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86



# MANAGING ECOSYSTEMS IN THE CONTEXT OF CLIMATE CHANGE MITIGATION: A review of current knowledge and recommendations to support ecosystem-based mitigation actions that look beyond terrestrial forests





UNEP WCMC

Convention on Biological Diversity

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# Managing ecosystems in the context of climate change mitigation: A review of current knowledge and recommendations to support ecosystem-based mitigation actions that look beyond terrestrial forests.

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# FOREWORD



Studies such as the fifth assessment report of the Intergovernmental Panel on Climate Change show that recent anthropogenic emissions of greenhouse gases are the highest in history, and that recent changes in the climate have had widespread impacts on human and natural systems. Many species have already been affected through shifts in their geographic ranges, seasonal activities, migration patterns, abundances and species interactions in response to ongoing climate change. At the same time, the fourth edition of the Global Biodiversity Outlook suggests that based on current trends, pressures on biodiversity will continue to increase at least until 2020, and that the status of biodiversity will continue to decline.

However we also know that the conservation and restoration of ecosystems play a key role in mitigating climate change by enhancing carbon sequestration and reducing greenhouse gas emissions. The Paris Agreement recognizes this important role and encourages Parties to conserve and enhance, as appropriate, sinks and reservoirs of greenhouse gases.

Indeed conserving natural terrestrial, freshwater and marine ecosystems and restoring degraded ecosystems, including their genetic and species diversity, is essential for achieving the overall goals of the Convention on Biological Diversity (CBD), the United Nations Framework Convention on Climate Change (UNFCCC) and the United Nations Conventions to Combat Desertification (UNCCD), including the land degradation neutrality goal, because ecosystems play a key role in the global carbon cycle and in adapting to climate change, while also providing a wide range of ecosystem services that are essential for human well-being and the achievement of the Sustainable Development Goals.

In decision X/2, the Conference of the Parties adopted the Strategic Plan for Biodiversity 2011-2020 and its Aichi Biodiversity Targets, including Target 15 which aims, by 2020, to enhance ecosystem resilience and the contribution of biodiversity to carbon stocks, 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.

In decision X/33, the Conference of the Parties to the CBD requested the Executive Secretary, in collaboration with relevant international organizations, to identify areas which, through conservation and restoration of carbon stocks and other ecosystem management measures, might have high potential for climate change mitigation, and make this information widely available.

The present report has been prepared by the United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC) for the Secretariat of the CBD, to summarize current knowledge on the potential contribution of ecosystem-based approaches to climate change mitigation, and additional benefits that such approaches can provide. Both the UNFCCC and the CBD have provided substantial guidance involving the conservation, sustainable use and restoration of forests, and actions related to these are already a part of many countries' strategies to address climate change. Therefore, the study focuses on a number of other ecosystem types, beyond forests, with a high potential to contribute to climate change mitigation. The purpose of the study is to provide biodiversity managers with a reference document on the additional benefits of managing these ecosystems for carbon sequestration and storage.

I hope this report will support Parties, other Governments, and stakeholders in implementing the Convention while also maximizing synergies with climate change mitigation and adaptation, disaster risk reduction, and sustainable development.

Braulio Ferreira de Souza Dias
Executive Secretary
Convention on Biological Diversity

## ACKNOWLEDGEMENTS

This document was developed through a combination of literature review and two rounds of expert consultation on the current state of knowledge about the potential of ecosystem-based approaches for climate change mitigation, taking into account the additional benefits that such approaches can provide. A draft version was published for peer review in January 2016, and revised based on comments received.

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# EXECUTIVE SUMMARY

Ecosystem management can play an important role in climate change mitigation and adaptation if current practices are evaluated and improved to move towards sustainability. Terrestrial and coastal ecosystems store more than five times as much organic carbon as there is carbon in the atmosphere, whilst net emissions from land cover change and ecosystem degradation are responsible for about 10 % of the total yearly anthropogenic carbon emissions.

Sustainable land use practices that maintain carbon stocks or enhance sequestration can provide a range of additional benefits that are crucial for sustainable development. Parties to the CBD have decided to promote the implementation of ecosystem-based approaches for climate change mitigation including the conservation, sustainable management and restoration of natural forests, grasslands, peatlands, mangroves, salt marshes[1] and seagrass beds. Aichi Target 15 calls on Parties to enhance ecosystem resilience and the contribution of biodiversity to carbon stocks, thereby contributing to climate change mitigation and adaptation.

This review summarizes current knowledge on the potential of ecosystems beyond terrestrial forests to contribute to climate change mitigation. It provides information on the capacity of existing management techniques for peatlands, grasslands and savannahs, coastal ecosystems and croplands to sustain and enhance carbon stocks and carbon sequestration. Recommendations are made for maximizing synergies with climate change adaptation, disaster risk reduction, sustainable development, environmental protection and biodiversity conservation, including through landscape-scale approaches that take into account the legitimate interests, knowledge and capacities of all stakeholders. Available evidence concerning the importance of biodiversity for ecosystem resilience and functioning, and thus the long-term effectiveness of ecosystem-based mitigation actions, is also presented.

A key message from this study is that knowledge is already available to guide concrete planning and target setting regarding ecosystem-based approaches to climate change mitigation. Relevant information has been compiled by the Intergovernmental Panel on Climate Change (IPCC), donor-funded projects, certification schemes and voluntary project standards. Lessons learned from climate change policies and actions targeting forests can inform actions related to other ecosystems.

A recommended first step in designing ecosystem-based mitigation approaches is to assess the extent and drivers of the degradation and conversion of ecosystems, together with opportunities for their restoration and sustainable use. Planning at landscape level, as well as active stakeholder engagement, can help to develop efficient and effective measures. A review of incentives related to land use can detect opportunities to make climate-friendly forms of management more economically viable. Donors who are interested in supporting integrated land management may wish to invest in the collection of regionally specific baseline data for the planning of mitigation and adaptation actions based on ecosystems. While many ecosystem-based mitigation measures can provide win-win solutions, some forms of ecosystem management such as afforestation of, or biofuel cultivation on, peatlands and natural grasslands may also pose significant risks. Thus, likely outcomes of such actions need to be carefully assessed to avoid unintended consequences for climate change mitigation and adaptation, disaster risk reduction, biodiversity conservation and local livelihood conditions.

We anticipate that the information provided in this document can support Parties in their implementation of CBD Decision X/33, as well as in their efforts to achieve Aichi Target 15.

1 Note that under the IPCC, the term 'tidal marshes' is used instead of 'salt marshes'.

# 1. INTRODUCTION

It is widely recognized that improving the way in which ecosystems are managed and used can be a key component in efforts to mitigate climate change and adapt to its consequences. According to recent estimates, terrestrial and coastal ecosystems store more than five times as much carbon in plant biomass and soil organic matter as is currently contained in the atmosphere, and net emissions from land use change and degradation of vegetation and soils are responsible for about 10 % of the total anthropogenic carbon emissions including those from fossil fuel combustion (see Box 1). Some forms of land use, especially those that affect fire occurrence, can also have an appreciable impact on emissions of non-carbon greenhouse gases (such as $N_2O$) and aerosols (including black carbon) (Smith et al. 2014). At the same time, terrestrial ecosystems not affected by land use change remove a net amount of around 2.5 gigatons of carbon (Gt C) per year from the atmosphere (Ciais et al. 2013). While in the past the terrestrial carbon sink has mostly been attributed to forests, a recent analysis of remote sensing data suggests that other ecosystems, in particular dryland systems such as tropical savannahs and shrublands, also make a significant contribution. The sink function of these water-limited ecosystems is very sensitive to climate variations (Liu et al. 2015).

A number of studies have further highlighted the fact that changes in land use can not only influence heat retention in the atmosphere through emissions and removals of greenhouse gases, but can also have an impact on global mean temperature through changes in biophysical characteristics such as surface albedo (i.e. the extent to which sunlight is reflected back from ground cover rather than absorbed and transformed into heat), evapotranspiration (increasing the moisture content of the atmosphere and providing local cooling) and surface roughness (affecting air movement) (see Myhre et al. 2013 for an overview of the discussion). Such effects are generally most pronounced in the case of transitions from one ecosystem type to another (e.g. conversion of forest to cropland), but can also occur when an ecosystem is significantly changed through management (e.g. replacement of broadleaved forest with conifer plantations, see Naudts et al. 2016). There are still large uncertainties around the net impact of these processes on global mean temperature. The current state of knowledge seems to suggest that impacts through changes in the hydrological cycle tend to offset the impacts of albedo changes, and that at the global scale both types of effects are significantly smaller than effects caused by greenhouse gas emissions from land cover change (Myhre et al. 2013).

Land use practices that contribute to climate change mitigation by maintaining carbon stocks and allowing additional carbon to be taken up from the atmosphere can often provide additional benefits for climate change adaptation, disaster risk reduction, sustainable development, environmental protection and biodiversity conservation. They can thus form a cornerstone of efficient policies for the integrated use of land and natural resources.

The concepts of ecosystem-based mitigation (i.e. managing ecosystems in a way that counteracts anthropogenic climate change, in particular by reducing emissions of greenhouse gases and enhancing removals of greenhouse gases from the atmosphere) and ecosystem-based adaptation (i.e. managing ecosystems in a way that uses biodiversity and ecosystem services to help people adapt to the adverse effects of climate change) are thus closely related, and can often be implemented in synergy.

Parties to the CBD have recognized the close interlinkages between biodiversity and climate change in a number of decisions. In decision X/33, the Conference of the Parties invited Parties and other Governments, according to national circumstances and priorities, to implement ecosystem-based approaches for mitigation through, for example, conservation, sustainable management and restoration of natural forests, natural grasslands and peatlands, mangroves, salt marshes[2] and seagrass beds. Decision XII/20 further encourages Parties, and invites other Governments and relevant organizations, to promote and implement ecosystem-based approaches to climate change-related activities and disaster risk reduction. Target 15 of the Strategic Plan for Biodiversity 2011-2020 aims to enhance, by 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks, through conservation and restoration, thereby contributing to climate change mitigation and adaptation.

The present study aims to support Parties to the CBD in their implementation of decisions X/33 and XII/20 and the achievement of Aichi Target 15, by reviewing available knowledge on the current and potential role of ecosystems in

2 Note that under the IPCC, the term 'tidal marshes' is used instead of 'salt marshes'.

**Box 1: The contribution of land use change and ecosystem degradation to anthropogenic carbon emissions**

The impact of land use change on global anthropogenic carbon emissions is determined by the balance between changes that cause emissions (such as conversion of natural ecosystems to agriculture), and changes that lead to increased carbon sequestration (such as abandonment or afforestation/reforestation of cropland and restoration of degraded forest).

According to the 5th Assessment Report of the Intergovernmental Panel on Climate Change (IPCC), the net impact of land use change and ecosystem degradation has been responsible for more than 1 Gt C of anthropogenic emissions per year during the period of 1980-2009 (Ciais et al. 2013). Gross emissions from land use change (i.e. the sum of all emissions from converted and degraded areas, without subtracting the carbon that is sequestered on areas where a reverse land use change leads to carbon uptake) are several times higher than the net figures. For example, it has been estimated that gross emissions from tropical deforestation and degradation amounted to 3.0 (+/- 0.5) Gt C during the 1990s, and 2.8 (+/- 0.5) Gt C during the 2000s.

It is considered 'more likely than not'* that net carbon dioxide emissions from land use change have decreased during the first decade of this century as compared to the 1990s. However, this change is within the uncertainty range of the estimates (see Table 1). The significant increase in carbon emissions from fossil fuel combustion and cement production over the past decades also contributes to an estimated decrease in the relative share of net emissions from land use change in total anthropogenic carbon emissions, from over 20 % during the 1980s to around 12 % during the 2000s (see Table 1).

**Table 1: Development of anthropogenic carbon dioxide emissions and carbon sequestration on areas not affected by land use change (the 'residual land sink') between 1980 and 2009 (all figures following Ciais et al. 2013)**

| | 1980–1989 | 1990–1999 | 2000–2009 |
|---|---|---|---|
| Net emissions from land use change and degradation (Gt C / year) | 1.4 +/- 0.8 | 1.5 +/- 0.8 | 1.1 +/- 0.8 |
| Emissions from fossil fuel combustion and cement production (Gt C / year) | 5.5 +/- 0.4 | 6.4 +/- 0.5 | 7.8 +/- 0.6 |
| Contribution of land use change and degradation to total anthropogenic $CO_2$ emissions (%) | 20.3 | 19.0 | 12.4 |
| Residual land sink (Gt C/year) | 1.5 +/- 1.1 | 2.6 +/- 1.2 | 2.6 +/- 1.2 |

* In the terminology adopted by the IPCC, 'more likely than not' indicates that a statement has high uncertainty associated with it, but its probability is assessed to be > 50 %. (See Ciais et al. 2013, p. 467).

climate change mitigation and providing advice on the management of ecosystems to maintain and enhance carbon stocks and carbon sequestration, and where relevant avoid or reduce emissions of other greenhouse gases, while maximising synergies with climate change adaptation, the conservation of biodiversity and sustainable development. If well designed, ecosystem-based mitigation actions can further establish synergies with the achievement of several other Aichi Targets (in particular Target 14 on restoring and safeguarding ecosystems that provide essential services; Target 5 on reducing the loss, degradation and fragmentation of natural habitats; and Target 11 on conserving areas of particular importance for biodiversity and ecosystem services through systems of protected areas and other effective area-based conservation measures), as well as with a number of the Sustainable Development Goals set out in the 2030 Agenda for Sustainable Development that was adopted by the United Nations Sustainable Development Summit in 2015 (see also UNEP/CBD/SBSTTA/20/10).

It is hoped that this information can be used by those involved in implementing the CBD to identify opportunities for such synergies and reach out to other stakeholders, including those working on climate change and land degradation issues, in order to promote the development of coherent policies and actions relating to ecosystem management. New alliances should be promoted at all levels, from the local to the international.

Among all ecosystem types, the importance of forests for the global carbon cycle has to date been most intensively studied, and actions involving the conservation, sustainable use and restoration of terrestrial forests are already a part of many countries' strategies to address climate change. This report therefore focusses on a number of other ecosystem types that were selected based on their potential to contribute to climate change mitigation and adaptation, their prominence in

land use-related policies, their biodiversity value, and the amount and quality of available literature. Where relevant, references to forest-based mitigation efforts are also made.

The list of ecosystems covered is not exhaustive. For example, inland waters, offshore marine ecosystems and urban ecosystems have not been dealt with, although there is an emerging body of evidence demonstrating their role in climate regulation, and some options to enhance their potential for climate change mitigation are being explored (see e.g. Laffoley et al. 2014; Lal & Augustin 2011; Lutz & Martin 2014; Raymond et al. 2013). Urban ecosystems are a special case, as they can contribute to climate change mitigation not only by sequestering and storing carbon, but also by reducing energy requirements for thermal regulation in buildings and for transport to natural areas for recreation (see Box 2).

**Box 2: Combining climate change mitigation and adaptation in the management of urban ecosystems**

More than half of the world's population now live in urban areas and cities are expected to absorb much of the population growth projected for the future (United Nations 2015). The influence of urbanization on greenhouse gas emissions from land use change as well as from fossil fuel usage is thus an important concern.

The expansion of built-up space causes the loss of considerable amounts of biomass carbon, while impacts on soil carbon have been found to be more variable (Pataki et al. 2006; Pouyat et al. 2006; Seto et al. 2012). Despite a paucity of data due to sampling difficulty, available evidence seems to suggest that the sealing of soils under impervious cover reduces carbon content and sequestration capacity as compared to soils under natural vegetation. However, whether and to what extent this reduction takes place under a wide range of conditions is still uncertain (Edmondson et al. 2012; Raciti et al. 2012). The potential of non-sealed urban soils to store and sequester organic carbon depends on a number of factors including climate, soil type and land use (Pouyat et al. 2006; Scalenghe & Marsan 2009)*.

Along with buildings and infrastructure, towns and cities host a variety of managed and unmanaged ecosystems such as parks and recreational grounds, gardens, brownfields, urban forests, green roofs, and plots used for urban agriculture. Although often overlooked, these ecosystems make a substantial contribution to resolving the environmental challenges faced by growing urban populations. Managing them as part of a 'green infrastructure' and planning for ecosystem services can enhance that contribution (Collier et al. 2013).

The management of urban ecosystems is a good example of the potential to achieve synergies between climate change mitigation and adaptation. Climate change is expected to exacerbate problems such as the urban heat island effect and associated health impacts, low air quality and the overloading of storm drains after heavy precipitation events (Campbell et al. 2009; Grimm et al. 2008; Pickett et al. 2011). As has been demonstrated in a number of studies, urban ecosystems can help to address these issues while at the same time providing additional benefits such as conservation of biodiversity or improved mental and physical well-being of local residents (e.g. Alexandri & Jones 2008; Bowler et al. 2010; Connop et al. 2013; Kitha & Lyth 2011; Mentens et al. 2006; Wong et al. 2003). By reducing the need for technological solutions to heating and cooling of buildings and making urban areas more attractive for recreation, adaptation measures based on urban ecosystems can at the same time reduce the consumption of fossil fuels (Castleton et al. 2010; Grimm et al. 2008; Pataki et al. 2006). These benefits for climate change mitigation come in addition to potential increases in carbon storage and sequestration in urban vegetation and soils (Davies et al. 2011; Lal & Augustin 2011; Pataki et al. 2006).

Incorporating native biodiversity into plans for urban green infrastructure can increase the resilience of urban ecosystems and further support the provision of multiple ecosystem services in cities, including cultural services (Connop et al. 2016).

A number of national and international research programmes and initiatives are currently working to enhance the body of knowledge and practical experience that can guide the development of sustainable and resilient urban structures that apply nature-based solutions to current and future challenges. Some examples from Europe include AMICA[3], GREENSURGE[4] and TURAS[5].

* A possible approach to climate change mitigation in urban ecosystems that has recently attracted attention is that of increasing the accumulation of inorganic carbon in artificial urban soils (e.g. Washbourne et al. 2015). However, a full discussion of this option (which could be classified as a geo-engineering approach) is beyond the scope of this report.

