Food and Agriculture
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VOLUME 2

RECARBONIZING GLOBAL SOILS

A technical manual of recommended management practices

HOT SPOTS AND BRIGHT SPOTS OF SOIL ORGANIC CARBON

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INTERGOVERNMENTAL
TECHNICAL PANEL ON SOILS

GLOBAL SOIL
PARTNERSHIP

VOLUME 2

RECARBONIZING GLOBAL SOILS

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HOT SPOTS AND BRIGHT SPOTS OF SOIL ORGANIC CARBON

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Forests

1. Forests

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1. Definition and description

Forest refers to land spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10 percent, or trees with the potential to reach these thresholds at these locations (FAO, 2020a). Forest lands that are temporarily treeless because of harvest or disturbance are included. Tree plantations are also included, unless they are in predominantly agricultural systems (FAO, 2020a). Forest does not include land that is predominantly under agricultural or urban land use, even though such land may have some tree cover.

Globally, forests cover 4.06 billion hectares, which is 31 percent of the total land area (FAO, 2020b). There are still at least 1.11 billion ha of primary forest – that is, forests composed of native species in which there are no clearly visible indications of human activities and the ecological processes have not been significantly disturbed (FAO, 2020b). More than 2.05 billon ha (greater than 50 percent) are covered under long-term management plans (FAO, 2020b).

About 1.15 billion ha of forest is managed primarily for the production of wood and non-wood forest products, while 749 million ha is designated for multiple use, which often includes production (FAO, 2020b). South America has the highest (2 percent total forest area) and Europe the lowest (0.4 percent total forest area) share of plantation forest. Almost half (44 percent) of plantation forests are composed primarily of introduced species; though this varies globally; South American plantations consist almost entirely of introduced species while North and Central American plantations mainly contain native species (FAO, 2020b).

2. Global distribution of hotspot

Forests cover 4.06 billion hectares (ha) of land globally, which is 31 percent of the total land area. The tropical domain comprises the largest proportion of the world's forests (45 percent), followed by the boreal (27 percent), temperate (16 percent) and subtropical (11 percent) domains (FAO 2020b).

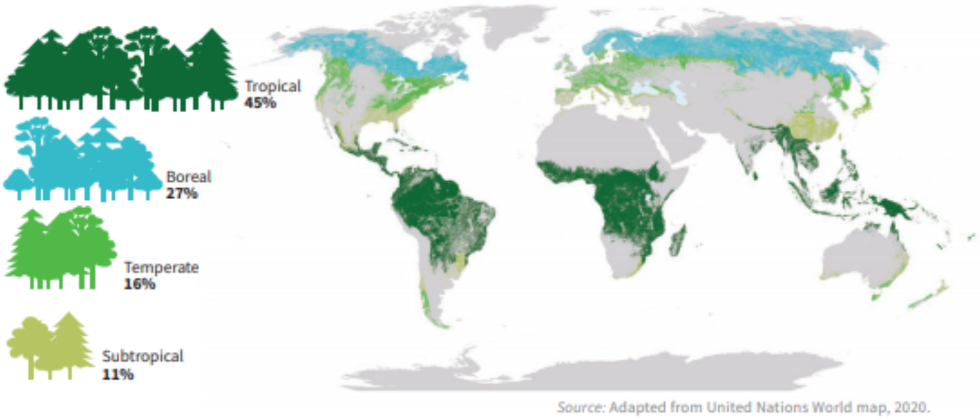


Figure 1. Proportion and distribution of global forest area by climatic domain, 2020 (FAO, 2020b)

Tropical forests also have the highest available timber volume (121 m^3/ha), carbon storage (91 tonnes/ha) and species diversity, while boreal systems have the least of these attributes (Kappen *et al.,* 2020). Temperate forests are the smallest biome (accounting for 15 percent of total forest area) but account for an immense share (29 percent) of global forest product harvest (Kappen *et al.*, 2020).

3. Global carbon stock and additional carbon storage potential

The total carbon stock contained in the world's forests is 662 gigatonnes (Gt), and the average carbon density of forests is 163 t/ha (FAO 2020b). The global average carbon density of forests is 74 tonnes/ha, with tropical forests containing the largest average carbon density (91 tonnes/ha), temperate forests intermediate (53) and boreal forests the lowest (41) (Kappen *et al.*, 2020). The highest densities of carbon are found in forests of South America and Western and Central Africa, storing about 120 tonnes of carbon per hectare in living biomass alone. On average, 44 percent of forest carbon is found in the living biomass (296 Gt of carbon in both above- and below-ground biomass), although the greatest proportion of forest carbon (45 percent) is found in soil organic matter, with an additional 10 percent in dead wood and litter (Figure 2). Soil C stocks (including litter) comprise the highest proportion of ecosystem C stocks in both boreal (202 Gt; 70 percent of the ecosystem C stock) and temperate forests (69 Gt; 60 percent), but only 30 percent of ecosystem C stocks (155 Gt) in tropical forests (Pan *et al.*, 2011).

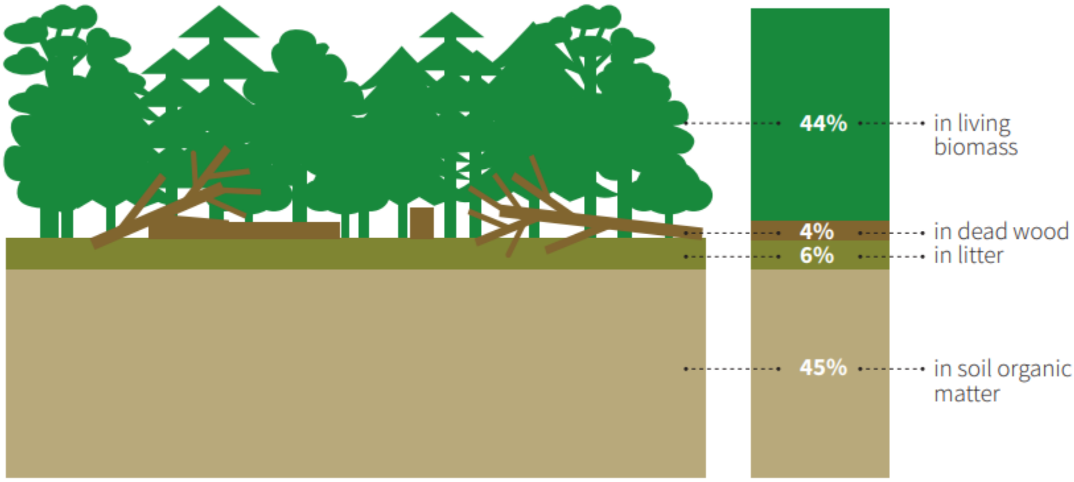


Figure 2. Proportion of carbon stock in forest carbon pools in 2020 (FAO, 2020b)

The total carbon stock in forests globally declined from 668 Gt in 1990 to 662 Gt in 2020; during the same period carbon density increased from 159 tonnes to 163 tonnes/ha (FAO, 2020b).

Globally, vegetation currently stores around 450 Gt of carbon (GtC), but could store around 916 GtC, in the absence of non-forest land uses (Erb *et al.*, 2018). Deforestation and other forms of land-cover conversion are responsible for around 55 percent of the difference between current and potential biomass stocks. The remaining 45 percent is from managed ecosystems where actual biomass only comprises 60 to 69 percent of the potential biomass stock per unit area. Forest management contributes two-thirds and grazing one-third to the management-induced difference in biomass stocks. The additional C storage potential of these managed ecosystems is about 396 Gt.

Carbon storage in forests can be increased by reducing deforestation, reforesting cleared forests, afforesting land that has been deforested, restoring land that has been degraded, and increasing C stocks in existing forests through forest management. Reduced deforestation, afforestation, and improved forest management globally could sequester an additional 3.8 GtC annually, of which about 1.6 GtC would result from reduced deforestation (Nabuurs *et al.*, 2007). Of 20 natural pathways to mitigate climate change, forest pathways offer over two-thirds of cost-effective mitigation needed to hold warming to below 2 °C (Griscom *et al.*, 2017; Figure 3). The maximum additional mitigation potential of all 20 natural pathways was 23.8 Gt CO_2eq/yr at a 2030 reference year, while cost-effective climate mitigation potential was 11.3 Gt CO_2eq/yr. Reforestation, avoided forest conversion, and improved forestry offered large and cost-effective mitigation opportunities with well-demonstrated co-benefits, including biodiversity habitat, air filtration, water filtration, flood control, and enhanced soil fertility (Griscom *et al.*, 2017). The reduction in GHG emissions that could be achieved by reducing deforestation and forest degradation has an estimated technical mitigation potential of 0.4–5.8 Gt CO_2/yr (IPCC, 2020).

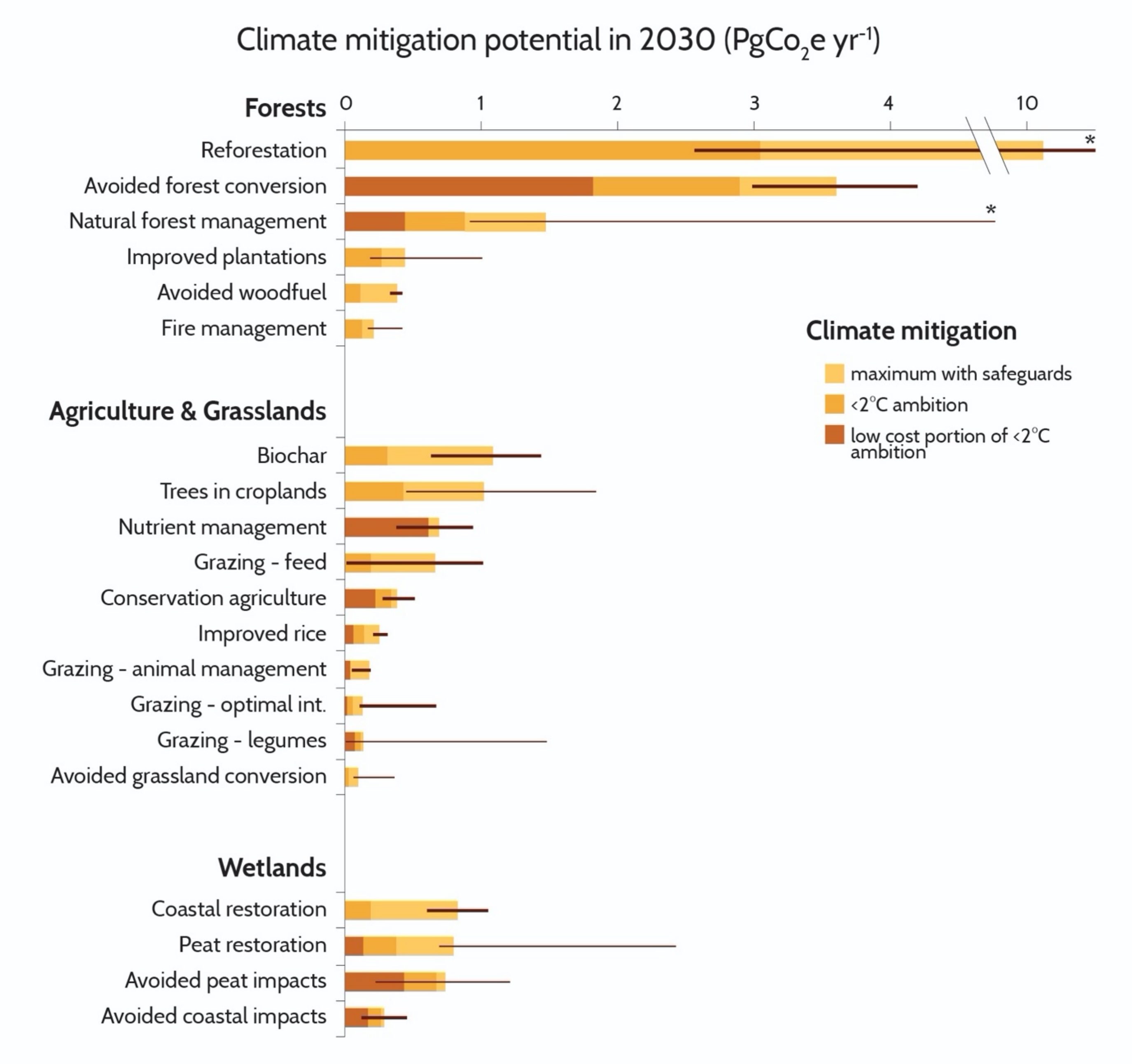


Figure 3. Of 20 natural pathways to mitigate climate change, forest pathways offer over two-thirds of cost-effective mitigation needed to hold warming to below 2 °C by 2030 (Adapted from Griscom *et al.* 2017)

By mapping the global potential tree coverage based on climate, Bastin *et al.* (2019) concluded that there is potential for an extra 0.9 billion hectares of canopy cover in areas that would naturally support woodlands and forests (a 25 percent increase in forested area; at maturity, these trees could store 205 GtC. The C storage potential for a largescale afforestation program that is economically, politically, and technically feasible is considerably lower (Nilsson and Schopfhauser, 1995).

4. Importance of forest conservation for the provision of specific ecosystem services

Forests provide critical ecosystem services including water regulation, temperature control, pollination and biodiversity.

Forests regulate streamflow, support groundwater recharge, filter water, enhance soil infiltration and soil water storage, and reduce soil erosion and sedimentation of water bodies. Over 75 percent of the world's accessible freshwater comes from forested watersheds, and over half of the human population depends on these areas for water (FAO, 2019). Evapotranspiration from forests and trees may reduce runoff at the catchment scale, but increase precipitation and water availability downwind (Ellison, Futter and Bishop, 2012). On average 40 percent of rainfall over land is recycled from evapotranspiration over land surfaces. Tropical and subtropical forests act as large conveyors of atmospheric moisture, providing a global circulation system that influences regional cloud cover and precipitation (Ellison *et al.*, 2017). The large-scale loss of these vast, contiguous tropical forests has been linked to decreased regional precipitation (Ellison *et al.*, 2017).

Forest cover directly affects regional surface temperature through exchanges of water and energy. Increases in forest cover in tropical regions increase evapotranspiration rates, resulting in cooler days during the growing season and reductions in the amplitude of heat-related events. Increased tree and shrub cover also has a wintertime warming influence in regions with seasonal snow cover, such as boreal and some temperate forests, due to reduced surface albedo (Shukla *et al.*, 2019).

Many wild pollinators depend on forests for nesting and foraging, and the extent of forests and other natural habitats in a landscape influences pollinator species composition (Krishnan *et al.*, 2020). A national assessment in Tanzania (Tibesigwa *et al.*, 2019) showed improved crop productivity in proximity to forests, and a positive association between forest cover and crop revenue. Pollinators are also vital for the regeneration of trees and plants used for timber and non-wood forest products. The decline in populations of both wild and managed pollinators can also hinder natural regeneration of forests (FAO, 2020b).

Forests also harbour most of Earth's terrestrial biodiversity and provide habitats for 80 percent of amphibian species, 75 percent of bird species and 68 percent of mammal species (FAO and UNEP, 2020). Tropical forests alone host at least two-thirds of terrestrial species. Globally, 424 million ha of forest is designated primarily for biodiversity conservation (FAO, 2020b).

The estimated value of the environmental services provided by forests, for example absorbing harmful particles from air, filtering water, providing protection from soil erosion, rock falls, high tides and tsunamis is about 2 percent to 7 percent of their total value (USD50 trillion to USD150 trillion; Kappen *et al.*, 2020).

4.1. Minimization of threats to soil functions

Table 1. Soil threats

Soil threats	
Soil erosion	Tree canopies and litter layer reduce impact of rain; tree root systems hold onto soil particles; woody debris slows overland flow on slopes.
Nutrient imbalance and cycles	Establishment of forest generates litter and soil organic matter; canopy intercepts gaseous, particulate and dissolved nutrients, roots access nutrients deep in soils and enhance soil weathering.
Soil salinization and alkalinization	Tree cover reduces evaporative losses but can cause problems if planted in very dry areas.
Soil contamination / pollution	Plant species with tolerance for contaminants can reduce soil concentrations through phytoremediation.
Soil acidification	Depends on species – some species increase pH and base saturation; others acidify soil. Some tree species solubilize Fe oxides, which is otherwise fixed in the weathered soils.
Soil biodiversity loss	Tree root exudates support diverse rhizosphere community; root and leaf litter support diverse decomposer communities; wood supports saproxylic organisms.
Soil compaction	Tree roots penetrate soil and reduce compaction; tree roots also increase soil aggregation and improve soil structure.
Waterlogging	Tree root channels enable water infiltration; canopy transpiration returns water to atmosphere.

4.2. Increases in production and food security

Forests contribute to food security by providing nutrient-rich foods, income, employment, energy and ecosystem services (FAO and UNEP, 2020). Around 1 billion people depend to some extent on wild foods such as wild meat, edible insects, edible plant products, mushrooms and fish. Forests also diversify dietary supplies for human populations (FAO and UNEP, 2020). Forests may also provide fodder, green manure, and compost for farming (FAO and UNEP, 2020). Forest ecosystems have the potential to enhance agricultural and fishery production through water regulation, soil formation, protection, nutrient circulation, biodiversity conservation, agroecosystem stability, pest control and pollination and so contribute to the food security (FAO and UNEP, 2020). Since trees are often more resilient to adverse weather conditions than agricultural crops, forest-based food items contribute to household resilience by serving as an important safety net in emergencies such as crop failure (FAO and UNEP, 2020).

The formal forest sector provides an estimated 45 million jobs globally and labour income in excess of USD 580 billion per year (including direct, indirect and induced employment; FAO and UNEP, 2020). Small and medium-sized forest enterprises account for about 20 million of these jobs, generating value of USD 130 billion per year. Globally, the reported value of non-wood forest products removals in 2015 amounted to almost USD 8 billion (FAO and UNEP, 2020). Globally, about 1.15 billion ha of forest is managed primarily for the production of wood and non-wood forest products and an additional 749 million ha is designated for multiple use, which often includes production. Forests and trees are also important livelihood components for many, including the estimated 2.5 billion people involved in smallholder agriculture (FAO and UNEP, 2020).

Woodfuel plays a critical role in ensuring access to affordable, reliable and modern energy by providing basic energy services to about 2.4 billion people worldwide, or one-third of the world's population (FAO and UNEP, 2020). Globally about half of total removals are for woodfuel; this ranges from 17 percent in high-income countries to >90 percent in low-income countries (FAO and UNEP, 2020).

4.3. Improvement of human well-being

Forests provide a wide range of products and services that contribute to human health, including medicines, clean water and air, shade and green spaces to exercise and relax in (FAO and UNEP, 2020). More than 28 000 plant species, many of which are found in forest ecosystems, are used as medicines (Willis, 2017). Traditional medicine systems contribute to the resilience of forest-dependent peoples around the world, often as the most available, accessible, affordable and sometimes culturally acceptable source of health care. Forests also indirectly decrease the occurrence of food- and waterborne diseases by filtering water and providing woodfuel for cooking food and sterilizing water (FAO and UNEP, 2020).

Exposure to forests positively affects human health, particularly in urban areas (FAO and UNEP, 2020). Forests improve urban air quality, reduce the urban heat island effect and buffer noise. Forests and green spaces have positive physiological effects, improve mental well-being and promote physical exercise which improves health.

Forests and trees are vital sources of income, livelihoods and well-being for rural populations, particularly indigenous people, smallholders and those living in close proximity to forests. Forests account for an estimated 45.15 million jobs globally and labour income in excess of USD 580 billion per year (FAO and UNEP, 2020). Recreation and tourism also contribute to rural cash economies, with about 8 billion visits to protected areas contributing in the order of USD 600 billion annually. Worldwide, 186 million ha of forest is reserved for services such as recreation, tourism, education research and the conservation of cultural and spiritual sites, and the global area designated for this forest use has increased at a rate of 186 000 ha per year since 2010 (FAO 2020b).

Kappen *et al.* (2020) estimated the social value of forests, based on: 1) the cost of housing and food for the nearly 200 million people who rely on forests for subsistence if they had to live in a non-forested rural community, 2) the personal income of the 12.6 million people worldwide who work in the forest industry, and 3) the travel costs people are willing to pay for access to forests for recreation. Social values constituted 2 percent to 7 percent of total forest value ($50 trillion to $150 trillion), mostly from subsistence use of forests and forestry employment. By far the largest share of global social value was from tropical forests in Asia and Africa, where the forest products industry is a major employer and large numbers of people also live in, and rely on the forest for their livelihood.

4.4. Mitigation of and adaptation to climate change

Forests play a crucial role in determining the accumulation of greenhouse gases in the atmosphere, as they absorb roughly 2 billion tonnes of carbon dioxide equivalent each year, storing the fixed C in long-lived tissues and soil. Deforestation is one of the biggest sources of carbon dioxide.

The IPCC's Fifth Assessment Report concluded that the most cost-effective GHG mitigation options for forestry are reducing deforestation, afforestation/reforestation, sustainable forest management and forest restoration (IPCC, 2014). These are defined below, along with their mitigation potential (Figure 4).

Categories	Practices and Impacts
	Forestry
Reducing deforestation	**C**: Conservation of existing C pools in forest vegetation and soil by controlling deforestation protecting forest in reserves, and controlling other anthropogenic disturbances such as fire and pest outbreaks. Reducing slash and burn agriculture, reducing forest fires.
	CH_4, N_2O: Protection of peatland forest, reduction of wildfires.
Afforestation/Reforestation	**C**: Improved biomass stocks by planting trees on non-forested agricultural lands. This can include either monocultures or mixed species plantings. These activities may also provide a range of other social, economic, and environmental benefits.
Forest management	**C**: Management of forests for sustainable timber production including extending rotation cycles, reducing damage to remaining trees, reducing logging waste, implementing soil conservation practices, fertilization, and using wood in a more efficient way, sustainable extortion of wood energy
	CH_4, N_2O: Wildfire behaviour modification.
Forest restoration	**C**: Protecting secondary forests and other degraded forests whose biomass and soil C densities are less than their maximum value and allowing them to sequester C by natural or artificial regeneration, rehabilitation of degraded lands, long-term fallows.
	CH_4, N_2O : Wildfire behaviour modification.

Figure 4. Summary of the most cost-effective GHG mitigation options for forestry (Adapted from IPCC, 2014)

Protection of existing forests (reduced deforestation and forest degradation) allows for conservation of existing carbon stocks, and reductions in carbon losses from biota and soils. Reducing deforestation and forest degradation lowers GHG emissions, with an estimated mitigation potential of 0.4 – 5.8 Gt CO_2/yr (IPCC, 2019).

Afforestation and reforestation (planting trees on non-forested land) can contribute to climate change mitigation by increasing stocking density in forests, carbon sequestration in soils, and wood use in construction activities. Afforestation and reforestation also generate changes in albedo resulting from land-use and land-cover change that increase reflection of visible light (IPCC, 2019). The many factors that must be considered in planning and implementation of afforestation schemes are touched on in Factsheet No. 6 on Afforestation (Volume 5, this manual).

Sustainable forest management practices aimed at providing timber, fiber, biomass, non-wood resources and other ecosystem functions and services, can lower GHG emissions and can contribute to adaptation. By providing long-term livelihoods for communities, sustainable forest management can reduce the extent of forest conversion to non-forest uses (e.g. cropland or settlements). Provision of products with low GHG emissions that can replace products with higher GHG emissions for delivering the same service (e.g. replacement of concrete and steel in buildings with wood, some bioenergy options) are other options to diminish climate change. Forest management has been estimated to have moderate mitigation value (0.3 - 3 Gt CO_2eq/yr; IPCC, 2019)

The capability of forests to regulate climate through carbon capture and storage accounted for 65 percent to 90 percent of the total value of forests (USD 50 trillion to USD 150 trillion). Tropical forests account for three-quarters of the total value, due to their area (58 percent of total forest area), high carbon density and high tree biomass (Kappen *et al.*, 2020).

5. General challenges and trends

Forest loss

Occurs through conversion to other land uses such as agriculture (cropping and pasture) or urbanization. The world has lost 178 million ha of forest since 1990 (FAO, 2020b). The rate of net forest loss decreased substantially over the period 1990–2020 due to a reduction in deforestation in some countries, plus increases in forest area in others through afforestation and natural expansion of forests. The rate of net forest loss declined from 7.8 million ha per year in the decade 1990–2000 to 5.2 million ha per year in 2000–2010 and 4.7 million ha per year in 2010–2020. The area of primary forest has decreased by 81 million ha since 1990, but the rate of loss more than halved in 2010–2020 compared with the previous decade (FAO, 2020b).

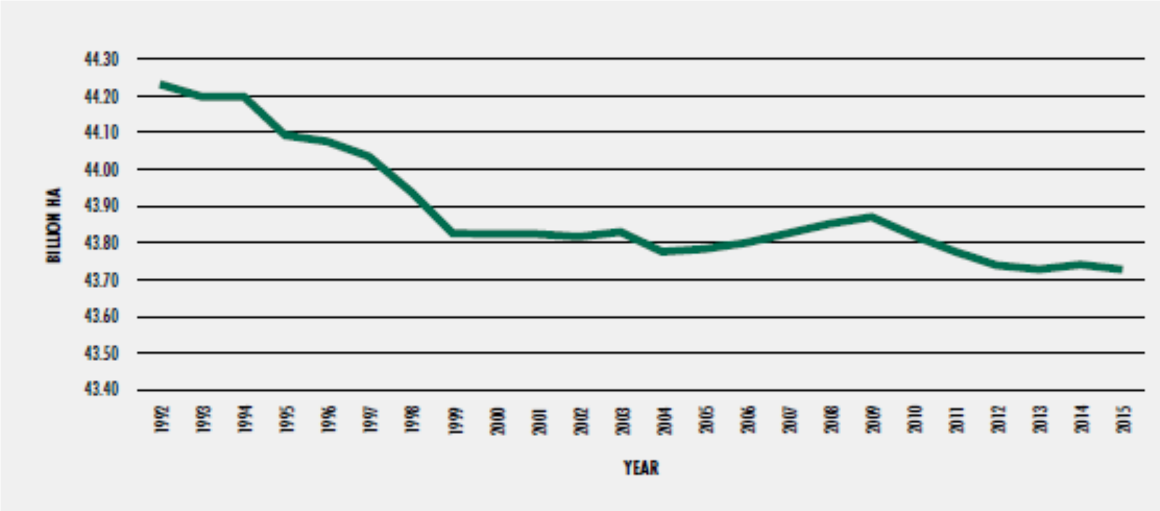


Figure 5. Trends in global tree cover between 1992 and 2015 (FAO and UNEP, 2020)

Africa had the largest annual rate of net forest loss in 2010–2020, at 3.9 million ha, followed by South America, at 2.6 million ha. The rate of net forest loss has increased in Africa in each of the three decades since 1990. It has declined substantially in South America, however, to about half the rate in 2010–2020 compared with 2000–2010. Asia had the highest net gain of forest area in 2010–2020, followed by Oceania and Europe. Nevertheless, both Europe and Asia recorded substantially lower rates of net gain in 2010–2020 than in 2000–2010. Oceania experienced net losses of forest area in the decades 1990–2000 and 2000–2010 (FAO and UNEP, 2020; Figure 6).

