Rattan Lal · Klaus Lorenz
Reinhard F. Hüttl · Bernd Uwe Schneider
Joachim von Braun *Editors*

Ecosystem Services and Carbon Sequestration in the Biosphere

IASS POTSDAM

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Foreword

Life on Earth and its inorganic environment forms a self-regulating complex system maintaining habitable conditions (*Gaia* hypothesis; Lovelock 1979). It is now well recognized that Earth's ecosystems provide many goods and services on which planetary life, including human society, depends. Goods provided to humans include food, feed, fiber, fuel, pharmaceuticals, and wildlife. Ecosystem services (ESs) essential to human society include maintenance of hydrologic cycles, denaturing of pollutants and cleansing (filtering) of water and air, regulating climate and weather, storing and cycling of nutrients, providing habitat, and spiritual/cultural, and aesthetical settings. However, human activities are increasingly acting as global geological forces by shaping the "Anthropocene" with significant footprint on global ecosystems and the goods and services they provide. For example, through expansion of agricultural and urban ecosystems along with combustion of biomass and fossil fuel, humans have drastically increased the atmospheric concentration of several greenhouse gases [e.g., carbon dioxide (CO_2)] attendant by global climate change.

Between 1959 and 2010, about 60 Pg of carbon (C) has been emitted to the atmosphere from land use and 290 Pg of C from fossil fuels (Ballantyne et al. 2012). About half of the anthropogenic C emissions were absorbed by C sinks in the oceans and in the terrestrial biosphere. To slow down or even reverse the increase in the atmospheric concentration of CO_2, options must be explored on how to transfer C from the atmosphere into sinks with long residence time in the oceans and in the terrestrial biosphere in a sustainable manner. Global forests are important terrestrial C sinks, but the only long-lasting C gain in terrestrial ecosystems occurs through the formation and stabilization of soil organic carbon (SOC).

The level of SOC within a particular soil can have profound influences on the capacity of the soil to produce food, feed, fiber, and fuel. Thus, by enhancing SOC sequestration and increasing SOC stocks, biological, chemical, and physical quality of the soil can be improved, which in turn can improve ESs (Franzluebbers 2010; Lal 2009). Enhancing the C sink in the soil for improving soil quality and other services such as climate change adaptation and mitigation must be assessed regarding its tradeoffs to other ecosystem goods and ESs. Attempts to enhance provisioning of a specific ES can particularly lead to several disservices or tradeoffs. For example, among disservices caused by intensification of agroecosystems are accelerated erosion, non-point source pollution, and loss of biodiversity. Thus, it is of crucial importance to avoid one-dimensional or "silo" approaches towards SOC. The general goal may be "sustainable intensification" to enhance productivity while maintaining and improving quality of soil/land and the ESs that they provide.

In a companion book *Recarbonization of the Biosphere*, based on a workshop held in March 2011 at the Institute for Advanced Sustainability Studies (IASS) in Potsdam, Germany, it was concluded that drained peat lands, soils degraded by erosion and salinization, agriculturally marginal lands, tropical rainforests and acid savanna soils, and urban ecosystems have a high priority for a positive impact towards recarbonization. The strategy of C sequestration in the terrestrial biosphere may have numerous co-benefits by generating ESs such as enhancing global food security and biodiversity, providing significant quantities of clean water, and adaptation and mitigation to climate change. However, effects of recarbonizing priority ecosystems and soils must be critically assessed towards their impact on ecosystems' goods and services before land use and soil management recommendations with less tradeoffs can be identified. Thus, to further elaborate on the consequences of recarbonization of the biosphere for ESs, about 20 participants from Australia, Austria, Brazil, China, Egypt, Germany, Kenya, South Africa, Thailand, UK, USA along with policy makers met at the IASS for the workshop "Carbon Sequestration and Ecosystem Services" from 27–28 October 2011.

With regard to provisioning ESs, it was concluded that the effects of recarbonization are generally positive except for water yield. Best management practices (BMPs) for recarbonization are in place to learn from, and subsidies and other policy instruments may be used to implement the practices on a case-by-case basis. The distribution of C in the terrestrial biosphere was identified as primary issue for the integrity of regulatory ESs and, thus, as a focus for further research. This can be assessed at places where the regulatory ESs are degraded and, with respect to C, where there is too little C (e.g., dry tropics, arid and semiarid subtropics, mountain areas), too much C (e.g., methane hydrates), and C stocks excessively vulnerable to disturbance (e.g., permafrost, peatlands, forest ecosystems). Activities at the local, regional, and global scales need effective policy interventions. However, the global community is currently not able to collectively agree on the management of the global commons in the Anthropocene and, thus, does not provide sufficient support and guidance for work on the ground. Nevertheless, waiting for a fully fledged global agreement is not an option either, and existing examples of favorable and less favorable practices should be used to learn from. This includes the evaluation of site-specific practices related to cultural ESs. However, cultural values of local communities must be thoroughly understood by understanding the human dimensions. Further, tax reductions for sustainable land use practices and certification of goods and products were discussed as incentives to maximize benefits from C sequestration and cultural ESs. Education was identified as key tool to raise awareness and enhance responsibility.

This volume is based on the papers presented at the workshop. The organization of the workshop and publication of the volume were made possible by the staff of IASS in cooperation with Dr. N. Lorenz. The volume is edited by an inter-disciplinary team of scientists comprising of Drs. R. Lal, K. Lorenz, R. F. Hüttl, B. U. Schneider, and J. von Braun. Special thanks are due to all authors for their contributions and willingness to share the knowledge and expertise with others. The efforts of all others who contributed to publishing this volume in a timely manner are greatly appreciated.

Executive Director Klaus Töpfer
IASS, Potsdam, Germany
1st November 2012

Reference

Lovelock J (1979) Gaia: a new look at life on earth. Oxford University Press, Oxford, 176 pp

Contents

Chapter 1
Societal Dependence on Soil's Ecosystem Services

Rattan Lal, Klaus Lorenz, Reinhard F. Hüttl, Bernd Uwe Schneider, and Joachim von Braun

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Abstract Through its natural capital, soil generates numerous ecosystem services for ecological functions and human wellbeing. These include provisional (feed, food, fiber, fuel, raw material), life support (cleansing, recycling) and cultural (aesthetical, intellectual, spiritual) services. Ecosystem services such as carbon (C) sequestration are generated through a close interaction of the pedosphere with atmosphere, hydrosphere, biosphere and lithosphere. Land misuse and soil mismanagement can degrade soil quality, and either reduces quantity and quality of ecosystem services or leads to disservices and creates large ecological footprints. By increasing the soil organic carbon (SOC) stock, soil biological, chemical and physical quality can be improved which in turn improves ecosystem services. There exist relationships among multiple ecosystem services, increase in one can decrease the other through trade-offs. Payments for ecosystem services, based on rational and objective criteria can minimize risks of overshoot of incentives for enhancing ecosystem services and promote sustainable use of finite and often fragile natural resources. Transdisciplinary collaborations including collaborations between scientists and extra-scientific actors and means of interdisciplinary collaboration are required to tackle the complexity of social-ecological issues associated with soil's ecosystem services.

Keywords Climate change • Food security • Soil security • Water security • Soil processes • Soil degradation • Desertification • Drought • Payments for ecosystem services

Abbreviations

EFs	Ecosystem functions
ESs	Ecosystem services
GHGs	Greenhouse gases
SOC	Soil organic carbon
SOM	Soil organic matter
RMPs	Recommended management practices
WUE	Water use efficiency
NUE	Nutrient use efficiency
ERD	Effective rooting depth
SQI	Soil quality index
PES	Payments for ecosystem services

1.1 Introduction

Ecosystem services (ESs) refer to attributes and processes through which natural and managed ecosystems sustain ecosystem functions (EFs), and create environments so that planetary life can thrive and flourish (Fig. 1.1) (de Groot 1987, 1992;

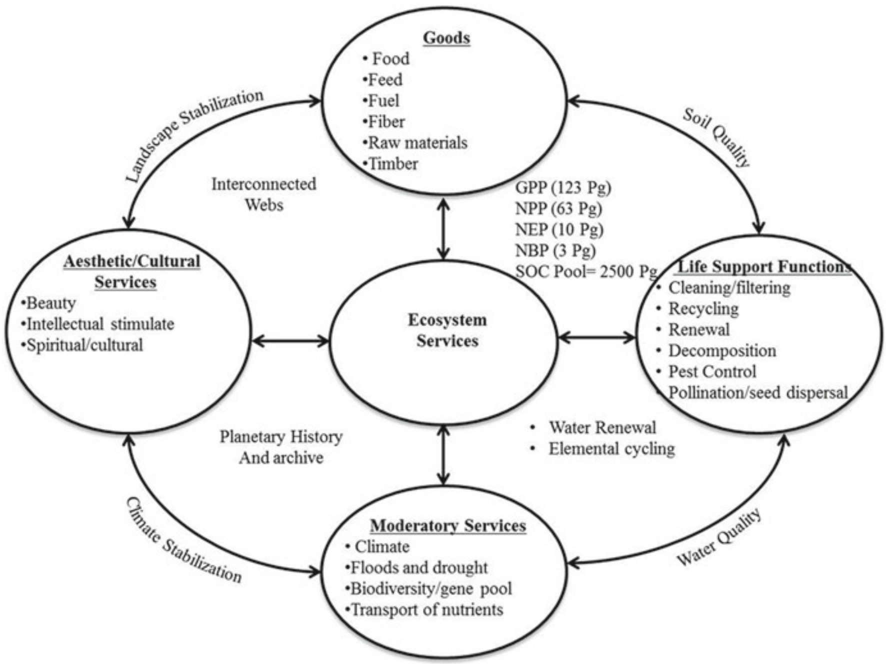

Fig. 1.1 Some examples of ecosystem services (Data on DPP etc. from Jansson et al. 2010; that on SOC pool from Lal 2004, 2010)

Turner 1988, 1991; MEA 2003, 2005). The concept of ESs is also in accord with the "*Gaia*" hypothesis proposing that Earth's organisms and their inorganic environment are closely integrated to form a single and self-regulating complex system maintaining the conditions for life (Lovelock 1979). Principal ESs are generated by a web of interacting processes within natural ecosystems and the terrestrial biosphere, and driven by solar energy (Daily 1997). There is a close inter-dependence between EFs and ESs. The EFs refer variously to the habitat system properties or processes of ecosystems which generate ecosystem goods such as food, feed, fiber etc. (Costanza et al. 1997). The diversity of terminology and definitions necessitate the strong need for standardizing the concepts and accounting units (Boyd and Banzhaf 2007).

Among the principal ESs is carbon (C) sequestration which can be defined as the uptake of C-containing substances and, in particular, carbon dioxide (CO_2) into a reservoir (IPCC 2007). Atmospheric C is initially fixed during photosynthesis and subsequently sequestered in terrestrial ecosystems in growing biomass of plants and other organisms. Globally, established forests are major terrestrial C sinks as they accumulate and store more C and for longer periods of time compared to non-forest ecosystems (Pan et al. 2011). Some of the aboveground C in terrestrial ecosystems moves belowground and may be sequestered in growing roots, soil (micro)organisms and, finally, during the formation of soil organic matter (SOM). However, only a very small fraction of the C initially fixed during photosynthesis enters the soil.

For example, less than 5 % of the C fixed by terrestrial ecosystems in Europe enters the soil and may eventually be sequestered (Schulze et al. 2010).

Soil is an integral part of each terrestrial biome and ecosystem, soil is a four-dimensional body (length, width, depth and time) at the interphase of the lithosphere and atmosphere, and is the basis of all terrestrial life. It is also the geomembrane that denatures and filters pollutants, and is a habitat for billions of known and mostly unknown species. Through an interconnected web of complex processes and properties, soil supports life, moderates climate, renews water resources, and recycles elements and disposes of the waste by-products.

Soil ecosystem services can be classified as (i) supporting (renewal, retention and delivery of nutrients for plants), (ii) regulating (regulation of major elemental cycles; buffering, filtering, and moderation of the hydrologic cycle; disposal of wastes and dead organic matter), (iii) provisioning (building material; physical stability and support of plants), and (iv) cultural (heritage sites, archeological preserver of artifacts; spiritual value, religious site, and burial grounds) (Daily et al. 1997; MEA 2005). Soil's ESs are created by its natural capital or endowment of inherent characteristics. The natural capital implies properties such as clay content and predominant clay minerals, soil organic carbon (SOC) stock and its turnover as influenced by its biological, chemical and physical characteristics, available water hold capacity depending on the retention porosity and forces of cohesion and adhesion within structural aggregates, effective rooting depth determined by the depth of solum and its life support processes, SOM stock and its dynamic, and plant nutrient reserves and their availability over spatial temporal scales. Soil's ESs can be improved, in particular, by an increase in SOC stocks (Franzluebbers 2010). In general, the level of SOC within a particular soil can have profound influences on the capacity of the soil to produce food, feed, fiber and fuel. By enhancing soil C sequestration, soil biological, chemical and physical quality can be improved which in turn improves ESs (Lal 2009).

1.2 Soil Processes and Ecosystem Services

Soil processes are mechanisms which govern physical, chemical and biochemical reactions within the soil solum through close interaction with the lithosphere, biosphere, atmosphere and the hydrosphere (Fig. 1.2). It is the net effect of weathering and renewal in the lithosphere, photosynthesis and respiration in the biosphere, gaseous exchange and the radiative forcing in the atmosphere, and erosion and deposition in the hydrosphere which determines terrestrial EFs, ESs and the natural capital of soil. Conversion of natural to managed ecosystems (agro, silvo, urban, mining) alters the processes, EFs and ESs. The strategy is to minimize the magnitude of adverse alterations so that the perturbations thus created are minimal, and thresholds on critical properties do not reach the tipping point of runaway degradation.

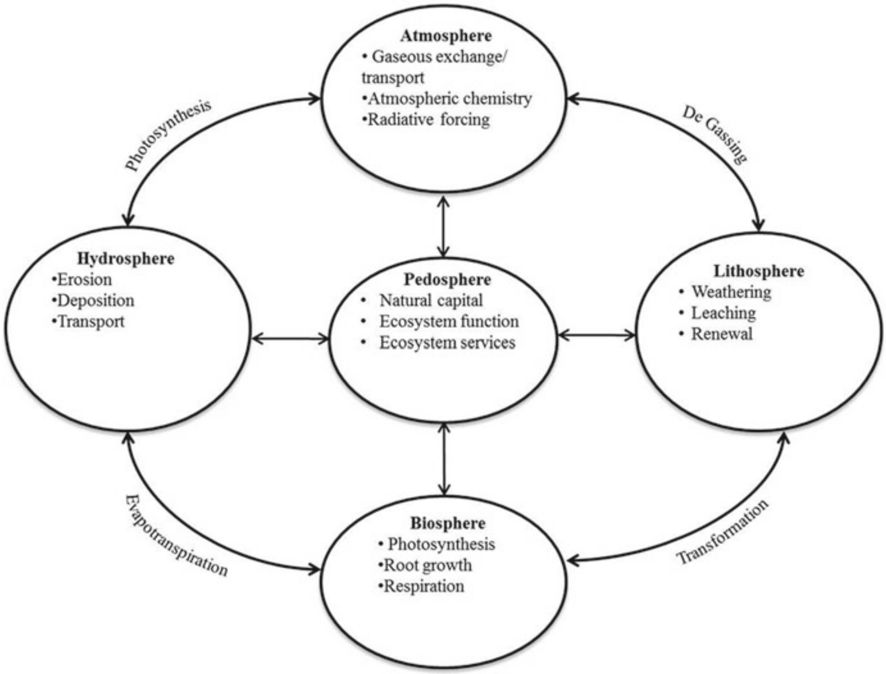

Fig. 1.2 Interactive pedospheric processes with biosphere, atmosphere, lithosphere and the hydrosphere which create natural capital of soil, moderate ecosystem functions

1.3 Soil Degradation Impact on Ecosystem Services

Soil degradation implies decline in soil quality and its natural capital by any perturbation(s) leading to reduction in its capacity to perform EFs and ESs. It is set-in-motion and caused by both natural and anthropogenic activities. Any perturbation that accelerates the rate of a process beyond that which occurs under the natural ecosystems can potentially lead to soil degradation. Some examples of degradation processes triggered by acceleration of the rate are those of erosion compared with soil formation, decomposition compared with net biome productivity (NBP), leaching compared with illuviation and deposition etc. It is the degradation of soil quality and its natural capital that jeopardizes terrestrial EFs and ESs.

1.4 Relationship Among Multiple Ecosystem Services

Most EFs and ESs are interlinked, and the magnitude and quality of one depends on those of another. Increase in one ES or EF can decrease others because there may be trade-offs and associated disservices. Thus, it is important to understand basic principles governing these interlinkages (Bennett et al. 2009; Tallis and

Kareiva 2005; Carpenter et al. 2009). What cannot be credibly measured cannot be adequately managed (EDI 2008). In the context of EFs and ESs, it is important to strengthen the understanding of the ecology underpinning the provision of diverse ESs (Kremen and Ostfeld 2005). This knowledge would also improve the managerial capacity to minimize threats to ESs (MEA 2005), especially those related to global food production (Tilman et al. 2001). The close link between SOC depletion and accelerated erosion caused by, for example, crop residues removal for cellulosic ethanol and other competing uses must be objectively and critically assessed (Lal 2008). There are numerous effects of agricultural practices on ecosystem services (Dale and Polasky 2007), and disservices such as loss of biodiversity, non-point source pollution, emission of greenhouse gases, soil degradation and desertification etc.

1.5 Soil Degradation and Desertification

Soil degradation is a major global issue, and is being exacerbated by land misuse and soil mismanagement (Bai et al. 2008). Improper agricultural practices are an important cause of disservice which accentuates risks of soil degradation, and especially desertification. Thus, there is a need to identify and use ecological indicators to estimate the magnitude of change and the degree of and response to the exposure (Suter 1993). Ecological indicators are chosen to assess environmental conditions, establish the cause-effect relationship and identify trade-offs (Dale et al. 2000, 2004; Dale and Beyeler 2001). It is also in this context that many have suggested the use of collective management. Stallman (2011) proposed cooperative solutions for specific ESs (i.e., pollination, flood control). However, collective management may involve a group of citizens who jointly manage a community-owned property to minimize on-site and off-site effects or disservices (Ostrom 1990).

1.6 Payments for Ecosystem Services

Mismanagement of soils does not only have long term costs for the soil manager on site but also adverse off site effects, that is, economic externalities. Such problems of externalities of soil mismanagement cannot be easily addressed by regulations and enforcement. Establishment of regulation regimes, their monitoring and enforcement costs can be high. An alternative is payments for ecosystem services (PES) which can serve to translate external, non-market values of the environment into incentives for local actors to provide environmental services (Engel et al. 2008). On the degraded lands and soils of the world an estimated 40 % of the poor are living (Nkonya et al. 2011). Poverty is one of the likely drivers of soil degradation, but enforcement of sustainable soil management is very difficult in that context. As PES is based on the beneficiary-pays rather than the polluter-pays principle it can be attractive for instance in settings where ES providers are poor, marginalized landholders (Engel et al. 2008).

Because land resources (soil, water, vegetation, biodiversity) are finite and fragile, the goal of sustainable management is to enhance ESs and EFs while minimizing risks of disservices associated with degradation, eutrophication etc. Enhancing productivity and other ESs of agroecosystems necessitate a widespread adoption of recommended management practices (RMPs). Most RMPs have additional or opportunity costs. For example, crop residues must be retained on the soil as mulch to reduce risks of water runoff and soil erosion and sequester C. Yet, it has numerous competing uses (e.g., feed, fuel, industrial material) and has a high economic value. Thus, performance based financial transfer from beneficiaries of ESs, or national/international organizations acting on their behalf, to providers of ESs will sustain the provision of these ESs (Ferraro 2011), and also maintain the finite recourse base. The strategy for PESs is also arguably (but debatably) more cost effective and with a simpler protocol (Simpson and Sedjo 1996). Not all PESs schemes need to be government financed, which may be needed when government acts upon ES users or the broader public interest; depending on circumstances, PES can also be directly user financed if institutional contexts permit so, for example in cases of soil related water system quantity and quality services.

The empirical data on effects of PES on adoption of RMPs and effectiveness of this strategy is scanty (Ferraro 2011). The research information is also scarce regarding the on-site (Arriagada et al. 2012) and off-site spillovers (Alix-Garcia et al. 2010). Rather than conventional approaches to PES, performance payments for protected ecosystems are also being proposed. As the experiences with soil related PESs are still at infancy, lots of experimental approaches may be helpful to establish robust institutional experiences for efficient solutions in well-defined context regarding PES performance measures, and payment modes and amounts. Specifics of program design and ecological and economic contexts, incl. property rights situation related to land and water, will determine the effectiveness and efficiency of PES for the complex soil related ES. Thus, well designed and tested implementation of PES programs may be an important strategy to enhance ESs, minimize risks of disservices, and advance sustainable management of finite and fragile natural resources.

1.7 Addressing Societal Issues Related to Ecosystem Services

People draw on scientific, institutional, contextual local or traditional knowledge forms to interpret problems and possibilities within their environment (Berkes 2008). However, diverse interests of individual stakeholders and disciplines may not be balanced when it comes to ecosystem services and their possible improvement. Thus, joint problem solving towards sustainable management of soil-based ESs requires mutual learning between scientists and external stakeholders, i.e., a transdisciplinary approach. Global developments such as climate change, increasing urbanization, and loss of biodiversity challenge ecosystems, their services and functions. Aside natural systems, however, social and artificial systems are also affected by these global developments. Thus, common themes among social, natural and artificial systems must be identified for the development of transdisciplinary collaborations to find ways to enhance ESs, minimize disservices and advance sustainable management of ecosystems.

Transdisciplinarity can be defined as a reflexive research approach that addresses societal problems by means of interdisciplinary collaboration as well as the collaboration between researchers and extra-scientific actors (Jahn et al. 2012). Integrating different scientific and extra-scientific insights produces new knowledge. Aim of transdisciplinary approaches is to enable mutual learning processes between science and society (Scholz 2001). Integration is the main cognitive challenge of the transdisciplinary research process. Novel and hitherto non-existent connections are established among the entities that make up the problem context (Jahn et al. 2012). Disciplinary, multidisciplinary, interdisciplinary and cross-disciplinary collaborations may not be sufficient to find solutions for the sustainable management of ESs. However, transdisciplinarity or any cross-disciplinary collaboration involves disciplinary practice. Incapacity to deal with problems such as climate change, health, land-use, forestry management, renewable and non-renewable resources, housing, poverty and urban planning is related to their complexity, to the compartmentalisation of scientific and professional knowledge, to the sectoral division of responsibilities in contemporary society, and to the increasingly diverse nature of the societal contexts in which people live (Lawrence and Després 2004). Thus, researchers and 'practitioners' dealing with the finite resource soil must cooperate in a transdisciplinary way to tackle the complexity of social-ecological issues associated with soil-based ESs.

1.8 Conclusion

A wide range of benefits and tangibles provided by functioning ecosystems depend on the stock materials inherent in soil. Sustainable management of soil implies maintenance of stock and flows at a level above the threshold or tipping point. Soil degradation by natural or anthropogenic factors reduces the provision of services and may even create disservices because of large ecological footprint. Increasing the SOC stock can improve soil quality and in turn soil-based ecosystem services. Payments for ecosystem services may be a viable tool to promote sustainability and minimize risks of soil and environmental degradation. Scientists and extra-scientific actors should integrate the multiple types of knowledge in a transdisciplinary way to address societal issues with respect to the management of ecosystem services.

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Chapter 2
Soils and Ecosystem Services

Rattan Lal

Contents

Abstract Soil's ecosystem services (ESs), provisions of tangible goods and intangible environments and processes, are governed by its capital endowment (inherent characteristics such as clay content) and capacity to transform managerial inputs into productivity. Thus, soil quality refers to its capacity to provide and sustain a range of ESs and functions of interest to human and for maintenance of ecosystem health. Provisions of these services and functions also depend on land use (e.g., arable, pastoral, silvicultural, urban, recreational, spiritual). Because of

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strong interactivity, management to enhance some ESs can jeopardize others or lead to some adverse degradation processes (e.g., accelerated erosion, water pollution, decline of biodiversity). Similarly, production of biofuels can exacerbate competition with food production for land, water, energy, and nutrients. The abrupt climate change (ACC), as an example of human footprint caused by the use of natural resources, also impacts ESs and the underlying processes. Developing mechanisms and identifying/implementing policies for payments to land managers for enhancing ESs such as carbon (C) sequestration can promote adoption of best management practices (BMPs) and facilitate restoration of degraded soils and ecosystems. Recarbonization of the biosphere and sequestration of soil organic carbon (SOC) are important strategies to enhancing and sustaining ESs and functions of natural and managed ecosystems.

Keywords Food security • Water quality • Soil degradation • Climate change • Ecological footprint

Abbreviations

ACC	Abrupt climate change
BMPs	Best Management Practices
BNF	Biological nitrogen fixation
ESs	Ecosystem services
GHGV	Greenhouse Gas Value
GHGs	Greenhouse gases
INM	Integrated Nutrient Management
IPM	Integrated Pest Management
MRT	Mean residence time
NT	No-till
NUE	Nutrient use efficiency
OM	Organic matter
PES	Payments for Ecosystem Services
SOC	Soil organic carbon
SQI	Soil Quality Index
SLM	Sustainable land management
WUE	Water use efficiency

2.1 Introduction: Terms and Basic Concepts

Soil is a four dimensional body (length, width, depth and time) at the interface of atmosphere and the lithosphere, and interacting very strongly with the biosphere and the hydrosphere (Fig. 2.1; Dokuchaev 1883; Jenny 1941). It is an integral component of land, and strongly interacts with hydrology, vegetation/biodiversity, landscape etc.

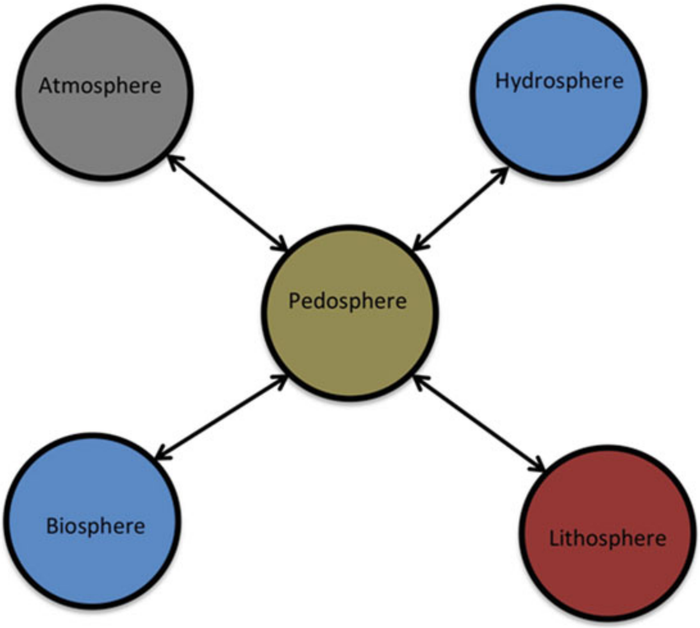

Fig. 2.1 Interaction of soil with the environment

The term ecosystem services (ESs) emerged in 1980s as a tool to describe a framework of structuring and synthesizing biophysical understanding of ecosystem processes with a focus on human well-being (Mooney and Ehrlich 1997) and ecosystem health.

A closer analysis indicates that it is not altogether a new concept. For example, Theodore Roosevelt (U.S. President from 1901 to 1909) created national parks and monuments to preserve nature and the goods and services that the land provides. Aldo Leopold (1949) advocated the concept of "land ethic" and emphasized the necessity of conserving natural resources. Leopold recognized the importance of connecting people with the land that provides them all the essentials for their well-being. At present, however, the term ESs has been given a more formal definition. It refers to the benefits of nature to households, communities and economies (Daily 1997; Costanza et al. 1997; Boyd and Banzhaf 2007; Dale and Polasky 2007).

In general, ESs are "the quantifiable or qualitative benefits of ecosystem functioning to the overall environment, including the products, services and other benefits human receive from natural, regulated or otherwise perturbed ecosystems" (Fig. 2.2) (MEA 2005). Although used erroneously as synonymous to ESs, the term ecosystem function or process implies transformation of inputs into outputs (Dominati et al. 2010).

Most ESs can be grouped into different categories (MEA 2005). An example of such a grouping can be into tangibles (food, feed, fiber, raw material) and intangible (aesthetic, religious, spiritual and cultural) values (SRR 2008). The intangibles are primarily perceptual in nature. For example, input of fertilizer or water into soil can transform its productivity. The rate of transformation of input into output is the use

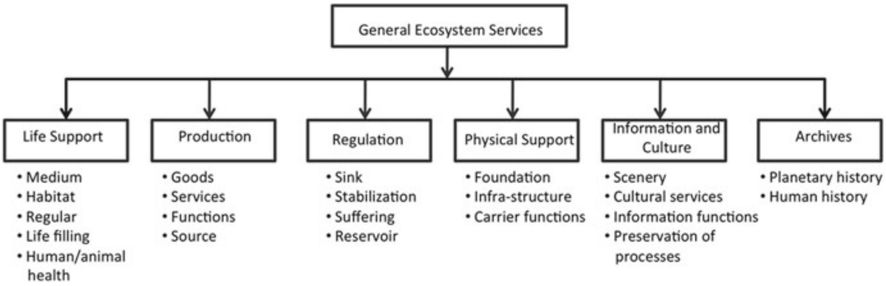

Fig. 2.2 General ecosystem services

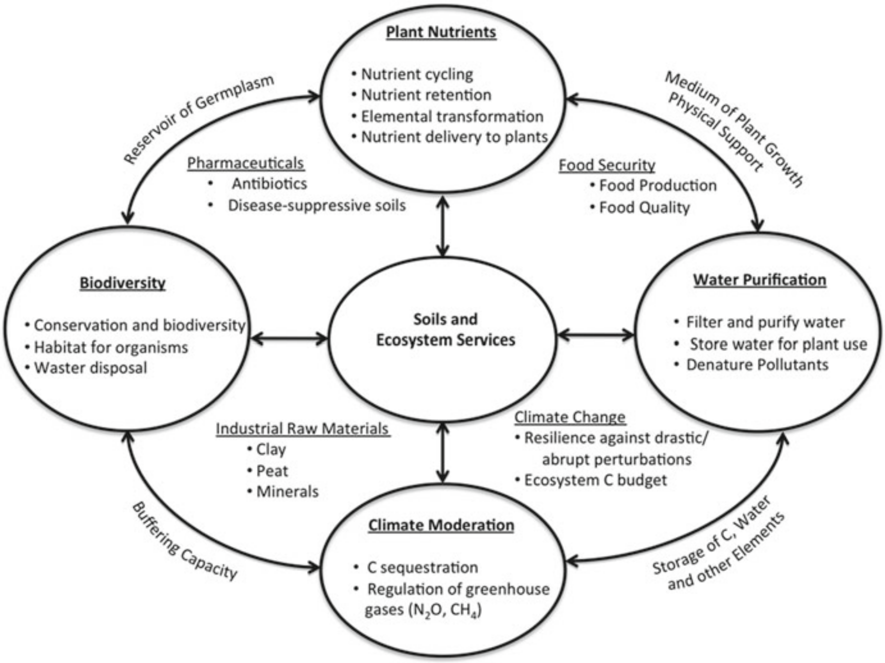

Fig. 2.3 Ecosystem services provisional by soil

efficiency such as water use efficiency (WUE), or nutrient use efficiency (NUE). Soil's numerous ESs (Fig. 2.3) depend on its quality, or its capacity to produce goods and services of economic and ecologic significance. Principal controls of soil's ESs are its capital (e.g., properties), land use and management (e.g., input, farming systems, non-agricultural uses). Management depends on land use (i.e., agricultural, forestry, urban, recreation, nature conservancy). Soils under agroecosystems (i.e., arable, pastoral), covering 38 % of the total ice-free land area are more intensively managed than those under forestry and nature conservancy. Thus, sustainable land management (SLM) implies maintaining or increasing ESs which is a grand challenge for agriculture (Stallman 2011).

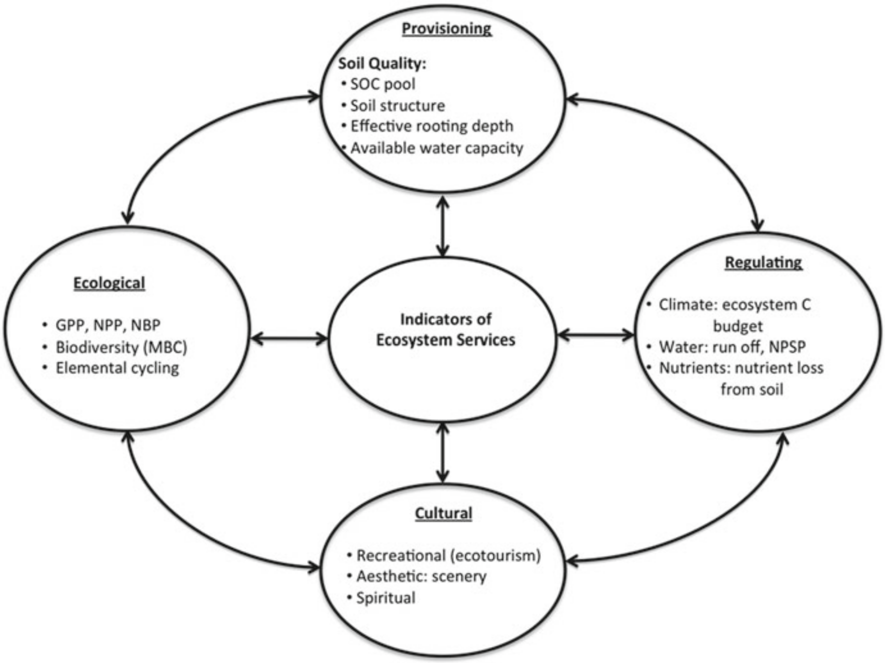

Fig. 2.4 Indicators of ecosystem services

Conversion of natural to agroecosystems can adversely impact some ESs. The magnitude and the direction of change can be assessed by using indicators (Dale and Beyeler 2001; Dale et al. 2000, 2004). These indicators are needed to assessing environment quality, detecting early warning signals of any drastic change, identifying disservices or trade-offs (Zhang et al. 2007; Dale and Polasky 2007), and developing strategies for ecological risk assessment and management (Suter 1993). Thus, there are specific indicators of ESs (Fig. 2.4). A land use that is managed to maximize a specific ES (i.e., food production from agroecosystem) may reduce the provision of other functions and services. It has been argued, for example, whether it is possible to have both food security and biodiversity (Chappel and LaValle 2011), and good water quality (Edwards and Harrold 1970). Thus, SLM must be based on understanding relationships among diverse ESs (Bennett et al. 2009). Therefore, the goal of sustainable soil/land management is to "create and maintain conditions under which humans and nature can exist in a productive harmony, and fulfill the social, economic, and other requirements of present and future generations" (United Nations 1987).

From a soil's perspective, sustainable management implies a 3-pronged approach: (i) replace what is removed, (ii) response wisely to what is changed, and (iii) predict what will happen "downstream" from anthropogenic influences anywhere in the landscape (Hammer 2010). Therefore, the objective of this chapter is to discuss the importance of soil use and management on ESs and trade-offs or disservices (Table 2.1).

Table 2.1 Relevant terms and definitions

Term	Definition	Reference
Ecosystem functions	Refer to the habitat, biological or system properties or processes of ecosystems	Costanza et al. (1997)
Ecosystem services	(i) Conditions and processes through which natural ecosystems, and the species that make them up	Daily (1997)
	(ii) The benefits of nature to households, communities and economies	Boyd and Banzhaf (2007)
	(iii) Benefits provided by functioning ecosystems	Leslie (2011)
	(iv) Consist of flows of materials, energy and information from natural capital stocks which combine with manufactured and human capital services to produce human welfare	Costanza et al. (1997)
Natural capital	(i) A stock of materials or information that exists at a point in time. Each form of capital stock can generate a flow of services that may be used to transform materials, or the spatial configurations of material, to enhance the welfare of humans	Costanza et al. (1997)
	(ii) It is a stock as opposed to flow	Dominati et al. (2010)
Ecosystem goods	These are tangible outputs from ecosystems that are provided to humans through human activities	Maczko and Hidinger (2008)
Ecosystem dis-services	Conditions and processes which reduce productivity or increase production costs, and detract from contributing to productivity	Zhang et al. (2007)
Ecological foot-print	The area of biologically productive land and water that a population (an individual, a city, or country or all of humanity) uses to generate the resources it consumes and absorb its waste under prevailing technology	Kitzes et al. (2008)

2.2 Soil Quality

The importance of soil was appropriately described by Franklin D. Roosevelt in his letters in 1937 to all governors by stating that "The nation that destroys its soil, destroys itself". Earlier, Thomas Jefferson (1743–1826) had outlined a visionary concept by advocating that "civilization itself rests upon the soil". Both leaders emphasized the importance of soil quality and its management in perpetuity. Management of soil quality, ability of a soil to perform functions that are essential to people and the environment, and understanding the nature and dynamics of its determinants are essential to maintenance and enhancement of ESs. Among principal determinants of soil quality are soil organic carbon (SOC) along with its quantity and quality, physical conditions and nutrient reserves (Powlson et al. 2011). These crucial determinants of soil quality are considered the "natural capital" of soil (Robinson et al. 2009; Leslie 2011). The latter consists of the stock of mass and energy and their transformations (both quantity and quality). Principal determinants

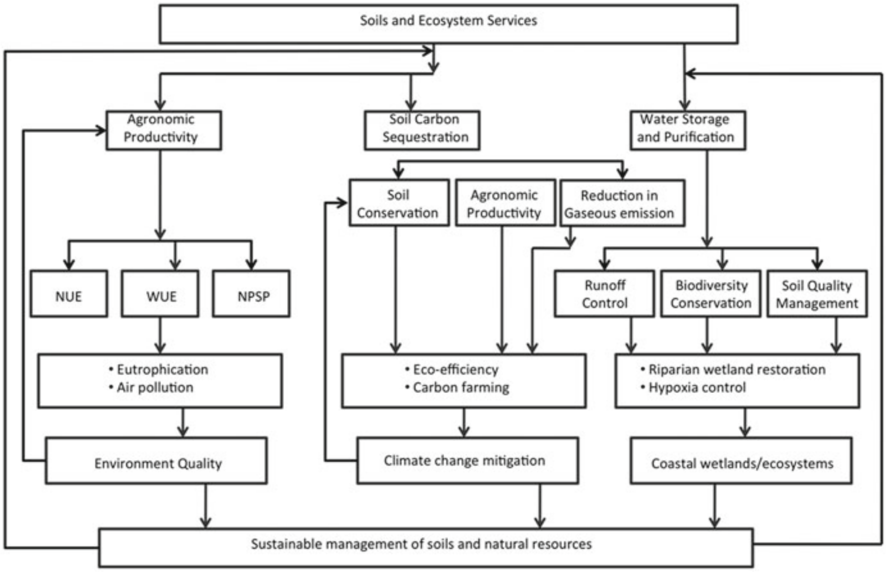

Fig. 2.5 Interaction among soil-related ecosystem services and functions

of soil physical quality are SOC, texture, structure, and aggregation, bulk density, and water retention and transmission (Dexter 2004a, b). Quality of soil physical characteristics is strongly influenced by biological processes (Lavelle 2000).

Ecological intensification (Doré et al. 2011) to enhance ESs can be achieved by implementing SLM options (Schwilch et al. 2011) which restore and enhance soil quality. Decline in soil quality, by natural or anthropogenic perturbations, can drastically alter soil capital, ESs and functions, and create trade-offs and disservices. Indeed, there is a strong interaction (Fig. 2.5) and inter-connectivity (Fig. 2.6) among soil-related ESs and functions.

2.2.1 Soil Degradation

The term soil degradation refers to long-term reduction in ESs caused by depletion of natural capital and the attendant decline in functions and productivity. Soil degradation has plagued humanity ever since the dawn of settled agriculture. However, credible statistics on the extent and severity of degradation (especially in relation to ESs) are not known. The extent of human-induced soil degradation was estimated at 1,964 million hectares (Mha) by the GLASOD methodology during early 1990s (Oldeman 1994), and land degradation at 3.5 billion ha (Bha) by using the normalized difference vegetation index (NDVI) by Bai et al. (2008). While being widespread and serious, it is also argued that the problem cannot be remedied simply by low-inputs. While the use of so called "participatory" approaches based on community

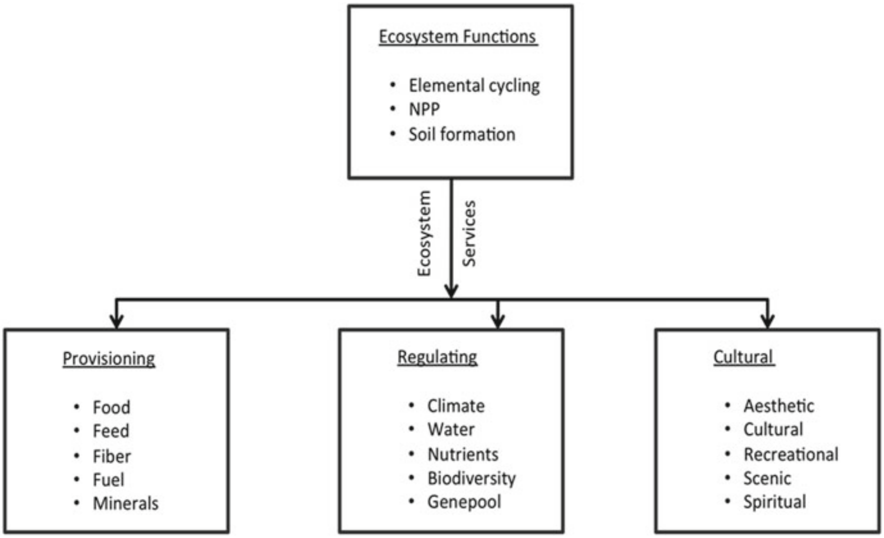

Fig. 2.6 Inter-connectivity among soil-relation provisions of ecosystem services

involvement at all levels (i.e., project planning, decision making and execution) is considered essential to combating desertification such as in southern Africa (van Rooyen 1998), a long-term and an effective solution to this complex problem necessitates a strong investment in infrastructure at appropriate level of external inputs (Koning and Smaling 2005) without which ESs cannot be restored. Indeed, nutrient mining in Sub-Saharan Africa, driven by population growth, indicates that Malthusian mechanisms are at work (Drechsel et al. 2001) and create disservices at the continental scale. There is a strong relationship between soil fertility decline by nutrient depletion and prevalence of hunger in Africa (Sanchez 2002). In addition to low external input and use of extractive farming practices, economic globalization has also triggered deforestation and elimination of ESs provisioned by natural vegetation (Lambin and Meyfroidt 2011).

2.2.2 Ecological Footprint

Ehrlich and Holdren (1971) proposed the concept that impact of population growth on the environment can be assessed by the equation $I = P \times A \times T$, where I is impact, P is population, A is affluence, and T is technology. This equation provides a conceptual basis of estimating the footprint of population and its lifestyle (Holdren and Ehrlich 1974). Protection of nature for nature's sake (McCauley 2006), for aesthetic and cultural services, is also needed because market-driven conservation approaches have not been entirely effective. Because many of the aesthetical/cultural/ethical services lack markets, they also lack the incentives for provision that come with prices (Swinton et al. 2007). Thus, sustainability has been appropriately defined as the possibility of all people living rewarding lives within the means of nature, and limiting human demands for ESs and goods within nature's capacity (Kitzes et al. 2008). The goal is to shrink humanity's present and future ecological footprint. In this context, the concept of "carrying capacity" (Leopold 1949) or ecological sustainability are relevant basic ecological indicators of biophysical sustainability. Sutton et al. (2012) proposed the use of "eco-deficit" as a measure of biophysical sustainability. A credible measure of ecological footprint of a land use or management system is an essential decision-making tool for land managers and policy makers.

2.3 Land Use

Land use refers to the human use of land, and involves the conversion and management of natural ecosystems or biomes to produce specific goods and services for meeting human demands. Land use is defined as "the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it" (FAO 1997; FAO/UNEP 1999). Principal land uses are estimated urban (1.5 %), arable (13 %), pasture (26 %), forests and woodlands (32 %) (FAO 2010). Conversion of natural to managed ecosystems has been accelerated since the middle of the twentieth century to meet humanity's need for living space, food, clean water, energy, industrial raw material etc. Yet, the habitat needs of other species must not be overlooked. The current and projected land use change can strongly influence soil's ESs (Nelson et al. 2010). Agricultural land use (e.g., cropland, pastoral land) is expanding to provide food, feed, fiber, fuel and other non-market services (Swinton et al. 2007). Thus, understanding the mechanisms and processes which generate these goods and services is critical. The concept of sustainable agricultural intensification (Omer et al. 2010) must be executed on basis of the knowledge and understanding of the mechanisms and processes. Attempts to mitigate climate change while maintaining ESs necessitate quantification of full greenhouse gas (GHG) effects of land use decisions. There is a lack of appropriate metric to account for emission of all GHGs within a tillage (land use) system. In this context, Anderson-Teixeira and DeLucia (2011) proposed the concept of greenhouse gas value (GHGV) of ecosystems. The GHGV accounts for potential GHG release upon conversion of vegetation cover (deforestation), annual GHG flux, and probably GHG exchanges caused by land use conversion and management. It provides a quantitative value of the GHG consequences of any land use decision.

2.3.1 Cropland

Intensive cropland management, to produce high yields per unit area and time, has a high ecologic footprint. Inputs of synthetic fertilizers, pesticides, supplemental irrigation and tillage operations have high carbon (C) costs (Lal 2004a). Therefore, testing and validating alternate management systems have been a high priority since 1980s.

The range of alternate management practices include no-till (NT) or reduced tillage systems, integrated nutrient management (INM) based on use of manure and leguminous cover crops in the rotation cycle, integrated pest management (IPM), complex (diverse) crop rotations including agroforestry and ley farming. Recommendations for alternative best management practices (BMPs) are soil and crop specific (Khakbazan et al. 2009). In terms of soil quality, earthworms and litter management can enhance soil quality and improve ESs (Fonte and Six 2010). Weeds, an important component of the overall biodiversity within crop fields (Marshall et al. 2003), must be managed in view of all ESs provisioned by an arable land use. Similarly, nutrient management is important to optimizing SOC sequestration. Use of organic sources of nitrogen (N) over a long time can enhance total SOC stock and also the resistant and slow fractions with longer mean residence time (MRT) (Fortuna et al. 2003).

Agricultural practices used in modern agroecosystems generate GHGs and have a large C footprint. Thus, it is pertinent to identify strategies of reducing the C footprint of agroecosystems without jeopardizing the productivity. Management strongly influences the ecosystem C and GHG budget (Ceschia et al. 2010; Dyer et al. 2010; Lal 2004a, b). Important among these strategies are (Gan et al. 2011a, b): (i) choosing appropriate crop species, (ii) diversifying cropping systems, (iii) including biological nitrogen fixation (BNF) to reduce the input of synthetic N fertilizer, (iv) improving use efficiency of nutrients, water and other energy-based inputs, and (v) using NT and crop residue management systems. The goal is to adopt integrated farming techniques which improve energy efficiency and reduce energy inputs without affecting energy outputs (Alluvione et al. 2011; Gruia 2011). Use of legumes and perennial grasses in a rotation can reduce emission of GHGs (Bremer et al. 2011). Similar to prudential management of nutrients and energy, management of SOC is also important to enhancing soil quality and improving agronomic productivity (Lal 2006, 2010a, b; Brock et al. 2011). Thus, organic farming with cash crops can enhance ecosystem/soil C stock and reduce emissions (Chirinda et al. 2010). Despite the carbon dioxide (CO_2) fertilization effect and other opportunities which may arise, global climate change may have adverse effects on agroecosystems because of decline in ESs and some of the projected benefits may be lost in a warmer climate (Fuhrer 2003). Increase in nitrous oxide (N_2O) emissions from agroecosystems such as from rice (*Oryza sativa* L.) (Asakawa 1993), is one issue and that of methane (CH_4) another (Shang et al. 2011).

The NT farming is gaining momentum in North and South America, and is globally practiced on more than 110 Mha (Kassam et al. 2009). The practice evolved during early 1960s in response to the severe problem of accelerated erosion and non-point source pollution on sloping lands of the U.S. At the North Appalachian Experimental Watershed in Coshocton, Ohio, Edwards and Harrold (1970) reported that average soil loss was 11.6 Mg ha^{-1} for conventional practice of low fertility and sloping straight rows, 3.2 Mg ha^{-1} for moderately high fertility and contour rows, 0.9 Mg ha^{-1} for minimum tillage (plow/plant system), and merely 0.02 Mg ha^{-1} for NT system. Similarly, NT farming may also be effective in minimizing the risks of wind erosion. Adoption of NT farming with residue mulch in dryland agroecosystems can enhance soil water storage in the root zone and increase crop yield, and make it possible to increase cropping intensification beyond the traditional wheat (*Triticum* spp.)-fallow system and thereby increase the WUE in the west central Great Plains of the U.S. (Peterson and Westfall 2004). Increase in SOC stock also leads to improvement in biodiversity and the provision of other ESs (Marks et al. 2009).

While the projected climate change may reduce SOC storage especially because of increase in mean annual temperature (Fantappiè et al. 2011), conversion of plow tillage and other degraded soils to NT farming can also increase soil C storage and mitigate GHG emissions (Franzluebbers 2010). Increasing SOC stock is important to improving ESs (Powlson et al. 2011), while also building a climate-resilient agriculture (Jackson et al. 2011). Irrigation management, especially done in conjunction with mulch-based NT farming, can also increase retention of SOC in the solum (Wang et al. 2010).

Despite numerous benefits, there are several constraints to adoption of NT farming especially by the resource-poor farmers and small land holders of the tropics and sub-tropics. Crop residues have numerous competing uses and are of such a high value that farmers are not prepared to leave it on the soil surface (Mchunu et al. 2011). The other concern is the net accumulation of SOC stock in the whole profile rather than just in the surface 0–20 cm depth. Some researchers have argued that, on a profile basis, there may either be no differences in SOC stock or it may be higher in plow tillage than in NT system (Kravchenko and Robertson 2011; Baker et al. 2007; Blanco-Canqui and Lal 2008).

There is also a question of gross versus net SOC sequestration. The latter may imply net SOC stock after the hidden C costs have been considered (Lal 2004a). The other is to consider emission of all GHGs by using the life cycle assessment of all components (Grant and Beer 2008). The magnitude and type of GHGs emitted depend on a range of factors including tillage methods and type of fertilizers (Smith et al. 2008). Use of NT system can accentuate emission of N_2O. It can also reduce emissions of CH_4 because of improvement in soil structure that enhances CH_4 oxidation (Lal 2004b). Obtaining reliable estimates of N_2O emission is a challenge, because of a high spatial heterogeneity and temporal variability (Röver et al. 1999). Emission of N_2O is also influenced by incorporation or plowing under of a winter cover crop (Petersen et al. 2011).

2.3.2 Pasture and Rangeland

Pastoral and agro-pastoral systems constitute an important land use in semi-arid and arid regions, and in marginal soils (e.g., steep lands, stony and rocky lands). Sustainable management of these land uses is important to minimizing risks of soil degradation (erosion, salinization) and GHG emissions, while preserving biodiversity and other ESs of rangelands. Because of the economic and ecologic significance, guidelines have been prepared to maintain and enhance ESs of the land use

(SRR 2008; Bernués et al. 2011). With adoption of BMPs, rangelands/grasslands can be net C sink (Zeeman et al. 2010), and yet produce food and generate other goods and services. Over and above the benefits of improving forage species and controlled grazing etc., genetic selection for efficient feed use for milk production can also reduce nutrient requirements, GHG emissions, and land use per unit product (Bell et al. 2011). The use of whole farm system modeling approach (Crosson et al. 2011) has also documented that improvement in animal productivity (i.e., live weight gain, milk production) and fertility (i.e., lower culling, lower replacement rates) can drastically reduce GHG emissions per kg of product. Similar to the benefits derived from sustainable intensification of arable lands, increasing output per ha of rangelands can also reduce GHG intensity (emission per kg product) as long as inputs of feed/fertilizer are optimal and not excessive (Crosson et al. 2011).

2.3.3 Peatland/Wetlands/Permafrost

Permafrost ecosystems are widespread in the arctic and boreal regions of the northern hemisphere, and these ecosystems occupy 16 % of the Earth's exposed land surface area (Schuur et al. 2008). Permafrost are important reservoir of organic C. Schuur et al. (2008) estimated that the total C stock in the northern circumpolar permafrost zone is 1,672 Pg C (1 Pg = 10^{15} g) of which 277 Pg is contained in peatlands. The 1,672 Pg is considered an underestimate of the total soil C in the permafrost region because deep soils have not been adequately considered (Schuur et al. 2008). In comparison with the global permafrost C reserve of 1,672 Pg, that in Siberian permafrost is estimated at 450 Pg C (Zimov et al. 2006). The large C reservoir is threatened by thaw due to abrupt climate change (ACC). The warming will accelerate microbial decomposition of organic matter (OM), release CO_2 (and CH_4) into the atmosphere, and enhance the positive feedback to ACC (Schaefer et al. 2011). Emissions to the atmosphere depend on a multitude of interacting factors including size of the C stock thawed by ACC, and by continuous (e.g., microbial decomposition) and episodic processes. Important among the episodic processes is fire, which can drastically alter the C stock of peatlands. It is widely feared that the carbon-climate feedback through thawing of the permafrost can enhance atmospheric CO_2 by additional 50–100 ppm by 2100 (Tarnocai et al. 2009). A global permafrost C flux of 0.8–1.1 Pg C year^{-1} is predicted by 2100 (Schuur et al. 2008). Schaefer et al. (2011) estimated a cumulative permafrost C flux to the atmosphere of 190 ± 64 Pg C by 2200, which will drastically change the permafrost from being a moderate C sink to a major C source. Jahn et al. (2010) reported that the arctic region may change from being a weak sink of 0.06 Pg C year^{-1} at present to a major source of 0.54 Pg C year^{-1}. The rate of permafrost thaw is a controversial topic (Yi et al. 2007) because of complex processes which are not yet fully understood. In addition to CO_2, CH_4 emissions are also projected to increase by melting of the permafrost (Koven et al. 2011).

The Himalayan region, called the Third Pole, may also be affected by the ACC. In addition to the controversial rate of glacial retreat, there may be a positive feedback due to degradation of grasslands in the Tibetan plateau (Du et al. 2004). The positive feedback is supposedly triggered by degradation of grassland by overgrazing.

There exists a close link between permafrost and peatlands, which provide numerous ESs (Kimmel and Mander 2010), and are vulnerable to ACC. Total area of northern peatlands store between 329 and 525 Pg C, or about 19 % of the global soil C stock, play an important role in the global C cycle (Gorham 1991), and are also source of 10 % of global CH_4 emissions. Similar to permafrost, there may also occur a rapid C response of peatlands to ACC (Bridgham et al. 2008). It is important to note that the global C stock in peatland is 52 times the annual fossil fuel emission of 8.7 Pg in 2008 (Le Quéré et al. 2009).

2.3.4 Forestland

Forest ecosystems, covering about 30 % of the Earth's land surface, provide numerous goods and services including timber, water resources, food (bushmeat), and minor products such as pharmaceuticals, and aesthetical and spiritual benefits. The ACC poses severe risks to forest ecosystems, which are also closely linked with the water security (Brown et al. 2008) as discussed in Sect. 2.2. The ACC may affect forest ecosystems both positively and negatively. A positive response may be attributed to the CO_2 fertilization effect, longer growing season in the higher latitudes, and increase in the WUE. In contrast, the negative effects may be caused by stresses such as drought (Bigler et al. 2006, 2007; Allen et al. 2010), pests and pathogens (Ayres and Lombardero 2000), and heat-induced tree mortality (Allen et al. 2010). Barber et al. (2000) reported that temperature-induced drought stress disproportionally affected the most rapidly growing Alaskan white spruce (*Picea glauca* [Moench] Voss) during the twentieth century. Barber and colleagues hypothesized that the climate warming and the drought may have been important factors limiting CO_2 uptake in a large portion of the North American Boreal forest. That being the case, the future C sink capacity of the forests in northern latitude (the so called missing C sink or fugitive CO_2) along with the observed increase in plant growth (Keeling et al. 1996; Myneni et al. 1997) may be severely reduced. Adverse effects of ACC on forests, mostly due to temperature and the water stress, are amplified by anthropogenic factors (Kaffa et al. 2008). Yet, both of these stresses caused by the ACC can also be mitigated or alleviated by the judicious management of forest within a watershed (Ford et al. 2010b). For example, structure and functions of a watershed and its water yield can be drastically altered by land use and land cover through forest management. By analyzing 75 years of stream flow data from six watersheds with diverse management histories, Ford et al. (2010a, b) documented that changing land use altered steam flow. Ford and colleagues reported that converting deciduous stands to pine drastically altered the stream flow response to extreme events because of increase in soil water storage during the wet years.

2.3.5 Coastal Marshes and Marine Ecosystems

Coastal marshes and wetlands are affected by ACC through sea level rise, increased temperature, changes in rainfall distribution and freshwater inputs, and frequency and intensity of storms and other extreme events. The ecomorphology of coastal and estuarine wetlands is also impacted by changes in sediments and nutrient inputs (Day et al. 2008). The effects of these factors are accentuated by human activities. Coastal regions are preferred zones of urban settlements and are important to region's economy. Coastal areas cover about 200 Mha in the EU with coastline extending to 170,000 km across 20 member states (Zanuttigh 2011). The world's most densely populated coastal regions are those in South Asia (e.g., Bangladesh), and are highly vulnerable to the sea level rise. Coastal marshes are extremely vulnerable to sea level rise, and reduced sediment delivery (Kirwan et al. 2009). Vegetation, which plays an important role in functioning of tidal marshes, is changing from those adapted to higher elevations to those associated with lower elevations. This change in vegetation is influencing the stability of coastal marshes and their functions. Management of human activities is essential to minimizing the impact of ACC on coastal wetlands (Day et al. 2008).

Similar to coastal marshes and wetlands, marine biodiversity and ecosystem functioning is also vulnerable to ACC (Beaugrand et al. 2010). This is consistent with the observation that oceans have absorbed 84 % of the heat added to the climate system since 1970 (Levitus et al. 2005). The ACC-induced alterations in marine systems include biogeographical and phenological changes (Beaugrand et al. 2010; Edwards and Richardson 2004). The change in marine biodiversity may have negative effects on the downward biological C pump (Beaugrand et al. 2010). There has been an exponential increase in dead zones (hypoxia or anoxia) around the world which now amount to 245,000 km^2 of coastal marine systems (Diaz and Rosenberg 2008). The oligotrophic waters and the least production zone have expanded by 6.6 million km^2 since 1990 because of the global warming (Polovina et al. 2008).

The regulation of climate by the oceans is an important ESs which may be drastically influenced by ACC. Ocean is a major sink of atmospheric CO_2, and annually absorbs about 2.3 Pg of CO_2-C from the atmosphere (Le Quéré et al. 2009). The thermo-haline circulation is another important function, which may be directly affected by ACC. Above all, marine biodiversity is drastically affected by warming and acidification (Perrings et al. 2011).

2.4 Tradeoffs

Management to enhance one ES can either reduce others or lead to disservices (e.g., accelerated erosion, non-point source pollution). For example, an intensive arable land use enhances food production but may reduce biodiversity and exacerbate soil erosion risks. Berry Commoner (1971) advocated that nothing comes from nothing. Thus, enhancement of one ES can lead to a decrease in another or cause some disservices (e.g., accelerated erosion, emission of GHGs). For example, agricultural intensification, increasing yield per unit area and also expanding land area allocated to food production, reduces ecosystem (soil and vegetation) C stock. Increase in agricultural production is necessitated to meet the needs of growing population, changing dietary preferences, and biofuel production. West et al. (2010) estimated that land use conversion leads to emission of 120 Mg C ha^{-1} in the tropics compared with 63 Mg C ha^{-1} in temperate regions, and produces crop yield of 1.7 Mg ha^{-1} $year^{-1}$ in the tropics compared with 3.84 Mg ha^{-1} $year^{-1}$ in temperate regions. Thus, C emission from newly converted land in the tropics is about 3 Mg C Mg^{-1} of grain yield. An intensive forest management can mitigate climate change through C sequestration but can use extra water and nutrients. The increased soil water storage in forests may reduce flood risks in wet years, but could create conditions that also exacerbate drought (Ford et al. 2010a, b).

There are also tradeoffs between biofuel production and other services such as: (i) food production, (ii) plant nutrient, water and land requirements, (iii) loss of biodiversity (Gomiera et al. 2010). Soil biota, activity and species diversity, are important to ESs and land productivity (Barrios 2007). Thus, trade-offs of reduction in biodiversity can be optimized through strategic interventions (Faith et al. 2010). In this context, the concept of multifunctional landscapes has been proposed (O'Farrell and Anderson 2010). The latter are landscapes created and managed to integrate human production and landscape use into the ecologic fabric of a landscape maintaining essential ecosystem functions and services (O'Farrell and Anderson 2010). The forest diversity is also a strong driver of numerous ecosystem functions and services (Nadrowski et al. 2010), and should be preserved. Therefore, a range of indicators have been identified that provide information on biodiversity and ESs across biomes and spatial scales (Feld et al. 2009). Accordingly, agrobiodiversity is promoted as a supporting service for sustainable agricultural intensification (Omer et al. 2010).

Sustainable management of agroecosystems is critical to human well-being, especially in view of the projected increase in population from 7 billion in 2011 to about 9 billion in 2050 and the global increase in affluence and income. ESs provisioned by agroecosystems include food, feed, fiber, fuel and pharmaceuticals, industrial raw materials, regulations of soil and water, C sequestration, biodiversity etc. Yet, agroecosystems can also cause numerous disservices (e.g., erosion, non-point source pollution, GHG emissions). Thus, the strategy of sustainable management is to identify win-win scenarios (Power 2010) which optimize ESs and minimize disservices. For most questions of global significance, sustainable intensification of agroecosystems require local answers (McCune et al. 2011) in the context of site-specific soil types and socio-economic and political factors.

2.4.1 Wildlife Habitat/ Biodiversity and Food

The ACC has a strong impact on biodiversity (Montoya and Raffaelli 2010), and the terrestrial vegetation redistribution can influence C balance under climate change (Lucht et al. 2006). Six mechanisms which affect biodiversity by ACC include the following: (1) It leads to a consistent trend of northward or westward expansion of species range and altitudinal shifts (Walther et al. 2002; Walther 2010), (2) It induces spring advancement of phenology (Parmesan 2006), (3) The ACC reduces body size by influencing metabolism (Moran et al. 2010), (4) It exacerbates the habitat degradation, (5) It accentuates and enhances population of invasive plants and animal species, and, (6) It accentuate pests and pathogens, such as the bark beetle outbreak (Kaffa et al. 2008). The rapid rate of warming has elevated species extinction (Stork 2010; Traill et al. 2010; Brook et al. 2008). The ACC-induced drought and the frequency of wild fire can exacerbate land degradation (Dude 2007). Furthermore, the rate of change in ecosystem processes under ACC is much less pronounced under high levels of biodiversity (Montoya and Raffaelli 2010). The most vulnerable species are those which have small population and have specialized habitat niches. Abundant or dominant species, major controller of ecosystem functions, may be less affected by moderate level of ACC. Nonetheless, greater the diversity of functional groups the less is the likelihood of cascading species extinction (Mooney et al. 2009; Borrvall et al. 2000; Elmquist et al. 2003). Indeed, an ecosystem function is the most resilient to ACC where species diversity or key functional species groups are maintained. Principal mechanisms of change in biodiversity through ACC include: net primary production (NPP), nutrient cycling, ESs, life history strategy and survival, phenological responses, coloration, behavior, range, complementarily, pollination and seed dispersal, predator–prey interactions, competition, invasive, parasite and pathogen-host symbioses, and ecological competitiveness through ACC, etc. (Traill et al. 2010). These mechanisms, and thus the impact on ecosystem functions, may be altered by the land use (Polasky et al. 2010). Urbanization is an increasingly important land use, and the nature of rivers and wetlands systems is severely degraded by the urbanization process. Within an urban center, the theory of island biogeography is useful for biodiversity planning, and the size of an isolated greenery is important to species richness (Morimoto 2011). Rather than the effect on individual and species, community and ecosystem responses are also influenced (Walther 2010).

Loss of biodiversity and the wildlife habitat are among the major consequences of conversion of natural to managed ecosystems. Jeopardizing biodiversity and shrinking wildlife habitat through intensive agricultural land use are like navigating in fog and commanding a lighthouse to move out of the way (Heberlein 1991). Peyton (2000) posed a serious question by stating whether we are “cropping to manage (wildlife/biodiversity) or managing to crop”? With a world population of seven billion and projected to increase strongly by 2050, the priorities for food production and preservation of biodiversity and other ESs must be objectively considered.

In the same context, wetland management also requires careful consideration. In general, wetlands are a net sink of GHGs, and mangrove ecosystems may have the highest rate of C sequestration because of a high biomass production and salinity-inhibited low CH_4 emission (Page and Dalal 2011). While drainage of some wetlands/peatland would increase agronomic productivity, it would also turn these lands into a major source. Thus, carbonization through inundation of drained lands would be a favorable strategy (Lal et al. 2012).

Table 2.2 Food and Feed *vs.* Biofuel Production (de Fraiture et al. 2008)

Crop	Global production of food and feed by 2030 (10^6 Mg)	Need to meet biofuel demand (10^6 Mg)	% Increase to meet biofuel demand
Maize	890	177	20
Sugarcane	2,136	525	25
Rapeseed (*Brassica napus* L.)	64	51	80

2.4.2 Biofuel and Ecosystem Services

Energy production through biofuels has a wide range of effects on ESs including agronomic production, and water and nutrient demands. Thus, there is a strong need to evaluate the food demand—biofuel production—climate change nexus. Change in land use for meeting energy demand is a useful strategy for the provision of fuel and moderating climate, but it compromises other ESs namely freshwater resources, plant nutrients, wildlife habitat and food. However, little is known about the processes by which ESs are produced or compromised by diverting land resources to different generations of biofuels. Thus, it is important to understand the drivers, impacts and tradeoffs of biofuels (Gasparatos et al. 2011). Similar to ethanol, a critical appraisal is also needed for biodiesel production because not all biodiesel sources make ecologic sense (Nogueira 2011). While providing many benefits, the rapidly growing bioeconomy can drastically alter the configuration of agricultural landscape (Raghu et al. 2011) with numerous but less understood ecologic consequences especially with regards to soil resources. Thus, there is a strong need for multi-dimensional and cross-disciplinary assessment of biofuel production (Raghu et al. 2011). Some effects on landscape ecological processes include those related to land productivity, surface cover, albedo, water/energy/nutrient fluxes, and micro-meso climate (Dale et al. 2011). Over and above the biophysical effects, also important are economic, social and human dimensions, especially those related to food demand and supply.

The 2007–2008 global food crisis caused by high prices has been linked by some to biofuel production (Timilsina and Shrestha 2011). While the non-food based (cellulosic or second generation) biofuel may have lesser impact than food-based (grain-based ethanol or biodiesel), competition for land, water and nutrients can aggravate the food supply. The strong link between food security and biofuel (Brandt 2011) cannot be ignored.

For example, removal of corn (*Zea mays* L.) stover for cellulosic ethanol or co-combustion can adversely impact soil quality, exacerbate erosion and runoff, and increase risks of non-point source pollution (Karlen et al. 2011; Blanco-Canqui and Lal 2008). De Fraiture et al. (2008) estimated that increase in production of food and feed to meet biofuel demand by 2030 will be 20 % for corn (890×10^6 Mg *vs.* 177×10^6 Mg) and 25 % for sugarcane (*Saccharum* spp.) (64×10^6 Mg *vs.* 51×10^6 Mg) (Table 2.2). Thus, there is a strong competition between food and biofuel, and the food-energy-environment trilemma cannot be ignored (Tilman et al. 2009). While

Table 2.3 Energy use and greenhouse gas emission per kg of food in Sweden (Gonzalez et al. 2011)

Food	Energy (MJ kg^{-1})	GHG (kg CO_2-Eq. kg^{-1})	Protein (g kg^{-1} CO_2-Eq.)
Beef	47	29	7.1
Mutton/Lamb (*Ovies aries* Linnaeus, 1758)	46	26	7.6
Pork (*Sus scrofa domesticus* Erxleben, 1777)	28	8.2	25
Chicken (*Gallus gallus domesticus* Linnaeus, 1758)	27	4.7	39
Fish	40	3.1	67
Eggs	14	3.0	42
Milk	3.0	1.0	31
Cheese	38	8.8	28
Beans	5.1	0.9	246
Wheat (*Triticum* spp.)	3.9	0.6	192
Potatoes (*Solanum tuberosum* L.)	1.8	0.2	89

forest protection combined with large-scale cultivation of dedicated bioenergy can meet the energy demand of 100 EJ (1 EJ = 10^{18} J) by 2055 and 300 EJ by 2095, it can also increase global food prices and exacerbate water scarcity (Popp et al. 2011).

Global food and energy demands are also linked with dietary preferences. The energy use (MJ kg^{-1}) in Sweden is 46–47 for beef and lamb (*Ovies aries* Linnaeus, 1758), and 2–5 for potatoes (*Solanum tuberosum* L.) and beans (Table 2.3). Thus, total emission of GHGs may be increased because of change in dietary preferences to meat-based diet (Fiala 2008).

Competition for the scarce water resources also necessitates a serious consideration. Soil quality, and the retention of crop residues on its surface, are important to the quantity and quality of fresh water resources. Intensification of bioenergy crops with a high water demand may have far reaching consequences on water resources and energy feedbacks. The biofuel production of 35 billion liter in 2008, which was merely 2.9 % of the 1,200 billion liter of annual gasoline consumption worldwide, is already causing a lot of strain on water resources (de Fraiture et al. 2008).

Stone et al. (2010) estimated that sugarcane and corn require 458 and 2,036 m^3 water m^{-3} of ethanol produced, respectively. Further, the water requirement for corn grain production to meet US-DOE Billion-Ton Vision may increase sixfold from 8.6 to 50.1 km^3 (Table 2.4) The water need for crop production may be exacerbated by the projected climate change and especially because of the associated extreme events. Water consumption is a major concern, and can off-set the benefits of biofuel use (Emmenegger et al. 2011). The concern for water use may be a serious constraint in populous countries such as China and India.

Yet, the principal goal of biofuels (i.e., abating the climate change) also requires an objective analysis. The impact of growing population on the energy equation cannot be ignored (Zuckerman 1992; Weisz 2004). The question is whether biofuels can be a solution to the complex issue of climate change? (Khanna et al. 2011). Indeed,

Table 2.4 Water requirements for biofuel production (Adapted from Stone et al. 2010)

				Water requirement	
	Feed stock	Type	Fuel	$m^3\ Mg^{-1}$ fuel	$m^3\ GJ^{-1}$
I	First generation				
	Corn (*Zea mays* L.) (World)	Grain	Ethanol	4.45	4.41
	Corn (Nebraska)	Grain	Ethanol	3.39	3.36
	Sorghum [*Sorghum bicolor* (L.) Moench]	Grain	Ethanol	16.3	16.1
	Sugarcane (*Saccharum* spp.) (World)	Sugar	Ethanol	1	1
	Sweet sorghum	Sugar/juice	Ethanol	1.61	1.59
	Soybean [*Glycine max* (L.) Merr.]	Grain	Biodiesel	16.9	11.8
	Canola (*Brassica* spp.)	Grain	Biodiesel	8.48	5.91
II	Second generation				
	Corn	Stover	Ethanol	4.25	4.18
	Switchgrass (*Panicum virgatum* L.)	Biomass	Ethanol	3.41	3.36
III	Mixed				
	Corn	Stover and grain	Ethanol	1.88	1.86

the GHGs from land use change may off-set any benefits of biofuels because of the large “carbon debt” (Searchinger et al. 2008; Melillo et al. 2009). The “carbon debt” is also high when the Conservation Reserve Program (CRP) grassland in the U.S. is converted to bioenergy production (Gelfand et al. 2011). Even the production of sugarcane has direct impacts on local climate (Loarie et al. 2011), which off-set the benefits of energy saving. There are direct climate effects of perennial bioenergy crops in the U.S. also (Georgescu et al. 2011). A complete life cycle assessment of the biofuel production process (Sanz Requena et al. 2011), and the uncertainties involved must be addressed (Soimakallio and Koponen 2011) over and above the CO_2 emissions from biomass combustion and for bioenergy (Cherubini et al. 2011), release of nitrogen oxides, sulfur dioxide and particulate matter must also be considered (Miller and Gage 2011; Bessou et al. 2011).

Given the availability of land area along with water and plant nutrients for production of biofuel feedstocks, it is possible to achieve the goal of a global 10 % biofuel share in the transport sector by 2030, and contribute to lowering of GHG emissions by up to 1 Pg CO_2 eq year^{-1} (Bessou et al. 2011). For this goal to be realized, however, harmonized policies must be in place to ensure that sustainability criteria are globally implemented. Nonetheless, consumption of fossil fuel must also be reduced.

2.5 Payment for Ecosystem Services

The value of the world’s ecosystems is large (Costanza et al. 1997) because of the provision of numerous ESs generated. Thus, adoption of BMPs must be promoted through payments of land managers for provision to ESs of global relevance.

Table 2.5 Ecosystem services and law of nothingness

The law or concept	Implications
1. Nothing is appropriated	There are always tradeoffs (give and take)
2. Nothing is permanent	Everything is in a dynamic equilibrium and a transient state
3. Nothing is absolute	All processes, properties and values are relative to a baseline
4. Nothing is a panacea	There is no silver bullet, there is a multitude/menu of options
5. Nothing is universal	Soil/site/region specificity is an important consideration which cannot be overlooked
6. Nothing tangible is free	Under valuing a commodity leads to "Tragedy of the Commons"
7. Nothing is empty (vacuum) in nature	All space is occupied, pores in solid rock contain water or air and injecting something (liquid CO_2), fracking solutions can create shock waves
8. Nothing is given or for granted	It is the judicious use and management which produce goods and services
9. Nothing is a waste	Everything in nature has a use
10. Nothing comes from nothing (Ex nihilo nihil fit)	There is no such thing as a free lunch

Agroecosystems cover between 28 and 37 % of Earth's land surface (Porter et al. 2009). Thus, they have a large potential to influence economic goods and functions at global scales (Porter et al. 2009).

There are several approaches to facilitate payments for ESs (Kroeger and Casey 2007). Important among these are legal tools (e.g., liability laws, property rights), economic incentives (e.g., tax reductions or the levying of taxes and fees), ethical issues (e.g., moral obligations), command and control approaches (e.g., product input, technology standards), and market-based tools. The latter include techniques which make soil quality enhancement financially attractive to the private sector. However, establishment of a just/rational price is critical to the success of the market-based approach. Undervaluing a system can lead to its abuse and degradation. Watanabe and Ortega (2011) evaluated the prices for biogeochemical flows by considering their energy content and converting it to equivalent monetary terms. The monetary value thus computed is useful information to policy makers involved in creating mechanisms for payments. In addition to establishing rational prices, institutional capacity (e.g., setting and functions) are also essential pre-requisites for payments for ESs (Kumar 2011).

2.6 Conclusions

The concepts presented in this chapter can be appropriately summarized by ideas outlined in Table 2.1. Choice of a judicious land use and management (soil, water, energy) can improve ESs while preserving or enhancing the quality and function of the natural resources base. Technological options and strategies of sustainable management depend on eco region, soil type and the land use (i.e., cropland, pasture land, forest land, coastal ecosystem). There are also tradeoffs or disservices caused by any unilateral strategy of enhancing a specific good or service (e.g., food *vs.* biodiversity, agricultural intensification *vs.* erosion).

Such disservices or tradeoffs can be minimized by the strategy of: (i) replacing what is removed, (ii) responding wisely to what is changed and, (iii) predicting what will happen from anthropogenic and natural perturbations. Without adopting these strategies of sustainable management aimed at advancing soil security is akin to navigating in fog and commanding a lighthouse to move out of the way. Management of natural resources for optimizing ESs requires a critical and an objective evaluation of the law of nothingness (Table 2.5). Of the ten laws or concepts listed, the most relevant is "*ex nihilo nihil fit*" or nothing comes from nothing.

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Chapter 3
Ecosystem Carbon Sequestration

Klaus Lorenz

Contents

Abstract Terrestrial ecosystems take up carbon (C) from the atmospheric carbon dioxide (CO_2) pool during photosynthesis, and return the majority of it back by combustion or the respiration of plants, animals and microorganisms. Some of the recently fixed C remains in terrestrial ecosystems in stabilized forms in biomass and soil after conversion to inert, long-lived, C-containing materials by biologically mediated processes (biosequestration). To slow or reverse the increase in the atmospheric CO_2 concentration an additional transfer of C into terrestrial C pools is needed, i.e., by C sequestration. The terrestrial C sink was about 2.6 Pg C (1 Pg = 10^{15} g) in 2010 but has a high interannual variability. However, world forests play a critical role, and are responsible for about half of the total terrestrial gross primary production (GPP) of 123 Pg C year^{-1}. Only between 0.3 and 5.0 Pg C year^{-1} remains as net biome production (NBP) in terrestrial ecosystems with the major long-lasting C gain by, rather poorly understood but important, soil organic carbon (SOC) stabilization. However, the terrestrial C sinks can be enhanced by soil and land-use management practices. The CO_2 mitigation potentials of croplands and grasslands may be about

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0.8 Mg C ha^{-1} year^{-1} and 0.2 Mg C ha^{-1} year^{-1}, respectively. The effects of forest management activities on C sequestration are also important but less well known. Enhancing the photosynthetic efficiency and increasing root C inputs particularly at deeper soil depths are potential approaches for terrestrial C sequestration. However, C sequestration is also a critical component of agricultural and forest ecosystem services (ESs) but unintentional consequences of management for C sequestration on ESs are not known. Thus, long-term agricultural and forest management experiments are needed to identify soil and land-use management practices aimed at enhancing C sequestration and ESs.

Keywords Ecosystem services • Carbon sequestration • Net biome production • Climate change mitigation • Agricultural ecosystems • Forest ecosystems • Soil and land-use management

Abbreviations

BVOC	Biogenic volatile organic carbon
CAM	Crassulacean acid metabolism
DIC	Dissolved inorganic carbon
DOC	Dissolved organic carbon
DOE	Department of Energy
ESs	Ecosystem service(s)
GPP	Gross primary production
IPCC	Intergovernmental Panel on Climate Change
MEA	Millennium Ecosystem Assessment
NBP	Net biome production
NECB	Net ecosystem carbon balance
NEE	Net ecosystem exchange
NPP	Net primary production
OM	Organic matter
PC	Particulate carbon
Pg	Petagram
R_a	Autotrophic respiration
SIC	Soil inorganic carbon
SOC	Soil organic carbon

3.1 Introduction

An ‘ecosystem’ can be defined as a “unit of biological organization made up of all of the organisms in a given area interacting with the physical environment so that a flow of energy leads to characteristic trophic structure and material cycles within the system” (Odum 1969). Among key ecosystem processes is the flux or flow of energy and materials from one pool or quantity to another (Chapin et al. 2002). Energy is entering ecosystems when atmospheric carbon dioxide (CO_2) is reduced to form organic carbon (C) compounds during photosynthesis, a process driven by solar energy. Both energy and organic C are closely linked as they move through ecosystems. Thus, energy is lost from ecosystems when organic C is oxidized to CO_2, and returned back to the atmospheric pool by combustion or the respiration of plants, animals and microorganisms (Chapin et al. 2002).

The short-term C cycle describes the continuous flow of large amounts of C among the oceans, the terrestrial biosphere and the atmosphere (Denman et al. 2007). This cycle controls the atmospheric concentrations of methane (CH_4) and CO_2 on short timescales whereas geological forces regulate the CH_4 and CO_2 concentrations over geological timescales (Berner 2003). Ecosystems contribute to the regulation of atmospheric concentrations by C fluxes involving biotic components. Thus, ecosystems also contribute to the regulation of Earth's climate as temperature and the C content of the atmosphere are correlated over geological time scales (Chapin et al. 2002). However, the basic understanding of the myriad of tightly interlinked biological processes that drive the global C cycle remains limited (U.S. DOE 2008).

Some of the recently fixed organic C in ecosystems is not rapidly returned to the atmosphere by respiration but remains in stabilized forms such as biomass, soil and deep ocean sediments. The biologically mediated uptake and conversion of CO_2 to inert, long-lived, C-containing materials is called biosequestration (U.S. DOE 2008). Biosequestration temporarily removes C from active cycling. More generally, C sequestration can be defined as the uptake of C-containing substances and, in particular, CO_2 into another reservoir with a longer residence time (IPCC 2007). Thus, any increase in the C content of a reservoir in an ecosystem might be referred to as sequestration as C is held in the reservoir and separated from other parts of the ecosystem (Powlson et al. 2011). However, it has become customary for the term C sequestration to imply a contribution to climate change mitigation. For this reason, C sequestration must slow or even reverse the increase in atmospheric concentration of CO_2. Thus, movement of C from one reservoir in the ecosystem to another should be appropriately termed accumulation whereas an additional transfer of C from the atmosphere into a reservoir should be termed sequestration as this process is a genuine contribution to climate change mitigation (Powlson et al. 2011).

Closely connected to the C flow within ecosystems are the goods and services which ecosystems provide to society and support nature's functions (U.S. DOE 2008). Goods include food, feed, fibre, fuel, pharmaceutical products, and wildlife. Ecosystem services (ESs) include maintenance of hydrologic cycles, cleansing (filtering) of water and air, regulation of climate and weather, storage and cycling of nutrients, provision of habitat, and aesthetics (U.S. DOE 2008). There are several classification schemes but no commonly accepted meaningful and consistent definition of ESs exist (Fisher et al. 2009). One of the most widely used is the classification of supporting, regulating, provisioning and cultural ESs (MEA 2005). The MEA defined ESs as benefits people obtain from ecosystems. As C sequestration affects the flow of C into and within ecosystems it may become critical component of ESs (U.S. DOE 2008). The focus of this chapter is to discuss C sequestration processes in terrestrial ecosystems in biomass and soil, and to elucidate how existing agricultural and forest ecosystems can be managed to enhance C sequestration. Finally, consequences of enhanced C sequestration on terrestrial ESs will also be explained. How soil microorganisms may be directly manipulated to enhance C sequestration is discussed elsewhere (e.g., King 2011).

3.2 Carbon Sequestration in Terrestrial Ecosystems

The principal 'reservoirs' in terrestrial ecosystems that can take up CO_2 and other C-containing compounds are animals, microorganisms, plants, soil, and water (Chapin et al. 2002). Terrestrial plants contain about 650 Pg C (1 Pg = 10^{15} g) and soils 2,300 Pg C (Table 3.1; Field and Raupach 2004). Soil microorganisms are estimated to contain 110 Pg C (Jansson et al. 2010). Forests hold 70–90 % of terrestrial above and below ground biomass. However, the knowledge on the amount of terrestrial biomass and, thus, biomass C stock is based almost entirely on ground measurements over an extremely small and possibly biased sample pool, with many regions still unmeasured (Feldpausch et al. 2012; Houghton et al. 2009). For example, plant roots may contain up to 280 Pg C which is significantly higher than the previous estimate of 160 Pg C (Robinson 2007). Even less certain is the C storage in nonliving components (Pan et al. 2011). For example, about 1,700 and 1,600 Pg C may be stored in soil to 1-m depth as soil inorganic carbon (SIC) and soil organic carbon (SOC), respectively (Eswaran et al. 2000; Jobbágy and Jackson 2000). Furthermore, permafrost and peatlands may contain an additional 2,300 Pg C (Tarnocai et al. 2009; Yu 2011; Jungkunst et al. 2012). Not well quantified is the C storage in human settlements including soil, vegetation, landfills and buildings. Carbon densities in human settlements in the conterminous United States may be as high as those of tropical forests but human settlements are unlikely to become net C sinks (Churkina et al. 2010).

Carbon sinks in the oceans and the terrestrial biosphere provide an extremely important ES (Raupach 2011). Since 1959, natural C sinks removed about half of

Table 3.1 Estimates for terrestrial ecosystem carbon storage (Pg C)

Storage component		Pg C	References
Plants	Total	650	Field and Raupach (2004)
	Forests	360	Pan et al. (2011)
Plant roots	Total	280	Robinson (2007)
	Forests	200	Robinson (2007)
Soil microorganisms	Total	110	Jansson et al. (2010)
Soil organic carbon	1-m soil depth	1,600	Jobbágy and Jackson (2000)
	3-m soil depth	2,300	Jobbágy and Jackson (2000)
Soil inorganic carbon	1-m soil depth	1,700	Eswaran et al. (2000)
Permafrost	Total	1,700	Tarnocai et al. (2009)
Peatlands	Total	600	Yu (2011)

Table 3.2 Estimates for terrestrial ecosystem carbon fluxes and sinks (Pg C year^{-1})

Flux/sink		Pg C year^{-1}	References
Gross primary production	Total	123	Beer et al. (2010)
	Forests	59	Beer et al. (2010)
Net primary production	Total	63	Roy et al. (2001)
	Forests	33	Roy et al. (2001)
Net biome production	Total	0.3–5.0	Jansson et al. (2010)
Carbon sink	Total	1.7–3.4	Canadell et al. (2007), Denman et al. (2007), Khatiwala et al. (2009), Le Quéré et al. (2009)
	Forests	2.5	Pan et al. (2011)

anthropogenic CO_2 emissions from the atmosphere and this uptake has doubled during the past 50 years (Ballantyne et al. 2012; Denman et al. 2007). However, the C fractions absorbed by the oceans and the terrestrial biosphere remain unclear (Levin 2012). Over the past 30 years, the net terrestrial C sink has increased likely as a result of both a decrease in emissions due to land-use change and an increase in the amount of CO_2 stored by the terrestrial biosphere (Gurney and Eckels 2011). However, the terrestrial sink can be highly variable among years in response to volcanic eruptions, climate fluctuations, and many different process that vary across biomes and through the cycling of the seasons (Ballantyne et al. 2012; Raupach 2011). For example, the terrestrial sink was 1.7 and 2.6 Pg C year^{-1} for the decade of the 1980s and 1990s, respectively (Table 3.2; Denman et al. 2007). Further, in 2010 the terrestrial C sink was more than 1 Pg C below the strength of the sink over the two previous years (Peters et al. 2011). A significant fraction of the global terrestrial C sink is supposedly located in the northern hemisphere (Raupach 2011; Stephens et al. 2007), and, in particular, in forests recovering from past disturbances. For the period 1990–2007, the entire terrestrial C sink may be accounted for by C uptake of global established forests (Pan et al. 2011). Thus, forests play a critical role as terrestrial C sinks. In contrast, nonforest ecosystems may have been collectively neither a major (>1 Pg) C sink nor a major source over the period 1990–2007 (Pan et al. 2011).

The term net ecosystem carbon balance (NECB) was proposed to describe the net rate of C accumulation in (positive sign) or loss (negative sign) from an ecosystem (Chapin et al. 2006). The NECB represents the overall ecosystem C balance from all sources and sinks – physical, biological, and anthropogenic, and is calculated as follows: $NECB = -NEE + F_{CO} + F_{CH_4} + F_{BVOC} + F_{DIC} + F_{DOC} + F_{PC}$. The net ecosystem exchange (NEE) is the net CO_2 exchange between the ecosystem and the atmosphere. Where, F_{CO} is the net carbon monoxide (CO) exchange and F_{CH_4} the net CH_4 exchange between the ecosystem and the atmosphere. The net exchange of biogenic volatile organic carbon (BVOC) between the ecosystem and the atmosphere is F_{BVOC}. Net exchanges of dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC) between the ecosystem and the atmosphere are F_{DIC} and F_{DOC}, respectively. The term F_{PC} is the net particulate (non-dissolved, non-gaseous) carbon (PC) exchange between the ecosystem and the atmosphere. It includes processes such as animal movement, soot emission during fires, water and wind deposition and erosion, and anthropogenic transport or harvest (Chapin et al. 2006).

When integrated over space and time, the NECB equals the net biome production (NBP). This is the amount of organic matter (OM) in a biome minus C losses or gains from disturbances such as fire, biotic stresses, and human land use (U.S. DOE 2008). Disturbances, in particular, can strongly influence C flows and stocks of ecosystems. For example, harvesting terrestrial ecosystems in geographic Europe exceeds NBP by threefold (Schulze et al. 2010). These disturbance-induced C losses and gains balance out in the long-term. However, changes in disturbance regimes, i.e., in frequency and/or severity can have profound effects on ecosystem C storage and dynamics (Weng et al. 2012).

The OM or NBP remaining in an ecosystem represents the long-term C biosequestration. Global annual values of NBP are currently about 3 Pg C but have varied considerably during the last decades between 0.3 and 5.0 Pg C (Jansson et al. 2010). Critical to biosequestration are interactions between plants and microbial species in soils (U.S. DOE 2008). As the biomass sink of wood in forests is particular vulnerable to harvest, the only long-lasting C gain occurs in soils of terrestrial ecosystems and in wetland soils (Brevik 2012; Schulze et al. 2010). Thus, the C storage in soils is a key component of NBP and biosequestration. However, anthropogenic activities may have strong effects on NBP. For example, by harvesting, more C is extracted from the natural C cycle of forest, grassland and cropland ecosystems in Europe than is remaining in the soil (Schulze et al. 2010).

3.2.1 Biomass Carbon Sequestration

Terrestrial plants absorb CO_2 as organic compounds during photosynthesis, a C flux also known as gross primary production (GPP) at the ecosystem level. Terrestrial GPP is the annual photosynthetic C uptake of all leaves over an area of land and is the largest C flux between Earth and the atmosphere (U.S. DOE 2008). Further, cryptogamic organisms such as cyanobacteria, algae, fungi, lichens and bryophytes on many terrestrial surfaces including soils, rocks and plants fix also atmospheric CO_2 and produce C-containing organic compounds (Elbert et al. 2012). The observation-based estimate of terrestrial GPP is 123 Pg C year^{-1} using eddy-covariance data and various diagnostic models (Beer et al. 2010). Tropical forests, and tropical savannahs and grasslands account for 60 % of this flux (40.8 and 31.3 Pg C year^{-1}, respectively). Croplands, temperate forests, temperate grasslands and shrublands, boreal forests, and deserts account for 14.8, 9.9, 8.5, 8.3 and 6.4 Pg C year^{-1}, respectively (Beer et al. 2010).

Photosynthesis takes place in plant chloroplasts and is driven by the availability of solar radiation and CO_2, by temperature and the availability of nitrogen (N) which is required to produce photosynthetic enzymes (Chapin et al. 2002). Differences in annual GPP among ecosystems depend primarily on the quantity of leaf area and the duration of its photosynthetic activity. Both, leaf area and photosynthetic season, are determined by the availability of soil resources (water and nutrients), climate, and time since disturbance (Chapin et al. 2002). Specifically, water availability has a large effect on GPP over 40 % of the vegetated land, and up to 70 % in savannahs, shrublands, grasslands, and agricultural areas (Beer et al. 2010).

The terrestrial GPP drives several ecosystem functions and, in particular, the uptake of C in growing plant biomass. The organic C produced by photosynthesis is either (i) incorporated into biomass or (ii) respired and released as CO_2 and water during energy generation for maintenance and ion uptake (Chapin et al. 2002). Thus, photosynthesis and respiration should be closely related to total leaf area (Mori et al. 2010). However, while the photosynthetic metabolism is comprehensively understood, basic information on key determinants of respiratory rates in photosynthetic and non-photosynthetic plant organs aboveground and belowground is scanty (Atkin et al. 2010).

The CO_2 uptake during photosynthesis is only temporary as about 25–70 % of the captured C is returned almost immediately to the atmosphere by autotrophic respiration (R_a) by plants (Lambers et al. 2005). Thus, total plant respiration may not be a relatively constant fraction of GPP as previously suggested (Zhang et al. 2009). Also, the percentage of captured C lost by R_a is variable among terrestrial ecosystems as it depends on the main climatic characteristics such as temperature and precipitation, and geographical factors such as latitude and altitude (Zhang et al. 2009). The rate of primary production that remains after accounting for losses through R_a is the net primary production (NPP) which is calculated as $NPP = GPP - R_a$. Growing biomass comprises NPP or the total amount of OM annually photosynthesized (U.S. DOE 2008). The variation in plant productivity may be controlled by (i) actual evapotranspiration, (ii) nutrient [N, phosphorus (P), potassium, calcium, sulfur (S), magnesium, silicon and other micronutrients] limitation, (iii) disturbance, (iv) water use efficiency, and (v) light use efficiency (Schulze 2006; Vitousek and Howarth 1991). For example, global average reduction in terrestrial plant productivity by nutrient limitation may be about 16 % (Fisher et al. 2012).

Partitioning and processing distributes OM to heterotrophs, and its subsequent assimilation and respiration eventually returns most of the remaining biomass C to the atmosphere. However, not all of NPP remains in terrestrial ecosystems as, for example, in the year 2000 humans appropriated 23.8 % of global terrestrial NPP, mostly above ground NPP (Haberl et al. 2007). Thus, only 3.7 % of potential vegetation NPP was available for heterotrophic food chains. The partitioning of GPP among NPP or biomass and R_a at the ecosystem scale remains poorly understood (Vicca et al. 2012). However, the NPP/GPP ratio among terrestrial ecosystems is not constant as suggested previously but lower for forests than for herbaceous and shrub ecosystems (Zhang et al. 2009). Further, dense or open ecosystems have lower NPP/GPP ratio than sparse or closed ecosystems, and evergreen ecosystems have lower NPP/GPP ratio than deciduous ecosystems. The NPP/GPP ratio increases with increasing altitude and with growing temperature for 60 % of land, and decreases with enhanced precipitation and with growing latitude in the Southern Hemisphere (Zhang et al. 2009).

The major components of NPP are new plant biomass (40–70 % NPP), root secretions (20–40 % NPP), losses due to herbivores and mortality (1–40 %), and volatile emissions (0–5 % NPP; Chapin et al. 2002). However, these data are uncertain as all components have never been measured in a single study and, depending on the ecosystem, some NPP components are difficult to measure or are of minor importance (Ciais et al. 2010b). For example, rarely measured are losses of NPP that occur during weed and seed production, emission of BVOCs, exudation from roots, and C transfer to root symbionts. Furthermore, not all of NPP remains in an ecosystem as some plant biomass may be lost by harvest and due to herbivores (Ciais et al. 2010b). Specifically, managed ecosystems such as agricultural ecosystems lose substantial portions of C by harvest. For example, up to 50 % of the aboveground dry mass is removed by harvesting cereal croplands and up to 40 % of total NPP is removed from grasslands by cutting (Ciais et al. 2010a; Johnson et al. 2006). Total harvest takes about 30 % of NPP from terrestrial ecosystems in Europe (Schulze et al. 2010). Globally, human biomass harvest alone is about 12 % of total potential vegetation NPP or 20 % of aboveground potential vegetation NPP (Haberl et al. 2007). Agriculture, in particular, is responsible for 78 % of global human appropriation of NPP (Haberl et al. 2007).

Herbaceous plants with extensive root systems and particularly perennial trees store large amounts of organic C in above- and belowground biomass (Jansson et al. 2010). Forestry is responsible for about 11 % of global NPP appropriated by humans (Haberl et al. 2007). In contrast to agricultural ecosystems, biomass C sequestration in forests is less affected by human disturbance as only a fraction of the total forest biomass growth is being harvested annually. In Europe, total woody biomass is increasing and forests currently accumulate C (70 % of total NBP; Schulze et al. 2009). Over decades to centuries forest ecosystems in Europe have been sequestering C in aboveground biomass in contrast to other land-use types (Schulze et al. 2010). Global forests contain currently 363 ± 28 Pg C in living biomass (above and below ground; Pan et al. 2011). The average above ground living biomass ranges from 59 Mg C ha^{-1} for boreal dry forests to 377 Mg C ha^{-1} for cool temperate moist forests (summarized in Keith et al. 2009). The highest known C density in the world was reported for Mountain Ash [*Eucalyptus regnans* (F. Muell.)] in a moist temperate region of Australia with an average of 1,053 Mg C ha^{-1} in living above ground biomass. Thus, factors that account for high biomass C densities in forests are (i) relatively cool temperatures and moderately high precipitation producing rates of fast wood growth, and (ii) older forests that are often multiaged and multilayered and have experienced minimal human disturbance (Keith et al. 2009). Specifically, pristine old-growth forests can continue to sequester C in living biomass for several centuries (Lewis et al. 2009; Luyssaert et al. 2008). However, forest biomass C is vulnerable to major natural disturbances such as large and intense wildfires, wind-throw, hurricanes, herbivore outbreaks, landslides, floods, glacial advances, and volcanic eruptions that can replace the forest stand. Furthermore, biomass C sequestration may weaken by climate change but there is no consensus on the response of the forest C cycle to climate change (Bonan 2008; Lorenz and Lal 2010).

3.2.2 Soil Organic Carbon Sequestration

A fraction of the C fixed by photosynthesis in ecosystems transferred below ground has the potential to persist for millennia in the SOC pool. However, the scientific knowledge about quantitative changes in the SOC pool over time is rather limited. Top-down modeling of NBP and observation of C exchange between the atmosphere and the biosphere are mostly used to predict NBP_{soil} or sequestration of soil C (Schulze et al. 2010). Accordingly, European grasslands sequester more C in soils than forests and forests sequester amounts similar to those in peatlands. In contrast, European cropland soils appear to be currently a small net C source in relation to the atmosphere but a verification of this flux through direct observations is needed (Ciais et al. 2010b).

The SOC sequestration depends primarily on the C input and soil stabilization processes. Plant root and rhizosphere inputs, in particular, make a large contribution to SOC (Schmidt et al. 2011). However, the link between plant litter quality and SOC is not well understood (Torn et al. 2009). Accumulation of SOC is mainly the result of partial degradation, microbial products and fire residues rather than humic substances. Physical disconnection (e.g., from enzymes, decomposers, e-acceptors), sorption/desorption (organo-mineral associations) and freezing/thawing govern SOC cycling and this process is shaped by environmental conditions (Schmidt et al. 2011).

Some surface residue C may be incorporated into the mineral soil by physical mixing and solubilisation, transport and subsequent adsorption (Lorenz and Lal 2005). However, plant roots (i.e., litter, rhizodeposition) are the primary vector for most C entering the SOC pool (Rasse et al. 2005). The relative importance of root litter and rhizodeposition *vs.* other incorporation processes for profile SOC distribution and dynamics depend on climate, soil and vegetation types (Rumpel and Kögel-Knabner 2011). Growing roots are major sinks of recently fixed C as up to 50 % of NPP is allocated to roots (Chapin et al. 2002). Roots in tropical forests may contain up to 123 Pg C, whereas tropical savanna and grassland roots may contain up to 112 Pg C (Robinson 2007). Temperate and boreal forests may contain up to 49 and 25 Pg C, respectively, stored in roots. Mediterranean shrublands, deserts, tundra, and crops may contain up to 15, 12, 3, and 1 Pg C of root C, respectively (Robinson 2007). However, inventory data on root biomass are uncertain due to spatial and temporal heterogeneity, uneven sampling and methodological differences among studies. Direct measurements of complete root profiles of forest trees down to the maximum rooting depth are scanty (Schenk and Jackson 2005). Thus, estimates of the forest root C pool, and of below ground C inputs from plant root litter to SOC are uncertain (Denef and Six 2006).

Rhizodeposition describes the release of organic C compounds by roots. About 5 % of the net fixed C can be recovered from soil as rhizodeposition and a small amount of the C partitioned below ground is lost by leaching (Jones et al. 2009). However, most isotopic labeling studies used to quantify the amount of photosynthate partitioned below ground have focused on young plants at a vegetative stage but partitioning is strongly affected by plant age. Furthermore, almost half of the published data on rhizodeposition are for wheat (*Triticum* spp.) and ryegrass (*Lolium* spp.), and 76 % of the studies are related to only five crop/grassland species. Thus, the knowledge of C rhizodeposition and, in particular, those in mixed plant communities is scanty. The quantification of C sequestration in forest soils is particularly hampered by the fact that the amount of rhizodeposition by trees in natural ecosystems is virtually unknown (Jones et al. 2009).

Adsorption of DOC in soil profiles is another direct below ground C input but represents only a small portion of profile SOC as the majority of DOC is ultimately returned to the atmosphere as CO_2 (Bolan et al. 2011). Throughfall, stemflow, recently deposited litter and humus in forest soils, and recently deposited crop residues and application of organic amendments such as manure and biosolids are important DOC sources in arable lands. The DOC flux may represent up to 5 % of NPP, but from surface organic layers to mineral subsoil DOC concentrations decrease strongly by up to 90 % through microbial consumption and adsorption (Bolan et al. 2011). Retention of DOC in subsoils is related to concentration of poorly crystalline iron and aluminum (hydr)oxides with a high specific surface area. Some DOC may leach from soils into adjacent aquatic ecosystems (Bolan et al. 2011). The C leaching losses may be particularly important for the NECB of agricultural systems (Kindler et al. 2011).

3.2.3 Additional Transfer of Atmospheric Carbon into Terrestrial Ecosystems

For a genuine contribution to climate change mitigation, C sequestration in terrestrial ecosystems should represent an additional C input from the atmosphere and not only a movement of C among terrestrial ecosystem pools (Powlson et al. 2011). Mitigation of CO_2 emissions can be achieved by enhancing sinks through maximizing the land-surface uptake of C (Schulze et al. 2009). This could mean replacing established land-use and soil management with practices that result in an additional net transfer of C from the atmosphere into biomass and soil compared to the previous practices.

3.2.3.1 Agricultural Ecosystems

In the year 2000, croplands covered about 12 % and pastures about 22 % of Earth's ice-free land area, respectively (Ramankutty et al. 2008). Improving cropland and grazing land management, and restoring degraded lands and cultivated organic soils are among the examples for enhancing C removals from the atmosphere (Smith et al. 2008). Croplands are often intensively managed and offer many opportunities for withdrawing atmospheric CO_2. However, long-term agricultural experiments especially those involving land-use changes to test these practices are scanty and SOC measurements rarely replicated (Falloon and Smith 2003, 2009).

Mitigation practices include improved agronomic practices that increase yields and generate higher residue C inputs such as (i) using improved crop varieties, (ii) extending crop rotations, notably those with perennial crops which allocate more C below ground, (iii) avoiding or reducing use of bare (unplanted) fallow, (iv) adding more nutrients when deficient, and (v) providing temporary vegetative cover ('catch' or 'cover' crops) between agricultural crops (Freibauer et al. 2004). Reducing soil disturbance by switching to reduced- or no-till agriculture, and retaining crop residues can potentially result in a net SOC gain (Franzluebbers 2010). Expanding the irrigated cropland area or using more effective irrigation measures can enhance SOC storage in soils through enhanced yields and crop residue returns (Lal 2004). Drainage of croplands in humid regions can promote productivity and, thus, SOC gains. Mitigation by C sequestration into biomass can occur by switching to agroforestry as the standing C stock above ground in trees is usually higher than that of the equivalent land use without trees (Nair et al. 2010). Also, planting trees may increase SOC sequestration (Laganière et al. 2010). Replacing the cropland with a land use similar to the native vegetation often increases SOC (Don et al. 2011; Poeplau et al. 2011) as does converting drained cropland back to wetland (Smith et al. 2008).

On grazing land, C accrual on optimally grazed lands is often greater than on ungrazed or overgrazed lands (Smith et al. 2008). The C storage in grazing lands can be improved by measures that promote productivity (e.g., fertilizers, organic amendments, irrigation) resulting in larger plant litter returns and, thus, SOC storage (Conant 2011). Reducing the frequency or intensity of fires in grazing lands leads to increased shrub and tree cover ('woody plant encroachment') and higher landscape C density in biomass and soil. Introducing grass species with higher productivity or C allocation to deeper roots, and introducing legumes can promote SOC storage (Smith et al. 2008).

Avoiding the drainage or re-establishing a high water table of organic soils protects or re-establish the removal of atmospheric CO_2 and accumulation of SOC because of suppressed decomposition under flooded conditions (Freibauer et al. 2004). The SOC storage in degraded agricultural soils can partly be restored by switching to practices that reclaim productivity (Bruce et al. 1999). Measures include re-vegetation, improving fertility by nutrient amendments, applying organic substrates such as manures, biosolids and composts, reducing tillage and retaining crop residues and conserving water (Smith et al. 2008).

Among agricultural practices, set-aside and land-use changes in croplands have an estimated CO_2 mitigation potential of about 0.8 Mg C ha^{-1} year^{-1} each in cool-moist and in warm-moist climates (Smith et al. 2008). Management of grazing, fertilization and fire in grasslands of cool-moist and warm-moist climates has a potential of about 0.2 Mg C ha^{-1} year^{-1} each. Restoration of degraded lands has an estimated mitigation potential of about 0.9 Mg C ha^{-1} year^{-1} independently of the climate zone. Applications of manures/biosolids have an estimated potential of about 0.7 Mg C ha^{-1} year^{-1} in cool- and warm-moist climates, and of about 0.4 Mg C ha^{-1} year^{-1} in cool- and warm-dry climates, respectively. Switching to organic agriculture has a mitigation potential from soil C sequestration of 0.27 Mg C ha^{-1} year^{-1} (Gattinger et al. 2012). The global technical mitigation potential from reduced soil emissions of CO_2 from agriculture by 2030 is between 4,895 and 5,340 Mt CO_2-eq. year^{-1}, which could offset about 18 % of total annual CO_2 emissions (Smith et al. 2008). However, barriers to implementation of this significant biophysical potential for mitigation in agriculture including options that can be immediately implemented must be overcome by policy/economic incentives (Conant 2011). It is particularly important to educate farm/land managers about the options for additional removal of atmospheric CO_2 (Smith et al. 2007).

3.2.3.2 Forest Ecosystems

Forests cover about 30 % of Earth's ice-free land surface and about 55 % of this area is managed as production forests or used to extract multiple ESs (FAO 2010). Forest management can promote C sequestration (Fahey et al. 2010). However, in contrast to agricultural ecosystems the environmental impacts of forest management on C sequestration are less well known (Lorenz and Lal 2010). Little is known about the effects of specific forest management activities on biomass C and SOC storage (Birdsey et al. 2007; Jandl et al. 2007). Also, few controlled long-term experiments have been established to study the impact of various forestry and silvicultural practices on C sequestration over time (Patterson and Coelho 2009). Thus, only a few general remarks are given in the following section.

Forest management can contribute to an additional withdrawal of atmospheric CO_2 into the biosphere by replacing lower-C-density forests with higher-C-density forests (Birdsey et al. 2007). The overall biophysical potential of management activities (e.g., longer harvesting cycles, reduced disturbances, fire suppression, harvest exclusion) to increase forest C density can be substantial (Canadell and Raupach 2008). Higher global forest C sequestration can be achieved by forest management to increase C density by enhancing young forests and improving degraded forests (Rautiainen et al. 2011). Changes in treatment intensity may result in more C stored in the forest after which a new equilibrium sustainable forest C balance may be attained (Lippke et al. 2011). Older forests store more C but the rate at which they remove additional C from the atmosphere can be substantially lower compared to the C assimilation rate of younger forests. Thus, commercial forest management involves relatively short rotations to capture the period of fast growth while also enhancing absorption of C from the atmosphere by higher yielding trees and storing C in woody products (Lippke et al. 2011). However, longer harvesting cycles may allow a progressive accumulation of C which causes an increase in forest C density but very long rotation lengths do not necessarily maximize the total C balance (Canadell and Raupach 2008; Jandl et al. 2007). Furthermore, there is no direct link between SOC and changes in forest cover or changes in rotation (Lippke et al. 2011). Otherwise, increase in site productivity by fertilization of nutrient-deficient forest stands and retaining wood on site has the potential to increase SOC storage (Lal 2005). However, impacts of forest treatments on SOC are uncertain as often only forest floors and shallow soil layers are studied and not the entire soil profiles (Harrison et al. 2011).

Forest management can increase C pool sizes by increasing production rates, slowing decomposition loss, reducing the amount of material transferred out of the forest stand, and extending the period between disturbances/management activities (Birdsey et al. 2007). Also, the less severe the disturbance, the more C is stored in forests. For example, forest C could increase assuming continued successful fire suppression efforts in some forests (Lippke et al. 2011). Theoretically, maintaining the forest in the maximal stages of NECB or NBP, e.g., by managing for maximum tree stocking or by high-intensity silviculture, can maximize removal of atmospheric CO_2 (Fahey et al. 2010; Markewitz 2006). Examples of enhancing C sequestration in living biomass include practices aimed at increasing NPP such as fertilization, genetically improved trees that grow faster, and any management activity that enhances growth rate without causing a concomitant increase in decomposition (Birdsey et al. 2007). Providing sufficient nutrients to trees can enhance the preferential allocation of C to above ground plant parts (Vicca et al. 2012). Productive C-dense forest systems can be supported by reducing fire losses and increasing the interval between disturbances. The technical potential for C sequestration in forest biomass is large but the economic potential, i.e., the amount of C that could be sequestered given a specified C price is significantly less (Conant 2011).

3.2.3.3 Advanced Approaches for Ecosystem Carbon Sequestration

Aside from the practices discussed in the previous sections, an additional net transfer of C from the atmosphere into ecosystems can probably be achieved by improving photosynthetic incorporation of atmospheric CO_2 into plant biomass, and enhancing C allocation into cellular C pools with low turnover and into deep roots (Jansson et al. 2010). Examples of practices include (i) phyto-engineering and cultivating plants with enhanced net photosynthetic C uptake, (ii) maximizing the photosynthetic uptake per unit area by optimum mixtures of species employing C_3 and C_4 photosynthetic pathways, and (iii) enhancing SOC sequestration by increased root C inputs through phyto-engineering and cultivating of perennial plants instead of their annual relatives, and of plants with deep and bushy root systems.

