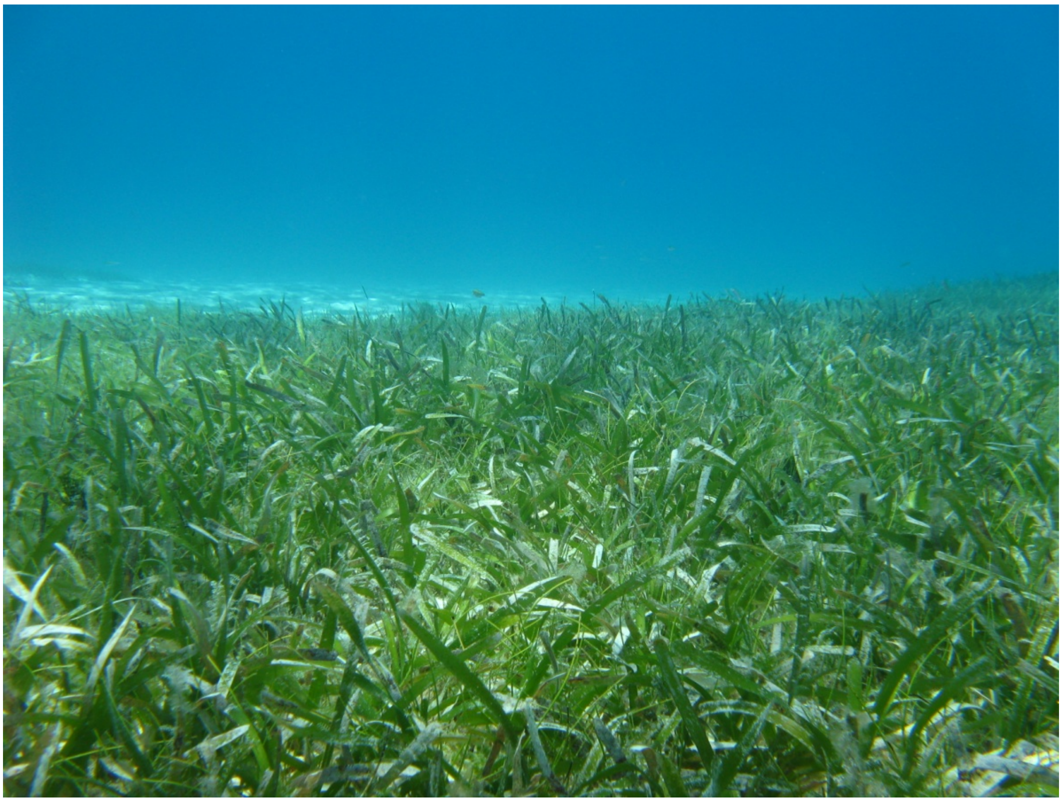
Cordula Epple

The climate relevance of ecosystems beyond forests and peatlands – A review of current knowledge and recommendations for action



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BfN
Federal Agency for Nature Conservation

Cover picture: Seagrass meadows (J. Fourqurean)

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This report was prepared by UNEP-WCMC as a background document and basis for discussion for the workshop "The climate relevance of ecosystems beyond forests and peatlands", which took place from 26-29 October 2011 at the International Academy for Nature Conservation on the Isle of Vilm, Germany. It was subsequently updated in collaboration with the participants to include additional information and ideas developed during and after the meeting. The workshop conclusions and recommendations are included as an annex to the document.

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Executive Summary

Many efforts to promote ecosystem-based approaches to climate change mitigation have so far focused on forests, agricultural ecosystems and – to a lesser degree – peatlands. However, there is a wide range of other options for ecosystem-based mitigation, and opportunities in non-forested dryland systems and coastal marine ecosystems deserve more attention.

The potential benefits from mitigation activities in different types of ecosystems can be assessed based on criteria such as typical carbon stocks per hectare, potential emission and sequestration rates of all types of greenhouse gases, area extent, rates of anthropogenic degradation, importance for other ecosystem services such as food provision or biodiversity conservation, and practical feasibility of interventions.

Enough information is available to demonstrate that a number of dryland and coastal marine ecosystems score highly against such criteria. However, margins of uncertainty in quantifying global and site-scale emission trends and mitigation potential are still greater for non-forest ecosystems than for forests, and are particularly large for some types of coastal marine ecosystems. This is partly due to the fact that in ecosystems other than terrestrial forests, the proportion of carbon stored in the soil is generally very high, and measurement methods for soil carbon are time- and cost-intensive.

Major research efforts that help to address these and other gaps in knowledge are under way as part of initiatives to improve the data base for understanding and monitoring the global carbon cycle and regional or national carbon budgets.

Where possible, plans for ecosystem-based mitigation should also take account of emerging knowledge on how climate change will affect the greenhouse gas balance of terrestrial and marine ecosystems, both directly and as a result of human adaptation or mitigation responses.

Results from a review of the current information base for assessing the mitigation opportunities linked to the major non-forested terrestrial biomes and to coastal marine ecosystems include the following:

In **tundra** ecosystems, despite high carbon densities and concern about climate change impacts on ecosystem carbon stocks, opportunities for direct intervention to increase the uptake of carbon or decrease greenhouse gas emissions are small, as land use change has so far been limited.

In **dryland** ecosystems, there are significant opportunities for linking climate change mitigation with other ecological and/or socio-economic benefits, including biodiversity conservation and poverty alleviation. Drylands encompass a large part of the world's temperate, subtropical and tropical **grassland**, **savannah**, **shrubland** and **desert** ecosystems, and are particularly susceptible to land degradation as a consequence of poor management. Their carbon densities can reach those of forests, but significantly lower values are also found, especially in the more arid regions. Current emissions from conversion are estimated to be particularly high in woodland savannahs in South America and Africa.

Options for climate change mitigation in drylands include avoiding further conversion of natural habitats, rehabilitating degraded landscapes, improving the management of overexploited croplands and pastures to maintain and restore soil carbon stocks and reduce emissions of all greenhouse gases, and improving fire management. The suitability of afforestation measures in drylands (i.e. planting trees in areas where natural woody vegetation would be lim-

ited) depends on the location and the methods applied, and impacts on biodiversity, water resources and other ecosystem services need to be taken into account.

Among **coastal marine ecosystems**, **mangroves**, **salt marshes** and **seagrass meadows** have the greatest potential for climate change mitigation. Vegetated marine habitats can contribute to carbon sequestration and storage through the growth and accumulation of plant biomass, as well as through the trapping and burial of organic carbon in their sediments. Estimates of average carbon burial rates for the three vegetation types investigated are higher than any soil carbon sequestration rates reported for terrestrial ecosystems. However, sequestration rates vary between sites, depending on a range of environmental factors. Total ecosystem carbon densities, too, are variable but generally high. Information on current distribution areas and rates of degradation and loss is of limited quality for all three vegetation types, but reflects strong anthropogenic pressure.

Considering the rapid decline of vegetated coastal marine habitats, both conservation and restoration are important mitigation approaches. Restoration efforts and the development of appropriate methods are most advanced for mangrove ecosystems, but there are also good examples of salt marsh restoration. For seagrass meadows, restoration is particularly costly, and a focus on conserving remaining areas is recommended. Mitigation efforts in coastal ecosystems can achieve significant additional benefits for society, as the range of environmental services provided by these systems is wide and includes natural coastal protection and potential for adaptation to sea level rise, biodiversity conservation, and habitat functions for commercially relevant fish species.

Major challenges in using the potential of dryland and coastal ecosystems for climate change mitigation will include reconciliation of competing demands for land at the local, national and global scale in order to arrive at management regimes that are both ecologically and socio-economically sustainable, and the further development of knowledge and methodological approaches to plan for, measure and verify the success of interventions.

The efficient application of incentive schemes requires development and improvement of default values and accounting methods to calculate the mitigation benefits of management interventions at site scale. Progress in these areas has been made for several types of mitigation activities in non-forest ecosystems, but further work is required to broaden the spectrum of activities covered.

Emerging standards and methodologies for ecosystem-based mitigation activities in grasslands, savannahs and coastal wetlands should be widely applied. In addition, pilot projects with appropriate scientific support could help to test and demonstrate the feasibility of new approaches and at the same time enhance the knowledge base on the climate relevance of a wider range of ecosystems.

By broadening the scope of ecosystem-based mitigation approaches, the available land area for mitigation activities, the number of stakeholders who can be involved and the overall mitigation potential can be increased significantly. Paying attention to all types of ecosystems within an area can also reduce the risk of unintended effects from mitigation strategies that are exclusively focused on forest carbon stocks.

1 Introduction

Living organisms play a decisive role in the greenhouse gas cycle, and ecosystems can be both sources and sinks of greenhouse gases, depending on the environmental and anthropogenic factors to which they are exposed. According to recent estimates, natural carbon sinks on land and in the oceans have taken up more than half of all anthropogenic carbon dioxide emissions between 1959 and 2008 through a combination of biotic and abiotic processes. At the same time, emissions from land use change made up a significant share of anthropogenic emissions, estimated at 20 % for the average of the period 1990-2000, and around 12 % in the year 2008 (Le Quéré et al. 2009).

It is therefore widely recognized that improving the management of ecosystems in order to preserve and enhance their carbon stocks and reduce greenhouse gas emissions should be an integral part of strategies to combat climate change. Estimates of the potential of measures in different sectors to achieve emission reductions and/or carbon sequestration, taking into account expected costs, indicate that changes in agriculture and forestry practices can make a contribution to climate change mitigation that rivals or surpasses that of technical options in industry or the transport sector (IPCC 2007). Many ecosystem-based approaches for climate change mitigation can also provide further environmental and social benefits by maintaining or restoring ecological productivity, ecosystem services and biodiversity, as well as by strengthening the resilience of ecosystems to impacts of climate change and other stressors, thus contributing to climate change adaptation.

Much of the attention given to ecosystem-based mitigation approaches has so far focused on forests. There are several reasons for this. First, forests are among the most carbon-rich ecosystems on Earth and many forest areas are facing high anthropogenic pressure (Trumper et al. 2009). The main share of carbon dioxide emissions from land use change as estimated with current methods is caused by tropical deforestation (Le Quéré et al. 2009)[1]. Second, the wide range of environmental, social and economic benefits delivered by forests has led to a long tradition of forest conservation efforts in policy and practice. Third, there is also a methodological aspect: temperate and tropical forests are among the few types of terrestrial vegetation which hold more carbon in the biomass than is stored in the soil (Trumper et al. 2009), making changes in carbon stocks both more conspicuous and easier to assess than for other ecosystems.

Another area where the development of methods and practices is well advanced is the enhancement of carbon stocks in agricultural soils. Effects of different cultivation practices on soil qualities, including organic carbon content, have been the subject of numerous studies, and management methods are available that reverse or avoid the depletion of soil carbon stocks caused by traditional farming practices (Smith et al. 2008).

In recent years, an increasing number of initiatives are also addressing emissions from peatlands. These systems are another globally significant emission source due to the notable

[1] Note however that the data base for these estimates is focused on detection of processes such as deforestation and forest burning, which may lead to an underestimation of emissions from other ecosystems.

share of degraded areas and the particularly high carbon densities found in these systems (Parish et al. 2008).

Although priority action in forests, croplands and peatlands is certainly justified by the potential for high emission reductions and/or carbon sequestration per unit area, the wide range of other options for ecosystem-based climate change mitigation should not be disregarded. By further developing the knowledge base and methodologies to address climate-relevant processes in dryland or coastal ecosystems, the available land area for mitigation activities, the number of stakeholders who can be involved and the overall mitigation potential can be increased significantly.

Moreover, mitigation strategies that focus solely on forest carbon stocks without considering the consequences for non-forest areas carry a substantial risk of unintended or even counter-productive side effects. Several authors have assessed the possible implications of a REDD+[2] mechanism for non-forest ecosystems. For example, the Terrestrial Carbon Group Project carried out a spatial analysis of the possible impacts of REDD+ on future agricultural conversion pressures for other natural landscapes, based on a simplified set of assumptions about legal, biophysical and economic factors influencing these pressures (Terrestrial Carbon Group 2009). Emphasizing the high potential for the displacement of land use change (so-called leakage), they conclude that under a scenario where REDD+ would succeed in protecting 50 % of the current forest area from conversion, displacement of land use change to other ecosystems could mean that emissions from land use change would decrease by no more than 10 % unless steps are taken to reduce overall land demand. Other authors have highlighted the risks that displacement of land use to non-forest areas poses for the conservation of biodiversity (e.g. Miles & Kapos 2008). Including the management of non-forest ecosystems in climate change mitigation strategies can help to reduce such risks.

Consideration of a wide range of ecosystems in mitigation efforts has also been supported by the governing bodies of several Multilateral Environmental Agreements, including the Convention on Biological Diversity (CBD) and the United Nations Convention to Combat Desertification (UNCCD). For example, the CBD's Strategic Plan for 2011-2020 calls for an enhancement of the contribution of biodiversity to carbon stocks through conservation and restoration of ecosystems in general, and CBD Decision X/33 invites Parties to consider implementation of ecosystem-based approaches for mitigation *inter alia* in natural forests, natural grasslands, peatlands, wetlands, mangroves, salt marshes, seagrass beds and agricultural ecosystems.

The present report aims to give an overview on the state of knowledge and relevant research initiatives concerning the status and trends of major ecosystem types with regard to their carbon stocks and greenhouse gas balance, taking into account human influence and possible future impacts of climate change. Recommendations are made on priorities for action and further research, based on the information reviewed.

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[2] REDD+ stands for reducing emissions from deforestation and forest degradation, conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries.

