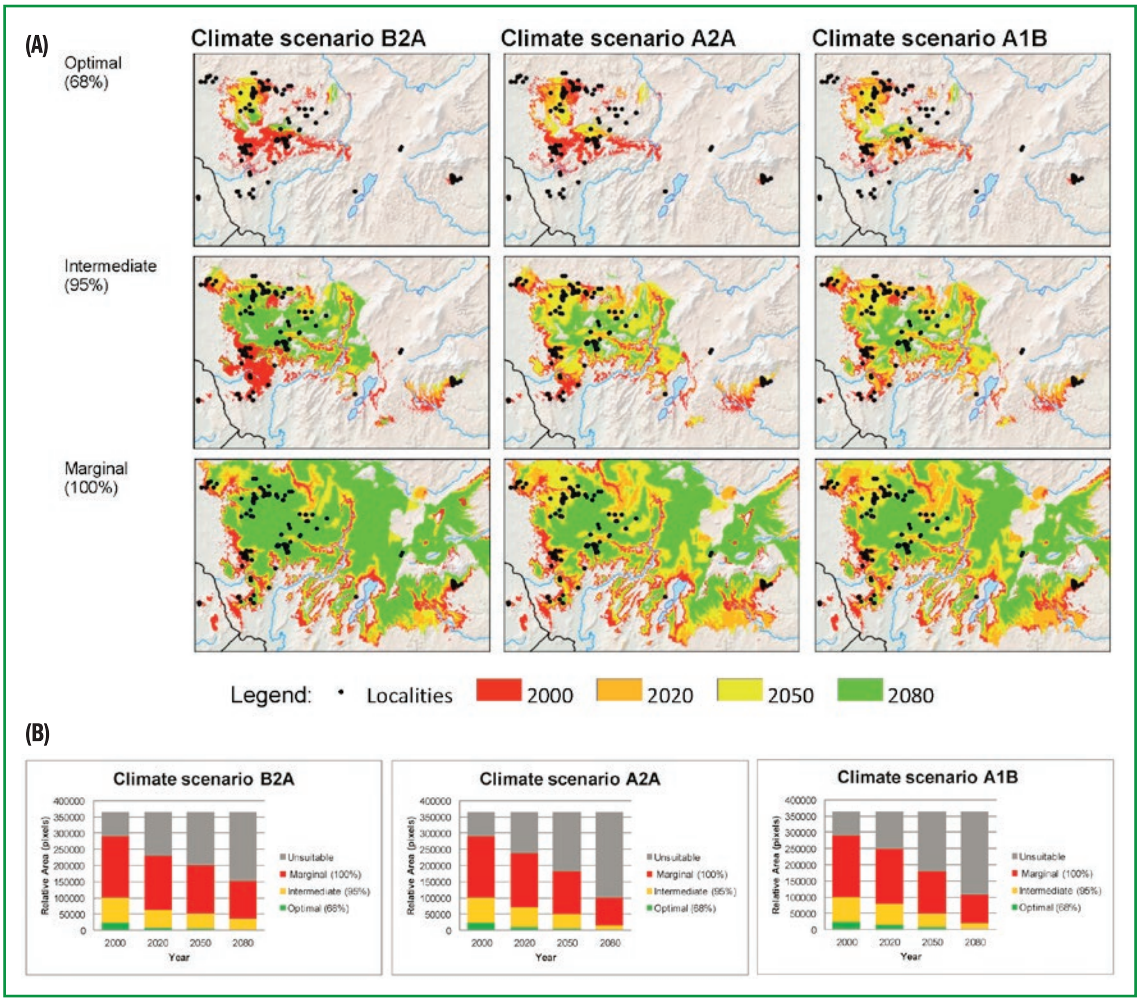


Figure 13.5: (A) Areas in Ethiopia predicted to have suitable bioclimate for Coffee Arabica in 2020, 2050 and 2080. Thresholds (of bioclimatic suitability) applied at 68% (optimal), 95% (intermediate) and 100% (marginal). Black dots show historical and present day localities. (B) Area analysis overview. Predicted climate change outcomes for indigenous Arabica for the year intervals 2020, 2050 and 2080. Stacked bar-charts based on area analysis maps (13.5A). Climate scenarios are IPCC SRES scenarios. Source: Davis *et al*. (2012).

Indigenous Arabica Coffee (*Coffea arabica*) has also been identified as being sensitive to climate change (Davis *et al*., 2012), and the productivity of the species is closely linked to climate variability, particular temperature. It was predicted that the suitable climate range in Ethiopia and Kenya will considerably retract by 2050 (even more by 2080), and many existing plantations will be negatively impacted (Figure 13.5).

Ex situ conservation of varieties may assist in buffering the effects of changing climate and environmental conditions, but the existing *in situ* genetic variation is maybe even more important for plant breeders to adapt to climate change, as new traits/new species may emerge/ be selected for from this genetic variation that can cope with the new environment (FAO, 2010a). However, some CWR might be driven to extinction by climate change.

Habitat change, mostly the expansion of agriculture will remain a threat, in particular for CWR (FAO, 2010a), leading to a further decrease in species and genetic variation. Invasive alien species are becoming more and more important as a threat for CWR, and the replacement of traditional with modern varieties (FAO, 2010a) will further reduce diversity of local crops. Conservation strategies for both local crop species, as well as CWR will need to be rethought to counteract these increasing threats.

13.4.3 Forest genetic resources

8,000 tree species are already under threat of extinction globally. Multiple factors are likely to continue to threaten trees and their genetic diversity over the coming 35 years. Climate change offers significant challenges as drought, fire and extreme weather events are predicted to become more frequent. Climate change is likely to influence plant species phenology and together with forest fragmentation present additional pressure on FGR's. In summary these factors suggest that increased rates of extinction in forest tree species, unless national strategies to enhance the resilience of Forest landscapes are strengthened considerably.

13.5 UNCERTAINTIES

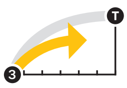
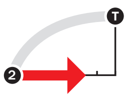
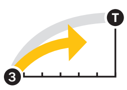
Numbers (and percentages) of breeds in a certain risk status are a snapshot in time only. For this report, we used the figures from the latest data base update, dated 1 June 2012. This does not necessarily reflect the actual number, and discrepancies may arise from a number of sources. Data managers may add or adjust data retro-actively, or re-allocate entries to other breeds. The Commission on Genetic Resources for Food and Agriculture requested FAO in its 14th Regular Session held in April 2013 to introduce, for the purpose of calculating breed risk status, a cut-off point of ten years, beyond which the risk status of a breed will be considered to be unknown if no updated population data are reported (FAO, 2013d). If population data have not been updated in a certain time period, the size of the actual population is unknown. Using this data to calculate risk status will not provide an accurate assessment, and thus the risk status of these populations is recorded as "unknown". The introduction of this cut-off point increases percentages of the risk classification "unknown", and reduces the percentages of the other categories accordingly.

Although there is general agreement that land use and land cover changes and climate change will have implications for genetic diversity, there is, to date, only limited evidence for the mechanisms how the environmental changes affect genetic diversity (Neaves *et al.*, 2013). In particular, evidence is lacking on how climate change, via changes in species distribution and population size, influences genetic diversity, and on how evolutionary adaptation might ameliorate climate (and other environmental) change impacts on biodiversity. Climate change is equally a major threat to species that are conserved *ex situ*, as these species do not experience any selection pressures, and will thus be maladapted to the "novel" environment when exposed to this.

Phylogenetic relationships among breeds also suggests that strategies to address Target 13 could prioritize conservation actions based on best-possible gains in conserved phylogenetic diversity among breeds (e.g., Simianer, 2003). These strategies can also integrate important information about within-breed diversity (Ginja *et al.*, 2013; Boettcher *et al.*, 2010).

13.6 DASHBOARD – PROGRESS TOWARDS THE TARGET

The FAO Global Plans of Action for Animal Genetic Resources and Plant Genetic Diversity for Food and Agriculture provide good frameworks for conservation of genetic resources. However, these Global Plans of Action need to be translated into national action plans, and this has been very limited to date.

Element	Current Status	Comments	Confidence
The genetic diversity of cultivated plants is maintained		*Ex situ* collections of plant genetic resources continue to improve, albeit with some gaps. There is limited support to ensure long term conservation of local varieties of crops in the face of changes in agricultural practices and market preferences	High
The genetic diversity of farmed and domesticated animals is maintained		There are increasing activities to conserve breeds in their production environment and in gene banks, including through *in vitro* conservation, but to date, these are insufficient	High
The genetic diversity of wild relatives is maintained		Gradual increase in the conservation of wild relatives of crop plants in *ex situ* facilities but their conservation in the wild remains largely insecure, with few protected area management plans addressing wild relatives	Medium
The genetic diversity of socioeconomically as well as culturally valuable species is maintained	Not evaluated	Insufficient data to evaluate this element of the target	
Strategies have been developed and implemented for minimizing genetic erosion and safeguarding genetic diversity		The FAO Global Plans of Action for plant and animal genetic resources provide frameworks for the development of national and international strategies and action plans	High

Author: Cornelia Krug, with contributions from Chris Kettle, Daniel Faith, and FAO

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TARGET 14: ECOSYSTEMS THAT PROVIDE ESSENTIAL SERVICES

By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable.

PREFACE

Target 14 focuses on the restoration and safeguarding of ecosystems that provide essential services. As noted in CBD Decision XI/3, appropriate metrics to track progress towards this target would include those relating to status and trends in ecosystems (or their components) which provide such 'essential services' as well as trends in the services themselves, particularly those relating to water. However, there is also an important equity or distributional element to the target. In order not to simply replicate information provided under other targets relating to safeguarding and restoring ecosystems, a more nuanced analysis would identify ecosystems that provide essential services on the basis of the presence/extent of explicit 'downstream' human beneficiaries, and would say something about how the services delivered by those systems, and the well-being of beneficiaries, including the poor and vulnerable, were changing. Yet it is important to acknowledge the weakness of our current ability to monitor trends in ecosystem services, and how critical it is to rectify this imbalance in order to make a thorough assessment of progress toward the implementation of the mission of the Strategic Plan.

According to the Millennium Ecosystem Assessment (MA), ecosystem services are all the benefits that people obtain from ecosystems (MA, 2005; other definitions are available). Ecosystems result from the interactions between living beings (biodiversity) and their abiotic conditions (water, energy, nutrients). It is important to point out at the very start, however, that this chapter is not a comprehensive update of the MA, and it does not come close to dealing with trends in the majority of ecosystems, their services and beneficiaries. The starting point for the chapter, indeed for every chapter, is to focus on selected global metrics corresponding to the indicators identified by Parties to the Convention on Biological Diversity (CBD), along with a relatively broad assessment of global-scale (and large sub-global) scenario studies of relevance to the target. Such an approach will necessarily be incomplete given data gaps, the large scale focus and the very wide breadth of the topic to which Target 14 relates. The examples selected should thus be taken as a selection of the many possible examples that exist. The chapter should be approached with that in mind.[1]

Background to ecosystem services

Four types of ecosystem services can be distinguished (MA, 2005). Provisioning services are the tangible resources that people obtain from ecosystems; these are finite, can be renewable, and can be directly consumed, appropriated, and traded (Maass *et al*., 2005). Crops, livestock, fish, wood, biofuels, harvested wild products and water are provisioning services. Regulating services result from the contribution of multiple ecosystem processes to ecosystem functioning, specifically to the regulation of the conditions where humans live and make a living (Maass *et al*., 2005); such regulation determines both the average and the variance in such conditions. The regulation of climate, water quality, soil fertility, soil erosion, coastal and flood protection, pollination, and pest control are regulating services. Cultural services are ecosystem's contribution to the non-material benefits that arise from the interaction between people and ecosystems; these benefits include a range of capabilities and experiences (Chan *et al*., 2012). Cultural ecosystem services include nature tourism, identity, inspiration and heritage. The contribution to human well-being that supporting services provide are often more indirect and complex but they are nonetheless essential to the proper functioning of all other ecosystem services (Haines-Young & Potschin, 2010). Soil, formation, nutrient cycling and primary production are examples of supporting ecosystem services.

Footnote

[1] One observation that may be made is that, partly as a result of globally available data, there is a disproportionate focus on provisioning services within the chapter. This is not uncommon but is potentially problematic. Reyers et al. (2012) note that "the focus of ecosystem-services analyses is often on provisioning services that lend themselves to economic valuation, such as timber and food. Use of these narrow definitions and measurements can make it appear that there is less connection and greater divergence between strategies to conserve biodiversity and strategies to promote ecosystem services than may in fact be the case."

Three different components of ecosystem services need to be clearly distinguished (Tallis *et al*., 2012). ***Supply*** refers to the potential contribution of a socioecological system to a service, which depends on the condition (e.g., amount of water) and functioning (e.g., primary productivity) of ecosystems, as well as the way ecosystems are managed. ***Delivery*** accounts for how the service is consumed (e.g., amount of timber harvested), delivered to societies (e.g., spatial location of those benefiting from flood regulation), and how societies have access to the services (e.g., laws that limit access to a service). ***Benefit*** accounts for the change in people's well-being that results from the service, such as changes in living standards, nutrition status, mortality rates, social conflicts, security in the face of extreme environmental conditions, or happiness, and can include monetary value.

Biodiversity, which is defined by the CBD as "the variety of life on Earth and the natural patterns it forms" from genes to species to ecosystems plays a paramount role in ecosystem service supply (Mace *et al*., 2012). *Biodiversity is a regulator of ecosystem processes*. Quantitative meta-analyses from experimental studies show the generality of the effects of species richness on ecosystem functioning (Balvanera *et al*., 2006; Cardinale *et al*., 2011)[2]. Evidence is available on the direct effects (experimental evidence) or correlations (observational evidence) between some provisioning and regulating services and biodiversity. Yet, limited data (i.e., the amount, consistency, or generality of the data) and a mismatch between the variables measured and the final ecosystem service that is relevant to stakeholders has precluded establishing a direct link between biodiversity and the supply of most ecosystem services. *Components of biodiversity itself represent provisioning ecosystem services* including fish, timber, biofuels and a wide array of non-timber forest products extracted from many terrestrial and aquatic ecosystems. *Biodiversity is valued by people*, and it contributes to human physical, mental and spiritual health as well as social relationships through non-tangible benefits that arise from the contact of biodiversity with people (Russell *et al*., 2013).

The list of essential services to support health and livelihoods of the poor and vulnerable is quite long. Yet, little information is available on many of them. Data deficiencies notwithstanding, this chapter attempts to capture how some ecosystems and their components that deliver services to people, including the poor and vulnerable, are changing. It focuses on a range of important service types for which broad scale temporal data were available (in line with the indicators suggested in Decision XI/3), provided by selected terrestrial/freshwater and marine ecosystems and/or their component species. Table 14.1 illustrates the breadth of those services and highlights those that constitute the primary focus of the status and trends section of the chapter. Each sub-section attempts to assess the status of change to the ecosystem (or component species) through reference to the services they provide to people, including services upon which the poor and vulnerable are most reliant. Selected available data at global, regional or even local scales is compiled here to provide an illustration of our current understanding.

Table 14.1. Examples of selected essential ecosystem services. Highlighted services in bold are those that constitute the primary focus of this chapter due to data availability. Modified from Chapter 4. Also see UNEP-WCMC, 2011.

Service/ Benefit	Metric	Ecosystems that provide them	Links to needs of women, indigenous and local communities, and the poor and vulnerable
Crops	Crop production (Tons)	Agroecosystems	Food amount (subsistence), food security, resilience of agricultural livelihoods
Livestock	Livestock and livestock products production (Tons)	Agroecosystems, rangelands	Food amount (subsistence), protein intake, food security, income security, resilience of pastoralist livelihoods
Aquaculture	Aquaculture production (Tons)	Modified waterbodies	Food amount (subsistence), protein intake, food security, income security, resilience of coastal and inland rural livelihoods
Fisheries	**Fish catch (Tons)**	**Coastal and inland water bodies**	**Food amount (subsistence), protein intake, food security, income security, resilience of coastal and inland rural livelihoods**

Footnote

[2] *However, many experiments show that often it is the number of common (experimentally available), dominant species, rather than large numbers of rare, uncommon species, that is related to ecosystem processes and in some cases to actual measured services.*

Service/ Benefit	Metric	Ecosystems that provide them	Links to needs of women, indigenous and local communities, and the poor and vulnerable
Wood	Harvested wood volume (m^3)	Forests	Income security, construction materials
Biofuels	Harvested biofuel, fuelwood or charcoal volume weight (Tons)	Forests, scrubs, agroecosystems, rangelands, mangroves	Household energy use, energy security, resilience of rural livelihoods
Harvested wild products	**Harvested wild products weight (Tons) – Including wild meat from vertebrates**	**All terrestrial ecosystem**	**Food amount, protein intake, food security, medicinal plants, construction materials, income security, resilience of rural livelihoods**
Water	**Volume of superficial or ground-water (total, domestic and agricultural uses) withdrawn (m^3)**	**All terrestrial ecosystem organised into watershed**	**Water security, food security**
Climate regulation (Carbon stocks and uptake)	**Emissions avoided and carbon uptake by vegetation from the atmosphere (Tons of C)**	**Tropical and temperate forests, grasslands, seagrass beds**	**Mitigation of impacts from climate change**
Regulation of marine and fresh water quality	Water conditions (mass of nutrients, organic matter, sediments, toxic organisms, toxic compounds, temperature, pH) in relation to standards	Riparian, inland and marine aquatic ecosystems	Regulation of water borne diseases, toxic compounds related illnesses and food poisoning
Regulation of soil fertility	Marginal contribution of soils (nutrients available) to agricultural, forestry and biofuel production	Soils	Food security, income security
Regulation of soil erosion	Mass of soils (Kg) retained	Agroecosystems, managed forests and rangelands	Food security
Coastal/ flood protection	**Area of avoided flood, loss of crops, cattle and infrastructure, erosion (ha)**	**Forests, riparian vegetation and soils, Mangroves, seagrass beds, coral reefs**	**Mitigation of impacts from extreme hydrometeorological events sea level rise and hurricanes**
Pollination	Marginal contribution of pollinators to crop production	Agroecosystems and surrounding terrestrial ecosystems	Food amount, food security, income security
Pest control	Regulation of pests contributed by their natural enemies	Agroecosystems and surrounding terrestrial ecosystems	Food amount, food security, income security
Health	Absence of disease, and well-being	Access to medicinal resources, societal productivity	Food amount, protein intake and food security, regulation of environmental toxins
Cultural services	Non-material benefits from ecosystems including nature-based tourism, identity, inspiration, heritage	Marginal contributions to income or well-being of visitors and to local inhabitants derived from aesthetic views	Physical, mental and spiritual health, social relations, happiness

Notwithstanding the challenges of measuring progress against Target 14, available data suggests that globally we are not yet progressing in the right direction. Achieving the target relies on actions to meet other Aichi Biodiversity Targets (including 3, 5, 7, 10, 15 and 18), and on the future development pathways taken, with some trade-offs inevitable.

14.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

Global assessments suggest that many ecosystem services are in decline as a result of ecosystem degradation – in the Millennium Ecosystem Assessment, 15 out of 22 categories of ecosystem services were judged to be in decline at a global level (MA, 2005). Sub-global assessments concur – for example the UK National Ecosystem Assessment concluded that around 30% of ecosystem services in the UK are declining, largely as a result of declines in the extent or condition of habitats delivering those services (UK NEA, 2011). The conclusions of these and other global, regional and national assessments suggest that, based on current trends, Target 14 is not likely to be met. However, headline messages like these mask significant variation, and do not disaggregate implications for different sectors of society such as the poor and vulnerable. This section of the chapter attempts to unravel some of the detail around some specific ecosystems and species and their associated services and benefit flows.

14.1.1 Status and trends

14.1.1.i Trends in species that provide food and medicine

While it can be difficult to determine trends for 'ecosystems that provide essential services' other than for broad ecosystem types as covered under Targets 5 and 10, it is possible to assess trends in populations of species that provide services, particularly those that provide provisioning services as a result of their direct use. Information about impacts on specific species can sometimes be used as a proxy for understanding change in the ecosystems of which they are a part. Findings are mixed – while many utilized species are declining, not all are, and although some are faring worse than non-utilized species, others are faring better.

The variety of species harvested for food and medicine contributes to health, livelihoods and well-being. While this broadly applies to the health of all people, it is particularly important for the poor and vulnerable who are often most reliant on natural resources yet frequently have access to fewer alternatives as primary sources of food, nutrition and health care (CBD, 2010). Species used for food and medicinal purposes include amphibians, birds, mammals, marine and plant species. In general, amphibian, bird and mammal species that are utilized for these purposes show declining IUCN Red List Index (RLI) trends, indicating that they are being driven ever-faster toward extinction (Figure 14.1). Both bird and mammal species used for food and medicines are on average more threatened with extinction than those that are not. Twenty-three per cent of all bird species used for food and medicinal purposes are classified as threatened compared to 12% for non-utilized species. In contrast to birds and mammals, amphibians used for food and medicine appear to be less threatened overall than amphibians not used for these purposes (2010 BIP, 2010). However, amphibians used for food and medicines are declining at a faster rate than the other vertebrate species assessed.

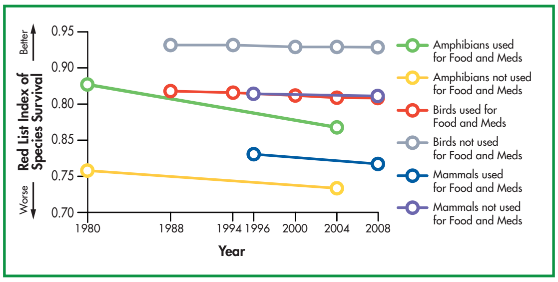


Figure 14.1. Red List Indices for birds, mammals and amphibians that are used by people for food and medicine compared with those that are not; declines indicates that species are being driven ever-faster towards extinction (Reproduced from 2010 Biodiversity Indicators Partnership, 2010). Source: TRAFFIC.

An estimated 3% of all plant species used for medicines have been assessed for global conservation status. Global trends between 1998 and 2008, indicate that, although assessed plant species did not show a notable decline over that period, an estimated 40-45% remained consistently threatened. The overall conservation status of medicinal plants may be particularly worrisome if this pattern is maintained in broader and more representative surveys of medicinal plants (2010 BIP, 2010). At the time of publication, regional efforts were ongoing to assess the conservation status of medicinal plants, including Europe, which is assessing this new group in 2014 for inclusion in the European Red List.

In terms of population abundance of utilized species, early analyses using globally available data and, more specifically for the Arctic, suggest that utilized species are faring better than other species overall (Tierney *et al.*, 2014). Although the global index revealed a decline of 14% between 1970 and 2007, this was significantly lower than the decline in populations overall as indicated by the Living Planet Index (Figure 14.2). Freshwater utilized species fared better than marine species, which in turn fared better than terrestrial species. More positively, utilized species in the Arctic were found to increase in abundance markedly over the same time period, again significantly higher than overall Arctic species trends (Figure 14.3a). This could be a consequence of better management of these populations, as indicated by more sustainable harvest levels in recent decades (Figure 14.3b).

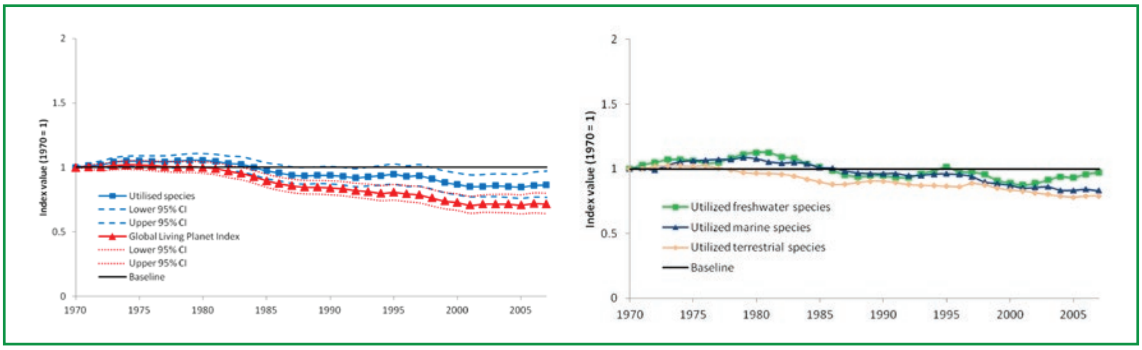


Figure 14.2. Trends (±95% confidence intervals) in a) Utilized Species compared to the Global Living Planet Index (WWF, 2012); b) Utilized Freshwater, Marine and Terrestrial Species. Source: Tierney *et al*. (2014).

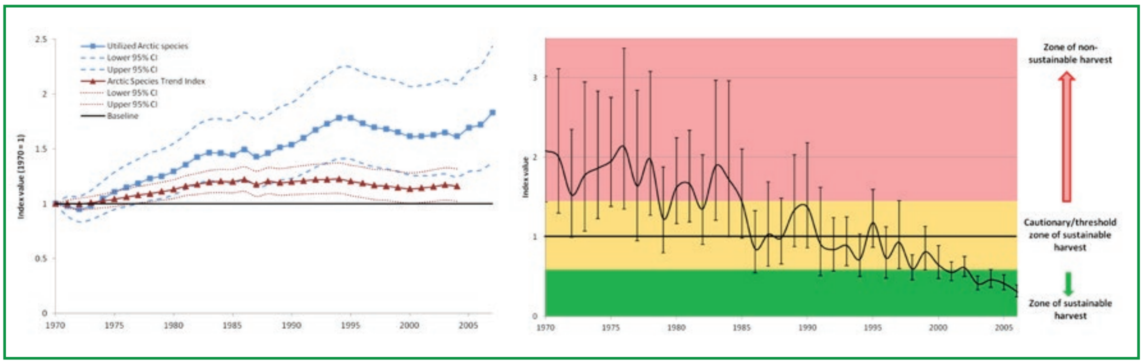


Figure 14.3. a) Trends (±95% confidence intervals) in Arctic Utilized Species compared to the Arctic Species Trends Index (McRae *et al*., 2010) **between 1970 and 2007; and b) Harvest Index of Arctic species between 1970 and 2006. Zones of unsustainable (light grey), cautionary (medium grey) and sustainable (dark grey) harvest levels shown.** Source: Tierney *et al*. (2014).

Plants harvested from the wild are used in traditional medicine in local health care, for food, and as source of income, but many (estimated 4,000-6,000, Iqbal 1993, Secretariat of the Convention on Biological Diversity, 2001) are important in international trade based on broader commercial use and value. It is estimated that the global trade in plants for medicinal purposes alone reaches the value of over US$2.5 billion (UN COMTRADE, 2013) and is increasing, driven by industry demand.

14.1.1.ii Trends in species that provide ecosystem services (pollination services)

Most services are difficult to link to individual species, but animal pollination is an exception, with multiple studies showing that exclusion of particular groups of pollinator species leads to reduction in crop productivity and value (Kremen *et al*., 2007; Potts *et al*., 2001)[3].

Most of biodiversity is comprised of insects, microbes and other invertebrate taxa, but relatively little is known about their distributions and abundance, including their dynamics and the threats they face. The information gap on the status and trends for species that play particularly critical functional roles, such as pollinators, is especially worrying. Pollination is essential to both food and nutrition security, and it plays a critical role in the maintenance of wild plant communities as well as agricultural productivity. Pollination services are reliant upon both domesticated and wild pollinator populations, and both may be affected by drivers of biodiversity loss and change, with unknown but potentially critical consequences for the health and well-being of all people, including the poor and vulnerable. Pollination does not only affect the overall quantity of foods such as fruits, seeds and nuts, but also the nutritional content, quality, and variety of foods available. Recent studies in this area have estimated that crop plants dependent on pollinator species contain most of the global availability of vitamins A, C and E and dietary lipids as well as an important proportion of minerals, calcium, fluoride, and iron (Eilers *et al*., 2011). These findings support the conclusion that the yield increase that can be attributed to animal pollinated crops are essential to nutritional diversity and human health and their resulting decline can have significant consequences for both food and nutrition security as well as human health.

Footnote

[3] *Noting, however, that many of the 'bulk' crops providing calorific value, such as cereals, are wind pollinated, whilst animal pollinated food plants are often those providing other nutritional value and flavour.*

Agricultural production is very reliant on ecosystem services such as pest control, soil fertility, and pollination, among others, and unsustainable agricultural intensification can disrupt beneficial functions of biodiversity (Power, 2010). It is estimated that the services of pollinator services affect over one third of global food supply (although staple crops such as cereals, corn and rice are predominantly self-pollinating) (Tscharntke *et al.*, 2012). Pollinators are also required for reproduction of almost 90% of angiosperms (flowering plants) and they improve the production of 70% of the most globally valuable crop species (based on data from 200 countries, 124 crop species were identified) (Tscharntke *et al.*, 2012; Klein *et al.*, 2007; Gallai *et al.*, 2009; Garibaldi *et al.*, 2011). Long-term global trends in crop yield and production imply increasing dependence on pollinators, particularly in the developing world (Figure 14.4). However, the RLI for pollinator species among birds (e.g., sunbirds and New World warblers) and mammals (e.g., some bats and rodents) shows declining trends, indicating that these species are moving faster towards extinction (Figure 14.5). Also, declining trends for many insect pollinator species have been observed (Biesmeijer *et al.*, 2006; Cameron *et al.*, 1998) and have raised concerns about the impacts of pollinator and managed honey bee declines on limiting crop yields (Garibaldi *et al.*, 2011). The pollinator species that is most commonly managed for agricultural production is the well-studied honey bee (*Apis mellifera*), although other species are also used in some contexts. This species been shown to have the ability to increase agricultural yield in up to 96% of animal-pollinated crops (Ibid). While global trends on managed pollinators are not yet available, there is clear evidence of recent declines in both wild and domesticated pollinators, with concomitant declines in the plants that rely upon them in various parts of the world (Potts *et al.*, 2010). In some regions, such as the United States and Europe, severe decline in domestic honey bee stocks were reported between 1947 and 2005, with up to 59% reduction of colonies in the United States and 25% reduction of colonies in central Europe over that period, potentially jeopardising agricultural crops, and wild plants (Potts *et al.*, 2010).

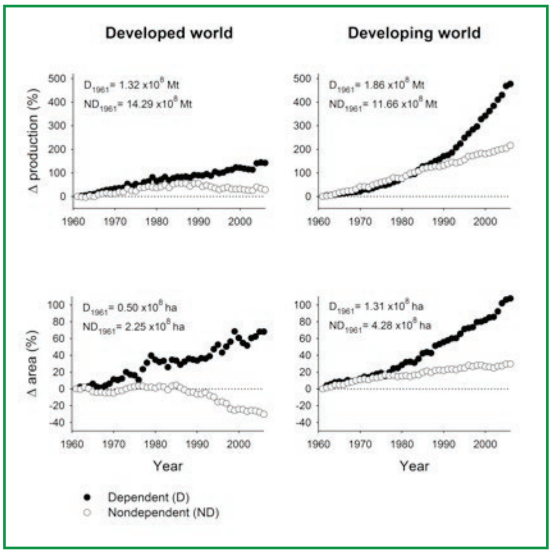


Figure 14.4. Change over time in the proportion of crop area and production that is pollinator dependent. Source: Aizen *et al.* (2008).

Overall, the pollinator species shown in Figure 14.5 are less threatened than non-pollinator species (for which the RLI has lower values), perhaps reflecting the fact that average body size is larger among non-pollinators, and that large-bodied species tend to be more threatened. However, rates of declines appear to be similar for both groups. Of all pollinators, birds and mammals form only a minority, but global data for the many pollinator species among insect groups are currently not available although a number of regional studies on select pollinator species, including those managed for agricultural production is increasing. As noted earlier, it is highly likely that many insect pollinator species are also in decline. Target 14 calls for "ecosystems that provide essential services" to be "restored and safeguarded". The decline in the RLI for birds and mammals pollinators indicates that ecosystems supporting them are not currently being adequately safeguarded.

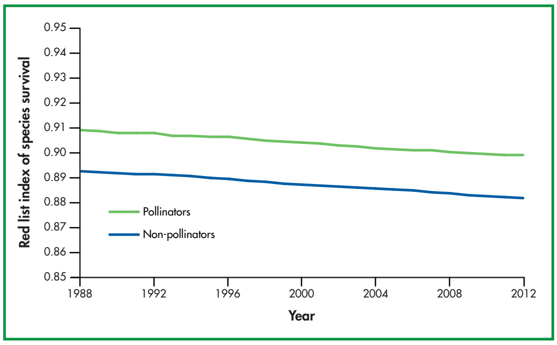


Figure 14.5. Red List Index for pollinators (880 birds and 308 mammal species) compared with non-pollinators (8,988 birds and 4,192 mammal species). Source: BirdLife International and IUCN.

14.1.1.iii Trends in selected forest ecosystem services (timber and wild meat)

Forests provide a range of services from the supply of goods such as timber, wild meat and medicines, through to the control of freshwater provision as well as disease regulation. Forests can be particularly important for the poor and vulnerable as a source of non-timber forest products (NTFPs) such as food, fibre and medicine (Golden *et al*., 2012; Schaafsma *et al*., 2014). In addition to provisioning services, forests provide a wide range on non-tangible goods and services such as cultural and spiritual enrichment, traditional food cultures, landscape, heritage, cultural identities and place attachments (which are particularly threatened among vulnerable populations (Daniel *et al*., 2012). The full range of benefits derived from forests also provide a critical safety net to the poor (Roe *et al*., 2013). The World Bank estimates that forest products provide roughly 20% of poor rural families' 'income' of which half is direct income and half is in the form of subsistence goods (Vedeld *et al*., 2004).

Whereas forest trends are described in detail in the chapter on Target 5, this chapter considers trends in the supply and delivery of some key forest provisioning services globally, including some potential implications of forest change for the poor and vulnerable.

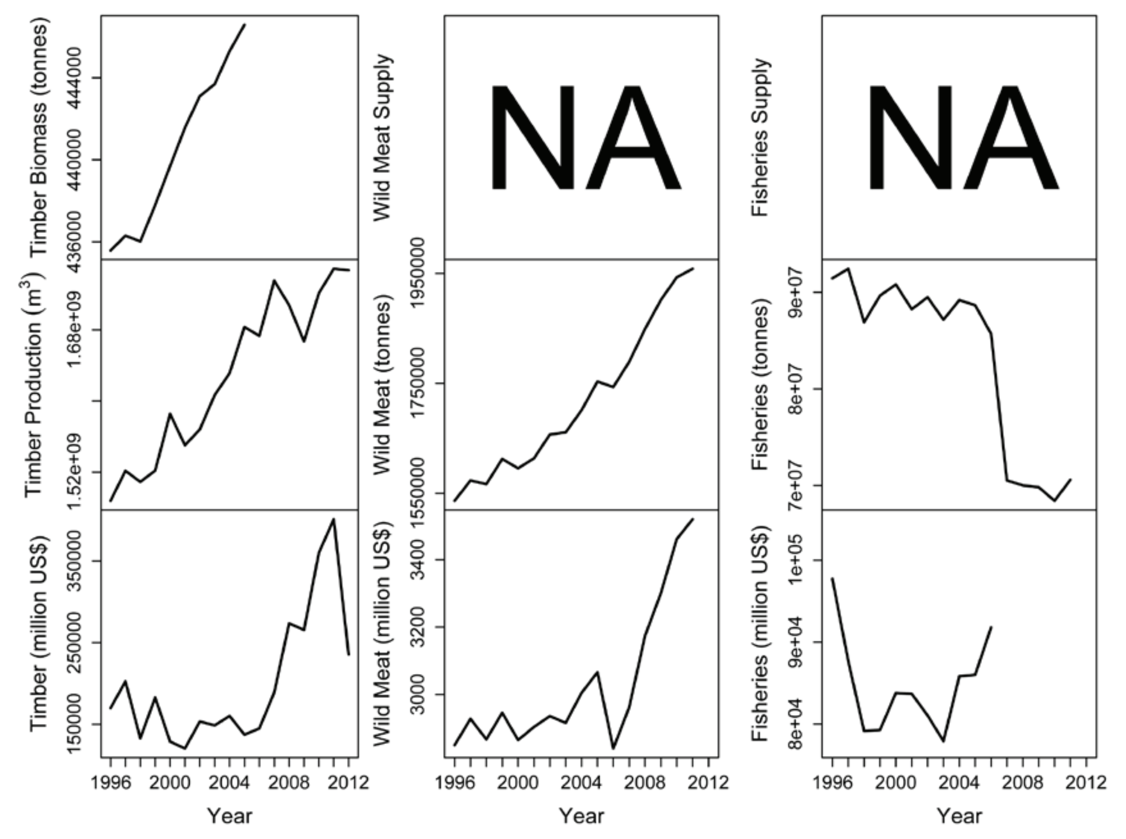


Figure 14.6. Timber, wild meat and fisheries trends over the last 20 years. Source: D. Karp, B. Halpern and K. Thonicke, (unpublished).

Timber

According to one model, global trends of timber biomass, i.e., the potential supply of timber, have generally increased over the last two decades (Figure 14.6).[4] The trend reported here indicates a recovery of forests and of potential timber supply through time. Annual timber production, i.e., the harvest from ecosystems, also shows an increasing trend over the last two decades[5]. There is considerable uncertainty associated with reported values because data are not consistently reported for each country and FAO often reports modelled and estimated data. Despite these uncertainties, a clear global trend toward increased timber harvest during this period was found.

Global timber value has also generally increased over time, although with a sharp decrease in the last five years[6]. The decline in total timber value is most likely given by changes in wood market value in the last five years.

In synthesis, the potential for timber supply has been increasing, probably more sharply than wood production. A global and regional balance between timber supply and harvest is still needed – the international timber trade is fuelling forest change in poorer countries as a result of demand in countries with emerging economies (e.g., China) that maintain their forests by importing wood from elsewhere (Kastner *et al*., 2011). This is significant in that it implies that, although timber as a product of forest provisioning services has increased globally, forest availability and resulting services to local forest-dependent communities in poorer countries with higher population densities may be declining. This signals the clear trade-off between timber production and other forest ecosystem services including regulating services and provision of NTFPs for other forest users. Other studies suggest that NTFP extraction itself can threaten the forest resource base, although any restrictions on forest use would have significant economic implications, particularly for poor households (Schaafsma *et al*., 2014). However, there are a number of best practice tools available to ensure the sustainable harvesting of wild resources alongside conditions for equitable trade in resources benefitting the harvesters. FairWild Standard is best practice guidance in wild harvesting and equitable trade in plant resources developed by TRAFFIC, WWF, IUCN and other partners, recognised by the CBD's Global Strategy for Plant Conservation. The FairWild Standard is being used by communities for resource management of medicinal plants (e.g., in Viet Nam and India), by industry for certification (e.g., in Kenya, Kazakhstan, Bulgaria, Poland) or as the best practice in pilot project with Traditional Chinese Medicine (TCM) industry in China, and governments to develop the regulations (e.g., in South Africa and Lesotho).

Wild meat

Potential supply of wild meat depends on population dynamics of all the species that are consumed by people. Data on this supply are not available globally, only locally for a few selected studies (e.g., Naidoo and Ricketts, 2006).

Wild meat production, that is the amount of wild meat that is harvested and marketed, shows an increasing trend over the last two decades (Figure 14.6)[7]. Wild meat value also shows a sharp increase in the past few years[8]. Accessibility is also changing. A seven-country pilot study suggested that wild meat, as well as animals and plants used for medicine, were becoming more affordable for the poorest in society in a number of cases, although less so in others (TRAFFIC, published in Chenery *et al*., 2013).

Footnotes

[4] Timber supply data were derived from the biophysical dynamic global vegetation and water balance model LPJmL (Lund-Potsdam-Jena Managed Land) (Bondeau et al., 2007). LPJmL relies on climatic and land-use change data to simulate main vegetation and hydrologic processes, i.e., photosynthesis and respiration, plant growth, soil moisture, runoff, evapotranspiration and vegetation structure. LPJmL estimates total aboveground carbon in woody biomass. This aboveground woody biomass is used to approximate timber availability based on a fix ratio between twigs and stem, without specification of tree species, and excluding areas of IUCN protection classes 1a, 1b and 2. We summed the global bole biomass across all grid cells to obtain annual estimates of global timber supply. Because LPJmL relies on global climate data that are reported with a considerable time lag, we only report total supply of timber biomass from 1996 to 2005. We acknowledge that inter-model comparison would be more robust that relying solely on one model such as LPJmL, and the findings should be interpreted accordingly.

[5] Total annual production of "roundwood" data were compiled from data reported by the Food and Agriculture Organization (FAO) of the United Nations for 1996 to 2012. Roundwood is defined as all wood acquired within a country, including wood obtained from natural and logging losses. It includes fuelwood, wood used in industry, and other uses. Roundwood is reported in m^3, excluding bark. We summed the total roundwood production across nations and report annual global production estimates from 1996 to 2005. The data are composed of national statistics, annually reported by countries to the United Nations.

[6] The FAO does not report total annual value of in-country roundwood production, but does report both the production and value (US$) of exported rounded. We calculated the value of in-country roundwood production (US$) though estimating in-country production prices as export production divided by export value.

[7] We obtained estimates of wild meat production (tonnes) from FAO data. The FAO defines wild meat as meat and offal of wild animals in all forms. Data are available and reported until 2011. We provide estimates of global production by summing statistics across all reporting countries.

[8] Value was obtained from FAO data (constant US$ 2004-2006) for individual countries and then summed.

Wild food sources are widely acknowledged as an important source of nutrition and are central to traditional food systems and cultures (Kuhnlein *et al*., 2009; Barucha and Pretty 2010; Fanzo *et al*., 2013). Based on 36 studies in 22 countries in Asia and Africa, the mean use of wild foods by agricultural and forager communities ranges between 90 and 100 species per location (Barucha and Pretty, 2010). Bushmeat and fish alone provide an estimated 20% of protein in at least 60 developing countries (Bennet and Robinson, 2000). Wild foods are also critical to addressing micronutrient deficiencies, particularly among vulnerable groups such as the poor and children (Powell *et al*., 2013). A recent study in Madagascar also showed that removing access to wildlife for food would likely increase levels of iron deficiency anaemia in children by up to three times in the poorest households, with consequent future disease implications (Golden *et al*., 2011). Despite their continued importance to food security, nutrition security, and human health, wild foods are frequently excluded from official statistics on economic values of natural resources. It has been estimated that their importance may be set to increase as pressures on agricultural productivity continue to rise (Barucha and Pretty, 2010).

In contrast to the increased dependence upon wild meat illustrated above, other studies report a decline in the availability of wild meat. For example, Brashares *et al*. (2001) highlights "sharp declines in biomass of 41 wildlife species. These findings indicate that while demand for wild meat is increasing, unsustainable harvest is causing substantial population declines in harvested species ("the bushmeat crisis"), in turn contributing to lower game meat availability (Nasi *et al*., 2008).

It is worth noting that in addition to wild meat being an important nutrition source for some populations, it is also a part of the international wildlife trade, destined for locations where nutritional alternatives are available. Analysing the patterns of wild meat consumption from tropical forests is critical to designing approaches to address threats to biodiversity and equally essential for the development of mitigation measures for the transmission of emerging infectious diseases, almost three quarters of which are transmitted via zoonotic pathogens (Wolfe *et al*., 2005); Myers *et al*., 2013). For example, HIV emergence was linked to nonhuman primate consumption whilst contact with chimpanzees intended for food spurred recent Ebola outbreaks (Karesh *et al*., 2005; Smith *et al*., 2012).

14.1.1.iv Trends in freshwater provision

Target 14 also refers explicitly to ecosystems providing services related to water which is closely linked to human health. Forests and wetlands are critical in regulating freshwater provision, the former particularly in relation to upstream watersheds and upland 'water towers' (Bullock and Acreman, 2003; Viviroli and Weingartner, 2004; Zedker and Kercher, 2005; Horwitz *et al*., 2011). Trends in forests and wetlands are presented in the chapter on Target 5, which indicates continuing declines in both. Here we consider how water provision, its use and dependency on natural freshwater water sources are changing.

Water yield

Global trends in water yield (a measure of surface runoff) show diverging patterns across the planet as a result of changes in climate and land use and land cover. At present wet tropical areas show the highest surface water yields while temperate areas in both hemispheres show the lowest ones (Figure 14.7A). These yields were calculated using LPJmL, a mechanistic model described below, from climatic, land use, and plant functional trait data.

Comparing this status against the 55-year trend, however, reveals that many African countries show a declining trend in water yield, using Kendall's tau statistics for trend analysis (Figure 14. 7B), whereas all countries in the Americas show an increasing trend. The dynamics behind this trend might not always be linear as is the case for Scandinavian countries, which have a positive long-term trend, but show cyclical changes as can be seen for the Sahelian countries Chad, Niger and Mali (Figure 14.7C).

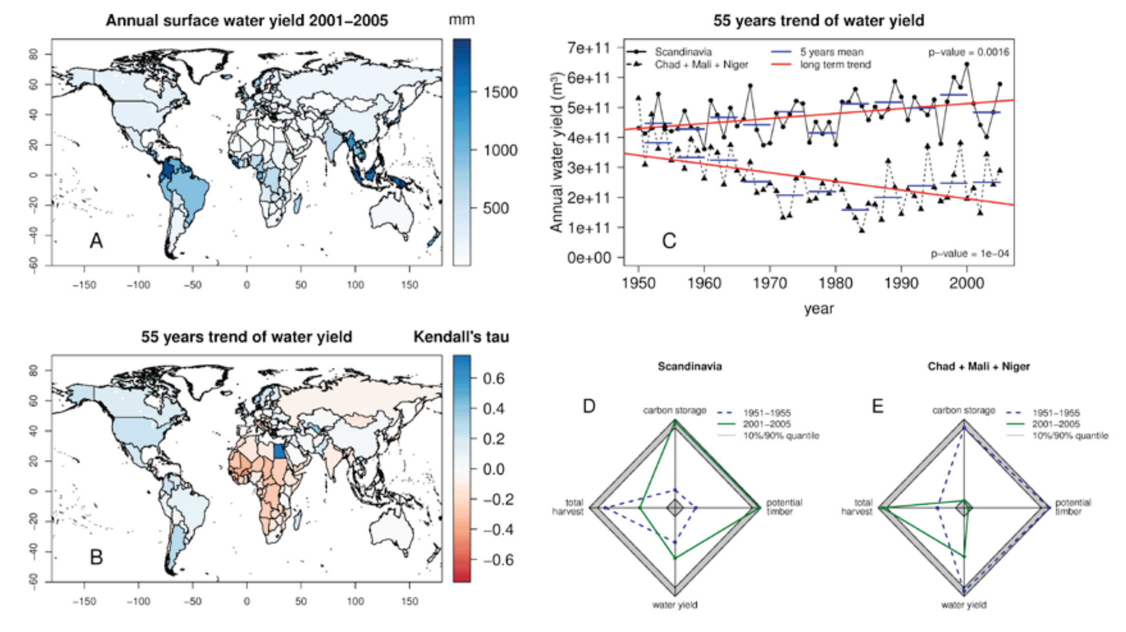


Figure 14.7. Water yield trends globally and for selected regions over time, derived from the LPJmL model. Source: K. Thonicke and A. Waltz, (unpublished).

Water use

The following is taken from Millennium Development Goal (MDG) indicators 7.5 and 7.8 which are listed as relevant to Target 14 in the CBD indicative list of indicators (Decision XI/3).

The proportion of the global population using improved drinking water sources (such as wells and piped water) increased from 76% to 89% in the two decades from 1990, suggesting that a small and declining proportion of the population still relies on unimproved, 'natural' freshwater sources from freshwater ecosystems and terrestrial surface runoff (Figure 14.8; UN, 2013b). However, 780 million people, the majority from rural areas, still relied on unimproved water sources in 2012 (WHO, UNICEF, 2012). Of these, over 180 million people rely on rivers, streams, ponds or lakes for freshwater. Some 40% of those without access to freshwater live in sub-Saharan Africa where coverage of improved water supply sources has only increased by 61% compared with 90% in Latin America and the Caribbean, Northern Africa and large parts of Asia. Moreover, the number of people without improved water sources has been found to be five times greater in rural versus urban areas. (WHO/UNICEF, 2012). This underscores the importance to the rural poor of freshwater supply and quality from natural ecosystems.

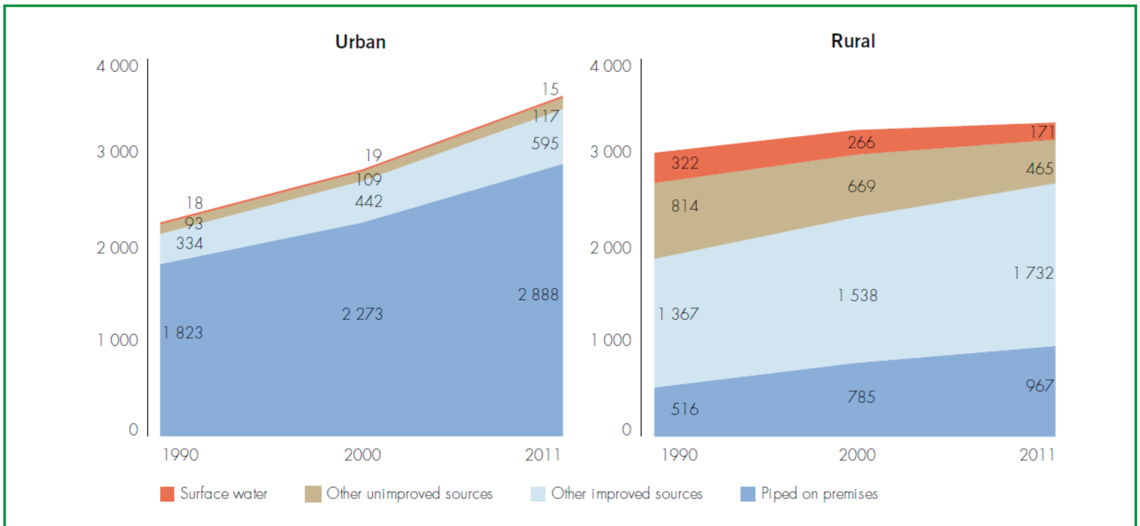


Figure 14.8. Population with access to drinking water, urban and rural areas, 1990, 2000 and 2011 (Millions). Source: UN (2013a).

Moreover, many of those people are exposed to periods of water stress. The proportion of total freshwater resources used (defined as surface water and groundwater withdrawal as percentage of total actual renewable water resources) was 9.2% globally in 2006, although this masks significant regional variation (Table 14.2). Some regions, notably Northern Africa (89%) and Southern, Western and Central Asia (53-55%) use significantly higher proportions of their available freshwater resources.

Table 14.2. Proportion of total water resources used. Source: UN (2013b).

	around 2006
World	9.2
Developing Regions	7.4
Northern Africa	89.0
Sub-Saharan Africa	3.2
Latin America and the Caribbean	2.0
Caribbean	15.2
Latin America	1.9
Eastern Asia	19.8
Eastern Asia excluding China	20.8
Southern Asia	52.9
Southern Asia excluding India	53.3
South-Eastern Asia	7.8
Western Asia	54.9
Oceania	0.06
Caucasus and Central Asia	55.1
Developed Regions	10.0
Least Developed Countries (LDCs)	4.5
Landlocked Developing Countries (LLDCs)	12.9
Small Island Developing States (SIDS)	1.5

14.1.1.v Trends in ocean (marine and coastal) services (including fisheries and flood defences)

Marine and coastal ecosystems provide a range of services, including carbon storage, food provision, coastal protection, marine organisms for medicines, pharmaceuticals, and other materials, cultural values, and many more (Worm *et al.*, 2006). This section considers overall ocean health in delivering services, and then focuses on two particularly important services relating to fisheries and coastal flood protection.

Ocean health

The Ocean Health Index (Halpern *et al.*, 2012) uses a portfolio of ten public goals for measuring overall condition of marine ecosystems. These goals broadly track ecosystem services and include: food provision (from wild-caught and mariculture sources), artisanal fishing opportunities, natural products, carbon storage, coastal protection, coastal livelihoods and economies, tourism and recreation, sense of place, clean waters, and biodiversity. The index score for the ocean within exclusive economic zone (EEZ) boundaries is 65 out of 100, providing an important benchmark and indicating substantial room for improvement across the goals (Figure 14.9). Index scores varied greatly by country ranging from 41 to 94, with many West African, Middle Eastern and Central American countries scoring poorly (below 50) and parts of Northern Europe, New Zealand, Australia and various island countries and territories scoring relatively well (above 70). Of all EEZs, 18% scored above 70, 91% score above 50 and 9% scored 50 or below.

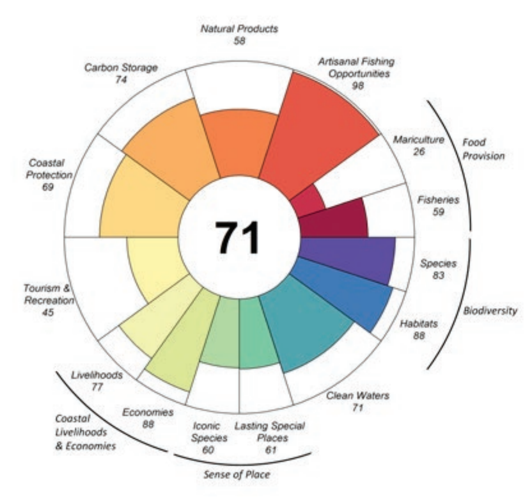


Figure 14.9. 2013 Ocean Health Index score (inner circle) and individual goal scores (coloured petals) for global area-weighted average of all countries. The outer ring is the maximum possible score for each goal, and a goals score and weight (relative contribution) are represented by the petal's length and width, except for 'food provision' sub-goals which are weighted by relative actual yield despite equal width of petals. Source: Halpern *et al.*, (in review). http://www.bipindicators.net/oceanhealthindex, accessed 26 August 2014.

Fisheries

Information on current status and trends of global fish stocks that contribute to the potential supply of fisheries depends on the scope and scale of assessment, with conclusions drawn from the same data depending on perceptions about risk and appropriate management strategies (e.g., Branch *et al.*, 2011). The Food and Agriculture Organization of the United Nations, which classifies all fish stocks, suggests that 30% of fish stocks are overexploited; in 1974 this proportion was just 10% (FAO, 2012; see Target 6 chapter). An additional 40-50% of stocks globally are fully exploited; from a fisheries management perspective, if catch targets are set appropriately and well enforced, then these stocks are being well-managed, whereas from a conservation perspective and/or if management is not effective, then these stocks are at risk of becoming overexploited. Recent work aimed at estimating the status of data-poor stocks suggests that estimates of global stock status may indeed by overly optimistic (Costello *et al*., 2012).

Global trends in fisheries production, i.e., the total amount of wild-caught fisheries harvested, seem to be declining (Figure 14.6)[9]. Global fisheries value (US$) shows a declining and then an increasing trend over the last two decades[10]. Regional assessments of fisheries status and production can vary dramatically from these global estimates, depending on the region. For example, many stocks in the United States are being well managed (Worm *et al*., 2009); regions and countries that can afford to implement formal stock assessments often have in place sustainable management plans. Unfortunately, most places in the world do not have formal stock assessments.

In synthesis, a decline in global fisheries production was observed, though these trends can partly be attributed to changes in reporting methods. These trends together with those on global fish stocks suggest that fisheries may have a declining ability to provide food by 2020.

Coastal flood protection

Coastal habitats such as mangroves, salt marshes, sea grasses and coral reefs provide protection from storm surges and flooding (UNEP-WCMC, 2006; Ferrario *et al*., 2014), and human communities exposed to such risks are inevitably more vulnerable (Danielsen *et al*., 2005). In many parts of the world these coastal habitats were degraded or destroyed by coastal human development decades if not centuries ago. This historic decline creates a management challenge in setting appropriate targets or reference points. Without data for historic extents of these habitats, it is difficult if not impossible to know what level of protection could be provided if habitats were fully restored. Even if these targets were known, they may be impractical in most locations given the permanence of many modifications (e.g., coastal in-fill to create urban areas). A recent study on coastal protection for the United States shows many regions with high vulnerability due to degradation or lack of protective coastal habitats (Arkema *et al*., 2013). Moreover, the first global synthesis and meta-analysis of the contributions of coral reefs to risk reduction and adaptation across reefs in the Indian, Pacific and Atlantic Oceans also reveals that coral reefs are very effective in protecting against natural hazards, by reducing wave energy by 97% on average. The study also estimates, that over 100 million people worldwide may receive risk reduction benefits from reefs or bear the costs of hazard mitigation and adaptation if they are degraded (Ferrario *et al*., 2014).

Even with limited historic data, recent trend data indicate that all coastal habitats have declined in extent or condition over recent decades (Target 5 chapter; Butchart *et al*., 2010). Given the expected future migration of human populations to coastal areas, this pressure will likely increase.

14.1.1.vi Trends for other ecosystem services.

Data are available today for a minimal set of ecosystem services including those that are particularly important for the livelihoods of the poor and vulnerable. Our ability to monitor many more services at local to global scales is increasing (Figure 14.10, Tallis *et al*., 2012; see also Carpenter *et al*., 2006). Also, a wider range of indicators of different services for the supply, delivery, value and contributions to well-being is increasingly available (Tallis *et al*., 2012; Table 14.3).

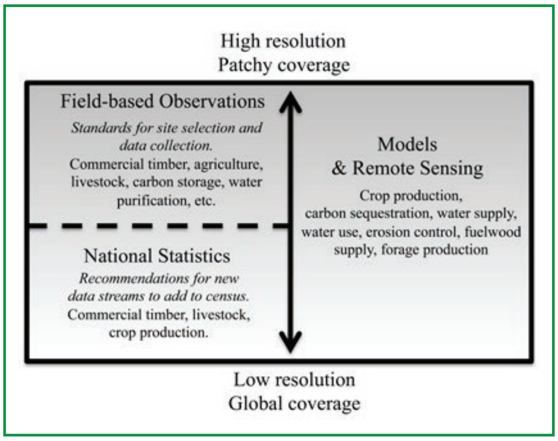


Figure 14.10. Tools available for monitoring ecosystem services at local to global scales. Source: Tallis *et al*. (2012).