Figure 6. Net forest area change by region between 1990 and 2020 (FAO and UNEP, 2020)

Figure 7. Annual rate of forest expansion and deforestation between 1990 and 2020 (FAO, 2020)

Loss of certain forests such as mangroves and moist tropical forests are of particular concern for SOC stocks. Mangrove forests can store 3 to 4 times as much carbon in soil as other forest types, and as much as 15 percent of marine organic carbon burial may occur in mangrove forests. Mangroves are being lost and degraded through urban development and overexploitation of timber and food sources such as fish, crustaceans and shellfish. As much as 20 percent of the global area of mangroves was lost between 1980 and 2005, and rates of loss are about 0.2 to 0.4 percent per year (see Hotspot n°5 on Mangrove forests, this volume). Tropical moist forest store about 650 m tonnes of carbon, which is about 30 percent of total carbon in terrestrial ecosystems, and average SOC stocks are greater than 100 tC/ha (see Hotspot n° 2 on Tropical Moist Forests, this volume). Tropical moist forests are being lost through unsustainable logging, conversion to agriculture and fire, and account for 32 percent of global forest cover loss.

Forest fragmentation

Is the division of continuous habitat into smaller and more isolated fragments –initiates long-term changes to the structure and functions of the remaining forest fragments (FAO and UNEP, 2020). Reduction of forest patch size and increase in patch isolation decrease the abundance of birds, mammals, insects and plants by 20 to 75 percent, impacting ecological functions such as seed dispersal and ecosystem services such as carbon sequestration, erosion control, pollination and nutrient cycling (Haddad *et al.*, 2015). Roughly 80 percent of the world's forest area is found in patches larger than 1 million hectares; this size class accounted for more than 25 percent of the forest area for all forest types. Tropical rainforest and boreal coniferous forest are the least fragmented forest ecosystems – more than 90 percent of the forest area in these zones is in patches larger than 1 million hectares. Almost 35 million forests (7 percent of the global forest area) occur in patches smaller than 1 000 hectares (FAO and UNEP, 2020). 70 percent of global forest area is within one km of the forest/non-forest boundary and therefore subject to fragmentation in the future, including some areas that are currently considered primary (Haddad *et al.* 2015).

Forest degradation

Is the reduction of the capacity of a forest to provide goods and services (FAO, 2020a). Degraded forests have lost the structure, function, species composition and/or productivity normally associated with the natural forest type expected at that site. Forests can be degraded through human activities such as unsustainable harvesting and through natural disturbances, which may be exacerbated by climate change.

Forest degradation can be monitored and measured using partial canopy-cover loss as a proxy (FAO, 2015). Other indicators or impacts of degradation include reduced growing stock, biomass, biodiversity, and production of forest goods, and increased soil erosion. From 2000–2012, the global area with partial canopy-cover loss was 185 million ha, most of which (over 156 million ha) took place in the tropical climatic domain (FAO, 2015; Figure 8).

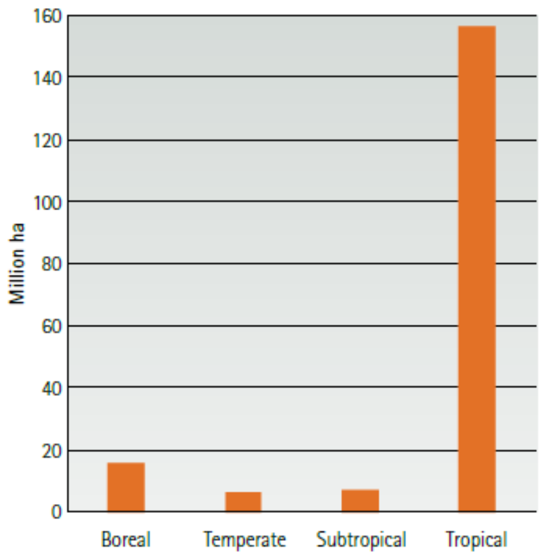


Figure 8. Estimated area with partial canopy-cover loss by climatic domain between 2000 and 2012 (FAO, 2015)

Forests are subject to a number of natural disturbances (e.g. wildfires, pests, diseases, adverse weather events) that can reduce their ability to provide the full range of goods and services. About 98 million hectares of forest were affected by fires in 2015 (FAO, 2020b). These fires occurred mainly in the tropics, where they affected about 4 percent of the forest area. Most fires are readily contained, but 10 percent of fires are not and these account for 90 percent of the burned area.

Disturbances other than fire affected 142 million hectares of forest between 2003 and 2012 (FAO and UNEP, 2020). In 2015, around 40 million hectares of forests were affected by such disturbances, mainly in the temperate and boreal zones (FAO, 2020b). Outbreaks of forest insect pests alone damage about 35 million hectares of forests annually. Invasive plant and animal species are now considered one of the most important causes of biodiversity loss, especially in many island countries. Higher temperatures, severe and extreme weather events and drought stress result in reduced vigour of trees, making them more vulnerable to outbreaks of native and introduced pests and diseases. Finally, more than 800 million hectares of forested area were destroyed or affected by weather disasters between 1996 and 2015 (FAO and UNEP, 2020).

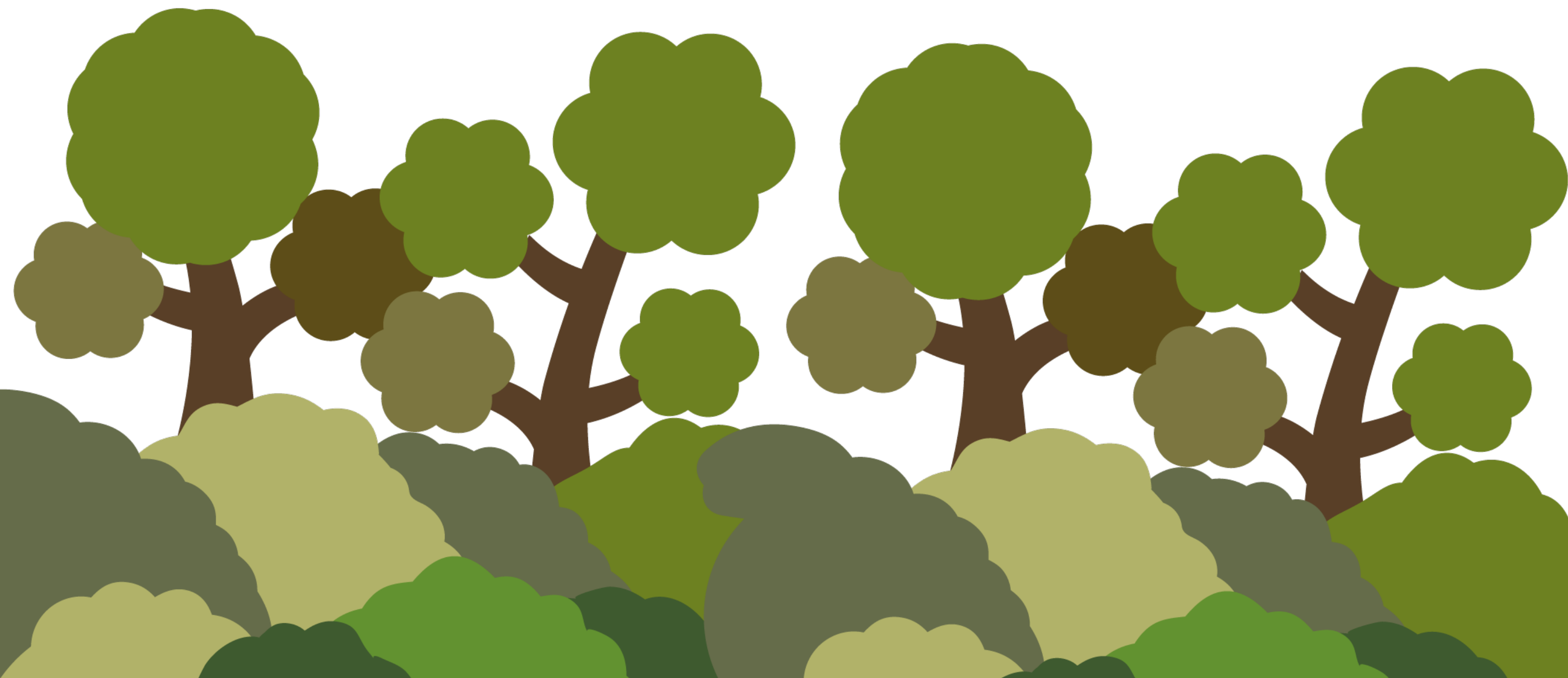
Increases in forest area may occur through natural processes such as regeneration of forest on abandoned agricultural land, or through reforestation or afforestation (including assisted natural regeneration; FAO and UNEP, 2020). For example, under the "Grain for Green" program initiated in 1999 to mitigate and prevent flooding and soil erosion, China planted 338,000 square kilometers of forests between 2013 and 2018 (Sheng, 2019). Forest and landscape restoration (FLR) is the process of reversing the degradation of soils, agricultural areas, forests, and watersheds thereby regaining their ecological functionality and improving their productivity and capacity to meet the various and changing needs of society (Besseau *et al.*, 2018). Under the Bonn Challenge, 57 countries, subnational governments and private organizations have committed to restore over 170 million hectares. The AFR100 African Forest Landscape Restoration Initiative aims to bring 100 million hectares of degraded land under restoration by 2030. These efforts will be bolstered by the declaration of 2021 – 2030 as the UN Decade on Ecosystem Restoration[1].

Photo 1. Old-growth montane temperate rainforest at Dakota Bowl, British Columbia, Canada

[1] https://www.unwater.org/the-united-nations-general-assembly-declare-2021-2030-the-un-decade-on-ecosystem-restoration/

Table 2. Related cases studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Soil fertility improvement of nutrient-poor and sandy soils in the Congolese coastal plains	Africa	7	6	1
Soil organic carbon stocks in forests of Singapore	Asia	Various	6	4
Reforestation of highlands in Javor Mountain, Republic of Srpska, Bosnia and Herzegovina	Europe	15	6	5
Natural afforestation of abandoned mountain grasslands along the Italian peninsula	Europe	23 to 72	6	6
Conservation of degraded forests of central and western Spain	Europe	22 to 80	6	9
Straw mulch and biochar application in recently burned areas of Algarve (Portugal) and Andalusia (Spain)	Europe	1	6	10



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Tropical moist forests

2. Tropical moist forests

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1. Definition and description

Different definitions for tropical moist forests (TMF) have been put forward by researchers and institutions. However, according to Myers (1980) there is not a universally accepted definition. Myers (1980) defines tropical moist forest as areas with ″evergreen or partly evergreen forests, receiving not less than 100mm of precipitation, in any month for two out of three years, with mean annual temperatures of 24+ °C and essentially without frost; in these forests some trees may be deciduous". Others (e.g. Raich *et al.,* 2006; Staal *et al.,* 2016) define TMF as forests that have dry seasons length of ≤ 3 months to 5 months. Whereas some authors distinguish between moist forests and rainforests others use the names interchangeably. However, rainforests and moist forests are co-distributed and often integrate rather than separate along abrupt boundaries (Staal *et al.,* 2016; Nave *et al.,* 2019).

Tropical moist forests are found on a wide range of soil types but are primarily found on soils that are highly weathered with high clay content and low in phosphorus. Predominant soils are Ultisols in Southeast Asia and, and Oxisols in Neotropics (Americas) and Africa (Fujii *et al.,* 2018). Many of TMFs are found near sea level, but others occur on high elevation such as montane and cloud forests (Holl, 2002). TMFs are the most diverse ecosystem and serve as home to more than 50 percent of the world's 5 –10 million species (Sommer, 1980; Holl, 2002; Thomas and Baltzer, 2002). Globally, the most important types of tropical moist forests identified include lowland evergreen rain forests, upper and lower montane rain forests, heath forests, peat swamp forests, freshwater swamp forests and mangroves (Thomas and Baltzer, 2002).

2. Global distribution of hotspot

Tropical moist forests are found within a band around the equator from 23.5 N (the Tropic of Cancer) to 23.5 S (the Tropic of Capricorn). The tropical moist forests occur in the equatorial zones of Africa, the Americas and Asia-Pacific, and in 2010 tropical humid forests were estimated to cover an area of 972 million ha, which was approximately 64 percent of the total tropical forest cover (Achard *et al.,* 2014).

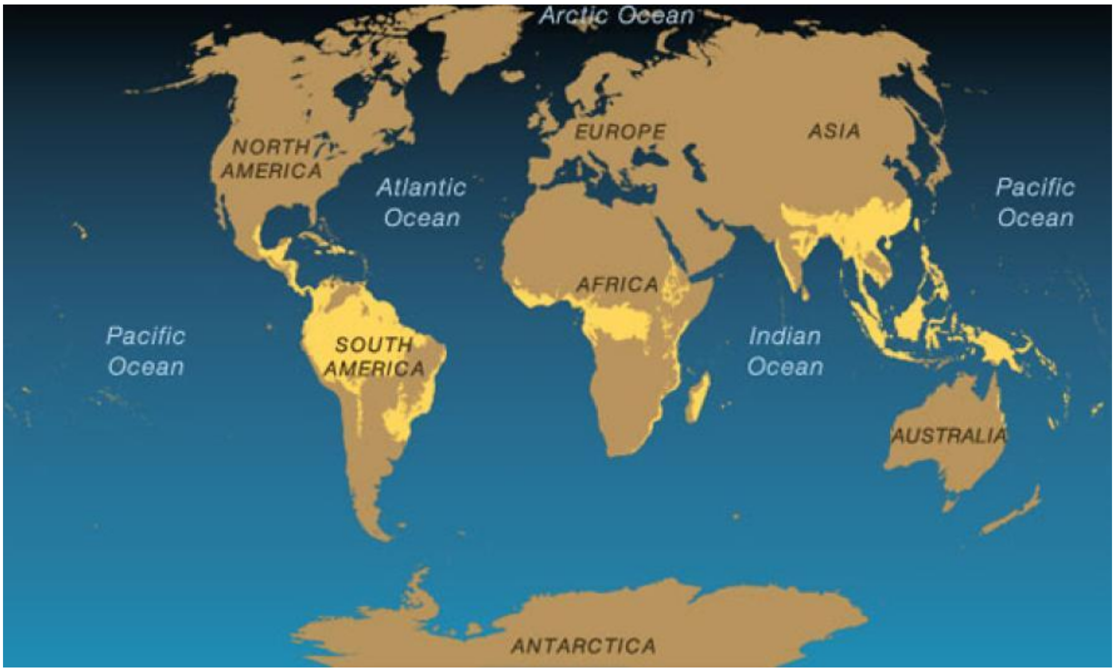

Figure 9. Location (area shaded yellow) of tropical moist forest (National Geographic/World Wildlife Fund)

3. Global carbon stocks and additional carbon storage potential

Soils of TMFs provide important goods and services that are relevant to human well-being, stability of ecosystems and the global climate. For instance, soils host myriad of micro-organisms, flora and fauna that are essential components of the Earth's biodiversity (FAO, 2017). Soil organic matter, and its largest constituent soil organic carbon play critical roles in the global carbon cycle (FAO and ITPS, 2015). Tropical moist forests store a large amount of carbon in the soil (about 30 – 60 percent of the total forest carbon); this may even exceed the carbon stored in the vegetation (Dixon *et al.*, 1994; Lal, 2005). In fact, these forests contain more carbon in the soil than soils under any other forest type in the world (Jobbágy and Jackson, 2000).

By analyzing existing SOC stock data in mature undisturbed tropical moist forests spanning 67 site observations from countries across Africa, Asia-Pacific, the Caribbean, Central and South America and North America, we estimate the average carbon stored in the 5 cm to 800 cm soil depth at 109.2 ± 10.19 tC/ha. The SOC estimates show thirty-five-fold differences, ranging from 9.9 tC/ha to 349.4 tC/ha (Appendix 1). In the last 40 years researchers have provided regional estimates for SOC stocks of tropical moist forests through synthesis, reviews and meta-analysis. For example, the average soil carbon storage in tropical moist and tropical wet forests was estimated to be 85 tC/ha and 115 tC/ha, respectively (Brown and Lugo, 1982). The above-mentioned soil carbon values are close to the average value that has been reported in this synthesis. The soil organic carbon stored in intact mature tropical moist evergreen forests found in areas where the elevation is higher than 500 m above sea level was estimated to be 166 tC/ha to 100 cm soil depth (Raich *et al.,* 2006), which is about one- and half times more than the average value the current synthesis found. Raich and his colleagues also reported that the SOC stock in the tropical forests they considered varied about tenfold, ranging from 31 tC/ha – 300 tC/ha. Soil organic carbon pool in predominant soils of tropical rainforest ecosystems, including Oxisols, Ultisols, Alfisols, and Inceptisols ranged from 120 – 123 tC/ha to 100 cm soil depth (Lal, 2005). The same study estimated the rate of organic carbon sequestration and the total potential of organic carbon sequestration in tropical forest soils to be 0.1 to 1 tC/ha/yr and 200 – 500 million tC/yr, respectively. A synthesis of tropical secondary forests soil carbon data from 81 studies estimated average SOC for 0-100 cm soil depth to be 164 tC/ha (Marín-Spiotta and Sharma, 2013). Another study reviewed soil carbon data of 17 dominant major soil types (IUSS WRB Reference Soil Groups) and reported that an average SOC pool of 193.3 tC/ha to 100 cm depth for major soil types (Acrisols, Ferralsols, Lixisols and Nitisiols) in tropical rainforests (Nave *et al.,* 2019). Other global studies that have synthesized soil organic carbon data provided a range of 115 – 210 tC/ha for tropical moist and wet/rainforests (Post *et al.,* 1982; Dixon *et al.,* 1994; Jobbágy and Jackson, 2000).

It could be observed that the soil organic carbon values that are reported by researchers for tropical moist forests have been different. This could be a result of several reasons, key among them are:

- the lack of consistency in the way "*Tropical moist forest*" is defined; there is no consensus on the definition for the term as explained under Section 1;
- researchers use different soil depth thresholds to assess soil organic carbon (see Table 3). There is no standard soil depth for reporting soil carbon. In many studies, the most common depth is 0 – 30 cm (Raich *et al.,* 2006). Researchers have tried to circumvent this problem by standardizing the data to 100 cm depth (e.g. Raich *et al.,* 2006). In recent times, however, there have been calls to use standardized methods to assess vertical distribution of soil carbon stocks at all relevant scales and to refine the methodologies for reporting on soil carbon (Bispo *et al.,* 2017; Smith *et al.,* 2020); and
- data coverage has increased to cover tropical regions which were hitherto underrepresented in such pantropic studies (e.g. Africa). As an example, the database for the current synthesis included 31 observations from Africa, which represents 46 percent of the data.

When the current database was separated into the different tropical regions (Africa, Americas and Asia-Pacific), there were large differences in the soil organic estimates within a region but the differences in organic carbon among the regions was not that wide (Table 3). The average soil organic carbon was 114.9 tC/ha, 116.5 tC/ha and 87.1 tC/ha for Africa, the Americas, and Asia-Pacific, respectively (Table 3). When regional soil organic carbon values are compared with estimates from other studies, the average organic carbon estimate for Africa was higher than the estimate of 57 tC/ha to 30 cm depth reported by Henry and his colleagues for African tropical and subtropical moist broadleaf forests (Henry *et al.,* 2009). The average soil organic carbon estimate for Asia-Pacific is slightly higher the value (50 tC/ha) reported by Abu Bakar (2000) for tropical dipterocarp forests in Malaysia.

Table 3. Soil organic C stocks (tC/ha) estimates (mean + standard error) for undisturbed mature tropical moist forests in Africa, the Americas and Asia-Pacific

Tropical Region	SOC Range (tC/ha)	Mean SOC (tC/ha)	N	Soil depth range (cm)
Africa	9.9–349.4	114.9 ± 15.9	31	10–100
Americas	20.2–330	116.5 ± 18.4	21	5–800
Asia-Pacific	27.5–300	87.1 ± 18.5	15	30–300

N = Number of sites; Africa includes sites in Madagascar, Americas consist of sites in the Caribbean, Central America, Hawaii, and South America; Pacific; sites in Papua New Guinea

Land use change in tropical moist forests can alter soil carbon stocks, which may affect concentrations of carbon dioxide (CO_2) in the atmosphere. To understand the impact that converting tropical moist forests to other land uses have on SOC, data was collated from 54 publications on changes in soil organic carbon stocks after the conversion of mature undisturbed tropical moist forests to three land use categories of crop lands, pastures and plantations from countries across Africa, Asia-Pacific, the Caribbean, Central and South America and North America (Appendix 2). On average, land use changes reduced soil carbon stocks by about 9 percent. However, when the data was separated into the different land use categories and examined, there was an increase in conversion to pastureland and reduction in conversion to cropland and plantations. Soil organic carbon increased by 8.1 percent after forest was converted to pasture (Table 4). Soil organic carbon declined by 12.9 percent after the conversion from natural forest to plantations (including tree plantations and palm oil plantations), and the highest loss was from forest to cropland by 22.2 percent.

Table 4. Change in soil carbon stocks after the conversion of undisturbed forest to cropland, pasture and plantation

Land use type	Changes in Soil Organic Carbon stocks (%)
Forest to cropland	-22.3 ± 6.55
Forest to pasture	8.1 ± 3.21
Forest to plantation	-12.9 ± 9.67

Note: Values in parentheses following the mean are bootstrapped 95 percent confidence intervals.

The changes in SOC reported here are close to values reported in other reviews and synthesis on soil carbon loss following conversion from undisturbed forests to different land-use categories. Guo and Gifford (2002) used data from 72 publications to assess the influence of land use change on SOC reported declines in SOC after conversion of forests to plantation by 13 percent, and forest to cropland by 42 percent, a difference of about two fold compared to the average value for conversion to cropland found in this synthesis. The authors also reported an increase in soil carbon stocks in pastures (8.1 percent). A global study of SOC changes for all major land use types in the tropics from 385 studies estimated that highest SOC losses was caused by the conversion of primary forest to cropland (25 percent) and perennial crops (30 percent) (Don, Schumacher and Freibauer, 2011). Another global study on soil carbon losses after conversion of forests to agricultural lands, estimated higher declines in tropical lands (41 percent) and temperate region (52 percent) compared to boreal regions (31 percent) (Wei *et al.,* 2014). In their review of soil carbon changes in 14 land-use transitions in the tropics, Powers *et al.* (2011) reported that while the conversion of forests to shifting cultivation and permanent croplands reduced soil carbon by 15.4 percent and 18.5 percent, respectively, conversion of forests to pastures and pastures to secondary forests increased soil carbon stocks. However, other studies reported a decline in carbon stocks when pasture is converted to secondary forests (Guo and Gifford, 2002).

Generally, the decline in carbon stock from the conversion of matured forests to croplands seems to be consistent with results reported by other researchers across the tropics despite discrepancies in the magnitude of change. Nonetheless, the potential for this trend to be reversed is remarkable for croplands. For example, in a meta-analysis of the impact of afforestation on croplands, Laganière, Angers, and Paré (2010) reported a 26 percent increase in SOC stocks following afforestation of croplands. Unlike croplands, the change in carbon stock values reported for other land-use transitions (e.g. pasture to secondary forests) have been variable. This variability could be due to differences in sampling procedures and methods, and underrepresentation of categories of land use in some tropical regions. For instance, in the current database changes in stock after the conversion of forests to pasture was dominant in the Americas but least represented in Africa (Table 5). This underrepresentation precludes extrapolating land use effect changes on soil carbon stocks to regional or global scales. In a review of 837 observations from 80 studies across the tropics on changes in carbon stocks following land use conversion, Powers *et al.* (2011) found that mean annual precipitation and clay soil mineralogy were key influencing factors but cautioned against extrapolating average values of changes in SOC due to geographic bias unless the distribution of field observations corresponds to the distribution of biophysical conditions.

Table 5. Change in soil carbon stocks in land use types across Tropical Regions

Land use	Africa	Americas	Asia-Pacific
Forest to cropland	-33.7 ± 10.95	-18.5 ± 7.64	-35.2 ± 7.67
Forest to pasture	-	7 ± 3.37	14.9 ± 9.44
Forest to plant	-17.7 ± 17.90	-10.4 ± 19.64	-7.2 ± 12.53

Note: Values in parentheses following the mean are bootstrapped 95 percent confidence intervals.

4. Importance of tropical moist forest conservation for the provision of specific ecosystem services

4.1. Minimization of threats to soil functions

Tropical forests are global host of the largest biodiversity and global carbon pool, which is vital for regulating the dynamic bio-geochemical processes and the exchange of greenhouse gases (GHG) with the atmosphere (Smith *et al.,* 2016; Eiserhardt, Couvreur and Baker, 2017). They deliver services locally such as provision of clean water, shelter, food, and fuel to local societies. They are important for soil conservation; through the prevention of soil erosion, soil salinization and alkalinization, soil contamination/pollution, soil acidification, soil biodiversity loss, soil compaction and soil water management. Tropical moist forest conservation prevents the movement of land mass because tree roots hold soil together, stabilize hills and mountains slopes which provides mechanical structural support to prevent shallow movement of land mass. Maintenance of forest cover through good forest management will reduce surface run-off and reduce the risk of erosion.

Soil provides nutrients such as Nitrogen (N), Phosphorous (P) and Potassium (K) that support biomass production essential for the supply of food for human and animal as well as energy and fiber (Smith *et al.,* 2016). Tropical moist forest play an important role in the cycling of these nutrients through different processes such as nutrient uptake and storage in vegetation perennial tissues, litter production, litter decomposition, nutrient transformations by soil fauna and flora and nutrient inputs from the atmosphere and the weathering of primary minerals (Foster and Bhatti, 2006). Forest conversion through clearing for other land use can negatively affect these processes and reduce the availability of these nutrients for root uptake (Foster and Bhatti, 2006). The balance between evapotranspiration and precipitation is important to avoid seasonal water deficit in forest landscapes. This balance is needed to ensure sufficient leaching in the soil to move salt from the soil profile (Schofield and Kirkby, 2003). Tree clearing disrupts the hydrological balance and lead to a buildup of salt in the subsoil (Schofield and Kirkby, 2003).