It should be noted that the kind and amount of additional benefits that can be obtained from ecosystem-based approaches to climate change mitigation (e.g. benefits for livelihood security, desertification control, biodiversity conservation or climate change adaptation) depends on the choice, design and location of activities. For example, measures to prevent further conversion of natural ecosystems are in most cases beneficial for biodiversity conservation, but their effects on local livelihoods depend on the exact choice of activities. Livelihood security can be enhanced if it is possible to decrease conversion pressure by making current land use methods more efficient, while preventing conversion through stricter land use regulations without additional measures can have negative effects on local livelihoods. The positive or negative impacts of measures to increase carbon sequestration through enhancement of carbon stocks, or to decrease fossil fuel emissions through the cultivation of biofuel feedstocks, can vary widely depending on their location and the restoration or cultivation methods used.

While considerations related to the potential for additional benefits are mentioned throughout this report, an exhaustive discussion would go beyond the scope of the document.

2 The research landscape

Several research communities are currently developing information that can help to assess the relevance of different ecosystem types for climate change mitigation.

A wide array of initiatives aim to improve the data base for understanding and monitoring the global carbon cycle and the carbon budget of individual regions or countries, including through the creation, strengthening and linking of regional carbon observation systems and the planned establishment of a global carbon observation system as part of the Global Earth Observation System of Systems (GEOSS). Examples of regional observation systems and related projects include CarboEurope, CARBOOCEAN, the US Carbon Cycle Science Program and the CarboAfrica project. Activities include the establishment of additional measurement infrastructure for monitoring carbon fluxes and pools, the further development and harmonization of measurement methods, the facilitation of data exchange and the closing of data gaps related to key processes affecting the greenhouse gas cycle. The EU Coordination Action on Carbon Observation Systems (COCOS) has worked on improving global datasets of variables such as fire emissions, soil carbon and feedbacks between climate and the carbon cycle, and on improving regional carbon budget estimates, including by reviewing data from existing biomass and soil carbon inventories.

Efforts are also underway to develop a global soil observing system as part of GEOSS.

The Global Carbon Project, a joint effort by the International Geosphere-Biosphere Programme (IGBP), the International Human Dimensions Programme on Global Environmental Change (IHDP), and the World Climate Research Programme (WCRP), aims to build an international framework for integrated research on the carbon cycle.

Unfortunately, because the initiatives described above mainly aim to improve overall global or regional estimates of carbon budget parameters, the collected data are not always published in a form that makes it possible to relate them to specific ecosystem types. Moreover, many of their component activities are focussed on forest ecosystems.

Further research activities related to various aspects of the role of ecosystems in the climate system are carried out by a broad spectrum of academic institutions and national research initiatives and programmes, such as the North American Carbon Program.

Targeted work on the greenhouse gas balance of individual ecosystem types under different uses or degrees of human impact has also been carried out in a number of smaller-scale studies synthesizing information from available literature and databases, often with the explicit aim of promoting ecosystem-based mitigation approaches. Examples include work supported by the FAO on carbon sequestration in drylands (e.g. Conant 2010) and work on carbon storage in coastal ecosystems (e.g. Laffoley & Grimsditch 2009).

In-depth studies linked to selected clearly defined ecosystem-based mitigation activities have been undertaken to support the development of accounting requirements for obtaining carbon credits that can be sold on the market for voluntary emission offsets. In order to develop such requirements for mitigation projects applying a certain type of activity, the main factors determining the impact of the activity on greenhouse gas emissions and sequestration need to be identified, and minimum methodological standards for measuring and reporting on these factors and calculating the resulting climate benefits need to be defined. Examples of certification standards that offer options for the validation of carbon credits from ecosystem-based mitigation projects include the Verified Carbon Standard (which has recently approved accounting requirements for peatland rewetting and conservation projects; work on requirements for wetlands conservation projects and projects avoiding the conversion of grasslands and shrublands is ongoing), the Protocols of the Chicago Climate Exchange (e.g. for soil carbon sequestration in sustainably managed rangeland and soil carbon sequestration through continuous conservation tillage and conversion to grassland), and a recent Australian standard for reducing greenhouse gas emissions from savannah burning (see Russell-Smith & Whitehead 2008).

The GEF Carbon Benefits Project has developed a web-based carbon measuring and modelling system that can be used to assess the impacts of land management interventions funded by the GEF or other donors on carbon stocks or greenhouse gas emissions.

Several projects have attempted to organize information on the multitude of research efforts that are underway. For example, the COCOS programme included a component aiming to compile an inventory of national and international research activities that are relevant to carbon cycle research. The Terrestrial Carbon Group has produced lists of major organizations working on, *inter alia*, estimates of land-based mitigation potential for major land classes, process level understanding of carbon dynamics and mitigation potential, and the scientific research base for alternative management practices (Terrestrial Carbon Group 2010).

A non-exhaustive list of relevant ongoing and completed research programmes and projects and related web resources is provided in Box 1.

Box 1: Research programmes and projects addressing the climate relevance of ecosystems (non-exhaustive list)

Global Carbon Project (2001 - ongoing): http://www.globalcarbonproject.org/, see also list of Regional Contributions at http://www.globalcarbonproject.org/research/regionalContributions.htm

Regional Carbon Cycle Assessment and Processes – RECCAP (2007 – 2011): http://www.globalcarbonproject.org/reccap/

FLUXNET (1998 – ongoing): http://www.fluxnet.ornl.gov/fluxnet/index.cfm

GEOCARBON project (2011 - 2014): see http://www.cmcc.it/research/research-projects/geocarbon-operational-global-carbon-observing-system

Global Earth Observation System of Systems (based on Group on Earth Observations Implementation Plan 2005 - 2015): http://www.earthobservations.org/geoss.shtml

CarboEurope (2004 - 2009): http://www.carboeurope.org/

CarboAfrica (2006 – 2010): http://www.carboafrica.net/index_en.asp

CarboEastAsia (2007 - 2012): http://www.carboeastasia.org/

CarboNA (2007 – ongoing): http://nacarbon.org/carbona/index.htm

CARBOOCEAN (2005-2009): http://www.carboocean.org/

Regional pilot platform contributing to a Global Soil Observing System - e-SOTER project (2008 – 2012): http://www.esoter.net/

COCOS - EU Coordination Action Carbon Observation System (2008 - 2011): http://www.cocos-carbon.org/

Integrated Carbon Observing System – ICOS (2008 – ongoing): http://www.icos-infrastructure.eu/

US Carbon Cycle Science Program (2003 - ongoing): http://www.carboncyclescience.gov/programs.php#Carbon_Program

North American Carbon Program (2002 – ongoing): http://www.nacarbon.org/nacp/

Large Scale Biosphere-Atmosphere Experiment in Amazonia (1998 – ongoing): http://lba.inpa.gov.br/lba/

Australian Soil Carbon Research Program (2009 – 2012): http://www.csiro.au/science/Soil-Carbon-Research-Program

African Carbon Exchange Project (2004 – 2007): http://www.nrel.colostate.edu/projects/ace/index.html

Greenhouse gas management in European land use systems (2010 – 2013): http://www.ghg-europe.eu/

Blue Carbon Scientific Working Group (2011 - ongoing): http://www.marineclimatechange.com/marineclimatechange/bluecarbon_2.html

3 Information needs

Despite the significant research efforts described in the preceding chapter, still not all information needs can be met. In their "roadmap for terrestrial carbon science" (Terrestrial Carbon Group 2010), the Terrestrial Carbon Group identified six areas requiring further work in order to provide the necessary knowledge base for effective climate change mitigation in terrestrial ecosystems:

1. enhancement of the process-level understanding of carbon dynamics and mitigation potential in ecosystems,
2. provision of knowledge on available alternative management practices,
3. development of feasible accounting tools for all lands, carbon pools and greenhouse gases,
4. development of a framework for integrating information from different scales into a tiered global information system,
5. supporting the establishment of national accounting systems, and
6. harmonization of reporting guidance across scales and sectors.

This report is concerned primarily with the baseline research needs covered by points 1 and 2.

The authors further note the unequal distribution of research and information synthesis across land classes, carbon pools and regions of the world. With regard to the availability of information on carbon dynamics, alternative management options and accounting tools for different types of ecosystems, they highlight the need for more comprehensive research on grassland, dryland, wetland and peatland systems as compared to forests and agricultural systems (see also further details in the individual chapters of this report).

As Janetos et al. (2005) point out, even basic information on the status and trends in different land cover classes is insufficient at the global level, with deforestation being the most intensively studied process of land cover change. Still, estimates of CO_2 emissions from deforestation and forest degradation are far from certain (van der Werf et al. 2009). One reason for the low quality of information on the status and trends of non-forest ecosystems is the fact that these are harder to detect with remote sensing methods. Estimating their aboveground carbon stocks via remote sensing is even more difficult, as illustrated by the range of values for vegetation biomass given by Goetz et al. (2009).

Measurements of soil carbon are time- and cost-intensive and constitute a particular bottleneck in assessments of carbon stock, carbon stock changes and mitigation potential (Conant 2009, Terrestrial Carbon Group 2010). There is also a need for harmonization of measurement and mapping methods. Henry et al. (2009) compared soil organic carbon estimates for Africa derived from four recent global digital soil maps and five soil properties databases, which held between 1799 and 4043 soil profiles. They found a variation of about ±30% when comparing the soil properties or the spatial data of the different databases.

In terms of regional information coverage, the difference in availability and quality of data on carbon budgets and ecosystem carbon stocks between developed and developing countries has been reinforced by the reporting requirements on greenhouse gas emissions and sequestration according to land categories and uses for industrialized countries under the United Nations Framework Convention on Climate Change (UNFCCC). These requirements have triggered significant investment in research, development of methodologies and information synthesis in developed countries in recent years. Information gaps are particularly wide with regard to the African continent (Williams et al. 2007, Henry et al. 2009, Ciais et al. 2011).

In order to assess the mitigation potential in a specific location, information is needed not only on current carbon stocks and greenhouse gas uptake and emissions, but also on their likely future development both in the absence of mitigation measures (i.e. a reference level) and with mitigation measures in place. The development of scientifically sound reference levels poses a significant challenge due to the uncertainties about socio-economic developments, changes in climate and impacts of climate change on ecological processes[3].

Readily available information on typical values for ecosystem carbon stocks and greenhouse gas balances under different kinds of management and environmental conditions can facilitate assessments and projections and reduce the need for field measurements. Use of default values is an important part of methods for the assessment of mitigation options and greenhouse gas estimation methodologies both at the scale of individual projects and at subnational or national scale (e.g. guidelines for national greenhouse gas inventories under the UNFCCC). However, in many parts of the world, only coarse estimates for major ecosystem types and land use activities like the default values provided by the IPCC (2003, 2006, see also IPCC Emission Factor Data Base, http://www.ipcc-nggip.iges.or.jp/EFDB/main.php) are available. The development of region-specific default values that take into account different subtypes of ecosystems and regionally typical forms of land management could significantly improve the results of assessments. In the case of projects aiming to earn carbon credits, reducing the uncertainty of default values can also reduce the need for discounting the credits that are awarded for a certain activity. If suitable input data are available, even more specific assessment results can be obtained through the use of appropriately calibrated process-based models that integrate a range of environmental factors (see e.g. Olander & Haugen-Kozyra 2011).

Assessments of the overall mitigation potential for larger regions or ecosystem types should differentiate between the biophysical mitigation potential and the mitigation potential that can be realized in practice, taking into account socio-economic factors such as the implementation and opportunity costs of measures. Methods for estimating implementation and opportunity costs, and for arriving at estimates of the feasible mitigation potential of different ecosystem-based approaches, vary between authors, with higher agreement in results for well-defined activities such as changes to specific agricultural practices, and wide disparities for approaches that involve large numbers of stakeholders and require fundamental changes to current patterns of natural resource use such as reducing emissions from deforestation (cp. IPCC 2007, Stern 2007).

Finally, if ecosystem-based mitigation approaches are to be designed in a way that promotes the delivery of multiple benefits, it is desirable to improve available knowledge on the current

[3] For examples of integrated assessments of likely future changes in carbon stocks, cp. Smith et al. (2005) and Müller et al. (2007). For an example of an attempt to model the combined interactions of ecosystem N and C dynamics with climate change, cp. Zähle et al. (2010).

and potential social and environmental benefits obtained from these ecosystems, and the way in which possible mitigation measures interact with such benefits.

4 Overview of available information on carbon stocks, sequestration and emissions in different ecosystem types

This chapter presents the results of a review of published data on the carbon stocks and greenhouse gas balance of major terrestrial and coastal ecosystem types. Data on carbon stocks, emissions and sequestration per hectare are provided in tonnes, whereas total global values are provided in gigatonnes (1 gigatonne = 10^9 tonnes). If emitted to the atmosphere in the form of carbon dioxide, 1 gigatonne carbon corresponds to an increase in atmospheric CO_2concentration of about 0.47 ppm.

When comparing data on ecosystem carbon stocks and greenhouse gas fluxes provided by different authors, a number of caveats need to be observed, as will be discussed in the following section.

4.1 Introductory remarks on the cited data sources

Delimitation of ecosystem types

A significant constraint for comparisons of data from various sources is caused by differences in the use of terms related to the delimitation of ecosystem types. For terrestrial ecosystems in particular, a variety of classifications are in use and sometimes the same designations are used with different meanings. Information on the applied classification systems or criteria is not always provided, making it impossible to assess possible causes for discrepancies between datasets. For example, Ciais et al. (2011) mention the difficulty of synthesizing information on ecosystem soil carbon stocks and biomass in Africa due to very different definitions being used for the same ecosystem types.

For the purpose of structuring the present report, terrestrial ecosystems are classified according to the seven biome types distinguished in Trumper et al. (2009)[4], based on the delimitation of biomes provided by Olson et al. (2001), which reflects the ecological and structural characteristics of dominant vegetation types and thus seems well suited as a basis for considering the role of ecosystems in the carbon cycle.