Footnotes

[9] *Data were derived from FAO statistics. The FAO reports these data in "major fishing areas" rather than by country or exclusive economic zones (EEZs). Data were reallocated to nations by the Sea Around Us project (1996-2006) and the Ocean Health Index project (2007-2011) (Watson et al., 2004; Halpern et al., 2008; Halpern et al., 2012). FAO data are reported in six taxonomic levels, from broad miscellaneous groups (e.g., "Finfishes" or "Pelagic Fishes") to actual species. The Sea Around Us and Ocean Health Index projects first matched inconsistent names between years to the highest resolution possible. Next, FAO data were matched to more spatially resolved Sea around Us data from 2006 to estimate the proportion of FAO data in each spatial country's exclusive economic zone (EEZ). These proportions were used to allocate fisheries to EEZs for other reporting years. Total annual fisheries production (tonnes) was calculated across all EEZs. Given that FAO methods changed for fisheries data in 2006, the big drop in that year has higher error bars than other year-to-year changes.*

[10] *Value was obtained from FAO data as described for fisheries production. Fisheries value was only available to 2006.*

Table 14.3. Examples of ecosystem services and their indicators of supply, delivery and benefit that can now be monitored. Source: Tallis *et al*., (2012).

Ecosystem Service	Type	Metric	Source	Available globally	Updated regularly
Fisheries Production	Supply	Biomass of abundance of all (commercially) important fishes			
	Service	Landings of (commercially) important species	FAOSTAT	X	X
		Caloric content of these landings			
	Benefit	Market value of the landings			
		Number or percentage of malnourished people			
Biofuel Production	Supply	n/a		n/a	n/a
	Service	Production of commercial oil seed crops	FAOSTAT	X	X
	Benefit	Market value of commercial oil seed crops	FAOSTAT	X	X
Water supply for domestic use	Supply	Volume of surface water or groundwater yield	LPJmL, InVEST	X	
	Service	Volume of freshwater withdrawals for domestic use	FAOSTAT		
	Benefit	Percentage of population with access to clean water	World Bank		
Water supply for irrigation	Supply	Volume of surface water or groundwater yield	LPJmL, InVEST	X	
	Service	Volume of freshwater withdrawals for agriculture	FAOSTAT		
	Benefit	Marginal market value of crops attributable to irrigation			
Nutrient retention for clean drinking water	Supply	Mass of nitrogen or phosphorus retained	InVEST	X	
	Service	Mass of nitrogen or phosphorus retained upstream of the extraction points	InVEST	X	
	Benefits	Avoided water treatment costs	InVEST	X	
Erosion control for reservoir maintenance	Supply	Mass of retained soil	InVEST, SWAT	X	
	Service	Mass of soil retained upstream of reservoirs	InVEST	X	
	Benefit	Avoided dredge costs	InVEST	X	
Flood regulation	Supply	Flood volume regulated by vegetation and soils			
	Service	Area of avoided flood damage due to regulation by vegetation or soil			
	Benefit	Avoided costs due to loss of property or infrastructure			
Nature-based tourism	Supply	Area with attractive natural features or high habitat quality			
	Service	Area with accessible attractive natural features or high habitat quality			
	Benefit	Income from nature-based tourism	IUCN-WCPA		

14.1.1.vii Trade-offs in the delivery of different services and between beneficiaries

Ecosystem change, whether managed or unintentional, affects services and sustainability. Yet patterns of change are not universal across all services. While some services may be in decline, others are stable or increasing, and there are often trade-offs in the delivery of different services between among beneficiary groups or segments of society. Additionally, in coupled social–ecological systems, actions that seek to address both biophysical and social characteristics can create a mutual benefit that improves human health and well-being by addressing both its socioeconomic and environmental determinants, while also promoting conservation and sustainable development (Bunch *et al.*, 2011).

Ecosystem service trade-offs

A trade-off occurs when one or more services decrease as a result of an increase in others (Figure 14.11). Supplying increasing demand for food, wood and water for the current generation often comes at the cost of decreasing the supply of regulating, supporting and cultural services and often with negative impacts on biodiversity and the functioning of ecosystem services (Foley *et al.*, 2005; MA, 2005). Increasing crop yield, for example, has negative effects on water quality if nitrogen is added to crops and then leached to adjacent water bodies (Tilman *et al.*, 2002; Zhang *et al.*, 2007). Timber extraction can have negative impacts on carbon stocks of the forest and the maintenance of particularly sensitive species (Putz and Romero, 2001; Nelson *et al.*, 2008).

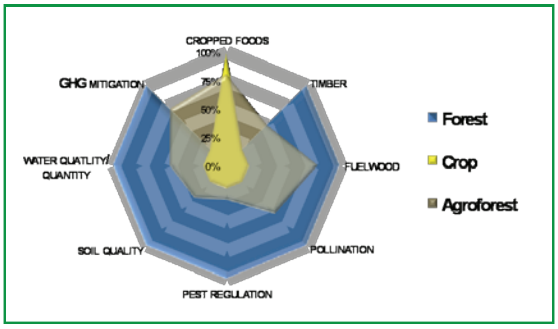


Figure 14.11. Trade-offs in service provision from forested land under different forms of management (Source: Smukler *et al.* 2012).

Trade-offs may be caused by simultaneous response to the same driver or by true interactions among services (Bennett *et al.*, 2009). For example, fertilizer use, the driver, leads to increases in crop yield and decreases in water quality, and yet these responses to fertilizer application are 'independent' in that one effect is not influenced by the other. True interaction among services occurs for example when afforestation enhances carbon sequestration, and when the process of tree growth increases net evapo-transpiration leading to decreasing water availability.

Synergies, instead, are situations in which multiple services increase or decrease together (Bennett *et al.*, 2009). For example, when conservation tillage improves erosion control, this will also improve crop yield). Bundles of services are the collection of services that tend to occur together through space or time as a result of direct interactions among them or concurrent responses to the same driver (Bennett *et al.*, 2009; Raudsepp-Hearne *et al.*, 2010).

Trade-offs among services change through time and across the globe. LPJmL was used to calculate these trends for the case of carbon, water, potential and actual wood harvest. Trade-offs observed here account for actual interactions among services as modelled by this process-based platform. For the case of Scandinavia, carbon storage and timber harvest are at its maximum today and surface water yield has been increasing over the last 55-years. In contrast, for the three Sahelian countries studied, harvest has been increasing at the expense of carbon- and water-related services. These changes in water-related services can also be set in context with changes in crop and timber harvest as well as carbon storage for selected countries or regions (Figure 14.7 D/E).

Trade-off configurations and bundles of services, that is, services that tend to co-occur, change under different management, biophysical and social-ecological conditions. For example, in rural areas of Quebec, proximity to urban centres, predominant economic activities, and current land cover conditions determine different configurations of such trade-offs (Figure 14.12).

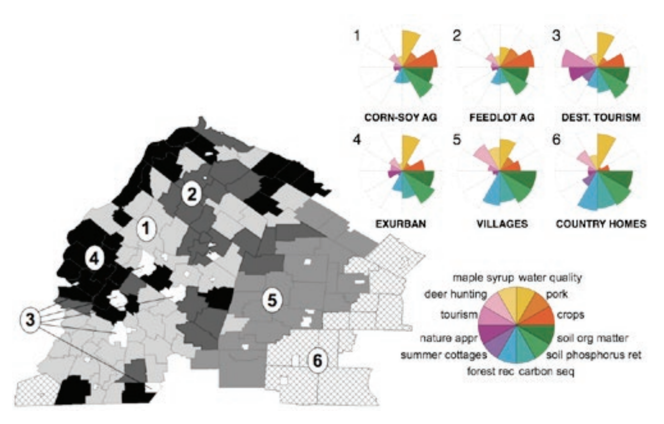


Figure 14.12. Different configurations of trade-offs among services and bundles of services are found among regions with different social-ecological dynamics. This is data from Quebec, Canada. 1 and 2 correspond to areas with predominance of agricultural activities, 3 and 5 to areas with unique natural or cultural features that draw tourism, 4 to areas close to urban centres, and 6 to heavily forested areas. Source: Raudsepp-Hearne *et al*. (2010).

While emphasis has been placed on trade-offs among services through space, much less is known on trade-offs among services through time. For example, what are the long-term consequences of timber or biofuel harvest on the supply of non-timber forest products? What are the long-term consequences of shrimp aquaculture on the maintenance of fisheries yields? The inter-linkages between ecosystem services can change over time as a result of changes in ecosystem processes themselves or the policies that address these services. Understanding the underlying mechanisms behind concurrent *responses* of several services to a driver, as well as the mechanisms behind *interactions among ecosystem services* are needed to manage the strength and efficacy of synergies and trade-offs (Bennett *et al.*, 2009).

Beneficiary trade-offs

Trade-offs may also occur between different groups of people benefitting from different services. This is often related to changing patterns of access and use. For example, as forests are protected for biodiversity and carbon storage, the global population receives greater benefits, as do those able to visit the forest for recreational activities. However neighbouring communities who were reliant on the forest for subsistence (food, fuel, medicines) and who may be denied access will lose out unless compensated in other ways (Schaafsma *et al.*, 2014; Milder *et al.*, 2010). There are not a lot of globally compiled data on changing patterns of access to, and benefit from, ecosystem services. It is well-established that the poor tend to be more directly reliant on local ecosystem services, which tend to be a higher proportion of the so-called 'GDP of the poor' (TEEB, 2010; Roe *et al.*, 2013). As these systems become more degraded, or access is appropriated by others, then these communities suffer.

14.1.2 Projecting forward to 2020

This section of each chapter presents comparable extrapolations of suitable indicators to explore how trends are likely to play out in the near future. It helps answer the question "Are we on track to meet Target 14?" As noted earlier, data are patchy, but the likely answer is that we are far from achieving the target. Extrapolations of suitable metrics (those that meet a particular set of criteria as laid out in Chapter 21 and Tittensor *et al.*, in review) in other chapters suggest that on current trajectories ecosystems will continue to decline, including those delivering essential services to society as a whole and the poor and vulnerable in particular. Extinction risk for pollinator species is also projected to continue to increase (Figure 14.13). This was the only additional metric suitable for extrapolation relevant to Target 14, which clearly only reflects one small part of the picture.

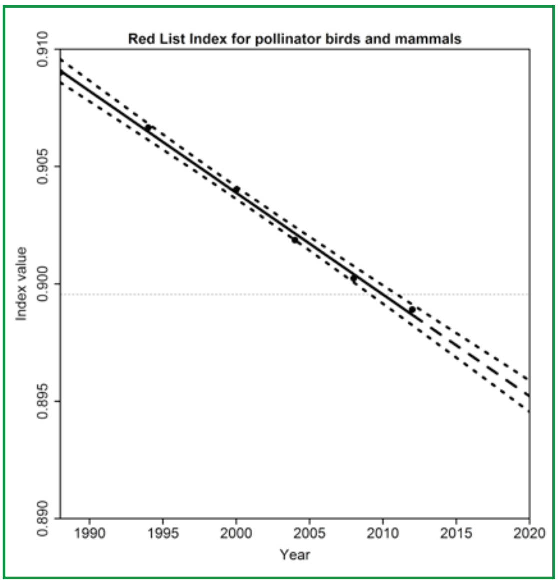


Figure 14.13. Statistical extrapolation of Red List Index for pollinator species (1,188 bird and mammal species) to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Source: Tittensor *et al*. (2014).

Habitat protection and restoration efforts are likely to counter this trend in some places, which will require actions to identify degraded areas suitable for restoration, to assess ecosystem services and the implications of management choices at site and landscape scale, and to develop decision support tools. However, in the absence of widespread evidence of restoration and improved ecosystem condition, the likelihood is that current progress will be insufficient to slow the global decline of essential services from natural ecosystems.

14.1.3 Country actions and commitments[11]

Few countries have set targets which explicitly address the elements in Aichi Biodiversity Target 14 (high). Japan set a target to strengthening the benefits received from biodiversity and ecosystem services including through the conservation and restoration of ecosystem and the Satoyama initiative. Those targets which have been set are generally focused on safeguarding the provision of ecosystem services (low) and few refer to the needs of women, indigenous and local communities or the poor and vulnerable (low). Two examples which are counter to this general trend are Brazil and Finland which have both set targets which refer to, among other things, protecting and restoring ecosystems that provide essential services and which refer to taking into account the need of indigenous communities. Mexico is currently launching a strategy to intensify biodiversity friendly practices in seven different rural economic activities (coffee, cocoa and honey production, agrotourism, etc.). Another example is the European Union, which has combined targets 14 and 15 and set up an ambitious target to maintain and enhance by 2020, ecosystems and their services by establishing green infrastructure and restoring at least 15% of degraded ecosystems.

14.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

Target 14 is very broad, and the full range of relevant actions, relating to each type of ecosystem and its various services and beneficiaries, would, at best, be difficult to address. Notwithstanding this difficulty, we provide several examples of key actions that are necessary (but not necessarily sufficient on their own) for the successful implementation of Target 14. Reference is also made to another major contemporary CBD assessment, the High Level Panel report on global assessment of resources for implementing the strategic plan for biodiversity 2011-2020. Further work and data are necessary for the development of a more comprehensive list of actions to address all linkages between biodiversity, ecosystems and the full range of critical services that they provide.

14.2.1 Actions

Meeting the objectives of Target 14 will not only require complementarity of actions between targets, but often also entail careful trade-offs, particularly when considering the added social dimension of human well-being. For example, preserving critical services provided by ecosystems such as food (which can be both a provisioning and cultural service), the dual objectives of food security and nutrition security must be achieved against a backdrop of rapidly rising demand for food and fuel from an increasing and generally wealthier and more urbanised population. Increasing food and nutrition security among the poorest, most vulnerable populations must be simultaneous with reduced demand for agricultural products among wealthier populations, a reduction of losses and waste, and an overall increase in global food production (Lucas *et al*., 2014). At the same time, a careful balance must be struck to conserve access to wild foods, that are often high in nutritional value, critical to traditional food cultures and upon which many poor and vulnerable populations are particularly reliant, not only as sources of food and nutrition but also the basis of critical cultural services (Fanzo *et al*., 2013; Barhucha and Pretty, 2010). Yet, increasing food production may in turn entail increasing productivity and the conversion of natural habitats to agricultural land, both potentially resulting in biodiversity loss, and possibly reducing access to wild foods as well as eroding other ecosystem services.

As Lucas *et al* (2014) and others have noted, targeted actions can range from the intensification of existing agricultural areas while reducing external impacts to imbedding agricultural production in a multifunctional landscape, optimising the use of biodiversity and ecosystem services. Both approaches may contribute to sustainable intensification of agriculture, but with potentially different consequences for biodiversity, ecosystem services, and livelihood options.

Footnote

[11] This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary and where relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.

Examples of priority actions required to achieve Target 14 include:

- Identifying ecosystems and priority areas that provide essential services (e.g., Key Biodiversity Areas, MPAs, KBAs, etc.), and contribute to livelihoods, health and well-being, with particular attention to ecosystems upon which vulnerable groups are directly dependent for their health, livelihoods and well-being.
- Developing integrated ecosystem assessments (addressing multiple issues including biodiversity, food security, water, health etc. In joined-up ways) including, where possible, multi-scale assessments at local, national and regional levels, with a view to implementing sound ecosystem-based management practices.
- Monitoring population trends and extinction risks of species that provide ecosystems services, with particular attention to species that contribute to essential ecosystem functions such as pollination, for the development of targeted actions.
- Developing new roles for agroecosystems and heterogeneous landscapes to ensure that food production systems and value chains are more equitable, sustainable and nutrition sensitive (see e.g., Fanzo *et al*., 2013). Additionally, the ability of diverse agroecosystems to deliver multiple ecosystem services including food production, climate regulation, erosion prevention, regulation of water quality needs to be sustained through the maintenance of the biodiversity involved in the delivery of these services (Jackson *et al*., 2007).
- Reducing pressures on and, where feasible, restoring the quality and health of ecosystems (e.g., forests, grasslands, wetlands, rivers, and coral reefs) which provide critical ecosystem services (e.g., improving water and air quality, attenuating and moderating flood water flows, regulating disease, etc.) and upon which the health, livelihoods and well-being of many local and indigenous communities and vulnerable populations depend.
- Enhancing ecosystem management for climate change adaptation and disaster-risk reduction (see e.g., Munang *et al*., 2013).
- Establishing systems for inventorying and monitoring the status of critical ecosystems, such as forests, which provide essential ecosystem services including wild foods, medicines and cultural services, with particular attention paid to poor and vulnerable populations most reliant on these for their health, livelihoods and overall well-being (see e.g., Hamilton, 2004).
- Establishing cross-sectoral policy platforms that bring together nutrition and food security, biodiversity, agriculture, health, development, conservation and land-use planning to inform the development of targeted cross-sectoral actions (see e.g., FAO, 2013).
- Improving understanding of the ecology of ecosystem services (=interactions and processes determining the supply of ecosystem services) in important ecosystems. Strengthening core scientific areas of research on ecosystem services, combining ecosystem service research by examining linkages with human health and well-being (including the health-benefiting services provided by ecosystems), agriculture, technology, and time lags (see e.g., Raudsepp-Hearne *et al*., 2010).
- Developing cross-sectoral "green growth" strategies that address the underlying drivers of ecosystem degradation, poverty, unsustainable production and consumption, and disease emergence (including infectious and non-communicable diseases)[12] to develop strategies that also promote equity, sustainable production and use.
- Reflecting community and gender perspectives in the development of policies and programmes aimed at managing essential ecosystem services.
- Removing perverse subsidies for infrastructure that harms, fragments, or degrades ecosystems.
- Investing in traditional ecological knowledge (TEK) including knowledge about ecological systems, processes, and uses held by traditional and indigenous and local communities, and strengthening community-based management regimes to safeguard essential ecosystem services at local level.

Footnote

[12] *There are major economic losses seen from recent emerging zoonotic infectious diseases (EIDs). For example, SARS cost the global economy US$30 billion to US$50 billion (Karesh et al., 2012). Thus, there is an economic argument for prevention of EIDs and potential diseases that could become established or at least their earlier detection (e.g., neglected or other persistently maintained diseases).*

14.2.2 Costs and Cost-benefit analysis[13]

The CBD's High Level Panel on global assessment of resources for implementing the strategic plan for biodiversity 2011-2020 estimated that average annual costs to implement key actions required to achieve Target 14 (particularly those relating to removal of perverse subsidies, investment in TEK and restoration of forests, wetlands and coral reefs) fall in the range US$10-102bn (Talberth and Gray, 2012).

However, it is worth noting that the costs of achieving individual targets are unlikely to be additive. Many of the actions described above directly support the achievement of other Aichi Biodiversity Targets including those relating to subsidies (Target 3), reducing ecosystem loss and enhancing restoration (Targets 5 and 15), sustainable agriculture, aquaculture and forestry (Target 7) reducing multiple pressures (Target 10) and using TEK (Target 18). There may also be synergies with Target 9 if restoration activities include those that address invasive alien species (Talberth and Gray, 2012). Efficient investments to meet multiple targets are therefore possible.

Despite limited data availability on costs associated with identified for Target 14, it is expected that costs are small relative to the value of the goods and services that biodiversity provides (McCarthy *et al.*, 2012). These have been estimated to range between 1% and 4% of the estimated net value of ecosystem services that are lost per year, itself estimated at US$2 trillion to US$6.6 trillion (Balmford *et al.*, 2002; McCarthy *et al.*, 2012). More recently, it has been argued that these and other values have been greatly underestimated. Based on new data from the TEEB study (TEEB, 2010), Costanza *et al.* (2014) compared results with earlier estimates and using different methods to provide updated estimates of the values associated with global changes in ecosystem services from land-use change over the period 1997–2011. The more dynamic simulation, which has made it possible to include a more comprehensive view of the complex interdependencies involved, have led to the conclusion that conservative estimated of the impact of global land-use changes between 1997 and 2011 have resulted in a loss of ecosystem services ranging between US$4.3 trillion and US$20.2 trillion per year.

While economic valuations such as these are not directly incorporated into policy making decisions, they nonetheless remain useful both in highlighting the magnitude of ecosystem services, and in supporting the claim that costs associated with the implementation of actions are likely to be considerably less costly than the continued degradation of ecosystem services. Leveraging partnerships, cross-sectoral collaborations, data sharing and tapping into a broader science base will also contribute to generating more robust and cost-effective actions to implement Target 14.

14.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

If ecosystems are restored and safeguarded this implies that biodiversity that forms part of them will be as well. However, if the focus of efforts to restore and safeguard ecosystems is on the continuing supply of specific services to specific beneficiaries, then the outcome for biodiversity may be more mixed. Services tend to rely on certain ecosystem functions, which may be a result of particular aspects of biodiversity which may not include the most threatened parts (Cardinale *et al.*, 2012; Mace *et al.*, 2012). The dependence of the poor on biodiversity relates to the amount (biomass) of particular harvested elements that are marketed (Roe *et al.*, 2013), but can also depend on a wide variety of plants and animals that constitute important sources of protein, micronutrients, medicine and diverse materials (Caballero *et al.*, 2011; Shackleton and Shackleton ,2004; Kaimowitz and Sheil, 2007). Also smallholders may rely on a wide variety of crop types (genetic diversity) which in turn increase resilience to shocks such as drought, floods and the outbreak of pests and disease (Cattivelli *et al.*, 2008; Hajjar *et al.*, 2008; Jamnadass *et al*, 2013). Whilst maintaining biodiversity helps to sustain the supply of ecosystem services (Balvanera *et al.*, 2004), an emphasis on sustaining key ecosystem services might focus only on some key species that contribute most to the particular functions and corresponding services (Cardinale *et al.*, 2011). It is then unlikely that achieving Target 14 would in itself help to achieve Target 12, for example.

Footnote

[13] *With further time and resources, more information on benefits could be added. This section could also draw on recommendations from other studies of actions required to maintain and restore ecosystems/ecosystem services (especially around water) and to improve benefit distribution.*

14.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

A multitude of quantitative and qualitative models and scenarios relevant to Target 14 have been constructed and applied across a range of disciplines, including those directly assessing ecosystem service futures as well as broader land use, agricultural and development studies. This section provides a summary of some of the global and major regional modelling and scenario studies in this field, to illustrate the range of plausible futures that have been identified and their likely effects on ecosystem services and human health and well-being.[14]

Global studies

Changes in various ecosystem services were modelled for the four future scenarios developed by the Millennium Ecosystem Assessment (MA) (Alcamo *et al*., 2005). Despite being almost a decade old, the MA scenarios represent something of a benchmark for more recent studies and are thus worth returning to. Overall this first assessment shows increased demand for provisioning services while supply of provisioning and regulating + supporting services appeared to decrease. These early results raise early concerns on how the flow of services can be maintained.

The global demand for ecosystem services is suggested to increase substantially (Alcamo *et al*., 2005). Cereal consumption is likely to increase by a factor of 1.5-1.7, water withdrawals by a factor of 1.3-2, and biofuel production by a factor of 5.1-11.3 (Alcamo *et al*., 2005). Large changes in cereal production are also predicted from 1995 to 2050. The largest changes are found in Asia as population and income increases. The second largest changes are found in sub-Saharan Africa where cereal production increases due to population increase and land-use change, especially under the "Order from Strength" Scenario[15] (Figure 14.14). However, other factors may influence this. For example, it has been shown that some geographical areas, particularly sub-Saharan Africa, risk an increase in soil erosion and lower water availability over the coming decades which could slow the rise in food production that would be needed to meet the demand (Figure 14.13; Alcamo *et al*., 2005). As can be seen in Figure 8, the difference in erosion risk is greater between geographical areas than between different scenarios. In sub-Saharan Africa the risk of water erosion almost doubles under the different scenarios because of the net increase in precipitation, widespread replacement of natural vegetation and the expansion of agriculture onto land susceptible to water erosion (Alcamo *et al*., 2005). Substantial but less extreme increases in the area under risk occur in Latin America and Asia. The highest risk of water erosion is found for the Order from Strength scenario because it entails conversion of natural vegetation into agriculture and larger increase in precipitation.

Data on climate change effects on food production suggests a wide projected range (between 5 million to 170 million) of additional people at risk of hunger by 2080, strongly depending on assumed socioeconomic development (Schmidhuber and Tubiello, 2007). Related to food production is pollination. One study has shown that, under the most extreme IPCC scenario, pollination services by managed honey bees are expected to decline by 14.5%, whereas pollination services provided by most native, wild taxa are predicted to increase, resulting in an estimated aggregate change in pollination services of +4.5% by 2099 (Rader *et al*., 2013). However this study focused on one pollinated crop and the findings contrast with current patterns of pollinator decline highlighted in section 2 above. Climate change is also predicted to influence disease risk as disease reservoir species alter their ranges in response to climate change thereby expanding the range of their associated virus (Daszak *et al*., 2012).

Footnotes

[14] *Noting for example that land-use change affects biodiversity and ecosystem services but also human health, being a major driver of disease emergence in humans (Jones et al., 2008; Karesh et al., 2012).*

[15] *The Order from Strength scenario represents a regionalized and fragmented world, concerned with security and protection, emphasising primarily regional markets, and paying little attention to common goods. Nations see looking after their own interests as the best defence against economic insecurity, and the movement of goods, people, and information is strongly regulated and policed. Source: http://www.greenfacts.org/en/ecosystems/toolboxes/scenarios-os.htm.*

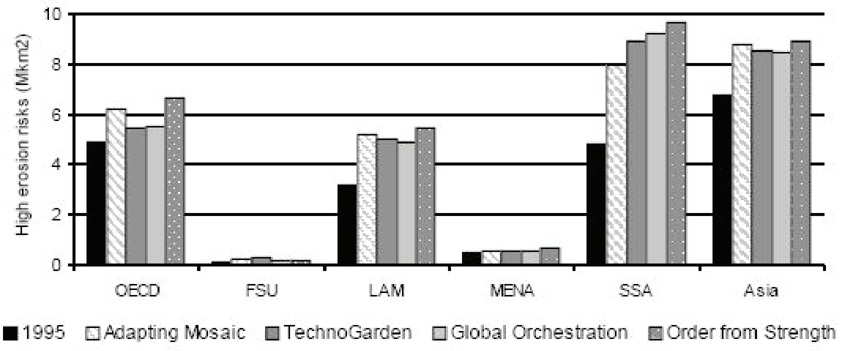


Figure 9.40. Global Area of soils with High Water Erosion Risk in 2050 in the MA Scenarios.
World regions: FSU: Former Soviet Union; SSA: Sub-Saharan Africa; LAM: Latin America; MENA: Middle East Northern Africa. (IMAGE 2.2)

Figure 14.14. Global area of soils with a high risk of water erosion in 2050, under the four different MA scenarios[16]. FSU = Former Soviet Union, SSA= sub-Saharan Africa, LAM = Latin America, MENA = Middle East & North Africa. Scenario names: GO=Global Orchestration; TG=TechnoGarden; AM=Adapting Mosaic; OS=Order from Strength. Calculations using UNEP-GEO methodology
Source: Alcamo *et al*. (2005).

Future fish catch show variable patterns across scenarios and regions although other studies more commonly suggest a decline in marine wild fish stocks. Meanwhile fish consumption is predicted to increase in all scenarios, especially under Global Orchestration. Future demands for protein for fish might then need to be increasingly provided by aquaculture. This corresponds with the findings of a number of more recent food and agriculture assessments described below.

As mentioned above, another ecosystem service that is predicted to change significantly by 2050 in the MA scenarios is water supply (Alcamo *et al*., 2007). By 2050, water stress is anticipated to become more severe in 62.0-75.8% of total river basins and become less severe in 19.7 -29.0% of this area, depending on the scenario and climate model used (Alcamo *et al*., 2005). The most important factor leading to the increase in water stress was identified to be the growth of domestic water use stimulated by income growth. In the MA scenarios population growth, contrary to many beliefs, was a much less important factor in the rise of water stress (Alcamo *et al*., 2007). Climate change is also likely to exacerbate water stress, for which more recent forecasts have been discussed by the IPCC.

Oborn *et al*. (2011) developed five scenarios at global and regional (European) scale with a time horizon of approximately 2050[17]. In the first, 'an overexploited world' where population growth and climate change are high, pressure on land resources is also high as a result of food demand, resulting in declining tropical forests, increased land under agriculture and decreasing soil fertility and yield. Freshwater availability declines and is unevenly distributed, whilst fish and seafood stocks also decline. Even in Europe, this scenario suggests increasing agriculture including biofuels, declining forests and a decline in ecosystem services. In contrast, under the scenario 'A world in balance' where population growth and climate change are low, poverty is reduced and there is strong regional cooperation, rapid technological development in the energy and agricultural sectors, pressures on land are reduced. As a result agricultural expansion is limited. "Soil fertility and production potential as well as availability of ecosystem services are good as a result of diversified production methods and well developed systems for use and management of the agricultural land." Availability of fresh water, and marine exploitation, remain similar to today. Under this scenario in Europe, "The total area of cultivated land is unchanged, but its location has shifted more to the north and east due to climate change resulting in improved growing conditions in northern and eastern Europe and growing drought problems in the south. The potential of fertile soils in the east is utilized to the full. Soil fertility and production potential are good and ecosystem services increase as a result of strong environmental policies. Availability of clean water and distribution of water resources are fairly good (approximately the same as today). Stabilization of the wild fish population is also noticeable in Europe." Other scenarios considered by Oborn *et al*. suggest geographic and other variances dependent upon the balance of factors considered.

Footnotes

[16] Brief descriptions of each of the MA scenarios can be found at http://www.greenfacts.org/en/ecosystems/toolboxes/scenarios-os.htm.

[17] These were developed, using morphological analysis, to create common conceptual frameworks for the purpose of identifying future research issues. Consequently, the scenarios are explorative and not intended to present the most desirable or probable visions of the future.

The Agrimonde assessment by Paillard *et al.* (2011)[18] constructed two global scenarios from the perspective of food and agriculture. One is based on the MA's Global Orchestration scenario (which prioritizes economic development, trade liberalization and technological development) while the other is a more 'desirable' scenario based on sustainability principles, assuming in particular that by 2050 the world will have been able to implement a sustainable food and agricultural system. This assumption builds in a consideration of the importance of forest-based ecosystem services to sustainable agriculture. Both scenarios envisage significant land-use change by 2050, including major increases in cultivated land, particularly in sub-Saharan Africa and Latin America (Table 14.4) which also see significant yield gains although it is recognized that soil fragility and aridification risk limiting yields. Where the two scenarios contrast relates to forests and pastures, with the sustainability scenario limiting forest loss in favour of increased grassland losses. As with other scenario studies, Agrimonde concluded that marine wild food sources will decline (with a concurrent rise in aquaculture) whilst tensions over fresh water use will worsen.

Table 14.4:. Regional land use rates of variation: 1961-2000 and 2000-2050 in the Agrimonde scenarios Source: Paillard *et al.* (2011).

Type of land use	Region	Rate of variation 1961-2000	Rate of variation 2000-2050		Region	Rate of variation 1961-2000	Rate of variation 2000-2050	
			AG1	AGO**			AG1	AGO**
Cultivated land*	**ASIA**	+23%	+23%	+11%	**FSU**	-15%	+53%	+10%
Pasture land		+36%	-9%	+30%		+19%	-16%	-41%
Forests		-5%	-10%	-11%		-3%	0%	+12%
Cultivated land*	**LAM**	+58%	+91%	+64%	**MENA**	+14%	+9%	+12%
Pasture land		+20%	-20%	-1%		+39%	-2%	-2%
Forests		-9%	-4%	-1%		-33%	0%	-35%
Cultivated land*	**OECD**	-2%	+18%	+12%	**SSA**	+33%	+76%	+58%
Pasture land		-8%	-23%	-19%		+2%	-12%	+49%
Forests		-9%	+10%	+13%		-10%	-9%	-31%
Cultivated land*	**World**	+12%	+39%	+23%				
Pasture land		+11%	-15%	+7%				
Forests		-9%	-1%	-1%				

* Cultivated land = food crop area + non-food crop area

** As the reference surface areas for the year 2000 differ in the MA and Agribiom data, a corrective factor has been applied to the MA gross surfaces (for the Global Orchestration Scenario), to be comparable with those of Agrimonde 1.

Regional predictions – Europe and North America

Scenarios for Europe, alongside that of Oborn *et al.* (2011) described above, suggest that there will be large changes in the supply of ecosystem services if climate and land-use change substantially. Ecosystem service supply during the 21st century was modelled by (Schröter *et al.*, 2005) from a range of ecosystem models and scenarios of climate and land-use change across Europe. Large changes in climate and land use were associated with important changes in service supply. The area dedicated to biofuels increased across Europe, while the choice of biofuel crops decreased in the South. Scenarios with increased population and climate change led to increased amounts of people living in water-stressed watersheds and exacerbated water deficiency for many already stressed areas such as southern Europe (Figure 14.15; see also Metzger *et al.*, 2006). Wood production was predicted to increase as climate change enhanced forest growth, especially in northern Europe, with a tendency towards decreased management intensity if demand decreased. Carbon stocks increased due to decreased land-use change and CO_2 fertilization, but warming led to carbon losses increasing by the end of the century.

Footnote

[18] *The aim of which was to produce scenarios of global and regional evolution in agricultural production, consumption and trade, as well as in scientific and technical knowledge on agriculture, with a view to drawing conclusions on the possible roles for research, public policies and international regulations.*

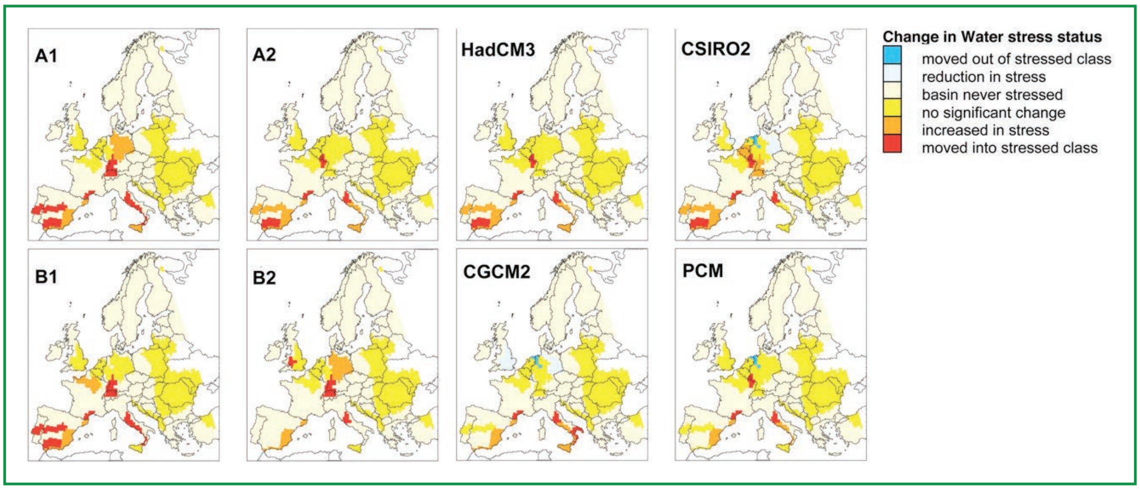


Figure 14.15. Stress status of water basins by 2080. Source: Schröter *et al*. (2005).

Based on an interpretation of IPCC emission scenarios, Rounsevell *et al*. (2006) suggest that in coming decades Europe is likely to witness small increases in urban areas and generally large reductions in agricultural areas for food production (as a result of assumptions regarding the role of technological development), partly compensated for by increases in bioenergy production, forest land and areas protected for conservation and/or recreation. Exact spatial patterns of land-use change vary between scenarios in this study, but nevertheless there is broad coherence in outcomes at a regional level (Figure 14.16).[19] The Agrimonde scenarios of Paillard *et al*. (2011) described above suggest forest gains and pasture/grassland losses in OECD countries, in line with Rounsevell *et al*. (2006) for Europe, although the contract in their anticipations of cropland gains and losses (Table 14.4).

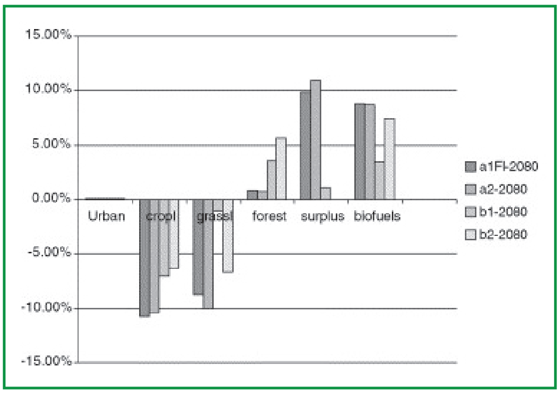


Figure 14.16. Aggregated land-use change trends in 2080 for Europe for the A1F1, A2, B1 and B2 (HadCM3) scenarios (the y-axis represents the absolute area as a percentage of the total European land area). Source: Rounsevell *et al*. (2005).

A sub-regional European study, the UK National Ecosystem Assessment also explored the implications of climate and land use changes within six contrasting scenarios (Figure 14.17). "Storylines that emphasised environmental awareness and ecological sustainability resulted in significant gains in the output of a broad range of ecosystem services, in contrast to storylines that emphasised national self-sufficiency or economic growth. Land-use change and pollution continue to be major drivers of change for biodiversity and ecosystem services, although by 2060 climate change is also predicted to be a significant driver of ecosystem services and of losses and gains of species throughout the UK." (UK NEA, 2011 p.44).

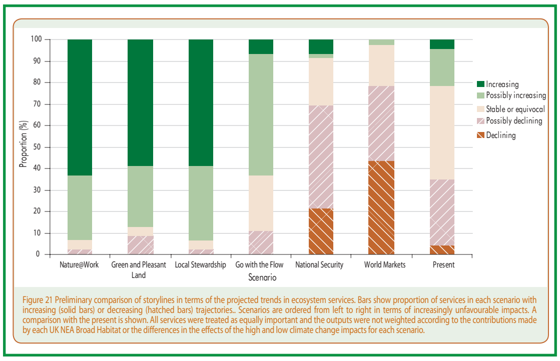


Figure 14.17. Results of six future scenarios showing proportion of ecosystem services increasing, declining or remaining stable by 2050. Source: UK NEA (2011).

Footnote

[19] *Amongst other things the paper discusses technical and conceptual difficulties in developing future land-use change scenarios. "Primary amongst these uncertainties are that scenario development involves interpretations based on judgements that may be subjective. In many cases different scenario developers would make a different judgement when faced with the same scenario framework." This important point should always be borne in mind.*

A US study of coastal protection and vulnerability predicted that climate change will increase coastal flooding and sea level rise significantly by 2050 (Arkema *et al*., 2013). Across all scenarios, the results suggest that more coastal segments will be highly exposed to hazards and that the amount of highly threatened people (particularly the elderly and the poorest) and property will increase by 30-60% by 2100 (Arkema *et al*., 2013). Figure 14.18 illustrates how vulnerability to coastal hazards and the importance of natural habitats vary across the United States in 2100. The highest vulnerability to hazards was found in the east and gulf coast in comparison to the west coast under all future scenarios. Coastal ecosystems provide protection from hazards to the largest amount of people and socially vulnerable populations and property in Florida, New York and California. The importance of protecting the coastal habitats, particularly in the states with the highest hazards risks is clear, as for some states the population at risk almost doubles if the habitat would disappear (Arkema *et al*., 2013).

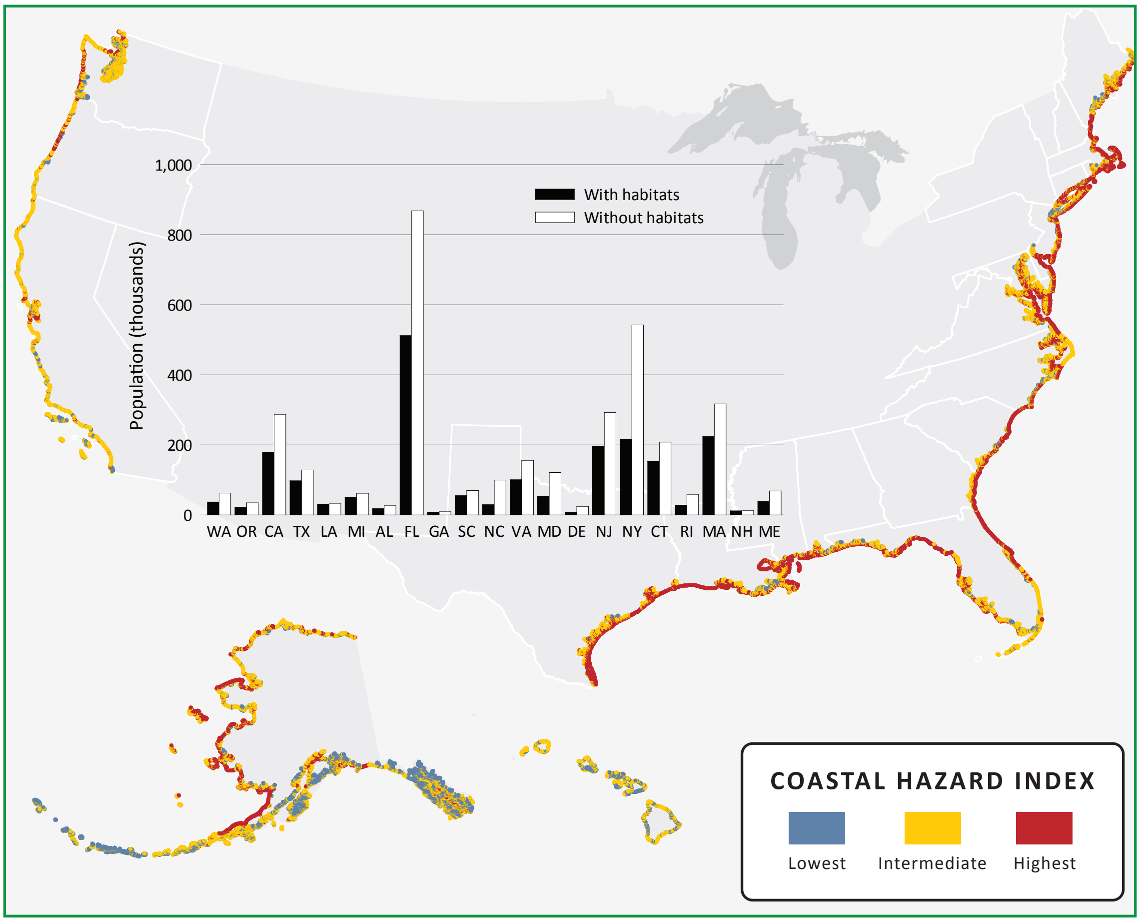


Figure 14.18. Exposure of the US coastline and coastal population to sea-level rise in 2100 (A2 scenario) and storms. Warmer colours indicate regions with more exposure to coastal hazards (index >3:36). The bar graph shows the population living in areas most exposed to hazards (red 1 km^2 coastal segments in the map) with protection provided by habitats (black bars) and the increase in population exposed to hazards if habitats were lost owing to climate change or human impacts (white bars). Letters on the x axis represent US state abbreviations. Data depicted in the inset maps are magnified views of the nationwide analysis.
Source: Arkema *et al*. (2013).

Regional predictions – sub-Saharan Africa

In one study, future changes in land biogeochemistry were modelled using LPJmL (see above) for east Africa (Doherty *et al*., 2010). Soil carbon increased in some scenarios but decreased after 2050 for some of the models (Figure 14.19). All the future scenarios showed a regional increase in net primary production (NPP) (18-36%) and total carbon storage (3-13%) by 2080-2099. According to Doherty *et al*. (2010) enhancement in NPP may lead to improved crop yields in some areas. In contrast, Thornton *et al*. (2009) suggest that both the production of maize and beans will suffer losses of 5-15% in yield by 2050 depending on emission scenarios used.

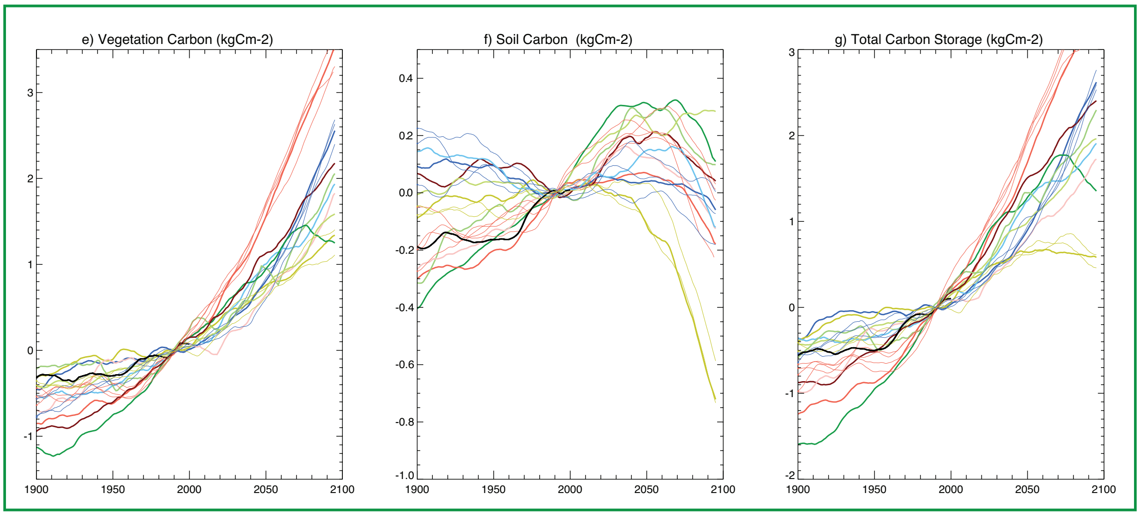


Figure 14.19: Vegetation and soil carbon storage projections. Source: Doherty *et al*. (2010).

In a very different study methodologically, Magnusson *et al*. (2012) applied four of the scenarios developed by Oborn *et al*. (2011, described above) in the context of sub-Saharan African agriculture. In a world of 8 billion people with relatively high climate change and technological development, sub-Saharan Africa is characterised by unevenly distributed inward investment and foreign utilization of land. Both increase in agricultural land use and areas of abandonment and desertification are apparent. "Soil fertility, production potential and ecosystem services are declining, but this is partially disguised by the availability of cheap fertilizers. Productivity is high when fertilizer is used. Pests and diseases of crops and livestock are increasing due to climate change. Newly cultivated areas where populations are neither resistant nor immune are especially vulnerable." Access to water is more unevenly distributed than today, wild fish availability is declining and climate-induced severe weather events are increasing. Women have little power and there is little effort to change this, whilst the gap between rich and poor is widening.

In a more extreme scenario, greater human population increase to 11 billion worldwide leads in sub-Saharan Africa to "pressure for new arable land, consequently in many areas grazing land and forest is brought under cultivation. Since states are weak, environmental policies are also weak or non-existent. Soil fertility, production potential and ecosystem services decline. Many cultivated soils are prone to erosion. Climate change results in crop and livestock pests and diseases posing greater problems than today. There is less access to water and it is more unevenly distributed than today. This is accentuated by more extreme weather events, such as recurring downpours resulting in flooding. Overfishing and widespread environmental destruction results in decreased availability of fish."

Conversely, in a world with greater cooperation, stronger environmental policies and more constrained population growth and climate change, sub-Saharan Africa is characterised by greater stability and more diversified production systems. Although arable land is increasing, "soil fertility, production potential and ecosystem services are improving because knowledge about sustainable cultivation is good, and sufficient inputs are available. Access to water is the same as today, but is more unevenly distributed. However technological development in this area lessens the effects of uneven distribution of water." Development, including rural business enterprise, has led to a more equal distribution of resources, with women gaining power in the home and in society.

In exploring African futures to 2050, Cilliers *et al.* (2011) noted a changing economic landscape and in particular the rise in influence of Asia, especially China, on the continent, a rapidly growing population particularly in Eastern and Western Africa (even as growth rates themselves decline), increasing urbanization and with it increasing human development as measured by the Human Development Index (albeit with pockets of stagnations and enduring inequality). According to this study, income inequality is likely to persist or worsen across the continent, and extreme poverty will remain high. It also noted that Africa holds 60% of the world's unused arable land and that agriculture is key to poverty alleviation and food security on the continent. Food production, particularly in East and West Africa, is therefore likely to increase markedly (Figure 14.20). However the effects of climate change create uncertainty regarding yields. Although water stress will rise in some areas, overall this study noted that Africa is relatively water-rich (albeit with variable and unpredictable supply). Changes in water availability could result in significant land degradation and population displacement in parts of the continent.

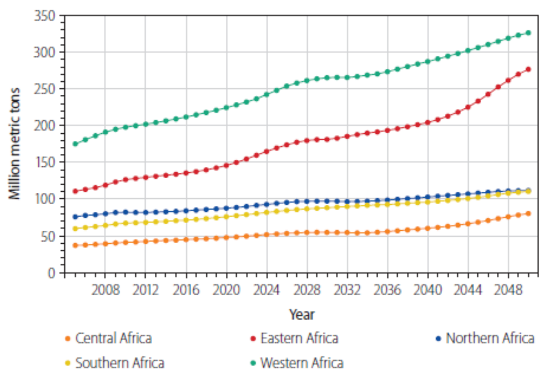


Figure 14.20. African food production by region. Source: Cilliers *et al.* (2011).

In the Agrimonde study described above (Paillard *et al.*, 2011), despite the focus on the maintenance of ecological infrastructure in the sustainability scenario, forests continue to decline in both scenarios in sub-Saharan Africa and Asia whilst grasslands in these regions increase under Global Orchestration and decline under the sustainability scenario. These high levels of conversion imply that ecosystem services will continue to be eroded alongside biodiversity and both water and carbon cycles. The authors note that, particularly in sub-Saharan Africa and even in the sustainability scenario, agricultural intensification is only likely to build in ecological concerns once environmental challenges outweigh development challenges in the region. Nevertheless a multifunctional agriculture that incorporates ecological infrastructure and efficient management of the diversity of ecosystems is viewed as plausible given appropriate innovation in farming practice, alongside the maintenance of natural areas in reserves and corridors.

Implications for biodiversity

Regarding implications for biodiversity on a global scale, scenarios for particular ecosystems in 2050 have been reported elsewhere in this report. Most predict continuing declines over the 21st Century, including for example in southern Africa (Biggs *et al.*, 2008). The global expansion of agriculture is likely to have major implications for biodiversity. Considering land-use scenarios on global forest area, models suggest only slight declines over the next few decades. However, the loss of forest area in the tropics and subtropics is predicted to be extensive, although these losses are partially offset by predicted increased forest cover in the Northern Hemisphere (Pereira *et al.*, 2010). Therefore, the overall impacts on biodiversity from tropical forest losses may be worse than the aggregate global forest projections might indicate. However, as with other changes described in the preceding sections, the projected changes in biodiversity vary substantially depending upon the scenario used, as illustrated for forests below[20] (Figure 14.21; Pereira *et al.*, 2010). This indicates that if the right policy decisions are made there is much opportunity to intervene and reduce the decline.

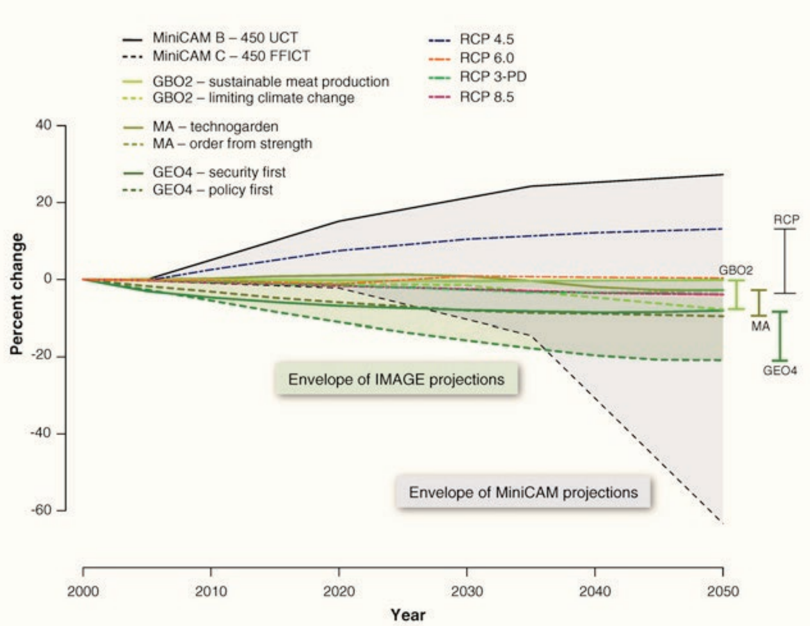


Figure 14.21. Change in the extent of forests to 2050 in different global scenarios: MA scenarios, GBO2 scenarios, GEO4 scenarios, MiniCAM scenarios, and RCP scenarios for IPCC-AR5. For each study, trajectories of the two most contrasting scenarios are shown. By 2050, the envelope of scenarios with the IMAGE model (MA, GBO2, GEO4) is narrower than the envelope of scenarios based on the MiniCAM model. Source: Pereira *et al.* (2010).

Footnote

[20] *Noting that forest area alone is not necessarily a good proxy for biodiversity per se.*

14.5 UNCERTAINTIES AND DATA AND KNOWLEDGE REQUIREMENTS AND GAPS

The major uncertainty relating to an assessment of progress towards Target 14 rests on the paucity of consistent global time series data on change in ecosystems, including ecosystem health, that supply important services and in particular those relating to the poor and vulnerable. Advances in remotely sensed data products will help (Secades *et al.*, 2014) but the need for greater *in situ* coverage of ecosystem health (functioning and condition) should not be overlooked given interest in this topic in the broader sustainable development agenda (Griggs *et al.*, 2013).

Data on the supply and utilization of ecosystem services is also very patchy (Layke *et al.*, 2012), although focus on this is increasing (Tallis *et al.*, 2012) and guidance is available (e.g., UNEP-WCMC, 2011). Perhaps most significantly, however, is the gap in our ability to connect ecosystem change to change in human well-being in meaningful ways.

As a result of these constraints, most ecosystem assessments rely on expert judgement to fill data gaps on ecosystem service supply, and most say little about distribution to different beneficiary groups. A few global models exist for some services. There is a need for significant efforts globally to boost integrated ecosystem science and social science to provide robust ecosystem service and beneficiary dynamics data to support decision making for sustainability.

Species trends: The RLI is only moderately sensitive; utilized species indicators are biased toward particular taxonomic groups. Trends for other taxonomic groups, particularly invertebrate pollinators are not yet available. For pollinating bees, a global assessment is needed, through the constitution of a global database of biodiversity, the analysis of potential distribution changes depending on ecological traits, and the definition of agricultural practices in favour of their conservation.

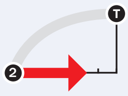
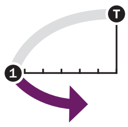
Timber analysis: One important limitation of this data is that the above assessment targets only the wood that is marketed. Illegal wood harvest is not considered in these lists but can account for a large fraction of total wood production in some countries (Lawson and MacFaul, 2010). Also, local wood harvest for self-consumption that is very frequent in many rural areas of many countries could not be considered here. FAO data can be very accurate for developed countries but is often not for many developing ones.

Wild meat: Wild meat harvest is likely to be even higher that the figures reported here considering that FAO statistics do not take into account illegal poaching or local wild meat harvest that is not marketed. FAO data can be very accurate for developed countries but is often not for many developing ones.

Modelled services: LPJmL models are based on global climate and vegetation maps that are more accurate for developed countries than for developing ones for which data for validation is lacking. Data from LPJmL allows for estimation of potential service supply but not actual delivery of the service to human populations. Further work is needed to assess the contribution of these services to meet the needs of women, indigenous and local communities and the poor and vulnerable.

With regard to the future scenarios on soil erosion by Alcamo *et al.* (2007), the estimation of risk of water erosion does not take into account management practices that have an important effect on the rate of water erosion. On one hand, the risk of erosion is magnified by soil tillage and other mechanical disturbances. On the other hand, it can be minimised by conservation measures, such as contour ploughing and terracing.

14.6 DASHBOARD – PROGRESS TOWARDS TARGET[21]

Element	Current Status	Comments	Confidence
Ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded...		High variation across ecosystems and services. Ecosystems particularly important for services, e.g., wetlands and coral reefs, still in decline	Low
...taking into account the needs of women, indigenous and local communities, and the poor and vulnerable		Poor communities and women especially impacted by continuing loss of ecosystem services	Low

Authors: Matt Walpole, Patricia Balvanera, with contributions from Stuart Butchart, Ben Halpern, Lisa Ingwall-King, Daniel Karp, Jennifer van Kolck, Sandra Quijas, Belinda Reyers, Cristina Romanelli, René Sachse, Kirsten Thonicke, Megan Tierney, Britta Tietjen and Ariane Walz.

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Footnote

[21] *This provides a current assessment of progress towards the Aichi Biodiversity Target based on the material presented in this chapter and the expert judgement of the authors of the GBO-4 Technical Report. It is subject to change as additional material becomes available, including information from national reports, NBSAPs and the BIP partnership.*

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TARGET 15: ECOSYSTEM RESTORATION AND RESILIENCE

By 2020 ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification.

PREFACE

Restored landscapes and seascapes can improve resilience contributing to climate change adaptation and generating ecosystem services and associated benefits for people, in particular indigenous and local communities and the rural poor. Ecosystem resilience is the capacity of ecosystems to absorb and adapt to disturbances while preserving their ecological functions and without moving to a new state governed by different processes and controls (Carpenter *et al.*, 2001). In some cases, restoration projects attempt to return ecosystems to states prior to regime shifts and thus recover lost ecological functions and processes (Suding *et al.*, 2004). The wider application of restoration efforts could also contribute to the achievement of the objectives of the Convention, and generate significant synergies with the United Nations Framework Convention on Climate Change (UNFCCC), the United Nations Convention to Combat Desertification (UNCCD) and the United Nations Forum on Forests (UNFF).

Different management strategies exist that contribute to enhance resilience in social-ecological systems.. Resilience can be addressed by focusing on keystone structuring processes that operate across scales and that increase the buffering capacity of ecosystems, on sources of renewal and reformation, and on multiple sources of capital and skills. However, no single mechanism can guarantee the maintenance of resilience (Gunderson, 2000). Within Target 15, we chose to address resilience via the restoration of degraded ecosystems rather than via ecosystem conservation and the limitation of land-use change and resource extraction, which are addressed in other targets.