Soil pollution and soil contamination are generally used interchangeably. FAO and ITPS (2015) defines soil pollution as the “presence in the soil of a chemical or substance out of place and/or present at a higher-than-normal concentration that has adverse effects on any non-targeted organism” In contrast soil “contamination occurs when the concentration of a chemical or substance is higher than would occur naturally but is not necessarily causing harm” (FAO and ITPS, 2015). Soil pollution as a result of activities in tropical forests such as mining, excessive use of agrochemicals (e.g. pesticides and fertilizers) by small holder farmers in forested areas can lead to contamination or pollution of the soils with elements such as arsenic, lead, and cadmium (Rodríguez-Eugenio *et al.,* 2018). Pollutants have the potential to affects soil biodiversity (soil microorganisms and larger soil-dwelling organisms) and affect the services that these organisms provide (FAO, 2017). Loss of forest cover through land clearing for agriculture negatively affected the responses of 60 percent of soil macrofauna and 51 percent soil microbial community attributes (i.e. abundance, biomass, richness, and diversity indexes) (Franco *et al.,* 2019).

Tropical moist forest soils deliver important ecosystem services to humankind through regulation of the water cycle, enhancing water purification and water holding capacity and reducing the risk of soil erosion through run-off. The quantity of water, which a soil can store, depends on a number of factors including the thickness of the soil layer and its porosity, which are influenced by the quantity of soil organic matter, and the macropores shaped by biotic activity (Kirkham, 2014). The overall quantity of water in streams and rivers may increase in areas where there is less forest cover due to higher peak flows in the rainy season and after heavy rain events. In addition, the quality of water is influenced by the amount of forest cover in a watershed. In a study (De Mello *et al.*, 2018) conducted in southeastern Brazil in which the quality of water was compared between watershed with 55 percent forest cover and 35 percent forest cover, watershed with less forest cover showed less water quality (high values of solid turbidity, nutrients and coliform). Thus, conversion of forest to other land use causes potential change in hydrology, water availability and quality (Smith *et al.,* 2016).

4.2. Increases in production and food security

Tropical moist forest and its associated high biodiversity and the ecosystem services it supports are vital for achieving sustainable agriculture. Agriculture relies on a myriad of ecosystem services including pollination, maintenance of soil structure and fertility, biological pest control, nutrient cycling, and maintenance of hydrological systems (Power, 2010). It is estimated that 75 percent of the world’s leading food crops, benefit from animal pollination for production of fruit, seed, and vegetables (FAO and UNEP, 2020). Forests indirectly contribute to food production by providing suitable microclimate for specific food and cash crop production (Jamnadass *et al.,* 2015). In addition, many people, especially in poor communities depend of TMF for food security, their livelihoods and general well being. Forests are essential sources of non-timber forest products for most forest-depended local communities. It has been estimated that around 1 to 1.5 billion people across the world depend on wild foods including wild meat, edible plants products, edible insects, fish and mushrooms (Vira, Wildburger and Mansourian*,* 2015; FAO, 2020).

Indeed, plant species in TMF provide important source of food and nutrients and help forest-dependent communities to meet the four pillars of food security, namely, availability, access, utilization and stability of food security (FAO, 2020). In most fringe communities, trees in the forest provide a range food such as fruits, leafy vegetables, nuts, seeds and edible oils that help to diversify diets and enhance continuous flow of foods products (Jamnadass *et al.,* 2015). This is particularly useful during periods (e.g. during drought year) when farmers face inadequate availability and access to food, which increases the risk to achieving nutritional security (Jamnadass *et al.,* 2015, Amissah and Aflakpui, 2020).

4.3. Improvement of human well-being

Tropical moist forests are very important from both economic and ecological viewpoints. They provide a myriad of ecosystem services for many millions of people and contribute to human wellbeing (MEA, 2005). These services range from temperature regulation and air filtration to provision of food and medicinal plants. In addition, they are important locations for recreation, aesthetic appreciation, and stress relief for people who especially reside in urban areas, which contribute to the health of an increasingly urbanized population. Few studies have established the relationship between natural environment such as moist forest and human wellbeing. Visually attractive and preferred environments are perceived to promote good mental health because it improves people ability to face uncertainty (Ivarsson and Hagerhall, 2008). Tropical moist forests provide a haven for disease carrying animals such as mammals, birds and insects and help to prevent the spread of diseases from animal to humans (Zell, 2004).

4.4. Mitigation of and adaptation to climate change

Tropical moist forest is a significant store of carbon (Sullivan *et al.,* 2017). The highest densities of global forest carbon are located in forests of South America and Western and Central Africa, storing about 120 tonnes of carbon per hectare in the living biomass alone, which is above the global average of 75 tonnes per hectare (FAO, 2020). Tropical forest soils also store about equal quantity (30 percent) of carbon (More details in section [3]). Forest dead wood and litter also store 10 percent of terrestrial carbon. They are vital for stabilizing and reducing the concentration of carbon dioxide in the atmosphere through the process of photosynthesis and respiration (Houghton *et al.,* 2015). Given their global significance, as carbon sink reducing deforestation and degradation will be an effective means to mitigate and adapt to the effects of climate change.

5. General challenges and trends

Tropical moist forest loss accounts for 32 percent of global forest cover loss, with almost half of this figure occurring in South American rainforests (Hansen *et al.,* 2013). Previous statistics on tropical deforestation rate has differed among many studies due in part to varied methodologies used in different countries for assessing deforestation as well as the quality of data countries provide for conducting such assessments (Dupuis *et al.,* 2020). Tropical forests deforestation continues to occur at an alarming rate, especially in tropical moist forests (Giam, 2017). Forest degradation and deforestation due to land use change have been estimated to account for 12 – 20 percent of global human-caused greenhouse gas emission (Harris *et al.,* 2012). Deforestation has been attributed to a myriad of factors notably, land use conversion to agriculture, mining, infrastructure extension and urban expansion (Malhi *et al.,* 2013; Putz and Romero, 2015; Garcin *et al.,* 2018). Forest degradation is driven by factors such as fires and unsustainable harvesting of forest products.

Globally, agricultural expansion into new forest frontiers has been identified as one of the main drivers of forest loss (Curtis *et al.,* 2018; FAO and UNEP, 2020). Large-scale commercial agriculture has been estimated to account for 40 percent of tropical deforestation between 2000 and 2010. Local subsistence agriculture also accounted for 33 percent of deforestation within the same period (FAO and UNEP, 2020). Within the moist tropical forest areas and especially in tropical Africa, agriculture production is the mainstay of the local economy. In most cases, slash and burn is the most common farming practice used by farmers to prepare their land through by clearing forest areas to plant their crops but the fields are abandoned after few years (mostly 2 – 4 years, depending on regions) because of low crop yields and weed invasion (Pedroso-Junior, Adams and Murrieta, 2009). Globally, an estimated 35 million to 1 billion people depend on this system of farming for their subsistence (Filho, Adams and Murrieta, 2013). This system, which is the conversion phase of the traditional shifting cultivation, has been practiced for thousands of years (Garcia *et al.,* 2018). There have been contrasting views on the sustainability of slash and burn agriculture, and shifting cultivation in general and their impacts on the conservation of tropical forest ecosystems (Pedroso-Junior, Adams and Murrieta, 2009; Filho, Adams and Murrieta, 2013). In some instances, the practice is considered unsustainable and a major driver of deforestation, especially in areas where population has increased and fallow periods have substantially reduced (Mertz *et al.,* 2009).

Mining is a threat to the conservation of tropical forests worldwide. Resources extraction including mining is seen by many political elite in some countries as a pathway towards development (Bebbington *et al.*, 2018). There are mining booms in developing countries in Africa, Latin America and Asia with significant tropical forests cover that have weak mineral governance systems and there is often low capacity to enforce regulations controlling mining activities (Sonter *et al.,* 2017; Sonter, Ali and Watson, 2018; Hund, Schure and van der Goes, 2017). Surface mining is the dominant form of mining in developing countries, and the different phases of the mining operations including exploration, exploitation, processing and closure contribute to the forest loss (Hosonuma *et al.,* 2012; Hund, Schure and van der Goes, 2017; Sonter, Ali and Watson, 2018). There are also indirect impacts on forests through other infrastructural development such as buildings, urban expansion to support a growing workforce associated with mining operation further contribute to deforestation (Bebbington *et al.*, 2018). For instance, in the Brazilian side of the Amazon, 10 percent (11 670 km^2) of deforestation between 2005 and 2015 was attributed to mining (Sonter *et al.,* 2017).

Tropical moist forests are not naturally adapted to fire and traditionally do not experience frequent and intense annual fires because they are characterized by high annual rainfall. However, in recent decades due to climatic extreme events such as El Niño associated with climate change coupled with human activities, tropical moist forests have become more susceptible to fires, with increasing fire events (Cochrane, 2003; Dwomoh *et al.,* 2019). Fires have destroyed forest areas, degraded forests, reduce biodiversity, soil microorganism and delivery of ecosystem services and thus affect human well-being and livelihood (Bonan, 2008). In 2015, an estimated 98 million ha of forest was affected by fire, which was predominantly in the tropical area, where about 4 percent of the total forest area was burnt (FAO, 2020). A high proportion (more than two-thirds) of the total forest area burnt by fire was in Africa and South America (FAO, 2020) Reduce microbial activity that results immediately after fires reduce soil porosity and pH value, which affects plant growth (IUFRO, 2018). In addition, carbon storage and climate regulation potential of the forest is either or reduced lost (Bonan, 2008).

Industrial logging is an integral part of forest management in the tropics. Globally, about 20 percent (3.9 million km^2) of tropical forests was allocated to selective logging between 2000 and 2005 (Asner *et al.,* 2009). Forest areas allocated for selective logging continue to expand across all tropical regions, and particularly in African countries like Ghana and Gabon, nearly half of forest resources have been allocated for timber leases (Hawthorne and Abu-Juam, 1995; Asner *et al.,* 2005). Furthermore, illegal logging activities have become widespread in many tropical regions (Hansen and Treue, 2008; de Lima *et al.,* 2018). Globally, logging accounts for more than 50 percent of tropical forest degradation (Hosonuma *et al.,* 2012). Logging opens up the forests through the construction of access road and provide access for farmers and hunters to expand their activities. In some tropical areas, forests that have been degraded by logging become vulnerable to fires due to fuel build up and access (through road construction) and are eventually deforested after repeated accidental fires.

Despite the fact that logging often contributes to tropical forest degradation, sustainable forest harvesting can create jobs for rural communities and provide incentives for maintaining forest land use and preventing deforestation. Logged forests have the capacity to sustain conservation values and provide critical environmental services and functions, including biodiversity, water regulation and carbon stocks (Putz *et al.,* 2012). Over the years, there has been an improvement in the application of sustainable forest management principles and enhanced harvesting and other silvicultural techniques in many tropical countries (Rutishauser and Herold, 2017; FAO, 2020). Indeed, recent global assessment of forest resources and the monitoring of the Sustainable Development Goals show that there has been significant progress made towards sustainable forest management, with continuing but declining rate of forest loss, more forest areas being protected, more forest areas under long-term management plans, and more production forests certified under international standards (FAO, 2020).

Photo 2. Moist forest in Ghana. Top: Bobiri reserve; Bottom: Birim forest reserve

Photo 3. Wet forest in Ghana, Ankasa Conservation Area

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Wetlands

3. Wetlands

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1. Definition and description

Wetlands consist of highly diverse ecosystems characterized by periods of standing water, seasonally or permanently saturated soils, and vegetation adapted to growing in saturated or flooded conditions (Mitsch and Gosselink, 2007). Globally, there are numerous wetland classification systems describing a myriad of wetland types such as peatlands, mangroves, marshes, swamps, bogs, fens, kettles (potholes), and upland embedded (Finlayson and van der Valk, 1995; Tiner, 2003; Mahdavi *et al.*, 2018). Wetlands generally are segregated into two overarching categories: 1) coastal/tidal and 2) terrestrial/inland/non-tidal which can be further subdivided into organic- and mineral-soil wetlands, as well as by salinity classes (e.g., freshwater, brackish, saline) (e.g. Cowardin *et al.*, 1979; Hiraishi *et al.*, 2014).

While wetlands cover roughly 5 to 8 percent of the global land area, they represent approximately 20 to 30 percent of the organic carbon stored in terrestrial soils (Mitsch and Gosselink, 2007; Lal, 2008; Mitsch *et al.*, 2013; Amendola *et al.*, 2018). Among the various wetland types, peatlands represent the largest global soil organic carbon SOC pool (see Factsheets Nos. 11, 12 and 13, Volume 5, this manual) and mangroves are important blue carbon systems that incorporate internally originated carbon, as well as carbon deposited from outside the system via sedimentation (see Middleton and Ward, *this manual*). This disproportionate storage of SOC is largely due to the flooded and low-oxygen conditions, which contribute to slow decomposition rates in comparison to terrestrial ecosystems. However, when wetlands are drained, their soils become aerated and SOC stocks are rapidly decomposed and released to the atmosphere as carbon dioxide (CO_2).

In addition to their substantial existing stocks of SOC, wetlands also are recognized for their considerable potential to remove CO_2 from the atmosphere through high levels of primary productivity, which accumulate additional carbon to their SOC stocks, leading to long-term carbon sequestration (e.g. Euliss *et al.*, 2006; Lal, 2008; Mitsch *et al.*, 2013). The process of carbon sequestration occurs in natural, intact wetlands, but is also very relevant to restored or rewetted wetlands where SOC stocks have been diminished through human alterations such as drainage or soil tillage (Photo 4; also see factsheet No. 9 "Wetland conservation" and 10 "Wetland restoration" of this manual [Volume 5]).

Policy and management actions focused on restoring degraded wetlands have potential to mitigate human greenhouse gas (GHG) emissions, although in some wetland systems SOC accumulation rates can take many decades. Protecting SOC stocks of existing, undisturbed wetlands is an effective management strategy to limit emission of CO_2 to the atmosphere (Tangen and Bansal, 2020).

Although wetlands represent significant global SOC stores, the flooded and low oxygen environments of some (i.e. non-saline) wetlands favor production and emission of methane (CH_4), a potent GHG that contributes to climate change (Saunois *et al.*, 2020). While wetlands account for a large proportion of global CH_4 emissions from natural sources (Whiting and Chanton, 2001; Bridgham *et al.*, 2006; Mitsch *et al.*, 2013; Saunois *et al.*, 2020), CH_4 is short-lived. Eventually, CH_4 comes into equilibrium with the atmosphere (i.e. wetland CH_4 emissions are equal to CH_4 destruction/consumption) so that wetlands generally are long-term carbon sinks (Mitsch *et al.*, 2013).

2. Global distribution of hotspot

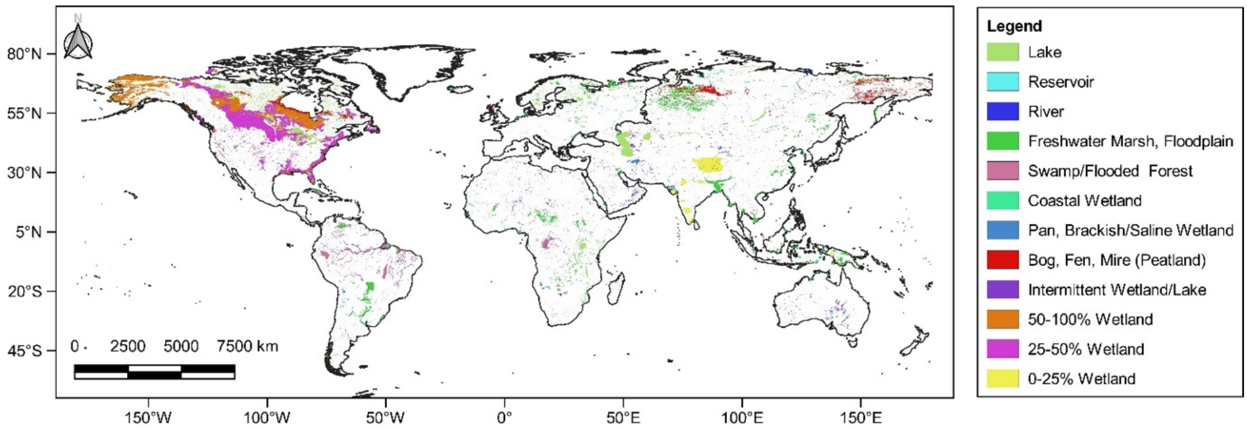


Figure 10. Distribution of global lakes and wetlands

Colors represent specific lake and wetland types, as well as wetland density classifications defined and presented by Lehner and Döll (2004).

The global lakes and wetlands database used to create this map was obtained from the World Wildlife Fund web site *(https://www.worldwildlife.org/pages/global-lakes-and-wetlands-database).*

3. Global carbon stocks and additional carbon storage potential

Table 6. Wetland soil organic carbon (SOC) stocks (mass or mass per area) and carbon sequestration (Cseq) rates (mass per year or mass per area per year)

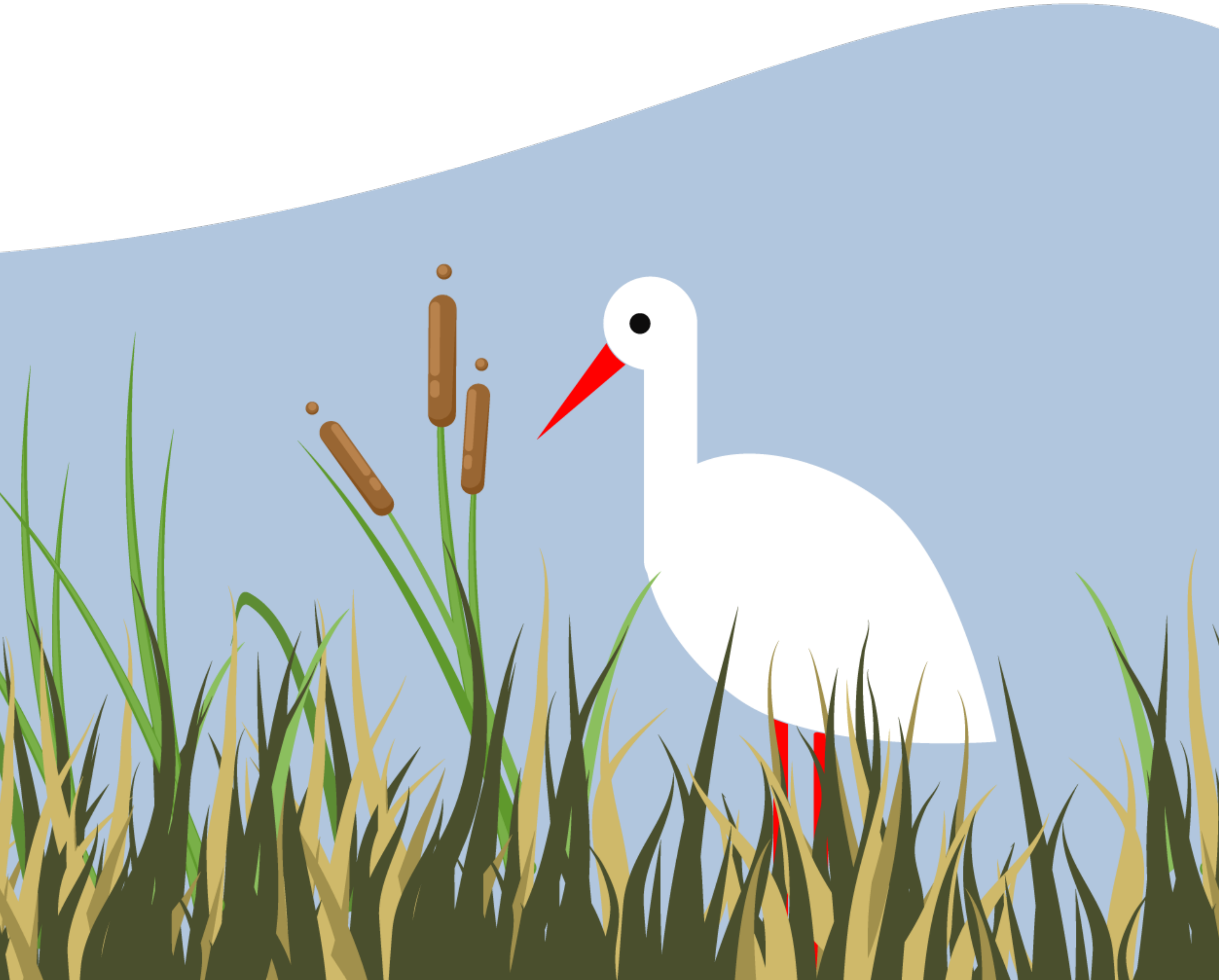
Location	Methodology	SOC stocks (Pg C)	SOC stocks (tC/ha)	Cseq Potential (Tg C/yr)	Cseq Potential (tC/ha/yr)	More information	Reference
Global	Stocks/ sequestration	120–646		137	0.3–1.8	Various wetland types. Estimates from literature reviews and modeling.	Scharpenseel (1993); Mitra *et al.* (2005); Bridgham *et al.* (2006); Köchy *et al.* (2015); Villa and Bernal (2018)
	Stocks		340–471			Coastal organic soil wetlands. Estimates from literature review.	Hiraishi *et al.* (2014)
			108–286			Coastal mineral soil wetlands. Estimates from literature review.	
			22–135			Inland mineral soil wetlands. Estimates from literature review.	
	Sequestration		237.4	42.6	2.1	Tidal wetlands. Estimates from literature reviews and modeling.	Chmura *et al.* (2003); Ouyang and Lee (2020)
	Stocks/ Sequestration	6.4		18.4–24	1.3–6.5	Mangroves Estimates from literature review.	Middleton and Ward, *this manual*; Chmura *et al.* (2003); Bouillon *et al.* (2008); Breithaupt *et al.* (2012); Alongi (2014); Hutchison *et al.* (2014); Hamilton and Friess (2018)

Location	Methodology	SOC stocks (Pg C)	SOC stocks (tC/ha)	Cseq Potential (Tg C/yr)	Cseq Potential (tC/ha/yr)	More information	Reference
	Stocks	600–644				Peatlands. Estimates from literature review.	Beer *et al.*, *this manual*; Yu *et al.* (2010); Leifeld and Menichetti, (2018)
Tropics	Stocks	200				Tropical wetlands. Estimates from literature review.	Neue *et al.*, (1997)
North America	Stocks/ Sequestration	161.0–215.0		57.2		Various wetland types. Estimates from literature reviews.	Bridgham *et al.* (2006); Kolka *et al.* (2018)
			48–82		0.8–3.1	Mineral soil wetlands (Prairie Pothole Region). Estimates based on measurements.	Euliss *et al.* (2006); Badiou *et al.* (2011)
		1.9				Tidal wetlands. Estimates from literature review.	Windham-Myers *et al.* (2018)
United States	Stocks	1.2–1.4				Tidal wetlands. Estimates from modeling.	Hinson *et al.* (2017)
China	Stocks	5–16.7	41.7			Various wetland types. Estimates from literature review, modeling, and measurements.	Zheng *et al.* (2013); Wang *et al.* (2014); Xiao *et al.* (2019); Han *et al.* (2020)

Here, we present global SOC estimates for wetlands in general, as well as for specific wetland types '(See 'Location' for scope of estimates). We also present regional SOC estimates to demonstrate specific hotspots of wetland SOC. The following SOC estimates are examples from a wide range of published literature, represent various wetland types and soil depth increments (e.g., 0-30 cm, 0-100 cm), and are not all-encompassing.

4. Importance of wetland conservation for the provision of specific ecosystem services

Wetland conservation is important because human activities can negatively impact wetland processes, functions, and ecosystem services provided to society. For example, wetlands often are drained to support urbanization, agriculture, forestry, grazing, and peat extraction, and the combined effects of these actions effectively eliminate the provisioning of standard ecosystem services by wetlands such as water and nutrient retention, wildlife habitat, biodiversity, and recreation (e.g. Millennium Ecosystem Assessment, 2005; Zedler and Kercher, 2005; Brinson and Eckles, 2011; Russi *et al.*, 2013). Ecosystem services provided by wetlands also can be diminished through hydrologic alterations associated with withdrawing surface water or groundwater for human uses such as irrigation. In addition to drainage and removal of water, wetlands can be affected by levees and dams, dredging and channelization, and water-level manipulation. Existing wetlands also can be degraded by activities associated with aquaculture or energy production. Therefore, preservation of existing wetlands, as well as restoration of degraded wetlands, is important for maintaining and enhancing wetland ecosystem services.



4.1. Minimization of threats to soil functions

Table 7. Soil threats

Soil threats	
Soil erosion	Human activities upstream or within watershed (e.g., drainage, agriculture, urban development) can result in erosion or sedimentation (e.g. Luo *et al.*, 1997; Gleason and Euliss, 1998; Craft and Casey, 2000; Gell *et al.*, 2009). Maintaining or reestablishing natural hydrology and vegetation (including buffers) can minimize these threats. Moreover, conservation of coastal or riverine wetlands can help prevent shoreline and bank erosion, as well as slumping (Millennium Ecosystem Assessment, 2005; Gedan *et al.*, 2011).
Nutrient imbalance and cycles	Natural microbial, plant, and animal communities of wetlands, as well as wetland soils, are responsible for cycling and storing nutrients such as N and P. The ability of wetlands to process nutrients results in downstream benefits through improved water quality (Howard-Williams, 1985; Bowden, 1987; Reddy *et al.*, 1999; Faulwetter *et al.*, 2009).
Soil contamination / pollution	Natural and created wetlands can act as landscape filters of contaminated/polluted waters by intercepting, processing/cycling (e.g., denitrification), and storing nutrients and contaminants in soils and biomass.
Soil biodiversity loss	Wetlands maintain soil biodiversity across the landscape by supporting microbial communities that are distinct from terrestrial and degraded systems.
Waterlogging	Wetlands can function as groundwater recharge sites.

4.2. Increases in production and food security

Wetlands provide water for human and livestock consumption (Photos 4, 5), as well as irrigation. Wetlands also provide food resources by supporting fish and wild game (e.g. Batt *et al.*, 1989), and facilitate the production of grains and other food. For instance, rice paddies cover over 1.5 million km^2 and provide grain for a large proportion of the world's population (Van Nguyen and Ferrero, 2006), and coastal wetlands support global fisheries (Barbier, 2019; Middleton and Ward, this manual). Moreover, populations of many developing countries rely on wetlands for subsistence agriculture, along with other ecosystem services (Silvius *et al.*, 2000; Irfanullah *et al.*, 2008; González-Marín *et al.*, 2017).

4.3. Improvement of human well-being

Wetlands provide numerous ecosystem services that benefit society (Millennium Ecosystem Assessment, 2005; Zedler and Kercher, 2005; Brinson and Eckles, 2011; Russi *et al.*, 2013). Wetlands can reduce impacts of flooding by storing and interrupting floodwater, improve water quality by removing excess nutrients and pollutants, and recharge groundwater. Wetlands provide a source of freshwater, as well as food, fiber, and goods produced from plants (e.g. Bansal *et al.*, 2019). In addition to supporting tourism and recreational activities, wetlands also help sustain biodiversity and provide wildlife habitat (Batt *et al.*, 1989; Szabo and Mundkur, 2017).