The enzyme D-ribulose-1,5-bisphosphate carboxylase/oxygenase (Rubisco), i.e., form I Rubisco catalyzes the fixation of CO_2 (carboxylase activity) into organic product once CO_2 enters the plant chloroplast cell (Nisbet et al. 2012). Virtually all the organic C in the biosphere derives from the CO_2 this enzyme fixes from the atmosphere (Ellis 2010). However, in its oxygenase activity form I Rubisco captures also oxygen to initiate the photorespiration pathway. Up to 40 % of the C fixed by C_3 photosynthesis may be lost by photorespiration but less from C_4 and crassulacean acid metabolism (CAM) photosynthesis (Chapin et al. 2002). Furthermore, only a part of the accessible CO_2 is chosen by I Rubisco for incorporation into organic products. Thus, to increase plant productivity research is under way to increase the limited catalytic efficiency of Rubisco under current and forthcoming CO_2 concentrations (Liu et al. 2010; Spreitzer and Salvucci 2002). Cultivation of plants with more efficient Rubisco carboxylation activity may result in enhanced C sequestration.

Aside Rubisco, other opportunities exist to overcome inefficiencies in photosynthetic energy transduction in plants from light interception to carbohydrate synthesis for enhanced biomass production (Zhu et al. 2010). These include improving the display of leaves in plant canopies to avoid light saturation of individual leaves and enhancing photorespiratory bypass. Long-term opportunities are to maximize C gains without increasing plant and particularly crop inputs by molecular optimization of resource investment among the components of the photosynthetic apparatus. Collectively, improving photosynthetic efficiency has the potential to more than double the yield potential of major crops (Zhu et al. 2010), and cultivating them may enhance C sequestration. Optimizing the C_3/C_4 plant species mixture for a maximum vegetated area C uptake can also potentially result in more ecosystem C sequestration. Theoretically, in C_3 plants 4.6 % of solar radiation is used for biomass production but 6.0 % of solar radiation is used in C_4 plants at 30 °C leaf temperature and an atmospheric CO_2 concentration of 387 ppmv (Zhu et al. 2010). However, the overall productivity of C_3 plants may be higher than those of C_4 plants at lower temperatures as C_4 photosynthesis comes with an extra cost (Jansson et al. 2010). Thus, increasing the proportion of C_4 plants in warmer climates and optimizing the C_3/C_4 plant species mixture in colder climates may also cause an additional transfer of atmospheric CO_2 into terrestrial ecosystems.

Soil is the only long-lasting C store in terrestrial ecosystems. Thus, replacing traditional practices with SOC-accreting land-use and soil management enhances ecosystem C sequestration. Additional plant derived C should be delivered to deeper soil depths where it may eventually be securely stored by SOC stabilization (Lorenz and Lal 2005). However, it is not known which root properties influence rhizodeposition rates or stability, and the effects of additional root-C inputs on subsoil SOC decomposition (Mendez-Millan et al. 2010; Sanaullah et al. 2011). For example, compared to their annual relatives perennial crops are characterized by an increase in seasonal light interception efficiency, an important factor in plant productivity (Glover et al. 2010b). Perennial crops have greater root mass particularly at deeper depths and store more SOC compared with annual crops (Glover et al. 2010a). However, the potential for replacement of annual with perennial plants may depend on site-specific reliable regrowth over multiple years, adaptation to abiotic stresses such as water and nutrient deficiencies, and resistance to pest and diseases (Glover et al. 2010b).

Additional storage of atmospheric C in soil profiles at deeper depth may be possible by replacing shallow rooting plants with those having deeper and prolific root systems. Active agricultural invention is required for breeding crop plants with deeper and bushy root ecosystems, and for their establishment and cultivation (Kell 2011). This would also stimulate photosynthetic yields (Zhu et al. 2010). However, some deep rooting cultivars may produce less aboveground biomass yields but there is no evidence that this is true in general (Fisher et al. 1994). The introduction of tree species with deep and bushy root ecosystems may also contribute to enhanced SOC sequestration in managed forests. The strategy is to select tree species that better divert root-derived C into stable mineral-associated SOC. Most tree root systems can be classified having a tap, heart or flat root type (Persson 2002). Thus, SOC sequestration though afforestation depends on the tree species planted (Li et al.

2012). Furthermore, common garden experiments with mono-specific tree stands on similar soil indicate that tree species contribute to the variation in SOC dynamics (Vesterdal et al. 2012; Hobbie et al. 2007). However, the ability to sequester C may depend not only on the amount, extent and turnover of plant roots, but also on the production rate and nature of rhizodeposits, the amount of N and other nutrients, and biophysical properties such as moisture content and compaction (Kell 2012). A higher stabilization of SOC contents may require an adequate supply of N, P and S, and potential costs for these nutrients 'locked-up' in each Mg SOC may be considerably higher than each Mg SOC is worth based on current CO_2 trading prices (Kirkby et al. 2011). However, all nutrients many not have to be supplied externally in all cases. In summary, while the present use of comparatively shallow-root plants may have approached a possible saturation of SOC sequestration depending on the inherent soil physicochemical characteristics such as the amount of fine particles (clay + fine silt; Hassink and Whitmore 1997), many soils and especially subsoils in agricultural ecosystems are probably far from being saturated with SOC (Angers et al. 2011; Kell 2011, 2012).

3.2.4 Additional Ecosystem Carbon and Ecosystem Services

Agricultural ecosystems provide humans with food, forage, bioenergy and pharmaceuticals and are essential to human wellbeing (Power 2010). These ecosystems produce a variety of ESs, such as regulation of soil and water quality, C sequestration, support for biodiversity and cultural services. Forest ecosystems provide timber, non-timber forest products, wildlife habitat, water quantity and quality, C sequestration and storage, climate regulation, recreational opportunities, aesthetic and spiritual fulfillment (FAO 2010; Ryan et al. 2010; Patterson and Coelho 2009). Thus, C sequestration is a critical component of agricultural and forest ESs. Any additional ecosystem C storage for climate change mitigation must, therefore, be assessed with respect to its effects on ESs and unintended consequences for ecosystem functions. Productivity increases in terrestrial ecosystems have been at most studied including two ESs at a time but how management regimes for C sequestration affect many ESs simultaneously is less well known (Bennett and Balvanera 2007). Some examples of possible effects are given in the following section.

Practices to enhance C sequestration in agricultural and forest ecosystems often result in increases in NPP. However, this impacts other ESs such as the provision of freshwater, air quality regulation, erosion control, and recreation opportunities (MEA 2005). For example, increasing application rates of fertilizers and manure to increase agricultural productivity has increased nutrient runoff, increased N and P fluxes to aquatic ecosystems, leading to low productivity, hypoxic zones in many estuaries, higher costs for processing lake water for use, and declines in fish and shrimp catches (Carpenter et al. 1998; Diaz and Solow 1999). Further examples of unintended consequences of intensive production systems on local and regional scale are desertification and soil salinization as a result of irrigation (Reynolds and

Stafford-Smith 2002; Rengasamy 2006). Air quality is also affected by agricultural intensification as N_2O and NO emissions from agricultural ecosystems increase with increase in N application rates (Bouwman et al. 2002). Furthermore, higher emission rates of both gases from soils with higher compared to lower SOC concentrations indicate that N_2O and NO emissions from agricultural soils can increase by practices aimed at increasing the SOC stocks. Methane emissions from rice (*Oryza sativa* L.) paddies increase with increased fertilizer application rates and crop residue return (Hatala et al. 2012; Zhang et al. 2011). Thus, enhanced C sequestration in flooded ecosystems may be associated with significant trade-offs. Erosion by wind and water from agricultural watersheds may increase when practices to enhance production result in increased soil disturbance, i.e., from increased tillage frequency and intensity (Fiener et al. 2011). In summary, sustainable intensification of agricultural production is needed to reduce any adverse impacts of increased food production on other ESs (Matson et al. 1997).

Storing more C by lengthening the harvest interval or reducing the amount removed in a harvest, e.g., by retaining key structural legacies and small forest patches (retention forestry) may store more C in forests (Gustafsson et al. 2012; Ryan et al. 2010). This increases structural and species diversity but increases the risk of C loss due to disturbance. Management practices to increase forest growth have the potential to introduce species and genotypes adapted to future climates. However, forest fertilization may enhance N_2O emissions, reduce water yield as faster growing species use more water and a loss in biodiversity when faster growing monocultures replace multi-species forests (Ryan et al. 2010). Also, forest conservation strategies to maximize C sequestration might deprive local people of access to forest goods and services such as timber and other forest products (Thompson et al. 2011).

Globally, there is no clear relationship between C sequestration and biodiversity but positive relationships may exist at local and regional scales (Midgley et al. 2010). Compared to the factors that determine the C distribution globally, those determining the latitudinal distribution of species are less well known. For example, there is a positive relationship between tropical forest C stocks and biodiversity suggesting that management practices to enhance C sequestration may also alter biodiversity (Strassburg et al. 2010; Thompson et al. 2009). Furthermore, it has been suggested that NPP controls plant and mammal species richness at local scales (Woodward and Kelly 2008). Thus, NPP increases due to enhanced C sequestration may change biodiversity at the local scale.

Forest ecosystems play an important role in the global water cycle (FAO 2008). With the exception of extreme floods, forests can also reduce flooding by acting as sponges, increasing the permeability of the soil and emitting water vapor into the atmosphere through evaporation and transpiration (Bradshaw et al. 2007). Increasing forest cover and density is positively related to the potential for higher relative humidity and potential precipitation (Ellison et al. 2012). Thus, practices aimed at enhancing forest production for C sequestration may further reduce less extreme flooding by forested watersheds. Further services of agricultural ecosystems such as water infiltration, nutrient cycling and protection of off-site water quality are hugely improved with soils having greater SOC stocks (Franzluebbers 2010). Sequestration of SOC provides additional ESs to agriculture itself, improving soil quality by conserving soil structure and fertility, increasing the use efficiency of agronomic inputs, and improving water quality by filtration and denaturing of pollutants (Smith et al. 2008). However, a maximum attainable C sequestration in soil and biomass plus food security would require an increased consumptive water use by 2050, reducing blue water flows strongly. Thus, the proposed global safe operating space for freshwater use would be surpassed causing societal problems related to water shortage and water allocation (Rockström et al. 2012).

Among unintended consequences of increased profile SOC stocks but particularly those at deeper depths may be enhanced SOC decomposition. For example, due to the rhizosphere priming effect the decomposition rate of SOC in the rhizosphere may increase three- to fivefold in response to root exudation but the response of the subsoil SOC pool is less certain (Kuzyakov 2010). Laboratory studies by Fontaine et al. (2007) indicated that adding an additional energy source similar to those of rhizodeposits to the subsoil may accentuate microbial decomposition of subsoil SOC. However, the stimulation of stable subsoil C decomposition in the field by addition of labile material may be small (Sanaullah et al. 2011). Further, a subsoil priming effect is not always observed but an increase in the subsoil SOC pool from inputs of root exudates may also be likely (Salomé et al. 2010). More complex compounds derived from increased root turnover may contribute indirectly to SOC pools by enhancing aggregation and stabilizing microaggregates (Rees et al. 2005).

3.3 Conclusions

The uptake of atmospheric CO_2 by terrestrial ecosystems results in the growth of biomass and build-up of SOC. Important terrestrial C sinks are forests and the only long-lasting C gain occurs in soils. The way management practices cause an additional uptake of atmospheric CO_2 has been better studied in agricultural than in forest ecosystems. Overall, long-term field experiments are needed to study C sequestration enhanced by soil and land-use management practices, and the effects of such practices on ESs.

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Chapter 4
Food Security Through Better Soil Carbon Management

Keith Goulding, David Powlson, Andy Whitmore, and Andy Macdonald

Contents

Abstract Soils store and filter water, prevent flooding, support the production of fuel and fibre, provide habitat, help to create landscape and are a major carbon (C) store. They are, thus, an essential component of supporting, provisioning, regulating and cultural ecosystem services and determinants and constituents of well-being, providing security, the basic material for a good life, health and good social relations. However, calculations based on inherent land quality classes show that fertile soil (that is, soil free of constraints for agricultural production) irregularly covers no

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more than 12 % of the terrestrial land surface. More generally, soil fertility/quality is determined by the interactions between land management interventions by humans and the inherent physical, chemical and biological properties of a soil. Land and soil management based on the understanding of these interactions is one part of delivering food security. Even small changes in C content can have disproportionately large impacts on key soil properties. Practices to encourage maintenance of soil organic carbon (SOC) are important for ensuring sustainability of most if not all soil functions. This chapter considers the relationship between SOC and soil fertility and structure, ways of increasing SOC, some disadvantages of increasing SOC and two proposed ways of increasing SOC, fertility and sequestering C: the soil application of biochar and the cultivation of deeper rooting crops.

Keywords Carbon sequestration • Climate change • Crop residues • Crop roots • Food security • Manures • Minimum-tillage • Soil fertility • Soil organic carbon • Soil structure • Zero-tillage

Abbreviations

ANOVA	Analysis of Variance
C	Carbon
CC	Climate Change
FAO	Food and Agriculture Organisation of the United Nations
FYM	Farmyard Manure
GHG	Greenhouse Gas
K	Potassium
N	Nitrogen
OM	Organic Matter
P	Phosphorus
SOC	Soil Organic Carbon
SOM	Soil Organic matter
S	Sulfur

4.1 Introduction

"The nation that destroys its soil destroys itself." US President F.D. Roosevelt (Letter to all State Governors on a Uniform Soil Conservation Law, 26 February 1937). It has been estimated that 90 % of all food is grown in soil (Swaminathan 2011). Soils also store and filter water, prevent flooding, support the production of fuel and fibre, provide habitat, help to create landscape and can be a major carbon (C) sink. They are, thus, an essential component of supporting, provisioning, regulating and cultural ecosystem services and determinants and constituents of well-being, providing security, the basic material for a good life, health and good social relations (Millenium Ecosystem Assessment 2005; Powlson et al. 2011a).

It should be noted that there is now discussion of ‘food sovereignty’ in addition to food security. At the forum for food sovereignty (http://forumfoodsovereignty.org/inglesepage.htm) meeting in Sélingué, Mali, in 2007, about 500 delegates from more than 80 countries adopted the ‘Declaration of Nyéléni’, which says in part: “Food sovereignty is the right of peoples to healthy and culturally appropriate food produced through ecologically sound and sustainable methods, and their right to define their own food and agriculture systems. It puts those who produce, distribute and consume food at the heart of food systems and policies rather than the demands of markets and corporations.” This clearly goes beyond food security and is an interesting and important development. It could have implications for soil C, but exploring these is beyond the remit of this chapter.

There is less high quality soil available for food production globally than one might think. Calculations based on inherent land quality classes show that fertile soil (that is, soil free of constraints for agricultural production) irregularly covers no more than 12 % of the terrestrial land surface (Eswaran et al. 1999) and is generally seen as a non-renewable asset on the timescale of a human life, soil formation usually taking thousands of years (Targulian and Krasilnikov 2007), although these authors describe ‘rapid’ soil formation that takes 10–100 years. Some soils are naturally fertile, such as the Black Earths of Russia and Ukraine, resulting from a combination of temperate climate and nutrient-rich rocks that slowly weather. More generally, soil fertility/quality is determined by the interactions between land management and the inherent physical, chemical and biological properties of a soil. Understanding this is one part of delivering food security.

Organic C in soil influences numerous soil properties relevant to ecosystem functioning and crop growth. Even small changes in organic C content can have disproportionately large impacts on key soil properties. Practices to encourage maintenance of SOC are important for ensuring sustainability of most if not all soil functions (Lal 2004; Powlson et al. 2011a, b).

The FAO suggested a list of key benefits of SOC to food security (Bot and Benites 2005):

- Drought resistance
- Reduced waterlogging
- Increased plant productivity
- Increased fertiliser use efficiency
- Increased yields
- Reduced herbicide and pesticide use
- Increased biodiversity
- Resilience

All of these are clearly vital for ensuring food security. In this Chapter, the results of some recent and older research from long-term experiments will be reported that link SOC to soil fertility:

- The contribution of SOC to soil fertility

- The benefits of increasing SOC for soil structure and tillage energy requirements
- Some disadvantages of increasing SOC
- The economic value of increasing SOC
- Increasing SOC using deep(er) rooting crops
- The potential benefits of biochar

But first, an important point needs to be made about terminology.

4.2 Terminology, Conversion Factors and Units

The term soil organic matter (SOM) is widely used to describe the sum of all non-living organic components in soil. By implication it refers to the total amounts of C, nitrogen (N), phosphorus (P) and numerous other elements contained therein. But modern analyses of soil actually determine the quantity of organic C present. This is generally termed SOC and it is now usual to quote a value for SOC rather than a numerical value for SOM. For many years it has been customary to assume an average C concentration within SOM of 58 % based on analyses conducted in the nineteenth century. This gives a conversion factor of 1.724 for converting SOC to SOM. However, it has recently been pointed out that this conversion factor is incorrect and an average C concentration in SOM of about 50 % is more reasonable, giving a conversion factor of 2 (Pribyl 2010), with a considerable range observed around that factor (e.g., Meersmans et al. 2009). For many purposes, such as studying management impacts on SOM as a basis of advice to farmers, this issue is unimportant. But because stocks of SOC in soil are central to studies of management impacts on this stock in the context of the global C cycle and climate change (CC), the use of a correct conversion factor has become critical. In a research context SOC is almost universally analyzed, so there is no problem. But in some large scale surveys more approximate analytical methods have often been used for the sake of speed when dealing with very large numbers of samples. Loss on ignition is one such method which leads a value for SOM rather than SOC. If such data is then used to derive values for SOC, and these are used to assess C sequestration rates for "carbon credit" schemes for CC mitigation, the results will be erroneous. In this context one should either use a conversion factor of 2 instead of the conventional lower value, or preferably base information directly on SOC analyses when these are available.

Another issue of terminology concerns whether SOC is expressed as a concentration in soil on a mass basis (e.g. in units such as mg C kg^{-1} soil) or as a stock within a defined soil depth on an area basis (e.g. in units such as Mg C ha^{-1}). Results of laboratory analyses of soil samples are normally expressed as concentration and for studies concerned with management impacts on soil fertility or quality this is satisfactory. But if changes in SOC stocks are being considered in the context of CC (mitigation or exacerbation) it is essential that C is expressed as a stock. To convert a measured concentration value to a stock, a measure of soil bulk density or soil mass to a defined depth is required. This and related issues of appropriate sampling and terminology for different purposes are discussed by Powlson et al. (2011b).

4.3 Soil Organic Carbon, Soil Organic Matter and Soil Fertility

SOM can contribute to soil fertility in a number of ways. It:

- is the source of N, P, and sulfur (S) and some trace elements at times during the growing season and positions within the soil profile that cannot be substituted by fertilizer;
- stabilizes/improves soil structure enabling good root growth and exploration of the soil, especially in poorly structured soils;
- increases cation and anion exchange capacity, especially in light textured soils;
- increases water-holding capacity, especially that of available water.

Figure 4.1 shows an example of the benefits of increased SOC on spring barley yields in the very long-term Hoos Barley Experiment at Rothamsted (Johnston et al. 2009). Best yields are achieved on the treatment that has received 35 Mg ha^{-1} farmyard (i.e. cattle) manure (FYM) since 1852 and now contains 3.5 % SOC to 23 cm (plough depth). These yields cannot be matched by the treatment that has received only fertiliser N (plus P and potassium, K) over the same period and contains only 0.9 % SOC, however much fertiliser is applied. The difference in grain yield of about 2.5 Mg ha^{-1} at the optimum N rate (150 kg N ha^{-1}) equates to about 40 kg grain ha^{-1} for each extra Mg SOC. This can be attributed to the better soil structure, water-holding capacity and early spring nutrient, especially N, availability, resulting from the long-term FYM application and much higher SOC content. However, on one part of the treatment receiving fertiliser, since 2001 the fertiliser has been replaced by FYM of the same quantity and quality, including N, P and K content, as that applied to the FYM treatment since 1852. It is interesting to note that applying FYM instead of fertilisers since 2001 has quite rapidly improved soil quality such that yields on this treatment are approaching those on the long-term manure treatment even though SOC content is still only half that of the long-continued FYM application (1.7 % SOC vs 3.5 % SOC, respectively). This indicates that poor soil structure, water-holding capacity and spring nutrient supply can be at least partially remediated by applying FYM, and relatively quickly, in this case in about 10 years. However, for maximum yields, more fertiliser N (about 50 kg N ha^{-1}) is still needed on the intermediate manured treatment than on the long-term manured treatment.

The particular benefits of higher SOC contents on the availability of P to crops are shown in Fig. 4.2 (Johnston et al. 2009). The availability of P fertilisers is a subject of much debate. Some claim that we are approaching or have even passed 'Peak P', i.e., the point in time of the maximum rate of extraction of rock P (Vaccari and Strigul 2011). Others dispute it (e.g. Rustad 2012) and calculate several

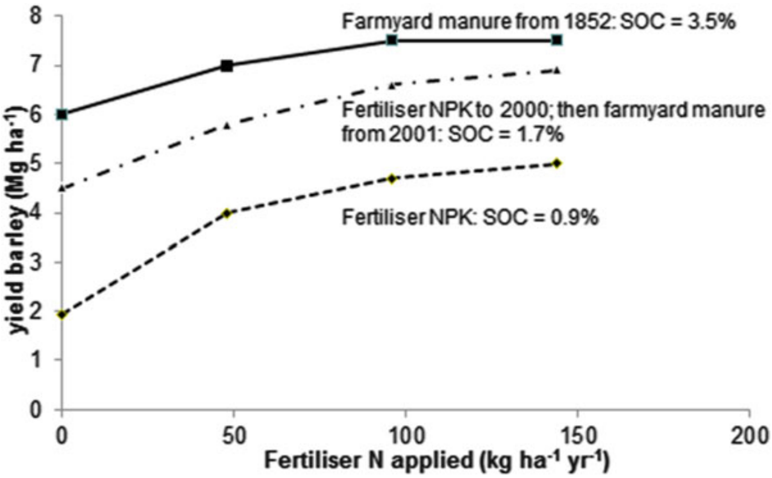

Fig. 4.1 Mean grain yields (Mg ha^{-1}; 2004–2007) of spring-sown barley (cv. Optic) (*Hordeum vulgare* L.) grown on selected treatments of the long-term Hoosfield Continuous Barley Experiment at Rothamsted, receiving fertiliser nitrogen since 1852 (plus phosphorus, potassium and magnesium); fertilisers from 1852 to 2000 and then 35 Mg ha^{-1} farmyard manure since 2001; 35 Mg ha^{-1} farmyard manure since 1852 (Johnston et al. 2009)

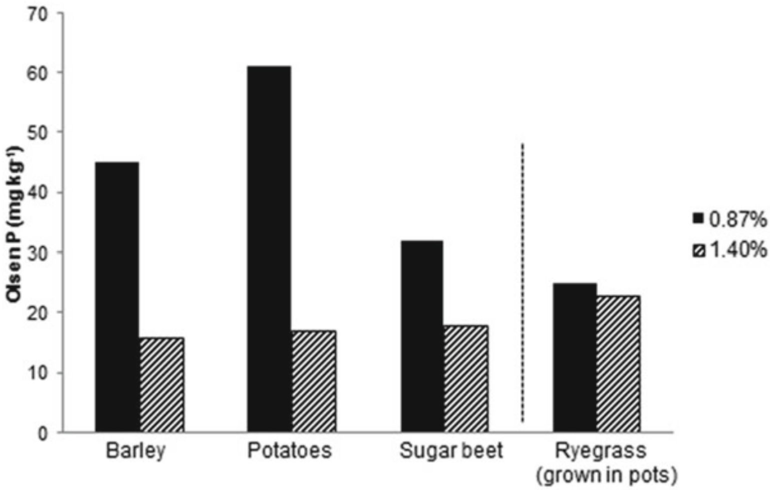

Fig. 4.2 Critical levels of available phosphorus (as measured by Olsen's method) in the same soil at two levels of soil organic carbon for obtaining 95 % of maximum yields of spring barley (*Hordeum vulgare* L.), potatoes (*Solanum tuberosum*) and sugar beet (*Beta vulgaris*) in the field, plus the same test for ryegrass (*Lolium perenne*) grown in pots of the same but thoroughly mixed soils (Johnston et al. 2009)

hundred years of mineable rock P (Hilton et al. 2010). Whichever is true, the cost of P fertiliser has increased dramatically in recent years, as have the prices of N and K fertilisers. Farmers would like to be able to add less and/or work at lower rates of available P in soil. Figure 4.2 shows the critical Olsen P value (i.e. available P as measured by Olsen's method using 0.5 M sodium bicarbonate as the extractant) for obtaining at least 95 % of maximum yield for three crops grown in the same soil with two very different SOC contents. The treatment with the higher SOC content required a much lower critical Olsen P value than did the soil with the lower SOC content by between two and four times. This can be attributed to the better structure on the higher SOC treatment allowing the roots of the crops to acquire P more easily than in a less-well-structured soil. This is supported by the results of the pot experiment with ryegrass (*Lolium perenne*) using the same soils. The thorough mixing of the soils before use has negated the structural differences in the field soils and no advantages of higher SOC content were observed.

4.4 Soil Organic Carbon and Soil Structure: Draught Force and Energy

There is good evidence that even a small increase in SOC can improve soil structure such that the energy needed to cultivate the soil is reduced. This reduces fuel use and improves the energy balance of tillage systems. The Broadbalk Experiment at Rothamsted was ploughed using a tractor fitted with a Doppler radar sensor to measure forward speed, laser proximity sensors to measure furrow depth and width, and a strain gauged frame to measure draught forces (Watts et al. 2006). Watts et al. (2006) produced contour maps of clay content and draught force for the whole field and combined this with SOC content on a plot basis. The clay and draught force maps are very similar. Variogram analysis and analysis of variance (ANOVA) showed that the dominant controlling factor of draught force is clay content in topsoil. The clay content varies across the field from about 18 to 38 % and explains about 58 % of the variance in draught force. However, SOC has an important moderating influence and varies with the amount of N fertiliser applied (N increases yield and so crop residue returns) and manure (cattle manure; FYM) applications. Adding fertiliser treatment as a plot number and as a factor to clay content in a linear regression accounted for 73 % of the variance in draught force. Figure 4.3 shows plough draught and SOC for selected plots of the Broadbalk Experiment, obtained from Watts et al. (2006). It shows a decrease in plough draught with increasing SOC, especially at SOC contents greater than about 1.2 % SOC.

This is clearly important for reducing energy costs and CC mitigation (both through reduced energy use and sequestering C in soils). Watts et al. (2006) calculated that the long-term application of FYM equated to a saving of 10 MJ ha^{-1} on the lighter soils, and the long-term application of fertiliser and increased crop residue

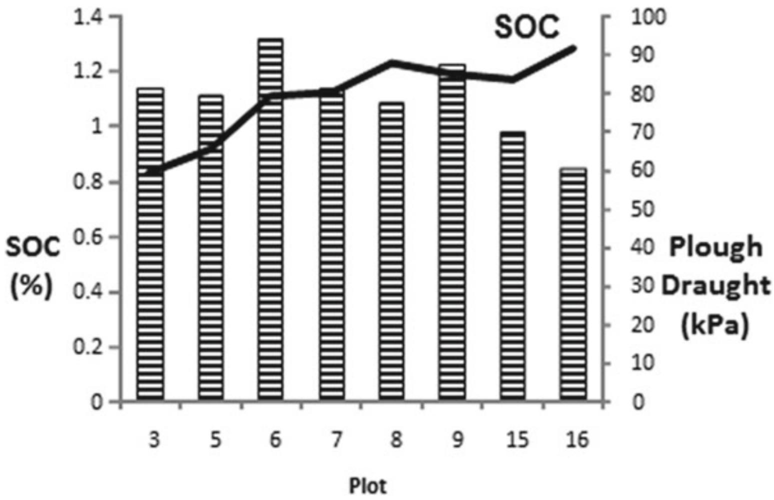

Fig. 4.3 Draught force needed to plough the soil on various treatments of the Broadbalk Wheat Experiment at Rothamsted Research and SOC contents of those treatments (Plot 3: no fertiliser or manure; Plot 5: P and K fertilisers; Plots 6, 7, 8, 9, 15 and 16: P and K plus fertiliser N at 48, 96, 144, 192, 240 and 288 kg ha^{-1} year^{-1}) (Watts et al. 2006) (SED of Plough Draught = 3.5 on 161 df; SED of SOC = 0.06 on 162 df)

returns to only slightly less at 6 MJ ha^{-1}. The extra SOC as a result of fertilisers has a sevenfold greater impact than FYM. This could be attributed to either the easily decomposable material (called the 'light fraction' by Sohi et al. 2001 and others) having a disproportionate effect on soil structure or the long-term application of FYM having a diminishing return on increasing SOC content.

4.5 Key Components of Soil Organic Carbon for Soil Structure and Soil Fertility

Researchers have for many years tried to determine the composition of SOC, especially that part of it called 'humus'. It would not be appropriate to go into detail about SOC components here, but the easily decomposable fraction of SOC (e.g. Sohi et al. 2001, 2005), is particularly important for soil structure and so, as shown earlier, soil fertility.

The labile fraction comprises about 10 % of total C and is thought to comprise microbial biomass plus microbial metabolites. It is increased by straw incorporation and, indirectly, through increased nutrients applications and so crop yields and residue returns. Labile C is correlated with increased aggregate stability and water infiltration rate (Blair et al. 2006). The latter is attributed to the effect of labile C (including exocellular polysaccharides) leading to a more stable and perhaps more continuous pore structure, enabling water to move more freely.

4.6 Critical Soil Organic Carbon Level with Regard to Soil Structure

There has been much discussion about whether there is a common critical SOC level below which serious structural damage occurs. For many years this has been thought to be 2 % SOC (Greenland et al. 1975). Loveland and Webb (2003) found little quantitative evidence for such a limit, and what they did find suggested that it was 1 % not 2 % SOC. They thought that research was needed to test this and especially the role of the light or active fractions of SOC (see above and Sohi et al. 2001). Some limited evidence has been found that supports a critical level of <2 % SOC.

Watts and Dexter (1997) found that simulated tillage in a laboratory test disperses increasing amounts of fine soil particles once a certain threshold of tillage energy is exceeded. This threshold energy increases with SOC content and decreases with soil water content. By fitting a model to Watt's and Dexter's data with the explicit aim of finding these thresholds, it has been shown that wet soils with less than 2 % SOC in Rothamsted's Highfield long-term ley-arable experiment were susceptible to damage (Fig. 4.4). The soils with less than 2 % SOC disperse even when quite dry. A fallow soil with only 1.1 % SOC disperses when the soil is as dry as the wilting point. The plastic limit is the water content at which soils cease to rearrange on disturbance (as the soil dries). It is to be expected then that almost all tillage in soils wetter than the plastic limit will cause damage.

Verheijen et al. (2005) and Schmidt et al. (2011) have found evidence that SOC does not fall below a minimum value which in turn depends on soil type. It seems almost certain that any critically low value of SOC will depend also on clay content or more generally on soil type. So identifying a universal value for a critical minimum SOC level regarding soil structure is probably unattainable.

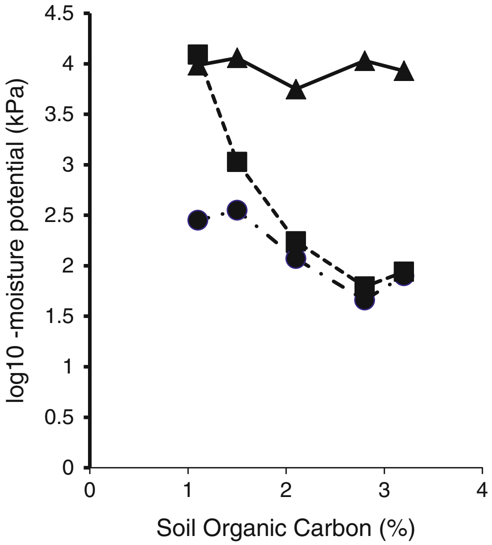

Fig. 4.4 Water potential above which tillage disperses clay in soils. ▲: shrinkage limit; ■: field moisture at which wetter soils are damaged; ●: measured plastic limit. Mean SE of $\log_{10}$ moisture potential = 0.1

4.7 Disadvantages from Increasing Soil Organic Carbon

The benefits of increasing the amount of SOC are bought at a cost and this should be realized. Much C and N is lost during the microbial decomposition of added organic matter (OM), either in the composting phase (in materials that undergo a degree of decomposition before adding to soil) or after application. This will result in emissions to air (CO_2, N_2O, NH_3) and losses of nitrate (NO_3^-) to water. It is important to note again here that, at the equilibrium SOC level for any soil and farming system, by definition, additions of OM will result in no net increase in SOC as the equivalent of all of the added C (N, etc.) will be lost (Johnston et al. 2009), although some of the added C may well be retained in exchange for the loss of an equivalent amount of stored SOC.

Figure 4.5 from Goulding et al. (2000) shows the grain yields of winter wheat (*Triticum aestivum* L.) on the Broadbalk Experiment at Rothamsted at a range of rates of N fertiliser and with cattle manure (FYM) and manure plus N fertiliser in spring. Current high-yielding wheat varieties cannot obtain enough nitrogen from even the 35 Mg ha^{-1} manure applied annually since 1843 despite growing better through the autumn; extra ammonium nitrate fertiliser must be applied in spring at 96 kg N ha^{-1} to match the yields from the optimum inorganic fertiliser application of between 240 and 288 kg N ha^{-1} year^{-1}, depending on climate and pest and disease impacts. Nitrate leaching from the treatments receiving the FYM is much greater

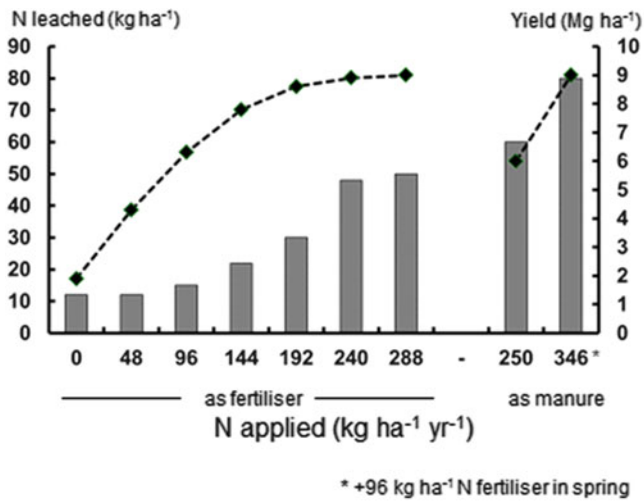

Fig. 4.5 Nitrate leached in kg ha^{-1} year^{-1} (*vertical bars*; mean of eight seasons, 1990–1998; mean SE over time = 26 kg N ha^{-1} year^{-1}) and annual winter wheat (*Triticum aestivum* L., cv. 'Hereward'.) grain yields in Mg ha^{-1} (*line*; first wheat in a rotation; mean of eight seasons, 1990–1998; mean SD over time = 1.1 Mg ha^{-1}) from the Broadbalk Wheat Experiment at Rothamsted Research (Goulding et al. 2000)

than that from treatments receiving only fertiliser at comparable rates. A way of expressing the environmental cost of additional OM in soil from FYM additions (referring to nitrate moving to drainage water) is to express nitrate-N leached per Mg of grain produced. For the treatment receiving FYM plus additional inorganic N fertilizer this value is about 2.6 times that for the treatment receiving inorganic N only and producing the same grain yield.

4.8 The Economic Value of Soil Organic Carbon

Farmers are aware of the inherent value of SOC but often find it hard to do those things that at least maintain SOC levels if not build them up. Moving from, say continuous arable to a ley-arable rotation, changing from ploughing to zero- or Minimum-tillage, applying straw or other organic materials or changing cultivation practices could cost money in equipment or manure, or be seen as risking loss of yield. Putting a value on SOC helps the farmer to appreciate the benefits, especially long-term, of building up SOC.

A project was carried out in the UK for its Department for the Environment, Food and Rural Affairs (Defra) that sought to value SOC (Defra 2004; Sohi et al. 2010). The most important factors in determining the value of SOC are soil type, the value of crops grown, the costs of any applied manures, composts, etc., the costs of application, and the lost value of any material applied that could have been sold off the farm. The project estimated the net value of SOC management Europe to be about €30–80 ha^{-1} $year^{-1}$. The project developed into a spin-off company that now advises individual farmers and growers on good SOC management (see: http://www.keysoil.com/).

4.9 Ways of Maintaining Soil Organic Carbon in Arable Cropping

Clearly a change from arable to permanent pasture or forest will increase SOC in the surface (0–23 cm) soil (e.g. Johnston et al. 2009; Don et al. 2011; Poeplau et al. 2011; Powlson et al. 2011b) but this is unlikely to be economic and removes land from food production, working against food security. One must therefore look at systems which maintain at least some arable cropping. The options are:

1. Ley-arable farming, including catch crops
2. Apply crop residues, manures or other organic “wastes”
3. Minimum-Tillage/No-Tillage
4. Grow plants with longer/larger root systems
5. Grow larger crops
6. Biochar?

4.9.1 Ley-Arable Farming (i.e. Crop Rotations with Some Pasture)

The most comprehensive recent review of the benefits of ley-arable farming was by Johnston et al. (2009) based mostly on the long-term experiments at Rothamsted. Much depends on the soil type, especially clay content (see Sect. 4.6 above), and the starting point for the rotation. If starting from long-term arable and a low initial SOC, then SOC begins to increase, but if starting from long-term (permanent) pasture and a high SOC then SOC will decrease even under a ley-arable rotation, but more slowly than in a continuous arable system. It is also important to remember the improvements in soil structure from even small but regular inputs of SOC, as noted in Sects. 4.4 and 4.5 above.

4.9.2 Crop Residues, Manures and Other Organic Materials

The results of applying organic materials including farm manures, digested biosolids, cereal straw, green manure and paper crumble were reviewed by Powlson et al. (2012). The application of organic materials increased SOC in the surface soil (0–23 cm; kg C ha^{-1} $year^{-1}$ Mg^{-1} dry solids added; mean $\pm$ SE) by 60 ± 20 (farm manures), 180 ± 24 (digested biosolids), 50 ± 15 (cereal straw), 60 ± 10 (green compost) and about 60 (paper crumble; no data, but C content and composition similar to that of farm manures so SOC increase assumed the same; Powlson et al. 2012). Long-term experiments show that the annual rate of SOC accumulation declines (>50 year) with farm manure applications as a new equilibrium value is approached. A review of 25 long-term experiments (6–56 years) on the impact on SOC of cereal straw management showed a trend towards increased SOC when straw was incorporated (Powlson et al. 2011b), but there was no significant effect. However, there was evidence that labile fractions within the total SOC increased proportionately more where straw was incorporated. It was concluded that continuous removal of straw (e.g., for bioenergy production) would be unwise as this could adversely impact soil physical properties – rational decisions on the frequency of straw removal should be taken in the light of local knowledge of these trends.

One other issue that needs to be addressed is the view that C sequestration can be an important element of CC mitigation. A genuine and at least semi-permanent sequestration, i.e., a net transfer of C from atmospheric CO_2, into soil by the addition of crop residues, manures and other organic materials will mitigate CC. However, biosolids, farm manures and cereal straw are typically already applied to soil in the UK and much of Europe, so resulting increases in SOC cannot be regarded as additional CC mitigation. Also large increases in SOC were deduced for paper crumble (>6 Mg C ha^{-1} $year^{-1}$) but outweighed by N_2O emissions deriving from the additional N fertiliser usually applied with this material. Application of green compost (also referred to as green waste compost), derived mainly from plant wastes from household and municipal sources, offers genuine potential for CC mitigation because application replaces disposal to landfill; it also decreases N_2O emission.

4.9.3 Zero- Minimum-Tillage

For the UK, Powlson et al. (2012) found that the average annual increase in SOC in surface (0–23 cm) soil deriving from reduced tillage was 310 ± 180 kg C ha^{-1} $year^{-1}$. However, even this accumulation of C is unlikely to be achieved in the UK and northwest Europe because farmers practice rotational tillage. Increases in SOC during the minimum or zero tillage period are mostly lost on periodic ploughing. Also, Powlson et al. (2012) found evidence that N_2O emissions may increase under reduced tillage in moist climates, counteracting increases in SOC.

4.9.4 Deep(er) Rooting Crops

Roots are a means of delivering C and a large range of potentially valuable small molecular weight compounds into the rhizosphere. The role of these in chemical signalling between plant roots and other soil organisms, including between the roots of neighbouring plants, is often based on these root-derived chemicals (Bais et al. 2006). These include C sequestration (at depth), biocontrol of soil-borne pests and diseases, and the inhibition of the nitrification process in soil (conversion of ammonium to nitrate) with possible benefits for improved N use efficiency and decreased N_2O emissions. Kell (2011) has recently discussed the possibility of breeding crops with deeper and bushy root systems that sequester more C, as well as being more effective at obtaining nutrients and water. If such phenotypes could be developed, they should have improved drought and flooding tolerance, deliver greater biomass yields, retain soils better against erosion, and produce better soil structure as well as sequestering C. Kell calculates that increasing SOC by 15 % would lower atmospheric CO_2 by 30 %. Clearly this would make a very significant contribution to mitigating CC. Of course, in the context of food security, one would have to ensure that diversion of additional photosynthate to roots does not cause a trade-off leading to decreased grain production.

4.9.5 Biochar: The Solution?

Biochar is the charcoal-like product of the pyrolysis of any organic material at carefully controlled temperatures. The time and temperature determine the properties of the biochar. There are many claims for the benefits of biochar in soil:

- Near-permanent increase in SOC
- Suppresses trace GHG emission
- Stabilises other SOC
- Increases water holding capacity
- Enhances pore-size distribution
- Improves crop N-use

But there are large gaps in knowledge, especially regarding the mechanisms that could be operating (Verheijen et al. 2010). Factors yet to be identified and understood include the rate of decomposition of biochar when added to soil, its nutrient and water retaining properties (i.e. cation exchange capacity, surface area), and its role as a microbial habitat. Also, published beneficial impacts on soil quality and crop growth are mainly confined to highly weathered soils in tropical regions and at least some of the benefits to yield appear due to the nutrient and liming effects of biochar rather than its C. As yet there are no reports of such benefits being reproduced in soil of temperate climates. There are also concerns about possible adverse environmental impacts such as the presence of contaminants and the land use implications if large areas were devoted to growing trees to produce biochar in large quantities.

A special 'virtual edition' of the journal *Plant & Soil* in late 2011 concluded that "… studies published over the past two years have demonstrated that biochar may be more or less effective depending on the crop and soil type." (Lambers and Lehmann 2011).

4.10 Conclusions

SOC is a key component of soil fertility and structure and so food security. It is also a major terrestrial C pool, containing an estimated 1,462–1,545 Pg SOC to a depth of 1 m (Powlson et al. 2011a). Thus effective management of SOC is essential for sustainable food supplies, mitigating climate change and overall ecosystem sustainability. There are well-known and effective ways of increasing SOC by returning crop and manure residues and crop rotations that include long-term grass or other leys. The application of biochar might improve soil fertility and sequester C, but good experimental evidence for this is lacking.

Acknowledgements The research reported in this chapter was funded in part by the UK Department for Environment, Food and Rural Affairs, the Lawes Agricultural Trust and the UK Biotechnology and Biological Sciences Research Council (BBSRC).

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Chapter 5
Soil Carbon and Water Security

Karl-Heinz Feger and Daniel Hawtree

Contents

Abstract The provision of hydrologic ecosystem services (HES) is critical for both human well-being and environmental sustainability. A key component in addressing current and future challenges in water resource management is the development of a comprehensive understanding of the complex relationships between soil properties, land use/management, and the hydrologic cycle. The soil-water interface is critical in determining the relative distribution of "blue" (i.e., irrigation, municipal supplies, aquatic ecosystems) and "green" (i.e., evapotranspiration) water usage for a given region, and therefore must be considered in the assessment of HES provisioning. The relationship between water security and food security plays a crucial role, since the agricultural sector consumes approximately 70 % of global water supply.

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In addition, it is projected that global agricultural yields will have to be doubled over the next 25–35 years to meet increasing demand, and that about 90 % of this increase will have to be produced on existing cultivated land. This can only be achieved with a more efficient use of the water resources and a substantial improvement and extension of water management systems.

The hydrologic functioning of soils is primarily a function of their physical, chemical, and biological properties, and in particular the amount and quality of organic carbon (C) present element. The partitioning of hydrologic fluxes into blue and green water is mainly a function of soil factors and processes that control the storage and transport of water, with soil organic matter (SOM) representing a key element. The amount of soil organic C (SOC) present will increase with specific land-cover/use changes (e.g., conservation tillage, mulching, agroforestry), and will be reduced or even eliminated by others (e.g., erosion, high-intensity fires), making proper land management crucial for sustaining beneficial soil properties. The SOM quality is decisive for the filtering, buffering, and transformation function capacities of soils. It also controls the mobilization of dissolved organic C (DOC) which has implications on the quality of water supply. Given the increased calls for management actions to address climate change which will impact soil functioning (e.g., C sequestration, afforestation), it is important to also consider how these changes will impact individual HES and water security, and what conflicts and tradeoffs will need to be addressed. Thus, provisioning of HES must be integrated in coordinated actions of resource planning and land management on the appropriate landscape scale (i.e., the watershed). Such a process may benefit from information resulting from integrated catchment modelling that systematically assesses land management/soil-feedback scenarios

Keywords Hydrologic ecosystem services • Water security • Soil organic carbon • Soil organic matter • Water cycle • Blue water • Green water • Dissolved organic carbon • Land use • Erosion • Soil functions • Soil quality • Soil degradation • Drinking water • Integrated land and water resource management • Flood retention • Soil structure • Infiltration

Abbreviations

C	Carbon
CEC	Cation Exchange Capacity
DOC	Dissolved Organic Carbon
DOM	Dissolved Organic Matter
ES	Ecosystem Services
HES	Hydrologic Ecosystem Services
OM	Organic Matter
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
WWP	Working for Water Program

5.1 Introduction: Defining Water Security

Freshwater is essential for human life and well-being, economic development, and ecosystem health (Falkenmark and Lundqvist 1997; Falkenmark and Rockström 2004; Postel 1998; Ripl 2003). Thus, water security is more than just the capacity of a population to have access to safe potable water, in a more comprehensive view it defines the availability of an acceptable quantity and quality of water for health, livelihoods, ecosystems, and production, coupled with an acceptable level of water-related risks to people, environments, and economies (WWAP 2012). This includes the sustainable use and protection of water systems, protection against water related hazards (i.e., floods and droughts), sustainable development of water resources, and safeguarding water functions and services for humans and the environment. Without water security, there will be no food security. Energy production will be compromised and poverty reduction and economic growth will not be sustainable. The human appropriation of freshwater has been identified as one of the nine environmental key processes within the concept of "Planetary Boundaries" that was introduced as a scientific framework for global environmental sustainability (Rockström et al. 2009). Water security is based on the critical regulatory functions provided by the water cycle (Vörösmarty 2009) and freshwater operates as the "bloodstream" of the biosphere (Falkenmark and Rockström 2004; Ripl 2003). Hence, freshwater availability represents a key mechanism for sustaining environmental conditions on Earth (Rockström et al. 2012).

Water security is rapidly declining worldwide. The uneven distribution of water resources over time and space, and the way human activity is affecting that distribution today are underlying causes of water crises in many parts of the world (Falkenmark and Lundqvist 1997; Vörösmarty 2009; WWAP 2012). Climate change is superimposed on the complex water cycling in catchments, making its signal difficult to isolate and its influence felt throughout the water supply, demand, and buffering system (Vörösmarty 2009). Notably the increase of extreme events like drought and heavy rainfall puts additional pressure on water supplies. There is increasing concern related to population growth, over-utilization of groundwater aquifers, water logging and salinization, pollution (through urban and industrial wastes, fertilizers and pesticides from agricultural land), and the flooding of cultivated, urban, and industrial areas. Many of these problems are related to changes in land use, such as urbanization, intensification of agricultural production, and reduction of close-to-nature vegetation forms, notably forests and wetlands. Hence, water security is at the core of management of sustainable ecosystems (Calder 2005; UNEP 2009).

5.2 Water Flow Components and Ecosystem Services

The concept of "blue" and "green" water usage can be applied to understand the partitioning of water resources at the soil surface (Fig. 5.1). Blue water is that which moves above and below the ground as surface or subsurface runoff in rills,

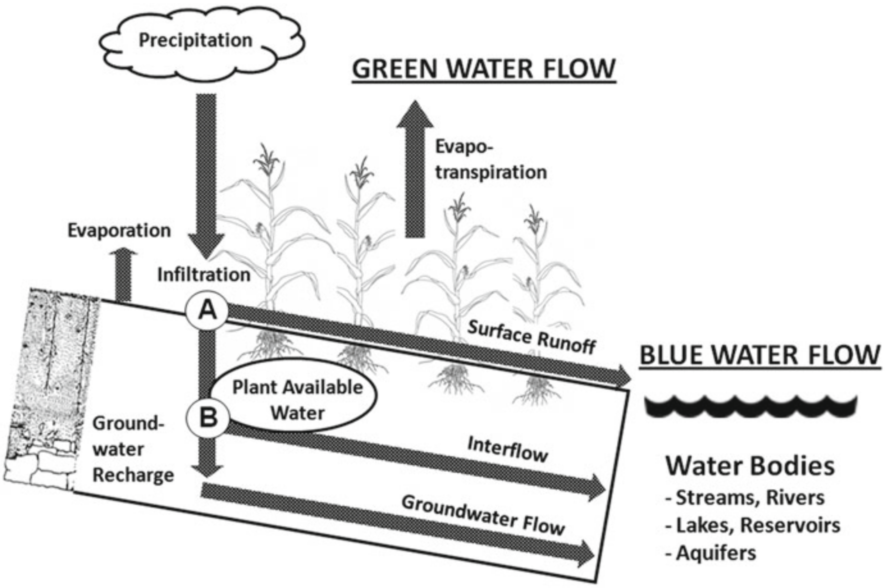

Fig. 5.1 The water cycle for a terrestrial ecosystem showing the partitioning of rainfall. At partitioning point A, rainfall is divided into surface runoff and infiltration. At the partitioning point B, soil water is divided into evaporation from the soil surface, transpiration and interception from plants, and groundwater recharge. Evaporation, interception, and transpiration together constitute the green flow component. Surface runoff and groundwater recharge together constitute the blue flow component (Modified from Falkenmark and Rockström 2004)

gullies, and rivers, or infiltrates into the soil to potentially recharge water tables and aquifers. Green water comprises the flow of vapor to the atmosphere in the form of evapotranspiration. Productive green water is the transpiration flow from plants and directly contributes to the growth of biomass, whereas non-productive green water consists of evaporation from soil and vegetation surfaces (Falkenmark and Rockström 2004).

The balance between the flows of green and blue water is a useful concept for assessing water availability in different regions. The extremely uneven distribution of global rainfall in conjunction with large differences in energetic factors controlling evapotranspiration results in very different patterns between regions of blue and green water partitioning. Table 5.1 provides typical values for the major biomes of the world. These numbers are synthesized from various studies and there is naturally high variability within the same ecological zone. Nevertheless, distinct trends for biotopes (ecosystems) in different climates can be recognized.

The occurrence of a certain type of ecosystem generally depends upon factors that are directly related to rainfall partitioning (Walter and Breckle 2002). To some extent this situation can be modified by soil biophysical properties (Ehlers and Goss 2004; Novák 2012), which are listed in Table 5.2. The biophysical factors are superimposed by numerous human influences mostly resulting from land use, including a great number of technical measures that affect both plant and soil systems (Fig. 5.2).

Table 5.1 Partitioning of rainfall in blue and green flow components for hydroclimates in the major ecosystem zones (biomes) of the world

Hydroclimate	Biome	Phytomass production (Mg ha^{-1} year^{-1})	Rainfall (mm year^{-1})	Blue water flow: Surface flow (mm year^{-1})	Blue water flow: Sub-surface flow (mm year^{-1})	Green water flow: Total ET (mm year^{-1})
Subtropical and tropical	Desert savanna	2–6	300	18	2	280
	Dry sub-humid savanna	4–12	1,000	100	30	870
	Wet savanna	8–20	1,800	360	240	1,200
Subarctic temperate	Tundra	1–2	370	70	40	260
	Taiga	10–15	700	160	140	400
	Mixed forests	10–15	750	150	100	500
	Wooded steppes	8–12	650	90	30	530
Equatorial	Wet evergreen equatorial forest	30–50	2,000	600	600	800

Data based on Falkenmark and Rockström (2004), adapted from L'vovich (1979)

Table 5.2 Biophysical and human factors that determine the partitioning of water flows in the water cycle

Water flow	Biophysical determinant	Human determinant
Partitioning point A		
Surface runoff	Vegetation/climate/biome	Land use
	Soil surface conditions	Tillage practices
	Rainfall intensity	
Soil water storage	Soil moisture regime	Soil management
	Water holding capacity	Soil management
Partitioning point B		
Evaporation	Atmospheric demand	Canopy cover
	Micro-meteorology	Mulching
	Wetness of soil	Timing of planting
Transpiration	Photosynthetic pathway	Crop management
	Plant available soil moisture	Crop management
	Atmospheric demand	
Groundwater recharge	Soil hydraulic conditions	Compaction
	Geologic conditions	Soil and crop management

Based on Falkenmark and Rockström (2004)
For explanations of the partitioning points see Fig. 5.1

The water cycle links terrestrial and aquatic ecosystems and contributes a series of ecosystem goods and services (Fig. 5.3). Because the ecosystem represents a useful framework to consider the many linkages between humans and their environment, an "ecosystem approach" has been advocated by many organizations and individuals as a means of addressing the interrelations between water, land, air, and

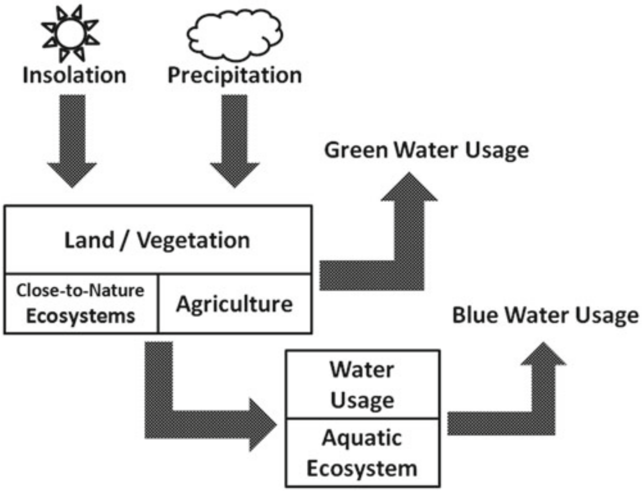

Fig. 5.2 Key linkages between land, water, and ecosystems with flows of blue and green water (Concept based on Falkenmark et al. 2009)

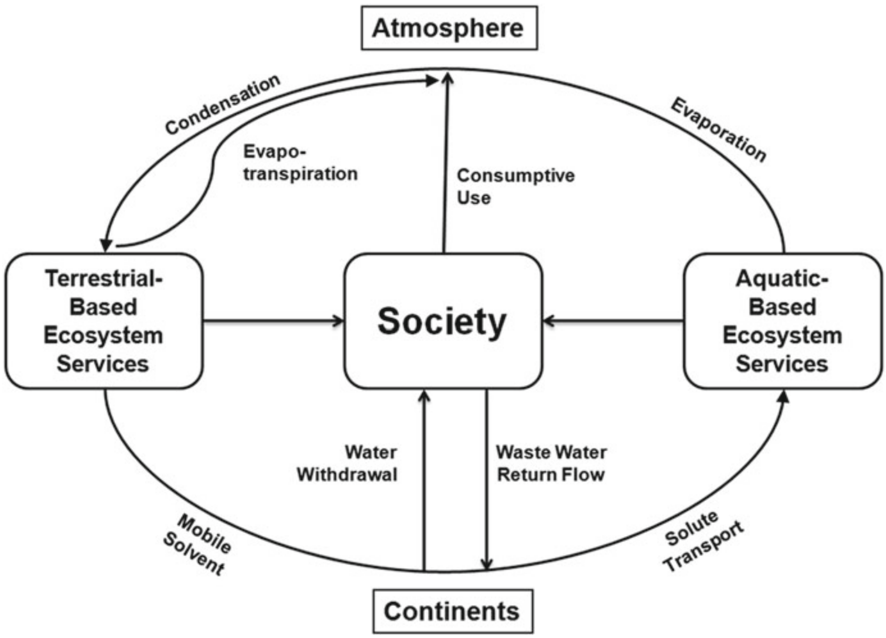

Fig. 5.3 Key linkages between land, water, and ecosystems with flows of blue and green water (Concept based on Rockström et al. 2012)

Table 5.3 Types of ecosystem and hydrologic ecosystem services

Ecosystem services (MEA 2005)	Hydrologic ecosystem services (Brauman et al. 2007)	Example for hydrologic ecosystem services
Environmental good (Provisioning service)	Extractive water supply	Municipal water supply Irrigation water supply
Regulating service	Water damage mitigation	Flood protection Groundwater protection
Supporting services	Ecosystem foundation	Plant growth Medium for aquatic habitat
Cultural services	Appreciative uses	Water based tourism Aesthetic/spiritual value

all living organisms, and encompassing ecosystems and their goods and services (Falkenmark and Rockström 2004; UNEP 2009).

In the most fundamental sense, ecosystem services (ES) are the processes and resources produced by ecosystems which are beneficial to humans. The Millennium Ecosystem Assessment (MEA 2005), characterizes ES according to the four categories:

1. Environmental goods/provisioning services (e.g., provision of food, freshwater, fuel, and fiber);
2. Regulating services (e.g., climate/flood/disease regulation, water purification);
3. Supporting services (e.g., nutrient cycling, soil formation, primary production);
4. Cultural services (e.g., aesthetic, spiritual, educational, recreational).

The ES concept has become a useful tool for connecting the goals of conservation and environmental health with the ultimate benefits which are provided to human societies (MEA 2005). Those ES which are related to water resources are known as hydrologic ecosystem services (HES). Water quantity and water quality in their spatio-temporal dynamics are affected by ecosystem characteristics and will often require a satisfactory standard of ecosystem health and functioning for their sustained provision. At the same time, the characteristics of an ecosystem will be affected by HES, particularly regarding the biotic community which can be supported. Brauman et al. (2007) characterized HES as falling into five categories. These HES are summarized in Table 5.3, along with how they relate to the categories of ES presented in the Millennium Ecosystem Assessment and with specific examples of each type.

In the context of regions with a pressing need for poverty alleviation and/or stabilization of food supplies, it is important that HES are considered as a critical element in development processes. The goal should be to facilitate the use of water supplies for the benefits of development, while minimizing the negative impacts on other HES. If an ecosystem becomes too degraded through poor land-use practices, then the underlying ES supporting the system (i.e., the supporting services) may be undercut, which will in the end defeat the purpose of additional water extraction used for development (WWAP 2012). Therefore, the goal must be to find the correct balance of allocation between extractive water uses for development, without the degradation of other HES functions.

An example where the HES concept has been applied in the context of economic development and poverty alleviation is in South Africa. There the Working for Water Programme (WWP) was designed in 1995 with the goals of both increasing water supply and providing much needed employment at the end of the apartheid system (Hobbs 2004). A major goal of WWP is the control of invasive plant species which are considerable consumers of water in the semi-arid regions of the country, thereby decreasing the hydrologic provisioning service of the ecosystem. In the fynbos biome, situated within the Cape Provinces of South Africa, the most problematic alien tree species are of the genera *Acacia*, *Eucalyptus*, and *Pinus*, which affect both riparian and terrestrial environments (Le Maitre et al. 2000). Some of those species are known to change the chemical and physical properties of the low-nutrient soils which characterize the fynbos biome. This conversion also affects the soil moisture regime and water yield, since invasive alien species (especially *Acacia*) also generally use more water than native fynbos. This is a particular concern in riparian environments, where invasive alien species can have effectively unlimited access to water, which may significantly reduce streamflow quantities.

5.3 Soil Functioning and Soil Organic Matter in the Water Cycle

Hydrologic ecosystem services related to soil properties fall under three categories: supporting, regulating, and provisioning services. The hydrologic soil characteristics controlling infiltration, transport, and storage of water are largely a function of the physico-chemical properties of a given soil.

Major soil functions have been identified and described in a concise form in the European Commission "Thematic Strategy" communication (European Commission 2006a, b). According to the strategy, soil delivers its services through seven main functions: (i) food and other biomass production, (ii) storing, filtering, and transformation of materials, (iii) habitat and gene pool of living organisms, (iv) physical and cultural environment for humankind, (v) source of raw materials, (vi) acting as a carbon (C) pool, and (vii) archive of geological and archeological heritage (Blum et al. 2006).

The storing, filtering, and transformation function capacities between the atmosphere, aquatic systems, and the plant cover, strongly influences the water cycle at Earth's surface (Fig. 5.1, Table 5.2). This encompasses the gas exchange between terrestrial and atmospheric systems and protection of the environment and society against the contamination of groundwater/surface water bodies and the food chain. The storing function of soils should be extended by that of transport (of water and solutes) as well. The interaction of both functions is crucial in the partitioning of rainfall into blue and green water (see Sect. 5.2) and runoff formation, including aspects of soil erosion and flood formation. The filtering, buffering, and transformation capacities play a major role in water quality issues (Fig. 5.4). Soils react through mechanical filtration, physical or physico-chemical adsorption, and precipitation on its inner surfaces and microbiological and biochemical mineralization and

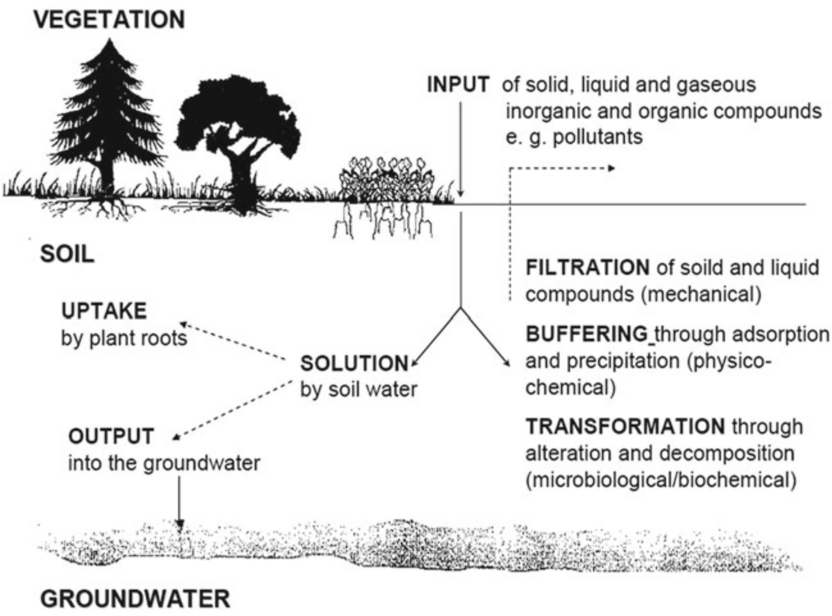

Fig. 5.4 Filtering, buffering, and transformation processes in the soil (Adapted from Blum 2008)

metabolization processes. As long as these filtering, buffering, and transformation capacities can be maintained, soils can play an important role in protecting water quality and the food chain. However, capacities of soils are naturally limited and vary according to specific soil conditions.

Evaluation of the main soil functions is at the core of the European approach to soil quality assessment, which is supplemented with the evaluation of the boundary condition of soil threats to allow for a complex description of the sustainability of the soil-use system (Tóth et al. 2008). Based on previous definitions of soil quality, in particular that proposed by the Soil Science Society of America (Allan et al. 1995), soil quality is defined as the soil's ability to provide ecosystem and social services through its capacities to perform its functions under changing conditions (after Tóth et al. 2007). This concept of soil quality allows for practical applications with regards to targeted social and/or ES. Targeted applications may be linked to special soil functions.

The amount and quality of the organic C component is a crucial soil characteristic, which in its vertical distribution in the soil profile influences many different facets of soil quality and functioning (Baldock and Nelson 2000; Janzen et al. 1997). Figure 5.5 displays the complex relationships between soil organic matter (SOM), other important soil properties, and connected HES.

Organic C is the main constituent of SOM that has many direct and indirect effects on soil physical, chemical, and biological properties. Those most relevant for water storage and transport (i.e., water amount) as well as filtering, buffering, and transformation (i.e., water quality) are discussed in the following section.

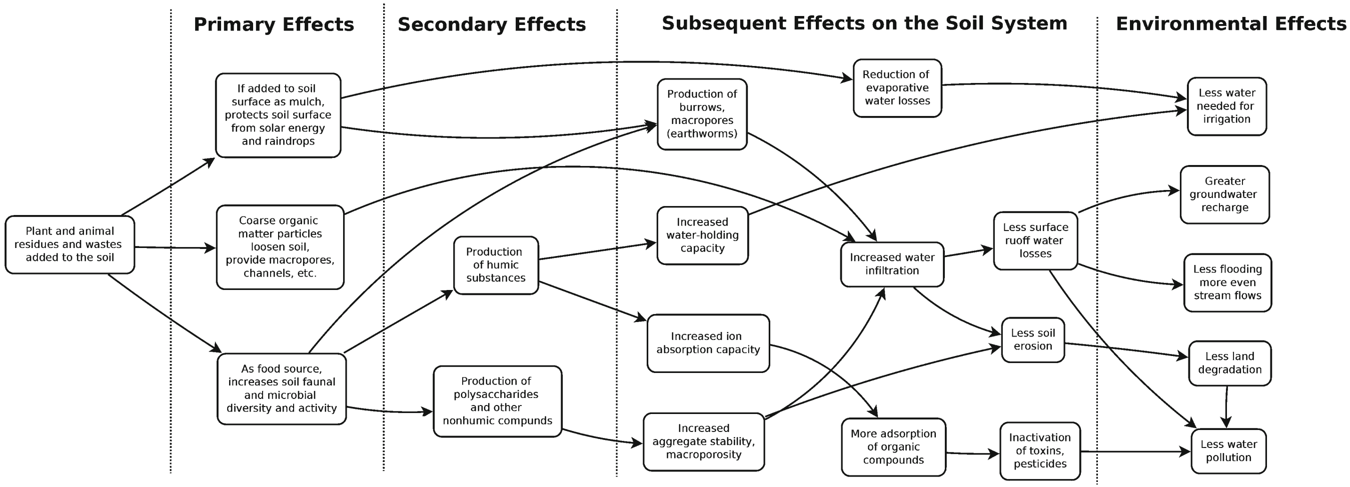

Fig. 5.5 Some of the ways in which soil organic matter affects soil properties, plant productivity, and environmental quality; notably related to hydrologic ecosystem services (Based on Brady and Weil 2001)

5.3.1 Structure

The grouping or arrangement of soil particles is mainly influenced by organic matter (OM). It is the major agent stimulating the formation and stabilization of granular and crumb-type aggregates (Brady and Weil 2001). In soils with high biotic activity, crumb formation is favored by gels and other viscous microbial products that are associated with bacteria and fungi procuring the decomposition of organic residues. Also, organic exudates from plant roots and associated mycorrhiza contribute to this aggregating action. Another important aspect of aggregate stability is the chemical interaction of organic compounds with mineral particles such as clay and iron and aluminum oxides/hydroxides thereby binding them together in water-stable aggregates. The presence of stable crumb-type aggregates guarantees high infiltration capacities and, thus, minimizes surface runoff flow. Further, soil surfaces are prevented from the detachment of soil particles, clogging the soil pores, and crust formation. This reduces accelerated erosion and flood formation. Erosion does not only lead to a considerable loss of SOM. Eroded soil particles have also negative impacts on water quality since they contribute to a higher turbidity and the mobilization of nutrients (such as nitrogen and phosphorus) and adsorbed contaminants (e.g., pesticides or heavy metals). This may jeopardize drinking water supplies from rivers, lakes, and reservoirs. Further, there may be negative effects on aquatic ecosystems resulting from eutrophication which is the over-enrichment of lakes and reservoirs with nutrients (notably phosphorus) leading to algal blooms and anoxic conditions (Carpenter 2005; Fraterrigo and Downing 2008).

5.3.2 Pore Size Distribution

The SOM and the associated biotic activity have a decisive influence on total pore space of a soil (as also reflected in its bulk density), but together with texture and structure (see above) also on the balance between macro- and micropores. This is reflected in the water holding capacity and the hydraulic conductivity which primarily define the behavior of a soil in the hydrologic cycle (Fig. 5.1). To some extent infiltration capacity also depends on the architecture of the pore system, notably the connectivity of macropores from the soil surface into mineral soil horizons (Hillel 1998). The concentration of SOM markedly increases the capacity of a soil to store and retain water, also in plant available form. Therefore, SOM together with texture is widely used as a predictor to estimate hydraulic properties of soils (Vereecken et al. 1989; Wösten et al. 2001).

5.3.3 Cation Exchange Capacity

Due to its large inner surface, SOM possesses a distinct capacity to adsorb cations. With respect to cation exchange, humic colloids have cation exchange capacity (CEC) that is 2–30 times greater (per mass unit) than that of the various types of clay minerals (Brady and Weil 2001). Thus, SOM may account for a considerable part of the CEC of mineral soils. CEC can be used as a good indicator for the filtering and buffering function of a soil in the water cycle. Further, SOM can strongly bind heavy metal cations (e.g., copper, lead) by the formation of stable complexes (i.e., chelates).

5.3.4 Microbial Activity

The transformation of organic contaminants (e.g., pesticides, air-borne micro-pollutants) in a soil diminishes the accumulation of those compounds and leaching to aquatic systems. This water-related soil function is mainly achieved by micro-organisms whose abundance in soils mostly depends on the amount and quality of SOM.

5.4 Dissolved Organic Carbon

A direct link between SOM and surface water quality is dissolved organic matter (DOM) which is a ubiquitous component of natural waters, and is normally measured as dissolved organic C (DOC). It is operationally defined as comprising any organic compound passing through a 0.45 μm pore size filter. In general, DOC includes a small proportion of identifiable, low-molecular weight compounds and a larger proportion of complex high molecular weight compounds, collectively termed humic substances (Steinberg 2003). To a great extent they originate from the soils in the catchment (McDowell and Wood 1984; Cronan and Aiken 1985; Hongve 1999; Kalbitz et al. 2000). DOC-rich water may produce problems when used for drinking water supply (i.e., moldy taste, color/low aesthetic quality, microbial contamination). Furthermore, DOC can play some role in the co-transport of problematic compounds such as heavy metals and pesticides. Appearance in drinking water can be unappealing, which results in increased consumer complaints. Therefore, there is a need to remove DOC according to drinking water standards (Council of the European Union 1998). This treatment procedure, including processes such as coagulation, adsorption and membrane filtration, represents a major cost to water providers (Sharp et al. 2006). Further, harmful by-products such as trihalomethanes can be formed when chlorine-based disinfection is applied.

High DOC concentrations in surface waters are mostly associated with extensive peat-covered catchments (Aitkenhead et al. 1999; Dawson et al. 2004), but can also occur in streams draining mostly forested catchments dominated by podzolic soils (e.g. McDowell and Wood 1984). As a consequence, land-use practices such as afforestation with conifers and/or excessive biomass extraction (e.g., fuel-wood, litter raking) may contribute to accelerated soil acidification and humus degradation favoring the formation of water-soluble organic substances (mostly fulvic acids) (Feger 1993). Peak values of DOC concentrations occur under high-flow conditions when a considerable portion of the water is transported laterally in the topsoil horizons. In general, differences between catchments relate to soil types and flow paths (Feger and Brahmer 1986). Since the 1990s, significant trends in increasing DOC concentrations in surface waters have been monitored in many regions of Central and Northern Europe, and North America (Driscoll et al. 2003; Evans et al. 2005; Monteith et al. 2007). Several potential causes for the observed trends have been discussed, including temperature, rainfall, acid deposition, land-use changes, nitrogen deposition, and increased atmospheric carbon dioxide (CO_2) concentrations. Most likely DOC has been increasing in response to a combination of declining acid deposition and rising temperatures. However, it is difficult to isolate mechanisms based on monitoring data alone. Further knowledge of the extent of natural long-term variability and of the natural "reference" state of soil-catchment systems are required (Evans et al. 2005; Sucker and Krause 2010). In some cases land-use activities appear to have resulted in increased surface water DOC concentrations (e.g., peatland degradation: Freeman et al. (2001); Kalbitz and Geyer (2002); forest liming: Kreutzer (1995); conifer harvesting: Neal et al. (1998)).

5.5 Effects of Wildfires

An important consideration with respect to the interaction of soil organic C (SOC) and HES is the impact of fire. Wildfires can lead to considerable changes both in the water flow dynamics and hydrochemistry, directly by changing soil structure and properties, and indirectly through the effects of changes to the soil and vegetation on hydrological and geomorphological processes (Shakesby and Doerr 2006). During a fire of low to moderate intensity, there will be a loss in organic material overlaying the soil, as well as a loss of SOC. In the case of more intense fires there is increased potential for more severe impacts on soil properties, both due to a higher loss of OM from burning and the potential development of a hydrophobic soil layer. A hydrophobic soil layer may form if the upper layer soil reaches relatively high temperatures (>175 °C), causing organic C compounds within the soil to volatize and to move down through the soil profile, where they will solidify on the surface of cooler soil particles (DeBano 2000). These volatized compounds are water repellent, wax-like hydrocarbons which form a highly hydrophobic soil layer (Fig. 5.6).

The creation of a hydrophobic layer during a fire will have a number of impacts on hydrologic behavior. At the site of the fire, the hydrophobic layer will decrease rates of infiltration, and, therefore increase the rate of surface runoff during storm events (DeBano et al. 1998). A second significant onsite effect is the potential for large increases in rates of soil erosion and associated nutrient leaching from the ashes that affects water quality (Neary et al. 1999; Shakesby and Doerr 2006). This can occur during a storm event if the wettable soil layer (i.e., above the hydrophobic layer) becomes saturated and water is unable to move either downward or laterally. This will increase the pressure on the stability of the soil cohesion in the saturated zone, and lead to the development of erosion rills or even mass wasting events.

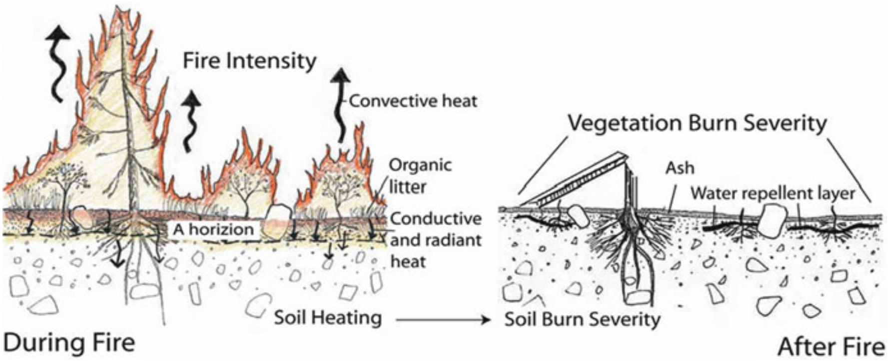

Fig. 5.6 Effect of fire intensity on above-ground vegetation and below-ground soil properties (Adapted from Parson et al. 2010)

Post-fire hydrological effects have generally been studied at small rather than large scales, with soil water repellency effects on infiltration and overland flow being a particular focus. At catchment scales, post-fire magnified peak-flow has received more attention than changes in total flow, reflecting easier measurement and the greater hazard posed by the former. Post-fire changes to stream channels occur over both short and long terms with complex feedback mechanisms, although research to date has been limited (Shakesby and Doerr 2006).

5.6 Soil Degradation Threats and Countermeasures

Degradation deteriorates soil quality and, thus, ecological soil functioning by partially or entirely damaging one or more of its functions (Tóth et al. 2008). The main threats to soil functioning abilities are identified as:

(i) decline in SOM
(ii) soil erosion
(iii) compaction
(iv) salinization
(v) landslides
(vi) floods
(vii) contamination
(viii) sealing

Loss or quality deterioration of SOM are directly or indirectly involved in most of the soil degradation threats listed above, and affect nearly all water-related ES. Therefore, sustainable soil management is essential for protecting water resources (Lal 2010). Ensuring the proper functioning of soils in a catchment is a major target within the frame of integrated land and water resource management (Calder 2005).

Loss of SOC leads to decreased cohesion between soil particles, which increases the susceptibility of soil to water or wind erosion, accelerates losses of bulk soil, and alters nutrient and water cycling. Degradation of soil structure reduces the soil volume for water storage and soil permeability for drainage. In turn, this can lead to greater volumes of overland flow, which exacerbates flooding and reduces groundwater recharge during rain events. Reduced groundwater recharge aggravates water shortages and drought conditions.

Keeping C in the soil is thus of utmost importance for ecosystem and agricultural sustainability and resilience (Lal 2009). A common source of an increase in OM is through an increase in plant and animal residues present in the soil. This may occur in an agricultural scenario through organic residue management methods, such as the adoption of conservation tillage or mulching practices. This could occur in a (semi)natural landscape through changes in land use/cover, such as the conversion of agricultural lands to forestry or establishing agroforestry (Nair et al. 2009) instead of cropland systems (see Sect. 5.7).

Some of the primary effects of this increase in SOC are an increase in soil microbial and faunal activity, a loosening of the soil structure through the addition of coarse material, and as a result an increase in the formation of crumb-type aggregates, macropores, and channels. Permanent plant cover and/or adding mulch or plant residue will protect the soil surface from rainfall impact and solar radiation. In addition, the increase in soil microbial and faunal activity will have the secondary impact of increasing the production of stable soil aggregates.

These primary and secondary impacts will translate into multiple beneficial effects on the soil system (Fig. 5.5) and, thus the partitioning into green and blue water (Sect. 5.2). The increased protection of the soil surface will reduce losses to evaporation from the soil on the surface and decrease erosion from rainfall impacts. A reduction in soil surface evaporation will be particularly significant in dry regions, where more than half of gross precipitation can be lost through surface evaporation. The protection of the soil surface in conjunction with an increase in coarse OM will encourage an increase in the production of burrows and macropores (i.e., through enhanced earthworm activity). Increased amounts of humic matter will lead to an increase in soil water holding capacity (particularly in sandy soils), and a higher adsorption capacity, increased soil aggregate stability, and a greater number of macropores. A higher adsorption capacity will result in the binding of more organic compounds and a greater inactivation of toxins and pesticides (Fig. 5.4). The improvement of soil stability and macropores will result in an increase in soil infiltration capacity and a reduction in surface runoff, with a decrease in soil erosion (or soil mass movement).

The beneficial effects on the soil system will then ultimately lead to positive impacts on the provision of HES. The reduction in evaporative losses from the surface along with increased water holding capacity means that more water will be conserved by the soil system, and made available for plant use and thus biomass production. In an agricultural system, this has the benefit of less water being required for irrigation and increased crop yields, and in a natural system for greater plant growth and productivity. Increased infiltration capacities result in more water entering the soil leading to more even streamflow through the regulation of baseflow. Another potential benefit of increased infiltration and decreased runoff is a reduced flood formation potential by routing water through sub-surface pathways (Wahren et al. 2009). However, this impact is likely to be reduced in the case of major events where the soil infiltration capacity is exceeded (Wahren and Feger 2010).

5.7 Role of Land-Use Changes

The effect of soil C and related soil processes and functions on HES mainly depends upon on the vegetation cover, which is typically a function of land use. The current rate of change in SOC is mainly attributable to worldwide land-use intensification and the conversion of new land for crop production (Foley et al. 2005; Lal 2010). Modern industrialized crop production relies on monocultures of highly efficient cash crops, which generally create a negative C budget. Alternative uses of crop residues for fodder, fuel or industrial applications exacerbate this trend of decreasing C return to the soil. Crop type also plays a role, for example the cultivation of soybean [*Glycine max* (L.) Merr.] monocultures, which have recently spread widely, accelerates SOM losses because their scant crop residues provide less cover to protect soils from wind and water erosion, are highly labile, and are rapidly oxidized to CO_2.

The conversion of forests or natural grasslands to cropland typically leads to losses of SOC. Traditional tillage methods break up soil aggregates, increase aeration, and, thus, enhance the decomposition of SOM particularly in cases of industrialized crop production, which relies on monocultures crops and intensive tillage practices (UNEP 2012). This process is further magnified in tropical regions, which have undergone significant conversion to cash-crops, coupled to warm climates with high decomposition rates (Henry et al. 2009; Don et al. 2011). Here, agroforestry can offer many possibilities to increase soil and water related ESs (Nair et al. 2009).

An ecosystem type with a particularly strong relationship with C stocks is peatland. Such wetlands provide many important ESs, including water regulation at the landscape scale, and C sequestration and storage (Joosten et al. 2012). Under natural conditions, peatlands tend to conserve SOC since the reduced oxygen availability in wet soils slows the decomposition of OM by soil microbes (Joosten 2009). In contrast, drier and well-aerated organic soils promote more rapid decomposition and accumulate less SOM. Therefore, when peat bogs are drained there is a rapid oxidation of stored SOM, which releases large amounts of CO_2 into the atmosphere. Furthermore, peatland degradation may also result in a higher export of DOM into the groundwater or streams (Kalbitz and Geyer 2002). Therefore, it is highly beneficial to restore degraded peatlands by rewetting, reforestation (where appropriate to the natural ecosystem conditions such as in the tropics), and subsequent conservation and/or adopted use (e.g., by paludiculture). Thus, restoration of peatlands may reduce greenhouse gas emissions, improve water regulation and water quality, benefit biodiversity, and opens other income options for local stakeholders (Joosten et al. 2012).

5.8 Conflicts and Tradeoffs

Ensuring water security depends on how well we can cope with an increasing demand both for water, land, and food (Rockström et al. 2012). Therefore, in water resources management it is important to consider what effect changes in management practices related to SOC will have on water resources (Calder 2005; Falkenmark and Rockström 2004; Lal 2010). In addition to crop improvements, conserving water to meet present and future water requirements (Falkenmark 2002; Rockström 2003) and making every drop count (Finkel 2009) are all important concepts for advancing food security. While emphasizing food supply is critical, there is also a need to address the demand side of food production (Lal 2010). One of the key-aspects is the relation between water security and food security. The agricultural sector is the largest water consumer (about 70 % of global water supply). Currently, about 45 % of global food production is generated without a water management system, and 55 % with either irrigation or drainage systems. However, it is projected that global agricultural yields have to be doubled during the next 25–35 years to meet increasing demand, and that about 90 % of this increase will have to be realized on existing cultivated land. This can only be achieved with a more efficient use of water resources and a substantial improvement and extension of water management systems (van Hofwegen and Svendsen 2000). According to Rockström et al. (2012), on the planetary level there is an evident conflict between the two parallel tasks, to (i) increase food production that depends on green water, and (ii) to secure blue water for human consumption and aquatic ecosystems. This may limit the potential for C sequestration through actions such as afforestation as a major climate change mitigation activity. Likewise, a strong increase in forest cover to combat soil erosion may be critical in dry regions. Due to the increased evapotranspiration (green water) of tree vegetation, the availability of blue water in such catchments can become critical (Wang et al. 2011). Hence, tradeoffs between different sectors/stakeholders have to be defined and included in land-use policies (Calder 2005; Wang et al. 2012).

5.9 Conclusions

Securing water resources and protecting hydrologic ecosystem services (HES) presents a significant challenge for the coming decades. HES encompass a range of different environmental goods and services. Their underlying sustainability however depends upon the complex interrelationships between land-use/cover, soils, the hydrologic cycle, climatic conditions, and socio-economic settings. A key component in addressing future environmental and development challenges is developing a comprehensive understanding of these complex relationships and to integrate them into simulation models. There has been considerable progress with plot-scale hydrologic models that are process-based and are able to quantify the effects of changed plant and soil properties (e.g. related to SOC). As many HES unfold their full potential not on defined plots or hill-slopes, but on larger spatial scales (i.e. catchments and basins) modelling of water and element fluxes has to be developed further as a valuable tool to link processes on various temporal and spatial scales. The application of such models on the landscape scale using GIS and regionalization techniques remains a major research challenge (Pilaš et al. 2011).

The soil-water interface is critical in determining the relative distribution of "blue" and "green" water usage for a given region, and, therefore, must be considered in the assessment of HES provisioning. A key element in this distribution is the amount and quality of soil organic matter (SOM) present. Organic C is the main constituent of SOM that has many direct and indirect effects on soil physical, chemical, and biological properties that control water storage and transport as well as filtering, buffering, and transformation. A loss or deterioration in the quality of SOM is directly or indirectly involved in most soil degradation threats and affect nearly all water-related ES. Therefore, sustainable soil management is essential for sustainable water resources management.

Concerns related to climate change have led to increased attention to soil C, and its potential role in C sequestration efforts. Given this potential for changes in SOC stocks, it is important to consider what impact this could have on the provision of HES in order to understand points of synergy and any potential trade-offs. Among the major processes influencing both water quantity and quality at the river basin scale are changes in land cover and land-use, notably the trend towards the intensification of agricultural production. Thus, the provisioning of HES in a wider context must be integrated into coordinated planning of resource and land management at the appropriate landscape scale (i.e., the watershed). By understanding potential points of synergy and conflicts, tradeoffs between different sectors/stakeholders can be defined and included in land-use policies in a comprehensive form. Such a process may benefit from information resulting from integrated catchment modelling that systematically assesses land management/soil-feedback scenarios.

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Chapter 6
Forests, Carbon Pool, and Timber Production

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Abstract Forests play an important role in the mitigation of climate change, and store substantial amounts of carbon (C). The living biomass contains 363 Pg C and the soils an additional 426 Pg C. Given that forests annually exchange about sevenfold more carbon dioxide (CO_2) with the atmosphere by photosynthesis and respiration than is emitted by burning of fossil fuels (currently 9.1 Pg C), the role of forests in the global C cycle is significant. Land-use change contributes 10 % or 1.1 Pg C to the annual CO_2-emissions and leads to significant changes in the C pool. Presently, the temperate forests are a C sink because the forest area increases annually by between 0 and 0.5 %, and the productivity of forests is increasing. Deforestation in the tropical zone is a source of CO_2. Ecosystem disturbances such as storm damages and insect infestations are causing economic loss, and destruction of forests leads to the loss of numerous ecosystem services. Disturbances are partially a component of natural ecosystem dynamics, partially they are triggered by climate-change effects, and partly by changes in forest management. The different effects are often difficult to disentangle. Foresters respond to climate change by developing strategies of adaptative forest management. The opinion on successful concepts is still unconsolidated, both due to differences in the anticipation of the extent of climate change, and due to different opinions on the resilience of different forest types. Simulation models and manipulative experiments are important tools for the development of strategies of adaptive forest management. With respect to the role of forests in the global C cycle two opposing opinions exist. Firstly, it is possible to focus on C sequestration in standing forests, alternatively, forest biomass can be intensively used in order to provide timber for the substitution of other materials, and forest biomass for energy. From a forester's perspective the active management of forests offers more opportunities than management towards old-growth forests with maximized C stocks in the standing biomass. Intensive forest management needs to follows the principles of sustainability. This paradigm is instrumental in forest politics. The use of criteria and indicators help to approximate and maintain a desired status of forest ecosystems.

Keywords Carbon • Forest ecosystem • Land use • Disturbance • Ecosystem services • Adaptive forest management • Sustainable forestry

6.1 Carbon in Forests

Forests play a major role in the terrestrial C cycle and in our efforts to manage the amount of C in the atmosphere. Forestry is, therefore, an important element of a voluntary C market and of government efforts of mitigating climate change. Presently, global forests are major terrestrial C sinks, whereas grasslands, crop lands, and many peatlands are C neutral or even a net source of CO_2. Accordingly, C sequestration in forest is an ecosystem service receiving considerable attention. Inventory data from around the world show the distribution of C sources and sinks, the importance of temperate and boreal forests as current sinks and the enormous

Table 6.1 Forest area, estimated total carbon pools in the forest biomass, soil carbon pool in forest soils to a depth of 1 m and carbon density for different climatic zones (Pan et al. 2011)

Climatic zone	Area (10^6 ha)	Living biomass (Pg C)	Dead biomass (Pg C)	Soil (Pg C)	Total (Pg C)	Carbon density (MgC ha^{-1})
Boreal	1,135	54	16	193	272	239
Temperate	767	47	3	69	119	155
Tropical	1,949	262	546	155	471	242
Total	3,851	363	73	426	861	224

1 Pg = 10^{15} g

fluxes (sources and sinks) contributed by tropical forests. Currently, forests absorb about 27 % of the annual CO_2 emissions from fossil fuels and are providing an arboreal discount on CO_2 emissions (Le Quéré et al. 2009). The absorption of CO_2 comes on top of a number of other ecosystem services that are provided by forests (Shvidenko et al. 2007). Factoring in the oceans and other terrestrial ecosystems the total absorption increases to over 50 %. Without these natural sinks, as result of increasing anthropogenic CO_2 emissions, the rate of CO_2 increase in the atmosphere would be substantially higher (Pan et al. 2011). Nevertheless, the terrestrial ecosystems encompass only one of several big sinks of CO_2. In the past 50 years, the fraction of CO_2 emissions remaining in the atmosphere has likely increased, presumably due to a declining sink strength of the oceans (Le Quéré et al. 2009).

6.1.1 Pools and Fluxes of Carbon

Organic C in forests is stored in the biomass of living and dead trees, in the forest floor and in the mineral soil. Forests are the only vegetation type where the biomass is a similar sized C pool as compared to the soil. By far the largest C pool is found in tropical forests where the deforestation dynamics are most dominant and where the available database on C stocks is partly sketchy (Dixon et al. 1994; Bonan 2008). In boreal and temperate forests the soils may store more C than the tree biomass, whereas tropical soils may contain about half as much C as the tree biomass. The total C pool in forests exceeds the amount of C in the atmosphere (Denman et al. 2007). Temperate forests that are thoroughly investigated store less than 15 % of the global C pool in forests. Due to the intense management the relative amount of deadwood is by far smaller than in other climatic zones (Table 6.1).

Many countries have access to long-term forest inventory data enabling to establish solid estimates of the C pool in the tree biomass (Lindner and Karjalainen 2007; Nabuurs et al. 2010; Tomppo et al. 2010). Their main purpose is traditionally the estimation of the harvestable stem volume. With biomass expansion factors and functions an estimation of the C pool in the tree biomass is possible (Zianis et al. 2005; Enquist and Niklas 2002). Substantial data gaps exist in tropical areas and in extensively managed boreal forests. For these regions, remote sensing approaches are the method of choice.

The soil organic carbon (SOC) pool contributes about 50 % to the total forest C pool, and exceeds in boreal and temperate forests even the C pools in the biomass. Forest soil inventory data are less abundant than information on the aboveground forest biomass. The data on temperate forest ecosystems in Table 6.1 are well supported (Food and Agricultural Organization of the United Nations 2006; Tomppo et al. 2010; Smith et al. 2009). Data gaps in the C pool in wetland soils and frozen soils prevail. Soil organic matter (SOM) is extremely heterogeneous with respect to its chemical properties, its decomposability, and consequently its age. Moreover, the small scale variability in the field is large. SOM forms from residues of plants, soil microorganisms, and soil fauna. The litter from roots is as relevant as the C input from needles and leaves. The assessment of the SOC pool may be derived from the total content of organic C, soil bulk density, rock content and soil depth as parameters, where soil bulk density is often estimated with pedotransfer functions (Rodeghiero et al. 2009; Schmidt et al. 2011). Remote sensing techniques are so far not useful for the assessment of SOC pools.

The major C fluxes are the fixation of C in the biomass by photosynthesis, the release of CO_2 by autotrophic and heterotrophic respiration, and harvesting. Photosynthesis and respiration depend on the productivity of the forest and hence ultimatively on the climate. The turnover of C in the living biomass is fast and highly sensitive to ecosystem disturbances. The biomass pool can expand only slowly by an increased growth rate and a lower frequency or severity of disturbances, but can be degraded quickly (Körner 2003). The net C fluxes within a particular period can be directly measured, e.g., by the eddy covariance method, or assessed from repeated inventories, or estimated by modeling approaches (Baldocchi 2003; Luyssaert et al. 2010). A large quantity of SOC is allocated to an arbitrarily defined inert pool. In the modeling exercises the soils are mostly represented by pools of different turnover rates. This approach is transparent but does not reflect the current understanding of SOC dynamics (Friedlingstein et al. 2006; Schmidt et al. 2011). While the eddy covariance technique provides the CO_2 budget of the entire ecosystem, the elements of the C fluxes need to be measured separately, such as C fixation via photosynthesis, litterfall, and soil respiration.

The quantification of C pool changes from repeated inventories is accurate for the aboveground biomass, because the relevant tree parameters such as stem diameter and height are directly measured. The estimation of the belowground biomass is less certain because it is often based on biomass expansion factors and functions that are supported by much fewer measurements. The assessment of a regional SOC pool is also uncertain because in most regions very few forest soil inventories are conducted. In undisturbed forest ecosystems, the decadal SOC pool changes are small and difficult to detect against the background of the large SOC pool. This challenge offers ample room for the application of SOC simulation models. The mechanistic understanding of the persistence of SOC is incomplete. The stability of the SOC pool depends not only on intrinsic soil properties, but is greatly controlled by the surrounding soil biological ecosystem. However, a precise assessment of

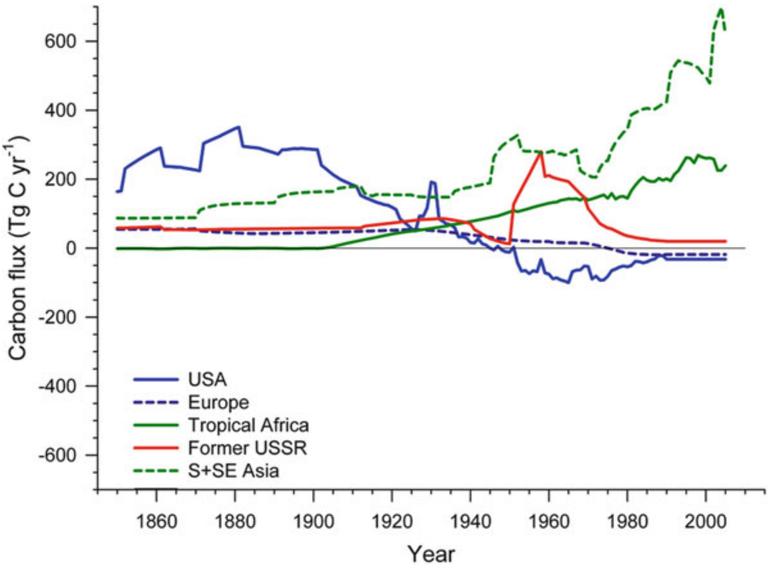

Fig. 6.1 Annual net flux of C to the atmosphere from land-use change for several regions (Data from Houghton 2008)

C pool changes is indispensable in the context of the monitoring and verification of the performance of C sequestration projects, and for the reporting of national greenhouse-gas (GHG) inventories.

6.1.2 Land-Use Change Effects

The size of the C pool in an ecosystem may be increased by improved land management and by the change in land use. By far larger than any modification of the land management is the effect of land use changes, and 30 % of the land surface has undergone anthropogenic land-use change (Houghton 2008; Schils et al. 2008). The replacement of forests by agricultural crops reduces the C pool in the biomass instantaneously, and leads to a decline in the SOC pool in the long run (Don et al. 2011). Deforestation and an exploitative use of natural resources is a known phenomenon of economies in transition. In Fig. 6.1, the C fluxes due to land-use change between 1850 and 2005 are shown. Europe had many periods of intense deforestation, starting already as soon as 1,000 years BC. The pattern of reforestation and forest loss reflects the demand for cultivatable land, political turmoils, and the effects of the bubonic plague. The reforestation/deforestation history in the Americas is clearly linked to the collapse of the indigenous populations and the arrival of Europeans (Pongratz et al. 2011). As a general pattern, it is proposed that deforestation rates are high in the initial phase of the development of a society, but decline as the knowledge in agriculture is enhanced and crop production is increasingly performed on soils with high productivity. In Europe, the temporal development of land cover and population density was uncoupled during the last century (Kaplan et al. 2009; Mather and Needle 1998; Reick et al. 2010).

The deforestation in the tropical zone is slowing and is currently about 1 Pg C year^{-1} or 10 % of the total CO_2 emissions (Friedlingstein et al. 2010). The downward trend of CO_2-emissions due to land-use change is partly a consequence of legal actions against deforestation in many regions (Tollefson 2012). Due to the implementation of improved soil management methods the C release is expected to be smaller than in the past. Nevertheless, soils will continue to lose C due to climate and land-cover changes (Eglin et al. 2010). With afforestation programmes the trend can be reversed. In the current offset of GHG emissions, temperate forests have a prominent role. The capacity of C sequestration due to afforestation is, however, limited. Replenishing the C pool is in some instances a one-time event such as the recovery of European temperate and boreal forests after intensive forms of land use, particularly in mountain areas and on marginal agricultural land. In addition, large tracts of Eastern European and Russian agricultural land were abandoned and have since reverted to forest (Vuichard et al. 2008). In China, million ha of afforestation land that had been used for other purposes or was barren, has now been turned back into forest (Food and Agricultural Organization of the United Nations 2006; Führer 2000; Seppälä et al. 2009; Wang et al. 2007). It will be difficult to find open land for afforestation in the future for maintaining the current sink, especially when considering an increasing demand for agricultural land for the production of food, feed, fibre, and fuel.

The major effect of C sequestration in afforestation programmes is mainly the quick accumulation of aboveground biomass. In the soil, the main effect is the formation of an organic soil layer, predominantly under coniferous forests. The accumulation of organic material has a duration of few decades and depends on the history of site management and on physico-chemical soil properties. The effect of land use change from agricultural land towards forests on the C stock in the mineral soil is small and simulation models tend to show a quicker response than indicated by field observations (Karhu et al. 2011; Laganière et al. 2010; Vesterdal et al. 2011).

When managing land with the objective of maximizing C sequestration there are several points for scrutinization. Afforestation/reforestation projects can decrease the biodiversity. An example is the gradual encroachment of forests into high-elevation pasture land in mountain regions of the temperate zone (Thuiller et al. 2005). The newly establishing forest ecosystems harbor less herbaceous species than the previous grassland ecosystems. In addition, the greater homogeneity of the landscape can reduce the scenic beauty. This argument is relevant in touristic areas. In regions with an intermediate snow cover such as the temperate and the boreal zones and many mountain ranges the effect C sequestration effect may be partially offset by the change in the surface albedo. A snow cover effectively reflects incoming radiation whereas the darker canopy of a forest plantation would absorb it. The climate forcing by albedo cooling dominates warming as a result of CO_2 emissions, particularly in the boreal region (Anderson et al. 2010). A forest cover would, therefore, have

Fig. 6.2 The dark surface of the forest canopy may absorb more radiation than the effectively reflecting snow cover on agricultural land and range land (*Picture*: *Robert Jandl*)

a warming effect despite the enhanced fixation of atmospheric CO_2 in biomass and soil (Fig. 6.2). An alternative concept to the maximization of the C pool in the living tree biomass is sustainable forest management because it ensures a continuous flow of wood products and bioenergy while maintaining or increasing the C stock in the ecosystem.

6.2 Climate Change

Climate change manifests itself in different ways. Most obvious is the increase in the global mean air temperature. More difficult to verify is the change in the precipitation regime, and the changing frequency of drought events. In addition, forests are responding to the fertilizing effect of the increasing partial pressure of CO_2 (Calfapietra et al. 2010). The currently observed warming trend is by far higher than during the paleocene-eocene warm period, and modern atmospheric CO_2-concentrations probably not occurred in the last 800,000 years (Kump 2011; Lüthi et al. 2008). Forests are an important element of climate-change mitigation efforts due to their ability to sequester C, especially when new areas are afforested or reforested, and due to the production of biomass for renewable energy replacing fossil fuels. Considering how well trees cope with seasonal changes and interannual variations of weather conditions it can be assumed that forests are able to adapt to slow changes in the site climate. However, rare extreme events have a strong effect on the development of ecosystems. The character and severity of impacts from extreme events depend on the extremes themselves, and on the exposure and vulnerability of the forests. Globally, a high probability of an overall decrease in the number of cold days and nights, and an increase in the number of warm days and nights has been predicted (IPCC 2012; Rahmstorf and Coumou 2011). There have been significant trends in the number of heavy precipitation events in some regions. While it is not currently possible to reliably project specific changes at the catchment scale, there is high confidence that changes in climate have the potential to seriously affect water management systems. However, climate change is in many instances only one of the drivers of future changes, and may be not necessarily the most important driver at the local scale (IPCC 2011).

The emissions of GHGs are still rising and so far it has not even been possible to reverse the trend. From 2000 to 2009, the C in the atmosphere rose annually on average by 4.1 Mg, and in 2010 the global C emissions increased by an unprecedented high rate of 5.9 %. The rate of increase in CO_2 concentrations in the atmosphere has never before occurred on earth (Kump 2011). For mitigation of climate change, forests will play an important role although it is well understood that the mitigation potential of forests is not the ultimate solution. A modeling exercise has clearly shown that the efficiency of terrestrial ecosystems to absorb the anthropogenic C perturbation will decline in a future climate. A larger fraction of anthropogenic CO_2 will stay airborne if climate change is accounted for. Should climate change advance too far and terrestrial ecosystems transform from C sinks to sources, and begin releasing vast quantities of C into the atmosphere, human efforts to mitigate climate change may be overwhelmed. Drastic emission reductions are required to limit global warming to the 2°C in order to reduce climate change risks, impacts, and damages (Friedlingstein et al. 2006; Meinshausen et al. 2009; Peters et al. 2011; Smith et al. 2012).

6.2.1 Forests in a Warmer World

Forests have proven to be able to adapt to changes in environmental conditions very well in the past. Only therefore have they been able to inhabit about 30 % of the global land areas (Food and Agricultural Organization of the United Nations 2006). Climate change may exceed the adaptive power of forests. There may be two vastly different possible responses of forests to climate change. An optimistic scenario indicates that in a warmer world, trees will grow faster, prosper in a longer growing season, flourish in areas where they had never before grown, take up more C and increase the C sink. A pessimistic scenario is that warmer temperatures will cause more forest fires, more insect infestation, more dead and dying trees, more peatland decomposition, thawing of permafrost and the transformation of forests into C sources. Both avenues of argumentation are possible based on experimental evidence. Thus, the effects of climate change on forests C storage may strongly depend on the geographical location of the forest (Lindner et al. 2010).

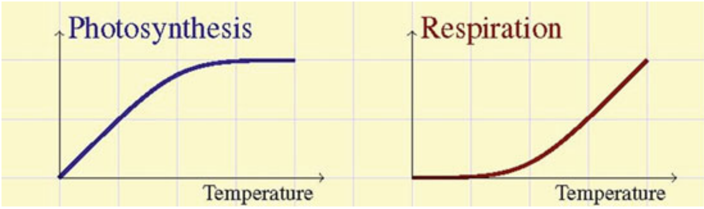

Fig. 6.3 Idealized responses of plant photosynthesis and soil microbial respiration to increasing temperature

Processes in forest ecosystems do not respond equally to warming. Figure 6.3 schematically shows that the warming effect on the productivity of trees tends to saturate with increasing temperatures whereas the heterotrophic respiration of soil microorganisms accelerates. Such textbook knowledge often fails to describe ecosystem dynamics. Under field conditions the temperature dependence can be modified by effects like drought and ecosystem disturbances, and by internal factors such as soil properties. However, generally a quicker turnover of C in terrestrial ecosystems is projected (Friedlingstein et al. 2006).

The heat wave in parts of Europe in the summer of 2003 has offered the opportunity to quantify the response of different tree species to hot and dry conditions (Ciais et al. 2005). In Germany, the productivity of Norway spruce (*Picea abies* (L.) Karst.) declined and did not return to previous levels in several years, partially because the summer of 2006 was also hot and dry. The growth decline of Scots pine (*Pinus sylvestris* L.) was by far smaller and mostly confined to 2003. The growth pattern of pine reflects the condition during the growing season and shows a quick recovery when the climate conditions return to average levels. European beech (*Fagus sylvatica* L.) did respond in the year after the summer drought with a strong growth decline and recovery phase of more than 3 years. In contrast, oaks (*Quercus sp*. L.) did not respond at all. These thermophilic species are obviously not affected by hot summers (Beck 2010). In a modeling exercise, the heat wave of 2003 led to reduction of the gross primary productivity over Europe of 30 %, and turned the terrestrial ecosystems into a temporary C source, reversing the effect of 4 years of C sequestration (Ciais et al. 2005).

6.2.2 Climate Manipulation Experiments

Simulation models play a dominant role in the prediction of the response of forests to climate change (Heimann and Reichstein 2008; Kurz et al. 2009; Luyssaert et al. 2010). There is broad consensus that a mere modeling approach is insufficient because models incorporate the state-of-knowledge and are built based on data of past experiences. Unprecedented process interactions are, therefore, likely not sufficiently represented. Moreover, the substantial simultaneous changes in forest ecosystems that may occur, such as climate change, nitrogen enrichment and the accelerated migration of potential pests as consequence of global trading of goods limits the opportunity to learn from the past. Field experiments are, therefore, important for constraining the modeling results and to indicate ecosystem responses to changing conditions that are not deductible from existing knowledge.

A challenge in climate change manipulation experiments is the size and longevity of mature forest ecosystems. One remedy is to focus on the compartments of the ecosystem that are considered to be particularly relevant. For the assessment of soil processes under conceivable warmer conditions, soil warming experiments have been set up in different forest ecosystems (Hagedorn et al. 2010; Jarvis and Linder 2000; Melillo et al. 2002; Schindlbacher et al. 2009). Their results are divergent with respect to the induced increase in CO_2 release from the soil. At some sites the increase in soil CO_2 emissions was transient and slowed down as the resource of easily decomposable SOM was exhausted. In other experiments warming elevated the CO_2 emissions from soils for many years. Another avenue of climate manipulation experiments is the reduction or exclusion of precipitation in order to simulate dryer conditions. Simulated drought strongly decreased the decomposition of SOM in temperate forests, whereas less to no effects on the soil CO_2 efflux were observed in the tropics (Davidson et al. 2008; Muhr and Borken 2009). In a temperate forest, the forest floor can become increasingly water repellent during drought and soil CO_2 emission were found to be substantially reduced for much longer as the actual drought lasted (Schindlbacher et al. 2012). Simultaneously, the uptake of C by tree-growth does not suffer during periods of reduced precipitation as tree roots can tap into deeper water saturated soil horizons. A combined warming-drought manipulation experiment showed that the increase in soil CO_2 emissions due to warming can be easily offset by the reduced soil CO_2 efflux during and after drought (Schindlbacher et al. 2012). The current trend towards multi-factorial climate manipulation experiments and attempts towards integrating the whole tree environment is a step forward to refine process related parameters in ecosystem models (Bronson et al. 2008; Niinistö et al. 2004). A universal manipulation experiment which includes changing climate parameters, increased atmospheric CO_2 levels and changes in nitrogen deposition in a mature forest has so far not been performed. It would challenge experimentalists technically and financially, but might be worth being conducted, as the compartments of forest ecosystems are closely interlinked and the total effect is not necessarily a linear combination of the individual effects. Such multi-factorial experiments need to be set up as parts of a long-term project, ideally extending beyond an entire rotation period (Fig. 6.4).

Another avenue of climate change experiments is the exposure of trees to elevated concentrations of CO_2. Several Free-Air-Carbon dioxide-Enrichment (FACE) experiments have often been conducted with young and vividly growing trees (Norby and Zak 2011). A maximum stimulation of tree growth occurred at concentrations up to 560 ppm unless other limiting factors such as water shortage and insufficient nutrient supply set in. For the long-term C sequestration potential of forest ecosystems the FACE experiments are not entirely relevant because the response to elevated CO_2 is probably transient (Körner et al. 2005).

Fig. 6.4 Automated chambers for the measurement of the soil carbon dioxide (*left*) and a roof to temporarily create drought conditions in a field experiment (*right*); Experimental site: mature, spruce-dominated forest, Mühleggerköpfl, Achenkirch, Tyrol (*Picture: Andreas Schindlbacher*)

6.3 Natural Disturbance of Forest Ecosystems

Complex ecosystems such as forests do not tend towards a state of equilibrium. Natural disturbances are part of forest ecosystem dynamics (Drever et al. 2006). The main disturbances in forests are the attack by damaging insects and pathogens, stand destruction by strong winds and snow, wildfire, and damages due to herbivores (Fig. 6.5). In an ecological context, disturbance is part of the natural forest dynamics, and the ability of an ecosystem to tolerate disturbances without undergoing fundamental changes is its "resilience" (Holling 1973). In regions of intensive forest management silviculture and harvesting has modified the natural disturbance patterns. Whether or not the frequency of natural disturbances is increasing and what factors are eventually responsible is subject to an ongoing debate. The available data on forest damages due to disturbance are by far less abundant than those due to harvest (Van Miegroet and Olsson 2011). The focus here is on disturbances caused by storm and insects. Fire is of regionally different relevance and the dynamics may be affected by many decades of fire suppression in actively managed forests (Certini 2005; Marañón Jiménez 2011).

6.3.1 Insects and Pathogens

Warmer site conditions and dry summers may accelerate the propagation rates of insects and pathogens, and also weaken host trees thereby increasing the susceptibility for attacks. Infestations by bark beetles, especially *Ips typographus* (L.), the dominant threat for Norway spruce forests in Central Europe, are projected to increase because under warmer conditions more generations by year develop (Wermelinger 2004). Especially vulnerable are secondary spruce forests, often established for economic reasons. The sites are often warmer than the ecological

Fig. 6.5 Natural disturbances are a part of ecosystem dynamics that are mostly excluded by active forest management; storm-damaged Norway spruce (*Picea abies* (L.) Karst.) forests in the High Tatra, Slovakia; the picture shows the situation in 2009, 5 years after the storm (*left*) and forest infested by bark beetle (*Ips typographus* L.) in Austria (*center*), burnt pine (*Pinus pinaster* L.) and (*P. nigra* L.) forests in the Sierra Nevada National Park, Spain. The Lanjarón wildfire occurred in 2005, the picture was taken in 2008 (*right*) (*Pictures: Magda Edwards, Christian Tomiczek and Sara Marañón Jiménez*)

Fig. 6.6 The Asian longhorned beetle; *Anoplophora glabripennis* (*left*), bark beetle; *Ips typographus* (*center*) and larva of the oak processionary; *Thaumetopoea processionea*, a moth (*right*) (*Pictures: Christian Tomiczek, Ute Tomiczek-Hoyer, Martin Brandstetter*)

optimum conditions for spruce. In addition, the global trading of wood and wood products facilitates the immigration of non-native insects and pathogens (Fig. 6.6). A prediction on the future pressure on forest trees is difficult but the concerns for more frequent epidemics are substantiated. The invasion of pathogens in new environments is not a novel process. The Dutch elm disease, caused by the ascomycete *Ophiostoma ulmi* (Buisman), and chestnut blight, caused by the fungus *Endothia parasitica* (Murr.), have afflicted tree populations in the United States and Europe in the past. The new challenge arises from the acceleration of these processes due to climate change and transcontinental transport of goods. Some thermophilic fungi that until now have been unproblematic are benefitting from warmer summers and infest trees more readily, summer droughts stress trees and make them more vulnerable to infection by fungi, and warmer winters may increase the activity of some weak pathogens, that are active only when the host is dormant (Lonsdale and Gibbs 2002). In addition, some fungi are performing host shifts that are believed to be a consequence of climate change (Gange et al. 2011). Moreover, the damage from insects needs to be seen in the context of storm damages. Mass propagations of insects are often observed when damaged forest stands are not immediately salvaged. The sufficiently moist stems are offering ideal breeding conditions for insects (Engesser et al. 2008; Hanewinkel et al. 2008).