Restoration activities are broad and range from ecological restoration of highly degraded areas such as former mining sites, restoration of wetlands for specific regulation services such as flood protection and water purification, changes in agricultural practices to promote soil restoration, to passive restoration of abandoned areas as in the case of rewilding. Some restoration activities are the result of environmental regulations that require developers and industry to revert or mitigate their impacts on ecosystems. But there is a growing set of voluntary interventions and incentive schemes that promote ecological restoration, including biodiversity offset schemes and carbon credit schemes, as well as schemes to promote afforestation to restore key ecosystem services (e.g., "Grain for Green" in China).

Relevant indicators for this target include: areas being restored to recover native habitats or specific ecosystem services; area of farmland undergoing passive afforestation; areas of highly degraded ecosystems being restored to respond to regulations. In each case an assessment of the proportion of extant degraded ecosystems may be required to quantitatively assess progress towards the 15% restoration goal defined in the target.

Our analysis discusses different approaches to assess degradation, but emphasize an analysis of ecological restoration approaches independently of a precise definition of a proportional restoration target. When possible, we assess the potential contribution of the restoration for climate mitigation and to combat desertification. We do not include marine ecosystems in this assessment as they are treated elsewhere in this report.

1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

15.1.1 Status and trends

There are several processes that may lead to ecosystem degradation including cultivation, overgrazing, logging, burning, eutrophication, hydrological disruption, species invasions, removal of native species and soil contamination (Rey Benayas *et al*., 2009; Rey Benayas and Bullock, 2012). Estimates of the extent of global land degradation (Figure 15.1) range roughly between 19 million km² land with reduced soil productivity up to about 112 million km² land that has been impacted somehow by humans through cultivation or other habitat conversion (Oldeman, 1998; Ellis *et al*., 2010; Caspari *et al*., 2014). This is about 15% and 78% of the global ice-free land, respectively. The lower bound can be even be smaller if one restricts out attention to soils that are considered highly degraded, which cover only 2% of the terrestrial ice-free surface (Figure 15.1 and Figure 15.2). And there are other possible metrics of degradation. Hoekstra *et al*. (2005) identified that within 142 out of 810 biomes, the percentage of area converted to human dominated land uses outweighs the percentage of areas protected, and estimated a total of 22% of the global land being converted and degraded. Bai *et al*. (2008) reported that 24% of the global land area exhibit a reduction in net primary productivity as measured by the Normalized Difference Vegetation Index (NDVI) between 1981 and 2004 (Figure 15.2). The range of these figures suggest that the assessment of the quantitative component of the target may require a precise definition of "degraded ecosystems", at least for each major ecosystem type.

The degradation of ecosystems by anthropogenic pressure results in habitat loss and fragmentation, leading to biodiversity alterations, including the loss of functional diversity (Pereira *et al*., 2012). These degradations call for specific actions designed to restore and enhance the resilience of those damaged ecosystems. Here we assess major trends in restoration activities along two axes: level of degradation of the ecosystem and effort required for the restoration (Figure 15.3).

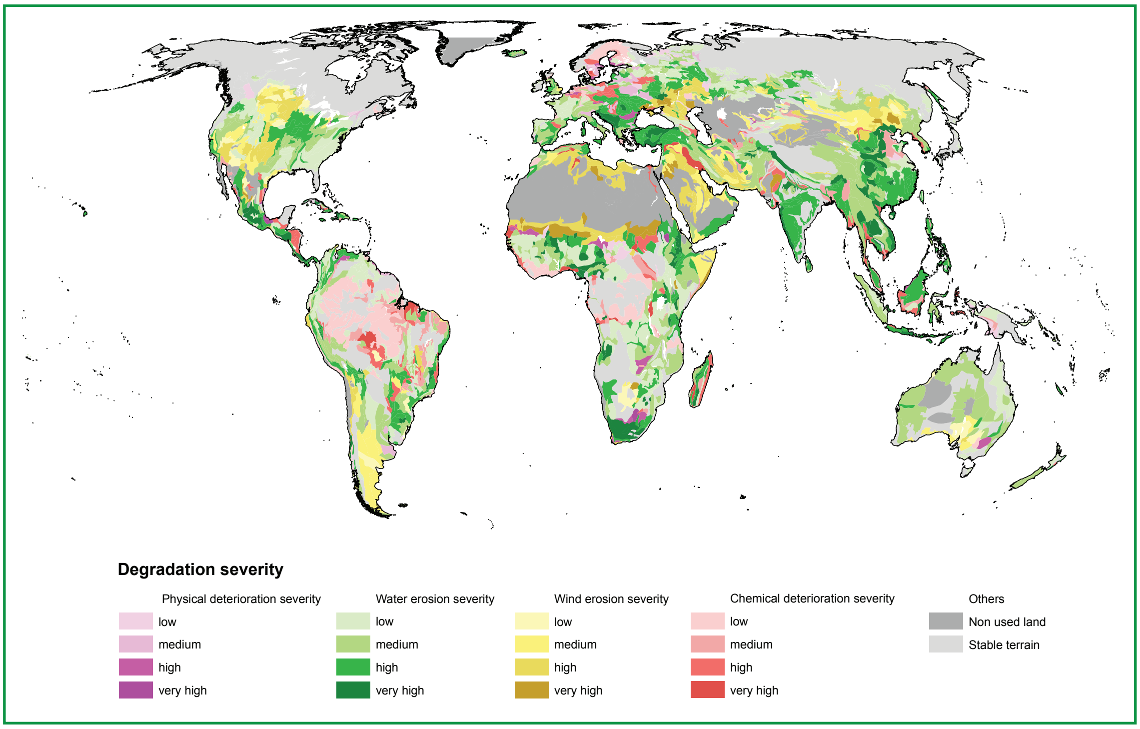


Figure 15.1. Global assessment of the status of human-induced soil degradation, by 1990[1] (Eckert IV projection). Types of degradation in this assessment include wind and water erosion, and physical and chemical degradation. The map illustrates the "dominant" degradation type for a given area, which does not exclude that other types of degradation can occur. Source: Adapted from Oldeman *et al*. (1998).

Footnote

[1] *Source: http://www.isric.org/projects/global-assessment-human-induced-soil-degradation-glasod*

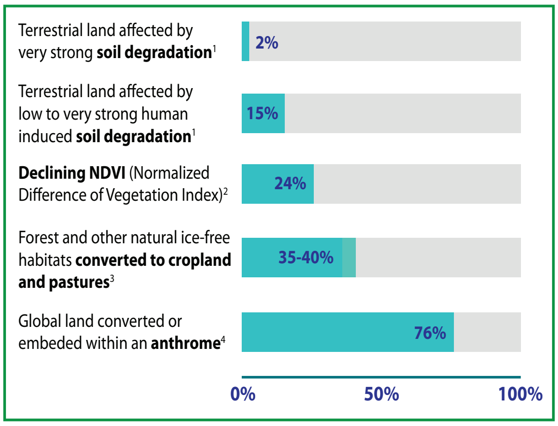


Figure 15.2. Percentage of global land affected by human pressure and/or human-induced land-use change. Source: Adapted from Caspari *et al*., 2014 ([1]Oldeman, 1998; [2]Bai *et al*, 2008; [3]Pereira *et al*, 2012; [4]Ellis *et al*., 2010).

Restoration ecology has matured in the last couple of decades and it has attracted increasingly more attention, with an increase of 700% of journal articles in the ISI Web of Science with the keyword "restoration" between 1995 and 2010 (Suding, 2011).The educational investment in restoration has also increased (Nelson *et al*., 2008). In terms of global assessments of restoration, comprehensive reviews have been published with an eye on ecosystem services and biodiversity outcomes (Jones and Schmitz, 2009; Rey Benayas *et al*., 2009). Nevertheless, the efficiency and the methodology of restoration are still debated, both in the scientific community (Palmer and Filoso, 2009) and by policy-makers (Nellemann and Corcoran, 2010).

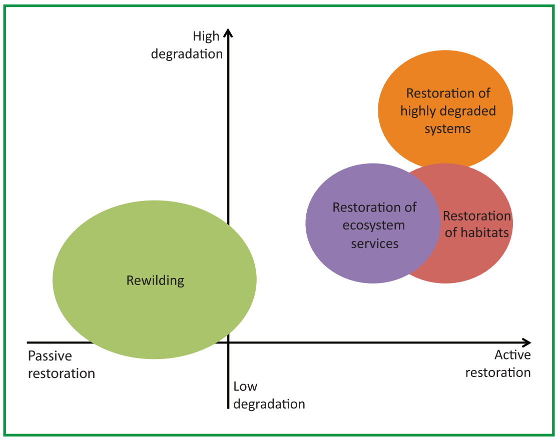


Figure 15.3. Restoration options discussed in this chapter. Each management option is placed according to a gradient of degradation and a gradient of restoration effort.

Science is important not only for designing restoration projects and proposing restoration mechanisms but also in establishing restoration goals (Choi, 2007) and in evaluating their achievement (Nakamura *et al*., 2014). Restoration success might rest more on establishing realistic goals than in finding miraculous tools for turning back in time before the degradation occurred (Suding *et al*., 2004). Moreover, current restoration goals should be adapted to the on-going global changes, especially projected climate change. There are recent advances in formulating new paradigms and models of restoration trajectories and these models help in clarifying what are the questions that need to be addressed in restoration projects (Lampert and Hastings, 2014). One of the new paradigms is the enhancement of ecosystem resilience as a goal. Still, resilience raises important challenges concerning quantification and data availability for proper monitoring (Carpenter *et al*., 2001; Folke *et al*., 2004). Therefore, we report here other restoration metrics such as area under intervention or area undergoing passive afforestation.

Goals cannot be discussed only in ecological terms. Considering the trade-offs inherent in restoration, the interests of different stakeholders, and the financial costs involved (Birch *et al*., 2010), the need for progress in social negotiation and stakeholder implication is evident (Aronson *et al*., 2010). Consolidated guidance for addressing many of these issues is increasingly being advanced (e.g., Keenelyside *et al*., 2012).

The set of active restoration tools has gradually evolved from approaches such as the planting of vegetation without any concerns for biodiversity (Rodrigues *et al*., 2009) to more complex approaches regarding planting methods and seed mixes. Additional restoration interventions, both biotic and abiotic, have been developed in the last years (Hobbs and Cramer, 2008): control of grazing and herbivory, structural alteration of vegetation, soil ripping, creation or removal of physical barriers for water flows, and fire regime management. Invasive species management (Funk *et al*., 2014), and the use of engineering species (Jouquet *et al*., 2014) have also been used successfully in restoration projects. Moreover, in many restoration cases, the reconstruction of the patterns of species interactions or the targeting of specific ecosystem functions seems to be crucial for the success of resilience restoration (Henson *et al*., 2009; Jouquet *et al*., 2014; Rodrigues *et al*., 2009).

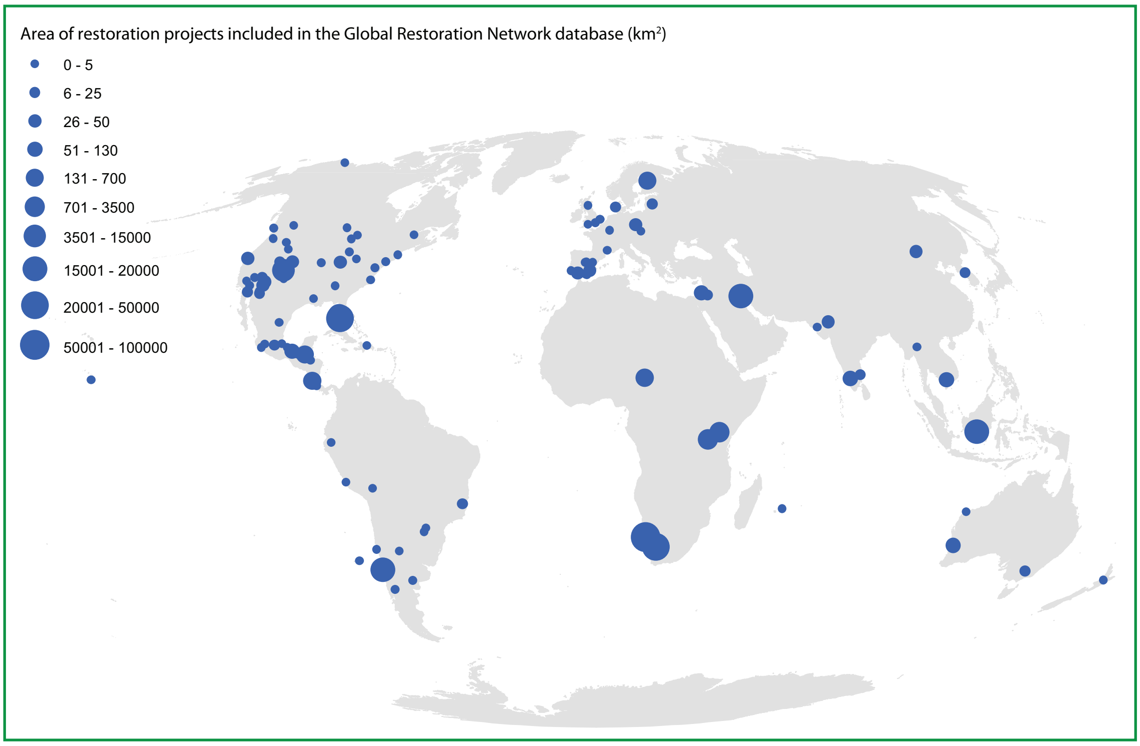


Figure 15.4. Active restoration projects in the Global Restoration Database (February 2014). We have excluded projects that seemed to refer to passive restoration, leaving 124 projects for analysis. The size of the dot represents the area of the restoration project. The projects without a specified area in the database (26 projects) were included in the first area class.

Considering the current accelerated damage to tropical ecosystems and the fact that large and increasing human populations in tropical countries depend directly on ecosystem services for daily sustenance (Aronson *et al.*, 2010; Kaimowitz and Sheil, 2007), tropical restoration should be a high priority. However, the geographical patterns of restoration projects seem to be different, with the highest investment levels in North America and Europe (Figure 15.4). Yet the low level of completeness of existing restoration databases limits a precise assessment of the geographical distribution of restoration projects (Aronson *et al.*, 2010). The geographical bias of restoration projects towards developed countries is likely related to economic resources, political willingness, and expertise (Wortley *et al.*, 2013), especially when restoration involves high investment. But the bias may also be related to land-use history. On one hand, the more recent habitat losses and larger remaining natural areas in tropical regions make them more resilient and passive restoration more likely (Rey Benayas *et al.*, 2009). On the other hand, pressures for degradation due to human population growth and global trade still remain high in the tropics, and restoration programmes, to be successful, need to better incorporate the local socioeconomic dynamics (Blignaut, 2009).

Active restoration targeted at specific habitats

Restoration directed at specific habitats is often justified by biodiversity concerns and the cultural significance we place on these particular habitats. Thus, different areas of the world have invested in restoring different types of habitats, depending on the cultural and natural history background of the areas. High-biodiversity grassland restoration represents a great focus for current European projects (Fagan *et al.*, 2008) while in the tropics the emphasis is on forest restoration (Hall *et al.*, 2011). The European grasslands have suffered from intensification or cessation of traditional agricultural practices and they are now one of the key aims of the European Union conservation policy and agro-environmental schemes, with a budget of over 2 billion euros (Fagan *et al.*, 2008; The Council of the European Union, 2005). Forest restoration, especially through plantations, was also important in several European regions in past centuries, especially for timber and firewood production. (Farrell *et al.*, 2000; Martin *et al.*, 2012). The tropics have experienced high deforestation rates in the recent past and restoration efforts are trying to keep up with the continuous degradation (Holl *et al.*, 2000) (Box 15.1).

In developed countries that experienced European settlement in the last few centuries, the reference point for habitat restoration is the habitat configuration before European arrival (Dodds *et al.*, 2008; Erskine *et al.*, 2007). Asia on the other hand has a diversity of ecosystems, both highly impacted by human activities and close to pristine state. Restoration on this continent has thus tackled a variety of systems, from abandoned farmland (Lee *et al.*, 2002) to forests and wetlands (Chua *et al.*, 2013; Nakamura *et al.*, 2014). For instance, in Japan, a meander restoration initiative was conducted in the Kushiro wetland, the largest in the country, and showed that both the natural landscape and its ecological function could be restored (Nakamura *et al.*, 2014). The Asian continent is also home to one of the most ambitious soil restoration projects in the world (Box 15.3) (Cao *et al.*, 2009a).

However, restoration of the original habitats is not always possible, particularly in the context of current global environmental changes (Harris *et al.*, 2006). Restoration can also be hampered by permanent changes in the environment at local scale and loss of resilience (Suding *et al.*, 2004).

It is extremely difficult to estimate how much of the area of degraded ecosystems has been restored, but data for some geographical areas are more readily available than for others. Dodds *et al.* (2008) offer estimates of the extents of native and restored habitats for the conterminous United States. The forests ecoregions have suffered the biggest impact of the European settlement but they have also benefitted the most from the restoration efforts. Perhaps only 1% of the native area of the Eastern temperate forests still remains intact while 146 millions of hectares have been restored. That represents around 60% of the pre-settlement area of the Eastern temperate forests (Dodds *et al.*, 2008). The Western forested mountains and the West coast marine forests have also only 5% and 3% respectively of the pristine area remaining. Restoration has been achieved for 72% of the historical area of the forested mountains and the 41% area of the historical West coast marine forests. The Great Plains ecoregion on the other hand, although having only 10% of the area remaining, has only been restored for an area of less than 5% of the pre-settlement extent.

Box 15.1: Brazilian Atlantic Forest

The Brazilian Atlantic forest is an area of more than 1.5 million km^2 inhabited by approximately 110 million people (Rodrigues *et al.*, 2009) and with high level of species endemisms (Wuethrich, 2007). The degradation of the Brazilian Atlantic forest began more than 500 years ago and only around 12% of the original area still retains natural vegetation (Ribeiro *et al.*, 2009), although highly fragmented (Oliveira *et al.*, 2004). Currently, this region is the focus of one of the biggest restoration efforts in the world, driven by actors at all levels from international (Alexander *et al.*, 2011) to local (Brancalion *et al.*, 2014; Wuethrich, 2007).

The history of the restoration of the Brazilian Atlantic forest mirrors in many ways the progress of restoration science across time (Rodrigues *et al.*, 2009). The first restoration project was triggered by the water supply crisis in Rio de Janeiro, and between 1862 and 1892 thousands of native and exotic trees were planted in the area (Rodrigues *et al.*, 2009). Ecological processes were largely ignored in these early restoration projects. Only some projects resulted in permanent forests and they required high maintenance costs (Rodrigues *et al.*, 2009). In the 1980s, the restoration evolved to planting only Brazilian tree species but not necessarily local, and using only a limited numbers of species. Thus the forests were not self-sustainable and the pioneer species were not able to ensure the conditions for the establishment of non-pioneer species (Rodrigues *et al.*, 2009). During the 1990s, with the maturation of restoration science, the Atlantic forest restoration progressed to using a wider variety of native forest species and functional groups (de Souza and Batista, 2004). In the early 2000s, the restoration goals had shifted from restoring the natural forest to restoring natural ecosystem processes and high species diversity, by accelerating natural succession. This shift towards process based restoration coincides with the progress in ecological science and literature towards a higher focus on ecosystem services (MA, 2005) and functional diversity (Díaz and Cabido, 2001). More recently, concerns with the conservation of intraspecific genetic diversity have lead to efforts of selecting local seeds and nurseries (Rodrigues *et al.*, 2009). However, and despite increased success in recent decades, success is still limited and restoration costs are high, at US$2,000 per hectare (Wuethrich, 2007). Moreover, the estimated time for the recovery of the biotic structure is hundreds of years, while for reaching the level of endemism of mature forests it is thousands of years (Liebsch *et al.*, 2008).

One of the strongest drivers of the restoration of the Atlantic Forest is the Brazilian legislative framework, even if not thoroughly implemented (Calmon *et al.*, 2011). For instance, the Brazilian Forest Act dates from 1934 (Calmon *et al.*, 2011), while the law on restoration in the state of São Paulo is grounded in scientific considerations, mentioning minimum numbers of species, seed origins, and monitoring indices for restoration actions (Rodrigues *et al.*, 2009).

Since much of the region is covered by farmland, trade-offs between the recovery of the habitat and agricultural yields are expected to occur. But the collapse in regulating ecosystem services due to habitat degradation led also to income reduction for the farmers and significant support for restoration now exists at local level (Wuethrich, 2007). One of the main negative impacts of the degradation of the Atlantic Forest is on water resources and this pressing problem unites stakeholders in support of restoration from both rural and urban areas (Brancalion *et al.*, 2014). Moreover, increases in biodiversity, cultural services and carbon sequestration in the restored areas of the Brazilian Atlantic Forest are of global importance (Tabarelli *et al.*, 2005; Alexander *et al.*, 2011; Brancalion *et al.*, 2014).

Box 15.2: Peatland restoration in Belarus

Peatlands represent at least one third of the global wetland resources and about 3% of the world land surface area (Parish *et al.*, 2008). They are characterised by the accumulation of organic matter (peat) as a result of dead and decaying plant material in a water-saturated environment. The incomplete cycling of matter in peatlands originate a positive carbon balance, hence, peatlands play an important role in trapping carbon (MA, 2005). In Europe peatlands have suffered large losses due to agricultural development and commercial extraction for fuel (Kimmel and Mander, 2010). Peatlands are now being restored in several regions. For instance, in Belarus, restoration projects have been developed with the support of the Global Environment Facility (GEF) and other organizations (Table 1). Peatland restoration can increase biodiversity, increase CO_2 sequestration and storage, reduce the risk of fire, bringing as well socioeconomic benefits to the local communities.

Table 15.1. Restored peatland in Belarus under the UNDP-GEF and other national and international programmes.
Source: Data provided by Alexander Kozulin, The National Academy of Sciences of Belarus

Peatlands by use type	Total area (ha)	Degraded (ha)	Restored (ha)
Peatlands drained for agriculture	1 085 200	250 520	0
Developed peatlands	292 400	255 600	21 333
Natural marshes	863 000	516 000	30 153
Other peat deposits	698 400	?	0
Total	2 939 000	1 022 120	51 486

Active restoration targeted at ecosystem services

Provisioning ecosystem services have been improving globally, but often this has been done at the cost of degrading regulating, supporting and cultural ecosystem services (MA, 2005). Increasing calls have been made to emphasise the restoration of degraded ecosystem functions and services, rather than only original species compositions (Choi, 2007). But restoration for ecosystem services dates back many centuries, as forest plantations have been a key management intervention in several regions (e.g., Europe) in order to provide timber and wood fuel and ensure soil protection (Farrell *et al.*, 2000). Ecosystem services can link ecology and economics (Suding, 2011) and help justify restoration costs (Birch *et al.*, 2010; Martinez-Martinez *et al.*, 2014). Despite this, in a recent literature review, ecosystem services were directly referred only in 12% of the studies analyzed and ecosystem function was mentioned as a goal in fewer than half of the studies (Rey Benayas *et al.*, 2009).

Restoration can be considered a policy option for ecosystem services such as water supply and wastewater management, prevention of disasters, carbon sequestration and climate change mitigation (Nellemann and Corcoran, 2010). Wetland restoration is a cornerstone for the supply of several major ecosystem services such as prevention of disasters (Sudmeier-Rieux *et al.*, 2006) and wastewater management (Day Jr *et al.*, 2004). Increasing the carbon sequestration capacity of ecosystems has also received significant attention, especially in the tropics (Miles and Kapos, 2008; Pichancourt *et al.*, 2014). Additionally, both water and carbon sequestration services are important by-products of restoration efforts directed at soil (Cao *et al.*, 2009a) and habitats (Rodrigues *et al.*, 2009).

Provisioning ecosystem services are much less often addressed in the restoration literature (Bullock *et al.*, 2007) as the supply of these services are often regarded as drivers of degradation (Ellis *et al.*, 2010). However, synergies between provisioning and non-provisioning services can occur, as for instance in the case of pollination and agricultural production (Dixon, 2009; Forup *et al.*, 2008). In other restoration projects, trade-offs between ecosystems services occur. For example, for short-term carbon sequestration, fast-growing species are a better choice, while long-lived and slow-growing species are better at promoting biodiversity and long-term carbon sequestration (Wuethrich, 2007). Afforestation of degraded lands can, in some instances, lead to a reduction of water yield at local level and to a reduction of agricultural output (Feng *et al.*, 2005).

There are indications that the ecosystem services provided by restored ecosystems do not match the ecosystem services provided by intact ecosystems, with the exception of provisioning services (Rey Benayas *et al.*, 2009). However, there are differences in recovery levels across ecosystem services. For instance, Dodds *et al.*, (2008) suggest that 10 years after the restoration efforts, services such as recreation, disturbance regulation and water supply returned to values almost equal to the services provided by the native ecosystems in restored land, in several biomes in the USA. In contrast, soil erosion or nutrient cycling services in restored lands are estimated at around 50% of the levels of the native ecosystems, while carbon and methane sequestration have recovered to a level below 80% of the native ecosystems value (Dodds *et al.*, 2008).

Box 15.3: Grain for Green policy in Western China

Desertification, sandstorms and floods in China have been attributed to the extensive land degradation and desertification in the west of the country, which also encompasses the upper reaches of the two largest rivers of China, the Yangtze and the Yellow river (Liu *et al.*, 2008). The Grain for Green policy has been initiated in 1999 as a pilot project and it was extended in 2002 to 25 Chinese provinces (Liu *et al.*, 2008; Xu *et al.*, 2006b). The programme was designed to afforest 15 million ha of low-yield farmland and 17 million more ha of barren lands (Feng *et al.*, 2005). Moreover, regulations establish that cultivated land on areas with slopes of more than 25° have to be terraced or restored with vegetation that will protect against erosion (Feng *et al.*, 2005). To compensate the loss of agricultural fields, the farmers receive subsidies and grains, and they keep all the profits arising from restored forests and pastures (Yan-Qiong *et al.*, 2003). The total planned investment of the Chinese Government in the Grain for Green programme is approximately US$70 billion (Liu *et al.*, 2008).

Although rising prices of agricultural products at national level have been blamed on the Grain for Green program, a study has estimated the impact on agricultural yield at only 2-3% of the national production, although at the local level the impact can be much stronger (Feng *et al.*, 2005). Environmental impacts have not been exclusively positive so far, and research suggests that afforestation with the wrong methods can have further negative environmental impacts. Although water runoff and soil erosion have been reduced in many areas (Liu *et al.*, 2008; Deng *et al.*, 2012), water shortages and further erosion has been linked to large scale afforestation in vulnerable arid and semi-arid regions (Cao *et al.*, 2009a; Sun *et al.*, 2006). Instead of planting tree seedlings, better customized restoration techniques such as planting of native steppe species, maximum water-use dwarf shrubs and even lichens are recommended in the context of arid areas (Cao *et al.*, 2009a). Biodiversity improvements are limited due to the low diversity of the planted tree species (Liu *et al.*, 2008).

Some of the lessons learnt are related to the need for a better articulation between different levels of government and for a longer timescale vision in order to maintain the current environmental gains (Xu *et al.*, 2006a). There are indications that although the local population recognizes the need for environmental rehabilitation (Cao *et al.*, 2009b), many gains would be reversed with the elimination of state subsidies (Hu *et al.*, 2006).

Active restoration of highly degraded ecosystems

Highly degraded ecosystem is any ecosystem that has been persistently and severely changed, either directly or indirectly, by a natural disaster or anthropogenic activities. Such ecosystems have been so profoundly altered to the point that they have lost partially or completely their defining characteristics and functions. This is particularly the case when tipping points have been crossed, leading to regime shifts in the systems, sometimes irreversible, which make mitigation and restoration considerably more difficult (Leadley *et al.*, 2014).

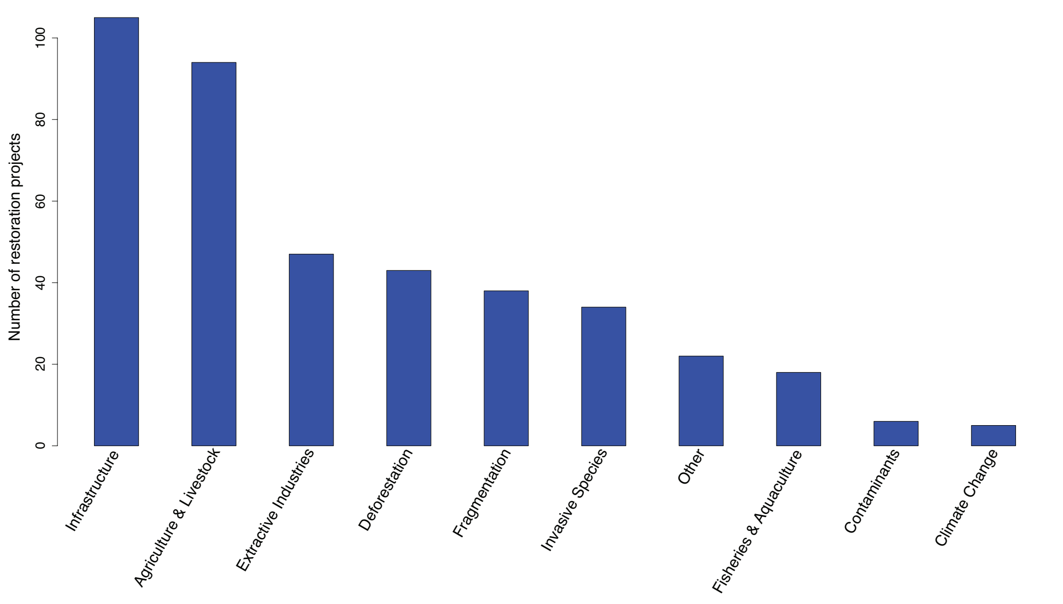


Figure 15.5. Number of restoration projects per degradation type in the Global Restoration Database (February 2014).

For example mining, the industry of extracting minerals and other geological materials from the earth, results in substantial environmental damage as it usually involves stripping of soil and vegetation, soil contamination, changes to hydrology and topography, and generates vast amounts of solid wastes (Li, 2006; Silva and Lucas, 2013; Deikumah *et al*., 2014). Surface mining typically damages more land than underground mining and former non-metalliferous mines (e.g., coal) have a higher restoration success rate than metalliferous mining sites (Li, 2006). Former mining sites are important targets of restoration (Figure 15.3) because they often show a legacy of environmental stress conditions which are hard to recover from within a reasonable time frame (e.g., extreme acidity, compaction, low nutrients, and heavy metals contamination). Former mining sites cannot be restored back to their original state. Instead the aim of the restoration is to deliver at least a functional ecosystem (e.g., a wetland), or a specific bundle of ecosystem services. One example is the restoration of former mining sites to recreational ecosystems such as lakes or parks.

Many developing countries are experiencing "mining blooms" (Deikumah *et al*., 2014), from which severe degradation results. Unfortunately, restorative action does not always follows, as the regulatory framework is often either weak or lacks enforcement. West Africa and China rely heavily on mining for socioeconomic development (Deikumah *et al*., 2014; Li, 2006). In both cases, small-scale mining is the source of much damage because such mines are difficult to trace and monitor. In China, even though mining activity is regulated since 1988, the restoration rate is very low (Li, 2006).

Another type of highly modified ecosystems, sometimes highly degraded, that could be the target of restoration activities are city ecosystems. Recently, the City Biodiversity Outlook (Elmqvist *et al*., 2013) has assessed the biodiversity and associated ecosystem services provided by city ecosystems. Important services that can be obtained by the increase in natural and green areas in the cities include recreation, local climate regulation, water infiltration and flood protection. *Ex situ* conservation of rare species could for instance be promoted in green roofs and other urban spaces (Rosenzweig 2003). Finally the development and expansion of urban and community gardens can contribute to support biodiversity, contribute to food security, and provide a wide range of ecosystem services (Cabral *et al.* 2014).

Passive restoration and rewilding on abandoned areas
Natural regeneration can be seen as the passive colonisation of abandoned land from the neighboring pool of species and the resulting secondary succession (Rey Benayas *et al*., 2007). When natural successions occur, several studies document that, with increasing time since abandonment, several ecological parameters resemble those of natural vegetation or old growth forest (Lebrija-Trejos *et al*., 2008; Lee *et al*., 2002).

Cases of natural regeneration after abandonment are documented across the globe, in most biomes (Figure 15.6). They vary in the type of activity that was performed on the land before abandonment, the time needed for regeneration, and the observed relative success of passive regeneration. Natural secondary succession is particularly successful in places where agricultural practices involved short term appropriation of the land (Bowen *et al*., 2007; Hobbs & Cramer, 2007). Nonetheless, independently of the duration of human activity, ecosystems can recover passively after abandonment, although the time to recovery may be very long (Lambin *et al*., 2003; Aide & Grau, 2004; Rey Benayas *et al*., 2007). The passive recovery of tropical ecosystems after abandonment has been studied in the dry deciduous forest of Mexico (Lebrija-Trejos *et al*., 2008), in the rain forests of Costa Rica (Finegan & Delgado, 2000), in tropical forests of Puerto Rico (Grau *et al*., 2003), and lowland forests of Brazil (Davidson *et al*., 2007). Successions on abandoned land are also observed in temperate forests, with several successful cases of passive regeneration, for both the vegetation, and the species that benefit from newly available habitat (McGrory Klyza, 2001; Lee *et al*., 2002; Deinet *et al*., 2013).

Recovery can take several decades, or even centuries, and it depends on the duration and type of activity that lead to the degradation. Plant communities may need longer time to fully recover compared to ecosystem functions and animal communities (Jones & Schmitz, 2009). Still, the resprouting ability of the vegetation after a disturbance can facilitate natural regeneration (Chazdon, 2003; Vieira & Scariot, 2006), though such ability decreases with the intensity and the duration of disturbances.

Rewilding is the passive management of ecological succession with the goal of restoring natural ecosystem processes and reducing the human control of landscapes (Pereira & Navarro, 2015). Unlike conservation practices centered on the restoration of given species or habitats, rewilding is focusing on the restoration of processes (Byers *et al*., 2006; Sandom *et al*., 2013), and the restoration of ecosystem resilience. If the land is not too degraded, once abandonment occurred, natural regeneration can passively restore the systems, with human intervention only preventing the appropriation of the land for new human activities (Clewell & McDonald, 2009). Nonetheless, if ecological filters are present, some forms of assisted restoration are needed (Navarro & Pereira, 2012). The study of pre-Holocene ecosystems can give guidelines to restore lost functions in the abandoned systems (Sandom *et al*., 2013), but should not be considered as an historical baseline describing the state to which the ecosystem must be returned to (McGrory Klyza, 2001). As a matter of fact, in many cases, such approach to restoration is not possible. The climate has considerably changed since the early Holocene, some species have been introduced, while others went (locally or globally) extinct, all of which hampers a return to pre-Holocene ecosystems (Gillson & Willis, 2004; Jackson & Hobbs, 2009).

Millions of hectares should be released from agriculture in Europe by 2030, which raises the case for rewilding on the continent (Box 15.4). The same reasoning can be applied in other regions of the world where a decrease in agricultural areas and the resulting land abandonment are projected to allow rewilding to happen (Figure 15.11). Typically, the lands released from agriculture are remote and less productive (MacDonald *et al*., 2000; Rey Benayas *et al*., 2007), hence their abandonment would have a low impact on the supply of food production. Where people cultivate a strong link with wilderness, its comeback after land abandonment does not raise concerns (McGrory Klyza, 2001), yet in areas marked by the age-old presence of traditional agricultural landscapes, and by conflicts between wildlife and human activities, opinions are divided (Bauer *et al*., 2009; Wilson, 2004).

The "inner" resilience of a degraded system, or "spatial resilience" (Cumming, 2011), will also guide the establishment of restoration goals and restoration processes. The resilience of disturbed ecosystems is dependent on the existence, or not, of ecological filters that can limit natural successions and rewilding. Those filters are determined by the "land-use legacy" of the land (Aide and Grau, 2004; Bowen *et al*., 2007; Cramer *et al*., 2008; Holl and Aide, 2011; Suding, 2011). The availability of nutrients in the soils, affected by harvesting, erosion, and fire, is an essential limiting factor for secondary successions (Hooper *et al*., 2005; Zarin *et al*., 2005; Davidson *et al*., 2007). Another ecological filter depends on the existence and proximity of remnants of natural vegetation, acting as "sources", and the potential for dispersal to the abandoned patch, or "sinks" (Bowen *et al*., 2007; Hooper *et al*., 2005; Meiners *et al*., 2007; Rey Benayas *et al*., 2008). The presence of invasive species in the vicinity of the abandoned land can also change the course of the secondary succession (Aide and Grau, 2004; Hooper *et al*., 2005; Shono *et al*., 2007), though in certain cases they can favor the establishment of shade tolerant seedlings of native species (Vieira and Scariot, 2006; Erskine *et al*., 2007). Finally, landscape dynamics rely on regimes of disturbances such as herbivory and fire and the alteration of these regimes can modify the succession trajectory. This is particularly relevant in places where sedentary agriculture has been established for centuries, replacing natural disturbance regimes to shape the landscapes as it is the case for traditional agriculture in Europe (Blondel, 2006).

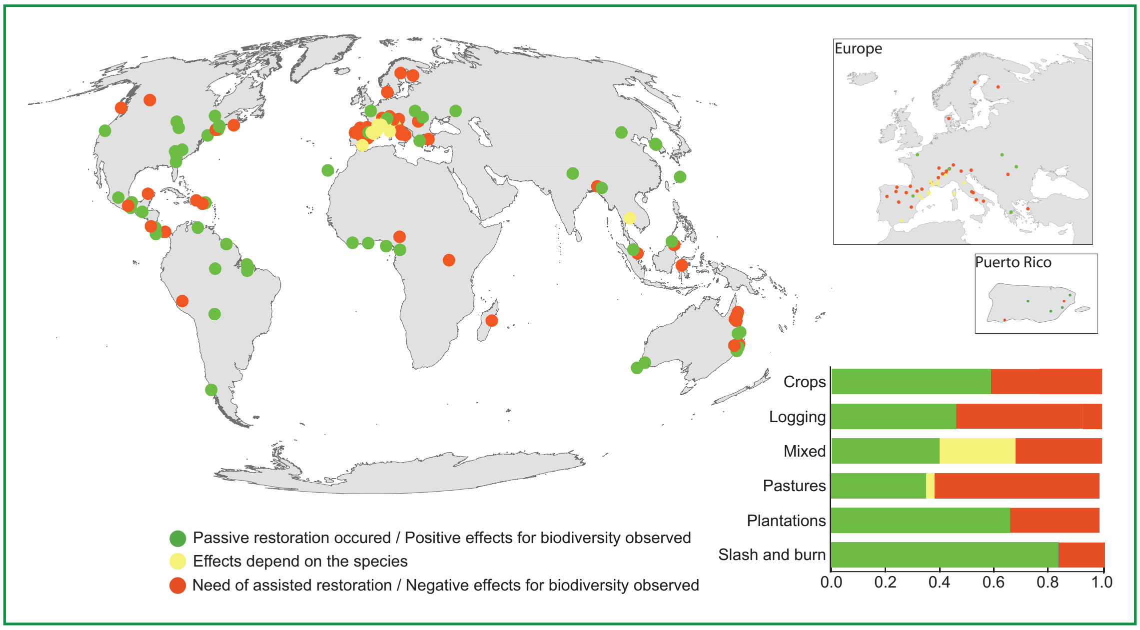


Figure 15.6. Localization of the 126 case studies of natural regeneration (Bowen et al., 2007; Davidson et al., 2007; Hooper et al., 2005; Jones and Schmitz, 2009; Lebrija-Trejos et al., 2008; Lee et al., 2002; MacDonald et al., 2000; Meiners et al., 2007; Navarro and Pereira, 2012; Rey Benayas et al., 2007; Sirami et al., 2008). The colour of the dot indicates the relative success of passive restoration identified by the authors of each study. Those relative successes are expressed separately for various type of degradation of the land: crops (n = 22), logging (n = 15), mixed activities (n = 47), pastures (n = 31), plantations (n = 3), and slash and burn agriculture (n = 6).

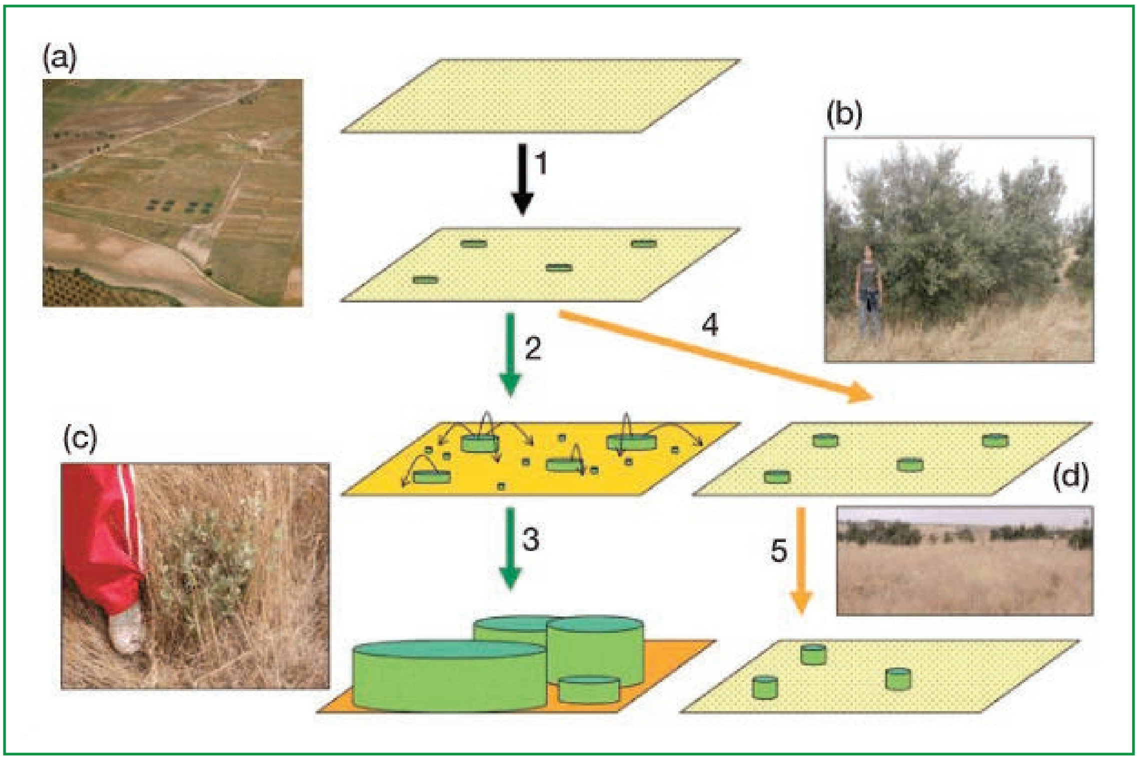


Figure 15.7. The woodland islets approach to assisted restoration. Schematic diagramme of the "woodland islets" approach, illustrated with the 15-year experimental site at La Higueruela Experimental Farm (Toledo, Spain). A denuded agricultural landscape is planted with a few small (eg 100-m²) woodland islets (1 and a). Targeted management of the islets allows the trees to establish, grow and reach sexual maturity rapidly (b). If the cropland is then abandoned, the islets can expand and export seeds (and other organisms established in them) to the surrounding land (2 and c – a holm oak seedling). The islets eventually coalesce to form closed woodland (3). Alternatively, the surrounding land remains in same or other uses (eg cultivation or pasture, d) while the islets remain as small patches of native woodland community as the trees grow taller (4). Because they are small, some islets may disappear through stochastic events (5). Source: Reprinted from Rey Benayas *et al.* (2008).

Assisted passive restoration can accelerate rewilding when ecological filters would hamper it (Shono *et al.*, 2007). The success of assisted passive restoration will depend on the presence of pioneer species in the abandoned land. In cases where patches of natural vegetation are not in the vicinity of an abandoned areas, and if the natural seed bank is depleted, "woodland islets" and/or hedgerows can be planted to assist passive restoration (Figure 15.7). The islets act as a seed bank, hence sources, for species to recolonise the abandoned patches, and can also play the role of "refuges" for dispersing animals, which can then colonise the newly available land (Rey Benayas *et al.*, 2007, 2008). This form of assisted restoration can be implemented while the land is still in use, and serve passive restoration if and once abandonment occurs (Rey Benayas and Bullock, 2012).

When the regime of natural disturbances are altered, assisted restoration can be needed. For instance, the maintenance of open habitats may require the use of prescribed burning (Driscoll *et al.*, 2010) or the restoration of populations of wild herbivores, through protection or reintroductions. However, special care should be taken to limit overgrazing and ensure that those population will be regulated in the wild by the presence of predators (Pereira & Navarro, 2015; Sandom *et al.*, 2013b). On the contrary, when forest regeneration is limited by the local populations of herbivores (Kamler *et al.*, 2010), fencing can be used to prevent the grazing of saplings of pioneer species. Similarly, firebreaks can be implemented (Shono *et al.*, 2007) to aid forest natural regeneration.

15.1.2 Projecting forward to 2020

We do not have global indicators on active restoration that can be used to project towards 2020. Still, it is likely from the recent policy commitments (see next section) and recent trends that active restoration activities will continue to increase until 2020.

For passive restoration and potentially rewilding, we can make some projections based on global land-use models. Between 2010 and 2020, millions of hectares of agricultural land may be released in several world regions, including South America, Central and West Asia, North America (Figure 15.11) (PBL, 2012). In Europe, abandonment until 2030 will potentially allow rewilding of up to 15% of habitats that have been historically converted to agriculture (Box 15.4).

Box 15.4: Agricultural abandonment and rewilding in Europe

The European landscape is marked by millennia of human pressure on the land. Over the last few decades, as market competition increased globally, agriculture became less profitable for European farmers in areas that are both less productive and harder to cultivate (Gellrich *et al.*, 2007; MacDonald *et al.*, 2000). This led to substantial rural depopulation since the mid 20th century, feeding a "circle of decline" of remote agricultural areas, only tempered by the subsidies system of the European Common Agricultural Policy (Stoate *et al.*, 2009). Between 1990 and 2000, nearly half a million hectares were converted from agriculture to (semi)-natural areas (EEA, 2012). Future scenarios predict that the aging rural population in remote areas will not be replaced, hence increasing the contraction in Europe's farmland area on semi-natural grasslands and mountain areas (Keenleyside & Tucker, 2010). Some scenarios project a further decrease of up to 15% in the total agricultural area of the EU27 by 2030 (Verburg and Overmars, 2009), consistent with projections of up to 20% loss in the area used by the main food crops in developed countries by 2050 (Balmford *et al.*, 2005). The areas projected to be abandoned are mainly located in mountain ranges, but also more generally in central Europe, Northern Portugal and Southern Scandinavia (Navarro & Pereira, 2012) (Figure. 15.8).

Rewilding aims at restoring natural ecological succession, leading to self-sustaining ecosystems and ecosystem processes (Navarro & Pereira, 2012), and emphasizes process-based conservation approaches. Most of European arable land would need 12 to 20 years to go from abandoned to (semi) natural, but some areas would require more than 40 years (Verburg & Overmars, 2009), to which another 15 to over 50 more years must be added until forest becomes the dominant cover. Moreover, the withdrawal of agriculture might leave a land vulnerable to species invasions and fire (Stoate *et al.*, 2009). These limits to passive restoration can be overcome by active measures in early post-abandonment stages, such as the localized establishment of seed banks (Rey Benayas *et al.*, 2008) or even the reinforcement or reintroduction of disturbance agents, i.e., grazers, browsers and prescribed burning (Sandom *et al.*, 2013).

A recent review (Navarro & Pereira, 2012), identified 60 bird species, 24 mammal species and 26 invertebrate species that would benefit from land abandonment and rewilding while another 101 "loser" species were identified. Europe is currently witnessing a wildlife comeback (Enserink and Vogel, 2006; Deinet *et al.*, 2013), especially of species of European megafauna, most of which were locally extinct in many regions, such as the Iberian ibex, Eurasian elk, roe deer, red deer, wild boar, golden jackal, and grey wolf (Deinet *et al.*, 2013; Navarro & Pereira, 2012). Nonetheless, land abandonment was also identified as a threat for some bird species such as the Barnacle goose, white stork, lesser kestrel, saker falcon, bearded vulture, and eastern imperial eagle (Deinet *et al.*, 2013). Still, the impacts of rewilding on farmland-associated species will be likely attenuated by their adaptation to alternative habitats and by the maintenance of habitat mosaics at regional scales (Proença & Pereira, 2013).

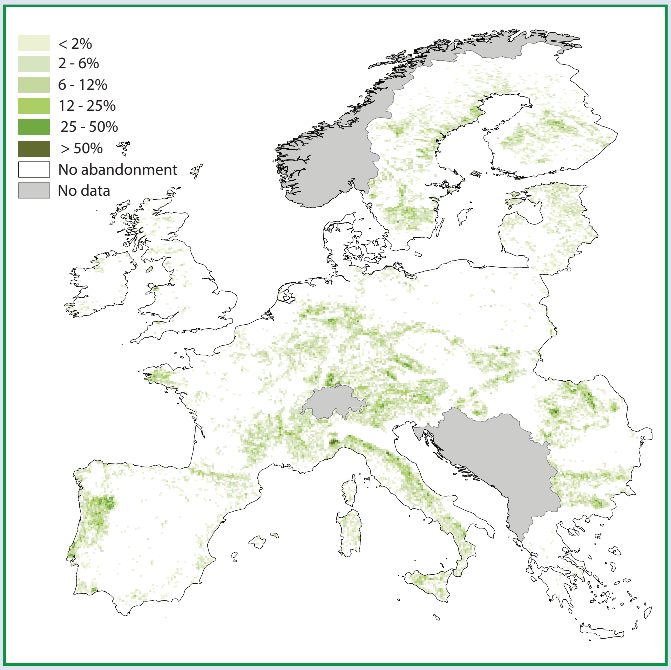


Figure 15.8. Localization of projected transitions from agriculture to forest and semi-natural habitats (i.e., land abandonment and rewilding) between 2000 and 2030 (after Verburg & Overmars, 2009 and Navarro & Pereira, 2012.). Abandonment and rewilding is expressed as a percentage of the area of each 100km² grid cell.

15.1.3 Country actions and commitments[2]

Most of the national biodiversity strategic and action plans examined contain national targets or similar commitments which are relevant to Aichi Biodiversity Target 15 (high). These are generally in line with the overall direction of Target 15. The majority of national targets refer to undertaking restoration activities (high) while approximately one third of the NBSAPs examined contain targets specifically referring to restoring 15% of degraded lands. Few of the national targets related to this goal explicitly refer to carbon stocks or climate change sequestration or mitigation (medium).

A number of countries refer to the restoration of specific habitats in their targets. For example Timor Leste has set a target related to the restoration of critical watersheds. Some countries have also specified how restoration is to be undertaken. For example in its NBSAP Belarus notes that they aim to decrease their use of forest plantations dominated by one species when undertaking restoration actions.

Given that the national targets contained in the examined NBSAPs are largely in line with Aichi Biodiversity Target 15, these commitments will help move the world community in the direction of this global objective. However, as only a few of the national targets which have been established so far are quantitative, how close these will bring us in attaining Aichi Biodiversity Target 15 is unclear.

15.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

15.2.1 Actions

In order for the target to be achieved, we propose a three-component approach. First, parties need to identify highly degraded ecosystems, and develop efforts towards restoring 15% of those highly degraded ecosystems. Ecological restoration for those areas often arises from developers' commitments taken under different environmental permits and environmental impact assessments for high impact projects such as construction of infrastructures (Cuperus *et al.*, 1999; Rundcrantz & Skärbäck, 2003). But these schemes face different implementation problems related to enforcement by governments (Reid & De Sousa, 2005), lack of procedure clarity (Tischew *et al.*, 2010) or lack of appropriate compensation leading to net losses in environmental benefits (Villarroya & Puig, 2010). The emergence of markets tools (Box 15.5) can potentially help solve some of these issues. However, such market-based mechanisms raise questions regarding effective implementation (Palmer and Filoso, 2009). Control procedures and independent evaluation bodies for market tools are regarded as necessary in order to get the best results in the context of market constraints (Bernhardt *et al.*, 2005; Tischew *et al.*, 2010).

Second, countries can also expand their programmes on active restoration of native habitats, although in many cases passive restoration or rewilding are cost-efficient alternatives. Rewilding has large potential in many regions, but requires that abandoned agricultural areas are not reallocated to other uses such as biofuel plantations. Rewilding can be accelerated by using assisted passive regeneration techniques and the appropriate management of disturbance regime, such as fire and grazing. In order to promote rewilding, countries and organizations need to recognize rewilding as a possible management strategy, and move from subsidies to maintain current agricultural practices toward incentives to promote natural succession (Merckx & Pereira, in review). When necessary, countries should envision the restoration of self-sustaining "novel" ecosystems (Hobbs *et al.*, 2006; Jackson & Hobbs, 2009), where resilience is restored, either passively or actively, but which are not identical to the native state of the ecosystem, prior to its degradation.

Footnote

[2] This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment will be further updated and refined to account for additional NBSAPs and as such these initial findings should be considered as preliminary and were relevant a level of confidence has been associated with the main statements. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets

Box 15.5: Wetland mitigation banking

Wetlands have been disappearing at an alarming rate worldwide over the last century, with perhaps as much as 50% global wetland area lost (Finlayson & D'Cruz, 2005). Clearing or drainage for agricultural development are the two most threatening human activities to wetlands, although urban development has also had an important toll (Finlayson & D'Cruz, 2005). In the USA, where approximately 2,606 km^2 of wetlands were lost between 1986 and 1997 (Dahl, 2000), various regulations and financial incentives have been developed to counteract that loss (Jenkins *et al*., 2010; Martinez-Martinez *et al*., 2014). Since 1990 a policy of No Net Loss (NNL) has been implemented under The Clean and Water Act (CWA). This regulation affords protection to aquatic resources, including wetlands, by requiring permits for impacts to resources. Permits typically follow a three step mitigation approach including avoidance, minimisation and compensation for unavoidable impacts, preferably through restoration (Kaza & BenDor, 2013).

When a development project results in unavoidable impacts to a wetland in the USA, the developer has essentially three options to meet its environmental liabilities: permitee-responsible mitigation (impact proponent delivers mitigation itself), in-lieu fees (impact proponent pays a third party to deliver compensation, typically after impacts have occurred) or mitigation banking (impact proponent buys credits from a pool of existing wetland habitats, i.e. the wetland bank). In the first case, the responsibility for compensation pends upon the impact proponent, while in the other two cases the responsibility is passed onto a third-party. Whenever a third-party is involved, there are at least two agents involved, the credit buyer (e.g., impact proponent) and the credit seller (e.g., landowner, bank).

In any case, geographically, offsetting may be delivered on-site, that is within the same watershed and corresponding to an area of at least that impacted by the development; or off-site, usually by an area greater than that impacted by the development project. The increase in area ratio is often justified by the increase in risk to meet NNL when compensating in an area outside the watershed where the impact occurred.

The regulatory demand to compensate for damages to wetlands has generated an important ecosystem services market, through wetland mitigation banking and the exchange of wetland credits or biodiversity offsets (Madsen *et al*., 2011; Kaza & BenDor, 2013). Today trades in this market generate US$3 billion in wetland and stream restoration per year (Kaza & BenDor, 2013). Similar markets for wetland restoration are now being developed in several other countries and for other habitats and species, leading to the creation of more general biodiversity offsets.

BOX 15.6: Restoration Ecology in Protected Areas as an adaptation strategy to climate change

Protected Areas (PA) play a fundamental role in national, regional, and global climate change adaptation strategies. It has long been recognized that healthy ecosystems provide a multitude of ecosystem services that support, for example, food security, clean air and water, and climate regulation (MA, 2005). They also act as buffers and reduce vulnerability to extreme events. (Colls *et al*., 2009; SCBD, 2009; Mooney *et al*., 2009; NAWPA 2012; Keenleyside *et al*., 2014). Protected areas can also contribute to climate change mitigation through their role in storing and sequestering carbon in healthy ecosystems (Sharma *et al*., 2013).

In some PA, natural, cultural, or other associated values have been compromised or lost, but through ecological restoration both within and outside the protected areas, it is possible to re-establish species, re-connect habitats, re-instate natural processes and recover cultural traditions and practices. In that order of ideas, the International Union for Conservation of Nature and other partners, published in 2012 the Guide: "Ecological Restoration for Protected Areas Principles, Guidelines and Best Practices" (Keenleyside *et al*., 2012). This document offers a guidance framework for ecological restoration and is intended to support managers and stakeholders efforts to restores natural and associated values in (an around) protected areas of all over the world in their efforts.

Based on the principles of the Guide, the Mexican National Commission for Protected Areas (CONANP), in partnership with Fondo Mexicano para la Conservación de la Naturaleza A.C. (FMCN) and with the financial support of Parks Canada Agency, worked in the Northeast and Eastern Sierra Madre region, one of the driest and most vulnerable areas of the country. Through a participatory process, with academic, local stakeholders and communities, robust adaptation measures were identified. Among these measures, on-the-ground restoration work was consistently considered one of the most important to reduce vulnerability within the PA (Keenleyside *et al*., 2014).

Following the same approach, restoration measures will be implemented by CONANP in the next five years through a GEF project: "Strengthening Management Effectiveness and Resilience of Protected Areas to Safeguard Biodiversity Threatened by Climate Change".

Restoration, along with sustainable resource management and conservation are part of an Ecosystem-Based Adaptation (EBA), which is one of the four principal concepts of the project. It recognizes that the loss of biodiversity directly influences the loss of ecosystem services that support human wellbeing, and values the role of ecosystems in providing a buffer from the impacts of climate change on human communities and infrastructure.

The project will engage local actors in activities, incentives or projects that promote good practices in restoration, connectivity and reduction of social/gender vulnerability in areas of conservation. Restoration and sustainable use strategies will be implemented in more than 4000 ha to promote resilience in natural ecosystems, productive landscapes and for human populations facing climate change.