4.4. Mitigation of and adaptation to climate change

Wetlands store a large amount of carbon and generally are considered an atmospheric carbon sink. Despite being considerable sources of CH_4, recent analyses into the short lifetime of CH_4 in the atmosphere shows that CH_4 is oxidized in the atmosphere and in soils at comparable rates as it is produced. Therefore, wetland CH_4 does not contribute to climate warming after a sufficient time period (> 50 years; Neubauer and Megonigal, 2015) following the formation/restoration of wetlands. Wetland drainage results in the emission of CO_2 and nitrous oxide (N_2O) to the atmosphere, which can exacerbate global climate change (Tangen *et al.*, 2015). Conversely, wetland conservation can avoid CO_2 emissions and restoration can result in removal of atmospheric CO_2.

5. General challenges and trends

Effects of human activities (e.g., drainage, aquaculture, urbanization) on wetlands and provisioning of their ecosystem services are wide ranging. Wetland drainage, land use, and pollution affect wetland microbial and wildlife communities, vegetation, soils, and carbon and nutrient cycling (Holden *et al.*, 2004; Blann *et al.*, 2009; Kayranli *et al.*, 2010; Gleason *et al.*, 2011; Islam *et al.*, 2016; Zhang *et al.*, 2016; McGonigle and Turner, 2017; Minick *et al.*, 2019). The primary threat to wetlands globally is artificial drainage to make lands available for urban and agricultural uses (e.g., Reis *et al.,* 2017). Estimates suggest that greater than 50 percent of the global wetland area has been lost, with much higher percentages reported regionally (Junk *et al.*, 2013; Dahl, 2014; Davidson, 2014; Hu *et al.*, 2017; Li *et al.*, 2018). Among the seven global geographical units, Asia and North America have the greatest area of wetlands (Reis *et al.,* 2017). While North American wetland losses have slowed considerably due to policy and management, rates of Asian losses remain high because of pressures from a growing human population (Reis *et al.,* 2017). Wetland drainage lowers the water table and desiccates soils, resulting in loss of SOC as CO2 to the atmosphere by stimulating microbial respiration (Armentano and Menges, 1986; Maltby and Immirzi, 1993). Drainage also can shift wetlands from CH4 sources to sinks and can alter abiotic conditions and microbial communities that regulate N and P cycling (Howard-Williams, 1985; Bowden, 1987; Reddy *et al.*, 1999; Bridgham *et al.*, 2006; Faulwetter *et al.*, 2009).

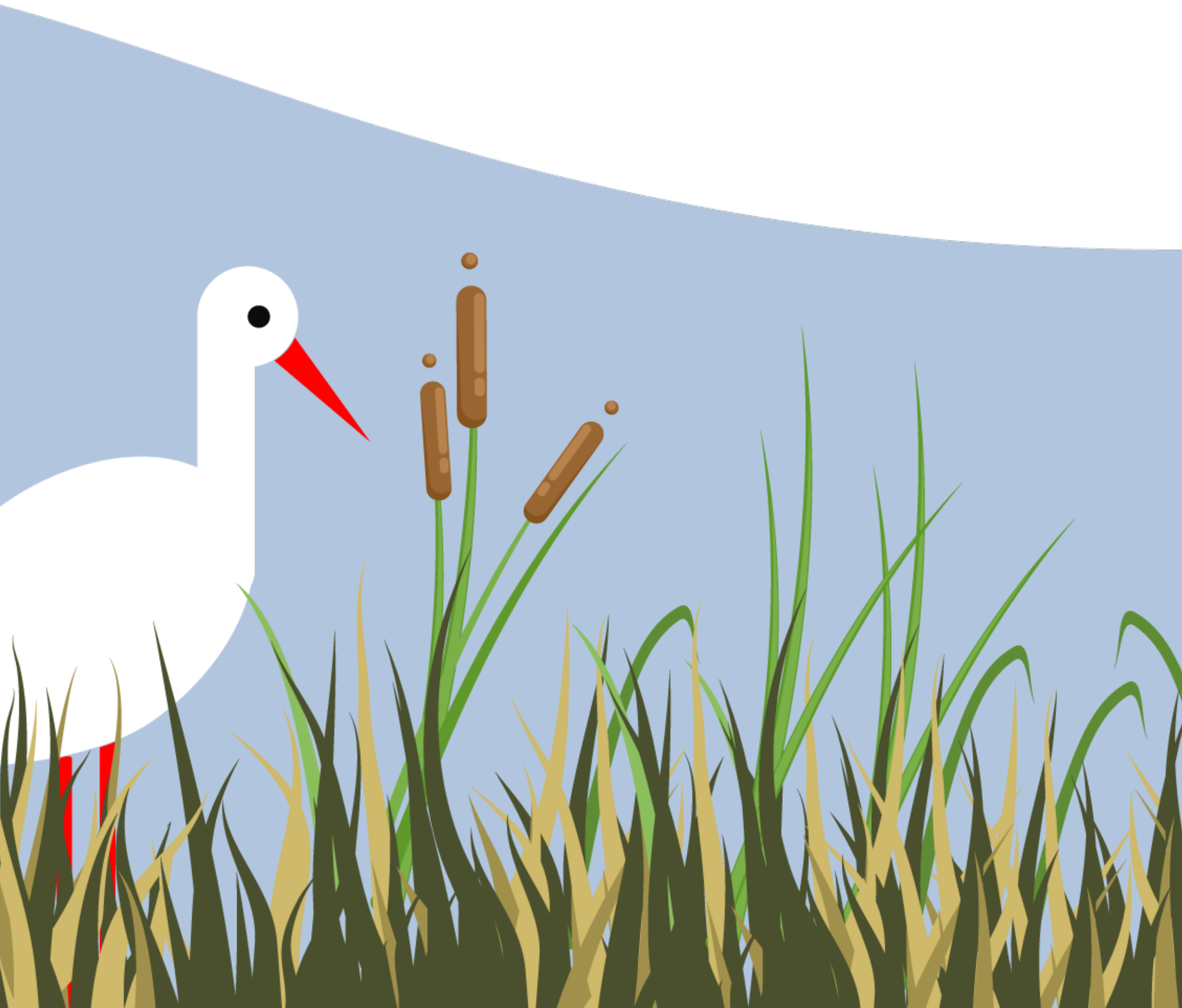
Among land use changes, human activities associated with artificial drainage often increase soil bulk density and compaction, with subsequent runoff and shifts in nutrient cycling (e.g., Fenstermacher *et al.*, 2016; Tangen and Bansal, 2020). Human activities also can result in contamination of wetlands by urban runoff, agricultural chemicals, and byproducts of energy production (Pascual-Aguilar *et al.,* 2015; Post van der Burg and Tangen, 2015; McMurry *et al.,* 2016; Schade-Poole and Möller, 2016). Aquaculture, specifically rice farming, can remove large amounts of potassium from wetland soils (Islam *et al.*, 2016). Salinization of wetland soils occurs from alterations to freshwater flows, land-clearance, irrigation, disposal of wastewater effluent, sea level rise, storm surges, and salts from road de-icing and oil drilling activities (Herbert *et al.*, 2015; Post van der Burg and Tangen, 2015). Salinization results in lower water quality, decreased carbon storage, and increased stress on wetland biota (Herbert *et al.,* 2015). High pH (>8.5) alkalinization of wetlands can follow hydrologic alteration designed to promote agricultural activities. Alkalinization reduces carbon uptake from impaired plant growth and increases carbon losses from elevated leaching and respiration (Jobbágy *et al.,* 2017). Wetland soil acidification can occur from increased sulfur deposition, desiccation from drainage or diverted water, increased groundwater acidification, and inputs from agricultural and mining runoff, which can negatively impact wetland fauna and flora (Lamers *et al.*, 1998). Additional threats to wetlands include nutrient and chemical pollution (Lee *et al.,* 2006; Verhoeven *et al.,* 2006; Post van der Burg and Tangen, 2015; McMurry *et al.,* 2016) and invasive species (e.g., Zedler and Kercher, 2004; Lavergne and Molofsky, 2006; Bansal *et al.,* 2019). Wetlands also can be affected by altered precipitation and temperature regimes, as well as by sea-level rise, associated with climate change (e.g., Cahoon *et al.,* 2006; Johnson and Poiani, 2016; Osland *et al.,* 2016; Gabler *et al.,* 2017; Chen *et al.*, 2018; Leng *et al.*, 2019).

Photo 4. Wetlands in a natural grassland (left) and cropland (right) setting in the Prairie Pothole Region (kettles) of central North America Natural wetlands store greater amounts of soil organic carbon than wetlands impacted by human activities such as drainage and tillage

Photo 5. Livestock in the Prairie Pothole Region of central North America utilize wetlands as a water source, as well as for grazing

Table 8. Related cases studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Management of Common Reed (Phragmites australis) in Mediterranean wetlands, Spain	Europe	Unknown	6	18
Preserving Soil Organic Carbon in Prairie Wetlands of Central North America	North America	Various	6	19
Maintenance of Marshlands in Urban Tidal Wetlands in New York City, United States	North America	100	6	31



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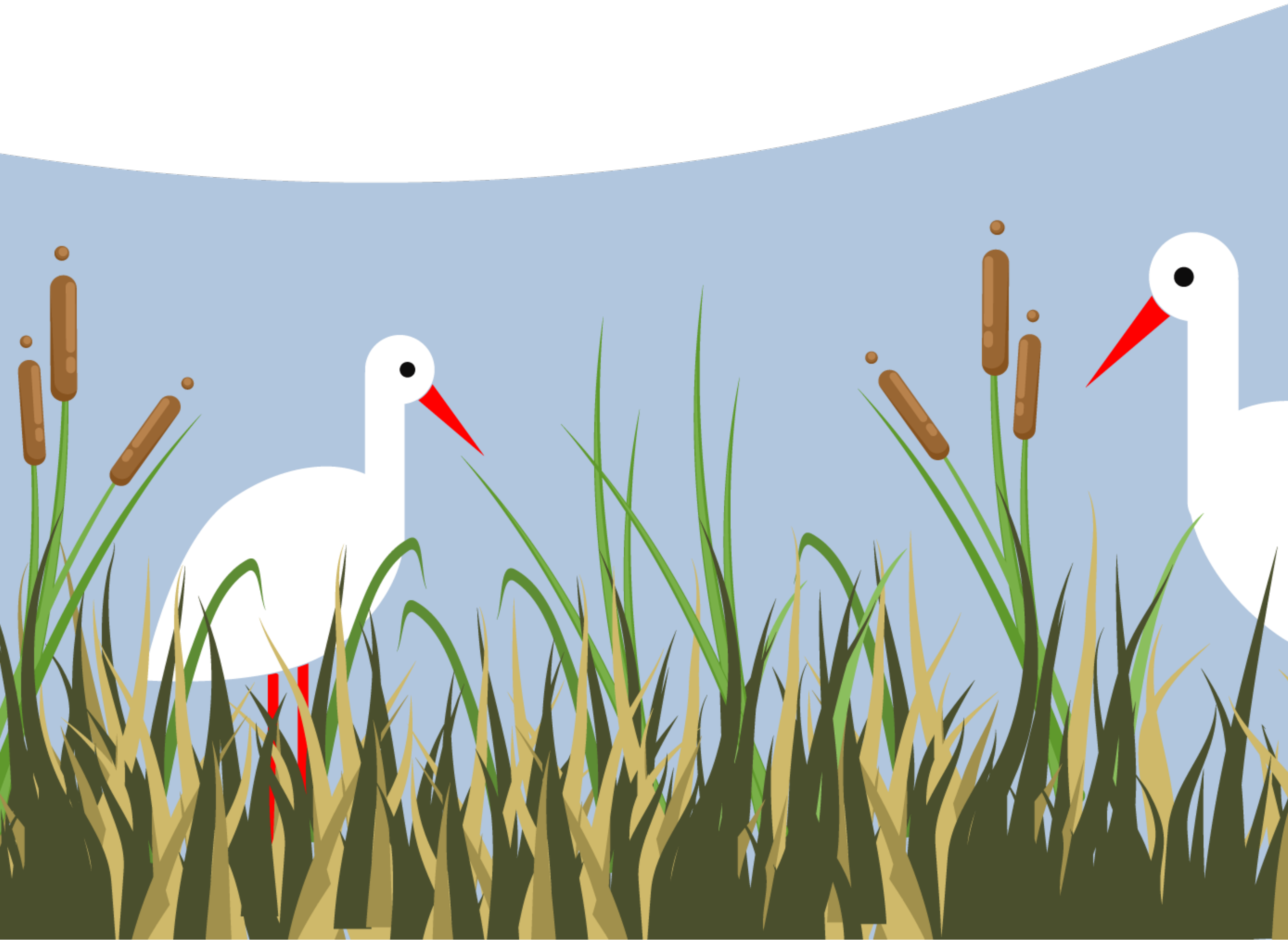

Peatlands

4. Peatlands

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1. Definition and description

Peatlands, also called organic soils, bogs, fens, swamps or mires, are the world's most carbon-dense terrestrial ecosystems and store most of the carbon in the soil (also called peat) (FAO, 2014). Peat soil – referenced in soil classification systems as Histosol (IUSS WRB, 2015) or included in organic soils (IPCC, 2014) – is composed mainly of organic matter from partially decomposed dead plant material that has accumulated in wet and oxygen-deficient soil conditions caused by a high water table. In their natural, water-saturated state, most peatlands are slow carbon sinks that sequester soil organic carbon (SOC), provide numerous ecosystem services (Figure 11), and are key in climate change mitigation and adaptation efforts.

Maintenance of the high water table level in a peatland is the key to avoid carbon losses and mitigate fire risk. Peatlands worldwide have been drained and degraded for agriculture, forestry, plantations, and for peat extraction for energy. Drainage or the artificial removal of surface and sub-surface water from an area, leads to lowering of the water table causing the drying of the peat soil. Drainage causes the exposure of the soil organic matter to oxygen and, when oxidized by microbial activity, the stored carbon is released as CO_2 and nitrous oxide (N_2O), while methane (CH_4) is emitted from the drainage ditches, which also transport dissolved organic carbon (DOC) out of the peatland.

Peatland degradation results in loss of biodiversity, subsidence, erosion, leakage of nutrients, and loss of hydrological properties (Joosten and Clarke, 2002; Silvius *et al.,* 2008) (Figure 13). The deeper the peat soil is drained, the higher the emissions (Couwenberg *et al.*, 2011), and the negative effects of peatland drainage are amplified by changes to the natural wetland vegetation that accumulates peat. Drained peatlands are prone to long-lasting fires that aggravate degradation and cause high carbon losses.

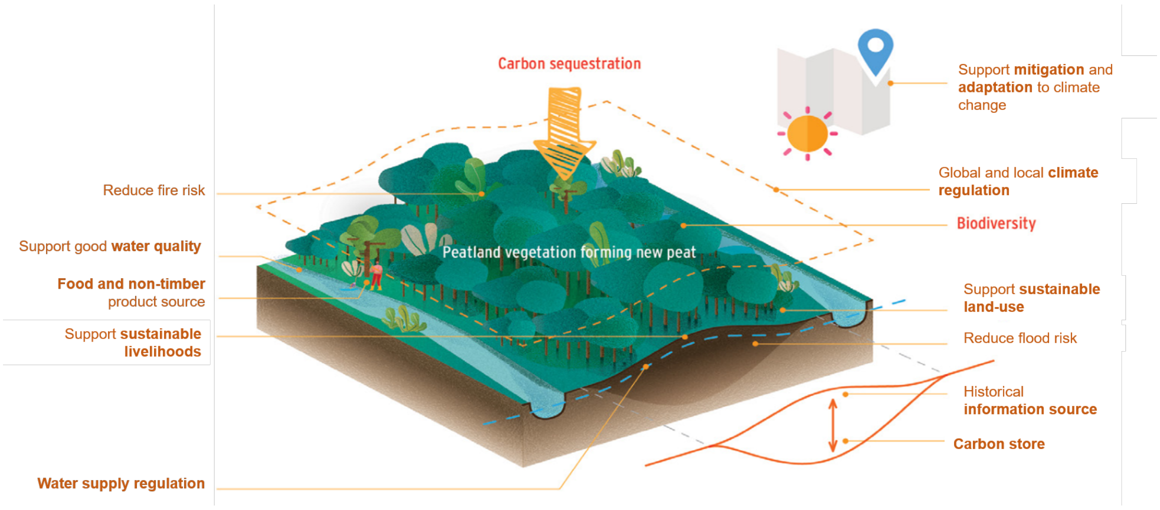

Figure 11. Ecosystem services provided by natural healthy peatlands (FAO, 2020)

2. Global distribution of hotspot

Peatlands exist in at least 180 countries (Parish *et al.*, 2008), in diverse climatic regions, altitudes and on all continents – tropical, boreal and temperate regions, in coastal, as well as inland and high mountains. Peatlands cover only three percent of the land surface – approximately 4.23 to 4.63 million km^2 (Figure 12) (Leifeld and Menichetti, 2018; Xu *et al.*, 2018).

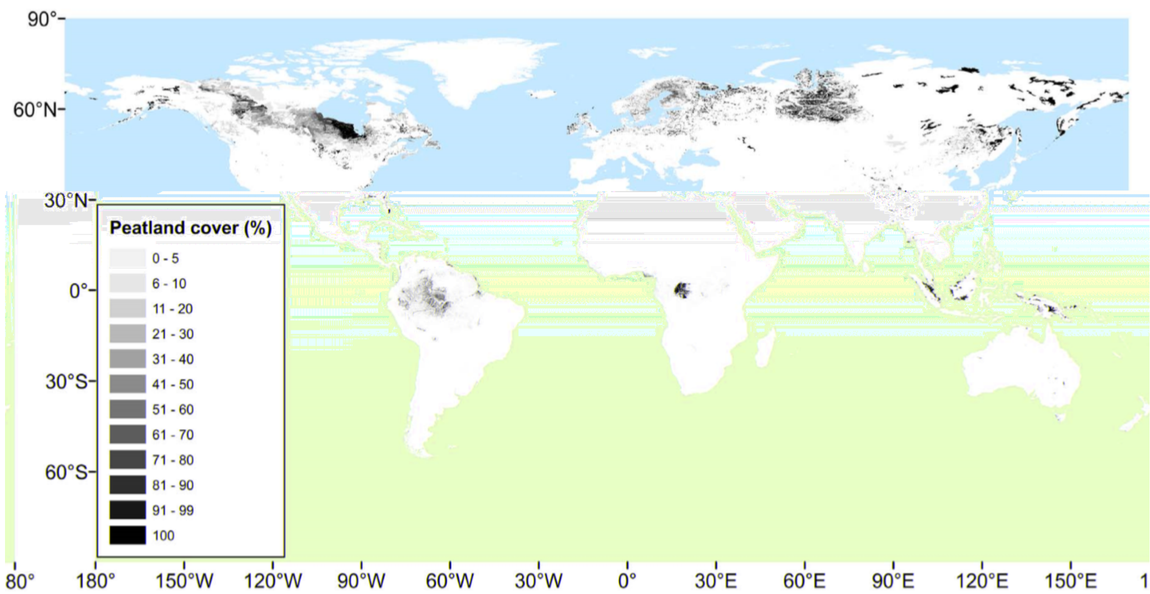

Figure 12. Global peatland distribution map derived from PEATMAP (Xu *et al.*, 2018)

3. Global carbon stocks and additional carbon storage potential

Peatlands, covering only 3 percent of the global land, store up to one fifth of the total global soil organic carbon stock – 600 – 644 Gt C (Leifeld and Menichetti, 2018; Yu *et al.*, 2010). A carbon stock that exceeds the carbon stored in the Earth's vegetation and may be equal to the carbon in the atmosphere (Turetsky *et al.*, 2015). The following estimates of C stock per ha should be taken only as examples (Table 9).

Table 9. Soil organic carbon stocks reported for peatlands

Location	C stock per unit area (tC/ha)	C stock total (over total peat depth) (GtC)	More information	Reference
Global		600–644		Yu *et al.* (2010); Leifeld and Menichetti (2018)
Northern boreal and sub-arctic regions	1 120	427–547	Includes peatland forests, estimates based on peat volume, carbon density and time history approaches	
Temperate regions	1 182	21.9	Time history approach	Leifeld and Menichetti (2018)
Tropical regions	2 850	104.7	Estimates calculated on different sources, C stock includes 124 (Dixon *et al.*, 1994) to 194 (IPCC, 2001) t C per ha of biomass	Dargie *et al.* (2017), Diemont *et al.* (1997), Immirzi *et al.* (1992)

4. Importance of peatland conservation for the provision of specific ecosystem services

Peatlands offer a variety of ecosystem services including regulation and production functions and shelter a distinct biodiversity (Figure 11). Peatlands contribute to global climate regulation through being a major sink of atmospheric carbon, and to local climate regulation by lowering temperatures during hot periods (Hooijer, 2005; Silvius *et al.*, 2008). Peatlands regulate water supply at a catchment level, securing drinking and irrigation water, and mitigate floods and droughts.

4.1. Increases in production and food security

Peatlands have a marginal agricultural capability because of their very high groundwater table, the low bulk density and bearing capacity, the high acidity and the low availability of nutrients in bogs. However, pristine peatlands provide many plant species that are utilized for food and fodder (Wichtmann *et al.,* 2016; Giesen and Sari, 2018). They are often significant sites for gathering berries, honey and mushrooms, hunting and fishing, representing a significant protein source for communities. Wet peatland management practices (see below "4.3" on climate change mitigation and adaptation, and Factsheets Nos.11, 12, 13 on Peatland management practices, Volume 5, this manual) allow reducing and avoiding CO_2 emissions, maintenance of SOC and support food security (Surahman *et al.,* 2018).

4.2. Improvement of human well-being

Pristine peatlands contribute to human well-being by lowering the risk of fires, regulating water supply and offering alternative livelihoods. In areas where peatlands are extensively drained, frequent fires threaten public health and the economy (Marlier *et al.,* 2019; The World Bank, 2016). Supply of drinking water in catchments dominated by peatlands depend on the management of these ecosystems (see e.g. Hooijer, 2005; Silvius *et al.*, 1984). The aesthetic and recreation values from peatlands and associated wetlands offer opportunities for income from ecotourism (Silvius *et al.*, 2008).

4.3. Mitigation of and adaptation to climate change

Conserved, restored and properly managed wet peatlands have a great climate change mitigation and adaptation potential. According to the IPCC (2014), conserving and managing peatlands in wet condition can avoid the emission of 0–20 tonnes CO_2eq /ha/year compared to conventional drainage-based peatland uses. Couwenberg *et al.* (2011) estimated that peatland conservation and paludiculture[2] in Central Europe can avoid the emission of 25–60 tonnes CO_2eq /ha/year, while undrained peatlands in Southeast Asia can avoid 70–117 tonnes CO_2eq /ha/year (Cooper *et al.*, 2020).

Conservation, paludiculture and wet management of peatlands boost adaptive capacity and help mitigate risks of extreme weather events like floods, droughts, and storms, especially in coastal peatlands. Land loss of riverine peatlands can be partly halted by avoiding drainage, while paludiculture and management practices can help diversify livelihoods and strengthen adaptive capacity (FAO, 2014). Wet-based extraction of timber and non-timber products – securing a high water table – are being developed to avoid drainage and consequent C losses (Wichtmann *et al.,* 2016).

5. General challenges and trends

Intact peatland ecosystems serve as carbon sinks, but when drained and degraded, they turn into long-term sources of GHG emissions (FAO, 2014), which continue until the peat is completely oxidized or until it is no longer drainable because subsidence lowers the peat surface until it reaches the water table level (Figure 13). The implementation of drainage-based land-use systems has often yielded short-term profits (Sumarga *et al.*, 2016) in exchange for long-term losses of ecosystem services and increased risk for neighbouring communities.

Draining, clearing of peat forests, plantations with fertilizer use, and land clearing by burning have led to a dramatic loss of SOC (Silvius *et al.,* 2008) and a range of other problems as per Figure 13. The current drained peatlands cover only 0.45 percent of the land surface but contribute at least 5 percent to the global GHG emissions (IPCC, 2014). Degradation also causes increased nutrient release from peat into water, and the reduction in the peatlands water-buffering capacity. In addition, drainage systems require constant maintenance, and intense fertilization is needed for production on peat (Hooijer, 2005). Additionally, due to the constant subsidence in combination with a rising sea level, increasingly large coastal peatland areas are going to be prone to regular, and partly permanent flooding during the next decades (Sumarga *et al.*, 2016; Hooijer *et al.,* 2015).

[2] Paludiculture produces biomass from wet or rewetted peatlands under conditions that maintain the peat integrity, facilitating peat accumulation and ensuring the provision of peatland ecosystem services. Also see Volume 5, factsheet n° 13

Eleven to 15 percent of global peatlands are estimated to be drained, mainly for cropping, forestry, grazing or energy use (FAO, 2020). The greatest areas of drained peatlands are in Europe and Southeast Asia (Crump, 2017). Since 2011, efforts by the scientific community, civil society and international organisations have increased (e.g. FAO, UNEP) awareness of the importance of peatlands. Heads of states, policy makers and agricultural producers have paid attention, although further actions are needed to stop wider-scale losses of SOC and other peatland services.

Peatland conservation and avoidance of drainage is encouraged by several global frameworks, conventions and multilateral environmental agreements. It is relevant for the fulfilment of the Paris Agreement and the Sustainable Development Goals (SDGs 6, 12, 13 and 15), the Ramsar Convention on Wetlands, the United Nations Framework Convention on Climate Change (UNFCCC), the Convention on Biological Diversity (CBD), The Sendai Framework for Disaster Risk Reduction 2015–2030 (SDFRR)[3], among other regional initiatives (FAO, 2020). These conventions state that peatland conservation is relevant to maintain vital ecosystem services and support human well-being, highlighting the importance of prioritizing this practice.