Examples of a pressure from formerly not present beetles are the infestations of trees with the Asian longhorned beetle (*Anoplophora glabripennis*) and the Citrus longhorned beetle (*Anoplophora chinensis*). Both species are native to Asia and have been introduced into Europe mainly with packaging wood material and ornamental plants from China or other Asian countries. In Europe, the Asian longhorned beetle was first detected in the city of Braunau/Inn in Austria, the Citrus longhorned beetle in Parabiago/Italy (Tomiczek and Hoyer-Tomiczek 2007). In the meantime many infestations and interceptions are recorded all over Europe. At the beginning of the introduction most infestations occurred on park trees in cities. Later on both species showed that they were able to spread to forest stands. This fact was already known from Canada and the USA, where their uncontrolled spreading allowed to invade natural forest landscapes and alter the tree species composition (Dodds and Orwig 2011).

Dealing with an increased pressure from autochthonous and immigrating pathogens and insects requires a complex training of forest practitioners, more intensive efforts for monitoring of insect outbreaks, and the flexibility to quickly adopt countermeasures. For the sake of efficiency the monitoring needs to be trans-national to capture critical events. The global exchange of goods with wood as packaging material and trading of ornamental plants and their seeds is an additional potential vector for insects and pathogens and needs more attention and control.

A well described mass propagation is the outbreak of mountain pine beetle (*Dendroctonus ponderosae*) in British Columbia. Due to the low management intensity of the affected forests the human influence on the dynamics of the disturbance is considered to be minor. The suggestion that the outbreak was triggered by climate change was adopted (Kurz et al. 2008). The immediate effect of the high rate of tree mortality was that the forest sector that previously was a small sink of GHGs, turned into a large source. The forests became significant for the Canadian budget of GHG emissions.

6.3.2 Storm Damages

Records of forest damages indicate that European forests have been noticeable damaged by on average 2 storms by year in the last 60 years (Gardiner et al. 2010). Storm Bora affected 46 000 ha in the Tatra Mountains in 2004, storm Gudrun in 2005 was the strongest storm hitting Southern Sweden in 100 years, downing an equivalent of the total annual harvest of the country. Emma and Paula hit Central Europe in 2008 and many regions have there own history. These storms are responsible for more than 50 % of the abiotic and biotic damage that have been recorded in these years. The extent and location of storm damage and the vulnerability of individual forest stands depends on meteorological conditions, location, site properties, stand composition, and past forest management. Currently, no reliable forecasting tool for storm damages is available. Spruce and poplar (*Populus spp*.) forests have been most severely affected in the past, whereas oak forests were damaged to a lesser degree. The projected future frequency of extreme events and eventual changes in the increase in frequency and severity of storms is controversially discussed, but a southward shift of storm activities is predicted by models (Ulbrich et al. 2009). In European forests the increased standing stock of trees and average forest age has contributed to forest damages (Gardiner et al. 2010; Jonášová et al. 2010; Schmidt et al. 2010; Seidl et al. 2011b).

Practical forestry can adapt to the problem of storm damages by proactive silviculture and knowledge is documented in classic textbooks of forestry (Assmann 1961; Mayer 1984; Oliver and Larsen 1990). Mixed-species forests are generally understood to withstand storms better than mono-species stands, early thinning interventions ensure the development of a favorable tree-height/stem-diameter ratio, and short rotation periods allow to harvest forests that later in their life cycle would become more vulnerable to storm damages. These silvicultural strategies have limits and even ideally managed stands cannot withstand extreme storm events. Once storms have damaged a part of forest the potential for further damages in subsequent events is high. It is often observed that an initial event opens the forest canopy and from the exposed edge of the stand the damaged area propagates into the stand and finally affects a larger area.

6.3.3 Human Influence

Climate change is occurring at a time when natural environments are becoming increasingly fragmented through habitat destruction, and when species are being moved inadvertently or deliberately around the globe at ever increasing rates. This means that climate change is occurring at a time when many forest ecosystems are already under pressure from invading species and disturbances. Fragmentation and invasions may also affect evolutionary processes by changing the way genes move around landscapes and by introducing novel genotypes into populations through hybridization (Hoffmann and Sgrò 2011).

Increased occurrences of disturbances are often linked to climate change and forest practitioners are willing to attribute unfavorable processes to it. However, storm damages have a long history in Central Europe. Already more than a century ago it was shown that secondary spruce forests were particularly vulnerable. The proposed remedy of establishing mixed-species forests has not always been adopted and the high production risk of a spruce-dominated forestry was obviously accepted for economic reasons (Assmann 1961). More recently, Central European forests have been subject to changes that occurred concurrent with global change. The stand density and the average stand age has increased because low timber prices made a proactive form of forest management unattractive. At the same time,

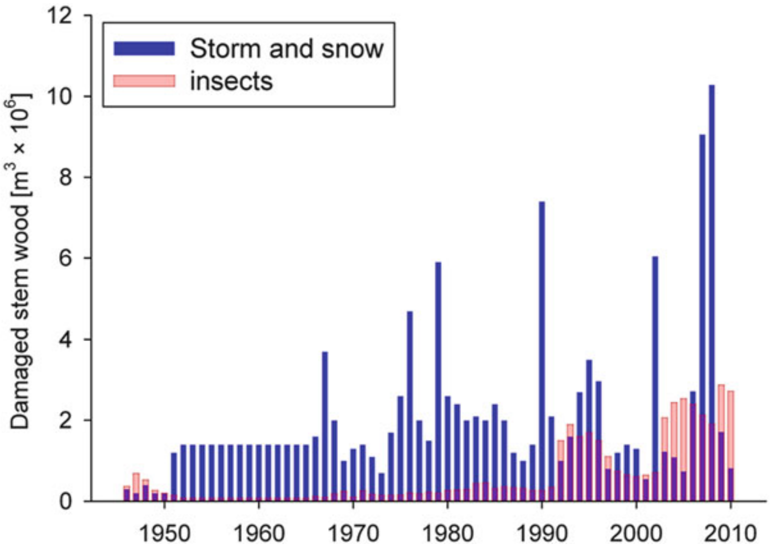

Fig. 6.7 Damaged stem wood volume in Austrian forests from 1946 to 2010. Storm and snow damages show an irregular pattern, damages caused by insects attacks are increasing. Bark beetle (*Ips typographus*) are the main cause for the insect damages (Data: *Christian Tomiczek*)

the growth rate has been increasing and nitrogen enrichment has been playing an important role (Sutton et al. 2011). Forests grew denser and higher and were more vulnerable to storm damage. Economical constraints in forest management led to decreasing monitoring efforts for insect-caused damages. Especially in areas with single-species forests large-scale insect outbreaks were observed. In a detailed study on reported forest damages in European forests it was identified that both storm and insect damages are enhanced by climate change as well as by management changes. The causes for wildfires were more difficult to disentangle due to insufficient data (Seidl et al. 2011b). The negative consequence is that forests are under increasing pressure. A positive aspect is that appropriate forest management strategies can alleviate the pressure partially.

An example for damages due to disturbances for Austrian forests is shown in Fig. 6.7. Storms and snow damages show the expected irregular temporal pattern because they are caused by singular extreme and rare climatic events. Quite typically, insect damages follow because the damaged timber is often not processed in time and is an ideal breeding ground for bark beetles. Bark beetle is indeed the by far dominating insect causing damage to Norway spruce forests. The figure displays the complete reliable record of forest damages in Austria and gives disconcerting evidence that damages due to bark beetle are no longer decreasing to pre-disturbance levels. Since the early 1990s, the insect-related damages are showing an upward trend. Apparently, the permanent stock of bark beetle is gradually increasing.

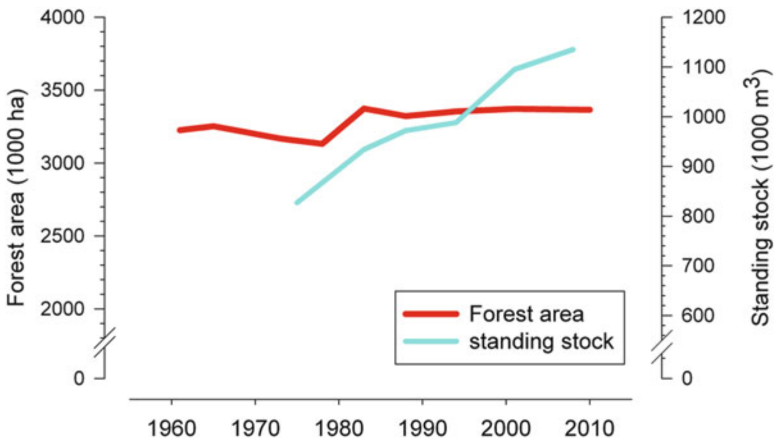

Fig. 6.8 Change of the forest area and the standing stock of stem biomass in Austrian forests from 1961 to 2010 (Data: Austrian Forest Inventory, http://bfw.ac.at/rz/bfwcms.web?dok=4303)

Central European foresters are successfully following the principles of sustainable forest management. The term encompasses a balanced form of forest management that ensure the provision of several forest functions under conditions of temporally differing demands for different forest products. The Ministerial Conference on the Protection of Forests in Europe offered a definition for sustainable forest management (MCPFE 2003):

> The stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems.

An example for the variable meaning of sustainability is shown in Fig. 6.8 for Austrian forests. Forest inventory data indicate that the forest area is slowly and significantly increasing, mostly due to the afforestation of marginal agricultural land. The standing stock increases at a higher rate. Reasons are the increased forest productivity due to nitrogen enrichment and the elongation of the growing season and the increase in stand age (Schadauer 1996; van Oijen and Jandl 2004). The low timber prices have given little incentives to harvest at the same rate as forests have been incrementing and the average stand age has increased. Despite substantial changes in the forests, forest management at any time during the last five decades may have been considered to be an example of sustainable management. Deriving a sustainable harvest rate yields 280 m^3 stem wood for the inventory conducted between 1986 and 1990 and 325 m^3 for the inventory period 2007–2009 (Russ 2011; Büchsenmeister 2011). In just one decade the amount of sustainably extractable timber would have increased by more than 15 % although the observed changes in the forested landscape would have been subtle and hardly detectable.

6.4 Adaptive Forest Management

The term 'adaptive management' was coined by Buzz Holling and encompasses the organization of the implementation of a forest management strategy in a series of alternating workshops and research periods. Adaptive management gives the opportunity of reviewing and modifying opinions and developing options for stakeholders and policy makers (Holling 1978). The concept ensures that the elaborated adaptive assessment and management develops as it proceeds. The approach surveys the features of the environment likely to be affected by the developments under consideration, analyses the information collected, tries to predict the impact of these developments and lays down guidelines or rules for their management. Upon completion of the process, managers receive well-articulated management plans and then monitoring begins in order to later evaluate the management plans.

Forest managers need to address climate change issues because, without action, some of the foreseeable impacts of climate change on forest ecosystems and their ecosystem services are socially inacceptable. Considering the role that forests play on the provision of water, hydropower, protection against natural hazards, it is difficult to imagine the establishment of technical structures, capable of resplacing forests (Beniston et al. 2011). Adaptation measures should either decrease the probability of damage triggered by climate change, identify upcoming opportunities associated with climate change, or increase the resilience of forests. Unsuitable forms of adaptive forest management bear the risk of economic losses, and may compromise the provision of ecosystem services.

6.4.1 Tools

Since more than 20 years forest growth models of different scales (tree level to entire watersheds) have been developed (Weiskittel et al. 2011). Earlier, the drivers of these models were based on steric interactions such as competition between trees for light or nutrients. New models focus on the integration of climate as stressor or catalyst, on species competition, and on ecosystem disturbances. Current research in mountain forest management is focusing on predicting impacts of future climates on forest growth and tree migration, using simulation models and assuming different climate scenarios. Another group of the models used are "management models" where the emphasis is on the description of competition between trees and the growth performance of different species. Another group of models are "succession models" emphasizing the regeneration of particular tree species under changing site conditions (Fig. 6.9). These simulations compare different adaptive forest management strategies. Ideally, such simulations of the productivity of forests are coupled with simulations of the SOC pool. An emerging challenge is the coupling of forest growth models, simulations of natural hazard propagation, and climatic and

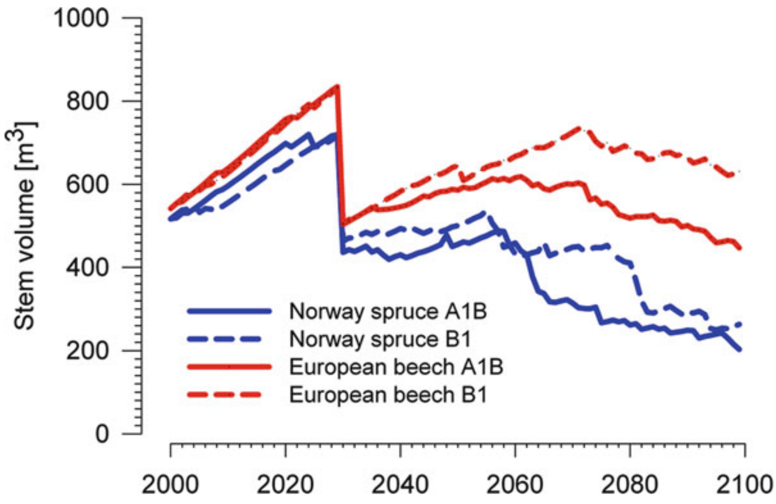

Fig. 6.9 Simulated growth of Norway spruce and European beech under the same management intensities and the site conditions of the Ossiach region in Southern Austria and the IPCC climate scenarios A1B and B1 from 2000 to 2100. The simulation indicates that the climate conditions of A1B are less favorable than B1. The growth performance of Norway spruce is stronger affected by climate change than European beech (*Data: Robert Jandl*)

socio-economical models. Such complex model ensembles enable the comprehensive analysis of strategies of adaptive forest management. The concepts of resistance, resilience and viability of ecosystems and of ecosystem functions have been improved via the use of models. Research increasingly tries to integrate the large uncertainty in how the climate and the timber market will develop, and how the demand for ecosystem services will evolve.

Another important set of information comes from forest management guidelines. Often, those are referring to existing site conditions and it is up to the forest practitioner to draw conclusions on the validity of the concepts under future conditions. The existing guidelines are often geared towards the forestry community of a specific region. Examples are available for Austrian and Swiss forests (Amann et al. 2010; Rigling et al. 2008). The texts are often not translated into English because they have a specific target audience.

An exemplary life cycle of the development of a concept of adaptive management consists of numerous elements. They are the assessment of the present forest management concept, projection of the future stand development, definition of indicators for sustainable forestry, evaluation of the foreseeable vulnerability based on defined indicators, development of future forest management with the intention of reducing the vulnerability. A decisive step is the interaction with stakeholders and the assessment whether the chosen concept of adaptive management performs well (Heinimann 2010; Seidl et al. 2011a).

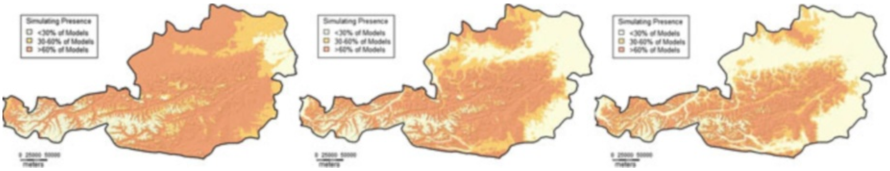

Fig. 6.10 Predicted occurrence of Norway spruce (*Picea abies*) in Austria. The *graph* shows the agreement of an ensemble of simulation models; present occurrence (*left*), predicted occurrence from 2020 until 2050 (*center*), and predicted occurrence from 2050 until 2080 (*right*) (*Figure: courtesy of Niklaus Zimmermann, WSL*)

6.4.2 Tree Species Selection

Adaptation may be an important strategy for natural populations to cope with rapid climate change. Threatened species can persist if they are unable to disperse naturally or through human-mediated translocation to climatically more suitable habitats. This process may also be essential in the case of dominant conifers being attacked by bark beetle populations benefiting from warming conditions. Any adaptation of tree species comes at a cost. It has been shown for birch (*Betula spp.*) and Scots pine that high mortality is able to reduce the adaptation lag between tree populations and the changing climatical optimum. However, adaptation cannot keep up when the environmental change occurs too quickly (Hoffmann and Sgrò 2011). Increasing periods of thermal stress and drought will produce a directional selection for resistance, particularly for species close to their physiological limits. An option to evaluate the ability of tree species in coping with future conditions is the use of *climate envelopes*. In its simplest case, climate envelopes are two-dimensional presence/absence graphs of tree species, derived from large data sets such as national forest inventories with the mean annual temperature and the precipitation as axes. More elaborate evaluations also take into account the influence of present and projected climate extremes and the competition between tree species. An ensemble of model consisting of different climate scenarios and projections of site conditions shows the current and predicted habitat of Norway spruce in Austria (Zimmermann et al. 2009) (Fig. 6.10). Presently, spruce is the dominant tree species. Only the summer-warm region in the east of Austria and the high-elevation zones in the Alps are not suitable for spruce. According to the simulations spruce will gain small areas in high mountains but will lose substantially in lower elevations. Even when scrutinizing the validity of the models the projections raise doubts that a business-as-usual scenario for forest management is justified.

One option of adaptive forest management is re-considering the list of managed tree species. Further, the debate continues whether drawing a distinction between "native" and "non-native" species helps in analysing the factors that affect species change. A potential issue with new tree species is that it is highly uncertain how they deal with new site conditions during an entire rotation cycle. It has been shown that some species do very well in a new environment because they escape their enemies when moving or being forced to move to a new range (Pyšek et al. 2009; Reif et al. 2011). The combination of release from enemies and inherent growth ability produces a synergistic effect that explains patterns seen in the abundance and types of invaders, and in the patterns of invasion seen in many ecosystems (Blumenthal et al. 2009; Seastedt 2009). Due to its high productivity and tolerance towards summer drought, the coast Douglas fir (*Pseudotsuga menziesii* var. *menziesii*) that is naturally not occurring in Central Europe is discussed as alternative to European beech. In Bavaria, Douglas fir covers only 0.7 % of the forest area but plans exist to increase it to 3 % in the next 50 years. It has been studied that mixtures of Douglas fir with other species are possible in order to maximize the delivery of ecosystem services such as timber production and C sequestration (Prietzel and Bachmann 2012). The displayed unsuitability of low-elevation sites for Norway spruce in Austria, as indicated by Fig. 6.10, raises doubts whether native tree species will be able to replace Norway spruce in future commercial forests. The potential natural tree species composition is useful as a reference state for the next decades but may fail for an entire rotation period.

There are ecosystems where the flexibility with respect to tree species is rather narrow. Many mountain forests in the Alps are protecting infrastructures and settlements. Often it is required that the protective function is permanently ensured. A long history of silviculture has shown some optimal tree species combinations and stand structures (Mayer 1976). In continental inneralpine valleys the dominant tree is Norway spruce whereas Cembran pine (*Pinus cembra*) forms the timberline. Presently, there is no known option to introduce tree species that have a similar year-round protective function similar to spruce.

The establishment of mixed-species forests is an integral part of concepts for adaptive forest management. It implies that different species are occupying different ecological niches. Upon a threat to the forest the risk is distributed over several species of different vulnerabilities. Even when one species is strongly affected or killed, a sufficient number of trees and tree species may survive. Thus, it is possible to continue with forest management and the positive effect of soil protection by the forest is ensured. In forest ecology mixed-species forests have been offered as a solution (Mayer 1984; Oliver and Larsen 1990; Puettmann et al. 2009). They have been recommended as remedy against acid rain, biodiversity loss, soil acidification and now as solution against adverse effects of climate change. A positive effect of mixed-species forests is that they allow considering the uncertainty of future site conditions. A mixture of tree species can exploit the nutrient and soil moisture pool of soil profiles better than a mono-species tree stand. The differences in rooting patterns among species stabilize the forest and make it more stable against storm damage. Moreover, new invading insects are expected to have a preference for particular species and may destroy single-species stands more easily than mixed-species stands. Overall, mixed-species stands are expected to offer a higher stability than single-species stands and are, therefore, superior when facing the threats of climate change.

In many European regions the forests are under pressure from a high animal population density. In Central Europe ungulates cause substantial damage to forests, but insufficient countermeasures are taken in many regions because the objectives of the hunting communities is not aligned with forest management. The selective browsing by deer in Central European forests inhibits the development of additional stand-stabilizing tree species and the entire discussion on the benefits of mixed-species forests remains futile. A potential remedy is fencing the newly establishing forests in order to offer protection in the early phase of stand development. However, fencing is expensive and not popular with a society that is entitled to access forests for recreation (Schulze and Schulze 2010).

6.4.3 Silviculture

Current research in forest management is focusing on predicting impacts of future climates on forest growth and tree migration, using simulation models and assuming different climate scenarios. These simulations serve to compare different adaptive forest management strategies. The simulation outcomes are associated with high uncertainties. However, uncertainty is not an excuse for not to act. Research projects integrate the large uncertainty in the interaction between climate change, timber market, and the demand for ecosystem services. Following the principles of adaptive management the collection of expertise from numerous stakeholders such as local forestry experts is required. In this participatory process we have to keep in mind that the future forest conditions may not be deductable from past forest dynamics, considering that numerous processes are changing simultaneously and taking into account that the present and future climate and – as a consequence of elevated rates of nitrogen input – trophic conditions may be unprecedented at many sites (Butterbach-Bahl et al. 2011; Sutton et al. 2011).

Considering the duration for silvicultural adaptation measures the window to a successful adaptation of forest management to climate change successively narrows. When establishing a new forest stand it is required to think about 100 years ahead and to evaluate the suitability of tree species for the entire rotation period. When adaptation to increased insect pressure is required the existing management concepts may be still appropriate even when the monitoring and intervention frequency needs to increase.

The age of forests and the management intensity play an important role on the C stock. Old-growth forests serve as a reference systems for the C retention under site conditions. The dominating position is that old-growth forests have maximum C stocks but negligible stock changes averaged over a region because inputs and losses are in close equilibrium (Franklin et al. 2002; Harmon 2001). The paradigm of zero-stock changes over time has recently been challenged and unexpected productivity rates of mature forests and ongoing increases in the SOC stock have been detected (Luyssaert et al. 2008; Zhou et al. 2006). Consequently a line of argumentation was opened in favor of discontinuing forest management in order to benefit both from maximized C stocks and long-term increases. The concept, however, falls short in accounting for avoided emissions due to different products of forest management (wood products, renewable energy). Sustainable forest management ensures ideally a neutral C cycle on forest land. Wood products can substitute for other goods that would be built from non-renewable materials. The biomass from forests is an important provider of bioenergy. Maintaining the stability of forest stands requires silvicultural interventions whose costs are covered by the revenue from timber sales (Malmsheimer et al. 2011).

6.4.3.1 Afforestation

The question of the choice of tree species propagates into the topic of establishing new forests. When natural regeneration is the method of choice the forest managers opts for maintaining the list of existing tree species and potentially accepting shifting dominances of the species. When strong doubts on the suitability of tree species in a future climate prevail, it may be wise to artificially regenerate stands with afforestation programmes. Such decisions can only be made on the basis of thoroughly established knowledge because errors will turn out costly. In the case that the traditional tree species cannot cope with future climate conditions the follow up costs of required silvicultural interventions and eventually a premature harvesting of the forest will hardly be covered by revenues from timber sale. Alternatively, establishing a costly afforestation even when the traditional tree species would have been successful represents an unnecessary expense. In addition to complete species shifts, afforestation measures might also be implemented to increase the number of tree species and to establish mixed-species forests. In particular, secondary conifer forests often consist of only a few tree species. Mixed-species forest – containing species with high and low risks under future climate conditions – are seen as a viable alternative because the uncertain future risk is proactively addressed and the present strategy of timber production is maintained.

6.4.3.2 Provenances

As another measure to maintain existing forest ecosystems, the planting of different provenances of the same tree species has been suggested (Matyas 1996; Rehfeldt et al. 1999). Up to now, the utilization of local seed sources for reforestation has been regarded as the most efficient way to avoid maladaptation under prevailing environmental conditions. However, for the changing climate recent analyses of provenance tests suggest that seed transfer from well adapted populations or populations with a particularly high potential for adaptation to the expected climate conditions can facilitate the maintenance of the productivity and vitality of presently used tree species (Kapeller et al. 2012; O'Neill et al. 2008; Wang et al. 2006). Although forest trees are sedentary and long-living organisms that have evolutionary developed means to adapt to changing conditions, a number of recent publications

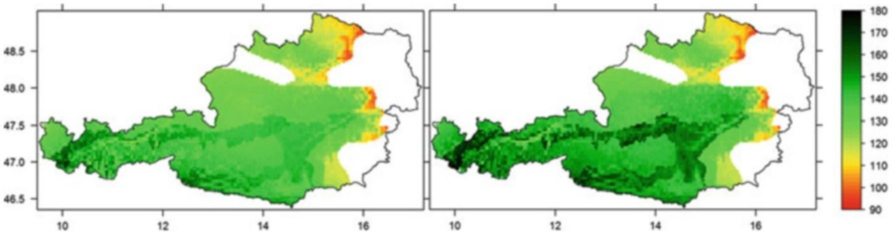

Fig. 6.11 Map of Norway spruce productivity within the current natural distribution range in Austria. The colors indicates the percental change of estimated tree heights from current conditions until the 2080s given that (**a**) local seed material of respective national provenance regions are used (*left*) or (**b**) seed material of the best performing population clusters are used (*right*) (*Data: Silvio Schüler*) (Color figure online)

corroborate the assumption that natural adaptation or range shifts may be too slow to keep up with the speed of climate change (Petit et al. 2008; Savolainen et al. 2007). A recent analysis of a large Austrian provenance test confirmed the impact of provenance selection for forest productivity in the future (Kapeller et al. 2012). The test series comprised tree height data of Norway spruce at the age of 15 years from 379 populations planted at 29 test sites across Austria. Connecting these data to climate information of the test sites and the provenance origin helps to calibrate climate response functions for groups of Norway spruce populations and to estimate the future productivity for a regionalized A1B scenario. Generally, the study did not reveal any declines in employed proxies for productivity throughout the current distribution range of Norway spruce in Austria. For most parts of Austria an increase of tree heights up to 45 % can be expected until 2080. However, the impact of a warming climate is different for individual population groups. Variation in climate response increases with higher temperatures and less precipitation. Thus, an optimized choice of seed material according to prospective future climate conditions has the potential for an additional increase of productivity up to 11 % (Fig. 6.11). The intra-specific variation in climate response can be explained by the local adaptation of provenances (Kapeller et al. 2012).

6.5 Conclusions

Due to the potential of forests for mitigating climate change the public attention on forest management has been rising considerably. Society expects that forests provide simultaneously numerous ecosystem services and the maintenance of high C stocks ranks high on the agenda. Forests store C in both the tree biomass and in the soils. Given that forests annually exchange about sevenfold more CO_2 with the atmosphere by photosynthesis and respiration than is emitted by burning of fossil fuels, the role of forests in the global C cycle is significant. However, forests are not a solution to climate change effects and do not interfere with the responsibility of reducing the anthropogenic emission of GHGs. A mere conservation of forest land can ensure the maintenance of a large C pool in terrestrial ecosystems. However, global population growth and increasing demands for food, feed, fuel and fibre exert increasing pressures on forest ecosystems. Particularly in developing countries and countries with economies in transition the pressure towards converting pristine forests to agricultural land is high. The establishment of a legal basis of forest protection and the enforcement of Forest Acts shows promising signals towards slowing the rate of the deforestation. In developed nations, the forest area is often increasing because marginal agricultural land is either actively afforested or encroached by shrubs and trees.

Forests themselves are vulnerable to climate change. Rapid warming and a change in the hydrological regime modifies site conditions. Forests may no longer be able to inhabit some areas because the site conditions are detoriating, but may be able to expand into lands that were so far not forested. For a successful migration of forests it is required that the changes are occurring rather slowly. An immediate effect of climate change is the migration of pests and pathogens. Their pressure on forests is already increasing. High uncertainty prevails on the relevance of changes in the frequency of extreme climatic events. The available data on increased frequencies of flooding, extreme precipitation events, storms and droughts are uncertain. However, when managing forest stand with rotation periods of several decades a cautious foresight needs to take into account climate dynamics that are likely, even when not yet statistically significant.

Forest practitioners can use many tools for "adaptive forest management". Simulation models parameterized with climate scenarios allow the visualisation of likely future forest conditions under chosen management scenarios, climate envelopes for tree species give indications on the expected shift of the potential habitat of trees. Silvicultural methods allow the creation of stable forests that are considered to withstand pressures efficiently. A successful concept is the establishment of mixed-species forests because the tree species have different abilities to tolerate stressors. The increased pressure from pests and pathogens both due to climate change and the global exchange of woody material calls for increased and transnational monitoring efforts in order not be caught by surprise by insect outbreaks. Another field is the choice of tree species. In a future climate the present distribution of tree species cannot be maintained. Remedies are the focus on other species or the selection of provenances that are expected to cope with future conditions.

Climate change is easily called upon as the reason for undesired dynamics of forest ecosystems. However, many forests have undergone recent changes that are unrelated to climate change. An example is the forestry in Central Europe. For several decades the harvest rates were lower than the annual increment. In consequence, forest grew older and denser and were, therefore, more vulnerable to storm damages. It is still disputed whether the frequency and severity of storms is increasing. In the last decades the response of forest practitioners to outbreaks of bark beetles were not sufficiently efficient and mass propagations caused substantial damages. An example for the Mediterranean region is the increase in forest fires.

Besides climate change, contributing factors are higher forest stand densities and a direct human influence. The conclusion is that the potential effects of forst management to decrease the vulnerability of forst stands has not yet been fully exploited and that adaptive forest management can play an important role for the forestry sector.

Setting aside forests for the sake of maximizing the C pool in the biomass and the forest soil is not seen as a successful concept of sustainable forest management. Unmanaged forests can only saturate the C pool once. Actively and sustainably managed forests are representing large C pools as well and can, in addition, provide wood products that replace other forms of construction materials, and can generate biomass as a renewable form of energy.

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Chapter 7
Ecosystem Carbon and Soil Biodiversity

Gerlinde B. De Deyn

Contents

Abstract Soils harbor a great diversity and abundance of soil biota, but because it is mostly invisible to our naked eye we are only just starting to discover it. This is not surprising given the estimated diversity of up to 10^6 species of bacteria per g of soil. The soil biota play key roles in ecosystem functions and hence in ecosystem services provided to humans, such as sustaining primary productivity through recycling plant nutrients and pest control. Soil biodiversity in turn depends on plants, directly through the influence by living plants and indirectly by after-life effects of dead plants, as primary producers provide energy and nutrients and create soil habitats. The soil food web (comprising bacteria, actinomycetes, fungi, soil fauna) itself represents only about 2 % of the total organic soil carbon yet it is central to soil carbon (C) sequestration through its impact on soil C turnover and stabilization by chemical and physical processes. However,

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how much or which soil biodiversity is needed to promote soil C sequestration remains an open question, yet key to develop management strategies for promoting soil C sequestration and biodiversity. Soils are of major importance in terms of ecosystem C given that terrestrial ecosystems contain more than 2,100 Pg organic C globally of which 1,500 Pg is located in soil. Moreover soils can serve as C source, i.e. C loss, or C sink, i.e. C sequestration in soil, and this balance is strongly dependent on soil management and soil biodiversity. Here we provide an overview of the current knowledge on the interdependency of soil biodiversity and ecosystem/soil C through interactions with primary producers. Overall most studies found that soil functional composition rather than species richness *per se* regulates soil C sequestration, through promoting ecosystem nutrient use efficiency, soil (micro)aggregate formation and soil depth development.

Keywords Soil biota • Ecosystem functioning • Ecosystem engineers • Key species • Functional groups • Soil microbes • Soil carbon sequestration • Soil structure • Stoichiometry • Nutrient use efficiency

Abbreviations

BVOC	Biogenic Volatile Organic Compound
C	Carbon
cm	Centimeter – 10^{-2} m
C: N	Carbon to Nitrogen ratio
CO_2	Carbon Dioxide
DOM	Dissolved Organic Matter
g	Gram
GSBI	Global Soil Biodiversity Initiative
GSP	Global Soil Partnership
mm	Millimeter – 10^{-3} m
N	Nitrogen
OM	Organic Matter
P	Phosphorus
Pg	Petagram – 10^{15} g
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
µm	Micro meter – 10^{-6} m

7.1 Soil Biodiversity

7.1.1 How Diverse, Where and Why?

The diversity of life on earth has been explored since centuries and to date the vast majority of plants and vertebrates have been described (Table 7.1). Plants and vertebrates, however, only represent a small part of all life on earth and the reason

Table 7.1 Global known (described) and expected biodiversity of major groups of organisms and their main role in soil carbon cycling

Body width	Organism group	Global number of described species	% known of expected	Role in soil carbon cycling	References
cm–m	Plants	270,000	84	Carbon input, soil structuring	Groombridge and Jenkins (2002)
	Vertebrates	52,500	95	Dung deposition	Groombridge and Jenkins (2002)
>2 mm, macrofauna	Termites	2,650	70	Soil structuring, decomposition, litter fragmentation	Groombridge and Jenkins (2002), Kambhampati and Eggleton (2000)
	Ants	14,700	50	Soil structuring, decomposition, litter fragmentation	http://www.antweb.org
	Earthworms	3,500	50	Soil structuring, decomposition, litter fragmentation	Lavelle et al. (1995)
	Beetles	350,000	30	Dung incorporation, decomposition, litter fragmentation	Gaston (1991), Davis and Scholtz (2001)
100 μm–2 mm, mesofauna	Mites	45,231	4	Decomposition, fungal channel	Walter and Procter (1999)
	Springtails	8,000	15	Decomposition, fungal channel	Hopkin (1998), http://www.collembola.org
	Potworms	691	10	Decomposition	Briones (2006)
<100 μm, microfauna	Nematodes	25,000	6	Decomposition, bacterial channel	Groombridge and Jenkins (2002), Boag and Yeates (1998)
	Protozoa	40,000	5–20	Decomposition, bacterial channel	Cox and Moore (2005), Finlay (2002)
<100 μm	Fungi	72,000	5	Decomposition, soil structure	Groombridge and Jenkins (2002), Fierer et al. (2007a)
	Bacteria	10,000	<1	Decomposition, soil structure	Groombridge and Jenkins (2002), Fierer et al. (2007a)
	Archaea	175	<1		Groombridge and Jenkins (2002), Fierer et al. (2007a)
	Viruses	5,000	5		Cox and Moore (2005), Fierer et al. (2007a)

Organisms are ranked from large to small body size and are mostly soil inhabiting. Empty cells indicate current lack of information why we know so much about these organisms is due to their large size and aboveground appearance. The smaller the organisms the less we know about their diversity as illustrated in Table 7.1. Most life on our globe is microscopic in size and can hardly be cultured and distinguished based on morphology. Only with the onset of molecular studies we started to become aware of how little of earths biodiversity we have discovered. During the last decades, molecular tools became widely available and completely changed our view on global biodiversity and evolutionary history by revealing the ancient origin of the tree domains the Bacteria, the Archaea, and the Eucarya (Woese et al. 1990).

Biodiversity in soil is more diverse than in sea water with 10^3–10^6 species of bacteria per g of soil (Torsvik et al. 2002; Gans et al. 2005), and very abundant with up to one billion cells of prokaryotes per g soil (Roesch et al. 2007). With respect to the Eucaryotes listed in Table 7.1 it has to be noted that the majority (if not all) of the fauna are typical terrestrial, living of dead organic matter (OM), fungi, bacteria or living plants. Only nematodes and protozoa also have a considerable number of species occurring in aquatic ecosystems. Fungi are also typically terrestrial organisms and, compared to bacteria, developed several ways to better exploit terrestrial habitats by being able to bridge air filled pores and to decompose recalcitrant OM (de Boer et al. 2005). It is well recognized that plant and vertebrate diversity peaks at equatorial latitude and decreases towards the poles, which can be explained by the distribution of energy and water availability (Hawkins et al. 2003). Biodiversity belowground does not seem to follow this pattern (De Deyn and van der Putten 2005; Fierer and Jackson 2006; Fierer et al. 2007a, 2009; Decaëns 2010). The reason for this discrepancy is attributed to the size of the soil organisms and the complexity of the soil environment in terms of physical and chemical composition in space and time, which affects potential organism densities, dispersal and reproduction rate and niche availability for food and shelter (Torsvik et al. 2002; Crawford et al. 2005; De Deyn and van der Putten 2005). Body size of soil biota span several orders of magnitude and range from <100 μm body width for the microbes (e.g., bacteria, fungi) and microfauna (e.g. protozoa, nematodes), to between 100 μm and 2 mm for the mesofauna (e.g., collembola, mites), to larger than 2 mm for the macrofauna (e.g., earthworms, myriapods, and insects) and up to body width larger than 2 cm for the megafauna (e.g., moles, mice) (Swift et al. 1979). The mode and level of active and passive dispersal capability is dependent on the body size of the organism, but also on whether it lives in the soils water or air phase.

Overall soil pH and soil carbon to nitrogen (C:N) ratio are of major importance for the structure and abundance of soil communities (Fierer et al. 2009; Wu et al. 2011). Biodiversity patterns, especially belowground, are scale dependent in space and time and so are the abiotic and biotic factors of influence (Ettema and Wardle 2002; Wardle 2006; Buée et al. 2009). As compared to aboveground biota, soil biota diversity is especially high at small spatial scale (Fierer and Lennon 2011). Soil biota distribution is, however, very heterogeneous and plant rhizospheres and dead OM patches are notable hotspots (Ettema and Wardle 2002; Buée et al. 2009; Bonkowski et al. 2009). At local scales, there is also a clear role for biotic interactions in space and time which lead to competition and facilitation (Wardle 2006).

Soils are typically three dimensional, even four dimensional given that soils are not static but dynamic in time, being slowly created and vulnerable to fast degradation. It is important to realise that soils consist of solid particles and pores filled with water and/or air. In this soil matrix soil biota live in and influence specific areas, depending on their body size and whether they are adapted to live in water, e.g., the nematodes, protozoa and bacteria, or in air filled spaces, e.g., the mites, collembolans and fungi. There is hence a clear degree of niche differentiation (Petersen and Luxton 1982).

7.1.2 Functions of Soil Biota

Soil biota play a key role in a multitude of ecosystem processes which provide indispensable ecosystem services to humans, such as sustaining primary productivity through recycling nutrients, pest control and water purification and as a potential source of new biochemicals for use in medicine and industry (Turbé et al. 2010). The best known and broad function governed by soil biota is the mineralisation of nutrients from dead OM, which makes nutrients available for plants to grow. This function requires a well-functioning soil food web where primary decomposer soil fungi and bacteria (1st trophic level) are being fed on by soil fauna (2nd trophic level) which are in turn being eaten by predatory soil fauna (3rd trophic level) (Petersen and Luxton 1982; Osler and Sommerkorn 2007; Holtkamp et al. 2011). Usually there are many species within each trophic level of the soil food web and many species consume many other species, i.e., the level of specificity is relatively low. However, there are two main energy channels, i.e., the bacterial and the fungal channel through which mineralization of nutrients and decomposition of OM occurs (Hunt et al. 1988; Holtkamp et al. 2011). The balance between these two energy channels has considerable impact on C storage in soil, an aspect that is discussed in more detail in Sect. 7.3.2.

To construct food webs soil biota are allocated into groups of organisms that share similar characteristics in terms of what they eat and by whom they are eaten, their efficiency in assimilation and production, their body C to nutrient ratio (i.e., their stoichiometry) and their death and feeding rates. Organisms that do not share these characteristics are hence functionally different from the perspective of energy and nutrient flow. The absolute contribution of the different functional groups is also very dependent on their abundances with in general the smaller the more abundant. The vast majority of soil biota is microbial representing about 90 % of the biomass (Fig. 7.1), but the soil fauna are essential in order to generate nutrient mineralization by freeing the nutrients that are locked up in the microbes (Osler and Sommerkorn 2007). The biomass distribution between fungi and bacteria can vary widely between different grasslands, forests and arable land in part due to different management; in contrast averages across studies indicate relative similar proportion of fungi and bacteria in grasslands, forests and arable land with somewhat lower fungal mass in arable and higher fungal mass in forest systems especially in the forests litter layer (Joergensen and Wichern 2008).

Mineralisation of nutrients by eating and being eaten clearly is a broad function in which all soil biota participate, so that it is a shared rather than a highly specialised

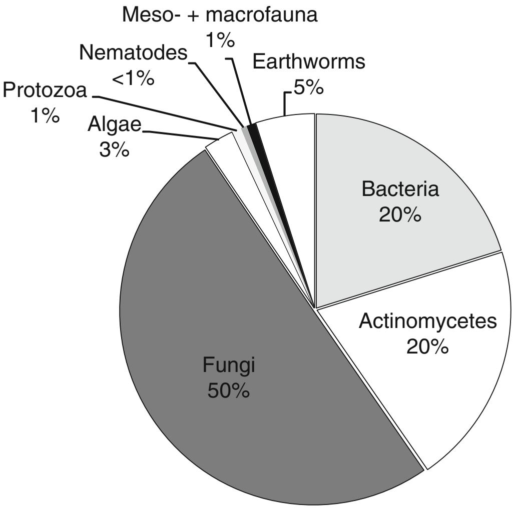

Fig. 7.1 Relative distribution of soil biotic carbon (excluding plants) per group of soil biota in a grassland soil (data based on Killham 1994 and personal observations)

function (Hunt and Wall 2002; Falkowski et al. 2008). It has to be noted that some metabolic steps in nutrient mineralisation can only be performed by a small number of soil biota species and without these the whole mineralisation process would stagnate, hence for these species there is a low level of redundancy (Falkowski et al. 2008). Other functions of soil biota that are governed by a limited number of species are the suppression of specific soil pathogens and the production of specific biochemicals (Weller et al. 2002; Garbeva et al. 2004). A subset of soil biota species can also alter plant hormone production or the provision of plant growth limiting nutrients through symbiotic association, which can all result in plant growth promotion. Loss of these soil species hence results in significant loss in functioning (Garbeva et al. 2004; van der Heijden et al. 2008). Also, amongst the soil invertebrates there are clear differences in contributions to ecosystem functions other than nutrient mineralisation. For example, there is a differential impact on pedogenesis and the larger soil mixing invertebrates play a particularly large role (Lavelle et al. 2006).

In order to sustain multi-functionality, diversity is required so that also for soil biota more diversity is needed when considering the set of functions they provide as opposed to when focussing on just one function in isolation (Gamfeldt et al. 2008; Mace et al. 2012). It is also important to note that diversity is not just about species presence but also about the relative abundances of the species that make up the soil communities. For example, Wittebolle et al. (2009) demonstrated that the evenness in abundances across species rather than species richness was of major importance for sustaining denitrification by microbes under salt stress.

The use of molecular techniques has revealed the vastness of microbial life that remained undiscovered due to the limitations of cultivation techniques (Schleper et al. 2005; Fierer and Lennon 2011). The great challenge now is to reveal how this genetic diversity relates to known and not yet discovered ecosystem functions (Schleper et al. 2005; Fierer et al. 2007b).

7.2 Ecosystem Carbon in Terrestrial Ecosystems

7.2.1 Primary Producers

Terrestrial ecosystems contain large amounts of organic C accounting globally for more than 2,100 Pg C (1 Pg = 10^{15} g, Schulze 2006), and when also inorganic C is included the amount of C in terrestrial systems at global scale exceeds 4,000 Pg C (Batjes 1996; Eswaran et al. 2000). In this chapter we focus on organic C as especially on this pool biodiversity and antropogenic factors can have a strong influence. Depending on the biome, a larger or smaller fraction of the total organic ecosystem C is stored in vegetation (Amundson 2001; De Deyn et al. 2008). Biomes are composed of co-occurring plant species that are well adapted to the temperature and precipitation regime of the geographic location where they prevail, and thereby indirectly to soil nutrient availability (Woodward et al. 2004; De Deyn et al. 2008). As forests (especially those in humid areas on well drained soil) may have very high aboveground standing biomass in the form of trees which are predominantly made up of cellulose and lignin (i.e., compounds with very high C concentration) it is clear that forest vegetation may be a large store of ecosystem C. Also in terms of C inputs to soil on annual basis for a given area forest rank high provided they are not limited by water availability (Amundson 2001). From an aboveground perspective grasslands do not appear to be large C stores, however, in grassland systems belowground plant biomass is much higher than aboveground (Jackson et al. 1996). Moreover, C input rates to grassland soils are considerable and comparable to those in tropical dry forest (Amundson 2001).

Primary producers (i.e., photosynthetic C uptake) are the entry point of C to soil, and depending on the vegetation type and climatic conditions this results in a certain amount of C input to soil during the growing season. When C inputs to soil exceeds C losses from soil then soil C will accumulate until a an equilibrium between in- and outputs is reached (Fig. 7.2), and this has been accruing across biomes over many years so that more than two-thirds of all terrestrial C is located in soils (Jobbagy and Jackson 2000; Amundson 2001). One has to realise that this soil C is vulnerable to losses with important implications for soil biodiversity, productivity and climate feedbacks (Davidson and Janssens 2006; Bardgett et al. 2008; Ostle et al. 2009).

7.2.2 Soil Organic Carbon

Soil organic C pools are highly variable in time and space as they result from the balance between C input via primary productivity and losses via decomposition of OM, root respiration, leaching of dissolved soil C and soil erosion (Amundson 2001; De Deyn et al. 2008) (Fig. 7.2). The soil C is found in a wide range of forms,

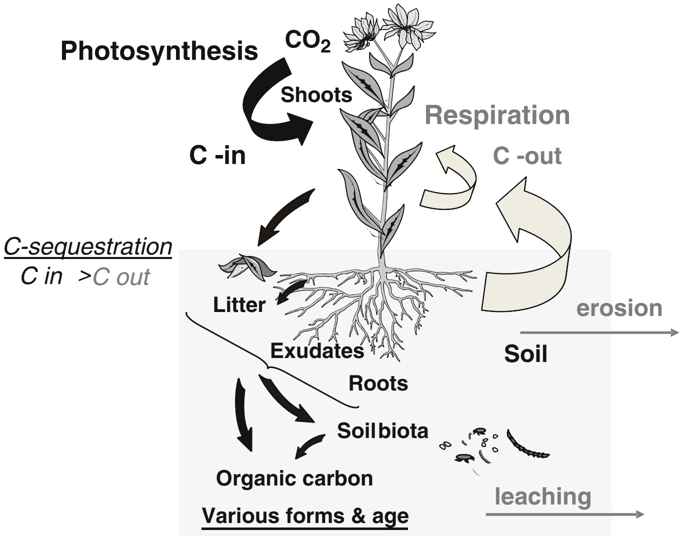

Fig. 7.2 Plant C input and loss pathways in soil. As C input only photosynthesis is considered, C output comprises respiration, erosion and leaching of C (in the figure), as well as harvest, herbivory, emissions of volatile organic compounds and loss by fire (not displayed in the figure)

most of which are organic. These organic forms are more or less easy to decompose by soil biota because they are more or less accessible physically, because of their location in soil, and/or chemically, because of the enzymes and nutrients needed to breakdown the compounds (Schmidt et al. 2011). These differences between soil C forms are important because of their differential responses to biotic and abiotic factors and their role in supporting soil food webs, plant growth and soil structure (Kuzyakov and Gavrichkova 2010).

There are two well established models to simulate soil C dynamics (CENTURY and ROTH-C), and these models distinguish three arbitrary C pools ranging from a very active pool with fast turnover rate to a virtually inactive (so called "passive" or "inert") pool with very slow turnover rate which represents a relative stable soil C pool. However, the large inert C pool may become active in response to altered environmental conditions (King et al. 1997; Davidson and Janssens 2006). The pools used in modelling studies are conceptual and not directly measurable, but C pools of different mean soil residence time can be isolated and measured based on chemical and/or physical fractionation (Davidson and Janssens 2006). The soil biota are often considered as part of the active soil C pool. In absolute terms soil biota (not taking plant roots into account) only represent about 1–2 % of the total soil organic C (SOC) pool, with 96 % of this in soil microbes (Tate 1987; Fierer et al. 2009) (Fig. 7.1). Nevertheless the soil biota are crucial in SOC dynamics.

7.2.3 Soil Organic Carbon Dynamics

Across biomes the concentrations of SOC in the top 1 m of soil are comparable in tundra, boreal and wet tropical forests (19 kg C m^{-2}), which may be surprising given their large differences in primary productivity and species composition (De Deyn et al. 2008). In deserts and savannas SOC pools are much lower (<10 kg C m^{-2}), and in temperate grasslands and forest SOC pools are intermediate (about 13 kg C m^{-2}) (King et al. 1997; Amundson 2001; De Deyn et al. 2008). The somewhat counterintuitive comparable SOC stocks in tundra and wet tropical forest is likely due to low input and even lower C output in tundra, and high input and high output rate of C from soil in tropical forests (King et al. 1997; Amundson 2001; De Deyn et al. 2008). The chemical and physical forms of SOC and their location in the soil profile are different between those systems, and so are the diversity, abundances and activity of the soil biota (Amundson 2001; Von Lützow et al. 2006; De Deyn et al. 2008). With respect to soil fauna, nematodes mostly dominate in grasslands in terms of diversity and abundance, and micro-arthropods in forests with low mean annual temperature, low soil pH, inorganic N and root biomass and N-poor litter (Wu et al. 2011). Also, for soil bacteria and fungi litter N concentration and moisture are of great importance with fungi being better capable of decomposing litter with low N concentration and operating under lower soil moisture than bacteria. However, for all soil biota their activity is strongly related to soil temperature with, for example, very low activity in the cold tundra and very high activity in tropical humid forest.

Also, chemically relative labile C can be protected from decomposition and hence can contribute to soil C sequestration, specifically by physical protection through its incorporation in soil aggregates. Soil aggregates are formed by mineral and organic soil particles that are being bound together by the processes of rearrangement of particles, flocculation and cementation (Six et al. 2000, 2004; Bronick and Lal 2005). These processes are mediated by SOC, soil biota, clay, carbonates and ionic bridging. It has been recognised for over a century that the interaction between soil biotic activity (i.e., by soil fauna, soil microorganisms and roots) and soil organic matter (SOM) are central to soil aggregate dynamics (Tisdall and Oades 1982; Six et al. 2004). The precise mechanism by which soil organic C is being stabilised is still under debate but the current view is that C is stabilised in microaggregates that are formed within macroaggregates that turn over at specific rates (Six et al. 2004; Von Lützow et al. 2006; Schmidt et al. 2011) (Fig. 7.3). If the turn over rate of macroaggregates is too high, for example due to tillage, micro aggregate formation is impeded and hence soil C stabilisation is impaired.

Amongst the soil biota ecosystem engineers are of particular importance for soil aggregate formation and C stabilisation in these structures. Ecosystem engineers are a group of soil biota that alter their environment to a great extent (Jones et al. 1994; Brown et al. 2000). In humid areas earthworms are the most common ecosystem engineers, while in more dry soils ants and termites proliferate

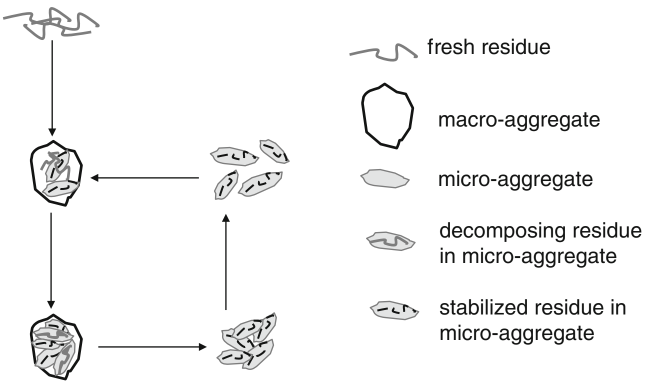

Fig. 7.3 Stabilisation of soil C in microaggregates formed in macroaggregates through the action of soil biota (After JW van Groenigen, based on Six et al. 2004)

(Jouquet et al. 2011; Brussaard et al. 2012). These soil biota burry organic material in soil and mix it with mineral soil particles and organic compounds produced by the soil biota (e.g., mucus, saliva), a process that greatly promotes aggregate formation. Also, soil fungi and roots can be viewed as ecosystem engineers as by their (network)structure and exudation of organic "sticky" compounds they enmesh and glue soil particles together (Tisdall and Oades 1982; Bronick and Lal 2005).

7.3 Soil Biodiversity and Ecosystem Carbon Relations

7.3.1 Species Richness and Functional Composition

Soil biota comprise a wide range of species of various sizes, forms and functions. Their impact on ecosystem C is several fold. First, soil biota are the prime agents of decomposition of soil C and hence of C loss by respiration. Secondly, soil biota can promote plant growth and hence C input by enabling plants to access nutrients and/or by protecting plants from adverse abiotic (e.g., drought stress) and biotic (e.g., pathogens) conditions. Thirdly, soil biota can help with stabilising soil C by promoting aggregate formation. These effects of soil biota are discussed in more detail in the context of soil biodiversity below. The relation between diversity and ecosystem processes can have different forms. The relation may be linear positive when species are singular and each species has a unique way of contributing to the ecosystem process, asymptotic when species are contributing in similar ways and are substitutable (i.e., there is redundancy), and idiosyncratic when the contribution of the species is unpredictable because it is dependent on the abiotic and/or biotic context (Loreau et al. 2002; Fig. 7.4).

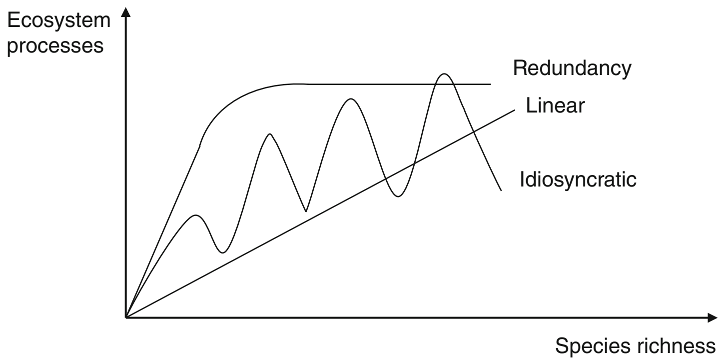

Fig. 7.4 Relation between species richness (or diversity) and ecosystem functioning, based on Loreau et al. (2002)

7.3.1.1 Decomposition

Given that microscopic soil biota are diverse, able to adapt quickly and are ubiquitous it is logical to assume high functional redundancy in soil microbial communities (Wertz et al. 2006). It has been shown that soils with only 1 % of the microbial diversity can still have similar C decomposition and N mineralization rates as the original soil with the intact diverse community, i.e. with 100 % of the microbial diversity present (Wertz et al. 2006). Also, several studies support the idea of redundancy in soil biodiversity as they found that that across saprophytic bacteria and fungi decomposition rates increased linearly with species richness at very low levels of diversity but the rates soon reached a plateau with further increase in species richness (Setälä and McLean 2004; Bell et al. 2005). In contrast, other studies have found idiosyncratic relations between microbial diversity and process rates and even negative relations, as in the case of potential nitrification rate (Griffiths et al. 2001), indicating that not species richness but the characteristics of the component species are of prime importance.

In a recent review Nielsen et al. (2011) compiled a wide range of studies that investigated the relation between soil biodiversity and C cycling. The 85 experiments they found could be grouped in experiments with less and in experiments with more than ten species. Across all the soil biota groups under study (bacteria, fungi, soil micro-, meso- or macro-fauna), linear positive relations between species richness and C mineralisation rates were found in studies with a small species pool (richness range between one and ten) while no or negative relationships were found when the species richness range started from more than ten.

There are also studies which indicate that decomposition rates are sensitive to the community composition rather than to the diversity of the microbial communities (Strickland et al. 2009). These authors found that soil microbes decomposed the litter from their natural habitat ("home litter") at faster rates than litter from other origins, despite the higher recalcitrance, i.e. higher C:N ratio and lignin content, of the home litter in terms of its chemical composition. Other support for the importance of community composition rather than species richness for decomposition rates comes from studies that investigated the impact of the assembly history of decomposer fungi on decomposition rates (Fukami et al. 2010; Dickie et al. 2012). These studies showed that even when the total fungal species pool is the same, the sequence of species arrival (i.e., which species starts to colonise the substrate first) does have significant impacts on the subsequent community composition and decomposition rates due to non-additive interactions between species and non-redundant functional characteristics of the species. Moreover, when also fungal grazers are included it becomes clear that the outcome of interactions between species of fungi and their impact on decomposition rates are dependent on the identity of the fungal grazers as well (Crowther et al. 2011). Clearly, community composition across trophic levels is an important ecosystem feature such that diversity-decomposition relations are often unpredictable (idiosyncratic) from species richness data alone.

High rates of decomposition indicate high rates of C loss via respiration, biogenic volatile organic compound (BVOC) formation, and potentially also via horizontal run-off and leaching of DOC, processes that reduce rather than promote soil C sequestration on the soil surface or in topsoil. However, if DOC is leached from the soil surface and infiltrates into the soil to be adsorbed on mineral surfaces at deeper soil depths, C can be stabilized for long periods of time. At the same time decomposition also results in nutrient mineralisation which supports plant growth and hence the assimilation of atmospheric C in new biomass, which can offset the losses of C incurred during decomposition. The amount of nutrient mineralization depends on the nutrient content of the substrate and on the structure of the soil food web, i.e., the abundances, efficiencies and feeding rates of the soil biota (Osler and Sommerkorn 2007; Holtkamp et al. 2011).

7.3.1.2 Productivity

Soil biota not only support plant growth indirectly via the mineralization of nutrients contained in SOM but also directly through symbiotic relations. Many plant species are hosts for root inhabiting organisms and these organisms can promote or suppress plant growth. Mycorrhizal fungi and N-fixing bacteria are root symbiotic organisms that generally promote plant growth by providing limiting nutrients to plants, especially phosphorus (P) in the case of arbuscular mycorrhizal fungi and N in the case of the N-fixing bacteria (van der Heijden et al. 2008). The ability to associate with N-fixing bacteria is mostly confined to leguminous plant species, and as the association is specific between host and bacteria species (or even genotype) the benefit for plant growth depends on the match between both partners and not on their diversity in the soil (Sprent 2001). On the other hand one can argue that the more diverse the community of N-fixers in soil, the higher the chance a good match between genotypes will be formed. In practice, in agricultural systems a more targeted approach is often sought and there are incentives to find the best bacterial strains and inoculate them with the host crop (e.g., Panzieri et al. 2000). As with any introduction of organisms into new areas one should clearly consider the potential non-targeted impacts on the ecosystem.

Mycorrhizal fungi associate with a much wider range of plant species than symbiotic N-fixing bacteria. Despite the wide occurrence and relative low species richness of mycorrhizal fungi the level of redundancy between mycorrhizal species is low. Support for the low redundancy between mycorrhizal species comes from studies demonstrating multi-functionality and complementarity of mycorrhizal species and high specificity of mycorrhizal host plants in their response to fungal species (Leake et al. 2005). The multi-functionality of mycorrhizal fungi operates through nutritional effects (i.e., facilitation of nutrients uptake of several elements), and non-nutritional effects, e.g., plant host protection against toxins in soil, drought stress and root herbivores and pathogens.

Free living soil microbiota can act as suppressors of pests and diseases. There are several mechanisms by which pathogen suppression can be established from little to highly specific namely through general competition for space and resources by saprophytic biota, through the production of suppressive metabolites and through active attack of the pathogens (Garbeva et al. 2004; Raaijmakers et al. 2009). To achieve general suppression microbial biomass is likely of major importance and diversity can help in occupying many niches, while for specific suppression effectiveness is directly related to the identity of pest and suppressor.

7.3.1.3 Soil Organic Carbon Stabilisation

The impact of soil fauna on soil physical properties is of major importance for the fate of soil C in terms of its location and residence time in soil. During decomposition of SOM the soil biota release part of the C from SOM back into the atmosphere through respiration, while another part gets mixed with soil mineral particles and becomes incorporated in soil aggregates. As mentioned earlier the formation of microaggregates in macroaggregates is key for longer-term soil C storage throughout the soil profile (but mostly happening where soil organisms are active) in agricultural as well as in natural soils (Six et al. 2004; Von Lützow et al. 2006; Schmidt et al. 2011). Especially ecosystem engineers aid in aggregate formation but one can wonder whether they are a key functional group with high levels of redundancy in the group or whether there is also complementarity between species.

The species richness of earthworms is rather limited at local and even at global scale (Table 7.1), nevertheless species have markedly different life history traits and impact on soil structure and soil C (Brown et al. 2000). It is therefore useful to distinguish between earthworm functional groups, namely epigeic, anecic and endogeic and soil compacting and decompacting species. The earthworm functional group identity and its abundance rather than species richness of earthworms *per se* have been shown to be the important parameters to explain leaching of DOC (and N) through macropores (Blanchart et al. 1999; Brown et al. 2000). Moreover the impact of the earthworm functional group on soil structure and soil C is dependent on the soil type. This interactive effect between earthworm species and soil type prohibits simple predictions so that knowledge of the system is warranted for developing earthworm mediated soil management strategies (Blanchart et al. 1999; Pulleman et al. 2005).

In arid and semi-arid systems the key ecosystem engineers are termites, representing up to 65 % of all soil macrofaunal biomass in some biotopes (Jouquet et al. 2011; Brussaard et al. 2012). The global species richness of termites is comparable to that of earthworms. Also, across termite species several ecological groups can be distinguished with respect to how they alter the distribution of ecosystem C. Most species are litter- or soil-feeding, and a smaller group of species are fungus-growing termites. The latter species collect OM to feed their fungal gardens which serve as food for the termites. The soil feeders are most abundant in semi-natural humid forests, while litter feeders usually dominate in other systems. Generally termites increase SOM concentrations and levels of mineral N in the soils or in constructs that they create (Jouquet et al. 2011). Compared to the influence of earthworms the impact of termites on SOM and soil C is longer-term and increasing habitat heterogeneity thereby contributing to landscape mosaics and patchiness of soil properties (Jouquet et al. 2006).

Ants are the most species rich group amongst the soil ecosystem engineers, have a global distribution, and contribute to a wide range of ecosystem functions (Brussaard et al. 2012). The impact of ant species on soil turnover and soil C is strongly determined by the ant colony size and activity, with some species-specific differences. In their review article, Wilkinson et al. (2009) show that the rates of (re)constructing soil by burial and mixing of organic and mineral particles is comparable between ants and termites, while earthworms appear to have highest activities. It has to be noted that soil bioturbation generates C losses as well as C gains, and that net effect (being positive, negative or neutral) results from these co-occurring processes. The impact of ecosystem engineers on soil C is for a substantial part also indirect, as the enhanced soil structure and soil depth resulting from activities of ecosystem engineers in their respective habitats promotes primary production (Lavelle et al. 2006).

7.3.2 Soil Food Web Characteristics

Soil food web models are very instrumental in simulating the rates of nutrient mineralization and soil C decomposition given the abundances of different functional (feeding) groups of microbes and fauna in the soil. Saprophytic bacteria and fungi are the two main entry points through which dead OM is decomposed and its nutrients mineralized (Hunt et al. 1988; Holtkamp et al. 2011). On average bacteria have lower C:N ratios in their body mass and lower C use efficiency than fungi so that substrate decomposition by bacteria generates more metabolic C loss and incurs more N immobilisation in microbial biomass than decomposition by fungi (Hodge et al. 2000). On the other hand, soil fauna feeding on bacteria mineralize more N than those feeding on fungi for the same amount of microbial biomass consumed

(Osler and Sommerkorn 2007). If plants can rapidly take up the mineralized nutrients as is the case in the rhizosphere where rhizobacteria are being consumed by protozoa, positive feedback loops to plant growth occur and these may not only involve nutrient but also hormonal effects (Bonkowski 2004). At larger scale it has been argued for natural and agro-ecosystems that the increase in abundance of fungal relative to bacterial biomass is beneficial for soil C sequestration (Six et al. 2006; De Deyn et al. 2008) and soil nutrient retention (de Vries et al. 2012). This benefit can be explained by their efficiency in C and nutrient use, the recalcitrance of fungal tissue and metabolites, and their beneficial impact on soil structure and aggregate formation, especially in the case of mycorrhizal fungi (Rillig and Mummey 2006; Wilson et al. 2009). As bacteria and fungi are species rich groups that can operate at different scales and locations in space and time there is a wide scope for functional complementarity between them as demonstrated by their differential roles in plant rhizospheres (Buée et al. 2009; Jones et al. 2009) or in bulk soil across seasons (Bardgett et al. 2005). Moreover the chemical properties of SOM and the metabolic capacities of the soil biota are diverse so that simple models cannot capture SOM dynamics, as they result from ecosystem properties including the soils biotic and physical characteristics, rather then solely from SOM composition (Six et al. 2006; Schmidt et al. 2011).

7.3.3 Soil Management

Soils function as a habitat for a wide array of soil biota as soils provide shelter and food to their biota. Soil organic matter serves as the basis of all soil life so when inputs of OM to soil are reduced soil biomass and soil biodiversity will decline as a consequence (Moore et al. 2004; Jeffery et al. 2010). It is well recognised that soil life and its diversity is sensitive to a wide range of factors amongst which soil over-exploitation, decline in soil organic matter, soil pollution, compaction, erosion and desertification rank amongst the highest treats (Jeffery et al. 2010). The decline in abundance and activity of soil biota has cascading effects on soil fertility because the processes of decomposition, mineralization and mixing of OM with mineral soil particles provide structure to soil and enables the retention of water and nutrients (Lavelle et al. 1997). Soil biota are not only sensitive to the input of organic substrates but also to physical and chemical disturbance, e.g., soil tillage and application of fertilizers and pesticides, responses that are notable at local and landscape scale (Culman et al. 2010). Tillage impacts are especially strong on the larger soil biota and soil biota that form networks such as (mycorrhizal) fungal mycelial networks (Lavelle et al. 1997; Galvez et al. 2001; Schnoor et al. 2011). As many of the biota that are sensitive to disturbance are important as ecosystem engineers, the self (re) structuring capability of soils declines with tillage. Tillage has also a particular strong impact on the C balance of soils as it disrupts aggregates, thereby exposing formerly physically protected SOC to decomposition by microbes while the disturbance suppresses the activity of soil biota that are instrumental in soil aggregate formation.

Global decreases in natural SOC due to cultivation by humans is evident and has been estimated to cause a loss of 60 (temperate regions) to 75 % (tropical regions) of the original SOC (Lal 2004). Especially the conversion of native forest, or pasture to crops generate large losses of SOC (Guo and Gifford 2002). In order to reduce further losses and restore SOC pools appropriate soil management is a prerequisite. In fact one should rather speak of ecosystem rather than soil management given the inherent tight coupling between C and nutrient inputs and cycling between plants and soil biota. A key management tool is the reduction in soil disturbance through the adoption of minimum or even no-till (Lal 2004; Holland 2004). Soil management using organic amendments and its incorporation by earthworms can also promote microaggregate formation and C and N retention in agricultural soils (Pulleman et al. 2005). These authors found that the impacts varied with land use and further studies need to elucidate which management is most beneficial in order to capitalise on the earthworms positive effects on soil structure and associated SOM dynamics. Management should be targeted not only towards abundances but also to accommodate the diversity of soil invertebrates as structuring activities of single species, especially when present in very high densities, can result in detrimental effects (Lavelle et al. 2001). It has to be noted that soil biodiversity restoration is a slow process with soil biota recovery often lagging behind changes observed aboveground (Kardol and Wardle 2010). Given that above- and belowground diversity and processes are interdependent it is being explored to which extend soil inoculations can speed up ecosystem recovery (Kardol and Wardle 2010; Carbajo et al. 2011). The maintenance of SOC, soil biodiversity and resistance and resilience to desertification are clearly interdependent (often in non-linear ways) and process-level understanding is required to design management strategies that optimise soil biotic and abiotic composition and functioning on the short and longer-term (Cowie et al. 2011).

The choice and use of vegetation is another important tool in direct and indirect management of soil C. The use of perennial rather than annual crops is beneficial for SOC, soil biota and productivity as they promote soil structure, reduce leaching of nutrients and are a ‘continuous’ food source for the soil food web (Glover et al. 2010). A recent study in restoration grassland also demonstrate a key role of legume species rather than application of mineral fertilisers to be beneficial for biodiversity and soil C sequestration via biotic and abiotic effects (De Deyn et al. 2011). Plant biodiversity studies in grasslands have also shown beneficial roles of combinations of species or plant functional groups for soil C and nutrient sequestration (Fornara and Tilman 2008; De Deyn et al. 2009). The benefits may be explained by complementing nutrient use, increased root production and decreased root loss to pathogens and herbivores. Also, in agricultural systems with limited plant diversity benefits for soil C and nutrient retention and biodiversity can be achieved. A major research challenge that remains is to elucidate which crop characteristics between intercropped species (or genotypes) are most beneficial for optimal yields and interaction with soil biodiversity for SOM accumulation under the given climate, abiotic soil characteristics and with minimal soil disturbance (Bronick and Lal 2005; Brussaard 2012). Changes in conventional agricultural practices aimed at promoting multiple ecosystem services may result in reduced aboveground biomass yield and income of farmers. It is therefore needed that political and economic incentives compensate farmers for the provisioning of common services such as soil C sequestration and biodiversity conservation (Lal et al. 2007; Brussaard et al. 2007, 2010).

7.4 Conclusions and Future Directions

Over the last decades scientists, policy makers and the public became aware of the vast biotic diversity soils contain, their role for supporting humanity and the potential of yet undiscovered functions, while at the same time soils are being globally degraded at fast rates. To halt and combat this loss of earths major resources global initiatives such as the Global Soil Partnership GSP (http://www.fao.org/nr/water/landandwater_gsp.html) and the Global Soil Biodiversity Initiative GSBI (http://www.globalsoilbiodiversity.org/) were launched in 2011. These initiatives are aimed at enabling a better use of knowledge on soil biodiversity and ecosystem services, promote global awareness of soil biodiversity and its values. Given the large diversity that is expected to be hidden in soils worldwide and the fast rate of global changes that are occuring and whichimpact on soil diversity and functioning interdisciplinary projects with integration from local to global scale are needed.

Relationships between soil biodiversity and C cycling processes are dependent on the soil biota functional composition and abundance rather than on species richness *per se*, as complementarity in their characteristics promotes ecosystem nutrient use efficiency, soil (micro)aggregate formation and soil depth development. In order to achieve ecosystem nutrient and water use efficiency it is warranted that plant resource demand and its provision by the soil biota are in tune which could be facilitated by larger use of mutualist symbiotic root biota (De Deyn et al. 2008). Striking examples of the importance of matches and mismatches in the plant-soil system come from invasion studies where new organisms, which initially increase local diversity as an extra species, disrupt the interactions of co-evolved species in natural ecosystems. These disruptions can take such proportions, for example as in the case of invasive earthworm species, that decomposition and nutrient mineralization occur at much faster rates than the native vegetation can handle, leading to leaky ecosystems and species loss, and eventually ecosystem collapse (Peltzer et al. 2009; Wardle et al. 2011). Such examples show that great care is needed when introducing organisms with new traits. At the same time it is important to realise that naturally more diverse systems are generally more resistant to invasions and resistant and/or resilient to perturbations, thereby providing a portfolio effect (Brussaard et al. 2007; Mace et al. 2012).

The main direct ways through which soils can be managed for biodiversity, productivity and SOC are through vegetation and reduced soil disturbance. Reduced soil disturbance enables the ecosystem engineers to perform their soil structuring capacities, while the vegetation characteristics (and the return of OM at harvest) directly affect the amount, location, diversity and nutritive value to the soil food web. Soil management for sustainable provisioning of multiple ecosystem services warrants an ecosystem approach, such that promotion of one service is not outweighed by deterioration of other services (Smith et al. 2008; Mace et al. 2012). Optimal strategies to promote soil biodiversity and the services they provide will require location specific measures, a challenging task but clearly the potential is there.

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Chapter 8
Ecosystem Services and the Global Carbon Cycle

M.R. Raupach

Contents

Abstract Ecosystem services associated with the global carbon (C) cycle must be assessed in the context of an Earth System that has been greatly modified by human activities, the ensuing great challenges of sustainability and wellbeing, and the generic role of ecosystem services in meeting these challenges. Given this broad context, a brief survey of the carbon-climate system highlights three features. First, total anthropogenic carbon dioxide (CO_2) emissions (the sum of emissions from fossil fuel combustion, other industrial processes and net land use change) have grown nearly (but not exactly) exponentially over the two centuries since the onset of industrialisation, at an average growth rate of 1.9 % year^{-1} over the period 1850–2010, to reach 10 Pg C year^{-1} in 2010. Second, and consequently, atmospheric CO_2

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concentrations have risen steeply since 1800, from a nearly steady level of about 278 to 389 ppm in 2010, increasing at 2 ppm year^{-1} (2001–2010 average). Third, less than half of total anthropogenic CO_2 emissions remain in the atmosphere, the rest being removed by land and ocean CO_2 sinks. The fraction remaining in the atmosphere, the CO_2 airborne fraction has been close to (but not exactly) constant over the period 1959–2010, at around 45 %, indicating that the global C cycle has a high self-regulating capacity. Given these features, the global C cycle supports two groups of ecosystem services: (1) services that protect the Earth System against carbon-climate vulnerabilities or reinforcing feedbacks that would accelerate climate change; and (2) services provided by C sequestration in the terrestrial biosphere, to contribute to mitigating climate change. All ecosystem services associated with the global C cycle have implications for safeguarding the "carbon cycle commons", the globally shared, self-regulating functions of the C cycle. The services provided by the carbon cycle commons to humanity are only partly in the form of direct benefits. They are also in the form of protection against vulnerabilities, or risks of potential changes in Earth System functioning that would be harmful to human wellbeing.

Keywords Carbon cycle • Carbon fluxes • Carbon stocks • Soil organic carbon • Ecosystem services • Land use change • Carbon sequestration • Sustainability • Vulnerabilities

Abbreviations

AF	Airborne fraction (of CO_2)
BC	Black carbon
CDR	Carbon dioxide removal
ENSO	El Niño-Southern Oscillation
FFI	Fossil fuel and other industrial processes
GDP	Gross domestic product
GHG	Greenhouse gas
LUC	Land use change
MEA	Millennium Ecosystem Assessment
RCP	Representative concentration pathway
SF	Sink fraction (of CO_2)
SIC	Soil inorganic carbon
SOC	Soil organic carbon

8.1 Introduction

We humans are a carbon-based life form, utterly dependent for sustenance on a carbon-based biosphere. Humankind is making progressively increasing demands on its biospheric life support systems, not only because of increasing human numbers but also because of increasing affluence. The result has been a transformation of the

Earth by human activities, particularly since the industrial age began around 1800, leading to stress on many aspects of the Earth System (Steffen et al. 2004; Millennium Ecosystem Assessment 2005).

There is a fundamental connection between the carbon (C) cycle and the transformation of the Earth by humankind. The seeds of the transformation were sown half a million years ago when our early ancestors in genus *Homo* began to exploit detrital C through fire as a source of extrasomatic energy, to gain enormous evolutionary advantages. These advantages grew as the use of detrital C increased, culminating in the use of fossil fuels for energy (Boyden 2004; Raupach and Canadell 2010). The interaction between humankind and the C cycle has now become a Faustian bargain, because the human modification of the C cycle in the industrial age has profound implications for the entire human-environment relationship.

The concept of ecosystem services (Millennium Ecosystem Assessment 2005, henceforth MEA) provides a framework for identifying and quantifying the dependencies of humans upon their environment, and the pressure points at human-environment interfaces. The MEA defined ecosystem services as "the benefits provided by ecosystems for human wellbeing", and identified 27 ecosystem services grouped into four categories: provisioning, regulating, cultural and recreational. Given the centrality of the C cycle in human-environment interactions, it is not surprising that the C cycle is involved directly in many of the ecosystem services identified by the MEA, and indirectly in all.

This chapter, a survey of ecosystem services associated with the global C cycle, is structured as follows. After this brief introduction, Sect. 8.2 reviews some underpinning concepts, including human impacts on the Earth System, the challenges of sustainability and wellbeing, and the generic role of ecosystem services in meeting these challenges. Section 8.3 surveys the carbon-climate system, including anthropogenic carbon dioxide (CO_2) emissions, the responses of the C cycle to its current disequilibrium, and self-regulation of the C cycle. Section 8.4 assesses two groups of ecosystem services associated with the global C cycle: (1) services that protect the Earth System against carbon-climate vulnerabilities or reinforcing feedbacks that would accelerate climate change, and (2) services provided by C sequestration in the terrestrial biosphere (the process of increasing the C content in biomass and/or soil C stocks by human management), to contribute to mitigating climate change. The chapter concludes with an assessment of the implications for safeguarding the C cycle commons.

8.2 Concepts

8.2.1 Human Impacts on the Earth System

Our planet functions as a single Earth System, an evolving complex system comprising the land, waters, air and ecosystems of the planet, together with human societies, cultures, knowledge and economies. All of these components are tightly connected, continually influencing and being influenced by each other.

The Earth System is now being transformed by human activities (Steffen et al. 2004; Millennium Ecosystem Assessment 2005; Rockström et al. 2009). Major direct impacts, some deliberate and some as a byproduct of other actions, have occurred in land cover, fire and disturbance regimes, the structure and function of terrestrial and marine ecosystems, the hydrological cycle, and atmospheric composition. The last through increases in concentrations of CO_2, methane (CH_4) and other greenhouse gases (GHGs), increases in levels of particulates such as sulphate aerosols and black carbon (BC), and the introduction of reactive anthropogenic substances such as chlorofluorocarbons. These direct influences have consequences throughout the Earth System, affecting global and regional climates, the great natural cycles of C, water, nitrogen (N), phosphorus (P) and other entities, atmospheric chemistry (e.g., stratospheric ozone depletion), water chemistry (e.g., the acidifying trend in ocean pH), and terrestrial and marine biodiversity. Collectively, human-induced transformations are modifying the functioning of the Earth at a planetary scale, to the extent that the epoch since the start of the industrial revolution around 1800 is often called the "Anthropocene" (Crutzen 2002).

The underlying drivers of Anthropocene transformations are increasing human population and affluence. Figure 8.1 illustrates the strong global economic growth since the start of the industrial revolution around 1800, accompanied by rapid growth in population. After 1800, the trajectory of world-average per-capita gross domestic product (GDP) changed within a decade or two from near stasis to exponential growth with a doubling time of 45 years, a pattern that continues unabated. This growth has led to huge but very unequally distributed improvements in affluence, health and material wellbeing, first in developed regions including Europe, North America and Australia, later in most parts of the developing world, prominently in China and Southeast Asia, and now in South America and Africa. Economic growth has become both self-sustaining and also essential to economic, societal and political stability, to the extent that any perceived threat to economic growth is widely interpreted as a sign of the end of civilisation as we know it.

The pattern of growth in Fig. 8.1 is so ingrained in our thinking that we take it as a natural state of affairs. However, the near-ubiquitous growth of the last 200 years is nothing short of astounding in historical terms. For over 99.9 % of the history of genus *Homo* on this planet, human groups and societies grew not at all or only very slowly. We regard growth as "business as usual" because the recent 200 years of strong growth, about eight human generations or three lifetimes, is long enough to exceed our capacity for direct intergenerational memory.

Focussing on the C cycle and climate, the interacting components of the Earth System that are the main subject of this chapter, there has been near-exponential growth in forcing from CO_2 (and other GHG) emissions and a similar response in both CO_2 concentrations and global temperature. Figure 8.2 shows total anthropogenic emissions of CO_2, atmospheric CO_2 concentration and global temperature for the period 1850–2010. These three quantities are respectively the largest single anthropogenic forcing on the carbon-climate system, the proximate response in atmospheric CO_2 concentration, and the further response in warming of the climate system. CO_2 makes by far the largest contribution to the present radiative imbalance

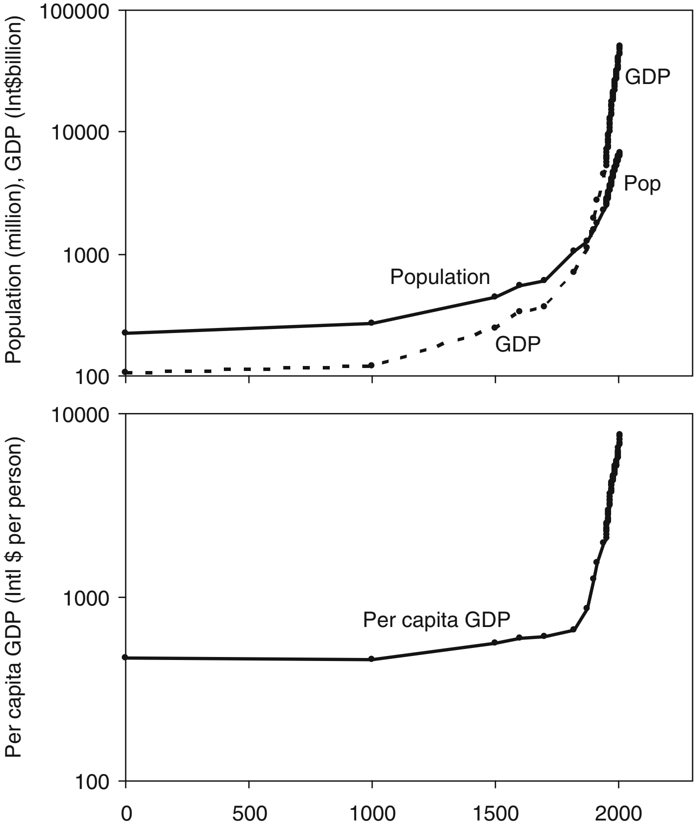

Fig. 8.1 *Upper panel*: Estimates of global population and global aggregate gross domestic product (GDP), from 1 to 2008 AD. GDP is measured in International Gheary-Khamis dollars, a unit that is constant in time and approximates the purchasing power of 1 US dollar in 2000. *Lower panel*: Per capita GDP over the same period. Approximate growth rates over 1800–2000 are 1.3 % year^{-1} for global population, 2.8 % year^{-1} for GDP, and 1.5 % year^{-1} for per capita GDP (Data source: Maddison 2010)

of the climate system, accounting for about 60 % of the forcing from all anthropogenic GHG emissions. Anthropogenic aerosols and other non-gaseous forcings exerting a net cooling effect comparable to the net warming caused by non-CO_2 GHGs (IPCC 2007a).

There is a crucial connection between Figs. 8.1 and 8.2, because CO_2 emissions are closely coupled with both energy use and economic activity or GDP (Raupach et al. 2007). The carbon intensity of energy (a measure of the amount of CO_2 emitted for each unit of energy generated, averaged across all energy sources) is a conservative number, changing only slowly in time and not varying much between countries, because it is determined by the technologies in the energy mix. The carbon intensity of the economy (a measure of the amount of CO_2 emitted for each unit of GDP

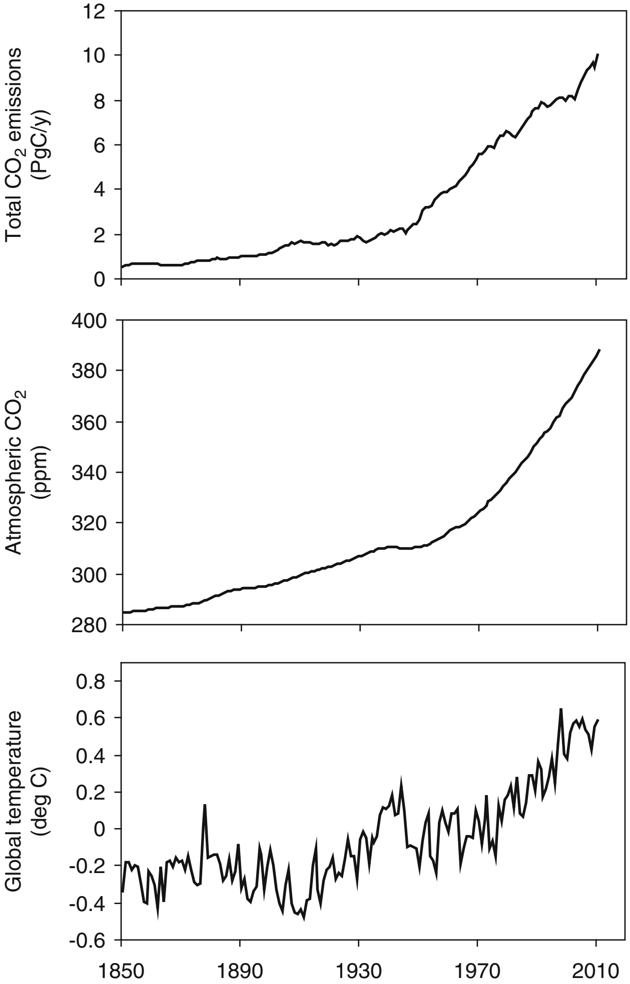

Fig. 8.2 Forcing and response of the climate system by human perturbation of the global carbon cycle, from 1850 to 2010. *Upper panel*: Total global CO_2 emissions from fossil fuel combustion and other industrial processes (mainly cement production), and from land use change. *Middle panel*: Atmospheric CO_2 concentration. *Lower panel*: Global temperature (Data sources: Global CO_2 emissions from fossil fuel combustion and other industrial processes from the Carbon Dioxide Information and Analysis Center (CDIAC) (Boden et al. 2009); Global CO_2 emissions from land use change from R.A. Houghton (GCP 2011); atmospheric CO_2 concentrations prior to 1959 from the Law Dome ice core (Etheridge et al. 1996), for 1959–1980 from averaged in situ measurements at Mauna Loa (Hawaii) and the South Pole (Scripps CO_2 Program 2012), and for 1980–2011 from globally averaged in situ data (NOAA-ESRL 2012); global temperature from the Climatic Research Unit, University of East Anglia (CRU 2012))

averaged across a whole economy for a nation or the world) is also conservative, tending to decrease (that is, improve) as economic development proceeds, but with major differences between nations (Raupach et al. 2007).

8.2.2 The Challenges of Sustainability and Wellbeing

Recent work (Rockström et al. 2009) has sought to define a biophysical "safe operating space for humanity", with dimensions associated with biodiversity, climate, nutrient cycling, water availability, the integrity of the ozone layer, and several other aspects of the natural world. This is the biophysical space within which the Earth has functioned through the Holocene, the climatically stable period of the last 8,000 years that has seen the independent emergence in four continents of urbanised human civilisations based on agriculture (Wright 2004).

The safe operating space envisaged by Rockström et al. is bounded by thresholds which, if crossed, risk a transition of the Earth System into a biophysical state that is much less favourable for human wellbeing than the present near-Holocene state. As a result of the multitude of human impacts on the Earth System, there are now signs that these boundaries are already exceeded in several critical dimensions, including biodiversity, climate change and nutrient cycling. Driven by ongoing growth, humanity is certain to further exceed these boundaries, and to cross others, in coming decades. This concept is a recent evolution of concerns that have been articulated since the landmark "Limits to Growth" report by the Club of Rome (Meadows et al. 1972). The resources of Planet Earth are finite and cannot support infinite growth. The world, therefore, faces a *challenge of sustainability* in the twenty-first century: to adapt the human enterprise, strongly shaped by centuries of near-continuous growth, to the realities of a finite planet.

The sustainability challenge exists in a social as well as an environmental context. A recent report (Raworth 2012) extends the idea of a biophysical safe operating space for humanity to encompass both a biophysical ceiling and a social foundation. Her definition of a social foundation includes 11 priorities which can be grouped into three clusters, focused on enabling people to be (1) *well*, through food security, adequate income, improved water and sanitation, and health care; (2) *productive*, through education, decent work, modern energy services, and resilience to shocks; and (3) *empowered*, through gender equality, social equity, and having political voice. These priorities spell out the global *challenge of wellbeing*: ensuring adequate individual and societal wellbeing for the world's peoples, especially for those with inadequate wellbeing now.

The challenge of wellbeing is defined in large part by the need to improve global equity. As the world has become more affluent over the last two centuries, inequity in wellbeing has grown. The richest are better off than ever before, while for the poorest, global development has been a curse rather than a blessing (Clark 2007; Collier 2007). Socially equitable societies, providing access to opportunity for all and fostering fairness and cooperation, are sources of wellbeing through all of the priorities for a social foundation defined above. There is evidence that societies function better when they are free from large and growing inequities (Wilkinson and Pickett 2009). This emphasises there are functional as well as ethical reasons for promoting equity and universal access to the social foundations of wellbeing (Raworth 2012).

The two great global challenges of sustainability and wellbeing are intimately entwined in many ways, through multiple human-environment connections in the Earth System that link human health and wellbeing, ecosystem functionality, biodiversity, climate, and productivity. One indicator of their nexus is population health, an essential component of the health and functionality of the entire social-ecological system and a key indicator of the asset base for progress towards environmental sustainability (McMichael 2012).

The nexus between the challenges of sustainability and wellbeing is also widely construed as a tension (Charlton 2011). For developing nations with a strong need to improve wellbeing through economic growth, concerns about environmental sustainability do not necessarily assume the same priority as in developed nations, a dichotomy that is particularly acute in efforts to forge a global consensus on mitigation of climate change. This division in priorities has been seen as a fundamental cause of the collapse of climate negotiations in Copenhagen in December 2009 (Charlton 2011). A recent paper from the developing-nation BASIC group (Brazil, South Africa, India, China) said "the criteria for equity and sustainability, or indeed the very necessity for both, are not immediately apparent to all in an unequal world" (BASIC experts 2011).

In the long run, any framing that demands a choice between environmental sustainability and social equity is creating a false dichotomy. Both challenges are now deeply entwined by the finitude of the resources of Planet Earth, the constraints of a safe biophysical operating space, and the need for a social foundation to ensure adequate global wellbeing, to the point where they are inseparable. Neither challenge can be met without the other. While the entwining of the two challenges takes many forms, some of the deepest entanglements occur around ecosystem services, at scales from local to global – including as prominent examples the ecosystem services associated with global C cycle.

8.2.3 Ecosystem Services

The MEA (Millennium Ecosystem Assessment 2005) defined ecosystem services as "the benefits provided by ecosystems" (p. 38), with understanding that the benefits are towards human wellbeing (p. vi). The MEA identified four categories of ecosystem service: *provisioning services* (e.g., food, fresh water, wood and fibre, fuel); *regulating services* (e.g., climate regulation, flood regulation, disease regulation and water purification); *cultural services* (e.g., aesthetic, spiritual, educational and recreational); and *supporting services* (e.g., nutrient cycling, soil formation, and biological primary production).

It is useful to consider ecosystem services within a broader framework for describing the state and function of a "social-ecological system" – a system with interacting natural and human components, the largest social-ecological system on the Earth being the Earth System itself. The state of a social-ecological system can be described by a set of stocks (or stores, or assets, or reserves) which can be aggregated into five essential "capitals". Different authors choose these capitals slightly differently, but the core concept is common. The following choice is common in the resilience literature (for example, Walker et al. 2004):

- *natural capital*: stocks of the natural resources that supply ecosystem services such as clean water, clean air, fertile soils for food production, and healthy biodiversity;
- *built or manufactured capital*: built environments from cities to homesteads, and the infrastructure that supplies their inhabitants with energy, water, food, transport and communications;
- *human capital*: the collective mental, physical and cultural capacities of the population;
- *knowledge capital*: collective knowledge in minds and encoded in libraries and on the internet, together with the skills to read the codes;
- *social capital*: the liberties, responsibilities, institutions, governance frameworks, and unwritten understandings that enable a society to function effectively, or "the features of social organisation, such as trust, norms and networks, that can improve the efficiency of society by facilitating coordinated actions" (Putnam, quoted in Manderson 2005).

In a social-ecological system, all of these kinds of capitals and their many constituent stocks are linked by flows and transfers of energy, matter and information. The system thus constitutes a vast network. Not only is there continual exchange of energy, matter and information through the network, but also the network itself evolves. Some parts grow and others fade away, new subsystems arise and others disappear.

Adequate functionality of a social-ecological system, encompassing the wellbeing of humans and the ecosystems that support them, depends on sufficient availability of all capitals. Shortage in any of the capitals impairs the ability of the system to function and evolve, for example in terms of social cohesion, food security, and health. All capitals interact with population, though not through an oversimplified "more people means more capital" logic.

The different capitals are not necessarily measured dollars. Some, such as social capital, are difficult to quantify other than through comparative indicators, yet remain essential for system function. Financial capital, in this perspective, is not one of the primary capitals but instead is a "secondary capital" providing a mechanism to facilitate some of the inter-capital and inter-temporal transactions needed for overall system functionality and development – for example, the creation of built and knowledge capital from human capital by paying wages and salaries, or the pricing of natural assets such as water and greenhouse gas emissions to maintain overall system functionality by regulating their use. Other inter-capital transactions are independent of the financial system, such as building natural and social capital from voluntary commitment of human capital, or the benefits from natural capital through "free" ecosystem services.

The selection of capitals varies slightly among different authors. Some (Pretty 2003; Porritt 2007) regard financial capital as one of the primary capitals, and lump knowledge and human capital together. Those who distinguish between knowledge and human capitals (as here) take the view that human capital is essentially people – their bodies, minds and spirits – whereas knowledge capital is embodied in libraries, disk drives and art works. Knowledge capital is coded in words, bytes and images, and can only be accessed by humans with the skills to read the codes and translate them into meanings and actions. Human and knowledge capitals are therefore deeply interdependent. Knowledge capital is the crucial for passing skills and wisdom down through generations and between societies, and thence for the development of civilisation.

Ecosystem services are benefits that flow both between and within all capital stocks, not only natural capital. For example, benefits from provisioning and cultural ecosystem services flow from natural capital to other capitals, via human and built capitals to social and knowledge capitals. Regulating and supporting ecosystem services mainly provide benefits that ensure the ongoing integrity of the natural capital base. In all these ways, adequate and resilient ecosystem services are essential for both sustainability and wellbeing.

Many – perhaps most – ecosystem services are supplied by the environmental commons, and their management is in large part a question of managing the commons. The "tragedy of the commons" (Hardin 1968) concerns all human-environment interactions involving use of a common resource for private gain, the tragedy being that rational maximisation of gain at an individual level leads to overexploitation and eventual ruin of the common resource. Much work has been devoted to the management of tragedy-of-the commons dilemmas. Broadly, solutions involve the emergence of "adaptive governance in complex systems". Known prerequisites (Dietz et al. 2003) are the ability for users of the commons to (1) monitor their common resource, (2) devise rules for sharing and nurturing it, (3) induce compliance with the rules, (4) resolve rather than escalate conflicts, and (5) adapt to change. All of this relies on a stock of social capital (Pretty 2003), in this context meaning the levels of trust, understandings and institutions that enable users of the commons to work together.

Solving the dilemma of the tragedy of the commons is central to the ongoing viability of many ecosystem services, since these are (often privatised) benefits to people and societies that are derived from (usually public) ecosystems and environmental resources. In particular, the challenge of climate change and the global management of GHGs in the atmosphere is now widely seen as a tragedy-of-the-commons issue. Energy production (with associated emissions) brings private benefits, but the consequent climate change is a global public cost. Later in this chapter I will develop the implications of these concepts for the "carbon cycle commons", the globally shared, self-regulating functions of the C cycle.

8.3 The Carbon Cycle and Its Disequilibrium

The global C cycle is the interacting set of C stocks in the Earth System, together with the C flows that connect these stocks[1]. Primary C stocks occur on land (including biomass and soils), in the ocean, in the atmosphere as CO_2, and in fossil fuels (Sabine et al. 2004).

8.3.1 Anthropogenic Carbon Dioxide Emissions

The C cycle has been heavily influenced by human activities, through CO_2 emissions to the atmosphere from the combustion of fossil fuels for energy, other industrial processes, and through the net release of C from terrestrial stocks to the atmosphere as a result of land use change, primarily deforestation for agriculture. Figure 8.3 shows the historic course of CO_2 emissions from fossil fuels and industrial processes (FFI, red line), net land use change (LUC, green line) and their total (FFI+LUC, brown line). The upper panel shows annual emissions, and the lower panel the cumulative emissions since 1751. Vertical axes in both panels are logarithmic, so that exponential growth in emissions would yield a straight line. The dominant contribution to total (FFI+LUC) emissions is from fossil fuel combustion. Other industrial processes, mainly cement manufacture, contribute 3–4 % of total CO_2 emissions (Andres et al. 2012), and emissions from land use change (associated mainly with tropical deforestation in recent decades) contributed around 10 % in 2010 (Canadell et al. 2007; Le Quéré et al. 2009).

The upper panel of Fig. 8.3 shows that the past trajectories for FFI and LUC emissions are different. LUC emissions levelled off over the decades since around 1970, and have decreased since around 2000 (Friedlingstein et al. 2010; Peters et al. 2011). By contrast, FFI emissions have increased strongly and nearly continuously for over a century, including in recent years, apart from a small (1.3 %) dip in 2009 relative to 2008, attributable to the 2008 Global Financial Crisis (Friedlingstein et al. 2010). After this dip, there was a very strong rebound in FFI emissions in 2010, at a near record growth rate of 5.9 % from 2009 to 2010 (Peters et al. 2011). Total (FFI+LUC) emissions have grown close to exponentially for over a century, falling nearly on a straight line on linear-log axes. The average growth rate of total emissions over the whole 160-year period from 1850 to 2010 is 1.9 % per year, corresponding to a doubling time of 37 year.

The cumulative emissions in the lower panel of Fig. 8.3 are the time integrals of the corresponding annual emissions in the upper panel. For well over 100 years, total cumulative emissions have grown nearly exponentially at 1.9 % per year, like

[1] In the carbon cycle community, carbon stocks are usually expressed in carbon mass units as Pg C and fluxes in Pg C year^{-1}, because carbon shifts between multiple chemical forms (coal, oil, biomass, CO_2 and others). Here 1 Pg C = 1 petagram = 10^{15} g of carbon.

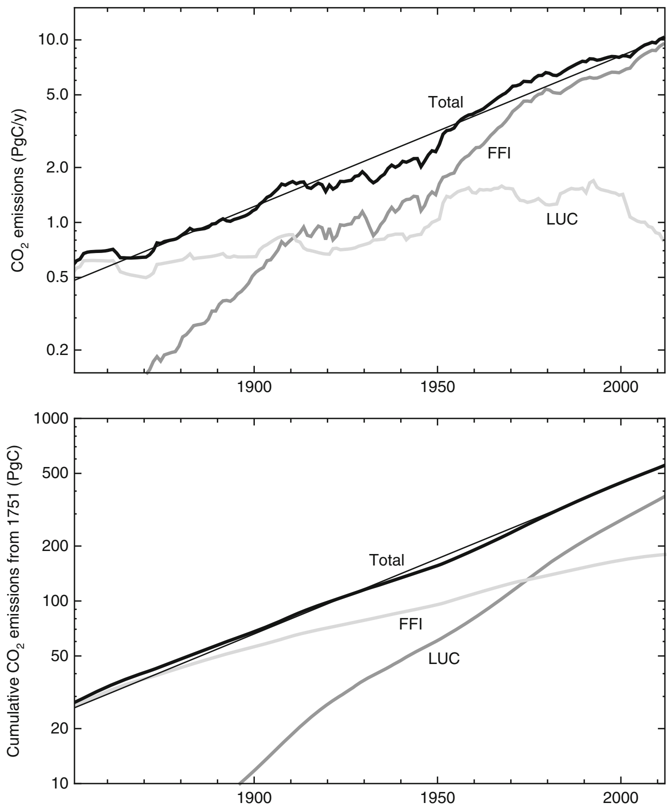

Fig. 8.3 *Upper panel*: Global carbon dioxide (CO_2) emissions from fossil fuels and other industrial processes, mainly cement manufacture (FFI), net land use change (LUC) and total (FFI + LUC), all in Pg C−1. *Lower panel*: Cumulative global CO_2 emissions from 1751, in Pg °C. *Vertical axes* in both panels are logarithmic, so that exponentially growing emissions appear as a *straight line*. In both panels, the *thin straight line* is a fit to the total (FFI + LUC) emissions data of an exponential-growth model with a growth rate of 1.9 % per year, corresponding to a doubling of emissions and cumulative emissions every 37 years (Data sources: Global CO_2 emissions from fossil fuel combustion and other industrial processes from the Carbon Dioxide Information and Analysis Center (CDIAC) (Boden et al. 2009); Global CO_2 emissions from land use change from R.A. Houghton (GCP 2011))

the annual total emission (since the integral of an exponential is another exponential with the same growth rate). The scatter in the total cumulative emission about the exponential-growth line is much less than for annual emissions, because of the smoothing effect of accumulation. The total cumulative emission to the end of 2010 was about 540 Pg C, rising at nearly 10 Pg C per year (Raupach et al. 2011). Of this total about 360 Pg C is due to FFI and 180 Pg C to LUC, but the share of the cumulative total due to FFI is increasing progressively.

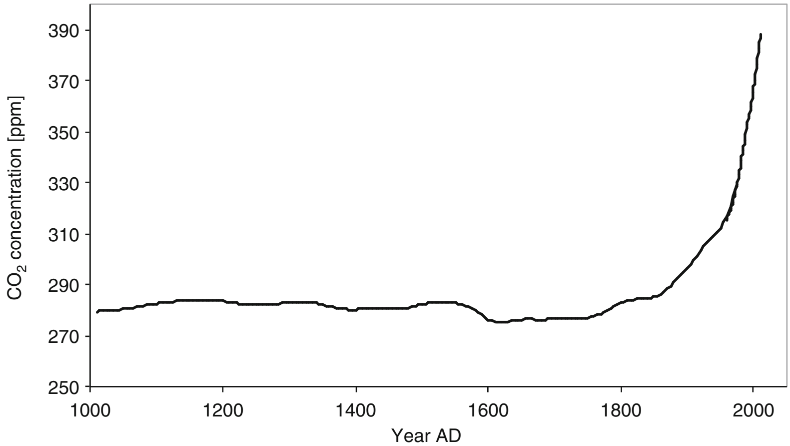

Fig. 8.4 Atmospheric carbon dioxide (CO_2) concentrations from 1000 to 2010 AD. Data prior to 1959 from the Law Dome ice core (Etheridge et al. 1996); for 1959–1980 from averaged *in situ* measurements at Mauna Loa (Hawaii) and the South Pole (Scripps CO_2 Program 2012); and for 1980–2011 from globally averaged *in situ* data (NOAA-ESRL 2012)

8.3.2 Responses of the Carbon Cycle

Figure 8.4 shows atmospheric CO_2 concentrations since 1000 AD, combining the record from the Law Dome ice core (Etheridge et al. 1996; MacFarling Meure et al. 2006) with modern *in-situ* measurements from direct atmospheric observations (from 1958 onward) at Mauna Loa (Hawaii), the South Pole, and other locations. Ice cores yield records of past CO_2 concentrations (among many other important palaeoclimate data) from the air trapped in tiny bubbles in the ice. The ice-core and modern instrumental records are in excellent agreement, together showing that CO_2 concentrations have risen steeply since 1800, from a nearly steady level of 278 ± 5 ppm through the prior millennium (and for several millennia before that) to 389 ppm in 2010. The recent (2001–2010 average) rate of increase in CO_2 concentrations is 2 ppm year^{-1}, with some fluctuation from year to year.

Growth in atmospheric CO_2 is a direct response to anthropogenic CO_2 emissions (Fig. 8.3). However, not all of the emitted CO_2 remains in the atmosphere. The amount of CO_2 accumulating there is determined by the balance between inflows and outflows, or a "CO_2 budget", that can be written

$$\begin{bmatrix}\text{atmospheric } CO_2\\ \text{accumulation}\end{bmatrix} = \begin{bmatrix}\text{FFI}\\ \text{emissions}\end{bmatrix} + \begin{bmatrix}\text{LUC}\\ \text{emissions}\end{bmatrix} - \begin{bmatrix}\text{Land}\\ CO_2 \text{ sink}\end{bmatrix} - \begin{bmatrix}\text{Ocean}\\ CO_2 \text{ sink}\end{bmatrix} \quad (8.1)$$

Figure 8.5 shows the terms in Eq. 8.1 from 1850 to 2010 (GCP 2011, updating Canadell et al. 2007 and Le Quéré et al. 2009). Inflows (above the horizontal axis) occur through FFI and LUC anthropogenic emissions (see previous subsection).

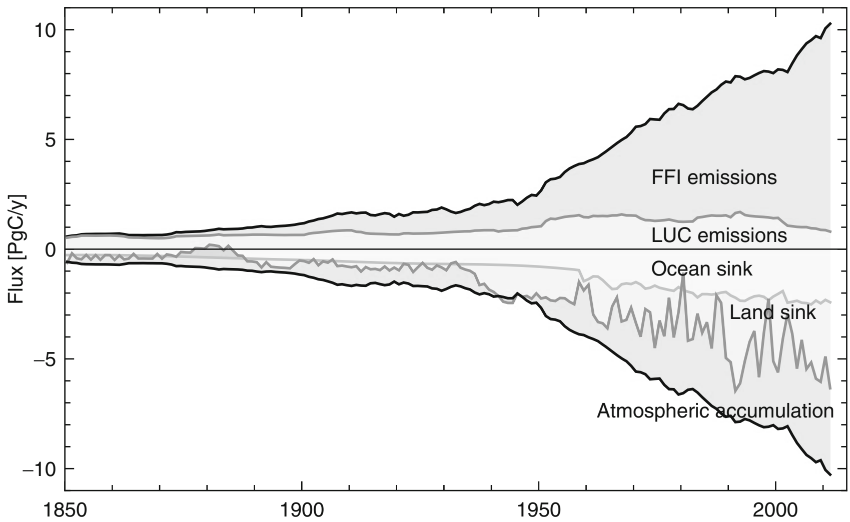

Fig. 8.5 Terms in the global budget of atmospheric carbon dioxide, [atmospheric CO_2 accumulation] = [FFI emission] + [LUC emission] − [Land sink] − [Ocean sink], from 1850 to 2010. As in Fig. 8.3, FFI denotes emissions from fossil fuels and other industrial processes, mainly cement manufacture, and LUC denotes emissions from net land use change or net deforestation (Data source: GCP 2011)

These are balanced by the sum of atmospheric accumulation and the natural CO_2 sinks in the land and oceans, shown as a matching total below the line.

The land CO_2 sink arises because the terrestrial biosphere is accumulating carbon in biomass, litter and soils. It is driven by several processes (Raupach and Canadell 2008), the main ones being the "CO_2 fertilisation" response of plant growth to the increase in atmospheric CO_2 (Sitch et al. 2008), responses to increased anthropogenic N deposition (Thornton et al. 2009), and changes to disturbance regimes, including regrowth after deforestation in earlier decades and reductions in fire regime. The land sink is associated mainly with forests and other ecosystems with significant woody vegetation, such as savannahs, because these systems contain large C stocks (Pan et al. 2011).

The global land sink varies strongly from year to year. This variability is driven mainly by interannual variations in the availability of water and light, and changes to patterns of disturbance from fire and deforestation. Much of the variability is associated with the effects of the El Niño-Southern Oscillation (ENSO) climate mode, which modulates tropical and subtropical precipitation, temperatures, surface direct and diffuse irradiance (through cloudiness and aerosol levels) and fire regimes on time scales from 2 to 7 years. This association is demonstrated by the observation that ENSO indices are well correlated with interannual fluctuations of atmospheric CO_2 accumulation (Keeling and Revelle 1985; Raupach et al. 2008). The ENSO climate mode is therefore the "global pulse" that drives much of the variability of the land CO_2 sink and atmospheric CO_2 accumulation (Fig. 8.5).

The land sink is also significantly affected by volcanic eruptions such as Agung (1961), El Chichon (1982) and Pinatubo (1991) (Jones and Cox 2001; Sarmiento et al. 2010), through several processes, prominently the increased scattering of solar radiation by stratospheric aerosols after a major eruption, leading in turn to an increased ratio of diffuse to direct surface solar irradiance, increased light use efficiency at canopy level, and increased plant photosynthesis and growth (Gu et al. 2003).

The ocean CO_2 sink occurs because CO_2 dissolves in ocean waters when atmospheric CO_2 levels are higher than the partial pressure of CO_2 (pCO_2) in the surface of the ocean. Transport of the added carbon from surface to deep ocean waters occurs both through overturning currents (the "physical pump") and sinking of organic detrital material (the "biological pump"). The added CO_2 is partitioned between dissolved CO_2, bicarbonate (HCO_3^-) and carbonate (CO_3^{--}) ions, a buffering process that allows the ocean to take up far more CO_2 than it would without repartitioning (the gain of this amplifier is called the Revelle factor). The repartitioning of the added CO_2 also alters the equilibrium between dissolved CO_2, HCO_3^- and CO_3^{--}, lowering CO_3^{--} and raising HCO_3^-, with the consequence that ocean pH is also lowered, so that the ocean becomes less alkaline and more acid. The ocean pH has change from about 8.2 in preindustrial times to 8.1 now, with an associated reduction in carbonate concentrations. Future further reductions in pH and carbonate concentrations are predicted to have significant implications for marine ecosystems, impairing the growth of calcifying organisms at the base of the food chain (McNeil and Matear 2008).