As far as possible, relevant information from the reviewed literature is linked to the ecosystem type which most closely corresponds to the classification unit used in the original source, and discussed in the respective chapter.

A particular case is the abundant literature considering the potential for climate change mitigation in "drylands". There are several definitions of drylands, the most widely applied being that of the United Nations Convention to Combat Desertification (UNCCD) and the one used

[4] i.e. tundra; boreal forest; temperate forest; temperate grasslands, savannas and shrublands; deserts and dry shrubland; tropical and subtropical grasslands, savannas and shrublands, and tropical and subtropical forests.

under the Programme of Work on Dry and Subhumid Lands of the Convention on Biological Diversity (CBD).

The UNCCD defines drylands according to an aridity index: the ratio of mean annual precipitation to mean annual potential evapotranspiration. According to this definition, drylands are areas with an aridity index value of less than 0.65. Drylands as defined by the UNCCD include a large part, though not all, of the temperate, subtropical and tropical grassland, savannah and shrubland areas and the desert and dry shrubland areas as identified in Trumper et al. 2009, as well as a small share of forest areas.

The CBD definition of drylands used under the Programme of Work on Dry and Subhumid Lands (document UNEP/CBD/SBSTTA/5/9) is wider than the UNCCD definition and includes additional areas on the basis of the dominant vegetation types, a large share of which are dry forests (see UNEP-WCMC 2007).

Most authors addressing climate change mitigation issues in relation to drylands are referring to the UNCCD definition.

As some of the available information on the climate relevance of dryland ecosystems cannot be disaggregated in order to relate it to the three main dryland biomes, it is presented in a separate section preceding the chapters on the grassland and desert biomes.

Comparability of methods and metrics used to measure carbon stocks

A wide range of different standards and methods are in use for measuring ecosystem carbon stocks, particularly with regard to below-ground biomass and soil organic carbon. A major problem in comparing estimates from different sources are the large differences regarding the soil depth up to which samples were taken. While some authors only provide values for the top 10 or 15 cm of the soil, other studies have examined soil profiles of up to 3 m depth. Where the depth of the analysed soil layers for the same ecosystem type varies among authors, it is not possible to make meaningful comparisons.

Data on carbon stocks and carbon densities within biomes

For each of the terrestrial biomes discussed in this report, values are provided for the total carbon stock in biomass and soils, as well as for the mean carbon density per hectare. These values were obtained by overlaying the biome boundaries derived from the map developed by Olson et al. (Olson et al. 2001, Trumper et al. 2009) with a global map of terrestrial carbon distribution (UNEP-WCMC 2008).

The latter was derived from two global datasets: a dataset of carbon stored in live biomass based on IPCC Tier-1 Methodology using global land cover data (Ruesch & Gibbs 2008), and a dataset compiled by the International Geosphere-Biosphere Programme on soil carbon up to 1 m depth (IGBP-DIS 2000[5]).

[5] note that this dataset is likely to underestimate carbon stored in peat soils

Although the use of IPCC default values for biomass involves significant simplification, this approach has the advantage of providing globally consistent results. When comparing the average carbon densities between biomes, it has to be kept in mind that the estimates derived from the map overlay reflect not only the carbon densities of the dominant natural vegetation types and subtypes within each biome and their relative share in the total area, but also the degree to which the natural vegetation has been converted to other, usually less carbon-dense land uses.

4.2 Tundra

Tundra ecosystems are found in Arctic and mountainous environments, particularly in Northern Canada, Scandinavia, Russia, Greenland and Iceland. The determinant climate parameters are low to very low temperatures for most of the year, with prolonged periods of snow cover and a short growing season. The active layer of soil, near the surface, tends to be waterlogged in summer and frozen in winter, and usually overlays a perennially frozen layer known as permafrost (Trumper et al. 2009). Tundra ecosystems are characterized by treeless vegetation consisting of dwarf shrubs and/or grasses and sedges. Alongside tundra ecosystems, land cover within the tundra biome includes barren areas with little or no soil cover and vegetation, and permanently ice-capped areas. Estimates of the areal extent of land cover types in the Arctic differ substantially among authors, depending on the classification of vegetation (Chapin et al. 2005).

Area extent of the biome, total carbon stock and mean carbon density

The tundra biome covers an area of approximately 12.9 million km^2. The total carbon stock within the biome is estimated at 155.4 Gigatonnes C (Trumper et al. 2009). The mean value of carbon density across the biome is 120.4 t C/ha.

Current land use and conservation status and expected trends

The magnitude of land use change in polar regions has so far been limited, although there has been a rapid increase in population after 1960 as a result of expanded resource development and government activity. Since 1990, population growth has slowed in the polar regions of North America and Greenland, and population has declined in Arctic Fennoscandia and particularly in Russia. Human impacts have mostly resulted from industrial development such as mining, construction and operation of metal smelters, and construction of pipelines and transport infrastructure leading to local loss of vegetation and damages resulting from pollution.

Overgrazing by reindeer herds has led to degradation in some areas. In parts of the Russian Arctic, the tree line has retreated southward as a result of forest harvest and anthropogenic burning, increasing the area of treeless vegetation (Chapin et al. 2005). It is difficult to assess future development trends, although some of the consequences of climatic warming might favour an increase in human activity.

Ecosystem carbon stocks and greenhouse gas balance

Tundra vegetation is formed by slow-growing hardy plants with low above ground biomass and a high proportion of plant carbon allocated below ground (De Deyn et al. 2008). Rates of decomposition are low and large amounts of dead plant material accumulate in the soil.

Estimates of above-ground plant biomass provided by Shaver et al. (1992) are highest for riparian shrub tundra, with about 19 t/ha, and lowest for evergreen heath tundra with about 2 t/ha. Amthor et al. (1998) cite an average value of 6.3 t C/ha for the total amount of carbon contained in both above and below ground tundra biomass.

Jobbagy and Jackson (2000) analysed the relationship between soil organic carbon distribution up to the maximum depth of 3 m, and climate and vegetation parameters, based on three global databases including more than 2700 soil profiles. They calculate an average value of 180 t C/ha for tundra soils. Amthor et al. (1998) cite an average content of 127.5 t C/ha for the top 1 m of tundra soils, while Amundson (2001) gives an average value of 218 t C/ha without specifying soil depth. Schuur et al. (2008) state that a typical tundra permafrost soil can contain up to 440 t C/ha in the top meter.

The thickness of the permanently frozen layer of arctic soils can be between several hundred meters and less than one meter, and due to cryogenic mixing and other soil processes, organic carbon can be found up to large depths. Estimates of the global carbon pool in permafrost soils are highly uncertain, but Schuur et al. (2008) provide a figure of 1,672 gigatonnes C as their best approximation, with 1,024 gigatonnes in the top 3 meters.

Under current conditions, both direct measurements and global carbon cycle models suggest that the Arctic is neither a large source nor a large sink of carbon dioxide. At the same time, high-latitude wetlands constitute a large natural source of atmospheric methane (Chapin et al. 2005).

There is little potential to increase the uptake of carbon or decrease greenhouse gas emissions in tundra ecosystems at present.

Likely impacts of climate change

The potential impacts of a warming climate on the greenhouse gas balance of tundra ecosystems are a matter of serious concern to scientists. One of the main issues discussed is the potential release of soil carbon through thawing of permafrost. Schuur et al. (2008) cite figures according to which thawing of permafrost as a consequence of climate change and subsequent decomposition of soil carbon could release 40 gigatonnes C into the atmosphere within four decades and 100 gigatonnes C by the end of the century, enough to produce a 47 ppm increase in atmospheric CO_2 concentration (Trumper et al. 2009).

As methane flux from soils increases with soil moisture and summer temperatures, thawing of permafrost is also likely to enhance methane release (Chapin et al. 2005). While the colonisation of tundra areas by shrubs and eventually trees would increase biomass carbon, the effect would not be enough to offset carbon losses from soils (Schuur et al. 2008). Also, the decrease in albedo would increase annual energy absorption, leading to a positive feedback. A northward movement of the treeline has already been observed in Alaska (Chapin et al. 2005).

Other potential impacts of global change that are discussed in the literature include an increased occurrence of fire in tundra regions, with observations already confirming such a trend (Mack et al. 2011), and reduced growth of some plant species as a consequence of higher levels of UV-B radiation.

Knowledge gaps

In addition to uncertainty about the total size of the carbon pool in permafrost soils and its horizontal and vertical distribution, the difficulty of predicting the hydrological conditions that will be created by the thawing of permafrost in combination with changes in precipitation constitutes a challenge for projections of the future greenhouse gas balance of tundra ecosystems. Depending on the landscape and site conditions, a variety of processes can be triggered by thawing permafrost, including both the formation and the drainage of wetlands and lakes (Schuur et al. 2008, Merbold et al. 2009, Koven et al. 2011). These processes have a strong impact on the rates and pathways of the decomposition of organic matter.

4.3 Grasslands and deserts: the dryland systems

The global distribution of natural grassland and desert ecosystems is largely determined by limitations in the availability of soil moisture as a result of low rainfall and/or high evaporation. As mentioned in chapter 4.1, the major part of the area occupied by the natural grassland and desert biomes is therefore included in the area identified as 'drylands' according to the definitions used by the UNCCD and CBD.

Dryland areas are particularly susceptible to land degradation, i.e. the reduction or loss of biological or economic productivity. The widespread degradation of dryland areas, also called 'desertification' has received significant attention over the past decades and has motivated the development and adoption of the United Nations Convention to Combat Desertification. As many of the processes leading to land degradation entail reductions in biomass and soil carbon, there is a large potential for synergy between efforts to combat desertification and climate change mitigation, which has been analyzed by a number of authors (e.g. Lal 2001, Trumper et al. 2008, Bai et al. 2008).

Drylands are commonly subdivided into hyper-arid, arid, semiarid and dry subhumid areas, depending on the severity of the moisture deficit. The distribution of desert, grassland, shrubland, woodland and forest ecosystems across the area of drylands is not directly correlated to these subtypes, because vegetation characteristics are also influenced by soil properties and geomorphological features. However, as a general rule, the area occupied by grassland increases from the hyper-arid to the semiarid zone, and forests mostly occur in the dry subhumid zone where they occupy a slightly larger share of the area than grasslands (Safriel et al. 2005).

According to the World Atlas of Desertification and based on precipitation and temperature data collected between 1951 and 1980, drylands cover more than 40 % of the Earth's land surface (Middleton & Thomas 1997, Safriel et al. 2005).

Current land use and conservation status and expected trends

The dominant land uses in the dryland region are agriculture and pastoralism. Rangelands and croplands jointly cover 90 % of the global dryland area and are often closely interlinked. Altogether, an estimated 25 % of the world's drylands are used as croplands. The share of croplands decreases from about 47 % of the total area in the dry subhumid zone to nearly zero in the hyper-arid zone, most of which is to some degree used as extensive rangeland. The importance of pastoral use peaks in the arid zone, as wetter areas are more important for agriculture and drier areas do not support significant numbers of livestock (Safriel et al. 2005).

Both agricultural and pastoral use can have a significant impact on the greenhouse gas balance of terrestrial ecosystems. For a more detailed discussion of the issues involved, see the *ex cursus* on the role of croplands and grazing lands in the terrestrial carbon balance on p. 19.

The transformation of natural grasslands to croplands continues, and some 15% and 14% of the natural habitats in the semiarid and dry subhumid zones were transformed between 1950 and 1990 (Safriel et al. 2005). In parallel to carbon loss, conversion of grasslands to poorly managed croplands in developing countries often results in loss of fertility and increased soil erosion. Common degradation processes in grassland under excessive pastoral use include reduction of vegetation cover, in turn leading to soil erosion, vegetation shifts towards scrubland of low productivity and depletion of soil nutrient reserves. Another process which can induce soil degradation are anthropogenic changes in fire frequency.

Despite the global concern about desertification, available data on the extent of land degradation in drylands are limited, and agreement is hindered by methodological problems (such as discerning between impacts of climate variability and anthropogenic impacts) and definitional issues. Safriel et al. (2005) cite earlier estimates of the proportion of dryland areas suffering from desertification that range from 10 to 70 %, and conclude that the true figure is likely to be between 10 and 20 %.

The Global Assessment of Land Degradation and Improvement (GLADA) considered changes in the net primary productivity of vegetation as an indicator of land degradation or improvement and used remote sensing to identify trends in land degradation between 1981 and 2006 (Bai et al. 2010). Their results indicate that during the period 1981-2006, about half of the global land area experienced no significant changes in degradation status. Significant negative change mainly due to human activities could be allocated to 4% and significant negative change mainly due to climate change to another 5% of the total area. Encouragingly, there were also areas experiencing significant improvement in degradation status: on roughly 10 % of the area, they found significant positive change that was mainly due to human activities, and a further 15 % improved due to benign changes in climate. These figures compare positively to the figures obtained from an earlier version of the analysis which only considered data up to the year 2003, and found a degrading trend on 24 % of the land and improvement on 16% (Bai et al. 2008).

Comparing their findings to the 1991 Global Assessment of Soil Degradation (GLASOD), which looked at degradation status rather than trends in degradation, the authors concluded that the land area identified as degrading hardly overlapped with the already degraded areas, meaning that new areas were being affected. At the same time, some areas of historical land degradation had been so degraded that they were now stable at very low levels of productivity. Degrading areas were found mainly in Africa south of the Equator, Southeast Asia and South China, North-Central Australia, the Pampas, and swaths of boreal forest in Siberia and North America. A disproportionately high share of degrading land globally (i.e. including non-dryland areas) was found to be cropland and forest (Bai et al. 2008).