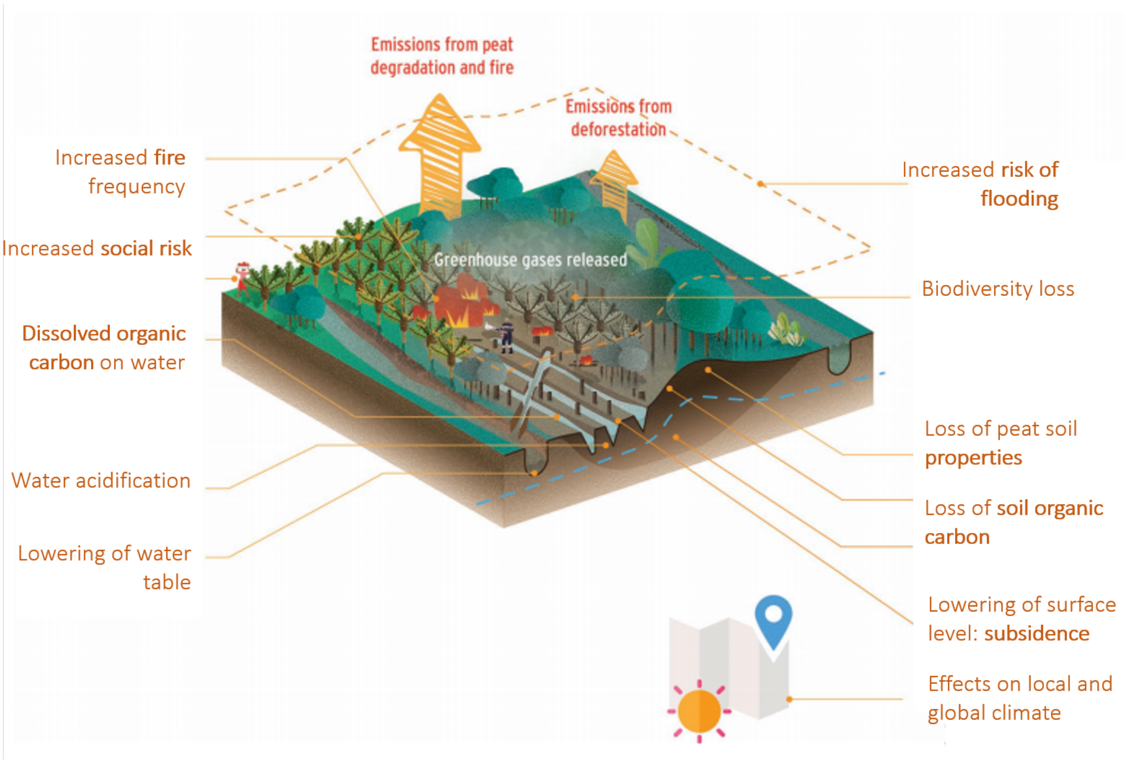

Figure 13. Effects of peatland drainage with canals to establish agriculture, plantations or other extractive activities (FAO, 2020)

[3] https://www.undrr.org/implementing-sendai-framework/what-sf

6. General recommendations

Targeted recommendations can be found in Factsheets 11 to 13 (Volume 5, this manual) related to specific practices on peatlands. Experts worldwide have agreed on the importance of:

- integrating peatlands into national policies and international monitoring and reporting frameworks to support peatland conservation, restoration and climate-neutral sustainable use at a landscape level (Factsheet No. 11, Volume 5 (this manual) on Conservation of pristine peatlands and avoiding drainage of peatlands);
- prioritizing peatland conservation strategies that involve communities and stakeholders (Factsheet No. 11, Volume 5 (this manual) on Conservation of pristine peatlands and avoiding drainage of peatlands), supported by policies, financial and legal mechanisms to safeguard natural peatlands from degradation;
- when conservation is not possible, facilitating the conditions for communities and stakeholders to transfer from drainage-based management to sustainable management practices, including knowledge and advisory networks, incentives, investment and consensus-based management approaches (Factsheet No. 12 on Restoration of peatlands (rewetting and revegetation), and 13 on Paludiculture);
- minimizing the loss of SOC and turn peatlands into carbon sinks again, water table levels should be established close to the soil surface (i.e. between 20 cm and 10 cm below the surface), and water-tolerating vegetation should be re-established and disseminated; and
- addressing research and knowledge gaps to generate data on peatlands' extent, contribution to SOC loss, effectiveness, costs and benefits of restoration, management and paludiculture, and explore competitive alternative crops to improve decision-making processes.

Table 10. Related cases studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Biomass from reeds as a substitute for peat in energy production in Lida region, Grodno Oblast, Belarus	Eurasia	Unknown	6	20
Sphagnum farming for replacing peat as horticultural growing media, Lower Saxony, Germany	Europe	10	6	21

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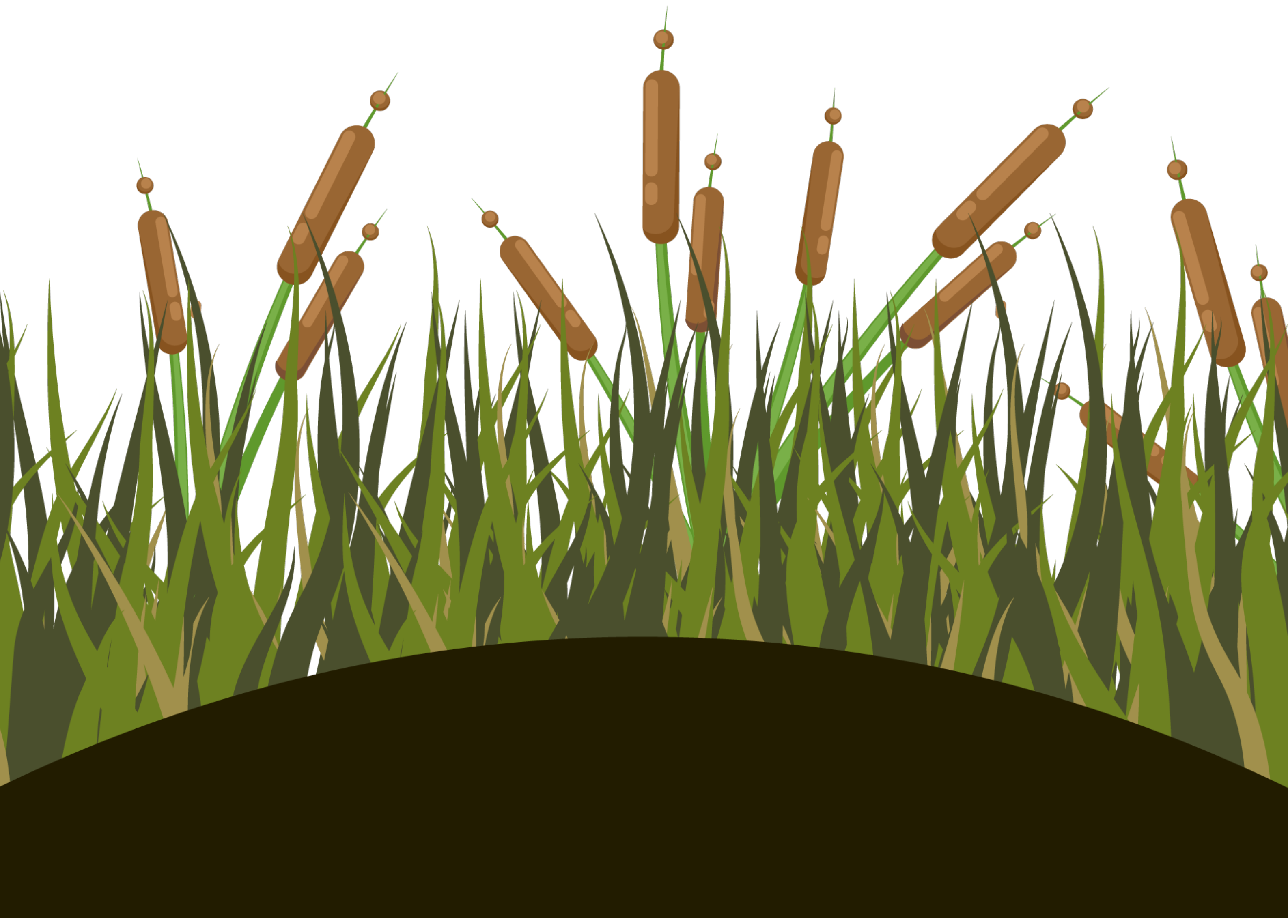

Mangroves

5. Mangroves

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1. Definition and description

Mangroves are forests with special assemblages of halophytic tree species, which occur along sub-tropical and tropical ocean coastlines in intertidal zones, estuaries and islands (Figure 14; Middleton and McKee, 2001). Their importance in carbon sequestration as well as their many other ecosystem services (e.g., fisheries production, nursery habitats, water filtering/detoxification, human livelihood, wildlife support, storm protection, timber, fuelwood, and food security; Barbier *et al.*, 2011) makes them priority areas for conservation and restoration (McLeod *et al.,* 2011). Mangrove forest can be very effective peat accumulators, with undecomposed root detritus building the foundation of certain islands such as in the Belizean Barrier Reef Complex (Middleton and McKee, 2001).

Globally, mangroves are among the world's most important blue carbon ecosystems because these capture carbon via primary productivity and sometimes sedimentation from outside their immediate setting (McLeod *et al.,* 2011). As much as 8-15percent of marine organic carbon burial may occur in mangrove forests (Breithaupt *et al.,* 2012). Mangrove forests are capable of storing 3 to 4 times as much carbon in the soil as other forest types (Sanderman *et al.*, 2018).

Most of the carbon in mangrove ecosystems is held in the top few meters of their organic soils (Donato *et al.,* 2011). While carbon held in their biomass can be substantial (Hutchinson *et al.,* 2014), recent research suggests that soil organic carbon stocks have been underestimated by as much as 50percent in soils with calcium carbonate and overestimated by as much as 86 percent in deltaic coastal settings (Rovai *et al.*, 2018). Mangrove area was lost at the rate of about 0.2–2.1 percent per year between 2000–2005 (Friess *et al.*, 2019), releasing 0.02 to 0.12 Pg carbon per year (Giri *et al.,* 2011), so that mangrove deforestation may contribute 10percent of total carbon emissions from loss of global forests (Donato *et al.,* 2011).

Through means such as aerial roots and viviparous embryos the trees in mangrove ecosystems have adaptation to survive saline and flooded conditions (Alongi, 2012). However, mangrove forests are relatively low in complexity compared to their freshwater counterparts, and include riverine, fringe, basin and dwarf (or scrub) forest types (Lugo and Snedaker, 1974). There are distinctly different mangrove species occurring in the New World vs. the Indo-West Pacific Region (Lugo and Snedaker, 1974; Duke *et al.*, 2008).

2. Global distribution of hotspot

Mangroves are forests that occur along sub-tropical and tropical ocean coastlines (Middleton and McKee, 2001)

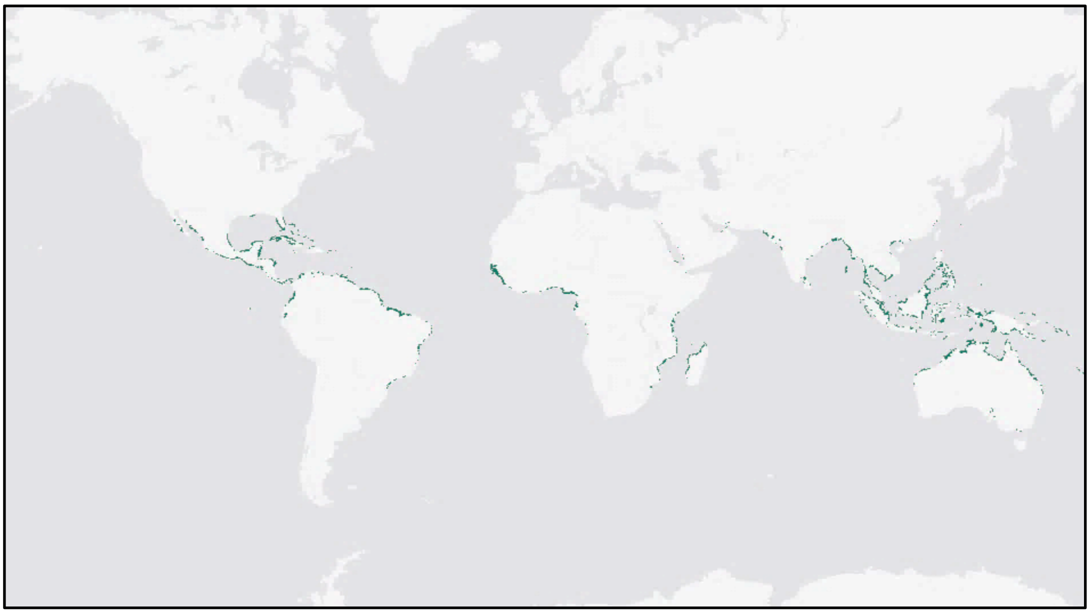

Figure 14. Global distribution of mangroves (dark green shading in coastal regions) (Giri *et al.*,2011)

General data license from UN WCMA Environment Program *(https://www.unep-wcmc.org/policies/ general-data-license-excluding-wdpa#data_policy; https://www.arcgis.com/home/webmap/viewer.html?useExisting=1&layers=62b6797f5091428fa89e10f7b3a1f73c)*

3. Global carbon stocks and additional carbon storage potential

Global carbon related to mangrove is substantial with significant storage in soil, and plant biomass (Hutchison *et al.*, 2014). Carbon emissions, especially methane, tends to be lower than in freshwater wetland types because of sulfate reduction (Windham-Meyers *et al.*, 2018), but mangrove forests are still viewed as carbon hotspots (Kolka *et al.*, 2018; Al-Haj and Fulweiler, 2020). Different land uses/covers have very different implications for 'blue carbon'[4] stocks of mangrove ecosystems, although regulations and assessments often lack this nuance (Friess *et al.*, 2020). Mangrove restoration can be an important avenue toward additional C storage (also see chapter: "Restoration of forest mangroves"). Table 11 provides some recent estimates of global C stocks in mangrove.

[4] Blue carbon is the carbon stored in coastal and marine ecosystems. Coastal ecosystems - mangroves, salt marshes and seagrass meadows - sequester and store large quantities of blue carbon in both the plants and the sediment below.

Table 11. Soil organic carbon stocks and annual C accumulation or emissions reported for mangrove forests

Location*	C stock per unit area (tC/ha)	C stock total (PgC)	Annual C accumulation or emission	Depth	More information	Reference
Global	-	6.4	-	Surface 1 m of soil	Global mapping, 30 m resolution data; Total soil C storage	Sanderman *et al.* (2018)
	Soil: 446.9 ± 175.4 Biomass: 244.2 ± 215.9	-	-	Various	Review; error is S.D.; C storage in soils and biomass	Hutchison *et al.* (2014)
	Mean: 515.9	-	-	Soil stocks to 1 m; 30 m pixels; five biomass eq.	C storage in soil and biomass. Biomass equations vary by latitude Map: https://dataverse.harvard.edu/dataverse/GMCSD	Hamilton and Friess (2018)
	-	-	24 TgC/yr	Various	Review; Total soil C accumulation	Alongi (2014)
	-	-	18.4 TgC/yr			Bouillon *et al.* (2008)
	-	-	1.74 tC/ha/yr			Alongi (2012)
	-	-	2.04 ± 1.53 tC/ha/yr		Review; error is S.D.; Total soil C accumulation	Hutchison *et al.* (2014)
	-	-	1.3 to 2.0 tC/ha/yr		Review; range is 95 percent CI; Total soil C accumulation;	Breithaupt *et al.* (2012)
	-	-	2.1 tC/ha/yr		Global review; Total soil C accumulation	Chmura *et al.* (2003)

Location*	C stock per unit area (tC/ha)	C stock total (PgC)	Annual C accumulation or emission	Depth	More information	Reference
Gulf of Mexico, US	-	-	2.0 – 6.5 tC/ha/yr		Review, loss on ignition (mostly); Total soil C accumulation	
Pacific and Indian Ocean	-	-	2.6 – 3.4 tC/ha/yr			
Herbert River Estuary, Australia	0.3-6.5 at individual plots	-	1.8 tC/ha/yr (mean);	0.8 to 1.6 m	SOC accumulation (not including calcium carbonate); Field surveys, grab samples and sediment cores	Brunskill *et al.* (2002)
Global	-	-	90–970 TgC/yr	Various	Review; Potential avoided loss by deforestation	Alongi (2014)
	-	-	0.01–0.02 tC/ha/yr		Review; non-normal distribution, Methane emission	Al-Haj and Fulweiler (2020)

The reviewed methodologies include a mix of approaches including loss on ignition for soil organic carbon, dry combustion, and methane emissions using chamber collection techniques.

1Pg = 1 000 Tg = 1 000 000 000 t.

*These systems form in positions along coasts and islands with latitudinal limits in the subtropics and tropics (20° C isotherm) where wave energy (Alongi, 2002) and freezing damage/mortality (Osland *et al.*, 2017) are low. The extent of mangrove habitat at a given location is a function of local rates of relative sea level rise, landform slope, tidal forcing, sedimentation rates and coastal subsidence (Feller *et al.*, 2017). Mangrove forests are also absent from arid coasts that receive little precipitation and freshwater input, where these ecosystems are replaced by hypersaline salt flats (Osland *et al.*, 2018).

4. Importance of mangrove conservation for the provision of specific ecosystem services

4.1. Minimization of threats to soil functions

Table 12. Soil threats

Soil threats	
Soil erosion	Coastal protection from tsunamis and storm surges (Alongi, 2012; Hutchison *et al.*, 2014; Sediment trapping (Kamal *et al.*, 2017)
Soil contamination / pollution	Water purification (Hutchison *et al.*, 2014); Shrimp farming can result in pollution to the pond from pesticides, antibiotics (Braun *et al.*, 2019) and nitrogen eutrophication (Burford and Longmore, 2001).
Soil acidification	Pond aquaculture can lead to acid sulfate soils (Alongi, 2002).
Soil biodiversity loss	Plant-soil-microbial relationships are understudied (Alongi, 2002).

4.2. Increases in production and food security

Mangrove forests provide support for timber and other valuable products as well as support for fisheries and biodiversity (Gunawardena and Rowan, 2005; Sathirathai and Barbier, 2001; Hutchison *et al.,* 2013). Also, mangroves provide food and fuel, and act as a nursery for semi-terrestrial and aquatic animals (Alongi, 2012). Humans extract various forest products including wood for fuel, construction, paper and fishing gear. Other non-wood products are collected for animal fodder, natural products (e.g. fish, crustaceans, honey, beverages, other food, drugs), and household items (e.g. clothing fiber, dye, incense) (FAO, 2007).

4.3. Improvement of human well-being

Mangroves support human well-being by delivering human necessities such as food, shelter and livelihood while contributing to the resilience of local communities (Barbier *et al.,* 2011).

4.4. Mitigation of and adaptation to climate change

Restored mangrove forests hold an equivalent amount of carbon as the original intact mangroves after 25-30 years (e.g. Cambodia; Sharma *et al.*, 2020), thus restoration programs have high potential in terms of carbon sequestration (Hutchison *et al.*, 2014) (Also see Factsheet No. 14, Volume 5, "Restoration of mangroves" of this manual). Mangrove forests have a role in coastal geomorphology via their high productivity, which contributes to peat formation and sediment deposition (Barbier *et al.*, 2011). Many mangrove restoration projects fail due to inappropriate species or site selection because of socio-economic limitations, but the number of large-scale successes has increased in recent years (Feller *et al.*, 2017; Friess *et al.*, 2020). Methane emissions may offset 8–20 percent of carbon sequestration and global warming potential in mangroves at the global scale at 20–100 year timescales (Rosentreter *et al.*, 2018). However, when calculating a total global warming potential, one must also consider avoided emissions, such as the reduced risk of carbon release from peat fires (Turetsky *et al.*, 2015) after hydrological restoration.

5. General challenges and trends

Mangroves are being lost and degraded because of land conversion for agriculture, urban development, and overexploitation of timber and food sources (fish, crustaceans, shellfish) (Alongi, 2012). Global mangrove deforestation rates during the late 20th century of 0.7 – 2.1 percent per year have decreased to 0.2 to 0.4 percent per year in the early 21st century (Friess *et al.*, 2019). Some estimates of overall mangrove loss between 1980 and 2005 are as high as 20 percent, with an annual decline of 1 – 2 percent (FAO, 2003) (see also the Global Mangrove Watch Viewer for specific annual breakdown of loss (Global Mangrove Alliance, 2020)). Regionally, nearly 80 percent of global mangrove loss between 2000 and 2016 was concentrated in Southeast Asia with conversion for agriculture and aquaculture as the most important deforestation driver (Goldberg *et al.,* 2020).

While deforestation rates have decreased from the late 20th century to the early 21st century, an unknown extent of mangrove area is degraded from hydrological alteration, urban pollution, and overharvesting of food and timber resources (Friess *et al.,* 2019).

Mangrove forests converted to other land uses may lose their ability to increase the elevation of their surface if undecomposed mangrove materials can no longer stay ahead of the rate of organic decomposition (McKee *et al.*, 2007). Conversion for agriculture and aquaculture is accompanied by loss of carbon stores from both biomass and soil (Friess *et al.*, 2020). Shrimp farming results in nitrogen eutrophication of ponds (Burford and Longmore, 2001).

Different land uses have very different implications for 'blue carbon' stocks, although regulations and assessments often lack this nuance (Friess *et al.*, 2020). Protecting mangroves from conversion to agriculture, including for rice, shrimp and oil palm cultivation, and restoring mangroves where biophysically and socio-economically feasible, can contribute significantly to reducing global CO_2 emissions from the land use sector (Lovelock *et al.*, 2011).

Photo 6. *Avicennia germinans* and *Rhizophora* mangle in the intertidal zone at Ding Darling National Wildlife Refuge, Sanibel Island, Florida, United States of America

Table 13. Related cases studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Mangrove restoration in abandoned ponds in Bali, Indonesia	Asia	10	6	17

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Black Soils

6. Black Soils

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1. Definition and description

Black soils are inherently productive and fertile soils that are critical for food production globally. Given favorable climatic conditions, these soils allow high crop productivity. However, inappropriate management practices of black soils can lead significant losses of SOC, decline in soil quality, and resulti in emissions of carbon into the atmosphere. Sustainable use and management of black soils toward maintaining or increasing SOC stock is crucial for ensuring global food security and mitigation climate change.

Black soils are mineral soils which have a black surface horizon enriched with organic carbon that is at least 25 cm deep. Two categories of black soils (1st and 2nd categories) are recognized. The categories are distinguished to recognize the higher value, and thus greater need for protection, of some soils (Category 1), while still including a wider range of soils within the overall black soil definition (Category 2) (FAO, 2019).

The first category of black soils (which are the most vulnerable and endangered, and need the highest rate of protection at the global level) are those having all five properties given below:

- The presence of black or very dark surface horizons typically with a chroma of ≤3 moist, a value of ≤3 moist and ≤5 dry (by Munsell colours);
- The total thickness of black surface horizons ≥25 cm;
- Organic carbon content in the upper 25-cm of the black horizons between ≥1.2 percent (or ≥ 0.6 percent for tropical regions) and ≤20 percent;
- Cation exchange capacity (CEC) in the black surface horizons ≥25 cmol/kg; and
- A base saturation in the black surface horizons ≥50 percent.

Most, but not all 1st category Black soils have a well-developed granular or fine sub-angular structure and high aggregate stability in the black surface horizons that are in a non- or slightly degraded state, or in the humus-rich underlying horizon which has not been subjected to degradation.

The second category black soils (mostly endangered at the national level) are those having all three properties given below:

- The presence of black or very dark surface horizons typically with a chroma of ≤3 moist, a value of ≤3 moist and ≤5 dry (by Munsell colours);
- The total thickness of the black surface horizons of ≥ 25 cm; and
- Organic carbon content in the upper 25-cm of the black horizons between ≥1.2 percent (or ≥ 0.6 percent for tropical regions) and ≤20 percent.

This category does not include the CEC and base saturation criteria of the first class.

2. Global distribution of hotspot

On a world-wide basis, soil scientists generally include as types of black soils, the Chernozems, Kastanozems and Phaeozems (WRB), Isohumosols from Chinese soil classification, and Mollisols of Soil Taxonomy (Liu, 2012). Although these are the main classes, other classes are included in the concept of black soils, such as soils with Chernic, Mollic, Umbric, Hortic and Pretic horizons.

Among the main soil types, four locations can be highlighted globally. Chernozems (Mollisols) occurring extensively in the central region of North America, across the central plains of United States and southern region of Canada. Kastanozems and Phaeozems appear as discontinuous belts, which extend across southeastern Europe and central Asia. The western belt begins in the sub-humid steppes of southcentral Europe and extends across Russia and into the eastern belt, which is best represented in northeast China (Isohumosols). The fourth major location corresponds to the Pampas of South America, covering most of central-eastern Argentina, most of Uruguay´s territory, and part of the southern region of Brazil. Thus, the of the world's 916 million hectares of black soils occur in three regions of the northern hemisphere and one region south of the equator, the Parana-La Plata basin of South America. The understanding of this distribution as well as the genesis, uses and management, and threats to these soils, is crucial given that, when considering the natural fertility and land use of the black soils, these four regions collectively form the one of main world's natural granary (Liu *et al.*, 2010).

3. Global carbon stocks and additional carbon storage potential

In the WRB system of classification, the soils identified as Chernozems, Kastanozems and Phaeozems are included in the concept of black soils. Although in natural conditions they have high organic carbon content, in reality large areas of these soils are now degraded.

The SOC stock evaluation by using GSOCmap and WRB classification provided a general C stock in the Black Soils, presented in a global level and including only the soil types: Chernozems, Kastanozems and Phaeozems (FAO, 2019; FAO, 2009). The results showed that total SOC stock of Black Soils is 54.8 Pg, with an average value of SOC stock of 66.4 t/ha (Table 14).

Table 14. Total SOC stock and mean SOC stock of black soils within 30 cm soil layer for the Chernozems, Kastanozems and Phaeozems classes in the WRB system

Soil Reference Group WRB	SOC stock (Pg)	Mean SOC (t/ha)
Chernozems	19.7	89.6
Phaeozems	18.2	62.2
Kastanozems	16.9	47.5
Total of Black soils	54.8	66.4

4. Importance of black soil conservation for the provision of specific ecosystem services

Black soils are the most productive carbon-rich soils and provide multiple benefits including ecosystem services, food production and security, human well-being, and climate change mitigation and adaptation (Figure 15).

4.1. Minimization of threats to soil functions

Black soils include those with abundant nutrients for crops´ growth and organic carbon as well as good physical properties. The distinctive characteristics of the first category of black soils are their dark-colored, humus-rich surface horizon, and the high base saturation (Eckmeier *et al.*, 2007). In addition, characteristics such as appropriate pH, adequate available nitrogen, potassium, and suitable levels of most micro-nutrients, allow Black soils to maintain or improve soil nutrient balance and cycling, when practices of sustainable management are used (Balashov and Buchkina, 2011; Zhang *et al.*, 2013). Black soils have good soil physical properties in terms of soil bulk density, soil aggregation, wet-aggregate stability, and water infiltration rate. Those characteristics allow these soils to regulate water supply in the field in terms of the mitigation of floods and droughts and of water quality (Balashov and Buchkina, 2011; Chen *et al.*, 2014). As carbon-rich soils they are a reserve of components such as sugar, amino acids and carboxylic acids, which are natural resources for growth of soil microbial community (Zhang and Han, 2015). Nutrients in black soils, such as nitrogen and phosphorus, also contribute to abundant soil biodiversity (Galloway, 2004).

4.2. Increases in production and food security

The high soil organic matter content, good soil fertility and physical structure of black soils makes them the most fertile and productive soils in natural conditions, and they are therefore intensively and extensively cultivated. Global analysis showed that out of the total land dedicated to growing crops, 19 percent of the farmland is currently comprised of black soils, and out of the total area covered by black soils, 62 percent is used as croplands (USGS, 2015; HWSD, 2009).