The interannual variability of the global ocean CO_2 sink is much less than that of the land CO_2 sink, and tends to have the opposite phase in relationship to ENSO indices (Feely et al. 2006). In consequence, most of the variability in atmospheric CO_2 accumulation is caused by variations in the land sink (Fig. 8.5).

8.3.3 *Self-regulation of the Carbon Cycle*

There are continual two-way exchanges between the C stored in the atmosphere as CO_2 and the C stored in the land biosphere and the oceans. During the Holecene epoch prior to the industrial era (8,000–200 years ago), these exchanges led to an equilibrium in which the atmospheric CO_2 concentration was nearly steady at 278 ± 5 ppm (Fig. 8.4). From around 1800 onward, excess atmospheric CO_2 has built up as a result of anthropogenic CO_2 emissions, leading to a C cycle response in which land and ocean CO_2 sinks redistribute excess C from the atmosphere into land and ocean C stocks, by processes sketched above.

Persistent land and ocean CO_2 sinks exist because of the disequilibrium of the global C cycle caused by anthropogenic CO_2 emissions, and represent the tendency of the C cycle to restore an equilibrium or zero-net-flux partition of C between atmospheric, land and ocean stocks. This is a form of self-regulation by the C cycle, in which the CO_2 sinks exert a strong negative or stabilising feedback.

From 1959 to 2010, land and ocean sinks respectively removed about 30 and 25 % of total (FFI + LUC) emissions, leaving about 45 % to accumulate in the atmosphere. This accumulating fraction is known as the airborne fraction (AF). The sink fraction, SF = 1 − AF, is the fraction absorbed by land and ocean sinks. From observations, the AF and SF exhibit three features (Canadell et al. 2007; Raupach et al. 2008): (1) the AF and SF have been close to constant on average, at AF ≈ 0.45 and SF ≈ 0.55, over the period 1959–2010 (the period of high-quality *in-situ* atmospheric CO_2 observations); (2) there has been significant interannual variability in both the AF and the SF; (3) although *nearly* constant, the AF and SF have not been *exactly* constant – there has been a small but statistically significant increase in the AF over the period 1959–2010, with an accompanying decrease in the SF. Each of these features provides important insights into the self-regulation of the C cycle, as follows.

The first feature, near-constancy, is significant because the AF and SF would be precisely constant under two idealisations: that the land and ocean sinks increase linearly with excess atmospheric CO_2 above the preindustrial equilibrium level, and that total CO_2 emissions are growing exponentially (Bacastow and Keeling 1979). The observed long-term near-constancy of the AF and SF indicates that these idealisations are quite good approximations through the period 1959–2010 (noting the independent direct evidence for near-exponential growth in total CO_2 emissions; Fig. 8.3). This simple argument suggests a near-linear response on decadal time scales of both land and ocean CO_2 sinks to excess CO_2, in which the sink strength is linearly proportional to the excess CO_2 concentration above the preindustrial level of 278 ± 5 ppm.

The second feature, the interannual variability of the AF and SF, arises mainly from interannual variations in the land sink, as noted above. These fluctuations are driven in turn by interannual climate variability and volcanic eruptions. Almost all of the interannual variability arising from these factors is rapid on multi-decadal time scales, occurring at periods of less than 5 years (Raupach et al. 2008).

The third feature, the observed fact that the AF is increasing and SF decreasing, is the most challenging. Over the period 1959–2010, the AF has a small positive (increasing) trend at a proportional growth rate of 0.2–0.3 % year^{-1}, with a confidence P > 0.9 (Canadell et al. 2007; Raupach et al. 2008; Le Quéré et al. 2009 and updates). Here the proportional growth rate is $(1/\mathrm{AF})d\mathrm{AF}/dt = d(\ln \mathrm{AF})/dt$, and the confidence or P value is the probability of a positive trend. Since a precisely constant AF would arise if land and ocean sinks respond linearly to excess atmospheric CO_2 and total CO_2 emissions grow exponentially, non-constancy of the AF indicates a failure in one or both of these idealisations. In view of this, three broad causes have been proposed for the observed positive trend in the AF:

- departure of total (FFI + LUC) CO_2 emissions from exponential growth (Gloor et al. 2010);
- effects of particular events, such as the volcanic eruption of Pinatubo in 1991, on the land C stock, with consequences for the distribution of anthropogenic CO_2 between atmosphere, land and oceans at decadal and longer time scales (Sarmiento et al. 2010);

- decreases in the efficiency of land and ocean C sinks through nonlinear responses to elevated CO_2, nutrient availability, land management, warming, precipitation and ecosystem changes (Le Quéré et al. 2009).

Of these factors, the last would indicate a weakening or loss of efficiency in the land and ocean CO_2 sinks, in the sense that while they are continuing to grow with rising in atmospheric CO_2, they are progressively "losing the race" against exponentially growing emissions, leading to a decline in the SF. Such a loss of efficiency may be occurring for many reasons. The efficiency of the land sink decreases as plants experience "diminishing returns" in CO_2 fertilisation of plant growth with increasing CO_2 (Sitch et al. 2008), and as nutrient (N and P) limitations constrain terrestrial primary production (Thornton et al. 2009; Wang et al. 2010). The ocean sink may lose efficiency due to changes in ocean ecosystems (McNeil and Matear 2008), and also changes in ocean circulations, particularly the exchange between surface and deep ocean waters (Le Quéré et al. 2007). This important question is not yet fully resolved. The trends in the AF and SF (respectively increasing and decreasing over 1959–2010) are statistically clear with high confidence ($P>0.9$), and analysis of the attribution is ongoing.

In summary, the global C cycle has demonstrated a high self-regulating ability over the 200 years since the onset of near-exponentially growing CO_2 emissions around 1800, with land and ocean CO_2 sinks together absorbing over half of all emissions ($SF \approx 0.55$), leaving less than half ($AF \approx 0.45$) to accumulate in the atmosphere. Without this strong self-regulation, the observed global temperature rise through the industrial era (about 0.8 °C, from Fig. 8.2) would have been around twice as great. There are signs that the AF has increased over the last 50 years, possibly implying a decrease in the efficiency of the land and ocean sinks, though alternative explanations exist for the AF trend. Whatever the resolution of this issue, the high self-regulating ability of the C cycle over the time from 1800 to the near present is well established.

A question of critical importance is whether the strong self-regulation of the C cycle will weaken significantly in the face of further climate change ensuing from continuing increases in anthropogenic CO_2 emissions. Initial responses to question have emerged from carbon-climate model intercomparisons (Friedlingstein et al. 2006; Sitch et al. 2008) that indicate significant enhancement of future climate change by climate-induced effects on land and ocean CO_2 sinks, under strong warming scenarios. For example, Friedlingstein et al. found that carbon-climate feedbacks engendered by coupling of climate and carbon-cycle models cause an increase of 0.1–1.5 °C in a median warming to 2,100 of around 4 °C above preindustrial temperatures, with the range (encompassing 11 models) due mainly to different predictions for the land CO_2 sink.

8.4 Ecosystem Services and the Carbon Cycle

The global C cycle participates directly in many ecosystem services and indirectly in all. It is fundamental to supporting services (nutrient cycling, soil formation, and biological primary production). Terrestrial primary production of C-compounds in biomass underpins the provisioning services that provide food, fibre and bioenergy from farming and forestry, and aquatic primary production of C-compounds sustains all fisheries. All the great natural cycles of the Earth System, including C, water and nutrients, together underpin cultural (aesthetic, spiritual, educational, recreational) and regulating services (flood regulation, disease regulation and water purification).

Acknowledging the direct and indirect roles of the C cycle in all ecosystem services, the main focus of this section is on the role of the C cycle in ecosystem services associated with carbon-climate interactions at global scale. These ecosystem services include (1) services that protect the Earth System against carbon-climate vulnerabilities or reinforcing feedbacks that would accelerate climate change, and (2) services provided by C storage and sequestration in the terrestrial biosphere, to contribute to mitigating climate change.

8.4.1 Protection Against Carbon-Climate Vulnerabilities

The term "vulnerability" has been used with different emphases in several disciplines. A definition for human-environment interactions in socio-ecological systems (Turner et al. 2003) is "vulnerability is the degree to which a system, subsystem, or system component is likely to experience harm due to exposure to a hazard, either a perturbation or stress/stressor". With an emphasis on climate change, the Intergovernmental Panel on Climate Change (IPCC) defined vulnerability as "the degree to which a system is susceptible to, or unable to cope with, adverse effects of climate change, including climate variability and extremes" (IPCC 2007b, pp. 883, 783). Both definitions implicitly or explicitly involve three components: a stress or forcing (e.g., CO_2 emissions), the hazard resulting from the response of the system to the forcing (e.g., physical climate change), and risk of harm to actors in the system, evaluated by some subjective criterion or value judgement (e.g., dangers arising from climate impacts).

The term "vulnerability" has also been applied more narrowly to carbon-climate interactions, to mean the extent to which the buildup of atmospheric CO_2 from anthropogenic emissions may be accelerated by climate change itself through reinforcing carbon-climate feedbacks (Cox et al. 2000; Gruber et al. 2004; Friedlingstein et al. 2006). This concept of vulnerability focuses on the relationship between the stress (anthropogenic disturbance of the carbon-climate system) and the hazard (the effects of reinforcing feedbacks or weakening of stabilising feedbacks on system stability), leaving implicit the issues around risk of harm.

The focus here is on the second, narrower meaning of "vulnerability". There are two kinds of potential carbon-climate vulnerability in this sense: (a) decreases in the efficiency of natural land and ocean sinks for atmospheric CO_2, and (b) CO_2 and CH_4 releases from previously inactive C stocks under disturbance by climate change or other anthropogenic influences (an inactive stock here meaning a stock that does not participate in exchanges with atmospheric CO_2 or CH_4). Protection against each of these vulnerabilities constitutes an ecosystem service.

The ecosystem service provided by land and ocean CO_2 sinks: As outlined in the previous section, natural land and ocean CO_2 sinks have ameliorated the buildup of atmospheric CO_2 due to human activities by absorbing over half of total CO_2 emissions throughout the industrial era to date (SF $\approx$ 0.55), thereby limiting climate change to date to about half of what it would have been in the absence of land and ocean sinks (that is, if the SF were zero or the AF were one).

This service by land and ocean CO_2 sinks is arguably the world's largest ecosystem service. With total CO_2 emissions of 10 Pg C year^{-1} or 37 Pg CO_2 year^{-1} in 2010 (GCP 2011) and at a C price of US$20 per tonne of CO_2, the monetary value of the C mitigation provided by land and ocean sinks is nearly US$800 billion per year, about 1.5 % of global total GDP. This ongoing global ecosystem service is free, unpriced, unregulated and largely unnoticed.

The ecosystem service by land and ocean CO_2 sinks cannot be taken for granted, or assumed to continue forever in the face of major climate change. Evidence from climate model intercomparisons suggests that strong self-regulation of the carbon-climate system (SF > 0.5) will weaken in the face of the climate change ensuing from continuing increases in anthropogenic CO_2 emissions (Friedlingstein et al. 2006; Sitch et al. 2008), particularly under emissions scenarios leading to strong warming (3 °C or more).

The ecosystem service provided by maintenance of inactive carbon stocks: Several hitherto largely inactive C stocks can be disturbed by climate change, leading to release of CO_2 or CH_4 to the atmosphere and a consequent reinforcing feedback that would accelerate climate change. Major potentially vulnerable C stocks are as follows.

- A major C stock is the organic C in permanently frozen or presently permafrost soils, estimated at nearly 1,700 Pg C in total (Tarnocai et al. 2009). Of this, around 100 Pg C may be vulnerable to release by thawing over the next century (Schuur et al. 2009).
- There is a significant stock of C in tropical peatland soils, mainly in the Southeast Asian archipelago, of which around 30 Pg C may be vulnerable to decomposition and fire following drainage associated with land use change (Hooijer et al. 2010).
- Net releases of C in temperate and boreal forest ecosystems are likely through fire, insect attack and ecological transitions (Kurz et al. 2008).
- Release of methane hydrates from reservoirs on the ocean floor and beneath permafrost is a high-impact, highly uncertain risk but is presently thought likely to occur over longer timescales than the other releases noted above. The biggest short-term vulnerability for this stock may well be through its use as a fuel source (Bohannon 2008).

8.4.2 Carbon Sequestration in the Terrestrial Biosphere

Carbon sequestration in the terrestrial biosphere (the process of increasing the C content in biomass and/or soil C stocks by human management) is one of a suite of strategies for mitigating anthropogenic climate change that are collectively called

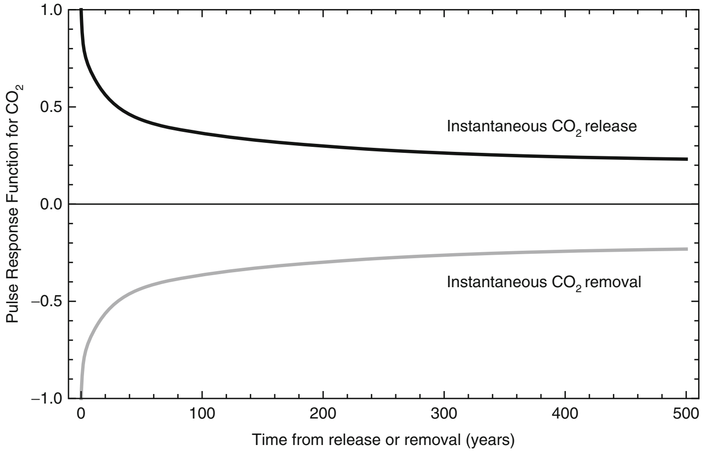

Fig. 8.6 The *black curve* shows the atmospheric pulse response function for CO_2, the fraction of an instantaneous pulse of CO_2 injected into the atmosphere that remains airborne after time *t* (IPCC 2007a, p 213). The CO_2 pulse into the atmosphere decays because land and ocean CO_2 sinks progressively remove CO_2 from the atmosphere to restore equilibrium. The *grey curve*, the negative of the *black curve*, is the fraction of a pulse of CO_2 instantaneously removed from the atmosphere that remains out of the atmosphere. In this case, CO_2 is progressively released back into the atmosphere because of their present disequilibrium

carbon dioxide removal (CDR). CDR strategies involve removal of CO_2 from the atmosphere by enhanced biological uptake and storage in terrestrial or oceanic systems, or by combining artificial CO_2 removal with geological storage.

Before turning to CDR through C sequestration in the terrestrial biosphere, it is important to recognise two points about CDR in general. The first is that the world already faces a choice between CDR and significant climate change. Of the four representative concentration pathways (RCPs) defined for the IPCC Fifth Assessment (Moss et al. 2010; van Vuuren et al. 2011), only the lowest constrains global warming to 2 °C or less (from a preindustrial baseline), and it requires significant CDR to meet this target.

The second general point is that the present disequilibrium of the C cycle has an important consequence for all CDR strategies. There is a "bounceback" of atmospheric CO_2 after an anthropogenic removal by CDR, just as there is a drawdown of CO_2 after an anthropogenic CO_2 emission into the atmosphere. This is illustrated in Fig. 8.6. The black curve shows the "atmospheric pulse response function" for CO_2, the fraction of an instantaneous pulse of CO_2 emitted into the atmosphere that remains airborne after time *t*, or (equivalently) the increment in CO_2 concentration caused by instantaneously adding 1 ppm of CO_2 (2.13 Pg C) to the atmosphere. The added CO_2 concentration increment decays with time because land and ocean CO_2 sinks progressively remove CO_2 from the atmosphere to restore equilibrium between atmospheric, land and ocean C stocks. The grey curve, the negative of the black curve, is the corresponding increment in CO_2 concentration arising from the instantaneous withdrawal of 1 ppm of CO_2 from the atmosphere by a CDR activity. In this case, CO_2 is progressively released back into the atmosphere from the existing excess C in land and ocean stocks, because the land and ocean sinks do not work as efficiently as they would have without the CDR withdrawal.

The implication is that just as land and ocean CO_2 sinks provide a major ecosystem service by taking away a proportion of all CO_2 anthropogenic emissions entering the atmosphere, the same processes also provide an "ecosystem disservice" by returning a proportion of CO_2 removals from the atmosphere by any CDR activity.

We turn now to CDR through C sequestration in the terrestrial biosphere. Natural terrestrial stocks of C are large, estimated at around 650 Pg C in biomass, 2,300 Pg C in soil organic carbon (SOC) in tropical, temperate and boreal ecosystems (Sabine et al. 2004), an additional soil inorganic carbon (SIC) stock that is largely inactive in the present context, and nearly 1,700 Pg C as organic C permafrost or frozen soils (Tarnocai et al. 2009). These stocks can be compared with an atmospheric stock of 820 Pg C (388 ppm) in 2010. From the magnitude of the terrestrial biospheric C stocks, it would appear that there is room to sequester a large fraction of annual anthropogenic emissions (10 Pg C year^{-1} in 2010 GCP 2011) by small proportional increases in biomass and non-frozen soil C stocks. However, several factors mean that this option is no magic cure.

To be realistic about the potential for C sequestration in the terrestrial biosphere as a mitigation option, it is necessary to distinguish between technical potential (the maximum biophysically possible sequestration) and attainable potential (the sequestration achievable under other unavoidable constraints including biophysical requirements for water and nutrients, competition for land, economic factors such as the price of C, and social choices). The attainable potential depends strongly on all these constraints (Raupach et al. 2004). Following are several high-level issues bearing on the attainable potential for C sequestration in the terrestrial biosphere as a mitigation option.

- *Longevity*: As reviewed in Sect. 8.3, terrestrial (and ocean) C stocks are continually exchanged with the atmosphere, and are not permanently sequestered. If conditions change through disturbance, climate change or changes in CO_2 concentration, C sequestered in the terrestrial biosphere can return to the atmosphere. This implies both that C sequestered in the terrestrial biosphere is vulnerable to disturbance (e.g., forest fire) and also that changes in management to sequester C (e.g., agricultural practices to increase SOC) need to be ongoing.
- *Additionality*: great deal of C uptake into the terrestrial biosphere is happening naturally (without direct human intervention) through the land C sink. As reviewed in the last section, the land sink has absorbed about 30 % of total anthropogenic CO_2 emissions on average over 1959–2010, with the oceans absorbing a further 25 %, together yielding a sink fraction (SF) of 55 %. This massive and unvalued ecosystem service is already implicitly counted as a benefit towards mitigation of climate change, in that its continuance is assumed. Carbon sequestration for purposeful mitigation needs to be additional to the ongoing natural land C sink, to avoid double counting.

- *Nutrient requirements*: The issue of nutrient (N and P) inputs is critical. Carbon in the terrestrial biosphere, either as SOC and biomass C, exists in a variety of chemical forms with turnover rates from highly labile (fast) to inert (slow), each form having a characteristic C:N:P ratio. Buildup of C stocks in the terrestrial biosphere therefore also requires significant nutrient inputs. Nutrient limitation may be a global constraint on managed terrestrial C sequestration, and also on the long-term future of the natural land CO_2 sink.
- *Measurement and verification*: Reliable measurement of sequestered C in the terrestrial biosphere is very challenging, especially in the presence of large fluctuations in sinks and terrestrial C stocks due to climatic variability (Fig. 8.5 and associated discussion).
- *Leakage and double-counting*: There is pressure on countries to use terrestrial C accounting for advantage in climate change negotiations, in ways that result in global-scale double counting. For example, prospective internationally traded C credits for avoided deforestation (where a developed country pays a developing tropical country to preserve forest that would otherwise be cleared, and in exchange receives C credits to allow emissions to continue) are often counted in scenarios for future mitigation by both the credit-buying and the credit-selling country.
- *Full radiative forcing and climate implications*: Any land management activity has multiple potential impacts on radiative forcing and climate, in addition to changes in C storage. These include release of other GHGs such as CH_4 and N_2O, direct radiative impacts through changes in albedo, and (if carried out at large scale) changes in regional circulation through alterations to the surface energy balance and roughness. Evaluation of these multiple impacts requires consideration of land use systems (such as agriculture, grazing or forestry) rather than individual terrestrial stocks (such as biomass or soil).

Recognition of issues like these has led to proposed “integrity standards” for the use of C sequestration in the terrestrial biosphere as a mitigation option. For example, the standards proposed by the Australian Government for its “Carbon Farming Initiative” (DCCEE 2010) involve seven criteria for C sequestration projects: additionality, permanence, avoidance of leakage, verifiability, conservatism (to ensure that abatement claims are not overestimated), consistency (with other national and international C accounting), and support by peer-reviewed science.

8.5 Conclusions

This chapter began with a broad view of human impacts on the Earth System, and then highlighted three aspects of the contemporary carbon-climate system: the near-exponential increase in anthropogenic CO_2 emissions since around 1800; land and ocean CO_2 sinks as responses of the C cycle to disequilibration by anthropogenic CO_2 emissions; and the self-regulation of the C cycle as indicated by the near-constancy of the CO_2 airborne and sink fractions at around 45 and 55 %, respectively. This background material introduced an assessment of two groups of ecosystem services associated with the global C cycle: services that protect the Earth System against carbon-climate vulnerabilities or reinforcing feedbacks that would accelerate climate change, and services provided by C sequestration in the terrestrial biosphere as a climate-change mitigation option.

Identification of the ecosystem services associated with the global C cycle highlights the significance of the “carbon cycle commons”, the globally shared, self-regulating functions of the C cycle. The carbon cycle commons are an important subset of the “Earth System commons”, the connected functioning of the land, waters, air and ecosystems of Earth, particularly those processes that naturally regulate and stabilise climate and the great natural cycles of energy, water, C and other entities.

The services provided by the carbon cycle commons to humanity – and more broadly by the Earth System commons – are only partly in the form of direct benefits. They are also in the form of protection against vulnerabilities, or risks of potential changes in Earth System functioning that would be harmful to human wellbeing. The changes to the Earth System in the Anthropocene epoch are sufficiently profound that continued health of these great global commons cannot be taken for granted.

Degradation of the carbon cycle commons can occur through decreases in the efficiency of land and ocean CO_2 sinks, and through vulnerabilities in previously inactive C stocks. The land CO_2 sink may decrease in efficiency because of “diminishing returns” in CO_2 fertilisation of plant growth with increasing CO_2, and the effects of nutrient (N and P) limitations on terrestrial primary production. The efficiency of the ocean CO_2 sink could decrease through changes in ocean ecosystems resulting from ocean acidification or other anthropogenic impacts, and changes in ocean circulation patterns caused by climate change itself. Carbon-climate vulnerabilities could occur in future through releases of presently inactive C stocks (in frozen soils, tropical peatlands, and forest ecosystems) under disturbance by climate change, leading to acceleration of climate change through reinforcing feedbacks.

Recognition of issues like these has led to calls for international monitoring of Earth System functions associated with the self-regulation of the C cycle and related biogeochemical cycles, and their critical role in climate regulation. In the case of the C cycle, this could occur through an “International Carbon Office” (ICO) (Le Quéré et al. 2010) to provide ongoing, robust and transparent monitoring and assessment of C and other biogeochemical cycles at global and regional scales, particularly regarding land and ocean CO_2 stocks, sinks and sources. The need for such an office to be mandated at international rather than national level follows directly from the global nature of the C cycle and Earth System commons, the shared benefits that all nations derive from them, and the shared responsibility that all nations have for protecting them.

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Chapter 9
Loss of Soil Carbon to the Atmosphere via Inland Surface Waters

Julian J.C. Dawson

Contents

Abstract Within the global carbon (C) cycle, there is still much debate as to the magnitude, location and turnover of the terrestrial C sinks (and sources). One of the major keys to closing this knowledge gap is that globally, the amount of C entering oceans maybe only *ca*. 33 % of the total C transported from terrestrial ecosystems to inland surface waters. Streams, lakes, rivers and transitional waters are areas for the active transformation and recycling of terrestrially-derived C indirectly back to the atmosphere (estimated range of 25–44 %). Understanding processes that control soil C losses to and its fate in surface waters is not only important in establishing accuracy of C fluxes, feedbacks and tradeoffs but also providing evidence to limit terrestrial ecosystem C contributions to atmospheric carbon dioxide (CO_2).

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The relationship between inland surface waters and C cycling are controlled by biogeochemical, physical and hydrometeorological metrics that integrate both lateral (soil to water) and longitudinal (along the riverine continuum) processes during C transport in its different forms, i.e., particulate, dissolved and gaseous C species. This chapter outlines processes affecting compositional "quality" of C within surface waters and in-stream physico-chemical and biotic mechanisms that are instrumental to understanding losses of C via the soil-surface water-atmosphere pathway.

Keywords Carbon • Dissolved organic carbon • Particulate organic carbon • Organic matter • Inland surface waters • Carbon dioxide outgassing • In-stream processes • Biogeochemical quality • Aquatic carbon cycle

Abbreviations

C	Carbon
CH_4	Methane
CO_2	Carbon Dioxide
DIC	Dissolved Inorganic Carbon
DOC	Dissolved Organic Carbon
DOM	Dissolved Organic Matter
FT-IR	Fourier Transform Infrared
GHG	Greenhouse Gas
GPP	Gross Primary Production
OM	Organic Matter
N_2O	Nitrous Oxide
NEE	Net Ecosystem Exchange
PIC	Particulate Inorganic Carbon
POC	Particulate Organic Carbon
POM	Particulate Organic Matter
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
UV	Ultra-Violet

9.1 Introduction

The global carbon (C) cycle is comprised of three main compartments: land (soil, vegetation and geological), ocean and atmospheric pools (Janzen 2004; Smith et al. 2008), each with their relative stocks of C that are constantly fluctuating. It is a complex system to understand and accurately estimate its C cycling processes, and there has been considerable disagreement over C stocks and the movement of C (i.e., C fluxes) between pools for disparate ecosystems (Dixon and Turner 1991; Malhi 2002). For example, total soil C content varies spatially within the same mapped soil unit and C stock estimates within a given area can vary depending on the scale of soil mapping (Dawson and Smith 2007). The myriad of spatio-temporal variations (e.g., soil characteristics, gaseous emissions, seasonality) combined with diverse land management strategies complicate derivation of C stocks, fluxes and processes in terrestrial (soil and vegetation) ecosystems (Chadwick et al. 1994; Cao et al. 2002). Furthermore, soil organic C stocks will be sensitive to changes in climate, but the magnitude and timescale of their response as well as potential feedbacks are not fully comprehended (Schmidt et al. 2011). Some discrepancies between literature estimates of the global C stocks and fluxes occur due to the different extrapolation approaches utilised. Limitation of sufficient representative data is one main source of uncertainty in estimates but further understanding of mechanisms and processes that underpin C cycling from microbial to global scales is also required to reduce the uncertainty that persists (Smith et al. 1993; Janzen 2004; Le Quéré et al. 2009).

It is important to note that the global C cycle does not operate independently but is linked to the nitrogen (N), phosphorus (P) and sulphur (S) cycles by biological processes such as mineralization and immobilisation (Richey 1983; Stevenson 1986). The 'Anthropocene' (Crutzen 2002) has been proposed as a name for the present epoch (commencing with the industrial revolution, *ca* 1750) in which consumption of fossil fuels releasing C, N and S to the atmosphere, together with N, P and ammonia fertilizer production and land-use changes (e.g., cultivation, clearing and harvesting of forests) have resulted in significant perturbations to these natural biogeochemical cycles (Bolin et al. 1983; Malhi 2002; Janzen 2004; Raupach and Canadell 2010). These perturbations have impacted climate, biodiversity and land cover resulting in multiple pressures on ecosystems that supply fundamental services for human well-being (Dawson and Smith 2010). One of the most striking examples of this ecosystem disequilibrium is the changing composition of the atmosphere as a result of anthropogenic consumption that has generated increasing amounts of greenhouse gases (GHGs, e.g., carbon dioxide [CO_2] as well as methane [CH_4] and nitrous oxide [N_2O]) as end products (Janzen 2004).

The distribution of C within the land-atmosphere–ocean system for the past 10,000 years (Holocene), prior to the Anthropocene, had remained relatively stable. Atmospheric CO_2 concentrations, determined from ice-cores, ranged between 260 and 280 ppmv (Barnola et al. 1987; Indermühle et al. 1999) and CH_4 concentrations at 650–700 ppbv (Chappellaz et al. 1990). Between 1850 and 1950, the average annual C flux from fossil fuel combustion was 0.6 Pg C year^{-1} (1 Pg = 10^{15} g; Stuiver 1978). This flux had increased to 5.4 Pg C year^{-1} by 1990 (Tans et al. 1990; Dixon et al. 1994); the average emission of CO_2 from fossil fuel combustion and cement manufacture for the 1990s was estimated at 6.3–6.5 Pg C year^{-1} (IGBP 1998; Schimel et al. 2001; Houghton 2003; Janzen 2004; Post et al. 2004). Since the turn of the century, rapid growth in these CO_2 emissions has continued, reaching 9.1 ± 0.5 Pg C year^{-1} by 2010 (Peters et al. 2012). Changes in human land management strategies have also caused a net release of C to the atmosphere (Woodwell 1978; Houghton et al. 1983). Until *ca*. 1960 this was higher than the C emitted by fossil fuel combustion (Houghton et al. 1983; Malhi et al. 2002). By 1980 land-use changes emitted between 0.4 and 4.7 Pg C year^{-1} (Buringh 1984; Schlesinger 1984;

Tans et al. 1990) and average annual emissions in the 1990s were between 1.7 and 2.2 Pg C year^{-1} (Houghton 2003; Janzen 2004). Net deforestation alone has been estimated to release 0.9–1.6 Pg C year^{-1} (IGBP 1998; Houghton 2000; Schlesinger and Andrews 2000; Raupach and Canadell 2010).

Although, the emission of CO_2 by fossil fuel combustion and land-use changes since 1750 have been partly balanced by oceanic and terrestrial sinks (about 50–60 % of anthropogenic CO_2 releases, Scholes and Noble 2001; Sabine et al. 2004; Le Quéré et al. 2009; Raupach and Canadell 2010), there was still a net increase of CO_2 to the atmosphere of 3.2–3.4 Pg C year^{-1} during the 1990s (IGBP 1998; Schimel et al. 2001; Janzen 2004) resulting in atmospheric CO_2 concentrations of 371 ppmv by 2001 (Malhi 2002; Janzen 2004; Post et al. 2004) and CH_4 concentrations of 2,000 ppbv (Fowler et al. 1995). Recent estimates indicate that up to 5 Pg C year^{-1} maybe accumulating in the atmosphere (Le Quéré et al. 2009; Friedlingstein et al. 2010). Monthly mean CO_2 concentrations, supplied by the US National Oceanic and Atmospheric Administration from the Mauna Loa Observatory in Hawaii, by April 2012 have reached 396 ppmv (National Oceanic and Atmospheric Administration 2012).

There is still much debate as to the magnitude, location and turnover of the terrestrial C sinks (and sources) (Dixon et al. 1994; Keeling et al. 1996; Houghton 2003; Aufdenkampe et al. 2011), particularly with changing climate and land use. As soil and vegetation combined (*ca*. 75 % of the terrestrial C stock occurs as organic matter [OM] in soils) contain three times as much C than in the atmosphere pool, small increases in net C losses from the terrestrial biosphere to the atmosphere could have significant impacts on atmospheric CO_2 concentrations. Hence, the response of soils, particularly organic-rich soils (e.g., peatlands), to climate and land management future scenarios is crucial when assessing climate-C cycle feedbacks (Smith et al. 2008; Evans and Warburton 2010). The more C that can be retained resiliently via reduced turnover rates within the global terrestrial sink moderates that amount exported to the atmosphere as GHG. To achieve this, requires accurate assessment of terrestrial fluxes and processes. This has led to numerous studies with regards to determining net ecosystem exchange (NEE) and net ecosystem production (NEE incorporating lateral export, Lovett et al. 2006; Aufdenkampe et al. 2011) across different soil and vegetation classes. Others studied sensitivity to decomposition of soil organic matter (SOM) pools and land use/management strategies that can reduce losses of GHG from land to the atmosphere (e.g. Cao and Woodward 1998; Cannell 2003; Janssens et al. 2003; Janzen 2004; Dawson and Smith 2007; Smith et al. 2008; Le Quéré et al. 2009; Schulze et al. 2010).

However, to balance the C budget of anthropogenic emissions, land and ocean sinks and the measured atmospheric pool, it was proposed that the losses of C via inland surface waters equated to the amount of missing C adsorbed from the atmosphere but not sequestered in the terrestrial biosphere on land (Siemens 2003). Moreover, in contrast to the NEE dynamic equilibrium between CO_2 inputs to land as gross primary production (GPP) and vegetation and soil biomass respiration losses of CO_2 (Cao and Woodward 1998), C losses to surface waters are – redeposition on floodplains aside – uni-directional and are now considered integral to improving estimates of C budgets within terrestrial ecosystems from landscape to global scales (Billett et al. 2004, 2010; Cole et al. 2007; Dawson and Smith 2007; Battin et al. 2009; Aufdenkampe et al. 2011). Key to this is that the amount of C entering oceans is only a part of the total C transported from terrestrial ecosystems to surface waters. The remainder is cycled within the riverine continuum and returned back to the atmosphere as CO_2 or buried in sediments as OM (Aufdenkampe et al. 2011).

Recent observations of increasing dissolved organic carbon (DOC) concentrations (Worrall et al. 2004; Evans et al. 2006a, b; Monteith et al. 2007; de Wit and Wright 2008) and increased erosion contributing particulate organic carbon (POC) (Evans et al. 2006a, b; Evans and Warburton 2010) within stream waters of upland (mainly organic-rich soils) ecosystems suggest that SOM stocks may be vulnerable and will potentially contribute to positive climate forcing. However, this assumes that the soil-derived OM entering surface waters is converted to CO_2 with subsequent evasion from the water column to the atmosphere (Hope et al. 2001; Billett and Garnett 2010). Conversely, non-respired OM transported from the land to oceans, with burial and incorporation in marine sediments would mean that losses of SOM would not contribute to CO_2 concentrations in the atmosphere. Therefore, understanding processes that control soil C losses to and its fate in surface waters is not only important in establishing accuracy of terrestrial and aquatic C fluxes, feedbacks and tradeoffs but also providing evidence for local, national and global land management and policy strategies that limit terrestrial ecosystem C contributions to atmospheric CO_2.

9.2 The Soil-Surface Water-Atmosphere Pathway

9.2.1 The Role of Surface Waters in the Carbon Cycle

Within the global C cycle, inland surface waters (streams, lakes, rivers and transitional waters) act as a conduit for the transport of C from the land to the ocean (Fig. 9.1). The total global riverine organic C flux has been estimated at *ca*. 0.40 Pg C $year^{-1}$ (Degens et al. 1991; Schlunz and Schneider 2000), of which particulate transport constitutes between 0.07 and 0.20 Pg C $year^{-1}$ (Ittekkot and Laane 1991; Hedges et al. 1997).

Global inorganic C fluxes from rivers are of an equal magnitude to organic fluxes and have been estimated at between 0.26 and 0.53 Pg C $year^{-1}$ (Meybeck 1993; Hope et al. 1994; Smith et al. 2008). Although these river fluxes are two orders of magnitude smaller than annual gross C fluxes between the atmosphere and land (*ca*. 120 Pg C $year^{-1}$) and atmosphere and oceanic (*ca*. 90 Pg C $year^{-1}$) pools of the global cycle (Janzen 2004; Smith et al. 2008), the global riverine flux of organic C to the oceans is comparable to the annual C sequestration in soil (0.4 Pg C $year^{-1}$, Schlesinger 1990; Hope et al. 1994; Roulet and Moore 2006), suggesting that terrestrially-derived aquatic losses of organic C may contribute to regulating changes in SOC storage (Hope et al. 1997; Cole and Caraco 2001; Billett et al. 2006).

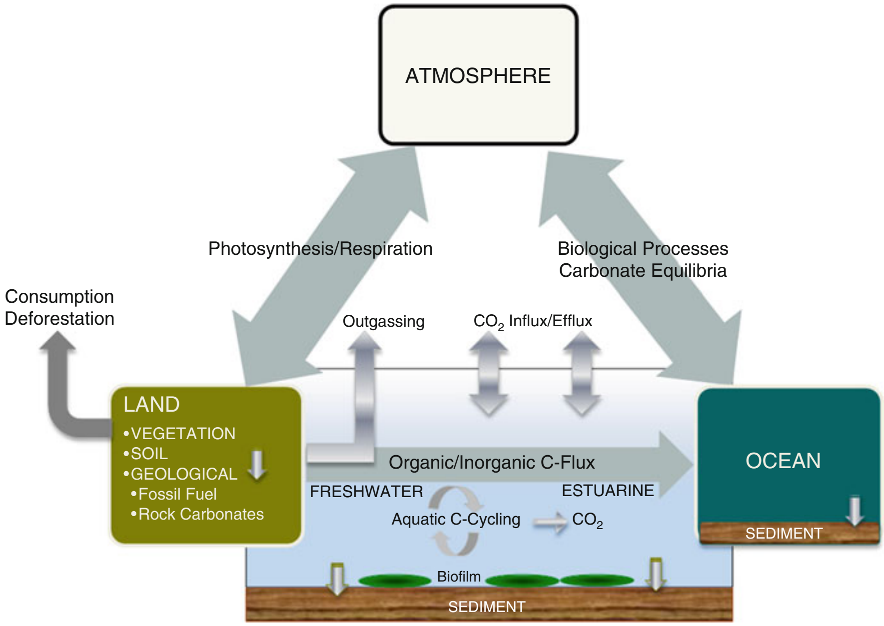

Fig. 9.1 Simplified global carbon cycle highlighting the uni-directional flux and aquatic cycling of carbon from the land to either atmosphere or ocean pools. See text for discussion of fluxes

In surface waters, C is transported in particulate (non-solubilised), dissolved and gaseous forms (Dawson et al. 2002). OM covers a continuous size spectrum of compounds ranging from free monomers such as amino acids, carbohydrates and fatty acids, fulvic and humic acids, to macromolecules such as proteins and colloids, to aggregates and large particles including bacteria (Thurman 1985). POC and particulate inorganic carbon (PIC) are retained on a filter with pore sizes mainly between 0.45 and 0.7 μm. The upper limit is operationally defined by the sampling device used and is usually about a few mm (Ittekkot and Laane 1991). DOC is associated with the filtrate that passes through the filter paper (Thurman 1985; Hope et al. 1997a). Dissolved inorganic carbon (DIC) is also contained in the filtrate and in surface waters comprises H_2CO_3, HCO_3^- and CO_3^{2-} . These anions are associated with gaseous free CO_2 via the carbonate equilibria. Their relative proportions dependent on pH and, to a lesser extent, on temperature (Stumm and Morgan 1981; Dawson et al. 1995). Although CH_4 concentrations in surface waters are usually much lower than CO_2, it is another important C-containing GHG associated with areas of anoxia within catchments (De Angelis and Scranton 1993; Jones and Mulholland 1998).

Continual C inputs to surface waters occur via tributaries, bank seepage and the hyporheic zone as headwaters – streams – rivers progress to estuaries, coastal environments and the deep ocean. However, surface waters are not inert channels but key hotspots for active transformation and recycling of C. Aquatic C has ecological significance as an energy source for both autotrophic and heterotrophic biota (Cole et al. 2007; Battin et al. 2008, 2009; Aufdenkampe et al. 2011). Inorganic and organic C, as a constituent of dissolved organic matter (DOM), influences the pH and ionic balance within surface waters (i.e., via organic acids and the carbonate equilibria). Both DOM and particulate organic matter (POM) are also involved in complexation mechanisms that affect solubility, transport and availability of ions, nutrients, heavy metals and organic pollutants (Hope et al. 1994; Dawson et al. 2002). Moreover, the quantity and quality of DOM can have economic consequences associated with treatment costs to remove colour for potable waters (Worrall et al. 2004; Dawson et al. 2009a).

9.2.2 Connectivity Between Soils and Surface Waters

In order to determine C losses to surface waters, most studies utilise the catchment as a functional unit. The catchment concept provides a natural environment that "links the atmosphere, plants, soils, groundwater, and streams through the convergence and interaction of material and energy flows" in an integrated manner (Lohse et al. 2009). Catchments provide a platform for maximising research within and between an infinite variety of different terrestrial systems (e.g., by area, characteristics or management), where hypotheses can be quantitatively assessed (Dawson and Smith 2010).

The climate, geology and land use/management of catchments and their interaction mainly determine soil type and vegetative communities and hence the terrestrial C stock. Topography, temperature, precipitation, hydrology and nutrients affect vegetative and soil-forming processes, such as primary production, respiration and biologically-mediated decomposition (Hope et al. 1994; Yoo et al. 2006), influencing the amount of potentially exportable C from the terrestrial pool. Decomposition and mineralization of litter and/or SOM converts primary production products to smaller organic components and inorganic forms, respectively, releasing C and nutrients for uptake by soil biota (plants, faunal and microbial communities). Factors, e.g., temperature, soil texture and moisture, and plant residue composition, that regulate OM stabilisation and decomposition rates in soils have been discussed extensively in the literature (e.g., Davidson and Janssens 2006; Von Lützow et al. 2006; Smith et al. 2008; Lohse et al. 2009; Schmidt et al. 2011). SOM decomposition differs considerably amongst terrestrial systems and results in losses of soil C as (i) mainly CO_2 and CH_4 to the atmosphere as part of NEE; (ii) particulate C and DOC by erosion processes and (iii) gaseous, dissolved and particulate C to surface waters (Dawson and Smith 2007).

Spatial and temporal (e.g., diurnal, seasonal) variations in concentrations and the type (particulate, dissolved or gaseous) of C exported in surface waters are determined by hydrological processes interacting with the biogeochemistry of the surrounding terrestrial environment (Hornberger et al. 1994; Lohse et al. 2009; Dawson et al. 2011). Within stream processes can constitute a secondary control on these inputs (Dawson et al. 2001a). Areas of SOM erosion, decomposition products within soil interstitial pore waters (e.g., temperature-related biological decomposition of available OM and solubility of DOC and gaseous components) and groundwater inputs contribute to C available for export from soils to surface waters (Dawson and Smith 2007). Connectivity of soils to the drainage network - controlled by soil distribution (Billett and Cresser 1992) and their hydrological properties (e.g., hydrology of soil types; Boorman et al. 1995) combined with hydrometeorological characteristics (e.g., antecedent conditions, rainfall-runoff ratios, mean transit times, discharge variability) are paramount in determining the extent and timing of C inputs (Brooks et al. 1999; Dawson et al. 2008, 2009b, 2011). However, the linkage between soil and surface waters is temporally dynamic and vulnerable to disturbance (Fraser et al. 2001; Billett et al. 2006).

9.2.3 Inputs of Particulate Carbon to Surface Waters

POC in surface waters originates from allochthonous (terrestrially-derived) sources such as soils and fragmented plant material (branches, cones, leaves, needles, twigs). It is also present in surface waters as microbial and algal biomass (biofilms) and from aggregation of DOM (Sollins et al. 1985; Ittekkot and Laane 1991; Hope et al. 1994). POC is less common, dependent on the nature of the underlying parent material within catchments, originating in areas containing carbonate minerals, such as limestone and dolomite (Meybeck 1982).

The main mechanism of POC inputs to surface waters is via erosion of topsoil aggregates containing SOM that are spatially close to receiving waters. Inputs of POC, as a variable component of suspended sediment (Dawson et al. 2012), tend to be episodic in nature. The majority of POC is transported via overland flow following intense rainfall events (and snowmelt), when runoff is greater than the infiltration capacity of the soil (Walling and Webb 1985, 1987; Reynolds 1986; Walling et al. 2002; Evans and Brazier 2005). In addition, sub-surface soil pipes can contribute substantially to POC content, particularly in peatlands that have been artificially drained (Warburton et al. 2004; Holden 2006). Bare soils, lacking in stability, are more liable to erosion than their equivalent soils that are naturally vegetated; e.g., under vegetation on arable land or have undergone peatland restoration. For riverbank soils, higher discharge and increased stream size enhances erosional processes, through weathering, fluid entrainment, "preparatory" bank weakening and eventual collapse of the bank (Lawler et al. 1997; Couper and Maddock 2001).

Soil erosion and production of suspended sediment is highly variable (e.g., Stott and Mount 2004; Dawson and Smith 2007). Altitude, slope and soil type have been correlated with extent of erosion. However, inadequate consideration of "best practice" protocols for land management influence losses such as tillage down slope by mechanised agriculture, burning and drainage of organic C-rich moorland, over-recreation, over-stocking and poaching by cattle are important contributors to erosional processes (McHugh et al. 2002; Dawson and Smith 2007). Temporally, the extent of erosion within a given area can also change markedly. Within the same locality, different stages of the forestry cycle comprising undisturbed peatland, ploughing and ditching, mature forests to harvesting have produced substantial changes to suspended sediment yields in adjacent surface waters (Stott and Mount 2004). The transport of eroded SOM also removes essential nutrients and minerals, which can affect the ability of the soil to support ecosystem functions (Neff and Asner 2001). Moreover, the mobilisation of soil to surface waters has other disadvantages as suspended sediment reduces light penetration, impacts on macroinvertebrate and fish communities, increases biochemical oxygen demand and contributes to diffuse pollution by controlling export of sediment-associated nutrients, pathogens and contaminants, such as trace metals and pesticides (Kronvang et al. 1997; Lawler et al. 2006; Collins and Anthony 2008).

9.2.4 Inputs of Dissolved Carbon to Surface Waters

DOC in surface waters originates from vegetation (leaching, rhizodeposition) and soil (leaching, OM decomposition and erosion) (Hope et al. 1994; Meyer et al. 1998). In-stream, processing of POM contributes to DOC concentrations (Dawson et al. 2004, 2012). DIC is derived from dissolution of carbonates and weathering of silicates in soils and bedrock. Speciation of DIC is also dependent on the carbonate equilibria (including its interaction with CO_2 dynamics) within ground and soil water pools (Hope et al. 2004). Soil interstitial pore water is modified as it moves preferentially along flow paths through organic and mineral soil horizons to ground or surface waters (Grieve 1990; Hinton et al. 1998; Hagedorn et al. 2000). In upland peatland streams, isotopic studies have suggested that soil-derived DOC was commonly of recent (decadal) origin. However, ground water DOC was much older, *ca*. 8,500 years and associated with underlying mineral layers (Schiff et al. 1997; Palmer et al. 2001; Tipping et al. 2010). In larger rivers, a terrestrially-derived combination of young and old DOC and mainly old POC (>1,000 year) was noted. It has been suggested that the younger, more labile DOC has undergone preferential mineralization resulting in a residual older OM component (Raymond and Bauer 2001).

Many studies in temperate and boreal catchments, but not all (e.g., Tao 1998; Dawson et al. 2008), indicate positive relationships between surface water concentrations of DOC (or POC) and hydrological discharge (Dawson and Smith 2007). However, seasonality and catchment-specific hydrological patterns (e.g., antecedent moisture levels, hysteresis related to rising and falling events on the hydrograph and OM source exhaustion following consecutive intense rainfall events) can affect DOC and POC concentrations for a particular discharge, increasing variation of individual concentration-discharge relationships (Walling 1977). Seasonal splitting of data can sometimes improve this relationship for DOC (Dawson et al. 2002, 2008, 2011). Surface water concentrations of DIC tend to show an inverse relationship with discharge as the contributions of ground water are reduced and the influence of sub-surface water derived from the upper organic soil horizons that contain less DIC increases at higher discharge (Hill and Neal 1997; Neal et al. 1997; Dawson et al. 2002, 2004). Understanding these relationships between discharge and concentrations is important for improving assessments of the total C lost from soils to surface waters.

9.2.5 Inputs of Gaseous Carbon to Surface Waters

Gaseous CO_2 in surface waters is derived from the terrestrial environment, in-stream mineralization of OM and by exchange at the atmosphere/water interface (Dawson et al. 2009b). Microbial communities also convert OM to CH_4 but only in areas of anoxia such as those in peat soils, riparian zones and bed sediments (Dahm et al. 1991; Jones and Mulholland 1998; Hope et al. 2004). In soils, gaseous components dissolve in soil water and are either trapped within soil pores, lost by diffusion to the soil surface, or hydrologically transported to ground or surface waters. The soil porosity and moisture levels control the proportion of gaseous C that is exported to surface waters (Skiba and Cresser 1991; Dawson et al. 2001a; Hope et al. 2004). In well-aerated, freely draining mineral soils, the rate of gaseous C efflux as CO_2 to the atmosphere from the soil surface is dominant, and gaseous losses to the surface waters are a minor component. However, in typically saturated organic C-rich soils, gaseous C effluxes to the surface are reduced as CO_2 and CH_4 dissolves in soil pore waters more readily from where it is then transported laterally to surface waters increasing their losses via hydrological pathways (Clymo and Pearce 1995; Fowler et al. 1995; Billett et al. 2004).

9.2.6 Spatial Variability of Carbon Inputs to Surface Waters

As streams increase in size to rivers, C inputs from the surrounding terrestrial ecosystem become less significant as autochthonous (derived from up-stream contiguous inputs and in-stream processes) C content becomes more dominant. Anthropogenic inputs are also important in areas with a higher population density and pollution (Hope et al. 1994).

Studies indicate strong correlations between C in surface waters and soil type, e.g., % area of peat (Hope et al. 1997; Dawson et al. 2011) or the soil C pool and SOM C:N ratios (Aitkenhead et al. 1999; Billett et al. 2006; Aitkenhead-Peterson et al. 2007).

However, these often occur in smaller scale (<5 km^2) catchments where soil-stream linkage is strongest (Aitkenhead et al. 1999; Dawson et al. 2001a; Hope et al. 2004; Billett et al. 2006). Figure 9.2 highlights an example of how changes in stream water DOC and CO_2 concentrations link to spatial changes in the soil C pool along a headwater stream. Initially, at the source, shallow, immature soils (Leptosols) with little C content only supply small amounts of DOC or CO_2 to the adjacent

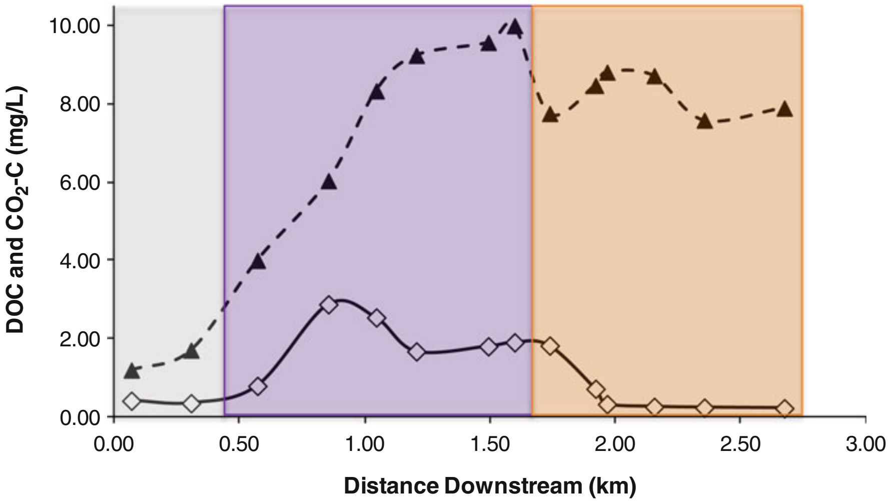

Fig. 9.2 Downstream spatial variability in DOC (▲) and CO_2-C (◇) concentrations along the main stem of a head water peatland stream, indicating connectivity with soil type (■ = Leptosol ■ = Histosol (Peat) ■ = Histic Podzol) (Data obtained on 14/07/97, Adapted from Dawson et al. (2001a) and Billett et al. (2006)) (Color figure online)

drainage network. As the stream flows through deep peat (>5 m in depth, Billett et al. 2006), increased DOC and CO_2 are supplied directly to the drainage channel. These inputs then decrease in the lower part of the catchment as the stream flows through more freely drained organo-mineral soils. At this stage, the relationship between organic C in the soil and stream becomes weaker as complex interactions occur, e.g., continual inputs of detrital DOC from upstream, lower DOC and CO_2 inputs from freely-drained minerals soils, continual outgassing of CO_2 to the atmosphere and in-stream processing of DOC (Dawson et al. 2001a; Billett et al. 2006). This inter-relationship between allochthonous and autochthonous inputs and processes, affecting both organic and inorganic C concentrations and fluxes in surface waters, occurs continuously downstream to oceans with an increasing influence of autochthonous inputs as terrestrial connectivity with the surface water is often reduced in larger river systems (Minshall et al. 1985; Meyer and Edwards 1990; McTammany et al. 2003; Dawson et al. 2004, 2009b, 2011; Webster 2007).

9.2.7 In-Stream Processes Affecting Carbon Concentrations and Speciation

Many processes modify both organic and inorganic C concentrations within surface waters prior to entering the oceanic sinks (Fig. 9.3). It has been estimated that of the total amount of organic C entering global rivers, 50 % is transported to oceans, 25 % is

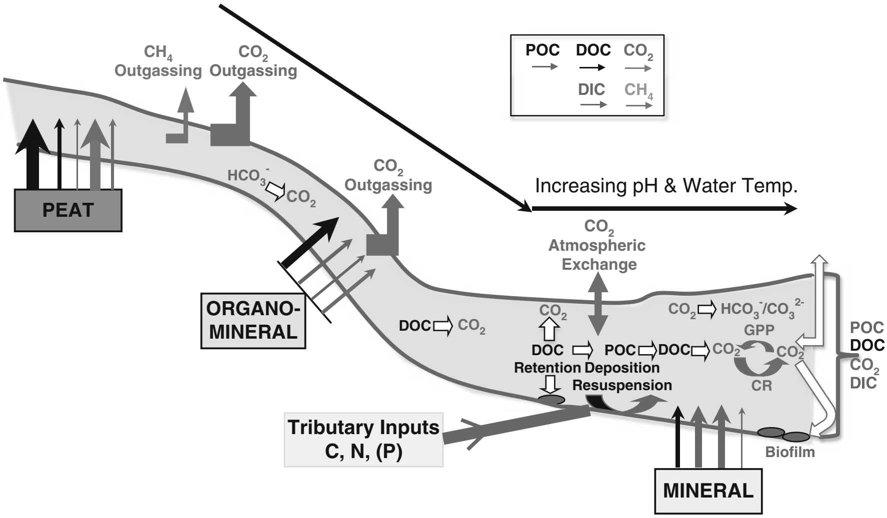

Fig. 9.3 Simplified diagram highlighting typical physical and biogeochemical processes affecting terrestrial inputs and in-stream carbon dynamics along an upland to lowland riverine continuum. The terrestrial *input arrows* give an indication of the relative magnitude of contribution for each carbon determinant; *GPP* Gross Primary Production, *CR* Community Respiration. Outputs of particulate organic carbon (POC), dissolved organic carbon (DOC), dissolved inorganic carbon (DIC) and gaseous components (e.g. CO_2) to estuarine/ocean (and atmospheric) pools are dependent on the overall spatio-temporal dynamics within individual catchments

oxidised and 25 % stored as sedimentary OM within impoundments, lakes, floodplains, and other wetlands (Hope et al. 1994). Recently, it has been suggested that *ca*. 33 % of global terrestrial C inputs to surface waters reach the oceanic sink and a minimum of 44 % enters the atmosphere as gaseous C at a rate of 1.2 Pg year^{-1} (range of 0.75–1.4 Pg year^{-1}) with the remaining 22 % retained in sediments (Battin et al. 2009; Aufdenkampe et al. 2011). However, these rough estimates have an inherent uncertainty due to the substantial spatio-temporal variability within and between ecosystems.

In organic C-rich uplands, POC is largely derived from low density eroded peat. Once this material enters a stream channel it is largely transported as washload which can be transported at very low discharge and is flushed downstream, only deposited once the flow weakens. Larger blocks of peat may be temporarily stored on the channel bed but these are soon broken down by in-stream microbial processes and rapid abrasion (Evans and Warburton 2001). OM deposited on the channel bed is re-suspended when discharge rises above a certain threshold. Sediment OM is liable to re-suspension (as POC) in an alternating sequence of deposition (with potential subsequent fates of burial or further decomposition) and re-suspension (Thurman 1985; Cushing et al. 1993; Evans and Warburton 2001, 2005) along the riverine continuum.

Whole stream metabolism of OM, related to the amount and type of biological activity within the stream ecosystem, exerts a significant control on C concentrations and is important for understanding differential speciation and fate of C within surface waters. Rates of in-stream GPP, community respiration (benthic and water column) and net ecosystem production rates are known to vary diurnally, seasonally and between different aquatic environments (Battin et al. 2008; Staehr et al. 2012). Increased input of nutrients in bioavailable forms to stream ecosystems may also stimulate primary productivity, increasing autotrophic organic biomass production such as phytoplankton and macrophytes, with concomitant increases in community respiration (Peterson et al. 1985; Neal et al. 2006; Stutter et al. 2008; Dawson et al. 2012). Physical aspects of the stream also play a role in determining OM processing (Fellows et al. 2001). This includes hydraulic parameters, such as water transient storage, hyporheic flow paths, pooling and flow characteristics, which can affect biotic establishment and stability, thus, influencing in-stream heterotrophic and autotrophic processes (Battin et al. 2008; Dawson et al. 2009b).

In detrital-based upland stream ecosystems and along organic C-rich rivers, heterotrophic decomposition of DOM often dominates (respiration > photosynthesis) acting as a source of CO_2 (Meyer and Edwards 1990; Marzolf et al. 1994; Mulholland et al. 1997; Dawson et al. 2001b, 2004). This is due to large inputs of respiratory substrate (OM) from either upstream or adjacent terrestrial sources that constitute a major energy source for aquatic communities. Biota that utilise OM are likely to be acclimatised to specific DOC sources found within particular ecosystems (Kuserk et al. 1984). POM deposited as sediments can be broken down physically by macroinvertebrates and microbes to finer particles. Collectors and filter feeders (e.g., *Diptera*, *Simulidae*), which are known to process large amount of POC, break down these finer particles contributing DOC to the water column (Vannote et al. 1980; Meyer and Tate 1983; Malmqvist et al. 2001; Monaghan et al. 2001). Strong relationships between filter feeders and bioavailable-P within POM confirm the importance and reactivity of this substrate (Stutter et al. 2007). DOC inputs can be removed from the water column either by biotic or abiotic processes (Hope et al. 1994). Adsorption of DOC as well as via aggregation to form POC is usually controlled by its physical chemistry at the solid/liquid interface. OM can be adsorbed to inorganic particulates, undergo deposition and be retained on the stream bed by biofilm communities that act as major transformers of DOC in headwater streams (McDowell 1985; Fiebig and Lock 1991; Staehr et al. 2012). Dissolved humic substances (a component of the DOC pool), although often considered relatively recalcitrant, are also an important biodegradable source of C (Volk et al. 1997). In addition, there is evidence that photolytic degradation of DOM to more readily assimilated compounds occurs stimulating ecosystem productivity under suitable conditions, e.g., in lakes (Lindell et al. 1995; Granéli et al. 1996; De Lange et al. 2003). Particulate OM has been less studied as a source of C to the atmosphere compared to DOM as it is transported towards sedimentary pools in the estuarine and oceanic environments where it constitutes a major substrate for heterotrophic metabolism (Ittekkot and Larne 1991; Tappin et al. 2003; Cole et al. 2007). Consequently, "far less is known about potential transformations of POM, which govern POC fate and interaction within surface waters" (Dawson et al. 2012). The major OM microbial transformation processes associated with in-stream POM require initial colonisation on particulate material, respiration (either decomposition to DOC or mineralization to CO_2) and nutrient recycling with formation of new biomass (Grossart and Ploug 2000; Zimmermann-Timm 2002; Battin et al. 2008).

Dissolved CO_2 content in surface waters is determined as a concentration (mg L^{-1}) or partial pressure (pCO_2) in the gas phase. The 'excess partial pressure of CO_2' ($epCO_2$) has been defined as 'the ratio of the pCO_2 in solution to the atmospheric pCO_2 value' (Neal 1988; Dawson et al. 2009b). Water in equilibrium with the atmosphere at current values has a partial pressure of dissolved CO_2 (pCO_2) of *ca.* 390 ppmv. A recent collation of estimates has stated that nearly all fresh waters contain CO_2 in concentrations that are supersaturated with respect to that of the atmosphere ($epCO_2$ values > 1). Measured pCO_2 values range from 1,000 to 12,000 ppmv in rivers and from 350 to 10,000 ppmv in lakes and reservoirs (Aufdenkampe et al. 2011). This supersaturated state is due to the allochthonous inputs of CO_2 from ground and soil pore waters, and because aquatic community respiration rates from OM mineralization exceed photosynthetic uptake (Cole and Caraco 2001; Dawson et al. 2001b, 2009b; Jones et al. 2003; Griffiths et al. 2007; Billett and Moore 2008; Teodoru et al. 2009).

The loss of supersaturated CO_2 in streams as it equilibrates with the atmospheric CO_2 concentration is termed the "outgassing effect" (Skiba and Cresser 1991). Like CO_2, CH_4 can also be supersaturated in streams (Jones and Mulholland 1998), where it undergoes oxidation to CO_2 and outgassing once it enters surface waters (Hope et al. 2001). Historically, losses of CO_2 via this soil-surface water-atmosphere pathway have tended to be unaccounted for in most inland surface water C flux estimations and ecosystem C budgets but it is now considered an important indirect transfer mechanism to the atmosphere from the terrestrial C pool (Kling et al. 1991; Cole et al. 1994; Cole and Caraco 2001; Richey et al. 2002). Considerable differences exist between studies that have investigated evasion rates of CO_2 and CH_4 from streams, rivers, lakes and wetland ponds due to spatio-temporal differences in catchment soils and their moisture levels affecting mineralization rates and hydrological flow paths e.g., lateral transport, as well as in-stream processes (Dawson and Smith 2007). Figure 9.2 illustrates how CO_2 concentrations in a headwater stream can vary spatially within the same network (and hence outgassing flux) along short distances in relation to changes in source areas (e.g., peat). Once the elevated levels of CO_2 have been lost from the stream by outgassing, further inputs are reduced as the stream enters areas adjacent to more freely drained peaty podzols. Often, significant outgassing 'hotspots' occur along a riverine continuum, such as organic C-rich soil pools, where soil C losses to the atmosphere are reduced and the soil-stream linkage is strong (Dawson et al. 2001a, 2004; Hope et al. 2001). In-stream physicochemical factors controlling CO_2 losses to the atmosphere include temperature-dependent solubility, volume, depth, velocity, gradient, turbulence and wind speed, as well as the initial concentration of CO_2 in the stream water (Neal 1988; Rebsdorf et al. 1991; Hope et al. 2001; Dawson et al. 2001a, 2009b).

Hope et al. (2001) showed that gaseous C evasion fluxes from a peatland stream to the atmosphere were similar to fluxes of the total C (particulate, dissolved and gaseous) transported within the stream itself. Other studies have indicated its importance as part of an overall total C balance (sink/source status) in landscapes (Billet et al. 2004; Dinsmore et al. 2009, 2010). Moreover, processes influencing CO_2 in surface waters, along with other chemical inputs, alter pH that determine speciation/calcite precipitation of the ground water and terrestrially-derived DIC (i.e. HCO_3^-/CO_3^{2-}) via the carbonate equilibria (Stumm and Morgan 1981). Isotopic studies of outgassed CO_2 from carbonate free waters suggest that a "small, rapidly cycling pool (2–5 year) of organic C is responsible for the large C fluxes from land to water to atmosphere". However, some systems show greater contributions from ground water sources which can increase the mean CO_2 age to several decades (Mayorga et al. 2005).

Even in highly heterotrophic streams and rivers, autotrophic processes continue, but their dominance is restricted to occasional periods of the day or year when conditions for photosynthetic processes are optimised. During these periods when *ep*CO_2 is <1, influx of CO_2 from the atmosphere can occur (Rosenfeld and Roff 1991; Dawson et al. 2001b, 2009b; Battin et al. 2008). Further downstream from organic C-rich headwaters, heterotrophic communities can potentially utilise C derived from "upstream processing inefficiencies" and autotrophic communities utilise CO_2 derived from continual ground and soil pore-water inputs as well as in-stream OM mineralization and solubility dependent gas exchange with the atmosphere (Vannote et al. 1980; Griffiths et al. 2007; Dawson et al. 2009b). There is a direct correlation between in-stream primary production and stream order (Minshall et al. 1985). The deposition of C increases as streams and rivers meander and hydraulic energy is dispersed to levels suitable for increased sedimentation. These physical changes that lead to increases in sedimentation, encourages colonisation and establishment of benthic autotrophic communities that sequester CO_2 from the water column. Along with the high input of allochthonous respiratory C substrate in organic C-rich upland systems, this produces a U-shaped curve of energy between contiguous headwaters, tributaries and main rivers ecosystems (Fiebig and Lock 1991; Webster 2007; Dawson et al. 2009b).

However, the dynamic equilibrium between efflux and sequestration of CO_2 within surface waters is spatially dependent as the dynamics of physical (outgassing/gas exchange) and biogeochemical processes vary (Dawson et al. 2009b). Moreover, different processes also dominate depending on the time-scale of the analysis (Hanson et al. 2006). For example, (i) atmospheric deposition, climate and land use changes influence DOC dynamics in soil and stream waters at different spatio-temporal scales which make it difficult to discern the overriding control on widespread long-term trends of DOC in surface water (Clark et al. 2010) and, (ii) photosynthetic consumption of CO_2 is light dependent but in-stream community respiration and CO_2 terrestrial inputs continue throughout periods of darkness and diurnal variations in concentrations of CO_2 concentrations can be significant (Guasch et al. 1998; Dawson et al. 2001b; Neal et al. 2004; Griffiths et al. 2007). Maximum diurnal amplitude of CO_2 concentrations tend to occur during summer when an increased prevalence of lower discharges and higher temperatures encourages biotic activity (Dawson et al. 2001b; Griffiths et al. 2007; Battin et al. 2008).

9.2.8 Biogeochemical Quality of Organic Matter

In the past, the majority of studies on organic C distribution within surface waters have concentrated on quantifying concentrations or fluxes and discerning processes that influence the main organic components, i.e., DOC and POC, and production of GHG. However, the biogeochemical composition and degradability of OM will affect its transformation dynamics and hence distribution within surface waters. This has implications for selective processing of OM and hence its fate within the aquatic environment (McCallister et al. 2006; Dawson et al. 2012).

Moreover, as OM mineralization and decomposition rates can depend on the physico-chemical characteristics of the OM itself, soil management and land use practices that affect spatio-temporal variability of the quantity and type of OM entering watercourses will influence whether terrestrially-derived C eventually resides in atmospheric or oceanic pools. Therefore, there is a need to understand SOM quality inputs and in-stream processes that control the proportions and kinetics of aquatic (POC and/or DOC) and benthic sediment OM transformations to GHG across a diverse range of environmental scenarios. Furthermore, nutrient enrichment from e.g., agricultural diffuse such as N and P pollutants (Stutter et al. 2008), influences the “biogeochemical environment” of aquatic biotic activity (both photosynthetic and respiratory processes) changing nutrient and C flows and ecological stoichiometry within food webs (Cross et al. 2005; Penton and Newman 2007; Manzoni and Poporato 2011).

Different functional forms of aquatic OM influence biological and physico-chemical processes, such as its biological decomposition potential e.g., labile to recalcitrant compounds; ecosystem energy, hydrophobicity, pH buffering and photochemical fading (Baker et al. 2008). The degradability of OM may also be limited by the availability of other elements such as N and P (Taylor and Townsend 2010). Stoichiometric ratios (C:N:P) of OM are generally a simple yet robust indicator of OM quality (Elser et al. 2000; Cross et al. 2003).

In order to monitor and assess the extreme spatio-temporal variability of OM quality, standardised biogeochemical techniques and assays are required that can be performed relatively easily. These assays should also relate to functional properties of OM that impact on their fate within the aquatic environment. In terms of DOC, The relationship between ultra-violet (UV) absorbance and DOC concentrations has been used to infer changes in the proportion of hydrophobic (aromatic, recalcitrant) C and hence potential biodegradability of DOC (Marschner and Kalbitz 2003; Chen et al. 2002; Weisharr et al. 2003; Dawson et al. 2009a). Ågren et al. (2008) have suggested that the bioavailability of the DOC maybe also be related to molecular weight (indicted by Abs_{254}/Abs_{365} ratios), encompassing both aliphatic and aromatic components. More complex characterisation of DOC can be achieved by fluorescence spectroscopy, which include determination of humic-like, fulvic-like, tryptophan or tyrosine (protein) like groups, chlorophyll as well as linking to functional characteristics of DOC (Baker and Spencer 2004; Baker et al. 2008). This technique has also been utilised to assess biodegradability of stream water DOC (Fellman et al.

2009). Furthermore, the use of Fourier Transform Infrared (FT-IR) spectroscopy has been utilised to assess chemical functional groups of humic substances (Lumsdon and Fraser 2005).

The inherent biodegradability of OM derived from surface waters and benthic sediment can be assessed by an integrated measurement of its respiration kinetics. Benthic sediment samples can be collected relatively easily and measured under defined conditions. However, respiration rates have also been measured from suspended sediments collected on filter papers using the MicroResp™ technique (Dawson et al. 2012). The principal of the MicroResp™ system is that a colour change forms in an indicator gel during incubation due to the CO_2 evolved from microbial-induced substrate decomposition (Campbell et al. 2003). Other measurements of "biogeochemical reactivity" of OM include stoichiometric (C, N, P) assessments of both DOM and POM as well as chlorophyll-α concentrations (Sobczak et al. 2002; Neal et al. 2006) that assess autotrophic OM contributions (autochthonous apportionment, Dawson et al. 2012). The amount of respired CO_2 produced as a proportion of the total C associated with POM on the suspended sediment and POM autotrophic activity could be related to contributory land use pressures as well as the biogeochemical water environment (Dawson et al. 2012).

9.3 Conclusion

The understanding and quantification of in-stream processes, including in-stream cycling of C derived from non-terrestrial sources, provides increased certainty of the proportions of terrestrially-derived aquatic OM transformed to GHG and that is eventually exported from streams, lakes and rivers to estuarine and marine environments.

The relationships between inland surface waters and C cycling can be dependent on the integration of many different processes that requires understanding of both lateral (soil to water) and longitudinal (along the riverine continuum) transport of C. This also requires an integrated understanding and assessment that incorporates particulate, dissolved and gaseous C species. These processes need to be explored to evaluate the ultimate fate of terrestrially-derived C in surface waters. Many surface waters are dominated by in-stream biological systems which may ultimately control the decomposition of complex natural OM. This is where an integrative quantitative assessment of these processes that link the compositional 'quality' of OM forms to biological accessibility for respiration under 'realistic' scenarios is required. There are, however, interconnected physical processes of stream morphology, deposition and re-suspension of particulates, photo-oxidation of DOM and outgassing of gaseous C. Hydrometeorological metrics affect connectivity of soil sources controlling the amount and nature of the material initially reaching the surface waters and the stability and hence processing capacity of in-stream biological communities. Moreover, climate change as well as progressive anthropogenic influences along the river continuum, such as bank disturbance, nutrient/pollutant inputs and water abstraction also influences OM degradation rates both in the water column and benthic sediments.

It must be noted that OM losses to inland surface waters are an important component of its natural ecosystem functioning. In-stream C provides energy for processing of other terrestrially-derived materials, e.g., processing and reduction of nitrate relies on metabolically available DOC. However, perturbation of biogeochemical cycles has meant that although C performs this important role, consequently being respired and lost to the atmosphere, it is appropriate to devise strategies that also reduce excessive point/diffuse pollutant terrestrial inputs to surface waters, resulting in a more balanced equilibrium between cycles.

Therefore, to reduce GHG production via the soil-stream-atmosphere pathway requires reduction in the first stage of this process, i.e., unnecessary losses of C (and nutrients) to waters, via soil erosion and hydrologically-mediated transport of SOM decomposition products. Strategies increasing SOM stabilization and reducing the rate of decomposition and mineralization to DOC and CO_2, respectively, are instrumental to this hydrological export. Therefore, land use and best management practices that mitigate against C losses from terrestrial environments (both directly to the atmosphere and via surface waters) are important as to which strategies have the ability to store the most C.

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Chapter 10
Why Pests and Disease Regulation Should Concern Mankind

Willis A. Oluoch-Kosura, Anne W. Muriuki, Florence M. Olubayo, and Dora Kilalo

Contents

Abstract Nature, through balancing mechanisms, provides ecosystem services, comprising provisioning, regulating, cultural and supporting services for the survival of mankind (MEA, Ecosystems and human well-being: biodiversity synthesis. World Resources Institute, Washington, DC, 2005). The balance and stability is usually upset by interventions or development activities, thereby threatening survival. Maintaining the balance guarantees sustainable development. Pests and disease regulation provides one component of managing the ecosystem. This chapter highlights why pest and disease regulation is important in contributing to sustainable agricultural production and development. Continued multidisciplinary research efforts are needed to enable understanding of the biological interactions between pests, beneficial and host communities on one hand and the interactions between and among cost-effective pest management methods, soil health, climate change, food security and human well-

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being on the other hand. The holistic approach in developing the understanding of the role of pest and disease regulation in the ecosystem will ultimately contribute to the development of appropriate strategies for the achievement of human well-being

Keywords Ecosystem services • Sustainable production • Crop protection • Biodiversity • Climate change

Acronyms

C	Carbon
CO_2	Carbon dioxide
CH_4	Methane
GHG	Green house gases
MEA	Millennium Ecosystem Assessment
K	Potassium
N	Nitrogen
N_2O	Nitrous Oxide
P	Phosphorus
SO	Soil organic matter
EGSAA	Environmental Guidelines for Small-Scale Activities in Africa
FAO	Food and Agriculture Organization
SADC	Southern Africa Development Community

10.1 Introduction

Mankind depends on ecosystem services, classified as provisioning, regulating, and cultural and supporting for survival (MEA 2005). The provisioning services are derived from products of the ecosystem, occurring naturally or produced through agricultural activities. They include food, wood fuel, fiber, biochemicals and genetic resources, amongst others. Regulating services are benefits from ecosystem processes and activities of climate, water, pests and diseases. Cultural services include non-material benefits from the ecosystem such as spirituality and religion, recreation, ecotourism, aesthetics, inspiration, education, sense and place as well as cultural heritage, amongst others. Supporting services are those that are necessary for production of all other services. They include soil formation, nutrient cycling and primary production, involving agricultural activities, amongst others. These Ecosystem Services are listed in Table 10.1.

Nature provides ecosystem services through balancing mechanisms. However, development activities and interventions threaten the ecosystem's stability and create imbalance, thereby making provision of the services uncertain or impossible and putting populations at risk. Therefore, it is necessary to maintain the balance of nature for sustainable development. This calls for "working with nature" and not

Table 10.1 Ecosystem services

Provisioning	Regulating	Cultural
Products from ecosystems	Benefits from regulation of ecosystem processes	Non-material benefits from ecosystems
Food	Pest and diseases	Spiritual and religion
Wood fuel	Climate	Recreation and eco tourism
Fibre	Water	Aesthetics
Biochemicals (Medicines)	Water purification	Inspirational
Genetic resources		Educational
		Sense and place
		Cultural heritage
Supporting		
Services necessary for production of other services		
Soil formation		
Nutrient cycling		
Primary production		

Source: Millennium Ecosystem Assessment (MEA) (2005)

"against it". Continued enjoyment of the benefits requires proper management of the ecosystem. The regulation of pests and diseases provide one component of managing the ecosystem. This paper highlights why pest and disease regulation is important in contributing to sustainable agricultural production and development.

10.2 Agriculture, Environment, and Pests and Diseases

Agricultural ecosystems comprising arable land and permanent pasture are estimated to cover 38 % of the ice-free land surface of the earth (FAO 2008a). They are a major component of the biosphere, housing a large portion of the earth's biodiversity. This diversity comprises the fauna and flora that have evolved and established in a particular environment and have an important role in the functioning and sustainability of agricultural ecosystems. The fauna and flora include beneficial and harmful organisms all of which are affected by natural processes and effects resulting from human activities. Every ecosystem, consisting of living things and the non-living environment, re-balances through self-regulatory processes. Hence, the action of one organism usually affects another (Guiterrez 2001; Guiterrez et al. 2006). Pests occupy a niche in ecosystems that may have been simplified, or disrupted through domestication, inputs of nutrients and toxic substances or by human mismanagement (Guiterrez 2001; Guiterrez et al. 2006). Nevertheless, regulation of pests takes place at such low levels that it often remains unnoticed by humans. For example, modern agriculture involves removal of problematic species through pest/disease control which continuously simplifies the productive species within a particular production ecosystem. Consequently, these actions lead to a decline in biodiversity within the ecosystem.

It is estimated that agriculture currently accounts for about 12 % of total global anthropogenic greenhouse gas (GHG) emissions, i.e., 60 % nitrous oxide (N_2O)

emissions, 50 % of the methane (CH_4) and negligible carbon dioxide (CO_2) emissions because the net flux between the atmosphere and agricultural land- is approximately balanced (Smith et al. 2007). The effects of climate change on agriculture are triggered through shifts in temperature, precipitation, soil quality, pests, diseases and weeds, seasonal growth patterns and loss of biodiversity (Kiritani 2007; Jarvis et al. 2008). The greatest impact is occurring in countries which depend on agriculture as the key driver of economic development while more diversified economies are less affected. Experts, however, agree that agriculture has a set of opportunities to cope with climate change. These involve application of mitigation and/or adaptation strategies.

Mitigation strategies that involve the reduction of GHG emissions are categorized into those that reduce GHG emissions (CO_2, CH_4 and N_2O) into the atmosphere (Paustian et al. 2004; IPCC 2001, 2007a, b), secondly those that enhance removal of GHGs from the atmosphere into the soil due to its large capacity to hold carbon (IPCC 2001), and thirdly those that avoid emissions of GHGs. Mitigation through reduction of GHG emission can be achieved for CO_2 by using more efficient food in livestock production systems, and for N_2O by incorporating crops with biological nitrogen (N) fixation into cropping systems (Bouwman 2001; Clemens and Ahlgrimm 2001). Avoidance of GHG emission can be achieved through agricultural management practices that preclude the cultivation of new lands mainly grasslands and forests (DeFries et al. 2004), and use of crops and residues as fuel sources (Cannell 2003; Edmonds 2004; Sheehan et al. 2004; Eidman 2005). Studies on climate change effects on agricultural plants show that elevated CO_2 levels generally affect plants negatively by causing changes in acquisition of N followed by changes in composition of secondary compounds such as sugars and vitamins (Idso and Idso 2001). Such changes may ultimately affect the nutritional quality of foods and feeds.

Mitigation via removal of GHGs from the atmosphere can be enhanced by any agricultural practice that contributes to C sequestration. This includes practices that enhance capture and retention of atmospheric CO_2 in soils such as agroforestry (Mutuo et al. 2007), conservation agriculture (Madari et al. 2005) and organic farming (LaSalle and Hepperly 2008), and/or retarding the release of stored CO_2 to the atmosphere via erosion, respiration and burning. Any practice that is geared towards coping with the negative impacts of climate change is contributing to adaptation. These strategies include use of water-efficient irrigation systems, crop diversification, creating opportunities for diversification of rural livelihoods, enhancing the resilience of soils through conservation agricultural practices that protect soils against runoff and erosion, promote biodiversity and conserve water, integrated pest/disease management and improving farmers' access to varieties that respond to increased climatic stresses (frost/drought) and/or have greater tolerance to pests and diseases (Padgham 2009).

10.3 Role of Pests and Disease Regulation

The pests most frequently encountered in agriculture are weeds, insects, mites and disease causing pathogens. Sometimes, nematodes, birds and mammals can be serious pests. Pests cause damage to target/desired species, resulting to crop losses that are estimated to be 25–40 %; even with modern crop protection practices (Oerke et al. 1994). The damage caused by pests varies from reduced growth or yield of crops, distress and/or reduced growth of livestock, spread of crop and livestock diseases, reduced quality of produce and livestock products and contamination of harvested products with toxic compounds. Severe infestations may lead to complete loss of crops or livestock. However, in most cases, pests damage the desired species seriously, significantly reducing yields or product quality which in return has a negative impact on food security. Animal and plant pests reduce the available quantities of food, whether obtained through domestic production or imports and food access. Pests may increase food-borne zoonoses, mycotoxins in food, presence of agricultural chemicals in food and feeds (due to higher application rates), consequently raising concerns about food safety to consumers (FAO 2008b).

In Africa, a large number of insects and mites attack crops from seedling to harvest stage reducing the potential yields significantly (Abate et al. 2000). Only a limited number are of economic importance depending on the agro ecological zones, cropping system and the crop in question. Up to 100 % yield losses due to these major pests have been reported under on-station conditions. However, there is no information available on the economic significance of these pests under field conditions (Goldman 1996; Abate et al. 2000). Field pests such as the stem borers affecting maize, sorghum and millets cause crop yields losses estimated at an average of 15–40 % but vary greatly between 0 and 100 % among agro ecological zones, regions and seasons (Sheshu Reddy and Walker 1990). The African bollworm (*Helicoverpa amigera*), a polyphagous and major pest affecting several crops in Africa is reported to cause estimated yield losses of 12–53 % on beans in Ethiopia, Kenya and Tanzania (Abate et al. 2000). The cassava green mite (*Mononychellus tanajoa*) and the cassava mealybug (*Planococcus manihoti*) are reported to cause 80 and 60 % direct yield losses in many African countries where cassava is cultivated. Accompanying secondary losses include reduced healthy leaves consumed as food, soil erosion and poor quality planting materials for the following season (Ortiz and Hartman 2003). Bean stem maggots (*Ophyomia spps*) frequent in major bean growing areas of Eastern and South Africa, are reported to cause yield losses of 8–100 % in these countries (Oerke et al. 1994). On the other hand, the bean aphid (*Aphis fabae*) has been reported to cause yield losses of 90, 50 and 37 % in Kenya, Zambia and Uganda, respectively (Abate et al. 2000). The millet head miner (*Heliocheilus abipunctella*) causes estimated yield losses of 8–85 % on pearl millet, in the Sahel region which includes Senegal, Burkina Faso, Gambia, Niger and Mali, depending on the crop season while the short horn grasshoppers cause yield losses of 70–90 % in a bad year that occurs every 5 years across crops in the Sahel zone (Wohllebher et al. 1996; Abate et al. 2000). Grain loss in Africa due to insect pest damage in storage systems is estimated at 20–30 % (Pingali 2001; Mallya 1992). Worldwide, *Sitophilus zeamais* cause more than 20 % grain loss for untreated maize (Giga and Mazarura 1991). Smallholder farmers in Africa experience considerable economic losses caused by *S. zeamais* and *Prostephanus truncatus* (Gueye et al. 2008; Oduor et al. 2000; Keil 1988).

In addition to field and storage pests, migratory pests such as locusts, armyworms and Quelea birds, cause substantial crop losses across Africa during the plague years

(Abate et al. 2000). According to World Bank (1998) while citing Botha, the potential economic loss caused by migrant pests on a crop in the Southern Africa Development Community (SADC) countries is more than tenfold that of the counterparts in the USA, since the rural population density of 370 people/km^2 is tenfold that of a developed country such as the USA (34 people/km^2). Exotic pests introduced in Africa have continued to establish and have become major pests of economically important crops. For example, the leaf miner (*Liryomza trifolii*) on vegetables, *Phylocnitis citri* on citrus, the fruit fly (*Bactocera invadens*) on several fruits and the Western flower thrips (*Frankniella occidentallis*) on horticultural crops.

Since pests, diseases and weeds can move across physical and political barriers, their control poses special challenges for agriculture because late containment may affect yields adversely, thus threatening food security and creating public concern nationally, regionally and globally . The warming trend of the Earth is already altering national distribution and contributing to trans-boundary spread of pests and diseases (Kiritani 2007; FAO 2008b) therefore, the need for coping strategies. Biodiversity can play a key role because genetically diverse populations and species-rich ecosystems are more resilient to a changing environmental stresses (FAO 2008b). Selection of cultivars that are tolerant to biotic stresses (weeds, pests and diseases) can contribute to food security by ensuring sustainable production while contributing towards climate change mitigation and adaptation by reducing reliance on fossil fuel produced pesticides. Crop diversification can improve resilience by creating greater ability of agro-ecosystems to suppress pest outbreaks and reduce pathogen transmission (Lin 2011).

Healthy soils produce vigorous plants that are able to withstand biotic and abiotic stresses. Plants under abiotic stress caused by drought, lack of nutrients and soil compaction are more vulnerable towards. Attacks caused by pests and diseases. Omorusi and Ayanru (2011) showed that NPK fertilization of cassava (*Manihot esculenta*) improved the plant's vigor making the plants fit to withstand infestation by green spider mites (*Mononychellus tanajoa* Bondar), cassava mosaic disease and cercospora leaf spot disease. Thus, soil fertility improvement through buildup of soil organic matter (SOM) may benefit cropping systems under the threat of reduced productivity due to climate change in several ways. C sequestration increases soil plant /biodiversity and helps to control pests and diseases.

The link between SOM and soil fertility improvement is well documented (Vanlauwe and Giller 2006). The importance of improved soil health, attainable through increased C sequestration and its relationship to reduction in disease/pest infestation can be conceptualized as shown in Fig. 10.1 below.

SOM can be increased by adopting conservation agriculture (Madari et al. 2005), agroforestry (Mutuo et al. 2007) and organic farming (LaSalle and Hepperly 2008). SOM contributes to soil fertility improvement by maintaining the physicochemical components that entail soil fertility such as cation exchange capacity and soil aggregation. Furthermore, SOM improves the capture and retention of water, reduces soil compaction, and susceptibility to soil erosion and leaching. SOM also acts as a source and reservoir for plant nutrients including C which drives all biological soil processes that control nutrient release (Kapkiyai et al. 1999; Lekasi et al. 2001;

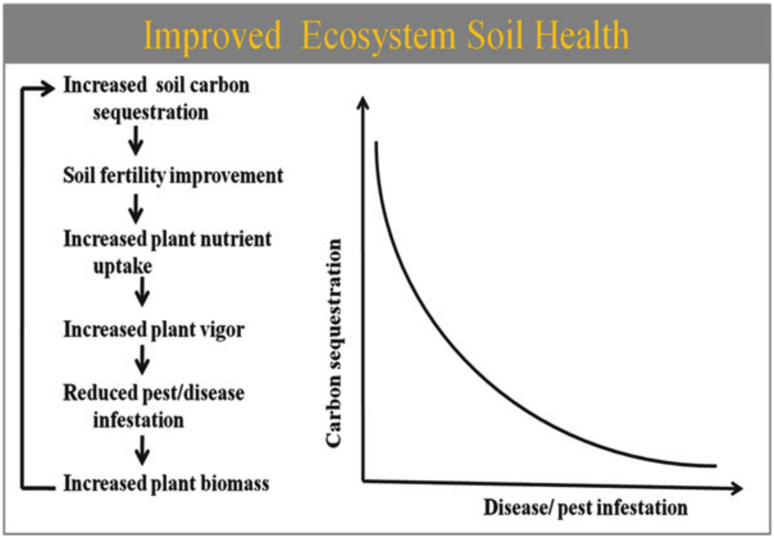

Fig. 10.1 Relationship between carbon sequestration and pest infestation

Taddesse et al. 2005; Ewulo et al. 2008). Thus, SOM impacts on ecosystem resilience by supporting greater biodiversity including rhizosphere microbial biomass such as mycorrhizae. Plant/mycorrhizal associations have been shown to improve plant acquisition of nutrients (especially P) and water, and increase tolerance to abiotic stresses such as heavy metal toxicity and soil acidification and biotic stresses including nematodes and plant diseases (Phirke et al. 2008; Finlay 2008; Richardson et al. 2009, 2011). Plant/mycorrhizal associations thrive best under favorable soil fertility status. Hence, farming methods that encourage the build-up of soil fertility and C sequestration via SOM formation is bound to feature more prominently in coming years as a coping strategy for boosting agricultural production in a changing climate.

10.4 Strategies for Promoting Sustainable Agriculture

Without some form of pest regulation, crop yield losses would be high and result in shortages of food and high food prices. There are two strategies for protecting plants against pests in ecosystems. These are natural control consisting of abiotic and biotic controls, and applied control. Natural controls are those measures taken to check pest populations without dependence on human intervention for success. They include climatic factors such as wind, sunshine, rain and temperature. Topographic features such as rivers, lakes and mountains also play a role in natural control. In addition, natural occurring enemies assist in regulating pest populations. However, when the natural methods are inadequate, humans have to intervene and apply pest management measures for control. Applied controls include physical/mechanical, cultural, biological, microbial, genetic, chemical and legal controls and integrated pest management. The objective of regulatory pest management is to prevent introduction and spread of pests through application of pest management techniques. To succeed, limiting movements of commodities and materials and treating commodities, materials and the environment is requisite (IPPC 2007b, Extension Bulletin E-2055, Michigan State University). It is essential for regulatory pest programs to differentiate natural spread of pests from artificial spread.

All farmers rely on some form of pest control practice especially mechanical, cultural, biological and genetic resistance. Cultural practices influence the survival of pests. Practices such as mowing, irrigation, mulching, aeration and fertilization are important for producing healthy crops and preventing pest build-up. Timing of planting and harvesting, crop rotation, sanitation and varied cropping systems help to reduce pest build-up in the ecosystem. In mechanical control, cultivation improves soil conditions and controls weeds and soil inhabiting pests (Vincent et al. 2003). In Kenya, biological control is gaining importance in horticulture. The flower farmers, especially rose growers have benefited from the use of biocontrol agents in managing pests in the greenhouse. Specifically, the use of *Phytoseiulus persimilis* a predatory mite identified within the country and is commercially reared by two indigenous companies, Dudutech and Real IPM (integrated pest management), has benefited the farmers. Those who have adopted the use of these predatory mites have reduced the annual cost of production ranging from $6,000–$15,000 the cost of a mite program through acaricide purchases to at least $6,000 per hectare per year. Other accompanying benefits include reduced pesticide use from 50 to 20 %, 20 % increase in yield, longer stems with an increase of about 10 cm and bigger buds (Labuschagne 2008: personal communication). Biological control agents are safe, environmentally friendly and inexpensive offering effective solutions for hardly manageable pests such as the red spider mites. When these natural methods are unsuccessful, farmers use pesticides which are effective, fast acting, broad-spectrum, less labor intensive, more energy efficient, economical and relatively safe if properly used (Larry 2002). Despite these benefits, pesticides are a health hazard for the environment and for humans. They affect the diversity of the ecosystem leading to unsustainable use of the natural resource for continued productivity. Even with the world-wide acknowledgement of the hazardous effects of chemical control, and the increasing acceptance of IPM, farmers continue to rely on pesticides as a means of reducing pests in the fields and in stores. They perceive pest control as a battle against the competitors (Kenmore et al. 1995; Banwo and Adamu 2003).

Producers/farmers will need to adopt/embrace IPM which seeks to consider the agro-ecosystem as a unit and works together with the natural factors. It seeks to understand the landscape ecology, local biodiversity and habitat management and enhance capacities of the beneficial organisms within the ecosystem (FAO 2008b). IPM seeks to breakthrough the vicious circle farmers find themselves in because of continuous and injudicious use of pesticides (van de Fliert 1997). By understanding the whole agro ecosystem, natural factors can be triggered before pesticide use is considered. IPM includes both chemical and non-chemical methods in regulating pest populations while balancing between pest management and the desire to maintain environmental quality. The key focus in IPM is diversity. Increased genetic and species diversity enhances resistance and resilience of an ecosystem to pest attack (van de Fliert 1997). Diverse composition of producers prevents specialized pests from becoming abundant and provides better conditions for natural enemy survival (Altieri 1995; EGSAA 2009). Ecosystem resilience should be enhanced whenever possible. To be successful, a pest management program must be cost-effective. It must correctly identify the pest, monitor pest population densities and extent of the problem to guide decision-making, choose appropriate pest control measures, and evaluate whether the IPM program is effective for the IPM goal and whether there are any harmful effects to the environment (Binns and Nyrop 1992; Quinsberry and Schotzko 1994). It is important to assess threshold pest infestation before applying chemical pesticides (as a last resort). In some cases, spot-spraying affected plants will be more effective rather than treating the whole field.

10.5 Conclusion

Sustainable agriculture is crucial for the survival of mankind. The farming system adopted in the process should be productive, profitable, energy conserving and environmentally friendly. Sustainability in agriculture depends on the single farmer, whether management systems are in place to improve and maintain soil quality, one of the key ecosystem supporting services. Continued multidisciplinary research efforts are vital to focus on the understanding of biological interactions between pests, beneficial and host communities and the interaction between and among cost effective pest management methods, soil health, climate change, food security and human well-being. This calls for adequate investment on innovative approaches for pest management for sustainable agriculture, building on and working with integrated pest management. It will also be desirable to invest in extension services to improve the knowledge base on best practices for the farmers, taking advantage of indigenous knowledge systems as well. At the same time, irresponsible promotion, distribution and use of harmful pesticides should be discouraged. With empowered farmers and more ecologically sound practices, agriculture is likely to become more and more sustainable offering food security necessary for an increasing world population. The holistic approach in developing the understanding of the role of pests and diseases regulation in the ecosystem, will ultimately contribute to the developing of appropriate strategies for the achievement of human well-being.

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Chapter 11
Natural Hazards Mitigation Services of Carbon-Rich Ecosystems

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Contents

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Abstract The year 2011 has been the costliest ever in terms of economic losses caused by natural disasters. This has been partly attributed to increasing natural hazard effects caused by global warming and land use changes (in particular deforestation). This chapter provides an overview of risk-relevant issues and an evaluation of the role of carbon(C)-rich ecosystems within the overall context of natural disaster risk. Hazard mitigation ecosystem services which are discussed include the regulation of global, regional and local climates (via C storage, evapo-transpiration, and albedo); the provision of structural stability to soil substrates (reducing risks of shallow landslides, and erosion during flooding); retention and transpiration of water (reducing flooding frequencies and intensities in catchments); and the buffering against solid and fluid mass impacts (landslides, rockfalls, snow avalanches, wind-driven sea waves, storm surges, and tsunamis). The information provided may serve to advance the valuation of ecosystems and support development-relevant decision-making, especially in regions prone to natural disasters. It is highlighted that heedless destruction of ecosystems will come at an increasingly higher cost to current and future generations, with progressively fewer options to maintain or reinstate their services. Hence, sensible foresight stipulates a precautionary approach when dealing with the remaining 'ecosystem capital'.

Keywords Natural disaster risk • Vulnerability • Climate change • Floods • Landslides • Wave hazards • Ecosystem valuation

Abbreviations

C	Carbon
CO_2	Carbon dioxide
CRED	Centre for Research on the Epidemiology of Disasters
ENSO	El Niño-Southern Oscillation
GHG	Greenhouse gas
GDP	Gross domestic product
SOM	Soil organic matter

11.1 Introduction: Retracing the Costs of Natural Disasters

11.1.1 Natural Disaster Losses in the Year 2011

Severe natural disasters have made headlines throughout the year 2011. The most damaging and harmful event was the Tohoku earthquake and tsunami in Japan, with 15,844 casualties, 3,394 missing, 5,893 injured, and 300,000 people displaced, and a record 210 billion US$ of damages (Munich Re 2012). This tragic event will also be remembered for triggering the worst nuclear disaster since Chernobyl, i.e., the explosion of the power plant Fukushima Daichii which caused widespread radioactive contamination within a radius of 30–50 km (MacKenzie 2011). Other exceptional disasters were massive floods in Thailand (40 billion US$ of damages, 813 casualties, 13.6 million people affected) and in other Southeast Asian countries, in Australia (32 billion US$, 35 casualties, 200,000 people affected), China, Brazil, Pakistan, in the USA and in Southern Europe. Powerful earthquakes struck in New Zealand (12 billion US$, 181 casualties) and in Turkey. Violent tornadoes affected the American Midwest, and tropical cyclones the Indo-Pacific and the Caribbean. Extensive wildfires burned in Siberia, California, Australia, and the Mediterranean (Munich Re 2012; Wikipedia 2012; Em-Dat 2012).

Several of these disasters had serious economic repercussions beyond the directly affected regions. For example, global automotive and technology supply chains were interrupted when floods brought the production in industrial estates to a standstill in Thailand, and food prices soared in Southeast Asian countries and Australia as crops failed following the Noachian rains and floods during the monsoon seasons (The Economist 2012a, b). Data presented by international insurance companies indicated that the year 2011 has indeed been the costliest ever in terms of economic losses caused by natural disasters. At a total of about 378 billion US$ the global losses were almost two thirds higher than those of the last record year, i.e., the year 2005 with losses of 262 billion US$ (Munich Re 2012; Fig. 11.1). As noted in an article by The Economist (2012a, p. 54) it appears that ‘multi-billion-dollar natural disasters’ are becoming more common as “five of the ten costliest [events], in terms of money rather than lives, were in the past four years”, and all of the ten costliest actually occurred within the last two decades.

To decrease the likelihood of future disasters and their respective costs, renewed efforts and investments in risk research and management are required. The objective of this chapter was to provide a structured review and synthesis of the various functions that ecosystems – in particular carbon(C)-rich ecosystems such as forests – may exert in influencing various natural disaster risks. A coherent framework for risk assessments and evaluations is introduced, and various issues of data management and interpretation are discussed. The hazards examined in the chapter include global and regional climatic changes and weather phenomena, catchment-based and seaborne floods, and mass movement hazards (landslides, rockfalls and avalanches). The chapter discusses hazard mitigation ecosystem functions within the context of global warming and various kinds of land use changes and impacts on ecosystems.

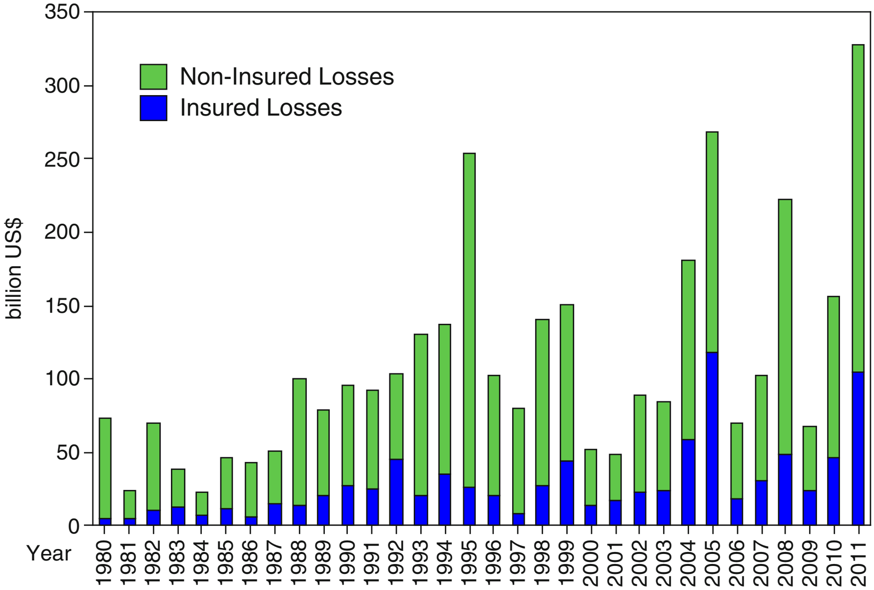

Fig. 11.1 Overall and insured losses in billion US$ (in 2011 values) due to natural catastrophes worldwide from 1980 to 2011, as reported by the NatCatService, Geo Risks Research of Munich Re Insurances (Adapted from Munich Re 2012)

11.1.2 Natural Hazard Challenges and Questions in a Carbon-Enriched Biosphere

The recent cataclysms elicit new questions and challenges in disaster risk research and management (Smith 2004; Keller and DeVecchio 2012). Are we living at a time when our planet becomes an ever more dangerous place due to natural disasters, or was the recent accumulation of severe events just by accident? Which elements of the risks are currently changing, and on which aspects should risk managers and the global community focus in order to revert the trends and achieve better, optimized outcomes? From a cost-benefit perspective, where and how should societies' limited resources (money, labor, energy) be invested in order to reap the highest benefits in terms of reduced disaster costs (lives, assets) relative to the costs of investments? In order to address these questions in an appropriate way we would need to consider and evaluate the risks in their entity rather than merely focus on selected aspects. Many risks are, however, considerably complex and, above all, context-specific. Exploring and examining specific elements of risks therefore helps in attaining a more coherent assessment, e.g., by determining specific indicators and assessing parameters which can then be used for risk approximations, including modeling and simulation.

Some of the most pressing issues that need more attention from risk researchers and managers relate to human dynamics and the question how humans are transforming their environment. The growth of the human society since the mid nineteenth century has been unprecedented in history. While in 1850 the total world population was around 1.2 billion, the mark of 7 billion people has been breached in the year 2011. Within the next 40 years the population is set to increase further to an estimated 9.3–10.6 billion (UNFPA 2011). Meanwhile, global per capita gross domestic product (GDP) has been increasing albeit with a high disparity among regions 10- to 22-fold from 300–700 US$ in 1850 to about 7,000 US$ in the year 2000 (in 1990 US$). This is equivalent to a 52- to 114-fold growth in the global economy between 1850 and 2000 (Maddison 2006; DeLong 1998). On the downside of human development and expansion lies an increased exploitation of natural resources, and a pervasive transformation of landscapes (especially deforestation, agricultural land development, and urbanization), often leading to severe and irreversible environmental degradation (especially losses of soil resources), including an unprecedented rate of species extinctions and the deterioration or even demise of entire ecosystems in fast developing regions (Cochard 2011a). Economic growth has largely been fueled by burning of fossil energy sources based on C, i.e. coal, oil, and gas. Since the start of the industrial revolution in the nineteenth century about 330 Pg C (1 Pg = 10^{15} g) [~1,200 Pg of carbon dioxide (CO_2) between 1890 and 2007; IEA 2009] have been unlocked from ancient geological deposits and released into the biosphere. This buildup of C within the biosphere – in particular atmospheric CO_2 – is unleashing new uncertainties and associated risks. This will bear new challenges and prospects for humankind, and may also promote innovation.

Many of today's questions in risk management relate to C issues. C is a life element which – bound in inorganic and organic compounds – performs vitally important and beneficial roles. It is the main element in the biomass of Earth's organisms. Organic biomass contains C at a mass ratio of about 0.5 (Smith et al. 2002), and therefore, productive ecosystems may store and sequester a lot of C. Such ecosystems play essential functions to humans in terms of provisioning, regulating, supporting, and cultural services (Hester and Harrison 2010, Turner et al. 2008). Provisioning services (e.g., the production of food, timber and medicinal products), in particular, may be important for human survival after disasters have happened. These services may be crucial for swift reconstruction and the recovery of affected societies (Adger et al. 2005; Cochard 2011b).

In contrast, atmospheric C in the form of gaseous CO_2 increasingly represents a threat as the most important greenhouse gas (GHG) held accountable for causing global warming. Concentrations of CO_2 in the atmosphere have increased between 1900 and 2010 from about 290 ppm to over 380 ppm. It is now well established and generally accepted among scientists that the observed increases in average global temperature (by 0.74 °C from 1906 to 2005; Trenberth et al. 2007) are a consequence of anthropogenic emissions of GHGs, in particular CO_2 (IEA 2009; IPCC 2007; Anderegg et al. 2010). Concentrations of CO_2 are projected to further increase to over 600 ppm until 2050 along with an estimated average global temperature increase of about 0.2 °C per decade (IEA 2009; Trenberth et al. 2007). One question of growing concern is whether and to what degree various types of weather-related hazards (severe storms, floods and other hazards) are becoming more dangerous as a result of global and regional climate change.

In this chapter the question of how climate change is expected to influence natural hazards will be addressed briefly, recognizing the many uncertainties which still exist. Enrichment of C within the biosphere may enhance global warming (and therefore the risks of weather-related natural hazards) only provided that C occurs in its gaseous form as CO_2. The conservation and management of C-rich ecosystems can, therefore, make a substantial contribution to mitigate the hazards by storing and sequestering C. At present, the conservation of the stored C pool via ecosystem conservation measures (in particular in forest ecosystems), and via measures to halt soil erosion and the losses of soil organic matter (SOM) should be a top priority. On a global scale, there may not seem much purpose in discussing the benefits of C sequestration as long as the current conditions of environmental degradation persist unmitigated. Within the previous decade about half of the Earth's mature tropical forests (7.5–8 million km^2) have been cleared, and the world loses forests at annual rates of 50,000–130,000 km^2 (equivalent to CO_2 emissions of 1.6 Pg C $year^{-1}$; FAO 2011; IPCC 2007; Asner et al. 2009; Laurance 2010). In addition, an estimated 4–6 million Pg of organic C may be lost from soils annually by erosion, mostly as a result of poor land management and deforestation (Lal 2003). Similarly, about half of the freshwater wetland areas have been lost globally – ecosystems which typically represent major C sinks (Zedler and Kercher 2005; Tockner and Stanford 2002). This chapter will, therefore, set a spotlight on C-rich ecosystems (and in particular forests), and a discussion of their functions relevant to climate change and natural hazards, including their stability under human pressure and exposure to natural hazards.

Many ecosystems are thought to play significant roles in directly mitigating the effects of natural hazards. The degradation or loss of ecosystems is, therefore, often feared to increase the hazards. The potential functions of ecosystems to protect communities against natural hazards have been viewed as a part of regulating ecosystem services. Yet, the term 'hazard regulation services' (Beresford et al. 2008) may be somewhat misleading as it implies control of hazards by ecosystems. No natural hazards can ever be completely 'regulated' either by humans or ecosystems. However, hazards may be mitigated in various ways by ecosystems, either by delaying the impact or lessening the hazard intensity. A major part of this chapter will, thus, address the question of whether and in which ways natural disasters are linked to or influenced by different types of C-rich ecosystems.

11.2 Risks of Natural Disasters: Overview of Concepts and Issues

11.2.1 Concepts and Frameworks for Risk Assessment

News broadcasts regularly report about disasters which have occurred in various parts of the world. Disasters are real and tangible events which directly affect human lives and stimulate strong emotions. In general terms, a disaster may be any "occurrence that causes great distress or destruction", and in its original sense the term signifies a "malevolent influence" caused by an unfavorable stellar constellation (*aster* meaning star in Latin) (Collins English Dictionary 2009). Disasters have not only affected humans directly, but have also influenced human beliefs and philosophical thought which sometimes changed the course of history. In certain religious texts disasters are often described as acts of punishment by God. However, natural disasters also inspired antithetic thoughts. For example, the earthquake and tsunami that destroyed the Portuguese capital Lisbon on a church holiday in 1755 AD shattered many tenets of the time, and stirred up debates on theodicy among European enlightenment philosophers, including Voltaire, Rousseau, Hume, and Kant (Paice 2008; Neiman 2002). The focus subsequently turned away from God to humans. In a more modern view, therefore, disasters are seen as a consequence of an unanticipated hazard or inadequately managed risk – irrespective of whether or not stars may be involved in creating the hazard. A radical positivist conception is that essentially all disasters can be seen as being human-made, since only human actions before and during the impact of a hazard determine whether the hazardous event will also result in disaster (Wisner et al. 2004).

In contrast to disasters, risks are rather elusive realities to humans. Risks are often shrouded by probabilities of debatable precision, or they are misvalued as a result of human perceptions. Nevertheless, an understanding of the nature and the elements of risks is key to appropriate risk assessment and, thus, effective disaster prevention and mitigation, including the reliable evaluation of ecosystem functions and appropriate measures to protect or restore ecosystems. 'Risks' have been broadly described as "a condition in which there is a possibility of an adverse deviation from a desired outcome that is expected or hoped for" (Vaughan 1997, p. 8). Within the risk, a 'hazard' characterizes a phenomenon (e.g., storm, flood, earthquake, etc.) that may, with a certain probability, produce a disaster of a certain magnitude (Cochard et al. 2008). 'Vulnerability' refers to an inability to withstand the impact of a given hazard. It is a condition or process resulting from physical, social, economic and environmental factors that determine the likelihood and scale of damage (Wisner et al. 2004; UNDP 2004). Finally, a 'disaster' represents an event (often impulsive and violent) which results in significant physical damages or destruction, loss of lives and property, disruption of the economic, social and cultural life of people and/or drastic changes in the environment. Disaster occurs when a hazard meets vulnerability (Wisner et al. 2004).

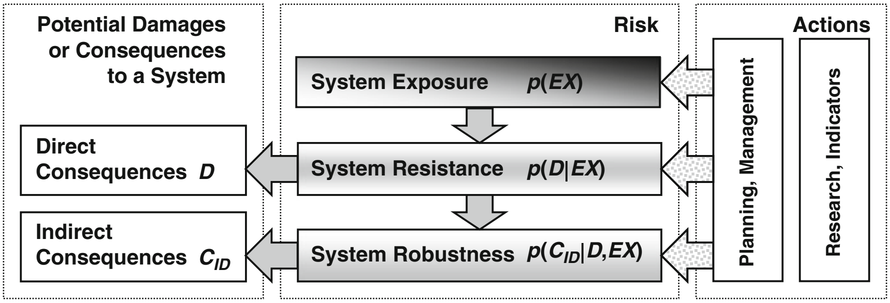

Fig. 11.2 A conceptual indicator-based risk assessment framework. The risk to a specific system is due to three sequential components. System exposure is determined by the probability p(EX) by which the system may be exposed to a hazard of a certain intensity. Resistance is determined by the probability p(D|EX) by which a damage D of a certain intensity will occur to the system given exposure EX. Finally, resistance is determined by the probability p(C_{ID}|D,EX) by which further indirect consequences will affect the system given damage D. Various components of the risk can be assessed by use of appropriate indicators. Knowing the risk, mitigation actions can be taken by managers, whereby new risk assessments can be performed if the parameters change (Adapted with modifications from Straub 2005; Bayraktarli et al. 2005)

Qualitative information of past disasters and lessons about the risks have been transmitted orally from generation to generation in traditional societies. Such information has often proven to be crucial for the survival of recurring hazard events. For example, based on transmitted instructions traditional societies on Simeulue Island, on the Nicobar and Andaman Islands, and in the Solomon Islands had largely escaped the impact of the tsunamis in 2004 or 2007, respectively, whereas many casualties occurred during these events in adjacent more modern or immigrant societies (McAdoo et al. 2006, 2009; Cochard et al. 2008). Nowadays, risks are assessed quantitatively. Estimations and predictions are made based on the evaluation of available databases of past disasters, and the application of state-of-the-art risk models.