As a general rule, land degradation in drylands is most widespread in the semiarid zone, as a consequence of relatively high sensitivity to human pressure (which increases with aridity) and human population density (which decreases with aridity) (Safriel et al. 2005).

As future projections indicate a continued rise in population densities and an increase in frequency and duration of droughts for many dryland areas, along with increasing water withdrawal rates especially in some arable regions (Siebert et al. 2010, UNEP 2012), it is expected that the vulnerability of drylands to degradation will grow over the coming decades.

Impacts of land degradation and efforts to combat desertification on dryland carbon stocks

Based on their assessment of changes in net primary productivity, Bai et al. (2008) conclude that the loss of carbon fixation from the atmosphere caused by land degradation between 1980 and 2003 amounts to almost one gigatonne. Emissions from the loss of soil carbon are likely to have been even greater (SCBD 2010). Lal (2001) estimated that dryland ecosystems contribute 0.23 – 0.29 gigatonnes of carbon a year to the atmosphere, as a result of desertification and related vegetation destruction, through increased soil erosion[6] and a reduced carbon sink.

Despite the remaining knowledge gaps concerning extent and severity of soil degradation at different scales, it seems likely that the mitigation potential of degraded lands is significant. Lal (2001) estimated that up to 0.4–0.6 gigatonnes of carbon a year could be saved if eroded and degraded dryland soils were restored and their further degradation was arrested. Measures that can help to restore desertified soils include water saving and harvesting practices, re-establishing vegetation cover and minimizing soil disturbance, improving soil nutrient budgets and diversifying land use systems. As Lal (2009) points out, these generic recommendations must be made site specific through local/adaptive research.

According to Lal (2009), reclaiming salt affected soils can further increase sequestration on degraded lands. For example, biofuel plantations consisting of salt-tolerant plants could sequester between one and three tonnes of carbon per hectare and year and if well-designed could have net positive effects on biodiversity by decreasing land use pressure on other areas.

Examples of salt-tolerant species that might hold potential for biofuel production include *Halopyrum mucronatum*, *Desmostachya bipinnata*, *Phragmites karka*, *Typha domingensis* and *Panicum turgidum* (Abideen et al. 2011).

Safriel et al. (2005) note that similar estimates of the potential carbon sink capacity of dryland ecosystems have been provided by other authors and point out that examples of changes towards more sustainable agricultural management in drylands exist and can be built upon.

Conant (2010) emphasizes the potential of good grassland management to reverse historical soil carbon losses and sequester additional carbon in soils. He further highlights the potential of agroforestry, and cites figures according to which the conversion of 630 million hectares of unproductive or degraded croplands and grasslands to agroforestry could sequester as much as 0.59 gigatonnes of carbon annually by 2040, noting that this would be accompanied by modest increases in N_2O emissions as more N circulates in the system.

[6] Trumper et al. (2008) mention the difficulty of assessing the fate of carbon lost through soil erosion. While some of the organic carbon contained in eroded soil material is buried and sequestered, it seems likely that a large fraction is emitted into the atmosphere (Safriel et al. 2005).

Afforestation of drylands (i.e. planting of trees in areas where the natural vegetation would support no or less woody vegetation) has been successfully implemented in some areas and can offer climate change mitigation as well as livelihood benefits. However, the suitability of afforestation measures depends on the location and the methods applied. Impacts on biodiversity, water resources and other ecosystem services need to be taken into account (Safriel et al. 2005).

Likely impacts of climate change

Due to the fact that they are already suffering from water stress, all types of dryland ecosystems are highly vulnerable to further decreases in precipitation or increases in rainfall variability, leading to periods of drought or greater frequency and extent of fire. Periodic deteriorations in climate may also force farmers to apply unsustainable practices such as increased use of groundwater for irrigation, which can exacerbate desertification processes. A higher frequency and duration of dry spells is projected for many dryland areas (Conant 2010), and Safriel et al (2005) state that there is a high degree of certainty that the combined effects of global climate change, land use developments, and land cover changes will lead to an accelerated decline in water availability and biological production in drylands.

The role of croplands in the terrestrial carbon balance – an *ex cursus*

Expansion of agriculture has been the single most dominant trend in land use since the beginning of human civilization. However, estimates of the proportion of natural ecosystems that has been converted to cropland vary considerably between authors. Reasons for this include difficulties in identifying some types of cropland from satellite imagery, definitional issues, and methods applied for classifying areas with a mosaic land use structure. Cassman et al. (2005) state that globally, agricultural land has expanded by around 13 million hectares per year over the past 25 years, predominantly at the expense of natural forests and grasslands. They estimate that some 24 % of the global land surface are now occupied by cultivated systems (including intensively managed grassland).

The largest and most rapid areas of cropland increase between 1980 and 2000 appear to have been located in Southeast Asia, followed by Bangladesh, the Indus Valley, parts of the Middle East and Central Asia, the region of the Great Lakes of eastern Africa, and the southern border of the Amazon Basin in Latin America. Main areas of decrease are found in the southeastern United States, the eastern part of China and parts of Brazil and Argentina (Janetos et al. 2005). It is expected that despite regional decreasing trends, the overall extent of cultivated areas will continue to grow (Cassman et al. 2005). The recent trend towards large scale foreign investment in agriculture that is observed in some parts of the world is likely to contribute to this development.

While intensive farming methods have enabled large increases in productivity in many agricultural areas, some countries, particularly in sub-Saharan Africa, face persistently low levels of productivity and continue to rely on the expansion of cultivated area to meet increasing food demand (Cassman et al. 2005).

Land clearance for agriculture greatly reduces above ground carbon stocks and as a rule also depletes below ground biomass and soil carbon stocks. WBGU (1998) cite figures according to which conversion of natural ecosystems to cropland leads to a mean decrease of 25 – 30 % in carbon content of the top 1 m of soil, but emphasize that rates of loss vary widely depending on farming methods and environmental conditions. They estimate that globally, agricultural conversion has led to a decrease in soil carbon stocks of 38 gigatonnes.

Ogle et al. (2005) compared published data on impacts of agricultural land use on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. The documented impacts were larger in tropical than in temperate regions, and areas with moist climate showed greater sensitivity than dry areas. For example, long-term cultivation caused the greatest loss of soil organic carbon in tropical moist climates, with cultivated soils having only 58% of the amount found under native vegetation. Guo and Gifford (2002) carried out an analysis of published data on impacts of conversion of pasture to cropland, and found that on average soil carbon stocks decreased by almost 60 %. Similar rates are reported by Conant (2010). Lal (2004) estimated that the conversion of natural to agricultural ecosystems causes depletion of the organic soil carbon pool by as much as 60% in soils of temperate regions and 75% or more in cultivated soils of the tropics, and that historic soil carbon loss could have been as high as 42 to 78 gigatonnes.

In principle, these losses are reversible. For example, Vuichard et al. (2008) report gains of 470 kg of soil carbon per ha per year in agricultural land abandoned after the breakup of the Soviet Union. Similar observations have been made for abandoned agricultural lands in Europe and North America (Trumper et al. 2009). In practice, it is to be expected that the potential for discontinuing agricultural use of lands will in many areas be highly constrained by rising demands for food and other agricultural products, as well as by socio-economic conditions and negative impacts of climate change on agricultural productivity (Cassman et al. 2005). It is therefore more realistic to consider the potential for carbon sequestration through improved soil management, set-aside of smaller land parcels and rehabilitation of degraded land. According to IPCC estimates, 50 to 75 % of the lost soil carbon could be regained by such measures, allowing for sequestration of 23 to 44 gigatonnes of carbon within 50 years (WBGU 1998). Lal (2004) estimates that the practically achievable carbon sink capacity of the world's agricultural and degraded soils would be 50 to 66% of historic carbon loss.

The choice of farming methods can strongly influence the climate impacts of agricultural land use, and a significant knowledge base on appropriate practices is already available despite gaps in regional coverage (Trumper et al. 2009, Conant 2010, Smith et al. 2008, Ogle et al. 2005). Further research may be needed to examine the distribution of sequestered soil carbon across different pools, and implications for the stability of carbon stores (Steffens et al. 2009, Terrestrial Carbon Group 2010). In addition to effects on soil carbon, other aspects such as energy demand, methane emissions and emissions of nitrous oxide need to be considered when assessing the greenhouse gas balance of agricultural management. There may be trade-offs between the different factors, as for example nutrient depletion due to low nitrogen input may contribute to loss of soil carbon.

It is also important to consider off-site impacts. For example, water abstraction in order to support irrigation may have impacts on the greenhouse gas balance of the source ecosystems, and addition of manure to fields may lead to carbon depletion on grazing lands. Even more complex is the question of balancing practices that reduce land demand per unit of crop output, thus potentially reducing pressure towards land use change in other areas, against practices that reduce greenhouse gas emissions per hectare of cultivated land[7]. The cultivation of biofuels is another issue requiring careful assessment, and many authors have warned that under some circumstances emissions from expansion of agricultural activity may more than offset the emissions saved through reduced burning of fossil fuels (Trumper et al. 2009, Cassman et al. 2005).

Cassman et al. (2005) state that most cropping systems have undergone a steady net loss of soil organic matter during the last century. Losses have been particularly high in some areas of the tropics, where erosion, tillage and removal of crop residues have led to soil degradation (see Chapter 4.3). Some soils in tropical agricultural systems are estimated to have lost as much as 20-80 t C/ha (Lal 2004). Bai et al. (2008) state that more than 20 per cent of all cultivated areas worldwide show a degrading trend.

Potentially beneficial practices that could be adopted include conservation tillage, crop rotation and use of cover crops, measures to avoid soil erosion, integrated nutrient management using compost and manure or nitrogen-fixing species, returning of crop residues to the field, integrated weed and pest management, irrigation and water management, and agroforestry (Lal 2009, Smith et al. 2008, Conant 2010, Cassman et al. 2005). Afforestation may also be an option on degraded lands and marginal soils although impacts on the water balance need to be considered.

Which approaches are best suited in a particular location depends on the specific environmental and socio-economic conditions and can therefore only be determined by small-scale assessment.

Smith et al. (2008) estimate that by 2030, 5.5 to 6 gigatonnes of carbon dioxide equivalent could be saved per year through soil carbon sequestration and avoided emissions if best management was widely adopted, making the agriculture sector broadly carbon neutral. About 70 % of the potential are seen in developing countries.

In addition to climate change mitigation, farming methods that maintain and restore soil carbon stocks can also produce other benefits that are of great value for climate change adaptation and food security, such as increasing soil fertility and stability and contributing to water storage capacity. Still, despite their documented advantages they are not yet common practice.

The role of grazing land in the terrestrial carbon balance

Estimated potential rates of carbon sequestration per unit area are lower for grazing land than for cropland, but the total sequestration potential of the two land use categories is comparable because of the large area covered by the former. Conant (2010) cites figures according to which it would be realistic to achieve sequestration of 0.2 to 0.8 gigatonnes of carbon dioxide per year in grassland soils by 2030, assuming prices for CO_2 of US$ 20 to 50 per tonne.

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[7] Cp. related discussions on integration or segregation of land use for agriculture and biodiversity conservation, e.g. Phalan et al. (2011).

As with croplands, a significant share of the sequestration potential in grasslands used as pasture is a result of previous degradation (see Chapter 4.3). Bai et al. (2008) concluded that about 16 percent of rangelands are currently undergoing degradation, but at the same time also noted a strong trend for improvement in other areas. Degradation in grazing lands is most commonly a result of overstocking and uncontrolled grazing, and major problems can arise in regions where land tenure rights are not well defined (Cassman et al. 2005).

Because livestock is at present the fastest growing agricultural sector, making up over 50 percent of agricultural GDP in many developing countries (Conant 2010), pressure on the land is likely to increase further in order to meet meat and milk demand.

Practices with the potential to increase livestock carrying capacity of grasslands and augment carbon stocks include fertilization, irrigation, sowing of favourable grasses and legumes, grazing management including appropriate timing and stocking densities, fire management and rehabilitation of degraded lands (Conant 2010).

As noted above, there is a need to consider the balances of all greenhouse gases and to take a systems perspective in order to identify appropriate land management regimes. A particular issue in the case of grazing lands are methane emissions from livestock. Overall emissions per unit area may even increase if enhanced productivity of grassland enables higher stocking rates and the rise in methane production offsets soil carbon sequestration. However, the increase in productivity may also reduce land demand (Conant 2010, Terrestrial Carbon Group 2010). Finally, research indicates that it is possible to identify grazing practices which reduce methane production per unit of product (Conant 2010).

4.3.1 Temperate grasslands, savannahs and shrublands

The temperate grasslands, savannahs and shrublands biome covers extensive areas of South America, the USA and Central Asia. The dominant natural vegetation in this biome are largely treeless grasslands locally known e.g. as steppe, prairie or pampa, which in some areas are interspersed with shrubland or form part of a forest-grassland mosaic. Plant growth in this biome is limited by water supply in what is typically a strongly seasonal semi-arid and continental climate. In their natural state, most temperate grasslands support large herds of grazing wildlife species which (along with burrowing soil animals) play an important role in the carbon cycle (Trumper et al. 2009). Field observations suggest that reductions in densities of wild ungulates through overhunting can lead to significant changes in vegetation communities and fire frequency of natural grassland

Area extent of the biome, total carbon stock and mean carbon density

The temperate grasslands, savannahs and shrublands biome covers an area of approximately 15.3 million km^2. The total carbon stock within the biome is estimated at 183.7 Gigatonnes C (Trumper et al. 2009). The mean value of carbon density across the biome is 119.9 t C/ha.