3 http://www.amica-climate.net/

4 http://greensurge.eu/

5 http://www.turas-cities.eu/

# 2. CARBON STOCKS AND FLOWS IN DIFFERENT TYPES OF ECOSYSTEMS

Globally, it has been estimated that living vegetation, dead plant matter and the top 2 m of soils together contain between 2,850 and 3,050 Gt C. In peatlands and permafrost soils, significant amounts of carbon (more than 2,000 Gt according to some current estimates) are also stored at depths greater than 2 m (Ciais et al. 2013). The spatial distribution of biomass and soil carbon[6] across different regions and biomes is highly uneven. See Figures 1 – 3 and Table 2 for (a) a global overview of carbon stocks and flows, (b) a map showing the distribution of terrestrial carbon stocks, and (c) a comparison of the areal extent and average carbon stocks of different ecosystem types.

An overview of carbon stocks and flows in different types of ecosystems is provided in the following sections. The potential future impacts of climate change and socio-economic developments on ecosystems are also discussed, bearing in mind that ecosystem-based mitigation efforts may need to anticipate and address emerging threats. Management activities that strengthen the resilience of ecosystems to climate change and other stressors can support the permanence of achieved mitigation outcomes.

When considering the information provided, it should be noted that any classification of ecosystems is to some degree subjective, and there are transitions and overlaps between the different types. For example, tundra areas and some tropical forests contain a large proportion of peat soils and can thus also be thought of as peatlands, and wooded savannahs may be considered forest areas or grasslands depending on circumstances[7]. Differences in the ecosystem definitions used by authors are part of the reason for the range of uncertainty for some of the estimates provided.

Differences in terminology can also be a source of confusion in communication between the different 'communities' involved in the development of policies and actions related to ecosystem management. For example, care should be taken not to confuse the term 'natural' (which is commonly used by ecologists to describe ecosystems whose species composition has not been significantly modified by humans, or whose vegetation is mainly composed of naturally regenerating species as opposed to planted ones) with the term 'unmanaged' as used under the United Nations Framework Convention on Climate Change (UNFCCC). A 'natural grassland' as described in this report, may well be 'managed' under the terminology of the UNFCCC, e.g. through livestock grazing. As anthropogenic emissions are mostly caused when previously unmanaged ecosystems come under human use, when management is intensified, or when unsustainable use of a managed ecosystem leads to ongoing degradation, this report is mainly concerned with ecosystems that are 'managed' in the terminology of the UNFCCC, or those that may change from being unmanaged to being managed. (See IPCC 2000 and IPCC 2010 for further background.)

6 Throughout this report, references to soil carbon refer to organic carbon only.

7 Savannahs are characterized by the co-dominance of trees and grasses in various proportions and compositions, ranging from tropical grasslands where trees are virtually absent to forest-like formations including a continuous grass layer such as the Miombo or Cerrado woodlands; exact definitions vary between authors (see e.g. Scholes & Archer 1997, Ratnam et al. 2011).

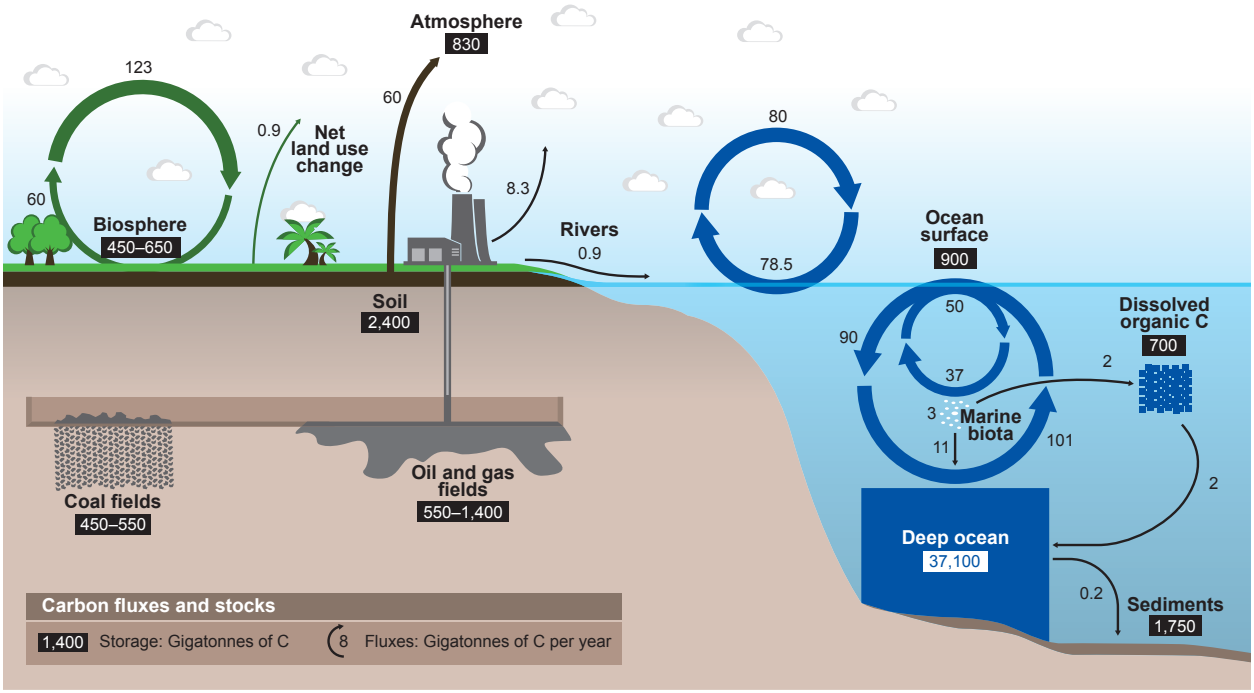



**Figure 1:** The global carbon cycle. Source: SCBD (2015)

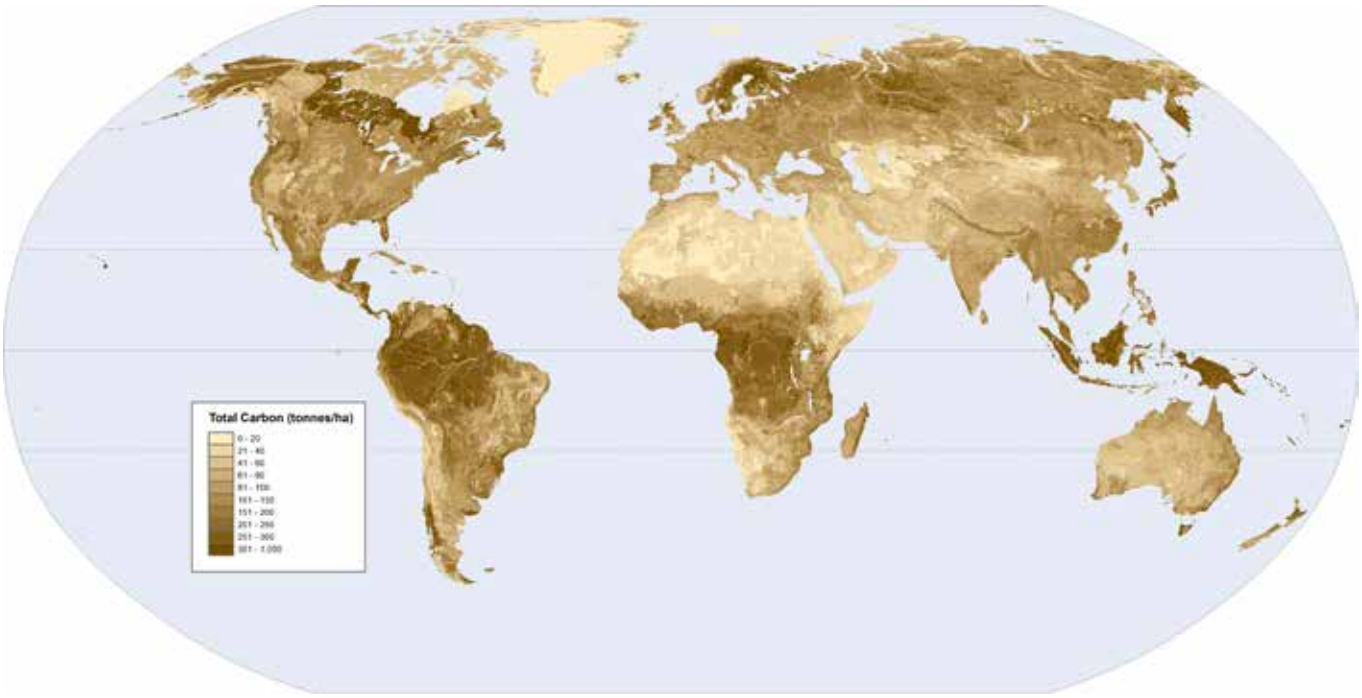



**Figure 2:** Global map of terrestrial biomass and soil organic carbon stocks. Source: Scharlemann et al. (2011)

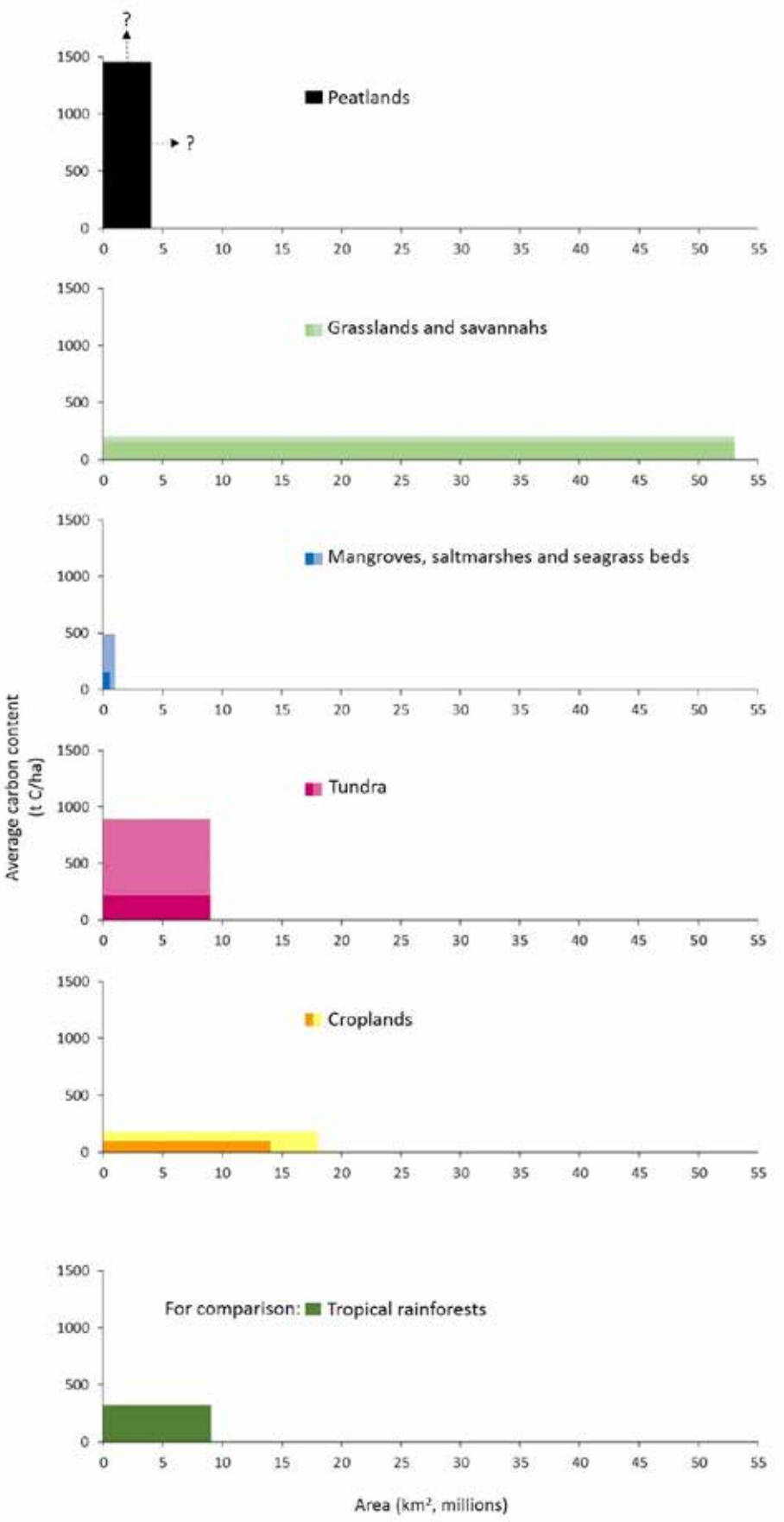



**Figure 3:** Comparison of major ecosystem types according to their global areal extent and average organic carbon stocks per hectare

Where the cited sources provide values as a range rather than a single figure, this is indicated by darker shading for the lower estimate and lighter shading for the upper values provided. Dotted arrows on the peatlands graph reflect the fact that new peat reserves have been discovered since the most recent global estimates of peatland area and average carbon stocks were developed. For the precise figures represented in the graphs, see Table 2. For sources regarding these figures, see next page.

**Table 2: Global areal extent and average organic carbon stocks of major ecosystem types**

| Ecosystem type | Peatland† | Grassland/ savannah | Mangrove, salt marsh, seagrass bed | Tundra | Cropland | Tropical rainforest | Total for global land area |
|---|---|---|---|---|---|---|---|
| Areal extent ($km^2$) | 4,009,238 | 52,500,000 | 489,000–1,152,000 | 8,800,000 | 13,500,000–18,766,440 | 9,400,000 | 149,000,000 |
| Average organic carbon stock (t C/ha) | 1,450 | 150-200 | 140-480* *soil carbon included up to 1m only | 218-890 | 95-177** **soil carbon only, biomass carbon not included | 320 | 191-205*** ***soil carbon included up to 2m only |

† Note that due to overlapping definitions (see above) some of the area identified as peatlands is also included in the area estimates for the other ecosystem types. At the same time, the area estimate for peatlands is likely to be too low, because due to incomplete information on the distribution of organic soils some peatlands are not identified as such, but instead accounted only under one of the other ecosystem types.

**Sources for Figure 3 and Table 2**: Tropical rainforest: Area: Joosten 2015 (for moist and humid tropical forests); average carbon stock: Parish et al. 2008; Peatlands: Area: Page et al. 2011; average carbon stock: Parish et al. 2008; Grasslands: Area: Suttie et al. 2005; average carbon stock: Grace et al. 2006 and Amthor et al. 1998 (for tropical savannah), Epple 2012, (for steppe); Coastal ecosystems: Area: Pendleton et al. 2012 (confident estimate and highest estimate); average carbon stock: Murray et al. 2011 (incl. soil carbon up to 1m depth only); Tundra: Area Joosten 2015; average carbon stock: Joosten 2015, combined with Tarnocai et al. 2009 (average soil carbon value for permafrost zone); Cropland soils: Area: Eglin et al. 2011, FAO 2014a; average carbon stock (soil carbon only): Eglin et al. 2011; Global land area totals: Area: Douglas et al. 2002; average carbon stock: own calculation based on Ciais et al. 2013 (incl. soil carbon up to 2m depth only).

## PEATLANDS

The carbon stock of known peat reserves has been estimated at over 550 Gt, despite peatlands covering only about 3 % of the global land surface (Parish et al. 2008, cf. Table 2). At the same time, new peat reserves are still being discovered in natural ecosystems, and not all peat soils in areas used for agriculture and forestry are recognized and/or recorded as such (see e.g. Draper et al. 2014; Parish et al. 2008; Scharlemann et al. 2014). On average, peatlands are estimated to hold about 1,500 tons of soil carbon per hectare, i.e. about 10 times as much as a typical mineral soil. For tropical peatlands, the values can be more than twice as high, depending on local topography and hydrological conditions (Parish et al. 2008).

Carbon sequestration occurs relatively slowly in many types of peatlands (with the notable exception of naturally forested peatlands, where biomass carbon plays a significant role). For example, Turunen et al. (2002) estimated that the average long-term carbon accumulation rate for undrained Finnish mire areas is around 185 kg per hectare per year. Dommain et al. (2011) calculated average soil carbon sequestration rates for peat domes in South East Asia over the Holocene period, and found values of 313 kg per hectare per year for Central Kalimantan and 770 kg per hectare per year for coastal sites. These figures may seem small if compared for example to the sequestration rate of 5 t C per hectare per year that can temporarily be reached in a young, fast growing forest stand (Malhi et al. 1999). However, they are quite comparable to the average carbon sequestration rate of 490 kg per hectare per year that Lewis et al. (2009) found for tropical old-growth forest. The relevance of carbon sequestration in peatlands becomes greater as longer time horizons are considered, since peat accumulation can continue at the same rate for millennia if environmental conditions remain beneficial.

Although a significant proportion of the known global peatland resource is still in a relatively undisturbed state, the rate of peatland disturbance has been steadily increasing, leading to significant greenhouse gas emissions from decomposition of organic matter in drained peat and from peat fires (Biancalani & Avagyan 2014; Cris et al. 2014; Parish et al. 2008). According to Joosten et al. (2012), around 15 % of the global peatland area is affected by disturbance. Of this, it has been estimated that 50 % can be attributed to agriculture, 30 % to forestry operations, 10 % to peat extraction, and 10 % to infrastructure development (Parish et al. 2008). The fact that many converted peatlands (e.g. former fenlands that have been claimed for agricultural or forestry uses) are no longer recognized as such often contributes to their inappropriate management. Most studies agree that the average annual loss of peat carbon has now gone up to more than 0.3 Gt per year (i.e. more than 3 % of all anthropogenic carbon emissions), while some estimate it to be as high as 2 Gt C in those years with a high incidence of peat fires (Biancalani & Avagyan 2014; Hooijer et al. 2010; Joosten 2015). Peat fires are in most cases a direct consequence of peatland drainage, and can have a major impact on total annual anthropogenic greenhouse gas emissions from land use change. For example, it has been estimated that the severe peat fires occurring

in Indonesia in 2015 alone caused emissions equivalent to the release of around 0.48 Gt C as carbon dioxide[8] (World Bank 2015, based on figures from the Global Fire Emissions Database). Global hotspots of anthropogenic emissions from peatlands are Southeast Asia (where peat is mostly drained for agroforestry and other forms of agriculture), and Europe (where peat is drained for agriculture, livestock grazing and forestry, and peat extraction also plays a role) (Joosten 2010; Joosten 2015).