Third, increasing the contribution of biodiversity to carbon sequestration and ecosystem services can be done through state or private sponsored passive and active afforestation programmes (Box 15.6). Restoration projects addressing carbon services are driven by global tools such as the REDD mechanism (Reducing Emissions from Deforestation and Forest Degradation) (Madsen *et al*., 2011; Kaza and BenDor, 2013) and other payments for ecosystem services schemes (Pascual *et al*., 2009), as well as national and regional governmental policies (Rodrigues *et al*., 2009). However, many challenges remain in implementing these tools. For instance, REDD deals with national governments as the main actors for preventing deforestation which raises questions regarding equity and benefits trickling down to local communities (Visseren-Hamakers *et al*., 2012). Schemes involving only governments as beneficiaries of payments can create incentives for increasing the centralization of forest governance which can threaten the progress achieved in community involvement (Phelps *et al*., 2010). Further issues are raised on the uncertainty of the estimates for avoided deforestation (Combes Motel *et al*., 2009) and the projected benefits towards biodiversity (Visseren-Hamakers *et al*., 2012). Although schemes of payments for ecosystem services have the potential to become game-changers, adoption and success is still limited (Venter and Koh, 2012). Finally, it is important to improve the monitoring of restoration activities, both to assess their success and to improve restoration approaches. In order to develop such systematic monitoring a harmonised approach such as the Essential Biodiversity Variables framework (Pereira *et al*., 2013) proposed by GEO BON (Group on Earth Observations Biodiversity Observation Network) could be used.

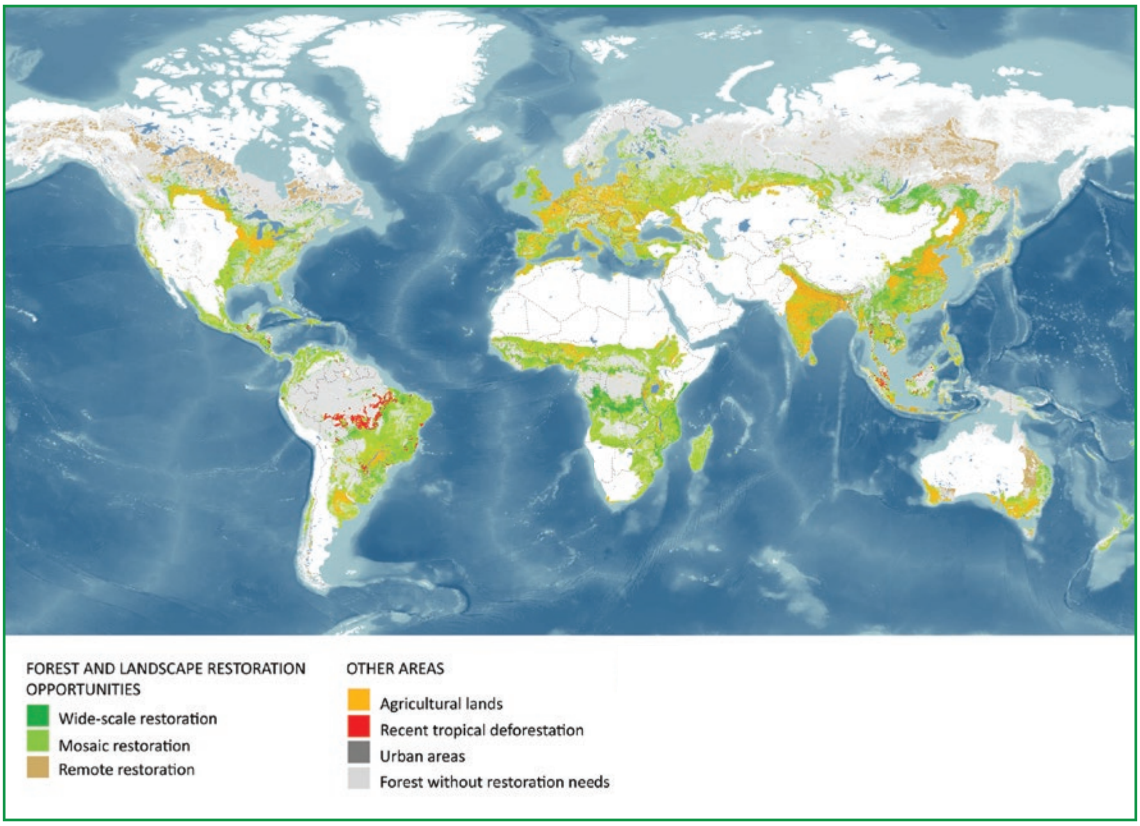


Figure 15.9. Forest and landscape restoration opportunities (Potapov et al., 2011). **The restoration potential was assessed by identifying the differences between current and potential forest extent, weighted by the current human pressure on the land (i.e., human density and land-use). Results must be interpreted at the landscape scale rather than to identify individual sites for restoration. The details of the different restoration types are given in the text.**

Recently, a global map of forest and landscape restoration opportunities was produced (Figure 15.9), distinguishing three types of restoration potential (Minnemeyer *et al.*, 2011; Potapov *et al.*, 2011). The "Wide-scale restoration" of closed forests in remote and less inhabited areas could restore 0.5 billion ha of land, for example in Canada, Scandinavia, Eastern Asia and in the Congo Basin. "Mosaic restoration", where restored forests are within a multiple-use landscape (e.g., with agroforestry, extensive agriculture), and could represent 1.5 billion ha globally, is identified as best suited for Europe, most of sub-Saharan Africa, Madagascar, Eastern Brazil, Central America, and the Eastern United States. Finally, "remote restoration", i.e., the restoration of remote and unpopulated, yet highly degraded areas, can be applied in the Northern boreal forests. The latter is described as the most difficult kind of restoration but represents an additional 0.2 billion ha of land. However, note that the degradation and deforestation of boreal systems may be exaggerated in this map due to the criteria used to define forest (e.g., forests regenerating from disturbances such as fires and pests and with a height lower than 5m are not included in the definition of "forests"). About two thirds of the restoration potential of forests is located in tropical systems, mosaic restoration being the most represented option there (Figure 15.9).

15.2.2 Costs and Cost-benefit analysis

All restoration works incur costs (de Groot *et al.*, 2013; Schleupner and Schneider, 2013). Direct costs, which include the costs of restoration itself, management and protection and indirect costs, such as missed opportunity costs (i.e., benefits missed from an alternative land use). Different types of restoration will have different costs: active restoration will be in principle more expensive than passive restoration, but it will also deliver the benefits in a shorter timeframe; certain habitats and ecosystems will be more time and resource consuming to be restored. Overall restoration costs will mostly depend on the level of degradation of the target habitat or ecosystem. More severely degraded ecosystems will require higher economic investments but on the other hand, these will be cases where most benefits from restoration will result (Jellinek *et al.*, 2014; Rey Benayas *et al.*, 2009).

A review of 316 case studies, reporting costs or benefits of ecological restoration across major world biomes, found that the majority of the restoration projects provided net benefits and should be considered as high-yielding investments (Figure 15.10, de Groot *et al*., 2013). A meta-analysis of 89 restoration assessments found that biodiversity and regulating services improved by 25% and 44%, respectively (Rey Benayas *et al*., 2009). There are also social benefits associated with restoration such as the range and quality of jobs and livelihoods (Aronson *et al*., 2010). In the EU alone about 110,000 jobs are directly related to restoration of ecosystems and/or green infrastructure (Smith *et al*., 2013). In some instances, passive restoration can be economically more efficient than active restoration, or than passive restoration with extra measures such as fencing and fire suppression (Birch *et al*., 2010).

However it is important to recognize two limitations of the assessments of the cost-benefit ratio of restoration projects. First, many of the benefits are associated with public ecosystem services that are not usually marketed, and therefore, for which landowners receive limited private benefit. Second, the uncertainty on ecological restoration is not fully accounted for in some studies. Therefore, from the point of view of individual investors, the cost-benefit ratio may not be always as high as the studies report.

A first assessment of the resources required to restore ecosystems (Target 14 and 15) estimated that initial investments between US$30 billion and US$300 billion would be necessary. Wetlands and coral reefs are the ecosystems whose restoration was estimated to be more costly in terms of initial investment. In addition, the annual recurrent costs associated with restoration of forests were estimated to be between US$6 billion and US$65 billion. These include, for example, costs of seeds, nursery establishment and planting, site protecting, and weeding (CBD, 2012).

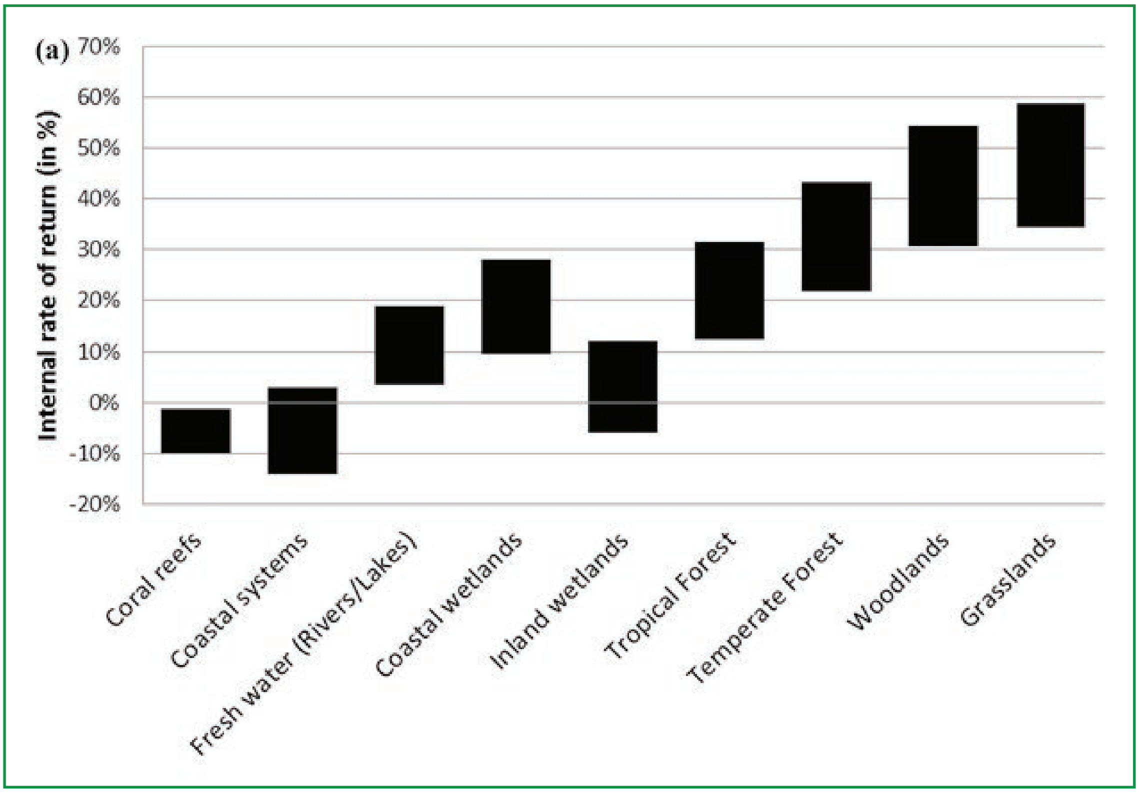


Figure 15.10. Benefit-cost ratios of restoration (bars, range of values: bottom of bars, worst-case scenario [analysis conducted at 100% of highest restoration cost reported, 30% of benefits, and social discount rate 8%]; top of bars, best-case scenario [analysis conducted at 75% of highest restoration cost reported and 75% of benefits and social discount rate of −2%]) across nine major biomes on the basis of 316 case studies over 20 years with a management cost component of up to 5% of the capital cost. Source: de Groot et al. (2013).

15.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

A fundamental question in restoration research is the link between biodiversity and ecosystem services (Swift *et al*., 2004; Turner *et al*., 2007). Understanding this link will allow to design restoration efforts and tools that are more precise and more effective. The debate has been so far divided. Some studies suggest that restoration projects focused on ecosystem services or on species could achieve each other's goals (Rey Benayas *et al*., 2009). For instance, Hector and Bagchi (2007) show that considering a large number of ecosystem functions in grassland systems means including large species richness to support those functions. This calls for increasing restoration efficiency by targeting multiple ecosystem services rather than addressing them one at a time (Wainger *et al*., 2010). Nonetheless, other studies indicate that species diversity and ecosystem services restoration do not necessarily go hand in hand, as it was the case in some wetlands in North America (Hoeltje and Cole, 2009) or in calcareous grasslands in the United Kingdom (Haines-Young *et al*., 2006).

Different taxonomic groups can benefit from different restoration paths. In Europe, a large number of species previously suffering from the reduction of their natural habitat and/or from human persecution are expected to benefit from land abandonment and rewilding (see Box 15.4). Species associated with farmland habitats are thought to suffer from land abandonment and secondary successions, although maintaining open habitat considerably limits those negative trends. From a restoration perspective, the consequences of rewilding for biodiversity will depend on the restoration baseline (i.e., "pre" or "post" abandonment in Queiroz *et al*., 2014), but also on the taxa and the scale (Navarro and Pereira, 2012; Plieninger *et al*., 2014). For instance, in the mosaic of old growth vegetation and secondary successions on abandoned land, biodiversity can be very high, with the occurrence of species thriving on different types of habitats.

Large scale projects such as the restoration of the Atlantic forest or the Grain for Green programme have the potential of creating large patches of habitat with high benefits for biodiversity but such an outcome seems more likely in restoration projects focused on both habitat and ecosystem services, as it is the case with the Atlantic forest (Rodrigues *et al*., 2009). Based on the current progress, Grain for Green seems to lead to few positive biodiversity outcomes as actions are directed almost exclusively at soil improvement (Cao *et al*., 2009a).

15.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

There is a high potential for restoration on the globe, with 2.2 billion ha of land identified as suitable (Figure 15.9). Yet, projecting active restoration projects for the next decades is an extremely difficult task. Nonetheless, assumptions on passive restoration can be made when looking into projections of decrease in agricultural areas, and increase of natural areas. For example, projections to 2050, using the four pathways scenarios of the Rio+20 (PBL, 2012), predict an increase in natural areas in some regions of the world. Such projected increase in natural areas will vary depending on the region, and could globally reach 146 million ha (with the Lifestyle scenario) to over 315 million ha (with the Global Technology scenario). The increase in natural vegetation could be partially explained by a decrease in agricultural areas (Figure 15.11 A to D), which could potentially be experienced in all world regions, depending on the scenario, with the exception of the Middle East and Sub-Saharan Africa. Such decrease in human activity on the land will thus give an opportunity for passive or active restoration.

Land restoration, whether passive or active, is also important when investigating CO_2 sequestration and mitigation of climate change. For example, though anthropogenic emissions are responsible for the largest part of the total emissions, in developing countries, forest degradation and deforestation can be the main source of carbon release (van der Werf *et al*., 2009). Nonetheless, estimating the carbon emissions triggered by deforestation and land degradation is subject to several uncertainties and approximations (Harris *et al*., 2012; van der Werf *et al*., 2009; Zarin, 2012). Van der Werf *et al.* (2009) estimated the carbon emissions resulting from deforestation and forest degradation to represent 12% of the total emissions (adding peatland degradation brings the value to 15%), while Harris *et al.* (2012) estimated that tropical deforestation between 2000 and 2005 was responsible for 7-14% of the global emissions.

Proportionally to the area that they occupy globally, wetlands such as mangroves and peatlands can be considered as the most carbon rich land covers of the planet (Murdiyarso *et al*., 2010; Donato *et al*., 2011; Hugron *et al*., 2013). While peatlands occupy 3% of the global area they store 15-30% of its carbon (Hugron *et al*., 2013). As a result, the degradation of these areas can have tremendous impacts in terms of carbon emissions. 2 Gtons of CO_2/yr are emitted by the 50 Mha of degraded peatlands (Joosten, 2009). The deforestation of tropical mangroves, which represent 0.7% of the global tropical forested area accounts for 10% of the global carbon emissions (Donato *et al*., 2011). Such estimates of the increase in carbon emissions resulting from the degradation of those ecosystems call for active measures to both hamper their deforestation, and restore the damaged ecosystems. If 15% of the degraded peatlands (i.e., 7.5 Mha) were to be restored, this would lead to an average reduction in the carbon emissions of 150 Mtons of CO_2 eq/yr globally (Joosten, 2009), though this estimates depends on the location of the restored peatland, tropical systems storing much more carbon than boreal systems for example. Drained peatland will need active restoration (Box 15.2), including the reintroduction of keystone species (Hugron *et al*., 2013). Their restoration could also increase the effectiveness of the REDD program (van der Werf *et al*., 2009). Additionally, the restoration of mangroves can mitigate the impacts of the projected sea level rise (Donato *et al*., 2011).

Natural regeneration and active restoration programmes will directly impact the biodiversity supported by ecosystems that are currently damaged. Over 80% of the species of mammals, amphibians and birds assessed by IUCN are threatened due to habitat loss (Pereira *et al*., 2012). Those species are for example located in Southeast Asia, Madagascar, North Africa and the Middle-East and would directly benefit from the restoration, either active or passive, of their natural habitat. Interestingly, some of these areas have been identified as having potential for "mosaic restoration" (*sensu* Minnemeyer *et al*., 2011, in Figure 15.9). The restoration of corridors could also increase the connectivity of natural habitats (McRae *et al*., 2012), and benefit species with either large home ranges or large dispersal abilities.

Another important consideration is the fact that scenarios of climate change could challenge the resilience of ecosystems, particularly when looking into the occurrence of fires. As a matter of fact, climate determines the probability of fire occurrence by influencing both the ecosystem productivity, hence the fuel quantity, and the temperature, hence the probability of ignition (Pausas and Ribeiro, 2013). In Mediterranean systems, though a decrease in precipitations would increase the ignition probability, it would also lower the systems' productivity, thus the fuel load, which in turn could reduce the occurrence of fires and their intensity (Batllori *et al*., 2013). Such elements are crucial when investigating the potential for restoring ecosystem's resilience.

Figure 15.11. Observed and projected variation in agricultural area per world region. A) to D): Relative cumulative change in natural area compared to 2010. The baseline data run from 1970 to 2010. The projections are obtained with the Rio+20 scenarios and baseline projections (PBL, 2012)**. The shaded areas represent the minimum and maximum values projected, regardless of the scenario. Agricultural areas include agricultural land, extensive grasslands and biofuel.**

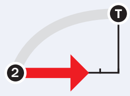
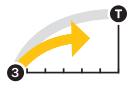
15.5 UNCERTAINTIES

There are several sources of uncertainties in assessing the success of restoration efforts, some of them are related to data-availability while others are conceptual. Firstly, restoration assessments lack a credible assessment of the global degraded area. As mentioned previously, the estimates of the degraded global area are very wide ranging. Moreover, estimates of the global or even regional restored areas are also lacking. Several specialized restoration databases exist at different geographical extents, mainly in North America and Europe, mirroring to some extent the geographical bias in restoration projects and the available resources (Benayas *et al.*, 2009; Figure 15.4). For example, river restoration in the United States benefits from several databases which collect projects details and results (Bernhardt *et al.*, 2005; Kondolf *et al.*, 2007). Europe also places high importance on river restoration projects (Hansen *et al.*, 1998). But many areas of the world remain poorly covered, and the need remains for a large-scale understanding of the successes and failures of restoration projects. The Global Restoration Network of the Society for Ecological Restoration (LeFevour *et al.*, 2007) aims to gather data about restoration projects at global level. So far it has registered 136 unique projects (in February 2014), a modest number considering the estimated thousands of restoration projects running in the world in any year (Suding, 2011). Besides the limitations regarding available data, further issues are raised about the scale of restoration projects and the definition of what represents a valid entry in a useful restoration database (Jenkinson *et al.*, 2006). Gathering these data for a significant number of projects at the global scale can help tremendously in answering some of the most important questions in restoration and evaluate the achievement of Target 15.

Further limitations reside in the feasibility of restoration, which must be assessed. For instance, when tipping points have been crossed, as it can be the case with severely degraded ecosystems, the return to a natural and self-sustaining state can be long and costly, if even possible (Leadley *et al.*, 2010). Additionally, active restoration must be done bearing in mind that different actors will benefit from a different bundle of services supplied by different ecosystems (e.g., provisioning services on agricultural land *versus* regulating services on a restored land), and that the beneficiaries of services are not necessarily located where those services are "produced" (MA, 2005).

From a conceptual point of view, one of the most important sources of uncertainty is related to defining degradation itself. For instance, in Europe farmland abandonment can be seen as an opportunity to reverse at low costs the deforestation that took place during the long agricultural history of the continent (Navarro & Pereira, 2012; Rey Benayas & Bullock, 2012; Sandom *et al.*, 2013). But others see the abandonment itself as the source of degradation. Acute differences can be seen as specific to certain controversial cases but finding a broadly accepted definition of degradation is very difficult even in less controversial contexts. For example, many see agriculture in general as a source of degradation, whereas for agriculture experts degradation occurs only when ecosystems are not able to supply provisioning ecosystem services at certain levels (Lal, 2002; Tilman *et al.*, 2002; Montgomery, 2007). Differences in definitions of degradation can also show up in matters of afforestation with monocultures or exotic plantations for ecosystem services restoration (Cao *et al.*, 2009a) in contrast with restoration projects for forest biodiversity (Wuethrich, 2007). Although not a solution for all restoration uncertainties, the conclusion of this assessment is that restoration with an emphasis on both ecosystem services and biodiversity yields the highest benefits for ecosystems and society.

15.6 DASHBOARD – PROGRESS TOWARDS TARGET

Target Element	Status	Comment	Confidence
Ecosystem resilience and the contribution of biodiversity to carbon stocks have been enhanced through conservation and restoration		Despite restoration and conservation efforts, there is still a net loss of forests, a major global carbon stock	Low
At least 15% of degraded ecosystems are restored, contributing to climate change mitigation and adaptation, and to combating desertification		Many restoration activities under way, but hard to assess whether they will restore 15% of degraded areas	Low

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TARGET 16: NAGOYA PROTOCOL

By 2015, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force and operational, consistent with national legislation.

PREFACE

The Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from Their Utilization (hereafter Nagoya Protocol) to the Convention on Biological Diversity (CBD) was adopted in Nagoya, Japan on 29 October 2010. It was adopted at the same time as the Strategic Plan for Biodiversity 2011-2020. Aichi Biodiversity Target 16 directly relates to the Nagoya Protocol and represents a commitment by Parties to the CBD to have the Protocol in force and operational, consistent with national legislation, by 2015.

The Nagoya Protocol is a legally binding, supplementary agreement to the CBD to advance the implementation of its third objective: the fair and equitable sharing of benefits arising out of the utilization of genetic resources. It sets out core obligations for its contracting Parties to take measures in relation to access to genetic resources and associated traditional knowledge, benefit-sharing and compliance.

By increasing legal certainty, clarity and transparency in access and benefit-sharing, the Nagoya Protocol contributes to building trust between users of genetic resources and traditional knowledge associated with genetic resources and those who provide them. As such, the Nagoya Protocol has the potential to open up new opportunities for the fair and equitable sharing of the benefits arising from the utilization of genetic resources.

The Nagoya Protocol also contributes to the other two objectives of the CBD relating to conservation and sustainable use of biodiversity, as the benefits that accrue from the utilization of genetic resources can act as an incentive to conserve and sustainably use biodiversity and to further enhance the contribution of biological diversity to sustainable development and human wellbeing. Article 9 of the Protocol also provides that Parties shall encourage users and providers to direct benefits arising from the utilization of genetic resource towards the conservation of biodiversity and the sustainable use of its components. In addition, by setting-out clear provisions on access to traditional knowledge associated with genetic resources, the Nagoya Protocol will assist in strengthening the ability of indigenous and local communities to benefit from the use of their knowledge, innovations and practices.

16.1 ARE WE ON TRACK TO ACHIEVE THE 2015 TARGET?

Aichi Biodiversity Target 16 provides that "*by 2015, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force and operational, consistent with national legislation*." Therefore, the target has two elements: the entry into force of the Nagoya Protocol and the Protocol being operational, consistent with national legislation.

The Nagoya Protocol enters into force on 12 October 2014 following its ratification by 51 Parties[1] to the Convention on Biological Diversity (see Figure 1.1). Thus this component of the target has been met in advance of the deadline set. This opens up new opportunities for the fair and equitable sharing of the benefits arising from the utilization of genetic resources.

The second element of Aichi Biodiversity Target 16 refers to the Nagoya Protocol being "operational, consistent with national legislation". The meaning and implications of this sentence has not been defined or established. However, at a minimum, having the Nagoya Protocol operational, consistent with national legislation, could be understood as requiring Parties to the Nagoya Protocol to have implementing domestic legislative, administrative or policy measures and institutional structures in place. Having this information available on the Access and Benefit-sharing Clearing-House will also contribute to the operationalization of the Protocol. Many countries are making progress to make the Protocol operational at the national level.

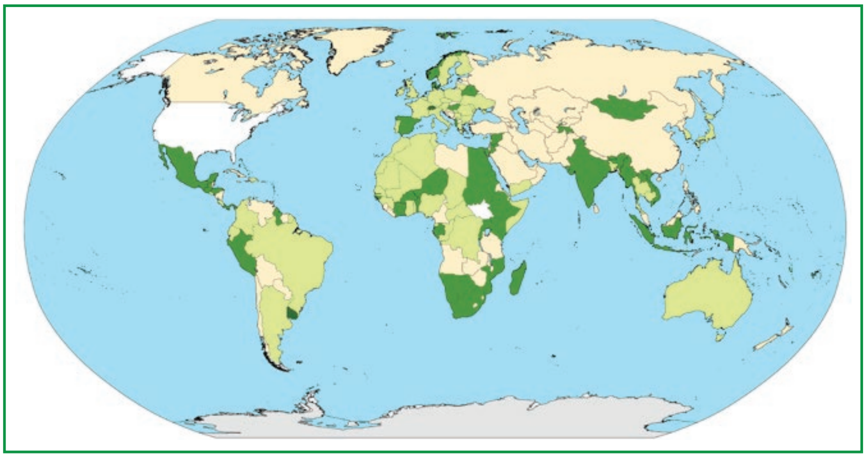


Figure 16.1. The Parties to the Convention on Biological Diversity that had ratified, approved or acceded to the Protocol by 14 July 2014, thereby enabling it to enter into force (dark green) or have signed it (light green). Source: CBD.

Box 16.1: The Access and Benefit-sharing (ABS) Clearing-House

Article 14 of the Nagoya Protocol establishes an ABS Clearing-House as part of the clearing-house mechanism of the Convention. The Secretariat of the CBD is currently implementing the pilot phase of ABS Clearing-House. Once fully operational, the ABS Clearing-House will serve as a means for Parties to share information related to access and benefit-sharing, including relevant legislative, administrative and policy measures, national focal points and competent national authorities, and permits or their equivalents, among other things. The ABS Clearing-House will play a key role in enhancing legal certainty and transparency and in promoting compliance. Having a fully functional ABS Clearing-House by the time of entry into force of the Protocol that includes existing national information is essential for making the Protocol operational, and will significantly contribute towards achieving the second element of this Aichi Biodiversity Target. The ABS Clearing-House can be found at https://absch.cbd.int/

16.2 WHAT NEEDS TO BE DONE TO ACHIEVE THIS AICHI TARGET?

16.2.1 Actions

The fair and equitable sharing of the benefits arising out of the utilization of genetic resources is one of the three objectives of the Convention on Biological Diversity. The Nagoya Protocol provides a transparent legal framework for the effective implementation of this objective. The Protocol covers genetic resources and traditional knowledge associated with genetic resources, as well as the benefits arising from their utilization by setting out core obligations for its contracting Parties to take measures in relation to access, benefit-sharing and compliance.

The Protocol will enter into force on 12 October 2014 and is expected to be fully operational by the target date of 2015, opening up new opportunities for benefits from biodiversity and ecosystem services to be more widely and fairly shared. Against this background, possible key actions, many of which are already underway in many countries, to accelerate progress towards this target include:

- For countries that have not yet done so, to deposit the instrument of ratification, acceptance, approval or accession of the Nagoya Protocol as soon as possible to ensure full participation in the Protocol;
- Putting in place, by 2015, legislative, administrative or policy measures and institutional structures for implementing the Nagoya Protocol; This may include revising existing legislative, administrative or policy measures or developing new measures to meet the obligations arising from the Protocol, as well as establishing a national focal point, one or more competent national authorities, one or more checkpoints, and actively sharing information through theABS Clearing-House;
- Reporting and sharing information, as required, through the ABS Clearing-House;
- Undertaking awareness raising and capacity-building activities, and engaging with indigenous and local communities and the private sector, to accompany the formal implementation of the Protocol.

Box 16.2: Building and developing capacity for the Nagoya Protocol

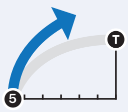
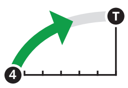
The Secretariat of the CBD is currently undertaking a number of regional and sub-regional workshops to support ratification and implementation of the Protocol, in collaboration with partners, including the International Treaty on Plant Genetic Resources for Food and Agriculture (ITPGRFA) and the Commission on Plant Genetic Resources for Food and Agriculture (CGRFA). There are also a good number of ABS capacity development activities and tools that have been developed under various initiatives and by numerous partners and organizations, including the United Nations Environmental Programme (UNEP), the United Nations Development Programme (UNDP), the International Union for Conservation of Nature (IUCN), and the ABS Capacity Development Initiative, among others, and with support of the Global Environment Facility (GEF) and other donors. These initiatives have significantly improved the awareness and capacities of Parties, indigenous and local communities and relevant stakeholders on related access and benefit-sharing issues and for achieving Target 16.

16.2.2 Costs and cost-benefit analyses

Precise information on cost-benefit analysis of implementing access and benefit-sharing and the Nagoya Protocol is not available. The financial costs related to the attainment of Target 16 are largely related to what is required to support the ratification process and to develop/revise relevant national measures and establishing institutional structures. The High Level Panel on Resource Mobilization has estimated that the resources required for meeting this Target range between US$55 million and US$313 million. This is an estimate of one-off investments over 2013 to 2020 without any estimates for recurring costs. These figures are for 197 countries.

Through enhancing legal certainty and transparency in access and benefit-sharing, the Nagoya Protocol has the potential to generate significant monetary and non-monetary benefits to be shared upon mutually agreed terms with the providers of the genetic resources and traditional knowledge associated with genetic resources.

16.3 DASHBOARD - PROGRESS TOWARDS THE TARGET

Element	Current status	Comments	Confidence
The Nagoya Protocol is in force		The Nagoya Protocol will enter into force on 12 October 2014, ahead of the deadline set.	High
The Nagoya Protocol is operational, consistent with national legislation		Given progress that has been made, it is likely that the Nagoya Protocol will be operational by 2015 in those countries that have ratified it	High

Author: Secretariat of the Convention on Biological Diversity

16.4 REFERENCES

Convention on Biological Diversity (2014). Status of Signature, and ratification, acceptance, approval or accession. https://www.cbd.int/abs/nagoya-protocol/signatories/default.shtml

ABS Clearing-House. https://absch.cbd.int/

Footnote

[1] *As of July 2014 the following Parties have now ratified or acceded to the landmark treaty: Albania, Belarus, Benin, Bhutan, Botswana, Burkina Faso, Burundi, Comoros, Côte D'Ivoire, Denmark, Egypt, Ethiopia, European Union, Fiji, Gabon, Gambia, Guatemala, Guinea Bissau, Guyana, Honduras, Hungary, India, Indonesia, Jordan, Kenya, Lao People's Democratic Republic, Madagascar, Mauritius, Mexico, the Federated States of Micronesia, Mongolia, Mozambique, Myanmar, Namibia, Niger, Norway, Panama, Peru, Rwanda, Samoa, the Seychelles, South Africa, Spain, Sudan, Switzerland, the Syrian Arab Republic, Tajikistan, Uganda, Uruguay, Vanuatu, and Vietnam*

TARGET 17: NATIONAL BIODIVERSITY STRATEGIES AND ACTION PLANS

By 2015 each Party has developed, adopted as a policy instrument, and has commenced implementing an effective, participatory and updated national biodiversity strategy and action plan.

PREFACE

Article 6a of the Convention on Biological Diversity (CBD) establishes "national strategies, plans or programmes for the conservation and sustainable use of biological diversity" or "adapt(ations) of existing strategies, plans or programmes which shall reflect, inter alia, the measures set out in this Convention relevant to the Contracting Party concerned" as the primary mechanism for achieving the three objectives of the Convention at the national level. Similarly, Article 6b recognizes that successful implementation of the national biodiversity strategy or equivalent would require "integrat(ion), as far as possible and as appropriate, of conservation and sustainable use of biological diversity into relevant sectoral or cross-sectoral plans, programmes and policies". Thus the cumulative impact of the implementation of all National Biodiversity Strategies and Action Plans (NBSAPs) and equivalent instruments represents the combined global effort to reverse the loss of biodiversity worldwide.

A wide-ranging assessment of NBSAPs completed in 2010 by the United Nations University Institute of Advanced Studies (Prip *et al*. 2010) noted that the great majority of Parties to the Convention had prepared an NBSAP – an achievement in itself, and that a significant number had revised their NBSAPs. It noted that second generation NBSAPs were generally better prepared, more focused, and more based on mainstreaming and on self-reliance than were the first NBSAPs. The study showed that most countries had invited the participation of most stakeholders, had created coordination structures to oversee implementation (although these did not always function well); and, importantly, that there was a trend towards greater political ownership at higher levels, such as at the level of Cabinet of Ministers or the Parliament, rather than only by the ministry responsible for the environment.

The UNU study concluded however that: "*the overall impact of NBSAPs on the driving forces of biodiversity loss continues to be limited*". Similarly, the third edition of Global Biodiversity Outlook[1] concluded that the 2010 Biodiversity Target of a significant reduction in the rate of biodiversity loss had been missed, in part, because of the limited ability of biodiversity policies to address the underlying causes of biodiversity loss. Underscoring the importance of NBSAPs in addressing biodiversity issues at the national level, the UNU study noted that few countries were using other tools for national biodiversity planning, and emphasized that getting the process of NBSAP development right is crucial to implementation.

Now, almost four years later, and almost halfway through the timeline adopted for achievement of the Strategic Plan for Biodiversity 2011-2020, this note attempts to assess whether the current round of NBSAPs under preparation is likely to be more effective in facilitating the desired biodiversity outcomes.

The wording of Target 17 has six interlinked components relating to NBSAPs – (i) "*developed*", (ii) "*updated*", (iii) "*adopted as a policy instrument*", (iv) "*has commenced implementing*", (v) "*effective*", and (vi) "*participatory*". Thus a full assessment of progress towards Target 17 requires the analysis of all six components.

The Conference of the Parties to the Convention has adopted several key decisions, among them Decision IX/8 which documents and offers as guidance from the Parties' collective wisdom on what NBSAPs should consist of, and how they should be prepared, in view to assure that they be effective instruments for implementation of the Convention and the Strategic Plan for Biodiversity 2011-2020. Key elements from this decision are outlined in Appendix 1.

Footnote

[1] *www.cbd.int/gbo3*

In decision X/2 (para (c)), the Conference of the Parties "*urged*" Parties to "*review, and as appropriate update and revise, their national biodiversity strategies and action plans, in line with the Strategic Plan and the guidance adopted in decision IX/8, including by integrating their national targets into their national biodiversity strategies and action plans, adopted as a policy instrument, and report thereon to the Conference of the Parties at its eleventh or twelfth meeting*".

This note uses a preliminary analysis of the six elements of Aichi Biodiversity Target 17, and elements from Decisions IX/8 and X/2 as reflected in the 26 NBSAPs submitted after COP 10[2] in order to assess progress to date in the achievement of Aichi Biodiversity Target 17.

17.1 ARE WE ON TRACK TO ACHIEVE THE 2015 TARGET?

Target 17 is one of the three Aichi Biodiversity Targets with a 2015 deadline. After a relatively slow start in the period immediately after the adoption of the Strategic Plan for Biodiversity 2011-2020 in October 2010, the majority of Parties are currently in the process of revising their NBSAPs, or equivalent biodiversity planning documents (see 17.1.1. Status and trends, below). The majority of Parties are reporting (informally) that they hope to complete their NBSAP revision process before the twelfth meeting of the Conference of the Parties (COP-12) in October 2014.

17.1.1 Status and trends

The analysis below assesses the degree to which the revised NBSAPs submitted to the Secretariat since COP-10 fulfill the 6 components of Target 17. As the Conference of the Parties has not defined the exact meaning of each of these, which could be subject to much debate, we provide an indicator for each of them for the purposes of this analysis, noting that the analysis is not comprehensive and alternatives could be used.

"Developed" – The number of Parties to the Convention that have developed an NBSAP in response to the obligation under Article 6 of the Convention.

Since 1993, 179 Parties (92%) have developed an NBSAP while 15 Parties have yet to submit their first NBSAP. Of the 179 Parties that have prepared NBSAPs, 45 have revised them at least once (Table 17.1).

Table 17.1: Status and Trends of NBSAP development (as of August 2014)

Status of NBSAP Development n=194				
Parties that have developed at least one NBSAP	Parties that have not developed an NBSAP	Parties that have revised NBSAP at least once	Parties that currently have NBSAP targets whose timelines[3] extend to 2014 or beyond.	Parties with NBSAPs adopted since 2010
179	15	45	57	29

"Updated" – *Indicator used in this analysis:* The number of Parties that have undertaken a revision of their NBSAP in the light of the Strategic Plan for Biodiversity 2011-2020.

Since the adoption of the Strategic Plan for Biodiversity 2011-2020 in Nagoya, Japan in October 2010, 26[4]Parties have submitted National Biodiversity Strategy and Action Plans (NBSAPs) to the Secretariat. Of these 26, fifteen indicate that they have taken the Strategic Plan for Biodiversity 2011-2020 into consideration during development, whilst the other nine Parties do not make this explicit and mostly do not appear to have taken the Strategic Plan into consideration. Some of these are NBSAPs developed before COP-10 but which were only approved and submitted after COP-10.

Footnotes

[2] *While this paper is based on the analysis of these 26 "revised" NBSAPs submitted after Cop 10, it is important to note that many Parties already had NBSAPs and targets with timelines extending beyond 2010 before the Strategic Plan was adopted that year. The document http://www.cbd.int/nbsap/targets/default.shtml an updated version of that originally presented as an information document to WGRI 3 lists these targets and notes that although these targets have not been set with the Strategic Plan for Biodiversity 2011-2020 in mind, they nevertheless contribute to its implementation (including to that of many of the Aichi Biodiversity Targets)*

[3] *Includes pre- and post-2010 NBSAPs*

[4] *Australia, Belarus, Belgium, Cameroon, Colombia, Democratic People's Republic of Korea, Dominica, Dominican Republic, El Salvador, Estonia, European Union, Finland, France, Ireland, Italy, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor-Leste, Tuvalu, United Kingdom, and Venezuela*

According to Parties' self-reporting on progress, 87 Parties are likely to have submitted revised NBSAPs to the Secretariat by the end of 2014. A more detailed breakdown of the status is provided in Table 17.2. The majority of these NBSAPs will be revisions/updates of prior NBSAPs. However, for a few Parties[5], this will be their first NBSAP submitted to the Secretariat. It is likely, given that an additional 57 Parties are currently working on their NBSAPs, albeit with no foreseen completion date, that the majority of Parties will have submitted their NBSAPs to the Secretariat, by the end of 2015.

Table 17.2: Status of post 2010 NBSAP revision (as of August 2014)

Status of NBSAP revision n=194						
NBSAP submitted	NBSAP completed awaiting approval	NBSAP completion expected prior to COP-12	NBSAP completion expected by the end of 2014	NBSAP in progress with completion date unknown	NBSAP update not started	No information available
29	13	40	17	16	16	36

A more detailed picture of progress with revision and updating of NBSAPs has been derived from information collected at the "*Global workshop on reviewing progress and building capacity for the NBSAP revision process*"[6], where NBSAP Coordinators were asked to self-assess their progress against a standard series of steps. The results of this assessment can be seen in Figure 17.1.

"Adopted as a policy instrument" – The NBSAP is given a status that enables the mainstreaming of biodiversity into all sectors and into cross-sectoral decision making. *Indicator used in this analysis:* The number of NBSAPs adopted at a whole-of-government level, i.e.: by the Parliament; Council of Ministers, Cabinet or equivalent; or Head of state or government.

Both the text of Target 17 and that of Decision X/2 request that Parties adopt their revised NBSAPs as a policy instrument. The intent is to enable NBSAPs to become whole of government policies thus facilitating the mainstreaming of biodiversity efforts across all sectors, and in particular, into cross-sectoral planning and decision making. Based on the information available, of 26 NBSAPs submitted to date, 10 have been adopted as whole of government instruments. For example Spain's NBSAP has been adopted as a Royal Decree, the NBSAPs of Japan, Myanmar and Tuvalu were adopted/endorsed by Cabinet and Belarus's NBSAP was approved by the Council of Ministers. In three countries the NBSAPs have been adopted as instruments applying to the environment sector while in three they have not been adopted as a policy instrument. In the remaining 10 countries there is insufficient information to place them in any of these categories (Table 17.3).

Table 17.3: Status of updated NBSAPs as policy instruments (as of April 2014)

Policy instrument n=26			
Adopted as instrument relevant to whole of government	Adopted as instrument relevant to environment sector	Not adopted as policy instrument	Not enough information
10	3	3	10

"Has commenced implementing" – *Indicator used in this analysis:* The number of countries reporting in their fifth national reports that they have begun to undertake activities identified in the updated NBSAP.

Many of the Parties that have revised their NBSAPs (56%) report having a formal coordination structure, or a working group for NBSAP related tasks, composed of different stakeholders. The role of these coordination structures vary. For example the committees/working groups of Ireland, Japan and Timor-Leste were/are responsible for reviewing/updating, monitoring and for overseeing implementation.

Information on implementation is also provided through the Parties fifth national reports to the Convention on Biological Diversity. All seven countries that had submitted, by early April 2014, both an updated NBSAP and the fifth national report[7], provide evidence that implementation of the updated NBSAP has commenced (Table 17.4).

Footnotes

[5] *Dominican Republic, Italy, Malta, Myanmar, Serbia, Timor-Leste, Tuvalu*

[6] *http://www.cbd.int/doc/?meeting=WSNBSAP-RPCB-01*

[7] *Belgium, Burundi, Colombia, Japan, Myanmar, Nepal, Spain*

Table 17.4: Status of implementation of updated NBSAPs as evidenced through fifth national report (as of April 2014)

Implementation n=7	
Fifth national report providing evidence that updated NBSAP is being implemented	Fifth national report not providing evidence that updated NBSAPs is being implemented
7	0

"Effective" - Whether an NBSAP is effective or not will depend on many variables and can be assessed in a myriad of different ways. For the purposes of this preliminary analysis, we have used COP guidance on NBSAPs as a benchmark taking "effective" to mean that NBSAPs contain national targets, and a monitoring system is in place to track progress.

Table 17.5a provides a summary assessment of the 26 revised NBSAPs' in relation to national targets. Table 17.6 provides a summary assessment of the revised NBSAPs' in relation to key elements of Decision IX/8.

By April 2014, 21 Parties had set national targets in their revised NBSAPs, and a further two Parties have sent sets of targets in advance of finalizing their NBSAP. Four Parties had not set clear targets[8]. Of the Parties which had set targets, eight had clearly linked their targets to the Aichi Biodiversity Targets. A majority, 21, of the 26 Parties that have submitted NBSAPs either have a monitoring system in place or plan to develop one as part of their further planning and implementation activities (Table 17.5a).

Table 17.5a: Effectiveness of updated NBSAPs (as of 10 April 2014)

Effectiveness of NBSAP n=26					
Updated NBSAPs containing national targets		Updated NBSAPs clearly linking national targets to Aichi Biodiversity Targets	Updated NBSAPs containing indicators[9]		NBSAP supported (or plan to be) by monitoring system
Yes	No	8	Yes	No	21
22	4		10	10	

Table 17.6: Assessment of 23 revised NBSAPs in relation to key elements of Decision IX/8

Key Elements of Decision IX/8 paragraph 8[10] N=23[11]	Yes	Plan to/or have a related target	Not enough evidence
NBSAPs are action-driven	18	4[12]	1
Actions Prioritized	13	0	10[13]
Mainstreaming Biodiversity			
National development/economic plan	13		
Poverty eradication/ development cooperation	3		
Sectoral plans	14		
Subnational	10		
National Sustainable Development Strategy	7	0	14, 2 n/a
Gender	2	0	21[14]
Resource mobilization plans	6	11	6
Ecosystem approach	14	0	9
CEPA strategy	10	8	5
Include indicators	10	0	13
Assessment of previous NBSAP done	12	0	5, 6 n/a[15]

Footnotes

[8] *Note that there is some subjectivity as to what is referred to/classified as a target.*

[9] *Six of these do not provide enough information to determine.*

[10] *Most of the six components analyzed in this document also form part of Decision IX/8 but have not been included here as they have been summarized in the other tables in throughout the text.*

[11] *Although there are 26 officially submitted revised NBSAPs, to date only 23 have been analyzed for these elements. This table will be updated when the analysis becomes available.*

[12] *These four NBSAPs are strategy documents pending the elaboration of action plans.*

[13] *Four out of these do not have action plans*

[14] *Two make a mention in the principles of the NBSAP.*

[15] *Six of these are first NBSAPs*

"Participatory" – The participatory nature of an NBSAP can be assessed, at least in part, by which stakeholders participated in its development/revision and how they participated.

Of the 26 Parties that have submitted a revised NBSAP, 12 formed an inter-ministerial committee that led or overlooked the revision process (Table 17.5). In three countries this committee has been assigned with revision, implementation and monitoring of the NBSAP. The remaining nine Parties do not provide enough information in their NBSAPs to make an assessment of the "participation" component (Table 17.5b).

Table 17.5b: Number of Parties having established inter-ministerial committees (as of April 2014)

Inter-ministerial committee n=26				
Revision/ preparation of NBSAP	Implementation	Monitoring	All three roles	Insufficient information
12	3	6	3	9

Collectively, the revised NBSAPs (that provide this information) have involved 15 different ministries in their NBSAP revision processes[16]. These range from those most often called upon: agriculture, forestry, fisheries, and tourism, to higher level ministries such as those responsible for development planning, finance and/or the economy, to the more rare participants such as infrastructure and public works, health, social affairs and sports. In most of the nine countries that have provided information on this, the government ministries involved were part of the inter-ministerial committee that led or looked over the process, and hence their participation went beyond mere consultation (Table 17.5c)

Table 17.5c: Number of Parties having involved stakeholders in the NBSAP (as of April 2014).

n=25	Government Ministries	ILCs	Civil Society/ NGOs	Private Sector	Academia Research
Consulted	9	6	12	6	7
Also formed part of committee	9	1	3	1	3

Footnote

[16] *List of ministries: Finance, Development planning/Economy, Agriculture, Forestry, Fisheries, Tourism, Infrastructure/Public works/Transport, Education, Culture, Social affairs/Welfare, Health, Sports, Trade and Industry, Science and Technology, Energy/Mining*

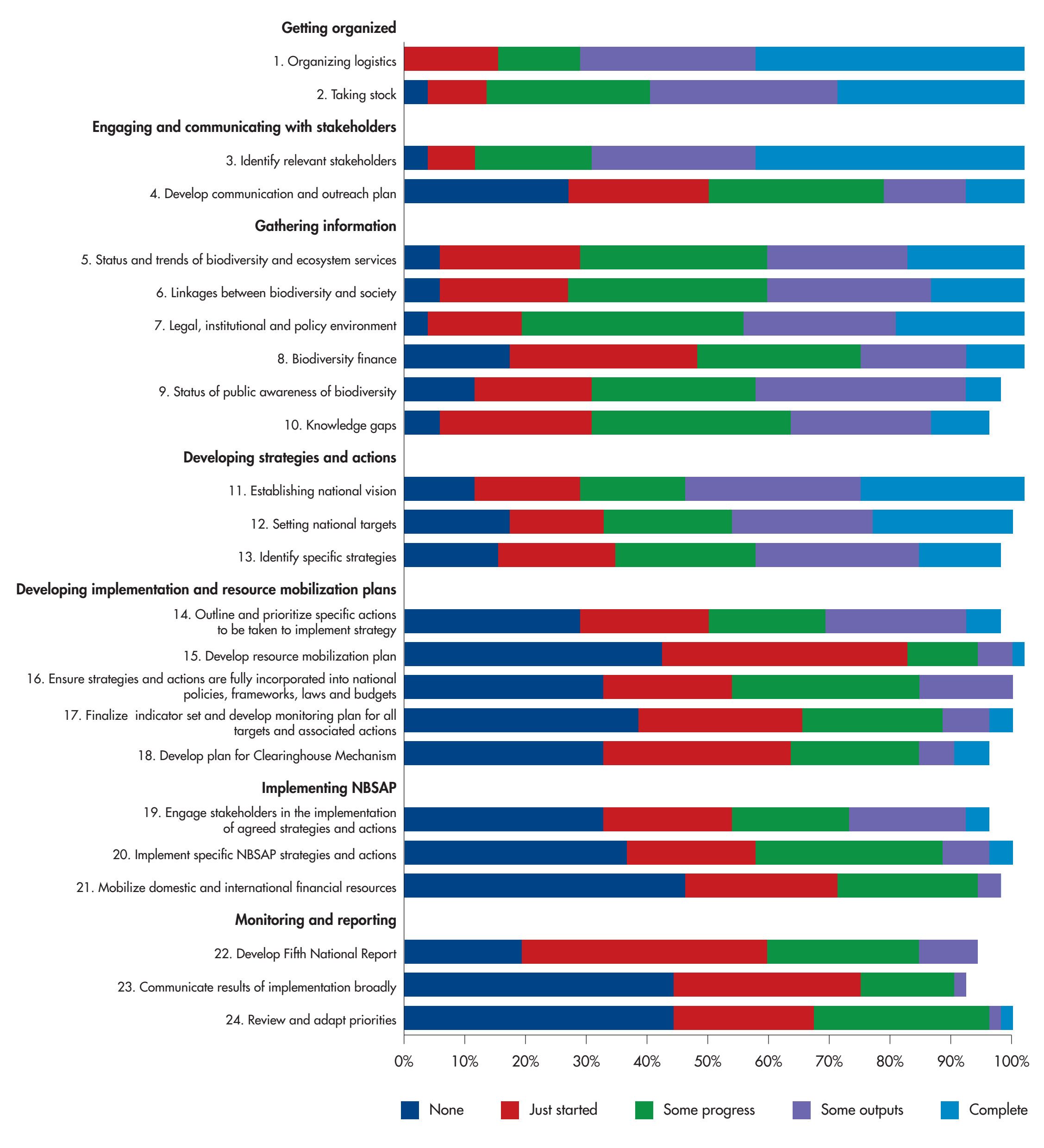


Figure 17.1: Actions towards the implementation of NBSAPs. Source: CBD.

17.1.2 Projecting forward to 2020

Whether or not Parties will collectively achieve Target 17 depends on the status of NBSAP revision in the 34 Parties for which we do not yet have information, and on whether the deadline for this target is interpreted to be the beginning or end of 2015. If we exclude the 34 Parties for which we do not have information, more than half of the remaining Parties expect to have submitted their NBSAP by COP-12 (Table 17.2). If the deadline for the target is interpreted to be the end of 2015, it is quite likely that the 57 Parties that have already started their NBSAP processes will have completed the revision between one and two years after having started for various reasons including;

- Most Parties are revising an existing NBSAP and thus have some experience with the process;
- Twenty one regional workshops for capacity building of NBSAP revision were convened since CoP 10, with support from the Japan Biodiversity Fund, in which more than 700 government officials from around 170 countries participated; this is also expected to have had a positive effect on the process of NBSAP revision

Historically, between 1996 and 2010, the mean time of GEF eligible Parties between approval of GEF finance for NBSAP development and final submission of the NBSAP to SCBD has been 3.17 years.[17] However, submission of revised NBSAPs is likely to be considerably higher in the period before December 2015 because:

- The GEF Implementing Agencies have greater experience in the GEF proposal development and approval procedures;
- In GEF-5, the majority of GEF-eligible Parties have had proposals approved in the time period between 2011-2012, with >90% having GEF proposals approved by March 2014.

Thus the period between 2015 and 2020 will focus on implementation, review, reporting and adjustments as appropriate. Historically, implementation of NBSAPs has proven to be challenging, especially for developing countries. Although developing countries are eligible for funding from the financial mechanism of the Convention, the Global Environment Facility, the System for Transparent Allocation of Resources (STAR) allocation, plus the requirement to demonstrate "global biodiversity benefits" arising from use of GEF funds, do not always correlate well with national resource needs and priorities. Moreover, resource mobilization needs of Parties to fully implement their revised NBSAPs will be considerably greater than in the past, given the broader scope of areas to be addressed.

On the other hand, where countries successfully mainstream biodiversity across sectoral and cross-sectoral policy areas significant positive biodiversity outcomes can be achieved with limited resources and overall savings can be made through enhanced policy coherence. Therefore the chances of successful and comprehensive NBSAP implementation are greatest in those countries that have followed the guidance arising from the target formulation and supplementary technical information.

17.1.3 Country actions and commitments

Country actions to achieve Target 17 can be documented by the completed or ongoing NBSAP updating, and by the completion of a number of elements of this process as detailed in Figure 17.1. The commitment and likelihood of making NBSAPs transformative instruments is evidenced by the degree to which the six "quality" components contained in the wording of Target 17, and additional guidance from decisions IX/8 and X/2 are taken into account in the NBSAP preparation/updating process.

With only 26 revised NBSAPs submitted to date, it is too early to make any global statements about the level to which these qualitative benchmarks have been achieved collectively by Parties. Looking only at the information available from these NBSAPs, and the seven National Reports that have been submitted by Parties that have also submitted revised NBSAPs, the analysis presented above indicates that components *i* (developed) and *iv* (started implementing) are well on their way to being fulfilled. Component *v* (effective) is being fulfilled in large part, although national targets are not being linked to Aichi Biodiversity Targets by the majority. Components *iii* (adopted as a policy instrument), *vi* (participatory), and *ii* (updated) could be cause for some concern as revised NBSAPs do not seem to be being adopted at a whole of government level by most Parties (although information from 10 of the 26 Parties is needed to confirm this). Similarly, with regards to component *vi* (participation) although the inclusion of an array of ministries in NBSAP committees is very promising, the apparent lack of medium and long term coordination structures that would follow the NBSAP through to implementation and monitoring is less so (again, more information would be required to confirm this). The numbers of Parties that report having involved or consulted other stakeholders in their NBSAP process is consistently very low across stakeholders. Finally, with regards to component *ii* (updated), as stated above under 1b, more than half of the Parties for which we have information on the status of their NBSAPs expect to have completed their revision process by October 2014. Ninety percent (90%) of Parties are expected to have completed by the end of 2015.

Footnote

[17] *Based on a random sample of countries for which GEF approval dates and NBSAP submission dates are known.*

17.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

17.2.1 Actions

In order to achieve Aichi Biodiversity Target 17 Parties need to continue (or in some cases start) to engage in a process to revise their NBSAPs in line with the Strategic Plan for Biodiversity 2011-2020.

Among other things, as outlined in Decisions IX/8 and X/2, this includes:

- Setting national targets within the flexible framework of the Strategic Plan, with corresponding indicators and monitoring mechanisms
- Establishing NBSAP revision committees and/or national coordination structures to oversee the revision and implementation process
- Consulting with a full range of governmental and non-governmental stake- and rights-holders
- Creating a mainstreaming strategy
- Creating communication, education and public awareness strategy
- Creating strategy for the (further) development of the Clearing-house Mechanism
- Identifying scientific and technical needs for implementation
- Creating a resource mobilization strategy
- Adopting the NBSAP as a whole of government policy instrument
- Creating and starting to implement a clearly costed, assigned and scheduled action plan.

From the CBD Secretariat, GEF Implementing Agency and supporting partners perspective, as more and more Parties complete their revision processes and submit completed NBSAPs to the Secretariat, capacity building and technical support can be focused on those Parties who are struggling to complete their NBSAPs, taking the six components of Target 17 into consideration, before the 2015 deadline or who have prepared NBSAPs that do not meet the quality criteria implied in the target and underlying guidance. Through capacity building activities and other communications Parties should be reminded of the importance of the six components of Target 17 and should ensure that they take them into account in the ongoing NBSAP revision as well as its implementation.

Footnotes

[18] *Six "quality" components contained in the wording of Target 17, and also included in prior decisions (IX/8 and X/2).*

[19] *http://www.cbd.int/financial/doc/id501-financial-planning-early-results-en.pdf*

17.2.2 Costs and Cost-benefit analysis

National Biodiversity Strategies and Action Plans are key instruments that set priorities, program implementation activities and provide the basis for monitoring of progress and communicating results. Having well-prepared NBSAPs implies being in possession of a tangible strategy document and of a realistic and costed action plan resulting from an appropriate process that ensures the support and participation of a large range of stakeholders and society at large. The updating of NBSAPs is therefore one of the enabling activities required for the achievement of the range of targets set at national level. The benefits of updating the NBSAPs in accordance with the guidance provided by the Conference of the Parties will therefore outweigh by far the investment necessary for the revision process.

The GEF-5 Biodiversity Strategy document [18] states that "*[Eligible] Countries will be able to access the focal area set-aside funds (FAS) to implement enabling activities for an amount up to $500,000 on an expedited basis. Amounts greater than that will be provided from a country's national allocation*." and that "*Enabling activity support could be provided for revising National Biodiversity Strategies and Action Plans (NBSAPs) in line with the CBD's new strategic plan to be adopted at COP-10, national reporting, and implementation of guidance related to the Clearing House Mechanism (CHM).*" Average GEF grant size allocated to date is just over US$200,000, with a range of US$175,000-900,000. Average co-finance is around US$130,000, with a range from US$40,000-4,300,000. Informal feedback from Parties indicates that many consider the average grant of around US$200,000 to be inadequate for a thorough NBSAP revision. Similarly, it is likely that the ranges of costs for full NBSAP implementation will greatly exceed both the eligibility criteria and total allocation of national STAR allocations for biodiversity under GEF-6. Thus it is likely that a funding gap will exist between costs of fully implementing revised NBSAPs and existing "*traditional funding mechanisms*" and that the full range of "*innovative financial mechanisms*" will need to be leveraged if this funding gap is to be bridged.

Four of the 26 revised NBSAPs submitted to the Secretariat have included a fully-costed Action Plan, or a Resource Mobilization Plan. Estimates range from US$18.1 mn. (Dominica), US$26.5 mn. (Suriname) through US$325mn. (Bangladesh) to just over US$1bn. (Spain) but the sample size is too small to provide any realistic guide to total costs of NBSAP implementation globally. The Secretariat has prepared an Information Digest on "Early Experience of Considering Finance in the Revised/Updated National Biodiversity Strategies and Action Plans" [19] from which some indication of the gap between costs and resource availability can be estimated.