Current land use and conservation status and expected trends

Due to the potentially high productivity of temperate grassland soils, much of their original area has been cleared for agriculture, while some of the remaining extent is used for intensive livestock grazing. It is estimated that more than 95 % of North American grasslands have been lost (SCBD 2010), and conversion rates have also been high in Eastern Europe and parts of the former Soviet Union and China. Overgrazing leading to degradation and soil erosion has been a serious problem across some of the areas used as grazing lands, such as parts of the Patagonian and Chinese steppes. The largest areas of remaining natural steppe vegetation are located in Central Asia.

The coverage of protected areas in the temperate grasslands, savannahs and shrublands biome is lower than anywhere else in the terrestrial realm (i.e. < 5%; Coad et al. 2009).

Pressure for further conversion remains a problem in many regions, and susceptibility to degradation is likely to increase as a consequence of impacts of changes in temperature and precipitation on ecosystem productivity. Increasing water abstraction rates, especially for irrigated agriculture, will regionally add to the problem.

Ecosystem carbon stocks and greenhouse gas balance

Generally, temperate grasslands have low levels of vegetation biomass, but high stocks of soil organic carbon. Grassland plants allocate much of their biomass below ground. Pendall et al. (2011) report an above ground biomass of 3 t/ha and below ground biomass of 7 t/ha for a South Australian native temperate grassland. Fan (2008) analysed field samples and literature values of biomass for 17 types of grassland communities in China. They give average values of total biomass carbon ranging between 21.66 t C/ha for alpine meadows, 17.8 t C/ha for alpine meadow-steppe, 12.57 t C/ha for alpine steppe, 10.58 t C/ha for temperate meadow-steppe, 7.99 t C/ha for temperate steppe, and 3.9 t C/ha for temperate desert-steppe. Amthor et al. (1998) cite an average value of 7.2 t C/ha for the total carbon content of above and below ground biomass in temperate grasslands.

For soil organic carbon, Jobbagy and Jackson (2000) provide an average value for temperate grassland of 191 t C/ha, while Amthor et al. (1998) cite a value of 236 t C/ha for the top 1 m of soil. Amundson (2001) gives 133 t C/ha as an average value for cool temperate steppe soils, and 76 t C/ha for temperate thorn steppe.

As mentioned above, conversion and degradation by overgrazing have a strong negative impact on soil carbon stocks. A drastic example is the heavy topsoil loss after cultivation that occurred in the North American central plains in the first quarter of the twentieth century (Safriel et al. 2005). Xie et al. (2007) conclude that degradation of grassland, particularly on the Qinghai-Tibetan Plateau, has led to the loss of 3.56 gigatonnes of soil organic carbon in China since the early 1980s. He et al. (2011) carried out a grazing experiment with seven stocking rates in temperate grasslands of northern China. They found that aboveground plant biomass decreased logarithmically with increasing stocking rates and ranged from 1.6 t/ha to 0.55 t/ha, while soil carbon stocks decreased linearly. The threshold between carbon sequestration in soils and carbon loss was found to occur at a stocking density of 4.5 sheep/ha. Reducing the stocking rate to 1.5 sheep/ha resulted in a carbon sequestration effect of 0.74 t/ha*a. Steffens et al. (2008, 2009) examined effects of grazing exclusion on different fractions of soil organic carbon in a steppe ecosystem in Northeastern China. Although total soil organic carbon content (and other soil parameters such as porosity and nutrient contents) recovered significantly after more than five years of grazing exclusion, they found that the additional carbon was stored in relatively unstable soil fractions which could be vulnerable to rapid loss under changing conditions, and cautioned against ignoring the distribution of carbon across soil carbon pools when considering sequestration potential.

Knowledge gaps

Information on the influence of different grazing systems on soil carbon in grassland systems is still lacking for many regions, in particular in developing countries. Additional field studies as well as efforts to synthesize knowledge from existing studies are needed. Further research is also needed on the likely impacts of climate warming and altered rainfall regimes on grassland vegetation and soil carbon (Conant 2010, Terrestrial Carbon Group 2010). Better information on the extent and severity of soil degradation at different spatial scales could also help to assess mitigation potential (Safriel et al. 2005).

4.3.2 Tropical and subtropical grasslands, savannahs and shrublands

The tropical and subtropical grasslands, savannahs and shrublands biome is characterised by a wide array of natural vegetation types in which trees and grasses co-dominate in various proportions and compositions, ranging from grasslands where trees are virtually absent to forest-like formations including a grass layer such as the Miombo or Cerrado woodlands. It is considered likely that human influences have decreased the share of woody vegetation in many savannah areas, especially through changes in the fire regime. Fires are a naturally recurring phenomenon in this biome, in particular in the subhumid savannah areas, where vegetation biomass is highest. Similar to the temperate grasslands, savannah ecosystems in their natural state host large herds of big herbivores, whose place is nowadays often taken by domestic animals. Africa contains by far the largest areas of savannah world wide, but significant tracts of savannah are also found in South America and Australia (Grace et al. 2006).

Area extent of the biome, total carbon stock and mean carbon density

The tropical and subtropical grasslands, savannahs and shrublands biome covers an area of approximately 20.8 million km^2. The total carbon stock within the biome is estimated at 285.3 Gigatonnes C (Trumper et al. 2009). The mean value of carbon density across the biome is 137 t C/ha.

Current land use and conservation status and expected trends

Many savannah areas, especially in the subhumid and semiarid zones, have been converted to cropland or are under intensive pastoral use (see Chapter 4.3). According to Safriel et al. (2005), dry subhumid African savannahs north of the equator now represent only 14 % of the original cover and are still retreating. Particularly high rates of conversion are currently reported from woodland savannahs in Brazil and Southern Africa. Cropland and pasture have already replaced nearly half of the Brazilian Cerrado. Between 2005 and 2008, the estimated yearly loss was more than 0.7 % of the original extent. In the Southern African Miombo woodlands, extraction of wood and land clearing for agriculture, as well as uncontrolled bush fire are leading to a continuous decline in natural land cover (SCBD 2010). Ciais et al. (2011) cite rates of deforestation of savannah woodlands of 1.1 per cent per year in Tanzania and 1 per cent per year in Zambia.

Human pressure on the savannah ecosystems is still increasing in other areas as well, and some sources estimate that even on global average more than one percent of savannah extent is lost annually to anthropogenic fires, cattle raising and agricultural activities (Trumper et al. 2009). In Africa and Latin America, there is still a significant share of potentially cultivable land which currently supports savannah ecosystems and might come under conversion pressure in future (Cassman et al. 2005).

Ecosystem carbon stocks and greenhouse gas balance

Above-ground biomass in savannah ecosystems is highly variable, depending on the degree of tree cover. Grace et al. (2006) report above ground carbon stocks ranging from 1.8 t C/ha where trees are absent, to over 30 t C/ha in areas with substantial tree cover. As a result of their study of literature data on savannah carbon stocks around the globe, they calculate mean values of 9.4 t C/ha for above ground biomass carbon, 19 t C/ha for below ground biomass carbon and 174 t C/ha for soil organic carbon. Amthor et al. (1998) cited values for tropical savannah of 29 t C/ha for total biomass carbon and 117 t C/ha for soil organic carbon.

Ciais et al. (2011) describe an example of a Mozambican miombo woodland with a tree biomass carbon content of about 30 t C/ha, and total soil organic carbon stock of about 110 t C/ha. They note that the soil organic matter distribution was not correlated with vegetation carbon stocks and was highly variable, presenting an additional complication for assessments of African carbon stocks. They also highlight the difficulty of monitoring changes in savannah woodlands with remote sensing and the fact that the uneven distribution of above ground biomass leads to a need for large sampling plots. Citing from published literature, they state that soil organic carbon stocks in African savannahs and woodlands range between 30 and 140 t C/ha.

Henry et al. (2009) calculated a mean soil organic carbon content of 31.5 t C/ha for the top 30 cm of soils in the African savannah biome, based on an analysis of four recent global digital soil maps and five soil properties databases. The values reported by Jobbagy and Jackson (2000) and Amundson (2001) are 230 t C/ha and 54 t C/ha, respectively.

As stated above, the main causes of pressure on savannah carbon stocks are agricultural conversion, heavy grazing and increased frequency or intensity of fire. Harvesting of wood biomass for fuel is another driver of carbon loss that can be locally or regionally significant (Prihodko et al. 2011, Kutsch et al. 2011). Zingore et al. (2005) measured long-term changes in soil organic carbon after the clearing of savannah woodlands for agriculture in Zimbabwe, and found losses of more than 20 t C/ha, depending on the soil type. Grace et al. (2006) assume that the loss and degradation of natural savannah vegetation constitutes a flux to the atmosphere that is comparable to that from the loss of tropical rainforests, and conclude that appropriate management could turn most savannah areas into significant carbon sinks, due to their high productivity and past degradation.

The effects of changing fire regimes on the greenhouse gas balance of savannah ecosystems are a subject of ongoing research. Fire events in savannahs can release huge amounts of carbon to the atmosphere (estimated at 0.5–4.2 Gt C per year globally as cited by Trumper et al. 2009), and according to Ciais et al. (2011), the African continent on average contributes 40 % of yearly global fire emissions, mostly from savannah burning.

Under favourable conditions, the carbon lost is mostly regained during the subsequent period of plant regrowth (Grace et al. 2006, Williams et al. 2007), which is why for accounting purposes the carbon balance of savannah fires is often assumed to be neutral (Ciais et al. 2011, Tropical savannahs CRC 2011). However, under a regime of frequent fires with high intensity, ecosystem carbon stocks can be depleted over time (Tropical savannahs CRC 2011). If all fires in the savannah biome could be excluded (which is a highly theoretical scenario), some authors suggest that the global area of closed canopy forest would double, with most of the expansion being in African savannahs. Long-term fire exclusion experiments confirm a trend towards development of closed canopies in the more humid types of savannah (Ciais et al. 2011).

In addition to effects on ecosystem carbon stocks, the burning of standing biomass also leads to loss of nitrogen from the system and emissions of nitrous oxide and methane. One approach that has been proposed to reduce these emissions is strategic early dry season burning, which causes less intensive fires and may also be more beneficial to biodiversity than regimes determined by fire prevention and ensuing low frequency but high intensity fires (Tropical savannahs CRC 2011, Douglass et al. 2011). Strategic burning approaches have been traditionally applied by aboriginal populations in Australia, who maintained a fine-grained spatial pattern of small fires with low on-site recurrence (Safriel et al. 2005). Another option for fire management is the promotion of certain levels of grazing intensity by wild or domestic herbivore species in order to reduce fuel loads.

Knowledge gaps

In addition to remaining questions about the influence of different management methods and fire regimes on ecosystem carbon stocks and emissions of greenhouse gases, Ciais et al. (2011) note that savannahs are not yet well represented in terrestrial ecosystem models, and thus estimates of their contribution to the global carbon cycle are highly uncertain. This is due to a lack of in situ data for model calibration and the development of allometric equations, and to the fact that many models were developed for other regions. For example, most fire modules within global vegetation models are parametrized and tested in boreal and temperate systems. They also mention methane emissions from termite activity in savannahs as a process which is not yet well understood.

Because of the importance of fire in savannah ecosystems, non-greenhouse gas factors such as emission of soot particles and albedo effects are likely to have more of an influence on climate than in some other ecosystems (cp. Williams et al. 2007, Randerson et al. 2006). More studies analysing total net radiative forcing (i.e. the climatic effects of greenhouse gas emissions and sequestration and other ecological processes taken together) are needed to determine their comparative significance.

Trumper et al. (2008) point at the technical and statistical challenges of measuring changes in above and below ground carbon stocks over large areas in drylands with the required accuracy, and conclude that further research is required to demonstrate the feasibility of large area measurement schemes.

4.3.3 Deserts and dry shrublands

Deserts and dry shrublands are found in regions of very low or highly seasonal precipitation, including many parts of Africa, the southern USA and Mexico, parts of Asia and South America, and large areas of Australia. Depending on the rainfall regime and soil conditions, the vegetation consists mainly of slow growing woody shrubs and succulents and/or species capable of highly sporadic growth such as desert grasses.

Area extent of the biome, total carbon stock and mean carbon density

The deserts and dry shrublands biome covers an area of approximately 28.1 million km^2, making it the largest of all terrestrial biomes. The total carbon stock within the biome is estimated at 178 Gigatonnes C (Trumper et al. 2009). The mean value of carbon density across the biome is 63.4 t C/ha.

Current land use and conservation status and expected trends

Although degradation as measured by negative trends in net primary productivity is less common in the desert and dry shrublands biome than in the more humid dryland zones due to lower densities of human settlement, substantial parts are used as extensive grazing land, and serious degradation of vegetation and soils occurs in some areas. Soils in arid regions may be stabilized by biogenic or mineral crusts which are sensitive to trampling by livestock (Safriel et al. 2005). Overharvesting of the slow-growing woody vegetation for firewood frequently leads to degradation of the shrub and tree components. Degradation is often localized around settlements and in locations with higher water availability and therefore higher potential primary productivity, such as along rivers or in areas with surface-near groundwater tables. Groundwater abstraction for irrigation is another possible cause of damage to the natural vegetation.

Ecosystem carbon stocks and greenhouse gas balance

Aboveground biomass in deserts and dry shrublands is dependent on the dominant plant forms but generally low. Fan et al. (2008) provide an average value of 1.17 t C/ha for above ground biomass carbon in grass-dominated temperate deserts in China, and 0.09 t C/ha for grass-dominated alpine deserts. They found average total vegetation carbon densities of 2.46 t C/ha in the temperate and 1.49 t C/ha in the alpine deserts. Amthor et al. (1998) cite an average value of 3.3 t C/ha for total biomass carbon in deserts, semi-deserts and scrubland and 0.3 t C/ha for total biomass carbon in extreme deserts. WBGU (1998) mention an average vegetation carbon density for deserts and semi-deserts of 2 t C/ha. In shrub- or tree-dominated areas, carbon densities can be significantly higher.