Expected impacts of climate change on peatlands depend on the climatic zone as well as on site conditions, and may lead to an increase in emissions or enhanced sequestration, depending on location. It is not yet possible to predict a general trend (Ciais et al. 2013; Parish et al. 2008; Smith et al. 2014; Strack 2008). However, peatlands where peat-forming vegetation is intact or has been restored are likely to be more resilient to climate change impacts than degraded ones (Parish et al. 2008).

## GRASSLANDS AND SAVANNAHS

Temperate, tropical and sub-tropical grasslands and savannahs occur naturally over an area that covers about a quarter of the world's terrestrial surface. In addition, semi-natural grasslands have formed in many other regions where forests were cleared to create space for grazing livestock, covering another 15 % of the Earth's land mass (Epple 2012; McSherry & Ritchie 2013; Suttie et al. 2005). Due to their large area, grasslands play a significant role in the terrestrial carbon balance (Grace et al. 2006; Liu et al. 2015; Poulter et al. 2014). The total amount of carbon stored in the natural grassland biomes has been estimated at around 470 Gt, i.e. around one fifth of the carbon contained in terrestrial vegetation and topsoils worldwide (Ciais et al. 2013; Trumper et al. 2009). Average grassland carbon stocks are on the order of between 150 and 200 t per hectare, with high variability depending on climate and soil type (Epple 2012; Grace et al. 2006). About 80 % of ecosystem carbon stocks in grasslands are stored in the soil (Ciais et al. 2011).

Among the main processes influencing greenhouse gas emissions and sequestration in grassland ecosystems are conversion to cropland, grazing by wild and domesticated animals, fire and climate variability and change (Liu et al. 2015; McSherry & Ritchie 2013; Poulter et al. 2014; Safriel et al. 2005; Victoria et al. 2012). In tropical savannahs, harvesting of wood can also be an issue. Information on the percentage of grasslands that is subject to livestock grazing is hard to obtain, particularly for extensive and mobile grazing systems in the natural grassland biomes (Sanderson et al. 2002). Luyssaert et al. (2014, Supplementary Material) assume that globally between 28 and 34.1 million $km^2$ of grasslands are used as pasture, which corresponds to between 53 and 65 % of the world's grassland area according to Suttie et al. (2005). Between 18.3 and 20.5 million $km^2$ of these grazed lands are situated in natural grasslands and savannahs (Luyssaert et al. 2014, Supplementary Material).

Because of their fertile soils, much of the original area of grassland ecosystems has already been cleared for the cultivation of crops, i.e. some 70 % of temperate grasslands and 50 % of tropical and sub-tropical savannahs, especially in North America, South Eastern Europe and Africa north of the equator (Epple 2012; Joosten 2015; Safriel et al. 2005). In some parts of Eastern Europe and Central Asia, this conversion trend has partly been reversed following the collapse of the former Soviet Union (see also the section on abandoned croplands) (Kurganova et al. 2015).

Overgrazing leading to degradation and soil erosion is a serious problem in the remaining grasslands of many regions, including sub-Saharan Africa, Central Asia, China and South America (Epple 2012; Golluscio et al. 2009; Jiang et al. 2006; Lebed et al. 2012). Overgrazing can be caused by a variety of factors, including high numbers of livestock per hectare as well as poor spatio-temporal management of livestock distribution that fails to take into account carrying capacity at the site level as well as seasonal changes in fodder availability and vegetation resilience (McGahey et al. 2014). A large part of the world's degraded dryland soils are found in areas whose natural vegetation is grassland, and the rate of desertification is estimated to be higher under pasture than under other land uses such as cropland (Steinfeld et al. 2006). It is further estimated that drylands affected by land degradation currently cover around 4-8 % of the global

8 Note that in order to facilitate comparisons, values for carbon stocks and carbon emissions are both provided in units of tons of carbon throughout this report. The conversion factor of carbon to carbon dioxide is 3.67, i.e. when 1 ton of carbon is released in the form of carbon dioxide, this will produce 3.67 tons of carbon dioxide.

land area (Safriel et al. 2005), and that around 0.3 Gt C per year are lost from dryland soils as a result of unsustainable agricultural and pastoral practices (Joosten 2015). As future projections indicate a continued rise in population densities and an increase in frequency and duration of drought in many dryland areas, it is expected that the vulnerability of grasslands to degradation will grow over the coming decades if management practices, as well as property rights and tenure regimes, remain the same (Safriel et al. 2005; Soussana et al. 2013).[9]

The effects of changes in species composition that will occur due to rising temperatures and carbon dioxide concentrations and altered precipitation patterns are still hard to predict (Smith et al. 2014). A number of authors expect that in many savannah areas, increased levels of atmospheric carbon dioxide will shift competition between woody plants ($C_3$ metabolism) and tropical grasses ($C_4$ metabolism) in favour of the former, leading to a potential for greater carbon storage resulting from increased coverage of bushes, shrubs or trees. However, other factors such as nutrient limitation, changes in fire frequency and levels of anthropogenic disturbance make more precise predictions difficult (Howden et al. 2008; Kgope et al. 2010; Lehmann et al. 2014; Midgley & Bond 2015).

## MANGROVES, SALT MARSHES[10] AND SEAGRASS BEDS

Coastal vegetation that is permanently or temporarily flooded by the sea can act as a trap for small particles of organic matter from the water column. This, together with root growth and accumulation of litter, creates highly carbon-rich soils (Donato et al. 2011; Fourqurean et al. 2012; McIeod et al. 2011). The carbon captured in these soils can be stored for centuries or even millennia, as the inundation with sea water slows down the decomposition of organic matter (Crooks et al. 2011; UNEP 2014). The high salt content also prevents the formation of methane. Mangroves, salt marshes and seagrass beds are therefore considered important carbon stores, despite covering only about 50 million hectares, i.e. about 0.1 % of the Earth's surface (Pendleton et al. 2012). Based on conservative estimates from recent literature, the total amount of carbon stored by these three ecosystem types is thought to be between 11 and 25 Gt. This means that coastal ecosystems hold between 0.5 and 1.2 % of the world's biomass and topsoil carbon.

Mean values of carbon stocks per hectare are highest for mangroves, as the tree biomass contains on average about 150 t C per hectare in addition to soil carbon stocks of around 320 t per hectare (Siikamäki et al. 2012). For some regions, even considerably higher stocks have been found. For example, Donato et al. (2011) arrived at an average total value of 1,023 t C per hectare for biomass and soil organic carbon in mangrove forests across the Indo-Pacific region. Factors influencing the spatial distribution of soil carbon in mangrove ecosystems include climate, exposure to waves and tidal fluctuation, salinity, sediment supply and nutrient concentrations (Adame et al. 2013; Jardine & Siikamäki 2014; McIeod et al. 2011). Conservative estimates of the mean carbon stocks in salt marshes and seagrass beds are on the order of 260 t per hectare and 140 t per hectare, respectively (Murray et al. 2011). Estimates of average carbon sequestration rates are around 1.63 t C per hectare per year for mangroves, 1.51 t C per hectare per year for salt marshes and 1.38 t C per hectare per year for seagrass beds (McIeod et al. 2011; Murray et al. 2011; Nellemann et al. 2009). One source of uncertainty in assessing the total contribution of coastal ecosystems to the global carbon balance is the limited understanding of the fate of exported organic matter, including soil and biomass particles or dissolved organic compounds originating from the site itself, as well as sediment particles of external origin that are either not retained, or re-suspended following disturbance (Ciais et al. 2013; Donato et al. 2011; Laffoley & Grimsditch 2009).

All three types of ecosystem are under high pressure from human activity, including conversion to agriculture, aquaculture, settlements or coastal infrastructure (especially for mangroves and salt marshes), changes in sediment transport due to flood control and coastal defence measures, and pollution with excess nutrients and chemicals contained in run-off from terrestrial areas (CEC 2016; Epple 2012; UNEP 2014; Valiela et al. 2009; Waycott et al. 2009). Between 30 and 50% of the area originally covered by the three ecosystem types is believed to have been lost over the last century alone (Irving

9 A new global synthesis of knowledge on land degradation (status and trends; impacts on biodiversity, ecosystem services and human well-being; restoration opportunities) will become available over the next few years through the Thematic Assessment on Land Degradation and Restoration of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), see http://www.ipbes.net/work-programme/land-degradation-and-restoration.

10 Note that under the IPCC, the term 'tidal marshes' is used instead of 'salt marshes'.

et al. 2011). Current average rates of area loss are estimated to be between 1 and 2 % per year for each ecosystem type, leading to annual global carbon emissions estimated at 0.02-0.12 Gt for mangroves, 0.01–0.07 Gt for salt marshes and 0.04–0.09 Gt for seagrass meadows (Donato et al. 2011; Pendleton et al. 2012). The reduction in area also decreases the potential for continued carbon sequestration in the future (Siikamäki et al. 2012).

Climate change poses an additional threat to coastal ecosystems, as sea level rise and coastal defence structures together are likely to reduce the area that is available for natural coastal vegetation. The coastal vegetation can in principle adapt to sea level rise through soil accumulation, as well as through area expansion on the landward side. However, the extent to which this adaptation is possible in reality will depend on the rate of change and on the availability of space for inland migration in the densely populated coastal regions (CEC 2016; Chmura 2011; Kirwan & Megonigal 2013; Spalding, McIvor et al. 2014). Rates of soil accumulation and associated surface elevation in coastal ecosystems vary over long timescales depending on environmental processes and sea level change. Observations show that sedimentation in mangrove forests is currently keeping pace with local rises in sea level throughout most of the tropics, but not in parts of the Caribbean and South Atlantic or on islands in the Pacific, which are dominated by fringe mangroves (Alongi 2014; Sasmito et al. 2016). Sasmito et al. (2016) reviewed published data on surface elevation change and accretion rates, and compared them with the sea level rise scenarios presented in the 5$^{th}$ Assessment Report of the IPCC (IPCC 2013). They conclude that hydro-geomorphic setting plays a key role in determining mangrove vulnerability to sea level rise, with basin mangroves potentially being less vulnerable. According to their analysis, both basin and fringe mangroves would be able to cope with sea level rise as projected under low scenarios, but their ability to keep pace with rates of sea level rise on the high end of the projections would be outstripped by 2055 and 2070 in fringe and basin mangroves, respectively.

Another expected impact of climate change is a shift in the geographical distribution of mangrove and salt marsh habitat, as warmer temperatures allow mangrove vegetation to expand towards the poles and encroach on habitats currently occupied by salt marshes. Under undisturbed conditions, this shift in vegetation type is likely to lead to an increase in carbon storage capacity (Doughty et al. 2016; Kelleway et al. 2016).

## TUNDRA ECOSYSTEMS

Tundra ecosystems cover just under 10 % of the global land area, mostly in the northern hemisphere (Joosten 2015). Many tundra ecosystems are characterized by peat-forming vegetation. Their role in the climate system is mainly determined by the fate of the large quantities of carbon stored in their soils, especially in the permanently frozen layers. It has been estimated that the permafrost soils of the tundra and boreal forest zone together contain at least 1,700 Gt of carbon, which makes them the largest reservoir of organic carbon worldwide. The spatial distribution of these carbon stocks is however highly uneven and not yet fully understood (Ciais et al. 2013; Tarnocai et al. 2009). There are serious concerns that tundra ecosystems will turn into a major source of greenhouse gas emissions within the next few decades, as climate change causes continued thawing of the permafrost layer, and that this will lead to a positive feedback further reinforcing climate warming (Ciais et al. 2013; Koven et al. 2011; Schuur et al. 2015). The situation is exacerbated by the fact that the regions at high latitudes and/or altitudes where tundra ecosystems occur are predicted to experience particularly strong climate warming.

Depending on local geology and hydrology, thawing of permafrost can lead to marked changes in the aspect of the landscape, including the formation or drainage of wetlands and lakes, and to an increase in coastal erosion rates (Chapin et al. 2005). This, in combination with the rising soil temperatures, can result in the release of a significant share of the stored carbon in the form of carbon dioxide or methane (Koven et al. 2011). Biomass carbon stocks in the tundra zone are expected to increase under climate change, as rising temperatures and changes in precipitation will continue to allow tall shrub and tree species to colonize the area (Frost & Epstein 2014; Myers-Smith et al. 2011). However, most authors expect that these carbon gains will not be large enough to compensate for the losses in soil carbon, and some draw attention to the fact that the lower albedo of tree canopies as compared to lower (and thus more often snow-covered) vegetation may further enhance warming (Smith et al. 2014). Increasing temperatures may also lead to a higher risk of fire, potentially affecting both soil and biomass carbon stocks (Mack et al. 2011). Pressures from human activity in tundra ecosystems are mostly linked to the extraction of fossil fuels and other mineral resources. Despite significant

impacts on the Arctic environment (AMAP 2010), these activities are currently not considered to be a major driver of greenhouse gas emissions due to their limited spatial extent (Chapin et al. 2005). This may change in the future as resource demand continues to grow, and tundra areas become more accessible for extractive activities due to reduced sea ice cover and milder temperatures (ACIA 2004; AMAP 2010). Growing suitability for forestry use could also increase human impact in the area (ACIA 2004).

## CROPLANDS

Lands used for the cultivation of crops (including annual as well as perennial crops and mixtures of crops and non-crop vegetation, as e.g. in some agroforestry systems) currently cover around 13 % of the global land surface, and are mostly located in areas formerly covered by forests and grasslands (FAO 2014a; Verchot 2014). Agriculture accounts for a significant share of global anthropogenic greenhouse gas emissions, mainly through the decomposition of soil organic matter and biomass following land use change and intensification, emissions of methane from livestock and rice cultivation, emissions of nitrous oxide caused by the application of fertilizers and manure management, and energy use for the operation of machinery, the production of agrochemicals and transport (Smith et al. 2014; Verchot 2014).

The amount of carbon stored in cropland soils can vary considerably depending on management practices as well as local factors such as geology and climate. However, where local environmental conditions are comparable, soil carbon stocks are usually significantly lower in croplands than in other types of ecosystems. The conversion of natural or semi-natural ecosystems to cropland leads to a decrease in soil organic carbon stocks, the extent of which depends on the soil and climatic conditions and the agricultural practices applied. In a meta-analysis of published data, Guo and Gifford (2002) found that a land use change from pasture to cropland resulted in an average decline of soil carbon stocks of around 60 %, while Lal (2011) reports long-term losses of between 25 and 75 % of soil organic carbon stocks from agroecosystems as compared to the original vegetation. Scharlemann et al. (2014) cite figures of 25–50 % for soil organic carbon loss in the top 1 m following conversion of native vegetation to cropland, noting that the impacts of land use change and management on soil organic carbon are dramatically different in mineral versus organic soil types. According to Joosten (2015), the period that it takes for soil organic carbon levels to stabilize after conversion (if management continues unchanged) is around 100 years for soils in the temperate region, whereas tropical soils may stabilize more quickly and boreal soils more slowly.

It has been estimated that over the course of human history, the expansion of agro-ecosystems has reduced global soil organic carbon stocks by 40 – 100 Gt C (Joosten 2015). Unsustainable practices have led to the degradation of large areas of land, often to the degree of making them unsuitable for further cultivation (Lal 2003). At the same time, changes in management practices can also lead to an increase in soil or biomass carbon stocks on lands that are already under agricultural use (Bernoux & Paustian 2015).

Due to the rising demand for agricultural products, it is projected that the use of existing croplands will be further intensified, potentially increasing the application of unsustainable methods, and intensive arable land uses will continue to expand into other ecosystems, especially savannahs and grasslands, tropical forests and peatlands (Victoria et al. 2012). The pressure for land conversion is likely to grow further as a consequence of climate change impacts on crop yields. Current projections indicate that many areas will suffer productivity losses due to declining water availability and stronger climatic fluctuations. Land degradation and loss of fertile soils through erosion are also exacerbating the problem. At the same time, rising temperatures will allow agriculture to expand poleward or into high-altitude regions that were previously unavailable for cultivation.

Depending on the way in which socio-economic development continues, it has been estimated that the demand for additional cropland for the production of food, fibre and biofuels will amount to between 320 and 850 million hectares by the year 2050, taking into account population growth and changing consumption patterns as well as the need to compensate for croplands that are lost due to land degradation and the expansion of built-up land (Banwart et al. 2015). Achieving a more efficient and sustainable use of existing cropland will be key to balancing environmental and agricultural outcomes and limiting the need for further expansion. Efforts towards climate change mitigation in agro-ecosystems

thus need to consider not only the potential for reducing greenhouse gas emissions or increasing carbon sequestration per unit of land, but also the impacts on total area requirements for commodity production (Banwart et al. 2015).

## ABANDONED CROPLANDS

When croplands are abandoned, under most circumstances they will turn into carbon sinks because the carbon losses that took place following conversion are partly or fully reversed. There is a variety of reasons why agricultural use of a site may be discontinued, including ecological factors (such as naturally unfavourable climate and soil conditions or drops in productivity following degradation) and socio-economic drivers (such as changes in land use-related policies or the emergence of new and more profitable livelihood opportunities) (cf. Benayas et al. 2007). Land abandonment took place at a globally significant scale across large areas of Eastern Europe and Northern and Central Asia following the political and socio-economic changes of the 1990s (Vuichard et al. 2008). It has been estimated that a total of 75 million hectares of cropland went out of use in Russia, Kazakhstan, the Ukraine and Belarus since 1990. Depending on location, most of this area has reverted to forest and grassland ecosystems. The average rate of carbon sequestration in vegetation and soils of the former croplands in Russia and Kazakhstan over the first 20 years following discontinuation of use has been estimated at 155 million tons per year (for Russia) and 31 million tons per year (for Kazakhstan) (Kurganova et al. 2014, 2015). If these areas remain uncultivated, sequestration will most likely continue, with a slowly decreasing rate, and carbon stocks close to those of undisturbed forests or grasslands should be reached after about 60–120 years in most regions. Large areas of abandoned croplands that are returning to native vegetation types are also found in parts of Western Europe and North America (Benayas et al. 2007; Smith et al. 2014). However, given that the global demand for cropland continues to rise, it is to be expected that many abandoned areas will be returned to agricultural use in the coming decades.