In the (current) absence of any possibility to provide a complete "bottom-up" costing for NBSAP implementation, an alternative picture might be provided via a "top-down" costing to estimate the gap. The *Report of the High-level Panel on Global Assessment of Resources for Implementing the Strategic Plan for Biodiversity 2011-2020* [20], prepared for COP-11 as an Information Document, estimated the range of potential global costs of achieving each of the global Aichi Biodiversity Targets. For Target 17 cost estimates ranged 10-fold from US$114 million to US$1.1 billion (see Table 17.7 below). For comparison, the resource requirements for implementing the 20 Aichi Biodiversity Targets are estimated at between US$150 billion and US$440 billion per year.

Table 17.7: Cost estimates for achieving the Aichi Biodiversity Targets based on the Report of the High-level Panel

Target	Investment needs (US$million)	Recurrent expenditure per annum (US$million)	Average annual expenditure (2013–2020) (US$million)	Other Aichi Targets impacted by the Target	Other Policy objectives linked to the Target
Target 17 NBSAPs	114–1,100	110–560	50–170	All Targets	Cross-cutting

17.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

Given that it has taken four years for the majority of Parties to complete their NBSAP revisions, mostly because of limited capacity and a reliance on external funding, the timely achievement of biodiversity outcomes as specified in national biodiversity targets by 2020 may be challenging. On the other hand, implementation of biodiversity measures continued even during the period of NBSAP updating and these measures would be expected to contribute to the achievement of targets formulated as part of the updating process. Moreover, the cross-sectoral implementation of NBSAPs is expected to result in a significant reorientation of existing revenue streams towards less biodiversity harmful development pathways, particularly where biodiversity considerations are effectively mainstreamed into other sectoral and cross-sectoral plans and programmes. Nevertheless, resource availability is dependent upon the development of the overall global financial and economic situation, which is difficult to predict more than five years in advance. Thus it is likely that significant improvements on the integration of the biodiversity agenda will be achieved. However, these may not take effect quickly enough with regard to positive outcomes for biodiversity to reduce pressures on biodiversity in the short term, and the rate of loss of biodiversity will continue to be determined primarily by land use change from natural to anthropogenically-influenced landscapes less compatible with the majority of species long-term survival needs.

17.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

While the past lack of mainstreaming biodiversity considerations into broader development strategies and policies has been a key factor that led to the failure to achieve the 2010 biodiversity target there is now evidence that this is changing gradually. The ongoing debate in the context of the development of the Sustainable Development Goals illustrates the connectedness of societal challenges and the need for solutions that take into account the range of issues being addressed by societies in a holistic way. There is increasing understanding that pursuing individual agendas without coordination leads to policy incoherence and brings unnecessary costs to taxpayers.

A major challenge for biodiversity has always been the fact that biodiversity loss does not appear in the balance sheets. This is gradually changing with countries introducing indices that go beyond the traditional gross domestic product calculations and with a significant research agenda on environmental economic accounts that would enable to reflect the values on non-provisioning ecosystem services and to assess environmental impacts of developments beyond physical parameters.

Footnote

[20] *http://www.cbd.int/doc/meetings/cop/cop-11/information/cop-11-inf-20-en.pdf*

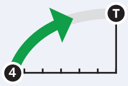
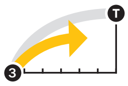
The development and implementation of effective NBSAPs is an ongoing learning process, both for countries individually as they improve upon previous strategies as well as by taking into account the experiences from other countries as they are being shared through the Convention process. It also needs to be borne in mind that the outcomes of activities undertaken in the recent past may not become visible until much later and that capacity and institution building are foundations for the ability of countries to act in future but do not have immediate effects on biodiversity. However, over a period of 30-35 years these actions, taken now, will bear fruit and are expected to have significant positive efforts for biodiversity.

17.5 UNCERTAINTIES AND DATA REQUIREMENTS/GAPS

NBSAPs, as physical documents, are one of the most prominent and easily accessible parts of the overall activities of Parties under the umbrella of the CBD. However, in many Parties, and especially in developing countries, a lack of resources and spatially-explicit biodiversity information severely limits the ability of Parties to bring to bear biodiversity concerns in national land use planning. Whilst this uncertainty about what biodiversity a country has, where it is, and what is happening to it persists, the impact of NBSAPs will continue to lag behind what is needed to reverse current trends in biodiversity loss.

The generation and implementation of effective NBSAPs depends on champions in each country that coordinate and pursue the agenda. In addition to resources and technical capacities this also requires a political climate that is supportive of such an agenda and that pursues the best interests of the people.

17.6 DASHBOARD – PROGRESS TOWARDS TARGET 17

Element	Current Status	Comments	Confidence
Submission of NBSAPs to Secretariat by (end of) 2015		For those Parties for which information is available, about 40% are expected to have completed their NBSAP by October 2014 and about 90% by the end of 2015	Medium
NBSAPs adopted as effective policy instrument		The adequacy of available updated NBSAPs in terms of following COP guidance is variable	High
NBSAPs are being implemented		The degree of implementation of updated NBSAPs is variable	High

Author: Secretariat of the Convention on Biological Diversity

17.7 REFERENCES

Harrop, S. R. (2012) Living In Harmony With Nature'? Outcomes of the 2010 Nagoya Conference of the Convention on Biological Diversity. *Journal of Environmental Law*, **vol.23**, no. 1, pp. 117-128

Prip, C; Gross, T; Johnston, S; Vierros, M (2010) *Biodiversity Planning: an assessment of national biodiversity strategies and action plans*. United Nations University Institute of Advanced Studies, Yokohama, Japan.

APPENDIX 1

Main Elements for COP Decision IX/8 paragraph 8[21]

8. COP "… *urges* Parties in developing, implementing and revising their national and, where appropriate, regional, biodiversity strategies and action plans, and equivalent instruments, in implementing the three objectives of the Convention, to:

Meeting the three objectives of the Convention:

(a) Ensure that NBSAPs are **action-driven, practical and prioritized**, and provide an effective and up-to-date national framework for the implementation of the Convention;

(b) Ensure that NBSAPs take into account the principles in the Rio Declaration on Environment and Development

(c) Emphasize the **integration of the three objectives of the Convention into relevant sectoral or cross-sectoral plans**, programmes and policies;

(d) Promote the **mainstreaming of gender considerations**;

(e) Promote synergies between activities to implement the Convention and **poverty eradication**;

(f) Identify **priority actions at national or regional level**, including **strategic actions** to achieve the three objectives of the Convention;

(g) Develop a **resource mobilization plans** in support of priority activities;

Components of biodiversity strategies and action plans

(h) Take into account the **ecosystem approach**;

(i) Highlight the contribution of **biodiversity and ecosystem services**, to poverty eradication, national development and human well being, as well as the economic, social, cultural, and other values of biodiversity

(j) Identify the **main threats to biodiversity**, including direct and indirect drivers of biodiversity change, and include **actions for addressing the identified threats**;

(k) As appropriate, **establish national**, or **where applicable, sub-national, targets**, to support the implementation of NBSAPs,;

Support processes

(l) Include and implement **national capacity-development plans** for the implementation of NBSAPs, making use of the outcomes of national capacity self-assessments;

(m) Engage **indigenous and local communities, and all relevant sectors and stakeholders**

(n) Respect, preserve and maintain **traditional knowledge**, innovations and practices;

(o) Establish or strengthen **national institutional arrangements** for the promotion, coordination and monitoring of the implementation of the NBSAPs,

(p) Develop and implement a **communication strategy** for the national biodiversity strategy and action plan;

(q) Address **existing planning processes in order to mainstream biodiversity** concerns in other national strategies, including, in particular, poverty eradication strategies, national strategies for the Millennium Development Goals, sustainable development strategies, and strategies to adapt to climate change and combat desertification, as well as sectoral strategies, and ensure that NBSAPs are implemented in coordination with these other strategies;

(r) Make use of or develop, as appropriate, **regional, sub-regional** or **sub-national networks** to support implementation of the Convention;

(s) Promote and support **local action for the implementation of NBSAPs**;

Monitoring and review

(t) Establish national mechanisms including **indicators**, as appropriate, and promote regional cooperation to **monitor implementation** of NBSAPs and progress towards national targets

(u) **Review NBSAPs** to identify successes, constraints and impediments to implementation, and identify ways and means of addressing such constraints and impediments, including revision of the strategies where necessary;

(v) Make available through the **Convention's clearing-house mechanism NBSAPs**, including periodic revisions, and where applicable, reports on implementation, case studies of good practice, and lessons learned;"

Footnote

[21] *This is an abbreviated version of the Decision. The full text can be accessed at: http://www.cbd.int/nbsap/guidance.shtml*

TARGET 18: TRADITIONAL KNOWLEDGE RESPECTED

By 2020, the traditional knowledge, innovations and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are respected, subject to national legislation and relevant international obligations, and fully integrated and reflected in the implementation of the Convention with the full and effective participation of indigenous and local communities, at all relevant levels.

PREFACE

Traditional knowledge refers to the knowledge, innovations and practices of indigenous and local communities around the world, usually associated with the natural environment. Developed from experience gained over time and adapted to the local culture and environment, traditional knowledge is passed from generation to generation. It tends to be collectively owned and may take the form of stories, songs, folklore, proverbs, cultural values, beliefs, rituals, community laws, local language, techniques and innovations. The practical applications of traditional knowledge reflect the complex worldviews and social systems of particular indigenous and local community cultures. Traditional knowledge is often of a practical nature, particularly in such fields as agriculture including forestry, water management, animal husbandry, fisheries, gathering, hunting and trapping, health, and environmental management in general.

Aichi Biodiversity Target 18 is regarded as both a cross-cutting issue and as an essential element of the "enabling" cluster of targets, which will assist achieving all other Targets. Indigenous and local communities are therefore recognized as key partners in the implementation of the Convention, the Strategic Plan for Biodiversity 2011-2020 and in planning and revision processes such as the development of national biodiversity strategies and action plans and their implementation. By encouraging the effective participation of the indigenous and local communities in its work, the Convention promotes and facilitates the use of traditional knowledge and customary sustainable use for the goals of the Convention.

18.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

18.1.1 Status and trends

Target 18 is extremely complex to measure and information is variable across countries and communities and frequently not easily accessible. To assess status and trends globally the following headline indicators have therefore been agreed as proxies:[1]

- Trends of linguistic diversity and numbers of speakers of indigenous languages;
- Trends in land-use change and land tenure in the traditional territories of indigenous and local communities;
- Trends in the practice of traditional occupations;
- Trends in which traditional knowledge and practices are respected through their full integration, safeguards and the full and effective participation of indigenous and local communities in the national implementation of the Strategic Plan.

A dataset is being advanced to determine trends over time only for the first of these headline indicators, that on linguistic diversity. Even for linguistic diversity, however, considerable uncertainty remains, primarily due to a lack of reliable data that is geographically and chronologically comparable. Advancing information and data on the indicators of traditional occupations and land change and tenure is under discussion with relevant international organizations including the International Labor Organization, the Office of the High Commissioner for Human Rights and the International Land Coalition, the United Nations Permanent Forum on Indigenous Issues, and the Working Group on Indicators of the International Indigenous Forum on Biodiversity. These organizations are also considering these indicators under the framework of the implementation of the United Nations Declaration on the Rights of Indigenous Peoples. The fourth indicator regarding the integration of traditional knowledge and participation of indigenous and local communities will be considered in the analysis of the fifth and future national reports.

Footnote

[1] *Decision XI/3 (http://www.cbd.int/decision/cop/default.shtml?id=13164)*

Language Indicator

Trends of linguistic diversity and numbers of speakers of indigenous languages are useful proxy indicators for measuring trends in traditional knowledge for two main reasons. Firstly, local and indigenous languages are a primary vehicle for the transfer of traditional knowledge (Larsen, Turner, and Brooks 2012). Just as one's native tongue is learned from parents, grandparents, and other elder family and community members, so too are the spirituality and cultural traditions, technical skills, and environmental expertise that traditional knowledge encompasses. Although traditional knowledge need not be expressed solely in traditional languages and concepts can survive translation into dominant languages or newly developing languages, many traditions, technical skills and related expertise are embedded in particular languages in ways that are not easily translatable. Secondly, general social changes that affect indigenous peoples and local communities around the world threaten both the diverse languages of the world and the wealth of ecological knowledge accumulated over centuries and millennia. For example, young Piaroa of Venezuela having greater competency in Spanish were also shown to lack skills in identifying and naming traditionally important plant species (Zent 2001). Like language loss, traditional knowledge loss is related to sociocultural, demographic and economic changes, which affect knowledge (and language) transmission in complex and unpredictable ways. Thus while the two phenomena of language loss and traditional knowledge loss are not always perfectly correlated, the loss of speakers of a language is generally suggestive of changes in the social dynamics that underpin traditional knowledge transmission.

Language erosion and loss can ultimately stem from a wide range of factors related directly to the environment and biodiversity, including catastrophic events, such as floods, droughts, disease or earthquakes, that may decimate communities or the gradual erosion of speakers over generations, due to loss of access to traditional territories and resources, displacement and migration, including urbanization, and partial or whole replacement with other languages. The decline may also be partially a consequence of negative pressure from exogenous elements, such as formal education (in the dominant language) and external models of economic development that may be voluntarily taken up by indigenous and local communities who are searching to improve their well-being in a rapidly changing world. In these circumstances traditional knowledge may change as communities incorporate new information and as they respond to a new range of problems. Mistaken beliefs that proficiency in an indigenous language will hinder acquisition of a national language or that speaking an indigenous language threatens dominant cultural values can lead to educational and public policies that encourage the loss of indigenous languages in favour of dominant languages. This phenomenon, known as language shift, is possibly the most serious form of language loss experienced by indigenous and local communities. These challenges are increased by the fact that many indigenous or traditional languages are spoken by small numbers of speakers, where a small loss of speakers will have a greater impact on the overall social dynamics that underpin language (and knowledge) transmission.

Despite persistent gaps in the quality and completeness of data, three different analyses offer useful insights into the current trends of linguistic diversity and numbers of speakers of indigenous languages.

UNESCO's *Atlas of the World's Languages in Danger* (Moseley 2010) represents a snapshot view of almost 2,500 endangered languages, based on an analysis of the degree of intergenerational transmission at the time of data collection. Based on this analysis approximately 40% of languages spoken in the world are vulnerable or endangered (Figure 18.1). Nonetheless, the data in the *Atlas* were not collected with the purpose of measuring trends, making it difficult to extrapolate how many of these languages may become extinct within a given period of time. Like *the Atlas, the Ethnologue* provides no longitudinal information and it is common for population figures to be carried over unchanged from one edition to the next, even for seriously endangered languages with few speakers (Hammarström 2005; Paolillo and Das 2006). For this reason, UNESCO developed a language database (Minasyan 2013).

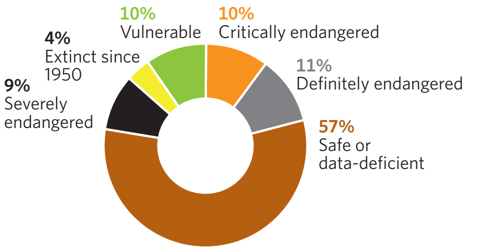


Figure 18.1. Breakdown of language threat level in UNESCO's Atlas of World Languages in Danger (UNESCO nd)[2]. Approximately 40% of languages are vulnerable or endangered, based on the degree of transmission between generations, at the time of data collection. Source: Moseley (2010).

UNESCO's language database currently contains data on over 3,000 languages in 123 countries, as well as other information about the sources of the data, how it was gathered, etc. The data were gathered from two primary source types: government censuses and non-governmental studies (including academic publications). While the dataset is not complete, it is

Footnote

[2] *See UNESCO Atlas of World Languages in Danger (http://www.unesco.org/culture/languages-atlas/index.php).*

more comprehensive than any alternative source, and it was gathered with the intention of informing on trends of language population growth or shrinkage, which can in turn provide information on language endangerment.

Analysing trends for 1,003 languages with at least two data points, it was found that approximately 38% of languages are shrinking (Paolillo, *in preparation*). It is worth noting that number of speakers has a significant influence on the growth trend of languages: larger languages tend to grow, while smaller ones tend to shrink. This finding leads to two important conclusions. Firstly, smaller languages are imperiled. Secondly, there is a critical size or "tipping point" below which a language is likely to lose speakers. The current data support only a broad estimate of this tipping point, which is centred on 8,239 speakers with a 95% certainty ranging between 2058 and 54,923 speakers. The wide range of this figure is likely to be due to the lack of accurate data, even in the UNESCO database, and the complex dynamics which determine language loss or gain.

A number of studies have shown that half of languages are spoken by fewer than 10,000 people (Grimes 1986, Paolillo & Das 2006, Minasyan 2013), roughly corroborating the findings of the UNESCO *Atlas* (Figure 18.1). Given the broad tipping point estimate, the finding that 38% of languages sampled in the UNESCO database are currently losing speakers generally supports this finding. However, these figures only take into account whether languages are gaining or losing speakers, not the speed at which this change is occurring. If trends for the 1003 languages for which longitudinal data is held in the database are extrapolated into the future, approximately 15% of languages are predicted to be extinct by 2100.

Although providing the best estimate currently available, these figures must be interpreted with caution. For many languages, only two data points are available, hence there is considerable uncertainty around these estimates. In addition, some of the estimates are clearly unrealistic; for example, a few small languages are projected to grow from the hundreds to the millions by 2100. This may reflect the relative paucity of information about smaller languages, changes in the counting of some languages, the early success of certain known revitalization efforts, or other factors. Finally, the sampling of languages in the database is still incomplete, with certain regions, notably Africa, being poorly represented. Taking all of these issues into account, the information in the database can be regarded as representing an optimistic view of language vitality and endangerment. It is likely that additional data would change this picture.

To summarise, several key conclusions may be drawn from UNESCO's *Atlas of the World's Languages in Danger* (Moseley 2010) and an analysis of UNESCO's language database (Paolillo *in preparation*). Firstly, best evidence suggests that 35-50% of languages are vulnerable or endangered, with at least 15%, if not many more, facing extinction by the end of the century. There is a tipping point of roughly 8-10,000 speakers below which languages should be considered particularly vulnerable, and should be prioritized for urgent support and revitalization. However, languages with more than 10,000 speakers may be shrinking, too, depending on the sociocultural and economic context of each language. Language dynamics are complex and small shifts in the social, cultural, economic, environmental or demographic context can have marked impacts on a language's vitality. This is particularly true of those languages with few speakers, for which changes can have particularly dramatic impacts either for the worse or, where language revitalization programmes are successful, for the better.

Further information on traditional languages is provided through the Index of Linguistic Diversity (ILD) developed by the non-governmental organization Terralingua using Ethnologue data. As described above, the *Ethnologue* data has some limitations as regards its applicability to a longitudinal study, but it provides some indication of what may be happening with smaller languages, especially in regions where national census data is absent or irregular, such as in Africa.

According to the Index of Linguistic Diversity, one-fifth of the world's linguistic diversity has been lost since 1970, as the human population shifts from speaking less-populous mother tongues to dominant languages (Harmon and Loh 2010). The Index of Linguistic Diversity (ILD)[3] is a metric that conveys the changes in the relative distribution of mother-tongue speakers, though it does not take into account multilingualism which is the norm rather than the exception (Tucker 1999). Of the approximate 7,000 languages on Earth, 1,500 were randomly selected from the 2005 15th edition of *Ethnologue*[4] for use in the ILD. An unchanged index value indicates that, within the language group being indexed, each language has maintained hypothetical stability over time, that is, its proportional share of all mother-tongue speakers has not changed.[5] A drop in ILD indicates a decline in the evenness of distribution of mother-tongue speakers among languages. The Global ILD declined 20% between 1970 and 2005 (see Figure

Footnotes

[3] *The methodology for and first iteration of the Index of Linguistic Diversity were developed in 2006-2010 as a Terralingua project (www.terralingua.org/linguisticdiversity).*

[4] *http://www.ethnologue.com/*

[5] *Some scholars point out that a language may be gaining speakers, but if it is not gaining as many speakers as English, for example, it is considered to be in decline.*

18.2), and Global Indigenous ILD declined 21% during the same period. As Harmon and Loh (2010) estimate that 80-85% of all languages are indigenous (as defined by ILO Convention 169), it is not surprising that ILD Global and ILD Global Indigenous show similar trends.

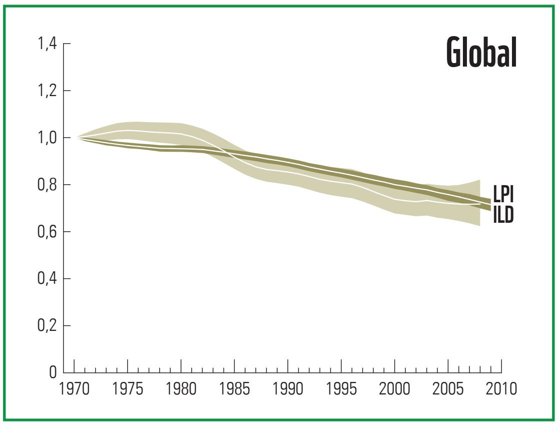


Figure 18.2. Global Index of Linguistic Diversity (ILD) and Living Planet Index (LPI) 1970 - 2012. The ILD measures loss of speakers of minority languages and mother tongues over time. At the global scale, linguistic diversity has been in decline since at least 1970, and most markedly since the mid 1980s. Upper and lower confidence limits, showing the boundary of the 95% confidence interval, are shown as thin lines above and below the main trend line. Source: Loh and Harmon (2014).

Land Indicator

Dispossession of traditional territories and natural resources continues to be a major problem confronting indigenous and local communities. Secure rights to land, territories and resources, including access, control and management of those resources, represent a fundamental requirement to enable communities to maintain and practise their customary use and traditional knowledge in their daily interaction with the biodiversity around them. Customary use and practices cannot be disconnected from the natural resources in traditional lands and territories. If communities do not have secure land and resource rights, this is a threat to their customary use systems[6] and traditional knowledge.

The global pace of commercial land acquisitions has increased dramatically since 2005, peaking in 2009 probably due to a spike in farmland purchasing following the 2007-08 food price crisis (see Figure 18.3) (Anseeuw *et al.* 2012). Africa is the region most targeted by the "land rush" since 2000. While there is yet no systematized metric for assessing the global status and trends in land-use change and land tenure relevant to indigenous and local communities, in most countries, land dispossession and other factors have displaced an estimated 50% at least of indigenous and local communities to urban areas (UN-Habitat 2011). Furthermore, an abundance of representative and indicative case studies reveals that traditional territories remain insufficiently protected from and broadly vulnerable to the high commercial demand for land. The local impacts of land grabbing schemes consistently depreciate food, water, housing, and livelihood security, especially in the short and medium terms (Anseeuw *et al.* 2012).

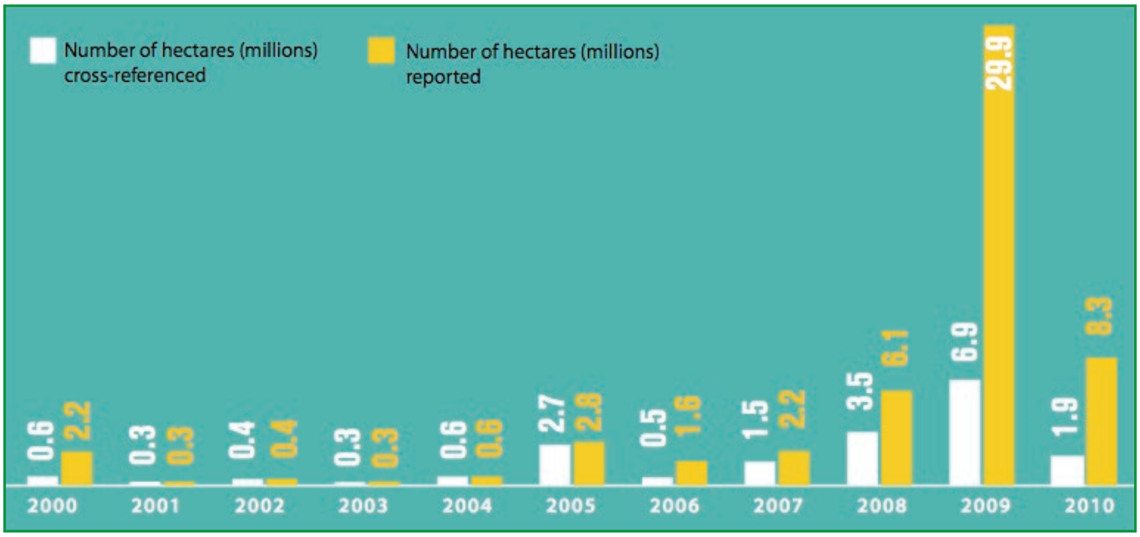


Figure 18.3. The global pace of land acquisitions. The year 2009 marked a dramatic spike in land grabbing, but otherwise, the reported area of land acquired every year has increased on average since 2000. Source: Anseeuw *et al.* (2012).

Footnote

[6] *Refer Alexandre Corriveau-Bourque at http://www.rightsandresources.org/publication_details.php?publicationID=6587*

Indigenous and local communities dispossessed of their traditional territories may or may not be evicted from their land (Anseeuw *et al*. 2012). However, in every scenario of land tenure dispossession, including privatization, indigenous and local users face a reduction in the amount or quality of resources that they may reap from their own territories. The loss of access to land is especially detrimental to the subsistence of pastoralists and people dependent on forest resources. Because they rely on very large, necessarily communal tracts of lands, they are extremely vulnerable to land seizure, development, and partitioning. Where traditional territories are seized from indigenous and local communities, compensation (if awarded at all) is often inadequate. Financial payments cannot compensate for the cultural losses experienced by disenfranchised indigenous and local communities, infrastructure and services provided especially to remote communities is generally of poor quality, and the potential for job provision during- and post-development is generally exaggerated. Jobs that are provided are often low-paid and temporary (Anseeuw *et al*. 2012). Intergenerational transfer of territories and resources is disrupted.

Unequal land rights, systematic discrimination, income disparities, and the profusion of sexual and domestic violence disproportionately expose women to the detrimental impacts of land tenure loss (Anseeuw *et al*. 2012).

Traditional Occupations Indicator

The practice of traditional occupations is a tangible component of the knowledge, innovations and practices of indigenous and local communities, theoretically making it relatively easy to assess. Common examples of traditional occupations include hunting, gathering, trapping, fishing, herding and grazing, shifting cultivation, weaving and carving. By preliminary definition, effective performance in a traditional occupation depends on the worker having knowledge of the traditional culture and practices (CBD, 2009). Beyond that aspect, the definition of a traditional occupation needs refining. Because many activities, tasks, and occupations within indigenous and local communities are distributed according gender, a full account of the status and trends in the practice of traditional occupations would not be complete without considering the differing roles of women and men (Ballard 2012).

As women are often the backbone of the indigenous community and usually responsible for raising children, the burden of meeting the immediate needs of the family in times of hardship tends to rest on them (Ballard 2012). Thus, the loss of traditional livelihoods and the resulting need to seek supplementary income disproportionately exposes women to poverty and exploitation. In some cases this may mean that women must forego traditional activities in order to earn income, thereby leading to an erosion of traditional knowledge (Heckler 2002), while in other cases, it may lead to an increased dependence upon traditional livelihoods. For example, when faced with economic hardships, single mothers in the Anishanabek community of Manitoba, Canada, rely more heavily on traditional activities, such as collecting berries and traditional medicines, reciprocal child-minding duties, or hunting and preparing game (Ballard 2012).

In conclusion, with political will and adequate financial support, the Working Group on Article 8(j) is likely to complete the development and adoption of several sets of guidelines, standards and tools by 2020 (see also section 2.a.), useful for the effective implementation of article 8(j)[7] and related provisions. However, on the basis of trends outlined above there is a risk that Target 18 may not be achieved until the products of the Working Group are adopted and effectively implemented at the national and local level, with the effective participation of indigenous and local communities.

Integration and Participation Indicator

Further assessment of what is needed to achieve Target 18 needs to more deeply consider where progress is being made in terms of the various indicators, including the fourth indicator "Trends in which traditional knowledge and practices are respected through their full integration, safeguards and the full and effective participation of indigenous and local communities in the national implementation of the Strategic Plan", as many of these changes are likely to be the result of policy changes and legal reforms happening at the country level, rather than from guidelines being developed by Working Group 8j of CBD.

Footnote

[7] *Article 8(j) Each Contracting Party shall, as far as possible and as appropriate subject to its national legislation, respect, preserve and maintain knowledge, innovations and practices of indigenous and local communities embodying traditional lifestyles relevant for the conservation and sustainable use of biological diversity and promote their wider application with the approval and involvement of the holders of such knowledge, innovations and practices and encourage the equitable sharing of the benefits arising from the utilization of such knowledge, innovations and practices;*

Indigenous and Local Community Initiatives in Support of Indicators

Indigenous and local Communities are advancing their own solutions to establishing status and trends in the indicators adopted for traditional knowledge including such initiatives as community based monitoring and information systems. Community-based monitoring and information systems (CBMIS) refers to the bundle of monitoring approaches related to biodiversity, ecosystems, land and waters, and other resources, as well as human well-being, used by indigenous and local communities as tools for their management and documentation of their resources. Community-based monitoring and information systems use an innovative methodology based on both traditional knowledge and new tools such as digital mapping using the latest technology, three-dimensional (3D) maps and printers and the countryside management software (CMS). The methodology is based on traditional knowledge and is particular for each indigenous or local community. CBMIS combines traditional knowledge and new technologies for use by communities in various assessments and it is a base for developing planning and decision-making. CBMIS could also contribute at national, regional and global levels through improved local, national and regional information systems. Further to this, the Swedish Resilience Centre is promoting a methodology using a Multiple Evidence Base approach which is compatible with CBMIS and which may also be very useful in arriving at a picture of status and tends in the indicators adopted for traditional knowledge. CBMIS and a Multiple Evidence Base approach may assist Parties in drafting of national reports, noting the guidelines for the fifth national reports[8] call for indigenous and local community participation.

18.1.2 Projecting forward to 2020

The survival of traditional knowledge is at a crossroad. Studies such as the composite report on the status and trends regarding the knowledge, innovations and practices of indigenous and local communities relevant to the conservation and sustainable use of biodiversity[9] have identified the use and transmission of traditional knowledge to be in decline and facing many obstacles to its retention and use in recent history. At the same time there is renewed interest by indigenous and local communities, Parties and governments, as well as the private sector in its retention and use. There are also excellent traditional language restoration and revival programmes in a number of countries. The adoption of the Nagoya Protocol in 2010 is also contributing to both the protection and promotion of traditional knowledge associated with genetic resources.

Parties have shown renewed interest in progressing tools, through the revised programme of work for article 8(j) and related provisions, to fully implement commitments under articles 8(j), 10(c)[10] and related provisions. Parties and governments are also increasingly reporting on related initiatives, both in their national reports and directly to the Working Group on Article 8(j) and Related Provisions.

Today there is a growing appreciation of the value of traditional knowledge. This knowledge is valuable not only to those who depend on it in their daily lives, but to modern industry and agriculture as well, where indigenous and local communities choose to grant access to such knowledge. Many widely used products, such as plant-based medicines, health products and cosmetics, are derived from traditional knowledge. Other valuable products based on traditional knowledge include agricultural and non-timber forest products as well as handicrafts.

The holistic nature of traditional knowledge places an emphasis on complex relationships that maximize and enable adaptive decision-making in local practices while providing feedback information on both short and long-term ecological and social trends. Traditional knowledge can make a significant contribution to global discussions concerning sustainable development goals and the post 2015 sustainable development agenda. Recognition and protection of customary sustainable use of biodiversity can contribute significantly to poverty alleviation. Many indigenous and local communities are situated in areas of high biological and genetic diversity. Many of them have sustainably managed and used biological diversity for thousands of years. Some of their practices have been proven to enhance and promote biodiversity at the local level and aid in maintaining healthy ecosystems. The contribution of indigenous and local communities to the conservation and sustainable use of biological diversity goes far beyond their role as natural resource managers. Their skills and techniques provide valuable information to the global community and a useful model for biodiversity policies. Furthermore, as on-site communities with extensive knowledge of local environments, indigenous and local communities are most directly involved with conservation and sustainable use and can be the first to notice and raise alarms about the erosion of biodiversity. Because of this, traditional knowledge finds itself interfacing with science more and more.

Footnotes

[8] Refer to decision X/10, paragraph 11.

[9] See UNEP/CBD/WG8J/5/3 http://www.cbd.int/doc/?meeting=WG8J-05; UNEP/CBD/WG8J/AG/2/2/ADD4 http://www.cbd.int/doc/meetings/tk/acpow8j-02/official/acpow8j-02-02-add4-en.doc; UNEP/CBD/WG8J/AG/2/2/ADD5 http://www.cbd.int/doc/meetings/tk/acpow8j-02/official/acpow8j-02-02-add5-en.doc; UNEP/CBD/WG8J/AG/2/2/ADD6 http://www.cbd.int/doc/meetings/tk/acpow8j-02/official/acpow8j-02-02-add6-en.doc

[10] Article 10(c) Each Contracting Party shall, as far as possible and as appropriate: (c) Protect and encourage customary use of biological resources in accordance with traditional cultural practices that are compatible with conservation or sustainable use requirements;

Despite important advancements at the policy level, for instance the United Nations Declaration on the Rights of Indigenous Peoples, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization, there is no concrete evidence that the erosion of traditional knowledge is slowing. This may be partly related to a lack of quantitative data, however concerns have also been raised about the many barriers to implementation of these instruments and to ensuring that increased appreciation for traditional knowledge improves the situation for indigenous peoples and local communities.

18.1.3 Country actions and commitments[11]

As evidenced in national reports and through submissions to the Working Group on Article 8(j) and related provisions, Parties increasingly recognize the importance of traditional knowledge and sustainable use (as cross-cutting issues) in reaching the goals of the Convention on the conservation and sustainable use of biodiversity, in light of the Strategic Plan for Biodiversity 2011-2020 and the Aichi Biodiversity Targets. Traditional knowledge and sciences can be complimentary and mutually beneficial to scientists and communities so long as there is recognition of the particular strengths and limitations of both types of knowledge" (Fraser, *et al*., 2006). Synthesizing effective strategies that can meet mutually defined conservation goals will require improved attitudes of mutual learning from multiple knowledge systems, more effective communication among academic disciplines, deeper analysis of what is working at the community level, and identification of where there are gaps in expertise and application" (Chan, *et al*., 2007)

Slightly more than half of the national biodiversity strategies and action plans examined contain targets or similar commitments related to traditional knowledge. These national targets are broadly in line with the Aichi Biodiversity Target 18. The targets generally focus on ensuring that traditional, knowledge innovations and practices are respected. By comparison there is less explicit emphasis on the integration of traditional knowledge innovations and practices into the implementation of the Convention or on ensuring the full and effective and participation of indigenous and local communities. Two examples counter to this trend are Brazil and Finland which have both established targets which reflect the various elements of the Aichi Biodiversity Target 18.

Some countries, for example Malta and Serbia, have established commitments in their national biodiversity strategies and action plans which relate to local communities. A number of countries, for example Suriname, have also included references to access and benefit sharing in their commitments related to Aichi Biodiversity Target 18.

18.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

To an extent the Parties have identified and prioritized tasks of the revised programme of work for article 8(j) and related provisions, as tools necessary for achieving target 18 by 2020. Their early completion and adoption is therefore desirable in global efforts to achieve Target 18. However, practical actions such as national action plans supporting community action plans for both traditional knowledge and customary sustainable use may be the best way forward in lieu of the finalized tools.

18.2.1 Actions

Traditional knowledge contributes to both the conservation and the sustainable use of biological diversity. This target aims to ensure that traditional knowledge is respected and reflected in the implementation of the Convention, subject to national legislation and relevant international obligations, with the effective participation of indigenous and local communities. Given the cross-cutting nature of this target, actions taken to fulfill it will contribute to several of the other Aichi Biodiversity Targets and the Nagoya Protocol. The GBO-4 concluded that processes are under way internationally and in a number of countries to strengthen respect for, and recognition and promotion of, traditional knowledge and customary sustainable use. Efforts to enhance the capacities of indigenous and local communities to participate meaningfully in relevant processes locally, nationally and internationally are progressing but limited funding and capacity remain obstacles. However, overall traditional knowledge continues to decline as illustrated by the loss of linguistic diversity and large-scale displacement of indigenous and local communities to urban areas, although this trend is reversed in some places through growing interest in traditional cultures and involvement of local communities in management of protected areas. Against this background, possible key actions to accelerate progress towards this target include:

Footnote

[11] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

- Developing national arrangements, aligned with relevant guidance under the Convention, on recognizing and safeguarding the rights of indigenous and local communities over their knowledge;
- Promoting local initiatives that support traditional and local knowledge of biodiversity and promote customary sustainable use, including traditional health care initiatives; strengthening opportunities to learn and speak indigenous languages; research projects and data collection based on traditional methodologies (*Target 19*); and involving local and indigenous communities in the creation, control and management of protected areas (*Target 11*);
- Raising awareness of the importance of traditional knowledge to conservation and sustainable use of biodiversity and applying it (Tar*get 1*);
- Supporting and cooperating in the organization of capacity-building activities on relevant issues under the Convention for indigenous and local communities, as well as cultural awareness-raising programmes; and
- Promoting effective participation of indigenous and local communities, at all levels, in issues related to biodiversity and of interest to them.

The guidance developed as part of the Convention's programme of work on traditional knowledge, innovations and practices (Articles 8(j) and 10(c) and related provisions) provides advice on how this target may be implemented.[12] More specifically, in light of the Strategic Plan for Biodiversity 2011-2020 and the Aichi Biodiversity Targets, and the adoption of the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization, the programme of work for Article 8(j) and related provisions (established by decision V/16, annex) was revised by the Parties in decision X/43 with a focus on:

- Task 7, Guidelines for mechanisms to ensure that: indigenous and local communities obtain a fair share of benefits arising from the use and application of their knowledge, innovations and practices; that prior informed consent of the knowledge holders is obtained by entities interested using such knowledge and; that obligations are identified for countries of origin off traditional knowledge, as well as users countries.
- Task 10, Standards and guidelines to report and prevent the unlawful appropriation of traditional knowledge.
- Task 12, Guidelines for national legislation or other mechanisms to implement Article8(j) and related provisions, including *sui generis* systems, that recognize, safeguard and fully guarantee the rights of indigenous and local communities over their traditional knowledge.
- Task 15, Guidelines that would facilitate repatriation of knowledge and related information to facilitate the recovery of biodiversity related knowledge, innovations and practices.

Parties have committed to completing these tasks in order to finalise tools needed by Governments to achieve Target 18. Parties have also committed to adopting and implementing a Global Plan of Action on Customary Sustainable Use as a major component of work of the revised programme of work for articles 8(j), 10(c) and related provisions.

Underpinning efforts by governments are efforts at the national level concerning equity and governance. In particular, realizing equitable governance of protected areas and the recognition and support of community conservation efforts should be paramount in achieving both Target 18 and Target 11 (on protected areas).

At the local level, there are several areas where certain types of initiatives have shown particular promise. Some of these are:

- Community Action Plans for the retention and inter-general transmission of Traditional Knowledge and promotion of Customary Sustainable Use;
- Indigenous and local community education or pluralistic education systems, which incorporate traditional languages and traditional knowledge, especially at the early childhood level[13].
- Traditional health care initiatives;
- Strengthening opportunities to learn and speak indigenous languages, including language revitalisation programmes;
- Culturally appropriate tourism policies and initiatives;
- Environmental research projects and data collecting based on the traditional methodologies of indigenous and local communities;
- Building of culturally appropriate business structures within communities (such as cooperatives);
- Developing technologies (such as agricultural tools) that focus on traditional methods of harvesting;

Footnotes

[12] *A Plan of Action on Customary Sustainable Use of Biological Diversity is currently under development. Once finalized this plan will provide an additional source of guidance on the possible actions that can be taken to reach this target.*

[13] *There are positive examples of intercultural bilingual (mother tongue) education available at http://www.rutufoundation.org/en/examples/*

- Reestablishment of traditional spiritual/religious institutions);
- Creation of media, such as radio, newspapers and television stations controlled by indigenous and local communities, in local languages and with local content;
- Initiatives bringing together youth and Elders for inter-generational knowledge and language transmission;
- Creation and promotion of businesses offering traditional products and services;
- Strengthening institutions that foster traditional collection and distribution of food and other resources.
- Recognition and/or establishment of community conservation areas and more broadly diverse arrangements between governments and indigenous and local communities regarding the management of protected areas (see for example Kothari *et al*., 2012).

In all types of mechanisms and measures used to promote traditional knowledge, as well as cultural, social and economic well-being, it seems that capacity-building is crucial. This involves a significant commitment to building the educational, governance, management and professional capacity of indigenous and local communities, as well as cultural awareness programmes for governments and other stakeholders such as scientists or the private sector. It is also important to build the strength, infrastructure and capacity of indigenous and local institutions, such as governance structures, research bodies, economic structures, health care systems and education systems.

Key mechanisms at national level include national-level strategies, mechanisms, legislation or other appropriate initiatives such as national action plans, including *sui generis* systems, for promoting/protecting traditional knowledge and the customary sustainable use of biological diversity. This may include such activities as reviews of legal frameworks and practices, adoption of law reforms measures and/or *sui generis* systems for the protection, preservation and promotion of traditional knowledge, as well as communication, education and public awareness activities with a focus on awareness raising of the value of traditional knowledge and customary sustainable use of biodiversity, including production of indigenous and local language educational resources and materials, maintenance of information portals, development and promotion of case studies and operationalizing agreed indicators for traditional knowledge at local/national levels (land tenure, traditional occupations and traditional languages).

In order to more accurately monitor trends in language endangerment and to be able to identify where support and resources are most needed, more data is required, especially in Africa, where many countries do not collect language data in their censuses. National censuses are an important tool for collecting data on such as things as traditional languages and traditional occupations, but capacity-building is required to ensure the inclusion of questions on languages and occupations in all regions that is geographically and longitudinally comparable.

Capacity building initiatives to foster effective participation of indigenous and local communities in the implementation of Articles 8(j), 10(c) and related provisions at regional, national and sub-national levels are a critical element in achieving the suite of activities listed above.

18.2.2 Costs and cost-benefit analysis

Target 18 is is regarded as both a cross-cutting issue and as an essential element of the "enabling" cluster, which will assist achieving all other Targets. Its effective attainment can assist the Parties in reaching other targets such as 11, 12, as well as 16.The effective implementation of Target 18 therefore can significantly contribute to preserving and promoting biological and genetic diversity, reduce the costs of protected areas management, as well as assist in climate change adaptation and maintain and improve eco-system services. Target 18 also contributes more broadly to the preservation and promotion of biological and cultural diversity.

The cost of not reaching Target 18, which would see traditional knowledge falling further into disuse, is unfathomable. In losing knowledge, humanity and the ecosystems on which we depend will become less resilient, less able to adapt to change and more prone to environmental shocks such as those brought on by climate change. Diminishing traditional knowledge directly impacts on food security and on plant and animal diversity. Moreover, indigenous peoples and local communities have the right to maintain and transmit their own systems of knowledge as recognised most explicitly in Article 31 the United Nations Declaration on the Rights of Indigenous Peoples.

The total amount of resources needed on average by Parties to be able to meet Target 18, accomplishing the three activities over the period 2013-2020 would require US$8.4 million to US$13.8 million per country or on average US$1.05 million to US$1.73 million per year. The grand total for the Secretariat costs associated with Article 8(j) and related provisions for the period 2013-2020 is estimated to be US$18,876,000[14]. There is no cost estimate on the actions required by indigenous and local communities although their contributions to the goals of the Convention are immeasurable.

Footnote

[14] *UNEP/CBD/COP/11/INF/20, Input to the High Level Panel on Global Assessment of Resources for the implementing the Strategic Plan for Biodiversity 2011-2020.*

18.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

Traditional knowledge can make a significant contribution to sustainable development, as well as conservation and sustainable use. Most indigenous and local communities are situated in areas where the vast majority of the world's biological and genetic resources are found. Many of them have cultivated and used biological diversity in a sustainable way for thousands of years. Some of their practices have been proven to enhance and promote biodiversity at the local level and aid in maintaining healthy ecosystems. However, the contribution of indigenous and local communities to the conservation and sustainable use of biological diversity goes far beyond their role as natural resource managers. Their skills, techniques and innovations provide valuable information to the global community and useful models for biodiversity policies. Furthermore, as on-site communities with extensive knowledge of local environments, indigenous and local communities are most directly involved with conservation and sustainable use. Indigenous and local communities are well placed to actively contribute to the management of protected areas, including their own Indigenous and local Community Conservation Areas, which can make a major contribution to achieving Aichi Biodiversity Targets 11 and 12.

18.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

The composite report (UNEP/CBD/WG8J/5/3) identified a significant number of international, national and local processes that may threaten the maintenance, preservation and application of traditional knowledge, innovations and practices, including:

- Environmental threats (including environmental damage, climate change, invasive species);
- Loss of indigenous languages;
- Coerced imposition of other religions and value systems;
- Continuing colonization and coerced assimilation;
- Imposed exogenous education systems;
- Objectification through tourism;
- Militarization, conflict, insecurity and war;
- Application of new technologies where there is a lack of opportunity for indigenous and local communities to adopt and adapt new technologies within their knowledge systems and to support respect, retention and maintenance of traditional knowledge;
- Social disintegration, including high rates of suicide, incarceration and violent death;
- Racism and discrimination;
- Degraded health and well-being including poverty, HIV, and restrictions on traditional health practices and practitioners;
- Destruction or reduced availability of traditional foods and medicines and food aid;
- Gender issues;[15]
- Lack of capacity, including infrastructure, training, and financial and social capital;
- Increasing populations – including young populations and low life expectancy;
- Increasing urbanization, forced relocations and coerced migration resulting from, among other things dispossession, environmental damage and lack of economic opportunities;
- Restrictions on self-governance and lack of participation in decision making processes;
- Lack of respect for traditional knowledge, customary sustainable use and customary law, including lack of formal recognition by government and academia, and denigration of traditional knowledge and traditional knowledge holders in the general public;
- Lack of security for indigenous and local communities' land tenure/usufruct rights and restrictions on access to traditional territories including sacred sites and protected areas;
- Unsustainable economic development and degradation of ILCs' traditional economic bases;
- Unsustainable exploitation of natural resources (with possible subcategories for fish, forests, etc.);
- Globalization, including concentration of political and economic power and homogenization of cultural influences;
- Misappropriation of traditional knowledge including through biotrade, bioprospecting, and weak/inappropriate intellectual property rights regimes.

Footnote

[15] *Gender issues, including gender specific knowledge, need to be considered against a standard of non-discrimination and affirmative action, noting that indigenous and local community women and girls are particularly vulnerable to both internal and external discrimination.*

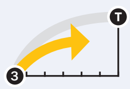
Traditional knowledge, customary sustainable use and biological and cultural diversity are all at risk if Target 18 is not met as soon as possible or by 2020 at the latest. Not reaching Target 18 will impact on other targets. Losing knowledge and diversity will also impact on food security, local resilience and adaptation to climate change. It is by no accident that the Parties to the Convention have identified articles 8(j), 10(c) and related provisions as cross-cutting issues, which can assist the implementation of the Strategic Plan and the Aichi Biodiversity Targets. At the same time, further progress is needed in the incorporation of traditional knowledge and customary sustainable use in practical ways that can advance the effective on-the-ground implementation of the strategic plan and the other 19 Aichi Biodiversity Targets.

18.5 UNCERTAINTIES AND DATA REQUIREMENTS/GAPS

The achievement of Target 18 depends on political will and on the broader societal arrangements for accommodating indigenous, local and traditional communities. At the same time, indigenous and local communities, in the face of adversity have shown remarkable resilience. In recent times, national reconciliation processes, anti-discrimination laws and social justice processes, including legal processes to address traditional land tenure issues and improvements in the health and social well-being of indigenous and local communities have combined to create stronger resilient communities who are actively engaged in cultural restoration, including revival and transmission of traditional knowledge. At the same time many of the obstacles and forces undermining traditional knowledge continue to grow stronger.

In the context of broad global issues facing humanity, including the post 2015 development agenda and sustainable development goals, the fate of indigenous and local communities, their knowledge innovations and practices, lay very much in the balance. Increasingly indigenous and local communities participate in global discussions to both to defend their rights and to provide input into the many perplexing issues facing humanity.

18.6 DASHBOARD – PROGRESS TOWARDS TARGET 18

Target Elements	Status	Comment	Confidence
Traditional knowledge, innovations and practices of indigenous and local communities are respected		Processes are under way internationally and in a number of countries to strengthen respect for, recognition and promotion of, traditional knowledge and customary sustainable use	Medium
Traditional knowledge, innovations and practices are fully integrated and reflected in implementation of the Convention ...		Traditional knowledge and customary sustainable use need to be further integrated across all relevant actions under the Convention	Low
... with the full and effective participation of indigenous and local communities		Efforts continue to enhance the capacities of indigenous and local communities to participate meaningfully in relevant processes locally, nationally and internationally but limited funding and capacity remain obstacles	Low

Authors: John Scott and Robert Höft with with contributions from Katherine Blackwood, Serena Heckler and John Paolillo

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TARGET 19: KNOWLEDGE, SCIENCE AND TECHNOLOGY

By 2020, knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied.

PREFACE

All countries need information to assess the status of biodiversity, identify threats to biodiversity and determine priorities for conservation and sustainable use. While nearly all Parties report that they are taking actions related to monitoring and research, most also indicate that the absence or difficulty in accessing relevant information is an obstacle to the implementation of the goals of the Convention. Actions taken towards this target will greatly facilitate the implementation of the Strategic Plan and the fulfilment of the other 19 Aichi Biodiversity Targets by encouraging new research, the development of new technologies and improved monitoring. Such actions will strengthen and improve the science-policy interface and will contribute to the fulfilment of the other elements of the Strategic Plan. Reaching this target will require substantial investment in global and national biodiversity observation networks, implementation of the Global Taxonomy Initiative, and further investment in research, including modelling and participatory research. With regards to the sharing of technologies related to biodiversity, this should be consistent with Article 16 of the Convention.

19.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

19.1.1 Status and trends

Countries need information to assess the status of biodiversity, identify threats to biodiversity, to assess the implications of biodiversity changes for human well-being and to determine priorities for conservation and sustainable use. However, very specific information on biodiversity is required to support such decisions. Today's knowledge about biodiversity is based on centuries of exploration, collections, descriptions and measurements, most of which stem from individual research projects responding to targeted research questions without being coordinated at a higher level. Systematic observations and monitoring were limited to selected places of interest and few were connected (see for example Jongman (2013)). Biodiversity scientists therefore struggle to draw on suitable and long-term datasets and analyse them in such a way as to respond to the needs of policy makers in a timely manner. Yet, coordination and harmonization of data collection, and harmonized storage, management and organized distribution, will significantly increase the value of biodiversity observations by enhancing the availability of relevant information from a variety of sources and enabling analyses across data sources and platforms.

Three main tracks are therefore being pursued more recently: (1) mobilizing and connecting existing data and observations; (2) establishing observation networks that provide the long-term monitoring data needed for enhancing the understanding about biodiversity change supported by platforms for biodiversity data aggregation, validation, and use; (3) setting and coordinating global research agendas and priorities to connect biodiversity research with decision support processes.

Mobilizing and connecting existing data and observations

This track aims to draw on and mobilize existing records, make them compatible and accessible and thereby discoverable for researchers in ways for which they were not necessarily collected. Major initiatives to enhance open access to data and analytical tools include the Global Biodiversity Information Facility[1] (Figure 19.1) and the Ocean Biogeographic Information System[2] (as well as many national initiatives) for sharing, organising and improving species occurrence data; the OGC Web Coverage Service (WCS)[3] for digital geospatial information provision and openModeller[4] for modeling software to support estimation of species distribution.

Footnotes

[1] *http://www.gbif.org*

[2] *http://www.iobis.org/*

[3] *http://www.opengeospatial.org/standards/wcs*

[4] *http://openmodeller.sourceforge.net/*

Regarding information from biological collections, there is the speciesLink[5], a distributed information system that integrates primary data from biological collections. Through the Biodiversity Heritage Library[6] major natural history collections have enabled open access to biodiversity literature.

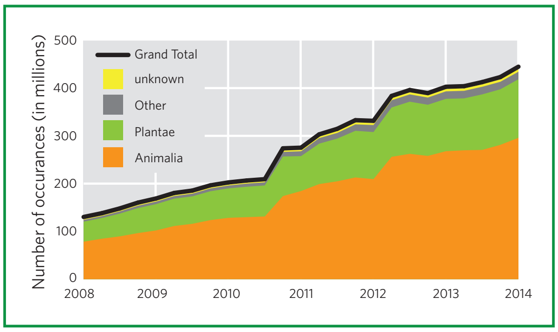


Figure 19.1. Growth in number of species occurrence records accessible through the Global Biodiversity Information Facility. Source: http://www.gbif.org (Tim Robertson)

Recent reviews have enabled better estimates of the total number of species on Earth (for example, Hamilton, Basset *et al*. 2010; Costello & Wilson 2011; Joppa *et al*. 2011; Mora *et al* 2011; Costello *et al*. 2012; Hamilton, Novotný *et al*. 2013;), now estimated at 5±3 million eukaryotes (Costello 2013). Previous estimates of 30 to 100 million species based on potential deep-sea diversity and estimates of insect host specificity now seem highly unlikely.

Molecular approaches and DNA barcoding (Vernooy *et al*. 2010) offer the promise of new data to understand species-level diversity and to help organizing other biodiversity data. However, historical data and most contemporary data relate to described species. Accordingly, a foundational component within this track is the completion of databases of published names for species and their organization into a working classification of all life. The Catalogue of Life is seeking to deliver such a comprehensive listing and classification for all described species. It has grown steadily and currently includes 1.58 million species, including known synonyms for each (Figure 19.2). This is an authoritative resource for managing all other classes of information related to species provided by the taxonomic community.

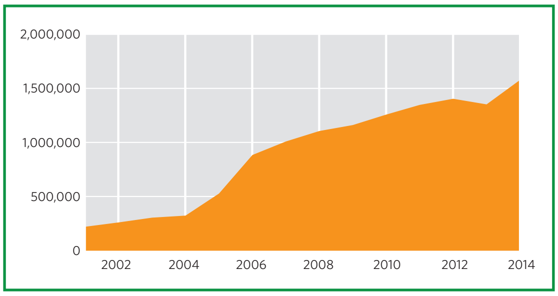


Figure 19.2. Number of species covered in the Catalogue of Life. Source: Catalogue of Life

Other databases, portals, initiatives and assessments contributing to the knowledge base include inter alia:

- National Biodiversity Information Facilities: These include national nodes for the Global Biodiversity Information Facility and major institutions established partly to respond to the need for information exchange and technical and scientific cooperation in line with Articles 17 and 18 of the Convention on Biological Diversity, for example CONABIO in Mexico[7], INBio in Costa Rica[8], Instituto Humboldt in Colombia[9], SANBI in South Africa[10] as well as other nodes of the clearing-house mechanism network [11]
- Regional networks, e.g., Spatial Data Infrastructures (SDIs) (e.g., Arctic SDI)[12]; the infrastructure for spatial information in Europe[13]; the European Biodiversity Observation Network EBONE[14], the Inter American Biodiversity Information Network[15], the Asia-Pacific Biodiversity Observation Network (AP-BON)[16], the East and Southeast Asia Biodiversity Information Initiative (ESABII)[17], the Digital Observatory for Protected Areas[18]; the Circumpolar Biodiversity Monitoring Program[19] and the International Centre for Integrated Mountain Development,[20] among others;
- Global networks, e.g., inter alia the Global Earth Observation System of Systems[21], and especially

Footnotes

[5] *http://splink.cria.org.br*

[6] *http://www.biodiversitylibrary.org/ and http://www.bhlscielo.org/en/*

[7] *http://www.conabio.gob.mx/*

[8] *http://www.inbio.ac.cr/en/*

[9] *http://www.humboldt.org.co/*

[10] *http://www.sanbi.org/*

[11] *http://www.cbd.int/chm/network/*

[12] *http://www.gsdi.org/*

[13] *http://inspire.jrc.ec.europa.eu/*

[14] *http://www.wageningenur.nl/en/Expertise-Services/Research-Institutes/alterra/Projects/EBONE-2.htm*

[15] *www.iabin.net/*

[16] *https://sites.google.com/site/asiapacificbon/*

[17] *http://www.esabii.biodic.go.jp/index.html*

[18] *http://dopa.jrc.ec.europa.eu/*

[19] *http://www.caff.is/monitoring*

[20] *www.icimod.org/*

[21] *https://www.earthobservations.org/geoss.shtml*

its biodiversity component, GEO BON, the Global Biodiversity Information Facility[22], the International Union for Conservation of Nature (IUCN) and its Commissions and major data products[23] or the Barcode of Life Data Systems[24].;

- Thematic networks addressing different aspects of biodiversity or associated concepts, including BirdLife International[25]; the Ocean Biogeographic Information System[26]; the Census of Marine Life[27]; the repository of Ecologically or Biologically Significant Marine Areas[28]; The IUCN Red List of Threatened Species[29]; the Gateway for the Global Invasive Alien Species Information Partnership[30]; the Global Genome Biodiversity Network[31], the Tree of Life Web Project[32]; the World Database on Protected Areas[33]; the World Data Centre for Microorganisms (WDCM) from the World Federation of Culture Collection (WFCC)[34] and the Scientific Collections International (SciColl)[35] and a range of community-based biodiversity monitoring initiatives;
- Assessment networks, such as the Global Mountain Biodiversity Assessment, the Millennium Assessment, the assessment function of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services[36];
- Species and taxonomic information, e.g., Species 2000[37]; the Encyclopedia of Life[38]; FishBase[39], the World Flora online project[40], the MycoBank[41] as well as culture collections, natural history museums and herbaria and national GBIF nodes;

Information reported by countries is accessible through the databases of various United Nations agencies, including FAOSTAT[42]; the UN Data Portal[43]; and UNESCO Atlas of the World's Languages in Danger[44], among others. Data holdings of national technical institutions and clearing-house mechanism nodes are at various stages of interoperability and often only exist in local languages. Linking these to other databases with a view to making the information discoverable is an on-going effort with varying progress in different parts of the world.