Due to low decomposition rates in the arid environment, organic soil carbon stocks can be quite substantial. Henry et al. (2009) give a figure of 15.3 t C/ha for the top 30 cm of African desert and dry shrubland soils. Jobbagy and Jackson (2000), who considered soil profiles up to a depth of 3 m, calculate an average soil organic carbon content of 115 t C/ha. According to WBGU (1998), an average organic carbon content for desert and semi-desert soils would be 42 t C/ha. Amthor et al. (1998) cite values of 80 t C/ha for desert, semi-desert and scrubland soils, and 25 t C/ha for extreme desert soils. Amundson (2001) provides soil organic carbon values ranging from 14 t C/ha for warm deserts, 20 t C/ha for tropical desert bush, 99 t C/ha for cool deserts and 102 t C/ha for boreal deserts. According to Trumper et al. (2009), carbon content estimates for dry shrubland soils can be as high as 270 t C/ha.

Knowledge gaps

In addition to uncertainties about the distribution of soil carbon, there is a lack of data and considerable uncertainty on carbon uptake by deserts (Wohlfahrt et al. 2008, Schlesinger et al. 2009), as well as on the effects of land degradation on desert and dry shrubland carbon stocks.

4.4 Coastal ecosystems

Vegetated marine habitats can contribute to carbon storage not only through the growth and accumulation of plant biomass, but also through the export of organic matter such as parts or dissolved components of leaves and stalks to other systems, as well as through the trapping and burial of organic carbon in their sediments. The fate of exported organic matter is difficult to predict (as is the case with eroded soil organic matter in terrestrial systems), and no reliable estimates exist of its contribution to carbon storage (cp. Bouillon et al. 2008, Bouillon et al. 2009, Duarte and Cebrián 1996, Reed and Brzezinski 2009). Carbon burial rates, on the other hand, can be measured in a comparatively straightforward way although some difficulties exist in discerning between short-term and long-term stores (Kennedy and Björk 2009). In coastal marine ecosystems that have a dense cover of higher plants and occur on soft substrates with limited wave activity, the rate of carbon sequestration in sediments can be substantial (Nellemann et al. 2009, Laffoley and Grimsditch, 2009). The carbon which is buried in these sediments may originate from within the ecosystem or be imported by currents or tidal activity and intercepted as a result of the reduced velocity of water flow in the vegetated area (cp. Kennedy et al. 2010).

The three main types of coastal ecosystems with a potential for sediment carbon accumulation are mangroves, salt marshes and seagrass meadows. It has been estimated that the average carbon burial rates in sediments under these three vegetation types are more than 6000 times greater than the average burial rate in the open ocean, i.e. 1.23 t C/ha and year as compared to 0.00018 t C/ha and year (Nellemann et al. 2009). However, there is a high variability in sequestration rates between sites, as these rates depend on a range of factors including the plant species composition, environmental parameters such as water temperature and depth and exposure to waves and currents, and the location in relation to potential sources of organic matter input. Estimates of the total contribution of vegetated marine habitats to global carbon sequestration are further constrained by large uncertainties in data about their areal extent. Nellemann et al. (2009) cite figures according to which the total yearly carbon burial in mangroves, salt marshes and seagrass ecosystems is likely to be between 0.114 and 0.131 gigatonnes C.

Other coastal systems dominated by macroscopic plants, such as kelp forests, can also show high rates of primary productivity, but their overall contribution to carbon sequestration is difficult to assess and probably significantly lower due to the fact that they occur on rocky substrates exposed to strong waves and currents, so any dead or fragile plant material is washed away quickly (Reed and Brzezinski 2009).

Around the world, coastal marine ecosystems are suffering degradation from a range of pressures, including land- and ocean-based pollution, eutrophication, changes in sediment input via rivers and ocean currents leading to siltation or erosion, direct conversion to uses like aquaculture, extraction of sediment, destructive fishing practices, coastal development for terrestrial or marine transport infrastructure, settlements or coastal protection measures, and land reclamation for building or agriculture. For mangroves, direct harvesting of wood is another major cause of threat. Feedback effects between loss of vegetation and erosion can reinforce degradation processes.

Estimated rates of loss for the three coastal marine ecosystems described in this report (cp. 4.4.1 – 4.4.3), are higher than the conversion rates reported for any terrestrial ecosystem, and some authors fear that they might disappear from most of their remaining distribution range within the next 20 years (Nellemann et al. 2009). Duarte et al. (2005) estimate that the decline of vegetated coastal habitats has already reduced the yearly carbon burial rates in the ocean by about 0.03 Gt C.

It seems very likely that pressures on ecosystems in coastal areas will continue to grow, as human population densities, already higher than in most inland areas, are projected to increase further and recreational use and demand for products from coastal agriculture, aquaculture and fisheries are on the rise. There is also a high certainty that anthropogenic climate change will place additional stress on coastal ecosystems through the combined effects of increasing air and water temperatures, changes in precipitation and ocean currents, and sea level rise (cp. Lovelock & Ellison 2007). According to current projections, the frequency and intensity of storm-related disturbances is also likely to increase (Agardy et al. 2005). The potential impacts on marine communities of ocean acidification as a result of higher concentrations of atmospheric carbon dioxide are a subject of intensive research but still not well understood (cp. Doney et al. 2009).

Efforts to restore natural coastal ecosystems are increasingly undertaken, most notably for mangroves. However, such activities may require substantial initial investment and need to be well planned in order to be effective (Nellemann et al. 2009, Agardy et al. 2005). The potential for carbon sequestration is not yet widely taken into account when locations and methods for restoration activities are chosen. Additional research is needed to improve guidance on best practice for the design of climate-friendly restoration projects (Nellemann et al. 2009, Bouillon et al. 2009).

Box 2: A note on coral reefs:

Judging from the amounts of carbon contained in the calcium carbonate skeletons of reef-building corals, some authors have concluded that coral reefs constitute a major carbon sink. However, this is not in line with the generally accepted scientific understanding of the chemical processes involved in the formation of reef carbonate.

Put very simply, the chemical compounds that play a role in calcification are carbon dioxide (as dissolved in the sea water), its hydrated form carbonic acid (H_2CO_3), the dissociated components of carbonic acid - bicarbonate anions (HCO_3^-), carbonate anions (CO_3^{2-}) and hydrogen ions (H^+) – and calcium cations (Ca^{2+}). The chemical processes linked to calcification are as follows: the formation of calcium carbonate from calcium cations (Ca^{2+}) and bicarbonate anions (HCO_3^-) that are dissolved in the sea water releases hydrogen ions (H^+). The resulting increase in hydrogen ion concentrations pushes the balance between hydrated carbon dioxide, i.e. carbonic acid (H_2CO_3), and its dissociated components, bicarbonate anions (HCO_3^-) and hydrogen ions (H^+), towards formation of the former. This in turn facilitates the release of carbon dioxide from its hydrated form (i.e. carbonic acid, H_2CO_3) to the non-hydrated form (CO_2) and its evasion into the atmosphere.

Thus, the formation of reef carbonate constitutes a small source of atmospheric carbon dioxide. This process could be reversed if the pH of sea water drops beyond a certain level and the carbonate starts to dissolve. Carbonate dissolution would then act as a temporary buffer to further declines in pH. The likelihood that ocean acidification will lead to the crossing of a threshold from net calcification to net dissolution in coral reefs is a matter of scientific debate.

For a fuller discussion of the issues addressed above, see Smith and Gattuso (2009). The main conclusion to draw from this is that coral reefs, while providing many outstanding services to humans including livelihood security and services that are highly relevant for climate change adaptation (cp. Wilkinson 2008), do not generally contribute to climate change mitigation.

4.4.1 Mangroves

Mangroves form the natural vegetation of shallow coastal areas in tropical and subtropical Asia, Africa, Latin America and Australia. It is assumed that about 30 % of the current global extent of mangrove forests are situated on the territories of Indonesia and Australia (Bouillon et al. 2009). Mangroves can tolerate a wide range of salinities and levels of exposure to waves and tides, resulting in the formation of subtypes with different species composition and ecological characteristics. Mangrove areas can be divided into three zones, each with distinctive physical, geomorphological and ecological characteristics. While riverine mangroves receive high levels of freshwater, nutrient and sediment input, fringing mangroves are in direct contact with the open ocean and are a net source of organic matter export. Interior basin mangroves are least exposed to physical impacts and characterized by net sedimentation (Agardy et al. 2005).

Area extent

Recent estimates of the current extent of mangrove ecosystems range between 0.11 and 0.24 million km^2 (Nellemann et al. 2009, see also Giri et al. 2011), with most of the published values tending towards the lower end of this range. Bouillon et al. (2009) considered 0.15 - 0.16 million km^2 the best estimate. Spalding et al. (2010) analysed data for 113 countries and arrived at an overall mangrove cover of 0.15 million km^2. Most recently, Giri et al. (2011), based on their interpretation of Global Land Survey data and data from the Landsat archive, calculated that the total global area of mangroves in the year 2000 was 0.14 million km^2, distributed across 118 countries and territories.

Current land use and conservation status and expected trends

According to some sources, mangroves once occurred along 75 % of all tropical coasts (Bouillon et al. 2009). Historic settlement patterns have led to the development of centres of urbanization in the vicinity of both major coral reefs and extensive mangrove stands. It is estimated that 64 % of the world's mangroves are currently situated within 25 km of urban centres with more than 100,000 inhabitants (Agardy et al. 2005), which indicates a high potential for anthropogenic pressure.

Estimates of the area that has been lost due to overharvesting, degradation and conversion to agriculture, aquaculture or other land uses range from 35 % since the 1940s to over 70 % in total (Bouillon et al. 2009, Agardy et al. 2005). Agardy et al. (2005) state that sufficient data to assess the extent of mangrove decline exist only for about half of the current distribution area, and that in nearly all countries for which multiyear data exist dramatic declines are documented. There is a high regional variation, and some countries in Southeast Asia are reported to have lost almost 90 % of their mangroves (Bouillon et al. 2009). About half of the overall reduction in mangrove area is believed to be a result of conversion to aquaculture (Agardy et al. 2005).

According to SCBD (2010), data provided by the FAO indicates that the annual global rate of loss of mangrove cover has decreased by 45 % over the past 25 years, from 1,850 km^2 per year during the 1980s to 1,020 km^2 per year between 2000 and 2005. However, this moderately positive trend has not been observed in Asia, which is the region holding the largest share of remaining mangroves.

The development of restoration techniques for mangroves is relatively advanced, but current restoration efforts are hardly sufficient to have a significant effect in slowing the decline (Agardy et al. 2005, Spalding et al. 2010)

Ecosystem carbon stocks and greenhouse gas balance

According to Bouillon et al. (2009), the average biomass carbon density of mangroves is on the order of 79 t C/ha, leading to an estimate of global biomass carbon stock of 1.2 gigatonnes C. The authors do not provide corresponding values for sediment carbon content, stating that there is insufficient information on sediment depths and densities. However, they note that the proportional carbon content of sediments varies widely, from less than 0.5 % of the dry weight to nearly 40 %, with a global median value of 2.2 %.

Donato et al. (2011) analysed biomass and soil carbon contents in transects across 25 mangrove sites of the Indo-Pacific region. They found above ground carbon pools with an average of 159 t C/ha and a maximum of 435 t C/ha. Below ground pools including soil organic carbon accounted for 71-98 % of total ecosystem carbon in estuarine sites and 49-90 % in oceanic sites. Average total ecosystem carbon within their sample of sites was 1,023 t C/ha, with estuarine sites showing slightly higher carbon densities than oceanic ones. Kauffman et al. (2011) studied two mangrove areas in Micronesia and found that they contained average ecosystem carbon stocks of 718 t C/ha and 1,062 t C/ha, respectively, with soils containing around 70 % of the total carbon stock.

Murray et al. (2011) state that biomass carbon densities in mangrove forests range between 65 and 153 t C/ha, and that average sediment organic carbon contents in the first meter are around 290 t C/ha for estuarine sites and 490 t C/ha for oceanic ones, adding that estuarine mangroves usually have deeper layers of organic soil than oceanic ones.

According to Nellemann et al. (2009), reported values of carbon burial rates in mangrove sediments range between 0.2 and 6.54 t C/ha and year, with an average value of 1.39 t C/ha and year. Murray et al. (2011) provide an average value for carbon sequestration in mangrove ecosystems of 1.7 t C/ha and year.

Clearing of mangroves can result in a rapid decline of sediment organic carbon stocks, particularly if the sediment is eroded or disturbed (e.g. in the establishment of shrimp farms). Bouillon et al. (2009) cite a study according to which up to 50 % of the sediment carbon can be lost within less than 10 years. Lovelock et al. (2011) found emission rates of around 29 t C/ha during the first year after clearance, with emissions gradually declining to around 8.2 t C/ha and year after 20 years. Donato et al. (2011) estimate that between 112 and 392 t C/ha are released after mangrove clearance, depending in large part on the subsequent land use and how deeply it affects soil carbon. Based on these values, they calculate yearly global emissions from mangrove deforestation of 0.02 to 0.12 gigatonnes of carbon.

In low saline areas, eutrophication can lead to significant emissions of nitrous oxide and methane from mangrove sediments (Allen et al. 2011).

With regard to the potential mitigation benefits of mangrove restoration, Bouillon et al. (2009) state that available information from mangrove plantations and sites undergoing rehabilitation indicates that the productivity of replanted mangrove ecosystems can reach a similar level to that of natural stands. However, restoration requires appropriate planning and design (e.g. choice of species and planting methods that are suited to the site, selection of sites with a high potential for mitigation) and can be problematic if the sediment conditions have changed as a result of the degradation process.