# 3. THE INFLUENCE OF BIODIVERSITY ON CARBON STOCKS AND FLOWS

The linkages between the biodiversity of an ecosystem and its capacity to store and sequester carbon have been the subject of intense scientific debate. The debate has focussed on a variety of questions, including whether or not there is a spatial correlation between the distribution of carbon stocks and biodiversity within specific ecosystem types, and whether observed correlations in distribution are an indication of causality (e.g. Hicks et al. 2014; Midgley et al. 2010; Strassburg et al. 2010; Sullivan et al. in review; Talbot 2010; Thompson et al. 2012).

For the purpose of informing decisions on the management of ecosystems, two questions are particularly relevant:

- Within a specific ecosystem type (e.g. steppe or coastal wetland), are those areas that have higher levels of species richness or genetic diversity likely to hold greater potential for carbon storage and sequestration? (And if so, should efforts for ecosystem-based climate change mitigation therefore focus on those areas?)
- Are forms of management that support the maintenance or restoration of natural species diversity likely to be more beneficial for carbon storage and sequestration than other management options?

There are two main mechanisms that could underpin the contribution of biodiversity to carbon sequestration and storage: increased efficiency of primary production due to complementarity between species with different ecological requirements and symbiotic effects; and increased resilience of ecosystems to disturbances that could reduce carbon stocks and sequestration capacity. In this context, resilience is understood as the ability of an ecosystem to maintain basic structural and functional characteristics over time despite external pressures. Resistance to fundamental change, i.e. change that alters the basic structure and function of the ecosystem into a new system, and recovery from disturbance are both mechanisms that can contribute to this ability (Epple & Dunning 2014).

Evidence from spatial correlation analyses comparing the distribution of biodiversity and carbon stocks for specific ecosystem types remains mixed (Hicks et al. 2014, Sullivan et al. in review), which may partly be due to the influence of non-biotic factors that affect the capacity of an ecosystem to take up and store carbon, such as hydrological conditions and disturbance regimes. However, case studies, experiments and the principles of theoretical ecology indicate that biodiversity has the potential to modify the turnover rate, magnitude, and long-term permanence of the terrestrial biosphere's carbon stocks (Diaz et al. 2009; Hicks et al. 2014; Isbell et al. 2011; Miles et al. 2010; Oliver et al. 2015).

Evidence has been established to support both the hypothesis that there is some degree of linkage between higher levels of species diversity and higher rates of carbon sequestration, and that higher biodiversity can increase the resilience of ecosystems and their carbon stocks to disturbance (Epple & Dunning 2014; Hicks et al. 2014). The evidence for the latter hypothesis is considered stronger due to a larger number of studies (Hicks et al. 2014). Studies also highlight that individual species (such as highly productive plant species) or functional groups (such as pollinators, seed dispersers or predators that control herbivore populations) can be disproportionately important for carbon sequestration and storage, and their loss can compromise ecological functions (Atwood et al. 2015; Bello et al. 2015; Hicks et al. 2014).

There is thus good reason to assume that both targeting ecosystem-based mitigation actions at areas of high biodiversity (all other conditions being equal) and choosing management methods that maintain or restore biodiversity can support the effectiveness of ecosystem-based climate change mitigation efforts, particularly with regard to resilience over the longer term. When selecting management options, it is also important to keep in mind that a number of other ecosystem characteristics, such as intactness or naturalness, have been shown to correlate positively with both ecosystem resilience and biodiversity (Epple & Dunning 2014; Miles et al. 2010).

While most ecosystem-based mitigation actions will to some degree meet the aim of Aichi Target 15 to 'enhance the contribution of biodiversity to carbon stocks'[11], they are likely to do so more efficiently if they are purposely designed to harness the potential of biodiversity to support ecosystem resilience and functioning. Other considerations that will

11 Target 15 of the Strategic Plan for Biodiversity 2011-2020: By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks have 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.

need to be taken into account to ensure the success of planned measures are the type and intensity of pressures on carbon stocks in a given area, the area's land use history, as well as cultural and socio-economic factors (see also section 5).

The information on management options provided in the following section is intended to be of use to governments and other stakeholders wishing to make a contribution to the achievement of Aichi Target 15.

At the same time, it may also be of interest in the context of the implementation of the Paris Agreement adopted under the UNFCCC (FCCC/CP/2015/10/Add.1), including the development of nationally determined contributions (in line with Article 4), as well as Article 7, paragraph 9 (e), which asks Parties, where appropriate, to "engage in adaptation planning processes and the implementation of actions (…) which may include: building the resilience of socioeconomic and ecological systems", and Article 8, paragraph 4 (h), which suggests that areas of cooperation between countries to enhance understanding, action and support may include the "Resilience of communities, livelihoods and ecosystems".

# 4. MANAGING ECOSYSTEMS TO SUPPORT CLIMATE CHANGE MITIGATION AND PROVIDE ADDITIONAL BENEFITS FOR BIODIVERSITY AND PEOPLE

Around the world, management options that support climate change mitigation are being developed and tested for a range of ecosystems, including non-forest ecosystems. An overview of the main options discussed in the literature, as well as the potential synergies and trade-offs linked to such actions, is given below. References to existing guidance manuals for the design of site-level interventions are included, noting that the planning of concrete interventions should take into account local conditions, as well as the knowledge and views of stakeholders. Section 5 addresses the steps that can be taken to integrate ecosystem-based mitigation actions into plans for the use of land at a landscape level, in order to maximize the delivery of multiple ecosystem services.

Where available, figures are provided on the amount of carbon emissions that can be avoided, or on the levels of carbon sequestration that can be achieved, through a particular type of management intervention. There are various forms in which such figures are reported in the literature, and different kinds of calculations may be appropriate to inform decision-making in different contexts:

- Information on typical values of emission reduction and sequestration rates per hectare is likely to be most relevant to those wanting to carry out an initial assessment of the likely feasibility and relevance of specific ecosystem-based mitigation actions in their own context.
- Estimates of the total contribution that a certain type of action can make to global climate change mitigation efforts may be of use in order to develop recommendations for action at the international level, or to prioritize research efforts.

When comparing estimates of global mitigation potential, it is important to differentiate between estimates that relate to the 'technical' potential, i.e. the mitigation benefits that could be achieved by implementing a measure across the full area covered by a land use or ecosystem type, and estimates taking into account some of the practical constraints (such as costs and availability of funding, land use designations and regulations, competition with other land uses, etc.). While these constrained estimates are likely to have more practical relevance, it can also be more difficult to compare the figures elaborated by different authors using different assumptions about socio-economic and political conditions. Some examples of constrained estimates are provided in Box 3 in the section on croplands.

## PEATLANDS

Avoiding or reversing drainage to reduce carbon emissions is the main mitigation option for peatlands (Trumper et al. 2009; Victoria et al. 2012). Studies have shown that in temperate regions, avoiding the further conversion of fenlands to cropland can prevent emissions from soils of between 2.9 and 17 t of C per hectare per year, depending on drainage patterns (Hooijer et al. 2010; Strack 2008). Emission savings in tropical regions can be on the order of 25 t of C per hectare per year when avoiding large-scale conversion of peat soils to plantations or cropland, which usually entails a drainage to about 1 m depth (Hooijer et al. 2010; Strack 2008). (Note that these are average figures, and actual values will depend on factors like climate and length of time since conversion.) Avoided conversion will also save the emissions that would result from the establishment, running and maintenance of drainage infrastructure, as well as potential substantial emissions of carbon dioxide, methane and nitrous oxides from peat fires. In some areas, reducing other pressures such as overgrazing or peat extraction can also contribute to emission reductions (Biancalani & Avagyan 2014; Parish et al. 2008).

Restoration of degraded peatlands to avoid further peat decomposition and reduce fire risk is another important mitigation option, but can be technically demanding and under some conditions involve long recovery times (Biancalani & Avagyan 2014). Where conservation of intact peatlands and peatland restoration are both possible, conservation is thus likely to be the more efficient approach. Depending on current land use and socio-economic context, re-wetting of drained peatlands can be carried out to restore the original water levels fully, or only partially. Full restoration involving the re-establishment of peat-forming vegetation provides the greatest emission savings and also has the potential to restore

the (naturally slow) process of carbon sequestration. However, where full restoration is not possible, switching to a land use that requires less intensive drainage may be a viable solution, e.g. changing from crop cultivation to pastoral uses, or to cultivation of reeds or tree species that can tolerate high water levels (Biancalani & Avagyan 2014; Smith et al. 2014). The latter approach, i.e. the cultivation of biomass on wet and rewetted peatlands, has become known as paludiculture, and has attracted attention as an option for combining the production of biofuels, food crops and other commodities with a reduction in soil carbon emissions from drained peat (Abel et al. 2012; Wichtmann & Joosten 2007). Measures to raise the water table in drained peatlands need to be planned and implemented carefully, because inundation of fresh or weakly decomposed plant matter, as well as nutrient-enriched soil layers, can lead to initially high emissions of nitrous oxides or methane. It may take decades for these to be offset by the subsequent savings in terms of avoided carbon dioxide emissions (Joosten et al. 2012; Smith et al. 2014).

Biancalani and Avagyan (2014) provide an overview of available guidance for the design of mitigation actions in peatlands, while Bonnett et al. (2009) and Page et al. (2009) address specific aspects that can be relevant for the design of peatland restoration measures. An interesting case study with regard to possible financing mechanisms is described in IUCN (2014), while a methodology for calculating emission savings from restoration of tropical peatlands is provided in VCS (2014).

The recent advances in knowledge about the spatial distribution of peatland carbon stocks and the emission factors associated with different forms of peatland management (e.g. IPCC 2014a) provide a good starting point for planning activities to avoid or reduce greenhouse gas emissions from peatlands, while achieving synergies with other policy goals including biodiversity conservation. While the focus on Southeast Asia in current discussions is justified by particularly high emission rates, the potential for mitigation action in peatlands of other regions should not be overlooked. Given the important role of agricultural production as a driver of peatland degradation, actions to support more sustainable forms of management and to direct development towards less sensitive areas will be crucial (Austin et al. 2015). These could include reforms to subsidies and mechanisms for land allocation, socially and environmentally responsible certification schemes, support to local livelihoods and raising awareness among companies and consumers. In the case of biofuel crops, initiatives should ensure that both short- and long-term emissions from soils in the location of production, as well as energy expenditure for drainage, and any indirect land use change impacts are included in the calculation of potential emission savings. This is all the more important because some of the ecological changes triggered by drainage can be irreversible. Considering the full greenhouse gas footprint of biofuel cultivation on peatlands is likely to reveal that it does not provide net benefits for climate change mitigation (Biancalani & Avagyan 2014; Hooijer et al. 2006; Parish et al. 2008; Trumper et al. 2009). Cultivation of bioenergy crops on tropical peatlands with methods that require drainage should be avoided, as the available evidence suggests it is likely to produce more emissions than the burning of an equivalent amount of fossil fuels, and crop cultivation may not be viable in the long term in many locations (Biancalani & Avagyan 2014; Hooijer et al. 2010; Hooijer, Vernimmen, Visser et al. 2015).

There is significant potential for ecosystem-based mitigation measures in peatland areas to achieve co-benefits. Many peatland areas play a key role in regulating the water cycle, including through buffering and control of floods. They can also contribute to water purification and may remove significant amounts of excess nutrients and other pollutants from the water that passes through them. These ecosystem services can be highly relevant for adaptation to climate change (Parish et al. 2008). Measures that reduce drainage will also lower the risk of peat fires, which in recent years have caused severe air pollution problems as well as loss of human life, disruption of economic activities and damage to infrastructure in both tropical and temperate regions (Betha et al. 2012; Biancalani & Avagyan 2014; Parish et al. 2008; World Bank 2015). Furthermore, mitigation measures in peatlands can counteract the process of subsidence, which occurs when peat decomposition leads to a shrinking of the soil profile and lowering of the soil surface (see Case Study 1). Depending on groundwater levels in the surrounding areas, subsidence can make drainage increasingly difficult and costly, lead to more frequent and intense flooding or saltwater intrusion, and eventually cause the loss of habitable and productive land (Joosten et al. 2012). Agricultural areas on peatlands in the tropics are particularly at risk, as decomposition proceeds much faster than in temperate or boreal climates (Hooijer, Vernimmen, Visser et al. 2015). However, increased flood risk with severe economic consequences has also been reported from drained peatland areas in North America and Europe (Joosten et al. 2012).

## CASE STUDY 1: LAND SUBSIDENCE ON DRAINED PEATLANDS

For many centuries, the drainage of peatlands has been a common practice across the world, driven mainly by agricultural development, urbanization, infrastructural expansion and forestry (Holden 2004; Parish et al. 2008). The greatest historic losses of undisturbed peatlands can be found in Europe. For example, in the Netherlands, peatland drainage was practised as early as in the 8th century to claim land for agriculture (Parish et al. 2008). The rate of disturbance of peatlands has recently been increasing, especially within tropical peat swamp forests. In Southeast Asia alone, more than 130,000 $km^2$ of peat forests have already been converted to other uses or severely disturbed, with associated impacts on human well-being across different geographic scales, e.g. as a consequence of flooding or peat fires (Biancalani & Avagyan 2014; Hooijer et al. 2010; Joosten 2015).

One of the unavoidable negative effects of peatland drainage is land subsidence, leading to a lowering of the land surface ranging from less than 1 cm per year to more than 30 cm per year (Fornasiero et al. 2002; Hooijer et al. 2012). This is caused by soil consolidation and compaction, and by the increase in decomposition prompted by new aerobic conditions allowing for greater microbial activity (Fornasiero et al. 2002; Hooijer et al. 2012). Subsidence directly threatens the stability of existing infrastructure and increases the risk of flooding when the surface settles to a level below adjacent river or sea levels (Boersma 2015; GEF et al. 2010; Holden 2004). For example, the Zennare Basin in Italy, which was claimed for agriculture in the 1930s, currently lies almost entirely below sea level, in some areas by up to 4 m. Arable land in the Everglades of Florida has subsided by about 2.5 cm per year during most of the 20th century, and it is thought that much of it would turn into a lake if active water management were to cease (Fornasiero et al. 2002; Ingebritsen et al. 1999). The impact of land subsidence also increases the amount of investment required for drainage, given that pumps and dykes are needed for mechanical drainage within urbanised areas and agricultural land (Boersma 2015; Hooijer, Vernimmen, Visser et al. 2015). For example, a total of 29 % of croplands in the Rajang Delta, Malaysia, is currently estimated to have drainage problems caused by land subsidence, and this effect is expected to grow over the next decades with subsequent drops in agricultural production (Hooijer, Vernimmen, Visser et al. 2015). Problems caused by significant land subsidence have been reported among others from the Netherlands, the United Kingdom, Italy, the United States, Southeast Asia and Israel (Hooijer et al. 2012; Hooijer, Vernimmen, Mawdsley et al. 2015). The subsequent loss of agricultural production prompted the end of peatland conversion in the United States and Europe during the 20th century. Amongst the countries where drainage continued is Indonesia, where over one million hectares of peatlands were opened to drainage for the Mega Rice Project during the mid-1990s. The project was finally abandoned in 1998, but its environmental impacts are still felt today (Hooijer et al. 2012; Page et al. 2009; Parish et al. 2008; Yustiawati et al. 2015).

Tropical peatlands suffer the impacts of drainage more rapidly and more severely than other peatland areas, because higher temperatures prompt faster decomposition rates. Hooijer et al. (2012), for example, estimated a rate of subsidence of about 28 cm per year for croplands in Indonesia.

The rewetting of peatlands is known to reduce flood risk, fire occurrence and economic impacts associated to drainage (Cris et al. 2014; GEF et al. 2010; Hooijer, Vernimmen, Visser et al. 2015; IUCN 2014). Given its capacity to restore hydrological functions, rewetting also contributes to reducing drainage-related carbon dioxide emissions and to re-establish carbon dioxide fixation in peatlands (Cris et al. 2014; GEF et al. 2010). Rewetting has been undertaken across different types of peatlands and land use conditions, and today data is available regarding its long-term performance for the restoration of ecological processes (Cris et al. 2014; Parish et al. 2008). Australia, Belarus, Canada, China, Germany, Indonesia, Ireland, the United Kingdom, Rwanda, South Africa and Sweden are amongst the countries where rewetting of degraded peatlands has been practised (Cris et al. 2014; Jaenicke et al. 2010; Page et al. 2009). In response to recurring issues around peat fires, the government of Indonesia established a dedicated Peatland Restoration Agency in early 2016, aiming to restore about 2 million ha of degraded peatlands.

A good example for the application of rewetting for the rehabilitation of wetland ecosystems comes from Belarus, one of the hotspots of greenhouse gas emissions from drained peatlands (Cris et al. 2014; GEF et al. 2010; Michael Succow Foundation 2009). Approximately 15 % of the country is covered by peatlands, and more than half of this area has been drained for mining, agriculture and forestry with associated impacts on soil quality, agricultural productivity and fire regimes (Cris et al. 2014). Since 2006, a series of restoration measures have been undertaken in Belarus focused on: (1) supporting sustainable management and rewetting as a restoration tool, (2) developing capacity for peatland management, and (3) promoting alternative income sources derived from the restoration of these ecosystems (Cris et al. 2014; GEF et al. 2010; Kozulin & Fenchuk 2012). So far, a total of 50,000 ha of degraded peatlands have been rewetted through these efforts (Kozulin & Fenchuk 2012). GEF et al. (2010) reported a reduction in carbon dioxide emissions equivalent to 87,500 t C/year, as well as numerous co-benefits. For example, the introduction of paludiculture allowed the production of renewable energy fuels through the harvest of biomass from rewetted peatlands (Cris et al. 2014). Peatland restoration also halted the occurrence of fires, which in turn saved public funds directed to fire-fighting and prevention, as well as to health care services for local communities who had been subject to yearly problems from smoke and dust (GEF et al. 2010). The restoration efforts further resulted in an increase in local biodiversity and opportunities for sport hunting.