Risk managers of various disciplines (e.g., city planners, engineers, foresters, etc.) often consider a specifically defined system (e.g., a city, a social community, a built object, a forest ecosystem, etc.; Fig. 11.2; Straub 2005; Bayraktarli et al. 2005). Then they try to determine the probability p(EX) by which such a system may be exposed to a hazard of a certain magnitude, as well as the probability p(D|EX) by which the hazard exposure EX may overcome the resistance of the system and lead to a direct damage D of a certain magnitude. The probabilities p(EX) and p(D|EX) obviously differ between types of hazards and systems, but also between the hazard and damage event magnitudes which may be considered. Hazards of low intensities are normally much more common (i.e., probable) than those of very high intensities, and system exposure to a low intensity hazard often bears no consequences, provided that the system is not exceptionally vulnerable (has a low resistance) to any exposure. Continuous frequency distribution functions of natural hazards and associated disasters have been determined from data collected about past events. Functions used to describe the probability of natural hazards p(EX) such as earthquakes, tsunamis, floods, storms and storm surges as well as associated disaster events have included the log-normal, three-parameter log-normal, the Gumbel, the Gamma and the Weibull distribution, as well as other specifically determined distribution functions (Emanuel et al. 2006; Cello and Malamud 2006; Malamud and Turcotte 2006; Kidson et al. 2005; Choi et al. 2002; Nishenko and Buland 1987). System robustness is another aspect of risk in a wider sense. It is assessed through the probability $p(C_{ID}|D,EX)$ which expresses the indirect consequences C_{ID} due to the damages D to the system. A robust system is characterized by a high resilience, which is the capacity of the system to return (rebound, recover) to the state held before the hazard impact. In contrast, a non-robust or unstable system is likely to further deteriorate after initial damages have occurred. This may be due to subsequent hazards which follow an initial disaster (e.g., disruptions and extinctions in an ecosystem, loss of state control and government services, diseases, conflict, excessive resource exploitation, etc.).

11.2.2 Natural Hazard Interactions and Probabilistic Network Models

Natural hazard risks are rarely due to a single and simple phenomenon. Normally, risks are the result of a chain of interlocking hazardous events, some of which may be more immediately dangerous than others. There are always certain elements along such a chain of events which can be mitigated better than others by using certain management approaches. Earthquakes, for example, cannot be predicted in an exact way, and there is nothing that can be done to decrease their seismic forces. The risk may only be mitigated by increasing knowledge and raising awareness about the potential hazard. A primary hazard such as an earthquake, however, rarely poses a threat to people directly, but only indirectly via secondary hazards such as collapsing structures or landslides. Thus, there is a lot that can be done to mitigate these secondary hazards via various engineering, planning and policy measures. In contrast to earthquakes, weather phenomena such as cyclones normally develop over several days. Provided that meteorological stations and media networks collaborate effectively to alarm the people, these hazards will not arrive entirely unexpected. There is a lot that can be done in anticipation of a potential impact, e.g., installing reinforcements of structures, and timely evacuation of people and vital assets to safer places. Cyclone-force winds can potentially be directly lethal, but most fatalities occur when houses are destroyed by winds or due to the inundation and impacts caused by coastal storm surges or catchment-based floods. There may, thus, be a large range of hazard components that may need to be assessed and addressed.

In risk management, each of the hazard components may be assessed separately based on certain assumptions. However, risks are often complex and some aspects may be quite unpredictable. For example, in June 2006 a minor earthquake occurred off the coast of Java in Indonesia, which resulted, however, in a significant tsunami disaster. The tsunami was so surprising and lethal because it was apparently triggered indirectly by a submarine landslide rather than directly by the earthquake itself (Kerr 2006). As the national authorities did not issue a tsunami warning people along the coast were taken by surprise, and more than 659 were killed in the disaster (Swissinfo in Cochard et al. 2008). Risk assessment is often an intricate, multifaceted process. During the past decades, however, powerful modeling methodologies have been developed. Using a multitude of risk-relevant factors and indicators risks can, for example, be assessed through Bayesian probabilistic network models (i.e., networks of conditional probabilities). Such models have also the advantage that they can be adapted continuously to include new or improved information on various parameters (Bayraktarli et al. 2005; Straub 2005; Faber and Maes 2005). For any particular location or system, all the natural risks may be assessed in order to advise decision-makers about necessary laws and policies, and how to best allocate the available resources in risk management.

11.2.3 Overview of Primary and Secondary Natural Hazards

Most natural disasters can be attributed to two categories of broadly defined primary hazard sources, (i) disasters which are the consequence of tectonic events, and (ii) disasters which are due to weather phenomena and climate variability (Fig. 11.3).

Primary tectonic hazard events are earthquakes and volcanic eruptions. These mostly occur along plate tectonic fault lines or near volcanic hotspots. As primary weather-related hazard events we may list severe rainfall (e.g., during the tropical monsoon), hail and snowfalls, tropical cyclones (hurricanes or typhoons), winter storms, and tornadoes. In most cases, these hazards entail strong winds and intense precipitation (rain, snow and/or hail). In addition, we may also refer to other weather-related hazards such as long-term water deficiencies leading to droughts and prolonged periods of extreme temperatures referred to as either 'heat waves' or 'frost waves'. Wildfires represent a special case of hazard (often human-caused) which is much influenced by climate and weather. Fire is primarily nourished by vegetation biomass, but it is promoted under dry, hot and windy conditions (Whelan 1995). Other hazards which in various ways may affect humans and their livestock and agricultural crops include biological pests and diseases. These too are often influenced by climatic parameters (Morens et al. 2004). Tectonic and weather-related primary hazards may directly impact structures by seismic shaking, extreme wind gusts (often particularly hazardous due to loosely flying debris), volcanic ash and debris, and volcanic lava flows. Tectonic and weather-related primary hazards most commonly also produce secondary hazards, sometimes through interaction.

Floods occur when waters spill over the confines of water bodies (i.e., river banks, lake shores, coastlines). Therefore, floods represent a potentially hazardous (but also often beneficial, e.g., for agricultural productivity) overflow of land adjacent to a water body. Catchment-based floods are a result of intensive rainfalls over a catchment. Floods may be due to long extended periods of rainfall or short but

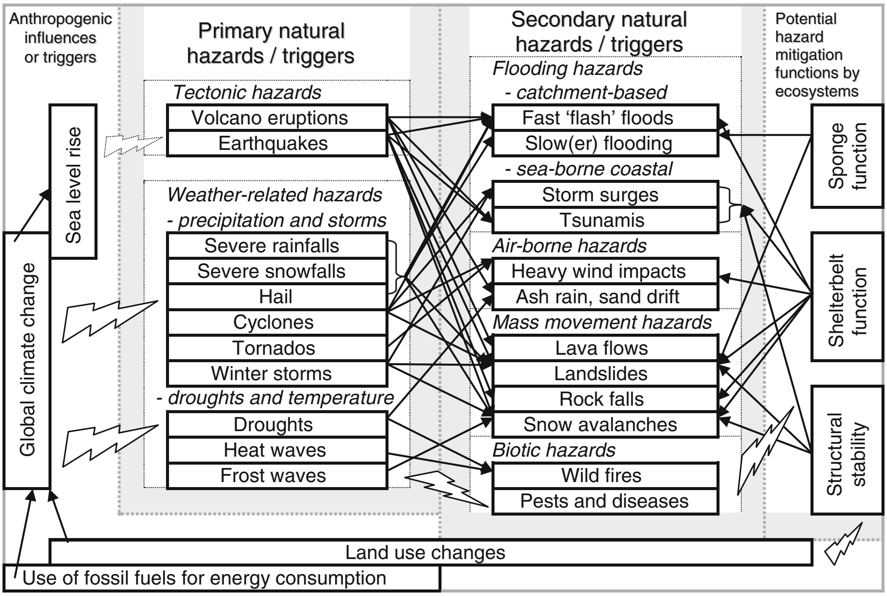

Fig. 11.3 Simplified overview of natural hazard interactions, the influence of human actions, and potential hazard mitigation functions by C-rich ecosystems. The *black arrows* indicate specific hazard triggers and influences of primary on secondary hazards, respectively potential hazard mitigation functions of ecosystems. The *flashes* indicate non-specific triggers and influences. Global climate change is primarily driven by the anthropogenic burning of fossil fuels. Land use changes add to GHG emissions but can also directly affect ecosystems which serve potential hazard mitigation functions. Climate change effects exert an influence on primary natural hazardous phenomena, in particular weather-related events such as storms and droughts. These in turn trigger secondary phenomena which represent the highest risks to human safety. Secondary hazards – catchment-based floods (Sect. 11.4), mass movement hazards (Sect. 11.5), and sea-borne floods (Sect. 11.6) – may or may not be mitigated by C-rich ecosystems

intensive rain showers (Merz and Blöschl 2002). In tropical regions, intense rainfalls may occur during the monsoon seasons. Severe floods may also be associated with cyclone storms. In temperate regions, flooding often occurs as a result of storms, commonly during late autumn and winter (Church 1988). However, flooding may also occur during spring and early summer, often as a result of warm rainfalls combined with fast rates of mountain snow melt (Merz and Blöschl 2002). Flash-floods may occur after exceptional heavy rain showers (sometimes in interaction with snow-melt) in steep locations in mountainous regions. Highly hazardous and violent flash-floods may also be the result of volcanic eruptions or earthquake-triggered landslides, rockfalls or ice-falls from glaciers, or breaking of constructed and natural dams. For example, the town of Armero in Colombia was completely destroyed in 1985 (with over 20,000 casualties) by a flash flood mixed with debris resulting from the melting of the glacial ice during the eruption of Nevado del Ruiz stratovolcano (Piersen et al. 1990). In mountain areas, lakes are often produced at the base of retreating glaciers. Such lakes may pose a threat to the valleys below when ice-cored moraines containing the lakes melt or collapse (glacial lake outburst floods). This risk is increasing as glaciers are retreating under climate change (Richardson and Reynolds 2000). Similarly, in 1911 an earthquake triggered a huge rockslide (2 km^3 volume) which blocked the Murgab River in the Pamir Mountains of Tajikistan creating the Lake Sarez. It is feared that the 17 km^3 of lake water could break out again if the natural dam collapses under the impact of a similar earthquake (Schuster and Alford 2004).

Coastal sea-borne flooding may occur as a result of extreme wind-driven sea waves (e.g., long-range ocean waves; Gibson and Swan 2006), especially along exposed coastlines with limited or degraded wave defense features. Storms and earthquakes are, however the two leading causes of catastrophic coastal flooding (Cochard et al. 2008). During intense storms, wind-driven waves can exceed 5 m in height, and waves up to 20 m high have been recorded during very extreme events (Earle in Woodroffe 2002). A major threat to coasts is, however, posed by storm surges which may develop during a low-pressure weather system – especially during tropical cyclones. Storm surges typically build in predominantly shallow waters (low bathymetry) due to the combined effects of strong winds (pushing water against the coastlines) and the low pressure of the weather system (pulling the water masses) (Cochard et al. 2008; Irish et al. 2008). Storm surges are a slow rise in sea level as the storm approaches landfall. Extreme storm surges may rise over two meters high (mean water level) and may penetrate several kilometers inland (Fritz et al. 2009). While the storm surge will lead to coastal flooding, it is mostly the violent wind waves (often persisting for hours) which cause the highest physical destruction. Storm surges may be particularly deep and hazardous if they coincide with high tides and catchment-based cyclonic flooding (Cochard et al. 2008; Irish et al. 2008; Keller and DeVecchio 2012). Tsunamis are commonly the result of a sea earthquake (e.g., the large tsunamis in 2011 in Japan and 2004 in Asia), but they can also be caused by a volcanic eruption (e.g., the Krakatoa eruption in 1886 in Indonesia), or an undersea landslide (e.g., the 1998 Aitape tsunami in New Guinea). Tsunami waves are generally much faster and violent over a short time period than storm surges. In the open ocean tsunamis may travel at speeds >800 km h^{-1}. Tsunamis significantly differ from wind-driven waves as the wavelengths may be in the order of tens to hundreds of kilometers and the wave energy is distributed throughout the entire water column (Yeh et al. 1994). During landfall, tsunamis may rush inland like a huge tidal speed-flood causing widespread destruction and loss of life. Wave build-up and run-up are significantly influenced by coastal bathymetry and land topography (Chatenoux and Peduzzi 2007; Cochard et al. 2008). Often, several tsunamis occur during a tectonic event (Gregg et al. 2006).

In mountainous or hilly terrain mass movements represent a common hazard. Mass movement hazards include landslides (soil and/or rocks), rockfalls and avalanches (snow and/or ice). A special case is represented by hot lava flows which may emanate from active volcanoes. Ground mass failure and movement may result in steep terrain due to essentially 'autogenic' processes (e.g., release of surface tensions, expansion of ice, etc.), but especially landslides are most typically influenced by abundant precipitation and infiltration of water (Sidle 2007; Guzzetti et al. 2007).

The most extensive landslides (i.e., multiple-occurrence regional landslide events; Crozier 2005) have been observed during tropical cyclone events and other storms with heavy rainfalls and wind impacts on forested hillsides, and also during earthquakes in steep mountain areas (Lin et al. 2008; Smith 2004; Peduzzi 2010). The snow and/or ice loads which are the origin of avalanche hazards are directly influenced by winter precipitation loads in mountainous regions. Certain weather conditions in interaction with terrain are most important in influencing the formation of avalanches as is illustrated by the daily Snow and Avalanche Bulletin of the Swiss Federal Institute for Snow and Avalanche Research in Davos (see SLF 2012). However, earthquakes and storms can also trigger multiple avalanches (Schweizer et al. 2003; Stethem et al. 2003).

11.2.4 Hidden Disparities in Data

Numerous databases of natural disasters and hazard impacts have been developed at national and international levels (GRIP 2012). Such databases can provide key information about the probabilities of hazards producing disasters. Depending on detail and accuracy of the data provided, the constituent components of the risks (hazards and vulnerabilities) can be further assessed. From disaster data (and information derived thereof), mathematical models can be developed and tested, and inferences can be drawn about the nature of disasters and risks as well as past and current trends in various regions. Databases are, therefore, a key tool for risk assessment and management. However, data provided from disaster audits need to be understood. Data need to be treated, analyzed and interpreted with due caution in order to deduct reasonable conclusions based on facts.

Various media articles, scientific literature and university textbooks convey a view that natural disasters – especially those related to weather and climate – have been increasing in recent decades and years. For example, in a major textbook (Keller and DeVecchio 2012, p. 4) it is stated that "during the past half century there has been a dramatic increase in natural disasters, as illustrated by Figure 1.6." This figure depicts a trend line showing a twofold increase in the number of natural disasters between 1987 and 2006 from approximately 200 to almost 450 disasters annually. According to data depicted in graphs in another standard textbook (Smith 2004) the numbers of flood events, those of people displaced by floods, as well as those of people killed by floods have all shown a worldwide upward trend between 1975 and 2002. The Economist (2012a, b, c) maintained that there is a clear upward trend in global economic losses due to natural disasters, but that there had been a decrease in the number of human casualties.

All the above statements on disaster trends were based on data from the Emergency Events Database (Em-Dat 2012) maintained by the World Health Organization Collaborating Centre for Research on the Epidemiology of Disasters (CRED). This is a major global database which contains data on the occurrence and the effects of over 19,000 disasters around the world since 1900. The database includes natural disasters (i.e., caused by earthquakes, volcanoes, tsunamis, landslides, mudflows, avalanches, cyclones, tornadoes, storms, catchment-based floods incl. flash floods, heat and frost waves, droughts, wild fires and different types of epidemics) as well as various types of technical disasters (Em-Dat 2012). The data are being compiled from various sources, including United Nation agencies, non-governmental organizations, insurance companies, research institutes and press agencies. Em-Dat is a key database. CRED continually works to improve the data input by tapping into new data sources, enhancing auditing standards, and incorporating new types of information. However, when using historical disaster databases such as Em-Dat or other databases several intricate issues need to be considered (cf. Smith 2004). First, there are various issues in the definition of disasters, second, there are biases in the reporting of disasters, and third, there are fallacies in the interpretation of the data which need to be avoided. Many of these issues are interconnected.

The criteria for Em-Dat to include a hazardous event as a disaster in the database are either that (1) ten or more people were killed during the event; (2) a hundred or more people were reported as affected; (3) a declaration of a state of emergency was issued; and/or (4) a call for international assistance was issued. Major issues are associated with criteria 2–4 (whereby virtually all disasters meeting criteria 3 and 4 meet criteria 1 and/or 2). 'Affected people' are defined by Em-Dat (2012) in general terms as "people requiring immediate assistance during a period of emergency." These may be injured, homeless, displaced or evacuated people, i.e., people which usually have been registered with an aid agency. The number of people recorded as "affected", therefore, depends on whether and in which ways aid agencies are involved in helping affected people. In disasters where no aid agencies are called to action all affected people may well remain unrecorded and the disaster may not be reported as such. Whether or not aid agencies are called to action and disasters are registered depends mainly on the type and the circumstances of the disaster and – in particular – on the location where it occurred. Various countries have different governments with different policies. For example, more than two times as many disasters have been recorded in Thailand as compared to its neighboring country Myanmar. Comparing North Korea with South Korea reveals a ratio of about one quarter. Even though the Democratic Republic of the Congo (DRC) covers a land area almost 2.5 times larger than that of Germany and France combined, more than ten times more natural disasters are recorded for the two European countries combined as compared to DRC (Em-Dat 2012). The main reason for these differences are obviously differences in reporting frequency rather than any differences in the number of disasters occurring.

Notably, none of the disasters listed in Em-Dat for the DRC have occurred before independence of the country in 1960. This is similarly the case for many other countries which were still under colonial rule at the beginning of the twentieth century. But even for the now industrialized, former colonizing countries in Europe and the USA documentation of disasters has increased throughout the twentieth and twenty-first century, with typically considerably more disasters recorded within the last two or three decades as compared to the preceding eight

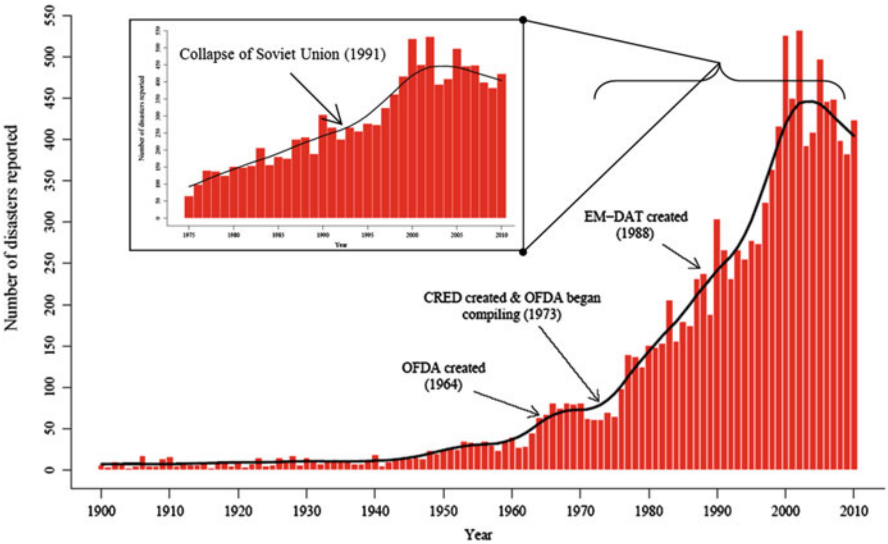

Fig. 11.4 The number of global records of natural disaster events in the Em-Dat (2012) database has been steadily increasing since data collection until about the year 2000 after which recording has remained constant or has even been decreasing. During the period 1975–2010 a steep increase occurred from 1995 to 2000. OFDA is the Office of Foreign Disaster Assistance, an organizational unit within the United States Agency for International Development (USAID). CRED is the Centre for Research on the Epidemiology of Disasters, a research unit of the Université Catholique de Louvainin Brussels (Em-Dat: the OFDA/CRED International Disaster Database, www.emdat.be, Université Catholique de Louvain, Brussels, Belgium)

decades (Em-Dat 2012). The frequency distribution of disasters reported between 1900 and 2010 globally is shown in Fig. 11.4. The trend as it was depicted by Keller and DeVecchio (2012) is probably also attributable to a general increase as well as to a historical artifact. In 1991 the Soviet Union collapsed and many successor countries and former Eastern Bloc allies changed to more open and democratic regimes and transparent governance. This may, at least partially, explain the steep increase in the worldwide reporting of disasters between about 1996–2000, whereas no significant increases occurred during the periods 1987–1996 and 2000–2006 (Fig. 11.4).

The first criterion for Em-Dat is considerably less equivocal as compared to the second criterion, albeit similar problems of biases in reporting apply, and ‘missing’ people may remain unrecorded. In the data provided, the ‘number of people killed’ is undoubtedly the most reliable of all the categories. For disasters before the 1970s it is often the only indicator (the other being ‘number of people affected’, ‘injured’ and ‘homeless’, and ‘damages in 1,000 US$’). However, even if mortality figures for an event are reported correctly, it is often difficult to interpret the exact causes and hazards explaining the fatalities. As already outlined in the previous section, many disasters may be the result of a complex hazardous ‘cocktail’. For example, the database rarely provides exact information about how people were killed during a tropical cyclone. This may be due to landslides and flash floods from steep terrain, due to storm surges in combination with catchment-based floods in low-lying coastal areas, due to wind impacts on structures, secondary effects resulting from the failure of structures, or any combination thereof. If it is difficult to identify the exact hazards involved, then it is even more difficult to determine the role that ecosystems may have played during any such event. Furthermore, fatalities during a disaster may not reflect the effects of a hazardous impact in an adequate way. The database does not provide information on "slow disasters", some of which, however, may be a direct consequence of a severe "fast" disaster. For example, severe chronic and potentially fatal diseases resulting from a nuclear power plant failure (or other environmental catastrophes) may only be recorded over time and persist over decades. If no people were killed directly during an event, then such a potentially severe incident may not even be recorded in the database. For example, a severe industrial accident and chemical spill at Schweizerhalle in Switzerland in 1989 raised health concerns as it exposed people to a foul-smelling cloud and killed off most of the fish stocks of the Rhine River downstream (Ackermann-Liebrich et al. 1992). Even though this was one of the worst and most expensive technical and environmental disasters in Switzerland's history it is not listed in the Em-Dat database. Certain disasters have serious impacts on the environment and may thereby indirectly affect communities in the long term (Cochard 2011b). Fatalities or injuries arising in the aftermath of an event resulting from secondary causes such as diseases, hunger, psychological stress resulting in suicide and other causes generally remain unrecorded.

11.2.5 Hidden Disparities in Human Vulnerabilities

The observation (factual or not) that natural disasters have increased is often seen as an indication that natural hazards have actually increased. As discussed in previous sections, disasters are, however, also a result of vulnerability. One obvious possible reason for an increase in disasters could be that increasingly more people and assets are placed in harm's way. Within the last decades the growth of the world's population and economy has been unprecedented. Agriculture has intensified, urbanization and industrialization has spread, and property and wealth has increased in many countries and has become concentrated in certain regions. Furthermore, societies, economies and information networks have become more globally interconnected and interdependent. More people and assets are, therefore, being exposed to natural hazards, and local impacts may have wider implications elsewhere. This may be one major explanation for the exceptional economic losses in 2011. For example, in Thailand most of the rapid growth in export-oriented industries during the last two decades has occurred in the floodplain areas along the Chao Phraya River. Albeit very large floods have occurred in the past, the factories were mostly unprotected and badly prepared for evacuation when they were hit by the floods. In contrast, in Japan coastlines were thought to be well defended against tsunamis before the event on 11 March 2011 when towering tsunami floods (over 30 m high in some places; Mori et al. 2011) overwhelmed many specifically engineered tsunami-defense levees and tsunami-buffer forests, and reached towns and settlements situated kilometers inland. Through the tragic event it became obvious that many defenses were insufficient. Scientific records now also show that similar tsunamis may have already occurred in the distant past (i.e., Jōgan earthquake and tsunami in 869 AD; Goto et al. 2011). Both events illustrate that knowing the risk as best as possible is crucial for appropriate disaster preparation. Infrastructural and economic developments, however, often proceed at a fast pace, leaving behind the advancements in risk research and operative arrangements in planning and management for disaster prevention and mitigation. In addition, the developments may lead to major land use changes and environmental degradation which may continually alter the nature and severity of natural risks as may cultural and institutional advancement and socio-political exchange, and conflict or solidarity.

The claim by The Economist (2012a, b, c) that economic losses caused by disasters have been increasing may partly reflect an economic bias as well as a reporting bias. Total losses of assets may well reflect the recent economic growth and increases in economic assets, rather than actual loss increases in relative terms. For example, data on annual hurricane damages in the United States show a clear increase between 1925 and 1995 only in absolute terms. However, if losses are weighted by total assets exposed to hurricanes at the respective times then past losses (period 1925–1975) match or even surpass the more recent (1975–1995) losses (Pielke and Landsea 1998). Similarly, increasing losses due to floods in Europe appear to be largely attributable to societal factors rather than to apparent effects of climate change (Barredo 2009). In the Em-Dat (2012) database costs are hardly reported before 1970, and reporting increases steeply between about 1975–1995. Even today, economic losses are normally reported by insurance companies and government institutions only for disasters in industrialized and emerging industrializing countries. Costs reported by Em-Dat (2012) for disasters occurring in developing countries are a great deal less reliable. Costs are often estimated from proxies, e.g., relief aid contributions by United Nations and other international agencies. The reported costs typically amount to much lower totals than those in rich countries, and frequently costs are not reported at all for smaller emergencies (cf. Em-Dat 2012). This does certainly not mean, however, that the overall suffering from disasters in poor countries is less. If costs are set in relation to the total wealth productivity in a country (e.g. weighted by the GDP) it becomes apparent that the losses due to natural disasters are considerably higher in developing countries than in developed countries (i.e., about 20 times as estimated by UNISDR 2009). According to Mirza (2003), economic losses in developing countries estimated at around 35 billion US$ annually are now more than eight times greater than during the 1960s. For small countries under high hazard exposure (e.g., Haiti, Honduras, Nicaragua, Ecuador, Bangladesh, etc.) these losses can have significant long-term consequences in terms of sustainable economic growth (Freeman in Mirza 2003). Figures of human casualties caused by disasters are usually much more reliable than estimates of economic losses, and they show differences between the rich and poor countries which are even more striking. According to UNISDR (2009) about 95 % of all the reported casualties globally due to natural disasters were reported in developing countries. Cyclone data show that mortality rates are a multiple factor higher in low-income countries as compared to high-income countries when similar numbers of people are affected by hazards of similar severity (Peduzzi et al. 2012; Em-Dat 2012). This illustrates the high hazard vulnerability of people living in poor countries.

The claim by The Economist (2012a, b, c) that human casualties during disasters have been decreasing is probably true to some degree, albeit there are equally huge discrepancies among various world regions, with conditions deteriorating in some countries. Especially industrialized countries have made considerable progress in risk and disaster management and decreased the overall number of casualties throughout the twentieth century. Considerable differences still exist between developing and industrialized countries. However, due to improved disaster preparation and international aid the numbers of casualties have also fallen in many developing countries (Em-Dat 2012). In Bangladesh, for example, an extensive program for cyclone warning and preparation (including the construction of shelters) has been undertaken during the last two decades. Even though many problems still exist the program has succeeded in substantially decreasing the number of cyclone victims. During Cyclone Gorky in 1991 about 140,000 people were killed whereas a similar event in 2007 (Cyclone Sidr) claimed only around 3,400 lives (Bimal 2009; Bimal et al. 2010). In addition to increasing efforts at national levels international disaster aid has become much more efficient in recent decades. A record ten billion US$ have, for example, been donated after the 2004 Asian tsunami which affected various countries in the Indian Ocean and killed 250,000–300,000 people (Cochard 2011b). Without much needed emergency aid probably many more people would have died. The success of national and international aid during large emergencies seems evident especially for slowly unfolding disasters such as large river floods and droughts which are responsible for some of the deadliest disasters in the twentieth century. For example, floods claimed tens of thousands or even millions of lives in China in 1911 (100,000 casualties), 1931 (up to 4 million), 1933 (18,000), 1935 (150,000), 1939 (half a million), 1954 (30,000), and 1959 (up to 2 million). Albeit similar or even larger floods occurred in recent years (e.g., in 1995, 1998, 2002, 2003 and 2011) on average less than 500 and maximally about 4,000 people were killed (Em-Dat 2012). Similarly, floods of the extent that hit Thailand in 2011 and Pakistan in 2010 would probably have resulted in many more casualties just a few decades ago. Despite an overall progress and an improving global network in disaster aid the numbers of casualties are, however, still disproportionately large in many countries for several types of disasters. State failure, repression and/or civil strive have gone hand in hand with the neglect of emergency-relevant institutions and disaster mitigation programs in several countries. Recent examples are Haiti (316,000 casualties after an earthquake in 2010), Myanmar/Burma (140,000 casualties due to Cyclone Nargis in 2008), North Korea (up to 3.5 million deaths 1990–1997 due to droughts and agricultural mismanagment), Somalia (tens of thousands of deaths during the East Africa famine 2011-present), and other afflicted countries. Many of the disasters in these countries did not come entirely unexpected (Spiegel et al. 2007; Webster 2008; Bilham 2010; The Economist 2011a, b; Zurich 2009). For many of these disasters which affected mostly poor people in insecure or failing states there are no detailed economic appraisals (Em-Dat 2012; The Economist 2012c). If losses of human lives could be measured up in monetary values (the 'economic value' of human lives have been estimated from 100,000 US$ to several million dollars; Mrozek and Taylor 2001; Shanmugam 2000) then, however, the costs would certainly be staggering.

While the numbers of victims and the economic costs of disasters are often unclear, there are rarely detailed appraisals about the impact of natural disasters on the environment and natural resources. Particularly in the poorest countries people are, however, highly dependent on natural resources which are frequently already severely overused and degraded. In the aftermath of disasters, such resources are often under increased pressure, and their exhaustion can mean that environmental rehabilitation from the disaster will be delayed or stalled – especially if international aid is not available in a sufficient amount or cannot effectively reach the affected communities. But even in regions where communities are not closely dependent on natural resources any gains in security by improvements in natural risk management may be increasingly threatened by intensifying hazards. We are only just starting to appreciate the scale by which human domination of the environment has altered the vulnerability of the environment and of societies to extreme events (Messerli et al. 2000). The risks of mass mortality during large natural disasters seem to have decreased due to the globalization of disaster aid (The Economist 2012a, b, c). Nonetheless, due to an increasingly hazardous environment the international community may still face an increasing frequency of smaller disasters (e.g., mass movement hazards) in terms of human casualties as well as large disasters (e.g., storms and floods) in terms of asset losses. Climate change and environmental degradation have been blamed to have contributed to large natural disasters in recent years. For example, deforestation and climate change have both been blamed for disasters such as the recent floods in China, Thailand, Australia, Pakistan (Yin and Li 2001; Deutsche Welle 2010; van den Honert and McAneney 2011; Bradshaw et al. 2007; FAO 2005). The devastation along coastlines caused in recent years by cyclonic storm surges and tsunamis have been partly blamed on the prior destruction of mangrove forests and other 'buffering greenbelt ecosystems' (Cochard et al. 2008; Gedan et al. 2010). An observed overall decrease of casualties due to large disasters is no convincing basis to invalidate the presumption that hazards may be on the increase as a result of changing environmental parameters.

11.3 Climate Change Mitigation Roles of Carbon-Rich Ecosystems

11.3.1 Primary Natural Hazards and Their Potential Mitigation by Ecosystems

In what ways can C-rich ecosystems contribute to mitigate the 'primary hazard sources', i.e., tectonic events and hazardous weather phenomena? Certain regions along the Pacific 'ring of fire' and elsewhere (e.g., regions in Japan, Indonesia and in the South Pacific) are currently under increased plate tectonic strain (Cochard et al. 2008; Beroza 2012; Heki 2011). During any hazardous event ecosystems may play a role as defense systems against tsunamis or safeguards against landslides in earthquake-prone regions. However, there is not much that ecosystems provide to mitigate tectonic hazards at source point, even if several scientists have suggested that sea level rise may increase faulting, seismicity and volcanic activity at tectonic boundary zones (McGuire et al. 2002; McGuire 2012; Fig. 11.3).

In contrast, the world's ecosystems are directly interacting with global, regional and local climates, and may, therefore, be of great relevance as agents 'regulating' climate variability and moderating the frequencies and intensities of extreme 'peak' events. Ecosystems may contain a large C pool. Their deliberate conservation and management can therefore make an important contribution to mitigate CO_2 emissions and thereby lessen the effects of global warming. One question which is relevant to risk mitigation and management is whether a C conservation strategy will also lead to a less hazardous climate and environment at global, regional and local scales. If a clear link can be established between climate-related hazards and climate change it may eventually be possible to estimate an approximate value for such a conservation strategy in terms of a reduction in losses due to disasters.

11.3.2 Global Warming and Weather-Related Hazards

According to Meehl et al. (2007), future climates are expected to be warmer and are, therefore, at higher risk of prolonged, intense heat waves. Various computer models indicate that in many regions summers may become more dry, and dry spells more prolonged (increasing the risk of severe droughts), whereas precipitation will increase during winters, especially at northern middle and high latitudes (Meehl et al. 2007). Within about the next 80 years the total precipitation may significantly increase in certain regions, e.g., in the tropical belts, in East Asia and in northern latitudes, and decrease in other regions, e.g., the Mediterranean, Northern and Southern Africa, Australia, the southern USA and Central America, and much of South America, except the Andean countries. This is expected to lead to overall increased levels of runoff in most of the regions with increased precipitation (Meehl et al. 2007). Warmer air can hold more moisture, and rainfall events may, therefore, become more intense, particularly in areas where overall increases will be observed, such as during winter storms in Europe and during the monsoon period in Southern and Eastern Asia (Meehl et al. 2007). It is projected by some models that the increase in precipitation extremes is disproportionately larger than the anticipated mean increases (Kharin and Zwiers 2005; Allen and Ingram 2002). Furthermore, some climate models predict harsh winters for the Northern Hemisphere as a result of increased snowfalls which can dynamically induce wintertime cooling over large scales (Cohen et al. 2012). Such a constellation apparently occurred in January and February 2012 in much of Europe and parts of Central and Northern Asia, when record snow falls (up to 6 m in certain regions in Eastern Europe) were followed by very cold temperatures (The New York Times 2012).

Models indicate that El Niño-Southern Oscillation (ENSO) inter-annual variability will continue under climate change scenarios, but there are still disagreements among scientists what changes – if any – will occur to ENSO in the future (Paeth et al. 2008; Meehl et al. 2007). Various indications exist, however, that ENSO events will significantly interact with monsoon rainfall patterns, possibly leading both to increased flood events or conversely to drought in various regions within the oscillation in the Indo-Pacific region (Yadav 2011; Ummenhofer et al. 2011; Kumar et al. 2006).

Even though there are still many uncertainties from the current databases (especially in the Indo-Pacific region), detailed analyses of past events and of time series data indicate that cyclones may be increasing in intensity as a result of climate change. There is more controversy regarding any change in cyclone frequencies with many studies suggesting an increase, but others suggesting a decrease (Knutson et al. 2010; Meehl et al. 2007; Rummukainen 2012). A study by Saunders and Lea (2008) showed that the two variables local sea surface temperature and atmospheric wind field may explain much of the variability of Atlantic hurricane frequency and activity between 1965 and 2005. An increase of 0.5 °C of the sea surface temperature corresponded with an intensification in cyclone frequency and activity of about 40 %. Models also suggest that tropical storms may become more common and intense at higher latitudes (Meehl et al. 2007; McDonald 2010); however, Lang and Waugh (2011) highlighted several inconsistencies among current models. According to Donat et al. (2011), storm frequencies in Europe have already increased since the end of the nineteenth century as evidenced by statistical time series analyses of storm data. Clearly, intensification in cyclone events will lead to higher risks of storm surges along the exposed coastlines, especially as coastlines are also increasingly affected by coastal erosion and flooding due to sea level rise (currently estimated at 3.3 mm $year^{-1}$; Rahmsdorf et al. 2007) and coastal development (Cochard 2011b; Thampanya et al. 2006). It is expected that climate change will also bring about an increase in extreme wave heights and peak winds, often associated with storm events (Wang et al. 2004; Pryor and Barthelmie 2010).

11.3.3 Global Warming, Ecosystems and Climate Feedbacks

By conserving and sequestering C, ecosystems may contribute to alleviate the increase in natural hazards associated with global warming (Canadell and Raupach 2008; Bonan 2008). However, ecosystems may themselves be affected and transformed by global warming, especially by changes in rainfall patterns and temperatures (Cochard 2011a). One of the greatest feedback threats of climate change is posed by the thawing of permafrost soils in cold climatic regions (in Siberia, the Tibetan Plateau, the Himalayas and other high latitude or mountain areas), and the resulting decomposition and release of previously frozen organic C. Such releases are estimated at 0.8–1.1 Pg C year^{-1} which accounts for 40 % of global C emissions due to land use changes (Schuur et al. 2008; Schuur and Abbott 2011). Global warming may also affect forest ecosystems in temperate and tropical regions. Forests and most other C-rich ecosystems only occur in regions with sufficient rainfall and soil water storage capacities. In environments where mean annual precipitation is less than 1 m, forests are commonly replaced by other vegetation types and may only persist in land depressions where soil water is being concentrated and stored over the dry season (Whittaker 1975). More intensive and prolonged droughts (projected to increase from 1 % to up to 30 % of land area within the next 80 years) could lead to widespread tree die-offs in severely affected regions (Breshears et al. 2005; Burke et al. 2006). Through changes in albedo and evapo-transpiration this could lead to further feedback effects such as more severe heat waves (Meehl et al. 2007; Bala et al. 2007). There are already indications that droughts are affecting sensitive forest ecosystems at the periphery of forest distribution (Allen et al. 2011). Net primary production in forests also depends on rainfall as well as temperature, ranging from averages of about 2,200 g C m^{-2} year^{-1} in lush tropical rainforests (trees of more than 30 m height and mean woody biomass of around 45 kg m^{-2}) to 800 g C m^{-2} year^{-1} in the northern taiga forests (stunted trees of less than 15 m height and woody biomass of around 20 kg m^{-2}; Gurevitch et al. 2006). In dry and hot environments productive evergreen forests are typically replaced by other types of more drought-resistant vegetation, such as deciduous types of woodlands, sclerophyllous thickets, savannas, steppes, open grasslands and desert vegetation.

Warmer temperatures, heat waves and more droughts may affect forests and other ecosystems in various ways. Apart from direct drought-related tree die-offs, climate-driven species shifts and extinction of keystone species may disrupt and alter ecosystems. Forests may, thus, become much more vulnerable to the impacts of fires and invasive alien species by a combination of direct and indirect effects of climate change (Cochard 2011a). In pre-historic and historic times humans have been dramatically transforming many regions by the use of fire. For this reason some ecologists working in regions with long-lasting fire influences on the vegetation have termed *Homo sapiens* a 'fire keystone species' (Bowman and Murphy 2010). Various ecosystems, especially some tropical savannas and woodlands, and Mediterranean vegetation types are presumably of anthropogenic origin. If fire would be excluded over a long time period (dozens to hundreds of years), then some of these ecosystems would likely change to more closed vegetation types such as forests under the current climatic conditions (Sankaran et al. 2005; Cochard and Edwards 2011a; Bowman and Murphy 2010). In many regions climate change is expected to facilitate fire regimes and, therefore, push back forest ecosystems and open up the landscape to fire-dominated ecosystems (Allen et al. 2011; Podur and Wotton 2010; Moriondo et al. 2006; Dimitrakopoulous et al. 2011). In contrast, it has also been reported that rainforests have expanded into fire-dominated ecosystems in wet tropical regions of Australia and other parts of the world. This has mostly been attributed to increases in mean annual precipitation and to increased levels of CO_2 (Bowman et al. 2010; Banfai and Bowman 2006). C fertilization has equally been hypothesized as an explanation for the observed expansion of bush- and woodlands into savanna and grassland ecosystems in many parts of the dry tropics (Bond and Midgley 2012). In several regions, however, such expansions can also be traced to the development and intensification of livestock grazing systems (Cochard and Edwards 2011b; Tobler et al. 2003). The impact of grazing may become more important with rising human populations and more open and intensively managed landscapes.

Forest and bush fires themselves represent a potential hazard to humans and property. The 2009 "Black Saturday" bushfire in Australia, for example, killed 173 people and destroyed 2,298 houses in the suburbs of Sydney. This was mostly due to the circumstance that houses had been built too close to pyrophytic myrtaceous bushlands (Crompton et al. 2010). In Australia and other regions with Mediterranean climates, many forest and woodland vegetation types have evolved with fire over many thousands of years of human interference. Fire protection of these ecosystems may, thus, lead to an increased build-up of biomass and an associated heightened risk of intense bushfires during extended dry spells. In other regions, however, wholesale destruction of tropical non-pyrophytic forest ecosystems by logging and subsequent burning can lead to substantial consequences not only in terms of increased CO_2 emissions, but also in terms of regional and local climate changes and associated ecosystem transformations, including the emergence of fire susceptibility and risk.

11.3.4 Regional-Level Climate Change Mitigation Roles of Ecosystems

In addition to their role in C storage and sequestration, forests are important in mediating temperatures and maintaining weather patterns and water cycles at local and regional scales (Bonan 2008). The albedo of dense forests is typically lower than that of ecosystems with less dense vegetation and lower biomass (Pielke et al. 2007). For example, the mean albedo of forests in Brazil has been measured to be 0.134, whereas adjacent ranch land had a higher value of 0.180 (Culf et al. 1995). According to Bala et al. (2007), tropical forests are, however, particularly important to combat global warming not only through their capacity to store C but also by their function to promote sunlight-reflecting clouds via large-scale evapo-transpiration. In tropical regions, forests also tend to cool temperatures through the direct effects of evapo-transpiration. In boreal forests, winter temperatures are slightly increased as compared to an open snow field due to the difference in albedo and wind velocity. Overall there are, however, still many unresolved questions relating to the net climate forcing from albedo and evapo-transpiration processes (Bonan 2008; Jackson et al. 2008).

Forest ecosystems may be particularly important in regional water cycles since forest soils store water, and trees return water to the atmosphere via their large transpiration surfaces (Fig. 11.5a). Measurements by Leopoldo et al. (1995) and Kumagai et al. (2004) in the Amazon and Bornean forests, respectively, indicated that tropical humid rainforests recycle as much as 68–79 % of the rainfall water into the atmosphere. This water vapor then precipitates over other parts of the forests as it generally condensates faster in cool air over forests than over open areas. Tropical forests may, thus, function like a 'conveyor belt' whereby water is transported inland by the force of trade winds and via cycles of convective precipitation and evapo-transpiration from forests. The relative role of forests as a 'conveyer belt', and the potential changes in rainfall patterns after deforestation are highly debated. For example, Makarieva and Gorshkov (2007) suggested a central role of forests as a 'pump of atmospheric moisture', but Angelini et al. (2011) maintained that precipitation over large continental areas are only partly influenced but not controlled by vegetation and local water sources (see also Meesters et al. 2009; Sheil and Murdiyarso 2009; Bonnell 2010).

A 'conveyer belt' function could potentially be disrupted under the effects of global and/or regional climate change, especially if the forests are destroyed or degraded by human activities over large areas. Due to changes in local albedo and temperature regimes deforestation may result in less frequent but more variable rainfalls with smaller precipitation volumes in tropical regions (Malhi and Wright 2004; Berbet and Costa 2003; Coe et al. 2009; Dubreuil et al. 2012; Gash and Nobre 1997; Ghazoul and Sheil 2010). Some regions may already have experienced significant climatic changes in pre-historic times. For example, late Pleistocene fire-driven woodland destruction and associated desertification in Central Australia probably caused significant declines in rainfalls during the monsoon seasons (Miller et al. 2005). Similar large-scale effects of deforestation may have occurred in the monsoon regions of Asia. Extensive deforestation of China's Loess Plateau thousands of years ago may have lead to desertification and regional climate changes (e.g., a weakening of summer monsoons). It certainly resulted in extensive erosion of soil resources on hillsides, giving the color to the sediment-loaded mighty Yellow River (Dallmeyer and Claussen 2011; Zhao et al. 2010; Xiubin et al. 2004). Equally, deforestation in the Middle East and in the Mediterranean region may have transformed regional climates (van Andel et al. 1990). In more recent times, large-scale deforestation and the destruction of forested wetlands in parts of Africa, Asia, and South America may have led to similar effects (Cochard 2011a; Coe et al. 2009; Hesslerová and Pokorný 2010; Wösten et al. 2006). In Southeast Asia, deforestation

Fig. 11.5 Flood hazard mitigation functions of forests. (**a**) Tropical rainforests can absorb and store significant amounts of rainfall, and – via evapo-transpiration – recycle water back to the atmosphere (Amazon rainforest, Ecuador, Dallas Krentzel, http://creativecommons.org/licenses/by/2.0/). (**b**) Such functions are largely lost, and regional climates may change after deforestation (2001 Modis satellite image of mainland Southeast Asia. The lowland regions of Thailand are visibly outlined by the deforested area, NASA, wikipedia.org). (**c**) Tropical monsoon forests have been largely replaced by plantation crops. These are often affected by soil erosion problems during intensive rainfalls (cassava plantation in Thailand, photo credit: Neil Palmer, CIAT). (**d**) Flooding can be partially a result of the lost forest cover (Central Plains, Thailand, October 2011, Daniel Julie, http://creativecommons.org/licenses/by/2.0/)

has been predicted to result in a precipitation decline of 8 %, with a decline of up to 17 % in Indonesia (Hoffman et al. 2003). In the mountainous areas of Northern Thailand, however, Walker (2002) found no evidence for a decrease in rainfall and runoff due to logging, even though rainfall appeared to decrease within various catchments. This may be due to the strong maritime influence which creates a significant degree of climatic stability within this region and reduces the impact of forest clearing. Local climate changes may be exacerbated in and around urban centers. Planting of trees along roads and in parks can make a considerable contribution to mitigate against hydro-meteorological hazards, especially summer heat waves (DePietri et al. 2012).

11.4 Roles of Carbon-Rich Ecosystems in Mitigation of Floods

11.4.1 Flood Claims and Claims About Floods

Catchment-based flooding has always represented a major natural hazard to humans. Currently, it is affecting about 20 million people and claiming around 20,000 lives annually (Smith 2004). There appeared to be an increase in the number of flood disasters, casualties and affected people over the past decades, but this may be partially due to increased reporting and exposure of people and assets (Milly et al. 2002; Brody et al. 2007). In addition, damages of floods are expected to further increase in several regions (Feyen et al. 2011; Mirza 2010). According to Smith (2004, p. 189) the baseline is that "most countries have found it difficult – even impossible – to reverse the upward trend." High hazard exposure can largely be blamed for the high number of disasters. Alluvial areas in low-lying floodplains and near estuaries are commonly the most fertile and productive land areas available, and have, therefore, always attracted humans to settle. In the past, people have not been unaware of the risks of flooding. If the possibility existed they built their settlements on higher grounds rather than within floodplains and other potentially endangered areas. For example, most of the old villages in the Rhine Valley or along the fertile Linth and Magadino Plains in Switzerland are located along the foot of the adjacent mountains and hills. Only after rivers had been straightened and channelized, and adjacent wetlands had been drained new settlements were built within the original floodplains (Vischer 2003). In many cases and especially in advanced industrialized countries, it is the failure of such engineering 'ameliorations', respectively human over-confidence in the reliability thereof, which represent a major risk during extreme events. In extensive tropical floodplain areas traditional societies have also adapted to flood risks by building houses on stilts such as the traditional stilt houses in SE Asia or even on floating boats (Roshko 2011). Significant increases in human population in many regions of the developing world, however, have forced many poor communities to reclaim and occupy increasingly marginal lands in order to make a living. Over the last century many cities have developed in fertile flood-prone river deltas, and suburbs and industries have rapidly expanded into increasingly unsafe, marginal lands (Hara et al. 2004). Mega-cities located in highly vulnerable locations include, for example, Bangkok, Dacca, Jakarta, Manila, Shanghai, Mumbai, Karachi and Brisbane.

Several countries which were affected by gigantic floods in recent years have undergone extensive deforestation in the past decades. For example, more than 70–80 % of the initial forest cover was lost in Thailand, China, Indonesia and in the Philippines, and less than 2 % of the land area is now covered with forests in Pakistan (FAO 2011; Sodhi et al. 2007). On various occasions, claims have therefore been made that flood hazards have increased as a result of catchment-based environmental changes, in particular up-stream large-scale deforestation and land degradation (Deutsche Welle 2010; van den Honert and McAneney 2011; Bradshaw et al. 2007; Yin and Li 2001). As a matter of fact, logging bans and forestry laws in many countries (including Thailand in 1989, China in 1998, and Switzerland at the end of the nineteenth century) were strongly influenced by the impression that severe flooding had increased due to extensive deforestation in catchments (Hegg et al. 2004; Usher 2009; Lang 2002; Lu et al. 2001). Many such claims have, however, been called into question by scientists and practitioners (FAO 2005; van Dijk et al. 2009; Calder and Aylward 2006; Walker 2002; Tran et al. 2010). The two major reasons for criticism are (1) many claims are not sufficiently substantiated by valid data and scientifically-sound studies, and (2) the claims divert attention away from the primary issues determining flooding risks, i.e., the increasing exposure and vulnerability as outlined above.

There may be some truth in these criticisms, but occasionally the argumentation is likewise skewed and based on a weak foundation. For example, in an influential public report by the United Nations Food and Agriculture Organization (FAO) and the Centre for International Forestry Research (CIFOR) (FAO 2005, p. 3; cf. also The Economist 2005; Laurance 2007) entitled 'Forests and Floods: drowning in fiction or thriving on facts?' it was stated that "the inherent uncertainties of many scientific findings and the absence of long-term research are downplayed [by the media]." Notwithstanding, FAO (2005, p. 11) then assert that "sound science provides little evidence to support anecdotal reports of forest harvesting or rural land-use practices leading to lower-basin catastrophic floods. When it comes to prevention of major floods, the 'sponge' theory is a historical *erratum*." This statement and others by FAO (2005) may be regarded critically, especially if they are meant to 'dispel myth' and educate the public about 'the reality'. First, several of the few examples which are cited as 'sound science' by FAO (2005) are either outdated or only from temperate regions. Second, the central question is not whether floods can be completely prevented through forest management. The issue is to what degree floods can be abated by forest cover and how the flood hazards will change after large-scale deforestation. Thus, if sound data would indeed be unavailable, then it could conversely be stated with a precautionary logic that "neither sound nor poor science provide any real evidence to disprove the perception that wide-ranging deforestation (often associated with losses of soil resources, especially in the wet tropics) and land mismanagement have significantly contributed to increase the flood hazard."

Many studies now available point to the conclusion that C-rich ecosystems such as tropical and temperate forests do not play an unimportant role in mitigating flood hazards. Nonetheless, it is evidently not easy to draw coherent and general conclusions from the available literature. There appear to be considerable differences between regions, ecosystem types, and context-specific situations. Areas of complexity include (1) the effects of deforestation on local and regional precipitation (i.e., total rainfall and variability, and intensity of rainfall events), (2) the patterns of precipitation which lead to certain types of floods (e.g., longer-term high rainfalls, intensive short-term pulses, etc.), (3) the ways and degree by which the local vegetation-cum-soil system (the so-called 'sponge') mediates water flows under various types of rainfall events and terrain situations (i.e., water interception, infiltration, retention capacity thresholds of soils, discharge velocity, etc.), and (4) how local catchment-based effects relate to large-scale regional floods. Furthermore, the methods of data collection and assessment of various earlier studies have been criticized as inappropriate and misleading, and some conclusions drawn from them may, therefore, still be open to discussion.

11.4.2 Precipitation, Floods and the Climatic Role of Forests

Clearly, the primary determinant of flood hazards is the amount of rainfall over a catchment (i.e., the total amount during the wet seasons, rainfall intensity and duration during specific events, as well as the occurrence of extreme peak events; Bruijnzeel 2004; Richey et al. 1989; Bradshaw et al. 2007). Some of the largest floods in South Asia have been linked to excess monsoon rainfall periods (Kale 2012; Wang et al. 2011). A major question relating to floods is, therefore, how climate change and large-scale deforestation and forest degradation will affect regional and local rainfall patterns (Bonnell 2010; Mirza et al. 2003). As climate change is predicted to result in more precipitation, and more intense rainfall events in various regions, flood hazards are expected to increase accordingly. Various studies have noted possible increases in flood events due to climate change effects in the Northern Hemisphere, e.g., increases in extreme rainfall intensity during summertime, more winter storms and associated rainfall, and generally wetter winters in Europe and in other regions (Meehl et al. 2007; Palmer and Räisänen 2002; Dankers and Feyen 2009; Christensen and Christensen 2003). In major river catchments, increases in winter snow depth may lead to higher risks of flooding during springtime (Milly et al. 2002). In the monsoon regions of South and Southeast Asia warmer climates are generally expected to result in higher amounts of rainfall over large areas (Palmer and Räisänen 2002; Guhathakurta et al. 2011; Mirza 2010).

If both the stable weather regime and the 'conveyor-belt' function of forests are disrupted over large areas, higher temperatures, more heat waves and less rainfall may lead to increased hazards of droughts and not necessarily to higher flood frequencies and pulses in river systems. Decreased rainfall effects can also occur at local scales due to decreased moisture interception and higher temperatures.

For example, annual streamflow from the Rio Pejibaye in Costa Rica reportedly decreased by over 300 mm after complete and fast deforestation within 5 years (Fallas in Bruijnzeel 2004). Precipitation and river discharge also appeared to decline in parts of Central Kenya due to extensive deforestation (Hesslerová and Pokorný 2010). Bruijnzeel (2004) listed several studies from Central America where logging of tropical cloud forests lead to apparent decreases in water flows from catchments.

11.4.3 Flood Abatement: Forests as 'Sponges'

Provided that mean rainfall patterns do not change over an area, the capacity to retain water will probably always be higher in a 'pristine' forest as compared to forest plantations (commonly lacking a protective understory and soil cover) or agricultural fields that replaced the forest within the same landscape (Fig. 11.5). With regards to flooding risks there are three central questions. First, how much water can different types of land cover hold in the soils? Second, how much water can be evaporated via surfaces or vegetation transpiration? And third, how do these differences relate to the flood hazards within a catchment, given certain rainfall patterns?

A 'sponge function' of forests may be defined as an 'ecosystem capacity to retain water'. However, in the context of flood hazards an extended definition may be a 'general capacity to keep water from flowing downstream into an open watercourse, especially during intensive rainfall events'. The focus may not only lie on 'inert' physical parameters of soils and plant biomass but also on the 'active' function of soils and plants to re-allocate water to the atmosphere (the gaseous part of the 'sponge') via evapo-transpiration.

Tropical rainforests, in particular, have a high evapo-transpiration rate with as much as >60 % of the water recycled back to the atmosphere (Fig. 11.5a). Kumagai et al. (2004), for example, estimated annual transpiration rates of 1,193 mm and evaporation rates of 362 mm in rainforests of Sarawak, Borneo. This was equivalent to about 61 % and 18 % of annual rainfall (1,970 mm) at the site, respectively. In lowland tropical rainforest in Brunei, 0.7 mm h^{-1} may be lost during rainfalls by evaporation alone (Dykes 1997). Such figures indicate that evapo-transpiration functions of rainforests cannot be dismissed as a factor in mitigating flood hazards. Apart from buffering and recycling water during rainfall events evapo-transpiration functions are particularly important to increase the soil water storage capacity prior to major rainfall events, in particular, between such events. This decreases the risk of development of surface water flow, and, therefore, also lowers the risks of soil erosion and flooding (Sidle et al. 2006; Bruijnzeel 2004).

In a comment on a study by Bradshaw et al. (2007) van Dijk et al. (2009, p. 112) noted that "local floods in tropical regions (e.g., an event of one or a few days with >10 years return period) are likely to be caused by a rainfall event well in excess of 100 mm. For such storms, the difference in interception loss between forest and non-forest vegetation is unlikely to exceed a few percent of rainfall and, therefore, is not likely to be a significant factor." Local floods and especially large regional floods (such as the Thailand floods in 2011 and the Pakistan floods in 2010) may, however, be the result of a succession of strong rainfall events (e.g., resulting in slower cumulative flooding at the conjunction of several streams) rather than just a single or a few extreme rainfall events (e.g., resulting in fast floods within small catchments). If rainfall is spread over longer time periods, more water volume can potentially be 'pumped away' via evapo-transpiration in forests. Compared to grassland or other vegetation, rainforest trees have a disproportionately higher capacity to move water from the 'buffer compartment' (i.e., the forest soil stores and vegetation surface) to the 'air compartment' of the 'sponge'. In an experiment in a lowland rainforest in northeastern Costa Rica, Parker (1985, in Bruijnzeel 2004) studied soil moisture contents during the dry season in pristine forest and opened gaps. During 50 days soil moisture decreased by almost 40 % from about 370 to 230 mm in pristine forest. In a large forest clearing the decline was only 9 % from 370 to 320 mm. The differences became less pronounced as secondary vegetation started to close the gaps. Similar patterns were also observed in many other places where forests were opened up and then allowed to regenerate (Bruijnzeel 2004). Evapotranspiration rates can also significantly increase in savanna ecosystems invaded by bushland. For example, in coastal Tanzania wet-season transpiration rates were estimated to be five times higher (0.1 l m^{-2} day^{-1}) in woodlands dominated by *Acacia zanzibarica* as compared to open grassland (0.02 l m^{-2} day^{-1}). This was mostly because the transpiration rates of the trees (0.143 l m^{-2} leaf area day^{-1}) were 3.4 times higher than that of grasses (0.042 l m^{-2} leaf area day^{-1}), whereas the actual leaf surface area indices were not markedly different between woodlands and grasslands (Cochard and Edwards 2011b; Cochard 2004).

In undisturbed forests the infiltration capacities of soils are typically high. Soils may absorb the water volumes delivered by most rainfall intensities. The generation of high and potentially hazardous river discharges largely depends on factors such as soil saturation and associated overland flows in depressions, or rapid subsurface flows through porous substrates (Bruijnzeel 2004). These factors in turn depend on soil characteristics and topography, as well as on the amounts of water which enter the soil 'sponge' per unit time. Especially in the tropics, mountainous, steep terrains are often prone to generate high discharges and fast flooding at the valley bottoms as orographic condensation on mountainsides may produce heavy rainfalls, and water may surface and accumulate fast at the valley bottoms. In such terrains the construction of weirs is a commonly used engineering tool to diminish the flow of water and debris (Di Pietro et al. 2011). Also, roots of different types of tree species can influence the rainfall-runoff response in certain areas. For example, Jost et al. (2012) studied the soil moisture dynamics and rainwater runoff processes at two sites with similar soil type but different tree species cover, i.e., Norway spruce (*Picea abies*) and European beech (*Fagus sylvatica*). After application of sprinkling they found that a water table developed at approximately 30 cm depth at the spruce site whereas water was directed to deeper soil horizons along the coarse roots of beech. These differences may have significant influences on overflow generation at the sites. Fritsch (1993, in Bruijnzeel 2004), for example, demonstrated in a study in ten undisturbed rainforest catchments how stormflow was influenced by soil characteristics and antecedent water storage. In catchments with a high groundwater table average volumes of stormflows were as high as 34.4 % of incident rainfall whereas on well-draining soils the volumes were about 7.3 %.

Various studies conducted at forest sites in temperate regions have not found any strong evidence for increasing flood hazards following logging within catchments (Hewlett 1982 and Bowling et al. 2000 in van Dijk et al. 2009; FAO 2005; Bruijnzeel 2004; Eisenbies et al. 2007; Cosandey et al. 2005). This may be due to specific site effects as discharge peaks are mostly controlled by topographic and soil parameters. Under certain conditions, flood generation from forested catchments may even surpass that of adjacent non-forested catchments. For example, within the Spergelgraben catchment in the Swiss Pre-Alps a strongly storm-damaged sub-catchment yielded generally more runoff than an adjacent catchment with intact forest. However, during short and very intensive rain showers discharge peaks were higher from the forested area as compared to the windfall areas. This was mostly explained by differences in geomorphic characteristics and the spatial distribution of the moist to wet forest site types (Badoux et al. 2006; Stähli et al. 2011). The findings of various studies on the effects of deforestation on floods were, however, also subject of methodological criticism. Alila et al. (2009, p. 1) noted that tests used extensively for evaluating the paired data of floods and meteorological input lead “to incorrect estimates of changes of flood magnitude because neither the tests nor the pairing account for changes in flood frequencies.” On this basis Alila et al. (2009, p. 1) called “for a re-evaluation of past studies and the [] paradigm that shaped our scientific perception of the relation between forests, floods, and the biophysical environment.”

There is nonetheless increasing evidence that deforestation does increase frequencies as well as intensities of flooding in catchments, especially in the tropics. According to Bruijnzeel (2004) post-logging upsurges in flooding response for small rainfall events tend to be in the range of 100–300 %, but may be less than 10 % for very large rainfall events. In addition, as vegetation quickly recovers on the logged sites flood levels may soon (sometimes within only a few years) return close to pre-logging conditions (Bruijnzeel 2004). On the other hand, if logged-over forests give way to other land uses such as agriculture or infrastructure development (as is often the case in the wet tropics) further soil erosion and compaction may exacerbate the increases in post-logging peakflows. Such increases due to soil degradation may, however, be less pronounced if the soils were already shallow and of poor hydraulic conductivity before logging, such as in clayey or rocky areas (Bruijnzeel 2004; Sidle et al. 2006).

In general, tropical forests differ from temperate forests in terms of (1) the amount and intensity of rainfall (overall annual averages and daily rainfall pulses), (2) the associated functional ecology of tree species (transpiration potential and productivity), and (3) the function, stability and resilience of soil resources (SOM and nutrient content, and susceptibility to erosion). Several studies in areas where rainfall has remained constant revealed increases in flooding rates from tropical deforestation. For example, in a paired experiment in the Eastern Amazon Basin comparing two small catchments of <0.1 km^2, Moraes et al. (2006 in Coe et al. 2009) found that the runoff to precipitation ratio increased from 3 % in the forested catchment to 17 % in the pasture catchment. In a similar experiment in the Central Amazon Basin, Trancoso (2006 in Coe et al. 2009) found an increase from 21 % in a forested catchment to 43 % in the paired pasture area (both catchments were about 1.2 km^2 in area). Similar increases were observed at the large scale. For example, in the 82,632 km^2 watershed of the Araguaia River in east-central Brazil increases in discharge of about 25 % were observed from the 1970s to the 1990s. Computer simulations suggested that about two thirds of the increase was due to deforestation of about 55 % of the area and one third was due to climate variability (Coe et al. 2011). Similarly, after deforestation of 19 % of the 175,360 km^2 Tocantins River watershed in the eastern Amazon Basin, Costa et al. (2003) found increases in the wet season river discharge of about 28 %. In addition, discharge peaks were measured about 1 month earlier. According to Bathurst et al. (2011) data of large catchments in Costa Rica (131 km^2) and in Chile (94–1,545 km^2) suggested that percentage change in forest cover must exceed 20–30 % in order to provoke a measurable response in peak flows.

Increases in river discharges do not necessarily lead to more flooding hazards in a linear fashion. Undoubtedly, however, large flood events will tend to increase in frequency, intensity and duration as soils remain thoroughly wetted throughout the wet seasons and river channels are filled up early in the wet seasons. Very large floods (e.g., the floods caused by monsoon rainfalls in 2011 in Thailand and Australia, and in 2010 in Pakistan) are typically the result of a large and persistent field of rainfall during the wet season. Bruijnzeel (2004, p. 205) noted that "under extreme conditions, basin response will be governed almost entirely by soil water storage opportunities rather than topsoil infiltration capacity or vegetation cover." This is most probably the case under persistent rainfalls, when transpiration rates of plants also tend to be much reduced. In fact, in a study of catchments with various forest cover in Central and South America Bathurst et al. (2011) found that differences between forested and deforested catchments virtually disappeared for extreme flood events of a return period of about 5–10 years, whereas forests had measurable and higher flood mitigation effects for smaller flood events. Over large areas in the tropics rainfalls do rarely persist over many days, but are frequently interrupted by short spells with sunshine when the plants' rates of productivity may be highest. The potential flood mitigating effects by forest ecosystems' transpiration capacities during large rainfall events has as yet not been thoroughly investigated. It appears unlikely, however, that it would be entirely negligible over areas of thousands of square kilometers.

Technically, there should be no fundamental difference regarding the vegetation effects in small catchments compared to large watersheds. By nature, large watersheds tend to be significantly more complex, however, and hence a precise and accurate assessment of vegetation effects will also be much more intricate and challenging (Gamble and Meentemeyer 1996). High stormflows generated in an upstream catchment may be attenuated further downstream. Essentially all flows from tributary catchments combine to generate a flooding risk along a certain section of interest (Kourgialas and Karatzas 2011; Mascarenhas and Miguez 2011). Therefore, if potentially vulnerable sections along large rivers are assessed (e.g., a town), sufficiently detailed data should optimally be available on rainfalls and river discharges as well as land use cover, land condition and wetland volumes within all upstream tributary river catchments. In order to make detailed assessments and compute reliable models, long-term data series would be very helpful. Such data are, however, rarely available even in many developed countries. It is therefore unsurprising that various studies focusing on large and rather complex river systems in the dynamic monsoon zones of Asia have yielded fairly inconclusive results regarding the effects of deforestation and land degradation (Mirza et al. 2001; Wilk et al. 2001; Marston et al. 1996; Hofer 1993).

11.4.4 'Sponge' Erosion Risks by Land Use Intensification

In most forests, soils are characterized by a relatively high infiltration capacity and hydraulic conductivity which is supported by organic matter on the soil surface and in the upper soil layers. If vegetation is lost, splash erosion during intense rainfalls often affects the upper soil layer resulting in decreased water infiltration (Sidle et al. 2006). It appears obvious, therefore, that large-scale deforestation of pristine tropical forest – such as has happened in many countries of Southeast and South Asia during the later half of the twentieth century (Cochard 2011a; Sodhi and Brook 2006) – would have an effect on the flows of large rivers in the region, especially during the monsoon periods.

Forest removal is often followed by poor land cultivation practices and rampant soil degradation (Sidle et al. 2006; Fig. 11.6), and there are signs in several regions that river discharges are increasing as indicated by widening river channels and eroding river banks (Bruijnzeel 2004; Li et al. 2007). There are, however, differences in land uses and management with differences in run-off during rainfall events. For example, according to Ziegler et al. (2009) swidden cultivation in montane mainland Southeast Asia does not necessarily lead to extensive soil erosion and degradation. Thus, no significant hydrological impacts are observed provided that cultivation is performed in a 'traditional subsistence-based' manner with organic management and extended fallow periods in between. In many regions, however, swidden practices are being abandoned for permanent cash cropping and monoculture plantations which require high inputs of water irrigation, fertilizers and pesticides. These intensive systems are prone to cause comparatively high soil erosion and compaction (with high irrigation and no fallowing to allow regeneration of vital soil properties) which in turn may lead to higher overland flows and risks of downstream flooding (Ziegler et al. 2009; Bruijnzeel 2004). In densely populated regions, agricultural expansion to steep hillside areas may also exacerbate erosion problems. For example, in upland farming areas in the Himalaya annual soil losses may be up to a hundred times higher in fields on highly sloping terraces (6–64 Mg ha^{-1} $year^{-1}$) compared to fields on low sloping terraces (0.3–0.66 Mg ha^{-1} $year^{-1}$; Sen et al. 1997).

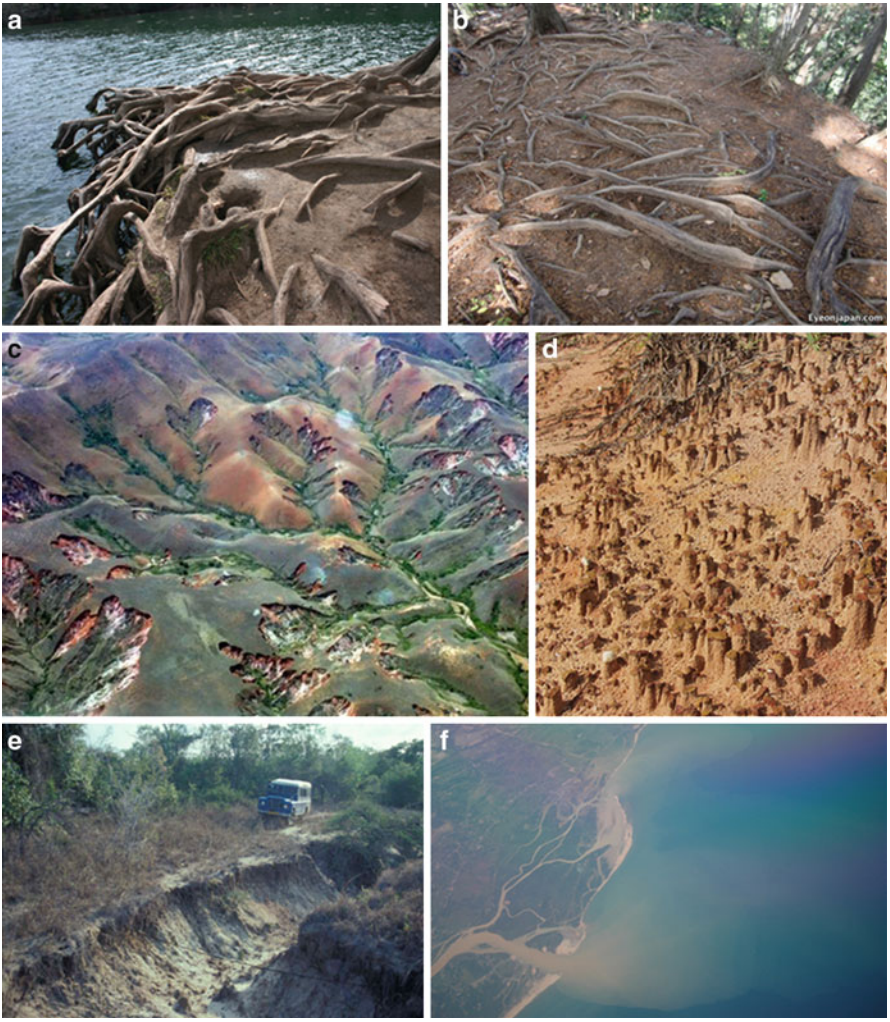

Fig. 11.6 Soil protection functions of vegetation. (**a**) Riparian forest vegetation plays an important role in river bank stabilization during floods (roots of *Taxodium distichum*, Austin, Texas, Wing-Chi Poon, wikipedia.org). (**b**) Equally, tree roots stabilize soils on mountain slopes (forest in Japan, Richie Johns, eyeonjapan.com, http://creativecommons.org/licenses/by/2.0/deed.de). (**c**) Deforestation of tropical rainforests on hillslopes often leads to fast soil nutrient depletion (vegetation cannot regenerate easily) followed by deep soil erosion and shallow landslides (deforested area in Madagascar, photo credit: Rhett A. Butler, mongabay.com). (**d**) Excessive removal of grass cover and soil compaction (e.g. on pastures) can lead to severe splash and sheet soil erosion (Soil Science, http://creativecommons.org/licenses/by/2.0/). (**e**) On compacted areas water runoff during a heavy rainfall can lead to deep gully erosion (eroded disused dirt road in coastal Tanzania, R. Cochard). (**f**) The eroded material often ends up in coastal areas, where it may affect sensitive offshore ecosystems such as coral reefs and seagrass beds (coastal sediment plume from Kelantan River, Malaysia, R. Cochard)

In well-managed agricultural systems vegetation ground cover may be maintained due to inputs of fertilizers and manure. However, in particularly sensitive areas, such as in many tropical regions, soil productivity may decline rapidly as a result of accelerated erosion during strong rainfall events. Much of the nutrient capital of the soils is concentrated in the organic-rich topsoil. After loss of much of the nutrients and organic matter plant cover may become thinner and soils may further disintegrate due to the direct impacts of rainfall (i.e., increased splash erosion) and higher surface water runoff volumes and dynamics (i.e., increased risks of sheet, rill and gully erosion; Fig. 11.6c–f).

In various studies in savanna rangelands in Australia it has been demonstrated that runoff as well as sediment loss is significantly influenced by ground cover. Bartley et al. (2006) found that runoff was 6–9 times higher, and sediment losses up to 60 times higher on hillslopes with small patches bare of vegetation (so-called 'scalds') as compared to similar hillslopes which were completely covered by vegetation. Most of the sediment which was primarily composed of fine material was lost during initial flushing. In addition, the authors suggested that initial 'historical' erosion rates would have been much higher, and that erodible materials were already largely exhausted. Similarly, Silburn et al. (2011) found that runoff was very high (30–50 % of rainfall) in locations with low grass cover, whereas on sites with >50 % grass cover it was much lower (about 6 % of rainfall). If rangeland vegetation was allowed to recover from grazing, higher vegetation cover resulted in a lower level of runoff during the first rainfall event and lower levels of sediment losses (Bartley et al. 2010). Erosion rates were not only related to vegetation cover but were significantly related to the patch size of bare area and the slope gradients (Silburn et al. 2011; Ludwig et al. 2007). Analyzing data sets of water quality from 514 different locations in Australia Bartley et al. (2012) indicated that the agricultural land uses causing the highest levels of total suspended sediments (TSS) in water bodies were horticulture (3,000 mg l^{-1}), dryland cropping (2,000 mg l^{-1}), cotton (600 mg l^{-1}), and grazing on native pastures (300 mg l^{-1}).

In addition to losses of vegetation cover, soil compaction by trampling (e.g., by livestock) or the impact of car tracks decreases water infiltration into soils. Runoff from logging tracks may substantially contribute to total runoff in an area (more than 50 % in some cases), often resulting in deep, irreversible erosion and contributing to flooding downstream. On hillslopes, road tracks and compressed soils can intercept subsurface flows. This may lead to increased levels of runoff and very high erosion rates or even surface landslide impacts (Woldie et al. 2009; Ziegler et al. 2007; Negishi et al. 2008; Fig. 11.6e).

11.4.5 Flood Abatement Ecosystem Services in Floodplains

In order to understand catchment dynamics and flooding risks forests and other types of natural or human-modified ecosystems should be considered in a systemic and functional way. Trees can only grow on an appropriate soil medium whereas soil formation (and especially the input of organic matter) is facilitated by the presence of trees and other types of vegetation. If the vegetation is lost, replenishment of organic matter by the input of tree litter as well as other biotic processes will significantly slow or cease. Soils will become more vulnerable to the risks of erosion, especially during intense rainfall events. The degradation or loss of the soil resources in turn will slow or impede the vegetation regeneration process. Therefore, the 'transpiration' capacity of the ecosystem may be permanently hampered. In Thailand, the highest rates of conversion of forestlands to intensive agriculture have not occurred in the montane areas but in the lowlands where only few pockets of native monsoon forests remain (Fig. 11.5b, c). According to Sthiannopkao et al. (2007) such land transformation has, for example, lead to severe soil erosion in the Phong River watershed, especially during wet-season storm events. During the dry season farmers often have suffered droughts on higher lands and many have, therefore, moved to alluvial areas where they cleared the vegetation and cultivated the land, exacerbating problems of soil erosion and flooding dynamics (Sthiannopkao et al. 2007).

Alluvial soils are typically highly productive for agriculture and, therefore, floodplain forests are often the first to be converted to cultivation fields. Such forests – especially riparian and riverine forests – are, however, important for stabilizing river banks and adjacent overflow lands during intensive flood events (Fig. 11.6a). According to Hubble et al. (2010) certain native species of riparian forest trees in Australia have rooting systems which are often deeper than 5 m, and sometimes even more than 20 m. For these and other reasons native riparian forests are particularly important to reduce the likelihood of riverbank erosion by mass failure (Hubble et al. 2010; Langendeon et al. 2009). Riparian vegetation can provide important river stability and ecosystem resilience against events of high peak discharges, but river banks may eventually erode if river discharge increases permanently. In contrast, under rather stable conditions riparian plants may act as ecosystem engineers by trapping and stabilizing sediments and organic matter, thereby delineating the river bed and forming land of a certain 'sponge' quality (Gurnell et al. 2012; Langendeon et al. 2009). Along headwater streams, riparian vegetation may buffer the effects of hillside deforestation and land degradation (i.e., resulting sediment and wood loads) (Gomi et al. 2006). Further downstream, riparian vegetation may also contribute directly to flood abatement by influencing and controlling water flows (e.g., by increasing flow resistance and preventing overbank flows) (Tabacchi et al. 2000; Järvelä 2004). Over large areas water storage and water return to the atmosphere may also significantly contribute to the abatement of flood levels (Tabacchi et al. 2000). Pristine vegetation growing along soil levees may play an equally important role of stabilizing those levees during high floods. For example, several unstabilized levees broke and caused widespread inundation during the 2011 floods in Central Thailand and during Hurricane Katrina in 2005 near New Orleans. Extreme flood flows are often underestimated, and under such conditions pressure on levees may be particularly high as levees reduce the area open to storage of flood waters. In addition, floodplain constriction may lead to reduced flow velocities and magnify discharge volumes above the confinements (Heine and Pinter 2012).

Riparian and wetland plant communities are highly susceptible to the invasion of alien plant species, especially when disturbed (Richardson et al. 2007; Mortensen and Weisberg 2010; Zedler and Kercher 2005; Weber 2003). For example, along the Mekong River and in other parts of Southeast Asia (as well as in tropical Africa and Australia) exotic *Mimosa pigra* invades wetlands, rice fields on alluvial plains and disturbed riparian forests, sometimes forming monocultures covering hundreds of square kilometers, and potentially displacing native plant communities in the long term. This may lead to associated bio-physical (e.g., increasing river bank erosion via displacing deep-rooted trees), hydrological and social consequences in the affected areas (Rijal and Cochard 2013). In North America, alien invasive *Tamarix* spp. represent a particularly serious problem along river sections where flows have been regulated by up-stream dams (Stromberg et al. 2007). In addition, invasions by exotic vines have the potential to smother riparian vegetation in many tropical regions (Radford et al. 2008).

11.4.6 Flood Abatement Ecosystem Services of Wetlands

Wetlands – natural or constructed – are particularly important for water flow ‘regulation’ throughout a catchment. During peak discharges wetlands serve as water storage areas, whereby wetlands situated in downstream areas tend to be more effective in reducing downstream flooding than wetlands located upstream of the potential flooding area (Smakhtin and Batchelor 2004; De Laney 1995; Ogawa and Male 1986). As the construction of levees has often proven to be an inadequate measure to contain floods, the restoration of lost wetlands has been suggested to improve the situation in several floodplains. Hey and Philippi (2006), for example, estimated that the most extreme floodwaters which have affected the city of St. Louis in 1993 would have been entirely buffered by wetlands which covered only about half of the original wetland area in the upper Mississippi Basin in 1780. According to Tockner and Stanford (2002), 90 % of the floodplains in Europe and North America are already cultivated and, therefore, ‘functionally extinct’. Floodplains in developing countries are similarly under increasing pressure due to rising populations, local climatic changes and transformations of the hydrological cycles, the degradation of riparian habitats, the increase in the discharge of pollutants, and development pressures, including the construction of hydropower dams and river basin ‘ameliorations’ (Jacobs 2009).

Peat wetlands are of particular significance for both C storage and sequestration, and flood mitigation (Holden 2005). Peat soils represent a most formidable ‘sponge’ system. Soils are composed mainly of porous but fairly cohesive organic material which may contain over 95 % water by volume at saturation (Boelter 1964). However, similar principles apply to peatlands as to other wetlands. The flood mitigation roles may only be really valuable in downstream areas. For example, most of the peat wetlands which cover about 13 % of the United Kingdom are located in the headwater areas of catchments. The flood abatement role of these ecosystems is limited since they act mainly as surface flow systems during heavy rainfalls (Haigh 2006). If such peatswamps in headwater areas are being drained this does not generally lead to any negative effects regarding flood hazards (Holden 2005). In contrast, Southeast Asian peat soils developed mostly downstream in extensive coastal forested floodplain areas as early as 30,000 BP (Page et al. 2004). The soils of many of those tropical peat swamp forests are up to >20 m deep and, therefore, constitute one of the largest ecosystem C stores (Phillips 1998). Evidently, they also represent formidable 'sponges' which absorb huge volumes of water and can also mitigate against peak discharges in those tropical areas. While their mitigation role against very large floods may be limited, the important thing to note is that the flooding hazards may exacerbate when large tropical peat swamps are destroyed. This has already happened in many places in Borneo and Sumatra. Logging of peat swamp forests is often followed by burning of peat soils and the conversion to paddy fields with often catastrophic consequences for the local ecosystems and massive emissions of GHGs (Page et al. 2002; Wösten et al. 2006; Hadi et al. 2005). Even without fire, logging tends to be highly detrimental to peat swamp forests. Subsequent drainage of the soils is typically followed by peat oxidation and associated annual land subsidence of around 2 cm (as measured in a peat swamp area in peninsular Malaysia; Wösten et al. 1997). In many cases this means that the land is lost. The area of open water may increase more than fivefold. Where surface flooding persists lakes are created where no plants can grow and peat can be replenished by litter fall (Wösten et al. 2006, 2008). Subsidence of peats can constitute a particular problem in coastal areas where catchment-based floods may be exacerbated by ocean spring tides which prevent the floodwater to drain into the ocean.

11.4.7 Flood Amplification Risks by Siltation of Waterways

The cumulative effects of agricultural land reclamation and 'corrective' measures – deforestation, stream channelization, levee construction, field terracing, and drainage channels in wetlands, etc. – have, thus, reduced the ability of most watersheds to absorb water, and detain soil resources and sequester C (Atapattu and Kodituwakku 2008; DeLaney 1995). Erosion from agricultural lands as well as increasing instability of river beds (sometimes exacerbated by the destruction or degradation of Riparian vegetation and wetlands) typically leads to high sediment loads in rivers (Fig. 11.6f). Sediment loads settle in calm sections of streams and can lead to raised river beds. This in turn can result in increased irregular stream dynamics and flood risks in floodplain areas (Dai and Lu 2010; Dudgeon 1992). For example, between 1970 and 2000 the forest cover in the upper reaches of the Yangtze River in China has been reduced by half and the area exposed to severe erosion has doubled. During the same period the storage capacity of downstream wetlands and floodplains has significantly decreased as lakes have been reduced in size due to land reclamation and severe siltation, and levees had been built that restricted the flood discharge capacities (Yin et al. 2007; Yin and

Li 2001; Zong and Chen 2000). This significantly contributed to the extreme floods in 1998, when the recorded water levels in the middle of the Yangtze floodplain basin were much higher than the previous historical maximum even though the precipitation over the catchment and the river discharge from the upper basin did not exceed the historical maximum (Zong and Chen 2000). Similar problems exist in many other parts of Asia and in the tropics. For example, according to Lorsirirat (2007) the annual sediment volume discharged into Lam Phra Phloeng Reservoir in Thailand was about 2.23 million m^3 per year during 1970–1983 when 74 % of the forest area in the catchment (820 km^2) was logged. During the period 1991–2000 when remaining forests remained stable and some areas (about 5 %) were reforested the annual sediment discharge was much lower, i.e., 0.37 million m^3 (Lorsirirat 2007).

11.4.8 Flood Amplification Risks by Land Subsidence

Flood risks in watersheds have also increased due to effects of land subsidence. For example, in the North China Plain over 70 % of the fields are irrigated. Due to excessive extraction of groundwater for irrigation the water table declined on average from about 7.2 m depth to 11.5 m between 1983 and 1993. This has lead to subsidence of vast tracts of land overlying cones of depression where large amounts of water have been extracted (Changming et al. 2001). How lowered groundwater levels have led to land subsidence has also been demonstrated in many other regions, especially in extensive floodplain areas. The levels of subsidence may also vary according to soil types. Land subsidence commonly represents a permanent land damage which is often associated with increased risks of flooding because of the lowered elevation, and decreased capacities for groundwater storage and recharge (e.g., lowered water infiltration) (Syvitski et al. 2009; Chen et al. 2010; Xue et al. 2005; Mori 2011; Lixin et al. 2010).

Land subsidence is a particular problem in large metropolitan areas located on former floodplains. Metropolitan areas not only experience excessive groundwater extraction for household and industrial uses, but the groundwater recharge is further diminished (and risk of flooding is increased) due to increased ground sealing by building and road surfaces (Mori 2011). In the early 1980s, for example, Bangkok has been subsiding at annual average rates as high as 0.12 m, and in some locations groundwater tables had been lowered by as much as 65 m (Phien-wej and Nutalaya 2006). Similarly, in Changzhou City in the Yangtze River Delta region annual subsidence rates of 0.147 m were measured during the 1980s. This, however, decreased to only 0.01 m $year^{-1}$ after groundwater extraction was restricted in 1995 (Wang et al. 2009). According to Rodolfo and Siringan (2006) recurrent flooding in Manila has been mostly attributed to global sea-level rise by Philippine government sectors, but a worse problem is actually land subsidence by several centimeters to more than a decimeter per year as well as a rapid expansion of the mega-city.

11.5 Mass Movement Hazard Mitigation Roles of Carbon-Rich Ecosystems in Mountainous Areas

11.5.1 Landslide Risks on Steep Inclines

Landslides are naturally occurring hazardous phenomena in steep terrain which are, however, often induced or augmented by the loss or degradation of vegetation (Montgomery et al. 2000; Fig. 11.7). Landslides are fast movements of materials (soil and/or rock) along slip surfaces. The main determinants of landslide risks are topography (mostly slope inclination), the weight of the soil (and/or rock) mass, and the 'rheological properties' of the mass (i.e., the properties referring to the deformation and flow of matter, including elasticity and viscosity). This determines mass stability as well as the dynamic development of the event once the slide is triggered (Smith 2004; McGuire et al. 2002; Casadei et al. 2003). The rheological properties are typically modified by soil water. Strong rainfalls and sometimes earthquakes are, therefore, the main triggers of landslides (Sidle 2007; Guzzetti et al. 2007; Gabet et al. 2004; Peduzzi 2010). In addition, road construction along hillsides may often amplify the risks of landslides due to water runoff from roads which increases pore water pressures in adjacent soils (Mirus et al. 2007).

Landslides are an under-recognized threat for two reasons. The impacts of single landslides are typically confined and may be comparatively small. On the other hand, when landslides occur *en masse* they do so in conjunction with larger hazards such as tropical cyclones and earthquakes. Hence, the effects of landslides are often not explicitly recorded and recognized in reports documenting regional disaster events. Especially in mountainous developing countries the overall impact of landslides can, however, be considerable (Dai et al. 2002; Jones 1992; Rautela and Paul 2001; Nadim et al. 2006). Shanty-towns of large and fast-growing cities such as Rio de Janeiro, La Paz, Quito, Caracas, Medellín, Tegucigalpa, Chongquing, Kathmandu and Kabul and many other cities are increasingly expanding onto unstable slopes along the surrounding fringes of the cities (Fig. 11.7a). Poor inhabitants are often putting additional pressures on remaining vegetation and soils on steep slopes. In Venezuela, a multiple occurrence of landslides in 1999 may have killed up to 50,000 people and caused damages of around 10 billion US$ (Swiss Re 2000). Mountainous areas along the Pacific 'Ring of Fire' (and especially in poor countries of Central and South America, and in Southeast Asia) are particularly at risk of landslide impacts, due to a combination of steep terrain, heavy rainfall and cyclone exposure, earthquake risks, rock types, and increasingly rapid land use changes (including deforestation) and high population densities (Bommer and Rodríguez 2002; Restrepo and Alvarez 2006; Glade 2003; Nadim et al. 2006). With increasing exposure and continuing high rates of deforestation in many of these countries landslide hazards appear to be on the rise on a global level (Smith 2004; McGuire et al. 2002). Severe multiple landslide events in mountainous areas may not only affect people

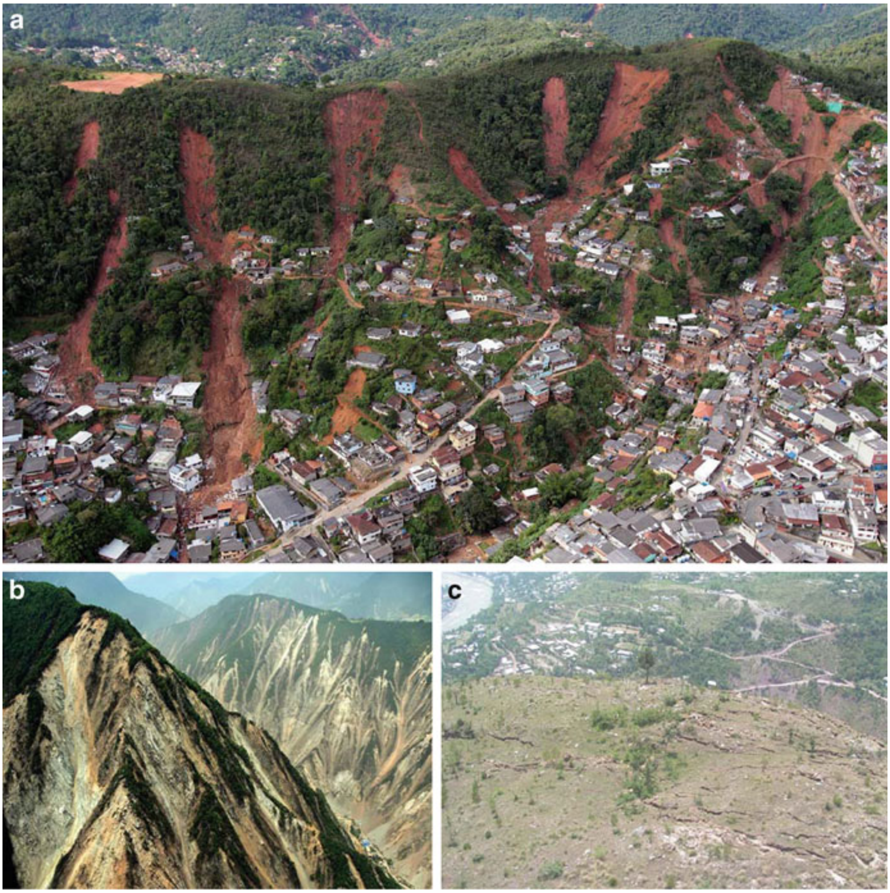

Fig. 11.7 Landslide hazard mitigation functions of forests. (**a**) In many developing countries cities are increasingly expanding into landslide hazard zones. On steep slopes the risk is particularly high during heavy rainfall events, especially in places where the vegetation has been degraded (shallow landslides and 'mudslides' near Teresopolis, Brazil, following heavy rainfall in January 2011, Keystone/EPA/Antonio Lacerda). (**b**) The main determinant of landsliding is slope inclination, but vegetation can provide significant soil reinforcements (landslide scars on mountain slopes after the 2008 earthquake near Wenchuan, China, photo credit: Runqiu Huang, Chengdu University of Technology). (**c**) Mountain slopes can pose a continuing threat of landslides after earthquake events (cracks on an unstable slope resulting from the 2005 Kashmir earthquake, Muzzafarrabad, Pakistan, photo credit: David Petley, Durham University)

directly but may also incur significant losses of productive land, and affect important ecosystem services in watersheds, including the siltation of waterways (Sidle et al. 2006; Peduzzi 2010; Philpott et al. 2008; Rautela and Paul 2001; Restrepo and Alvarez 2006; Glade 2003).

11.5.2 Landslide Mitigation by Forests and Other Types of Vegetation

Deep-seated landslides may result from mostly geologically controlled phenomena, such as surface tensions in mountainous areas, and deep cracks and fractures in the soil mantle or bedrock. Such landslides may eventually be triggered after a gradual deep buildup of pore water pressure over many weeks during an intense rainy season. The influence of vegetation (positive or negative) on deep-seated landslides (i.e., deeper than about 3 m) may thereby be rather limited since tree roots are concentrated in the upper soil layers and may reach only several meters deep (Sidle et al. 2006; Bruijnzeel 2004).

In contrast, vegetation may be linked more directly to the risk of shallow surface landslides. Vegetation plays a crucial role in forming and stabilizing soils with roots, and enriching soils with organic matter (Fig. 11.6b). *Vice versa*, pristine stands of forests can only find a foothold in a sufficiently deep and rich soil substrate. If such soils are situated on steep slopes and relatively impermeable substrate, high pore water pressure may develop in soils during strong rainstorms and trigger landslides even in the presence of fully developed, natural vegetation (Restrepo and Alvarez 2006; Scatena and Larsen 1991; Fig. 11.7a). Similarly, earthquake-triggered landslides may potentially denude large areas of lush forests (Garwood et al. 1979; Lin et al. 2008; Fig. 11.7b). In contrast, in unvegetated areas which do not support deep soils and other erosive material the danger of catastrophic surface mass movements may be comparatively limited. However, in areas where lush vegetation was cleared or degraded the risk of destructive slides may be particularly high since various protection functions of the vegetation may have been lost or diminished whereas the physical soil 'mass' is still in place constituting a potential hazard (Fig. 11.7c).

The roots of forest trees and other plants protect against surface landslides as they (1) provide structural cohesion to soil materials, (2) improve water permeability in the soils, and (3) decrease soil pore water pressure via transpiration of the plants (Sidle et al. 2006). Structural cohesion is generally believed to be the most important factor, especially in the case of shallow landslides. Tree roots may penetrate deeply into the soil mantle and act like anchors into more stable substrate (bedrock, etc.). Lateral roots form a dense safety network which stabilizes lose matter in the upper soil horizons (Fig. 11.6b). Furthermore, tree roots traversing planes of weaknesses and shear zones may provide increased stability to deeper soils by lateral reinforcement (Sidle et al. 2006; Stokes et al. 2009). Some tree and other plant species, and mixed vegetation combinations thereof, may be better suited than others to mitigate landslides under certain climatic conditions. Plant root traits which are typically important encompass long and thick tap roots of certain species and a dense net of shallow roots (Stokes et al. 2008, 2009). In addition, the spatial distribution of trees of various sizes and root types, and the presence of undergrowth vegetation may also influence landslide risks (Roering et al. 2003; Fattet et al. 2011).

Vegetation is also important to buffer the impact of rains, facilitate water infiltration into the soils and lower the soil water volume via interception and transpiration (Sidle et al. 2006; Keim and Skaugset 2003; Ghestem et al. 2011). Vegetation, therefore, plays an important role not only in erosion and flood abatement, but also in the mitigation of landslide risks, especially for shallow surface landslides, but possibly also for deeper landslides. The overall (longer-term) role of these functions, however, need to be better quantified regarding their contribution to landslide mitigation. Accumulating subsurface water flows during persistent rainfall often lead to slope instability, especially by increasing the weight of the material, but also by decreasing mass cohesion and friction (e.g., granular blocks of masses become more elastic and/or viscous due to increasing pore water pressure) (Smith 2004; McGuire et al. 2002). High evaporation of tropical vegetation, therefore, acts to alleviate the risk of landslides by preventing pore water pressure from reaching critical thresholds. Also the period of landslide activity may be shortened (Sidle et al. 2006; Dhakal and Sidle 2004).

Soil C *per se* appears to play a somewhat two-sided role in regards to landslides. For one part soils rich in organic matter typically have the capacity to absorb high amounts of water, especially if undisturbed (i.e., high soil porosity and other properties). This can potentially increase the risk of mass failure. On the other hand, however, SOM (especially finely decomposed matter) can be a valuable agent for cohesion within coarse-textured soils, especially at medium water saturation (Karami et al. 2012; Abiven et al. 2009; Six et al. 2004). Furthermore, water drainage may be facilitated from well-aerated porous soils rich in organic matter, plant root density and biotic activity (Sidle et al. 2006; Fattet et al. 2011).

The predominant role of tree roots in terms of soil structural cohesion is illustrated by the observation that the risk of landslides is often not highest directly after clear-felling operations, but increasing rates of landslides tend to occur with a delay of several years. According to Sidle et al. (2006; see also Imaizumi et al. 2008; Alcántara-Ayala et al. 2006) many studies noted up to a tenfold increase in landslide occurrence 3–15 years after deforestation. This is due to a delay of root strength deterioration after logging. If forests are allowed to regenerate, the risk of landslides diminishes substantially 15–25 years after felling as the roots of newly growing trees again start to provide cohesion (Fig. 11.8). In a study of landslide scarps in the Oregon Coast Range, Schmidt et al. (2001) determined median lateral root cohesion strengths of 7–23 kPa in industrial forests and deciduous vegetation, and 26–94 kPa in natural forests dominated by coniferous vegetation. In contrast, lateral root cohesion in clear-cuts was on average around 10 kPa. Similarly, in a model study by Sakals and Sidle (2004) selection cutting of Canadian Douglas-fir (*Pseudotsuga mentziesii*) forests resulted in a decrease of the minimum root cohesion to 81 %, whereas clear-cutting resulted in a decrease to only 38 % of the cohesion in pristine stands. Permanent deforestation or replacement with plantation forests can, therefore, lead to increased landslide risks over many decades to come as illustrated in Fig. 11.8. Similarly, landslide risks may also increase as a result of species invasion in natural forests. For example, the highly invasive exotic woody weed *Miconia calvescens* has the potential to replace deep-rooted native vegetation with

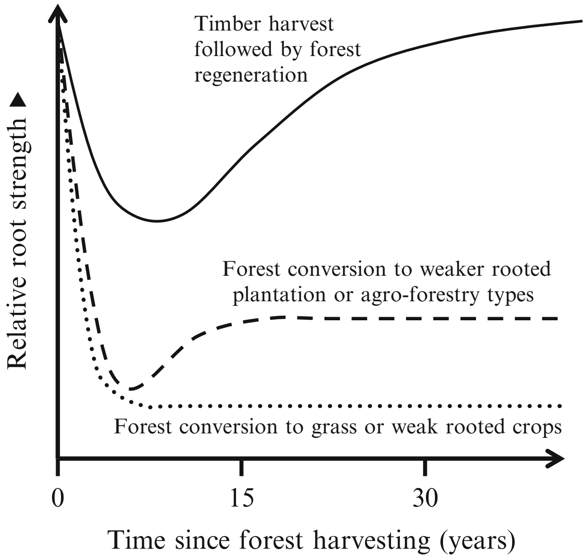

Fig. 11.8 Hypothetical changes in vegetation rooting strength related to different scenarios of timber harvesting and subsequent forest regeneration, respectively conversion to plantations or agricultural uses (Adapted from Sidle et al. 2006)

mono-specific shallow-rooted stands on steep mountain slopes, thereby increasing the incidence of landslides (Burnett et al. 2007). Conversely, in agricultural areas hurricane-induced landslide risks may be mitigated appreciably by increasing the vegetation complexity and other measures of careful farm management (Philpott et al. 2008). Similarly, conservation of pristine forests on steep mountain slopes can significantly mitigate against landslides during severe earthquake events. According to Peduzzi (2010) forest vegetation cover explained about 10 % of landsliding during the 2005 Kashmir Earthquake, whereas the minimum distance to the rupture fault line and the maximum inclination on mountain slopes each explained about 20 % of the landslide area in the multivariate regression model.

11.5.3 Forests as Shelterbelts in High Mountainous Areas

Even before the enactment of national forestry legislation in Switzerland in the late nineteenth century many communities in the Alps maintained and enforced local logging bans and use restrictions of mountain forests which were located above their villages (Sablonier 1995). Such forests were recognized as vitally important to ensure the safety of villages not only against landslides and rockfalls, but in particular against winter snow avalanches (Fig. 11.9). In addition to landslide mitigation, dense mature forests represent a fairly reliable safety measure against avalanche

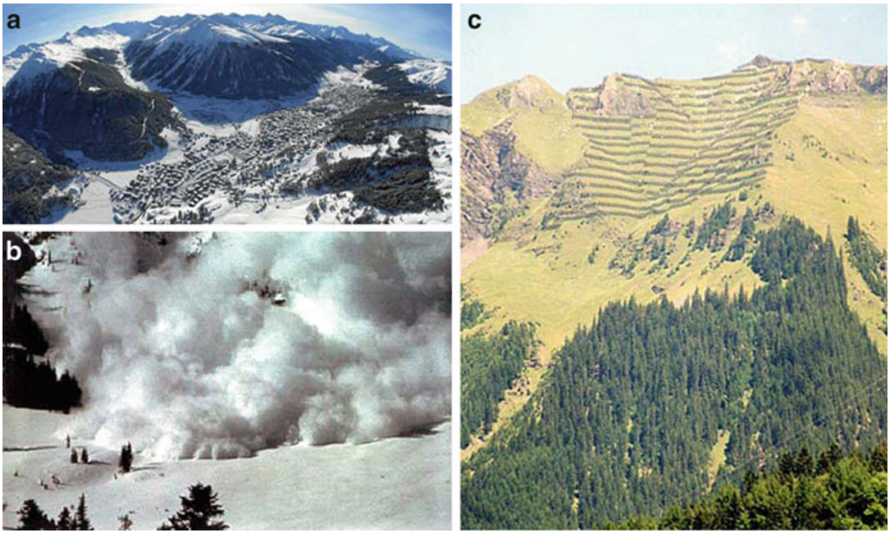

Fig. 11.9 Avalanche hazard mitigation functions of forests. (**a**) Mountain forests are an important safeguard against winter avalanches (the mountain resort of Davos, Switzerland, World Economic Forum, http://creativecommons.org/licenses/by-sa/2.0/deed.de). (**b**) Forests may, however, provide an insufficient shelter against large avalanches which are released from far above the timber line (nps.gov, wikipedia.org). (**c**) Snow fences can reinforce the protection provided by forests above the timber line (Swiss Alps, Roland Zumbühl, picswiss.ch, wikipedia.org)

initiation below the timberline as trees and forest undergrowth act to intercept snowfall, and tree boles locally stabilize the developing snow field (McClung 2001; Schneebeli and Meyer-Grass 1993; Frey and Thee 2002; Schönenberger et al. 2005). Forests may, however, also serve an important role as ‘shelterbelts’ against mass movement hazards emanating from above the timberline. In the Alps, about 60–80 % of the forests serve a potential shelterbelt function, and in a national forestry inventory in Switzerland 37 % of mountain forests showed signs of substantial (and potentially hazardous) snow movements, 31 % of rockfalls, and 16 % of erosion, mudflows and landslides (Wehrli et al. 2007).

The protective capacity of mountain shelterbelts is due to the presence of dense stands of strong and healthy trees. Depending on the intensity and frequency of impacting mass hazards these biotic buffers may be severely weakened, and forest rejuvenation may be permanently impaired in forest aisles (so-called *couloirs*) formed by recurrent winter avalanches or other mass movement impacts (Bebi et al. 2009; Dorren et al. 2004a). In addition, increasingly frequent storm impacts in forests (e.g., in autumn), higher amounts of snow-fall (during winter) and more frequent intermittent warm spells (in late winter and early spring) may increase the incidence of hazardous snow gliding and creep in forest gaps. This may affect forest regeneration in key locations and weaken the effectiveness of the protection forests (Frey and Thee 2002; Höller et al. 2009). Risk management using forest shelterbelts, therefore, depends on an appropriate forest management that mitigates against natural, human and ecological hazards (e.g., storm impacts, excessive forest uses, pests and diseases, fires, etc.), and ensures timely tree regeneration via the maintenance of a rich and diverse forest structure and composition. Additional measures using engineering tools (e.g., fixed snow fences or rockfall nets, etc.) and management (e.g., setting felled tree trunks diagonally to the slope direction in *couloirs*, leaving fallen trees in place after a storm, etc.) may temporarily help to restore forest vegetation in disturbed and in recurrently exposed areas (Brang et al. 2001; Bebi et al. 2009; Dorren et al. 2004a, b; Wehrli et al. 2007; Frey and Thee 2002; Höller et al. 2009; Schönenberger et al. 2005).

Trees buffer the kinetic energy of falling rocks, intercept rock jumps, and may stop the movement of rock materials after a certain distance. Depending on the intensity of the rockfall event and the tree stand structure and species composition forest shelterbelts can, therefore, act to mitigate effectively against rockfalls (Dorren et al. 2007). In a paired trial in the French Alps, Dorren et al. (2005) evaluated the effects of a forested site and a non-forested site on each of 100 experimental rockfalls. The rocks had a mean diameter of 0.95 m and volume of 0.49 m^3 (range 0.1–1.5 m^3). The tree-less distance before the forested site was 35 m. They found that the mean maximum velocity decreased from 15.4 m s^{-1} at the open site to 11.7 m s^{-1} at the forested site. While at the open site 95 % of the rocks traversed the entire slope length (224 m; slope gradient 38° for both sites), only 34 % reached the valley bottom at the forested site. Forest shelterbelts will be disproportionately less effective if exposed to rockfalls with increasingly higher kinetic energy, e.g., emanating from far above the timberline (Dorren et al. 2007).

Similarly, forests may provide little protection against the impacts of very powerful snow avalanches which started high above the timberline. Powder clouds of such avalanches may exceed by far the largest trees (Fig. 11.9b), and if trees are broken by the impact then the hazard may increase as the avalanche is being mixed with debris. Nonetheless, over large areas forests generally mitigate against the force of exogenous avalanches by stabilizing local snow fields and by providing resistance against the impacting snow cloud. Considering various impact scenarios and using standard models, Teich and Bebi (2009) studied the effectiveness of the avalanche protection forest above Andermatt, Switzerland. Their models indicated that deforestation (e.g., tree falls caused by an autumn storm) in the lower parts of the forest would result only in a slight increase of the avalanche risk. In contrast, deforestation in the upper parts of the forest more than doubled the risk. The risk from avalanches was estimated to an average annual damage of about 80,000 US$ under the actual setting. According to the models this risk would increase about sevenfold (to 550,000 US$ $year^{-1}$) if the current snow fences above the forest would be removed and a further sixfold (to 3.3 million US$ $year^{-1}$) if the entire forest would be cut. In this particular case, the annual risk mitigation value of the avalanche protection forest was thus estimated at 200,000 US$ per hectare. Mostly due to the high damage potential this was, however, very high as compared to other studies (Teich and Bebi 2009).

11.6 Coastal Hazard Mitigation Roles of Carbon-Rich Ecosystems

11.6.1 Unravelling Waves of Coastal Disasters

In contrast to often locally confined disasters caused by landslide or flood events, coastal disasters are less frequent but typically encompass regional scales and are of dimensions which seldom can be ignored by the international community and the media (Fig. 11.10). Casualties from coastal disasters repeatedly range in the thousands, especially in developing countries, and some disasters in recent decades were among the deadliest and costliest in human history. These include several severe cyclone events such as the 1970 Bhola Cyclone (>300,000 casualties) and the 1991 Cyclone Gorky (~139,000 casualties) in Bangladesh, the 2008 Cyclone Nargis in Myanmar (~138,000 casualties, >10 billion US$ of damages), and the 2005 Hurricane Kathrina (1,833 casualties, 108 billion US$ of damages), 2008 Hurricane Ike (103 casualties, 38 billion US$ of damages), and 1998 Cyclone Andrew (26 casualties, 41 billion US$ damages) in the United States (Smith 2004; The Economist 2011b; Wikipedia 2012). Some of the fatalities and damages during these disasters resulted from cyclone wind storm damages and excessive rainfall (e.g., triggering river floods and landslides). Most of the casualties and damages were, however, caused by catastrophic sea-borne flooding due to cyclonic storm surges and associated extreme wind-driven ocean waves, sometimes exacerbated by the confluence with catchment-based floods in coastal floodplain areas. For example, over 70 % of the 10,000 victims of the 1999 Orissa Cyclone in India drowned in the storm surge (Gupta and Sharma in Das and Vincent 2009). Similarly, the vast majority of the victims during the 2008 Cyclone Nargis were due to storm surges which penetrated up to 50 km inland (Fritz et al. 2009; Webster 2008). Storm surge hazards can also occur during severe winter storms in temperate regions. For example, the 1962 storm surge in the North Sea left over 300 people dead and caused widespread damages along the coastlines near Hamburg, Germany (von Storch and Woth 2008). It is expected that cyclones as well as winter storms are becoming more frequent.

Two of the most destructive disasters during the last decade (and in recorded human history) resulted from the impacts of tsunamis. The 2004 Indian Ocean Tsunami affected several countries (Indonesia, Thailand, India, Sri Lanka and others) along the Indian Ocean Basin, and led to up to 300,000 casualties and an estimated 14 billion US$ of damages. The 2011 Tohoku Tsunami in Japan caused more than 16,000 casualties and a record 380 billion US$ of damages (Wikipedia 2012). To determine global annual average numbers of casualties and damages is not straightforward for coastal hazards, especially in the case of tsunamis. Disastrous mega-tsunamis such as occurred in 2004 and 2011 are – fortunately – very rare. The last comparable event occurred more than 100 years ago, e.g., a tsunami triggered by the Krakatau volcanic eruption in 1883 in Indonesia which caused about 37,000 casualties (Levy and Gopalakrishnan 2005; Cochard 2011c). The unpredictability

Fig. 11.10 Storm flood and tsunami hazards mitigation functions of coastal vegetation. (**a**, **b**) Coastal ecosystems may buffer some of the impact of wind driven waves during a tropical storm. However, their effects on large storm surges are still controversial (flooding by storm surge during Hurricane Ike, Galveston, Texas, September 2008, Chuck Simmins, http://creativecommons.org/licenses/by/2.0/deed.de). (**c**) Coastal vegetation cannot adequately protect against the impact of a large tsunami. Trees and other obstacles may, however, slow the advance of water masses to some degree, potentially providing more time for fast evacuation of the coastal population (2011 Tohoku tsunami, Japan, Keystone/Xinhua). (**d**) Equally, mangroves may not protect humans from a tsunami, but their roots can provide stability to the substrate, preventing excessive land erosion during and after such a violent flood (roots of *Rhizophora* mangroves that withstood the 2004 tsunami, Banda Aceh, Indonesia, R. Cochard)

and intensity of such events, however, pose particular challenges for risk management. In addition, uncertainties about this risk may rise if these geologically triggered coastal disasters become more common as an indirect result of climate change as some scientists hypothesize (McGuire et al. 2002; McGuire 2012).