In Russia, among the 221 million hectares of agricultural lands, 60-70 percent is of soils with Chernozemic horizons (Avetov *et al.*, 2011). In Slovakia, black soils are covering an area of 474 885 hectares (Kobza and Pálka, 2017), which is approximately 20 percent of total area of agricultural soils in the country. In China, the total area of black soils is 35 million hectares (Liu *et al.*, 2012), black soils have been important food basket since 1950s, producing 15.9 percent, 33.6 percent, and 33.9 percent of rice, corn, and soybeans of whole China in 2014 (Bureau of statistics of China, 2015). In the United States, black soils (Mollisols) cover about 196 million hectares, 36.9 percent of which are used for livestock and crop production (Wickham *et al.*, 2014). Most of the Mollisols in South America are used for grain and oilseed crops, orchards, forage, and crops for fiber production. They are also used for cattle raising and dairy farming, feeding the cattle with grains, forage crops or natural pastures (Durán *et al.*, 2011).

4.3. Improvement of human well-being

Black soils contribute to human wellbeing by providing nutritious food, enriching folks culture and offering alternative livelihoods. Multiple nutritious foods are produced in black soils region globally including cereals, beans, meat, etc. In Brazil, the contribution of pre-Colombian indigenous communities, which for hundred years cultivated the low lands in the Amazon region, adding materials such as charcoal, fish bones, with organic matter, formed fertile soils now called Amazonian Dark Earths (Schmidt *et al.*, 2014; Anne, 2015; Kern *et al.*, 2019). In northeast China, people associate the black soils with a symbol of healthy and positive characters to enhance the value of their personality, products and culture (Cui *et al.*, 2017). The aesthetic and recreation values of black soils also offer opportunities for increasing income of farmers.

4.4. Mitigation of and adaptation to climate change

Black soils have a high potential to mitigate climate change due to their inherently high SOC content. For example, according to the results of Global Soil Organic Carbon map (GSOCmap), average SOC stock of Black soils is 66.4 t/ha in top 30 cm, which is higher than the average of SOC stock in all soil types (57.34 t/ha) (FAO and ITPS, 2019). Although at the global level there is to date little information on potential GHG emissions from black soils, it is known that black soils are both extensively and intensively farmed (cereal, pasture, range and forage system) resulting in significant losses of organic carbon. According to various estimates, black soils lost 20 to 50 percent of SOC in 50 to 100 years after conversion from natural system to intensive farming system. For example, in the United States of America, in intensive continuous corn cropping system, SOC decreased by more than 50 percent in 100 years (Gollany *et al.*, 2011). The significant losses of SOC in black soils are typically the result of inappropriate land use and poor management practices, leading to a decline in soil quality and soil structure as well as increased soil erosion, resulting in emissions of carbon into the atmosphere. On the other hand, appropriate land use and soil management can lead to an increase of SOC and improved soil quality and multiple benefits (Figure 15) that can partially mitigate the rise of atmospheric CO_2 in black soil regions (Liu *et al.*, 2012). In conclusion, sustainable use and management of black soils toward maintaining or increasing SOC stock could be crucial for climate change mitigation and adaptation.

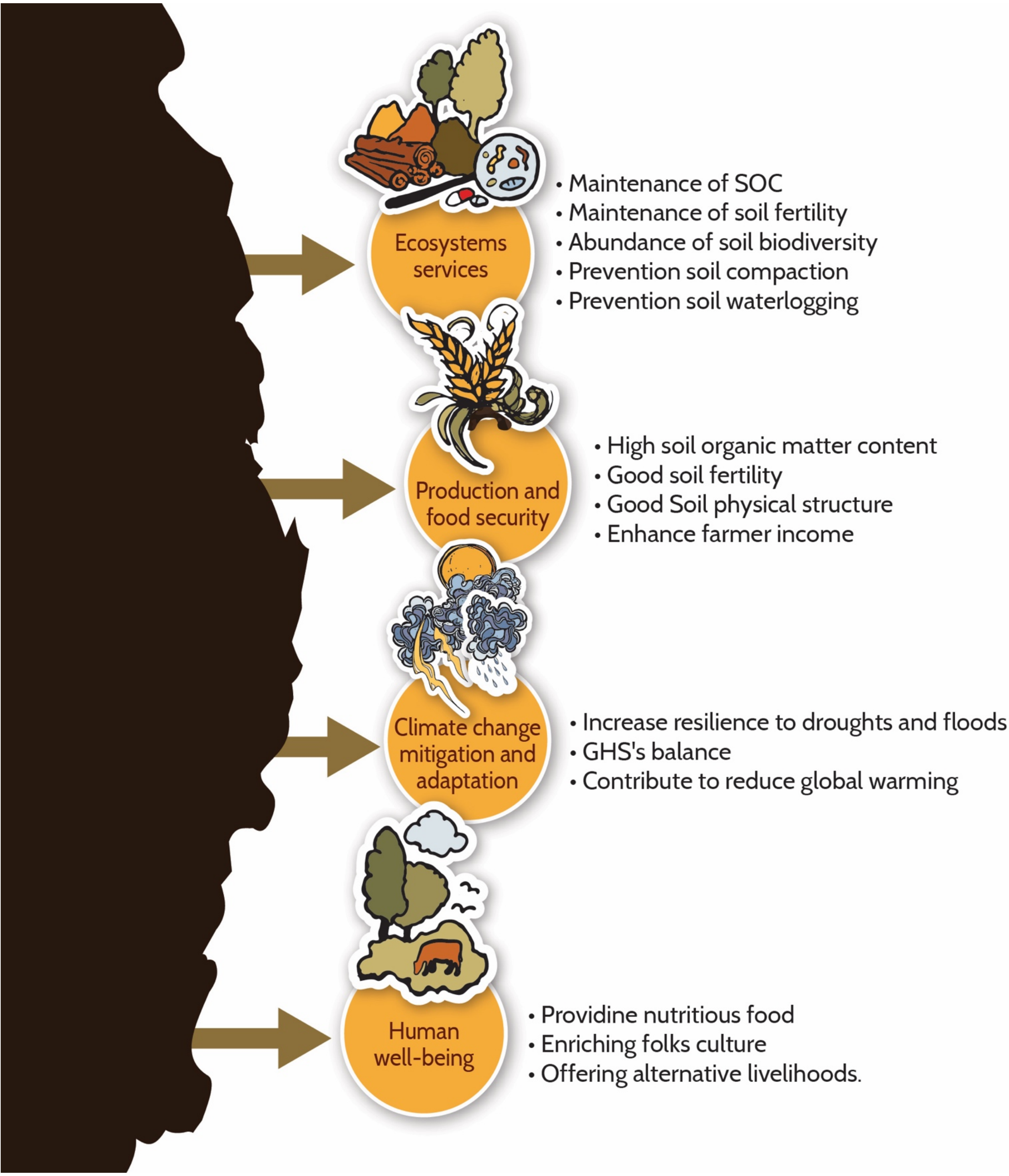

Figure 15. Mutiple benefits of black soils

5. General challenges and trends related to the black soils

5.1. Soil organic carbon loss

Land use change and inappropriate use and management soils lead to significantly decrease of soil organic carbon in Black Soils, in all regions of the world. The amount of soil organic matter content in weak, medium and severely eroded black soils has declined by 15, 25, and 40 percent, respectively in Russia (Iutynskaya and Patyka, 2010). Another study showed that 30 percent of organic matter has been lost in black soils of Ukraine (Balyuk and Medvedev, 2012). Black soils of the Republic of Moldova lost about 30 – 45 percent of carbon from the 0 – 25 cm layer over a period of 100 – 125 years (Krupenikov, 1992; Ciolacu, 2017). Chinese black soils have experienced an average annual rate of decline in soil carbon of 0.91, 0.97 and 0.48 percent, under monocropping systems of corn, soybean and wheat, respectively, in the 0 to 90 cm soil layers (Liu *et al.*, 2005). Excessive cultivation and summer fallowing have caused a 50 percent decline in soil organic matter in the Canadian prairie soils (Agriculture and Agri-Food Canada, 2003). Deforestation and subsequent cultivation have resulted in a pronounced depletion at the values of organic carbon (60–85 percent) in regions of black soils in Brazil (Rezapour and Alipour, 2017). Soil organic matter has decreased by 35.6 – 52.5 percent after a long cropping period in black soils in Argentina; all this demands the establishment of conservation practices to reduce losses of SOC and deterioration of soil quality (Durán, 2010). In Uruguay, SOC decreased more than 50 percent after only 50-year period (Baethgen and Morón, 2000).

5.2. Soil erosion

Erosion induced by rainfall and wind degrades the quality of black soils. A study showed that about a third of arable land is eroded from the black soils of Ukraine (Balyuk and Medvedev, 2012). From 1979 to 2014, cropping system conversion from forestry to dry lands aggravated erosion from 204.4 to 420.9 tons per km^2 per year in the black soil region of northeast China (Ouyang *et al.*, 2018). Changes in particle-size distribution and mainly organic matter of soils due to deforestation were responsible for a significant increase in the values of soil erodibility factor (a rise of 10–270 percent) on a study on these soils in Brazil (Rezapour and Alipour, 2017). Loss of soil by wind were observed in the lowlands with black soils in Eastern Austria, and new windbreaks are planted annually, thus increasing the protected areas by several thousand hectares per year (Strauss and Klaghofer, 2006).

5.3. Soil nutrients imbalance

With the intensive use of black soils, without proper management of fertility, the levels of nutrients decrease significantly. Increasing deficiency of labile nutrients, especially nitrogen (declining from -41.4 kg/ha in 2001 to - 56.4 kg/ha in 2009) and potassium (declining from -32.9 to - 64.2 kg/ha between 2001 and 2009) has been observed in black soils of Russia (Grekov *et al.*, 2011; Medvedev, 2012). Stocks of nutrients have noticeably decreased in black soils of Ukraine (Balyuk and Medvedev, 2012), and excessive cultivation and summer fallowing have lowered soil nutrients in the Canadian prairie soils (Agriculture and Agri-Food Canada, 2003). A pronounced depletion was observed in values of total N (67–88 percent), cation exchange capacity (9–18 percent) and exchangeable cations (4–60 percent) after deforestation of black soils in Brazil (Rezapour and Alipour, 2017).

5.4. Soil compaction

Soil compaction is a common cause of black soil degradation. After 75 years of cultivation, the total amount of water-stable aggregates declined by 26.9±1.0 percent and the clay content by 26.9±1.0 percent in black soils from Russia (Balashov and Buchkina, 2011). A study in Ukraine showed that approximately 40 percent of the black soils have a compacted layer (Balyuk and Medvedev, 2012). Excessive cultivation and summer fallowing have degraded the Canadian prairie soils, resulting in poor surface structure (Agriculture and Agri-Food Canada, 2003). Without significant variation, 14–20 percent higher bulk density and 10–22 percent lower porosity values were observed in cultivated black soils compared to forest lands in Brazil (Rezapour and Alipour, 2017).

5.5. Salinization and acidification

Salinization and acidification are consequences of both natural (primary) and human-induced (secondary) processes. But human induced salinization and acidification by inappropriate soil and fertilizer management are the main challenges in regions of black soils. Secondary salinization of irrigated soils, accompanied by a reduction of the humus rich layer depth was reported in Russia (Grekov *et al.*, 2011; Medvedev, 2012). Acidification of black soils, especially in the regions of Cherkassy and Sumy in Ukraine, was observed, where the value of pH dropped 0.3–0.5 units after 40–50 years cultivation (Grekov *et al.*, 2011; Medvedev, 2012). A decrease of soil pH by a factor of 0.27 was observed in Northeast China black soils region, from 2005 to 2014, showing a trend of acidification due to overuse of nitrogen fertilizers in intensive cropping systems (Tong, 2018).

In conclusion, Black soils are facing services threats in terms of soil organic carbon loss, soil nutrient imbalance, soil compaction and salinization and acidification. Actions are needed for sustainable management on black soils for insuring the productivity and ecological services of these carbonrich soils (Figure 16).

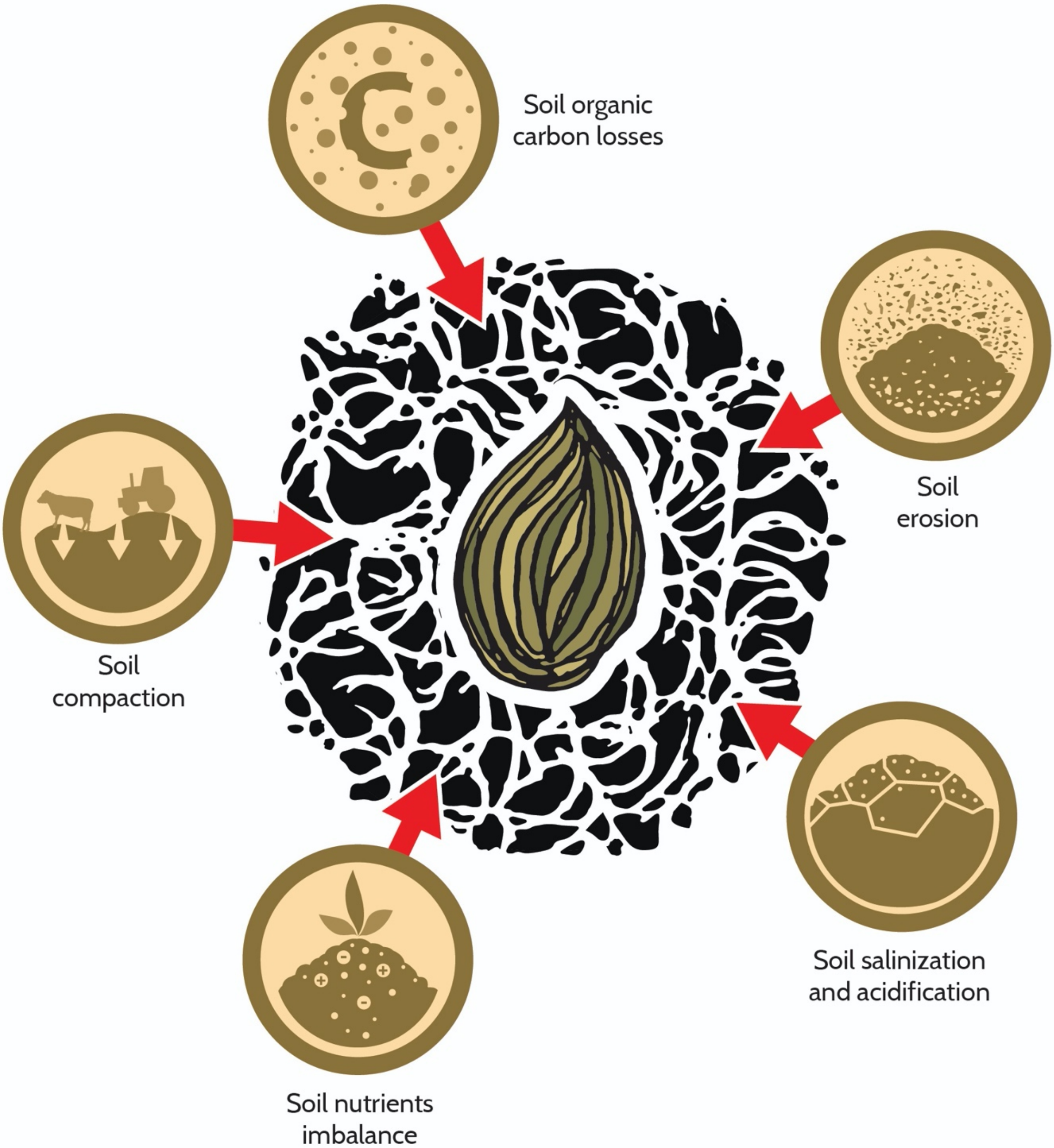

Figure 16. Main challenges to black soils

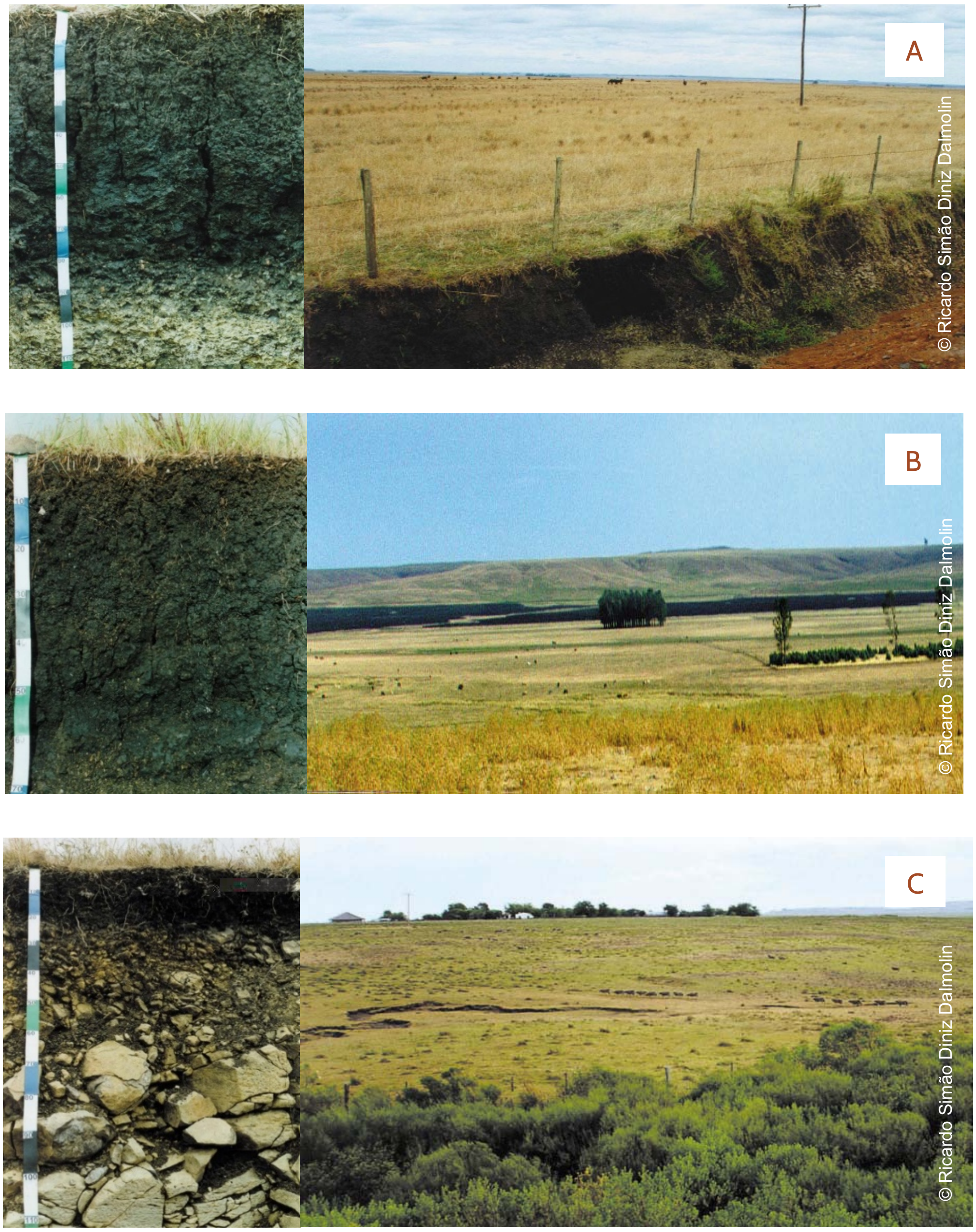

Photo 7. Profiles of Chernossolo (A), Vertissolo (B) and Neossolo (C) and landscape associated in the Pampas Biome, southern Brazil

Table 15. Related cases studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
16 years of no tillage and residue cover on continuous maize in a Black soil of China	Asia	16	4	10
Organo-mineral fertilization on a Ukrainian black soil	Europe	5	4	26
Application of swine and cattle manure through injection and broadcast systems in a black soil of the Pampas, Argentina	Latin America and the Caribbean	1	4	30
No tillage and cover crops in the Pampas, Argentina	Latin America and the Caribbean	2 to 8	4	31
Crop-pasture rotation on Black Soils of Uruguay and Argentine	Latin America and the Caribbean	10 to 48	4	39
Willow Riparian Buffer Systems for Biomass Production in the Black Soils of Elie, Manitoba, Canada	North America	6	4	42

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Grasslands

7. Grasslands

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1. Definition and description

The term "grassland" is used here in the wider sense of grazing land, consistent with the UNESCO definition of "land covered with herbaceous plants with less than 10 percent tree and shrub cover and wooded grassland as 10–40 percent tree and shrub cover". Grasslands are characterized by grasses and forbs normally occurring where there is sufficient moisture for grass growth but where environmental conditions do not support tree cover. Regionally other names also describe grassland systems, including *prairies* in North America, Asian *steppes*, *veldts* in Africa, *savannahs* in Australia and Africa, and *pampas*, *llanos* and *cerrados* in South America.

Grasslands are highly diverse in terms of the characteristics of the climate and natural landscape, their land use and management, and in the level and stability of their soil carbon. Grasslands can contain high SOC densities/stocks compared to other managed ecosystems, but net losses have occurred in both intensively managed pasture and extensive grazing lands with inappropriate livestock production practices such as stocking density (Viglizzo *et al.,* 2019) in combination with soil and climate factors (Smith *et al.,* 2019; Xu *et al.,* 2017). Identified hotspots for management include (i) areas with high soil organic carbon (SOC) stocks that are at or close to the natural equilibrium levels so that there is less potential to increase SOC but a risk of loss, particularly through LUC: (ii) areas with low to medium levels of SOC where past management has resulted in some depletion and there is a high potential to build stocks; and (iii) degraded areas where substantial losses of organic matter have occurred due to natural or anthropogenic factors where potential to restore soil health and soil carbon levels is highly uncertain. Here, the hotspot analysis focusses on identification of grassland areas with high C sequestration potential. In general, variables influencing the spatial distribution of SOC sequestration hotspots in grazing lands include plant type (life-form), vegetation cover and soil clay content.

In general, LUC from forest to grassland may not reduce SOC levels (Guo and Gifford, 2002), other than relatively small depletion in the short-term following conversion, whereas high rates of loss can follow conversion of forest or grassland to cropping. There is potential for conversion of croplands to permanent grassland to represent hotspots for soil C sequestration (Sanderman, Hengl and Fiske*,* 2017). Moreover, improved management practices can increase organic matter input in intensively grazed pastures and those used for production of grass silage and hay (Khalil *et al.,* 2020). Adoption of integrated crop-livestock-forestry systems and restoration of degraded grasslands and savannah livestock farming systems, especially in semi-arid regions similarly represent opportunities for SOC sequestration hotspots.

2. Global distribution of hotspot

Hotspot identification methods and mapping are highly variable. For example, Khosravi *et al.* (2015) used a direct sampling approach and spatial statistics to assess the drivers of C sequestration hotspots in semi-arid rangelands of the Kerman province of Iran, while Sanderman, Hengl and Fiske (2017) identified hotspots for potential SOC sequestration globally by estimating amounts and spatial distribution of SOC loss due to historic land use and land cover change (Figure 17). High SOC rundown relative to historic native levels (some losses >50 percent), give the greatest potential for soil C sequestration including by conversion of cropland to permanent grasslands.

Based on modelled carbon debt the global potential soil carbon sequestration is 133 GtC for the top 2 m of soil, with 48 GtC being in grasslands and savannahs (Sanderman, Hengl and Fiske, 2017). Rangelands of Argentina, southern Africa and Australia stand out as hotspots of SOC loss (Figure 17) and also hotspots for potential soil carbon sequestration through improved management practices.

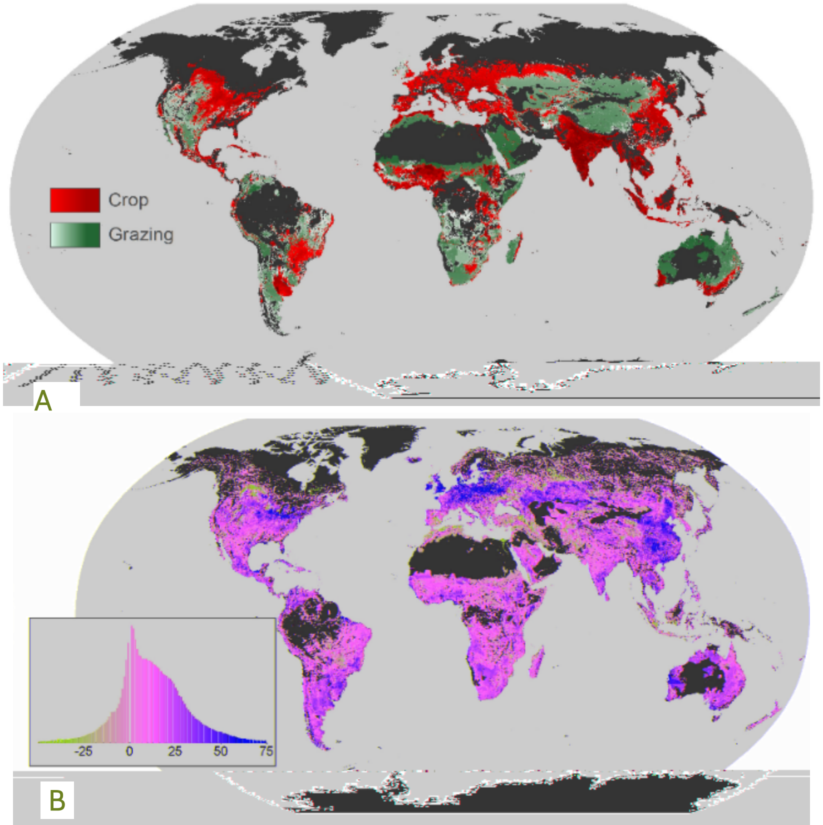

Figure 17. Global distribution of cropping and grazing in 2010, (A) and modelled SOC change in the top 2 m (B)

In A, colour gradients indicate proportion of grid cell occupied by given land use. In B, legend is presented as a histogram of SOC loss (tC/ha), with positive values indicating loss and negative values net gains in SOC

(Adapted from Sanderman, Hengl and Fiske, 2017) *Permission given 25 March 2020*

Note: the potential carbon sink is, realistically, not 100 percent of the SOC debt but, at best, may be 10 – 30 percent of the long-term loss (Sanderman, Hengl and Fiske, 2017) (Figure 18)

3. Global carbon stocks and additional carbon storage potential

Quantifying SOC sequestration potential and defining hotspot boundaries in grasslands can be challenging. Major contributing factors are data limitations and the complex interactions that determine vegetation productivity and soil C responses for different grassland management systems. While it has been estimated that managed grassland ecosystems can act as carbon sinks, storing on average 0.7 ± 0.16 tC/ha/yr (EFDC, 2018), actual rates are highly variable and uncertain (Table 16).

Figure 17B and Supplementary data from Sanderman, Hengl and Fiske (2017) show that grazing lands (grassland and savanna IGBP land classification categories) collectively lost more SOC than any other land use with the greatest contribution from arid and semi-arid regions. The rangelands of Argentina, southern Africa, and Australia represent hotspots of SOC loss when estimated as a percent of historic SOC.