Knowledge gaps

In order to improve the knowledge base for mitigation activities in mangroves, further research is needed into the factors that determine the distribution and accumulation of sediment organic carbon at various scales (cp. Alongi 2011), as well as into the impact of different kinds of disturbance (including eutrophication) on fluxes of carbon and nitrogen between mangroves, adjacent ecosystems and the atmosphere. Addressing the uncertainties around the current area extent, geographic distribution and rate of decline of mangroves will require the use of consistent protocols for the analysis of data from remote sensing in combination with ground-truthing and review by local experts (Fitzgerald et al. 2011). Further work is also needed on restoration methods that are efficient for achieving mitigation as well as other benefits.

4.4.2 Tidal saltmarshes

Salt marshes occur on sheltered sections of marine and estuarine coastlines in the intertidal zone, which is regularly exposed to the atmosphere. Although their distribution ranges from the sub-Arctic to the tropics, they are most common in temperate climates (Chmura 2009).

Area extent

Laffoley and Grimsditch (2009) point out that there are particularly large uncertainties around the total area extent of tidal salt marshes, and cite a figure of 0.22 million km^2. Nellemann et al. (2009) state that the global coverage of tidal salt marshes is likely to be on the order of 0.4 million km^2.

Current land use and conservation status and expected trends

Between 25 % (Nellemann et al. 2009) and 50 % (Crooks et al. 2011) of the area originally covered by salt marshes are believed to have been lost, with current rates of loss at about 1 to 2 % per year. The main processes leading to salt marsh destruction are land reclamation for agriculture, dredging and coastal development. Pollution and eutrophication through agricultural runoff and sewage water can also contribute to degradation. In the context of climate change, salt marshes may be particularly vulnerable to 'coastal squeeze', i.e. the loss of area due to sea level rise without the possibility to shift inland because of coastal infrastructure.

Efforts to restore salt marshes have been carried out to a limited extent, mostly in Europe and the USA. In recent years, the possibility to use salt marshes as a 'soft' measure of coastal protection has received increasing attention. Up to a certain rate of sea level rise, salt marshes can adapt through accretion of sediment. They can also form a buffer zone protecting coastal infrastructure against damage from storm surges. The restoration of drained saltmarshes can slow or reverse the process of land subsidence caused by mineralisation of organic matter under aerobic conditions, and thus combine carbon savings with increased resilience to flooding. For case studies on the carbon emissions and land subsidence that occurred as a consequence of drainage in a number of major river deltas around the world, and an assessment of the potential for restoration, see Crooks et al. (2011). Reduced sediment inputs as a result of river regulation or coastal protection measures can decrease the ability of salt marshes to adapt to rising sea levels.

Ecosystem carbon stocks and greenhouse gas balance

Chmura (2009) cites a global estimate of 0.43 gigatonnes of carbon for the organic carbon content of the upper 50 cm of tidal salt marsh soils, stating that this is likely to be an underestimation as salt marsh soils usually contain significant amounts of carbon up to several meters of depth and the applied estimate of total global area of salt marshes is likely to be too low. Murray et al. (2011) cite average values for biomass carbon density in salt marshes of around 3-16 t C/ha, and for soil organic carbon content in the first meter of salt marsh sediment of 250 t C/ha.

According to Nellemann et al. (2009), reported values of carbon burial rates in tidal salt marshes range between 0.18 and 17.3 t C/ha and year, with an average value of 1.51 t C/ha and year. The significant variation is partly due to differences in species composition. Chmura (2009) provides a figure of 2.1 t C/ha and year for long term carbon accumulation in the sediment, and points out that a beneficial feature of salt marshes is their low potential for the production of methane, due to the presence of sulphates in the soil which inhibit the activity of methane-producing microbes. In tidal marshes characterized by brackish conditions (as they occur in estuaries), rates of methane production are often significantly higher and can be quite variable, which constitutes a challenge for the estimation of greenhouse gas balances (Poffenbarger et al. 2011).

Following drainage and conversion, a significant part of the carbon stored in tidal salt marsh soils can be released as carbon dioxide (Murray et al. 2011).

Enrichment of coastal wetlands with nitrogen may increase the release of nitrous oxide (Chmura 2009). The global warming potential of nitrous oxide over a period of 100 years is considered by the IPCC to be about 298 times higher than that of carbon dioxide (Forster et al. 2007).

Knowledge gaps

In addition to more precise information on the geographic distribution, rates of loss and degradation, total organic carbon stocks and sequestration potential of tidal saltmarshes, an improved understanding is needed of the likely impacts of sea level rise on saltmarsh ecosystems, and of their potential to deliver simultaneous benefits for climate change mitigation and adaptation. It is likely that critical thresholds for the capacity of saltmarshes to adapt to sea level rise will depend not only on the extent, but also on the speed of the change in sea levels.

4.4.3 Seagrass meadows

Seagrass meadows are found in shallow waters of all continents except the Antarctic on soft-bottom substrates. They are made up of a range of different species from various genera. Seagrasses form highly persistent roots and rhizomes, leading to large stores of underground biomass.

Area extent

The area extent of seagrass meadows is not well known. While the documented area is 0.12 million km^2, upper estimates based on suitable distribution areas amount to 0.6 million km^2 (Green & Short 2003). Particularly large areas of potential seagrass habitat with limited data availability are found in South East Asia (Nellemann et al. 2009). Nellemann et al. (2009) cite 0.33 million km^2 as the most likely figure, taking into account recently observed declines.

Current use and conservation status and expected trends

Seagrasses require high light intensities and clear water, and are particularly susceptible to eutrophication and siltation. Other anthropogenic pressures include dredging, coastal development, destructive fishing practices, pollution, and shifts in community composition as a result of overfishing of fish and invertebrates that feed on algae. Major losses of seagrass habitat have been reported from the Mediterranean, Florida Bay and Australia (Agardy et al. 2005).

According to Waycott et al. (2009), the global extent of seagrass meadows has already been reduced by around one third, with an accelerating trend. The authors estimate that the rate of area loss was still at 0.9 % per year in the 1970s and has increased to more than 7 % per year since 2000. This would make seagrass meadows one of the most threatened ecosystems in the world. Kennedy and Björk (2009) even cite figures according to which the areal extent of seagrasses has been reduced by 50 % over a period of about 15 years. It is expected that present losses will accelerate in particular in Southeast Asia and the Caribbean (SCBD 2010).

Restoration of seagrass meadows is technically difficult and costly and is currently not practised on a significant scale (Nellemann et al. 2009). Reducing pressure on still existing meadows seems to be a more practicable option.

Ecosystem carbon stocks and greenhouse gas balance

Kennedy and Björk (2009) cite estimates for average biomass carbon density in seagrass meadows of around 1.84 t C/ha, and for soil carbon of around 70 t C/ha, noting that in the most productive types of seagrass beds soil carbon stocks can be as high as 1,600 t C/ha. Using an intermediate estimate of seagrass distribution area, this would yield a value for total global carbon stock in seagrass meadows of 2.16 gigatonnes C. Murray et al. (2011) cite average values of 0.1-5 t C/ha for biomass carbon and 136 t C/ha for soil organic carbon content in the first meter of seagrass meadow sediments.

Knowledge on carbon accumulation in seagrass meadows is limited, and the main part of the available data originates from the Mediterranean Sea (Kennedy and Björk 2009). The seagrass species *Posidonia oceanica*, which is endemic to the Mediterranean, is capable of accumulating underground root mattes of several meters' depth and is considered to be the most effective species in terms of long term carbon storage. It is not yet clear whether and which other species make similarly high contributions to carbon burial.

According to Nellemann et al. (2009), reported values of long-term carbon burial rates in seagrass meadows range between 0.56 and 1.82 t C/ha and year, with an average value of 0.83 t C/ha and year. Kennedy and Björk (2009) calculate global rates of carbon sequestration in seagrass meadows of 0.027 – 0.04 gigatonnes per year, while the calculations provided by Kennedy et al. (2010) lead to a global estimate of carbon burial in seagrass meadows between 0.05 and 0.11 gigatonnes per year.

Management activities that reduce nutrient loads to coastal waters can allow for a recovery of degraded seagrass beds. However, the potential effects of such activities on carbon sequestration are difficult to estimate.

Knowledge gaps

As indicated above, more accurate information on the distribution and conservation status of seagrass beds especially in South East Asia, and on the influence of species composition on ecosystem carbon stocks and sequestration rates, are among the most pressing research needs with regard to the potential of seagrass ecosystems for climate change mitigation (cp. Duarte et al. 2011). Ongoing research into the biogeography of seagrass species and the capacities of different species for carbon burial may help to significantly improve estimates of the total global carbon stock and sequestration potential of seagrass beds. More information is also needed on rates of carbon loss from degraded seagrass beds and on effective methods for seagrass restoration, including the identification of suitable areas.

5 Analysis of possible priorities for action

Many criteria can be used to rank options for ecosystem based mitigation activities, both in terms of the ecosystems to be addressed and the activities to be implemented. With regard to the potential benefits for climate change mitigation from individual projects, the possible emission and sequestration rates per hectare come to mind as most directly relevant:

- Information on rates of sequestration in natural or sustainably managed ecosystems, as well as on possible emissions through their degradation or conversion, can be used to estimate the potential benefits from approaches to reduce pressures acting on these ecosystems. It is worth noting that mitigation benefits in the form of avoided emissions may be high even where sequestration rates are low.
- Information on possible rates of sequestration during the recovery or rehabilitation of degraded ecosystems can be used to estimate the potential benefits from approaches to restore ecosystem carbon stocks, and the time scales over which these benefits could be realized.

However, other types of information should be considered as well:

- Scenarios and projections related to the future development of pressures (including land use as well as climate change) on ecosystem carbon stocks and greenhouse gas balances are important in order to estimate the actual mitigation benefits of interventions as compared to business as usual. They can also help to assess the long-term viability of achieved results.
- The potential off-site impacts and indirect effects of different types of measures and activities need to be taken into account, taking a systems perspective in order to assess the risk of displacement/leakage effects and other unintended outcomes, and the net benefits that would be achieved. For a discussion of some of the complexities surrounding predictions of the net impacts of land use-related policies, see Angelsen (2010).
- The nature and extent of possible contributions to the achievement of other environmental or socio-economic goals, such as biodiversity conservation, food security, poverty reduction and climate change adaptation, are likely to be of importance both for the funding institutions and for stakeholders in the project area and at the national level. This links with the wider question of political acceptability.
- Assessments of the practical feasibility of interventions should be carried out, taking into account technical, socio-economic and political aspects. Implementation and opportunity costs will be another key determinant of the likelihood of success.
- The quality of the knowledge base and available implementation and accounting methodologies for a specific activity will determine whether it is appropriate for one-off interventions, or whether it could most usefully be addressed as part of a comprehensive pilot programme including a scientific component.
- The total area extent of the ecosystem type and the relative importance of the pressures to be addressed can provide an indication of the potential for upscaling lessons learned from projects and for applying the developed methods and tools more widely.

A preliminary assessment of options for action with regard to the ecosystem types considered in this report, taking into account the above considerations, is given as follows.

Tundra

Although the high-end estimates of the carbon density of tundra ecosystems are comparable to carbon densities in forest ecosystems (estimates for which range broadly between 150 and 450 t C/ha, depending on forest type; Trumper et al. 2009), the main threats to tundra carbon stocks are related to climate change. There seem to be no major opportunities for beneficial interventions other than climate change mitigation more broadly.

Temperate grasslands, savannahs and shrublands

The reported estimates of the carbon density of ecosystems in the temperate grasslands, savannahs and shrublands biome are comparable to the lower range of estimates for carbon densities in forest.

Due to the extent of past degradation, projects aimed at rehabilitating degraded landscapes could have a high potential for carbon sequestration as well as for socio-economic benefits and climate change adaptation, incurring very low opportunity costs and avoiding leakage risks. However, required initial investments for implementation are likely to be high and biodiversity benefits may be moderate unless restoration activities can take pressure off other valuable habitats. Depending on the causes of degradation, rehabilitation may need to involve restoration of vegetative cover as well as reversal of changes in hydrological regimes.

Projects improving the management of over-utilized grazing lands could be highly cost-efficient by combining increased productivity with climate change mitigation benefits and increased resilience to climate change (cp. Conant 2010).

For some temperate grassland areas, improved fire management may also offer opportunities for climate change mitigation, especially where fire regimes have been changed as a direct or indirect consequence of human activity (e.g. increasing frequency of fires due to reduced densities of wild grazing animals).

Projects aimed at avoiding further conversion of natural habitats could yield high benefits in terms of avoided emissions and biodiversity conservation where appropriate, and are likely to require low public investment. Associated socio-economic impacts and risks of displacement of activities would need to be assessed. Projects could build on emerging accounting methodologies.

Conversion of temperate grasslands to tree plantations has been discussed as a further ecosystem-based mitigation option. However, such projects are unlikely to produce biodiversity benefits and location, design and management regimes need to be chosen carefully in order to avoid negative ecological and socio-economic impacts.

Tropical and subtropical grasslands, savannahs and shrublands

Most estimates of the carbon density of ecosystems in the tropical and subtropical grasslands, savannahs and shrublands biome are slightly lower than the values reported for forests. At the same time, due to currently high rates of degradation and conversion, emissions from unsustainable land use activities in this biome may equal emissions from tropical deforestation (cp. Grace et al. 2006).

Interventions aiming to achieve sustainable land use patterns will face major challenges in this biome due to high population pressure and often low institutional capacities for environmental management, but could yield significant socio-economic, adaptation and biodiversity benefits if they succeed.

Due to the fact that pastoral systems in savannah areas are often not supported by clear property rights, attempts to create financial incentives related to carbon sequestration through improved grazing regimes are likely to be difficult and innovative solutions may need to be found (Conant 2010).