Due to the unique array of species harboured by peatland ecosystems, measures that support their conservation will generally have positive impacts on biodiversity (Parish et al. 2008). In the case of restoration measures, or of the introduction of new uses of peatland that can be carried out without drainage, the implications for biodiversity depend on how and where these are implemented. Positive outcomes for biodiversity can be enhanced if restoration is carefully designed to improve habitat conditions for native species, and if measures that introduce the cultivation of water-tolerant crops or trees are focussed on degraded areas and areas suffering from subsidence (Joosten et al. 2012; Wichtmann & Joosten 2007). Where afforestation of naturally treeless peatlands or use of peatlands for biofuel production is considered for mitigation pruposes, trade-offs between climate and biodiversity goals, as well as consequences for the supply of other ecosystem services, should be carefully assessed. Care should also be taken to evaluate the full climate footprint of such measures, as their possible benefits for climate change mitigation are often overestimated (see above).

## GRASSLANDS AND SAVANNAHS

Mitigation approaches in grassland ecosystems can include adjusting grazing intensity (including through better management of the spatio-temporal distribution of livestock), regulating fire frequency, avoiding conversion to croplands, restoring degraded grassland, and in the case of savannahs, reducing extraction of woody biomass (Conant 2010; Epple 2012; Gerber et al. 2013). Due to the extent of degradation that has already occurred, grassland soils offer a potentially large carbon sink (Conant 2010). It has been estimated that full rehabilitation of the world's overgrazed grasslands, mainly through adoption of more moderate grazing intensities and better distribution of livestock, could sequester about 45 million tons of carbon per year (Conant & Paustian 2002).

What intensity of grazing is most beneficial for carbon stocks depends on climate, soil type and vegetation type. In some grassland systems, especially those dominated by tropical grasses, the greatest rates of carbon sequestration are achieved at intermediate levels of grazing, while in others even moderate grazing can lead to losses of soil carbon. If grazing is optimally adjusted to the characteristics of the ecosystem, annual sequestration rates can be as high as 1.5 t C per hectare (McSherry & Ritchie 2013). Differring approaches have been suggested in order to optimize grazing management on permanent pastures, and there is an ongoing debate on the advantages of rotational versus continuous grazing; further research (including long-term studies) could be beneficial to identify the best strategies under a range of conditions (Badgery et al. 2015; Briske et al. 2008; Machmuller et al. 2015; McSherry & Ritchie 2013; Sanderman et al. 2015). For many dryland systems, mobile pastoralism can be a good way to ensure efficient use of natural resources, due to the variability of rainfall and plant growth (McGahey et al. 2014).

Grazing by wild or domesticated animals can also reduce fire occurrence by decreasing fuel loads, thus potentially avoiding significant emissions of carbon and nitrous oxides. In some regions, strategic burning causing more frequent but less intensive fires (a management technique that has been used traditionally by indigenous communities for example in northern Australia) has been applied successfully as an approach to reduce carbon emissions (Douglass et al. 2011; Fitzsimons et al. 2012).

Where avoided conversion to cropland is an option, this offers the largest possible carbon savings per hectare, as soil carbon stocks have been shown to decline by up to 60 % following conversion (Guo & Gifford 2002; Joosten 2015). The effects of conversion from croplands back to grassland are generally more moderate, but can still lead to an increase in soil organic carbon of about 20 % over a timescale of several decades (Guo & Gifford 2002; Soussana et al. 2004). Impacts on soil carbon from potential mitigation activities that would involve conversion of grasslands, such as cultivation of biofuels or afforestation, should therefore be carefully assessed.

Recently, some initiatives for more sustainable management of grasslands have produced quantified emission reductions and obtained carbon credits from the voluntary market (Ducks Unlimited undated; McGahey et al. 2014; USDA 2014). Experiences from these pilot projects can inform the development of similar initiatives in other regions, and/or be applied to other types of management interventions. In savannah areas where wood extraction is an issue, methodologies can also be transferred from forest-based projects, for example to support activities that reduce pressure on the tree layer through alternative approaches to charcoal production (Epple 2012; Iiyama et al. 2014). Given the urgency of sustainable development challenges in many grassland regions and the significant co-benefits that mitigation actions in grasslands

## CASE STUDY 2: AFFORESTATION IN NATURAL GRASSLANDS AND NATURALLY TREELESS PEATLANDS

Deforestation and forest degradation account for substantial ongoing emissions of greenhouse gases (Ciais et al. 2013; IPCC 2014b; Le Quéré et al. 2009). Many initiatives are working to counteract this trend through reforestation, forest restoration and afforestation. While the restoration of previously disturbed forest ecosystems is likely to provide both climate and biodiversity benefits, concerns have been raised that actions to increase carbon stocks in naturally treeless or open-canopy ecosystems could pose threats to key areas for biodiversity conservation and the provision of important ecosystem services, as well as to local livelihoods (Bremer & Farley 2010; Putz & Redford 2009; Veldman, Buisson et al. 2015; Veldman, Overbeck et al. 2015b).

A review by Bremer and Farley (2010) showed that afforestation of shrublands and natural grasslands decreases plant species richness on average by about 30 %, and is particularly detrimental for native species and endemic species (which were found to decline in richness by 38 % in the shrublands and 47 % in the grasslands). This concurs with previous research on the subject. Ecological mechanisms behind these effects involve the reduction of sunlight availability, which influences plant species richness and primary productivity, and in consequence the availability of habitat for associated species (Veldman, Buisson et al. 2015). Afforestation of open-canopy ecosystems also has the potential to affect stream flow and water quality (Farley et al. 2005; Farley et al. 2008; Jobbágy & Jackson 2004). Farley et al. (2005) estimated an annual reduction in runoff of about 44 % and 31 % when grasslands and shrublands were afforested respectively. Jobbágy and Jackson (2004) reported a reduction in the water table by an average of 38 cm and an increase in groundwater salinity of up to 19-fold as a consequence of the introduction of *Eucalyptus camaldulensis*, i.e. a non-native species, in the Pampas ecosystem of Argentina. Impacts vary depending on the species used in afforestation projects, and may be less pronounced if native tree species co-existing with the respective grassland ecosystem are employed.

It is often assumed that planted forests would store more carbon than open-canopy ecosystems, but this has been challenged e.g. by Conant (2010) and Veldman, Buisson et al. (2015). According to the former, improvements in the management of grasslands could prompt similar levels of carbon sequestration to that of forest ecosystems, mainly due to increased soil carbon storage. In addition, plantations typically alter nutrient cycles and can reduce soil carbon storage, a fact which is sometimes overlooked in calculations of potential carbon gains from afforestation (Berthrong et al. 2009; Berthrong et al. 2012; Guo & Gifford 2002). It has further been highlighted that soil carbon stocks are less vulnerable to disturbance from fire than those above ground, which may be a relevant consideration in the dry and fire-prone climates occupied by many natural grassland ecosystems (Bremer & Farley 2010; Veldman, Buisson et al. 2015). While it has been suggested that afforestation could reduce nitrous oxide emissions of abandoned peat soils drained for agriculture, Maljanen et al. (2012) showed that for boreal peatlands, $N_2O$ emissions on afforested sites were similar to those measured in active agricultural plots, and higher than those on abandoned plots.

Conservation of open-canopy ecosystems can thus secure the protection of threatened and unique species, whilst providing key ecosystem services, including soil carbon storage and sequestration (Bremer & Farley 2010; Veldman, Overbeck et al. 2015a). In the Northern Andes, for example, more than 10 million people rely on water supplies from tropical alpine grasslands (i.e. páramos) (Farley et al. 2013). Belowground carbon stocks within these ecosystems could match those of planted forests (Gibbon et al. 2010), with the additional benefits of providing habitat for plant communities that include about 60 % of endemic species, and ecosystem services required at the national level (Vásquez et al. 2015).

As these examples show, afforestation of open-canopy ecosystems may not contribute to climate change mitigation to the extent expected, and it frequently leads to detrimental ecological effects which may include decreased delivery of a number of ecosystem services (Bremer & Farley 2010). Arguments for promoting afforestation of naturally treeless or open-canopy ecosystems therefore need to be scrutinized carefully, and potential benefits, for example from increased access to timber or altered vegetation structure (which could be desirable in specific locations, e.g. to provide erosion control) should be weighed against the possible risks. Decisions on potential afforestation measures should also take into account the landscape context and possible impacts on adjacent areas (see also section 5).

can achieve, funding for programmes to improve the management of natural resources in grasslands could be sought from a variety of sectoral budgets, and incentives could be provided for example in the form of enabling activities, carbon payments or payments for ecosystem services.

Due to the importance of grasslands for local livelihoods, any change in management that leads to avoided degradation or to the recovery of ecosystems is likely to enhance the sustainability of current economic activities, as well as the capacity of often poor local populations to adapt to future impacts from climate change (Conant 2010; Millennium Ecosystem Assessment 2005; Stringer et al. 2012). Higher soil organic carbon stocks are linked to greater infiltration capacity and nutrient retention, which may have beneficial effects on water regulation and quality. By avoiding soil erosion and maintaining vegetative cover, climate change mitigation measures in grasslands can also prevent aridization of local climates and increased sediment loads in rivers and lakes (Conant 2010; Victoria et al. 2012). Trade-offs between climate change mitigation and socio-economic development may need to be managed where optimal grazing intensities for maintaining or enhancing soil carbon stocks are lower than the carrying capacity of pastures for livestock keeping.

The impacts of mitigation actions in grasslands on biodiversity can be both positive and negative. Reduced degradation or conversion of grasslands, as well as grassland restoration (especially through natural regeneration), are likely to be desirable approaches from the perspective of biodiversity conservation (Millennium Ecosystem Assessment 2005). Biodiversity impacts of mitigation approaches involving fire management depend on the practices used, as well as the natural fire regimes to which species in the area are adapted. Negative impacts could result from approaches that affect wild herbivore populations, or from intensive grassland management involving fertilization, irrigation or re-seeding with high performance grasses (which may also lead to the degradation of regulating and cultural ecosystem services) (Berry et al. 2008). Grassland biodiversity can further be threatened by afforestation schemes, or 'reforestation' efforts that are wrongly directed towards natural grasslands (Veldman, Buisson et al. 2015, see also Case Study 2). And finally, the risk of negative impacts through displacement of pressure as a result of mitigation activities targeting forests is particularly high in savannah or steppe ecosystems (Miles & Dickson 2010).

Given the range of opportunities and risks presented by mitigation actions in grassland ecosystems, biodiversity stakeholders should engage with the climate change community to identify mutually beneficial solutions and ways to manage trade-offs between climate change mitigation and the delivery of other ecosystem services where these cannot be avoided.

## MANGROVES, SALT MARSHES AND SEAGRASS BEDS

Given the high current rates of loss of coastal ecosystems, the most important option for climate change mitigation is to address the drivers of conversion, habitat degradation and pollution. As demands on coastal areas are multiple and intense, this is likely to require integrated and in some cases transboundary land use planning, which should also take into account the main factors that will influence the future availability of space for coastal vegetation. Such factors include human population growth, sea level rise and changes in coastal currents that lead to shoreline regression and the lateral movement of erosion and accumulation zones (Gilman et al. 2008; UNEP 2015). Where processes for Integrated Coastal Zone Management or other integrated planning approaches exist, these may offer a good avenue for ensuring that the full range of values offered by coastal ecosystems is reflected in decisions about their management, and that opportunities for ecosystem-based mitigation are taken up (cf. UNEP 2015). Care should be taken to design integrated planning processes so that the perspectives and knowledge of local communities are appropriately taken into account, as local stakeholders may not only be significantly affected by the outcome of decisions, but can also be key actors in the implementation of ecosystem-based mitigation and adaptation actions (cf. Case study 4 and section 5).

One way to reduce the overall amount of pressure on coastal areas is to develop more efficient and sustainable management methods for major land uses. There is considerable scope for improvements to current practice with regard to aquaculture, which is a major driver of habitat loss in coastal areas. Better forms of management could increase the timespan for which aquaculture installations can operate, and reduce their environmental impacts (Primavera 2006, see also Case Study 3). Better planning and site selection can also help to ameliorate environmental outcomes (see e.g. Bricker et al. 2016). Ways to support the uptake of improved techniques could include a variety of regulatory and non-regulatory approaches, including reforms to subsidies, permitting requirements and certification schemes that set out social and environmental

criteria for good practice. In the case of seagrass beds, it is crucial to address the land-based causes of nutrient pollution and siltation, including erosion in areas under agriculture and forestry (Short & Wyllie-Echeverria 1996).

There is also significant scope for restoration of coastal ecosystems, as between 30 and 50 % of the area originally covered by mangroves, salt marshes and seagrass beds is thought to have been lost over the last century alone. Restoration methods have been developed for all three ecosystem types, and have proven effective in terms of restoring both the vegetation cover and the soil accumulation processes that are the basis for carbon sequestration (Crooks et al. 2011; Marbà et al. 2015; Nam et al. 2016; Osland et al. 2012). However, restoration requires more effort, resources and technical skill to be successful than interventions to halt further loss and degradation (Bosire et al. 2008; Fonseca et al. 1998; Primavera & Esteban 2008; Thayer et al. 2003; Twilley et al. 2007; see also Case Study 4), and has less immediate mitigation benefits. It will also fail if the causes that originally led to degradation and destruction of the vegetation cover are not addressed before re-planting or re-seeding is undertaken. Restoration initiatives should therefore prioritize areas where there is a high demand for the ecosystem services that can be re-established, and be planned in a participatory manner (UNEP 2014).

Efforts to establish ecosystem-based mitigation actions in coastal areas are facilitated by the high values of carbon stocks and sequestration rates per unit area, leading to a comparatively low required investment per ton of carbon saved (Duarte et al. 2013; Siikamäki et al. 2012). There are also approved methodologies that can be applied to estimate changes in carbon stock. The Wetlands Supplement to the IPCC Guidelines for National Greenhouse Gas Inventories, adopted in 2013, provides guidance for calculating carbon emissions and savings from a range of management practices in coastal ecosystems (IPCC 2014a). This information can be used as an input to the design of individual projects and larger programmes. In the case of mangroves, relevant mitigation measures could also be supported as part of countries' emerging REDD+[12] activities or activities to implement joint mitigation and adaptation approaches for the integral and sustainable management of forests, as set out in the Paris Agreement adopted under the UNFCCC (FCCC/CP/2015/10/Add.1). Given the high potential to manage coastal ecosystems for multiple benefits, there may be scope to combine funding from sources that address various purposes, including climate change mitigation and adaptation, biodiversity conservation, coastal protection and sustainable development. Wylie et al. (2016) carried out a global review of coastal blue carbon projects and provide recommendations for the development of future projects as well as the identification of policy opportunities.

Coastal ecosystems provide a wide range of ecosystem services that are relevant to climate change adaptation, disaster risk reduction, human health, food security and local livelihoods (UNEP 2014; UNEP 2015). These are all the more important because many coastal regions have a high density of human settlement (Kirwan & Megonigal 2013; UNEP 2014). For example, it has been estimated that over 100 million people around the world live within 10 kilometres of a large mangrove forest, mostly in developing countries in Asia and West and Central Africa.

All types of coastal vegetation offer some level of protection for the coastline by reducing wave intensity and stabilizing the ground with their roots, thus preventing coastal erosion (Guannel et al. 2015; McIvor et al. 2012; Möller et al. 2014; Spalding, Ruffo et al. 2014). The processes of filtration and sedimentation that contribute to carbon sequestration in coastal ecosystems can, at the same time, help to maintain or improve water quality. Coastal ecosystems are also important habitats and breeding grounds for animal species used by humans, including fish, molluscs and seabirds. The vegetation itself, if used sustainably, can provide materials for a number of uses, such as roof thatch, fuel, animal bedding, or even, in the case of mangroves, timber (Orchard et al. 2016; UNEP 2007; UNEP 2014).

The potential of coastal ecosystems to contribute to climate change adaptation and disaster risk reduction has been studied most intensely at the example of mangroves. Research has shown that mangrove forests can significantly reduce storm wave intensity, and that wide belts of mangrove can attenuate the impacts of storm surges and even tsunamis (Spalding, McIvor et al. 2014; Spalding, Ruffo et al. 2014; UNEP 2014). The potential of mangroves to provide food, fuel and building materials can also be important for local populations during recovery from an extreme event. The protection and restoration of mangroves, especially if combined in an appropriate way with other elements such as early warning systems and hard infrastructure, can thus make a key contribution to strategies for climate change adaptation and disaster preparedness in almost any coastal setting (Spalding, McIvor et al. 2014).

12 Reducing Emissions from Deforestation and forest Degradation, plus conservation of forest carbon stocks, sustainable management of forests and enhancement of forest carbon stocks.

## CASE STUDY 3: ALTERNATIVE MANAGEMENT APPROACHES IN AQUACULTURE THAT CAN REDUCE AREA REQUIREMENTS

Aquaculture is the farming of aquatic flora and fauna to produce food, medicine, ornaments and other products (Shumway et al. 2003). It is currently growing faster than any other sector supplying animal protein to the global market (Olesen et al. 2011; Pattanaik & Narendra Prasad 2011). In 2008, aquaculture provided incomes and livelihoods for a total of 10.8 million people across the world, particularly in Asia (Klinger & Naylor 2012). According to FAO (2014b), this activity reached a peak in productivity in 2012, supplying 157 million tonnes of produce, with a value of over US $ 140 billion. Aquaculture is considered to be crucial to ensure future food security, as well as to accommodate the increasing demands for seafood from emerging economies (Klinger & Naylor 2012; Olesen et al. 2011).

A significant intensification of aquaculture took place during the last two decades of the twentieth century, with a yearly rate of increase of 8.6 % between 1980 and 2012 (FAO 2014b; Pattanaik & Narendra Prasad 2011). The global aquaculture production augmented by approximately 80 % between 1990 and 2012, with China and South East Asia hosting the largest share of this growth (FAO 2014b).

The rate and extent of conversion of coastal ecosystems (e.g. mangroves and salt marshes) for aquaculture is an important source of concern. In 1999, FAO (1999, in Páez-Osuna 2001) estimated that 1-1.5 million ha of natural ecosystems and agricultural lands had already been converted to aquaculture. Giri et al. (2008) estimated the loss of 12 % of the entire mangrove cover in Asia to aquaculture, whilst about 62 % of the area of brackish water available in India is now used for shrimp farming (Pattanaik & Narendra Prasad 2011).