Table 16. Soil carbon sequestration potential reported for pastures and rangelands

Location	Methodology	Depth (cm)	Cseq potential (tC/ha/yr)	More information	Reference
Global (rangelands)	Modelling using Century, Daycent, GLEAM*	Not specified	0.23 (Av) 0.13–0.32 (Range)	Modelled C sequestration potential with improved grazing management	Henderson *et al.* (2015)
Global (pasturelands)			0.16 (Av) 0.05–0.32 (Range)		
Argentina (semi-arid savanna)	Measurement of SOC		1.9–2.75	Restoration of highly degraded sites. Potential total gain of 58 tC/ha equal to loss due to overgrazing by cattle	Abril and Bucher (2001)
Australia (temperate pasture)	Measurements in 3 long-term trials over 15-25yrs	0 –30	0.5 –0.7	Improved nutrient, grazing management in permanent pastures	Chan *et al.* (2011)
Europe (temperate)	Modelled estimates based on 89 observations in 24 studies	0–23.5 (av)	1.05 (100 yrs) 3.23 (20 yrs)	Conversion of cropland to grassland. (% SOC change used to calculate rates)	Poeplau *et al.* (2011)
United States of America (south-east)	Av. results from 35 long-term studies on conversion of annual cropland to grassland	0–25	0.84 ± 0.11	Trials averaged 17 yrs	Franzluebbers (2010)
N. Ireland, UK (County Down)	Measured vs DNDC** modelled rates as the mean across treatments	0–15	0.46 ± 0.06 (measured) 0.37 ± 0.01 (DNDC model)	Long-term fertilisation trial (42 yrs) in grassland managed for fodder (silage/hay)	Khalil *et al.* (2020)
New Zealand (Grazing hill country)	SOC measured at 23 sites with archived samples	0–30 0–90	0.60 ± 0.16 0.9	Average period 23 years; N gains of 0.07 tN/ha/yr	Schipper *et al.* (2014)

Location	Methodology	Depth (cm)	Cseq potential (tC/ha/yr)	More information	Reference
Brazil (Southern Amazon)	Measurements for 12 yrs after tree planting in clay oxisol	0–100	1.47	Eucalypts planted in pasture (12 yrs). Nutrients not limiting.	Oliveira *et al.* (2018)

Meaningful comparisons of results are limited by differences in methods, depths and time periods. For any change in management practice or land use, the change in SOC stocks will depend on soil and climate factors, baseline SOC stocks (degraded soils have a higher potential to gain SOC), and the timeframe of the study.

*GLEAM - Global Livestock Environmental Assessment Model (Gerber *et al.*, 2013);

**DNDC - The Denitrification-Decomposition (DNDC95 version)

4. Importance of grassland conservation for the provision of specific ecosystem services

4.1. Minimization of threats to soil functions

Table 17. Soil threats

Soil threats	
Soil erosion	Soil carbon sequestration makes soils less prone to water and wind erosion by improving soil structure; practices such as increasing cover for increasing SCS reduce vulnerability to landslides (Keesstra *et al.*, 2016).
Nutrient imbalance and cycles	Increased soil organic matter aids in storing, cycling and transforming nutrients (Smith *et al.*, 2019)
Soil salinization and alkalinization	Avoiding overgrazing on grasslands with saline groundwater helps manage salinity risk (Lavado and Taboada, 1987). Conversion from cropping to grassland in semi-arid regions increases SOC and decreases soil salinity and sodicity (Yu *et al.*, 2018).
Soil contamination / pollution	Soil carbon sequestration enhances soil health and buffers pollutants, also protecting water quality (Smith *et al.*, 2019)

Soil threats	
Soil acidification	Conversion of grasslands to forest results in soil acidification (Jobbágy and Jackson, 2003)
Soil biodiversity loss	Soil carbon sequestration improves soil health, enhancing habitats for soil biota (Keesstra *et al.*, 2016).
Soil compaction	Permanent grasslands with low stocking rates generally have lower bulk density than heavily grazed areas (Miao *et al.*, 2015).
Soil water management	Practices for soil carbon sequestration can also help regulate timing and magnitude of peak water flows (Smith *et al.*, 2019).

4.2. Increases in production and food security

Soil organic /matter is essential for good soil quality, functionality, and health. Through supply of macro- and micronutrients, the level of SOC is a strong determinant of global food and nutritional security (Lal, 2016). Practices that sequester SOC also tend to improve food security and climate change adaptation. With increasing SOC, co-benefits for yields (ca. 0.07 ton (t) of dry matter/t SOC sequestration) could be obtained each year under tropical conditions (Soussana *et al.*, 2019). Ruminant livestock production on the estimated 2.6 billion ha of grazing lands worldwide (Henderson *et al.*, 2015) contribute to the livelihoods of the billion people who depend on livestock production (Herrero *et al.*, 2009).

4.3. Improvement of human well-being

SOC sequestration in grasslands positively affects the physical and cultural environment for humans (Smith *et al.*, 2019). A healthy soil is important in defining large areas of grasslands that provide ecosystem services for some of the world's poorest peoples. Through effects on nutrition and water quality, provision of certain plant-based medicines and aesthetic benefits, soil health in these landscapes determine human well-being.

4.4. Mitigation of and adaptation to climate change

Soil carbon sequestration represents a potentially large sink for atmospheric CO_2, supports resilience to climatic changes, and underpins adaptation of food production to climate change (e.g. Lal, 2016; Soussana *et al.,* 2019). Estimates of the potential climate change mitigation in grasslands vary. Henderson *et al.* (2015) estimated that, with improved management and legume sowing, grazing lands could sequester about 80 Mt C/yr while increasing forage production by over 900 Mt dry matter, resulting in economic and food security benefits. However, increased production involves a trade-off with GHG emissions in the form of higher enteric methane from ruminant livestock.

5. General challenges and trends

While the biophysical potential for gains in SOC is generally greatest in grassland soils where past management has depleted natural levels of organic matter, there are often socio-economic barriers to large scale adoption of best practices in grazing management (Figure 18). Growth in demand for animal protein (meat and milk) is projected to continue (Herraro *et al.*, 2009). While expansion in production was historically driven by increasing the extent of grazing through LUC, the current trend is more towards intensification on current grassland areas. Higher grazing pressure in combination with climate changes, such as warmer temperatures and less reliable precipitation, increases the risk of overgrazing with loss of grass cover and erosion and, hence, exacerbates the challenge of increasing or maintaining SOC stocks. Examples of specific challenges are:

Forest

Historic deforestation for livestock production in grasslands in temperate and tropical regions (Figure 17) contributes to hotspots for SOC sequestration (Sanderman, Hengl and Fiske*,* 2017). Conversely, use of fire to manage forest encroachment and grazing productivity in savannahs particularly in the tropics and neotropics (e.g. Burrows *et al.*, 2002) can result in non-CO_2 GHG emissions and also loss of stored nitrogen and carbon from soils. Surface SOC in plots burnt frequently over 64 years were estimated to have declined by 36±13 percent Compared to unburnt plots after 64 years (Pellegrini *et al.*, 2018).

Peatland and Wetlands

Organic soils such as histic Gleysols that are drained for grassland livestock production can be hotspots for CO_2 loss (Leiber-Sauheitl *et al.*, 2014) with the net GHG balance reaching 7–9 t C/ha/yr on soils in northern Germany.

Climate change

Anthropogenic warming increases the challenge for sequestration in grasslands as higher soil temperature initiates a positive feedback loop as soil microbe convert more SOC to CO_2 (Amundson *et al.,* 2015). Moreover, the scope to increase organic matter inputs will be limited by projected more intense droughts in future exacerbating land degradation due to over-grazing.

All

A meta-analysis of changes in SOC for 2001–2019 examined the effects on SOC and underlying mechanisms of 33 influencing factors, including LUC and LU for livestock grazing (Xu *et al.*, 2020). It was concluded that increasing C inputs is one of the best measures to sequester SOC, consistent with the identification of hotspots in Figure 17.

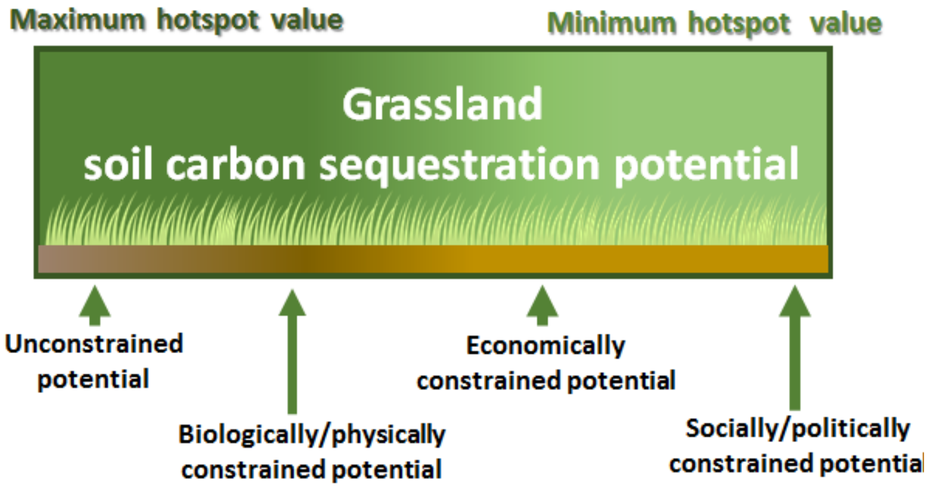

Figure 18. Constraints on grassland sequestration potential (Adapted from Sanderman, Farquharson and Baldock, 2010)

At any location, the maximum feasible sequestration potential in grasslands and, therefore, the absolute and relative significance of livestock farming hotspots is dependent on soil and climate conditions and economic, social, and political constraints. These constraints may change over time, e.g. with local impacts of climate change

Table 18. Related cases studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case - study n°
Grazing management in rangeland grassland systems in South and East Australia	Southeast Pacific	4 to 10	4	9
Mediterranean savanna-like agrosilvopastoral grassland system in Spain, Italy and Portugal	Europe	4, 22 and 37	4	17
Increasing Yield and Carbon Sequestration in a Signalgrass Pasture by Liming and Fertilization in Sao Carlos (SP, Brazil)	Latin America and the Caribbean	6	4	32
Integrated farming in tropical agroecosystems of Brazil	Latin America and the Caribbean	4 to 12	4	34
Mitigation of SOC losses due to the conversion of dry forests to pastures in the plains of Venezuela	Latin America and the Caribbean	5 and 18	4	40

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Mountain soils

8. Mountain soils

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1. Definition and description

Soils can act as relevant carbon (C) reservoirs and/or sinks, as they can store C for variable times, removing it from the atmosphere, thus contributing to the regulation of climate at the planet scale by reducing GHG emissions. Mountain soils in particular can store high amounts of organic C (soil organic C - SOC). Not only the amount of OC stored in soils is important, but also its quality, which in turn affects its persistence (e.g. Cotrufo *et al.,* 2019). In particular, SOC can be retained in different pools such as particulate organic matter and mineral-associated organic matter, which show different turnover rates in soils, depending on soil type and management. These general considerations hold for all soils, but they are particularly relevant in mountain areas where the natural vegetation cover is preserved, soils are mainly undisturbed, and sealing (urbanization) is less pronounced. In such conditions, the organic residues (litter, dead organisms) are not removed from the system, and can enter the carbon cycle by mineralization and humification, and be stored into the soil (mostly as SOC) for long time periods.

The SOC stock is generally defined as the amount of organic C stored in a fixed land surface (e.g., 1 hectare). This amount of SOC refers to a given soil depth (e.g., 30 cm, 1 m) or the whole soil profile, and the input data for the calculation are the soil bulk density (measured in the field or estimated), the stone content (measured or estimated), and the SOC content.

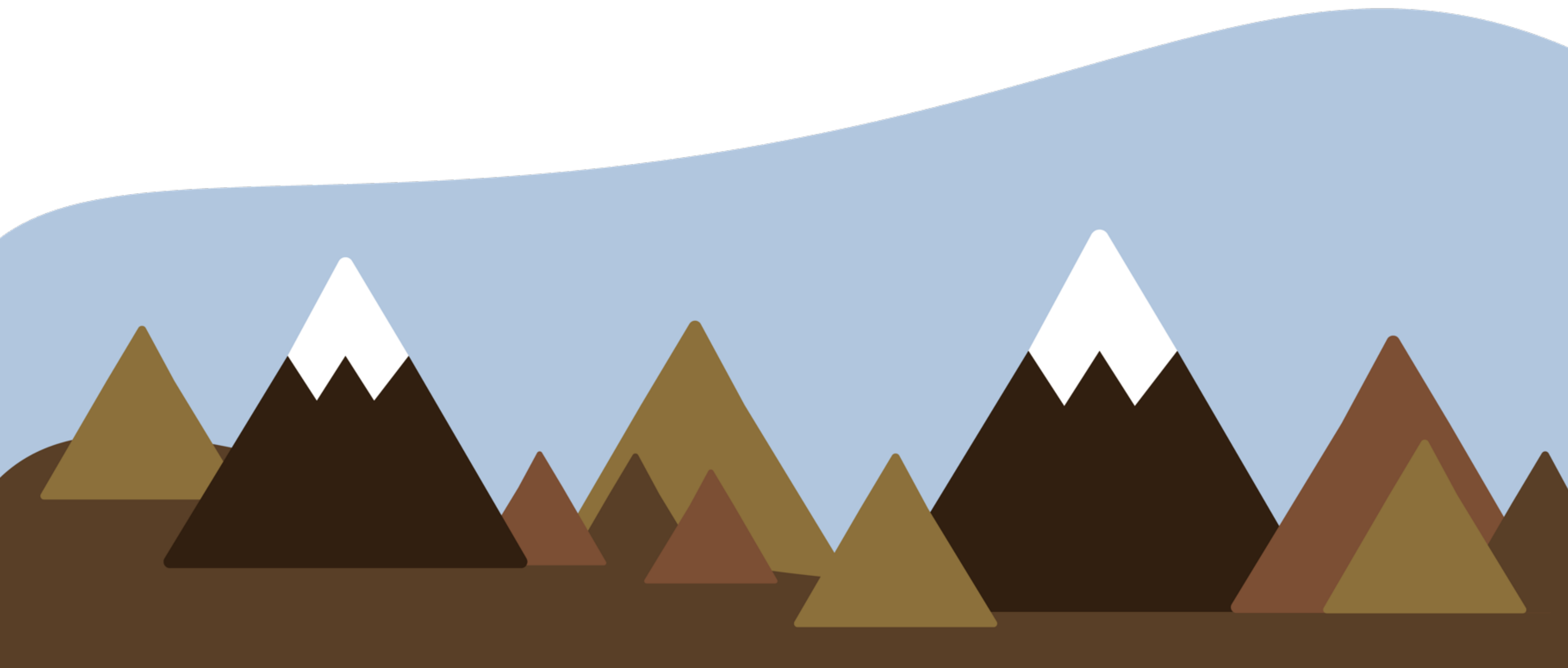
The extent to which this storage function is fulfilled depends on the soil type (and thus on several soil properties such as total depth, organic matter content, texture, aggregation and porosity, humus type), but it is also affected by environmental and site conditions (e.g. climate and slope), land-use and management history (Wiesmeier *et al.*, 2019). In particular, land-use change and soil management practices (e.g., deforestation, overgrazing, tillage practices enhancing erosion) can affect the ability of soils to store carbon, and its residence time in soils.

For this study, the UNEP-WCMC definition of mountains was used, i.e., mountain land is estimated from a digital elevation model based uniquely on elevation (when >2500 m a.s.l.), or a combination of elevation, slope, and local elevation range when < 2 500 m a.s.l. (Kapos, 2000; UNEP-WCMC, 2002). The global mountain area is 39.3 million km^2, or 27 percent of the Earth's land surface (FAO, 2020, forthcoming); of this surface, a large part is covered by forests. According to the Mountain Green Cover Index Data (SDG Indicator 15.4.2), as of 2015 mountain forests covered about 12 924 600 km^2.

Given the extent of mountain forests in the world and their potential for biomass production, it is evident that mountain areas are potentially active as hotspots of SOC stocks, and could stock even more carbon. As underlined by Lehmann *et al.* (2020), a deep understanding of C dynamics in soils is needed to mitigate climate change, but constant care is to be preferred to one-time actions to prevent C emissions into the atmosphere. Mountains and mountain soils are also very vulnerable to climate change, which can increase erosion or enhance the fast mineralization of organic matter due to higher temperatures and modifications in the precipitation regimes (Hock *et al.*, 2019). Detailed inventories of SOC stocks can help to simulate climate and land-use changes, and their consequences on soils (e.g. Shi *et al.*, 2020). Thus, the management of mountain land and soils is a crucial issue in climate change mitigation strategies.

2. Global distribution of hotspot

In Table 33 (Annex 2), we collected relevant literature, focusing on SOC stocks in mountains all over the world (Figure 19). One relevant drawback for comparing the estimations of average SOC stock values in the different mountain ranges is that the authors used a broad variety of methods (Table 33). For example, bulk density can be measured from specific samples or estimated using pedotransfer functions (often leading to an overestimation of bulk density values, and thus of stocks). Moreover, each study considers different soil depths for C stocks calculation: some researchers measure it only in topsoil (generally, 0–10 cm or 0–30 cm), others consider different depths (0–40, 0–50, 0–60 cm, etc.), some divide topsoil from subsoil stocks, others equalize all stocks to 1 m depth, and others calculate it on the whole soil profile (Table 33). When dealing with forest soils, some researchers calculate stocks included in organic horizons, while others neglect this amount. Thus, a precise comparison between SOC stocks between different mountain ranges, dependent on climate and vegetation or land-use differences, is impossible. Only broad differences and relationships with environmental properties can be estimated.



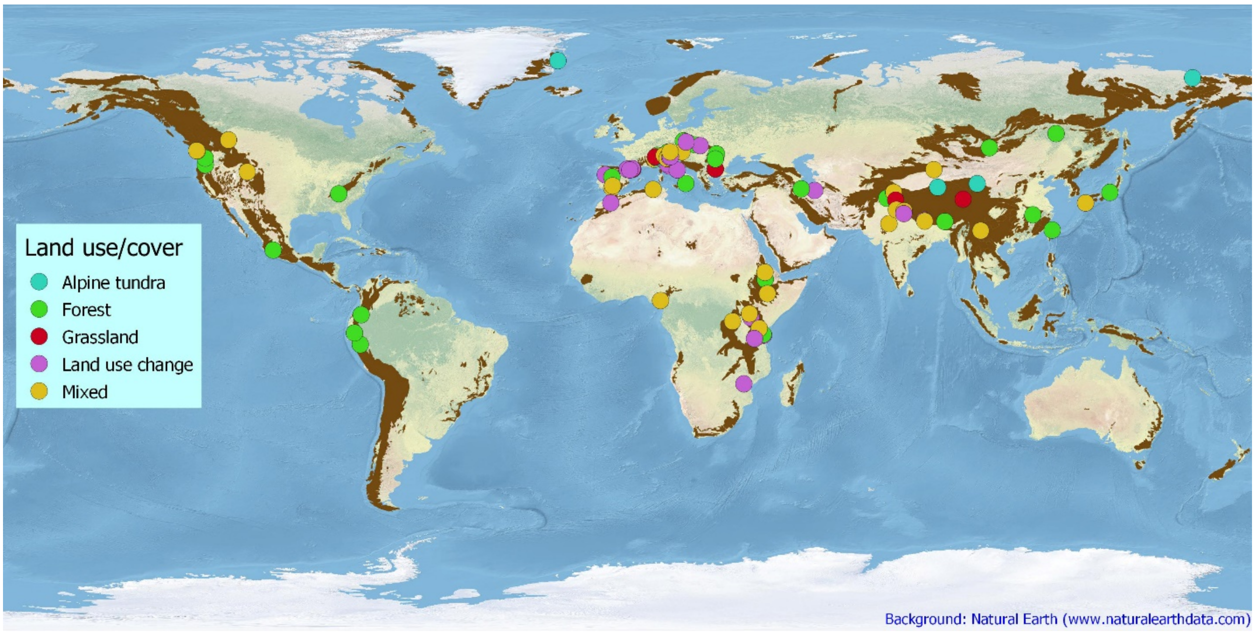


Figure 19. A map showing the localization of the literature data presented in Table 33 (Annex 2)

Shapefiles were produced by the Global Mountain Biodiversity Assessment (GMBA) for the Mountain Portal representing mountain ranges (University of Bern, 2020). The category "Mixed" refers to the presence of more than one land use/cover.

3. Global carbon stocks and additional carbon storage potential

As visible in Annex 2, the large variation among OC stocks within single mountain ranges (and study sites as well) tends to hide broader trends related to environmental factors, such as land-use and climate. For example, based on data from Table 33, in the European Alps, forest soils stock between 61 and 278 t/ha of OC, while grasslands (Photo 8) have average OC stocks in mineral horizons below 100 t/ha. Mountains in Mediterranean areas (such as the Apennines in Italy, Photo 9), the Pyrenees between France and Spain, and many mountain ranges in Spain store between 38 and 165 t/ha, with minor difference between forest and grassland soils.

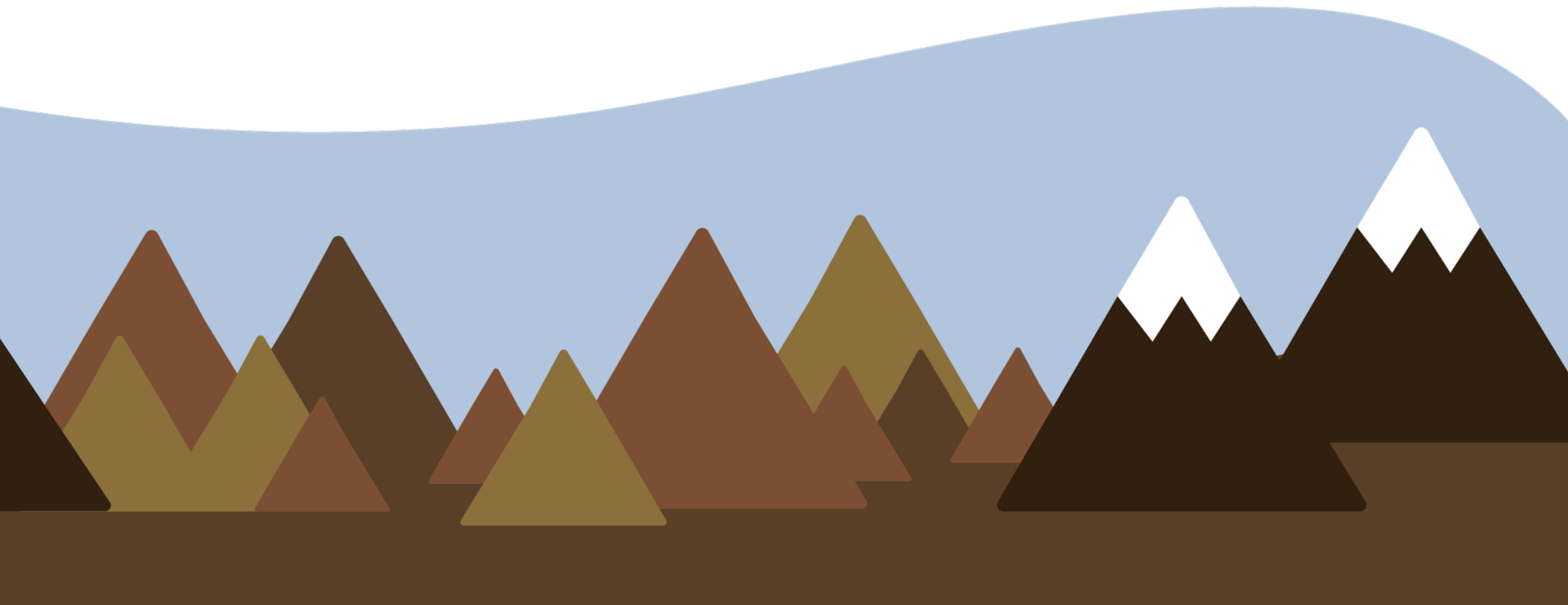


Photo 8. Entic Umbric Podzol at 2600 m a.s.l on the southern slope of Mont Blanc (4810 m a.s.l. NW-Italian Alps)

Photo 9. Shallow and stony but OC-rich Rendzic Leptosol in the alpine elevation belt in the Apennines, Central Italy

The SOC stock ranges shown in each work, however, are sometimes even larger. All these data exclude Histosols (peatland soils), as they are treated in a specific chapter (See hot-spot n°4 "Peatlands", this volume). In the Himalayas (9 articles), soil profiles store on average between up to ~ 400 t/ha of SOC (Photo 10); the huge elevation and climatic gradients, however, make these data poorly representative. Mountains in tropical areas tend to stock more SOC: for example, the first 70 cm of soil on Mount Cameroon stores 150–300 t/ha (with maximum values in rainforests, Tsozuè *et al.* 2019), while the steep slopes of SW Uganda (Twongyirwe *et al.* 2013) store up to 170 t/ha in the top 30 cm (Photo 11). Soils in the Usambara Mountains (Eastern Arc Mountains, Tanzania) store up to 270 t/ha in the top 100 cm.

Loss of OC is usually observed when land-use is changed from native forest to cultivation, particularly in the tropics. On the slopes of Mt. Kilimanjaro, for example, SOC stocks are reduced by ca. 23–38 percent when natural vegetation is removed to make space for cultivations (maize or coffee, Pabst *et al.* 2016). Tea plantations, however, can sometimes help soils store larger quantities of SOC compared to both degraded and primary forests (Chiti *et al.* 2018). Dinakaran *et al.* (2018) show how agricultural soils in the Himalayas are characterized by extremely depleted OC pools compared to forest or grassland soils.

The effect of land abandonment followed by shrub encroachment or reforestation, which are dominating processes in European mountains, is less clear, particularly in temperate areas (e.g. Campo *et al.* 2019).

Photo 10. A gentle slope along the Khumbu Valley (5065 m asl, Nepal), characterized by a C-rich Brunic Dystric Arenosol (Aeolic, Raptic)

Photo 11. Steep slopes covered by tropical montane rainforests in Bwindi National Park (SW Uganda), and nearby cultivated areas

4. Importance of mountain soil conservation for the provision of specific ecosystem services

Mountain soils provide many ecosystem services ranging from primary production to global climate regulation, by controlling C emissions into the atmosphere. They also provide a wide number of services related to water provision, filtration, and regulation that control the availability of pure water and groundwater recharge. Geitner *et al.* (2019) reviewed the main ecosystem services performed by mountain soils (in the Alpine Region) as the product of a unique combination of environmental conditions and soil properties, and listed them as:

- agricultural and forest biomass production;
- water retention;
- surface run-off regulation;
- local climate regulation;
- global climate regulation;
- water filtration and purification;

- nutrient cycle regulation;
- soil habitat & biodiversity;
- cultural & natural archives;
- recreational & spiritual services.