Projects aiming to improve fire management could combine socio-economic and environmental benefits, incur relatively low cost and build on emerging accounting methodologies.

Projects focussing on the maintenance or restoration of the tree component in savannah woodlands can earn carbon credits by building on existing methodologies for reducing emissions from deforestation and enhancing forest carbon stocks. On the downside, the risk of negative impacts through displacement of pressures from forest areas as a consequence of measures to reduce deforestation and forest degradation is particularly high in savannah ecosystems, and REDD+ initiatives should address this.

With regard to afforestation of savannahs (i.e. planting trees in a density that would not occur without human intervention), the same ecological and socio-economic considerations need to be taken into account as for afforestation of temperate grasslands.

Deserts and dry shrublands

Average carbon densities in this biome are lower than for any other area. However, variability among different types of desert and dry shrubland ecosystems is high, and some dry shrubland systems in particular come close to the carbon stock values for dry forest. Addressing degradation of such systems could be a worthwhile activity providing substantial climate change mitigation, biodiversity and socio-economic benefits. Due to low rates of productivity, the potential for restoration projects to contribute to carbon sequestration is likely to be relatively small.

Mangroves

Reported carbon stocks, sequestration capacities and potential emissions from conversion of mangrove ecosystems are higher than those estimated for terrestrial forests, and high rates of decline imply that successful initiatives for conservation and restoration could achieve significant mitigation benefits. At the same time, mangroves provide a wide range of livelihood benefits and other environmental services including coastal protection and disaster prevention, timber and non-timber forest products, as well as habitat and nursery functions for commercially relevant fish species and threatened biodiversity. Depending on environmental conditions, rates of sediment accretion in mangrove stands can be high, contributing to the potential for adaptation to sea level rise (cp. McKee 2010). Appropriate management of mangroves can further enhance their resilience to climate change (cp. McLeod & Salm 2006).

Projects aimed at maintaining or rehabilitating mangrove ecosystems have been shown in a number of case studies to provide net economic gains to society, even though they may restrict the economic opportunities of individual stakeholders (cp. Spalding et al. 2010, Millennium Ecosystem Assessment 2005, TEEB 2009). Because of the importance of mangrove ecosystem goods and services for local populations, supporting sustainable livelihood options should be a central component of interventions.

Activities can be linked with REDD+ initiatives and build on emerging specific accounting standards. A review of methods to quantify impacts of human activity on greenhouse gas emissions in wetlands, including tidally influenced wetlands like mangrove, saltmarsh, seagrass and tidal freshwater systems, is currently being carried out by the IPCC and is expected to be concluded by 2013.

Tidal saltmarshes

Reported carbon sequestration rates in tidal saltmarshes are comparable to those of mangroves, and estimated rates of loss are giving similar reason for concern. Projects combining climate change adaptation and mitigation in tidal saltmarshes could bring significant advantages where they are realistic given current coastal settlement and infrastructure patterns. Due to the fact that most tidal saltmarshes are situated in developed countries, the possibility for synergies with poverty reduction is low. Activities can build on emerging accounting standards (see above).

Seagrass meadows

Seagrass meadows have important ecological functions (including forage, habitat and nursery functions for fish, marine mammals and reptiles, molluscs and other species, sediment stabilization, and buffering of coral reefs against siltation and pollution) that are estimated to deliver large economic benefits to society (cp. Duarte et al. 2008), and appear to be highly threatened. However, due to the extent of uncertainty with regard to their carbon contents and storage rates, it seems too early to make a conclusive assessment of their mitigation potential. Also, projects trying to address climate change by conserving or restoring seagrass ecosystems will face large difficulties related to monitoring as well as to demonstrating causality, as the pressures acting on seagrass meadows mostly originate from land-based activities which can take place at a large distance from the area of impact. Where such projects are undertaken, they should have a strong research component.

Efforts to restore seagrass ecosystems should be concentrated in areas where the root causes for decline (e.g. eutrophication and sedimentation) can be addressed, as they are unlikely to succeed otherwise.

6 Conclusion

As discussed in this paper, with few exceptions most ecosystem types have the potential to serve as a basis for ecosystem-based mitigation approaches, due to the fact that they hold significant carbon stocks and are suffering from a variety of anthropogenic pressures that lead to the depletion of these stocks. This potential should be used more widely, and measures for the conservation, restoration and sustainable use of ecosystems and their carbon stocks should be implemented in a way that provides multiple benefits for climate change mitigation, adaptation and other social and environmental goals.

Challenges are likely to include the reconciliation of competing demands for land in order to arrive at management regimes that are both sustainable and acceptable to stakeholders, and the development of knowledge and methodological approaches in order to plan for, measure and verify the success of interventions.

With regard to remaining gaps in knowledge, there are good reasons to invest in the improvement of data on carbon stocks and greenhouse gas fluxes for understudied regions, ecosystems, carbon pools (e.g. soil carbon), greenhouse gases or activities, both through further field studies and through desk reviews of existing literature, as well as in the development of simple accounting tools and methodologies, including regionally appropriate default values, indicator-based approaches and computer-based modelling.

More information is also needed on the likely impacts of climate change and associated phenomena, such as sea level rise and elevated carbon dioxide concentrations, on ecosystem carbon stocks.

Finally, the potential synergies and trade-offs between the climate-related benefits and other positive environmental and social outcomes from different kinds of ecosystem-based mitigation measures should be investigated more closely.

Pilot projects with appropriate scientific support could help to test and demonstrate the feasibility of new approaches and at the same time enhance the knowledge base on the climate relevance of ecosystems other than forests and peatlands, and on best practices for strengthening their role in climate change mitigation. In the long run, such projects could pave the way for the development of results-based incentive systems for a broad portfolio of ecosystem-based mitigation activities.

Where suitable accounting methodologies are already available, existing and/or emerging mechanisms that can (and should) be used to support ecosystem-based mitigation activities beyond forests and peatlands are the development of carbon credits for the voluntary market, the development of Nationally Appropriate Mitigation Actions (NAMAs) in developing countries, and direct support to climate change mitigation from governments, environmental funds and organizations.

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Annex: Recommendations of the workshop "The climate relevance of ecosystems beyond forests and peatlands", 26 - 29 October 2011

The German Federal Agency for Nature Conservation and the UNEP World Conservation Monitoring Centre convened a workshop at the International Academy for Nature Conservation Isle of Vilm to review and explore the climate change mitigation potential of ecosystems other than terrestrial forests and peatlands, and identify knowledge gaps and research needs.

32 participants from science, national conservation authorities, the European Commission, UNDP, NGOs and consultancies and an observer from UNFCCC participated in the workshop. UNEP-WCMC had produced a draft background report that reviews relevant literature and was discussed during the workshop.

This discussion is in line with interest expressed by some Parties within the UNFCCC process in the potential of various types of ecosystems for climate change mitigation. In the CBD process, making use of the mitigation potential of a range of ecosystems has been supported by several COP decisions and technical reports.

The presentations and discussions at the workshop focussed particularly on temperate and tropical grasslands and savannahs, deserts and dry shrublands, mangroves, seagrass meadows and tidal marshes.

The participants came up with the following conclusions and recommendations:

A. General conclusions

All ecosystems can contribute to climate change mitigation, and some have a higher potential than has previously been recognized, due in part to the often overlooked importance of soil carbon. The full potential of ecosystem-based climate change mitigation beyond forests and peatlands should be realised, including the opportunities for the multiple benefits to society which can be provided by such ecosystem-based approaches.

Considering the soil carbon pools and their stabilisation and sequestration capacities is important for any mitigation activities in temperate and tropical grasslands and savannahs, deserts and dry shrublands and mangroves, seagrass meadows and tidal marshes.

Maintenance of intact ecosystems, ecosystem restoration and sustainable forms of land management are all relevant mitigation strategies. The most appropriate strategy for any intervention should be identified depending on the context taking into account i.a. the following factors: mitigation potential as compared to business as usual over relevant time scales, availability of suitable areas, risk of leakage with regard to carbon and biodiversity, cost effectiveness incl. opportunity and implementation costs, technical feasibility, availability of information and measuring tools, and potential for multiple benefits (e.g. biodiversity conservation, poverty reduction, food security, water supply and purification, climate change adaptation).

The Ecosystem Approach of the CBD[8] is a useful tool when designing ecosystem-based mitigation strategies and activities, also with regard to participation of local communities.

Ecosystem-based approaches to mitigation in ecosystems other than forests and peatlands can at present be funded through the voluntary carbon market, governments, specific programmes such as the International Climate Initiative (ICI) and donors including environmental funds and NGOs. For developing countries, the development of Nationally Appropriate Mitigation Actions (NAMAs) using ecosystem-based approaches could emerge as one option to support and promote such mitigation measures.

For those ecosystems with relatively low climate change mitigation potential but high potential for multiple benefits, support from governments and donors plays an important role and should be encouraged.

Finding ways to reward the provision of multiple benefits can provide key additional incentives for mitigation approaches based on all types of ecosystems. Such additional incentives can enhance the uptake of ecosystem-based mitigation approaches even for ecosystems with very high mitigation potential due to their carbon stocks and sequestration capacity, such as some coastal marine systems.

B. Recommendations to Policy-Makers

- Enabling legal and policy frameworks for ecosystem-based mitigation need to be adopted in-country (e.g. regulations for land use planning) – the Rio Conventions (UNFCCC, CBD, UNCCD) can help promote this process.
- National implementation of UNFCCC, CBD and UNCCD should take place in a more integrated manner in order to promote multiple benefits and increase synergies and efficient use of resources. In addition, application of ecosystem-based approaches contributing to both climate change mitigation and adaptation should be enhanced.
- Mechanisms to fund ecosystem-based mitigation measures should be identified, developed and implemented.
- Guidance on how to address ecosystem-based mitigation as possible part of NAMAs should be developed. Assistance should be provided to countries that wish to develop such measures.
- Payment for Ecosystem Services (PES) schemes should be applied to provide incentives to institutions, land managers etc. for sustainable land management.
- In order to facilitate enhanced generation and use of voluntary carbon offsets, the development of accounting standards for mitigation activities and/or ecosystem types which are not covered by already existing standards should be supported, building on the existing standards.

[8] see http://www.cbd.int/ecosystem/

- The special management needs of wooded ecosystems not traditionally seen as forests (i.e. ecosystems with sparse tree cover) should be taken into account in existing and planned funding mechanisms, and the potential and risks of REDD+ and CDM for the conservation of savannahs and mangrove forests and their carbon stocks should be considered.
- Avoided conversion of terrestrial and coastal marine ecosystems, beyond those covered by REDD and LULUCF, should be addressed by UNFCCC, UNCCD and CBD.
- The downstream impacts of land based pollution and degradation on coastal ecosystem-based climate change mitigation should be recognized.

C. Recommendations to Science

- Improve global mapping of current and potential carbon stocks and fluxes.
- Enhance knowledge on greenhouse gas sequestration and emission rates for available management options across all types of ecosystems.
- Make available knowledge more easily accessible, also for decision-makers, donors, project developers and conservation practitioners.
- Carry out further research on soil carbon stocks, sequestration potentials and stabilisation capacities. Soil carbon stocks in general, their spatial heterogeneity and their dynamics including the temporal aspect need much greater attention. Internationally standardised methods need to be developed for measuring soil carbon across all types of ecosystems.
- Enhance the process-level understanding of dynamics related to all major greenhouse gases and carry out Integrated Net Radiative Forcing analysis (not just GHGs) for important processes and land use options.
- For temperate grasslands and savannahs, carry out research into the consequences of fire management and afforestation for mitigation, land use and biodiversity.
- Pay attention to the particular knowledge gaps for coastal ecosystems with regard to monitoring of trends, mapping of current and potential carbon stocks and fluxes, identification of drivers leading to loss and degradation of carbon stocks, and development of standardized methods for estimation of carbon stocks, sequestration and emissions.
- Develop feasible accounting tools for all types of ecosystems, carbon pools and greenhouse gases, including regionally appropriate default values, indicator-based approaches and computer-based modelling.
- Carry out further research on the likely impacts of climate change and associated phenomena, such as sea level rise and elevated carbon dioxide concentrations, on ecosystem carbon stocks, paying attention to the issue of critical thresholds.
- Improve the representation of non-forest ecosystems in process-oriented models on carbon stock changes and their interaction with climate change.
- Carry out further research into impacts of man-made changes of hydrology on carbon stocks and interactions with climate change.
- Develop and calibrate models to make sensitivity analyses of how ecosystem feedbacks to climate are affected by interactions with ecosystem-based mitigation measures.

- Carry out further research into the fate of carbon transported by erosion processes.
- Intensify research on management options that link climate change mitigation with multiple benefits, including for adaptation.
- Enhance knowledge on the relationship between soil carbon stocks and agronomic/biomass productivity of different land use and management systems.
- Assess direct and indirect impacts of policies related to production and use of biofuels, including impacts on soil carbon stocks.
- Develop tools for decision-makers to assist with the prioritisation of options for action, including through assessment of project feasibility and potential for multiple benefits, opportunity cost mapping and scenario mapping.
- Improve models to identify suitable areas for restoration actions.

D. Recommendations to Donors and other Actors in Project Funding and Project Development

- Donors and other actors are encouraged to actively promote new ecosystem-based mitigation approaches, and test and demonstrate their feasibility, i.a. through pilot projects with appropriate scientific support; such projects should at the same time enhance the knowledge base on the climate relevance of ecosystems other than forests and peatlands, and on best practices for strengthening their role in climate change mitigation. Multiple benefits should be addressed in both project design and scientific monitoring and evaluation.
- Support and capacities for the maintenance and enhancement of climate change mitigation benefits of non-forest ecosystems should be increased through communication of project results, capacity building, awareness raising and environmental education.
- Innovative funding instruments should be developed, tested and used.