The high conversion pressure is partly caused by unsustainable practices, which lead to a frequent need for shifting locations. For example, the average lifetime of a shrimp plot ranges between 7 and 15 years (Páez-Osuna 2001; Pattanaik & Narendra Prasad 2011; Primavera & Esteban 2008). In many cases, practices applied for the production of shrimp and fish have caused large-scale environmental impacts that in turn have brought disease outbreaks and productivity reductions, risking the overall future sustainability of this activity (Olesen et al. 2011; Páez-Osuna 2001). From a climate perspective, documented impacts from the conversion of coastal ecosystems to shrimp farms include major losses of biomass and soil carbon and potential significant emissions of $N_2O$ (Hu et al. 2012; Kauffman et al. 2014).

Poor planning and management, lack of regulations, as well as weak enforcement, contribute to the negative social and environmental impacts associated with aquaculture today (De Silva 2012; Klinger & Naylor 2012; Páez-Osuna 2001). Nonetheless, under sustainable forms of aquaculture many of these impacts can be avoided. Molluscan shellfish aquaculture, for example, can provide incentives to secure water quality, as growers depend on clean water to ensure adequate production (Shumway et al. 2003).

Overall, aquaculture is a highly dynamic sector that can potentially focus on a variety of commercial species, and a range of innovations and management practices can be used or are under development to assist in overcoming its negative effects (De Silva 2012). Klinger and Naylor (2012) carried out an in-depth review of currently available and proposed solutions to avoid environmental problems associated with aquaculture. According to the authors, the available solutions can be classified into three main categories: changes to culturing systems, feed strategies, and species selection. The first type of approaches has direct relevance for efforts to reduce pressure on natural ecosystems, by decreasing the extent of land and the amount of water required for aquaculture through sustainable intensification.

Recirculating Aquaculture Systems (RAS) have the capacity to reduce water usage down to 16 litres/kg of product in marine environments, while conventional aquaculture systems require between 3,000 and 45,000 litres/kg (Klinger & Naylor 2012; Verdegem et al. 2006). The reduced requirements in terms of water supply and waste water disposal allow for more efficient land use, as the installations can be located in areas that are unsuitable for other types of use (e.g. on degraded land). However, these systems have high energy demands and costs associated with the removal of waste (Klinger & Naylor 2012). Offshore aquaculture, in areas with deeper water than those colonized by coastal vegetation, is another alternative for reducing land conversion and pressure on freshwater resources and sensitive ecosystems (Naylor 2006). Potential obstacles are related to the high levels of initial investment required, and the conflicts that can arise regarding the use of public waters (Naylor 2006).

Integrated Multi-Trophic Aquaculture (IMTA) has been championed as a viable alternative to overcome the social and environmental impacts associated to commercial aquaculture, as it is less expensive than other forms of ecologically sustainable intensification (Chopin 2011; Klinger & Naylor 2012). IMTA focuses on farming a number of species together that represent different levels within the trophic web, mimicking, to a certain extent, the ecological processes of aquatic environments (Chopin 2011). Pilot studies suggest that IMTA has the capacity to be developed commercially, considering also that interest in sustainable sources of seafood is growing amongst consumers (Klinger & Naylor 2012). Nobre et al. (2010) compared the ecological and socio-economic outcomes of single-species aquaculture and IMTA in the production of abalone in South Africa. The authors report a reduction in nitrogen discharges with the incorporation of seaweeds as part of the IMTA, together with a decrease in the harvest of natural kelps and of greenhouse gas emissions. IMTA adoption also increased profits from 1.4 to 5 %. Thus, the approach could potentially reduce land conversion if similar schemes are applied elsewhere (Nobre et al. 2010). Today, IMTA programmes are at different stages of development and trial in at least 40 countries (Chopin 2011; Soto 2009). The economic viability of IMTA could be enhanced if the costs of waste disposal were internalized within the overall costs of all aquaculture operations, which is often not the case (Klinger & Naylor 2012).

## CASE STUDY 4: MANGROVE RESTORATION

Mangrove forests not only play an important role for climate change mitigation, but also provide habitat for many plant and animal species, including both nursery and breeding grounds (Spalding et al. 2010). Human populations across the world depend heavily on these ecosystems (Primavera & Esteban 2008; Spalding et al. 2010). Many local communities rely on mangrove forests for their supply of food, firewood and timber (Bosire et al. 2008; Kairo et al. 2001; Spalding et al. 2010). National and sub-national governments also receive significant income from fishery and tourism revenues, and benefit from the coastal protection services that mangroves provide (Bosire et al. 2008; Kairo et al. 2001; Spalding et al. 2010).

There is a general consensus that mangrove forests once covered over 200,000 $km^2$ of the Earth's surface (Spalding et al. 2010). However, a large share of this area has been lost. For example, in the Philippines and Thailand, about 76 % and 54 %, respectively, of the total original mangrove forest cover was cleared mainly for the expansion of aquaculture (Khemnark 1995; Macintosh et al. 2002; Primavera & Esteban 2008). The first documented efforts at restoration of mangrove forests were undertaken during the 19th century, but initiatives were stepped up worldwide in the 1980s, following growing concern about the consequences of their disappearance, and an increasing recognition of the ecosystem services that mangrove forests provide to support human communities (Bosire et al. 2008; Dahdouh-Guebas & Mathenge 2000; Kairo et al. 2001; Primavera & Esteban 2008). In 1983, UNDP and UNESCO launched a regional project across Asia and the Pacific to raise awareness about the value of mangrove forests, prompting restoration initiatives worldwide, which in many cases involved large-scale international investment (Bosire et al. 2008; Primavera & Esteban 2008).

Detailed studies evaluating the performance of programmes focused on mangrove forest restoration are scarce and information on successful outcomes is limited (Kairo et al. 2001; Lewis & Gilmore 2007; Salmo et al. 2013). Existing literature reviews show mixed results, and indicate that most accomplishments relate to projects restoring mangroves to harvest wood and increase the provision of ecosystem services for agricultural development (Kairo et al. 2001; Lewis & Gilmore 2007). Local community involvement has also been reported as crucial to secure positive project outcomes (Primavera & Esteban 2008). Lewis and Gilmore (2007) noted, however, that many projects did not take into account the ecological requirements for restoration, and that a lack of consideration of hydrological processes has reduced the chances of success of many initiatives. Lewis (2009) emphasized that seasonal water fluctuations should be a key consideration for any restoration attempt, together with the micro-topography of the site, given that salinity variations can cause significant dieback of the planted trees.

The species composition of plantations is another important point. Monospecific mangrove planting has been a standard practice of many restoration attempts, but survival rates are generally low (< 20 %) and ecological characteristics of the plots fail to resemble those of natural ecosystems (Lewis & Gilmore 2007). Primavera and Esteban (2008) have assessed the causes behind the limited success of several large-scale projects undertaken in the Philippines since the 1980s. They conclude that low survival rates could be due to the use of non-pioneer mangrove species (i.e. *Rhizophora* sp.) and the selection of intertidal or subtidal planting sites where mangroves are challenged by natural conditions. Primavera and Esteban (2008) further highlight the lack of monitoring activities for many restoration initiatives, which limits the possibility to evaluate their effectiveness. Dale et al. (2014), too, point towards missing or ill-designed monitoring as one of the greatest practical weaknesses in many rehabilitation efforts. They further conclude that inconsistencies in policy, insufficient information, and failure to involve local communities are among the reasons for the limited success of some initiatives.

Despite such setbacks, existing examples of successful practices demonstrate the importance and potential of mangrove restoration (Bosire et al. 2008; Lewis & Gilmore 2007; Salmo et al. 2013; Thornton & Johnstone 2015).

Community efforts to ensure wood supply and coastal protection through mangrove restoration have proven effective for achieving these goals, whilst providing alternative incomes for local people (Primavera & Esteban 2008). In the Philippines, for example, a mixed initiative that brought together villagers and the local government reversed a poverty trend that had resulted from mangrove forest conversion to aquaculture. By expanding the remaining mangrove forests, the initiative secured habitat for 38 species of migratory birds and for more than 15 species of fish of commercial interest. They also generated additional revenue by developing a popular site for ecotourism (Primavera & Esteban 2008). Survival rates for some community-led initiatives in the Philippines were estimated to be over 90 %, possibly due to: (1) the use of natural colonizing species such as *Avicennia* and *Sonneratia* sp., (2) ecologically appropriate site selection, (3) prospects of tenure, and (4) monitoring efforts and other human-related factors (Primavera & Esteban 2008).

The recovery of mangrove forests on the coast of Florida is another good example of successful restoration (Lewis & Gilmore 2007). During the 1950s and 1960s, thousands of hectares were cleared in an attempt to control mosquito populations (Lewis & Gilmore 2007). This brought changes in the local vegetation, reductions in fish species richness and increasing fluctuations in salinity and dissolved oxygen levels (Gilmore et al. 1982). However, the subsequent hydrological restoration brought back fish, invertebrate and flora communities through natural succession, and allowed the recovery of commercial and sport fisheries. A key feature of this project was that simple tidal reconnection restoring the conditions suitable for mangrove growth allowed ecosystem restoration without the need for replanting (Lewis & Gilmore 2007).

Actions for climate change mitigation that involve the conservation and sustainable use of coastal ecosystems such as mangroves, salt marshes and seagrass beds are likely to generate strong benefits for biodiversity, as these systems provide critical permanent and seasonal habitat for large numbers of plant and animal species. In the case of actions aiming to restore lost or degraded coastal vegetation, the biodiversity impacts will depend on the methods applied. Restoration methods that are designed to promote natural species diversity and are suited to the conditions of the site can not only achieve better short- and medium-term outcomes for biodiversity and ecosystem services, but also enhance the long-term resilience of the restored ecosystems to climate change (UNEP 2014).

## TUNDRA ECOSYSTEMS

The potential for mitigation actions in tundra ecosystems is limited, as no feasible approaches are known that could help to slow the process of permafrost thawing, and the extent of direct human impacts on carbon stocks that can be addressed is relatively small. In the current situation, climate change mitigation through other activities thus seems to be the most promising option for reducing greenhouse gas emissions from tundra areas (Epple 2012; Schuur et al. 2015). However, given the expected rise in human influence on the tundra, approaches for the management of anthropogenic pressures to limit their negative impacts on soils, hydrology and vegetation should be developed now. In areas with increasing fire risk, realistic mechanisms to control and manage fires should also be put in place. Generally, the complex nature of the challenges caused by climate change in the remote but resource-rich tundra regions calls for the development of approaches and strategies that involve coordination and collaboration across sectors and stakeholder groups and between countries, and that address the anticipated environmental and socio-economic trends.

Despite the low density of human population in the tundra regions, adaptation to the impacts of climate change presents significant challenges both for public and private economic investment and for local communities, many of which are engaged in subsistence livelihoods. This is largely due to the fundamental and only partly predictable landscape changes that are caused by permafrost thawing, as well as to the impacts of climate change on populations of the large mammals that form the basis of many local livelihoods (Chapin et al. 2005). Strategies to manage the impacts of human intervention in tundra ecosystems on carbon stocks could be designed to take these processes into account and provide synergies with adaptation goals.

The biodiversity of tundra ecosystems is very sensitive to disturbance, mostly because of the long recovery times needed under the harsh climatic conditions. Mitigation approaches that manage the impacts of human intervention on tundra soils are therefore likely to yield biodiversity benefits as well. Risks to biodiversity could result from mitigation options that involve the manipulation of hydrological site conditions or the establishment of tree plantations.

## CROPLANDS

For the purpose of the present study, only those cropland management options that address greenhouse gas emissions from, and carbon sequestration in, soils and biomass have been identified as ecosystem-based mitigation approaches. Other approaches to mitigation in agriculture, for example through more efficient use of energy and chemical inputs or through better waste management, are beyond the scope of this document. Nevertheless, it is noted that such technological improvements should go hand in hand with the ecosystem-based approaches. For a comprehensive overview of mitigation options in agriculture, see Smith et al. (2014), p. 830 ff., and on the specific question of water management in rice cultivation to reduce methane emissions, see Tyagi et al. (2010).

It has been estimated that the total greenhouse gas mitigation potential that would be technically achievable within agriculture (including the management of livestock and grazing lands) corresponds to a net emission reduction of 1.2 to 1.6 Gt C per year by 2030, and that about 90 % of this potential is linked to soil carbon sequestration (Bernoux & Paustian 2015, see also Box 3 for more detail on available estimates of economically achievable mitigation potential in agriculture).

Among the main options for maintaining or increasing soil and biomass carbon stocks on croplands are reduced tillage, addition of organic matter to the soil, adjusting crop rotations to include cover crops and fallow periods, combining different crops on the same field, and agroforestry or the inclusion of hedgerows and forest buffers in agricultural landscapes (Banwart et al. 2015; Bernoux & Paustian 2015; FAO 2013; Haddaway et al. 2015; Smith et al. 2007; World Bank 2012). These practices have the potential not only to enhance the build-up of organic matter, but also to reduce carbon losses through soil erosion, and to contribute to the restoration of degraded agricultural land. The 'conservation agriculture' approach integrates many of the techniques identified above, and has been suggested as a useful way forward to combine climate change mitigation and adaptation while sustaining crop productivity (e.g. FAO 2013). Enhanced soil carbon stocks have also been observed as a consequence of conversion from conventional to organic farming (e.g. Gattinger et al. 2012). In addition, agroforestry can help to protect carbon stocks in adjacent forest areas by providing sustainable supplies of woody biomass for a variety of uses, including household energy production and construction (Neufeldt et al. 2015).

**Box 3: Estimates for the economically achievable mitigation potential of changes in cropland management**

Mitigation options for the agriculture sector are comparatively well researched, and a significant number of studies are available that include estimates of the potential global contribution of changes in cropland management to climate change mitigation. Smith et al. (2014) have compared estimates of mitigation potential across a range of studies. They classify estimates into ones that relate to 'technical mitigation potentials' (which do not consider the cost of measures), 'economic potentials' (that consider costs but not other information such as capacity and willingness to pay for and implement measures), and 'market potentials' (expected outcomes under current or forecast real-life market conditions, taking into account all relevant barriers and constraints). The authors found that average estimates of the global economic mitigation potential in cropland management are on the order of between 0.2 and 0.22 Gt C per year by the year 2030 for a carbon price of up to 20 and up to 50 dollars per ton of carbon dioxide, with a significant standard deviation of +/- 0.16 and 0.13, respectively. They note that the large variation between estimates is partly due to differences in the range of management options and greenhouse gases considered.

In order to achieve their aims, changes in agricultural practices need to be closely tailored to the specific environmental, socio-economic and cultural context in which they are to be implemented. Local knowledge and traditional forms of land management can often inform the selection of appropriate approaches. This is demonstrated by success stories of agricultural restoration and sustainable use around the world (see Case Study 6).

When choosing the best approaches for a more sustainable and efficient use of existing cropland, it may sometimes also be necessary to consider trade-offs between mitigation outcomes per unit of land and per unit of product. For example, productivity increases through changing production methods in areas of low-yielding agriculture may under some conditions come at the cost of rising emissions per hectare, but could still lead to a net mitigation benefit if less land now needs to be cultivated to meet the same demand, and if the new methods are socially and ecologically sustainable in the long term (Burney et al. 2010; Smith et al. 2014; Tilman et al. 2011). However, the possibility of rebound effects that increase demand or provide incentives to take further areas under production also needs to be considered (Angelsen 2010; Matson & Vitousek 2006; Smith et al. 2014). The concept of sustainable intensification has been used to describe approaches that aim to increase crop production per unit area while avoiding negative social and environmental impacts. Given the complex dependencies between the different actors in commodity production, global and regional markets, local livelihoods, overall socio-economic development and environmental conditions, the best ways to achieve this balance are still under discussion and need to be considered against the local context in which they are to be applied (FAO 2013; Garnett et al. 2013; Godfray & Garnett 2014; Smith et al. 2014).

The impacts of agriculture on ecosystem carbon stocks can also be addressed through measures that aim to reduce food waste and the demand for area- and energy-intensive products, in particular meat and dairy. It has been shown that livestock products generally have much larger requirements in terms of land and water use than vegetal products, and are associated with higher greenhouse gas emissions. For example, it has been estimated that the production of beef protein requires about 50 times more land than the production of vegetable proteins (Nijdam et al. 2012), and greenhouse gas emissions (excluding those from land-use change) are about 100 times higher. Numerous authors (Bajželj et al. 2014; Hedenus et al. 2014; Popp et al. 2010; Smith et al. 2013; Tilman & Clark 2014; Westhoek et al. 2014) have

suggested that reducing excessive consumption of meat and dairy products among affluent populations and changing nutrition patterns toward more healthy diets could help to solve the challenge of feeding a population of 9-10 billion people by 2050 (Bajželj et al. 2014; Stehfest et al. 2009), while reducing the area requirements and climate impact of food production. When discussing such recommendations for changes in diets and consumption patterns, there is however a need to differentiate between meat produced as an output of traditional pastoralist livelihoods in areas of semi-natural and natural grasslands that are mainly suitable for grazing (see also the section above on grasslands and savannahs), and products sourced from intensive livestock keeping systems, in which beef and dairy cattle are generally at least partially fed with feed coming from cropland (McGahey et al. 2014; Nijdam et al. 2012). Overall, according to Smith et al. (2013) the potential of demand-side measures in agriculture for climate change mitigation could be greater than that of all of the supply-side measures taken together.

While changes to agricultural practices along with measures that address demand can decrease the need for further cropland expansion, emissions from the conversion of other ecosystems to croplands can also be reduced by directing

**Box 4: Identifying suitable areas for the cultivation of bioenergy crops – the role of 'degraded' or 'marginal' lands**

The substitution of fossil fuels with bioenergy (i.e. energy derived from biomass), potentially in combination with carbon dioxide capture and storage, is considered an important option for climate change mitigation and features prominently in many of the global mitigation scenarios developed by the IPCC (Smith et al. 2014). At the same time, many authors have raised concerns about the effectiveness and sustainability of bioenergy pathways due to possible competition for land with food production and expected impacts on greenhouse gas emissions from land, as well as on water resources, biodiversity conservation and livelihoods (see Chum et al. 2011; Coelho et al. 2012; Gibbs et al. 2008; HLPE 2013; Smith et al. 2014).