Increasingly research funding bodies support calls for biodiversity research that promotes the integration of data and information. The European Research Framework Programme (FP5–FP7) supported major initiatives on habitat monitoring in Europe (BioHab[45]), the development of a European Biodiversity Observation Network (EBONE[46]), and the preparation of a directory of monitoring projects in Europe (EuMon[47]), monitoring of agricultural biodiversity (BIO BIO[48]), Remote Sensing for biodiversity surveillance (BIO SOS[49] and MS.MONINA[50]) and a project on data operability (EU BON[51]). More recently, the Belmont Forum, an international group of funding agencies for global environmental research, has started to issue calls for international research projects that address and investigate the impact of global environmental change on the earth system and society/humanity, and aiming at responding to societal and policy needs.

Significant efforts have been made to develop and promote data and metadata standards for biodiversity observations, to resolve uncertainties about ownership and usage of digital data, objects and tools on the Internet and to promote free and open access to publicly funded research findings. The Conference of the Parties to the Convention on Biological Diversity, in decision VIII/11, invited Parties and other Governments, as appropriate, to provide free and open access to all past, present and future public-good research results, assessments, maps and databases on biodiversity, in accordance with national and international legislation.

Footnotes

21 *https://www.earthobservations.org/geoss.shtml*
22 *http://www.gbif.org*
23 *http://www.iucn.org/*
24 *http://boldsystems.org*
25 *http://www.birdlife.org/*
26 *http://www.iobis.org/*
27 *http://www.coml.org/*
28 *https://chm.cbd.int/database/?schema=marineEbsa*
29 *http://www.iucnredlist.org*
30 *http://giasipartnership.myspecies.info/*
31 *http://data.ggbn.org*
32 *http://tolweb.org/tree/phylogeny.html*
33 *http://www.wdpa.org*
34 *http://www.wfcc.info*
35 *http://www.scicoll.org*
36 *http://www.ipbes.net/*
37 *http://www.sp2000.org/*
38 *http://eol.org/*
39 *http://www.fishbase.org/*
40 *http://www.missouribotanicalgarden.org/plant-science/plant-science/world-flora-online.aspx*
41 *http://www.mycobank.org*
42 *http://faostat.fao.org/*
43 *http://data.un.org/*
44 *http://www.unesco.org/culture/languages-atlas/*
45 *http://www.edinburgh.ceh.ac.uk/projectpages/biohab_page.htm*
46 *http://www.wageningenur.nl/en/Expertise-Services/Research-Institutes/alterra/Projects/EBONE-2.htm*
47 *http://eumon.ckff.si/index1.php*
48 *http://www.biobio-indicator.org/*
49 *http://www.biosos.eu/*
50 *http://www.ms-monina.eu/*
51 *http://www.eubon.eu/*

The decision to provide free access to the Landsat data in the USGS archive[52] has led to a significant uptake, usage and development of value-added products and services as evidenced by the numbers of downloaded scenes and the economic benefits from the expansion of services from their analysis for a range of users (Figure 19.3).

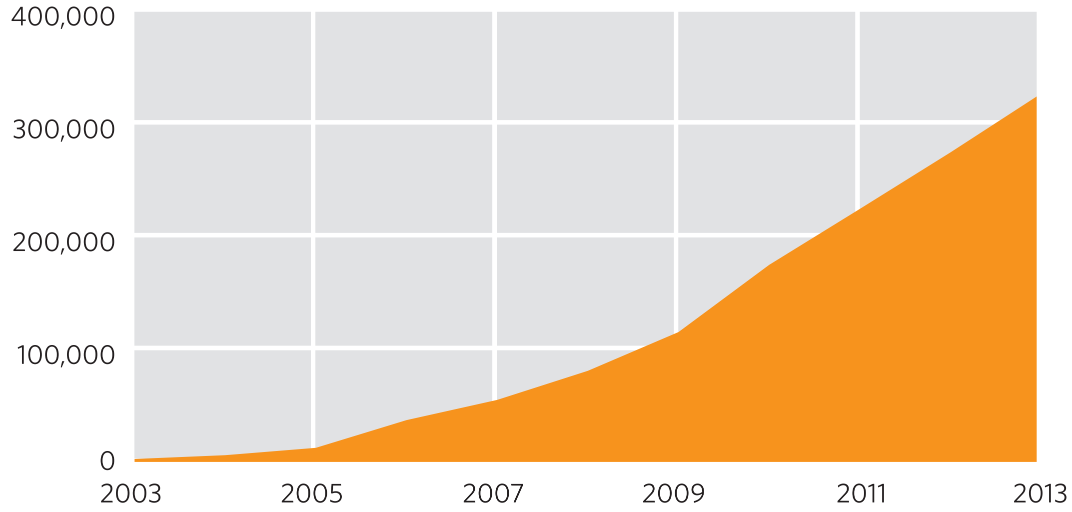


Figure 19.3. Cumulative number of Landsat scenes distributed between 2007 and 2013. Note the hundred-fold increase in delivery since the adoption of the free data policy in October 2008. Source: USGS.

The Instituto Nacional de Pesquisas Espaciais (INPE) in Brazil reports that the free availability of near real-time imagery of the Amazon forest has led to increased awareness of, and participation by stakeholders in conservation and sustainable management of forests, and enhanced compliance with forest legislation. Calls have been made to other agencies to follow these examples and the European Commission regulation on Copernicus is another step towards the free, full and open dissemination of Copernicus dedicated data and service information. France also announced the provision of open access to non-commercial use of 27 years from the SPOT family.[53]

The International Barcode of Life project (iBO[54]) is a global network of research institutions and government agencies undertaking barcoding – a molecular biodiversity approach towards identifying living organisms using short standardized DNA sequences. The library is accessible through the Barcode of Life Data System (BOLD)[55] – the public online repository, analytical workbench and taxonomic identification engine for DNA barcode data. It currently holds nearly 3.2 million records for 210,000 species (Figures 19.4-5). This effort has provided a new set of tools and operational framework for addressing taxonomic impediments and tackling practical challenges in food security, human health, and biodiversity conservation.

Footnotes

[52] *http://landsatlook.usgs.gov/*

[53] *http://www.earthobservations.org/documents/ministerial/geneva/statements/gms_european%20commission_statement.pdf*

[54] *http://ibol.org*

[55] *http://boldsystems.org/*

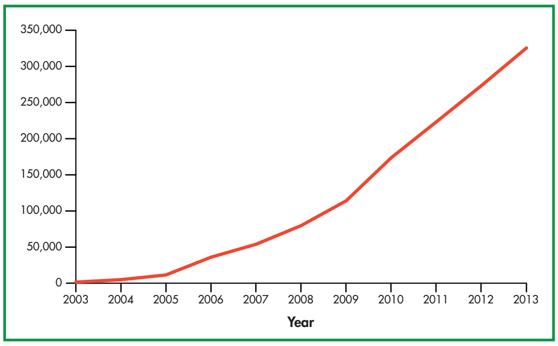


Figure 19.4. Growth of the number of DNA Barcode records in the Barcode of Life Data System global reference library. Source: http://www.boldsystems.org

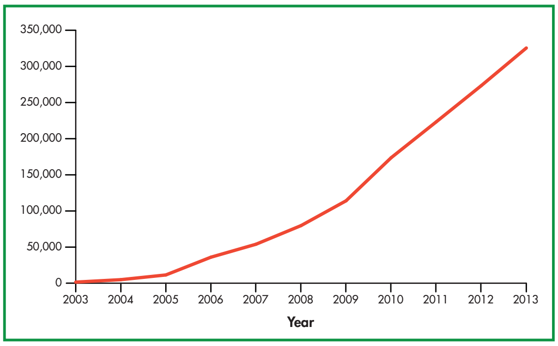


Figure 19.5. Number of animal species represented in the Barcode of Life Data System global reference library. Source: http://www.boldsystems.org.

Promoting biodiversity observation networks and monitoring

Effective monitoring of the status and trends of biodiversity is necessary to enable Parties, individually, and collectively through the Conference of the Parties, to review the implementation of the Convention, the Strategic Plan for Biodiversity 2011-2020, national biodiversity strategies and action plans and assess progress towards the Aichi Biodiversity Targets and related national targets. Indeed monitoring is an obligation of Parties to the Convention (Article 7(b)), and the global monitoring of biodiversity is identified in the Strategic Plan as one of a number of key elements to ensure its effective implementation. Specifically paragraph 25 (a) of the Strategic Plan notes that "work is needed to monitor the status and trends of biodiversity, maintain and share data, and develop and use indicators and agreed measures of biodiversity and ecosystem change.

Well-designed monitoring activities are essential for the development of stable indexes and indicators of biodiversity patterns and changes. Combining these contemporary streams of data with historical sources also offers the possibility of modelling longer-term changes.

Biodiversity is typically monitored on a site, community or species basis. Standardized or harmonized monitoring protocols across sites are being used in networks of long-term ecological monitoring sites[56]. Typically, site-based measurements are integrated with data from other sources and observation platforms and modelled to yield data products that are meaningful for management and decision making (for example see Box 19.1).

The Global Biodiversity Information Facility and the Group on Earth Observation Biodiversity Observation Network (GEO BON[57]) are promoting and facilitating the harmonization of disjointed observation efforts, their integration into observation networks at different scales, and the linking of *in situ* and remotely sensed information. GEO BON thereby envisages a coordinated, global network that gathers and shares information on biodiversity, provides tools for data integration and analysis, and contributes to improving environmental management and human well-being. Efforts thereby focus on observing and analysing changes in biodiversity over time. It seeks to achieve this through regional and national biodiversity observation networks[58], the promotion of tools and resources for their development and operation[59], and the identification of a limited set of Essential Biodiversity Variables (Pereira *et al.*, 2013) that would improve the efficiency of monitoring by focusing observations on a limited number of key attributes.

Footnotes

[56] *See for example http://www.lternet.edu/, http://www.ilternet.edu/, http://www.neoninc.org/*

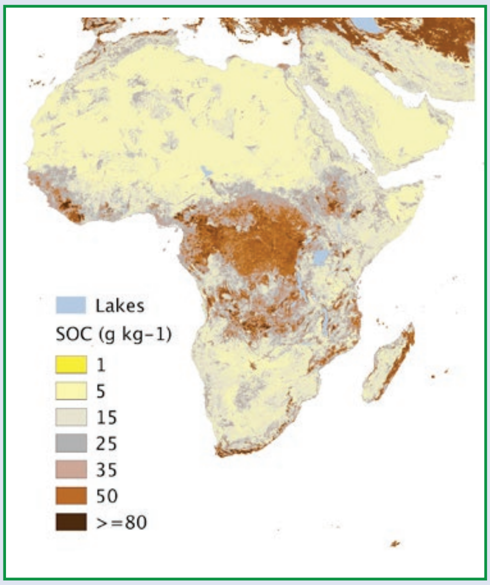
[57] *http://www.earthobservations.org/geobon.shtml*

[58] *Regional networks have been established in the Arctic, Asia/Pacific and Europe and are developing in other regions.*

[59] *At its seventeenth meeting the Subsidiary Body on Scientific, Technical and Technological Advice noted that a toolkit ("BON-in-a-Box") that can be tailored to national and regional needs would fill a major gap (see http://www.cbd.int/doc/meetings/cop/cop-12/official/cop-12-02-en.doc)*

Box 19.1: Landscape based approaches for assessment of ecosystem health at multiple spatial scales

Comprehensive information is needed to understand habitat loss and degradation and to design interventions to reduce them. In the past, this has necessitated considerable soil sampling and laboratory analysis to build up a picture of the health of the ecosystem being studied. The World Agroforestry Centre is leading the development of techniques that are being used to catalogue and map ecosystems health across the global tropics. The Land Degradation Surveillance Framework (LDSF) database currently holds detailed data from more than 20,000 plots sampled from 125 sites in more than 30 countries. The sites cover major climatic zones, different agricultural systems, natural forests and protected areas. The database provides access to important metrics needed to assess the current health of the ecosystem. The development of the database has been made possible by recent advances in laboratory techniques (including the rapid assessment of soil characteristics using infrared spectroscopy), improved satellite remote sensing technologies and advances in statistical techniques including data mining from complex data sets. The information available from LDSF is rich in nature including bio-physical data on land use, vegetation, vegetative cover, biodiversity, soil health and risk of land degradation linked with socio-economic data all mapped together and spatially-referenced. LDSF is a unique asset for land management planning and understanding the factors that are driving land degradation[60].



Map showing soil organic carbon (SOC) concentrations and soil erosion hotspots for Africa in 2012

Advancing biodiversity research in support to decision making

Knowledge on biodiversity has increased tremendously over the last 20 years, and research priorities have evolved with this increasing knowledge from the production of knowledge on biodiversity per se (i.e., which species exist, where and how they evolve) and identification and understanding of the drivers of biodiversity changes, to a greater understanding of the role of biodiversity in ecological processes, and their relations to the production of ecosystems goods and services. Although a considerable amount of knowledge still needs to be produced in these fields, the achievement of the Aichi Biodiversity Targets requires a more integrated and interdisciplinary research that focuses on socioecological systems, and contributes to a better understanding of the role of human behaviour in shaping biodiversity and its uses. This research will inform decision-making to move from a loss of biodiversity and natural resources to sustainable use of these resources, and then human population development and human well-being. Over the last 20 years, DIVERSITAS[61], through its core projects and cross-cutting networks, has been instrumental in bringing the biodiversity and ecosystem research community together in a coordinated manner, sparking novel biodiversity and ecosystem research of societal relevance, and informing the science-policy processes on a number of topics. Research conducted within the DIVERSITAS networks informs global assessments such as the GBO and IPCC reports, and will provide valuable input into the IPBES assessments.

Footnotes

[60] *Vågen, T.-G., Davey, F., Shepherd, K.D., 2012. Land health surveillance: Mapping soil carbon in Kenyan rangelands, in: Nair, P.K.R., Garrity, D. (Eds.), Agroforestry - The Future of Global Land Use. Springer, pp. 455–462.; Vågen, T.-G., Shepherd, K.D., Walsh, M.G., Winowiecki, L.A., Tamene Desta, L., Tondoh, J.E., 2010. AfSIS Technical Specifications - Soil Health Surveillance, Africa. CIAT (the AfSIS project), Nairobi, Kenya.; Vågen, T.-G., Winowiecki, L.A., Abegaz, A., Hadgu, K.M., 2013. Landsat-based approaches for mapping of land degradation prevalence and soil functional properties in Ethiopia. Remote Sens. Environ. 134, 266–275.*

[61] *www.diversitas-international.org*

Tools in support to decision making

A variety of policy support tools and methodologies have been developed under the Convention on Biological Diversity and its Protocols. These tools and methodologies complement the guidance provided through the Convention's various programmes of work and are designed to facilitate their implementation. In addition to policy support tools and methodologies developed under the Convention, a large number of relevant tools and methodologies have been developed by Parties and other partners.

At its seventeenth meeting the Subsidiary Body on Scientific, Technical and Technological Advice identified among the key scientific and technical needs related to the implementation of the Strategic Plan for Biodiversity 2011-2020 the need for better integration of science and policymaking and for improved science-policy interfaces, particularly at local and national levels and through the use of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, and the improved and wider use of tools to promote policy coherence and policy evaluation and to produce scenarios and options relevant to policymakers. The meeting further noted the need to foster improved scientific and technical cooperation among Parties, scientific networks and relevant organizations, in order to match capabilities, avoid duplication, identify gaps and achieve efficiencies and the need to enhance the clearing-house mechanism of the Convention to make scientific and technical cooperation more effective[62].

Indicators are recognized as a key tool to support decision making. The Biodiversity Indicators Partnership, launched in 2007 to support the assessment of progress towards the Strategic Plan of the Convention on Biological Diversity, is adding indicators and partners to eventually reflect all targets of the Strategic Plan for Biodiversity 2011-2020. However, global indicators are not suitable to support national decision making processes. Through an online reporting tool as part of the Convention's clearing-house mechanism, Parties will report progress made towards national targets, how these relate to the global Aichi Biodiversity Targets and which indicators or other assessment methods are used to track progress. Information on indicators in use at national level is expected to enable targeted capacity building and the identification of commonalities and synergies with regard to underlying datasets and methodologies.

The establishment of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) is a major step towards enhancing biodiversity knowledge and facilitating the use of biodiversity-related information, including information from different knowledge systems, in decision making processes through assessments at various levels. It will also develop policy support tools and methodologies for scenario analysis and modelling and guidance for making these policy tools relevant for policy-making. Moreover, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services develop guidance for the production and integration of assessments from and across all levels. It is envisaged that the processes leading to and the outcomes from these practical activities will make a significant contribution to an enhanced science-policy interface and both strengthen the contribution of science in decision making and improving decision making processes through the use of appropriate types of information.

Significant progress is also being made on advancing approaches to ecosystem services valuation and on developing ways to integrate these into national accounting and reporting systems though there is still limited ability to monitor ecosystem services trends and use this information for planning purposes, e.g. in impact assessments. Nevertheless, methodological advances are being made through inter alia The Economics of Ecosystems and Biodiversity (TEEB) initiative[63] and the increasing number of TEEB country studies as well as the Wealth Accounting and the Valuation of Ecosystem Services (Waves) partnership[64]. Volume 2 of the revised System of Environmental-Economic Accounting (SEEA) contains a framework for experimental ecosystem accounts, which will help provide a better understanding of the market and non-market goods and services provided by ecosystems[65]. Indicators to evaluate resilience in production landscapes and seascapes have been developed through the International Partnership for the Satoyama Initiative[66]. A research agenda on the development and testing of ecosystem accounting is underway.

Footnotes

[62] *See documentation under http://www.cbd.int/doc/?meeting=SBSTTA-17, in particular documents UNEP/CBD/SBSTTA/17/2, UNEP/CBD/SBSTTA/17/2/ADD1, UNEP/CBD/SBSTTA/17/2/ADD2, UNEP/CBD/SBSTTA/17/2/ADD3, UNEP/CBD/SBSTTA/17/2/ADD4, as well as the report of the meeting (http://www.cbd.int/doc/meetings/cop/cop-12/official/cop-12-02-en.doc)*

[63] *http://www.teebweb.org/*

[64] *www.wavespartnership.org/*

[65] *http://unstats.un.org/unsd/envaccounting/workshops/int_seminar/note.pdf*

[66] *https://satoyama-initiative.org/wp/wp-content/uploads/2013/08/Indicators-of-resilience-in-sepls_ev.pdf*

19.1.2 Projecting forward to 2020

The outcomes of the seventeenth meeting of the Subsidiary Body on Scientific, Technical and Technological Advice list a range of scientific and technical needs for Aichi Biodiversity Targets 1-15. Many of these call for action under Target 19 which is understood as enabling progress on all other targets. The importance of advancing implementation of Target 19 is therefore generally understood and significant progress can be expected in this field by 2020. This is confirmed by the analysis of NBSAPs and National Reports in the next section.

The implementation of the Convention benefits from a large number of highly competent institutions and experts willing to assist partners who have specific scientific and technical needs. The Global Biodiversity Informatics partnership has developed the Global Biodiversity Informatics Outlook (GBIO[67]) as a framework and concept to promote access, use, analysis of primary data and distilling policy-relevant information[68]. It identifies the need for organized activity to ensure 1) that relevant data are made available and published through stable open repositories, 2) that all historical and contemporary streams of biodiversity data are published in standardized digital formats, 3) that all of these streams are combined and organized to deliver comprehensive, evidence-based data sets, and 4) that models are developed to exploit all of these data and deliver necessary indicators. The Decadal View of Biodiversity Informatics[69] discusses many of the necessary technical steps. Ongoing implementation of this framework is expected to provide another significant element to the achievement of Target 19. GEO BON provides a broader framework for long-term monitoring of biodiversity (at different levels i.e., genetics, species, ecosystems) jointly with monitoring other environmental factors allowing for deeper analyses of the drivers and consequences of biodiversity changes over time.

The implementation of the IPBES work programme will provide a tremendous push, both conceptually and in terms of knowledge, assessments and underlying research efforts. It is thus expected to significantly contribute to the achievement of Target 19.

Finally, the launch of the new scientific initiative Future Earth[70] by the Alliance for Global Sustainability, that integrates ICSU's global change programmes DIVERSITAS, IGBP and IHDP, and collaborates with WCRP, coordinates and provides research opportunities not only on biodiversity and its drivers of change, but also on the role of biodiversity and ecosystem services for global sustainability and human well-being.

More generally, the amount of data and knowledge available is increasing at a tremendous rate. This can lead to a challenge of having too much rather than too little information ("big data"). New analytical tools and other new technological advances are continually being developed to facilitate distilling useful information from the mass of underlying data and enabling growth in knowledge.

The scope of Target 19 is immense. The potential exists for major Internet companies to make a significant contribution to biodiversity knowledge for the planet if one or more of them could be engaged to apply its technological strengths to delivering a global online "Biodiversity Knowledge Service", enabling the addition and presentation of biodiversity knowledge items to specific locations in the world.

The target formulation does not include a quantitative element. Evidence suggests that the potential exists for achieving Target 19 by 2020 (see Dashboard assessment in section 6 below).

This assessment is supported by projected trends in scientific research and transference of scientific knowledge through an analysis of scientific publications on the topic of biodiversity (Figure 19.6).

Footnotes

[67] *http://www.biodiversityinformatics.org/*

[68] *http://www.gbif.org/resources/2251*

[69] *http://www.biomedcentral.com/1472-6785/13/16*

[70] *www.futureearth.info*

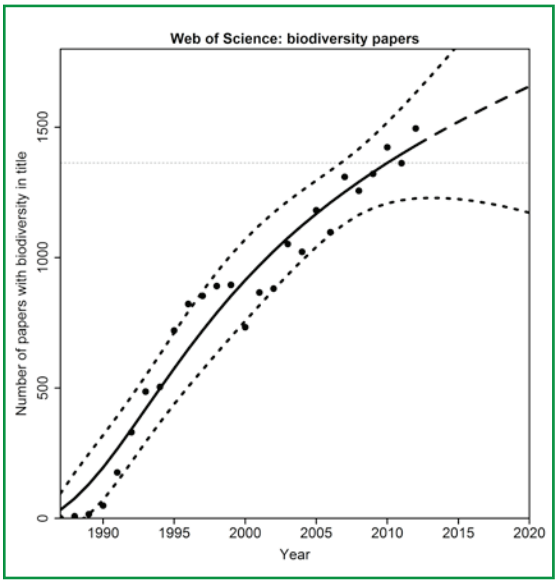


Figure 19.6. Modelled trend in the Knowledge transfer (number of biodiversity papers published over time) 1980-2013 and statistical extrapolation from 2013-2020. The trend suggests a non-significant increase in the underlying trend between 2010 and 2020. Solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant. Searches for the word 'biodiversity' in the title of the publication were undertaken through the Web of Science search engine. Each year was specified and the number of manuscripts published per year were recorded. Searches were undertaken from 1970 to 2013, but only searches from 1980 onwards (around the time of the first use of the term biodiversity) produced any records. Source: http://thomsonreuters.com/thomson-reuters-web-of-science/

19.1.3 Country actions and commitments[71]

Almost all of the national biodiversity strategies and action plans examined have targets, or similar commitments, relevant to this target. These targets are generally in line with Aichi Biodiversity Target 19. Further many targets or similar commitments, which are related to other Aichi Biodiversity Targets, refer to the need to increase the amount of biodiversity knowledge that is available and/or to conduct specific research activities. Overall, the targets which have been set in relation to Aichi Biodiversity Target 19 focus both on increasing the available amount of information related to biodiversity and on improving the quality and understanding of this information. However, the national targets that have been adopted appears to have placed relatively less emphasis on issues related to developing mechanisms to make better use of biodiversity information and tools in decision making. Such mechanisms depend on application of appropriate statistical methods, models, and scenarios in addition to use of the best and most complete available data.

Many national targets address issues that are not explicitly contained in Aichi Biodiversity Target 19. For example a number of countries, including Belarus and Belgium, have elements in their national biodiversity strategies and action plans related to the development or further strengthening of monitoring systems for biodiversity or specific elements of biodiversity. Some countries have also set targets in relation to the identification of national biodiversity research priorities. For example, Australia has set a target of having nationally agreed science and knowledge priorities for biodiversity conservation guiding research activities by 2015.

A number of national strategies also contain commitments related to making relevant information more available. For example, a few countries, including Japan and Timor Leste, refer to the further development or enhancement of their national clearing-house mechanisms. A number of countries have also established targets or similar commitments related to providing support to various organizations so that they can generate and collect biodiversity information. For example an intervention priority in Myanmar's national biodiversity strategy and action plan is to develop mechanisms for coordination and information sharing among nongovernmental organizations and academic institutions. Similarly one of the strategic lines in Colombia's national biodiversity strategy and action plan relates to the inclusion and harmonization of biodiversity and ecosystem services research priorities in national policies and plans related to science, technology and innovation.

Footnote

[71] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

19.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

19.2.1 Actions

Biodiversity-related information is vital to identify threats to biodiversity and determine priorities for conservation and sustainable use. Given this, progress towards this target can contribute to the attainment of the other Aichi Biodiversity Targets. This target is a general commitment to increase the amount and quality of biodiversity relevant information and technologies available, to make better use of these in decision-making, and to share them as widely as possible. Some data and information on biodiversity are being shared much more widely through initiatives promoting and facilitating free and open access to digitized records from natural history collections and observations, including through citizen science networks; however, much data and information remain inaccessible and capacity is lacking to mobilize them in many countries. Against this background, possible key actions to accelerate progress towards this target include:

- Developing inventories of existing biodiversity information as a means of identifying knowledge gaps and defining research priorities and making greater use of existing national and international research networks to help address these;
- Strengthening and promoting the further mobilization of and access to data by, for example, encouraging the use of common informatics standards and protocols, promoting a culture of data-sharing (for example, through requirements on publicly-funded research projects and academic recognition for publication of datasets), investing in digitization of natural history collections and promoting citizen scientists' contributions to the body of biodiversity observations;
- Facilitating the use of biodiversity related information by decision makers at national and local levels;
- Establishing or strengthening monitoring programmes, including monitoring of land-use change, providing near-real time information where possible, in particular for "hotspots" of biodiversity change;
- Engaging local and indigenous communities (*Target 18*) as well as relevant stakeholders in information collection and use;
- Supporting communities of practice and stakeholders in relevant skill fields, and strengthening cooperation among relevant national institutions, national and regional centres of expertise in biodiversity; and other relevant stakeholders and initiatives; and
- Improving national, regional and international clearing-house mechanisms, strengthening thematic information-based services and establishing interconnections in order to contribute to the development of a global biodiversity knowledge network.

The Convention's cross-cutting issue on identification, monitoring, indicators and assessments can provide a starting point for work towards this target. Similarly the clearing-house mechanisms[72] and the Global Taxonomy Initiative provide guidance on actions that can be taken to implement this target.

Promote generation of and access to biodiversity observation data

This can be achieved through:

- Promoting a common observation framework as a basis for biodiversity observing systems;
- promoting and facilitating the interoperability of data sets;
- advancing the digitization of historic biodiversity collections;
- systematically providing free and open access to publicly funded data;
- systematically promoting the use of data and metadata standards across disciplines;
- development of cost effective species identification methods for use *in situ* and *in silico*;
- promoting citizen scientists' contributions to the body of biodiversity observations; and
- encouraging major Internet companies to provide online services that enable and accelerate world-wide sharing of biodiversity information and knowledge.

Footnote

[72] *Through decision X/15 the mission, goals and objectives of the clearing-house mechanism for the period 2011-2020 were adopted by the Conference of the Parties. Document UNEP/CBD/COP/11/31 outlines the proposed work programme for the clearing-house mechanism in support of the Strategic Plan for Biodiversity 2011-2020.*

Promote appropriate monitoring programmes

This can be achieved through:

- developing cost-effective monitoring programmes (i.e., monitoring as little as possible to get the necessary answers, detect hotspots of biodiversity change or the impacts of pressures, use and management regimes);
- incorporating standardized concepts such as Essential Biodiversity Variables (EBVs) to maximize the compatibility and usefulness of data collected;
- engagement of citizen scientists, including knowledgeable local and indigenous experts and those working in primary industry, through well-planned *in-situ* monitoring schemes; and
- promoting and applying speedy and inexpensive species identification tools.

Improve assessment and knowledge management

This can be achieved through:

- making greater use of, and improving access to, case studies, tools and guidance taken from national reports, other submissions by Parties and organizations, and workshops; and
- making better use of evidence-based assessments of the effectiveness of various approaches to implementing the Convention by drawing upon the case studies, tools and guidance referred to above, along with academic publications, project reports, etc., thereby linking general conclusions emerging from the scientific literature with a wider range of grass-roots experience.

Develop networks of institutions:

This can be achieved through:

- fostering communities of practitioners in relevant skill fields (e.g., risk assessment, strategic environment assessment, economic valuation, systematic biodiversity planning, taxonomy and biological collections), building upon existing networks and professional associations;
- strengthening cooperation among relevant national, regional and global institutions and partners, including members of the Consortium of Scientific Partners, with a view to promoting exchange within and among regions and identifying possible models for the further development of institutions at the national and regional levels; and
- promoting the participation of institutions and experts in international cooperative initiative such as GEO BON, GBIF, and Future Earth, avoiding effort duplication and favouring sharing of tools, knowledge.

Enhance the Clearing-house Mechanism of the CBD

This can be achieved through:

- modernizing the architecture and enhancing the information technology capabilities of the Secretariat to provide Parties with the best possible service, drawing upon best practices;
- developing clearing-house functionalities that facilitate inputs of information by Parties and partners as well as networking and collaboration among them;
- developing a facility within the clearing-house mechanism to enable Parties to express their specific technical and scientific needs and to enable Parties and scientific networks, relevant organizations and funding bodies to indicate their areas of competence and expertise, thereby facilitating the matching of needs and capabilities; and
- building more effective national clearing-house mechanism nodes, such as by including activities related to the enhancement of national clearing-house mechanism as a component of countries' GEF-funded enabling activities.

Develop and promote thematic pilot activities

This can be achieved through:

- facilitating access to tools, information and expertise though international institutions and specialized networks such as the Global Oceans Biodiversity Initiative, the Global Invasive Species Information Partnership, the Global Taxonomy Initiative, or the Global Partnership for Plant Conservation among others; and
- promoting activities such as the Global Invasive Alien Species Information Partnership efforts in making global information on introduced and invasive species available from a range of distinct data holdings (IUCN, CABI, DASIE, NOBANIS), reviewing the data by experts, including with the help of molecular identification tools such as DNA barcoding, and data mobilizing projects of GBIF in the Pacific, Africa and Caribbean regions.

Develop regional and sub-regional pilot activities

This can be achieved through:

- building upon the experience, expertise and knowledge bases of existing national and regional institutions working on biodiversity, within an appropriate enabling framework that would involve relevant regional cooperation organizations with the aim of facilitating technical and scientific cooperation at various levels, through, inter alia: access to good practice cases, tools and methodologies; regional networking and help desks; training workshops; and the direct exchange of experts;
- strengthening information systems and knowledge management;
- promoting and enhancing networks of institutions;
- financial support; and
- a political or institutional governance framework.

19.2.2 Costs and Cost-benefit analysis

Target 19 is an enabling target that benefits the achievement of all other targets. Investing in the achievement of Target 19 is therefore a cost-effective strategy and will underpin progress in all relevant multilateral environmental agreements and many regional agreements. Many of the other targets may be unachievable, and it may be impossible to assess progress toward most targets, unless significant advances are achieved toward Target 19. Small catalytic funding can make significant advances in enhancing collaboration, networking and inter-institutional cooperation. Moreover, many of the technological advances that improve access to and sharing of information are relatively low-cost. Systematic data mining and the mobilization of existing data through web-based technologies may therefore offer a significant expansion of the pool of useful data at relatively low cost.

On the other hand, biodiversity monitoring, field research and taxonomic advances are extremely expensive undertakings. It has been estimated that US$0.5 billion to $1billion per year would provide a tenfold increase in taxonomic effort globally and result in the description of all taxa within 50 years (Costello, 2013).

Projected trends in funding spent on environmental research suggest an increase, but there is a large amount of scatter in the data. Thus it is uncertain whether funding will increase, remain static, or decrease (Figure 19.7).

The discovery of new species may inspire society to protect biodiversity and basic research on biodiversity strengthen and broaden the foundation of biodiversity knowledge necessary to underpin the attainment of the Aichi Biodiversity Targets. Further efforts to reduce the cost of new species descriptions and its identification are expected. The development of molecular technologies and standardized DNA-based identification approaches, such as DNA barcoding, respond to the need for large-scale routine biodiversity identifications. However much more is needed in terms of generating knowledge and monitoring of genetic diversity or ecosystem services, and large, coordinated global research efforts are required to generate this knowledge, and to feed this into science-policy.

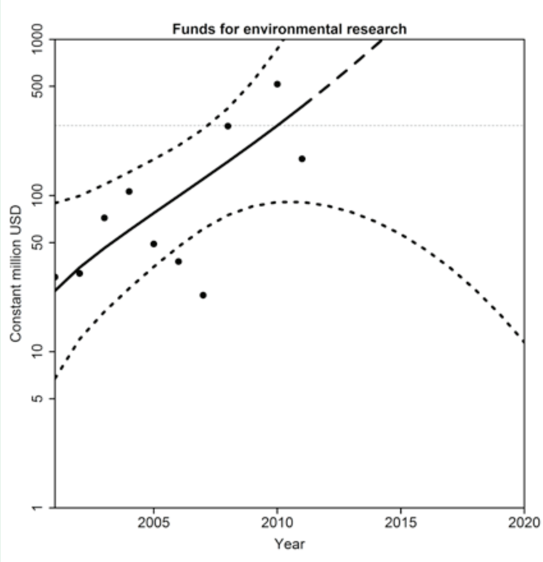


Figure 19.7. Statistical extrapolation of Funds committed to environmental education and research to 2020. Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. The trend suggests no significant change in the underlying trend between 2010 and 2020. Note the log scale on the y-axis.
Source: Tittensor et al. (2014).

19.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY IN 2020?

Target 19 is critical to enable targeted and cost-effective conservation action and the development of adequate policies and management approaches. The achievement of Target 19 is therefore expected to have significant implications for biodiversity. Moreover, the uptake of scientific and technical information in decision-making is expected to be enhanced through improved science-policy interfaces.

19.4 WHAT DO SCENARIOS SUGGEST FOR 2050 AND WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

The most important trend in the area of science and technology is the continuing information and communications revolution and its implications. The fastest computers perform trillions of operations per second at time of writing with strong signs that Moore's law will uphold up to 2020+. If continued in the future, the computers will have reached the computational power equivalent to one and possibly all human brains before 2050. Decisions and their consequences will increasingly be simulated before being taken, identifying options and providing decision support systems to start with. The public sector agencies encourage citizen participation by developing online systems gathering together also free and timely information concerning the work of government and politicians[73].

A lot will be achieved through these technological advances. However, experimental and basic research will still be needed gather new data, and to provide data to test scenarios and models. In addition, an increase in our understanding of the interlinkages and feedbacks between human societies and biodiversity and ecosystem services is crucial to improve biodiversity conservation and sustainable use of natural resources.

19.5 UNCERTAINTIES AND DATA REQUIREMENTS/GAPS

In the longer term, scientific and technical advances that are difficult to imagine today can lead to quantum leaps in the further implementation of the ideas behind Target 19 and these would likely have significant implications for the conservation and sustainable use of biodiversity in the long term.

On the other hand, there is a degree of uncertainty regarding the implementation of the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization and fears about obstacles faced for taxonomic research and international interchange of new information about species, despite the provision in Article 8(a) that: "In the development and implementation of its access and benefit-sharing legislation or regulatory requirements, each Party shall: (a) Create conditions to promote and encourage research which contributes to the conservation and sustainable use of biological diversity, particularly in developing countries, including through simplified measures on access for non-commercial research purposes, taking into account the need to address a change of intent for such research".

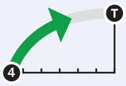
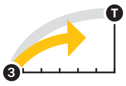
Data coverage is variable for most indicators and data sets as evidenced in the technical documents on the other Aichi Biodiversity Targets. Limitations also exist with regard to temporal coverage with limited or no time-series information for many aspects addressed in the Strategic Plan for Biodiversity 2011-2020. Where possible, future efforts should focus on the provision of spatially explicit information on biodiversity change, if possible in near-real time, and at appropriate scales for managers and policy makers.

With today's technology, large amounts of data and information can be captured and stored. A negative side of such progress is the increased risk of storing poor or erroneous data. The fact that data quality remains a major challenge should be acknowledged and addressed. Appropriate resources should be allocated to data curation and other knowledge review processes in order to succeed in achieving Target 19.

Footnote

[73] *http://ec.europa.eu/research/social-sciences/pdf/global-europe-2050-summary-report_en.pdf*

19.6 DASHBOARD – PROGRESS TOWARDS TARGET 19

Element	Current Status	Comments	Confidence
Knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved		Significant effort on delivery of information and knowledge relevant to decision makers is being made, and relevant processes and institutions are in place	High
Biodiversity knowledge, the science base and technologies are widely shared and transferred and applied		Improvements in analysis and interpretation of data gathered from disparate collecting and monitoring systems. However, coordination to guarantee models and technologies that can integrate this knowledge into functional applied systems needs to be improved	Medium

Author: Robert Höft with contributions from David Cooper, Junko Shimura, Olivier de Munck, Kieran Mooney, Scott Miller, Alex Borisenko, Sujeevan Ratnasingham, Robert Hanner, Olaf Banki, Tim Robertson, Alan Paton, Anne-Hélène Prieur-Richard, Fawziah Gadallah, Donald Hobern, Cornelia Krug

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Vernooy R, Haribabu E, Muller MR, Vogel JH, Hebert PDN, *et al.* 2010. Barcoding Life to Conserve Biological Diversity: Beyond the Taxonomic Imperative. *PLoS Biol* **8**(7): e1000417. doi:10.1371/journal.pbio.1000417

TARGET 20: FINANCIAL RESOURCES

By 2020, at the latest, the mobilization of financial resources for effectively implementing the Strategic Plan 2011- 2020 from all sources, and in accordance with the consolidated and agreed process in the Strategy for Resource Mobilization, should increase substantially from the current levels. This target will be subject to changes contingent to resources needs assessments to be developed and reported by Parties

PREFACE

Many countries indicated in their fourth national reports that limited capacity, including financial capacity, was a major obstacle to the implementation of the Convention. Recognizing this obstacle, the Parties to the Convention on Biological Diversity adopted Aichi Biodiversity Target 20, the overall purpose of which is to increase the amount of financial resources available to implement the Strategic Plan for Biodiversity.

The Strategy for Resource Mobilization, adopted by the Conference of the Parties at its ninth meeting, provides guidance on the needed international and domestic action on biodiversity finance. Preliminary targets for resource mobilization were adopted by decision XI/4 of the Conference of the Parties, which include, inter alia, the doubling, by 2015, of international biodiversity finance against a 2006-2010 benchmark, the development of national financial plans for biodiversity, and the reporting of domestic biodiversity expenditures as well as of funding needs, gaps and priorities.

Furthermore, Article 20 of the Convention provides that each Party to the Convention should undertake to provide, in accordance with its capabilities and national plans, priorities and programmes, financial support and incentives for national activities, which are intended to achieve the objectives of the Convention. The Article also specifies that developed countries will provide new and additional financial resources to enable developing countries to meet the incremental costs of them fulfilling their obligations under the Convention and that the extent to which developing country Parties will effectively implement the Convention will depend on this commitment.

Due to a lack of comprehensive data availability, it is difficult at local and global levels to estimate the financial resources required to achieve the goals and targets of the Strategic Plan as well as the size of existing financial resource flows to biodiversity from all sources. Consequently it is challenging to put a number on the global biodiversity finance gap. However the estimates that exist, such as from the first report of the high-level panel on biodiversity finance, generally indicate that the shortfall is significant, and quite possibly of an order of magnitude.

Correspondingly, assessing progress towards the attainment of this target is equally challenging. Available information, in particular on some funding streams, allows for some limited assessments. At COP-11, Parties agreed to use a preliminary framework and associated methodological and implementation guidance (COP/11/14/Add.1) to report on and monitor resources mobilized for biodiversity at national and global level. Through this process, it is expected that additional information on biodiversity financing will eventually become available and allow for more detailed estimates to be developed.

20.1 ARE WE ON TRACK TO ACHIEVE THE 2020 TARGET?

20.1.1 Status and trends

Biodiversity financing is multifaceted and what constitutes biodiversity, or biodiversity-related, funding is not always clear. However generally, two broad sources of biodiversity financing can be identified: international flows of financial resources and domestic funding.

20.1.1.i International Flows of Financial Resources

In the context of Aichi Biodiversity Target 20 international flows of financial resources constitute those resources which are provided by one country or organization to another country. International financial flows can be delivered through a variety of mechanism including official development assistance (ODA), non-ODA public funding, (also called "other official flows") as well as funds provided through NGOs, foundations and academia and the private sector. International financial flows can also include cooperation among developing countries (South-South cooperation). With the exception of ODA, there is relatively little comprehensive global information that allows assessing the status and trends in the funding delivered through these channels.

Official Development Assistance is financing "administered with the promotion of the economic development and welfare of developing countries as the main objective, and which are concessional in character…"(OECD 2014). ODA can be either bilateral (directly from a donor country to a recipient country) or multilateral (resources channelled through international financial institutions and United Nations organization, funds and programmes). According to some sources, total ODA (both bilateral and multilateral) remains the most significant source of finance for biodiversity in developing countries (Waldron *et al.*, 2013). Moreover, ODA can also play an important role in building the capacity in partner countries to develop plans and policies to increase domestic finance for biodiversity, and to attract and accommodate other forms of external financing for biodiversity.

Assessing tends in biodiversity-related ODA is possible through the reporting of OECD member countries of the Development Assistance Committee (DAC). Information is compiled in the OECD DAC Creditor Reporting System (CRS). Reporting started in 1998, and has been mandatory since 2007. Every aid activity reported is screened and marked using a variety of makers. With respect to the Convention on Biological Diversity, relevant activities are marked as targeting the Convention as a 'principal' objective, as a 'significant' objective, or as not being an objective.

There has been a general increase in bilateral biodiversity-related ODA. According to the information from the OECD DAC Creditor Reporting System total biodiversity related bilateral official development assistance (the combined total of significant and principle) has generally increased against a 2006-2010 baseline. However there have been year to year fluctuations. The amount of resources devoted to activities that have biodiversity marked as a principle objective has remained relatively flat between 2006 and 2012. Principal official development assistance refers to funding which is provided specifically to address issues related to biodiversity. The general increase in bilateral biodiversity related ODA exhibit between 2006 and 2012 is largely attributable to an increase in ODA marked as targeting biodiversity as a "significant" objective (see Figure 20.1). A "significant" biodiversity objective refers to those activities which have other primary objectives but which are also addressing biodiversity concerns. While there was a small decline in biodiversity-related aid in 2012, overall, aid to developing countries reached an all-time high in 2013.

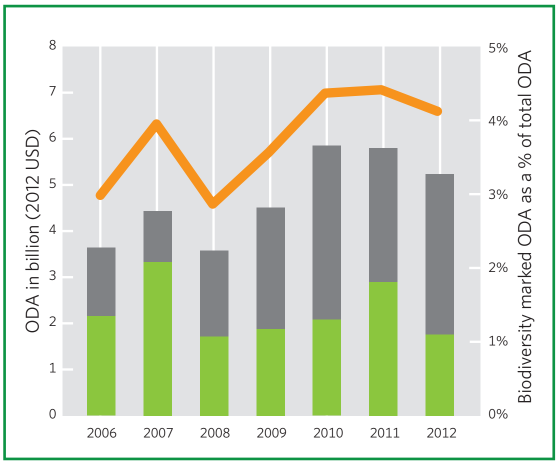


Figure 20.1. Biodiversity marked ODA between 2006 and 2012 in billion of USD (2012 constant prices) and as a percentage of total ODA. OECD Creditor Reporting System Source: Data extracted on 27 May 2014 from OECD.Stat.

Multilateral ODA is also a significant source of funding for biodiversity, however, there is limited information on the total amount of funds provided through this channel. One example of multilateral ODA is the funding provided through the Global Environment Facility (GEF). This funding supports projects in more than 140 countries to tackle a broad range of threats to the global environment. The GEF is also the main global mechanism to support developing countries' in taking action to fulfill their commitments under the world's major multilateral environmental agreements, including the Convention on Biological Diversity.

The amount of resources to the GEF has been increasing over time, with a particularly large increase between GEF-4 and GEF-5. However the amount of resources provided specifically to the biodiversity focal areas has remained relatively flat in absolute terms since GEF-3 (Figure 20.2). During the GEF-6 replenishment meeting donor countries pledged to provide US$4.43 billion to support developing countries over a four year period, in preventing the degradation of the global environment, including US$1.30 billion for biodiversity (Global Environment Facility (2014)).

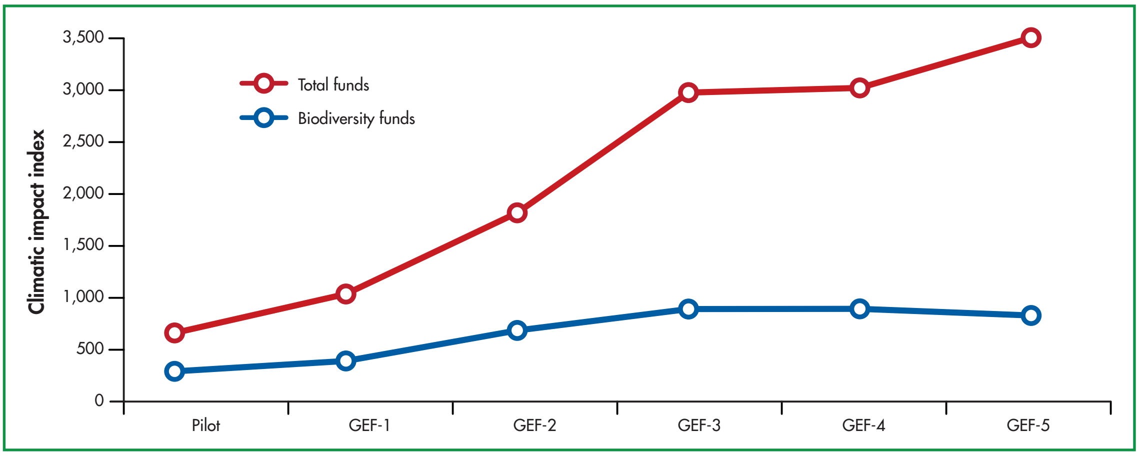


Figure 20.2. Total GEF funding and the funds for the biodiversity focal area between the pilot phase and GEF-5 expressed in millions of dollars as of September 2013 (red and blue line) Where possible mutlifocal areas funds have been disagregated and attributed to the biodiversity focal areas as appropriate. Source: Drawn from data contained in Global Environment Facility Independent Evaluation Office (2014).

20.1.1.ii Domestic funding

Domestic financial resources for biodiversity are provided through three main channels: the public sector (public budgets for government departments and agencies at all levels), the private (for-profit) sector (mediated through markets) and NGOs, foundations and academia (the non-profit sector).

There is limited information on government spending on biodiversity, in particular on spending that is indirectly related to biodiversity, making it difficult to assess the global status and trends of resources delivered through this channel. However government budgets are generally regarded as accounting for the majority of biodiversity funding(Parker *et al.,* 2012). It is important to note however there are significant differences between developed and developing economies in this regard and issues related to purchasing power parity also need to be considered

Estimates suggest that globally domestic funding in support to biodiversity is around US$20 billion a year or more. For example one estimate suggests that funding provided through general government budgets was on the order of US$25.6 billion in 2010[1] (Parker *et al.*, 2012). Of this funding approximately 78% is generated in developed economies while 22% is generated in developing economies or economies in transition. A further assessment estimated average annual expenditure on biodiversity to be on average US$21.5 billion per year for 2001-2008 (in 2005 dollars) (Waldron *et al.,* 2013)[2]. Of this amount US$16 billion could be analysed and of this US$14.5billion , or about 90%, constituted domestic funding for biodiversity. Ninety-four per cent of this funding was used in upper income countries, 6% was used in middle income countries, and 0.5% was used in lower income countries (Waldron *et al.,* 2013).

More than 30 Parties reported on domestic biodiversity funding through the preliminary reporting framework under the Convention mentioned above, sometimes making cross-references to their fifth national reports and providing, in some cases, comparative numbers. While the information does not allow for a comprehensive global assessment of domestic biodiversity funding at this stage, the evidence seems to indicate that government expenditures directly related to biodiversity tend to be small relative to total government budgets.[3]

In addition to using the existing portofolio mix of revenue sources for funding activities realted to biodiversity, government have a variety of so-called ‘innovative’ financial mechanism are at their disposal, including, according to the strategy for resource mobilization, environmental fiscal reforms, payments for ecosystem services, green markets, inclusion of biodiversity into climate finance, and others (Box 20.1).

Footnotes

[1] This assessment is largely based on information related to the establishment and management of protected area estates.

[2] This assessment drew on information from domestic and international flows of global conservation investment largely from the 1990s to 2008. The assessment looks at domestic financial flows as well as international donor disbursements. Information on spending by grass-roots conservation groups by local communities was not included as information at global level is limited.

[3] For further information see the numbers and comments provided in the submissions from India, Namibia, and Uganda, available under http://www.cbd.int/financial/statistics.shtml .

Environmental fiscal reform with a biodiversity focus provides the opportunity to use taxation, pricing measures and policies to tap into private sector financing to raise fiscal revenues, while also achieving biodiversity conservation and sustainable use goals. These measures can include taxes and charges on natural resource use (such as forestry, fishing and access to natural parks), taxes and charges on pollution, (such as those on pesticides and fertilizers), rents from resource extraction (such as on mining activities), and the reform of environmentally-harmful subsidies (such as those on agriculture and fossil fuels). Undertaking environmental fiscal reform has the potential to generate and unlock a large amount of revenue; for instance, eliminating or reforming subsidies in economic sectors are identified to be important potential contributors to to biodiversity finance (TEEB, 2011; Parker *et al.*, 2012)[4].

Box 20.1: Innovative fiancial mechanisms

To date there is insufficient information to allow for a comprehensive assessment of the status and trends of the funds mobilized by *payment for ecosystem services* schemes globally. However the evidence seems to suggest that there has been, in the last decade, a significant increase both in the number of such schemes as well as in the funds channelled through them. For example, a 2010 OECD assessment notes that there are more than 300 known PES programmes that have been implemented worldwide and that national PES programmes in China, Costa Rica, Mexico, the United Kingdom and the United States have channelled more than US$6.5 billion (OECD 2010). However, it is also noted that many of these schemes, in particular the larger ones, use some form of public funding sources, possibly combined with resources provided by international donors including NGOs. In these cases, the additional financial resouces mobilized cannot be deduced directly from the funds disbursed.

Markets for green products are a possible means to access private finance for biodiversity conservation and sustainable use. They include those that are based on ecosystem services, such as eco-tourism, goods that have been produced using production methods that have a lower impact on biodiversity then other methods (such as certified timber and certified agriculture), and goods whose consumption will have a lower impact upon the environment then standard goods of that kind (such as biodegradable detergent, TEEB, 2011). Such markets can be facilitated by eco-labelling and certification schemes, which inform consumers of the products' biodiversity-friendly qualities, and by green public procurement, which stimulates demand for these products. These markets have been growing strongly over the past few years (OECD, 2013b), and this is expected to continue. Parker *et al.* (2012) estimate that they may generate US$10.4 billion to 29.9 billion per year by 2020, up from the estimated US$6.6 billion that they are estimated to generate today.

20.1.2 Projecting forward to 2020

The information on official development assistance derived from the OECD-DAC suggests that on average total official development assistance has been increasing, though there have been considerable year to year fluctuations. While this is a positive trend, a considerable push would be needed in order for the mobilization of financial resources for effectively implementing the Strategic Plan 2011-2020 to increase substantially from all sources.

20.1.3 Country actions and commitments[5]

The need for resources for implementing national biodiversity strategies and action plans, and for ensuring the conservation and sustainable use of biodiversity more generally, is noted in the majority of national biodiversity strategies and action plans.

The targets or similar commitments that have been set in relation to Aichi Biodiversity Target 20 are, for the most part, general. They tend to refer to commitments to increase capacity to implement national biodiversity strategies and action plans. A number also note the need to create enabling environments to support the attainment of national biodiversity goals as well as

Footnotes

[4] *Cases of successful reform initiatives do exist both in developed and in developing countries; see for instance the cases collected in CBD Technical Series no. 56 and under https://www.cbd.int/financial/fiscalreform/*

[5] *This assessment is based on an examination of the national biodiversity strategies and action plans from the following countries: Australia, Belarus, Belgium, Colombia, Democratic People's Republic of Korea, Dominican Republic, El Salvador, England, The European Union, Finland, France, Ireland, Japan, Malta, Myanmar, Serbia, Spain, Suriname, Switzerland, Timor Leste, Tuvalu and Venezuela. In addition it considers the set of national targets developed by Brazil. This assessment focuses on the national targets, objectives, priority actions and similar elements included in the NBSAPs in relation to the international commitments made through the Aichi Biodiversity Targets.*

to enhance national contributions to biodiversity protection. However only a few of the NBSAPs submitted include fully-costed Action Plans.

Given that Aichi Biodiversity Target 20 calls for financial resources for implementing the Convention, from all sources, to be increase substantially by 2020, national commitments will likely need to be further substantiated if this target is to be achieved – such as through the establishment of national financial targets in the context of the development of national resource mobilization strategies – the latter being one of the global targets established by the Conference of the Parties at its eleventh meeting.

20.2 WHAT NEEDS TO BE DONE TO REACH THE AICHI TARGET?

20.2.1 Actions

Funding for biodiversity from all sources needs to be substantially increased if this target is to be met. The magnitude of the required increase is such that it is unlikely that it could be met by a single source of biodiversity funding. Therefore in addition to scaling up the biodiversity funding provided through government budgets and development assistance there is also a general need to broaden the scope of biodiversity funding. This includes exploring new innovative ways of generating funds as well as making better or more efficient use of existing funding mechanism. In this regard, not all of the needed funds need to originate from new spending. Examples of mechanisms that could be further explored are provided in the global strategy for resource mobilization, adopted by COP-9, and include fiscal reforms, payments for ecosystem services, and greening markets, among others. What mechanisms will be the most effective will depend on the specific national circumstances and priorities of each country and given this a mix of activities would likely be required. Importantly, in developing countries, funding provided by bi- or multilateral ODA can also be used to catalyze and leverage other sources of finance, both public and private, to support the conservation and sustainable use of biodiversity and ecosystem services, as part of the broader agenda to mainstream biodiversity consideration into national development strategies and priorities. Further, the use of instruments that account for the full range of values provided by biodiversity may be useful in generating the required funds.

As part of the discussions at COP-11 on the issues of financial resource several targets were adopted, which if attained by 2015 as foreseen, would contribute to the achievement of Aichi Biodiversity Target 20. These targets spell out some of the actions that countries can take to reach this target. These include developing national financial plans and reflecting biodiversity in national priorities or development plans and making appropriate domestic financial provisions, reporting on domestic biodiversity expenditures, as well as funding needs, gaps and priorities in order to improve the robustness of biodiversity funding baselines and to further guide discussions in this regard, as well as assessing and/or evaluating the intrinsic, ecological, genetic, socioeconomic, scientific, educational, cultural, recreational and aesthetic values of biological diversity and its components. The BIOFIN initiative of the United Nations Development Programme, which supports a total of 19 pilot countries in developing national finance plans for biodiversity, has developed a methodology for the preparation of such plans.

Parties have also agreed to use a preliminary reporting framework to report on and monitor the resources mobilized for biodiversity at national and global level. The information gathered through this process should help to both further clarify issues related to the status and trends of biodiversity financing as well as facilitate further discussions on this issue.

Box 20.2: India's assessment of funding support for biodiversity conservation

India has undertaken a detailed assessment of the amount of funding that it provides to biodiversity conservation. The assessment considered various sources of funding including direct core and non-core funding from the Ministry of Environment and Forests as well as indirect peripheral funding, which comprises resources that are allocated by other scientific and Ministries and departments that have an impact on biodiversity conservation. The funding provided through peripheral sources was calculated using a multiplier that expressed how directly related to biodiversity conservation the resource use was. Resources provided through state governments were also considered. The assessment found that during 2013-2014 more than 1.48 billion was spent on biodiversity conservation, 56% at the state level, 17% through the Ministry of Environment and Forests, and 26% through other ministries and departments at the national level. (Ministry of Environment and Forests, Government of India 2014)

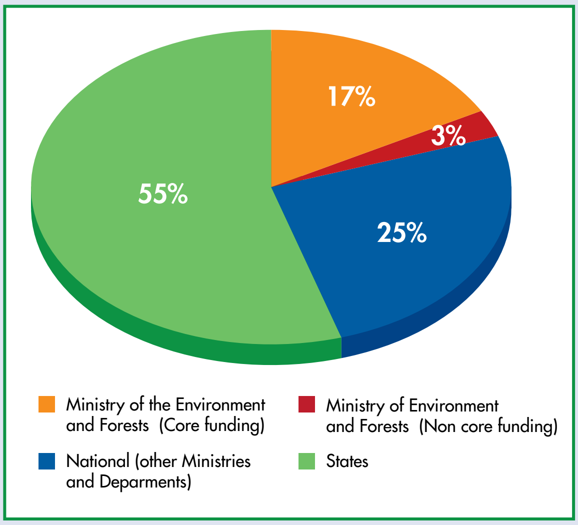


Figure 20.3. Funding provided to biodiversity conservation during 2013-2014, through different channels expressed as a percentage of total funding.
Source: Government of India (MoEF, 2014).

The information on biodiversity funding in India illustrates that while funding provided through the ministry of environment is significant it is not the largest source of funding. It also illustrates that funds for biodiversity conservation are delivered through multiple channels.