Over one third of the world's population lives in coastal zones which cover however only about 4 % of the land area (UNEP-WCMC 2006). In most coastal regions already densely concentrated human populations will further increase in the future, potentially putting more people in the way of harm, and increasing pressures on already heavily exploited coastal resources and acutely stressed ecosystems. Several of the world's largest cities (e.g., Shanghai, Bangkok, Jakarta, Manila, Rio de Janeiro, New York, Miami, Brisbane, Lagos, Dar es Salaam, etc.) are located in coastal zones and floodplain areas near the estuaries of large rivers. These cities may not only be threatened by land subsidence and river flooding, but increasing frequencies and intensities of sea-borne flood and wave hazards as well as heightened vulnerabilities due to sea-level rise, coastal land use changes, and ecosystem degradation are potentially set to augment the risks of coastal disasters.

The 2004 Asian tsunami may be seen as an example of a "black swan event", i.e., a massively fatal and destructive event which occurred fairly suddenly (in many places within less than half an hour after the earthquake) and which came utterly unexpected for most of the affected communities (Cochard 2011b). After the event, coastal ecosystems (in particular mangroves and coral reefs) attracted the interests of international NGOs, donor agencies and the wider public as a potential means to buffer the impacts of extreme wave hazards (Cochard et al. 2008; Cochard 2011b). UNEP-WCMC (2006, p. 4), for example, wrote that the tsunami was a "wake-up call for the global community, dramatically drawing attention to the vulnerability of coastal ecosystems and the dangers of undermining the services they provide to humankind." Various reports speculated or claimed that the widespread destruction of mangrove forests through the previous expansion of shrimp farming and other economic activities had lead to a higher vulnerability of communities in various affected regions.

Coastal ecosystems, such as coral reefs, seagrass beds, mangrove and beach forests, and sand dunes may provide effective buffering against certain types of coastal wave and flood hazards, but many of these ecosystems have already been severely degraded, especially in the vicinity of dense human habitations (Cochard et al. 2008; Cochard 2011b). For example, about 19 % of the mangroves have been completely lost just between 1980 and 2005, and a similar amount may have been lost before. In addition, many remaining mangrove areas are being severely degraded (Spalding et al. 2010). Losses at national and regional levels vary widely. In the Gulf of Thailand almost 90 % of the mangrove forest cover was lost between 1961 and 1997, which is mostly attributable to a shrimp farming boom during the 1980s and 1990s. This was to a large degree responsible for coastal erosion of up to 0.9 km^2 per year (Thampanya et al. 2006; Aksornkoae and Tokrishna 2003). At the Chao Phraya estuary south of Bangkok, the destruction of mangroves led to approximately 25 m of land erosion per year between 1969 and 1987, whereby at Bang Khun Tien the coastline now follows the layout of abandoned former fishponds (Winterwerp et al. 2005). In several estuaries in Sri Lanka and elsewhere mangroves have been degraded due to regime changes in river estuaries (hydrology and sediment loads) caused by the construction of inter-basin canals (Dahdouh-Guebas et al. 2005). Along several coastlines, beach forests have been cut and sand dunes have been replaced by coastal infrastructure or – reportedly – by exotic tree plantations (Cochard 2011b; Feagin et al. 2010).

Emissions of chemicals (e.g., antibiotics) from coastal shrimp farm industries add to increased loads of catchment-borne coastal pollution, including agricultural effluents such as nutrients, pesticides, organic matter and sediments, and industrial and household wastewater from cities (Tabucanon 1991; Dierberg and Kiattisimkul 1996). Chronic pollution can weaken and kill offshore ecosystems such as coral reefs and seagrass beds either directly or indirectly via smothering with sediments and/or algal growth. Increasingly common incidents of coral bleaching are also thought to be linked to climate change as well as to pollution and other stressors (van Oppen and Lough 2009). Other threats to offshore ecosystems include overfishing, destructive fishing (e.g., using dynamite or poisons) and trawling, coral mining, dredging and coastal development (Spalding et al. 2001; Green and Short 2003). According to Burke et al. (2002) human activities threatened 88 % of South East Asia's, and 85 % of Indonesia's coral reefs. For about 50 % of the reefs the level of damage was estimated to be high or very high.

11.6.2 Mitigation of Storm Waves and Surges by Coastal Ecosystems

Cyclonic storms are typically most destructive along coastlines due to the combined impacts of storm surges and extreme wind-driven waves (Fig. 11.10a, b). Scientists commonly agree that coastal ecosystems can buffer some of the energy of wind-driven waves – even under intense storm conditions. Several scientists, however, still question the capacity of reefs and mangroves to provide any protection against storm surges (Feagin et al. 2010; Kerr and Baird 2007).

Coral reefs may absorb as much as 90 % of the average wind-driven surface wave energy (Kench and Brander 2006; Brander et al. 2004). During storms, reefs (and to some degree probably also sea grass beds) may act as wave filters, i.e., extreme waves are reduced in height to less than the maximum depth of the water over the reef flat (Lowe et al. 2005; Cochard et al. 2008). The dissipation of wave energy is highest over a long and intact reef flat (mostly due to bottom friction). However, the shoreline becomes more exposed to waves if parts of the reefs are destroyed or weakened (Frihy et al. 2004; Sheppard et al. 2005). The capacities of reefs to buffer waves also much depend on tidal regimes and the formation and effects of a storm surge, since these influence water depth (Madin et al. 2006).

Wind-driven surface waves are attenuated to various degrees in the web of stems and aerial roots of mangroves. For example, average wave energy decreased by over 95 % after passing through a 1.5 km wide belt of *Kandelia kandel* mangroves in Vietnam (Mazda et al. 1997). Similar results were found for other species. However, the effectiveness decreases disproportionately when trees are smaller, tree stands are less dense, the water levels are elevated (e.g., during high or spring tides and/or a storm surge), and if the waves are very large (Massel et al. 1999; Mazda et al. 1997, 2006; Cochard et al. 2008).

Mangroves and other vegetation may not be very effective to buffer very large storm surges generated near the eye of a cyclone. Under less extreme storm conditions (e.g., areas farther away from the cyclone eye), however, vegetation (i.e., parts above the water) may reduce the speed of local wind fields and, thus, wind stress on the water surface. According to Danard and Murty (1994) locally wind-generated surges may account for up to 50–80 % of the total surge during storms, and in some measure even dense reed vegetation may be able to reduce surges during storms in temperate regions – provided the vegetation is not already submersed. It appears plausible that large and extensive stands of mangroves (e.g., the Sundarbans mangroves in the Bay of Bengal) function in a similar way and can reduce surges and waves even during so-called 'super cyclones' – albeit there is not yet any study which quantified the effectiveness of such a function. On the other hand, storm surges may also be concentrating in shallow estuarine areas and afflict communities living along the rims of mangroves and along large river channels. Furthermore, mangroves along these rim zones may be severely damaged due to wind and wave impacts (Cochard et al. 2008; Cochard 2011b). Cyclone early warning, evacuation schemes and the construction of elevated shelters are, therefore, certainly the most effective measures to reduce casualties in areas at high risk of cyclonic storm surges (Bimal 2009; Bimal et al. 2010).

As shown in surveys (e.g., Badola and Hussain 2005; Walters 2004; Akter 2010), many poor coastal communities value mangroves and other coastal vegetation as a protective component against storm winds and associated wave impacts. However, there appears to be only one study which attempted to demonstrate a protective role of mangroves against a storm surge by analyzing actual event data: Das and Vincent (2009) collected data on the deaths and house damages in 409 coastal villages which were affected by the 1999 Orissa Cyclone in India. The villages were all situated near a coastline which was originally covered with mangroves. Half of the mangrove area was, however, lost between 1944 and 1999. The authors found that the width of remaining frontal mangroves significantly reduced the number of casualties as confirmed in regression models which controlled for distance to the coast, rivers and roads, original mangrove width, presence of sea dykes, and the effects of socio-economic parameters. The models suggested that the number of casualties (totaling 256) may have been three times higher if mangroves would have been completely destroyed (Das and Vincent 2009). The models did, however, not include an unambiguous control for village elevation. This may weaken the validity, respectively the precision of the findings. Furthermore, the authors did not discuss the observation (Table S6 in Das and Vincent 2009) that the influence of frontal 'mangrove width' on the level of damage caused to housing was non-significant in the regression models – in contrast to the numbers of casualties.

11.6.3 Tsunami Mitigation Functions of Coastal Ecosystems

Tsunami waves cause catastrophic coastal flooding which tends to be much faster, higher and contain more energy than storm surges and associated wind-driven waves (Fig. 11.10c). Tsunamis are triggered by geologic events. The resulting floods are, therefore, not influenced by wind and weather, but coastal bathymetry and topography are dominant controls on the development of the hazard along the coastlines (Chatenoux and Peduzzi 2007; Cochard et al. 2008). Coral reefs may act as protective buffers against tsunamis (McAdoo et al. 2011), but tsunami interactions with reefs are generally complex and dependent on various parameters. Reefs which have a high surface roughness (causing wave dissipation via bottom friction) over a wide area, a high reef front, and which are not interrupted by gaps and channels may be most effective in lessening the wave energy. In contrast, reefs with smooth surfaces and gaps (which may concentrate wave flows) may actually amplify the hazard in certain locations (Gelfenbaum et al. 2011; Cochard et al. 2008).

Despite the worldwide media attention, in the aftermath of the 2004 Asian Tsunami relatively few reports went beyond presenting fragmentary and anecdotal observations on the role of mangroves and other types of vegetation. Several frequently cited scientific studies emerged to be of limited use as they were either incoherent in their methodology and/or ambiguous in their results (Cochard et al. 2008; Cochard 2011b; Kerr et al. 2006, 2009; Vermaat and Thampanya 2007; Baird and Kerr 2008; Feagin et al. 2010). However, there are now two fairly informative studies, i.e., a cross-regional study by Chatenoux and Peduzzi (2007) and a local study from Aceh by Laso Bayas et al. (2011).

Chatenoux and Peduzzi (2007) collected data for 62 locations in Thailand, Aceh, Sri Lanka, India and the Maldives which were hit by the tsunami (i.e., tsunami intensities ranging from a few meters to over 30 m run-up heights). Using multivariate regression models (but not controlling for potential spatial autocorrelation) they investigated the effect of 36 variables – tsunami source distance (3), near-shore bathymetry (18), terrestrial geomorphology (10), and biophysical obstacles (5, including a vegetation resistance index and % mangrove cover) – on tsunami inundation distance. Laso Bayas et al. (2011) collected data from 180 villages along a straight coastline in South Aceh. The authors investigated whether and how vegetation cover (mostly rubber plantations and natural rainforest) in front and at the back of the villages influenced total tsunami inundation distance, and the casualty rates and damages to housing in the villages. The study controlled for initial tsunami wave height at the shoreline, topographical variables, and spatial autocorrelation. Both studies did not find any effect of vegetation on tsunami inundation distance and run-up, which means that the protective effect against tsunami flooding (in terms of area) is probably unimportant (Cochard 2011c). Furthermore, Laso Bayas et al. (2011) did not find any protective effect of vegetation in regards to housing damage. The data even indicated that damage may have been increased if forests were situated at the back of the villages. This was interpreted by potentially increased impacts of high debris loads in the backwash of tsunamis (Laso Bayas et al. 2011; Cochard 2011c). Similarly, the casualties were increased by about 3–6 % if the vegetation lay behind the villages. This probably indicates that villagers were either more likely to be caught in the waves and/or that they were more likely to be killed if being swept against trees. In contrast, if the forests lay in front of the villages 3–8 % fewer residents tended to be killed by the tsunami. The trees may have slowed or diverted the waves, allowing more villagers to flee in time to a safe location (Laso Bayas et al. 2011; Cochard 2011c). Similar to coral reefs, the buffering effects of frontal vegetation are, however, not uniform. In locations where the flooding water masses are confined and channeled (e.g., along open mangrove creek channels in estuarine areas) the hazard may be increased (focused and faster flows of water masses). On the other hand, the flow energy may be decreased behind dense stands of large trees which may increase the time margin for escape for fleeing persons, whilst not providing ultimate protection from the flood (Cochard et al. 2008; Cochard 2011c; Tanaka et al. 2007; Fig. 11.10c).

11.6.4 Land Protection Functions of Coastal Vegetation Under Flooding

Fast evacuation to high ground is clearly the most important life-saving strategy in the advent of coastal flooding caused by a tsunami. In the case of a storm surge, evacuation to high ground may be similarly important, albeit, due to the longer time span from storm warning to impact, complete evacuation to the hinterland is generally possible and also safer. Nonetheless, houses, infrastructure and environmental assets cannot be evacuated, and an understanding of the interaction of sea-borne flooding hazards with the landscape is therefore needed for safer planning and management in potentially exposed areas. Evidently, the effects of coastal topography necessarily need to be accounted for in any meaningful analysis on the effects of coastal vegetation for hazard mitigation (Cochard et al. 2008). Large stands of forest may buffer some wave energy, but being 'permeable' they may ultimately offer little resistance to flooding – at least in the case of a tsunami, but possibly also in the case of a storm surge. In contrast, geophysical elements, such as high sand dunes, will offer a more compact and effective barrier against sea-borne wave hazards than vegetation cover as indicated in models (Borthwick et al. 2006; Kanoglu and Synolakis 1998) and also observed in the field (Cochard et al. 2008; Synolakis et al. 2005). Beach vegetation may thereby play a significant role in dune formation and by providing stability to sand dunes and the coastline during exposure to sea-borne flooding. Similar to sand dunes, extensive tidal flats stabilized by seagrasses may also offer some buffering against tsunamis in the offshore zone (Chatenoux and Peduzzi 2007; Cochard et al. 2008). The role and utility of vegetation and soil C in regard to such more indirect mitigation of sea-borne wave hazards is, however, still insufficiently understood.

Vegetation cover on sand dunes reduces wind speed and thereby stabilizes and traps loose sand particles (Lancaster and Baas 1998). In moist tropical zones rapid vegetation growth in sand dunes prevent prolonged periods of sediment transport inland, and coastal dunes are therefore commonly less well developed as compared to temperate zones (Morton 2002). Dunes in temperate zones may in general be larger. On the other hand, vegetated dunes may be more resistant against the impacts of sea-borne flooding since vegetation contributes in several ways to consolidate and stabilize loose sand material. Foliage increases water retention and fallen leaves facilitate the formation of an organic soil layer. Roots provide a structural binding medium and stimulate bioactivity through inputs of nutrients and organic matter (Chen et al. 2005; Martinez et al. 2001; Mailly and Margolis 1992). A high diversity of specialized colonizing plants typically indicates a higher dune stability and resistance to wave and flooding impacts (Garcia-Mora et al. 2000).

In a laboratory flume study and controlled field experiment, Feagin et al. (2009) found that the lateral soil erosion rate was not significantly influenced by the root density of salt marsh plants. The primary variable influencing erosion was the soil type. The authors concluded that vegetation mostly indirectly reduced erosion via modifying soil parameters, i.e., by facilitating the accretion of fine sediments and organic detritus. However, during large-scale flooding over land vegetation may play additional important roles by modifying parameters of water flow, waves and soil matter suspension. For example, according to Duarte (2002) seagrasses stabilize sediments with their extensive root systems, but they have also a significant effect on local current flow rates. While strong water currents pass over the seagrass, canopy water movement near the bottom is attenuated in dense (and mostly flexible) seagrass meadows. For this reason, sedimentation rates in seagrass meadows (as well as other types of submersed wetland vegetation) are high under normal conditions whereas rates of re-suspension (respectively soil and sediment erosion) during violent storms and floods are mitigated as compared to areas without vegetation cover (Duarte 2002; Granata et al. 2001; Madsen et al. 2001).

Similar effects may also be expected for larger types of vegetation under severe flooding impacts. For example, in the absence of forest cover tsunamis typically cause significant land erosion near to the shoreline (from the beaches up to ~50 m inland) and near to the maximum flooding distance. Tsunamite sediments may be deposited in between, particularly in swales (Moore et al. 2006; Cochard 2011b). Provided that trees are not being uprooted by the impact of the floods, trees are likely to buffer and modify flow energy and protect soil resources, especially in the landward direction. At Banda Aceh, *Rhizophora* mangroves near to the shore were broken above the stilt roots, but the remaining roots largely protected the mangrove sediments, whereas nearby treeless land areas were mostly lost to the open sea (Cochard 2011b; Fig. 11.10d). Impacts may, however, be more severe in adjacent locations where water flows are concentrated or in front of forests where flood edges and waves are being broken (Feagin et al. 2009; Cochard 2011b).

11.7 Concluding Remarks: Towards Sustainable Risk and Resource Management

Throughout the last century much has been achieved to reduce the risks posed by natural disasters. Especially in developing countries, however, natural disasters continue to take a heavy toll on human lives and livelihood assets. In many regions and cities the loss potentials are rising because populations and economies are fast growing. In addition, climate change and worsening environmental degradation increasingly threaten the progress made in risk management. If present trends continue, disaster losses may soon surpass the annual economic growth in several countries. Disaster risk reduction, therefore, must be central to policies aimed at 'climate change adaptation'. Several measures can significantly help to improve risk management at a relatively trivial cost as compared to the overall losses due to disasters.

Data collections need to be improved in order to better assess risks, identify trends and their driving factors, and evaluate disaster policies. Bouwer et al. (2007) suggested setting up an open-source peer-reviewed database. Such a database would need to be standardized throughout. Data gaps should be made explicit, and further research efforts should be encouraged and funded to fill these gaps. In areas where data remains unavailable, the data sets may be supplemented with data derived from models in order to improve risk approximations and estimates. The model assumptions will need to be explicitly stated, and data derived from such models should be available separately from the original 'primary' data. Coherent procedures should be applied to refine, amend and update the database.

Based on the best available information, risk management needs to be improved and funding increased. Efforts need to be better coordinated at national and international levels. In developing countries the benefits of investments in disaster risk mitigation may exceed their costs by a factor of about 2–4 (Mechler 2005). The potential benefits of investments aimed at risk management are, however, often inadequately appraised. While large sums of international aid are often dispensed in the wake of a disaster event, aid is generally spent most efficiently on *ex ante* risk reduction (Bouwer et al. 2007; Cochard 2011b). In poor and/or autocratically led countries, however, available funds are sometimes not used resourcefully by institutions. Aid programs may be most effective if they aim to reduce both the vulnerability and poverty of rural and urban communities, thereby strengthening civil society, cooperative management within communities and the 'human capital' (Erikson and O'Brien 2007; Heltberg et al. 2009; Wood 2003). The objectives of poverty reduction may, however, not necessarily equate with those of vulnerability reduction, if considered separately. For example, the replacement of mangroves and other wetlands with shrimp farms may increase the incomes and wealth of communities at least in a short term outlook. On the other hand, the communities are likely to become more vulnerable to natural hazards such as storms and coastal erosion; equally, their sole income source (shrimps) may become more vulnerable to diseases (Adger et al. 2005; Thampanya et al. 2006; Dierberg and Kiattisimkul 1996).

In the longer term, therefore, the resource base may become degraded to such a degree that communities will end up more impoverished than before the development of shrimp farming (Barbier et al. 2011). In contrast, after the 2004 Asian Tsunami, programs to restore coastal ecosystems with the aim to reduce communities' future vulnerability to coastal hazards were sometimes implemented in a poorly planned and wasteful way. Many of these restoration programs failed completely, and essentially nothing was gained in either material or educational ways (Cochard 2011a, b, c; Cochard et al. 2008; Wibisono and Suryadiputra 2006). However, even if mangroves can be restored successfully, the sustainability of such reforestations may not be guaranteed, provided that the communities remained impoverished and overly dependent on scarce resources (e.g. wood from mangroves). Hence, development programs need to consider the various long-term consequences of planned interventions. Programs should aim at the diversification of livelihood options and the increase in overall welfare within a context of best-practice sustainable resource management. Maintaining the options provided by diverse ecosystems (and their services) can provide significant socio-economic stability as well as safeguard future opportunities, especially under the increasing risks posed by climate change.

Within national and international agencies as well as among various decision makers, economists and scientists there has been an increasing awareness of the potential value of ecosystem services and the need to account for 'ecosystem capital' in development projects. This is reflected in the compilation of several significant policy documents during the last years, including, for example, the 'Millennium Ecosystem Assessment' report (MEA 2005), 'The Economics of Ecosystems and Biodiversity' series (TEEB 2010, 2011, 2012), and the 'Corporate Ecosystem Services Review' (Hanson et al. 2012). While these and other works provide useful frameworks for the valuation and economization of hitherto largely discounted ecosystem services, many uncertainties still remain regarding the practicality and equitable application of various concepts. Ecosystem capital and its flow are often not well understood, partly since ecosystems are mainly common property resources. Various stakeholders may frequently provide differing and even conflicting assessments with regard to the benefits and opportunities provided by ecosystems. Such differences can be due, for example, to the stakeholders' socio-economic status and role within specific development schemes, or their divergent interpretations of ecosystem functions and dynamics at differing spatial scales and at different times (Corbera et al. 2007; Hein et al. 2006; Skourtos et al. 2010). Knowledge based on ecological understanding can provide a less partial basis for assessing the overall trade-offs between land development interventions (satisfying immediate human needs such as food and energy) and ecosystem conservation measures (maintaining wider ecosystem functions) (DeFries et al. 2004; Dale et al. 2011). Daily et al. (2000, p. 400) noted that "valuation is a way of organizing information to help guide decisions but is not a solution or end in itself. It is one tool in the much larger politics of decision-making. Wielded together with financial instruments and institutional arrangements that allow individuals to capture the value of ecosystem assets, however, the process of valuation can lead to profoundly favorable effects."

Understanding ecosystem functions is fundamentally important to make realistic, objective valuations of ecosystem services, respectively to properly assess and communicate the consequences of different choices on overall human well-being. Potential ecosystem functions to mitigate natural hazards either directly (as shelter-belts, structural reinforcements, ‘sponges’, or climate regulators) or indirectly (by mitigating against global warming via C storage and sequestration) have been noted as highly valuable. Yet, a closer look reveals that the scientific sources presented in relevant public documents and reviews (e.g. MEA 2005; TEEB 2010, 2011, 2012; Hanson et al. 2012; Barbier et al. 2011) are generally still limited, and sometimes inadequate as a constructive basis for economic valuation. Elmqvist et al. (2010, p. 71) for example, list a series of studies and then note that “the available evidence for some of these [hazard mitigation] effects [by ecosystems] is still scarce, and in some cases controversial.” The material presented here shows that the various ecosystem functions relating to catchment-based floods and mass movement hazards are considerably well described and documented by scientific studies despite several ongoing debates and remaining knowledge gaps. In contrast, there are still only a few useful studies on tsunami and storm surge protection functions by coastal ecosystems. Despite a sizeable quantity of studies, the roles of climate hazard mitigation functions by ecosystems are inherently complex, especially concerning various feedback mechanisms between global warming and C-rich ecosystems. What can be said with certainty, however, is that the relentless, improvident destruction of ecosystems which we currently observe in many regions around the world will come at an increasingly higher cost to current and future generations. While the option may exist in principle to restore ecosystems and reinstate their services, in practical experience the services are often lost permanently as the costs for ecosystem restoration tend to become exceedingly high. Sensible foresight therefore stipulates a precautionary approach when dealing with the precious, limited and potentially irretrievable ‘ecosystem capital’.

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Chapter 12
Preserving Regulating and Cultural Ecosystem Services: Transformation, Degradation and Conservation Status

Benis Nchine Egoh and Joachim Maes

Contents

Abstract For many years humans have benefited from provisioning services such as meat from hunting of wild animals, raw material and livestock grazing; regulating services such as water and climate regulation; supporting services such as soil fertility; and cultural services such as recreation. These Ecosystem Services (ESs) are now being degraded and used unsustainably around the world. Understanding the levels of threats facing various ESs and their conservation status is important for safeguarding them. In this study, the degradation and transformation of five regulating ESs in South Africa have been examined. Results showed that at least 10 % of the total hotspot area and 20 % of the total area that provide substantial amount (the range) of all five ESs has been transformed or degraded. The range of water regulation and

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supply had the highest level of transformation (30 and 27 %, respectively). The range of carbon storage revealed the highest degradation (10 %) followed by surface water supply (9 %). Amongst the hotspots, again, water flow regulation and supply showed the highest levels of transformation (33 and 25 %, respectively). The hotspot of water supply was the most degraded (12 %). More than 60 % of all transformations that occurred within the hotpots and the ranges of all five ESs could be attributed to cultivation. The second most common driver of transformations were plantations. Although protected areas presented an opportunity for safeguarding ESs, they are rarely included in the identification of areas for the establishment of protected areas. Important areas for providing several ESs in Europe and Africa continue to be found outside protected areas. Apart from recreational services, at least 80 % of such areas remain outside protected areas. Conservation strategies for ESs need to be urgently developed to safeguard them. Effective strategies should consider the benefit of multiple ESs such as provisioning, regulating and supporting services inside and outside protected areas.

Keywords Ecosystem services • Degradation and transformation • Protected areas • Carbon • Recreation • Water

Acronyms

C	Carbon
ESs	Ecosystem Services
EU	European Union
GHG	Greenhouse gas
IPCC	Intergovernmental Panel on Climate Change
MA	Millennium Ecosystem Assessment
N	Nitrogen
PES	Payments for ES
REDD	Reduced Emissions from Deforestation and Degradation
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
WAF	Department of Water Affairs and Forestry
WMO	World Meteorological Organization
WRI	World Resource Institute

12.1 Introduction

Ecosystem services (ESs), i.e., the benefits that humans receive from ecosystems contribute to the livelihood of about a billion people around the world (World Bank 2006). For many years humans have benefited from provisioning services such as meat from hunting of wild animals, raw material and livestock grazing; regulating services such as water and climate regulation; supporting services such as soil fertility; and cultural services such as recreation. The millennium ecosystem assessment (MA), a global study commissioned by the World Bank was instrumental in highlighting the contribution of natural ecosystems to human well-being and in measuring the status and trends of ESs around the world (MA 2003, 2005). According to this report, 60 % of ecosystems and their services are being degraded or used unsustainably around the world. The findings from the MA supported previous reports from the World Resource Institute (WRI) of declining ecosystems around the world (WRI 2000–2001).

An important step towards safeguarding ESs is to identify areas that are crucial for providing ESs. Secondly, the condition of such areas should be examined and thirdly, an assessment of the threats facing them needs be conducted. Several studies have quantified ESs spatially and assessed the relationship with biodiversity, analysed trade-offs amongst services, or evaluated ESs in monetary terms (Chan et al. 2006; Turner et al. 2007; Egoh et al. 2008; Naidoo et al. 2008; Reyers et al. 2009). However, limited research has evaluated the degradation and conservation status of ESs. In addition to assessments that have been published as part of the MA (2003, 2005), only a few studies have evaluated the degradation and transformation status of areas providing ESs and the associated impacts on service provision (Van Wilgen et al. 2008; Reyers et al. 2009). Understanding such degradation and the associated threats from local to continental scales is important in designing management strategies to safeguard them.

Many ESs are supported by biodiversity and face the same threats as biodiversity. In such cases, conservation measures for biodiversity could safeguard ESs as well. Sometimes the link between biodiversity features and ESs is direct. For example, many woody plant species are used to produce medicines, for roofing or firewood, and thus directly impact livelihoods of many villagers (Dovie et al. 2002). This is the case for most provisioning services and some regulating services such as pollination by insects. In such cases, conservation measures for biodiversity will directly enhance ESs. Sometimes this relationship is less straight forward and biodiversity combined with other abiotic elements together provide an ecosystem service. In any case, conservation measures tailored to conserve biodiversity could be beneficial for many ESs.

An important strategy for biodiversity conservation is the establishment of protected areas (Chape et al. 2005). Protected areas are managed in order to reduce land degradation, and to improve biodiversity features such as species abundance or vegetation condition. If important areas for ESs are found inside protected areas, they are expected to continue providing ESs. It is therefore essential to understand if valuable areas that deliver multiple ESs are situated in protected areas or not. In fact, the establishment of protected areas presents an opportunity for safeguarding multiple ESs alongside biodiversity.

The aim of this chapter is to examine the degradation and transformation levels of regulating ES and the threats facing them using the case study of South Africa. Also, the conservation status of some regulating and cultural ESs have been examined.

Finally, the congruence between regulating services and cultural services are discussed. Here, the regulating service of carbon (C) sequestration and the cultural service of recreation were emphasized.

12.2 Cultural Ecosystem Service: Recreation

The most comprehensive definition of cultural ESs is given in the MA report (MA 2003). These are the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences; those include cultural diversity, spiritual and religious values, knowledge systems, educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values and recreation, and ecotourism. Recreation is sometimes used similarly or combined with tourism services. Recreational activities include hiking, biking, swimming or game viewing. Recreational activities may be carried out in natural or semi-natural areas. However, protected areas remain a key element for recreation and tourism. For example, it is estimated that about 84 million people have visited protected areas in Australia in 2006 (Pickering and Hill 2007).

A quick review of the indicators used globally for mapping ESs using peer reviewed literature from 1998 until June 2011 was carried out. Cultural ESs found were aesthetic enjoyment, cultural inspiration, recreation and tourism. Among the cultural ESs, recreation and tourism were the most studied with about one third of the 67 studies considering recreation or tourism. Recreational services were often expressed as areas with forest cover, clean water resources, natural areas, protected areas, areas with fish abundance or areas with high visitor numbers (Egoh et al. 2012).

12.3 Regulating Ecosystem Service: Carbon Sequestration

Regulating ESs are the benefits people obtain from ecosystem processes, including air quality maintenance, climate regulation, erosion control, regulation of human diseases, and water purification (MA 2003). C sequestration, i.e. the removal of O_2 from the atmosphere, is a regulating service under the GHG regulation ESs (De Groot et al. 2002; MA 2003). C storage is the storage of the removed GHG in the soil, vegetation or sediments. Many scientists use C storage capacity of soils and biomass to quantify C sequestration services (Chan et al. 2006; Egoh et al. 2008; Anderson et al. 2009; Reyers et al. 2009; Eigenbrod et al. 2010). The underlying concept implies that C released due to land transformation affects climate regulation negatively as associated carbon dioxide (CO_2) exacerbates climate change (Foley et al. 2005).

Amongst the services that have been quantified and valued, C sequestration is one of the most assessed services (Egoh et al. 2012). Over 80 % of studies that have mapped regulating services have considered C sequestration (Egoh et al. 2012).

A third of these studies used aboveground biomass to quantify this service, while others used soil organic C, forest biomass, and net primary production (NPP). The frequent assessment of this ecosystem service can be linked to the increasing evidence of climate change as a result of increasing anthropogenic CO_2 emissions, and associated national and global policies. An example is the Kyoto protocol and the United Nations Programme, REDD. The Kyoto protocol is a protocol to the UNFCCC which is a global treaty aiming to achieve the stabilization of GHGs. REDD is also a global effort to improve C stored in forests, offering incentives for developing countries to reduce emissions from deforestation (http://www.un-redd.org). The IPCC has also been set up by the UNEP and the WMO to perform scientific research on climate change and to report on trends. Scientific support for such policies includes the understanding of emissions from different areas, C storage in different ecosystems as well as rates of deforestation.

The aim of policy instruments such as REDD is to establish financial incentives to curb emissions. This is part of a larger concept known as PES which is gaining momentum around the world (Pagiola et al. 2005). PES is a scheme which is encouraging enhancers of ES financially to continue sustainable land management by those who benefit from these ESs provided. In addition to PES, a financial market for trading CO_2 is already in place. C trading markets were developed to bring buyers and sellers of C credits together using standardized rules of trade (http://www.carbontrading.com). These financial incentives coupled with various policy instruments have resulted in increased interest in the quantification of climate change related ESs such as C sequestration. The overall goal is to encourage sustainable land use practices and reverse land degradation to ensure the continuous provision of C sequestration.

12.4 Degradation of Regulating Ecosystem Services

12.4.1 Effects of Land Degradation on Ecosystem Services

Land degradation and unsustainable uses of ESs have different effects on these services. For example, land cover change may influence the water cycle such as groundwater recharge and surface water runoff (Schulze 2000; Scanlon et al. 2005). The effects of cultivation and plantations on water resources include changes in water runoff, reduction of water flows, and for example increased deposition of minerals and other harmful materials in water bodies (Schulze 2000; Tong and Chen 2002). Cultivation and plantations mainly affect soils by causing nutrient depletion and soil erosion (Houghton et al. 1999; Post and Kwon 2000; Canadell 2002; Wu and Tiessen 2002). Continuous cultivation of land would induce net soil C losses mainly by emissions into the atmosphere (Lal 2004; Chaplot et al. 2009). Change in land cover and the subsequent effects on ecosystem processes such as water runoff and soil properties do not only affect the quality of the service produced, but reduce the total area and thus the quantity of services produced.

12.4.2 Case Study: Transformation of Hotspots and Ranges of Five Ecosystem Services in South Africa

12.4.2.1 Study Area

South Africa is approximately 1.22 million km^2. The country is semi-arid, with an annual rainfall that varies between 50 and 3000 mm. A significant proportion of the country's 47 million people (Stats-SA 2005) live in rural areas, where livelihoods directly depend on ESs. The country is exceptionally rich in biodiversity with high levels of endemism, containing three global biodiversity hotspots (Mittermeier et al. 2005). About 16 % of the country's total area is not in a natural state due to land uses such as conversion of natural land to agricultural land, mining areas and plantations (Rouget et al. 2004). In highly productive areas such as grasslands higher levels of land conversion (35 %) have been reported (Reyers et al. 2005).

12.4.2.2 Quantification of Five Ecosystem Services in South Africa

Egoh et al. (2008) mapped the hotspots and extent (from now on referred to as range) of five ESs in South Africa (including C storage, soil retention and accumulation, water regulation and supply). The range of each service was defined using the areas that produce an average amount of the service while hotspots were areas that produce a significantly higher than average amount of the service. C storage was mapped using the experience of experts in the field who estimated C storage within different vegetation types (Mills and Cowling 2006). South Africa has reliable spatially explicit data on soil properties which includes soil organic matter (SOM), soil erodibility, soil structure, soil N content, soil depth and others (Schoeman et al. 2002). This dataset and data on vegetation types were used to map soil retention and accumulation, one of 23 ESs listed by De Groot et al. (2002). Soil retention is the ability of ecosystems to protect soils from erosion, while soil accumulation is the ability of the environment to retain soils and nutrients. Soil protection was mapped using the soil vulnerability to erosion and the vegetation cover. The soil erodibility data was extracted from Schoeman et al. (2002) while experts scored data on vegetation types based on soil retention. Soil accumulation was mapped by using the litter accumulation index from Schulze (2004) and by considering soil depth (Schoeman et al. 2002). Water regulation was mapped using data including ground water contribution to base-flow from DWAF (2005). Water supply was mapped using water runoff data from a revised South African environmental Atlas (Schulze 2004). The ranges and hotspots were delineated in each of the maps. The full details of the mapping approach are given in Egoh et al. (2008), where mostly proxies were used. The use of proxy data in mapping ESs has been criticized but in the absence of primary data, proxy data is useful for identifying broad trends in ESs (Eigenbrod et al. 2010).

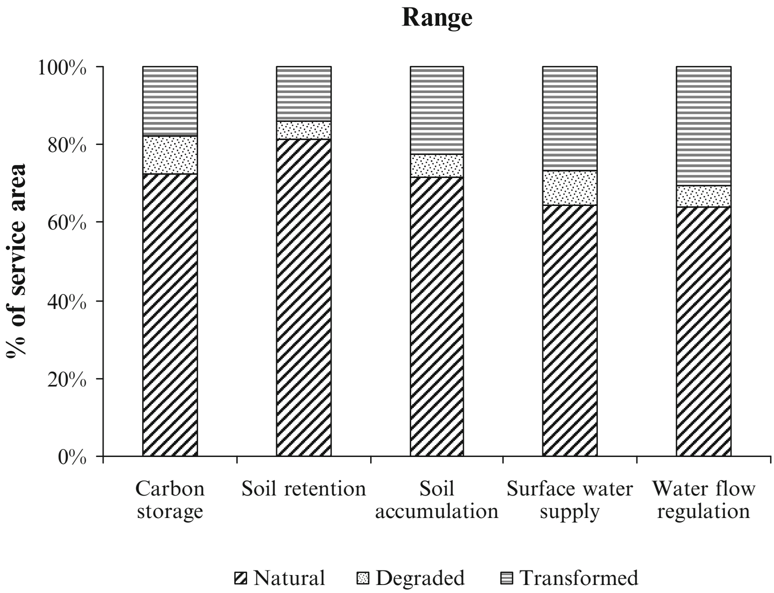

Fig. 12.1 Transformation and degradation status of range of five ESs in South Africa

12.4.2.3 Degradation and Transformation Analysis

Two datasets were used in this analysis including land cover data (Thompson 1996) and maps of five ESs (Egoh et al. 2008). The land-cover map consisted of natural areas and various forms of transformations (e.g., cultivated land, plantations and built-up area). This first data set has been used extensively in the conservation sector in South Africa (Reyers et al. 2002; Van Jaarsveld et al. 2005), although a newer land cover data has been published (Fairbanks et al. 2000). This is because the land-cover data from 1996 is believed to be more accurate than the land-cover dataset from the year 2000.

The land cover data was classified into three broad categories, natural, degraded, and transformed (cultivated areas, mining areas, forest plantations and urban areas; Thompson 1996). The land cover map was overlaid with the ESs range and hotspots maps from Egoh et al. (2008) using ArcGIS. The percentage of the natural, transformed or degraded areas were calculated considering the range and hotspot. Calculations were also performed for the percentage of each land use type within the transformed areas in each hotspot or range for each ecosystem service.

12.4.2.4 Results and Discussion

Results showed that at least 10 % of the total area of the hotspot and 20 % of the total area of the range of all five ESs have been transformed or degraded (Figs. 12.1 and 12.2). The range of surface water supply and water flow regulation were the

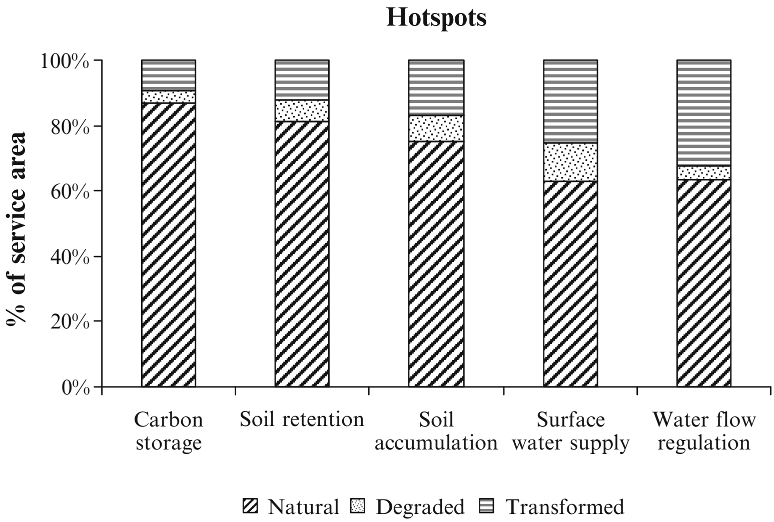

Fig. 12.2 Transformation and degradation status of hotspots of five ESs in South Africa

most transformed. At least 25 % of the total area of the range of these two ESs was transformed. Such levels of transformation combined with at least 7 % degradation for both services are unsustainable for a semi-arid country like South Africa. This problem is compounded by invasive plants which affect the country's ground water resources negatively (Le Maitre et al. 2000). Van Wilgen et al. (2008) showed that South Africa's ground water resources have been reduced by about 7 % from invasive alien plants alone. Water supply is an essential ecosystem service in South Africa because of its importance in agriculture, tourism and mining. About 30 % of the country's crop production rely on irrigation, more precisely about 1.3 million hectares of land under irrigation (Bennie and Hensley 2001). Degradation and transformation of areas which are important for water production combined with the problem of water scarcity could result in serious negative effects for the economy. There is an urgent need to stop any further transformation and degradation in areas important for capturing and storing water in South Africa.

Other trends observed showed that although the water was the most transformed, the range of C storage was the most degraded. At least 10 % of the range of the C storage service has been degraded while more than 15 % is transformed. These results are supported by findings of Reyers et al. (2009) who showed that the supply of the services of forage production, C storage, and tourism was reduced by at least 25 % in their study area (Little Karoo, South Africa). Transformation of areas important for storing C contributes to climate change due to increased emissions of CO_2. Reyers et al. (2009) also reported higher levels of degradation for soil erosion (44 % of its original extent). Soil erosion depletes soil nutrients and decreases productivity. Soil erosion is not only detrimental to soils that lose their nutrients as mentioned above, but eroded soil material may also end up in water bodies

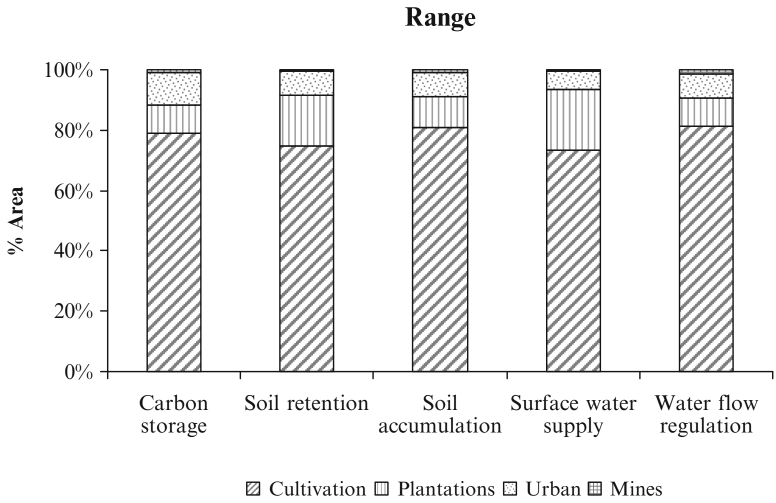

Fig. 12.3 Percentage area of range of five ESs transformed by cultivation, plantations, urban areas or mines in South Africa

loading them with nutrients and promote eutrophication making them unsuitable for human use.

Within the transformed areas of the five ESs that were considered in this study, cultivation remains the biggest threat to the ESs considered. More than 60 % of all transformation that occurred within the hotpots and ranges of all five ESs could be attributed to cultivation (Figs. 12.3 and 12.4). These results illustrate the kinds of trade-off that exist between ESs. While cultivation is important for food production it may be detrimental to regulating services such as water regulation and soil retention as discussed above. The presence of cultivated land may not necessarily influence recreation and tourism. Many tourists may enjoy flowers on agricultural land or landscape sceneries with both natural and cultivated areas (Allali 2006). Another threat is the existence of plantations in areas important for the provision of ESs in South Africa. These plantations, which consist mostly of non-native plants, have been shown to affect water provision negatively because they mostly use more water than native trees (Le Maître et al. 2000). However, plantations support C storage. Again, the importance of understanding trade-offs could not be overstated. Tree planting is encouraged in many restoration projects around the world for the benefit of C capture and storage (Omeja et al. 2011). Given the potential of negative effects caused by alien trees on biodiversity and other ESs, it is important that operating companies consider planting native species.

Despite the low threat posed by urbanization and mining to all services, it was interesting to observe that 19 % of the hotspot and 10 % of the range of areas important for C storage which have been transformed are urban areas. Urbanisation and urban sprawl affects almost all ESs negatively. Despite these negative effects, urban areas around the world still continue to grow (Cohen 2006). Planting of trees

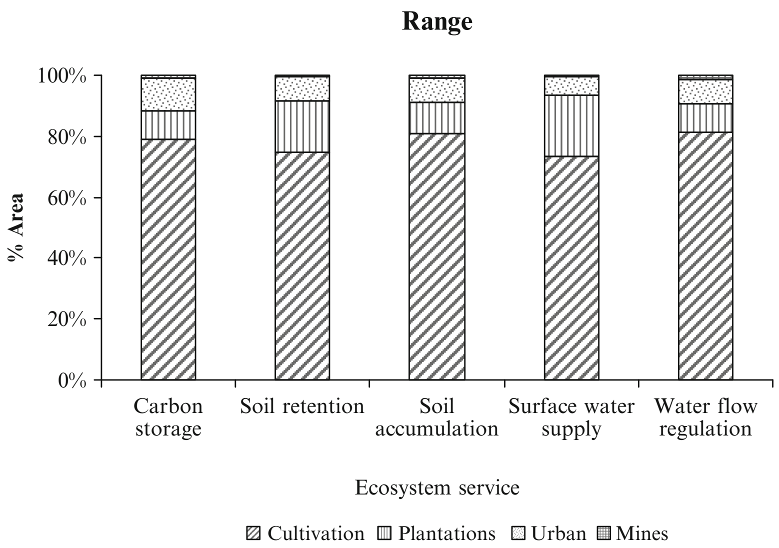

Fig. 12.4 Percentage area of hotspots of five ESs transformed by cultivation, plantations, urban areas or mines in South Africa

in urban areas should highly be encouraged thus offsetting some of the losses of C storage through the conversion of natural into urban areas.

There is an urgent need for identifying and managing areas that are important for the provision of ESs (Balvanera et al. 2001; Kremen and Ostfeld 2005; Chan et al. 2006). Important areas for ES delivery managed with restrictions by law (e.g. protected areas) will prevent land degradation and unsustainable use of ESs in the future. Conservation action that could be employed to safeguard ESs includes land management such as the establishment of protected areas or restoration of land to improve its ecological functioning. In fact, restoration of land can take place in protected areas. However, a single management approach is mostly not sufficient for all ESs.

12.5 Ecosystem Services in Protected Areas

A key management approach that has been employed for maintaining biodiversity is the establishment of protected areas which involves setting aside land regulated by law for protection (Chape et al. 2005). The identification of land for protection, unfortunately rarely considers ESs (Egoh et al. 2007).

Historically, the selection of areas for conservation in many countries around the world was mostly driven by socio-economic and political aspects and less thought has been given to biodiversity. A major reason for land conservation was the relative lack of value for major commercial land uses or for human settlements or other important reasons such as scenery, recreation, tourism potential, the influence of lobby groups, and historical protection for uses such as hunting or water supply (Pressey 1994). Frequently, this resulted in a selection of areas which were not representative for local or regional biodiversity (Rebelo 1997; Reyers et al. 2001; Rouget et al. 2003) and biased towards unproductive and economically marginal landscapes (Turpie et al. 2003).

In an attempt to ensure that protected areas contained the necessary biodiversity elements, many other approaches have been advanced including scoring approaches, expert approaches, minimum set of sites, and complex systematic conservation planning tools (Sarkar et al. 2006). Scoring systems were developed in the 1970s to provide an explicit and rational basis for selecting conservation areas (Margules and Usher 1981; Terborgh and Winter 1983; Smith and Theberge 1986). The minimum-set approach introduced in the 1980s aimed to identify the smallest area that conserved the greatest number of species. At present, the identification of protected areas has become a more complex process. Systematic conservation planning comprises a selection of sites which are based not only on biodiversity species data but also on the contribution of areas to ecological processes and in some cases to the delivery of ESs (Margules and Pressey 2000).

Although some protected areas have been established previously for the purpose of ESs such as recreation or water regulation and provision, ESs are rarely included in prioritization procedures (Egoh et al. 2007). Two analyses were used to evaluate the extent ESs are being delivered by protected areas and to assess the opportunities for conservation of areas important for ESs. The first analysis used ecosystem service maps created by Egoh et al. (2008) mentioned above and overlaid with a map of protected areas in South Africa. The protected area data were obtained from the South African National Parks (SANParks) which manages a system of parks representing the indigenous fauna, flora, landscapes and associated cultural heritage of the country. The data consisted of the names of category 1 and 2 protected areas in South Africa. Category 1 reserves are strict nature reserves consisting of some outstanding or representative ecosystems, including geological or physiological features and/or species, which are managed mainly for scientific research and monitoring or which are wilderness areas managed mainly for wilderness protection. Category 2 reserves are national parks managed mainly for ecosystem protection and recreation. These two categories are the most well managed protected nature reserves in South Africa. Although private nature reserves (category 3) can provide a significant level of protection for biodiversity and ESs (Gallo et al. 2009), these were excluded because of the lack of reliable information.

Results showed that less than 20 % of the hotspots and 20 % of the range of five ESs in South Africa were inside protected areas (Fig. 12.5). C storage had the highest area of hotpots and range (16 and 13 %, respectively) in protected areas. All other ecosystem service considered in the study had 10 % or less of their hotspot or range in protected areas. These results show that ESs in South Africa will have to be managed outside protected areas. Conservation strategies for ESs will have to be developed apart from biodiversity if ESs continue to be excluded in the selection of protected areas or protected areas expansion strategies. An example is a restoration

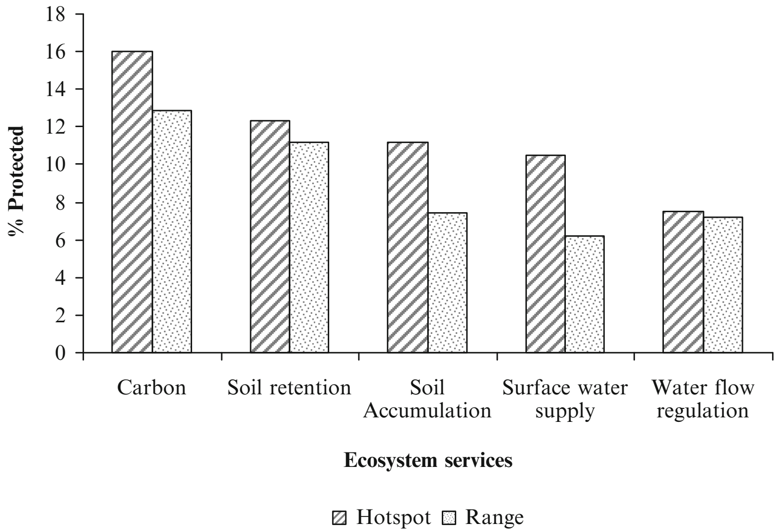

Fig. 12.5 Percentage area of hotspots and ranges of five ESs in protected areas in South Africa

project in South Africa that encourages the planting of trees for C sequestration (http://www.ercap.co.za).

The second analysis considered the spatial distribution of the proxies used for mapping ESs in Europe and retained areas that delivered ESs at values above the 10th percentile of the range. Subsequently, we assessed the overlap of these hotspot areas with NATURA 2000 sites, the most important conservation network in Europe. Again, C storage proxies had the most areas in the network of protected areas in Europe. Twenty percent of hotspot areas important for wood biomass production and 19 % of hotspot areas for soil organic C coincided with NATURA 2000 sites (Fig. 12.6). This level of securing ESs is generally low compared to levels of protection of habitat and species in protected areas. The European Union (EU) has included ESs in its conservation strategy and aims to restore at least 15 % of degraded land by 2020 to improve biodiversity conservation which delivers ESs. This target provides an opportunity to prioritize areas important for the provision of ESs as well as for biodiversity restoration.

Areas identified as conservation priorities for the establishment of protected areas could provide multiple benefits if both biodiversity and ESs are included in the planning process. Egoh et al. (2010, 2011) showed that there was much flexibility in reaching targets set for ESs compared to biodiversity and biodiversity targets can be met together with targets set for ESs such as C sequestration in one plan. These findings support the potential that exist in identifying areas for the establishment of protected areas, where both biodiversity and the delivery of multiple ESs are considered. Although protected areas provide a unique opportunity for safeguarding ESs, PES initiatives which promote financial incentives for managing land sustainably are also important towards safeguarding ESs. Truly, many ESs operate in scales larger than the scale at which protected areas are implemented. This justifies

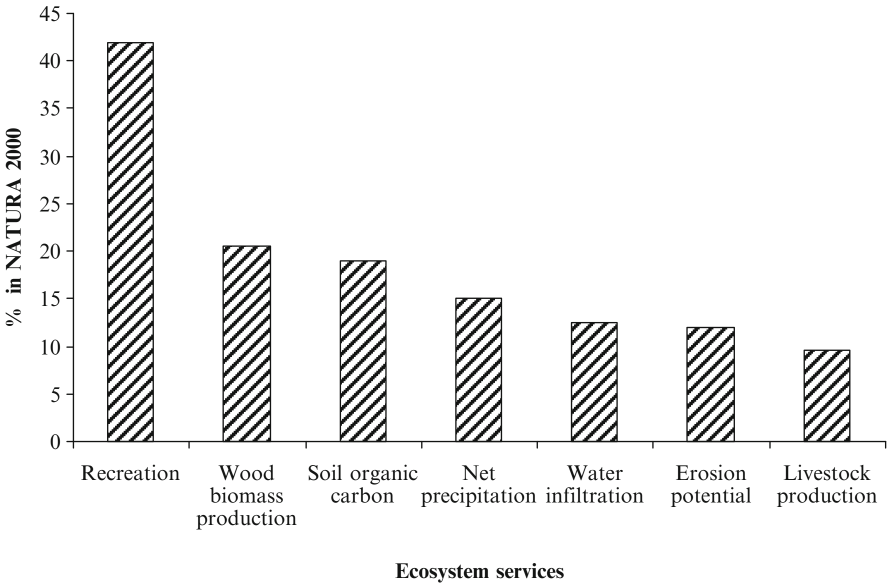

Fig. 12.6 Percentage area of top 10 % of ecosystem service indicators in NATURA 2,000 sites in Europe

a more integrated approach to landscape management. Some cultural ESs such as recreation may be managed at the scale of protected areas for hiking and wildlife viewing. However, the water regulating ecosystem service which provides clean water serving multiple purposes requires management at landscape scale including agricultural areas where a nutrient surplus through fertilization occurs.

The scales at which ESs operate are quite important in developing management strategies. Protected areas such as nature conservancies which usually cover larger areas than strict protected areas may become a very useful management strategy for regulating services. However, they are often neglected in biodiversity conservation because of the management style, which allows use of resources. Current initiatives in establishing trans-boundary conservation areas may proof to enhance many landscape scale ESs (http://www.tbpa.net). Whatever management approaches are developed, it is important to understand congruence between different ESs and if such management approaches will deliver multiple benefits.

12.6 Congruence Between Cultural Services and Carbon Sequestration

An important conservation action for biodiversity and for ESs is the restoration of land where such services have been degraded and where restoration could improve the delivery of ESs. For deciding where such restoration should take place, it is important to understand the effects of restoration on multiple ESs and with biodiversity to maximize benefits. If restoration is targeted towards C sequestration, it is important to know if such restoration will benefit other services such as recreation. It becomes necessary to understand the relationships among different ESs and between ESs and biodiversity.

Biodiversity underpins most ESs, but the relationship between the two is in general positive but weak (Turner et al. 2007; Naidoo et al. 2008; Egoh et al. 2009). The general impression on relationship among ESs is that, despite the often weak positive correlation, ESs are not necessarily congruent with each other and there is little evidence that safeguarding one service will deliver benefits to the others (Chan et al. 2006; Egoh et al. 2008; Anderson et al. 2009; Eigenbrod et al. 2010). Egoh et al. (2008) found a positive but weak correlation between various ESs in South Africa. They show that C storage was not congruent with any other ecosystem service except for a weak correlation with water retention. Chan et al. (2006) found a positive correlation between recreation and C storage and again, the explained variance was very low. They also found low (36 %) overlap of planning units shared between recreation and C storage. In contrast, Anderson et al. (2009) reported a negative correlation between recreation and C storage. Eigenbrod et al. (2010) also found a negative correlation between primary recreation data (measured as leisure trips) with both primary and proxy C storage data. These results suggest that increase in C storage will not increase recreational activities and recreational activities may even decrease with an increase in C storage.

Despite the findings above, there is evidence that there could be congruence between certain ESs. For example, forests are known to be suitable for C storage, water regulation and recreational activities. Bai et al. (2011) found an overlap of 66 % of hotspots of C sequestration and forest area, and a 48 % overlap of forest area and water provision hotspots. Forests in Tanzania which store large amounts of C are also used for tourism and recreation (Zahabu et al. 2005). Ongoing research in Chile has started to demonstrate the importance of native forests in the provision of ESs that directly or indirectly benefit society, such as water supply (both quantity and quality), tourism, recreational fishing opportunities and biodiversity conservation (Nahuelhual et al. 2007; Lara et al. 2009).

The reason for different trends in relationships between different ESs could be the data used in these studies. It appears that different indicators used to quantify ESs relate differently to each other. For example, recreation and tourism could be mapped using only accessibility, or using accessibility and naturalness, or a combination of these two indicators. Also water quality data or using visibility from touristic routes could be used to map recreation services (Chan et al. 2006; Reyers et al. 2009; Maes et al. 2011). If only water quality is used to map recreational activities, it is very likely that it will not correlate with C storage or sequestration. But if forest cover in natural areas is used to map C storage and protected areas in a forested region is used to map recreational services, it is very likely that those two parameters are highly correlated. At present, many different indicators are being used to map the services of C sequestration and storage. These include NPP, above ground biomass, soil organic C storage and SOM (Egoh et al. 2012). Consequently, congruence analysis with combinations of different indicators is more likely to produce different results.

12.7 Conclusions

Many regulating ESs continue to be transformed. In South Africa, cultivation and plantations remain the biggest threats. This problem is compounded by the fact that range and hotspot areas of many ESs remain outside protected areas. The lack of inclusion of ESs in prioritization procedures to establish new nature reserves is an essential reason for the lack of ecosystem service hotspot areas in the networks of protected sites. Urgent conservation actions such as restoration of ecosystems are needed to safeguard these services. Such conservation measures should take congruent patterns among different ESs into consideration and ideally work with aggregates or bundles of ESs rather than single ESs. The establishment of protected areas could be instrumental for safeguarding multiple ESs which include regulating ESs such as C storage and cultural ESs such as recreation. However, conservation measures for ESs will have to be developed in addition to the establishment of protected areas to safeguard them.

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Chapter 13
Human Appropriation of Net Primary Production, Stocks and Flows of Carbon, and Biodiversity

Helmut Haberl, Karl-Heinz Erb, Simone Gingrich, Thomas Kastner, and Fridolin Krausmann

Contents

Abstract The human appropriation of net primary production (HANPP) is an integrated socioecological indicator of land-use intensity. HANPP is defined as the alteration of the availability of biomass in ecosystems resulting from (1) changes in net primary production (NPP) induced by current or past land use and (2) biomass harvest. In this chapter we discuss how HANPP can be extended with data on carbon (C) stocks in biota and soils to forge an integrated stock-flow account of C in ecosystems. Using comprehensive data for Austria in the period of 1830–2000 as an example, we illustrate the usefulness of such accounts to improve our understanding of human impacts on the stocks and flows of C in ecosystems – an important component of the human alteration of the global C cycle. Austria's agrarian-industrial transition was accompanied not only by a tremendous growth of fossil-fuel related C flows, but also by profound changes in C stocks and flows in biota and soils. Fossil-fuel related emissions increased from almost zero to 14.6 Tg C year while, at the same time, a land-based C sink of 2.6 Tg C year emerged. These trends were related causally because the use of fossil fuels in agriculture supported agricultural intensification

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which helped raising yields. This in return resulted in shrinking farmland and growing forest (as well as settlement) areas. Stocking densities in forests grew as well. While the strength of an integrated C accounting system is being analyzed and discussed, we also review the current state of knowledge regarding potential interrelations of HANPP and biodiversity.

Keywords Human appropriation of net primary production • Biomass • Carbon flow accounting • Carbon sink • Carbon emission • Carbon stock • Long-term socio-ecological research (LTSER) • Socioeconomic metabolism • Austria

Acronyms

ΔNPP_{LC}	Productivity changes induced by land use
C	Carbon
EROI	Energy return on investment
GDP	Gross domestic product
GHG	Greenhouse gas
HANPP	Human appropriation of net primary production
LTSER	Long-term socio-ecological research
NPP	Net primary production
NPP_h	Biomass harvest
SOC	Soil organic carbon

13.1 Introduction

Through land use, humans alter patterns and processes in terrestrial ecosystems. Biota play a central role in the global carbon (C) cycle as vegetation absorbs and plants as well as heterotrophic organisms release huge amounts of carbon in photosynthesis and respiration, respectively. Because land use alters both of these processes, it affects the global biogeochemical cycle of C.

The magnitude of these land-use effects has become quite noteworthy. Approximately three quarters of the earth's ice-free land surface is used by humans for settlements, cropping, grazing and forestry or is at least substantially affected by human activities or infrastructures (Sanderson et al. 2002; Erb et al. 2007a). Land-use related changes in the C cycle vary around the globe, especially between industrialized and developing countries (Houghton 2005). According to a recent study (Richter and Houghton 2011), the yearly gross C flux from vegetation to the atmosphere amounts to approximately 4.3 Pg C year^{-1} (1 Pg = 10^{15} g), 70 % of which is emitted in the tropics. The land-based gross C sink is estimated at 2.8 Pg C year^{-1}, most of which is located in the northern hemisphere. Both numbers are subject to considerable uncertainties (Denman et al. 2007). The net terrestrial source of C to the atmosphere resulting from land-use change is hence approximately 1.5 Pg C year^{-1}. In contrast, results from inverse modelling suggest that terrestrial ecosystems absorb approximately 1.3 Pg C year^{-1} which is often interpreted as a result of atmospheric change (C fertilization) and climate change (Field and Raupach 2004; Denman et al. 2007; Pan et al. 2011; Richter and Houghton 2011).

Hence it may be said that C flows in terrestrial ecosystems are simultaneously affected by natural and socioeconomic drivers. Ecological C flows are co-determined by factors such as geomorphological conditions, climate and climate change, the atmospheric C concentration, species assemblages or pedology. Additionally, human activities play a central role for these C flows; such activities include: the replacement of natural vegetation with infrastructures, agroecosystems (croplands, meadows and grazing land) or managed forests and management measures such as plowing, fertilization, application of pesticides, and irrigation. The recognition of the manifold systemic interlinkages between socioeconomic and natural drivers in co-determining the trajectory of terrestrial systems has motivated the formulation of the interdisciplinary research agenda of 'integrated land-change science' which also aims to contribute to a systemic understanding of the global C cycle and its natural as well as socioeconomic drivers (GLP 2005; Turner et al. 2007).

C is equally important for ecosystem functioning as well as for socioeconomic processes. Activities such as agriculture and forestry primarily aim at providing biomass (food, feed, timber, etc.) and hence C for socioeconomic metabolism, i.e., the material and energy flows associated with all human activities (Martinez-Alier 1987; Ayres and Simonis 1994; Fischer-Kowalski 1998; Fischer-Kowalski and Hüttler 1998). Next to the C emissions that result from the combustion of C-rich materials such as fossil fuels and processes such as cement manufacture (i.e., socioeconomic metabolism), the use of biomass is associated with alterations of stocks and flows of C in ecosystems (Canadell et al. 2010; Raupach and Canadell 2010). Thus, changes in society's C metabolism as well as in stocks and flows of C in biota and soils over time are inextricably entangled (Erb et al. 2008; see Fig. 13.1 and discussions below).

One prominent indicator of human-induced changes in ecological C flows is the human appropriation of net primary production (HANPP). HANPP is an indicator of land-use intensity that assesses human-induced alterations of the net primary productivity (NPP) of ecosystems. It accounts for the difference between (1) the NPP of ecosystems in the hypothesized absence of human land use (approximated as 'potential natural vegetation') and (2) the NPP remaining in the ecosystem after harvest under current conditions, i.e., the NPP remaining in ecological cycles (food webs, build-up of ecological C stocks; Vitousek et al. 1986; Wright 1990; Haberl et al. 2007a; Erb et al. 2009a). This chapter presents the HANPP indicator and explains its relevance for ecological C flows, demonstrates how HANPP can be combined with data on ecological C stocks to form a comprehensive C accounting system that can help understanding the various interrelations depicted in Fig. 13.1 and discusses the example of Austria 1830–2005 to show what can be learned from such accounts.

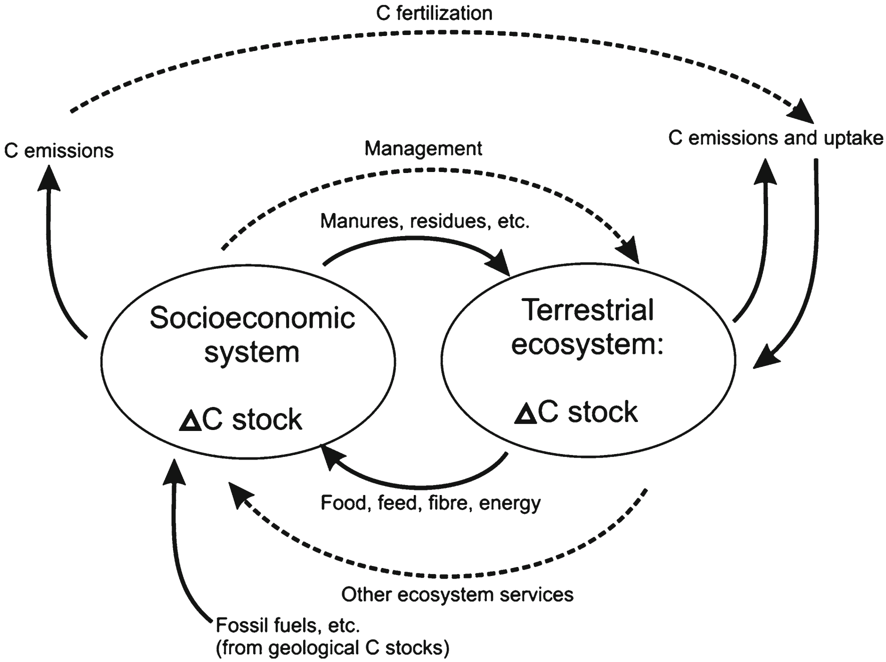

Fig. 13.1 Socio-ecological C interactions. Total human-induced C emissions to the atmosphere include C emissions from socioeconomic resource use (fossil fuels and biomass) as well as the effects of human activities on C emissions and uptake of terrestrial ecosystems (biota and soils). *Solid arrows* denote C flows, *dashed arrows* denote systemic feedbacks in the coupled socio-ecological system

13.2 The Concept of Human Appropriation of Net Primary Production

Plants use radiant energy from the sun to convert inorganic substances (CO_2, H_2O, nitrogen, potash, phosphorous, etc.) into organic materials (biomass) that store chemical energy. The amount of biomass produced by green plants per year minus the amount of fixed C spent on the plants' own metabolic needs is net primary production (NPP). NPP can be measured as annual flow of biomass (g year^{-1}), as annual C flow (g C year^{-1}) or as annual energy flow (J year^{-1}).

The organic material produced enters ecological food chains or accumulates within the vegetation (i.e., in living plants) or in the soil. If C absorbed from the atmosphere accumulates in biota and/or soils, ecosystems act as a C sink. Ecosystems may also represent a net source of C to the atmosphere, e.g., when forests die back, plants use their reserves during dry periods and also as a result of human activities such as deforestation. In the absence of climate change or human interference, most larger regions are on average C neutral relative to the atmosphere over long periods of time, although C in- and outflows occur at different speed ('slow in, fast out',

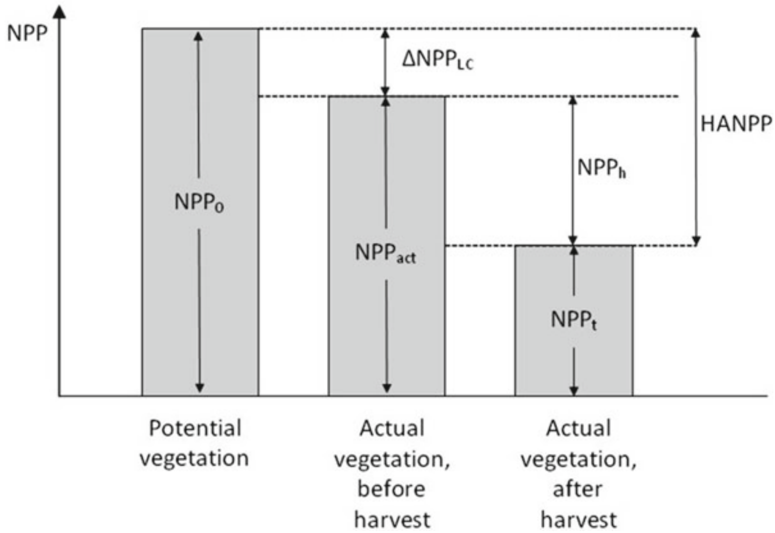

Fig. 13.2 The human appropriation of net primary production is defined as (1) the sum of productivity changes induced by land use (ΔNPP_{LC}) and biomass harvest (NPP_h) and (2) the difference between the NPP of potential natural vegetation (NPP_0) and the fraction of current NPP (NPP_{act}) that remains in the ecosystem after harvest (NPP_t)

Körner 2003, 2009). In addition to its importance for C stocks and flows, NPP is needed in ecosystems as ultimate energy resource. NPP is indicative of the amount of trophic energy available in ecosystems for animals and microorganisms ('heterotrophic organisms').

Humans alter the amount of NPP available in ecosystems through two closely related processes. Firstly, human influence on ecosystems potentially changes their NPP. For example, if natural vegetation is replaced with settlements, industrial areas or infrastructure, sealed surfaces will have little, if any, NPP while adjacent un-sealed land may be more productive (e.g., gardens, parks). In agro-ecosystems, NPP is also directly influenced by management activities such as irrigation, fertilization, etc. As a result, the NPP of agro-ecosystems is often different from the NPP of the natural ecosystems they replace. This process of direct human alteration of NPP is denoted as the change in productivity resulting from land conversion (ΔNPP_{LC}). Secondly, humans extract biomass from ecosystems for their own use, thereby rendering it unavailable for C sequestration and as trophic energy source for other heterotrophic organisms. This process is denoted as the harvest of NPP (NPP_h). HANPP may then be defined as the ΔNPP_{LC} plus NPP_h (e.g., Haberl et al. 2007a). HANPP can be expressed in the same units as NPP, i.e., flows of biomass, C or energy.

This basic concept (Fig. 13.2) has been implemented using varying definitions and names or acronyms (Whittaker and Likens 1973; Vitousek et al. 1986; Wright 1990; Imhoff et al. 2004). The above-quoted HANPP definition ($HANPP := \Delta NPP_{LC} + NPP_h$) is used in many current studies, and was also used in deriving a global HANPP map for the year 2000 (Haberl et al. 2007a). In this study, NPP_h was defined rather inclusively and encompassed all biomass extracted by humans or livestock, parts of plants killed during harvest (e.g. roots of trees or annual crop plants) that are not recovered, and biomass burned in human-induced fires. Depending on the inclusiveness or exclusiveness of the definitions used, most studies found global HANPP to be between 14 and 39 % of NPP_0 (Vitousek et al. 1986; Wright 1990; Imhoff et al. 2004; Haberl et al. 2007a).

HANPP represents changes in C flows resulting from past and present land use and is, therefore, relevant for global biogeochemical cycles. For example, an increase in HANPP indicates an increasing 'human domination' of the biosphere (Vitousek et al. 1997), including the significance of human activities for the global C cycle. However, the relation between HANPP and C flows to the atmosphere is complex. For example, HANPP calculations do not distinguish C which society rapidly returns to the atmosphere (e.g., C in food, feed or bioenergy) and C which is sequestered in long-lived socioeconomic stocks such as timber used for housing and construction or furniture. Moreover, land use is not only associated with a certain harvest level (NPP_h) and a change in the productivity of the vegetation (ΔNPP_{LC}), but often also with changes of the amount of C stored in biota and soils. As will be discussed below for Austria, land-use change has a strong impact on the standing crop (i.e., the amount of C stored in living plants) and soil organic carbon (SOC) if forests are replaced by croplands or grasslands and vice versa. Hence, land use affects not only NPP and its availability in ecosystems but also the turnover of organic C, e.g., through changes in decomposition (Haberl et al. 2001; Erb 2004a). An increase of HANPP, therefore, does not necessarily result in C loss. A reduction of HANPP may be associated with a land-based C sink, but this is not necessarily the case. In order to understand the interrelation of land use with sources and sinks of C it is useful to extend the HANPP concept by also including data on C stocks in biota and soils.

13.3 Integrated Socio-Ecological Carbon Stock-Flow Accounts

As discussed in the previous section, HANPP is useful to keep track of changes in important annual C flows in terrestrial ecosystems related to human activities. However, it is by far not sufficient to understand the human-induced C flows that need to be accounted for in an integrated socioecological stock-flow account. Figure 13.3 presents a scheme of the C stocks and flows that need to be considered in such an accounting system.

One essential component that needs to be added to the flows comprised in the HANPP account are the C flows related to socioeconomic metabolism, e.g., fossil-fuel combustion, or production processes such as cement manufacture. As HANPP only comprises biomass flows related to domestic extraction (harvest within the territory), horizontal C flows related to trade of food and feed commodities have to be added. Data on these socioeconomic C flows can be derived from data included in material and energy flow analyses (MEFA; Haberl et al. 2004a). The MEFA

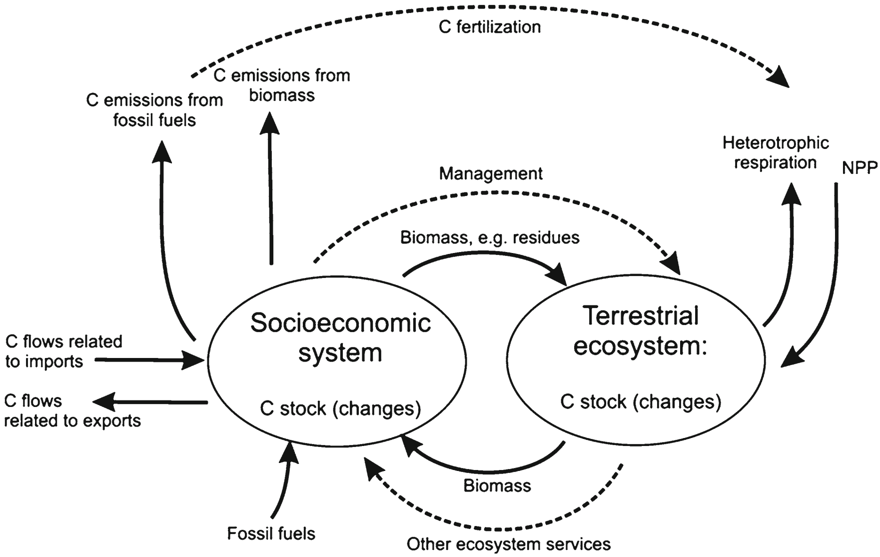

Fig. 13.3 Scheme of terrestrial C stocks and flows to be accounted for in an integrated socio-ecological C account at the national level. Stock changes result from the net balance of C input and output in each subsystem (*NPP* Net primary production). *Solid arrows* denote C flows, dashed arrows denote systemic feedbacks

framework provides a basis to establish accounts of the flows of materials, energy or substances such as C related to socioeconomic processes, production and consumption. Methods of material flow analysis (Eurostat 2007) and energy flow analysis (Haberl 2001) are based on common definitions of system boundaries and consistent accounting principles. They allow tracing the flows of materials and energy through national economies or other socioeconomic subsystems (Weisz et al. 2006; Fischer-Kowalski et al. 2011).

Based on such accounts it is also possible to establish C flow accounts that allow tracing the flows depicted at the left-hand side of Fig. 13.3, i.e., C contained in fossil fuels and the carbon dioxide (CO_2) from their combustion, biomass flows from ecosystems to society as well as backflows to nature, and CO_2 from food and feed consumption and biomass combustion. Some of these flows are readily available from material flow accounts (MFA), some need additional data research and calculations (Erb et al. 2008). Accounts of socioeconomic C stocks as well as their changes over time can also be derived from such databases and accounts, but consistent and comprehensive C stock accounts are missing (for examples of partial accounts see Winjum et al. 1998; Churkina et al. 2010; Dias et al. 2011).

Trade flows merit special attention. First, trade is an important source of fossil energy carriers (coal, oil and natural gas) for many countries with little or no domestic resources (such as Austria). Secondly, biomass trade also plays an increasing role. Most densely populated countries derive a considerable fraction of the biomass-based products (food, feed, fibre and bioenergy) from imports whereas countries with a low population density tend to be net exporters (Erb et al. 2009b). Thirdly, CO_2 emissions associated with the provision of imported and exported products need to be considered when establishing a national C balance. MFA-based methods only include data on products that actually cross the border. Thus, they can be used to account for C in traded biomass or fossil energy, but not for the CO_2 emissions in the respective 'upstream' product chain. These can be calculated using two distinct approaches, (1) life-cycle analysis (see Zhang et al. 2010; Lewandowska 2011) or (2) input–output analysis (Lenzen et al. 2007; Wiedmann et al. 2007). While greenhouse gas (GHG) accounts related to the fossil fuels 'embodied' in traded products have recently become available (Peters and Hertwich 2008; Hertwich and Peters 2009; Peters 2010), similar accounts are missing for the CO_2 emissions related to biomass trade (see Gavrilova et al. 2010; Kastner et al. 2011a, b for first steps in this direction).

Turning to the right-hand side of Fig. 13.3, it is obvious that HANPP can help to address the issue of management-induced changes in NPP as well as, when combined with material flow analysis (MFA), the biomass flows between the ecosystem and the socioeconomic system (harvest as well as backflows). However, as noted above, land use also affects ecological C stocks as well as C turnover. This is not covered in the HANPP framework which only assesses C flows. For an integrated socio-ecological C account, however, a consistent integration of C stocks with flows is imperative (Körner 2009). Most of the C absorbed by green plants during photosynthesis is metabolised either by plants or by heterotrophic organisms and, thereby, released back to the atmosphere. Compared to these annual flows, net changes in C stocks – either in the soil, e.g., as SOC, or aboveground in the standing biomass stocks ('standing crop') – are usually small. Estimating the net release ('source') or net absorption ('sink') of C, therefore, requires the assessment of C stocks in biota and soils at different points in time. If the stock is growing, one can assume that biota and soils have acted as a C sink, while in the opposite case they acted as a source, i.e., emitted C to the atmosphere.

13.4 The Example of Austria for the Period 1830–2000

The usefulness of such an extension of the HANPP framework can be illustrated through an empirical example. This section presents and discusses results from recent research on changes in C stocks and flows in Austria from 1830 to 2000.[1] The analysis of changes in C stocks and flows in Austria through almost two centuries provides an example of how the above-discussed methods and approaches add value to studies of national C flows in Long-Term Socio-Ecological Research (LTSER; see Singh et al. 2012).

[1]For details see Krausmann (2001), Krausmann and Haberl (2002), Erb (2004a), Gingrich et al. (2007), Erb et al. (2008).

The data situation for Austria allows for in-depth analyses of temporal trends: extended historic databases exist, reaching back to the early nineteenth century and covering the majority of the municipalities in the Austrian-Hungarian Empire. Additionally, 50-year time series of forest inventories (Weiss et al. 2000), and detailed land use and energy statistics (Haberl et al. 2003), provide the basis for the development of an integrated C account at the national scale. Austria is a small, highly industrialized Central European country, with medium population density (area 83,000 km^2, current population about 8 million, 96 inhabitants per km^2). The majority of the country's surface is dominated by the Alpine mountains, which implies the existence of complex, three-dimensional continua of abiotic gradients, e.g., along mountain slopes, and a heterogeneous mosaic of land cover and land use. Austria currently has a comparably high share of forest cover of 47 % (the average for the EU-27 is 37 %). Forests prevail mainly in hilly and mountainous areas of central and western Austria, whereas the lowlands are mostly dominated by intensive agriculture, with 15 % of the total area being croplands and about 4 % infastructures and settlements (Krausmann 2001).

During the period 1830–2000, Austria underwent a sociometabolic transition from an agrarian to an industrial society (Krausmann et al. 2008). Population grew by a factor of 2.3 from 3.6 to 8.1 million. The agrarian share of the population (i.e., farmers and their families) dropped from 75 to 5 %, and the contribution of agriculture to gross domestic product (GDP) decreased to 1.4 % in the year 2000, while total GDP rose by a factor of 28 and per-capita GDP by a factor of 12 (Krausmann and Haberl 2007).

In the first half of the nineteenth century, Austria was still a predominantly agrarian country.[2] In 1830, biomass accounted for 99 % of the socioeconomic energy input for food, feed and fibre but also for mechanical work, light and heat. Other energy sources played only a minor role. Hydropower, for instance, used by water mills for processes such as grain milling or metal works, contributed less than 1 % to the total socioeconomic energy input. Similarly, coal was used at almost negligible levels at that time.

After 1850, the use of coal increased tremendously. The new energy source rapidly became the dominant fuel for Austria's industrialization until World War I. After World War I, the newly formed Republic of Austria had lost access to the coal reserves in the former empire. This required a restructuring of the Austrian industry, with less emphasis on heavy industry after the war; in consequence, levels of coal use decreased. After World War II, Austria's rapid economic growth was mostly powered by fossil fuel products, later by natural gas and by a considerable hydropower programme that led to the utilisation of about three-quarters of the economically usable potential, continuing into the present day (for details see Krausmann and Haberl 2002, 2007).

[2] Note that before 1918 the current territory of Austria was part of the much larger Austro-Hungarian monarchy. For data reasons, we had to use data that refer to a territory that is similar, but not exactly identical to Austria's current territory. We had applied extrapolation techniques to extrapolate these datasets to Austria in its current boundaries, in order to generate a consistent time series (for details, see Krausmann and Haberl 2007).

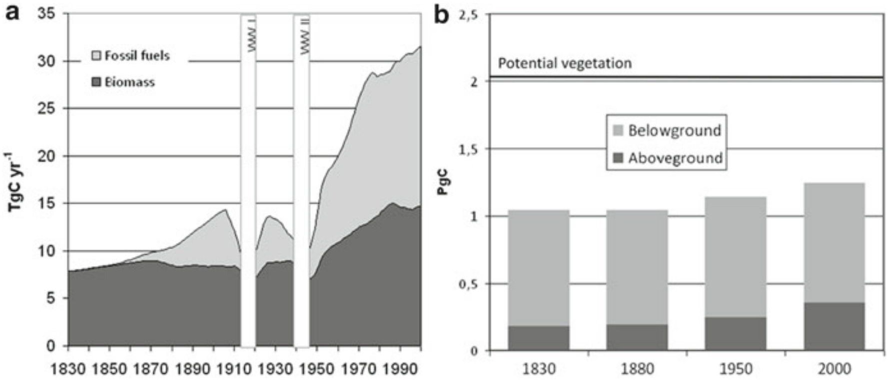

Fig. 13.4 Stocks and flows of C in Austria for the period 1830–2000. (**a**) Socioeconomic C flows per year (5-year moving average). WWI and WWII denotes the first and the second world war. (**b**) C stocks in biota and soils in petagrams of C for the years 1830, 1880, 1950 and 2000 ('above ground' are aboveground parts of plants, 'belowground' includes SOC and belowground parts of plants) (Source: Redrawn after Erb et al. (2008), Gingrich et al. (2007))

All of this resulted in fundamental changes in Austria's socioeconomic C flows. In 1830, almost all of the C metabolised by the Austrian society came from biomass, mainly harvested on Austria's own territory. By contrast, in the year 2000, fossil fuels played a major role, although the C contained in biomass was nowhere near negligible (Fig. 13.4a). Almost all the C metabolised by the Austrian economy enters the atmosphere, mostly as CO_2, but at the same time plant growth also removes CO_2 from the atmosphere through photosynthesis. The historic alterations in socioeconomic energy system and, thus, of socioeconomic C flows had fundamental effects on the C stocks and flows in Austria's ecosystems.

While the amount of C stored in biota plus soils did not change between 1830 and 1880, it started to grow steadily after 1880 (Fig. 13.4b), indicating a considerable net uptake of C. The net C uptake increased from almost zero to approximately 2.6 Tg C year^{-1}for the period 1986–2000 (Fig. 13.5). The reason for this increase was that cropland and grassland areas were shrinking, and that forest area and stocking density, i.e., C stocks per unit area, were increasing (Gingrich et al. 2007; Erb et al. 2008). This phenomenon is observed in many parts of the world and is denoted as 'forest transition' (Mather 1992; Kauppi et al. 2006; Lambin and Meyfroidt 2010). In Austria, forest area grew by more than one-fifth in the last 170 years. In this period, infrastructure areas grew by a factor of four, while cropland area was reduced by one-third and pastures and meadows by one-fifth (Krausmann 2001).

Figure 13.5 illustrates the magnitude of the C flows of the early, pre-industrial period (Fig. 13.5a) and the late, industrialized period (Fig. 13.5b). Next to the tremendous increases of the fossil-fuel related C flows (originating mainly from imports), ecological C flows through ecosystems are also considerably increased in the course of the 150 year period resulting in a significant decrease of HANPP. An empirical analysis suggests that aboveground HANPP was reduced by 15–20 %

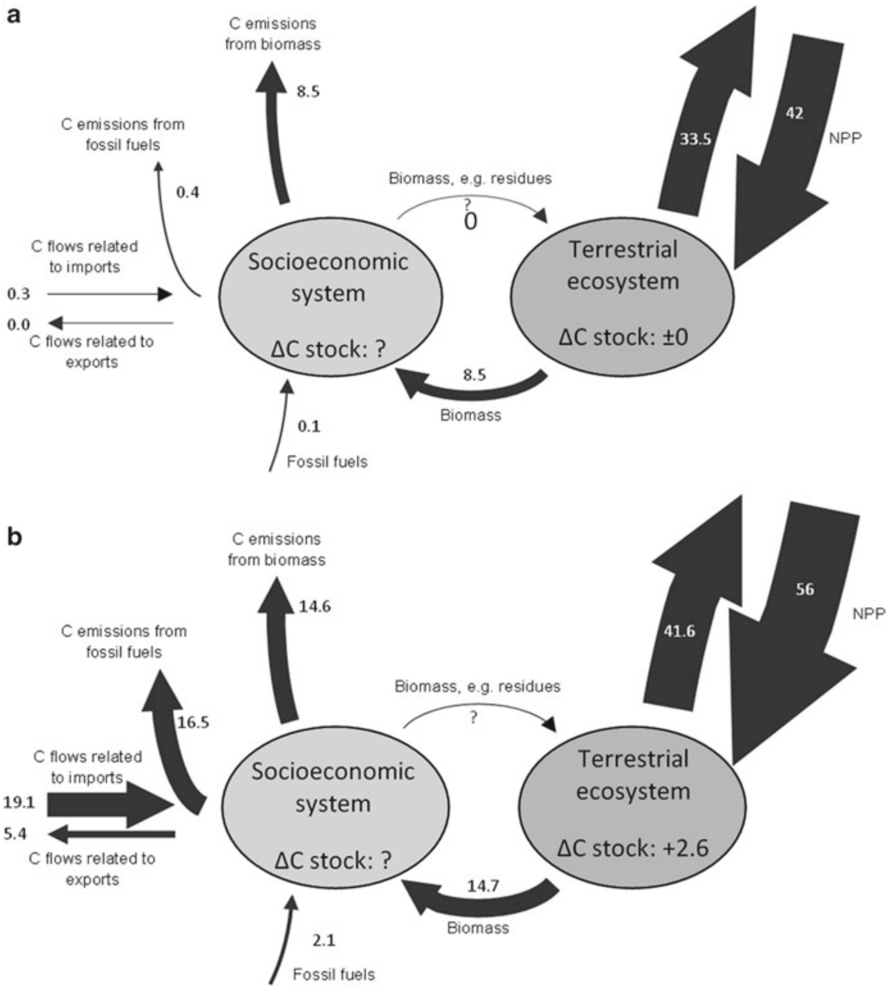

Fig. 13.5 Average yearly flows of C (Tg C year^{-1}) in Austria in the periods (**a**) 1830–1880 and (**b**) 1986–2000. *Arrow sizes* are drawn proportionally to the magnitude of the C flow (except those which are unknown, marked with '?'). Import and export include C in fossil fuel products and biomass products. Biomass transfers from the socioeconomic system to the terrestrial ecosystems, as well as C stock changes in the socioeconomic system are unknown (*NPP* Net primary production) (Redrawn after Erb et al. (2008))

over this period, largely because agricultural intensification and productivity increases (NPP_{act}) were larger than increases in harvest (NPP_h), and the lands freed up by this process were allowed to recover from intensive land use and gained in productivity. Furthermore, the fraction of NPP_h that could be used as commercial product increased as well, due to more efficient biomass utilization chains (Krausmann 2001).

In the later period (Fig. 13.5b) emissions from fossil energy consumption reached a magnitude similar to that of overall socioeconomic biomass consumption. While terrestrial C stock changes were negligible in the early period, in the later period approximately 5 % of total NPP was allocated to stock accumulation (biomass growth). As a result, C stocks in vegetation were substantially (18 %) larger in 2000 than in 1830 (Fig. 13.4b).

Figure 13.4b also shows that the amount of C stored in Austria's biota and soils was substantially lower than it would have been in the absence of human use of the land. The reason is that most of Austria's area would be forested if not used by humans, and only around 10 % of the territory would be covered by non-forest vegetation or be bare (Erb 2004a). From the Middle Ages onwards, a substantial proportion of these natural pristine forests have been replaced by humans either by agro-ecosystems (cropland, grasslands), by infrastructure areas, or by managed forest stands. The development of the terrestrial C sink can thus be interpreted as a recovery from past C losses resulting from land use.

The fact that biota and soils in Austria act as C sinks could be misinterpreted as a justification for assuming that C releases through biomass combustion to the atmosphere were indeed 'C neutral'. In fact, Austrian ecosystems did not only deliver substantial amounts of biomass to the socioeconomic system, they even sequestered C while providing biomass. Such an argument, however, would not be valid.

First, this interpretation of the data neglects the possibility that Austria's biomass consumption causes C releases elsewhere. The analysis of this issue is intricate, however, due to the complexity and data-requirements related to the assessment of upstream flows related to biomass consumption, hence no robust results are available at the moment. Nevertheless, previous research has shown that the magnitude of these C flows is considerable and has the potential to influence the overall greenhouse-gas balance at the national level (Gavrilova et al. 2010; Kastner et al. 2011a).

Figure 13.6 explores the potential magnitude of this effect. This graph is based on the consistent and double-counting free calculation procedure for wood trade presented by Kastner et al. (2011a). Figure 13.6a displays the development of Austria's aboveground biomass C stocks over time, Fig. 13.6b the respective annual C flows. The solid line in Fig. 13.6 shows the development of Austria's aboveground C biomass stocks in aboveground vegetation between 1960 and 2008 (data until 2000 from Gingrich et al. 2007, data after 2000 based on FAO 2010). The other three lines present trajectories based on the hypothetical assumption that no international trade would have taken place and all wood processed in Austria would have been produced domestically.

The bold dashed line in Fig. 13.6 shows the net effect of a hypothetical 'no trade' assumption, implying Austria would have had to rely on its domestic wood stocks only to meet its domestic consumption. In such a case, forest C would have increased to slightly higher levels than in the actual development. This reflects the fact that Austria has been a net exporter of wood products since 1961 and therefore the overall effect of 'no trade' assumption would have been a lowering of domestic wood harvest pressure. The grey lines in Fig. 13.6 show how the growth in aboveground biomass C stock would have developed in the hypothetical case without any imports (all biomass consumed and exported would have been produced from domestic forests; darker grey

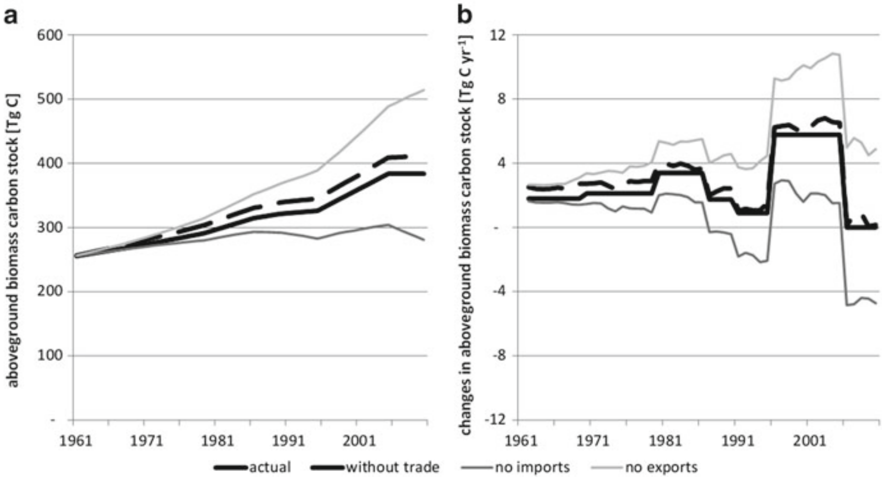

Fig. 13.6 Actual (*solid line*) and hypothetical developments of Austria's (**a**) aboveground biomass C stocks (wood), and (**b**) net C exchange of these stocks for the period 1961–2010; details see text

line) or a second hypothetical 'no export' case (Austria would only produce for the domestic consumption; lighter grey line). The area between those two lines illustrates the magnitude of gross trade flows compared to forest change. Under a "no exports" assumption, Austria's forest stocks would have grown considerably faster, and total aboveground biomass C would be about 20 % higher in the year 2000 than is actually the case today. A 'no import' assumption would not allow Austria's forests to act as C sink due to the high harvest levels. This would have resulted in a stagnation of the C stock in forests after 1980.[3] Figure 13.6 also illustrates the continuous growth in gross trade volumes, whereas for Austria the trade balance is close to zero which is a rare case at the global level (see Kastner et al. 2011a).

The drastically altered picture in the no-import assumption, however, highlights an important issue related to trade. It refers to the economic system of Austria that has been detached from domestic resources for a long time and is adding value by importing cheap raw materials, processing them to goods of higher prices and exporting these refined goods (Erb 2004b). In consequence, one can conclude that Austria is economically relying on non-territorial forest resources, thereby impacting forests in other countries (including the related effects on C sources/sinks in forests outside its own territory). The magnitude of this effect, however, is currently unknown due to lacking data availability.

[3]It is important to emphasize at this stage that a crucial assumption of Fig. 13.6 is that a reduced wood harvest would directly result in increased forest stocks. This assumption is not generally valid and can be considered a reasonable approximation only for forests that are in a stage far from the natural climax-equilibrium. Forests, by maturing and approaching climax, will reach a level of maximum standing stock and all carbon flows would be balanced (at least over larger areas) because biomass production would be equal to heterotrophic respiration in herbivore and detritivore food chains.

A second important caveat related to an oversimplified interpretation of the increasing C stocks in Austria's biota and soils is the following. The reduction in agricultural areas that allowed forests to gain both area and C stocks per unit area was made possible by massive changes in agricultural technology that allowed to drastically increase crop yields and improve conversion efficiencies in the livestock sector (e.g., feed to meat ratios). The empirical data presented above suggest that these technological improvements relied on massive large-scale inputs of fossil fuels in agriculture, both directly (e.g., mechanization) and indirectly (e.g., the supply of industrial N fertiliser). These changes have resulted in a massive reduction of the energy return of investment (EROI) of agriculture from around 6 : 1 in 1830 to approximately 1 : 1 in the year 2000 (Krausmann 2004). In other words, the very same input of fossil fuels that resulted in the massive increases in total GHG emissions also helped to turn Austria's biota and soils into a C sink. It is thus fully justified to speak of a 'fossil-fuel powered C sink' (Erb et al. 2007b). In consequence, terrestrial C sinks cannot serve as silver bullets for solving the climate change problem but can, at best, play only a minor, transitory role, because the emergence of the C sink is intrinsically linked to the fossil-fuel based mode of subsistence.

13.5 Relations to Other Ecosystem Services: The Example of Biodiversity

Beyond the above-discussed interrelations with ecological stocks and flows of C, HANPP is also related with many other ecosystem functions and services. Exemplary, here we discuss the current state of knowledge on the relationships between HANPP and biodiversity.

The 'species-energy' hypothesis suggests that energy flow in ecosystems is positively related with species richness (Brown 1981, 1995; Gaston 2000; Wright 1983, 1990). The species-energy hypothesis has been proposed by macroecologists to explain patterns in species diversity, in particular the gradient of species richness from the equator to the poles. Here species richness is highest in low latitudes and decreases towards both poles in parallel with shrinking availability of energy (Gaston 2000; Wright 1983). The mechanisms behind the relation of energy and species richness could be the higher availability of resources which support higher population densities (larger populations are less prone to become extinct) and the subdivision of resources between more specialized species which allows co-existence of a larger number of species. Monopolization of resources by few dominant species is thought to be limited by the 'costs of commonness', e.g. attractiveness of very abundant species for predators, parasites or pathogens (Brown 1981). Empirical analyses have supported this hypothesis, although they suggested different mathematical models to explain the interrelation between energy flow and species richness (Waide et al. 1999).

This line of thought can be carried forward when analysing human-dominated landscapes. Once the flow of trophic energy is reduced through HANPP, it can be assumed that this reduction of available energy will also reduce species richness

(Haberl et al. 2007b). Species richness – the number of species present in an ecosystem – is an important, and relatively easily measurable component of the larger concept of biodiversity (Magurran 1988).

Direct empirical tests to verify this hypothesis are, however, difficult to pursue because HANPP should reduce species richness compared to its initial value according to the species-energy hypothesis. But while current levels and patterns of species richness are accessible for empirical studies, species richness of the past is generally not well documented, therefore, we indeed know only little about changes in the spatial patterns of species richness. Thus, direct tests of the hypothesized relation between HANPP and loss of biodiversity are not possible at present (Haberl et al. 2007b).

Indirect tests of the interrelation between HANPP and species richness proceeded by calculating the correlations between the spatial patterns of energy remaining in the ecosystem (NPP_t) and spatial patterns of species richness in GIS grids (Haberl et al. 2005, 2009) and correlations between NPP_t and measured species richness in 38 experimental plots (Haberl et al. 2004b). These studies confirmed strong correlations between energy availability and species richness; such interrelations were observed in natural systems as well as in landscapes heavily modified by humans. Nevertheless, more research is needed because different levels of HANPP are also associated with changes in landscape patterns. Those changes in landscape may in return affect biodiversity and thus potentially confounding the effects of energy availability (Wrbka et al. 2004). Overall, the studies discussed above demonstrate the high relevance of HANPP for patterns and processes in ecosystems.

13.6 Conclusions

An extension of the HANPP framework, based on an integration of stocks and flows of C of a socio-ecological system, can aid analysing systemic interlinkages between the socioeconomic energy system, the land use system and patterns and processes in ecosystems. These insights are useful as they improve our basic understanding of the complex and intricate ways society interacts with natural systems, including the feedback loops from the (altered) natural systems to society.

The historical long-term example of Austria's land use and the stocks and flows of C related to biota and soils on its territory reveals that a systemic approach is indispensable for the development and evaluations of strategies that aim to foster sustainable development. Similar trajectories of forest cover and C stocks as found in Austria, are described for many countries (e.g., Kauppi et al. 2006; Kuemmerle et al. 2011), which suggests that such complex interrelations and feedback loops between land intensification, forest growth, and the overall socioeconomic energy system are frequent rather than being an exception. Our understanding of the spatial and temporal interrelation of these feedback loops, however, is still limited, as many parameters and causal chains show time lags and are subject to trajectories that operate at different spatial scales, e.g., meta-trends such as the globalisation of production and consumption.

An extension of the HANPP framework by establishing consistent accounts for ecological C stocks can help to advance our understanding of these complex relationships. This is because it provides accounts that are resistant to problem-shifts over time and space and allows to identify feedback loops of higher magnitudes.

Such comprehensive, integrated approaches also question common sense, such as the concept of biomass being a C neutral energy carrier. The assumption that bioenergy use would be C neutral has long been recognized as being imprecise, because amongst others it ignores the fact that fossil energy is utilized for the production, harvest and processing of biomass (Schlamadinger et al. 1997). In fact, a 'C-neutrality' assumption may even result in major flaws, especially in cases where large C stock changes are caused, such as through the conversion of pristine forests to used forests or to agricultural fields. The use of biomass, for instance, can significantly reduce the strength of ecosystems to serve as C-sinks. This adverse effect may very well wipe out any emission reduction from bioenergy if additional wood needs to be harvested for processing instead of using fossil fuels (Haberl et al. 2003; Searchinger et al. 2009). Moreover, theoretical considerations supported by empirical evidence also suggest that HANPP is related to biodiversity. Increased land-use intensity resulting from large-scale implementation of bioenergy programs could therefore contribute to species loss and other detrimental effects on ecosystem functions and services.

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Chapter 14
Soil Carbon and Biofuels: Multifunctionality of Ecosystem Services

Iris Lewandowski

Contents

Abstract Biofuels can be solid, liquid or gaseous and can be made from various biomass resources (or feedstocks). Biofuels produced today for the transportation sector are mainly made from oil crops (biodiesel) or sugar and starch crops (ethanol), of which many are annual crops. Future generation biofuels may use lignocellulosic biomass from trees and grasses, all perennial plants, as raw material.

The impact of biofuels on soil carbon (C) (i.e., soil organic carbon, SOC) depends on the characteristics of the crop, the management system and the previous land use. Soils under cultivation of annual crops such as sugar beet (*Beta vulgaris*), wheat (*Triticum aestivum*) and rapeseed (*Brassica napus*) experience a loss of soil C unless a system of reduced cultivation intensity is in place. Residues from annual crops, e.g., straw, can help to maintain soil C stocks if left in the field after harvest but are also deemed a low-cost resource for biofuels that does not require land for production. For corn (*Zea mays*), not more than 25 % of the residue biomass (corncobs) should be removed for energetic use if the soil C level is to be maintained. Under perennial crops soil carbon may accumulate at about 1 Mg C ha^{-1} annually.

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Land-use change for bioenergy production has potentially the strongest impact of all management measures on soil C. In the most severe case of converting rainforest to oil palm (*Elaeis guineensis*) plantations for biodiesel production, it is estimated that about 25–170 Mg soil C are lost when the rainforest grows on mineral soil and more that 700 Mg C when the rainforest grows on peatland, the latter resulting in a carbon dioxide (CO_2) emission payback time of more than 400 years. On the other hand, more than 10 Mg C ha^{-1} can be accumulated if imperata (*Imperata cylindrica*) grassland is converted to oil palm plantations.

It is concluded that depending on the type of cropping system under which biomass for biofuels is produced the effect on soil C can be positive (i.e., an increase or maintenance of soil C stocks) or negative (i.e., a decrease in soil C stocks). Positive effects can be expected where biomass is produced in perennial systems or where annual crops are grown in low-tillage intensity systems and not more than 25 % of the residue mass is removed. Strong negative effects can be expected where land-use changes result in a reduction of soil C.

Keywords Biofuels • Second-generation biofuels • Ethanol • Biodiesel • Oil crops • Sugar crops • Starch crops • Lignocellulosic crops • Perennial crops • Crop management • Agricultural management • Land-use change • Tillage intensity • Residues

Abbreviations

BtL	biomass-to-liquid
C	carbon
°C	degrees Celsius
CO_2	carbon dioxide
DME	dimethyl ether
FT	Fischer Tropsch
GHG	greenhouse gases
Ha	hectare
Mg	mega gram (equivalent to metric ton)
SOC	soil organic carbon
SRC	short rotation coppice
R&D	Research and Development
w/w	weight by weight
y	year

14.1 Present and Future Biofuels

The term ‘biofuels’ is often used as a synonym for all biogenic fuels produced for the transportation sector. A more general definition would be “all liquid, gaseous and solid fuels derived from biomass”. Biofuels can be produced from various biomass sources or feedstocks (Fig. 14.1).

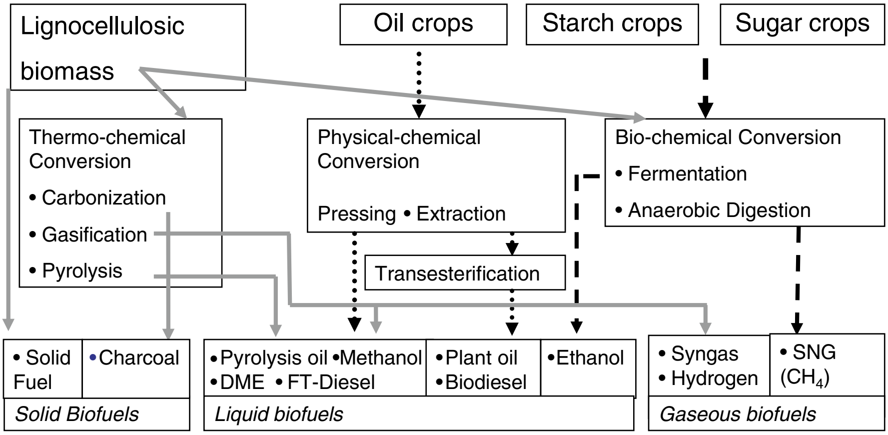


Fig. 14.1 Overview of conversion routes from biomass to biofuels. *DME* dimethylether, *FT* Fischer Tropsch, *SNG* substitute natural gas

Fig. 14.2 Sugar cane[a] (*Saccharum officinarum*) stalk ([a]all plant illustrations by Uli Schmidt http://www.uli-schmidt-paintings.com/)

Today's liquid biofuels are mainly produced through either the fermentation of sugar or starch-rich biomass or the processing of oil-rich biomass. Sugar cane (*Saccharum officinarum*; sugar crop, Fig. 14.2) and corn (*Zea mays*; starch crop, Fig. 14.3) are the most important energy crops for ethanol production and for the production of liquid biofuels in general (Table 14.1). Ethanol is also produced from other sugar crops such as sugar beet (*Beta vulgaris*; Fig. 14.4) and sweet sorghum (*Sorghum bicolor*; Fig. 14.5) and from other starch crops, e.g., wheat (*Triticum aestivum*; Fig. 14.6) and cassava (*Manihot esculenta*; Fig. 14.7) (Table 14.1).

Fig. 14.3 Corn (*Zea mays*) plant with corncob

Table 14.1 Global annual production of liquid biofuels as an average of the years 2008–2010 (OECD 2011)

	Billion litres
Ethanol production from:	
Coarse grains (mainly corn)	47
Sugar cane	27
Wheat	2
Molasses	4
Non-agricultural feedstock	3
Sugar beet	1.5
Others	8
Biodiesel production from:	
Vegetable oil (mainly rapeseed, others: soybean and palm oil)	17
Non-agricultural feedstock	2

In the process of ethanol production, sugar is fermented using adapted yeasts. The fermentation process with glucose as starting material can be described by (Eq. 1):

$$C_6H_{12}O_6 \rightarrow 2C_2H_5OH + 2CO_2 \tag{1}$$

The oil palm (*Elaeis guineensis*; Fig. 14.8) is the most important oil crop globally. However, only 4.7 % of its oil is currently used for the production of biofuels or bioenergy. Most is used for food (71.1 %) or industrial purposes (24.2 %) such as the production of detergents and cosmetics (FNR 2011).

Rapeseed (*Brassica napus*; Fig. 14.9) oil is the major feedstock used for biodiesel production in Europe.

Fig. 14.4 Sugar beet (*Beta vulgaris*)

Fig. 14.5 Sweet sorghum (*Sorghum bicolor*)

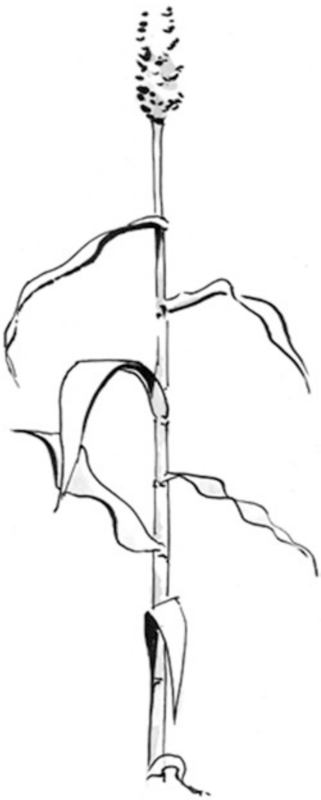


Fig. 14.6 Wheat grain (*Triticum aestivum*)

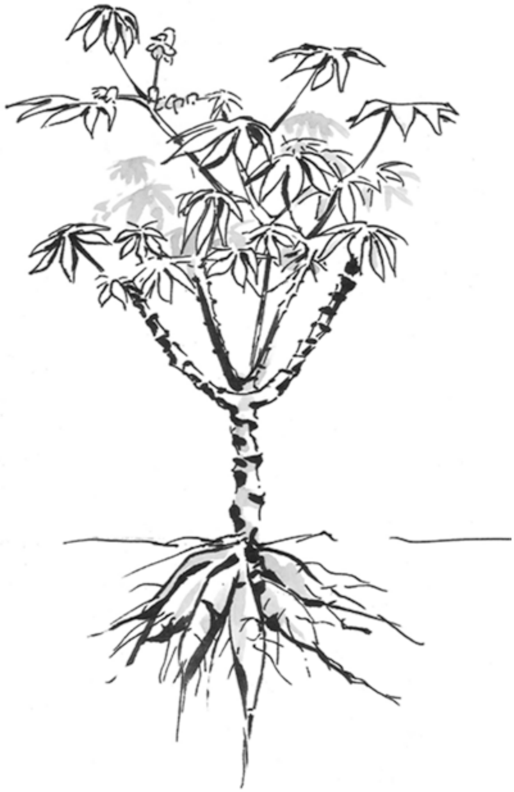


Fig. 14.7 Cassava (*Manihot esculenta*)

Fig. 14.8 Oil palm (*Elaeis guineensis*) with fruit

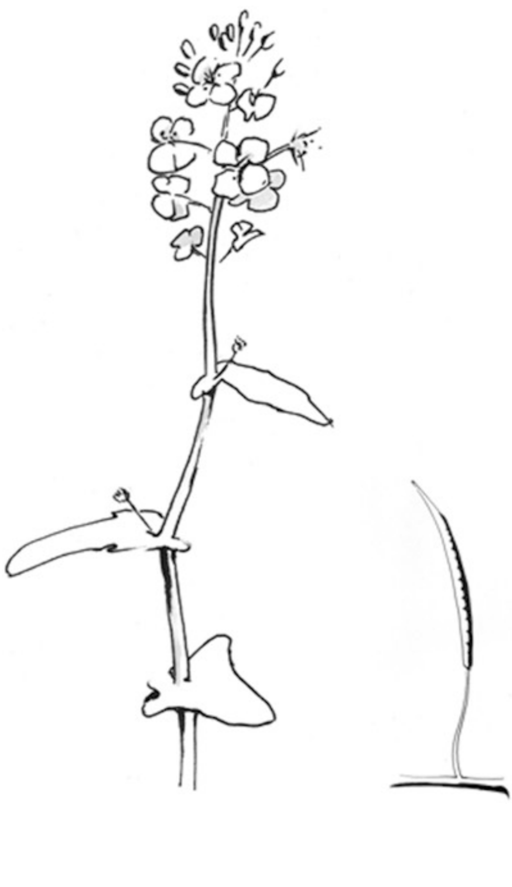


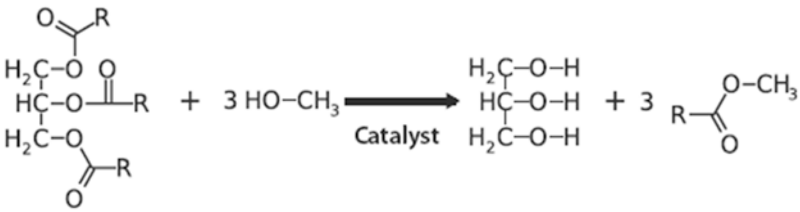
Fig. 14.9 Rapeseed (*Brassica* napus) with pod

Fig. 14.10 Sunflower (*Helianthus annuus*)

The main purpose of producing soybean (*Glycine max*) is as protein-rich animal feed and vegetable oil is only the minor share of the product. The beans contain 40 % (w/w) protein and 20 % oil.