The land area that will be required to produce a certain amount of bioenergy depends on the choice of feedstocks (i.e. what share of the energy is derived from wastes and residues or different types of dedicated energy crops), the techniques applied for cultivation and processing, and the productivity of the croplands used. While the available evidence suggests that some options with low lifecycle emissions (such as the use of sugar cane, *Miscanthus*, fast growing tree species, and sustainably sourced biomass residues) will be effective at reducing greenhouse gas emissions, the scientific debate about the overall climate impact related to land use competition effects of specific bioenergy pathways is still ongoing (Smith et al. 2014).

Many studies have investigated the availability of 'degraded', 'marginal' or 'underutilized' land that could be used for bioenergy crop cultivation without compromising food security and other social and environmental goals, both at the global level and for specific countries or regions (e.g. Coelho et al. 2012; Gelfand et al. 2013; Haberl et al. 2010; Liu et al. 2012; Nijsen et al. 2012; Zhuang et al. 2011). The outcomes from these studies vary widely depending on data sources and models used, as well as on the sustainability considerations included (Coelho et al. 2012; Eitelberg et al. 2015; Gibbs & Salmon 2015; Lewis & Kelly 2014; Smith et al. 2014).

For example, the review of published estimates of global cropland availability by Eitelberg et al. (2015) found that these ranged from 1,552 to 5,131 million ha, in each case including 1,550 million ha already being used as cropland. Thus, the lowest estimates provided would indicate that there is almost no room for further cropland expansion, while the highest estimates indicate that cropland could potentially expand to over three times its current area. Similarly, Smith et al. (2014) reviewed a large number of studies in order to identify a likely range for the global technical potential for bioenergy production under the condition of a 'food / fibre first' principle, and found that most studies support an estimate of at least 100 exajoule (EJ) per year by 2050, while some modelling assumptions lead to estimates exceeding 500 EJ per year.

Several authors have highlighted that some of the approaches used for the identification of available land for bioenergy crops (1) underrate the value of land categorized as degraded, marginal or underutilized for local livelihoods (especially those of poor and vulnerable groups), biodiversity conservation and ecosystem services, and potential future conversion to food production; (2) overrate their productive potential; or (3) fail to account for high initial emissions that would be incurred as a consequence of land conversion (Chum et al. 2011; Coelho et al. 2012; Cotula et al. 2008; Creutzig et al. 2013; Gibbs & Salmon 2015; HLPE 2013; van der Horst and Vermeylen 2011). At the same time, the evaluation of possible social, environmental and economic implications of bioenergy options conducted by Smith et al. (2014) identified good examples of win-win situations where introduction of bioenergy crops achieved emission reductions in combination with other positive social and environmental impacts (e.g. through restoration of land affected by salinization or erosion).

These findings, together with the high uncertainty of estimates of available land, suggest that mitigation efforts based on bioenergy should be planned carefully and where possible draw on approaches with low area requirements such as use of wastes and residues and integration of bioenergy production with food and fibre production (e.g in agroforestry systems).

expansion towards areas with a low share of vulnerable carbon stocks. For bioenergy crops in particular, the use of lands that are considered degraded or of marginal value for food production has been discussed as a promising approach to avoid major negative impacts on the environment and socio-economic conditions. However, care should be taken to consider the full value of such areas for local livelihoods, biodiversity and ecosystem services before decisions are made (see Box 4). Also, other possible uses of the land such as reforestation and forest restoration (in the case of former forest lands) should be considered, as these may under certain conditions provide larger benefits for climate change mitigation than the cultivation of bioenergy crops (see e.g. Albanito et al. 2016).

It seems likely that a combination of all of the strategies outlined above (applying practices that maintain or increase soil and biomass carbon stocks on existing croplands and restoring degraded croplands; aiming for sustainable and site-adapted levels and forms of intensification in terms of energy and chemical inputs; taking measures to reduce demand for area-intensive crops and livestock products; and directing cropland expansion to less vulnerable areas) can be useful to achieve a reduction in net emissions from crop cultivation and the conversion of other ecosystems to cropland.

A useful step towards creating the necessary impulse for the uptake of more sustainable agricultural practices (i.e. practices that avoid ecosystem degradation and reduce greenhouse gas emissions and/or land demand as described above) can be a review of current economic and fiscal incentives, in order to identify any perverse or ill-designed incentives that could be reformed or redirected to support more climate-friendly land management (see Case Study 5). The role of property rights and tenure regimes in shaping agricultural practices should be considered when addressing incentive design. Better targeting of incentives can also be aligned with efforts to achieve a more efficient allocation of land to different uses through landscape-level planning and/or regulatory approaches (see section 5). As agriculture is often critical to strategies to reduce pressure on forests and other ecosystems, there is a large potential for linking mitigation measures in agriculture with efforts to implement REDD+ or other forest-based mitigation actions. The recent progress in methods for measuring or estimating carbon stock changes on croplands can be of use for such approaches (Batjes & van Wesemael 2015; Vågen & Winowiecki 2013; World Bank 2012).

In terms of key changes to land management that could be promoted, some major opportunities exist in dryland regions. In these areas, ongoing losses of soil organic carbon through decomposition and erosion are often high due to unsustainable land use patterns, and there is a particularly strong potential to combine management for soil and biomass carbon with benefits for food security and adaptation to climate change (see Case Study 6). As described in the section on peatlands, there are also great opportunities to reduce emissions from cultivated peat soils. Building the capacity of farmers to apply good practices can help to overcome the barriers to their adoption, especially in situations where such practices may lead to economic benefits in the medium to long term.

Mitigation approaches that maintain or enhance soil and biomass carbon stocks on croplands are likely to provide benefits both for current livelihoods and food security and for adaptation to climate change. Higher contents of soil organic matter not only improve soil fertility, but also enhance water storage capacity, water infiltration, and resistance to soil compaction and erosion. This can create better conditions for the growth of crops, support groundwater recharge, and reduce sediment loads, pollution levels and flood risk in downstream areas (Bernoux & Paustian 2015; FAO 2013; Harvey et al. 2014; Scharlemann et al. 2014; Victoria et al. 2012).

If techniques for improving soil condition are strategically applied in combination with water saving and harvesting practices in order to prevent or reverse land degradation in drylands, they can provide significant economic benefits. They can further help to avoid the environmental damage and potential social conflicts related to displacement of land use. This has been demonstrated for example in degraded dryland areas of Africa and Asia (Reij et al. 2009; UNCCD 2015). Management practices that increase carbon sequestration in biomass, especially agroforestry, can also support food security, income diversification and livelihood stability, while contributing to the protection of soils and improving microclimates (FAO 2013; Mbow et al. 2014; Thorlakson & Neufeldt 2012; van Noordwijk et al. 2014).

By increasing structural diversity and the diversity of crop species in agricultural landscapes, many approaches for the enhancement of soil and biomass carbon stocks are beneficial for biodiversity, including that of non-cultivated species. Management practices that increase soil organic carbon contents often also support a higher diversity of soil organisms (Victoria et al. 2012). However, the most important mechanism through which mitigation actions in agro-ecosystems

can provide synergies with biodiversity conservation is likely to be that of reducing pressure on natural ecosystems, as farming on existing croplands becomes more sustainable and land degradation leading to lower yields is avoided. Risks to biodiversity are most likely to arise as an unintended side-effect in cases where the introduction of new and more profitable forms of management eventually provides an economic incentive for further land conversion (Angelsen 2010).

## A SPECIAL CASE: ABANDONED CROPLANDS

When a trend towards abandonment is ongoing in an area, the scope for mitigation actions is often relatively small. This is because most areas of former cropland will spontaneously revert to a vegetation type that is similar to the natural vegetation that was prevalent before conversion, and carbon sequestration will occur without further intervention. In some cases, especially on lands that have been abandoned in a degraded state or in landscapes where little natural vegetation is left, restoration efforts may be useful to speed up the recovery of soil and biomass carbon stocks. Depending on the climatic zone, fire management in abandoned croplands may also become an issue. This is especially the case in areas naturally covered by grasslands, where controlled burning or grazing with wild or domesticated animals can reduce emissions and enhance the build-up of soil organic matter (see also the section on grasslands and savannahs).

Yet, the main opportunities for ecosystem-based mitigation with regard to former croplands arise when increasing profitability of land use causes a trend towards re-conversion. In such a situation, greenhouse gas emissions can be reduced by directing conversion towards areas that have been abandoned more recently and hence had less time to regain their natural levels of carbon stocks. There is also the potential to avoid emissions by applying sustainable agricultural practices that protect soils and retain soil organic matter as far as possible (see preceding section). In many parts of the world where abandonment of cropland has occurred on a significant scale, the agricultural methods that have been applied in the past, as well as their ecological consequences, are well documented. This means that there is a good starting point for identifying more sustainable forms of management that can be applied in the future if re-cultivation of abandoned areas becomes necessary or desirable. Countries with a large share of abandoned lands that are likely to be returned to agricultural use should develop strategies early on to ensure that re-cultivation takes place in an efficient and sustainable way. Such strategies might include the establishment of policies, regulations, incentives, or governance and tenure arrangements that support the application of good practices for the conservation of soil carbon and other values provided by the ecosystem.

Many ecosystem services related to water regulation and other functions of the soil are enhanced when the agricultural use of an area is discontinued. This is particularly true for marginal lands, which are often among the first to be abandoned, as well as for lands that have been cultivated with unsuitable methods, often leading to increased water and wind erosion and an aridization of the local climate. By promoting the use of more sustainable methods where the re-cultivation of abandoned lands becomes necessary, these improvements in the supply of ecosystem services may be maintained. This can increase the resilience of farmers' livelihoods to climate variability and change.

Following the abandonment of cropland, a shift in species composition takes place, with species typical of more natural ecosystems becoming more frequent. However, a full recovery of species assemblages that are comparable to those of unconverted areas can take decades or even hundreds of years. In some cases it may not be possible at all, depending on the location of the area in relation to remnants of natural vegetation that can serve as a starting point for recolonization. Restoration measures with appropriate methods, which may include the transfer of individuals or seeds, can help to improve the biodiversity outcomes. The abandonment of large areas of cultivated former steppes in Eastern Europe and Central Asia provides unique opportunities for the restoration of ecosystem types that had almost fully disappeared from many regions. Biodiversity considerations should also be taken into account when appropriate areas for re-conversion to cropland need to be identified.

## CASE STUDY 5: REFORMING AGRICULTURAL SUBSIDIES

Around the world, subsidies[13] are used by governments as a means to promote activities that are considered to be in line with the achievement of certain policy goals, often in the social or economic sphere. Agriculture is among the economic activities that are most heavily subsidized (OECD 2003; OECD 2013). In 2012 alone, agricultural subsidies within a group of the world's top food-producing countries, who together account for almost 80 % of global agricultural value added (i.e. the 14 OECD countries plus the non-OECD EU countries, as well as Brazil, China, Indonesia, Kazakhstan, Russia, South Africa and the Ukraine) reached US $ 486 billion (OECD 2013; Potter 2014).

Due to their fundamental role in many countries' social and economic policies, the funds available for agricultural subsidies generally surpass those available for support to ecosystem-based climate change mitigation actions by a wide margin. For example, Norman and Nakhooda (2014) estimated a global commitment to REDD+ finance of about US $ 8.7 billion since 2006, whilst domestic agricultural subsidies at the receiving end greatly exceeded these contributions (McFarland et al. 2015). Thus, if the incentives provided by agricultural subsidies act against the aims of climate change mitigation policies, the effectiveness of the latter can be seriously compromised. On the positive side, changes to subsidy design can often be a highly cost-effective way to promote more climate-friendly agricultural practices and production patterns.

Concerns about the environmental sustainability of subsidies related to the use of natural resources have prompted Parties to the CBD to call for the elimination, phasing out or reform of subsidies that negatively affect biodiversity by 2020, through Aichi Target 3 of the Strategic Plan for Biodiversity 2011-2020 (UNEP/CBD 2011). Subsidies may affect the use of natural resources through their influence on investment, productivity and consumption, and by setting prices below societal costs. This can increase human pressure on biodiversity and ecosystem services due to effects such as inefficient production, overconsumption, and capacities and fund allocation that are increased beyond sustainable practices (McFarland et al. 2015; Pearce 2003; Valsecchi et al. 2009). At the same time, well-designed subsidy schemes can also help to promote the uptake of more sustainable practices, for example by compensating farmers for the delivery of ecosystem services from agricultural land (cf. Kurkalova et al. 2006; Merckx & Pereira 2015). According to OECD (2013), subsidy policies directly addressing environmental concerns continue to represent a small part of countries' portfolios, although an increasing number of countries make use of cross-compliance requirements, linking the provision of payments to farmers to the compliance with certain environmental standards above the legal minimum.

Agriculture is one of the main drivers of deforestation in many regions, and it has been estimated that about 80 % of global deforestation is caused directly by agricultural expansion (Bajželj et al. 2014; Houghton 2012; Kissinger 2015), with subsidies being an indirect driver at both the national and international levels (Geist & Lambin 2002; Goers et al. 2012; Kissinger 2015; McFarland et al. 2015). Similar effects occur with regard to the conversion of other ecosystems, such as peatlands, savannahs or grasslands (Joosten et al. 2012; McAlpine et al. 2009; Russi et al. 2013).

Reforms to agricultural subsidies have been called for given: the role of agriculture as a driver of ecosystem conversion and degradation; the fact that unsustainable agricultural practices are widespread in many parts of the world, leading to depletion of freshwater resources, nutrient pollution in terrestrial and aquatic ecosystems, loss of agrobiodiversity, and rising emissions of greenhouse gases; and the fact that many subsidy schemes are seen to favour large producers over small enterprises and subsistence farmers (Bajželj et al. 2014; Kissinger 2015; Lamb et al. 2016; McFarland et al. 2015; Potter 2014; TEEB 2015; UNEP/CBD 2011).

Worldwide, there is a growing number of good examples of national initiatives to reverse the effects of unsustainable agricultural subsidies. These range from the abolishment of a pesticide subsidy scheme in Indonesia or the removal of subsidies for wetland drainage in Austria to adjustments in India's intergovernmental fiscal transfer system that are designed to encourage forest conservation (Busch 2015; Kissinger 2015; SCBD 2011; TEEB 2015).

To elaborate on one example, government action in Brazil has helped to reduce Amazon deforestation during the first decade of the 21st century by restructuring agricultural subsidies, with impacts on cattle grazing and soy production. This involved introducing or amending existing legislation and promoting a change in management practices (Kissinger 2015). Between 1990 and 2004, Brazil experienced high rates of deforestation with an average loss of 2.7 million hectares of forest a year (McFarland et al. 2015). Conversion to cattle pastures was responsible for about three-quarters of this figure. During the same period, soy crops expanded to cover 34 % of the country's arable land (McFarland et al. 2015). According to Assunção et al. (2012), reforms undertaken in 2004 and 2008 were key to reversing the "perverse incentives" of the country's agricultural subsidies.

The first step of the change in Brazilian policy towards agricultural subsidies was the launch of the "Action Plan for the Prevention and Control of Deforestation in the Legal Amazon" in 2004 (Assunção et al. 2012). Through this plan, agricultural and forestry incentives were reviewed and modified to promote sustainable use and management (Assunção et al. 2012). This was followed by a set of Presidential Decrees and Reforms undertaken between 2007 and 2008, which included a provision to award rural credits only when the receiving entities were in compliance with legal and environmental regulations (Macedo et al. 2012; Nolte et al. 2013). Assunção et al. (2012) estimated that this provision alone reduced forest loss by 15 % between 2008 and 2011, in conjunction with a decrease in the allocation of credits of about US $ 1.4 billion (McFarland et al. 2015).

As set out above, the reform of agricultural subsidies can be an important contribution to efforts to reduce greenhouse gas emissions. Thus, the effectiveness of investments in climate change mitigation can be increased by simultaneously revising national and international incentives for unsustainable agricultural practices (Kissinger 2015). Current examples of success and the framework provided by the Aichi Targets could promote assertive action in this regard.

13 The World Trade Organization (WTO) defines subsidies as "a financial contribution by a government or any public body" where funds or liabilities are directly transferred, a funding mechanism or private trust is created, goods or services are provided, revenue is waived and/or income or prices are supported (WTO 1994).

## CASE STUDY 6: RESTORATION OF DEGRADED AGRICULTURAL LANDS IN DRYLAND AREAS

According to Safriel et al. (2005), between 10 and 20 % of the world's drylands are currently degraded, and 6 to 12 million km$^2$ suffer from desertification. Much of this degradation has been caused by unsustainable agricultural practices, which have prompted droughts, soil erosion, salinization and reductions in agricultural productivity (Safriel et al. 2005). This, in turn, has caused large scale human migration and related societal impacts (Lal 2002; Reij et al. 2009; Safriel et al. 2005; UNCCD 2009).

Restoration and rehabilitation of degraded drylands is thus a crucial endeavour for sustainable development, also bearing in mind that over the next 20-50 years, negative impacts caused by drought are projected to increase on a global scale (United Nations General Assembly 2011). Drought and related impacts affected 36 % of people suffering from environmental disasters between 1974 and 2003, and it is estimated that water-related factors have caused the displacement of between 24 and 700 million people (Guha-Sapir et al. 2004; World Water Assessment Programme 2009).

A range of land and water management practices have proven successful at reducing and preventing desertification and soil erosion, rehabilitating degraded drylands and sustainably intensifying agricultural production within arid ecosystems (Lal 2002; Reij et al. 2009; Safriel et al. 2005; Stene 2007; WOCAT 2007). Some examples are provided below.