20.2.2 Costs and cost-benefit analysis

In its first assessment The High-Level Panel on Global Assessment of Resources for Implementing the Strategic Plan for Biodiversity 2011-2020 concluded that the cost of attaining the twenty Aichi Biodiversity Targets would be somewhere between US$150 billion and US$440 billion per year (High Level Panel 2012). However, the Panel also noted that these figures needed to be regarded as broad approximation of the resources required to attain the targets rather than exact estimates. The second assessment of the High Level Panel concluded that the available evidence broadly supports these estimates but that for some targets the estimates may be conservative (High Level Panel – in preparation). Both assessments came to the conclusion that most of the investments required to attain the targets will deliver multiple benefits and should not be financed from biodiversity budgets alone and that many activities could be jointly funded through budgets for agriculture, forestry, fisheries, water, pollution control and climate action as these benefits would extend to biodiversity.

Other estimates related to funding, both at the national level and in relation to specific Aichi Biodiversity Targets, have also been undertaken. These estimates generally support the conclusion that there is currently a significant funding gap related to the implement of the Strategic Plan. For example in assessing the financial costs of associated with Aichi Biodiversity Target 12, McCarthy *et al.*, (2012), determined that the over the next decade the cost of reducing the extinction risk of all globally threatened bird species was between US$0.875 billion and US$1.23 billion per year. In their assessment they also determined that of this amount 12% is currently funded. Reducing the extinction risk of other threatened species is estimated to cost between US$3.41 billion and US$4.76 billion per year. These estimates provide further evidence that biodiversity funding increases of an order of magnitude are needed to reach the Aichi Biodiversity Targets.

While significant increases in biodiversity funding are needed to reach to the Aichi Biodiversity Targets it is important to note that these resources have the potential to generate significant societal benefits. This case has been made through the various assessments of The Economics of Ecosystems and Biodiversity (TEEB) assessments (TEEB, 2010).

20.3 WHAT ARE THE IMPLICATIONS FOR BIODIVERSITY?

Recent trends and the limited information available, suggest that while some progress has been made towards this target, progress to date is not sufficient to meet the target by 2020. The limited availability of financial resources has consistently been regarded as one of the main obstacles to the implementation of the Convention. The need for resources for implementing national biodiversity strategies and action plans, and for ensuring the conservation and sustainable use of biodiversity more generally, is noted in the majority of national biodiversity strategies and action plans. It is also noted as an obstacle in many national reports. Further the attainment of Aichi Biodiversity Target 20 will have significant implications for attaining the other 19 Biodiversity Targets.

Without immediate action, the social and economic costs of biodiversity loss and the loss of ecosystem services will be felt at an accelerating rate in the future and will limit growth and stability. Investments made now will reduce resource requirements in the future. Funding for biodiversity needs to be significantly scaled up in order to secure the full range of social and economic benefits that biodiversity provides. The costs of securing these benefits would likely significantly outweigh costs. As such the costs associated with attaining this target should be viewed as part of wider investment needs for promoting sustainable development.

There is increasing evidence from both the development and biodiversity communities that poverty reduction, and development more generally, and biodiversity conservation and sustainable use, are positively linked (Roe *et al*., 2013). Further there is a growing body of evidence pointing to the potential synergies that can be achieved between poverty reduction and biodiversity and sustainable use, through development planning, policy and cooperation.

20.4 UNCERTAINTIES AND DATA REQUIREMENTS/GAPS

The information that is available on biodiversity financing is limited to certain types of data. Current information allows for reasonably robust assessments of biodiversity-related official development assistance however information on funding from other sources such the private sector, NGOs, or academia is less readily available. Nor are government expenditures that are indirectly related to biodiversity generally well known. This makes it challenging to determine the status and trends of biodiversity funding from all sources as stipulated by the target.

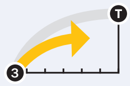
A further challenge relates to determining what constitutes biodiversity funding. Actions in support of biodiversity can take a variety of forms, and the boundaries of what can be considered as financing for biodiversity are not always distinct. Some actions are designed to have a direct and intended impact on biodiversity. In other cases, the impacts on biodiversity from certain actions can be considered as co-benefits, that is, there are benefits to biodiversity even when the actions may only be partially or indirectly aimed at biodiversity. Furthermore some actions also have unintended positive impacts on biodiversity and these expenditures are even more difficult to identify on a regular basis. This issue of funding scope makes it challenging to obtain certain estimates of funding for some types of finance, as some activities that are relevant to the objectives of the Convention and the Strategic Plan have different primary purposes.

There are a number of ongoing processes that could help remove some of the uncertainties and data gaps related to the monitoring of progress towards this target. This includes[6] (i) the BIOFIN initiative of the United Nations Development Programme, mentioned above; (ii) on going work at the OECD to further improve the Rio markers methodology; (iii) suggested work to further improve the resource mobilization reporting framework; and (iv) the forthcoming report of the High-Level Panel on Global Assessment of Resources for Implementing the Strategic Plan for Biodiversity 2011-2020.

Footnote

[6] *See UNEP/CBD/WG-RI/5/4 for further details on these elements.*

20.5 DASHBOARD – PROGRESS TOWARDS TARGET 20

Element	Current Status	Comments	Confidence
The mobilization of financial resources from all sources has increased substantially		Limited information on domestic funding. Pledges during GEF-6 show modest increase in absolute terms. General increase in ODA against 2006-2010 baseline.	Low

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INTEGRATED ANALYSIS OF THE 2020 STRATEGIC GOALS: TIME LAGS, INDICATORS AND INTERACTIONS

PREFACE

The overarching mission of the Strategic Plan for Biodiversity, adopted in Nagoya, Japan in 2010 is to "*take effective and urgent action to halt the loss of biodiversity in order to ensure that by 2020 ecosystems are resilient and continue to provide essential services, thereby securing the planet's variety of life, and contributing to human well-being, and poverty eradication*". Such an ambitious, yet crucial overarching mission requires the implementation of measures across several fronts, structured under five Strategic Goals and supported by the twenty Aichi Biodiversity Targets (Table 21.1).

Chapters 1 through 20 of the Global Biodiversity Outlook 4 (GBO-4) technical report provide detailed assessments of progress towards each of the individual Aichi Biodiversity Targets. However, interactions between Targets and their contribution for each Strategic Goal are sporadically treated in these chapters. The objective of this chapter is to provide an overview of progress towards the Strategic Goals using extrapolations to 2020 based on recent indicator trends and to present a detailed analysis of the interactions between Targets.

The first section examines the time lags between actions (i.e., responses) and outcomes as measured by pressure and status. When examining the relationships among strategic goals it is essential to keep in mind that there are numerous time lags involved in moving from current trends towards more sustainable stewardship of biodiversity and ecosystems. These time lags must be borne in mind when interpreting progress towards Targets and Strategic Goals and may influence how parties prioritize actions.

The second section of this chapter examines the progress towards each of the five Strategic Goals based on current trends and their statistical extrapolations to 2020. This analysis is based on 55 global indicators, from the Biodiversity Indicators Partnership (BIP) partners and other sources that are described in detail in Chapters 1-20. The goal of this analysis is to provide a broad-brush overview of progress towards the overarching mission based on the full set of extrapolated indicators.

The third section of this chapter provides a comprehensive analysis of the interactions between targets. Understanding the relations between targets is essential for developing an effective action plan because it helps to identify potential synergies and trade-offs and may also allow for increased implementation efficiency, thereby reducing the overall burden of progressing towards the achievement of the Strategic Plan for Biodiversity by 2020.

21.1 TIME LAGS BETWEEN ACTIONS AND OUTCOMES

In order to appreciate how specific actions under each of the targets may impact the overall achievement of the Strategic Plan, it is important to acknowledge and understand the likely length of time between actions being taken and outcomes being observed (noting that not all actions will be effective or sufficient to achieve the desired outcomes even at some future point in time). Even with full implementation of the Aichi Biodiversity Targets, there are incompressible time lags that make improvements in the status of biodiversity lag behind the implementation of measures of protection of biodiversity and ecosystems, or reduction in pressures. For example, it typically takes several years to decades for collapsed fisheries to show signs of recovery after fishing efforts have halted (see Chapter 6). Understanding these temporal lags is important to prioritize actions and to ensure the sustainability of the Strategic Plan itself.

Achieving the targets of Strategic Goal C is projected to have the most immediate effects on the status of biodiversity (Figure 21.1), since these targets directly aim at safeguarding ecosystems, species and genetic diversity. For example, conservation actions directed at species on the brink of extinction can have immediate effects in improving the species status. In other cases, lags between conservation actions such as the establishment of protected areas and the improvement of biodiversity status are likely to occur in some situations due to long-term ecosystem and species population dynamics. Moreover, without broader efforts to reduce pressures on biodiversity and ecosystems (e.g., climate change and pollution), these measures can only be partially and temporarily successful in halting the loss of biodiversity.

The aim of Strategic Goal B is to start reducing these direct pressures on biodiversity and ecosystems. However, broad scale reduction in some pressures may take years or even decades due to the slow pace of change in socioeconomic systems and in the biological and physical dynamics of the Earth system (Figure 21.1). For example, reducing the inputs of agricultural pollutants at global scales will require major social and technological transitions that will take time to implement, and even then many pollutants persist in the environment at high levels for decades. Still, many actions taken to achieve Targets under Strategic Goal B will have an effect on the short-term, and we can expect that their effect on biodiversity state will be, after actions associated with Strategic Goal D, the fastest to be felt.

The aims of Strategic Goal A are to initiate the groundwork necessary for the socioeconomic transitions that are required to reduce direct pressures on biodiversity. A greater understanding and appreciation of biodiversity and its benefits for all is essential for reducing direct pressures and implementing safeguards, especially since many of the actions required for the implementation of the Aichi Biodiversity Targets are not without substantial short-term costs and changes in behaviour. The Targets within Strategic Goal A are essential for attaining long-term sustainability, but in some cases they are likely to be associated with long time lags before changes in biodiversity outcomes are observed. For example, integration of biodiversity values into national and local strategies will eventually translate into more sustainable pathways of development, which in turn will decrease the pressures caused by consumption and production on biodiversity. Recognising the inherent time lags described above, Strategic Goal D aims to ensure that, in the meantime, the most essential ecosystem services are, as far as possible, not compromised. Goal D also promotes ecosystem restoration that can be initiated quickly and have short-term positive effects on biodiversity and ecosystem services, but may take several decades to reach full benefits (Figure 21.1).

Goal E contains enabling actions that can be implemented quickly to support the other Goals. They include the implementation of the Nagoya Protocol, the adoption of National Biodiversity Strategies, the protection of traditional knowledge, the development of biodiversity science and the sharing of the science between countries, and the mobilization of financial resources. These actions have mostly indirect effects on biodiversity condition by contributing for other strategic goals, and therefore have a time scale for producing effects somewhere between Strategic Goals B and A.

These time lags should be kept in mind when reading Sections 2 and 3 of this Chapter, as one potential component affecting the pattern and interpretation of results.

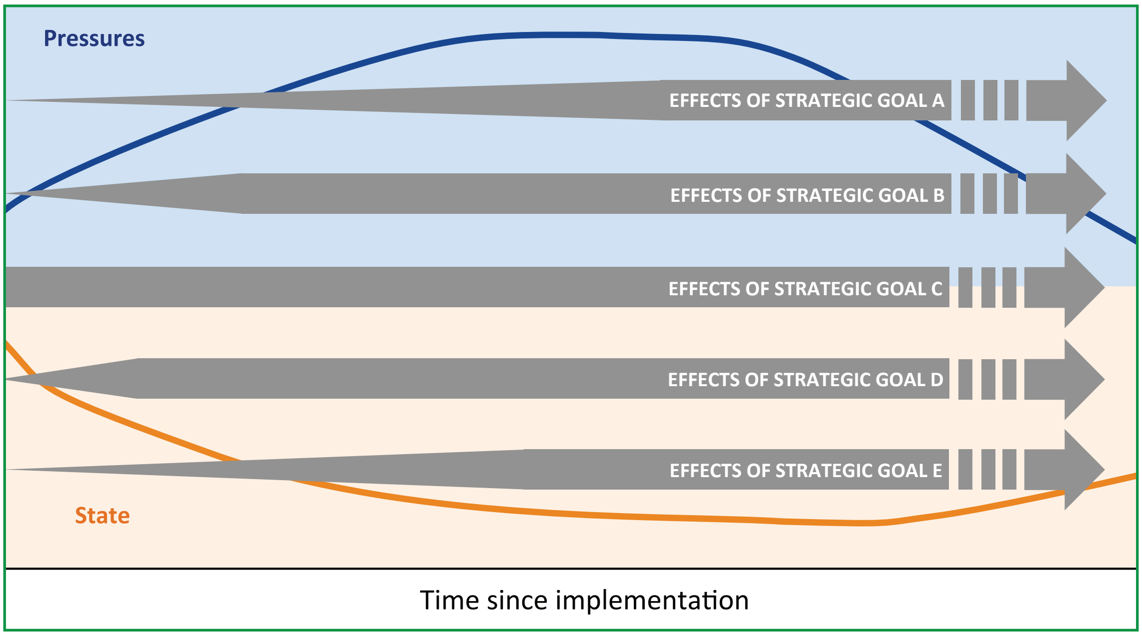


Figure 21.1. Conceptual timeline between implementation of measures to address all Strategic Goals and the outcomes of those measures. The blue line represents the evolution of the pressures on biodiversity and the orange line the evolution of biodiversity status. The grey lines represent the evolution of the outcomes of the actions under the different Strategic Goals, since implementation. The dashed ending of the arrow represents the prevalence of the outcomes.

Table 21.1. The five strategic goals and 20 targets that comprise the Aichi Biodiversity Targets. All targets start with "By 2020..." or "By 2020 at the latest..." unless otherwise noted.

Strategic Goal or Target	Abbreviated title
A Address the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society	
1	...people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably.
2	...biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.
3	...incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the Convention and other relevant international obligations, taking into account national socioeconomic conditions.
4	...governments, business and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits.
B Reduce the direct pressures on biodiversity and promote sustainable use	
5	...the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.
6	...all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits.
7	...areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.
8	...pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.
9	...invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.
10	By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning.
C To improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity	
11	...at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes.
12	...extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained.

Strategic Goal or Target	Abbreviated title
13	...the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socioeconomically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.
D Enhance the benefits to all from biodiversity and ecosystem services	
14	...ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable.
15	...ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification.
16	By 2015, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force and operational, consistent with national legislation.
E Enhance implementation through participatory planning, knowledge management and capacity building	
17	By 2015 each Party has developed, adopted as a policy instrument, and has commenced implementing an effective, participatory and updated national biodiversity strategy and action plan.
18	...the traditional knowledge, innovations and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are respected, subject to national legislation and relevant international obligations, and fully integrated and reflected in the implementation of the Convention with the full and effective participation of indigenous and local communities, at all relevant levels.
19	...knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied.
20	...the mobilization of financial resources for effectively implementing the Strategic Plan for Biodiversity 2011-2020 from all sources, and in accordance with the consolidated and agreed process in the Strategy for Resource Mobilization, should increase substantially from the current levels. This target will be subject to changes contingent to resource needs assessments to be developed and reported by Parties.

21.2 INDICATORS FOR AICHI BIODIVERSITY TARGETS BY 2020

In the preceding chapters, we provided a target by target analysis of recent trends in multiple indicators. For some indicators, we also provide projections for 2020 based on statistical extrapolations. Here, we provide some methodological details of how the projections were made and how indicators were selected. We also develop an integrated analysis of those projections by Strategic Goal.

Indicators for extrapolation to 2020 were chosen to ensure that the broadest possible range were included while still ensuring that they were sufficiently data-rich for statistical extrapolation. The following criteria were applied to select indicators for extrapolation:

(i) relevance to a target;

(ii) scientific and institutional credibility, either through publication in the peer-reviewed literature, or through having an strong institutional basis;

(iii) having an end data point after 2010, although this was relaxed where Targets had few indicators or where an indicator was particularly relevant;

(iv) having at least five data points in time; and

(v) broad geographic coverage.

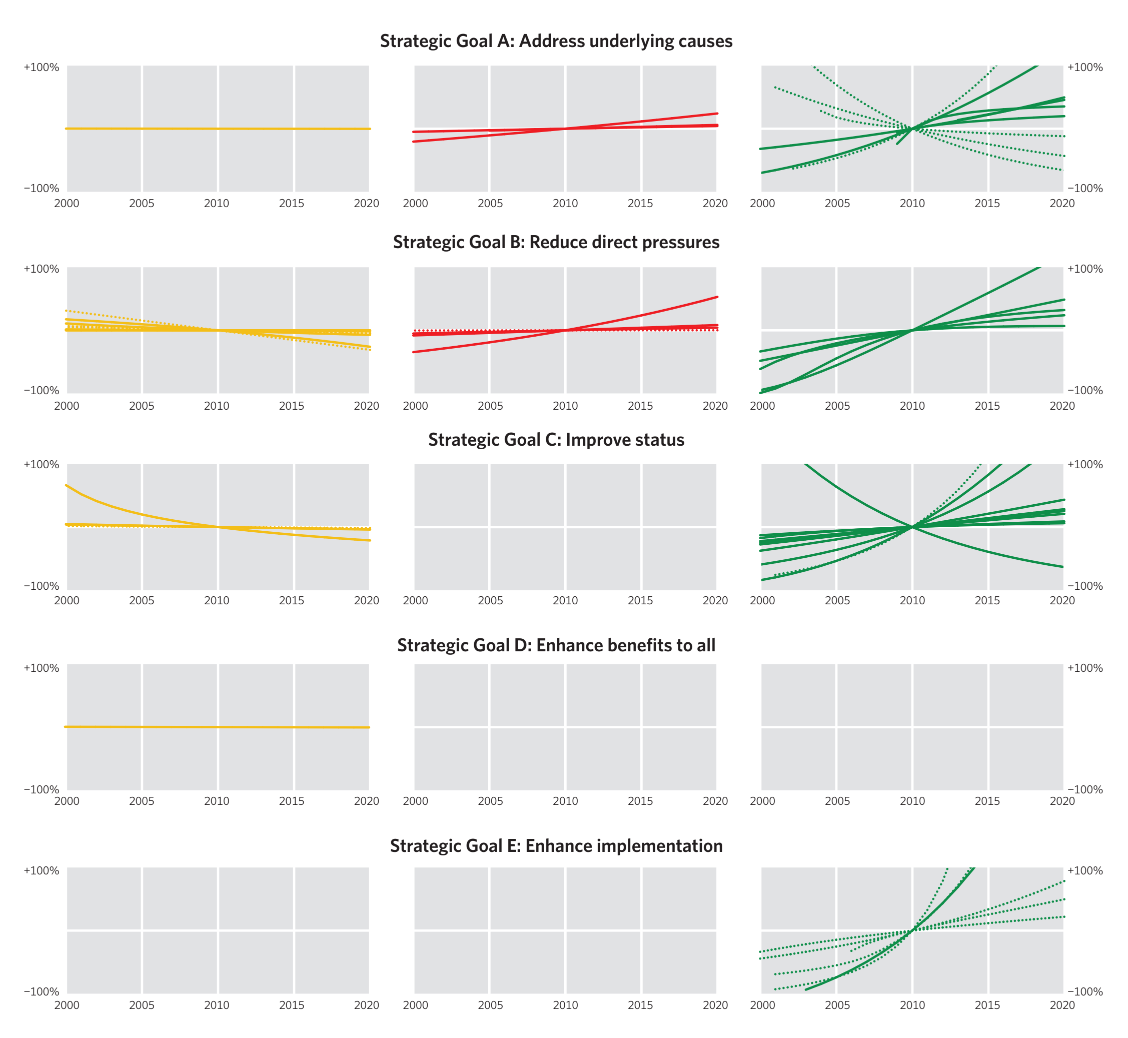
Fifty-five indicators out of a potential list of over 150 were selected as suitable for extrapolation across the 20 Aichi Biodiversity Targets. It is however worth noting that many more of those indicators not meeting the criteria for extrapolation can be used to illustrate recent trends as part of the wider assessment of progress towards particular Aichi Biodiversity Targets, and these are included in the relevant target chapters.

The analysis of the indicators is structured around the five Strategic Goals (Figure 21.2; Table 21.1). The indicators have been further divided into three groups within each Strategic Goal based on whether they provide information about pressures, states or responses. This analysis provides a more synthetic overview of progress towards the overarching mission than an analysis by individual targets. A summary of the indicators for the individual targets is provided in Annex 1.

In order to assess the progress towards 2020, the 55 indicators were extrapolated using an adaptive statistical approach. Extrapolated indicators were identified for 16 of the 20 Aichi Biodiversity Targets; indicators suitable for extrapolation could not be sourced for Targets 15, 16, 17, and 18 (Targets 16 and 17 have too short a time window to permit extrapolation). The quality of the publically available indicators to cover each aspect of the Aichi Biodiversity Targets is variable. Some reflect well the wording of the target, whilst others are weaker proxies. For instance, the indicators for Target 14 did not include the component "… taking into account the needs of the poor". Important gaps remain, with Strategic Goal D in particular being poorly covered (Figure 21.1).

There is encouraging evidence of positive action ('responses') to support biodiversity across most of the Aichi Biodiversity Targets where indicators could be extrapolated (Figure 21.1). Nineteen of the 32 response indicators showed a significant projected increase by 2020 providing evidence for notable progress on response components of the majority of targets. There are response indicators for all the Strategic Goals, suggesting that relevant actions are being taken in all Strategic Goals. Nevertheless, 12 of the 16 biodiversity state indicators exhibit significant declines, and only one state indicator shows a (non-significant) improvement. This is not surprising given that six of the seven indicators of pressures on biodiversity show a significant projected increase (i.e., rising pressure on biodiversity) by 2020 relative to the 2010 value. These indicators are primarily associated with Aichi Biodiversity Targets 4, 6, 8 and 9 (Strategic Goals A and B) which are focused on addressing the underlying causes of biodiversity loss and reducing the direct pressures on biodiversity.

We now provide an integrated analysis for each goal.



Trends in normalized indicators from 2000 and projected to 2020 for the five different CBD 2020 Goals; State measures are coloured orange, Pressure measures are coloured red, and Response measures are coloured green. The horizontal dotted line represents the modelled indicator value in 2010. For state and response indicators, a decline over time represents an unfavourable trend (falling biodiversity, declining response) whereas for the pressure indicators a decrease over time represents a favourable trend (reducing pressure). A dashed coloured line represents no significant trend, whereas a solid coloured line represents a significant projected change between 2010 and 2020. Values are normalized by subtracting the modelled mean then dividing by the modelled standard deviation. For individual extrapolations on their original scale and for more detailed interpretations of the extrapolations, see target-by-target chapters. Note that many time series include data prior to the year 2000; the x-axis has been limited to this date. Source: Tittensor *et al.*, (2014).

Goal A (Address Underlying Causes):

Goal A focuses on addressing the underlying drivers of biodiversity loss, and hence most indicators address responses. The increasing number of measures taken to promote sustainable consumption and production and increasing public awareness of biodiversity is encouraging. Overall, it appears to be possible to achieve significant progress in addressing Strategic Goal A by 2020 if current measures are continued and reinforced. One concern is the apparent stagnation in the funds available for investment in for example environmental education and environmental impact assessments.

Despite progress in actions addressing the targets, it is projected that, based on current trends, all Targets under Goal A will be missed. For some aspects of the targets, in particular harmful environmental subsidies, little or no progress has been made. In other cases, actions being taken are still insufficient. The three indicators of pressure for this goal all show increases (human appropriation of net primary productivity, water footprint, and ecological footprint) and the single indicator of state under Goal A (Red List Index for birds, mammals and amphibians showing trends driven by utilisation) is declining.

Goal B (Reduce Direct Pressures):

All response measures under Strategic Goal B are increasing – strong increases in many cases – and are projected to continue increasing to 2020, suggesting progress towards the promotion of sustainable use of biodiversity. Examples include efforts to improve the sustainability of extractive uses of biodiversity such as Forest Stewardship Council and Marine Stewardship Council certifications.

Surprisingly, there are relatively few pressure indicators for Goal B, but they are highly pertinent. Three of the four measures of direct pressures on biodiversity are all increasing, and are projected to do so until 2020. The indices include measures of trawling effort in marine fisheries, nitrogen surplus and introductions of invasive species. Overall, these indicators plus assessments of additional pressures indicates that either no progress is being made toward targets, or indicators are moving away from targets within Goal B. Given that Goal B focuses on reducing direct pressures, this analysis indicates that current responses are insufficient and that substantial additional efforts to reduce pressures will be required to start moving towards achieving the "Reduce direct pressures on biodiversity…" component of Goal B.

Given that the pressure indicators are still increasing, it is not surprising that of the 11 indicators of state for Goal B, covering habitats and species, all but one are declining and projected to continue to decline based on current trends. These indicators include declines in wetland extent, natural habitat extent, and several indexes of wild bird status.

Goal C (Improve Status):

Strategic Goal C focuses on the status of biodiversity by directly safeguarding ecosystems, species and genetic diversity. Indicators of response for Goal C include terrestrial and marine protected area coverage, representation of key biodiversity sites, and funding for species conservation and protected areas. All but funding are showing positive trends, highlighting the development of networks of protected lands and seas around the world as one of the major responses of the global community to the biodiversity crisis. This indicates that substantial efforts are being made towards the "…safeguarding ecosystems, species and genetic diversity" component of Goal C, noting that protected area designation alone without effective management does not guarantee that biodiversity will be safeguarded.

Despite the positive trends in responses, indicators of state in Goal C are declining and extrapolations suggest further declines. These include the Living Planet Index and the Red List Indices. The rate of decline in the Living Planet Index appears to be slowing, but rapid declines in the Red List Indices continue unabated. These indicate that the "Improve the status of biodiversity…" component of Goal C will not be met based on current trends. However, there is evidence of progress towards preventing extinction of bird and mammal species, although the news is bleaker for amphibians and fishes (see Target 12).

There are no indicators of pressure for Goal C, because the targets within this goal focus on improving the status of biodiversity.

Goal D (Enhance Benefits to All):

Due to the deficiency of indicators applicable under Strategic Goal D (only one indicator, the Red List Index for pollinators was available) progress towards this goal is the hardest to measure. Other data and analysis in the individual target chapters of this report suggest that many actions are being taken to restore degraded ecosystems, but little progress has been made in safeguarding ecosystems that provide key services, particularly those benefiting the poor and the most vulnerable. Indicators for other Strategic Goals can also be used as proxies for measurement of progress towards Goal B - these include Wetland habitat and Natural habitat extent. These indicators suggests that ecosystems and the services that they provide (for example, pollination) are in decline and are projected to continue to decline up to 2020.

Goal E (Enhance Implementation of the Strategic Plan):

All the indicators compiled for Goal E relate to responses and measure availability of data and knowledge, funding for conservation, and development assistance by governments. None of these indicators show significant change other than records in the Global Biodiversity Information Facility that are increasing.

Caveats and conclusions

The statistical extrapolations used in this analysis make the assumption that the underlying processes will remain on their current trajectories until 2020. If the underlying processes, for example the rate of growth of per capita consumption, which is one of the major drivers of changes in natural systems, shifts significantly from current trends between now and 2020, then the extrapolations might not be valid anymore.

It should be kept in mind that each of the indicators in this analysis is explored in more detail in the chapters on individual targets. Comparison with non-statistical projections and knowledge of underlying factors mediating trends in chapters 1-20 provides more nuanced interpretations of the likely future trajectories of these indicators.

Although the 55 indicators analysed above achieve reasonable coverage of the 20 Aichi Biodiversity Targets, they do not represent all aspects of the Strategic Goals. For example, the text of Target 1 reads "By 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably". The indicators analysed for this target provide information on people's awareness of the values of biodiversity, but no indicators are available for assessing whether people are aware of the steps they can take to conserve and use it sustainably.

A final caveat is that this analysis focuses on generalized global trends and does not account for large geographical variation in the trends of some indices at sub-global scales. Chapters 1-20 provide more detailed analyses that frequently provide addition insight into differences in regional trajectories.

Despite the incomplete coverage, the number of indicators and data sets used has increased since the Global Biodiversity Outlook 3 and associated studies (e.g. Butchart *et al.*, 2010; Leadley *et al.*, 2010) were published: the suite of 55 extrapolated indicators and other non-extrapolated indicators used to assess progress in GBO4 is greater than the 31 used to assess progress towards the 2010 Biodiversity Target in GBO3. Although the amount of data on most dimensions of biodiversity at the global scale is increasing, improving systematic collection of good quality and harmonized data is required to obtain a more complete overview of the state of biodiversity, pressures on biodiversity and our responses to its loss (Pereira *et al.*, 2013).

Despite the above caveats, there are clear messages emerging from the indicator extrapolations. These are, by goal:

Despite progress, moves towards sustainable practices do not appear sufficient to meet all Targets under Goal A.

Goal B focuses on reducing direct pressures and this analysis indicates that current responses to reduce pressures are insufficient.

Despite the positive trends in responses, indicators of state in Goal C are declining and extrapolations suggest further declines.

Despite progress towards Goal D being the hardest to measure, the indicators suggests that ecosystems and the services that they provide are in decline and are projected to continue to decline up to 2020.

Although responses are increasing, none of the indicators compiled for Goal E show significant change other than records in the Global Biodiversity Information Facility that are increasing.

Taken together, these results suggest that the impacts of responses supporting biodiversity conservation and sustainable use cannot yet be discerned in the form of reduced pressures or improved state of biodiversity – and that actions therefore need to be stepped up and accelerated if the goals of the Strategic Plan are to be achieved.

21.3 INTERACTIONS BETWEEN THE AICHI BIODIVERSITY TARGETS

The Aichi Biodiversity Targets are deeply inter-connected, but the relationships between targets vary in strength and are often asymmetric. In this section, the main interactions and synergies between the Aichi Biodiversity Targets are identified and explained. These interactions are particularly important when designing National Biodiversity Strategic Action Plans (NBSAPs). Coordinated actions that maximise the positive interactions amongst targets can potentially reduce the overall costs of implementation of a NBSAP and optimise its implementation and execution time.

21.3.1 Synthesis of interactions between Targets and Strategic Goals

To determine the potential interactions among the twenty Aichi Biodiversity Targets, a group of 18 experts (composed of GBO-4 Technical Report authors and reviewers) qualitatively assessed how the achievement of any given Aichi Biodiversity Target could influence the achievement of the other targets. The following ordinal scores were used by each expert to qualify all the target interactions in a matrix: 1 - low influence, 2 - intermediate influence, and 3 - high influence. For each entry the mode value was used as the final score of the interaction, and the relative agreement for each entry was estimated, as the percentage of experts that have chosen the mode value (Figure 21.3). Finally, for each target we calculated the difference between the sum of downstream interactions, or the effect of attaining one target on the other targets, and the sum of upstream interactions, or the effect induced by other targets in one target (Figure 21.4).

	T1	T2	T3	T4	T5	T6	T7	T8	T9	T10	T11	T12	T13	T14	T15	T16	T17	T18	T19	T20
T1		2	2	2	2	1	2	2	2	2	2	2	1	2	2	1	1	2	2	2
T2	2		3	3	3	3	3	2	2	3	3	3	3	3	3	3	3	3	2	3
T3	1	3		3	3	3	3	3	1	3	2	2	2	3	3	1	1	1	1	2
T4	1	1	3		3	3	3	3	2	3	2	2	3	3	3	1	1	1	1	1
T5	1	1	1	1		1	3	1	1	3	3	3	2	3	3	1	1	1	1	1
T6	1	1	1	3	2		1	1	2	3	1	3	3	3	1	1	1	1	1	1
T7	1	1	1	3	3	1		3	2	3	2	3	3	3	3	1	1	1	1	1
T8	1	1	1	2	1	2	2		1	3	1	2	1	3	2	1	1	1	1	1
T9	1	1	1	1	1	1	1	1		3	1	3	1	3	2	1	1	1	1	1
T10	1	1	1	1	3	2	1	1	1		3	3	1	3	2	1	1	1	1	1
T11	1	1	1	1	3	3	1	1	1	3		3	3	3	2	1	1	1	1	1
T12	1	1	1	1	2	2	1	1	2	2	1		3	2	1	1	1	1	1	1
T13	1	1	1	1	1	2	2	1	1	1	1	2		2	1	1	1	1	1	1
T14	1	2	1	2	2	3	3	2	1	3	1	2	1		3	1	1	1	1	1
T15	1	1	1	1	3	1	3	1	2	2	1	2	2	3		1	1	1	1	1
T16	2	1	1	1	1	1	1	1	1	1	1	1	3	1	1		1	3	1	2
T17	3	3	2	2	3	2	2	2	2	2	3	2	2	2	2	2		2	2	3
T18	2	2	1	2	2	2	2	1	1	1	2	2	2	3	1	1	1		3	1
T19	3	2	2	1	2	2	2	2	2	2	2	2	2	2	2	1	2	1		1
T20	3	3	2	2	3	3	3	2	2	3	3	3	3	3	3	3	3	3	3	

Figure 21.3. Strength of interactions between the Aichi Biodiversity Targets, based on expert opinion, depicted as effect of row (downstream interactions) on column (upstream interactions). Numbers indicate the mode of the strength of the relationship (1 – low, 2 – intermediate, 3 – high). For example, the impact of Target 2 (T2, integration of biodiversity values) on Target 10 (T10, protection of vulnerable ecosystems) is strong, while the impact of T10 on T2 is rather weak. Colours represent the relative agreement on the strength of the interaction. White cell – less than 50%, light green – more than 50% and less than 75%, dark green – more than 75%. Source: Authors of this chapter, produced by Alexandra Marques.

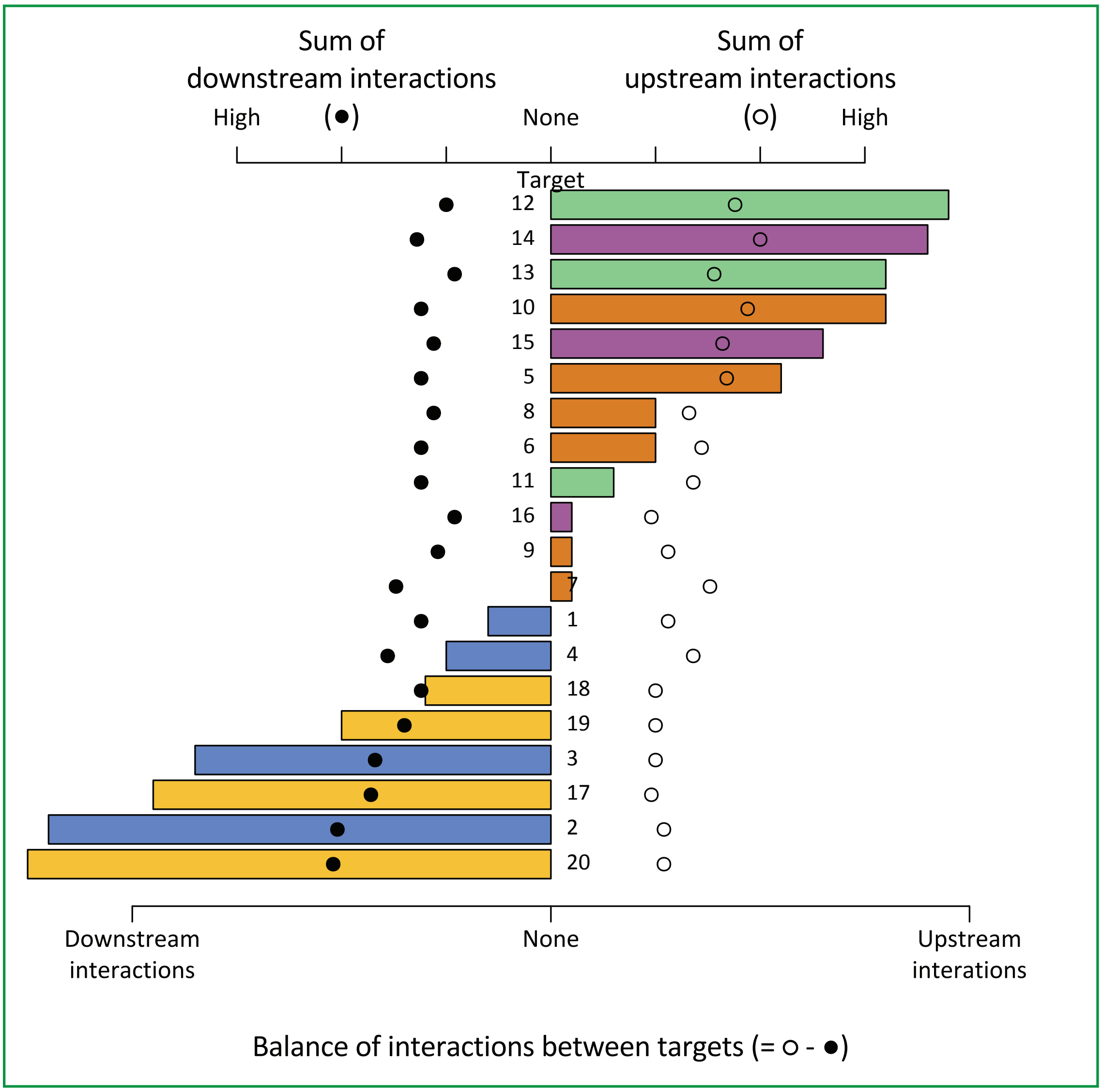


Figure 21.4. Net interactions of the different targets (bars) measured as the difference between the sum of downstream interactions, or the sum of impacts exerted on other targets (●) ; and the sum of upstream interactions, or the sum of impacts received from other targets (○) . Actions towards targets with high net downstream interactions will impact other targets, whereas targets with high net upstream interactions will be impacted by other targets. Source: Authors of this chapter, produced by Rainer Krug.

Overall, targets under Strategic Goals A, B and E (see Table 22.1 for a description of Strategic Goals and Targets) received the highest scores in terms of having a greater impact on the achievement of other targets.

All targets in Strategic Goal A are felt to have a high influence on the achievement of the other targets (i.e., have many high scores along the rows). Target 1 is considered to have an intermediate influence on 15 (out of 19) other targets, Target 2 has a high influence on 15 targets, Targets 3 and 4 have a high influence on 9 targets.

Targets under Strategic Goal B (5 to 10) are considered to have high influence on targets under Strategic Goal C and D (11 to 13 and 14 to 16, respectively). Notably, all targets under this goal are thought to have the highest impact on Target 14, and all but one have high influence on Target 12. Also, Targets 5 to 9 are felt to have a high influence on the achievement of Target 10.

Under Strategic Goal C (11 to 13), Target 11 has been assessed as having a higher impact on other targets, scoring 3 for the influence on Targets 5, 6, 10, 12, 13 and 14. In Strategic Goal D (14 to 16) Target 14 is identified as having a higher influence on other targets, scoring 3 for Targets 6, 7, 10 and 15.

Finally, for targets under Strategic Goal E (17 to 20), Target 17 is considered to have a high influence on 5 targets and an intermediate influence on all other targets, Target 20 is considered to have a high influence on 15 out of 19 targets.

When looking at the difference between upstream and downstream interactions, some targets mostly have impacts on other targets, while others are primary impacted by other targets (Figure 21.4). In particular, actions taken within Targets 20 (Financial Resources), 2 (Biodiversity Values), 17 (Adoption of NBSAPs), 3 (Incentives), 19 (Knowledge Base) and 4 (Production and Consumption) are considered to have large effects on other targets. These targets, under Strategic Goal A and E, should therefore be seen as strategically important because they influence the achievement of a broad range of Targets and Strategic Goals. For example, Targets 2 and 4 ensure that governments, business and stakeholders properly account for biodiversity, while Targets 17 and 20 aim at knowledge generation and funding for implementation. Many of the actions undertaken in Strategic Goals A and E may take substantial time to influence biodiversity status. The later two targets are of particular importance, as they provide the resources and tools required in the initial stages of implementation of the Strategic Plan (see Section 21.2.1). Target 3 which aims at reducing harmful subsidies and enhancing positive incentives could have both short-term and longer-term effects on biodiversity status.

Several targets are primary impacted by other targets (Figure 21.4). For example, Targets 12 (Species Conservation), 13 (Genetic Diversity), 10 (Vulnerable Ecosystems) 14 (Ecosystem Services), 5 (Habitat Loss) and 15 (Ecosystem Restoration and Resilience) are heavily affected by actions taken in other targets, so they benefit most from progress towards all other targets, albeit indirectly. Nevertheless, implementing actions that are directly related to a particular target (e.g., implementing policies to maintain genetic diversity of livestock, or preventing further extinctions of species) are the first, urgent steps to making progress towards these targets and are amongst the actions which will produce faster positive effects on biodiversity (Figure 21.1). The maintenance of the effects of these targets can be sustained by actions taken to achieve other broader reductions in pressure with longer-term effects. This does not mean that actions taken within these targets have little influence on other targets. For example, restoring ecosystems (Target 15) can contribute to reversing habitat loss (Target 5). Box 21.1 explores the effects a policy programme to protect essential ecosystem services (Target 14) has on other Targets. Conversely, preventing or reversing habitat loss (Target 5) can reduce species extinctions (Target 12) and relieve pressures on vulnerable ecosystems (Target 10).

21.3.2 Interactions between Targets: synergies and trade-offs

We now illustrate for selected targets how they interact with other targets, identifying important synergies and trade-offs.

21.3.2. i Examples of interactions of Target 2 on other Targets

The Targets under Strategic Goal A provide the basis for a structural change of the relationship between the socioeconomic system and the environment. Such a change can have an impact across all other Targets. An important component of Target 2 is the incorporation of biodiversity and ecosystem values (physical and monetary) into national accounts.

Australia has been in the forefront of the development of experimental biodiversity and ecosystem accounts[1] (see Target 2 and UN, 2013), the Great Barrier Reef area has been extensively covered (ABS, 2012; 2013). The Great Barrier Reef land accounts showed that since 1750 around 8 million hectares of native habitats have been lost (Target 5), due to cleared, non-native vegetation and buildings (ABS, 2012). Currently, industry occupies the largest fraction of land surrounding the reef (ABS, 2012).

Footnote

[1] *Biodiversity and ecosystem accounting is a new and emerging field dealing with integrating complex biophysical data and its changes and linking it to economic and other human activities (UN, 2013).*

Agriculture practices in the vicinity of Great Barrier Reef can increase the levels of sediment, nitrogen and phosphorous in the reef, with consequences for its environmental condition. Land accounts include information on Land Management Practices. The Mackay Withsunday Natural Resource Management region has the highest number of agricultural holdings applying fertilizers and chemicals, 70% and 82% respectively (Target 7 and 8) (ABS, 2012). Per year, the discharge of nitrogen and phosphorous to the Great Barrier Reef Lagoon is estimated to be 14.000 and 16.000 tonnes, respectively (ABS, 2012). On average 20% of the holdings in the Great Barrier Reef actively control stock access to riparian areas and 43% ensured that at least 40% of groundcover remained on the paddocks at the end of the dry season (Target 7 and 14). The biodiversity accounts assessed status in 2000 and how it has changed in 2011. They provided information on the number of introduced species (Target 9) and their change, and the threat status of native species and its change (Target 12).

By keeping track of the changes in ecosystem and biodiversity, and linking these changes to the economy, it is possible to reduce the direct pressures on biodiversity. Management practices can subsequently be shifted to follow a more sustainable course (Strategic Goal B). Information on ecosystem condition, and the status of the services provided by them, can be used to ensure adequate planning measures for the safeguarding of ecosystems at risk of degradation. This could in turn lead to an improvement of the status of biodiversity and hence enhance the benefits to all from biodiversity and ecosystem services to all (Strategic Goal C and D).

21.3.2.ii Examples of interactions of Target 3 on other Targets

Target 3 is aimed at removing incentives that are harmful to biodiversity, and to develop and apply incentives that support conservation and sustainable use of biodiversity. Removing harmful subsidies is a key mechanism for reducing overfishing (Milazzo, 1998). It has been shown that although removing subsidies might reduce total catch and total revenue, it can increase the overall profitability and the total biomass of commercially important species of North Sea fish (Heymans *et al*., 2011).

Incentive payment schemes are a common strategy to conserve biodiversity and to enhance the supply of ecosystem services (Armsworth *et al*., 2012). Payments for Ecosystem Services (PES) can be used to protect species of special interest (Target 12), for example by protecting nesting sites of endangered species (Cambodia; Clements, 2013), or vulnerable habitats (Target 10) such as biodiversity-rich pastures in montane regions (Switzerland; Huber *et al*., 2013). Elsewhere, PES agreements are used to maintain habitats that are crucial for wildlife migration (Target 5) (Tanzania; Nelson *et al*., 2010).

Bundling biodiversity with ecosystem services such as water provision or carbon sequestration can make PES even more (cost-)effective and successful (Wendland *et al*., 2010). Such integrated approaches are for example used in Madagascar, where sites high in biodiversity and standing carbon where selected for PES programmes (Wendland *et al*., 2010), or in Bolivia, where biodiversity conservation, watershed protection and water supply are linked (Asquith *et al*., 2010). In Costa Rica's PES programme, mitigation for GHG emissions, hydrological services, biodiversity and scenic beauty, all ecosystem services provided by forest ecosystems, are integrated (Pagiola, 2008).

The UN-REDD+ programme also provides the opportunity to link biodiversity to the provision of ecosystem services, as the scope of REDD includes the reduction of emissions from deforestation, reductions of emissions from forest degradation, the conservation of forest carbon stocks, the sustainable management of forests, and the enhancement of forest carbon stocks (Gardner, 2012). The programme has the potential to contribute to the achievement of a number of Aichi Biodiversity Targets. The main contribution will be made to Target 15 (ecosystem restoration), but REDD+ can also contribute to achieving Target 5 (e.g., slowing of habitat loss, increase of forest area), Target 10 (recovery of forest structure and composition), Target 11 (spatial planning that includes protected areas for biodiversity, increased forest connectivity) and Target 12 (recovery of forest composition and return of specialist forest species).

21.3.2.iii Examples of interactions of Targets 10 (Vulnerable Ecosystems), 11 (Protected areas increased and improved), 12 (Species conservation status and extinction)

Improving the status of threatened species (Target 12), and minimizing anthropogenic pressures on vulnerable ecosystems (Target 10), are the Targets perceived as benefiting most from progress achieved in all other Targets. The creation of protected areas (Target 11), has strong influence on Targets 5, 10, 12, 13 and 14, which are key targets relating to pressures (habitat loss and pressures on vulnerable ecosystems), the state of biodiversity (extinction risk, conservation status and genetic diversity) and ecosystem-service benefits.

Target 12 (Species conservation status and extinction) represents an important long-term goal of the conservation community. Actions to achieve Target 11 (increased areal extent, increased ecological representation and improved management of protected areas) can help achieve Target 12, as do actions to achieve Target 10 (reduction of anthropogenic pressures on coral reefs and other vulnerable ecosystems). In turn, since all threats to biodiversity ultimately depend on human activities, the reduction of anthropogenic pressures through the establishment of protected areas (Target 11), is expected to result in an overall improvement of the conservation status of vulnerable ecosystems (Target 10) and threatened species (Target 12). The extent of this effect depends on the effectiveness of protected areas and on the coverage of vulnerable ecosystems and threatened species by protected areas. The achievement of Target 10 can provide additional positive interactions with Target 12, where the reduction of anthropogenic pressures is not obtained through the establishment of protected areas but through other means, e.g., the reduction of fishing pressure.

Effects of Target 11 on 10

The majority of Marine Protected Areas (MPAs) are located in areas of high human impact (Halpern *et al.*, 2008; Mora and Sale, 2011). Protected areas cover 28% of the world's coral reefs, 27% of mangrove forests and 16% of seagrass beds. Ocean acidification and global warming are a major threat to coastal waters (Halpern *et al.*, 2008). MPAs can contribute to the reduction some forms of biological stress, such as overfishing, that enhance corals competition with macroalgae (Mumby and Harborne, 2010), although pressures are often displaced elsewhere rather than overall abated (Mora and Sale, 2011). Marine protected area networks can also provide temporal *refugia* that increase ecosystem resilience to acute stresses (Mumby *et al.*, 2011).

Reduction of human pressures to coastal environment will also require making the "land-sea connection" (Álvarez-Romero *et al.*, 2011; Stoms *et al.*, 2005). Strategic placement of terrestrial protected areas to protect endangered coastal habitats and minimize the runoff of pollutants and fertilizers can therefore contribute to achieving Target 10 (Beger *et al.*, 2010).

Effects of Target 10 on 12

The ability of biodiversity, and time required, to recover after threat abatement is highly context-dependent. This could limit the magnitude of the effects of protection (Target 11) to biodiversity persistence (Target 12) via reduction of direct pressures to coral reefs (Target 10), given that the benefits of protection would be mostly visible after 2020. However, in areas where the main threat to coral reefs is overfishing, some biodiversity benefits can accrue rapidly, for example in the Great Barrier Reef of Australia (McCook *et al.*, 2010).

Cold-water, or deep-sea coral reefs, complex habitats of considerable biodiversity, are likely to be at risk from ocean acidification this century. Such reefs are also extremely vulnerable to trawling (Ramirez-Llodra *et al.*, 2011) therefore establishing no-take areas (Target 11) and sustainable fishing practices (Target 6) are both expected to contribute to mitigate direct pressures (Target 10) and improve the conservation status of cold-reef species (Target 12).

Effects of Target 11 on 12

The current global network of terrestrial protected areas still falls short of adequately representing biodiversity (Butchart *et al.*, 2012; Cantú-Salazar *et al.*, 2013). In addition, the successful achievement of area-based targets for protected areas designation (Target 11) does not in itself guarantee a desirable outcome in terms of biodiversity conservation (Target 12; see Mora and Sale, 2011), as the majority of protected areas are still seeing ongoing declines in plant and animal populations, although at lower rates than in surrounding areas (Craigie *et al.*, 2010; Laurance *et al.*, 2012; Geldmann *et al.*, 2013). Extinction risk of species whose most important sites are protected is also lower (Hoffmann *et al.*, 2010; Butchart *et al.*, 2012). Furthermore, protected area management effectiveness is fundamental in determining the extent to which Target 11 can contribute to Target 12, as increasing protected area coverage alone does not provide better biodiversity outcomes. Rather, improving management effectiveness (Nicholson *et al.*, 2012).

Box 21.1: Improving ecosystem services - South African *Working for Water* programme

The Working for Water (WfW) programme was established in 1995, with the aim to restore and maintain water provision in the Western Cape of South Africa that have been altered by invasive introduced tree species (Marais and Wannenburgh, 2008). Conceived and funded to provide poverty relief (Turpie *et al.*, 2008), the programme creates jobs by employing people from previously disadvantaged communities to clear invasive alien plants (Marais and Wannenburgh, 2008, Turpie *et al.*, 2008).

This example illustrates how tackling a single target, when appropriately implemented, can contribute to moving towards a variety of other targets.

Introduced tree species (in particular *Hakea sericea*, *Acacia longifolia* and *Acacia saligna*) not only have a negative impact on surface water runoff and ground water recharge (Enright, 2000; van Wilgen *et al.*, 2008) but also intensify the effects of fires and floods (Richardson and van Wilgen, 2004). Other species negatively impact on agriculture and fresh water fisheries (e.g., Water hyacinth (*Eichhornia crassipes*) Richardson and van Wilgen, 2004). Invasive species threaten biodiversity, mainly in the fynbos biome (van Wilgen *et al.*, 2008), and have negative impacts on the ecological functioning of ecosystems (Richardson and van Wilgen, 2004; Le Maitre *et al.*, 2011), as well as water security. They also cause millions of South African Rand (ZAR) in damage as a result of fires and floods, or losses incurred in agriculture and fisheries (le Maitre *et al.*, 2004).

Prioritized, systematic eradication of invasive species (Target 9) of a range of taxa is at the core of the WfW programme, making use the best use of financial and human resources available. Efforts are under way to prioritize eradications across the range of invasive species, using a combination of different methods (mechanic clearing, herbicide application, biocontrol agents (Forsyth *et al.*, 2010, Roura-Pascual *et al.*, 2012). Eradication of invasive tree species leads, for example, to the recovery of endemic red-listed dragonfly species (Samways and Taylor, 2004), or the recovery of indigenous riparian vegetation in the Western Cape (Target 12, e.g. Ruwanza *et al.*, 2013). Clearing of invasive trees species in the fynbos region contributes to increases in stream flow (Marais and Wannenburgh, 2008), and reduces evaporation (Dye and Jarmain, 2004), thus increasing water provision (Target 14). Clearing and integrated control of invasive species, coupled with "natural" or assisted restoration contributes to the re-establishment of the natural hydrological regimes, and reduces fire and flood risks (Target 14 and 15). Clearing of invasive species also improves the productive potential of the land (Turpie *et al.*, 2008).

The WfW programme is based on close cooperation and collaboration between scientists from different research institutions, conservation and land managers and government departments (Target 19). In 2008, a unit at the South African National Biodiversity Institute – Invasive Species Programme was established with the aim to (1) detect and document new invasions, (2) provide reliable and transparent post-border risk assessments and (3) provide the cross-institutional coordination needed to successfully implement national eradication plans (Wilson *et al.*, 2013).

The WfW programme contributes towards achieving a range of Aichi Biodiversity Targets through the integration of biodiversity and ecological concerns into a poverty alleviation programme (achievement of Aichi Biodiversity Target 2), by mainstreaming biodiversity across a number of national and provincial government departments for agriculture, conservation and environment, by introducing a PES system aimed at restoring public and private lands (Turpie *et al.*, 2008) (achieving Aichi Biodiversity Target 3), and by using water provision as umbrella service for biodiversity conservation (Turpie *et al.*, 2008).

21.4 CONCLUSIONS

An integrated view on the Strategic Plan for Biodiversity 2011-2020, and of its Strategic Goals and Aichi Biodiversity Targets identifies aspects that should be taken into consideration in policy design. One is the occurrence of time lags between implementation of actions and their outcomes. Another is the occurrence of interactions between actions taken towards one Target and other Targets.

Our analyses indicate that there is variable progress towards Targets within Goals. The greatest progress lies within responses in Goal C (especially Target 11). Response indicators across all Strategic Goals (and nearly all targets) are increasing, showing that efforts are being made to relieve pressures on biodiversity, and to implement measures for the conservation of biodiversity and sustainable use of resources. Nevertheless, the trajectory of biodiversity status is still declining. The recovery of biodiversity and ecosystems will not occur immediately. This make difficult to evaluate the contribution of the current and projected increased responses. However, there are numerous examples throughout the chapters that indicate that responses, when properly implemented, have substantially improved biodiversity status at local to national scales. Strategic Goal C is the one whose outcomes will be felt sooner and contribute in the short-time to avert biodiversity loss and ecosystems degradation.

Across all Targets and Goals there appear to be considerable efforts needed to move from current trajectories to those that would attain the 2020 targets. The majority of measures of state continue to decline and pressures continue to increase, but there are signs in some cases that policies and other responses highlighted in Chapters 1-20 are starting to have traction at local and national levels. As such, at this mid-point between 2010 and 2020, we conclude that important progress has been made in moving towards many of the 20 Aichi Biodiversity Targets, however current rates of progress are mostly insufficient to reach the targets. Additional efforts will therefore be needed to achieve the targets by 2020.

Our analysis also highlights interactions among the targets. Targets under Goal A are particularly important in terms of their broad impacts on other targets and, therefore, in paving the way to achieving targets under Strategic Goals B, C, and D. It will take considerable political will to mainstream biodiversity, as well as to remove harmful subsidies and incentives. Achieving Target 5, and thereby addressing the currently largest pressure on terrestrial biodiversity loss will require a concerted approach that draws upon actions under most of the other targets (see Target 5). This applies similarly to Targets 6, 7 and 10. Under Strategic Goal E, Targets 17 and 20 have been identified as having a large influence on the progress towards achieving other targets. While good progress has been made in developing and implementing NBSAPs, there is so far, no evidence that financial resources are being mobilized to fully implement NBSAPs and to support measures for the conservation of biodiversity and natural resources.

The picture that we can paint of the progress towards biodiversity targets has improved substantially since the last Global Biodiversity Outlook. It is expected that this will continue to improve as a number of pertinent indicators will have more complete data over longer time series closer to the 2020 CBD deadline, presenting a richer set of information from which to draw conclusions.

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ANNEX 1

Table A1. Indicators used within the analysis. For full details and individual plots for each time series, as well as a list of sources, see Appendix S1. Note that temporal coverage indicates the date span of the raw data; in some cases, the full data spans were not used for extrapolation due to data certainty issues; these are indicated within the data descriptions in Appendix S1. Projected change is relative to 2010 modeled value (underlying trend).