Sunflower (*Helianthus annuus*; Fig. 14.10) is the fourth largest oil crop globally in terms of vegetable oil production volume, after oil palm, soybean and rapeseed. However, the quality of the oil is less suitable for biodiesel production and sunflower oil production costs are high compared to other vegetable oils. Therefore sunflowers do not play an important role in biodiesel production.

Only a minor share of biofuel derived from vegetable oil is used in an unmodified form. Most vegetable oil is transesterified before being used as engine fuel. For this process, the oil is pressed from the seeds or extracted chemically (using hexachloride). In a second step, the oil is transesterified in a reaction with alcohol in the presence of a catalyst to produce glycerol and fatty acid methyl ester (see Eq. 2).



(2)

Due to its high content of monounsaturated oleic acid (C18:1) and low levels of both saturated and polyunsaturated acids (Table 14.2) rapeseed oil is the ideal

Table 14.2 Fatty acid composition in percent (w/w) and carbon chain length of major vegetable oils

	Fatty Acids										Suitability for biodiesel production
Feedstock	8:0	10:0	12:0	14:0	16:0	16:1	18:0	18:1	18:2	18:3	
Rapeseed					3–5		1–2	55–65	20–26	8–10	+++
Oil palm	0.5			1–2	40–48		4–5	37–46	9–11	0.3	+
Soybean					11–12		3–5	23–25	52–56	6–8	+
Sunflower					6		3–5	17–22	67–74		+
Jatropha		0.1		0.1	14–15	1	7	34–45	31–43	0.2	++

The first figure indicates the number of carbon atoms in the chain, the figure after the colon indicates the number of double bonds in the chain (data from Mittelbach and Remschmidt 2010). Empty cells indicate no occurrence of this fatty acid or contents are below 0.1 %. +, suitable; ++, very suitable; +++, most suitable

feedstock material with respect to combustion characteristics, oxidative stability and cold temperature behaviour (Mittelbach and Remschmidt 2010).

Sunflower oil has a high content of linoleic acids (C18:2), which limits its use for biodiesel production. Pure sunflower oil methyl ester shows low oxidative stability (Mittelbach and Remschmidt 2010). Similar to sunflower oil, the fatty acid composition of soybean oil is not ideal for biodiesel production because of its high share of polyunsaturated fatty acids. However, it is an important source for biodiesel production in the US (Mittelbach and Remschmidt 2010).

In climates where freezing occurs, unsaturated vegetable oils are needed for biodiesel production because saturated oils can flocculate or solidify at low temperatures leading to the clogging of filters and pipes in engines. Therefore, rapeseed oil provides the best feedstock for biodiesel in temperate climates. In subtropical and tropical climates, oil from oil palm or jatropha (*Jatropha curcas*) is also suitable for biodiesel production. Palm oil is an important resource for biodiesel production in Asia.

Although jatropha is currently being developed as an energy crop and an estimated area of 900 000 ha is presently under cultivation (Renner et al. 2008), until now, there is no significant supply of jatropha oil for biodiesel production available. Grown on marginal land, the perennial crop can lead to an increase of soil C. But jatropha can still be considered a wild crop because high-yielding varieties do not exist and the crop management system has not yet been fully developed. This is reflected in various assumptions which have been made about its yield potential. Therefore, more research and development is needed before jatropha can be considered as a crop for large-scale biodiesel production. Due to the presence of curcine and various toxic phorbol esters in its leaves and fruits and also its ability to survive under marginal (especially drought) conditions, jatropha is considered an ideal energy crop that does not compete or competes very little with food production. However, this is only true if jatropha is not grown on prime agricultural land.

Gaseous biofuels are produced through the process of anaerobic fermentation yielding so-called “biogas” (Fig. 14.1). A diverse range of biomass feedstocks and

Table 14.3 Biogas and methane (CH_4) yield from various organic substances and biomass feedstocks

Feedstock	Gas yield [L kg^{-1}]	CH_4 [% of yield]
Corn	350 (250–450)	54
Grass (fodder grass)	300 (200–450)	56
Sweet sorghum	250 (200–340)	51

Data from LfL (2007)

Fig. 14.11 Miscanthus (*Miscanthus* × *giganteus*) stalk

organic compounds can be used for biogas production including sugars, starch, cellulose, proteins and oils (Table 14.3).

Various energy crops can serve as biogas substrates with corn being the dominant biogas crop in Europe (Fig. 14.3). However, organic wastes, e.g., from food industries, or slurry may also be used. Biogas is mainly combusted in Combined Heat and Power Plants (CHPs) for power and heat production. If cleaned and compressed, biogas can also be used to fuel vehicles.

Lignocellulosic biomass is produced by trees such as willow (*Salix viminalis*; Fig. 14.12) and grasses such as miscanthus (*Miscanthus* ssp.; Fig. 14.11). It is also present in agricultural residues, e.g., straw. The major use of lignocellulosic biomass is its direct combustion as solid fuel for heat and power production (Fig. 14.1).

At present, the majority of liquid biofuels are produced from sugars, starch or oils (Table 14.1). However, in the future so-called 'second' or 'next' generation biofuels

Fig. 14.12 Willow (*Salix viminalis*) tree

are expected to be released. These biofuels are produced from biomass such as lignocellulosic biomass which is non-edible for humans. Some also consider jatropha oil as a next generation biofuel because it is non-edible. The question arises as to whether second or next generation biofuels are defined by the (non-edible) feedstock used or by the conversion technology applied.

The two major technology pathways for converting lignocellulosic biomass into liquid biofuels are 1) the production of diesel hydrocarbons via gasification and Fischer-Tropsch Synthesis and 2) the production of the gasoline additive ethanol via hydrolysis and fermentation of the carbohydrates contained in the lignocellulose (Lange et al. 2010). Other second generation biofuels such as dimethylether (DME) and methanol can also be produced from lignocellulosic biomass (see Fig. 14.1).

As shown in Fig. 14.13, only biodiesel, ethanol and biogas-derived biofuel technologies are fully commercialised but none of the next generation biofuels. The biofuels produced using lignocellulosic biomass or biomass from algae are, until now, only being demonstrated in the laboratory and their full commercialisation will take at least another decade.

The use of lignocellulosic biomass for direct combustion and the production of heat and power are already commercialised. Present technologies allow the co-combustion of lignocellulosic biomass in coal combustion power plants. In Denmark and the UK, wheat straw and miscanthus biomass are currently burned to run biomass heat and power plants. It can be expected that the use of biomass for combined

	2nd or next generation biofuels			1st generation biofuels
	Basic and applied R&D	Demonstration	Early commercial	Commercial
Bioethanol		Cellulosic Ethanol		Ethanol from sugar and starch crops
Diesel-type biofuels	Biodiesel from microalgae Sugar-based hydrocarbons	BtL-Diesel (from gasification + FT)	Hydrotreated vegetable oil	Biodiesel by transesterification
Biomethane		Bio-synthetic gas		Biogas (anaerobic digestion
Hydrogen	All other novel routes; Gasification with reforming	Biogas reforming		

Liquid biofuel Gaseous biofuel

Fig. 14.13 Commercialisation status of biofuels (according to IEA 2011) *R&D* Research and Development, *BtL* biomass-to-liquid, *FT* Fischer-Tropsch

heat and power production will increase in future because greenhouse gas (GHG) emission reductions of more than 90 % are possible if biomass from short rotation coppice, miscanthus, straw or forest residues are used (JRC 2009).

The major technical challenge in pure biomass combustion lies in the ash-melting behaviour of particularly herbaceous biomass. The melting of ash at comparatively low temperatures (800 °C) leads to slagging and fouling of the combustion chamber or boiler. The presence of slag impedes the heat transfer and has to be removed manually (Lewandowski and Kicherer 1997).

14.2 How Different Scenarios of Biofuel Production Influence Soil Carbon

In the following text 'soil carbon' is used synonymously with 'soil organic carbon' (SOC).

14.2.1 Choice of Crop or Biomass Resource

The energy crops described in the previous section have varying characteristics which are described in Table 14.4.

All crops require specific climatic conditions. Oil palm is a tropical crop and grows best in the same climates as rainforests. Sugar cane grows in subtropical to tropical climates and requires a relatively high annual precipitation level of more than 1000 mm. Jatropha is a subtropical crop that can also survive with annual precipitation levels as low as 300 mm. However, yields are rather low at that level

Table 14.4 Characteristics of important global energy crops

Crop	Growth climate	Annual precipitation requirement [mm]	Life form	Photosynthetic pathway	Crop's lifetime [years]
Oil palm	Tropical	1,500–3,000	Perennial	C_3	25
Rapeseed	Temperate	600–800	Annual	C_3	1
Soybean	Subtropical	500–750	Annual	C_3	1
Sunflower	Temperate to subtropical	450–500	Annual	C_3	1
Jatropha	Subtropical	500	Perennial	C_3	25–40
Sugar cane	Subtropical	1,000–1,250	Perennial	C_4	3–6
Sugar beet	Temperate to subtropical	500	Annual	C_3	1
Sweet sorghum	Subtropical	500	Annual	C_4	1
Corn/maize	Temperate to subtropical	500–700	Annual	C_4	1
Wheat	Temperate	500–600	Annual	C_3	1
Cassava	Subtropical to tropical	750	Annual	C_3	1
SRC[a] willow	Temperate	600	Perennial	C_3	20–25
SRC[a] poplar	Temperate	550	Perennial	C_3	20–25
SRC[a] eucalyptus	Subtropical	>400	Perennial	C_3	20–25
Switchgrass	Temperate to subtropical	500	Perennial	C_4	10–15
Miscanthus	Temperate to subtropical	600	Perennial	C_4	20

Data from Lewandowski et al. (2003, 2009), Rehm and Espig (1984), Rockström (2003)
[a]*SRC* short rotation coppice

and for reasonable yields annual precipitation should reach 600 mm or more. Other subtropical energy crops are soybean, cassava, sweet sorghum and eucalyptus (*Eucalyptus* ssp.). Rapeseed, sunflower, sugar beet, corn, wheat, willow, poplar (*Populus* ssp.), miscanthus and switchgrass (*Panicum virgatum*) are all crops that grow in moderate and some also in subtropical climates. If any crop is grown under unsuitable conditions it will either not establish, e.g., due to too low temperatures, or it will not yield. Therefore, crops must always be chosen according to site conditions (climate and soil type).

The major energy crops also include the so-called C_3 and C_4 crops. These differ with respect to their photosynthetic pathway. In C_3 crops the first product is a molecule with three C atoms whereas for C_4 crops it has four C atoms. Especially in warmer climates, C_4 photosynthesis is more efficient in converting solar radiation into biomass due to its more effective binding mechanisms for CO_2 and lower C losses during dark respiration (Zhu et al. 2010). However, at lower temperatures and low intercellular CO_2 partial pressure (Ehleringer and Pearcy 1983) the overall efficiency of C_3 plants is greater. The photosynthetic use efficiency of nitrogen is often higher in C_4 than in C_3 plants and this is reflected in an inherently lower nitrogen concentration in C_4 than C_3 plants (Greenwood et al. 1990; Lattanzi 2010). Also,

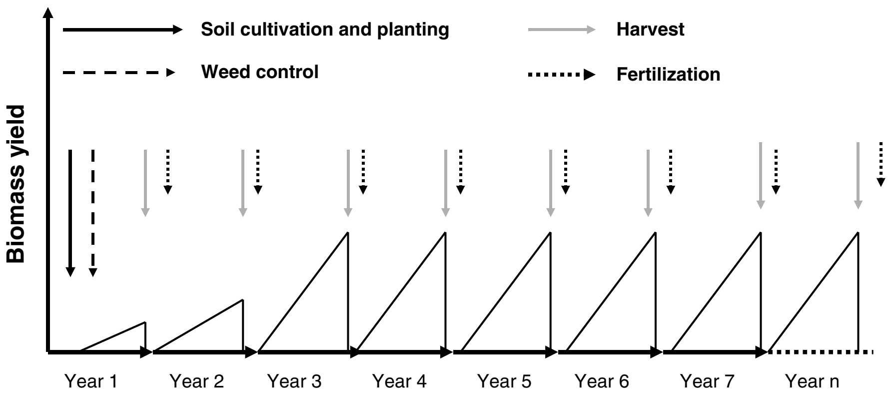


Fig. 14.14 Production cycle and management for perennial grasses

Table 14.5 Soil carbon sequestration potential of land management options in Europe

Practice	Soil carbon sequestration potential (Mg C ha^{-1} y^{-1})	Estimated uncertainty (%)
Zero tillage	0.38 (0.29)*	>50
Set-aside	<0.38	>>50
Permanent crops	0.62	>>50
Cereal straw	0.69 (0.21)*	>>50
Conversion of cropland into grassland	1.2–1.69 (1.92)*	>>50
Conversion of cropland into woodland	0.62	>>50

Data from Rees et al. (2005)
All estimates are based on extrapolation from Smith (2004), except those marked by an asterisk (*), where the figure in brackets is derived from Vleeshouwers and Verhagen (2002)

higher water-use efficiency is reported for C_4 plants. However, under severe stress conditions both these advantages of C_4 crops eventually disappear (Lattanzi 2010; Sage and Pearcy 1987).

With regard to soil C, the life form of the crop – annual or perennial – is the most important factor. Because they are sown or planted every year, annual crops require yearly soil cultivation. Generally soil C levels do not increase, but rather decrease, under annual crops due to intensive soil cultivation (Fig. 14.15).

The SOC level under annual crops depends on the soil texture, the intensity of soil cultivation and the kind and amount of biomass that is produced and remains in the field.

Perennial crops are planted and sown in year one and harvested without replanting for a period of 3 to 25 years. Fig. 14.14 shows the simplified production cycle for a perennial grass (e.g., miscanthus, switchgrass, reed canary grass (*Phalaris arundinacea*)).

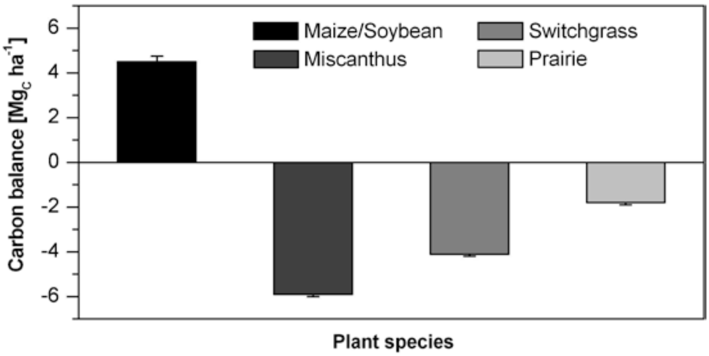


Fig. 14.15 Net carbon balance for the production of various first and second generation biofuel crops (data from Zeri et al. 2011)

There is an increase in soil C levels under perennial crops because of the long period of reduced soil disturbance (Table 14.5). Figure 14.15 shows the C balance for four different energy cropping systems. Zeri et al. (2011) measured the C balance of the soil during the establishment of an annual crop rotation, perennial grasses and prairie sown on arable land previously used for oat (*Avena sativa*) production. Over a period of four years approximately 4.2 Mg C were lost from the soil when the maize-maize-soybean rotation was grown. Under miscanthus, switchgrass or prairie around 6, 4 or 2 Mg C were added to the soil C pool, respectively (Fig. 14.15).

The accumulation rate for SOC measured over 12 years showed that it can reach 2.75 Mg C y^{-1} for miscanthus (during the initial period after planting), and be higher than that for permanent grassland or afforestation (Müller-Sämann and Hölscher 2010). In a review by Anderson-Teixeira et al. (2009), an average SOC accumulation rate of 1 Mg ha^{-1} y^{-1} in the top 30 cm was calculated for miscanthus. The SOC increase calculated for sugar cane was 0.29 Mg C ha^{-1} y^{-1} and for switchgrass 0.4 Mg C ha^{-1} y^{-1} (Anderson-Teixeira et al. 2009). However, if the plantations were established through the conversion of native ecosystems such as forest or grassland, an initial SOC loss occurred, which was largest under sugar cane, but with the establishment of sugar cane the SOC levels rose again (Anderson-Teixeira et al. 2009).

The SOC accumulation rate depends on the soil texture and the previous soil C levels (Poeplau et al. 2011). In a study on sugar cane, Cerri et al. (2011) measured an annual rate of soil C stock change of around 1, 1.5 and 2.2 Mg C at soil clay contents of about 10, 30 and 65 %, respectively. The major increase of soil C occurs in the first 5–10 years after the conversion to permanent cropping systems or grassland (Fig. 14.16). After a period of 60–100 years the soil C content may reach a saturation level (Jenkinson 1988; Umweltbundesamt 2010). However, whether soil C saturation occurs in agroecosystems is still under discussion (e.g. in Blair et al. 2006a, b). Poeplau et al. (2011) concluded that for temperate zones the establishment of grassland on formerly cultivated soil did not reach a new equilibrium within 120 years after conversion.

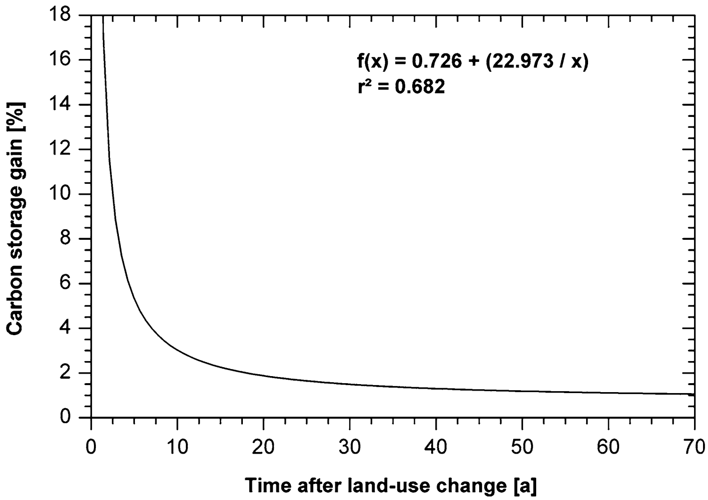


Fig. 14.16 Simplified relationship between yearly soil carbon accumulation rate and length of period after land-use change from annual crops to grassland, perennial crops or forest measured for German sites (according to Umweltbundesamt 2010)

14.2.2 Crop Management

14.2.2.1 Soil Cultivation Intensity

With regard to crop management, the intensity and method of soil cultivation have a particularly relevant influence on soil C dynamics under energy crops. The long periods of soil rest possible under perennial energy crops can be partly reached in annual production systems by reducing soil cultivation intensity (Table 14.5).

In North and South America there are increasing proportions of low-till or no-till systems. The adoption of conversion tillage has slowly gained momentum in Europe. Reduced tillage is practiced on only 20–25 % of arable land in Germany, France and Portugal (SoCo 2009). The 'no-till' rate is 2.5–4.5 % of arable land in UK and Spain (Lezovic 2011). No-tillage can control soil erosion and improve water supply which can result in higher yields under water-limited conditions (Triplett and Dick 2008). However, if no-till leads to reduced crop densities or high weed pressure, yields may also be lower compared to tilled systems (Kravchenko and Thelen 2007; Boomsma et al. 2010). In no- or low-till systems the soil is not ploughed but either loosened superficially or not cultivated at all. In both cases, the main problem for crop management is high competition with weeds (ploughing buries weeds and their seeds), which have to be controlled by herbicides.

14.2.2.2 Use of Agricultural Residues as Biomass Resource

The amount and type of biomass produced and left in the field as harvest residues, roots or stubble has a strong influence on the C balance of the soils in energy cropping systems. For every Mg of cereal grain about one Mg of straw is produced. These agricultural residues can be used for energetic purposes. In Denmark, about 1.5 million Mg wheat straw are combusted annually for heat and power production. Residues with a high water content, such as the leaves of sugar beet, can be used in biogas plants. In the future, lignocellulosic-rich residues should be able to be used for the production of second generation biofuels. Residues generated as "by-products" in food or fodder production can become a sustainable resource for bioenergy production. However, their removal from the field can have a negative effect on the C balance of the soil. Residues are the major SOC source in annual energy cropping systems and lignin-rich residues in particular result in C accumulation in the soil (Paustian et al. 1992). Therefore, it is recommended that not all residues are removed from the field.

The impact of residue removal and the sustainable removal rate depend on site-specific factors such as climate, soil type and topography, but also on management practices and crop yield (i.e., residue yield) (Andrews 2006; Blanco-Canqui and Lal 2007). Don et al. (2011) pointed out that for the SOC balance of a land-use system, the harvested fraction of the produced biomass seems to be more important than tillage, climate or soil characteristics. In a review by Johnson et al. (2006) on the possibilities of cereal straw removal it was concluded that in moldboard-plow systems an annual above-ground source of 2.5 ± 1.0 Mg C ha^{-1}, and in no-tillage and chisel-plow tillage systems 1.8 ± 0.4 Mg C ha^{-1} was needed to maintain soil C levels, respectively. This also indicates that the amount of removable straw depends on the straw yield (i.e., overall crop yield). Blanco-Canqui and Lal (2007) assessed the effect of the removal of corn stover removal in long-term no-tillage corn systems. The removal of more than 25 % of corn stover led to a reduction of SOC and soil productivity. The increase in SOC loss is approximately 0.2 % for every 1 % increase in residue removal (Anderson-Teixeira et al. 2009). However, no general rules for sustainable removal rates can be given because these have to be determined for each location (Andrews 2006).

14.2.2.3 Residues from Bioenergy Production

Organic residues are generated in the process of biofuel production, for example fermentation residues from biogas production. These fermentation residues are rich in C and mineral nutrients (Table 14.6). However, the effect of biogas residue application on SOC has been evaluated differently. Terhoeven-Urselmans et al. (2009) assumed that biogas residues do not have a positive effect on SOC due to the decomposition of the biomass during the fermentation process. In a field study by Petz (2000), an increase of 27 % in SOC was recorded with biogas residues compared to a non-fertilized control.

Table 14.6 Composition [% (w/w)] of oil biogas fermentation and oil mill residues

	N	K	P	Reference
Fermentation residues	7.4	4.0	0.8	LTZ (2008)
Rapeseed cake	4.8–5.8	1.2	1.0–1.2	Widmann et al. (2009)
Empty oil palm fruit bunches	0.8	0.24	0.06	Elbersen et al. (2005)
Jatropha press cake	3.2–4.4	1.2–1.7	1.4–2.1	Kumar and Sharma (2008)

Residues incurring in the process of biodiesel production are mainly originating from the pressing of the seeds (rapeseed) or fresh fruit bunches (oil palm). Rapeseed cake is rich in protein and thus mainly used as animal feed. Its composition allows using it as an organic fertilizer (Table 14.6). Palm mill residues and jatropha press cake are also rich in plant nutrients and are returned to the plantation soil as organic fertilizers.

The production of ethanol from sugar cane delivers cellulose-rich bagasse as residue. Bagasse contains 40–60 % cellulose and 20 % lignin and is therefore suitable as a solid fuel. In practice, it is used for fuelling the sugar mills and ethanol plants. These would be suitable as C-rich soil amendments. However, their most economic use is being solid fuel for heat and power production.

14.2.3 Land-Use Change

Land use has a strong influence on the C level in soils (Table 14.5). Under perennial vegetation, the C level is much higher than under cropping systems harbouring annual crops. An example of land-use change in Germany regarding the production of biofuels is the conversion of grassland for the production of corn as a substrate for the generation of biogas. In this process, about 63 Mg C are released from the soil, equivalent to about 230 Mg CO_2 emissions (Fritsche and Wiegmann 2008). Since biogas from corn can help reduce GHG emissions, the C lost from the grassland can be compensated. However, as a C debt (amount of carbon that is released from the ecosystem due to land-use change over a period of 20 years and which is calculated as part of CO_2 emissions when assessing the GHG balances of biofuels) of more than 60 Mg C ha^{-1} is generated, the compensation time can easily reach more than 20 years. Table 14.7 gives examples of 'carbon debts' generated by various types of land-use change effected for the subsequent production of biofuels.

It is obvious that the conversion of rainforest to palm oil production causes the highest C debt, especially when they are growing on peat soils. About 148 ± 76 Mg C ha^{-1} are stored above-ground by a rainforest compared to 30 ±10 Mg C ha^{-1} by an oil palm plantation. Below ground, tropical forests store 24 ±13 Mg C ha^{-1} compared to 4–8 Mg C ha^{-1} in oil palm plantations (Germer and Sauerborn 2008). Thus, there is a loss of about 60–210 Mg C ha^{-1} when rainforest is converted into oil palm

Table 14.7 Carbon debt, biofuel debt allocation, annual carbon repayment rate and years to repay carbon biofuel debt for nine scenarios of biofuel production

		Carbon debt [Mg C and Mg CO_2 ha^{-1}]						
		Soil carbon loss		Above-ground biomass carbon loss				
Biofuel produced	Previous ecosystem	C	CO_2	C	CO_2	Debt allocated to biofuel [%]	Annual repayment [Mg CO_2 ha^{-1}]	Time to repay biofuel carbon debt [y]
Palm oil biodiesel	Tropical rainforest (Indonesia/Malaysia)	47	171	145	531	87	7.1	86
Palm oil biodiesel	Peatland rainforest (Indonesia/Malaysia)	796	2,921	145	531	87	7.1	423
Soybean biodiesel	Tropical rainforest (Brazil)	56	205	146	531	39	0.9	319
Soybean biodiesel	Cerrado grassland (Brazil)	19	69	4	16	39	0.9	37
Sugarcane ethanol	Cerrado wooded (Brazil)	29	106	16	59	100	9.8	17
Corn ethanol	Central grassland (US)	35	128	2	6	83	1.2	93
Corn ethanol	Abandoned cropland (US)	0	0	2	6	83	1.2	48
Prairie biomass ethanol	Abandoned cropland (US)	1	2	1	4	100	4.3	1
Prairie biomass ethanol	Marginal cropland (US)	0		0	0	100	7.8	No debt incurred

Data from Fargione et al. (2008)

plantations. Fargione et al. (2008) estimated that the loss of below-ground C reaches more than 700 Mg C if oil palm plantations are grown on peatland. According to their calculations, the payback time for C losses from peatland rainforest is 423 years if oil palm plantations for biodiesel production are grown on deforested land. There are also long C payback times for other biofuels if land-use changes occur. For ethanol produced from corn grown on former grassland in the US, the calculated payback time is 93 years (Fargione et al. 2008, Table 14.7). The conversion of primary forest to cropland (−25 %) causes twice as high SOC losses as its conversion to grassland (−12 %) (Don et al. 2011).

However, land-use change does not always have negative effects on soil C. Conversion of arable land to perennial cropping systems may result in an increase in soil C. There are also examples where marginal lands converted to the production of perennial energy crops can increase soil C levels. For example, an increase in soil C levels is recorded when imperata grasslands growing on land deforested decades ago is converted into oil palm plantations. In a greenhouse gas balance of this land-use change scenario, a reduction of 136 ± 37 Mg CO_2 equivalents (sum of the greenhouse gases CO_2, CH_4 and N_2O weighted according to their global climate-change impact) can be achieved (Germer and Sauerborn 2008). Don et al. (2011) assessed that cropland (re)conversion to grassland increased SOC stocks by 8 Mg C ha^{-1} (+26 %). This was more than the SOC lost when the cropland was established on grassland. A study by Anderson-Teixeira et al. (2009) shows that although the conversion of grassland to miscanthus and switchgrass initially results in moderate soil C losses (on average 5.8 Mg C ha^{-1}), the following SOC accumulation leads to an annual net SOC gain of 1 Mg ha^{-1} in subsequent years. The study by Don et al. (2011) showed that soils in regions with higher precipitation may be more vulnerable to land-use change effects than those in drier regions.

The evaluation of C loss through indirect land-use change in a GHG balance is a matter of debate. Indirect land-use change can occur when the production of palm oil for food purposes encroaches onto areas formerly not used for agricultural production (e.g., rainforest) because the palm oil being presently produced on the land is used for biodiesel production. In that case, the direct land-use change is caused by palm oil production for food purposes. However, indirectly this change is caused by withdrawing palm oil for biodiesel production. The indirect land-use effect is not quantifiable. Therefore, discussions about introducing a “risk adder” arose to account for losses of below-ground and above-ground C that is caused through indirect land-use change. This risk adder is an average value that has been assessed for potential indirect land-use changes occurring globally. Fritsche et al. (2010) discuss values of 2.8–8.5 Mg CO_2 equivalents ha^{-1} y^{-1}.

Presently, the concept of a risk adder is not an obligatory part in GHG balances for biofuels because it is very difficult to estimate an emission value for indirect land-use change. It is also debatable whether it is justified to “punish” biofuels in GHG balances on account of potential indirect land-use effects. The EU directive 2009/28 EU describing the evaluation of GHG balances for the sustainability assessment of biofuels does not include a risk adder. Instead the production of biofuels on degraded land is given a greenhouse gas bonus of 29 g CO_{2equiv}. MJ^{-1} biofuel energy (referring to the energy content of the biofuel).

14.3 Conclusions

The production of annual crops for biofuel supply can positively contribute to SOC levels under three scenarios: (1) low soil-tillage intensity, (2) sufficient residues are left on the field or (3) the organic residues generated in the process of biomass to biofuel conversion are returned to the field. However, while not fulfilling these duties, the effect of cropping may be negative towards SOC (= decrease of SOC levels).

Generally, biomass production in a perennial cropping system is to be preferred compared to annual systems if the aim is to increase the SOC content. For first generation transport biofuels, sugar cane, oil palm and jatropha are suitable as perennial crops delivering sugar or vegetable oil. Whereas for sugar cane and oil palm the production system and varieties are well developed, efficient and deliver reliably high yields, there is little known about jatropha. Also, there are no varieties available for jatropha and the management system needs to be further developed before high and reliable yield levels can be attained.

Although the production of palm oil can be managed sustainably, the extension of oil palm plantations has led to unsustainable land-use changes in the past. If rainforest on peatland is converted for oil palm production, an estimated 230 Mg C ha^{-1} of above and below-ground C is lost and converted into CO_2. Only the conversion of imperata grassland into oil palm plantations may lead to an increase in SOC at an estimated accumulation rate of 13 Mg C ha^{-1} over 20 years. However, as long as the expansion of palm oil production leads to deforestation, very high losses of SOC will occur. Because the establishment of oil palm plantations is financed through wood felling, the practice of deforestation is likely to continue unless prohibited by clear governance.

Solid fuels for the production of heat and power are generally produced from lignocellulosic perennial crops, i.e., trees and grasses, or from residues. The biomass for the production of second generation transport fuels can also be derived from trees and grasses. Among energy crops, perennials in general have the highest impact on an increasing SOC content. As such, the most positive effects on soil C can be expected when solid fuels or second generation transport biofuels are produced from biomass of perennial, lignocellulosic energy crops or forest. Although technically feasible, due to the high production costs of second generation biofuels, their large-scale production is still 10–15 years away, whereas solid biofuels are already used for heat and power production by applying state-of-the-art technology. Therefore, heat and power produced via direct combustion from lignocellulosic biomass appears a potentially more sustainable option than first and second generation transport biofuels.

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Chapter 15
Land Degradation and Ecosystem Services

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Abstract Destructive land use is driving long-term losses of ecosystem function and productivity. Satellite measurements of Normalized Difference Vegetation Index (NDVI) since 1981 provide a global yardstick, revealing that a quarter of the land surface has been degrading over the last quarter of a century; every continent and biome is affected with Africa south of the equator, southeast (SE) Asia and south (S) China hardest hit. The loss of primary productivity is equivalent to more than a billion Mg C but the associated emissions from loss of biomass and soil organic carbon are much greater. Degradation is not confined to farmland (18 % of the degrading area is cropland; 47 % is classed as forest); neither is it strongly associated with drylands, population pressure or poverty. A case study using more detailed data for China explores the effects of soil resilience and the association between land degradation and land use. NDVI can only be a proxy measure of land degradation; assessment of ecosystem services is a further step removed. Remotely-sensed data can be used along with climatic and topographic data as an input to models that predict the provision of these services but the processes, drivers and effects beyond NPP are unseen and more importantly, unmeasured. This is an issue for emerging markets in environmental services.

Keywords Land degradation • Normalized Difference Vegetation Index • Land use • Soil • Climate • Ecosystem services

Abbreviations

AVHRR	Advanced Very High Resolution Radiometer
CGIAR-CSI	The Consultative Group on International Agricultural Research, Consortium for Spatial Information
CIESIN	Center for International Earth Science Information Network, Colombia University
CRU TS	Climate Research Unit, University of East Anglia, Time Series
EUE	Energy-Use Efficiency
FAO	Food and Agriculture Organization of the United Nations, Rome
GIMMS	Global Inventory Modelling and Mapping Studies, University of Maryland
GLASOD	Global Assessment of Human-Induced Soil Degradation
HANTS	Harmonic Analyses of NDVI Time-Series
JRC	European Commission Joint Research Centre, Ispra, Italy
MOD17A3	MODIS 8-Day Net Primary Productivity data set
MODIS	Moderate-Resolution Imaging Spectroradiometer

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NDVI	Normalized Difference Vegetation Index
NPP	Net Primary Productivity
RESTREND	Residual Trends of sum NDVI
RUE	Rain-Use Efficiency
SOC	Soil organic carbon
SOTER	Soil and Terrain database
SRTM	Shuttle Radar Topography Mission

15.1 Context

Ever-more-pressing demands on the land are driving unprecedented land-use change. In turn, unsustainable land use is driving land degradation – long-term loss of ecosystem function and productivity from which the ecosystem cannot recover unaided, and which requires greater and greater external inputs to repair. Symptoms include soil erosion, nutrient depletion, water scarcity, salinity, pollution, disruption of biological cycles, and loss of biodiversity (Dent et al. 2007). This is a global issue recognised by United Nations conventions on climate change, biodiversity and, specifically, the UN Convention to Combat Desertification – but it is contentious. There is little agreement, even about definition of land degradation beyond the bald description 'loss of usefulness for human beings' (Wasson 1987). Plato's evocative account from the fourth century BC describes process, proximal cause, and the resulting loss of ecosystem services and the soil itself. However, modern technical work has focussed on specific processes of degradation and ways and means to arrest them while maintaining productive use of the land. The concept of ecosystem services codified by the Millennium Ecosystem Assessment incorporated production, awkwardly, as production services – and defined land degradation as the 'loss of its services, notably the primary production services' (Millennium Ecosystem Assessment 2005).

Policy development for food and water security, environmental integrity and economic development needs comprehensive, quantitative, up-to-date information. The following questions need to be answered in a scientifically justifiable way. Is land degradation a global issue or a collection of local problems? Which regions are hardest hit and how hard are they hit? Is it mainly a problem of drylands? Is it mainly associated with farming? Is it related to population pressure – or poverty? And in the context of this book, can we relate loss of ecosystem services to any global yardstick of land degradation? Measurement is contentious too. The first harmonized assessment, the Global Assessment of human-induced Soil Degradation (GLASOD) was a compilation of expert judgements of the kind and degree of land degradation, e.g. soil erosion by water or by wind, or salinity (Oldeman et al. 1991). GLASOD is a map of perceptions, not a measure, of land degradation. Its judgments have proven inconsistent and hardly reproducible, and the relationships between land degradation and policy-pertinent criteria were unverified (Sonneveld and Dent 2007).

15.1.1 Proxy Measurement of Land Degradation

Land degradation nearly always affects vegetation. The first opportunity for global vegetation measurement was provided by daily measurements by the Advanced Very High Resolution Radiometer (AVHRR) carried by US National Oceanic and Atmospheric Administration weather satellites since 1981. Its 4 km resolution is not very high but, fortuitously, NDVI (a normalised ratio of red to near-infrared radiation, [NIR-RED]/[NIR + RED]), is a good measure of green biomass (Tucker 1979). NDVI is one of few consistent indicators available at the global scale over a meaningful period and can be translated in terms of net primary productivity (NPP)[1] which can be subjected to economic analysis.

The Global Assessment of Land Degradation and Improvement (Bai et al. 2008) used the Global Inventory Mapping and Modelling Studies (GIMMS) dataset which is corrected for calibration, view geometry, volcanic aerosols and other effects unrelated to vegetation cover. Daily NDVI data are summarised fortnightly at 8 km resolution using a maximum value composite technique to remove the effects of cloud and haze, and regularly updated with new measurements and enhanced calibration (Pinzon et al. 2007). Newer sensors like MODIS (Running et al. 2004) provide higher resolution and a more direct measure of NPP but the value of 30 years consistent AHVRR data can hardly be overstated.

As a first approximation, land degradation and improvement may be measured by long-term NDVI trends. However, a decreasing NDVI trend does not necessarily indicate land degradation, nor does an increasing trend necessarily indicate land improvement. Biomass depends on several factors including climate (especially fluctuations in rainfall, sunshine, temperature and length of growing season), land use, large-scale ecosystem disturbances such as fires, and the global increase in nitrogen (N) deposition and atmospheric carbon dioxide (CO_2). To interpret NDVI trends in terms of land degradation or improvement, false alarms have to be eliminated, particular those arising from climatic variability and land use change. For climate, we can draw upon a century's consistent data but we have no global time series for land use.

Where productivity is limited by rainfall, rainfall variability is accommodated by rain-use efficiency (RUE), *i.e.* the ratio of NPP to rainfall (le Houérou 1984; le Houérou et al. 1988). The combination of satellite-based estimation of NDVI and station-observed rainfall has been used to assess land degradation at various scales (Holm et al. 2006; Prince et al. 2007). Although short-term fluctuations in RUE say more about rainfall patterns than about land degradation, we judge that its long-term trends distinguish between the two. To take account of the lengthening and warming of the growing season at high latitudes and high altitudes, we applied a similar test of energy use efficiency (EUE), represented by the ratio of annually aggregated NDVI to annual accumulated temperature (day degrees above 0 °C). There are

[1] Net primary productivity is the total amount of organic matter created annually (US Department of Energy 2008).

caveats when applying these data globally: The NDVI signal can be saturated at closed vegetation canopy so it is more sensitive for cropland and rangeland than for forest; further, NDVI may be underestimated in perennially cloudy areas; the spatial variability of rainfall in dry regions makes interpolation of point measurements problematic – and measurements are sparse in many of these areas.

We might also expect that the land's resilience against degradation will depend on soils and terrain. The Shuttle Radar Topography Mission (SRTM) digital elevation model provides almost global coverage at a horizontal resolution of 30 m but there is no information on soils that is compatible with NDVI data. Regional information is provided by Soil and Terrain (SOTER) compilations of field surveys in a common format (van Engelen and Wen 1995), and we have used ChinaSOTER at scale 1:1million for a case study. For Zhejiang Province in south China, we also used Landsat TM imagery to assess the influence of land use change.

NDVI is simply a ratio of reflected red and near-infrared light. To provide a more tangible measure of land degradation that may be subjected to economic analysis, we have translated NDVI to NPP using data from MODIS measurements for the overlapping period 2000–2006 (Running et al. 2004; Zhao et al. 2005; Zhao and Running 2010). This translation is approximate.

Trends of greenness or productivity cannot be other than a proxy for land degradation. They identify areas where significant biological change is happening but greenness and productivity tell us little about the nature of the changes and nothing about the drivers. The resolution of the GIMMS data is a limitation in two senses. First, an 8 km pixel integrates the signal from a wider surrounding area. Many symptoms of quite severe land degradation, such as gullies, rarely extend over such a large area. They must be severe indeed to be seen against the signal of the surrounding unaffected areas. More detailed analysis is possible for those areas that have higher resolution time series data (e.g., Wessels et al. 2004) but even a 1 km pixel cannot be checked by a windscreen survey. And we cannot inspect the situation in previous years.

15.2 Data and Methods

15.2.1 NDVI and Net Primary Productivity

Starting with the most recent GIMMS data, we applied the Harmonic Analysis of NDVI Time-Series (HANTS) algorithm (Verhoef et al. 1996; de Wit and Su 2005) to replace outliers, such as any residual cloud effects, and to reconstruct smoothed NDVI time-series (de Jong et al. 2011). Subsequent analysis employed the reconstructed data.

For analysis of linear trends, the fortnightly NDVI data were aggregated as annual $NDVI_{sum}$. The NDVI time series was translated to NPP by correlation with MODIS MOD17A3 data for the overlapping years of the GIMMS and MODIS (from year 2000), re-sampling the MODIS NPP at 1 km resolution to 8 km resolution using nearest-neighbour assignment.

15.2.2 Climate

Monthly rainfall data from 1981 to 2009, gridded at resolution of 0.5°, were taken from the Global Precipitation Climatology Centre full data re-analysis product (Schneider et al. 2008). Mean annual temperatures, also gridded at 0.5° resolution, are from the CRU TS 3.0 dataset (Mitchell and Jones 2005).

RUE was calculated as the ratio of annual $NDVI_{sum}$ and annual rainfall. EUE was calculated as the ratio of annual $NDVI_{sum}$ and accumulated temperature (day-degrees Celsius above zero). Aridity index was calculated as P/PET where P is annual precipitation in mm and $PET = P/\sqrt{(0.9+(P/L)^2)}$ where $L = 300 + 25T + 0.05T^3$ and T is mean annual temperature (Jones 1997).

15.2.3 Land Use, Land Cover and Soils

GLC2000 global land cover data (JRC 2006) and Land Use Systems of the World (FAO 2008) were generalised for comparison with NDVI trends. The SOTER database for China was compiled by identifying terrain units based on SRTM 90 m digital elevation data (CGIAR-CSI 2004) and combining these with the digital Soil Map of China (Shi et al. 2004). The original 67 000 mapping units were generalized to a minimum polygon size of 50 km^2.

15.2.4 Population, Urban Areas and Poverty

The CIESIN Global Rural–urban Mapping Project provides data for population and urban extent, gridded at 30 arc-second resolution (CIESIN 2004). Sub-national rates of infant mortality and child underweight status and the gridded population for 2005 at 2.5 arc-minutes resolution CEISIN (2007) were compared with indices of land degradation.

15.3 Analysis

Areas of land degradation and improvement are identified by sequential analyses of the remotely sensed data:

1. Annual $NDVI_{sum}$ was translated to NPP; trends were calculated by linear regression.
2. Drought effects were masked by:
   (a) Identifying each pixel where rainfall determines productivity, i.e. there is a positive relationship between NDVI and rainfall;

(b) RUE has been considered for those areas where rainfall determines productivity: where NDVI declined but RUE increased, we may attribute declining productivity to declining rainfall; those areas are masked (urban areas are also masked);
(c) For the remaining areas with a positive relationship but declining RUE and also for all areas where productivity is not determined by rainfall, NDVI trends have been calculated as RUE-adjusted NDVI; land degradation is indicated by a negative trend and may be quantified as RUE-adjusted NPP.

3. EUE was calculated for all pixels. In practice, it hardly affects the estimation of land degradation but it does affect the identification of land improvement, which is indicated by a positive trend in both RUE-adjusted NPP and EUE and quantified as climate-adjusted NPP.
4. Seasonality and autocorrelation of the data violate the assumptions of ordinary least-squares regression. As an alternative, we have applied the non-parametric seasonal Mann-Kendall test which is unaffected by serial auto-correlation (Mann 1945) with NDVI normalised for variations in the start or length of the growing season by vegetation development stages (de Jong et al. 2011).
5. To assess the effects of soil and terrain, residuals from the NDVI trend of each SOTER unit as a whole were analysed for the China case study:

(a) The annual mean NDVI of the SOTER unit from 1981 to 2006 was calculated and re-sampled to the GIMMS pixel size;
(b) Residuals of annual NDVI (difference between the annual sum NDVI and the annual mean NDVI of the SOTER unit) were calculated for each pixel and the trend of these residuals (RESTREND-SOTER) calculated by linear regression.

6. Indices of land degradation and improvement were compared with land cover and land use; soil and terrain; rural population density; and indices of aridity and poverty.

15.4 Results and Discussion

15.4.1 Global Greenness Trends

Judging by NDVI, the last quarter century has been a good time for vegetation. Overall, global greenness increased by about 3 % ($P < 0.05$) (Fig. 15.1).

But the picture of NDVI or NPP is uneven. For the period 1981–2006, Fig. 15.2 depicts zonal contrasts in greenness and Fig. 15.3 changes in NPP which show clear regional trends, both decreasing and increasing.

Bai et al. (2008) flagged caveats concerning the reliability of GIMMS data in perennially cloudy regions and the significance of the trends where there is autocorrelation of the data. However, removal of cloud and seasonality using HANTS made no difference to the computed regression compared with the original annual aggregated GIMMS data (de Jong et al. 2011).

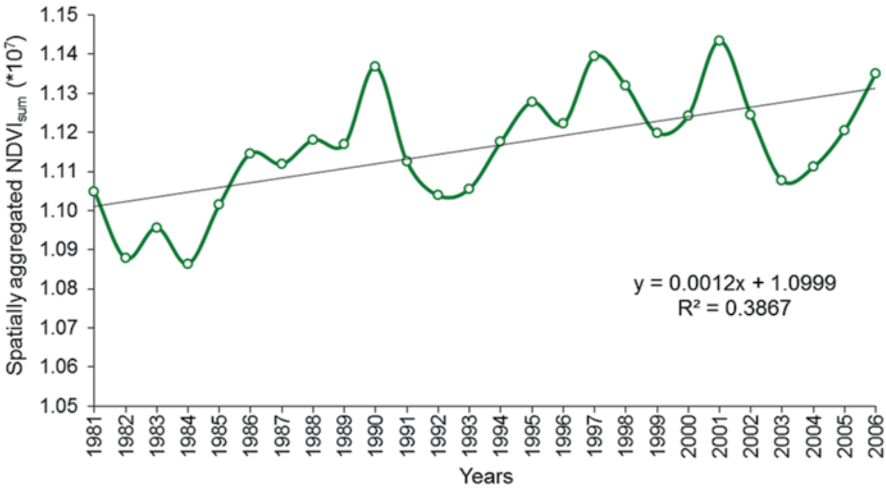

Fig. 15.1 Spatially aggregated annual $NDVI_{sum,}$ 1981–2006

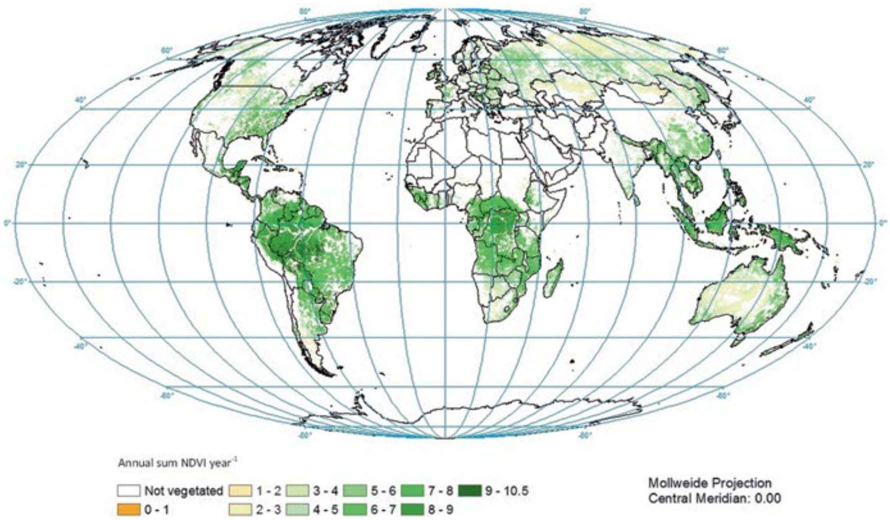

Fig. 15.2 Mean annual $NDVI_{sum}$ $year^{-1}$, 1981–2006

The use of annually aggregated data counters autocorrelation but reveals only a fraction of the information within the full dataset. For example, we cannot interpret changing seasonal responses of the NDVI signal that may indicate the nature of any degradation. Figure 15.4 depicts an alternative approach, making use of the full dataset and applying the non-parametric Mann-Kendall model that is unaffected by autocorrelation. We have normalised the data for variations in phenology from one year to another by considering vegetation development stages rather than calendar years

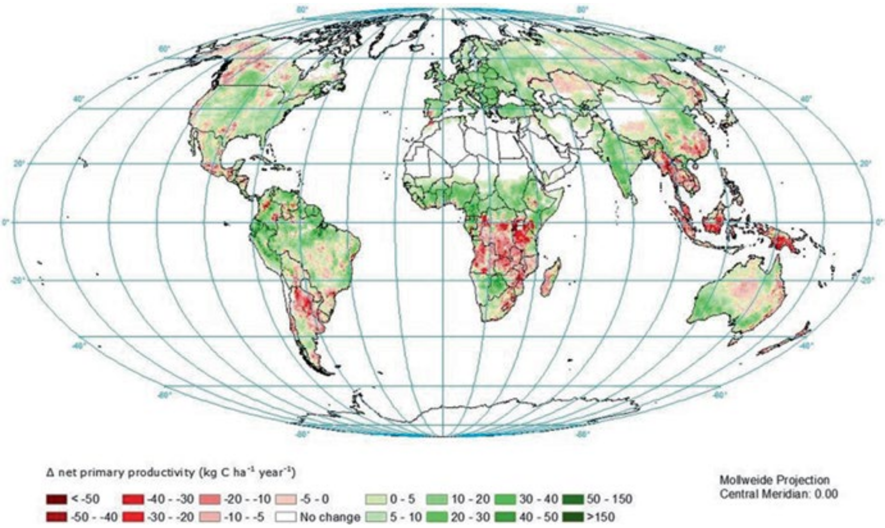

Fig. 15.3 Global change in net primary productivity, 1981–2006 (extreme desert with NPP less than 1 g cm^{-2} are designated as no change)

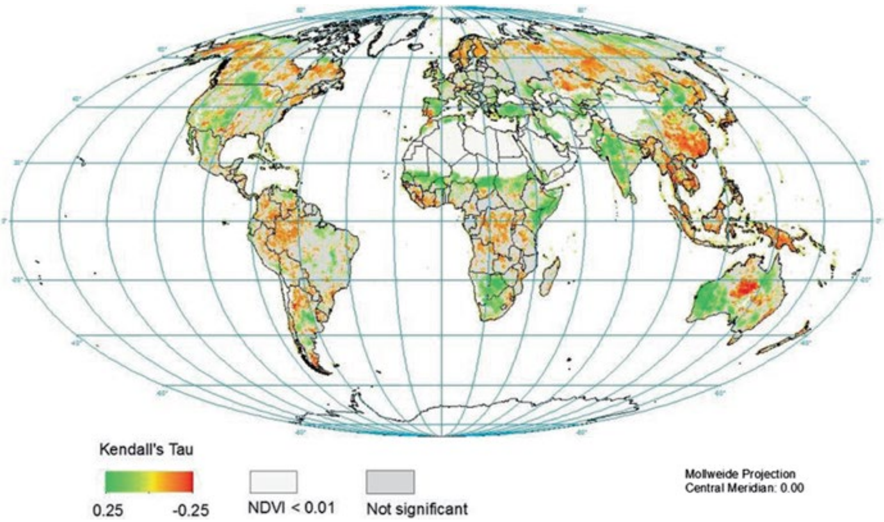

Fig. 15.4 Kendall's tau for NDVI adjusted by vegetation growth stages (Data from de Jong et al. 2011)

(de Jong et al. 2011). Whereas the linear regression measures annual accumulated photosynthetic activity, the Mann-Kendall model measures the photosynthetic intensity of the growing season. If the Kendall's tau value is significantly different from zero, we may conclude that there is a negative (tau < 0) or positive (tau > 0) trend in

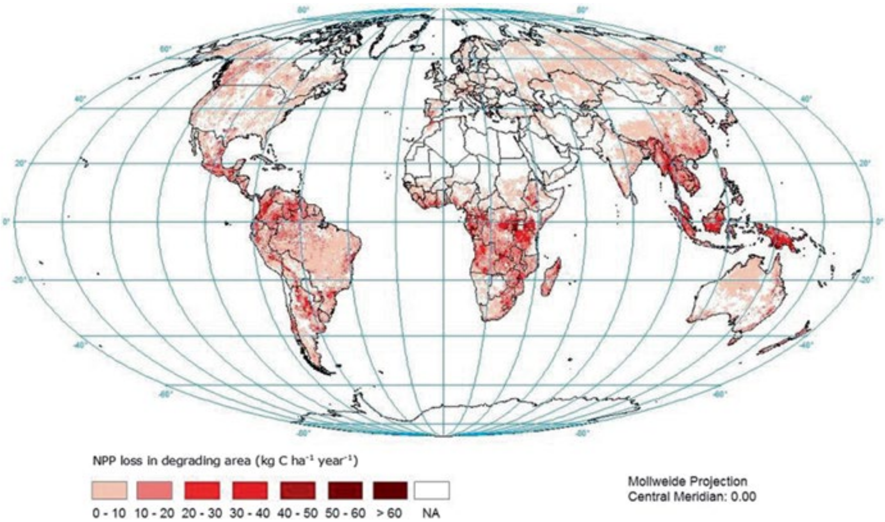

Fig. 15.5 NPP loss in the degrading areas 1981–2006

photosynthetic intensity over time, which we may equate with land degradation or improvement.

These two analyses do not measure the same thing. Each has its own tale to tell but the close similarity of the patterns of greening and browning revealed by the two models suggests that both are robust and we can have confidence in the results.

15.4.2 Land Degradation

Figure 15.5 shows losses of NPP since 1981, adjusted for drought effects using RUE.

Globally, the degrading area amounts to some 35 million km^2 or 24 % of the land surface. The population of the areas directly affected is more than 1.5 billion. Lost NPP over the period 1981–2006 amounts to 1.08 Pg carbon (C),[2] but the unrecognized loss of standing biomass and soil organic carbon (SOC) is likely to be an order of magnitude greater.

The results challenge conventional wisdom. Certainly, they are very different from the GLASOD assessment which compounded current land degradation with the legacy of centuries past. We are dealing only with the period since 1981; there are no comparable data for earlier periods. To answer the questions posed at the outset, we compare RUE-adjusted NDVI (and its translation to NPP) with global

[2] 1.08 billion tonnes (1 Pg = 10^{15} g).

data for land cover and land use, aridity, population density and, as proxies for poverty, infant mortality rates and proportion of underweight children under the age of five.

15.4.3 Which Regions Are Hardest Hit?

Area data refer to pixels showing any declining trend, irrespective of degrees of confidence. By and large, the areas identified as high confidence show the most extreme trends. Thus, intensity of degradation may be ranked more meaningfully according to total NPP loss than by gross degrading area. The hardest-hit regions include:

- Africa south of the Equator (13 % of global degrading area and 18 % of lost global NPP);
- Burma, Indo-China, Malaysia and Indonesia (6 % of the degrading area and 14 % of lost NPP);
- S China (5 % of the degrading area and 5 % of lost NPP);
- N-central Australia and the western slopes of the Great Dividing Range (5 % of the degrading area and 4 % of lost NPP);
- The Pampas (3.5 % of the degrading area and 3 % of lost NPP);
- Swaths of the high-latitude forest belt in North America and Siberia.

The data give useful detail down to country level. The national rankings of severity of land degradation by percentage of the global degrading area are Russia (16.5), Canada (11.6), USA (7.9), China (7.6), Australia (6.2); by loss of NPP (Tg C) rankings are Canada (94), Indonesia (68), Brazil (63), China (59), Australia (50); by percentage of the country affected rankings are Swaziland (95), Angola (66), Gabon (64), Thailand (60), Zambia (60); and by rural population affected (million) China (457), India (177), Indonesia (86), Bangladesh (72) and Brazil (46).

The usual suspects, *i.e.* the dry lands around the Mediterranean, the Middle East, South and Central Asia, are represented only by relatively small areas of degradation in southern Spain, the Maghreb, Nile delta, Iraqi marshes, and the Turgay steppe. Most areas of historical land degradation have become stable landscapes with stubbornly low productivity. In contrast, the greening of the Sahel may reflect both recovery from drought and abandonment of agriculture.

15.4.4 Is Land Degradation a Global Issue?

Over the last quarter century, a quarter of the land area has been degrading – over and above the legacy of thousands of years of mismanagement in some long-settled areas. In 1991, GLASOD estimated that 15 % of the global land was degraded. Those areas are, by and large, not the same as the areas highlighted by the new analysis. Land degradation is cumulative: that is the global issue.

Table 15.1 Global degrading/improving lands in the aggregated land use systems

Land use system	Total pixels (TP) (5′×5′)	Degrading pixels (DP) (5′×5′)	DP/TP (%)	DP/TDP[a] (%)	Improving pixels (IP) (5′×5′)	IP/TP (%)	IP/TIP[b] (%)
Forestry	661,932	194,321	29.3	46.7	65,207	9.9	23.5
Grassland	666,668	105,380	15.8	25.3	111,458	16.7	40.1
Agricultural land	329,862	73,104	22.2	17.6	65,909	20.0	23.7
Urban	52,640	9,114	17.3	2.2	6,152	11.7	2.2
Wetlands	42,572	10,637	25.0	2.6	4,759	11.2	1.7
Bare areas	400,220	11,800	2.95	2.8	19,617	4.9	7.1
Water	62,893	11,904	18.9	2.9	4,571	7.3	1.6
Undefined	499	14	2.8	0.0034	11	2.2	0.004

[a]*TDP* total degrading pixels
[b]*TIP* total improving pixels

Degrading areas currently support 1.5 billion rural people. In terms of C, these areas represent a loss of NPP of 1.08 Pg C relative to the 1981–2006 mean. More than a Pg C not removed from the atmosphere which is equivalent to 20 % of the global CO_2 emissions for 1980. At the shadow price used by the British Treasury in July 2009 (US\$ 51 Mg C^{-1}) this amounts to \$US 55 billion in terms of lost C fixation. But the cost of land degradation is an order of magnitude greater in terms of C emissions, especially from loss of soil organic matter (Bolin et al. 1986; Don et al. 2011; Poeplau et al. 2011). As much as one third of the human-induced increase in atmospheric CO_2 and 23 % of global C emissions over the period 1989–1998 is related to land use change (IPCC 2000; Houghton 2008).

15.4.5 Is Land Degradation Mainly Associated with Farming?

The short answer is no. Cropland occupies 12 % of the land area (and part of a further 4 % of mixed cover) but makes up 18 % of the total degrading area by land cover classes and by land use systems (Table 15.1). Most of the increase in farmland over this period has come from expansion of irrigation (FAO 2011) but irrigated areas fare no better than the norm.

Forest is also over-represented. Forests in total occupy 28 % of the land but 47 % of degrading land is classified as forest. Declining NPP is seen across 30 % of natural and supposedly protected forest, 25–33 % of grazed forests and 33 % of plantations. We can only speculate about the reasons. Certainly, high-latitude *taiga* is subject to catastrophic fires and outbreaks of pests such as the mountain pine beetle (Kurz et al. 2008) that affect huge areas; since the rate of recovery is slow in cold and dry regions the 26 year period will not encompass the whole cycle. This may not be land degradation and we should expect recovery but if these events are, themselves, related to climate change the ecosystem may not recover. Surely, much of the decline of tropical forests is caused by logging and clearance for farming.

Twenty-five percent of degrading land is grassland where the natural and protected areas do seem to be faring better than grazed areas.

Degradation of farmland is still a serious matter. In Kenya over the period 1981–2006, NPP increased in woodland and grassland but hardly at all in cropland. Across 40 % of cropland, NPP decreased – which is a critical situation in context of a doubling of human population during the same period. In the same vein, in South Africa NPP decreased overall and 29 % of the country suffered land degradation, including 41 % of all cropland. Some 17 million people, 38 % of the population, depend on these degrading areas (Fig. 15.6).

False alarms about land degradation may be generated by change of land use. Conversion of forest or grassland to arable, pasture or even perennial crops will usually result in an immediate reduction in NDVI but may well be profitable and sustainable, depending on management. Lack of consistent time series data for land use and management precludes a generalised analysis of land use change but this can be undertaken manually for the potential black spots.

15.4.6 Is Land Degradation a Dryland's Issue?

With the exception of Australia, drylands don't figure strongly in ongoing degradation. Indeed, the Sahel has recovered strongly from the droughts of the 1980s (Olsson et al. 2005). Globally, there is little correspondence between land degradation and Turc's aridity index correlation ($r = -0.12$). 78 % of degradation by area is in humid regions, 8 % in the dry sub-humid, 9 % in the semi-arid, and 5 % in arid and hyper-arid regions.

15.4.7 Is Land Degradation Related to Population Pressure?

Comparison of rural population density with land degradation shows no simple pattern. Globally, the correlation coefficient is –0.3; in general, the more people the less degradation. But in some contexts population pressure is positively related to land degradation. For South Africa (Fig. 15.7), the correlation between land degradation and $\log_e$ population density is positive ($r = 0.25$) but the former apartheid homelands have more than their fair share of degrading land; so we cannot use a simple rural population density *vs*. degradation relationship to explain land degradation in populated areas.

15.4.8 Is Land Degradation Related to Poverty?

Taking infant mortality rate and percent of children under five who are underweight as proxies, there is some global relationship between land degradation and poverty. Correlation coefficients are 0.20 for both infant mortality and for underweight

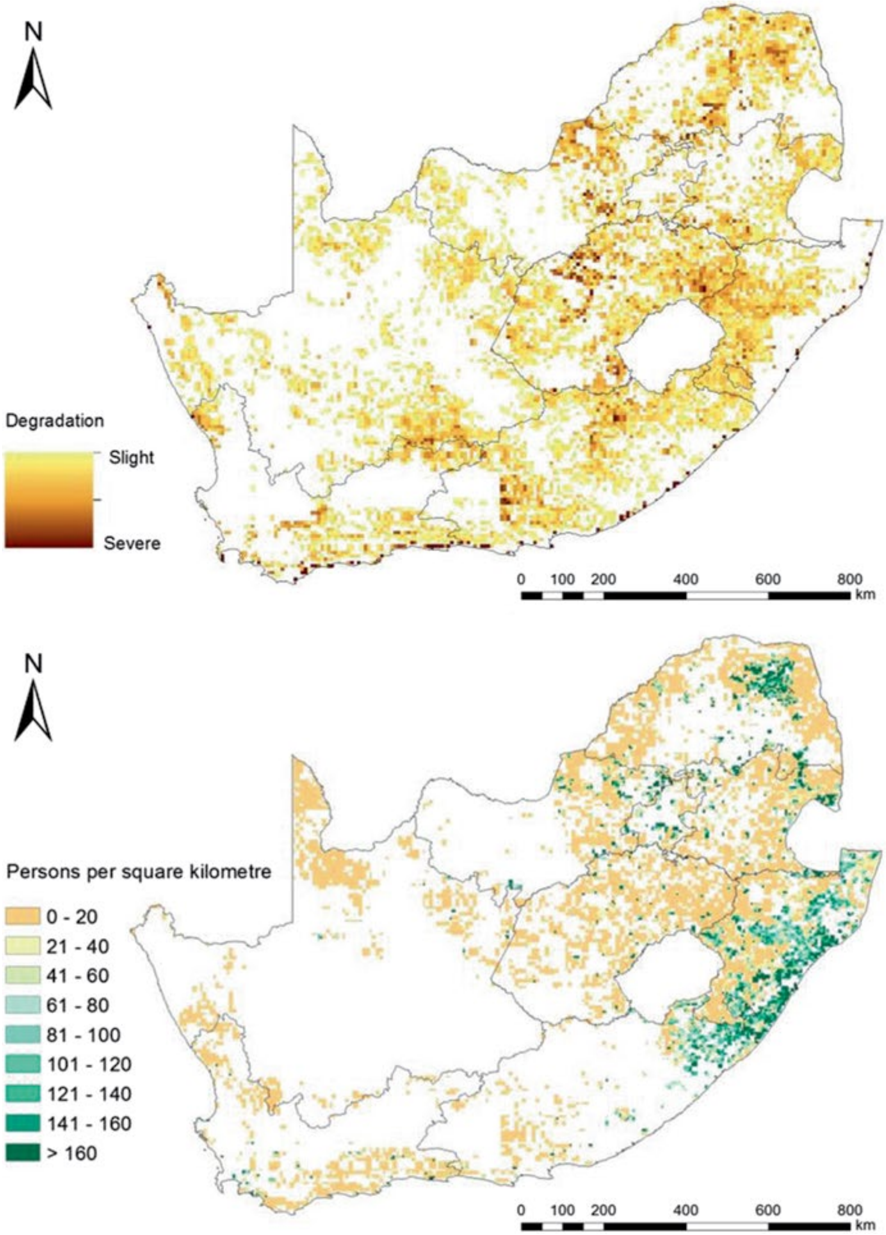

Fig. 15.6 South Africa, land degradation (*top*) and population affected (*bottom*), 1981–2006

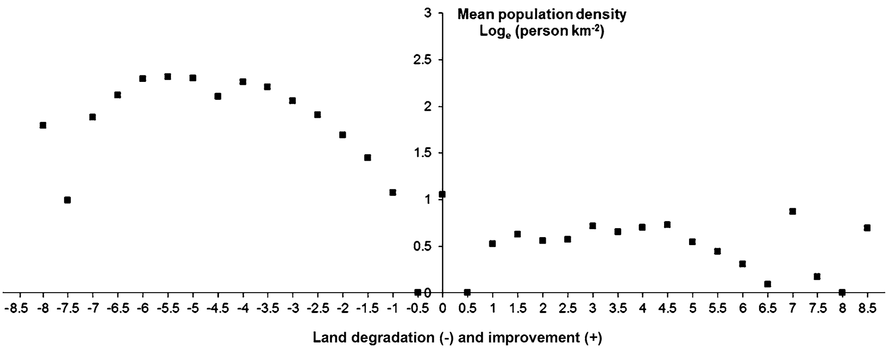

Fig. 15.7 South Africa: relationship between population density and land degradation/improvement

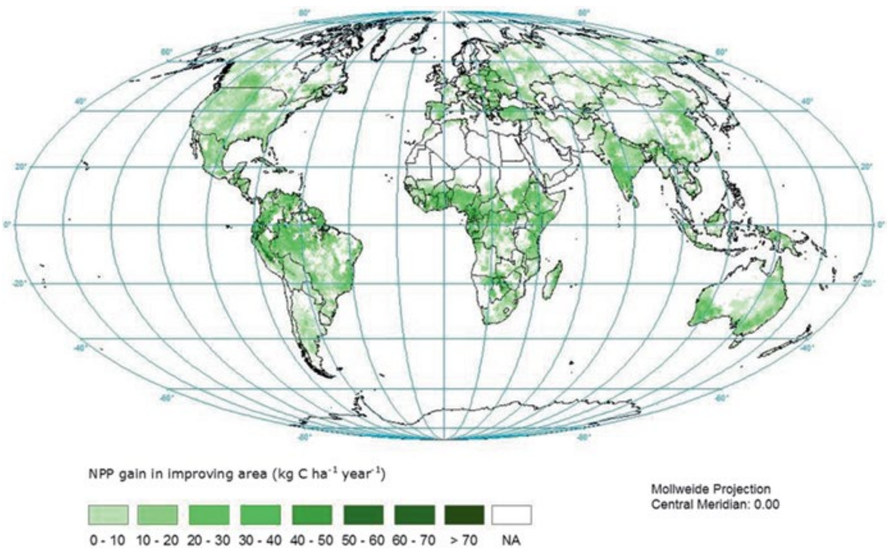

Fig. 15.8 Global increase in climate-adjusted NPP, 1981–2006

children. However, a much more rigorous analysis is needed to tease out the underlying biophysical and social and economic variables, which might be done using more specific geo-located data.

15.4.9 Land Improvement

Land improvement is identified by both a positive trend in RUE-adjusted NDVI and a positive trend in EUE. Figure 15.8 depicts improving areas in terms of climate-adjusted NPP. These areas account for 15.7 % of the land area. Eighteen per cent is cropland (20 % of the total croplands), 23 % is forest, and 43 % rangeland. Many gains in cropland are associated with irrigation but there are also swaths of apparent improvement in rain-fed cropland and pastures in the Great Plains of North America.

Some NDVI gains are a result of forest plantations, especially in Europe and North America (FAO 2006) and some significant land reclamation projects, for example in North China. However, some gains represent farmland tumbling down to grass, for example in the former Soviet Union, and bush encroachment into rangeland which are not generally regarded as land improvement. The increasing trend across the Sahel probably includes an element of recovery from the devastating drought of the early 1980s. Increases in biomass in the Amazon basin may be related to decrease in cloudiness but global data for net incoming radiation are not available to check this.

15.4.10 Urban Areas

Whether urbanisation is land degradation is arguable. It brings a huge increase in financial value but where it involves sealing of the soil surface it is degradation according to our criterion of partial loss of loss of ecosystem functions. Only few urban sewerage systems achieve effective cycling of nutrients. The CIESIN Global Rural Urban Mapping Project shows 3 % of the land area as urban[3] and these areas are masked in the global maps but this makes little difference to the results - a reduction of 0.5 % for the identified degrading land, and a reduction of 0.2 % for the improving land.

15.5 China Case Study

Looking in more detail at China, Figs. 15.9 and 15.10 present trends in climate-adjusted NPP and Kendall's tau, respectively. Figure 15.11 depicts the RESTREND SOTER analysis.

Several big programs of land restoration in the drylands of the north and west of China can be seen as areas of increasing NPP. However, contrary to common perception, most of the degrading areas are not in drylands but lie in the humid and more densely populated south-east region of the country. Based on Fig. 15.9 we calculate a total loss of NPP attributable to land degradation amounting to almost 60 Tg C over the period 1981–2006.

Comparison of degrading land with the Global Land Cover 2000 and FAO Land Use Systems shows that 33–35 % of the degrading area is forest (about 40 % of forests). Agricultural land makes up 16 % of the degrading area (17–18.5 % of

[3]The figure mapped from MODIS data is 0.5 % (Schneider et al. 2009).

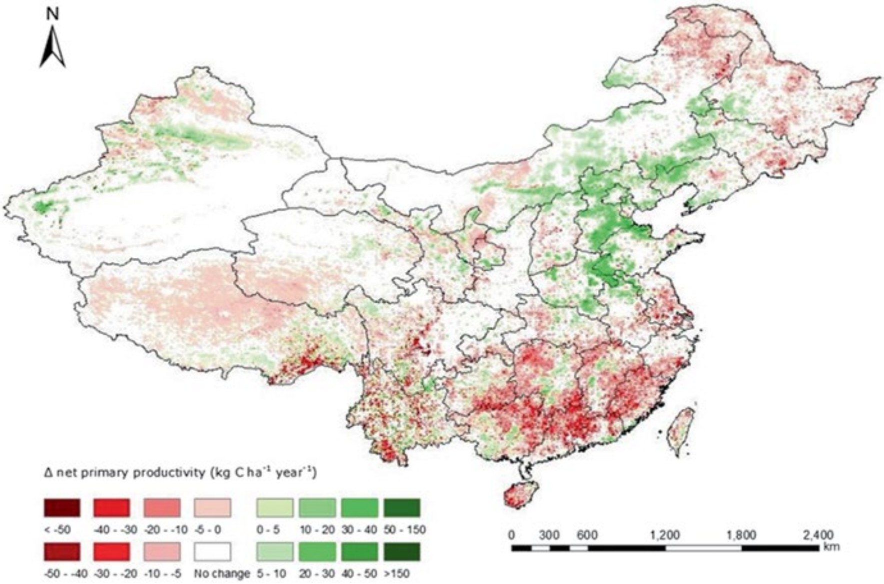

Fig. 15.9 China: trends in climate-adjusted NPP, 1981–2006

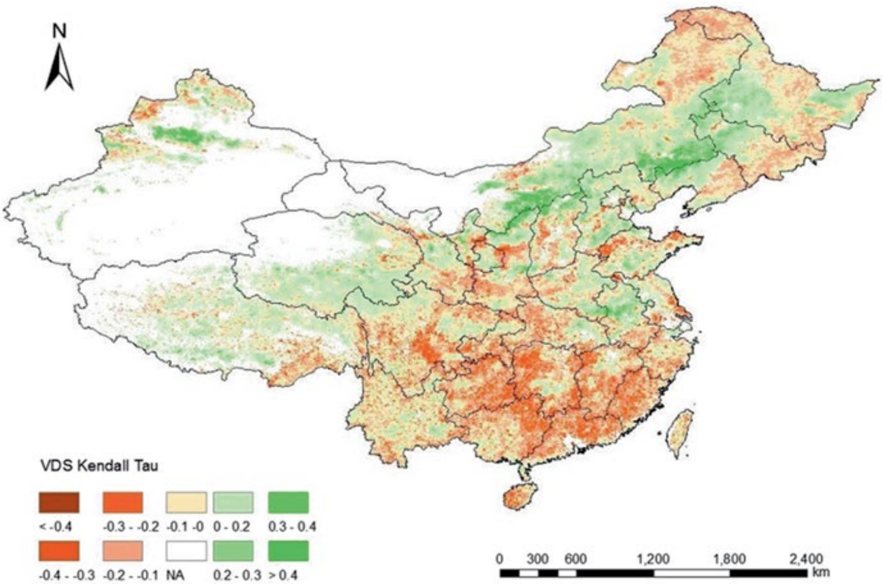

Fig. 15.10 China: Kendall's *tau* with NDVI adjusted by vegetation development stage (Data from de Jong et al. 2011)

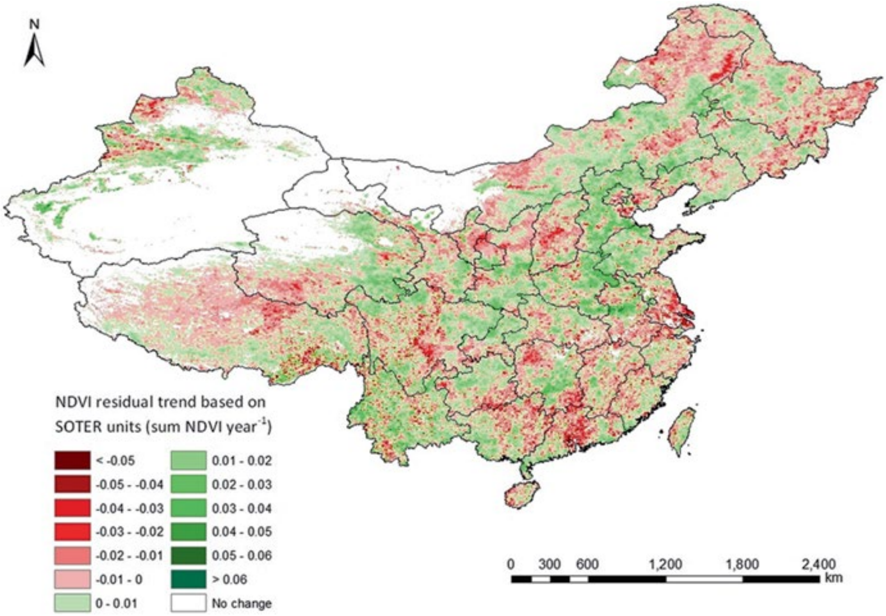

Fig. 15.11 China: trends of NDVI residuals from SOTER units, 1981–2006

farmland, depending on definition) – rather less than the global norm of 19 % although irrigated and protected land fare hardly better than the national average. Thirty-four percent of the degrading area is grassland (28 % of grassland). Of the improving areas, 43 % is grassland, 34 % agricultural land, 14 % forest, and 6 % is classified as bare!

15.5.1 Soil and Terrain Effects

Comparison of degrading areas with terrain shows that all landforms are affected although hills and mountains have more degradation in proportion to their area, and plains and depressions very little. Figure 15.11 shows the relative departure of each pixel from the trend of its SOTER unit. This gives insight into land degradation and improvement within each landscape. This is probably the most useful interpretation for local and regional land use planning. If SOTER units were homogeneous, this procedure would eliminate soil and terrain effects. However, the mapping units are by no means homogeneous. Thus, soil and terrain are accounted for only to the extent that differences within mapping units are less than those between units.

Comparing Figs. 15.9 and 15.11 indicates that about 8 % of degrading land shows a positive RESTREND-SOTER, *i.e.* is less degraded than the norm for its soil and terrain. Most such areas are steepland (15–45 % slopes) that is not intensively managed. The remaining degrading lands show a negative residual trend. Forty-two percent of this land is level, 32 % sloping and 26 % steepland. We may attribute degradation of these areas directly to land use and management. The differences between the areas of negative trend in Figs. 15.9 and 15.11 indicate that soil resilience has a significant effect on the ultimate outcome in the order of 25 %.

15.5.2 Relationships Between Land-Use Change and Land Degradation

Over the past quarter century, China has achieved unprecedented economic growth which has been accompanied by burgeoning cities and infrastructure. If we compare the extent of urban areas with areas of declining NDVI, it is obvious that in rapidly developing areas, like the Yangtze delta and the Pearl River delta, urban development and linking infrastructure account for much of the decline in NDVI.

For Zhejiang Province in south-east China, we have used Landsat imagery from 1985 and 2005 and ground survey to chart land use change. Figure 15.12a plots the trend of climate-adjusted NDVI at the 95 % confidence level on the vertical axis against the numerical change in NDVI ascribed to land use change on the horizontal axis. Figure 15.12b is a similar plot of the trend of SOTER residuals, on the vertical axis, against land use change, on the horizontal axis.

The extension of points in the first quadrant (positive/positive) and the third quadrant (negative/negative) indicates that there is a relationship between land use change and land degradation. Apparently contradictory trends (positive/negative) and the mass of pixels showing degradation without land use change might be explained by changes in management practices without change in the kind of land use, and by ecosystem disturbance such as forest fires and outbreaks of pests and disease that cannot be identified from our data.

15.6 Land Degradation and Ecosystem Services

We have defined land degradation as a long-term loss of ecosystem function and productivity, and have used trends of remotely sensed NDVI as a globally consistent yardstick. The results have proven disconcerting and politically inconvenient.

Various NDVI derivatives show significant regional trends, both decreasing and increasing, that identify areas where significant biological change is happening. As a proxy, they may be interpreted as land degradation or improvement but they are a step removed from tangible symptoms of land degradation like soil erosion, salinity, or nutrient depletion. Land degradation means a loss of NPP but a decrease in productivity, even allowing for climatic variability, is not necessarily degradation. For example, urban development is generally considered to be just that. Land use change from forest or grassland to cropland of lesser productivity invariably leads to a loss of standing biomass and soil organic carbon. However, depending on management it may or may not be accompanied by other symptoms of degradation – and may

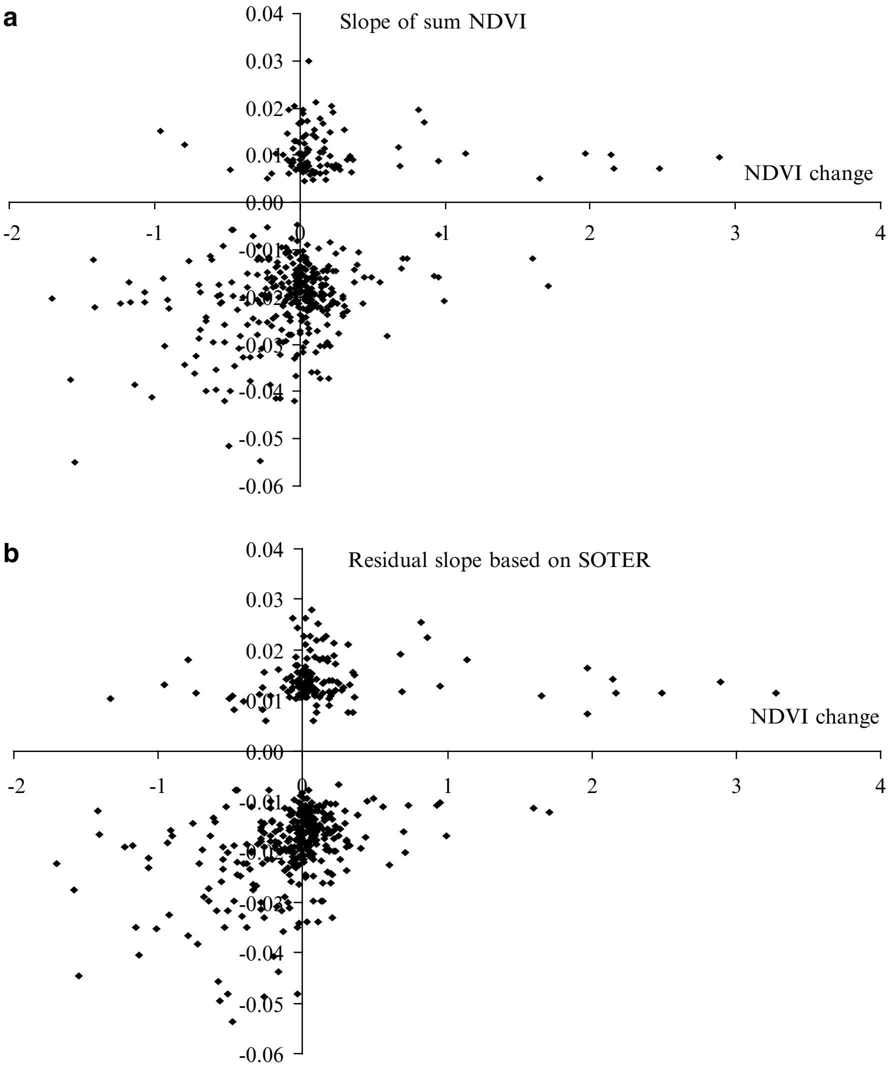

Fig. 15.12 Zhejiang Province: relationships between the slopes of $NDVI_{sum}$ and NDVI changes ascribed to land use change (**a**); residual slopes of $NDVI_{sum}$ and NDVI changes ascribed to land use change (**b**)

well be profitable. Similarly, an increasing trend of NPP means greater biological production but may reflect arable land tumbling down to grass or bush encroachment in rangeland which is not land improvement as commonly understood.

Assessment of any loss of ecosystem services is a further step removed. We may deduce direct relationships between productivity and nutrient cycling, C sequestration, infiltration of rainfall and groundwater recharge; and a more complex relationship with biodiversity. Information that can be extracted from NDVI (leaf area index, the length and intensity of the growing season, as well as NPP) their trends and changes can be used along with climatic and topographic data as an input to models that predict the provision of these services (e.g. for C storage, Milne et al. 2007; Easter et al. 2007). But the processes, drivers and effects beyond NPP are unseen and more importantly, unmeasured. This is an issue for emerging markets in environmental services.

The clearest link is between land degradation and climate but short-term biological productivity is by no means the whole story. The stock of C in standing vegetation is much the same as that represented by CO_2 in the atmosphere. The stock of soil organic carbon (SOC) is at least twice as much (Bolin et al. 1986). It is recognized that, up to 1960, the net emission of CO_2 from vegetation and soils as a result of land use change exceeded that from combustion of fossil fuels. Currently, the annual contribution from deforestation and changing land use is estimated to be 1.6 Pg CO_2-C, about 23 % of the total annual emission of 7 Pg CO_2-C to the atmosphere (Houghton 2008).

There are many well-documented field studies of loss of C through land use change (e.g. Don et al. 2011; Poeplau et al. 2011). In Indonesia, one of the black spots identified in Fig. 15.5, Palm et al. (2005) estimated that logging of primary forest decreases the standing biomass from 306 to 93 Mg C ha^{-1} and under cultivation this is reduced to 2 Mg C ha^{-1}. In tropical forest, below-ground biomass and soil organic carbon, together, is of the same order as the above-ground biomass and most of it is lost over a few years under cultivation. But the greatest stocks of soil carbon are not in tropical forest but in peatland which is not explicitly included in global climate models although it accounts for about half of organic C storage; Batjes (1996) estimates a carbon stock in the top two metres of 679 Pg C but their extent is, perhaps, the least well known and extensive areas are much thicker. Tropical peatland has been drastically reduced in the last 50 years by drainage for agriculture and unstoppable fires and the surface effects can be detected by various sensors, including AVHRR. However, loss of boreal and subarctic peat through global warming is accompanied by greening rather than browning. Another great stock of SOC is in black earth soils inherited from native grasslands, where some 80 % of the ecosystem C is held below ground (Krupenikov et al. 2011). Long term field experiments demonstrate that all current farming systems are mining soil organic matter and, in the case of the chernozem, 30–40 % of the original SOC stock has been lost during one hundred years under the plough. If we consider their global extent (250 Mha), a humus stock of 360–450 Mg C ha^{-1} and an average loss of 33 % of the initial SOC stock, we arrive at total emissions of 19–24 Pg C *which are invisible to remote sensing because these soils are still productive*. We may deduce from Fig. 15.5 that tracts of less resilient soils have already reached a tipping point where we can see degradation in terms of a collapse of productivity.

Extensive evidence for links between land degradation and ecosystem services has been well reviewed for policy makers, for instance by McDonagh et al. (2006), and with respect to soils and climate change by Lal and others (2003 and Chap. 2 in this volume). The problem is that there is no regional or global framework to link the various, often incompatible local observations – so it is hard to upscale local results to a regional and global level (Lal and Bruce 1999) and the results do not command confidence. Furthermore, although technical means preventing land degradation and restoring degraded land and its ecosystem services are well-researched, they are hardly put into practice for the simple reason that they do not pay, especially within the short time horizon of many millions of poor smallholders. This is a market failure.

The recent emergence of C markets and market-related mechanisms to secure water resources (green water credits) has highlighted the deficiency of fundamental information. At present, only the Alberta carbon market deals in SOC as opposed to the more-easily-verified overground sequestration in forest plantations. In the foreseeable future, both carbon and water markets will depend on verification of approved management practices by high-resolution data from the new generation of satellite-mounted sensors and modelling of the resulting benefits in terms of ecosystem services. The participation of myriad smallholders in markets for ecosystem services also depends on many individuals banding together in self-policing cooperatives both to negotiate with buyers and to undertake the approved management practices (Dent and Kauffman 2007).

15.7 Conclusions

Satellite measurements of NDVI since 1981 provide a proxy measurement of land degradation and improvement over the last quater century. Over this time and against a background of globally increasing primary production, a quarter of the land surface has been degrading. Every continent and biome is affected with Africa south of the equator, SE Asia and S China hardest hit. The loss of primary productivity is equivalent to more than a billion tonnes of carbon but the associated emissions from loss of biomass and soil organic carbon are much greater. Degradation is not confined to farmland (18 % of the degrading area is cropland; 47 % is classed as forest); neither is it strongly associated with drylands, population pressure or poverty – but unsustainable land use and management is strongly implicated.

In the face of burgeoning demand for ecosystem services, all the evidence points to a deteriorating global situation that is not being effectively addressed. This is a market failure – ecosystem services are taken for granted and it does not pay to do the right thing. Market-related mechanisms to address the situation, like carbon credits and green water credits are stymied by the difficulties of measuring the benefits of best practice. NDVI is only be a proxy measure of land degradation; assessment of ecosystem services is a further step removed.

Remotely-sensed data can be used along with climatic and topographic data as an input to models that predict the provision of these services. The new generation of satellite sensors yield both higher resolution and more direct measurement of primary productivity but the processes, drivers and effects beyond NPP are unseen and more importantly, unmeasured.

Emerging markets in environmental services have to rely on models that are accessible only to specialists and there is no framework within which detailed case studies can be up-scaled to significant areas of land. Starting from what can be measured consistently, new research is exploring the field relationships between primary productivity and important biological measures like biodiversity (Michael Schaepman, personal communication). The current UNEP Carbon Benefits project is addressing the thorny issue of measuring soil carbon gains by combination of modelling and cheap, reproducible measurement of soil carbon by IR spectroscopy. Assessment of water benefits also depends on models which have proven robust, in themselves, but which depend on good climatic, soil and land use information which, for most parts of the world, is not available at an appropriate scale.

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Chapter 16
The Human Dimensions of Environmental Degradation and Ecosystem Services: Understanding and Solving the Commons Dilemma

Ajay Singh, Robyn Wilson, Jeremy Bruskotter, Jeremy Brooks, Adam Zwickle, and Eric Toman

Contents

Abstract Human drivers of environmental degradation occur as a result of predictable errors and biases in decision making at the individual social, and institutional levels. A better understanding of these human drivers can help policy makers and managers pinpoint the causal forces of environmental degradation as well as implement more effective policies, plans, and management practices that limit degradation and conserve ecosystem services. This chapter briefly outlines select theories and concepts from social science at the individual, social, and institutional level and highlights how multidisciplinary social science perspectives can contribute to the creation of sustainable solutions to environmental problems.

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Abbreviations

C	Carbon
CBRM	Community-based Resource Management
CPR	Common Pool Resource
ES(s)	Ecosystem Service(s)
REDD+	Reduced Emissions from Deforestation and Forest Degradation plus

16.1 Introduction

In his 1968 article, Garrett Hardin illustrated a dilemma that has shaped much subsequent research and debate regarding the relationship between humanity and our environment (Hardin 1968). Assuming every individual on Earth behaves in a way that maximizes the benefits they acrue from shared, i.e., "common pool" resources, the collective effect of individual maximization would lead to depletion of resources and degradation of the environment. Hardin's article effectively illustrated how environmental problems can arise when individual goals do not align with the broader interests of society. Hardin's "tragedy of the commons" provided an excellent foundation for approaching the so-called "human dimensions" of environmental issues.

Hardin described the tragedy through a metaphorical village with a commonly owned cow pasture in which each villager grazed cattle. Hardin argued that each villager had an incentive to add more cows to the pasture because the benefits of adding a cow accrued exclusively to each individual villager, while the costs were distributed equally across villagers. Each rational villager, Hardin concluded, would continue to add cows until the pasture was over-grazed and the soil compacted, leaving the land incapable of producing enough grass to support grazing. Lacking a means by which to exclude individuals from the resource (e.g., via privatization) or constrain individuals' behavior (e.g., via regulation), common pool resources would degrade to a point of ecological collapse. Hardin's warning spawned intense research across numerous scientific disciplines and drew attention to the need for developing approaches to environmental problems that considered human attitudes and behavior.

Hardin proposed two solutions to the commons dilemma: (i) privatize the common resource (e.g., by fencing off and distributing sections of the pasture for individual use) or (ii) regulate resource use through government intervention—what he called, "mutual coercion mutually agreed upon". Either approach to the dilemma faces numerous challenges. While the privatization approach may work for land, it is more problematic for other common pool resources (e.g., air, water, wildlife). The privatization of resources also assumes that there is no geographic variability in the quality or quantity of resources available to the owner. For example, precipitation or soil fertility varies geographically so one plot of land will not be as productive and may create social inefficiencies. In addition, the actions of private individuals on privately-owned lands often have non-monetized impacts on other resources. For example, burning one's pasture to increase regrowth releases carbon (C) into the air while allowing one's livestock to graze along waterways may affect water quality downstream. Unless all of these impacts are known and accounted for, the collective actions of individuals attempting to maximize the utility of privately owned resources may still have negative impacts on society.

In contrast, the use of coercion via governmental regulation relies on the compliance of all users. Because the incentive to maximize utility remains even when regulations are in place, individuals may choose not to comply with regulations or act to undermine them, increasing the need to invest in enforcement mechanisms. Thus, neither privatization nor regulation is a panacea for averting the tragedy of the commons—neither approach provides an ultimate solution.

To address these continuing challenges, efforts have been made in recent years to establish the economic value of ecosystem services (ES(s)) in order to incentivize behaviors that promote environmental stewardship (Hein et al. 2006; Fisher et al. 2009). Ecosystem services is a term that has been used to describe the "benefits humans derive, directly or indirectly, from ecosystem functions" (Costanza et al. 1997; Fisher et al. 2009). Costanza et al. (1997) identified 17 types of ES(s) including water supply, soil formation, waste treatment, pollination, climate regulation, and the provision of raw materials, among others. Designating such economic values would address a shortcoming of privatization as proposed by Hardin by providing a means to ensure that the benefits and costs of resource use accrue to the individual. Once a value is established, the market system could be used to not only trade the resource as a good but also provide payments for resulting ecosystem functions. Moreover, establishing an assessed value for ES(s) may also assist policy-makers by providing a common denominator for negotiating the complex trade-offs that underlie any decision about how to manage common pool resources. That is, by quantifying the impacts of proposed decisions on ES(s), policy makers can make informed judgments about both the costs and benefits (i.e., tradeoffs) associated with that decision. Thus, valuation of ES(s) can be a useful tool that informs both of Hardin's approaches of privatizing or regulating resource use.

However, there remain challenges when quantifying the value of ES(s) as well as estimating the complex trade-offs between valued services that will result from any change in management strategy or policy. Thus, ES(s) are often given little weight in policy making (Costanza et al. 1997; Daily and Matson 2008). The economic valuation of ES(s) has been the major human dimensions approach to date, but the value of ES(s) also depends on addressing individual and social attitudes and behaviors, as well as the institutions that regulate and manage the use and implementation of ES(s). Thus, additional research from the social and behavioral sciences can contribute to the

Table 16.1 Example behaviors or actions from the forest management context from multiple sources that can directly or indirectly impact the environment

	Type of behavior/action	
Source of the behavior	Direct environmental impact	Indirect environmental impact
Individual	Plant a tree	Write your congressperson
Societal	Harvest a private forest	Collect signatures for a ballot
Polity	Conduct a prescribed burn	Pass a law, Establish a precedent

understanding of how an ES(s) approach can be effectively designed and implemented to encourage behaviors that enhance, restore, and protect ecosystem functions. In particular, we posit that human impacts on the environment emanate from three sources: (a) the behaviors of individuals (Individuals), (b) the actions of social groups, interest groups or more generally, "collectives" (Societal), and (c) the actions of political institutions (Polity). Importantly, these types of actions are not mutually exclusive; rather, they overlap and often affect one another. Thus, for example, cultural forces such as norms exert pressure on individuals to act in accordance with the collective, blending individual and collective action (and the disciplines of sociology and psychology). Similarly, an individual may be motivated to take actions when they attend an interest group meeting or hear about a new policy. As a result, it is important to address the most salient influences on behavior at the appropriate level and craft the best possible solutions based on the source of the behavior (see Table 16.1 for a summary of types of behavior occuring at different scales).

This chapter is not a critique of an economic approach to valuing ES(s) but rather a discussion of additional factors that should be considered when creating policies and programs that address environmental degradation and the resulting loss of valued services. Our goal is to describe how increased understanding of the human dimension (i.e., individual behavior, social dynamics and institutional structures) is essential to developing more effective solutions. The remainder of the chapter will (1) outline three scales for thinking about the human dimensions of ES(s), i.e., the individual, the societal, and the institutional level of analysis; (2) identify how behavior and decision making at each level contributes to environmental degradation; and (3) offer additional ways to develop more effective solutions that can be implemented in concert with the valuation of ES(s) and creation of market-based incentives.

16.2 Assessing the Human Drivers of Environmental Degradation

16.2.1 The Limitations of Individual Rationality

It is important to consider what contributes cognitively to the pursuit of individual goals (e.g., resource use), in particular, the pursuit of these goals in the immediate or short-term, as opposed to maintenance of those goals over time. Early models of decision making maintained that human beings were capable of calculating the utility, or the relative value, of a particular choice when deciding between multiple courses of action (Von Neumann and Morgenstern 1947; Baron 2004). Calculating utility not only requires complete knowledge of the consequences of a particular decision, but also the ability to accurately weigh the short and long-term costs and benefits of a particular action. Given this complexity, it is not surprising that judgment and decision making research indicates humans do not actually calculate the decision utility in most decision processes (Gilovich et al. 2002; Tversky and Kahneman 1974). Accordingly, Simon (Simon 1986; Simon 1990) argued that humans are incapable of being fully "rational" actors in an economic cost-benefit sense, rather they are boundedly rational, meaning they make the best possible decisions given the limited resources available to them (including the time required to make a decision, knowledge or projections concerning the expected costs and benefits, as well as the cognitive ability to process and carefully consider and weigh all of the relevant information).

Bounded rationality results in individual decisions that do not necessarily maximize for a specific goal, but rather, merely *satisfice*. In contrast to the intense, deliberative decision-making often necessary to optimize goal attainment, satisficing occurs when individuals reach some threshold where they believe their efforts and information are "good enough" to make a decision (Simon 1947). Degradation of the commons can result from such boundedly rational decision-making. Simply put, individuals do not always account for, nor are they able to appropriately weight, the long-term economic and environmental costs of their resource use decisions. Instead, they make trade-offs that frequently negatively impact the long-term sustainability of the resource in order to reach their immediate goal, i.e., usually, maximization of short-term economic benefits. From this perspective, the actions of individual cattle grazers in Hardin's metaphor arise from their inability to recognize that the short-term utility gained by adding cattle will ultimately result in the long-term degradation of the common pasture, and individual negative consequences in the future.

Literature in judgment and decision making (i.e., behavioral economics, cognitive, and social psychology) offers insight into the many predictable biases in decision making that contribute to this tendency to satisfice, and, in particular, only consider the immediate, most salient objectives or goals during decision making. Two of the most prevalent biases that are relevant here are loss aversion and discounting. Loss aversion refers to the idea that expected (and past) losses and gains are not equal. That in fact people make very different decisions depending on whether the choice involves the pursuit of a gain or the avoidance of a loss (Kahneman and Tversky 1979). Specifically, individuals are averse to losses and will work harder to avoid potential losses than to pursue potential gains. One particular implication for human behavior has to do with how individuals respond to risk when losses or gains are at stake. When an individual is considering the potential losses associated with a choice, they tend to be willing to take risks, willing to gamble in order to avoid the loss, while potential gains invoke a tendency to be risk averse or unwilling to gamble in order to achieve a gain (Tversky and Kahneman 1981).

In general, the tendency to make choices that avoid loss is a reliable phenomenon (Kuhberger 1998). In the context of managing forest resources, Wilson et al. (2011)

found that federal fire managers exhibit loss aversion, choosing the safe option during a hypothetical wildfire event more often when the consequences of the choice were framed as potential gains (e.g., saving homes) compared to potential losses (e.g., losing homes). Further, Wilson et al. (2012) found that individuals living in the Lake Tahoe Basin in Nevada, USA, were willing to accept a greater likelihood of agency management failure when the need to manage was framed as necessary to restore forest health (i.e., recover a loss) as opposed to improve forest health (i.e., achieve a gain). In the context of the commons, whether an individual thinks about avoiding current losses versus achieving future gains, in terms of available resources, will influence the amount of risk they are willing to take to achieve their goals. Perhaps the adoption of a loss frame results in overly risky behavior, and the potential for potentially greater losses over time.

This tendency to avoid short-term loss is also reflected in the second bias, namely discounting, which results when an individual assigns relatively less meaning to the future costs and benefits of a decision (Strotz 1956). The individual essentially places greater weight on the short-term costs and benefits of a decision with limited consideration of the long-term. Such behavior is reflected in the proverb that originates from the 13th century, “a bird in the hand is worth two in the bush”, reflecting the idea that one should take what they can get now because the future is uncertain and the potential to earn or achieve more in the future is not worth the risk. Discounting provides another explantion for why Hardin’s pastoralists would continue to add cattle, even if they understood the long-term consequences of overgrazing. Specifically, they may simply discount long-term benefits relative to short-term benefits.

The construal level theory of psychological distance (Liberman et al. 2002; Trope and Liberman 2003) helps to explain why people are prone to the discounting bias (and perpetuating commons dilemmas). People can only directly experience that which is happening to them in the here and now, everything else is removed from them by some amount of “psychological distance.” Events that are immediately experienced by someone (using a saw to cut down a tree) are construed concretely, while events that are psychologically distant are construed more abstractly. Events can be removed from direct experience on any of four domains, i.e., (i) temporal (this tree will be cut down in 30 years), (ii) spatial (a tree was cut down in a distant forest), (iii) social (someone unlike you cuts down a tree), or (iv) hypothetical (there is a possibility that the tree was cut down). Most importantly, how one construe’s an event (or situation, idea, or message) affects how to perceive, evaluate, and respond to that event, with additional decision weight being given to those outcomes that are *congruent* with the invidual’s current level of construal. For example, in the case of soil degradation, farmers have to choose between the more concrete benefits of applying chemicals to their fields, with the short term payoff of increased yields and decreased weed and insect pressure and the abstract costs of decreased soil fertility and impaired structure, not to be realized for several years into the future. A farmer who is focused on more immediate, concrete concerns (i.e., this season’s yields, current weed and insect pressure) would be more likely to apply chemicals in order to maximize short-term benefits than a farmer who is more focused on abstract goals (i.e., long-term sustainabilty of soil health and structure) (Zwickle and Wilson, in press).

This brief review of the limits of individual rationality provides a warning for those who assume the valuation of ES(s) provides a panacea for addressing environmental problems. Even if decision makers have perfect information of the short and long-term benefits associated with a policy, practice, or decision, psychological biases may prevent adequate consideration of long-term effects. Specifically, the combination of loss aversion, discounting, and construal level effects help explain why individuals would attempt to maximize personal gains in the short-term, in an attempt to avoid future loss, or forego future gains that are uncertain. A recognition of these biases is necessary to design policies to encourage behaviors to protect ES(s). In particular, such efforts should work to help individuals more accurately assess the future costs and benefits of a decision, helping to ensure that their personal goals and values will be met not just in the short-term but over time as well.

16.2.2 The Power of Social Context

In addition to the cognitive biases that affect individual decision-making, humans are also influenced (sometimes subconsciously) by contextual information from our social environment. Thus, making decisions that contribute to the maintenance and provisioning of ES(s) is not only influenced by the individual-level characteristics of the decision-maker but also by societal level influences—the lenses through which we see our social environment (e.g., social norms, identity and trust). Tetlock (1985) described decision-makers as “politicians” who are ultimately accountable to their “constituents”—that is, their family, friends and peers. One way in which our social affiliations affect our judgments and behavior is through “norms”. Put simply, a “norm” is an individual’s perception about the social acceptability of a particular behavior. Sometimes, society sends clear, consistent messages about what is expected of individuals, and in these cases, the norms—or evaluations of behavior— that emerge tend to be internalized as expectations about how to behave (Grasmick et al. 1991). Some researchers refer to such evaluations as “moral” norms (e.g., Stern et al. 1985). However, often what is acceptable or appropriate behavior in a particular situation is unclear—and in these cases, we look to our social environment for cues to discern how we should behave (Khoury 1985; Cialdini et al. 1990).

Schwartz’s *Norm Activation Model* (Schwartz 1968, 1977) proposes that in order for a person’s norm to impact their behavior, it must first be “activated”. Norms are said to be activated when (i) people have some awareness that their actions are likely to have consequences for the welfare of others (i.e., *awareness of consequences*), and (ii) when an individual ascribes at least some of the responsibility for their actions to his or herself (i.e., *ascription of responsibility*) (Schwartz 1968). Schwartz’s model (or variants thereof) has been used in a host of studies that examine pro-environmental behavior with findings generally indicating that the stronger the social norm, the more likely one is to act in accordance with that expectation.

It also plays a key role in the *Value-Belief-Norm Theory* developed by Stern and colleagues (Stern et al. 1999; Stern 2000). Findings from multiple studies examining environmentally-relevant behaviors indicate that societal expectations— a function of dominate culture—can profoundly influence individual behavior (Stern et al. 1999; Kaiser et al. 2005; Oreg and Katz-Gerro 2006).

Returning to Hardin's dilemma, a social expectation that individuals act in a way that is compatible with the long term sustainability of the biophysical environment could help over-ride failures in processing that bias judgment toward short term thinking. However, the inverse is also true. That is, when environmentally-unfriendly management practices become the norm (as when individuals are unaware of the detrimental consequences of their behavior), then future decisions are likely to be biased toward maintenance of the status quo (see further discussion below). Importantly, individual perceptions of societal norms are most influenced by important others (e.g., family, friends, peers). Norms can also prove problematic when the judgment of such important others conflicts with the expectations of society at large resulting in mixed messages for decision-makers.

Here the work of Cialdini and colleagues (Cialdini et al. 1990, 2006) is relevant as they have sought to examine behavior in contexts where what was socially appropriate was not necessarily clear. They posited that two types of normative information were relevant to understanding behavior in these contexts, i.e., *descriptive norms*, which refer to the behavior exhibited by others around an individual (or "what is commonly done"), and *injunctive norms*, which refer to what is appropriate (or what "ought" to be done). In one experiment, Cialdini et al. (1990) gave unwitting subjects the opportunity to litter (i.e., a flyer left on their car windshield) either in an environment littered with flyers or one that had clearly been cleaned of littered flyers after witnessing another individual who either dropped a flyer on the ground or simply walked through. Thus, they sought to manipulate the descriptive norm (clean *vs.* littered) by changing the state of the environment, while they used the other individual to draw attention to the injunctive norm (i.e., what *should* be done in this context). Consistent with study predictions, they found (i) people were more likely to litter in a littered environment, (ii) people were most likely to litter when they saw someone else litter into a littered environment, and (iii) people were least likely to litter when they saw someone else litter into a cleaned environment as this scenario drew subjects attention to the inappropriateness of that behavior. These results show the profound effect that social forces can have on environmentally relevant behavior. While only 6 % of subjects littered under the condition cleaned environment and confederate littered, more than half littered when they saw the confederate drop their flyer into an already littered environment reinforcing the descriptive norm that littering is acceptable.

These studies emphasize how people's behaviors are influenced both by the state of the environment and their perceptions about what others are doing. Put simply, when the environment looks like a dump, people are likely to treat it like a dump—especially when their observations of others' behavior reinforce this perception. Individuals perceptions about what is commonly done provide a powerful decision shortcut or "heuristic" for decision-makers that serves to perpetuate existing behaviors. When existing behaviors promote the protection of ES(s), individuals' reliance on what others are doing can be advantageous. However, such reliance becomes problematic when the normally accepted practices contribute to degradation of ecological conditions.

Social identity refers to an individual's definition of the self, relative to his/her membership in a social group (Haslam et al. 1999). Brewer and Hewstone (2004) explain that "social identities are aspects of the self that have been particularly influenced by the fact of membership in specific social groups or categories and the shared socialization experiences that such membership implies". For example, identification with a particular group may lead to favoritism toward that group (i.e., in-group bias) (Tajfel and Turner 1979). One of the best examples of in-group bias involves people's attachment to sport teams. Across cultures and types of sports, many people exhibit substantial emotional investment in sporting teams despite the fact that these teams have little to no economic impact on people (Cialdini et al. 1976). Strong shared team identities has been linked to an increase likelihood of participation in sports riots (Russell 2004), illustrating how identification with in-groups can have important consequences for behavior.

Likewise, whether we define ourselves as hunters, farmers, environmentalists, or affiliated with a particular political party this identity can impact how we receive, process and ultimately act upon new information. For example, one study found that identification with key interest groups (e.g., hunters, ranchers) affected what people believed about the potential ecological outcomes associated with species restoration (Bruskotter et al. 2009). Likewise, political party affiliation has been shown to impact what people believe to be true about climate change as well as potential policy responses (Hart and Nisbet 2011). Other research ties social identity directly to environmentally-responsible behavior (Terry et al. 1999; Mannetti et al. 2004; Sparks and Shepherd 1992). For example, Terry et al. (1999) found that self-identification as someone who recycles helped explain recycling behavior along with more traditionally measured norms, attitudes, and perceived ability. Sparks and Shepherd (1992) found a similar effect of identification with "green consumerism" on individuals' intentions to consume organic vegetables.

Psychological research illustrates the important role that social identification plays in how individuals receive, process, interpret and act upon information tied explicitly to our identities. These studies can assist decision-makers attempting to deal with common pool resource dilemmas by, for example, understanding how best to frame communications, and when prescriptive, top-down policies (i.e., Hardin's *mutual coercion*) are likely to succeed or fail.

In addition to the social influences on behavior described to this point as described in Hardin's example, the environmental condition and ES(s) provided by the pasture depend on the actions of all those involved in the pasture's management. In such scenarios, while each villager makes an individual decision regarding the number of cattle to place on the pasture, the condition of the pasture depends on the collective decisions made by the group. This characteristic of commons dilemmas is known as interdependence (Komorita and Parks 1994; Kollock 1998). The interdependence increases the uncertainty regarding the future outcomes and encourages actions that have the greatest likelihood of maximizing short term benefits.

The adoption of agricultural practices to sequester C, for example, falls into a particular type of dilemma known as a *social fence*. Social fences represent those situations where an individual action may result in two outcomes, separated in time. Initially, a particular action results in an immediate negative cost with a delay prior to the realization of future benefits (Komorita and Parks 1994). Agricultural practices that contribute to C sequestration can impose short-term costs (e.g., reduced yield, discontinuance of traditional practices, need for new equipment) that will be borne by each farmer. However, the future benefits will be shared by the group as a whole. This leads to a phenomenon, known as free-riding, that creates a strong disincentive to adopting such behaviors (Komorita and Parks 1994). While a few farmers may choose to forgo some short term gains and adopt practices that contribute to C sequestration, in the absence of compensating mechanisms (e.g., payments for ES(s)), the majority are more likely to adopt practices that provide greater short term benefits. Such actions will ultimately result in limited benefits being realized as the long term benefits of C sequestration depend upon adoption of appropriate practices by a sufficient proportion of farmers.

Because of this interdependence, individual's decisions are strongly influenced by the existence of trust between participants (Gambetta 2000). Specifically, to make a decision to accept the short-term costs, the different actors have to trust that the other participants will do likewise. Otherwise, they will be less successful in the short-run and potentially fail to achieve the desired outcome in the long run (Kollock 1998).