In 1974, the Baringo District in Kenya was categorized as an "ecological emergency area", given that its semi-arid lands were subject to worrying levels of desertification and the Lake Baringo was drying up (Stene 2007). By 2001, about 50 % of the forest within this watershed had been cleared, and agricultural intensification was degrading the dryland ecosystem to an even greater extent (Stene 2007). Land degradation resulted in famine in dry years, and severe flooding in wet years, both of which generated increasing social unrest (Stene 2007). In the 1980s, the Rehabilitation of Arid Environments Trust (RAE), a local non-governmental organization working to overcome environmental degradation, started to work with local communities to reclaim and manage degraded land through soil and water conservation techniques, tree planting and introduction of native grasses tolerant to drought (Chabay et al. 2015; Stene 2007; RAE 2007). This, over time, allowed the recovery of grazing with sustainable practices, and thus, the main source of income for local communities (Chabay et al. 2015; Stene 2007; RAE 2007). Farmers shared the costs of this endeavour and the rehabilitated fields provided additional benefits such as seeds, cash income from surplus fodder, construction materials and a wide range of ecosystem services such as erosion control and soil recovery (Chabay et al. 2015; Feeding Knowledge 2015; Stene 2007). Stene (2007) showed that soils within protected fields had more capacity to absorb water inputs than those in areas not subject to rehabilitation (Stene 2007). Overall, this initiative restored more than 1,600 ha of degraded semi-arid land across Kenya and benefited more than 15,000 people directly (Chabay et al. 2015; Feeding Knowledge 2015).

The Three Northern Shelterbelts Project was established in 1960 across the Inner Mongolia Autonomous Region of China, to reduce wind erosion on drylands dedicated to crop production (WOCAT 2007). This agroforestry project undertaken by the Forestry Department was focused on establishing shelterbelts (i.e. rows of tall-growing tree species) to protect fields from erosion, sandstorms, droughts and freezing temperatures (WOCAT 2007). Shelterbelts were planted with both deciduous and evergreen species, and some usage of these was allowed under a rotational felling scheme that facilitated cash income through quality and high yield of tree products, whilst maintaining the protection offered by this "green infrastructure" (WOCAT 2007). The subsequent increase in crop yields promoted an extension of the shelterbelt planting reaching up to 500 km$^2$. Apricots (*Prunus armeniaca*) and Chinese dates (*Ziziphus jujuba*) were increasingly introduced as an alternative source of income (WOCAT 2007). Over 22 million hectares of vulnerable cropland have been protected through this project. It has been suggested that benefits to local farmers could be further enhanced through the establishment of sustainable harvesting systems to use the additional resources provided by the shelterbelts (WOCAT 2007). Species selection is crucial in the establishment of these "green infrastructures", to ensure that the usage of already scarce land and water resources by the planted trees is balanced as much as possible by increased availability of tree products (WOCAT 2007).

Concerned with the degree of erosion caused by agricultural production during the mid-twentieth century, the Government of Queensland in Australia established a service aiming to preserve the soil (Thomas et al. 2007). The methods promoted included adequate land use and crop selection and runoff management (Thomas et al. 2007; WOCAT 2007). Strip cropping and stubble mulching were later also encouraged to improve water infiltration, reduce runoff speed and counteract wind erosion. Subsequently, techniques to retain crop residues were introduced, and both research and extension programmes were undertaken to strengthen the use of sustainable agricultural

practices (Thomas et al. 2007). In 1985, the Conservation Farming Information Centre (currently, Conservation Farmers Inc.) was established to coordinate actions between different stakeholders. This enabled the increased adoption of no-tillage farming in the early 2000s together with other "conservation farming" practices (WOCAT 2007). Long-term research on active agricultural plots under no-tillage showed an increase in soil water storage of 20 mm and of 250 kg/ha in yield (Thomas et al. 2007). This was reported to bring an annual net benefit of approximately US $ 60/ha (Gaffney & Wilson 2003 in Thomas et al. 2007). In addition, crop residue prompted greater soil water retention allowing for longer sowing times after prolonged dryness (Thomas et al. 2007). By 2005, no-tillage techniques were applied on about 50 % of the main cropland areas in parts of Queensland and these management techniques are now considered standard practice, despite groups of local farmers remaining opposed to their adoption (Thomas et al. 2007; WOCAT 2007).

The Sahelian "*Green Revolution*" is another good example of dryland restoration. Farming communities across Burkina Faso and Niger adopted enhanced traditional practices to rehabilitate over 5.2 million hectares of degraded dryland (Reij et al. 2009). This was initiated at the end of the 20th century in response to major droughts and prolonged periods of dryness, which caused social disruption due to generalized labour migration (Reij et al. 2009). In Burkina Faso, it was estimated that groundwater levels within the Central Plateau region were dropping by up to 100 cm/year during the early 1980s (Reij et al. 2009).This reduced the agricultural productivity and generated an annual household food deficit of about 50 % (Reij et al. 2009). In response to this situation, "planting pits", "contour stone bunds" and the use of manure, all traditional techniques within the Sahel, were reintroduced and their application enhanced (Reij et al. 2009). This helped to capture soil and organic matter eroded by the wind, as well as improving soil structure and mineral content (Reij et al. 2009). In this way, at least 200,000 ha of dryland have been rehabilitated across the Central Plateau and crop yields in low rainfall conditions have increased to about 300-400 kg per ha per year (Reij et al. 2009).

In Southern Niger, "*Farmer-managed Natural Regeneration*" (FMRN) was initiated to ensure the continued provision of fodder, food, construction materials and firewood by adapting traditional techniques that avoided constant replanting. Under this model, farmers regenerate native trees and shrubs within their plots based on a simple process: when clearing the land, farmers select tree stems to protect based on their utility, and remove and/or prune other stems, creating a form of parkland where native trees and shrubs grow alongside the crop (Reij et al. 2009). As a result of these practices, replanting due to wind-blown sand is no longer required and the trees supply at least six months' worth of cattle forage each year to local farmers. Food availability has also been enhanced, and households are provided with medicines and surplus income from the sale of tree products. Sorghum yields have increased by 36-169 % and additional cereal yields range between 400 and 500 kg/ha (Reij et al. 2009). At the national scale, FMRN supplies an additional 500,000 tons of cereals per year through the regeneration of 5,000,000 ha of cropland, and the initiative has already been expanded to other regions (Reij et al. 2009).

Over the second half of the 20th century, severe deforestation and water shortages prompted extreme poverty within the Atiquipa community of Peru (Canziani & Mujica 1997; FAO et al. 2011). By the 1980s, about 90 % of the highly diverse forests found in humidity pockets across the Peruvian Coastal Desert had been cleared. At the same time, annual precipitation dropped to 40 mm (Caziani & Mujica 1997; FAO et al. 2011). The loss of the forest, which fulfils an important hydrological function by capturing humidity from the frequent fogs in the coastal Andes, led to increased soil erosion and water shortage threatening subsistence agriculture and livelihood conditions (FAO et al. 2011). Thus, the local community with support from the Universidad Nacional de San Agustín de Arequipa, initiated forest restoration activities using the tara tree (*Caesalpinia spinosa*) with recognized commercial and ecological values. In addition to capturing water from the fogs, tara, which is a plant from the legume family, facilitates nitrogen fixation in the degraded soils, and due to its size and root system it also contributes to erosion control (De la Cruz 2004). In addition, tara trees provide pigments and gums that are a source of income for the Atiquipa community (De la Cruz 2004; FAO et al. 2011; Torres Guevara & Velásquez Milla 2007). As a result of this initiative, the condition of about 400 ha of land affected by soil erosion on slopes of the Peruvian Coastal Desert has already been improved (Torres Guevara & Velásquez Milla 2007).

All of these success stories start from an initial stage of severe degradation of important natural resources prompting the awareness of key stakeholders. These stakeholders have then sought alternatives that balance revenue and sustainability, reaching a final middle ground that in all cases has allowed the restoration of drylands, together with a significant improvement in living conditions at the local level and overall agricultural productivity. Climate change mitigation was not a central consideration when the initiatives started (in many cases several decades ago) and for most of the examples presented here, mitigation benefits have not been quantified. However, it is clear that the restoration of croplands has helped to maintain and increase carbon stocks, both by enhancing soil carbon content on the plots themselves and by halting further degradation and thus the need to open up additional land for agriculture.

# 6. INTEGRATING ECOSYSTEM MANAGEMENT AT THE LANDSCAPE SCALE

There are many possible approaches to promote more climate-friendly management of ecosystems, ranging from individual projects and community-based conservation initiatives to changes in legislation or large-scale programmes aiming to provide capacity-building and incentives to certain groups of land users or concession holders, to name only some examples. However, the efficiency of ecosystem-based mitigation efforts can be enhanced greatly through landscape-level planning, taking into account the demands of different sectors and stakeholder groups for land and ecosystem services, as well as the suitability and availability of different parts of the landscape for various uses and the linkages between ecosystems and their surroundings (DeFries & Rosenzweig 2010; Harvey et al. 2014; Scherr et al. 2012).

A number of studies have investigated the possibility to reduce trade-offs between goals such as commodity production, local livelihoods and food security, climate change mitigation and adaptation, and biodiversity conservation, in specific landscape contexts (e.g. Austin et al. 2015; Koh & Ghazoul 2010; Law et al. 2015; Siikamäki & Newbold 2012; Venter et al. 2009). Despite differences in approaches and the range of factors considered, all authors conclude that integrated planning for multiple purposes can deliver substantially greater overall benefits than scenarios based on individual sectoral priorities. Often, large gains in the potential to deliver one ecosystem service can be achieved at the cost of moderate reductions in another (Austin et al. 2015; Harvey et al. 2014; Siikamäki & Newbold 2012; Venter et al. 2009).

In order to increase the likelihood of success, initiatives for the management of ecosystems at the landscape scale should be developed with the engagement of all relevant sectors of government and stakeholder groups, paying particular attention to marginalized and vulnerable populations and ensuring a gender-balanced approach. Ensuring that plans for the use of land and natural resources reflect the legitimate interests, needs, capacities and perspectives of all stakeholders can increase the long-term viability and cultural appropriateness of agreed measures and reduce the risk of 'leakage effects' through displacement of activities from one area or ecosystem type to another (Scherr et al. 2012; UN-REDD 2013).

Possible barriers to integrated landscape management include:

- Gaps in knowledge about the ecological and socio-economic implications of different spatial configurations in the allocation of land to specific uses, and about the available management options;
- Unclear, conflicting or unsupportive tenure regimes or rules for access to resources, which prevent resource users from engaging in long-term planning or adopting management practices that require initial investment before they become profitable;
- Limited capacity on the part of certain groups to participate in a multi-stakeholder coordination process;
- Governance structures that inhibit decision-making at the landscape level, do not encourage cross-sectoral planning or are unable to accommodate traditional governance mechanisms and access rights recognized by indigenous peoples and local communities, including collective rights;
- A lack of rules and incentives to support implementation of agreed plans.

(Compiled from Runsten & Tapio-Bistrom 2011, Scherr et al. 2012 and Shames et al. 2011.)

Overcoming these constraints may require action on a number of levels, including the identification of institutional mechanisms to support multi-stakeholder planning processes; clarification and/or reform of tenure rights and rights to the use of resources; establishment of regulatory instruments or incentives that are appropriate to the national and local context, e.g. through land use zoning, fiscal incentives, financial support schemes or product certification schemes; and identification of mechanisms to track the outcomes of plan implementation as a basis for the documentation of best practices and adaptive management.

Positive examples of integrated management of ecosystems at the landscape level exist from a variety of geographical and ecological contexts, ranging from coastal zones to watersheds and river basins or mountain ranges. Initiatives that explicitly try to accommodate mitigation and adaptation goals are still a relatively recent development. However, Scherr

et al. (2012) analyze existing case studies of such 'climate-smart' landscape initiatives and conclude that results from these early-stage initiatives can already inform future efforts in their development of stakeholder and institutional capacities.

The integration of ecosystem-based mitigation actions with other forms of land use at the landscape level can be supported at the national level through the harmonization of sectoral policies and programs related to climate change, agriculture, forestry, biodiversity conservation and economic development.

# 7. AREAS FOR FURTHER RESEARCH

As described above, important progress has been made in recent years in improving the state of knowledge on the global distribution of organic carbon stocks and rates of greenhouse gas flows to and from ecosystems under different land use intensities and in different ecological settings. There are, however, still many areas where better understanding could support the planning of concrete actions that use the potential of ecosystems to contribute to climate change mitigation, biodiversity conservation and sustainable development. Areas for further targeted research include:

- The spatial distribution of soil carbon stocks, including stocks below 1 m depth in peatlands, permafrost areas and coastal ecosystems; a global-level effort to increase knowledge on peatland distribution (including under agricultural land uses) and peat depths could provide much-needed baseline information for initiatives to conserve soil carbon;
- The climate impact of non-carbon dioxide emissions and albedo effects resulting from wildfires, vegetation changes and changes in hydrology, especially in peatlands, grasslands and tundra ecosystems;
- The fate of soil organic matter that is exported from terrestrial and coastal ecosystems as a result of erosion, in particular with a view to assessing the share of eroded carbon that is re-deposited in other locations versus the share that is oxidized and emitted to the atmosphere as carbon dioxide;
- Additional studies to identify good practice for specific approaches to ecosystem-based mitigation. These studies should run over a sufficient period to generate knowledge on long-term outcomes, which is currently often incomplete. Topics for such studies could include:
  - improvement of land use practices in those peatlands that are currently under intensive use;
  - management of grazing by wild and domestic animals in various types of grasslands (also taking into account methane emissions caused by grazing animals) and under different forms of governance;
  - sustainable enhancement of cropland productivity to reduce emissions from agricultural expansion and conversion of other ecosystems; and
  - restoration of mangroves in a way that provides good results for climate change mitigation and adaptation, disaster risk reduction and livelihoods.

  Reviews of traditional knowledge and practices of indigenous and local communities related to ecosystem management could contribute to such studies;
- Improvement of models to predict the impacts of climate change and different forms of management on multiple ecosystem services and carbon stocks and flows, both at the global scale and at site level; again, the incorporation of data from long-term studies could greatly enhance the accuracy of such models;
- Scenario analysis of possible impacts on ecosystems of different socio-economic development trajectories and related changes in drivers of ecosystem degradation and conversion, as well as their implications for the feasibility and long-term likelihood of success of ecosystem-based approaches to mitigation; and
- Further development of cheap and efficient approaches for estimating and measuring changes in ecosystem carbon stocks and flows for both terrestrial and coastal systems.

# 8. CONCLUSIONS AND RECOMMENDATIONS

As can be seen from the information contained in this document, some general lessons are starting to emerge from research and practice on ecosystem-based approaches to climate change mitigation.

1. A perceived lack of knowledge about the mitigation benefits that can be achieved through managing non-forest ecosystems often hinders the uptake of such actions, as well as their mainstreaming across climate, biodiversity and other policies. However, there is a growing body of information, data and methodologies that can provide the basis for concrete planning and target-setting, as well as for communication and awareness-raising among decision-makers. This information can be drawn from guidance documents adopted by the Intergovernmental Panel on Climate Change (IPCC), as well as documents developed in the context of voluntary project standards, certification schemes and donor-funded projects. The references provided throughout this document can be used as a starting point.

2. Efficient land use policies are those that integrate climate change mitigation and adaptation, disaster risk reduction and sustainable development, while also providing biodiversity benefits. Research from a wide variety of ecosystems and socio-ecological settings shows that management options that avoid or reverse greenhouse gas emissions from ecosystems are in most cases also beneficial for biodiversity and the continued delivery of important ecosystem services. At the same time, the available evidence suggests that higher levels of biodiversity within an ecosystem type can enhance ecosystem resilience and function, and thus the permanence, and possibly size, of the ecosystem carbon pool.

   Successful mitigation of climate change, including through ecosystem-based approaches, can also create a positive feedback loop, as it reduces the risk of negative impacts of climate change on ecosystems and their carbon stocks. Thus, using the full potential of ecosystem-based approaches to climate change, and designing these measures to enhance the contribution of biodiversity to carbon stocks in line with Aichi Target 15, can help to address several development challenges simultaneously.

3. Lessons from policies and actions targeting forests can inform the design of interventions in other types of ecosystems. In recent years, many developing countries have made significant efforts to establish policies, institutional arrangements, methodologies and baseline data for REDD+[14]. Developed countries have also improved their capacity to monitor forest-based emissions and sequestration rates. A significant number of countries have included actions targeting forests in their response to climate change, as exemplified by the Intended Nationally Determined Contributions (INDCs) announced under the UNFCCC. There are good examples where possible synergies between these efforts and biodiversity policies have been reflected in strategies and plans on climate change and/or in National Biodiversity Strategies and Action Plans (NBSAPs). Many of the lessons learned from these processes about success factors and possible challenges, on matters such as the assessment of pressures and identification of options to address them, as well as the definition of targets, social and environmental safeguards, mechanisms for participation, and incentive systems, may be transferable to initiatives involving other ecosystems.

The following recommendations can be made:

1. Countries should assess the extent and drivers of processes leading to ecosystem degradation and conversion, as well as opportunities for the restoration and sustainable use of ecosystems, and act on identified opportunities for integrated land use management providing benefits for the climate, biodiversity and ecosystem services. Possibilities to transfer lessons learned from forest-based mitigation efforts to other ecosystems should be explored.

2. Where ecosystem-based measures to address climate change are envisaged, they should be based on landscape-scale planning involving active and equitable engagement of stakeholders across sectors and

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scales, including indigenous people and local communities. This can enhance the efficiency, viability and local ownership of measures, given competing demands on terrestrial and coastal areas and the fact that the most suitable areas for different uses are distributed unevenly across landscapes and may be covered by a range of property and tenure rights and legitimate stakeholder interests. This is particularly true for areas where access to resources is shared between large numbers of people, or where use rights are unclear or overlapping, as is often the case in grassland or coastal ecosystems.

3. A review of the incentives (and disincentives) that are in place for different land uses should be carried out to identify opportunities where reforms could make a transition to more sustainable management approaches economically viable and enable positive contributions to local and national economies. Other possible policy options include regulatory approaches such as land use zoning or permitting requirements, the establishment or improved management of protected areas, and demand-side measures for agricultural products.
4. Donors interested in supporting integrated land management in a particular region should invest in initiatives to make baseline data available for the planning of mitigation and adaptation actions based on ecosystems, as location-specific and ready-to-use information can facilitate action, leveraging large gains for biodiversity and sustainable development.
5. While many options for ecosystem-based approaches to address climate change are likely to benefit biodiversity, some risks are also becoming apparent, in particular for natural grasslands; these should be taken into account when looking for actions that provide multiple benefits. Where measures carrying potential risk such as afforestation or the cultivation of biofuels are considered, the likely outcomes in terms of carbon sequestration and greenhouse gas emissions, climate change adaptation, disaster risk reduction, biodiversity conservation and support to local livelihoods should be carefully assessed.

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