Strategic Goal	Aichi Target	Indicator name	Units	Type	Temporal coverage	2020 projection (95% confidence interval)	Projected change relative to 2010
A	1: By 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably.	Biodiversity Barometer (respondents that have heard of biodiversity)	Percent	Response	2009-2013	72.28 (64.08, 80.48)	Significant increase
		Biodiversity Barometer (respondents giving correct definition of biodiversity)	Percent	Response	2009-2013	29.40 (26.41, 32.39)	Significant increase
		Online interest in biodiversity (Google Trends)	Unitless	Response	2004-2013	44.26 (37.89, 50.63)	Non-significant decline
		Investment in environmental education	Constant million US$	Response	1965-2012	25.81 (6.11, 108.83)	Non-significant decline
	2: By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.	Investment in Environmental Impact Assessments (EIAs)	Constant million US$	Response	1970-2011	52.25 (2.65, 1029.23)	Non-significant decline
		Number of research studies involving economic valuation	Unitless	Response	1974-2010	3451.43 (1779.93, 6696.62)	Significant increase
	3: By 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the Convention and other relevant international obligations, taking into account national socioeconomic conditions.	Funding towards institutional capacity building in fisheries	Constant million US$	Response	2000-2011	1067.67 (45.69, 24,946.44)	Non-significant increase

A	4: By 2020, at the latest, Governments, business and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits.	Ecological Footprint	Number of earths needed to support human society	Pressure	1961-2009	1.46 (1.40, 1.51)	Significant increase
		Red List Index for birds, mammals & amphibians showing trends driven by utilisation	Index value	State	1986-2012	0.84 (0.84, 0.84)	Significant decrease
		Percentage of Category 1 parties to CITES (Convention on International Trade in Endangered Species)	Percent	Response	1994-2013	64.96 (55.66, 74.27)	Significant increase
		Human Appropriation of Net Primary Productivity	Pg C/Year	Pressure	1910-2005	16.05 (15.33, 16.77)	Significant increase
		Water footprint	1,000 m^3	Pressure	1995-2009	1.5e10 (1.34e10, 1.66e10)	Significant increase
B	5: By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.	Wetland Extent Index	Index value	State	1970-2008	0.52 (0.37, 0.68)	Significant decrease
		Wild bird Index for habitat - specialist birds	Index value	State	1968-2011	71.77 (67.95, 75.60)	Non-significant decrease
		Natural habitat extent	Percent global area	State	1961-2011	60.67 (60.00, 61.33)	Significant decrease
	6: By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits.	Marine Stewardship Council engaged fisheries (Tonnage)	Tons	Response	1999-2013	15,997,948 (8,431,701, 23,564,194)	Significant increase
		Fish stocks in safe biological limits	Percent	State	1974-2009	67.91 (61.94, 73.60)	Non-significant decrease
		Red List Index for seabirds	Index value	State	1988-2012	0.83 (0.82, 0.83)	Significant decrease
		Global effort in bottom- trawling	kW sea days	Pressure	1950-2006	3.2e10 (2.4e10, 4.4e10)	Significant increase

Strategic Goal	Aichi Target	Indicator name	Units	Type	Temporal coverage	2020 projection (95% confidence interval)	Projected change relative to 2010
B	7: By 2020 areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.	Area of forest under sustainable management: FSC and PEFC certified	Million hectares	Response	2000-2012	485.44 (404.23, 566.65)	Significant increase
		Wild Bird Index for farmland birds	Index value	State	1980-2011	44.31 (42.83, 45.79)	Significant decrease
		Area under organic agriculture	Million hectares	Response	1999-2011	44.46 (38.89, 50.03)	Significant increase
		Area under conservation agriculture	1,000 hectares	Response	1990-2010	191,613 (161,559, 221,667)	Significant increase
	8: By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.	Insecticide use	Tonnes	Pressure	1992-2011	295, 573 (241, 613, 349,533)	No significant trend
		Red List Index for birds showing trends driven by pollution	Index value	State	1988-2012	0.92 (0.92, 0.92)	Significant decrease
		Nitrogen surplus	Tg N/Year	Pressure	1970-2005	141.64 (136.73, 146.54)	Significant increase
	9: By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.	Red List Index for birds showing trends driven by invasive alien species	Index value	State	1988-2012	0.92 (0.92, 0.92)	Significant decrease
		Cumulative number of invasive alien species introduction events	Unitless	Pressure	1500-2012	5,205 (5,039, 5,371)	Significant increase
		Adoption of national legislation relevant to the prevention or control of invasive alien species	Number of countries	Response	1967-2009	60.25 (48.56, 71.38)	Non-significant increase
	10: By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning.	Glacial mass balance	Mm water equivalent	State	1970-2012	-1050.11 (-1382.81, -717.42)	Non-significant decrease
		Mean polar sea ice extent	Million km^2	State	1979-2013	22.97 (22.67, 23.27)	Significant decrease

C	11: By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes.	Coverage of marine protected areas	Percent	Response	1911-2011	4.92 (2.92, 7.36)	Significant increase*
		Coverage of terrestrial protected areas	Percent	Response	1911-2011	13.34 (11.86, 14.89)	Significant increase*
		Protected area management assessments	Number assessed	Response	1966-2011	28, 813 (19, 866, 38, 759)	Significant increase
		Funds towards nature reserves	Constant million US$	Response	1976-2013	2,641 (29, 238,505)	Non-significant increase
		Protected area coverage of bird, mammal and amphibian distributions	Percent	Response	1990-2012	23.19 (22.5, 23.88)	Significant increase
		Protected area coverage of freshwater ecoregions	Percent	Response	1935-2012	59.99 (56.63, 63.31)	Significant increase
		Protected area coverage of marine ecoregions	Percent	Response	1990-2012	70.95 (46.58, 90.16)	Significant increase
		Protected area coverage of terrestrial ecoregions	Percent	Response	1990-2012	83.93 (79.64, 87.80)	Significant increase
		Protected area coverage of Important Bird and Biodiversity Areas	Percent	Response	1900-2012	27.01 (26.01, 28.01)	Significant increase
		Protected area coverage of Alliance for Zero Extinction sites	Response	Response	1990-2012	24.45 (23.19, 25.73)	Significant increase
C	12: By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained.	Living Planet Index	Index value	State	1970-2011	0.80 (0.79 0.82)	Significant decrease
		Red List Index (birds, mammals, amphibians and corals)	Index value	State	1986-2012	0.80 (078, 0.81)	Significant decrease
		Funds towards species protection	Constant million US$	Response	1976-2013	849.4 (10.5, 68,853.4)	Non-significant increase
		Mammal and bird extinctions	Num. extinctions per 25 years	State	1800-2000	18.00 (11.41, 28.40)	No significant change

Strategic Goal	Aichi Target	Indicator name	Units	Type	Temporal coverage	2020 projection (95% confidence interval)	Projected change relative to 2010
C	13: By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socioeconomically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.	Genetic diversity of terrestrial domesticated animals	Percentage of breeds at risk	State	1996-2013	17.36 (16.35, 18.37)	Significant increase
D	14: By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable.	Rural population with access to improved water resources	Percent	Response	1990-2011	88.68 (87.72, 89.64)	Significant increase
		Red List Index for pollinators (birds & mammals)	Index value	State	1988-2012	0.90 (0.89, 0.90)	Significant decrease
	15: By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification	No indicators available for extrapolation					
E	16. By 2015, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force and operational, consistent with national legislation	No indicators available for extrapolation					
	17. By 2015 each Party has developed, adopted as a policy instrument, and has commenced implementing an effective, participatory and updated national biodiversity strategy and action plan.	No indicators available for extrapolation					

E	18. By 2020, the traditional knowledge, innovations and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are respected, subject to national legislation and relevant international obligations, and fully integrated and reflected in the implementation of the Convention with the full and effective participation of indigenous and local communities, at all relevant levels.	No indicators available for extrapolation					
	19: By 2020, knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied.	Funds committed to environmental education and research	Constant million US$	Response	1959-2013	3,636.6 (63.3, 208,798.8)	Non-significant increase
		Knowledge transfer (number of biodiversity papers in Web of Science per year)	Unitless	Response	1987-2013	1,503.96 (1,289.69, 1,718.22)	Non-significant increase
		Number of Global Biodiversity Information Facility (GBIF) records over time	Millions	Response	2001-2013	1,146.9 (894.8, 1,398.9)	Significant increase
	20: By 2020, at the latest, the mobilization of financial resources for effectively implementing the Strategic Plan for Biodiversity 2011-2020 from all sources, and in accordance with the consolidated and agreed process in the Strategy for Resource Mobilization, should increase substantially from the current levels. This target will be subject to changes contingent to resource needs assessments to be developed and reported by Parties	Official Development Assistance provided in support of the CBD	Constant billion US$	Response	2002-2012	7.87 (2.94, 12.80)	Non-significant increase
		Funding provided by the Global Environment Facility	Constant million US$	Response	1998-2013	6,996.3 (1,536.7, 31,852.5)	Non-significant increase
		Global funds committed towards environmental policy, laws, regulations and economic instruments	Constant million US$	Response	1976-2013	8,373.6 (2,287.2 30,655.7)	Non-significant increase

** While significantly increasing, projections imply that terrestrial and marine protected area coverage will be insufficient to meet 2020 goals of 17% and 10% global area respectively based on current trajectories.*

SYNTHESIS OF TRENDS, STATUS AND PROJECTIONS FOR 2050 AND BEYOND

The 2050 Vision: "By 2050, biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people."

PREFACE

The most recent United Nations estimates of human population growth indicate that the global population will rise from slightly more than 7 billion currently to between 8.1 and 10.9 billion people by 2050 (UN, 2013a). The United Nations has also set a wide range of objectives to improve human well-being globally (Millennium Development Goals; UN, 2010; UN, 2010b) which are currently being revised and renewed (Sustainable Development Goals; UN 2013c, UNDP & UNEP 2013). Meeting human development objectives, in the light of the combined effects of population growth and increasing average per capita consumption, will require substantial increases in global food and energy supply by 2050 (IPCC WG3 2014). This means that attaining the 2050 vision will require meeting substantial challenges above and beyond those needed to achieve the 2020 Aichi Biodiversity Targets (see sections "4. What do scenarios suggest for 2050 and what are the implications for biodiversity?" in all chapters). In particular, five major challenges will come to the forefront by 2050 and beyond. These challenges are highlighted below in section 22.1, as are the risks of not meeting these challenges. The actions that can be taken to rise to these challenges are then outlined in section 22.2 of this chapter.

22.1 FIVE MAJOR CHALLENGES FOR 2050 AND THE RISKS OF NOT RISING TO THESE CHALLENGES

Climate change is projected to become a major driver of biodiversity and ecosystem change by 2050. The most recent Intergovernmental Panel on Climate Change (IPCC) report projects further global temperature increases of between 0.4°C to 2.6°C by 2055 and 0.3°C to 4.8 °C by 2090, depending on greenhouse gas emissions scenarios and other uncertainties (IPCC WG1 2013, Figure 22.1). Global warming will be accompanied by rising sea levels, changes in precipitation patterns, substantial loss of summer Arctic sea ice and increasing ocean acidification. Climate change will have a broad range of impacts on biodiversity at genetic, species and ecosystem levels including shifts in the distribution of species and ecosystems, changes in species abundance and increased risk of extinctions (Chapters 5, 6, 9-12; IPCC WG2, 2014). For example, in high greenhouse gas emissions scenarios, terrestrial and aquatic species in some regions will need to move more than 10 km/year to stay in favorable climates, generating large impacts on ecosystem services, reducing the efficacy of protected areas and increasing species extinction risk (Chapters 5, 6, 9-12; IPCC WG2 2014). Efforts to mitigate climate change could have very large impacts, both positive and negative, on biodiversity (Chapters 3 & 7; IPCC WG2 2014).

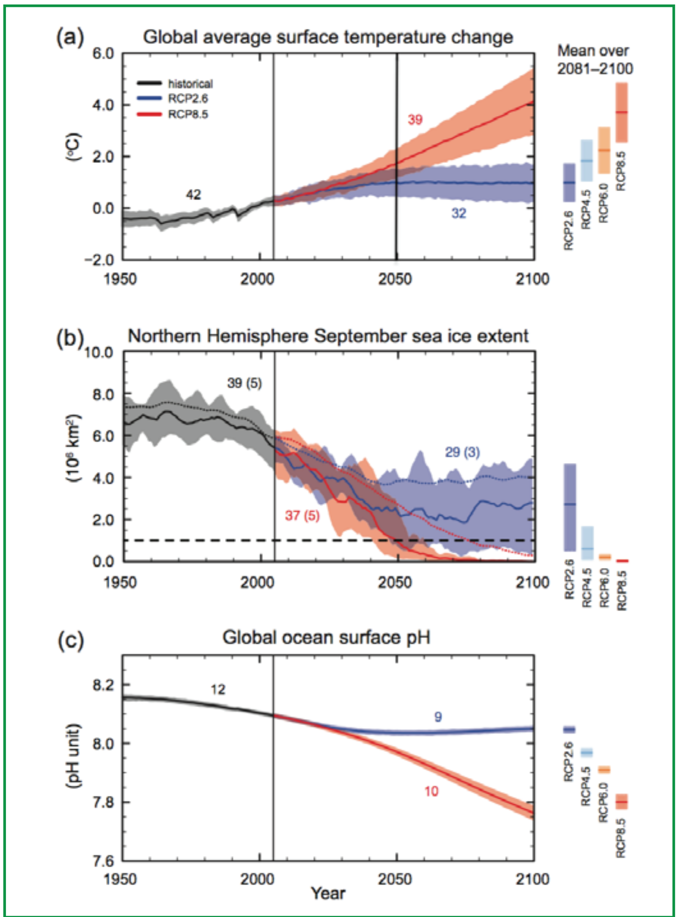

Figure 22.1. Trends, status and projections of global temperature (a), Arctic ice extent (b) and ocean pH (c) Figure SPM.7 from IPCC AR5 WG1 2013. The four RCP scenarios correspond to very low (RCP2.6, blue), low (RCP4.5), medium (RCP6.0) and high (RCP8.5, red) greenhouse gas emissions scenarios. Solid lines indicated multi-model means and the shading indicates the range of uncertainty between models for each RCP scenario. For sea ice extent a subset of models that best reproduce current trends were used for the multi-model means and uncertainty. For completeness the dotted lines indicate the average of all models. Source: IPCC AR5 WG1 (2013).

Demand for fertile land is projected to substantially increase by 2050. Habitat loss in terrestrial ecosystems, which is currently low or slowing in several regions of the world, is projected to increase substantially in many scenarios by 2050 due to increasing pressures to convert natural habitats in areas with good potential for agricultural and bioenergy production (Chapters 4, 5 & 7; Lambin & Meyfroidt 2011; IPCC WG3, 2014). Scenarios that account for realistic increases in agricultural productivity often require substantial increases in land converted to agriculture by 2050 in order to meet per capita food demands of a growing human population (Chapters 5 & 7; Foley *et al*., 2011; Schmitz *et al*., 2014). These scenarios are based on the assumptions that waste in the food supply chain remains high and that diets converge on current "Western" standards. In addition, many scenarios that achieve the dual goals of keeping global warming below 2°C and meeting energy requirements for human development goals rely on massive deployment of bioenergy (Chapters 3-5; Harfoot *et al*. 2014, IPCC WG3 2014). The combination of expanded agriculture and bioenergy results in a projected global land squeeze in which there is not sufficient room to conserve natural terrestrial habitats, leading to large declines in biodiversity (Chapters 3-5 & 7, Lambin & Meyfroidt 2011).

Aquaculture is foreseen to dominate fish production by 2050. The majority of wild-capture marine fisheries are currently at or beyond their maximum sustainable capacity, meaning that the large increases in global fish production foreseen for 2050 are projected to come primarily from aquaculture (Chapters 6 & 7). If harmful subsidies are not reduced and management of territorial and non-territorial marine systems do not improve, negative impacts of wild-capture marine fisheries are projected to substantially increase by 2050 in many regions (Chapters 3 & 6). These impacts include the collapse of exploited fish populations and increased pressure on aquaculture to produce fish for a growing human population (Chapter 6). Recent projections suggest that aquaculture production will be nearly double by 2050 and surpass that of wild-capture fisheries within a few years from now (Chapter 7). This rapid expansion of aquaculture raises a variety of concerns including pollution, increased demand for high protein feed and competition for land or coastal areas (Chapter 7).

Water scarcity is foreseen to increase in many regions of the globe by 2050. Global water withdrawals from freshwater systems are projected to nearly double by 2050 in most scenarios (Chapter 4; Hejazi *et al.*, 2014; Figure 22.2). Water for food production currently accounts for 84% of global water consumption and dominates projected future global water consumption. Additional water use for biomass production makes a significant contribution in scenarios with massive deployment of bioenergy (Chapter 4, Hejazi *et al.* 2014). This results in reduced water flow for freshwater ecosystems, which are highly dependent on water flow to maintain biodiversity and ecosystem functions (Chapter 4). Climate change will also alter water flow with highly variable impacts depending on the region (IPCC WG2, 2014). Dams, pollution, invasive alien species and freshwater habitat modifications are foreseen to increase in many regions of the world, leading to very high projected biodiversity loss in many freshwater systems (Chapters 5, 8, 9, 11).

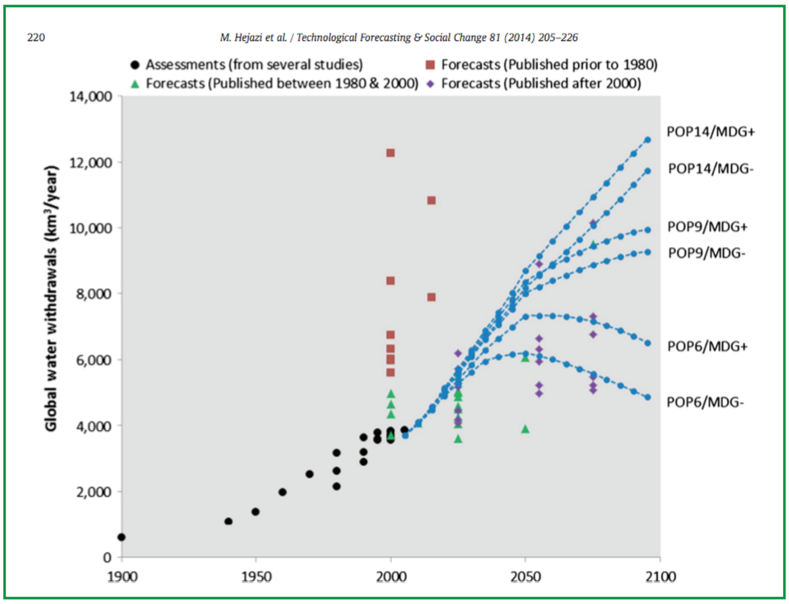

Figure 22.2. Synthesis of trends, status and projections of global water withdrawals. Points correspond to assessments and forecasts classified by their date of publication. Blue lines with points correspond to scenarios of Hejazi et al where "POP/MDG" corresponds combinations of population growth and meeting or not the Millennium Development Goals. For example POP6/MDG+ is a scenario with low population (ca. 5.5 billion in 2100), high sustainability actions and MDG goals achieved; whereas POP14/MDG- is a high population scenario (ca. 14 billion in 2100) that is described as a "crowded chaos" scenario. Source: Hejazi *et al.* (2014).

Combinations of drivers could push some systems beyond tipping points at regional scales by 2050. There is evidence that several large-scale regime shifts have already started and scenarios suggest that these could cause substantial disruption of social-ecological systems by 2050 (Leadley *et al.*, 2010; Leadley *et al.*, 2014; IPCC WG2 2014). The two most clearly understood examples are degradation of coral reefs due to combinations of pollution, destructive fishing, invasive alien species, ocean acidification and global warming, and loss of summer Arctic sea ice due to global warming (Chapter 10; IPCC WG1 & WG2 2014; Leadley *et al.*, 2014). More speculative regime shifts include degradation of the Amazonian tropical humid forest due to combinations of deforestation, use of fire and global warming, and collapse of some tropical fisheries due to combinations of overfishing, pollution, sea level rise and global warming (Leadley *et al.*, 2010; IPCC WG2 2014; Leadley *et al.*, 2014). These relatively rapid and large shifts in ecosystem structure and function at regional scales are projected to have large negative impacts on biodiversity, ecosystem services and human well-being if not averted (Leadley *et al.*, 2010; Leadley *et al.*, 2014).

22.2 PLAUSIBLE PATHWAYS TO THE 2050 VISION

Scenarios for 2050 indicate that very substantial changes from business as usual trends are needed in order to meet three key global objectives: slow and then stop the loss of biodiversity; keep global warming below 2°C; and attain human socioeconomic development goals (MA, 2005; Leadley *et al.*, 2010; PBL, 2010 & 2012; IPCC WGIII, 2014). As several examples of recent environmental successes illustrate, solutions for a sustainable future will require a wide range of deep societal transformations (see examples in most chapters; PBL, 2010, 2012; Figure 22.3). Global scenarios developed in the context of the Rio+20 United Nations Conference on Sustainable Development are used in several chapters of the Global Biodiversity Outlook 4 to illustrate the diversity, complexity and feasibility of pathways to a sustainable future (see especially Chapters 5, 6, 7, 8 & 12; PBL, 2012; Box 22.1). Together with a variety of other regional and global scenarios they provide insight – key elements of which are summarized below – into the major transformations required to meet the three 2050 objectives. These major transformations in development pathways will need to be fully engaged over the next decade in order to meet these objectives because of the long lag times inherent in social and technical transitions and in the biological, climate and oceans systems of the Earth (PBL, 2012; IPCC WG3, 2014).

Figure 22.3 illustrates several facets of achieving the 2050 vision in the Rio+20 pathways. For example compared to business as usual scenarios:

Marine fisheries undergo rebuilding due to reduced fishing pressure in the Rio+20 scenarios, resulting in a much smaller fraction of collapsed fish stocks than is currently the case (Chapter 6; Figure 22.3A).

Loss of terrestrial biodiversity is substantially reduced - While halting terrestrial biodiversity loss compared to current levels is not achieved (for most of the indicators that have been explored), the Rio+20 scenarios show an improved biodiversity status when compared with business as usual (Trend) scenarios (Chapter 12, Figure 22.3B).

As outlined below, the actions required to achieve these goals vary substantially between socioeconomic scenarios, but all require substantial changes from current trends (see also Box 22.1). These biodiversity goals are attained within the context of reaching broader socioeconomic objectives that include strong climate mitigation (Figure 22.3C) and increased food production (Fig 22.3D), improved diets and the eradication of hunger.

The actions that contribute most to attaining sustainability include:

Climate change and energy systems - Halting deforestation and appropriately implementing reforestation can make important contributions to climate mitigation and protection of biodiversity (Chapters 3, 5 & 15; IPCC WG3, 2014). Bioenergy and hydroelectric power can make significant contributions to future decarbonization of energy (Figure 22.3C; IPCC WG3 2014), but scenarios indicate that biodiversity objectives cannot be attained in scenarios of massive deployment of these (Chapters 3, 4 & 7; Kraxner *et al.*, 2013). As a result, tremendous gains in technology and implementation of decarbonisation of energy and in energy use efficiency are required to keep global warming below 2°C while at the same time reaching human development goals (PBL, 2012; IPCC WG3, 2014; Figure 22.3C). A substantial degree of climate change by 2050 and beyond is already committed due to long lags in the Earth's climate system, so adaptation plans for biodiversity are needed (IPCC WG2 & WG3, 2014). For example, adaptation will require anticipating climate change in the design of protected areas (Chapter 11).

Food systems – Scenarios suggest that major transformation of food systems is one of the most important keys to achieving sustainability (Chapters 3-7; PBL, 2010; Figure 22.3D). First, roughly a third of harvested food is lost either in the food transport and transformation chain (primarily in developing countries) or in the home (primarily in developed countries) (Chapter 5). Second, diets in developed countries, with some notable exceptions, are typically very high in calories and rich in meat. This combination of factors is correlated with high rates of obesity and related health problems, in addition to which diets rich in meat have a very high environmental footprint (Chapters 4 & 7). Diverse diets combined with global convergence to moderate levels of calorie and meat consumption would improve health and food security in many developing and developed countries (Stehfest *et al.*, 2008; PBL, 2010; Foley *et al.*, 2011; Mozaffarian *et al.*, 2011). This would also substantially reduce impacts on biodiversity and other aspects of the environment compared to business as usual scenarios (Chapters 5, 7 & 8; Figure 22.3). Scenarios indicate that when losses in food systems are reduced, together with changes in diet, only modest expansion of global land use for agriculture would required in some regions and demands on marine fisheries and aquaculture is diminished (Chapters 5-7; Figure 22.3).

Another essential element for achieving sustainability is management of the food production component of food systems. This involves improved management of agriculture, aquaculture and wild-capture fisheries. Agriculture consumes a large fraction of global freshwater resources and produces a significant fraction of pollution including nitrogen, phosphorus, greenhouse gases and micropollutants (Chapters 4, 7 & 8). Scenarios indicate that realistic changes in management of crops and livestock could substantially reduce both water consumption and pollution compared to business as usual scenarios by 2050 without significantly impairing food production capacity (Chapter 7; Figure 22.3D). Scenarios for marine wild-capture fisheries suggest that significant reductions in fishing pressure and changes in fishing techniques in most marine fisheries would lead to rebuilding of fisheries over the next one to two decades (Chapters 3 & 6; Figure 22.3A). This would come at the cost of reductions in jobs for fisherman in the short-term, but the long-term benefits include healthier and more resilient marine ecosystems, greater sustainable yields and stability of fisheries related livelihoods (Chapters 3 & 6).

Freshwater resources – Differences in scenarios for food and biomass energy production entirely dominate the projections of future water use, meaning that much of the focus on achieving sustainable use of water resources requires major transformations in the agricultural sector (Chapter 4; Hejazi *et al.*, 2014). Efforts in other sectors, such as municipal and energy production can also make small but important contributions. Scenarios indicate that widespread deployment of water conservation techniques could substantially reduce water withdrawals and consumption by 2050. For example, optimizing irrigation or shifting to crops that do not require irrigation would substantially reduce water withdrawals and consumption. Technological responses for increasing water supply such as desalinization may be an option to overcome water scarcity in a few cases, but there is little evidence that this can make significant contributions to water needs at large regional or global scales by 2050.

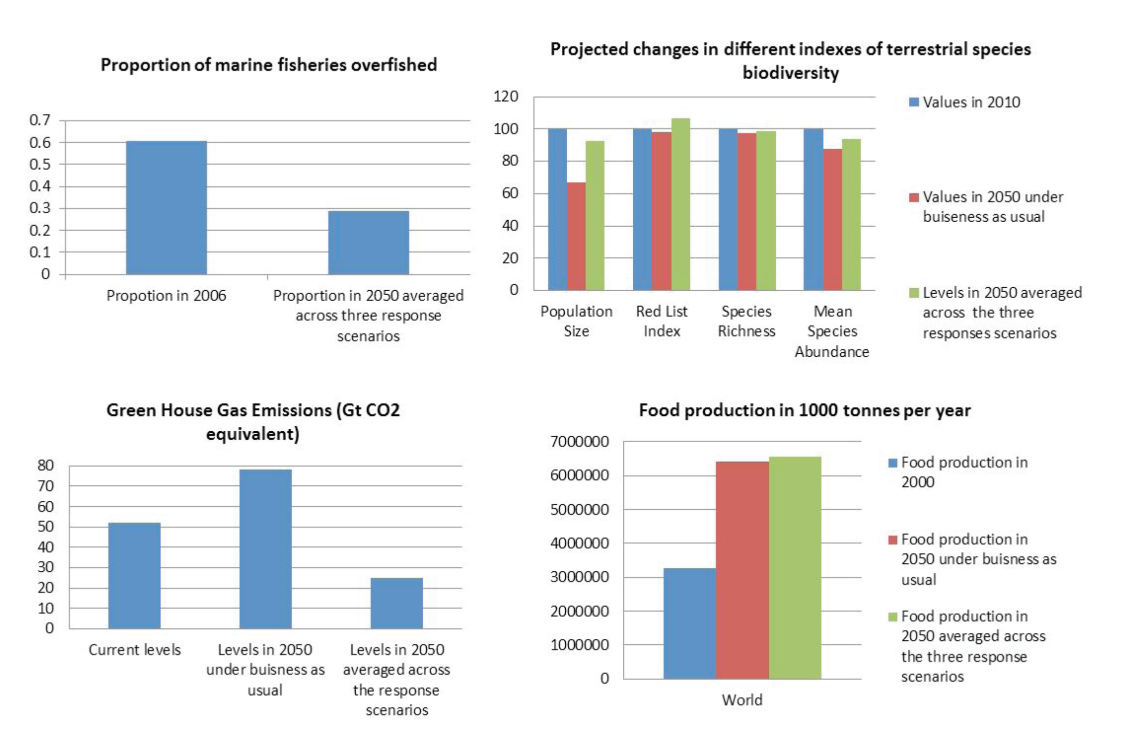

Figure 22.3 – Multiple indicators of response to actions taken in the Rio+20 socioeconomic scenarios. In all cases, the indicators at 2050 have been averaged across all three Rio+20 response scenarios (see Box 22.1). A) proportion of fish stocks overfished in four key regions of the oceans currently (2006) and for an average of the three Rio+20 scenarios in 2050 (see details in chapter 6 for details). These four regions currently have the highest fraction of stocks overfished of all global regions. B) Four indicators of terrestrial species response to socioeconomic scenarios following the Trend (=business-as-usual) or Rio+20 scenarios for 2050. Population size and IUCN Red List status are for carnivores and ungulates; species richness and mean species abundance (MSA) are for a wide range of species groups. MSA measures the degree to which species abundances differ from a "natural" reference ecosystem (see chapter 12 for details). C) Greenhouse gas emissions currently, and for the Trend and Rio+20 scenarios for 2050 (adapted from PBL 2012). D) Global food crop production: current, Trend scenario for 2050 and the three Rio+20 scenarios for 2050 (from PBL 2012). All Rio+20 scenarios achieve eradication of hunger by 2050 as set out in the Millennium Development Goal 1. Source: PBL, 2012, Teh *et al.* (2014) (chapter 6).

The feasibility and viability of these actions strongly depends on their economic costs and benefits and their social acceptability. The UNEP's Green Economy Report (UNEP, 2011) provides insights into these issues at a global scale. The objectives and outcomes of this analysis are broadly coherent with the analyses carried out using the Rio+20 scenarios and summarized above. The Green Economy Report compared two green investment scenarios – G1 and G2, promoting resource efficiency and low carbon development – with business as usual (BAU) scenarios (more conventional use of resources and fossil fuels, following trends of the past 40 years). Green investments in these scenarios of 1 (G1) or 2% (G2) of GDP respectively (related to Aichi Biodiversity Target 20) included reducing deforestation and increasing reforestation (related to Target 5 and Target 15), reducing extractive capacity in fisheries sector and supporting restoration of fish stocks (related to Target 6), reducing fertilisers and reorienting towards conservation agriculture (related to Target 7 and Target 8), focusing on renewable energy (related to Target 10) and water management including ecosystem services (related to Target 14). The sectorial targets for the green investments align closely with Aichi Biodiversity Targets (e.g., 50% reduction in deforestation; restoring fish stocks to sustainable yields) and MDG/SDG relevant goals (e.g., increasing nutrition levels; increasing water availability; increasing energy efficiency and reducing waste).

BAU scenarios foresee increased population and gross domestic product leading to, amongst other things, increased water consumption above sustainable withdrawals, expansion of agricultural land leading to a net loss of 6 million hectares of forest per year and a resultant decline in carbon storage in forests of 7% between 2010 and 2050. CO_2 emissions are projected to increase with atmospheric greenhouse gases approaching 1000ppm by 2100, in line with the A1FI and A2 IPCC SRES scenarios. Natural resource depletion in the BAU scenarios results in falling GDP growth and, combined with climate effects, leads to water stress and food insecurity particularly in Africa and other developing parts of the world. In comparison, under green investment scenarios, both poverty and water stress are reduced alongside consumption levels, whilst economic growth, although slower in the short term, is ultimately higher and more sustainable (Figure 22.4).

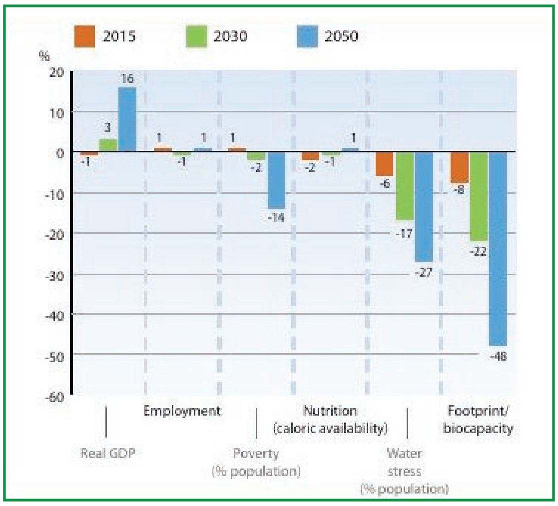

Figure 22.4. Outcomes of the Green Economy analysis showing effects of a 2% GDP green investment socioeconomic scenario (G2) relative to a business as usual scenario (BAU2). Indicators include GDP, Employment, Poverty, Nutrition, Water status and Biocapacity. Source: UNEP 2011.

The differences between the G1/G2 and BAU scenarios hinge on the effects of investment on projected future stocks of natural capital such as forests and fish stocks, which under the green investment scenarios show a recovery compared to an ongoing decline under BAU (Figure 22.5). These global scenarios suggest that by adopting a low carbon, resource efficient and sustainable development pathway, natural capital (including aspects of biodiversity) can be maintained or enhanced without compromising, and in many cases improving long-term employment and growth as well as food, water and energy security.

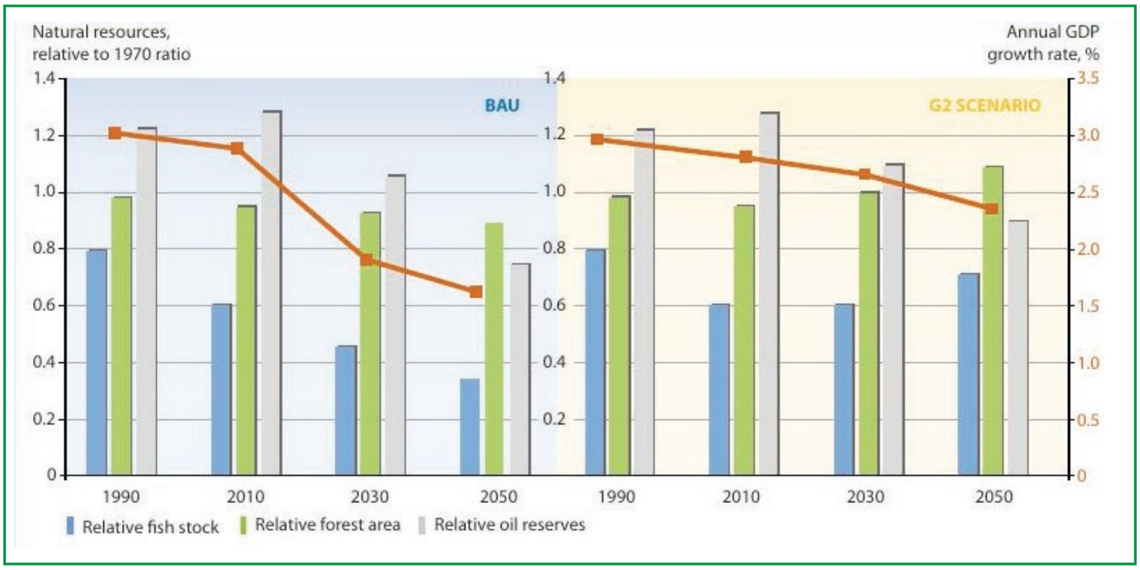

Figure 22.5. Outcomes of the Green Economy analysis showing effects of a 2% GDP green investment socioeconomic scenario (G2) relative to a business as usual scenario (BAU). Indicators include GDP, Fish stocks, Forest Area and Oil Reserves. Source: UNEP 2011.

Box 22.1: A diversity of pathways for reaching the 2050 vision: the "Rio+20" scenarios

The pathways presented here were designed to achieve a broad set of targets (PBL, 2012) that are based on existing international agreements on environmental and development topics (see also Ozkaynak *et al*., 2012). In a way this set of goals could be considered as 'sustainable development goals' – *avant la lettre*. The overarching goal with respect to biodiversity might be phrased as 'by 2050 eradicate global hunger while avoiding further biodiversity loss'. The goal is based on the Convention on Biological Diversity 2050 vision, the Aichi Biodiversity Targets (CBD, 2010a) and the Millennium Development Goals target 1c 'Halve, between 1990 and 2015, the proportion of people who suffer from hunger' (UN, 2001). The 2050 vision is interpreted as slowing the rate of biodiversity loss until 2030 and bringing it down to zero loss by 2050. The MDG hunger target is extended to zero hunger by 2050. These targets are accompanied by goals to limit global long-term mean temperature increase to 2°C, providing universal access to safe drinking water, basic sanitation and modern energy sources, and reducing urban air pollution and fertilizer use. This forced the analysis to take into account synergies and trade-offs with goals in other themes. The trade-offs include limited biofuel use for climate mitigation to avoid competition for land and improved fertilizer-use efficiency to reduce nitrogen emissions resulting from agricultural intensification. Synergies include reduced deforestation due to lower fuel-wood demand resulting from the transition to modern energy sources, and reduced meat consumption reduces biodiversity loss and climate change.

Three pathways that all meet these goals are distinguished:

- *Global Technology*: Focus on large-scale technologically optimal solutions, such as intensive agriculture, and a high level of international coordination;
- *Decentralized solutions*: Focus on decentralized solutions, such as agriculture that is interwoven with natural corridors and national policies that regulate equitable access to food;
- *Consumption Change*: Focus on changes in human consumption patterns, most notably by limiting meat intake per capita and by ambitious efforts to reduce losses in food systems.

Box 22.1: A diversity of pathways for reaching the 2050 vision: the "Rio+20" scenarios *continued*

The pathways differ in their emphasis on human behavior as leverage for change, in the relative weight of regulation versus markets, in coordination versus competition and on the characteristics and scale of the stimulation of technology. The analysis is based on a back-casting approach, addressing the level of effort required to achieve the above described set of sustainability goals taking into account social, economic and technical constraints, and concentrates on the biophysical changes required to achieve the goals (see Chapter 1 for a comparison with other types of socioeconomic scenarios).

The pathways point to five important elements – albeit included in different 'amounts' – to meet sustainability goals (PBL, 2012): 1) increase access to food; 2) alter demand for agricultural products including consumption change and reduction of losses and waste; 3) increase agricultural efficiency; 4) change agricultural land allocation and management, including fragmentation; and 5) protect the most important ecosystems and their goods and services.

All three pathways eradicate hunger and substantially slow biodiversity loss globally, although the route of each varies significantly. The analysis shows that long-term terrestrial biodiversity goals can be met as part of an integrated agenda of land use, food production, hunger, biodiversity protection, access to drinking water, sanitation and modern energy and mitigating climate change. It also shows that achieving the long-term biodiversity goal constrains the types of development in the agricultural sector and how the eradication of hunger can be achieved. Although this is not the scope of this chapter and there are many caveats in the quantitative analysis, the analysis gives input for quantitative target setting, including the rate of agricultural productivity increase and benefits of lifestyle change.

The lifestyle change pathway emphasizes the role of changing consumption patterns to reduce the demand for food and other products. The Global Technology pathway puts emphasis on increasing yields in large-scale agricultural landscapes and the strict separation of land-use functions. The Decentralised Solutions pathway emphasises more ecologically oriented agriculture where technology is adapted to smaller-scale agriculture. Differences between the Global Technology pathway and the Decentralised Solutions pathway include a lower production intensity and related larger claim on land in the latter. However, the Decentralised Solutions pathway also includes an increase in biodiversity and ecosystem services in agricultural fields and surrounding areas, lower fragmentation and reduced emissions of nutrients. Thus, in the Global Technology pathway biodiversity loss is more concentrated in current agricultural areas, whereas in the Decentralised Solutions pathway biodiversity is much higher in agricultural areas, but biodiversity loss is more spread out.

22.3 CONCLUSIONS

All scenarios based on current trends, i.e., business as usual, result in large projected net negative impacts on biodiversity and a broad range of ecosystem services. As in previous Global Biodiversity Outlook reports, there is very strong agreement that a transition from business as usual development pathways is necessary to achieve sustainability by 2050 and beyond. For some of the five key challenges there are indications that some countries and regions are moving away from business as usual development pathways. For example, dramatic reductions in deforestation in some regions suggest that deforestation could be substantially slowed at the global scale in the near future. Modest progress is visible in other areas such as the expansion of sustainable fisheries, aquaculture and agriculture, but very substantial efforts are still required and must be coupled with major transitions in diet and food systems to achieve sustainability. Little progress, however, is being made for other key challenges. Greenhouse gas emissions in particular show no signs of deviating from high emissions scenarios (IPCC WG1 2013), and lack of strong action on greenhouse gas emissions is projected to seriously jeopardize a wide range of positive actions taken to reduce other key drivers of change in biodiversity and ecosystems.

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THE AICHI TARGETS AND BIODIVERSITY IN THE CONTEXT OF THE POST-2015 DEVELOPMENT AGENDA

FOCAL POINTS OF THE ANALYSIS

The original question posed in the technical report outline was "What is the contribution of meeting the Aichi Biodiversity Targets and the 2050 Vision with respect to human well-being and in particular the Millennium Development Goals?" However, given that the Millennium Development Goals mature in 2015, given also the fact that GBO-4 will be published at the time that the political discussion on the post-2015 agenda will be in full swing and in order to provide longer term relevance we have chosen to extend this to include the post-2015 development agenda and the proposed Sustainable Development Goals. The analysis now focuses on the question of how the Aichi Biodiversity Targets and the 2050 Vision can contribute to inclusion of biodiversity in the post-2015 development agenda. The chapter is thus primarily forward looking, literature-based, drawing heavily from recent syntheses, in particular the Biodiversity Issues Brief for the Open Working Group on the post-2015 development agenda meeting in February 2014. Scenarios related to these issues can be found in Chapter 22.

KEY MESSAGES

Biodiversity and ecosystem services can contribute to economic growth and poverty reduction. Equally, biodiversity loss has negative consequences for society worldwide. Actions to reduce pressures on biodiversity, and to conserve and sustainably use biodiversity within safe limits, thus support a broad range of societal issues.

Whilst the importance of biodiversity conservation as a contribution to environmental sustainability is recognized in the Millennium Development Goals (MDGs) under MDG-7, its direct contribution to other goals is not made clear, which may have hindered attention and action on biodiversity loss.

The post-2015 agenda provides an opportunity for mainstreaming biodiversity in a universally relevant development agenda, in more explicit and integrated ways than has been the case to date. Meeting the Aichi Biodiversity Targets would help achieve goals for other global development priorities such as eradication of poverty and hunger, improved health and a sustainable supply of clean energy, food and water.

Biodiversity could be included as part of various kinds of goals and targets currently being considered under the Sustainable Development Goals, including (i) overarching goals on poverty eradication, well-being and sustainable development, (ii) 'topical' goals on food security and nutrition, sustainable water use and universal clean energy, (iii) foundational goals relating to ecosystem health and the planetary life support system, and (iv) goals that relate to "enabling factors" like education, governance and empowerment. Elements of the Aichi Biodiversity Targets provide a good basis for this.

To overcome shortcomings of the MDGs, where a separate goal on environmental sustainability was not conducive for an integrated approach on poverty eradication and environmental sustainability, a combination of the four types is most desirable. Such an approach complements overarching and topical goals that address the pressures and benefits of biodiversity with a foundational goal that addresses the underlying global "life support systems".

23.1 INTRODUCTION

Sustainable development and poverty reduction have been two related and overriding concerns of the global community in recent decades. Where sustainable development is about meeting the needs of the present without compromising the ability of future generations to also do so (WCED, 1987), poverty reduction aims to increase human well-being of those most in need. Since 2000 these concerns, and international commitments to tackle them, have been embodied in the United Nations Millennium Development Goals (MDGs), which, although framed in a sustainable development context, are chiefly concerned with poverty reduction in developing countries.

With the MDGs coming to an end in 2015 and the United Nations resolved to establish an inclusive and transparent intergovernmental process to develop Sustainable Development Goals (SDGs), discussions on how to define, design and implement long-term sustainability goals have taken centre stage. Countries agreed that both the follow-up to the MDGs and the SDGs should be integrated in a single framework and set of goals by the end of 2015 (UN, 2013a). This framework should be a universal agenda applicable to all countries, addressing both poverty eradication and sustainable development, integrating in a balanced manner the social, environmental and economic dimensions of sustainability (Nilsson *et al*., 2013).

Where does biodiversity fit? The role of biodiversity in supporting development and poverty reduction is recognized in broad terms in the Strategic Plan for Biodiversity 2011-2020. "The vision for the plan is: *"Living in Harmony with Nature" where "By 2050, biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people"(CBD, 2010a)*. The rationale behind this vision is that biodiversity "underpins ecosystem functioning and the provision of ecosystem services essential for human well-being. It provides for food security, human health, the provision of clean air and water; it contributes to local livelihoods, and economic development, and is essential for the achievement of the Millennium Development Goals, including poverty reduction" (CBD, 2010a; Millennium Ecosystem Assessment, 2005). The post-2015 agenda provides an opportunity for mainstreaming biodiversity to ensure it is properly included both as part of a broader agenda, integrated with relevant issues like poverty reduction, agriculture and water, thereby providing greater traction for biodiversity in these sectors and as a stand-alone issue.

This chapter discusses how the Strategic Plan for Biodiversity 2011-2020, and in particular the 2020 Aichi Biodiversity Targets and the 2050 vision, might help to meet broader societal objectives of sustainable development and poverty alleviation. More specifically, it explores how the SDGs can draw from what is already agreed in the Convention on Biological Diversity. It begins with a brief summary of knowledge regarding the links between biodiversity, economic development and poverty reduction, before looking to what extent biodiversity has been part of the MDG-agenda and how biodiversity, based on the Aichi Biodiversity Targets, could be included in the SDGs.

23.2 THE LINKS BETWEEN BIODIVERSITY, ECONOMIC DEVELOPMENT AND POVERTY REDUCTION

It is widely acknowledged that biodiversity underpins ecosystem functioning (e.g., Cardinale *et al*., 2012) and that the role of ecosystem services in providing food, water, energy and other, non-material benefits (Butler and Oluoch-Kosura, 2006; Nelson *et al*., 2011) is essential for human well-being (Millennium Ecosystem Assessment, 2005; TEEB, 2011). These services all depend on the ecological processes of functioning ecosystems (De Groot *et al*., 2010; TEEB, 2011).

However, the relationship between biodiversity and ecosystem services is not straightforward and depends largely on the type of ecosystem service considered (Mace *et al*., 2012). Biodiversity plays a crucial role in the provision of regulating services; examples include the role of pollinators and a large variety of predator species that reduce outbreaks of pests in agricultural fields (e.g., Cardinale *et al*., 2012). Furthermore, biodiversity is important to some degree for cultural services. However, there are often choices to be made between the delivery of one kind of service over another – management decisions that favour the provision of agricultural goods, for example, may do so at the expense of maintaining regulating services (Foley *et al*., 2005; Millennium Ecosystem Assessment 2005; see also Target 14 chapter) with consequent implications for the kinds of biodiversity that are maintained as well as the aspects of human well-being that are addressed.

Nevertheless it is recognized that we all depend in different ways on biodiversity, and that the poor rely more directly on biodiversity than others because of their limited ability to purchase alternatives (TEEB, 2011). In many regions people are dependent on food, water and energy derived directly from natural areas such as forests, coral reefs, etc. (FAO and CINE, 2009). Evidence suggests that biodiversity often acts as a safety net for the poor in times of crisis, although it may provide a route out of poverty in some circumstances. In the short term it is the availability of natural resources that is most beneficial to the poor, although diversity, including for example different crop varieties, is important from a risk management perspective and for sustaining benefits by ensuring resilience to shocks and longer term change (Roe *et al*., 2011, 2013).

Various economic sectors rely on biodiversity and ecosystem services such as fisheries, agriculture and tourism (OWG TST Paper, 2013). Yet both poverty and economic development can negatively affect global biodiversity and the provision of important ecosystem goods and services (MA, 2003). More food, water and firewood are needed to sustain on-going population growth in especially the poorer parts of the world, that are not always endowed with the resources and technologies to produce these in a sustainable matter. At the same time, continuing economic growth, including growth of the global middle class, will add to the demand for products like meat, timber, bio-energy and paper. Our historical development pathway has been built on transforming natural capital (and eroding biodiversity) to fuel economic growth. Thus, under prevailing production and consumption patterns, biodiversity loss and natural resource degradation will continue unabated or accelerate without additional policies (CBD, 2010b; PBL, 2010; OECD, 2012), with the poor being disproportionally affected (Millennium Ecosystem Assessment, 2005). Lack of sustainable access to food, drinking water and modern energy forms a major part of the global problem of poverty and impacts directly on human well-being (PBL, 2009), while the provision of food, water, and energy to the poor becomes more difficult when available natural resources are not managed sustainably or degrade due to global environmental change, including climate change, land degradation and water scarcity (Millennium Ecosystem Assessment, 2005; IPCC, 2007; UNEP, 2012a). The existence of thresholds and tipping points increases the risk of difficult-to-reverse negative biodiversity change, with societal implications (CBD, 2010b).

However there are alternative development pathways, with more promising potential futures as are illustrated in the previous chapter. The Green Economy concept (UNEP, 2011) places sustainability and resource efficiency at its core. Moreover, evidence suggests that actions to conserve biodiversity offer solutions to a range of societal challenges including climate change, food and water security, and can benefit the poor if designed appropriately (Koziell, 2001; Sachs *et al*., 2009; Roe *et al*., 2011). A pro-poor policy orientation is necessary, for example to ensure access to clean energy (UNEP, 2011) or access to agricultural extension services and markets (IFAD, 2013) amongst others. However, the possible contribution of biodiversity and natural capital for the greening of the economy in emerging economies and industrialised countries should also receive more attention.

The relationships between biodiversity and development and between biodiversity and poverty reduction are not simple, and mutually beneficial outcomes are by no means assured. Measures to conserve biodiversity and reduce poverty can be complementary, although trade-offs are sometimes inevitable (Tekelenburg *et al*., 2009). However, many of the underlying causes of both sustained poverty and biodiversity loss are similar and stem from the way that economic growth and development has progressed. Addressing those causes will help both agendas, and within the right enabling environment biodiversity itself can be a foundation for sustainable development and poverty reduction. To be able to deal with trade-offs and capture synergies between biodiversity and ecosystem services and the provisioning of water, energy and food, the development of integrated responses that target multiple objectives is increasingly advocated (Bazilian *et al*., 2011; Hoff, 2011; World Economic Forum, 2011; European Report on Development, 2012).

23.3 RELATIONSHIPS BETWEEN BIODIVERSITY AND THE MILLENNIUM DEVELOPMENT GOALS

The Millennium Development Goals (MDGs) came into being in September 2000. They represent a global commitment to tackling poverty and meeting the needs of the world's poorest, by formulating eight clear goals, accompanied by targets and indicators to monitor progress and accountability. The MDGs prioritize basic needs in global efforts to reduce poverty. Hence, MDG1 focuses on poverty and hunger, MDGs 2 and 3 focus on education and empowerment, MDGs 4-6 focus on health, whilst MDG7 (environmental sustainability) and MDG8 (global partnership for development) provide something of the enabling environment.

The importance of biodiversity for development is recognized by the MDGs under goal 7 (ensure environmental sustainability) that includes the CBD 2010 biodiversity target to 'reduce biodiversity loss, achieving, by 2010, a significant reduction in the rate of loss'. Yet it is widely held that MDG 7 lacked political voice, as well as a means to integrate different components of environmental sustainability into the broader development agenda (UNDG, 2010). In the implementation of the MDGs, and in particular through the creation of a distinct, 'separate' goal for environmental issues, the importance of biodiversity for the achievement of the other MDGs (including the high-profile goals on poverty, food, and health) has not been sufficiently recognized and promoted.

Although it is difficult to test relationships, biodiversity is clearly directly relevant to some of the other MDGs (Pisupati and Warner, 2003). This relation goes both ways; biodiversity provides important opportunities for poverty reduction and economic development, while loss of biodiversity and natural resources will exacerbate current risks. For example, actions to conserve biodiversity can positively contribute to:

MDG1 – Eradicate Extreme Poverty and Hunger. As described above, the rural poor often depend on biodiversity, whether cultivated or wild harvested, for their food and income. Maintaining biodiversity that is used in these ways therefore has the potential to contribute to MDG1, particularly Target 1C on reducing hunger. To achieve this would entail amongst other things improving poor peoples' access to, and tenure of, biodiversity resources; providing market linkages and sustainable use practices, and; involving the poor in decision and policy making (Pisupati and Warner, 2003).

MDG2 – Achieve universal primary education, and MDG3 – Promote gender equality and empower women. Although the role of biodiversity is more marginal for these goals, it is clear that depletion of natural resources, including biodiversity, increases the effort and travel distance required to access household necessities such as water, fuel wood and other forest products. The burden of this falls disproportionately on women and children. Reducing this burden through improved biodiversity management would free up time for other activities including education.

MDG6 – Combat HIV/AIDS, malaria and other diseases. Biodiversity is a source of traditional medicines relied upon by a great majority of people in developing countries (Chivian, 2002), In addition, although natural ecosystems, particularly in the tropics, support pathogens and disease vectors, there is increasing evidence that biodiversity loss increases disease transmission (Keesing *et al*., 2010). This suggests that maintaining biodiversity and intact ecosystems may contribute to Target 6C in reducing major diseases.

Looking back, missing the 2010 Biodiversity Target didn't help the realization of the MDGs. Nevertheless, elements of the MDGs are likely to persist into the post-2015 development agenda (UN, 2013b), which provides a fresh opportunity to more clearly integrate biodiversity and development priorities.

23.4 BIODIVERSITY AND THE AICHI TARGETS IN THE POST-2015 DEVELOPMENT AGENDA

How might biodiversity be incorporated into the post-2015 development agenda and build upon the CBDs strategic plan and 2050 vision? In the Rio+20 outcome document existing commitments to biodiversity were reaffirmed. However, some note an increasing marginalization of biodiversity and warn that the emphasis on mainstreaming biodiversity in, for example, the green economy tends to marginalize the specificity of biodiversity conservation issues (Carrière *et al*., 2013).

Noting the weaknesses of isolating biodiversity in the MDG process, the post-2015 United Nations development agenda might usefully consider how biodiversity could be more integrated into broader development objectives in ways that break down the silos created by separate thematic goals (OWG TST paper, 2013).

Discussions on the structure of the post-2015 development agenda have identified a range of potential SDGs. Four broad classes can be distinguished, into which biodiversity could be integrated at the level of the goal or under a target (Lucas *et al*., 2014; OWG TST paper, 2013; see also Melamed and Ladd, 2013; Boltz *et al*., 2013; CBD Secretariat, 2013):

- Overarching goals and targets that encompass multiple dimensions of sustainable development such as poverty eradication and green economy.
- Goals and targets dealing with specific development topics such as food security and nutrition, sustainable water use, universal clean energy and access to medicines, that all have a direct relationship with biodiversity.
- Goals and targets that relate to the underlying global "life support systems" which are a foundation for society and development. Such goals recognize the importance of 'healthy ecosystems' and address natural resource limits or environmental limits ('planetary boundaries', Griggs *et al*, 2013). They could take the form of an integrated landscape goal (see TST Issues brief on forests) or the equivalent for oceans. Such goals are in some ways similar to MDG 7 on environmental sustainability, in their separation of environmental or ecological issues from other, parallel goals on food, health, energy etc.
- Goals and targets relating to "enabling factors" such as education, equality, gender equity, governance, participation and human rights, which do not depend directly on biodiversity but which may influence and be influenced by biodiversity change.

The Aichi Biodiversity Targets and associated indicators (along with means of implementation included in the Strategic Plan for Biodiversity 2011-2020) can provide specific inputs across the range of possible SDG types outlined above. This is because, as noted elsewhere:

- Conserving biodiversity can directly help to achieve human development targets.
- Many of the pressures and threats to biodiversity and sustainable development are the same, so there are often co-benefits for both biodiversity and development in acting on those pressures.

Considering the four types of potential SDG in turn:

1. Overarching goals

A goal or goals to reduce poverty, in its broadest sense, or to enhance human well-being requires inclusive measures of 'wealth' or progress that recognize natural and human capital alongside economic capital. Target 2 encourages the integration of biodiversity values into national accounting systems, which would support the development of such measures and ensure that biodiversity is recognized as an important component. Target 14 focuses on maintaining natural capital of particular importance for well-being, and so is also relevant.

2. Topical goals

Considering the Aichi Biodiversity Targets and the development themes stemming from the MDGs that are likely to form part of the post-2015 agenda, what can be seen is that, in particular, biodiversity targets are relevant to issues including food security, water security and health.

Achieving Targets 7 on sustainable agriculture, aquaculture and forestry and 13 on safeguarding genetic diversity (particularly cultivated plants and livestock) would help to address hunger by helping to ensure long term food supply. In the same way, Target 6 on rebuilding fisheries in a sustainable way enhances food security whilst also contributing to marine biodiversity conservation. The establishment and effective management of marine protected areas (under Target 11) has been shown to be important in rebuilding fisheries. Other targets, including those dealing with perverse subsidies (Target 3) and sustainable production and consumption (Target 4) also support sustainable food systems that are the key to both long term development and the maintenance of biodiversity.

Achieving Target 8 on reducing pollution has a range of co-benefits including for enhanced water quality (water security) and air quality (health and climate change).

3. Foundational (life support system) goals

Maintaining biodiversity is central to ecosystem health and biodiversity should inherently be a fundamental part of any goal relating to planetary life support systems. Targets 5, 11, 14 and 15 relate directly to protection, maintenance and restoration of 'ecosystems', whilst Target 12 relates to preventing species extinction.

4. Enabling factors

The potential goals and targets discussed above have to be further supplemented with targets that address structural barriers and create the enabling conditions for the goals to be achieved. A range of 'other' kinds of goal may be included here, such as those relating to equity and empowerment, including education and gender issues.

Enabling conditions will need to be realised in areas such as capacity and knowledge, institutions and governance, public policy, and investment and finance (Nilsson *et al*., 2013). Achieving the Aichi Biodiversity Targets under strategic goal E (especially Targets 18, 19 and 20), that address various means of implementation of the Strategic Plan for Biodiversity 2011-2020, would provide a basis for such goals. Equally, Target 1 on awareness of biodiversity and actions required to conserve it would contribute to education and commitment to act in support of sustainable development.

23.5 CONCLUSIONS

This analysis, drawing heavily on early scholarly analysis on the SDGs as well as input documents to the formal SDG discussion process in 2013-4, suggests that there are many entry points for biodiversity in what may be proposed as a structure for the SDGs, and illustrates the importance of biodiversity being included. It is important however to note the potential misalignment between actions to maintain biodiversity and the achievement of development goals. For example, efforts to halve or bring to zero habitat destruction (Target 5), increase protected areas (Target 11) or bring extinction rates to zero (Target 12) could in the short term reduce access to ecosystem services whilst diverting economic resources that could be used to meet other development objectives. Considerable attention to human well-being must be paid when developing and implementing plans to reach these targets.

The role of biodiversity and ecosystem services for human well-being and development are complex areas for policy formulation and universally relevant goals and targets will be inherently difficult to measure and monitor. Practical considerations regarding availability of indicators may therefore influence the choice and/or formulation of targets. However such goals and targets are formulated, this chapter highlights critical interlinkages between different development sectors – a key debate in the post-2015 preparation process. How to deal with these interlinkages in a pragmatic way is open to discussion. Possibilities include separate or integrated goals on poverty eradication and environmental sustainability, while a mixed approach might be conceivable where for example an integrated goal on food security and sustainable use is complemented with a separate goal on earth system functioning and/or limits. Experience with the MDGs suggests that encapsulating biodiversity solely within a distinct, stand-alone environmental goal may not be sufficient to ensure an integrated approach to sustainable development. The Aichi Biodiversity Targets provide a plurality of existing global commitments that can be incorporated in a range of ways into an emerging SDG framework, thereby strengthening their visibility as well as the SDGs to which they are applied.

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