Substantial social science research has examined the concept of trust. A recent review by Earle (2010) examined 132 empirical studies of trust conducted between 1986 and 2009. Across these studies, Earle identified three primary approaches to conceptualizing trust, i.e., (i) an emphasis on relational trust based on relationships between people (or between people and an organization), (ii) calculative trust (often referred to as confidence), which emphasizes the demonstrated abilities and past performance of the other actors, and (iii) a combined approach that includes both relational and calculative items. In regards to land management decisions, farmers are likely to use a combined approach and base their consideration of trust on existing relationships with other farmers and government agencies with whom they interact as well as their past practices and performance keeping previous agreements. Such considerations are possible at a local level. Farmers are likely to have existing relationships and have at least some understanding of the practices used by those located geographically nearby. However, it will be more difficult to draw such judgments on a larger scale (e.g., watershed-wide, nationwide, globally). If achieving the benefits from some action, such as C sequestration, requires cooperation at these broader scales, establishing trust among the actors is likely to be a significant challenge. Given the multiple participants and their relative anonymity, even farmers who are individually motivated to adopt such practices may choose not to due to concerns about whether a high enough proportion of the rest of the group will engage in cooperative behaviors to provide the long-term benefit (Kollock 1998). The challenge in engaging in such actions is evident in the challenge of developing an acceptable international agreement on climate change mitigation and adaptation.

16.2.3 Institutional Failure

Just as individual decisions can be constrained and influenced by social and environmental contexts, decision-making can also be influenced by institutional structure and mechanisms. Institutions can be defined as shared rules or norms that describe what is required, forbidden or permitted and are predictably enforced (Ostrom 1990) or more generally as the constraints that people devise to shape human behavior and interactions (North 1990). One can distinguish between informal institutions, such as the social norms discussed above, and formal institutions, such as written policies, laws, and constitutions. In reality, there is a continuum from informal to formal institutions (North 1990), though this section will largely focus on the latter.

Both of the solutions to the collective action problem proposed by Hardin (1968) involve institutions. The institution could be in the form of rules about the number of cattle an individual is permitted to graze or rules about property ownership that structure how the costs and benefits of decisions accrue and to whom. Institutions that ultimately constrain individual behaviors can be present at a number of scales ranging from the international-level to the regional, state, and district-level, and down to the community or social group. While institutions can be an important tool for avoiding or minimizing environmental degradation and averting the tragedy of the commons, institutions do not necessarily always produce positive environmental outcomes. There are a number of reasons that institutions may fail to produce positive environmental outcomes or even result in greater environmental degradation. First, institutions are not easy to develop and maintain. Scholars of the commons have identified several factors that tend to inhibit the ability to develop institutions to manage common pool resources. These factors include, but are certainly not limited to, highly mobile resources, poorly defined resource or governmental boundaries, high transaction costs, low social capital, lack of monitoring, and difficultly enforcing rules (Agrawal 2002; Agrawal and Yadama 1997; Ostrom 1990). These obstacles may make it challenging for groups of people at many scales to reach agreements, and enforce those agreements, to ensure sustainable resource use or protect ES(s).

Secondly, institutions may be intentionally crafted to maximize profit or some other goal that does not benefit environmental conservation and may be antithetical to the protection of ES(s). The way institutions function depends on the goals and aspirations of those crafting the specific rules and regulations, as well as the quality and quantity of monitoring and enforcement to ensure the agreed upon rules are followed. When the people who are crafting institutional rules are more interested in maximizing resource extraction than in conservation, those rules will likely not shape behavior in ways that supports ES(s) protection.

A third problem is that institutions can be captured by individuals, organizations, or interests who then re-shape them for their own purposes to the detriment of the environment. One example of this process is called *agency capture*, which occurs when an industry or specific group of individuals is able to dominate the agency decision-making processes to favor the goals and interests of that industry or group (Mullen 2007; Mank 1993). In some cases, such capture may lead to a weakening of environmental regulations or perverse the interpretation of standards, statutes, policies, or laws in order to allow for activities that otherwise would have been forbidden or restricted. In the context of community-level institutions around the world, this has been called *elite capture* (Wyckoff-Baird et al. 2001). Elite capture occurs when political leaders, or the wealthiest and most influential members of a community are able to accrue the benefits of an institutional structure (e.g., being able to earn most of the money from an eco-tourism venture). Because only a small proportion of the community benefit elite capture can lead to dissatisfaction with the institution among the broader community, to non-compliance with institutional rules, and, subsequently, to increased degradation of local environments.

Citizens may also view such capture as corruption of government agency officials leading to a loss of credibility and legitimacy and influencing the institution's ability to shape resource management (Ostrom 1999). Corrupt officials facilitate free-riding by turning a blind eye to those who do not contribute and create local social norms which may delegitimize rules regulating conservation of the common pool resource. The accountability of resource monitors and officials responsible for resolving conflicts is critical for ensuring that agreements are not only monitored but properly enforced (Agrawal 2002; Ostrom 1990). Agency capture, elite capture, and corruption are barriers to implementing an ES(s) approach because in all cases an industry, group, or individual may be able to work around or within the institutional structure to ensure that their interests or goals are given priority over the well being of others or the condition of the environment.

Fourth, even when institutions are intended to reduce environmental degradation, they may lead to unintended consequences. One type of unintended consequences is perverse incentives. In a conservation context, perverse incentives occur when an institution that is meant to protect the environment instead creates conditions that lead to increased environmental degradation (Wunder 2007; Adger and Vincent 2005). A commonly cited example of perverse incentives comes from the U.S. Endangered Species Act (ESA), which is designed to protect threatened and endangered species, in part, by requiring the protection of habitat for those species and limiting the land management practices that are allowed if a protected species is present. Out of concern that regulations on land use and development will be imposed if an endangered species is found on private property, the ESA has created an incentive to preemptively clear habitat to avoid land use restrictions. Thus, while the policy goal is to protect threatened and endangered species, ESA regulations encourage the destruction of habitat that those species require.

The fifth way in which institutions can fail to produce positive environmental outcomes may come in the form of institutional "crowding out". Crowding out refers to situations in which institutions are imposed in a way that ends up reducing (or "crowding out") cooperative behavior that was already in place. One form of crowding out can occur when an institution imposes punishment in a situation in which cooperation had been previously established. Punishment by an outsider has been shown to create distrust and crowd out voluntary cooperation (Fehr and Falk

2002; Bohnet and Baytelman 2007). Another form of crowding out, and one that is perhaps more common, occurs when economic incentives crowd out voluntary cooperative behavior (Frey 2000). In this case, economic incentives remove the moral impetus for the behavior and the ability of an individual to signal that he/she is a good citizen through voluntary cooperative behavior. When economic incentives are not great enough to make up for the loss of this signal, cooperative behaviors decline. For example, Titmuss (1971) found that paying blood donors led to a drop in donations. More recently, research has indicated that incorporating money, or even the notion of money, into social experiments resulted in less pro-social and more individualistic behaviors (Vohs et al. 2006), i.e., precisely the kinds of actions that can contribute to the tragedy of the commons and other forms of environmental degradation.

Finally, locally-devised institutions can be disrupted and undermined with the intrusion of central governments (Agrawal 2002). One example of this process comes from Nepal, where traditional resource management institutions were ignored when forests were nationalized in the 1960s and again when Sagarmatha National Park was created in 1976. The Sherpas living in the area had long-standing rules about where they could cut fuel wood for themselves and trekkers (e.g., not from steep slopes or near temples), how much could be taken, and under what conditions. Local forest guards who lost their role when the forests were nationalized had enforced these rules that are now enforced by state officials. The Sherpa lost their sense of ownership over the local forests and their motivation for responsible resource management. Ultimately, timber extraction increased. Nationalizing forests and establishing the National Park while ignoring traditional resource management practices essentially turned what had been a common property regime managed by the collective users of the resource into an open-access regime with little control of use, which often results in a tragedy of the commons (see Borgerhoff and Coppolillo 2005 for a brief description and Stevens 1997 for a more in-depth discussion of this series of events).

In general, institutions at any scale are not easy to develop and maintain, and when they do emerge, they are certainly not always geared towards environmental conservation or may not have the consequences that were intended. In short, institutions are not a panacea, and can do more harm than good if not crafted and enforced properly.

16.3 Solutions for Conserving Ecosystems and Implementing an Ecosystem Services Approach

Environmental degradation results from human behaviors that are driven by complex interactions between human needs, desires, and values and environmental characteristics and processes. As the discussion above illustrates, the drivers of environmental degradation occur at multiple scales—as a result of (1) individual cognitive errors in judgment, (2) the tendency for individuals to use society as a guide for acceptable behavior, and (3) a variety of institutional failures that codify what is acceptable behavior. Effectively addressing environmental degradation will depend on overcoming individual biases in judgment, changing social norms, building social trust, and developing and implementing more effective policies or plans. In this section, it is discussed how to address environmental degradation by applying solutions at each scale.

Heberlein (1974) proposed three strategies for addressing environmental degradation (i) using technology to alter the physical environment, or (ii) using cognitive or (iii) structural strategies to change human behavior. Technological strategies are attractive because technology can circumvent the need to address human behavior (Heberlein 1974). However, technological strategies have been criticized because this approach makes simple assumptions about human behavior and in some cases may postpone the impacts of human behavior on the environment. In order to create longer-term solutions that can make technological strategies more efficient and effective, human behavior should be taken into account. The following sections discuss cognitive (modifying the use of biases and beliefs) and structural (modifications to the social context) strategies to addressing environmental degradation.

16.3.1 Cognitive Fixes

Solutions to commons dilemmas at the individual-level primarily fall under the umbrella of Heberlein's "cognitive fix." As noted earlier, humans are "boundedly rational" (Simon 1986, 1990), making decisions with a finite amount of cognitive resources and using only a part of the available information. A host of biases influence the decision making process, with loss aversion and discounting being particularly relevant to commons dilemmas, also known as social fences or traps (Platt 1973). As previously introduced, social fences are problems that require one to pay an immediate cost in order to enjoy a benefit sometime in the future. A social trap is the opposite, i.e., one enjoys an immediate benefit, but that benefit comes with a long term cost. As individuals are prone to discounting costs and benefits that occur in the future, and overweighting costs (i.e., losses) when compared to benefits (i.e., gains), there is a tendency to avoid immediate costs and discount the future costs that might otherwise be motivating. These tendencies make social fences difficult to overcome and social traps difficult to avoid.

The classic cognitive fix is to inform and educate (Heberlein 1974). However, more recent research in the context of environmental conservation and management suggests that individuals who clearly understand the benefits of a particular service or resource (see Zajac et al. 2012 for an example involving black bear conservation) or the benefits of a particular action (see Paveglio et al. 2009 for an example involving wildfire management) are more likely to support a conservation or environmental policy. Although educating the public is certainly a useful tool, such information must also be carefully framed in order to be most effective (Tversky and Kahneman 1981; Chong and Druckman 2007; Stapel and Semin 2007). For example, by taking

Table 16.2 Example message frames for presenting risk information aimed at reducing the psychological distance between an individual and a hazard

Dimension	Getting the audience to think concretely	Getting the audience to think abstractly
Temporal	X deaths per day	X deaths per year
Spatial	Provide a city scale map	Provide a national scale map
Social	You or your family	People
Hypotheticality	Probable	Possible

into account predictable decision biases (like loss aversion where losses are given more weight than gains), it is possible to influence people's perception of ecosystem restoration or preservation efforts, and ultimately their support for such efforts. As people are more driven to recover from a loss than pursue a gain, they will be more willing, for example, to support potentially risky efforts to "restore" some measure of lost forest health than they would be to "gain" the same measure of health in that same forest (Wilson et al. 2012). Thus, framing a message encouraging behaviors that will contribute to ES(s) in terms of restoring degraded conditions is more likely to be effective than simply describing benefits expected to result from taking action.

Research conducted over the last ten years in the field of psychology has also shown that it is possible to encourage individuals to put more emphasis on the long term outcomes of their decisions thereby reducing the amount they are discounted. This myopic focus is a key factor in explaining individual decision making among commons dilemma scenarios (Thompson 2000). In terms of social fences or social traps, people must be motivated to pay a cost (or forgo a benefit) in the short-term to realize a benefit (or avoid a cost) in the long-term. People naturally find this difficult to do because of the discounting bias, where the value of any costs or benefits which are to be realized in the distant future are discounted (decreasing the utility of options that focus on future conditions). As discussed earlier, construal level theory explains how a person viewing a situation concretely is more likely to discount those features which are abstract. But if the same person is viewing the situation abstractly, he or she will *augment* the weight placed on those abstract features (such as long-term goals, costs and benefits), thus negating the effects of the discouting bias (Trope and Liberman 2000; Fujita and Roberts 2010). Individuals can be encouraged through careful message framing to view the decision at hand more abstractly, influencing them to behave in a manner more in line with long term goals (see Table 16.2 for examples of message framing that vary from concrete to abstract across multiple dimensions). For example, if a forest manager determining harvest levels for a section of land views the decision as an incremental step towards a long term goal of sustainability as opposed to, say, the primary method of fulfilling the budget within the next fiscal year, they would be more likely to forgo a larger income in the short-term in order to ensure long-term timber production (Fujita et al. 2008; Liberman and Trope 2008; Torelli and Kaikati 2009).

Applying an ES(s) approach allows for a decision-maker to take current as well as future services into account while making benefits more concrete, and, thus, personally relevant. Creating payments to offset past or future activities, such as planting trees to sequester C, allows for decision makers to not have to forego a benefit today in order to ensure a long-term or distant future benefit. Without the legal or technological clout of Heberlein's other fixes, to influence individual decision making through a cognitive fix one must carefully frame the issue in an appropriate and context specific manner. If care is taken as to how that information is presented, it is possible to get more people to "see the bigger picture," and act more in line with their stated goals and values over time.

16.3.2 Structural Fixes

As mentioned previously, institutions can be thought of as social or political in nature. In terms of social institutions, norms can shift behavior in desired directions. A recent meta-analysis of psychologically-based interventions for inducing pro-environmental behavior indicates that social factors such as norms play an important role in modifying behavior (Osbaldiston and Schott 2012). The study examined data from 87 publications involving more than 250 experimental treatments that assessed changes in observed behavior. Interestingly, the study found that treatments that included a normative influence had a greater average effect size (g = .63) than those that provided incentives (g = .46) and approximately equivalent to monetary incentives (g = .68), meaning that the observed behavior can be motivated by monetary incentives and social norms. These results demonstrate the potential strength of normative influences in modifying behaviors and suggest that both social norms and monetary incentives should be working in concert in order to ensure that policies and programs are not subverted. These findings may be important when applying an ESs approach; if there are local social norms that are counterproductive, monetary incentives may not be always be sufficient to motivate an individual to implement conservation or restoration practices that enhance ecosystem services. If, for example, monetary incentives for implementing ES(s) practices were to be given only for a number of years, prevailing social norms counter to environmental stewardship may create social incentives to remove ES(s) practices. In order to create long-term, sustainable practices, policies and programs must establish monetary incentives and address underlying social norms.

Given the complex interactions between resource and social characteristics that influence the health of ecosystems, formal institutions can also play an important role by managing human behavior. Formal institutions provide a structure and process for developing collective agreements on how resources should be managed and distributed. Successful institutional arrangements are able to make simple and easily understood rules that are enforced with penalties reflecting the severity of the infraction (Agrawal 2002). Self-governance and control over the management of the resource has been shown to help sustain the resource and the population who depends on the resource.

One proposed solution to the tragedy of the commons and related environmental degradation is decentralizing rights and responsibilities for resource management from central governments to local communities. This process of decentralization is often a part of strategies commonly referred to as Community-Based Natural Resource Management (CBNRM). The logic behind CBNRM is that local communities are better situated for developing institutions that can incorporate local behaviors, norms, and contexts than a centralized government. This perspective challenges Hardin's emphasis on external government regulation. The idea behind decentralization and CBNRM is that (i) transaction costs for developing rules and for monitoring and enforcing those rules are lower in local communities, (ii) there are higher levels of trust and cooperation in smaller, more homogeneous communities, (iii) there is more detailed knowledge of local resource dynamics, and (iv) local communities are more responsive and can more quickly and efficiently respond to changes in resource conditions (Ribot and Agrawal 2006).

A CBNRM approach may help to reduce environmental degradation by creating nested institutions that address global environmental problems but limit noncompliance to international agreements by localizing regulation and enforcement. To address the need to understand the cooperation and coordination between global, national, and local institutions the Institutional Analysis and Design (IAD) framework was developed to show the connections between larger scale institutions and local implementation. Ostrom (1990, 2005; Ostrom et al. 1994) developed and described how different institutional scales coordinate to implement social goals through action arenas at the constitutional, national, and operational levels. The hierarchical framework structures interactions between institutional scales through rules which help to frame more localized agreements on how to reach the goal of distributing natural resources at the local level. The constitutional level outlines broader social goals while the local institutional level implements policies and management practices that should take into account the local physical and cultural context. The IAD framework and research can be a tool in developing institutions that manage the implementation of international agreements at the local level in a effective and efficient manner.

In a review of Common Pool Resource (CPR) research, Agrawal (2002) found there to be common characteristics of collective action institutions that facilitated sustainable resource use and management. Characteristics include rules that are easy to understand and locally developed, maintaining local access to the resource, ability to enforce the rules, graduated sanctions, availability of low cost adjudication, and accountability of monitors and other officials to users (see Table 16.3 for additional details).

For mobile resources such as oceans and atmospheric gases, international agreements are needed. Much like local institutions there must be institutional structures that create stable and predictable processes that can reach effective agreements. Unlike local institutional arrangements, international agreements must take into account their impacts on multiple levels of institutions as well as the quality and quantity of the resource. To be effective, international agreements can create incentives that reduce environmental degradation.

A program that is currently being used to address global climate change is the program *R*educed *E*missions from *D*eforestation and Forest *D*egradation *plus*

Table 16.3 Characteristics for Sustainable CPR Institutions (Agrawal 2002)

Characteristic type	Specific characteristics
Resource System	Well-defined boundaries
	Small geographic size
	Low level of resource mobility
	Predictability
	Ability to store resource benefits for future use
Group	Shared norms
	Small size
	Clearly defined boundaries
	Successful prior experiences- social capital
	Appropriate leadership
	Interdependence among group members
	Heterogeneity of interests and identities
Relationship between Resource System and Group	Geographic overlap between resource and user group location
	High levels of resource dependence of user groups
	Fairness of resource allocation
	Low level of resource demand
Institutional Arrangements	Rules are simple and easy to understand
	Locally devised access and management rules
	Ease of enforcement of rules
	Graduated sanctions
	Availability of low cost adjudication
	Accountability of monitors and other officials to users
Relationships between Resource System, Institutional Arrangement and External Environment	Match restrictions on harvests to regeneration of resources
	Recognition of legitimacy of local institutions from external institutions
	Central governments should not undermine local authority
	Appropriate levels of external aid to compensate local users for conservation activities
	Nested levels of appropriation, provision, enforcement, and governance
	Low cost of exclusion technology, time to adapt technology
	Low level of articulation with external markets

sustainable forest management, conservation, and enhancing the ability of existing forests to sequester C (REDD+). Under this program, developing countries and communities are paid through a variety of funding sources to manage their forests for C sequestration. Funding and involvement can come from the EU emissions trading scheme, the US carbon market, national governments, bilateral aid organizations, international NGOs, local NGOs, investment banks, and private companies (Wertz-Kanounnikoff and Angelsen 2009). Projects can be designed to provide payments to national governments, sub-national entities (e.g. local communities), or a combination of the two and, much like other recent conservation approaches, can rely on community-based approaches (Angelsen 2009). As such, REDD+ represents (i) international, inter-governmental agreements, (ii) payments for ES(s), and, (iii) decentralization of natural resource management to local-level institutions.

Projects may include strategies ranging from replanting local forests, requiring lower levels of fuelwood use, reducing forest clearance for agricultural expansion, preventing wildfires or restricting the scope and timing of burning, restoring hydrological systems in peat domes, or adjusting the length of cultivation and fallows in swidden agriculture systems (Sills et al. 2009). Despite the wide variety of approaches used, all projects share the goal of reducing C emissions or increasing C stocks in local forests.

Over 40 countries are developing REDD+ projects and hundreds of projects have already been initiated (Angelsen 2009). Despite the progress being made on REDD+ projects, a number of questions remain. Perhaps the most critical questions relate to monitoring, reporting and verification of project success. Because payments will may be made based on project outcomes rather than inputs, it is crucial that C reduction/sequestration is accurately measured and is monitored using consistent methods across projects. Other questions relate to the scale at which payments are made and concerns that local communities who bear the costs of restricted forest use will not see the economic benefits (Phelps et al. 2010), as well as uncertainty about what the global policy architecture for REDD+ will ultimately look like (Angelsen 2009).

While these and other questions remain, REDD+ projects in the early stages have proved promising. This global-scale payment-for- ES(s)-program combines economic incentives and, in some cases decentralized governance approaches, to combat global climate change, making use of several of the potential solutions introduced throughout the chapter. With sufficient attention paid to the politics and mechanics of REDD+ policies and projects, and an attention to international and intra-national equity, this approach could succeed where others have failed.

16.4 Conclusion

Solutions that attempt to protect or restore ecosystems and associated services will require taking into account the diverse influence of the human dimension. Ultimately, preventing environmental degradation and conserving ES(s) depends upon both a knowledge of ecological conditions and the psychological, social, economic, and institutional factors that influence human behaviors. The brief review of the theories and concepts presented here illustrates how such factors may contribute to the degradation of ecological resources but also demonstrates how a knowledge of the human dimension can contribute to improved ecological conditions. One of the continuing challenges of addressing human-environmental issues is the complexity of causes and projected effects between human activity and environmental responses. The theories discussed in this chapter provide a brief overview of the body of literature that has been applied to environmental problems. Although addressing human contributions to envrionmental problems will never be easy, this literature provides great insight for educators, managers and policy makers who want to avoid the potentially negative influence of social and behavioral factors and establish more effective and efficient institutions.

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Chapter 17
Soil Organic Carbon, Soil Formation and Soil Fertility

Thomas Gaiser and Karl Stahr

Contents

Abstract Soil organic carbon (SOC) plays a vital role in soil formation. The accumulation of SOC is one of the initial soil forming processes and is determined by physical, chemical, biological and anthropogenic factors with complex interactions. On the other hand, SOC and its composition influences other soil forming processes like leaching of cations, soil acidification, gleying including Fe-reduction and podzolization. As SOC is strongly correlated with soil organic nitrogen (SON) and nitrogen being the most widespread constraint for biomass production on cropland, SOC content and composition is a determining factor for soil productivity on well drained soils. Thus, SOC is an effective contributor to the supporting ecosystem services of soil formation on the global land surface and at the same time it positively affects the provisioning ecosystem services (ESs) for supplying food, feed and fiber.

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Keywords Soil forming factors • Carbon accumulation • Soil acidification • Podzolization • Iron reduction • Nitrogen supply • Crop yield

Abbreviations

C	Carbon
ESs	Ecosystem Service(s)
N	Nitrogen
OM	Organic matter
P	Phosphorus
SOC	Soil Organic Carbon
SOM	Soil organic matter
SON	Soil Organic Nitrogen

17.1 Introduction

Soil formation is an environmental process which transforms the geological substrate (the so-called parent material) into soil through the combined influence of physical, chemical, biological and anthropogenic factors. Jenny (1941) classified the factors of soil formation into the five categories (i) climate, (ii) parent material, (iii) topography, (iv) organisms (including humans) and (v) time (Fig. 17.1).

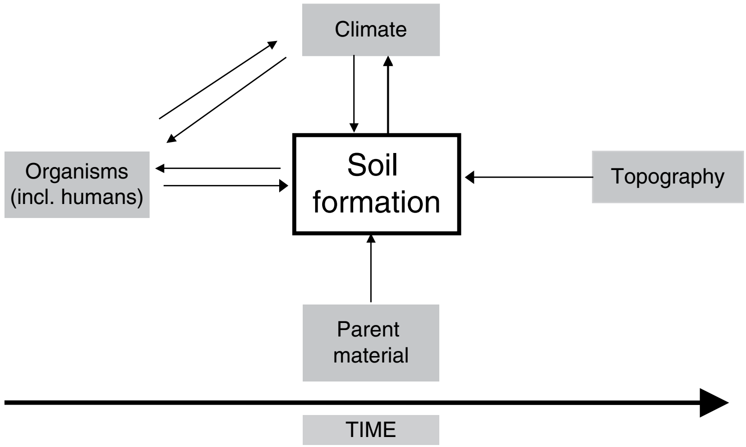

Fig. 17.1 Five determining factors for soil formation (*arrows* denote the major direction of influence) (adapted from Jenny 1941)

These factors drive many different soil forming processes which determine the development of soil properties over time. Thus, soil formation is the result of dynamic processes though at longer time scales (from decades to millennia). Accumulation of soil organic carbon (SOC) is among the initial soil forming processes and is, as such, subject to the factors of soil formation. For example, the organisms living in and growing on a soil determine, in interaction with climatic conditions and time, the amount of carbon (C) which is transferred from the atmosphere through C assimilation (i.e., photosynthesis), into the soil via root exudation, root residues and surface litter. At the same time, the amount and quality of SOC influences other soil forming processes.

17.2 Accumulation of Organic Carbon in Soils

The initial state of soil formation is characterized by the decomposition of the parent material through physical and chemical weathering. This process leads to the break-down of larger mineral structures into smaller and less solid fragments. A distinct pore system develops between these fragments which offers space for organisms including the roots of higher plants. If the parent material consists of non-consolidated material, a pore system is already in place, which allows colonization by soil biota and roots. As the organisms on and in the soil die, their biomass, which is composed of about 50 % C bound in organic components, remains in the soil and is subject to various chemical transformations. These include oxidation (in the presence of sufficient oxygen) or fermentation (in the absence of oxygen) catalyzed by different soil organisms. Roots of higher plants release exudates consisting of organic compounds which are easily decomposable. In all cases, some portion of the organic C is released as carbon dioxide (CO_2) to the atmosphere, whereas the remaining organic C is either incorporated into the biomass of the organisms or, in the case of some larger molecular organic compounds, may accumulate in the soil. Under natural conditions, accumulation of SOC is the result of an excess C input compared to C output, the C output consisting mainly of CO_2 as a product of oxidation of organic compounds metabolized by the soil biota. There are three dominant factors which cause a reduction of the C output and hence result in an accumulation of SOC. In mineral soils which are aerated for the major part of the year, the decomposition of organic compounds is reduced by physical or chemical protection against the decomposing activity of soil organisms (Von Lützow et al. 2006). Also, extremely high soil acidity may cause a reduction in decomposition rate. In organic soils which are saturated with water for more than 300 days per year, the decomposition of organic compounds is reduced by oxygen deficiency.

Organic soils which are almost permanently saturated with water are showing soil layers with high concentrations of organic C. Those soil horizons containing at least 12 % (w/w) of organic C are classified as histic horizons (H-horizon). In many cases the amount of organic C exceeds 30 % and organic layers may extend to a thickness of several meters. According to the World Reference Base for Soil

Fig. 17.2 Leptosol (*left*), Chernozem (*middle*) and Histosol (*right*) with contrasting humus horizons due to different patterns of carbon accumulation (World Soils 2002)

Resources, soils with an H-horizon are classified as Histosols (Fig. 17.2). They occur usually in the humid regions of the world where an excess of water in the soil is favored by higher annual precipitation compared to potential evapotranspiration. However, saturation of the soil can also be caused by shallow groundwater or stagnating water, which is the case in e.g. the permafrost zone, where water from the melting snow cannot percolate through the soil profile due to the impermeable permafrost layer in the subsoil.

Generally, in mineral soils, organic C accumulation is highest in the topsoil due to a denser root system and due to litter fall which, in the first instance, is incorporated into the surface soil. Thus, with increasing accumulation of SOC within a mineral matrix over time, a soil layer develops which is called the A-Horizon and which is distinctly different from the layers below due to its dark-grey to blackish color. Shallow soils (soil depth below 30 cm) that consist exclusively of an A-Horizon with underlying hard rock are classified as Leptosols (WRB 2006, Fig. 17.2). Leptosols are typical soils at the initial state of soil formation. As soil weathering proceeds to deeper layers or on non-consolidated parent material, the accumulation of organic C in the soil may continue. In continental climates with hot, dry summers and extremely cold winters, bioturbation is observed. The latter describes the incorporation of organic compounds into the subsoil through the activity of larger soil organisms (macro-fauna). Bioturbation favors the extension of the A-horizon into deeper soil layers forming dark-colored, humus-rich, well-structured soils which are classified either as Chernozems, Kastanozems or Phaeozems (WRB 2006, Fig. 17.2).

Human activities have brought along other types of SOC accumulation in mineral soils, which resulted from the application of large amounts of charcoal and ashes or organic and mineral material from livestock. The former case has

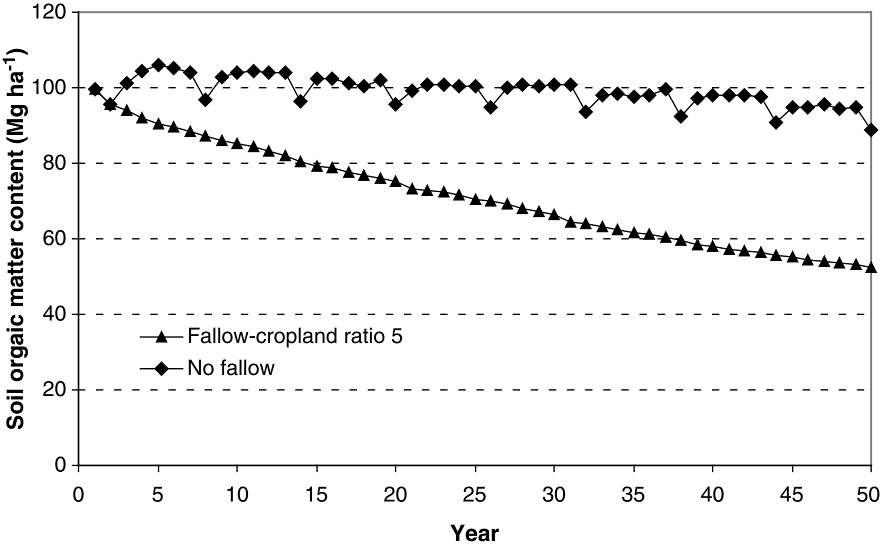

Fig. 17.3 Temporal evolution of soil organic matter content under permanent maize cropping (no fallow) and a fallow-maize rotation with 1 year maize (*Zea mays* L.) and 5 years of fallow (Fallow-cropland ratio 5) (Gaiser et al. 2010)

been observed in the Amazon basin where indigenous practices of soil fertility restoration have promoted the formation of so-called Terra preta soils. During medieval times, farmers in some regions of Northern Europe used to collect litter material and mineral soil in the forest, brought it home and mixed it with animal manure. Then, this mixture was applied in large quantities to the cropland leading to elevation of the plots and forming so-called Plaggen soils. Both soil types are known to be more fertile and productive compared to the associated soil types which have developed under natural conditions.

In peat bog areas under permanent water saturation, organic C accumulation may proceed over longer time scales, forming deep soils consisting almost exclusively of organic matter (OM). There, organic C accumulation slows down when the water saturation is reduced either by changing climatic conditions or when the H-horizon is elevated above the level of the ground or perched water table. In mineral soils, depending on climate, soil texture, groundwater influence and vegetation, SOC accumulation reaches steady-state conditions with an equilibrium between C input and output. Stable boundary conditions over a longer period of time, result in an equilibrium level of SOC in mineral soils. However, as soon as one of the boundary conditions changes, the equilibrium is disturbed and the SOC concentration will shift to a new equilibrium level. Due to the relatively fast break-down of organic compounds compared to their synthesis, reaching a higher equilibrium concentration requires more time than dropping to lower concentration (Fig. 17.3).

17.3 Interaction of Soil Organic Carbon with Soil Forming Processes

As soon as organisms start to colonize the soil the organisms as well as the accumulating organic C interact with various processes of soil formation. Polysaccharides produced by roots or by the excreta of earthworms promote soil structure formation, i.e., the aggregation of individual mineral particles. In addition, high-molecular humic acids create bounds with primary or secondary clay minerals to form stable organo-mineral complexes.

The breakdown of OM by soil biota produces large amounts of CO_2 in the soil which transforms into carbonic acid under the presence of water filled soil pores. Subsequently, carbonic acid causes a decrease in pH and an increase in proton concentration in the soil solution. The protons are in equilibrium with other cations like Ca^{2+}, Mg^{2+} and K^+ at the cation exchange sites of the soil matrix. The change in proton concentration leads to desorption of the cations from the soil matrix into the soil solution which makes them prone to leaching. Thus, SOC and its transformation processes promote soil acidification and leaching of neutral cations.

In contrast to soil acidification caused by organic acids and leaching we observe processes which generally buffer acids. There are two simple rules in the soil ecosystem. First, all oxidizing processes produce H^+ and cause acidification. Second, all reduction processes consume H^+ and raise the pH. How does it work? The oxygen dissolved in soil solution is used for respiration by roots, soil macro- and mesofauna and especially by microorganisms. Under water saturated conditions, respiration decreases the oxygen concentration in the soil and due to slow diffusion of oxygen in the soil water, diffusion is not able to replace the consumed oxygen at a sufficient rate. When almost all oxygen is used for respiration, specialists can use chemically bound oxygen for anaerobic respiration. To facilitate these processes, easily available organic soil material must be present. Then, SOC is oxidized and the oxidant is reduced as shown in the following equation using the example of iron oxides (Fe_2O_3):

$$C_6H_{12}O_6 + 4Fe_2O_3 \rightarrow 6CO_2 + 6H_2O + 8Fe^{2+}$$

The organic component could be sugar or other water soluble organic substances. As shown in the formula above, all these processes produce CO_2 and water. They also produce energy. This is the main principle for all aerobic or anaerobic respirative processes. Stable organic molecules like cellulose or lignin will not be mineralized under these conditions, but rather accumulate in the soil. Therefore, under those conditions, OM and thus SOC is enriched in soils. Quite often this is involving material belonging to the group of highly polymerized organic compounds with phenolic ring structures produced from organic matter transformation, but these compounds can also originate from physically or chemically protected litter components. When oxygen levels are too low, we observe a certain chain of substances (ions) which can be reduced by anaerobic reduction, like NO_3^-, Mn^{4+}, Fe^{3+}, SO_4^{2-}, CO_2, and H_2O. The corresponding products which will be reduced by the reduction processes are N_2O, N_2, Mn^{2+}, Fe^{2+}, S^{2-}, CH_4, and

H_2 (Stahr et al. 2008). There are many more ions, which can be involved in the reduction and oxidization chains, like Cu, Co or Mo. They generally do not play a significant role because of low concentrations or contents. With reducing and oxidizing processes, several other products, besides OM, are accumulated. This is iron oxide together with manganese oxide within a fluctuating ground water table (gleying conditions), but also sulfides can accumulate, particularly under permanent reduction. Beside these reaction products, other substances of high climatic relevance can be formed like N_2O and CH_4. All these soil processes can occur in different soil depths. Generally, the processes decelerate with increasing soil depth, because they need special easily available organic C sources. Soils which are temporarily saturated with water either by the influence of ground water (gleyic) or surface water (stagnic) can be very productive with favorable temperature regime and nutrient supply. Then they also tend to be rich in SOC. On the other hand, when these soils are poor, they tend to accumulate less SOC.

Podsolization is a process which cannot occur without contribution of SOC. However, it is completely different from the above described aerobic and anaerobic processes. Podzolization occurs under humid to per-humid climates in soils with free drainage and unbuffered conditions. While keeping an eye on the occurrence of those soils which are characterized by a spodic horizon being enriched in Fe, Al hydroxides and/ or SOC, we generally find these soils under either of the three following conditions: These are, first the boreal coniferous forests, secondly the alpine tree line, and thirdly the Fe-poor and extremely sandy soils in other humid climates. The podzolization process is one of the early soil forming processes, which has been described in the nineteenth century by the Russian father of soil science V. V. Dokuchaev. Podzols are soils which have an esthetically beautiful morphological appearance with a strongly leached pale albic horizon overlaying a dark, SOC-enriched, horizon as well as a bride reddish horizon colored by the Fe-oxides (Fig. 17.4).

Podzolization is a combination of leaching and translocation of SOC and cations. The humid conditions force a vertical transport of water. This process may dissolve cations like Na^+, K^+, Ca^{2+}, Mg^{2+}, and others through cation exchange and hydrolysis and leach them deep into the soil or even into the ground water and further away. When the conditions of Al-mobilization and later Fe-mobilization through decreasing soil pH are met, soluble organic C compounds will complex Al, Mn and later Fe, and transport it into lower horizons. The typical situation for a Podzol is that the mobilized materials are immobilized in the subsoil which may be only 20–40 cm deeper than their place of origin. With respect to translocation of matter, the process is very picturesque. It does not affect large amounts of Fe and SOC. Generally, the translocated SOC is 0.6–1.2 kg C m^{-2}, which means 6–12 Mg C ha^{-1}, while the Fe loss is often only 200–400 g Fe m^{-2}, which is 2–4 Mg Fe ha^{-1} (Stahr 1979).

At the scale of the horizons, the process is prepared by leaching of basic cations from soil solution and from exchange complexes. The pH drops down below pH 5. Then organic compounds which are water soluble and leached out from the litter above the mineral soil are forming chelates first with Al-ions. If the pH drops below 4, it additionally mobilizes Fe-oxides. The acidification is caused first by carbonic acid (H_2CO_3) later by organic acids like oxalic, citric or acetic acid, and finally by strong mineral acids like nitric or sulfuric acid. The ions of Fe and Al are liberated

Fig. 17.4 Podzol with typical sequence of organic carbon rich litter layer and mineral topsoil, followed by a grayish, leached intermediary horizon, a thin carbon enriched layer and an iron-oxide rich subsoil

by acid weathering, then bound in organic complexes and finally leached down into the sub-soil. There are multiple reasons for re-precipitation of these compounds. There is either polymerization of SOC in the subsoil, or simply the deceleration of the water fluxes, over-saturation of the organic complexes with Al and Fe or mineralization of OM by fungi. One of the interesting side-effects of podzolization is the non-stoichiometric transport. Some mobilized compounds are not immobilized in the spodic subsoil horizon but transported laterally below the soil profile or vertically out of the catchment (Sommer et al. 2001). As the Podzols are generally very poor and unfertile soils, they do not promote much turnover of organic material. Therefore, even a litter production of only 1 Mg ha^{-1} will gradually lead to an enrichment of SOC, especially when the litter is not readily decomposable. Therefore, the turnover time of SOC in Podzols is higher than in other mineral soils and may be up to 2,000 years (Stahr 1979).

17.4 Soil Organic Carbon and Soil Fertility

Soil fertility refers to the amount of nutrients in the soil which is sufficient to support plant life (Derek and Bogs 2009). SOC alone does not provide any essential nutrient to crops. However, since organic C is closely linked to organic compounds

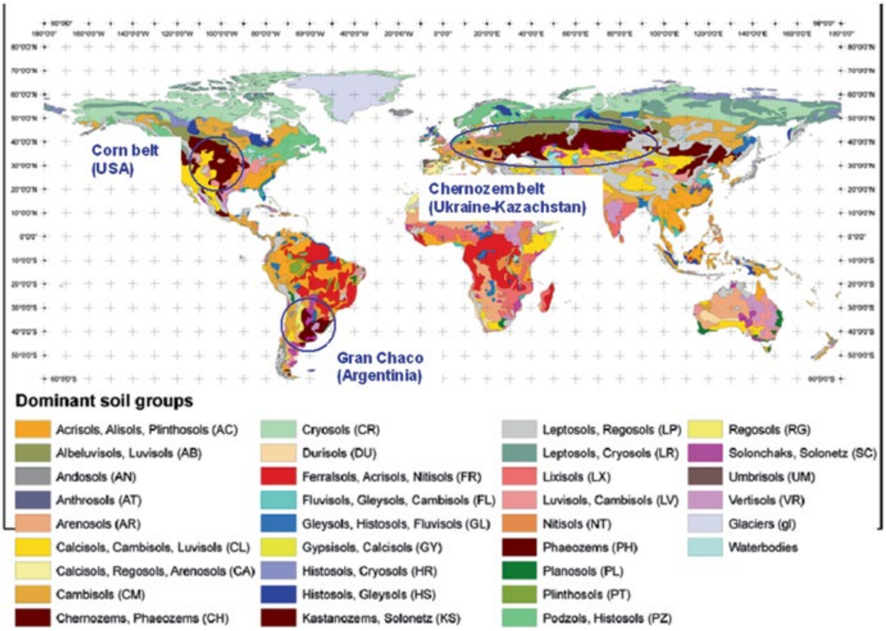

Fig. 17.5 Global distribution of Chernozem, Kastanozem and Phaeozem areas (*black* color) and major wheat growing regions (according to FAO/AGL 2003)

containing either N or P, there are close relationships between, e.g., SOC and soil organic N over a wide range of soils. This relationship is particularly valid for cultivated soils where the C:N ratio in the SOM is generally in a narrow and stable range between 10 and 15. Thus, large stock of SOC is equivalent to large amounts of organic N in the soil and hence high N availability to crops, under the assumption that N mineralisation rates are similar. In fact, at the global scale, well drained mineral soils with highest amounts of SOC storage tend to be the food baskets for quite a number of countries (Fig. 17.5). The corn (*Zea mays* L.) belt in Northern America coincides with regions where Chernozems and Phaeozems predominate. The same is true for the major wheat producing region in Argentina (Gran Chaco), as well as for the Chernozem belt stretching from Eastern Europe to Central Asia. The cropland in these regions is managed using large amounts of inputs especially fertilizers and in some cases water for irrigation, which may reduce the effect of N supply from soil organic N to the crops. Nevertheless, Bauer and Black (1994) report from long-term fertilizer experiments in the Great Plains (US) with 55–125 kg N ha^{-1} year^{-1} that 1 Mg of SOC increased wheat (*Triticum aestivum* L.) dry matter production by 35 kg ha^{-1}. On the other hand, in low input fallow systems of Western Africa with less than 5 kg N ha^{-1} year^{-1}, Gaiser (1993) observed an increase of 320–440 kg DM ha^{-1} by 1 Mg of SOC with comparable total SOC stocks. The tenfold impact on biomass production in West African cropping systems cannot be explained exclusively by the higher temperature or the physiological differences between wheat and maize. This illustrates the importance of the total amount of SOC on soil

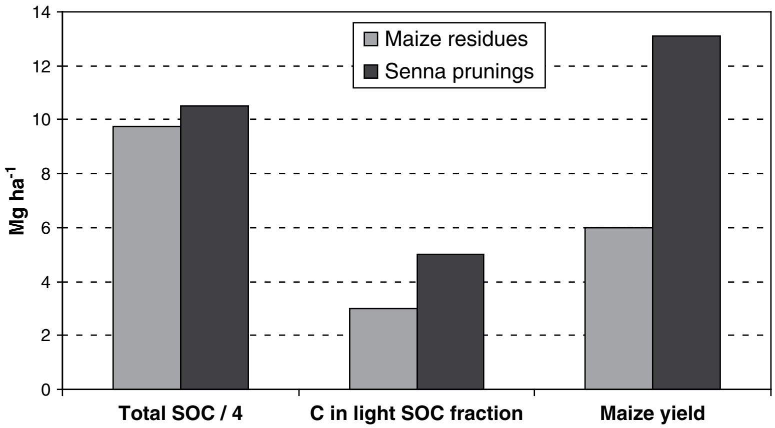

Fig. 17.6 Comparison between total soil organic carbon, the light fraction of soil organic carbon and maize dry matter production in a tropical Acrisol (based on data of Gaiser 1993)

fertility and crop productivity especially in low input systems, because they almost entirely depend on SOM as a source of available N for the crops.

However, besides the total amount of SOM its composition is also relevant for the amount of mineral N to be released during the cropping season. For organic amendments to the soil the C:N ratio is considered to be the major determinant regarding its effect on the subsequent crop. If C:N ratio of applied organic materials is above 20–30, there is a high risk of N immobilization following the first months after its application, causing reductions in N availability and yield (Parnas 1975). High amounts of organic manure may cause a change in SOM composition which results in changing N availability. In this case, total SOC is disconnected from N availability and hence its correlation with crop yields as shown in Fig. 17.6.

With almost the same amount of SOC, the soil amended with prunings of a leguminous tree (*Senna siamea*) produces more than twice as much maize dry matter than the one amended with maize residues. However, the light fraction of the SOC, which is defined as the floating organic material with a grain size fraction between 0.5 and 2 mm, is clearly correlated with maize dry matter production. This is supported by the fact that N mineralisation in these soils was closely linked to the amount of N in the light fraction (Fig. 17.7).

17.5 Soil Organic Carbon and Supporting Ecosystem Services

Soil formation and nutrient cycling are recognized as supporting ecosystem services Millennium Ecosystem Assessment (2005). Supporting ecosystem services are not directly affecting humans and the function of ecosystems but act through all other ESs because the other ESs (provisioning, regulating and cultural) depend partly or

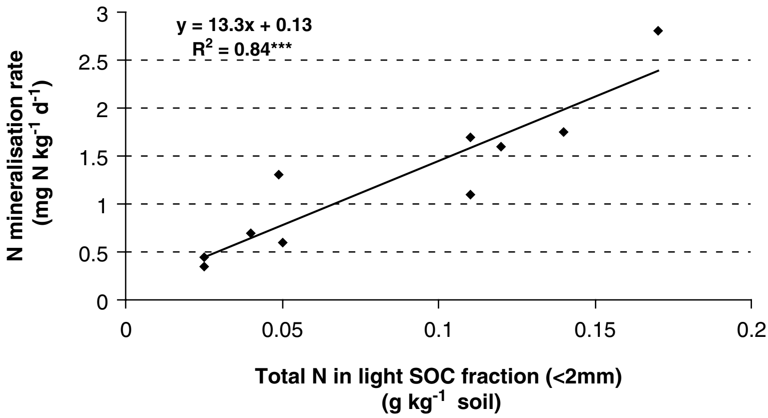

Fig. 17.7 Relationship between amount of nitrogen in the light soil organic carbon fraction and the nitrogen mineralisation rate (Gaiser et al. 1998)

entirely on the supporting ESs. In the previous sections, we have shown that SOC is an important driver in soil formation and nutrient cycling and thus, SOC plays a vital role in this supporting ecosystem service. Many soil types with their inherent properties are related to SOC accumulation like the Histosols. In the global C cycle, these soils act as C sinks, as long as they are not drained or their natural vegetation is not converted into arable land. In addition, Histosols are typical wetland soils and recognized hotspots of biodiversity. The same is true for Podzols, whose genesis strongly depends on the presence and composition of SOC. These soils with their specific properties are unique habitats and are rich in distinct above-ground and below-ground faunal and floral species, thus supporting ecosystem biodiversity. Other well-drained soils with SOC accumulation (Chernozems, Kastanozems) constitute the national breadbaskets of many countries and they support the provisioning ESs of specific biomes. Finally, it has been shown that SOC is an essential driver in the reduction of a series of soil compounds like nitrate, iron or manganese oxides, thus influencing the regulatory ESs in the global nitrogen and iron cycle and favoring the emission of green house gases from partly or temporarily anaerobic soils.

17.6 Conclusions

In conclusion, the interactions between SOC, soil formation and soil fertility may be summarized as follows:

- The factors of soil formation determine the intensity of soil formation and the amount of SOC stored in a soil
- SOC, in turn, influences many soil forming processes, thus leading to unique habitats for organisms both above- and belowground
- SOC is strongly influencing soil fertility in well-drained soils

The effect of SOC on crop productivity is mainly due to improved N supply to crops. However, the magnitude of the soil C effect depends on (i) the input intensity of the cropping system and (ii) the composition (or quality) of the SOM pools.

Acknowledgements We are grateful to Andreas Lehmann and Yakov Kuzyakov for providing soil profile pictures in Figs. 17.2 and 17.4.

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Chapter 18
Managing Soil Organic Carbon for Advancing Food Security and Strengthening Ecosystem Services in China

Mingsheng Fan, Jian Cao, Wenliang Wei, Fusuo Zhang, and Yansen Su

Contents

Abstract China's economy underwent great changes since 1949, especially since China initiated economic reforms and the open-door policy in the 1980s. The growth of agricultural production has been one of the main national accomplishments. By 1999 China was feeding 22 % of the global human population with only 7 % of the world's arable land. It has been acknowledged widely that the crop yield increase was accomplished by greater inputs of fertilizers, irrigation, new crop strains, and other technologies of the "Green Revolution". However, soil quality improvement in most of the arable land, indicated by increased soil organic carbon (SOC) concentrations, in return might result in an increased crop yield. Looking towards 2030, further increases in crop production on the remaining arable land to meet the demand for grain and to feed a growing population will be more problematical than it has been for the last 50 years. The availability of high quality soil is one of the

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major limiting factors in China. Thus, we propose that the advancement of food security in twenty-first century in China will depend on a continuous improvement of soil quality. The approach to improve soil quality, accomplished by management of SOC may provide multiple benefits, it improves agronomic productivity, reduces the net rate of atmospheric carbon dioxide (CO_2) enrichment and it supports other ecosystem services.

This paper summarizes the trends in crop production and crop yields and reviews changes of SOC in most arable land and its possible effects on soil quality and crop yields in China. In addition, the approach of SOC management for an advancing food security in China and for strengthening other ecosystem services in the new century will be discussed.

Keywords Soil organic carbon • Food security • Ecosystem services • Carbon sequestration • Environmental protection

Abbreviations

C	Carbon
SOC	Soil organic carbon
CO_2	Carbon dioxide
N	Nitrogen
N_2O	Nitrogen dioxide
NCP	North China Plain
P	Phosphorus
S	Sulphur
Zn	Zinc

18.1 Agricultural Intensification, Soil Organic Carbon and Environment in China

18.1.1 Increased Crop Production and Yields Since 1960

China's agricultural production has experienced a remarkable growth in the last 50 years (National Bureau of Statistics of China 1950–2010). The net increase in cereal production was 391 Tg with an annual growth rate of 3.7 % from 1961 to 2009 (Fig. 18.1a), which is substantially higher than the world mean growth rate in cereal production of 2 % during the same period. In 2009, China was providing about 29 % of global rice (*Oryza sativa*) production, 20 % of maize (*Zea mays*) and 17 % of wheat (*Triticum astivum*) production (National Bureau of Statistics of China 2010; FAO 2010). The success of crop production in China resulted in the so-called

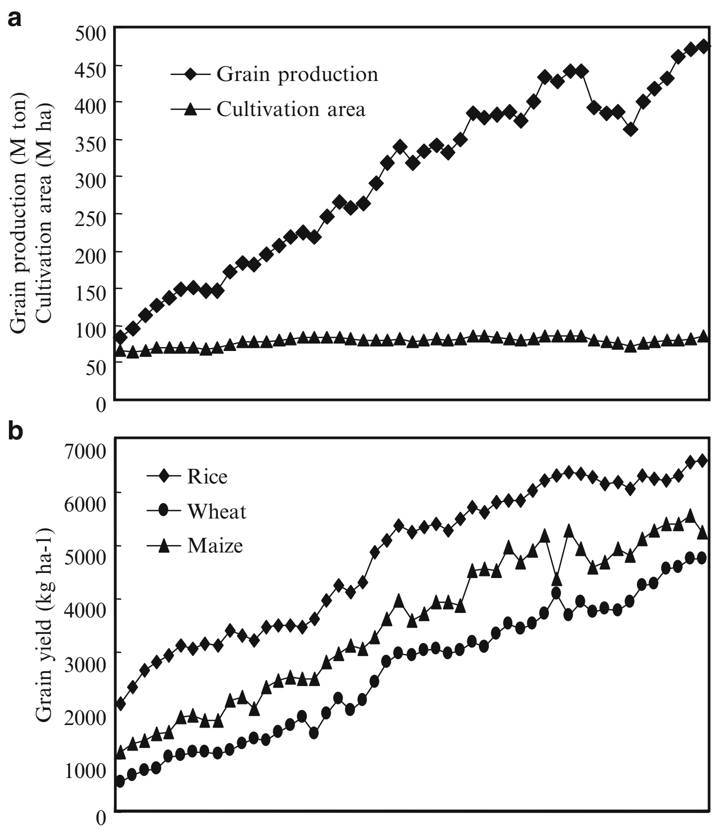

Fig. 18.1 Production and cultivation areas of cereal crops (rice+wheat+maize) (**a**), and grain yields of rice, wheat and maize (**b**) in China from 1961 to 2009 (Source: National Bureau of Statistics of China 1950–2010) (Adapted from Fan et al. 2012)

'Miracle in China' with 7 % of the world's arable land feeding 22 % of the world's population, and impacted on both global food supply and on the use of natural resources, and these changes have gained global attention (Fan et al. 2012).

The increase of crop production in China was caused mainly by an increase of yield per area rather than from an increase of the cultivated area. For example, from 1961 to 2009 there was a 3.2 fold production increase of rice, an 8.5 fold production increase of wheat, and a 4.6 fold increase in the production of maize (Fig. 18.1b). In the same time, the total cultivated area of cereals increased by only 30 %, e.g. from 65.5 Mha in 1961 to 85.1 Mha in 2009 (National Bureau of Statistics of China (1950–2010), Fig. 18.1a).

The intensification of the crop production over the last 50 years has been recognized as the 'Green Revolution', and it has been achieved by using modern high-yielding varieties, greater inputs of chemical fertilizers, irrigation and weed and pest control (Fan et al. 2012). The total consumption of chemical fertilizer in China (calculated using production + imports–exports) has increased linearly since 1961. Total consumption of chemical fertilizers exceeded 64 Tg in 2009, and this was almost 35 % of the total global fertilizer consumption. The area of irrigated farmland has expanded by 32 % since the 1970s, and has now reached 58.5 Mha (National Bureau of Statistics of China 1950–2010). Chemical use in agriculture, such as application of pesticides and herbicides increased from 0.76 Tg in 1991 to 1.76 Tg in 2005 (National Bureau of Statistics of China 1950–2010). Without the use of chemical fertilizers, irrigation and agricultural chemicals, China's food production could not have increased at the rates that have been recorded.

18.1.2 Soil Quality, Soil Organic Carbon and Crop Yields

Despite the effect of agricultural intensification on cereal crop yields over last 50 years. Crop productivity is strongly depending on soil quality. The yield decline phenomenon that has occurred in a number of long-term experiments with annual double- and triple-crop irrigated rice systems is an example that small changes of soil properties can have a large impact on productivity (Cassman and Pingali 1995).

Soil organic carbon (SOC) is considered to be a key attribute of soil quality (Larson and Pierce 1991; Gregorich et al. 1994) and also environmental quality (Smith et al. 2000). The enhancement and maintenance of SOC and soil quality are fundamental to ensuring global food security (Lal 2004). The present understanding asserts that SOC content is positively correlated with soil quality, and higher grain yields tend to occur in soils with higher SOC content. For example, an increase of 1 Mg ha^{-1} of SOC increased wheat grain yield by 27 kg ha^{-1} in North Dakota, U.S.A. (Bauer and Black 1994), by 40 kg ha^{-1} in the semi-arid pampas of Argentina (Diaz-Zorita et al. 2002), and by 17 and 10 kg ha^{-1} in maize in Thailand and western Nigeria, respectively (Petchawee and Chaitep 1995; Lal 1981, 2008).

A recent study in China evaluated the changes in SOC at the national scale over the previous two decades. For example, Huang and Sun's (2006) calculations for each region indicated that the SOC in more than 60 % of soil samples or monitoring sites increased in the East, North, Northwest, South and in central China. About half of the soil samples and/or the monitoring sites showed an increase in SOC in the Southwest region. This was attributed to an amendment of crop residues and organic manures on soils combined with synthetic fertilizer applications and the optimal combinations of nutrients and the development of no-tillage and reduced-tillage practices, which were calculated to contribute to the increase by 76, 22 and 2 %, respectively. This conclusion is supported by other studies in China (Xie et al. 2007; Lu et al. 2009). Further investigations showed that the topsoil (0–20 cm) SOC of croplands in China's mainland increased by 358–463 Tg C, equivalent to about a 30 % increase in SOC concentration from 1980 to 2000 (Huang and Sun 2006).

The increase in SOC in most of the arable land in China indicates soil quality improvement and consequently may lead to increases in cereal crop yields. For example, Pan et al. (2009) showed at the national scale that a 1 % increase in SOC

Table 18.1 Contributions of increased soil organic carbon (SOC) to wheat and maize yields in fertilizer omission plots on-farm at the North China Plain (NCP)

	Wheat	Maize
Region	ΔYield-control/ΔSOC	ΔYield-control/ΔSOC
Henan	517	324
Hebei	540	793
Shandong	598	960
Beijing and Tianjing	749	789
Whole NCP	577	693

Note: ΔYield-control represents the changes in yield-control during the 1980s and 2000s, which are the differences of yield-control between1980s and 2000s [kg ha^{-1}] ΔSOC represents the changes in soil organic Carbon (SOC) content [g kg^{-1}]

on average would lead to an increase in total cereal productivity of 0.43 Mg ha^{-1} and a decrease of yield variability against disturbance by 3.5 %. Recently, Su (2012) evaluated changes in wheat and maize yields in plots without fertilizer addition on the farm scale in several provinces of the North China Plain (NCP) since 1980. Both yields in plots without fertilizer addition and SOC showed significant increase between 1980s and 2000s, ranging from by 2.91 to 4.57 g kg^{-1} for SOC and 811 to 2,516 kg ha^{-1} for yields with fertilizer omission. Improvement in SOC concentration increased yield for treatments without fertilizer addition by 331 kg ha^{-1} for wheat, 398 kg ha^{-1} for maize since 1980s, respectively (Table 18.1). The positive effects of the increase of SOC concentration on crop yields may be attributed to improvements in soil aggregation and tilth, better root system development, higher water-holding capacity, a higher water infiltration rate, and a better availability of essential plant nutrients such as Phosphorus (P), Sulphur, (S), and Zinc (Zn) (Lal 2008; Johnston et al. 2009).

18.1.3 Environmental Consequences of Agricultural Intensification

Despite the achievements attained in agricultural production, agricultural intensification is increasingly deteriorating China's environment (Fan et al. 2012). For example, increase of fertilizer nutrient input had significant effects on crop yields in China; however it has been realized that fertilizer overuse and high nutrient loss resulting form inappropriate timing and methods of fertilizer application, especially in high yielding fields have caused serious problems. Annual synthetic fertilizer N-induced nitrogen dioxide (N_2O) emission from China's croplands which has increased from 120 Gg N_2O-N year^{-1} in the 1980s to 210 Gg N_2O-N year^{-1} in the 1990s (Zou et al. 2010). Another case study showed that soil pH in the major Chinese crop-production areas has declined significantly from the 1980s to the 2000s because of excessive N fertilizer inputs (Guo et al. 2010). Agricultural production accounts for 30 % of the national total CO_2 emission (Weifeng Zhang, personal communication 2012). Losses of N and P through leaching and run-off have led to drinking water pollution which affects 30 % of the population and results in eutrophication of 61 % of lakes in the country.

Water pollution and increased water shortage associated with overuse of surface water is threatening the sustainability of agricultural production. Annual water shortage in agriculture amounts to 30×10^{10} m^3 in China. By 2030, China's total water deficit could reach 130×10^{10} m^3 (Li 2006). The outlook for water shortage is especially dire on NCP. This plain comprises 33.8 % of the national arable land, but has only 3.85 % of the national water resources. Over the past 40 years, NCP's water table has fallen steadily as some 120×10^{10} m^3 more water has been used from the land compared to the amount replaced by rainfall (Li 2010).

18.2 Managing Soil Organic Carbon for Advancing Food Security and Strengthening Ecosystem Services

18.2.1 Continuous Improvement of Soil Quality Enhances Food Security

To meet the future demand for grain and to feed a growing population on the remaining arable land in 2030, annual crop production must reach 580 Tg and yield in China has to increase by 2 % annually over the next 20 years (Fan et al. 2010). However, further increases in crop production will be even more problematical than has been the case for the last 50 years. The availability of high quality soil is one of the major limiting factors in China. Agricultural input needs to be reduced; especially N and P fertilizer overuse which have led to environmental problems in China.

A long history of arable farming and steady increases of human population have led to a depletion of arable land reserves in China (Li and Sun 1990). The per capita arable land area is 0.1 ha at present, which is 45 % of the world average (Wang et al. 2009). China has used almost every piece of available land for agriculture: from favorable to marginal land (Fig. 18.2). The potential to increase the cropping area is therefore limited in the future, and more food will need to be produced from the same amount of (or even less) land. Now, it is clear that it will become more important to adopt technological and policy measures to improve the sustainability of agriculture as well as to increase grain yield per unit area of arable land (Fan et al. 2012).

To enhance crop production in China with efficient resource utilization, improvement in soil quality is crucial, because most of the arable land in China has a poor soil quality. Furthermore, some of the arable land, such as slopes on the Loess Plateau of the Northwest and small lots of land in the mountainous area of the Southwest are not suitable for agricultural production, as shown in Fig. 18.2 (from (d) to (f)). Moreover, in the Northeast of China, which is the main agricultural production area with relatively fertile soils, grain yields in low productivity soils

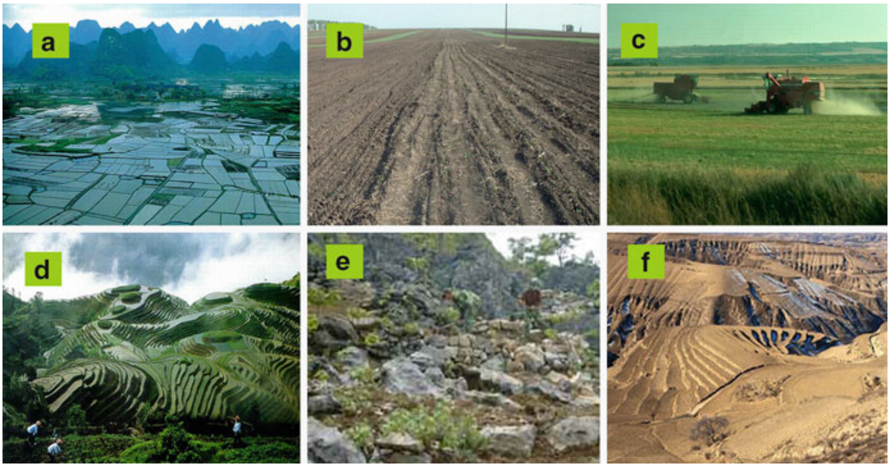

Fig. 18.2 Crop land types in China. (**a**) Paddy soil in the South; (**b**) upland in the North China Plain; (**c**) upland in the Northeast; (**d**) terraces in Southwest; (**e**) small lots of arable land in the mountainous area of the Southwest; (**f**) slopes on the Loess plateau, Northwest China

were less than 1,500 kg ha^{-1}, but the corresponding average value was 7,595 kg ha^{-1} on highly productive land (Fan et al. 2010). The areas of high, medium and low productivity account for 28.7, 30.1 and 41.2 % of the total arable land in China, respectively (Wang 2005). Fan et al. (2012) suggested that existing knowledge and technology can improve management practices of farmers to a certain extent, but will unlikely lead to an increase in production to the level that is needed to double food production by 2030. Greater advances in crop production, which must depend on continuously improved soil quality, improved crop management and improved crop varieties, are needed during the next 20 years to ensure substantial increase in cereal yield to achieve food security (Fig. 18.3). Improvements of soil quality especially in those areas with favorable and semi-favorable growth conditions may lead to an increase of agronomic production. As a result, marginal land could be saved for natural vegetation, which provides great potential ecosystem services such as biodiversity conservation and C sequestration.

18.2.2 Soil Organic Carbon Management for Food Security and Ecosystem Services

The approach to improve soil quality, accomplished through management of SOC in soils is especially important: It improves agronomic productivity, reduces the net rate of enrichment of atmospheric CO_2 (Lal 2004), and serves other ecosystem services such as water purification, flood mitigation and biodiversity conservation. Despite of the increase in SOC in croplands since 1980s (Xie et al. 2007; Huang and Sun 2006; Lu et al. 2009; Piao et al. 2009), the average content of SOC in topsoil from cropland

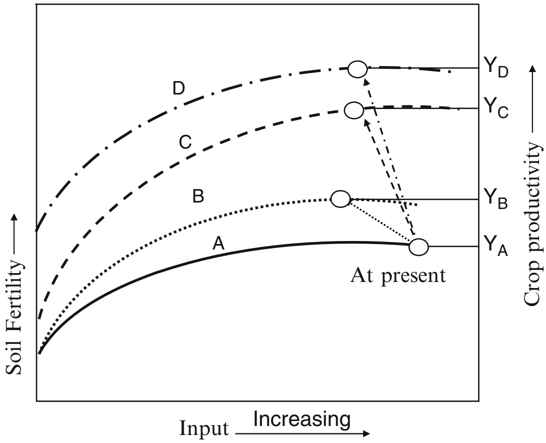

Fig. 18.3 Conceptual model for optimal crop production to achieve synchronously increasing crop productivity, improving resource use efficiency and protecting the environment in China. *A*, the current status in crop productivity on farm fields; *B*, scenario of crop productivity upon application of the existing technologies; *C*, scenario of crop productivity upon improved soil and crop management such as integrated soil-crop systems management, in existing crop varieties; *D*, scenario of crop productivity upon improved soil and crop management and improved crop varieties (Adapted from Fan et al. 2012)

is 10 g kg^{-1} in China compared with 25–40 g kg^{-1} in European countries and the United States (Fan et al. 2010). Furthermore, the loss of the SOC pool has been widely reported in China's croplands. Huang and Sun (2006) estimated that SOC in 31.4 % of monitoring sites in China experienced some loss due to water deprivation and soil erosion combined with low C inputs. In the Northeast region of China, SOC decrease was observed in 74 % of total soil samples and/or monitoring sites. Huang et al. (2000) estimated that 45.6 Mha were affected by severe erosion on the Loess Plateau, representing 70 % of the total area of the Loess Plateau. Each year 1.6 Pg (1 Pg = 10^{15} g) of sediment are transported into the Yellow river (Wang et al. 1991). The loss of equivalent topsoil depth is 0.2–2.0 cm. These sediments contain 4.18 g kg^{-1} of SOC (Wang et al. 2001). The SOC sink capacity in cropland in China can therefore be greatly enhanced by effective erosion management and restoration of degraded soils, potentially by 14–28 and 14–28 Tg C $year^{-1}$, respectively (Lal 2002).

18.2.3 Technologies and Policy Issues Regarding Soil Organic Carbon Management

Improving the recycling of agricultural by-products (i.e., animal and human wastes, crop residues, green manure, and city sludge) is essential to improving natural resources, enhancing SOC, and adapting to a changing climate. Novel and innovative soil management practices (i.e. biochar addition, conservation tillage etc.) should be validated, fine-tuned, and promoted for site-specific conditions (Fan et al. 2012). For example, incorporation of biochar may represent a measure for sequestrating C and there is increasing evidence that although there may be some negative effects of incorporation, it may also reduce nutrient leaching and impact positively on the slow release of nutrients to enhance crop yields (Marris 2006; Lehmann 2007). Until now, zero-till or reduced till practices have rarely been practiced in China but have reportedly allowed sustained yields with largely positive effects on ecosystem services. These practices may be especially beneficial on low productivity land with low SOC content. Furthermore, a comprehensive strategy is needed, which integrates soil quality management through SOM improvement and C sequestration into the intensification process.

Policy interventions are needed to promote adoption of SOC pool management technologies including those which involve payments for ecosystem services. Due to small-scale farming, economic benefits derived from SOC management practices generally may not translate into economic incentives which induce farmers to adopt these technologies voluntarily. Therefore, incentive measurements such as subsidies may be useful to encourage farmers to adopt C farming technologies and change inappropriate management practices for soil C conservation.

18.3 Concluding Remarks

China's agricultural production has experienced a significant increase since the past half century. However, agricultural intensification is increasingly deteriorating China's environment. Soil quality improvement for most of the arable land in China may play an important role for achieving an increased agricultural production. Thus, advancing food security without environmental integration in twenty-first century, China must adopt a strategy towards continuous improvement of soil quality. Managing SOC may provide multiple benefits for soil quality improvement and other ecosystems services. Because of the strong interaction amongst several factors on agronomic yield (e.g., improved varieties, crop management, climate change, infrastructure, and market), additional research is needed to assess the net impacts of soil quality on crop yields and total agronomic production. It is also important to assess quantitatively ecosystem services function derived from SOC sequestration on arable land. This will help to evaluate how we may better intervene to achieve food security and strengthen ecosystems services in the near future.

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Chapter 19
Research and Development Priorities for Global Soil-Related Policies and Programs

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Abstract While scientific principles for sustainable soil management are known, a widespread adoption of recommended management practices (RMPs) has not happened, especially in the developing countries. Economic disincentives and constraints, and lack of knowledge and of capacities to implement sustainable soil management are fundamental causes. As a result, there are severe problems of soil and environmental degradation, low agronomic productivity, and malnutrition and hunger globally affecting >1 billion people. Appropriate policy interventions are needed to incentivize the process of the stewardship of natural resources while enhancing soil-related productivity. Related ecosystems services (ESs) [i.e., for carbon (C) sequestration, water resources management, biodiversity conservation] are undervalued. Payments to land managers for provisioning and enhancement of ecosystem services can be important part of promotion of conversion to a restorative land use and adoption of RMPs. Appropriate payments based on a fair price and transparent system, should be made not only for the tangible goods, but also for non-materialistic benefits (e.g., cultural, recreational, spiritual, and aesthetical services). Prioritization and implementation of research, development and outreach programs at local, regional, and global scales are essential to sustainable management of soils and other natural resources. Researchers and 'practitioners' dealing with the finite resource soil must cooperate in a transdisciplinary way to tackle the complexity of social-ecological issues associated with the management of soil in relation to other natural resources.

Keywords Ecosystem Services • Science-Policy Interface • Provisioning Services • Cultural Services • Regulatory Services • Policy Interventions

Abbreviations

C	Carbon
ESs	Ecosystem Services
Gt	Gigaton
GHG	Greenhouse gas
MDB	Murray Darling Basin
NPP	Net primary productivity
N	Nitrogen
NT	No-till
RMPs	Recommended management practices
SOM	Soil organic matter

19.1 Introduction

Through interaction with hydrosphere, atmosphere, biosphere, the management of the pedosphere (Fig. 19.1) creates numerous ecosystem services (ESs) (i.e., provisioning of food, water) and support ecosystem functions. The primordial

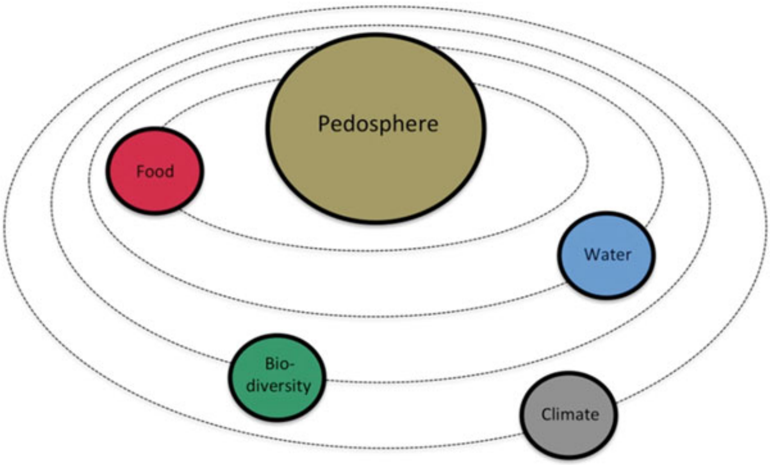

Fig. 19.1 Ecosystem services provided by soils

bond between man and nature needs to be explored (Schneider and Morton 1981) so that humans can live in harmony with nature. Soil, an important component of nature and the essence of all terrestrial life, plays a crucial role in the provisioning of goods of interest to humans, and also generates numerous ESs essential to functioning of nature (Tallis and Kareica 2005). Any drastic reduction in ESs can lead to societal collapse (Diamond 2005). Thus, there is a need to understand the scientific basis of environmental functions by unifying the concepts of ecology and economics (de Groot 1987, 1992), and implementing these concepts to address practical problems facing humanity. The goal is to go beyond the Millennium Assessment (Carpenter et al. 2009), and to measure, analyze and manage ESs (Kremen and Ostfeld 2005). However, diverse interests of individual stakeholders and disciplines may not be balanced when it comes to ESs and their sustainable management (Berkes 2008). A transdisciplinary approach involving mutual learning between scientists and external stakeholders is needed to support joint problem solving towards sustainable management of ESs (Lawrence and Després 2004).

With ever increasing global population and burgeoning of its demands, a judicious management of soil is essential to sustaining soil quality and productivity. Drastic anthropogenic (and natural) perturbations can lead to strong alterations in the provisioning of ESs, and even to several disservices and tradeoffs. These perturbations include deforestation, draining of peatlands, residue removal, tillage and the attendant increase in risks of soil erosion, indiscriminate use of agrochemicals, agricultural intensification based on monoculture, and uncontrolled/excessive grazing (Fig. 19.2). Urbanization can also have significant negative impacts on ESs on local and regional scales as cities are often located in highly productive agricultural areas (Bairoch 1988). Soil degradation is especially a serious problem in the tropics and sub-tropics (Stocking 2003) where soils are fragile and the climate is harsh. The institutions are weak and infrastructure is poor. Furthermore, the resource-poor farmers cannot invest in soil/land restorative measures, such as the use of fertilizers, other soil amendments, and supplemental irrigation. Thus, nutrient depletion and the negative elemental budget are serious issues which adversely affect agronomic productivity (Stoorvogel and Smaling 1990; Vitoustek et al. 2009). Removal of crop residues, for numerous completing users, such as biofuel production (Somerville 2006), can exacerbate soil erosion on sloping lands (Harrold and Edwards 1972), reduce soil organic carbon (SOC) concentration (Sandhu et al. 2010), and jeopardize soil quality (Doran et al. 1984). These perturbations can cause drastic alterations and adversely impact quality of soil, water, air, and biodiversity.

Thus, the objective of this concluding chapter is to discuss tradeoffs among ESs, and identify research, development and outreach strategies of soil management to enhance and sustain ESs and functions. The goal is to identify research/development/outreach priorities, policy interventions, and a road map for their implementation.

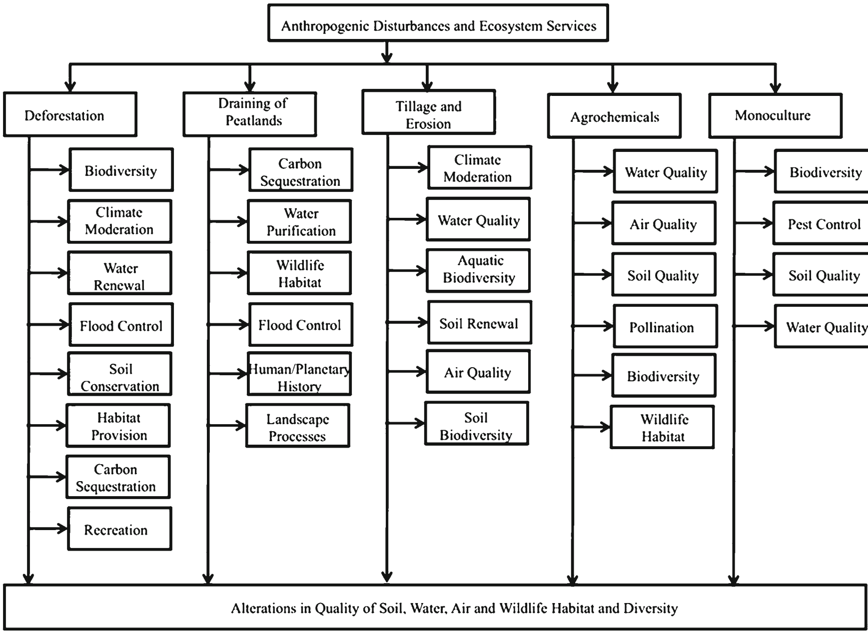

Fig. 19.2 Anthrophogenic disturbances and their impacts on ecosystem services

19.1.1 Tradeoffs Among Ecosystem Services

There exists a strong inter-connectivity among different ESs. For example, attempt to enhance food production through agricultural intensification can strongly impact water resources which are already a scarce resource (Postel 2000), and are likely to be constrained even more by the global warming (Brian et al. 2004). Water resources are also in greater demand because of the increasing emphasis on biofuel production (Ulhenbrook 2007), especially for growing of warm season grasses with irrigation during summer (Robins et al. 2009). Thus, there is a need to adapt to a right strategy of irrigation of biofuel plantations (Bauder 2009) so that the effects of drought stress and water deficit are minimal (Mastrorilli et al. 1999).

There is a strong link between the above-and-below-ground biodiversity (Wardle et al. 2004). Both of these are crucial to several ecosystem functions (Brussaard et al. 1997; Wall and Virginia 2000; Loreau et al. 2001; Hunt and Wall 2002; Ritz et al. 2003), processes (Wall and Moore 1999), and ESs (Swift et al. 2004).

Attempts to enhance provisioning of a specific ESs can lead to several disservices or tradeoffs (Fig. 19.3). Notable among disservices caused by intensification

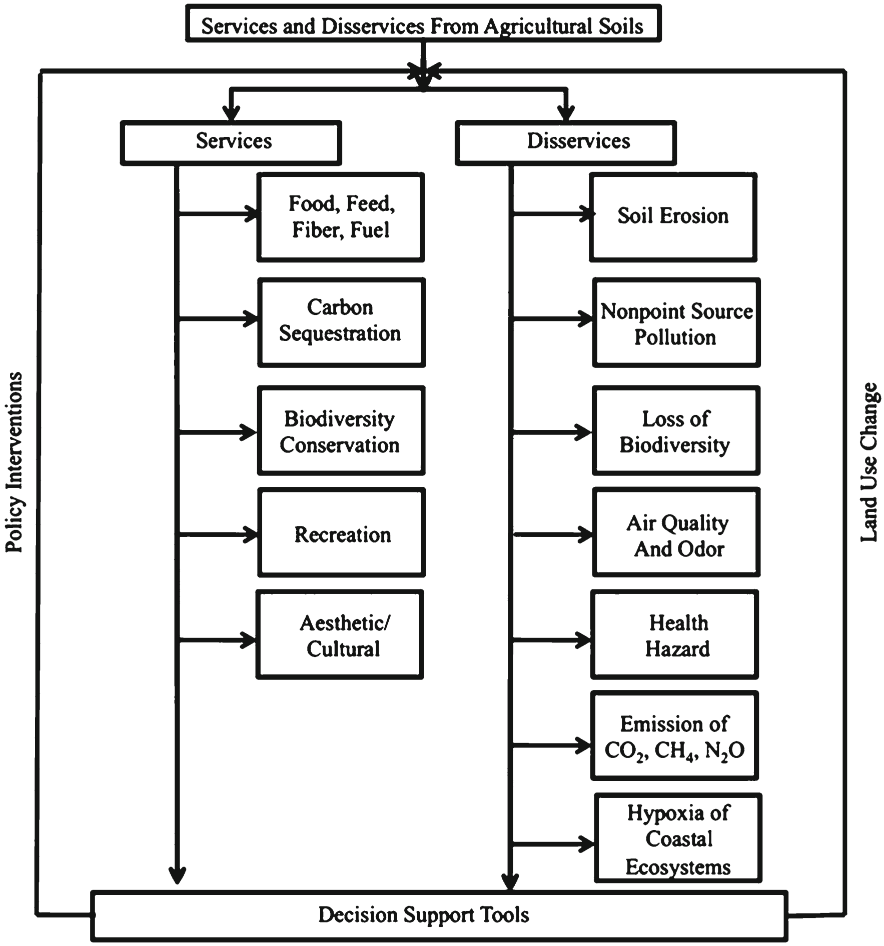

Fig. 19.3 Impacts of soil degradation on ecosystem services

of agroecosystems are accelerated erosion, non-point source pollution, and loss of biodiversity. Thus, the goal is "sustainable intensification" to enhance productivity while maintaining an improving quality of soil/land and the ESs that they provide.

19.1.2 Soils and Climate Change

Agriculture plays an important role in the climate change (Tilman et al. 2001). While world soils have a large carbon (C) sink capacity, the soil-based strategy of mitigating climate change by off-setting anthropogenic emissions has its own limitations (Lal 2008). Yet sustainable management of terrestrial ecosystems (Obersteiner et al. 2010), and sustainable intensification of agriculture are important options (Royal Society 2009) which must be critically and objectively assessed. Indeed, a widespread adoption of improved agricultural practices has to be provided to both mitigate and adapt to climate change (Smith et al. 2008).

Adoption of no-till (NT) system or conservation agriculture has been widely discussed as an option to mitigate climate change through enhancing SOC sink capacity (West and Post 2002; Lal 2004a; Smith et al. 2008; Oorts et al. 2007; Six et al. 2002). Yet, there are questions about the net gains in SOC and nitrogen (N) stocks because of differences in surface soil *vs*. sub-soil accumulation of SOC under NT *vs*. conventional tillage and emission of nitrous oxide (N_2O). Several experiments have documented differences in N_2O emissions in NT *vs*. conventional tillage systems (Kandeler et al. 1999; Dalal et al. 2003; Baggs et al. 2003; Ball et al. 2008; Chatskikh et al. 2008; Rochette 2008). In general, N_2O emissions are more under NT than conventional tillage systems. Similar to residue management and tillage methods, there are also hidden C costs and a large ecological footprint of supplemental irrigation (Lal 2004b). Whereas use of incorporating cover crops in the rotation cycle is important to enhancing SOC and improving soil quality with NT system, fallowing also affects emissions of GHGs (Liebig et al. 2010), and must be accounted for.

19.1.3 Payments for Ecosystem Services

Challenged by ever increasing consumer demands of the growing and richer world population, sustaining Earth's life support systems (Moore 2000) is a high priority. Adoption of improved technologies can be promoted, among resource-poor farmers and small landholders and large scale commercial farmers alike, through payments for provisioning of ESs (Ferraro 2011; Arriagada et al. 2011). The risks of severe soil degradation, the so called 'death of nature' (Merchant 1980), can be minimized by governing the commons through collective action (Ostrom 1990) through conversion to restorative land use and adoption of recommended management practices (RMPs) through incentivizing by payments for ESs and by disincentives for non-sustainable soil management practices. Such programs towards the enhancement of the environmental effectiveness have been proposed for developing countries (Alix-Garcia et al. 2010). Transdisciplinary collaborations must be developed to find ways to enhance ESs, minimize disservices and advance sustainable management of ecosystems. Some have argued that the organic farming can be promoted through payments for ESs (Sandhu et al. 2008) but its productivity effects remain disputed. Also, conservation of endangered species has been proposed through payments for ESs (Simpson and Sedjo 1996).

Therefore, the strategy is to adapt to any possible perturbation and also mitigate its severity and impact. In this regards, adoption of judicious land use and soil management can also lead to adaptation/mitigation of climate change (Fig. 19.4).

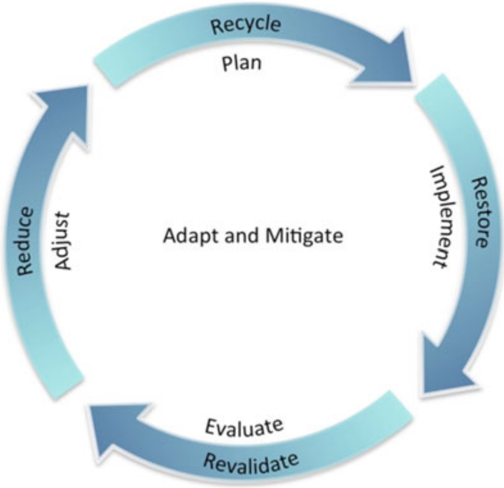

Fig. 19.4 Strategies of soil management

ESs are not an alternative to technological innovations, as both belong together in a strategy for sustainable soil management that aims to facilitate reduced degradation, recycles biomass, restores, and enhances productivity. Although strongly interconnected, there are several types of ESs. Basic concepts of strategic management and prioritization of research and development needs of ESs are discussed below.

19.2 Regulatory Ecosystem Services (M.R. Raupach, R. Cochard, and J.J.C. Dawson)

The objectives are: (1) to identify regulatory ESs with a focus on soil C and soil organic matter (SOM), (2) to assess the integrity of these ESs, (3) to identify knowledge gaps and research directions, and (4) to identify potential points for policy and management intervention.

19.2.1 Scope

The Millennium Ecosystem Assessment (MEA) defined ESs as "the benefits people obtain from ecosystems. These include provisioning services such as food, water, timber, and fiber; regulating services that affect climate, floods, disease, wastes, and water quality; cultural services that provide recreational, aesthetic, and spiritual benefits; and supporting services such as soil formation, photosynthesis, and nutrient cycling" (Millenium Ecosystem Assessment 2005). The MEA defined 24 ESs in these categories, of which nine are classed as regulating services: air quality regulation, climate regulation, water regulation, erosion regulation, water purification, disease regulation, pest regulation, pollination, and natural hazard regulation.

The wide range of ESs can be grouped as follows:

- Regulation of climate
- Regulation of nutrient cycling
- Maintenance of biodiversity
- Regulation of water and air quality
- Regulation of water quantity (amount, timing)
- Erosion suppression (water, wind)
- Hazard regulation and minimization of vulnerability (examples of hazards include floods, earthquakes, tsunamis, fire and drought; a distinction is also made between fast or acute hazards such as earthquakes, tsunamis and flash floods, and slow or chronic hazards such as drought and some floods).

This list includes some services that the MEA classifies as "supporting services", such as nutrient cycling.

19.2.1.1 Systems View

The analysis presented herein is based on a holistic, system-oriented perspective on the Earth System, with a focus on the terrestrial biosphere as a key subsystem. The terrestrial biosphere subsystem includes plants, animals, soil, surface water, groundwater, and the lower atmosphere adjacent to the surface.

The terrestrial biosphere can be considered from several perspectives:

- A set of C (inorganic and organic) pools linked by flows of energy and matter, such as living (above, below ground), detrital and soil pools containing C;
- A set of functions or processes which mediate and link the great cycles of water, energy, C, macro nutrients (N, phosphorus, sulfur), and micronutrients (trace elements);
- A set of life-forms making up terrestrial ecosystems and biota, maintaining plant, animal and soil biodiversity and species richness.

19.2.1.2 Systemic Vulnerability and Resilience

To characterize the integrity of regulatory ESs, critical system-level attributes are those of vulnerability (Turner et al. 2003; Raupach et al. 2011) and resilience (Walker et al. 2009; Folke et al. 2010) to environmental changes and major perturbations such as extreme weather, climate change, pressures from pests and diseases, flood, drought, fire and seismic disturbances. For general definitions and analyses of these concepts, see also (Rockstrom et al. 2009; PMSEIC 2010).

The integrity of regulatory ESs depends not only (or even mainly) on their performance in typical conditions or "good times", but also their performance under stress. The stress may be external to the terrestrial biosphere or ecosystem (e.g., global climate change or a flood) or internally generated (e.g., endogenous pest outbreaks).

19.2.2 Earth System Disequilibrium and its Pressure Points

The phrase "recarbonization of the biosphere" has been used in this volume to describe an imperative for the maintenance and restoration of ESs (regulatory and others). However, this concept requires a careful interpretation.

19.2.2.1 Earth System Disequilibrium

In the Anthropocene epoch, the Earth System is in severe disequilibrium. The Anthropocene (Crutzen 2002; Steffen et al. 2007) is the epoch in which human activities are significantly modifying the great natural cycles of C, water and nutrients, concomitantly influencing climate, biodiversity, land cover and other properties of the state and function of the Earth System. The start of this epoch is considered by some as the dawn of the industrial era in the mid- eighteenth century.

The disequilibrium of the Earth System in the Anthropocene is evident in many ways (Steffen et al. 2004). One of the most profound is the changing composition of the atmosphere through the buildup of carbon dioxide (CO_2) and other gases emitted by human activities, leading in turn to human-induced climate change. This set of processes has led to the injection of a vast amount of C into the atmosphere, terrestrial biosphere and oceans, totaling about 550 Gt C (1 Gt = gigaton = 10^9 ton = 1 billion ton = 1 Pg = 10^{15} g) since 1750 (Raupach and Canadell 2010). Of this, about 220 Gt C remain in the atmosphere and the remainder (340 Gt C) has been absorbed into the terrestrial biosphere (180 Gt C) and oceans (130 Gt C), through land and ocean CO_2 sinks. Of the nearly 10 Gt C year^{-1}, total CO_2 emissions in 2010 from the sum of fossil fuel combustion (9.1 Gt C) and net deforestation (0.9 Gt C), the ocean sink absorbed 2.3 Gt C year^{-1} and the land sink 2.7 Gt C year^{-1}, leaving only 5 Gt C year^{-1} to accumulate in the atmosphere (Le Quere et al. 2009; Friedlingstein et al. 2010).

The implication of this massive injection of previously immobile C into the Earth System is a planetary carbonization of the terrestrial biosphere. The current disequilibrium of the Earth System has increased the total pre-Anthropocene C store in the terrestrial biosphere (about 3,000 Gt C; plants 700 Gt C, soil 2,300 Gt C; Sabine et al. 2004) by nearly 200 Gt C. However, the consequences of an accumulating CO_2 and methane (CH_4) concentrations in the atmospheric pool, requires solutions that optimize C sequestration processes within appropriate sinks.

19.2.2.2 Spatial Patterns and Pressure Points

At a planetary scale, recarbonization of the biosphere is not an imperative. For the integrity of regulatory ESs, the primary issue is the distribution or spatial pattern of C in the terrestrial biosphere rather than its global total. Human pressures on the terrestrial biosphere, through deforestation and agricultural production, have led in some places to severe depletion of terrestrial biospheric C and SOM sinks.

It is, thus, proposed that the integrity of regulatory ESs is best assessed in terms of pressure points, i.e., places where the functioning of regulatory ESs is degraded, relative to a baseline needed for adequate whole-system function. With a focus on C in the terrestrial biosphere, these pressure points are locations where there is too little C, too much C, or where C pools are excessively vulnerable to disturbance. Examples are given in the following section.

19.2.3 Examples of Pressure Points

19.2.3.1 Carbon Pools Vulnerable to Disturbance

Several hitherto largely immobile C pools can be disturbed by climate change, leading to C release to the atmosphere. The following examples include, but are not restricted to, vulnerable soil C pools.

- A major potential pool is the organic C in permafrost soils, estimated at nearly 1,700 Gt C in total (Tarnocai et al. 2009), of which about 100 Gt C may be vulnerable to release by thawing over the next century (Schuur et al. 2008, 2009).
- There is also a significant pool of C in tropical peatland soils, mainly in the Southeast Asian archipelago, of which around 30 Gt C may be vulnerable to decomposition and fire following drainage (Hooijer et al. 2010).
- The C in temperate peatland soils, coastal and upland areas, is likewise vulnerable to decomposition, e.g., following drainage, peat extraction and moorland burning (Dawson and Smith 2007).
- Net releases of C in some forest ecosystems are also likely through fire, insect attack and ecological transitions (Kurz et al. 2008a, b).
- Release of CH_4 hydrates from reservoirs on the ocean floor and beneath permafrost is a high-impact, highly uncertain risk but is presently thought likely to occur over longer timescales compared to the other releases noted above (Bohannon 2008); the biggest short-term vulnerability for this pool may well be through its use as a fuel source.

Pressure points associated with vulnerable C pools require protection from disturbance and monitoring of potential environmental changes that may influence their C source/sink dynamics.

19.2.3.2 Degradation and Desertification

In the dry tropics, arid and semiarid subtropics, and in mountain areas, there are many examples of regions where soil C and SOM have been lowered to inadequate levels for maintaining net primary productivity (NPP), often by inappropriate land management over decades or centuries. Such practices include overgrazing, "mining" of soil C and soil nutrients by cropping without replenishment of these resources and lack of control of water or wind erosion (see below).

Pressure points associated with degradation and desertification are cases where a recarbonization of the terrestrial biosphere is required.

19.2.3.3 Inappropriate Spatial Distribution of Soil (and Plant) Carbon

Examples include:

- Removal of wetlands: Wetland drainage for afforestation is associated with increased flood risk. In these cases, wetland restoration reduces the risk.
- Carbon forestry in grassy catchments: Afforestation of a grassy catchment generally leads to reduced runoff, other parameters being equal (Zhang et al. 2004). This is a case where too much C in the terrestrial biosphere can impact negatively on other ESs (Dawson and Smith 2007).

19.2.3.4 Vulnerability to Erosion

Landscapes vulnerable to erosion by water or wind are also vulnerable to the associated loss of C and nutrients, to reduced air or water quality, and to damage at receptor points where particulates are deposited – e.g., smothering of riverine or estuarine aquatic ecosystems by deposited particulates in water. Benefits of management to reduce erosion include nutrient and soil C retention, and reduced off-site particulate pollution.

19.2.4 Knowledge Gaps

19.2.4.1 Process Knowledge

This includes knowledge of the basic physical, biological and ecological processes needed to assess and maintain regulatory ESs.

The task of assessing the integrity of regulatory ESs, generic process knowledge is broadly adequate, but that many important details remain worthy of further research.

19.2.4.2 System Knowledge

This includes the capacity to understand and manage whole-system function, vulnerability and resilience, recognizing both the biophysical and human dimensions of the system.

In contrast with process knowledge, system-level knowledge in most areas is very poor, and much better integrative capacity is required (PMSEIC 2010). This is a priority area where the efforts of multiple groups and agencies are needed.

19.2.4.3 Knowledge Needed to Identify and Manage Pressure Points

This is a critical area. Key questions are: where are pressure points? Are they getting better or worse? How can they be managed effectively to achieve adequate outcomes in multiple areas?

Answers to these questions involve several components: robust "pressure indices" to quantify the relative performance of regulatory ESs in space and time; spatial biophysical and biogeographic data at high-resolution, and well-calibrated time series to identify trends. For the above examples of pressure points, these knowledge gaps are as follows.

- Carbon pools vulnerable to disturbance: A possible pressure index is a measure of risk of release, quantified as (hazard) × (pool size) × (vulnerable pool fraction). The hazard involves climate factors such as rate of warming, and land management factors such as rates of lowering of water tables through drainage. The pool size is available in broad terms, but the vulnerable pool fraction is very hard to quantify in many cases (e.g., for thawing of hitherto frozen organic soils) and requires better process knowledge.
- Degradation and desertification: A possible pressure index is the ratio of actual to potential soil C levels, and in case of desertification, the combined effect of C level decline with disruption of soil moisture beyond a tipping point, constraining ecosystem functioning. Spatial maps of actual soil C are available but two aspects need much better quantification, i.e., mapped trends (in addition to levels) in actual soil C, and maps of potential soil C.
- Inappropriate spatial distribution of soil (and plant) C: The examples given above were (1) drainage of wetlands, and (2) carbon forestry in grassy catchments. A possible pressure index for wetland removal is the resulting change in water tables and flood heights, assessed from flood models. For carbon forestry, a possible index is the resulting change in modeled streamflow. In both cases, a combination of models with sophisticated biogeographic data is required.
- Vulnerability to erosion: A possible pressure index is a typical or extreme erosion flux relative to other system fluxes, considering both potential and actual erosion (the difference being the effect of ameliorating agents such as vegetation cover). There is a global need for maps of erosion risk (PESERA for Europe; Kirkby et al. 2008) and quantitative assessments of sediment yield and eroded area, e.g., by remote sensing.

19.2.5 Benefits from Interventions

There are three scales, each with a distinct set of benefits, challenges and communities of engagement for intervention.

19.2.5.1 Local Scale

Obvious benefits include yield improvements for farmers, holding on to soil C and nutrients, reduction in nutrient replenishment costs, and reduction in erosion (water, wind) damage both on-site and off-site.

Engagement is through local catchment management authorities, farming advisory groups, industry bodies, and the government and non-government agencies that interact directly with such bodies.

19.2.5.2 National Scale

Approaches include agreements and regulations to govern public-good (common) assets which currently do not have value through property rights, but need protection to ensure the integrity of regulatory ESs. Examples include C and GHG emission rights as constrained by national inventories, water for ecosystem maintenance, especially in water-challenged areas and, reducing risks from extreme events.

Engagement is essentially through government departments and agencies, and also with political bodies who set the rules under which departments and agencies operate. This is often a contested process. A recent case being the effort to return about 30 % of available water to environmental flows in the Murray-Darling Basin (MDB), Australia, which has caused burnings of MDB Authority reports in irrigation towns.

19.2.5.3 Global Scale

The recent history of global efforts to curb GHG emissions indicates that the global community is still at an early stage in learning how to collectively manage the global commons in the Anthropocene.

19.3 Provisioning Ecosystem Services (T. Gaiser, M. Fan, K. Goulding, H. Haberl, I. Lewandowski, and R. Siddique)

The following services are considered as provisioning ESs: Production of (1) food and feed, (2) fibre and construction materials, (3) biofuels, (4) medicinal plants, (5) specialized materials, and (6) provision of drinking water.

19.3.1 Effects of Carbon Sequestration

On average, the effect of additional C sequestration (or recarbonization) in the soil is considered to be positive for the first five provisioning ESs listed above. However, the magnitude of the effect depends on several factors:

- Species (crops/utilized wild plants),
- Climate conditions,
- Soil properties (in particular texture, drainage, amount of biomass and composition),
- Nutrients added in the process of recarbonization or major nutrient constraints at a given site, and
- Quality of C sources added for recarbonization.

Food quality partly relates to micro-nutrient density of foods, such as iron. Micro-nutrient deficiencies are an important part of the global food problem, mainly among the poor. It can also be affected by recarbonization of the soils depending on the recarbonization measures and related impacts on micronutrients. With the use of manure, the content of micronutrients essential to human nutrition can be higher compared to exclusive mineral fertilizer applications. Plant breeding for efficient nutrient uptake and harmonized supply of nutrients promotes healthier plants, which are more resistant against diseases and pests. On the other hand, when sewage sludge or slurry are applied in an inappropriate way, the risk of pathogens on the harvested crops or of heavy metal uptake is increased. If recarbonization of soils and above-ground biosphere are accomplished in an appropriate way then it will have positive effects on drinking water quality. However, above-ground recarbonization can have both negative and positive effects on water yield depending on the environmental context. In any case, above-ground recarbonization is expected to increase the buffering capacity of the land against floods and droughts.

19.3.2 Practices Needed to Restore Both Biosphere Carbon Pools and Provisioning Ecosystem Services

Recarbonization of soils should only be attempted where appropriately integrated with the provision of other ESs, and on marginal lands. Practices include:

- Adding various forms of organic C (farm yard manure, compost, green manure, biosolids, biochar, paper),
- Increasing photosynthetic efficiency,
- Breeding/use of crops with deeper and bushier root systems,
- Adopting high yielding cropping systems (more residues left after harvest),
- Using crop rotations instead of monocultures, and
- Adopting intercropping and agroforestry.

Recommended accompanying measures including the following:

- Reduction of erosion through, e.g., reduced tillage, contour tillage, mulching, cover crops, agroforestry, and
- Avoid removal or burning of crop residues.

19.3.3 How Can These Practices Be Implemented for Maximizing Benefits from Recarbonization of Soils and the Biosphere?

For harnessing of maximum benefits including direct and indirect costs of recarbonization integrated interdisciplinary research is recommended (see recommendations below).

19.3.4 How Can Stakeholders Be Incentivized to Achieve This?

Some of the options include subsidies combined with legal requirements, carbon market with control instruments to maintain and restore other ESs, community participation, multi-stakeholder approaches, and increasing public awareness. The latter requires strengthening the knowledge base and extension services.

19.3.5 Research Recommendations

Notable recommendations for scientists include the following:

- Quantify synergies and trade-offs between recarbonization and other ES, as affected by management practices (e.g., crop species, materials used as soil amendment), and their short- and long-term net-benefits for farmers and society,
- Quantify the effect of soil recarbonization on soil biota and their modifying effect on provisioning ESs,
- Assess catchment scale water budgets as affected by recarbonization of soil and biosphere components, in contexts of variable water scarcity,
- Conduct integrative assessment of pros and cons of biochar application, and
- Adopt integrative interdisciplinary research involving stakeholders (e.g., Rural Economy and Land Use program in UK).

19.3.6 Policy Recommendations

Some recommendations for policy makers include the following:

- Implement appropriate recarbonization measures with experimental approaches through institutions at a suitable scale to account for variation in local conditions,
- Concentrate incentives to encourage recarbonization to those practices that also have positive synergies with other ESs (e.g., pressure points of regulating and provisioning ESs),
- Develop incentives as well as regulatory mechanisms that optimize trade-offs where trade-offs of recarbonization with other ES are inevitable, and

- Develop mechanisms for broad stakeholder participation in negotiating balanced optimization of trade-offs between recarbonization and other ESs (e.g., building food sovereignty).

19.4 Cultural Ecosystem Services (G.B. De Deyn, B. Egoh, K. Lorenz, J. Wesemeir, and T. Tilmann)

Cultural services include non-materialistic benefits returns for the good of humankind. Thus, key cultural services are cognitive (understanding soil systems and its relation to humanity), spiritual/religious (soil stewardship), recreational (including eco-tourism), heritage (archaeology, history of civilizations), and educational (learning resource). The cultural attributes of soils differ a lot across cultural contexts and nations.

19.4.1 Effects of Carbon Sequestration

The impact of enhanced C sequestration on cultural services will be highly dependent upon the methods applied or approaches taken to achieve C sequestration as well as on the locations where C sequestration incentives will be imposed. The incentives for C sequestration need to match the local possibilities in terms of people's acceptance. Certain incentives may appear very plausible from a purely economic, ecological and rational point of view but will encounter resistance for its adoption because of religious beliefs or cultural traditions (e.g., applications of human based manure/waste products even when health issues are taken care of by pre-treatment).

19.4.2 Management Practices Needed to Restore Both Carbon Pools and Cultural Ecosystem Services

Thorough understanding of local peoples' values with respect to their relation to soil and the landscape is of primary importance and can be very specific. This understanding will require direct 'on site' socio-cultural investigations directly speaking to local communities. In this regard, education is a key tool for raising awareness and responsibility for the status of (world) soils and the benefits they provide to current and future generations. Let children (especially in urban areas) experience and learn where the food actually comes from and how soils play a central role in this will raise respect for soils, its amazing diversity and properties "magic box".

Besides education, also the role of religion/spiritualism for elevating or reinvigorating care taking of soils by societies should be explored.

19.4.3 Implementation of Land Use and Soil Management Practices for Maximizing Benefits from Carbon Sequestration and Restoration of Cultural Ecosystem Services – How Can Stakeholders Be Incentivized to Achieve This?

By knowing what people and markets value (and not value) in terms of soils and landscapes, strategies can be optimized for multiple benefits. There will be need for custom made rather than general practices in order not to lose on cultural services because of differences in people's religious and spiritual beliefs, places of worship and cultural heritage.

By selecting areas with low cultural value in its degraded state and restoring these for soil C and valued properties such as biodiversity, aesthetic and/or recreational (e.g., eco-tourism) services win-win strategies may be achieved.

19.4.4 Policy Interventions Needed to Enhance Cultural Ecosystem Services in a Re-carbonized Biosphere

Cultural services in a re-carbonized biosphere should generally not need policy interventions because the cultural and especially the spiritual values are highly personal experiences of individuals that can and should not be imposed by policy. However, the educational services of soils will need a policy that supports investments in bringing soils back closer to the essence of being human and the responsibilities for future generations.

19.4.5 Monetary Value of Cultural Ecosystem Services

The monetary value of cultural services is difficult to determine because the definition of cultural services implies that returns to humans are non-materialistic. The topic is also debatable from an ethical and spiritual point of view. How can you put a value on the essence of life on earth? Do we own the land or does the land owns us and just tolerates us (for the time being)? Indirect valuation approaches, such as existence and insurance values can in principle be applied, as being used in biodiversity valuation.

Recreational values may be easier to quantify, e.g., in terms of the money people are prepared to pay to visit certain ecosystems (eco-tourism) or to make use of recreational infrastructures. If recreational areas are supported by public money (government), then the number of visitors to certain recreational areas can be used to reflect its value.

The value of soils as logbook of human history should be highly valued because no other medium is so rich on resources to help us reconstruct the evolution of humans and civilizations. Understanding our past makes us more human. It can make us more aware of threats that humans have faced through history and the ingenuity by which they were solved, or not which should facilitate learning from historic mistakes.

Understanding of soil systems (cognitive and inspirational values) may be partly reflected in soil-system based discoveries and their economic value (e.g., bio-physico-chemical networks, structures, products).

19.5 Economics and Policy Services (W. Oluoch-Kosura, P. Mayer, H. Bliddal, and F. Glante)

The strategy is the implementation of land and soil management practices for maximizing benefits from C sequestration and restoration of ESs, and how stakeholders can be incentivized to achieve this. In order to provide a basis for this, economic assessment of soil degradation is required, and to incentivize policy a costing actions versus inaction is called for (Nkonya et al. 2011).

19.5.1 Institute Relevant Policies

Policies to be hinged on strategies agreed on by stakeholders (actors) encourage interdisciplinary (multidisciplinary) approaches to develop strategic plans including soil management, and encompass the following:

- Fostering political will through information and demonstrated evidence regarding net-benefits of policy options for sustainable soil management and the key role of soil carbon.
- Integration of science of soil and technology policy for a prioritized agenda that aims at sustainable productivity.
- Linking the role of soils to potential or emerging crises (e.g., food and fuel crises) is needed to get attention of the public and other stakeholders.
- Identifying policy instruments for specific contexts that enhance both, the public goods aspects of soils and the private returns for investment in sustainable soil management practices. Potential conflicts between the two need to be addressed by public policies and ESs can play key roles among them.

19.5.2 Understanding the Science/Policy Interface

The critical interaction between scientists and policy makers is important in cultivating/building trust.

There exists a strong need to have stakeholder analysis to know who are affected by various interventions from the soil research outcomes and recommendations, and how decisions are made at the local, national and regional and global scales. It is pertinent to exploit the notion that "scientists" provide credible and evidence based data and messages that can be received and acted upon accordingly. It is also pertinent to develop clear and simple messages on research outcomes, giving tradeoffs and scenarios for needed actions and the cost for inaction for different potential or actual interventions. In this regards, the U.S. President F.D. Roosevelt (1937) stated that, "The nation that destroys its soil destroys itself"! Furthermore, 90 % of all food is grown on soil, and healthy soils are the basis for food security and food sovereignty. These identifying messages of the stewardship can be effectively communicated by partners to work with to convey relevant messages. It is also important to maintain the integrity and objectivity of scientific research, in the interdisciplinary collaborative mode, and involving diverse and new stakeholders. It is also pertinent to take advantage of initiatives (e.g., the FAO initiative on Pillars for the Global Soil Partnership GSP, the Global Soil Biodiversity Initiative GSBI, the Economics of Land Degradation Initiative ELD) and develop cooperative programs.

19.5.3 Toward Global Policy Frameworks

The challenge of sustainable management of soils must be on major international agendas. Thus, developing a "Soil Convention" at a global scale may be relevant. At the same time it must be noted that modern global policy frameworks are less a matter of global conventions and more a complex process of interaction among stake holders and government to government (GtoG) relationships and informal consensus seeking. As sustainable soil management is very much a local matter at global scale, a combination of global action and of globalised cooperation of national and regional mechanisms of action should be called for.

Both of these complementary approaches require to report more on global trends as well as systematic small scale events based on typologies that facilitate policy learning. Therefore, the strategy is to report and popularize success stories for scaling up, based on sound country and regional level assessments and success narratives.

19.5.4 Incentives

Developing a reward system is essential for promoting research, development and outreach programs on soils through:

(a) Sharing valuable information on incentives and their partly soil enhancing and soil threatening effects with prospective beneficiaries. Negative external effects from incentives need to be internalized in costs through active policy. Information is only the first step.
(b) Building capacity of: (i) scientists-soft skills-to distil messages from complex research results, (ii) farmers such as farmers Field Schools-Kenya for sharing information between farmers-indigenous knowledge systems and appropriate and workable new scientific approaches, and (iii) trainers for others.

(c) Developing legislation for securing land tenure for land owners to improve upon responsible stewardship and governance of land including soil and other natural resources.
(d) Providing subsidies for interventions deemed to have positive net benefits/impact to society and human well-being (e.g., subsidizing farmers compliant to maintain a positive C balance).
(e) Increasing productivity, profitability and improving livelihoods.
(f) Developing investment codes (Public Goods) such as by publicizing BMPs such as those in U.K., or developing technical guidelines for responsible stewardship of the environment (soils).
(g) Identifying winners and losers in relation to the desired interventions and rewarding appropriately by using public money for public goods and principles. Payments depend on the political good will, and
(h) Creating strong and functioning institutions with sound understanding of technical and socio-economic understanding of soil policy.

The overall strategy involves determining trade-offs for interventions and developing clear scenarios for costs/benefits/impacts in the short/medium/long term by stakeholder groups and for the ecosystems.

19.5.5 Others

There exists a strong need to promote monitoring, quantification, measurement and analyses of costs/benefits/effectiveness (impacts). This would require building capacity for developing appropriate methodologies for valuation of non-tangible ESs. Some feasible options as also pursued by the Economics of Land Degradation Initiative are:

(a) Developing scenarios for cost of inaction (doing nothing),
(b) Assessing cost/benefit (net benefits) from interventions and document impacts,
(c) Creating maps of global hot spots (e.g., South Africa), and
(d) Identifying knowledge gaps and developing research agenda to address these issues.

19.5.6 Ecosystems That Do Not Need Policy Intervention

It is equally important to concentrate efforts and focus on ecosystems which need policy interventions. There are several ecosystems which are at the equilibrium states (where policy is not required), and which may be very short-lived, or where collective action has facilitated sustainable land and soil use practices. It is prudent to avoid possibilities of backlashes due to temporary equilibrium. Even if sustainable agricultural intensification is achieved, it is still important to monitor possibilities for ecological backlash.

19.6 Conclusions

Measuring, monitoring and assessing ESs are needed to identify policy interventions for sustainable management. Enhancing one ES can cause some disservices on trade-offs off site because of a strong interconnectivity. The quantity and quality of soil and ecosystem C pools are essential to provisioning of several ESs and functions. Thus, identification of key questions (for research, development, and outreach) and their prioritization is important at local, regional and global scales. Over and above the choice of land use and practices which ensure sustainable intensification and development, key policies must be identified and implemented by applying trans-disciplinary approaches addressing societal problems by means of interdisciplinary collaboration as well as the collaboration between researchers and extra-scientific actors. It is also important to define winners and losers, map global hotspots, and identify those ecosystems which do not need policy interventions.

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