FAO (2019) estimated that, as of 2017, mountains were home to around 15 percent of the world population (about 1.1 billion people). Mountain soils benefit, in many ways, not only the people living in the world's mountains but also billions more living downstream. Mountains provide food, fodder, medicinal plants and other wood and non-wood forest products, and are recharge areas for water in aquifers used by a consistent part of the world's population. However, mountain people are still economically marginalized and heavily affected by poverty and high rates of vulnerability to food insecurity. According to the recent FAO data, as of 2017, one in two rural mountain dwellers living in developing countries were vulnerable to food insecurity (FAO, 2020). Vulnerability to food insecurity in rural mountain areas has been increasing since 2000, the first year when such data was monitored. The number of vulnerable people has increased in all regions of the developing world, but some have suffered more than others. Indeed, during the last 5 years, rural mountain dwellers in Africa saw the biggest increase in food insecurity. Between 2012 and 2017, more than 25 million rural mountain people have become vulnerable to food insecurity. Hence, the relevance of investing in mountain areas and protecting mountain soils to achieve the zero-hunger SDG goal becomes evident.

A wide range of soil ecosystem services is related with the water cycle, including the retention of water available to plants and the soil biota; the regulation of run-off, and thus the reduction of flooding and erosion risks through the balance with infiltration; water filtration, (the ability of soils to filter water, neutralizing or degrading potentially harmful substances), and contributions to the groundwater recharge. In particular, run-off regulation limits erosion rates and consequent nutrients loss on mountain slopes, where the soil formation rates are lowered by the harsh climate.

Mountains soils also act in climate regulation both at the local scale (through evapotranspiration, which cools down the air temperature and is especially relevant in urbanized areas) and on the global scale, through the storage of organic C that prevents its emission into the atmosphere as GHG. Additionally, forests are also known by being sinks of methane, a GHG with high warming capacity (e.g. Delmas *et al.,* 1992; Zhao *et al.* 2019b).

Among the other regulating services performed by mountain soils, is the ability to store, cycle and exchange nutrients with the other ecosystem components, thus preserving soil fertility and keeping healthy soils for food production. Soil management can largely contribute to enhance OC storage both in terms of amount and residence time (OC sink function).

Mountain soils also greatly contribute to biodiversity, hosting a huge variety of organisms living and growing in soils. The soil biodiversity expresses itself not only in terms of number and variety of species and individuals, but also in terms of the gene pool, and it's estimated to exceed the aboveground biodiversity.

Finally, soils also perform less visible services related to the aesthetic perception and the quality of the landscape, that are important for human well-being. Because of limited development in mountain regions, they can also host remnants of the past and give scientists many insights on past climate, vegetation, etc.

Given the variety of ecosystem services performed by mountain soils, the goal of sustainable soil management is fundamental if we want to keep mountain soils healthy and functioning to their full potential.

Photo 12. Top: Vulnerable and eroded slopes after deforestation in the Kisoro District, SW Uganda (Bwindi mountains, with the Virunga Volcanoes in the background). Bottom: Diffused erosion affecting recently deforested slopes along the Rift Valley escarpment, Elgeyo-Marakwet County, Kenya

5. General challenges and trends

Besides soil ecosystems services, Geitner *et al.* (2019) also listed the main soil threats that can put mountain soils in danger. As soil ecosystem services are affected by chemical and physical properties such as texture, organic matter content, permeability, and structure, they can be negatively impacted by changes in these properties after natural or anthropogenic disturbance. Among the main threats putting at risk the functions of mountain soil, erosion and nutrients loss are particularly relevant in mountain areas where the effect of slope, combined with slow soil formation rates, can limit soil development. As well compaction (i.e. a significant reduction of soil porosity resulting from improper soil management practices (e.g. timber harvesting techniques, overgrazing…) can heavily affect mountain soils, compromising the infiltration capacity and the regulation of surface run-off, thus favoring accelerated erosion. Other threats are related to the loss of organic matter, which can affect both organic and mineral soils and can contribute to the release of greenhouse gases into the atmosphere. The loss of biodiversity can also alter mountain soils, and affect other services which are fundamental for healthy soils. Very often, soil threats (e.g. erosion and compaction) are triggered by land-use changes such as deforestation, and overgrazing (Photo 12 and Photo 14), urbanization, the building of infrastructures, or other disturbances (for example, wildfires). Soil sealing can also have negative effects on soil ecosystem services such as run-off regulation and biodiversity. This is particularly true in densely settled areas such as touristic resorts in the Alps, where the surface available for services and infrastructures is naturally limited by land conformation. The growth of tourism (for example, construction of ski runs, lifts, and hotels), the expansion of traffic, energy projects (including power lines, dams, hydroelectric plants, and reservoirs), intensified agricultural use, settlement development, and human-induced climate change are placing a growing impact on the environment. The soil in the Alps is not exempt from this development. Land use change and anthropogenic climate change result in severe sealing, erosion, and degradation.

The Member States of the Alpine Convention (AC) thus adopted the Soil Conservation Protocol at the 5th Alpine Conference in 1998, which is an instrument under international law that deals specifically and directly with soil conservation in a particular region (Markus, 2017). Specific attention has been dedicated to the protection of soils with particularly characteristic features, such as in wetland and moors, with the designation and management of endangered areas and areas threatened by erosion.

Attention and efforts by the scientific community, civil society, and international organizations have increased the awareness of the importance of mountain soils (for example, FAO, national and international soil science societies). In 2015 (UN International Year of Soils) FAO, a collaboration with the Mountain Partnership Secretariat[5], the Global Soil Partnership[6] and the University of Turin (Italy) promoted an awareness-raising campaign focused on mountain soils, with the publication of a book called "Understanding Mountain Soils". More than 100 authors contributed to this book, through the presentation of worldwide case studies on the specificities of mountain soils, including their potential for climate change mitigation.

The case studies ranged from oceanic alpine landscapes in Scotland (Britton *et al.*, 2015) including ecosystems typical of many mountain areas on the northwestern fringe of Europe, to the mountain wetlands of Lesotho

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[5] https://alpinesoils.eu/gspesp/mountain_p_secretariat/

[6] http://www.fao.org/global-soil-partnership/en/

(Mapeshoane, 2015). From these studies it appears clear that a better understanding of the mechanisms underlying the spatial variability in soil C stocks and fluxes is urgently needed, to predict the fate of mountain soil C under a changing climate and land-use.

6. General recommendations for the hotspot

To increase the effectiveness of mountain soils in OC sequestration, we can act in several ways to promote SOC accumulation and increase its residence time in soils that is, to delay its return into the atmosphere. Several guidelines can be proposed, but all of them are related to one or more of the following processes (Post and Kwon, 2000):

- increasing the input of organic matter (grazing, cover crops, afforestation...)
- changing the decomposability of organic matter, which is usually low in mountain areas due to harsh climate, by favoring its incorporation into the soil (for example, by enhancing mixing by organisms or by direct below-ground input)
- favouring the interaction between organic matter and the soil mineral phase, that is, promoting aggregation types that protect organic matter from fast mineralization
- developing effective guidelines aiming at maintaining and increasing the OC sequestration potential in mountain soils (Links4Soils, 2020)
- avoiding excessive irrigation, leading to nutrients loss and erosion. Drip irrigation or sub-irrigation are optimal systems in terms of water-use efficiency. Additionally, they minimize nutrients leaching and water erosion
- increasing organic matter content in agricultural soils with suitable fertilization. Apply animal manure and/or compost to improve soil aggregation through the input of organic matter. When possible, cover manure to limit the decline of soil fertility (i.e. via ammonia volatilization) and the dilution effect caused by rainfall and irrigation
- whenever possible, encouraging integrated systems (e.g. crop-livestock systems or crop-livestock-forest-systems, Photo 13), as well as reduced- or no-tillage practices that slow down the mineralization rate of the soil organic matter
- preserving the areas with carbon-rich soils such as peatlands and forests not only as OC reservoirs but also as unique sources of biodiversity
- avoiding or limit agricultural burning, especially in areas affected by erosion
- avoiding overgrazing leading to soil erosion on slopes and floods in lowlands. Organize pasture rotation and haymaking practices where possible.

Photo 13. Agroforestry in the Pare Mountains, Eastern Arc mountains, NE Tanzania

Photo 14. Erosion due to overgrazing (Kisimiri Chini, Mount Meru, Arusha District, Tanzania)

Table 19. Related cases studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Reforestation of highlands in Javor Mountain, Republic of Srpska, Bosnia and Herzegovina	Europe	15	6	5
Natural afforestation of abandoned mountain grasslands along the Italian peninsula	Europe	23 to 72	6	6

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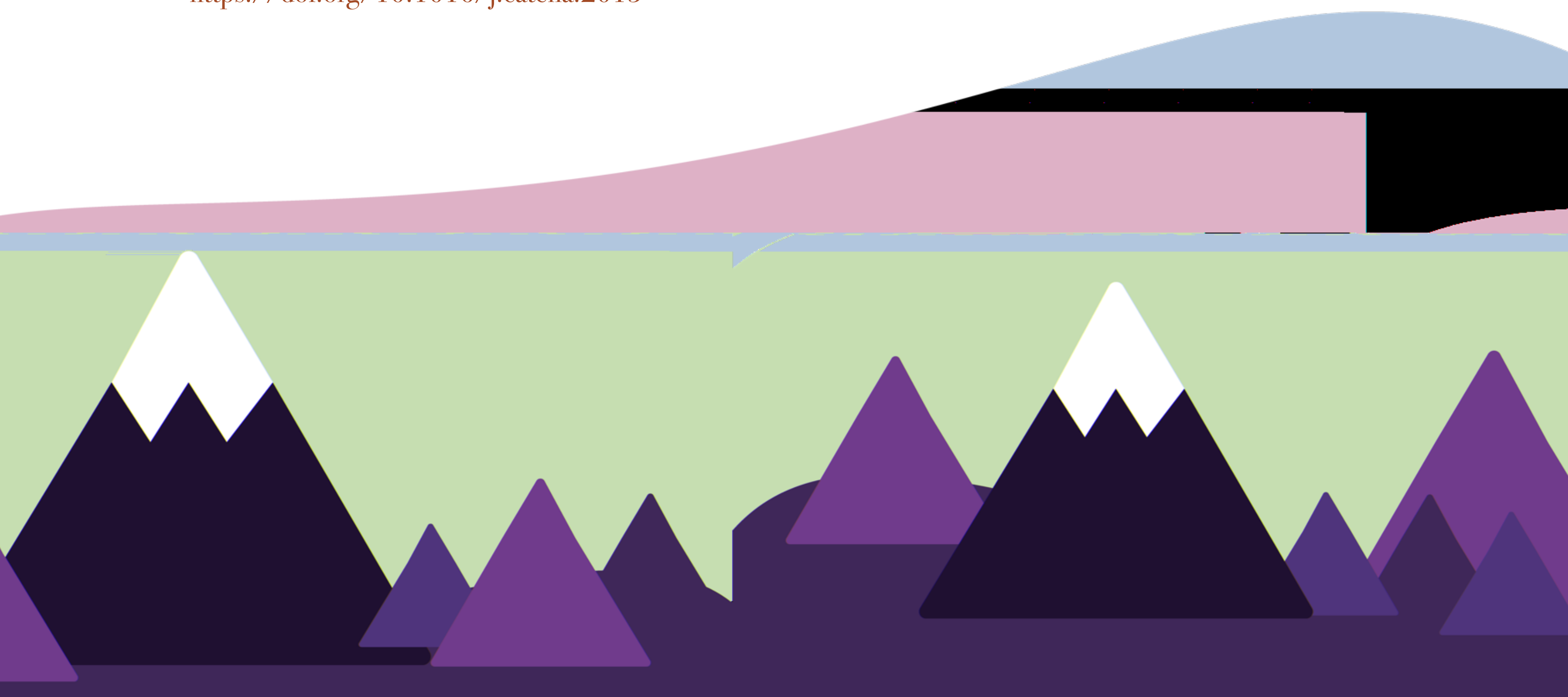

Permafrost

9. Permafrost

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1. Definition and description

Permafrost is perennially frozen ground, such as soil, rock, and ice. In permafrost regions, plant and microbial life persists primarily in the near-surface soil that thaws every summer, called the 'active layer' (Figure 20). The cold and wet conditions in many permafrost regions limit decomposition of organic matter. In combination with soil mixing processes caused by repeated freezing and thawing, this has led to the accumulation of large stocks of soil organic carbon in the permafrost zone over multi-millennial timescales. As the climate warms, permafrost carbon could be highly vulnerable to climatic warming.

Permafrost occurs primarily in high latitudes (e.g. Arctic and Antarctic) and at high elevation (e.g. Tibetan Plateau, Figure 21). The thickness of permafrost varies from less than 1 m (in boreal peatlands) to more than 1 500 m (in Yakutia). The coldest permafrost is found in the Transantarctic Mountains in Antarctica (−36 °C) and in northern Canada for the Northern Hemisphere (-15 °C; Obu *et al.*, 2019, 2020). In contrast, some of the warmest permafrost occurs in peatlands in areas with mean air temperatures above 0 °C. Here permafrost exists because thick peat layers insulate the ground during the summer. Most of the permafrost existing today formed during cold glacials (e.g. before 12 000 years ago) and has persisted through warmer interglacials. Some shallow permafrost (max 30–70m depth) formed during the Holocene (past 5000 years) and some even during the Little Ice Age from 400–150 years ago.

There are few extensive regions suitable for row crop agriculture in the permafrost zone. Additionally, in areas where large-scale agriculture has been conducted, ground destabilization has been common. Surface disturbance such as plowing or trampling of vegetation can alter the thermal regime of the soil, potentially triggering surface subsidence or abrupt collapse. This may influence soil hydrology, nutrient cycling, and organic matter storage. These changes often have acute and negative consequences for continued agricultural use of such landscapes. Thus, row-crop agriculture could have a negative impact on permafrost (e.g. Grünzweig *et al.*, 2014). Conversely, animal husbandry is widespread in the permafrost zone, including horses, cattle, and reindeer.

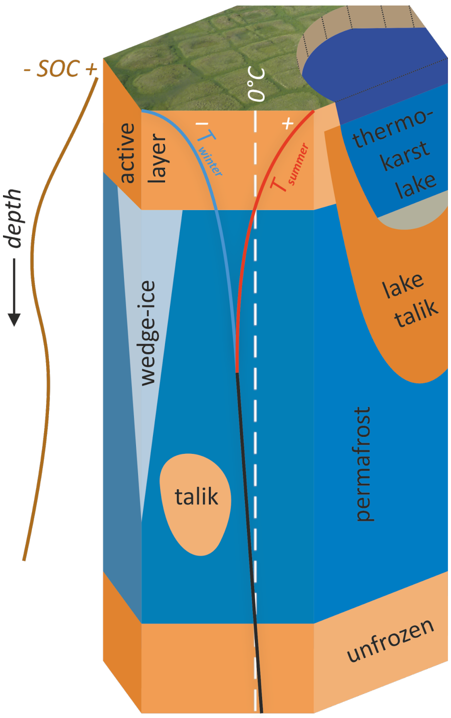


Figure 20. Diagram of the vertical structure of permafrost consisting of the active layer, permafrost including ground ice such as ice wedges, and unfrozen parts called taliks

The red and blue curved lines down the center of the diagram show the typical ground-thermal regime, indicating maximum (T_{summer}) and minimum temperatures (T_{winter}), the point of zero annual amplitude (intersection T_{winter} and T_{summer}), the increase in temperature with depth (geothermal gradient), and the depth of seasonal thaw (the active layer). Taliks are unfrozen areas within the layer of frozen material. The density of soil organic carbon (SOC) with depth is shown on the left by the brown line, based on Harden *et al.*, 2012 (top 3 m) and Strauss *et al.*, 2015, 2017 (deeper SOC deposits).

2. Global distribution of hotspot

The global permafrost distribution is controlled by long-term mean air temperature. Locally, the distribution of permafrost is also affected by the properties of the ground surface and various ecosystem factors. Permafrost is more likely to occur in areas of low snow cover, insulative soil (e.g. peat) or vegetation, and absence of surface water. Permafrost regions are commonly subdivided by the proportion of the land area underlain by frozen material (Figure 21): continuous permafrost with >90 percent coverage, discontinuous permafrost with 50–90 percent coverage, sporadic permafrost with 10–50 percent coverage, and isolated permafrost, which has <10 percent coverage (not included in Figure 21).

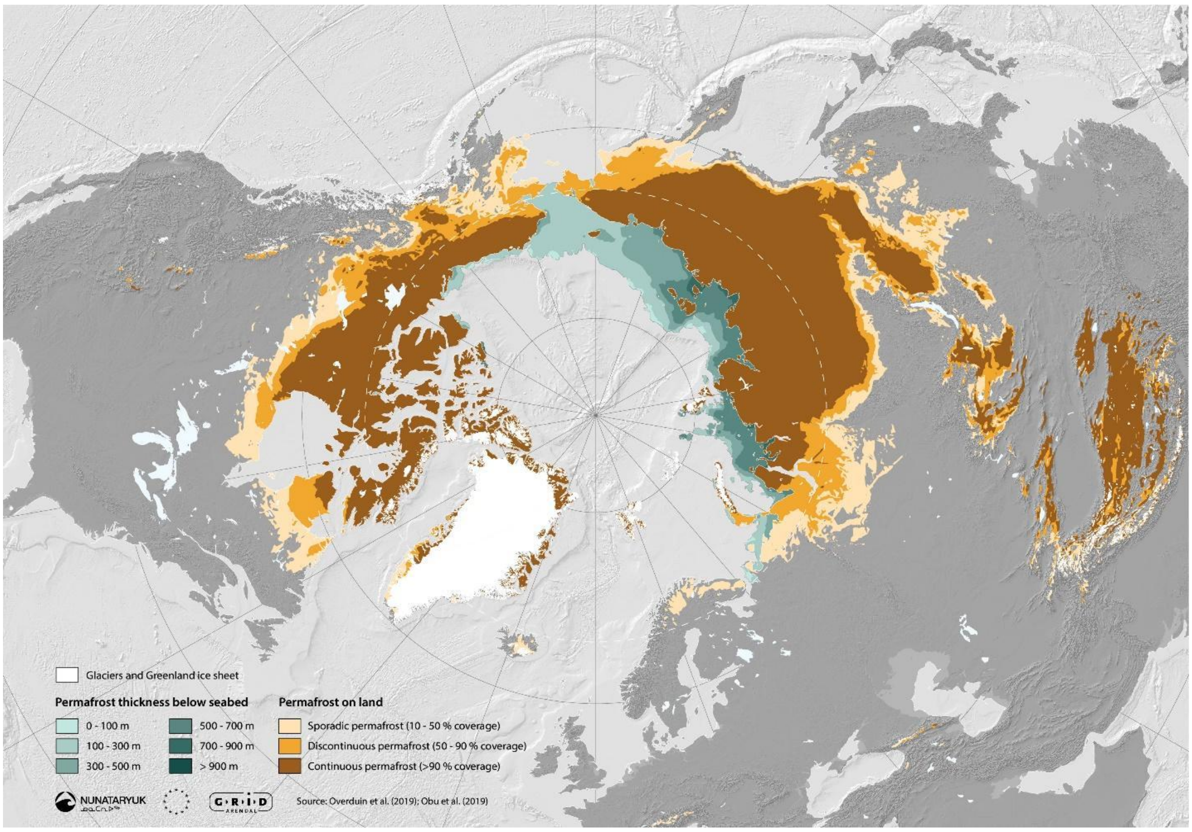


Figure 21. Extent of permafrost on the Northern Hemisphere

This map has been graciously adapted by G. Fylakis from GRID-Arendal based on data from Overduin *et al.* (2019) and Obu *et al.* (2019) and a product of the NUNATARYUK project in collaboration with GRID Arendal.

Permafrost occurs on land in polar and high mountain areas, and as submarine permafrost in the bottom sediments of shallow shelf regions of the polar oceans (Figure 21). Estimating its total coverage is challenging because permafrost occurrence is spatially heterogeneous and difficult to measure remotely. For example, the permafrost region (<u>including</u> permafrost-free patches) of the Northern Hemisphere is estimated to be 21 million km² (22 percent of exposed land area, brownish colors in Figure 21), but modelling studies indicate that only 13.9 million km² of this area is actually underlain by permafrost (Obu *et al.*, 2019). Lowland (non-alpine) permafrost accounts for 10.1 to 19.6 million km², mountain (alpine) permafrost accounts for 3.6 to 5.2

million km², and subsea permafrost accounts for about 2.5 million km² (Obu *et al.*, 2019; Overduin *et al.*, 2019). The Southern Hemisphere has three orders of magnitude less permafrost than the Northern Hemisphere, most of which occurs in Antarctica, where 21 700 km² is underlain by permafrost (IPCC, 2019). The Tibetan Plateau is the largest alpine permafrost area outside the polar regions, covering 1.1 million km² (IPCC, 2019). The 2.5 million km² of submarine permafrost formed when sea level was more than 100 m lower during past glacial periods. Though it has been degrading since inundation, subsea permafrost persists in areas of the Arctic continental shelves (Figure 21 blue-greenish colors, Overduin *et al.*, 2019).

3. Global carbon stocks and additional carbon storage potential

The cold temperatures and unique soil processes of permafrost have led to the accumulation of deep deposits rich in organic matter (Figure 20 and Figure 22, Table 20, Hugelius *et al.*, 2014). Understanding the amount and degradability of soil organic matter stored in permafrost is crucial as increasing temperatures in northern high latitudes lead to permafrost thaw and loss (Figure 23 and

Figure 24). This permafrost degradation can accelerate decomposition of organic matter previously stored in permafrost. Microbial decomposition produces carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), the three most influential long-lived greenhouse gases (Schuur *et al.*, 2015; Voigt *et al.*, 2020).

Globally, permafrost regions store ~1460–1600 Gt[7] of soil organic carbon (SOC; Hugelius *et al.* 2014, IPCC, 2013, 2019, Schuur et al 2015; Figure 22, Table 20). This represents approximately twice as much carbon as is currently present in the atmosphere (Figure 22). The rest of Earth's biomes, excluding the Arctic and Boreal regions, are estimated to contain 2 050 to 2 800 Gt SOC in the top 3 m of soil (Schuur *et al.*, 2015, Jackson *et al.*, 2017). This means that even though these northern regions account for only 15 percent of global soil area, they contain approximately 42 percent of global soil carbon (taking the 2 050 Gt from Schuur *et al.*, 2015). Recent studies suggest that up to half of the global soil carbon pool (estimated at 2 800 Gt C to a depth of 3 m; Jackson *et al.*, 2017) is stored in the permafrost region (Figure 22). In addition to these relatively well-constrained SOC pools, there could be an additional deep permafrost pool of 350-465 Gt C (mean ~400 Gt, Figure 22). This would be in addition to the already included deep SOC from yedoma (Strauss *et al.*, 2017) and Arctic delta estimates. This additional pool is estimated using a depth interval of 3-10 m and carbon content of 11–14 kg C/m³ (Schuur *et al.*, 2015).

Most of the SOC in permafrost regions occurs in circumarctic ecosystems (Figure 21). However, we estimate that alpine permafrost zones outside the circumarctic contain 83.2 Gt SOC (Table 20). This estimate includes SOC in global mountain permafrost (IPCC, 2019) and an updated estimate of SOC in the top 3 m of the Tibetan Plateau (36.6 Gt C; Ding *et al.*, 2019). We note that 46 percent of this Tibetan C is estimated to be in permafrost. There is less SOC in alpine permafrost compared to circumarctic permafrost because of its smaller area and lower C density (kg C/m³) (IPCC, 2019; Hugelius *et al.*, 2014). The same elevational pattern holds

[7] 1Gt = 1 billion tons

within the circumarctic, with mountain regions showing 50 percent less C density compared to tundra lowlands (Schuur *et al.*, 2015; Strauss *et al.*, 2017).

The permafrost coverage can be patchy and discontinuous, especially in the southern edge areas of the permafrost zone and/or areas of lower altitude. Because of this, only ~1000 Gt C (derived from Hugelius *et al.*, 2014, Strauss *et al.*, 2017, and mountain permafrost estimate in IPCC, 2019) of the global permafrost region C stock is stored in permafrost, while up to ~600 Gt C are stored in permafrost-free soils or sediments within the region (Table 20).

Besides C, nitrogen (N) stocks of permafrost soils are estimated to range between 22 to 106 Gt N, with a best estimate of 66 Gt N (Harden *et al.*, 2012). This N is of concern because it could constrain the loss and uptake of C and potentially cause a climate feedback via N_2O. If only a minor portion of this soil N is released as N_2O during nitrification and denitrification, the climate feedback loop from permafrost thaw and resulting greenhouse gas production would be even larger.

Table 20. Soil organic C stocks reported for permafrost

Unit	Depth (cm)	Region	SOC stock (Gt C)	stock uncertainty range (Gt C)	Reference
Turbels	0–300	lowland permafrost	476	359–593	Hugelius *et al.* (2014)
Orthels	0–300	lowland permafrost	98	61–135	Hugelius *et al.* (2014)
Histels	0 –300	lowland permafrost	153	139–167	Hugelius *et al.* (2014)
Histosols	0 –300	lowland permafrost	149	130–167	Hugelius *et al.* (2014)
Non-Gelisols, mineral	0–300	lowland permafrost	158	131–185	Hugelius *et al.* (2014)
Permafrost deep peatlands	>300	lowland permafrost	32	21–43	Hugelius *et al.* (2020)
Deltaic alluvium	>300–5400	lowland permafrost	91	39–143	Hugelius *et al.* (2014)

Unit	Depth (cm)	Region	SOC stock (Gt C)	stock uncertainty range (Gt C)	Reference
Yedoma region*	>300–5000	lowland permafrost	297	297–436	Strauss *et al.* (2017)
Mountain permafrost excl. Tibetan plateau	0–300	high altitude	47	na	IPCC (2019)
Tibetan plateau	0–300	high altitude	37	34–39	Ding *et al.* (2019)
Frozen in permafrost**		global	1024	920–1132	Hugelius *et al.* (2014) combined this synthesis;
Total permafrost region		global	~1538	1460–1600	This synthesis; IPCC (2019), Schuur *et al.* 2015
additional other deep deposits***	300–1000	lowland permafrost	400	unknown	Schuur *et al.* (2015)

*Lower boundary of the yedoma region minus the uppermost 3 m causes the difference to Strauss *et al.* (2017) estimate for full 0–50m yedoma pool (327 Gt C).

**Estimated assuming an active layer depth of 30 cm or more in all Gelisols/High Arctic soils and 46 percent of the Tibetan Plateau C perennial frozen.

***Rough estimate of potential permafrost carbon in regions with additional thick sedimentary overburden. Not included in any calculations yet due to very high uncertainties.

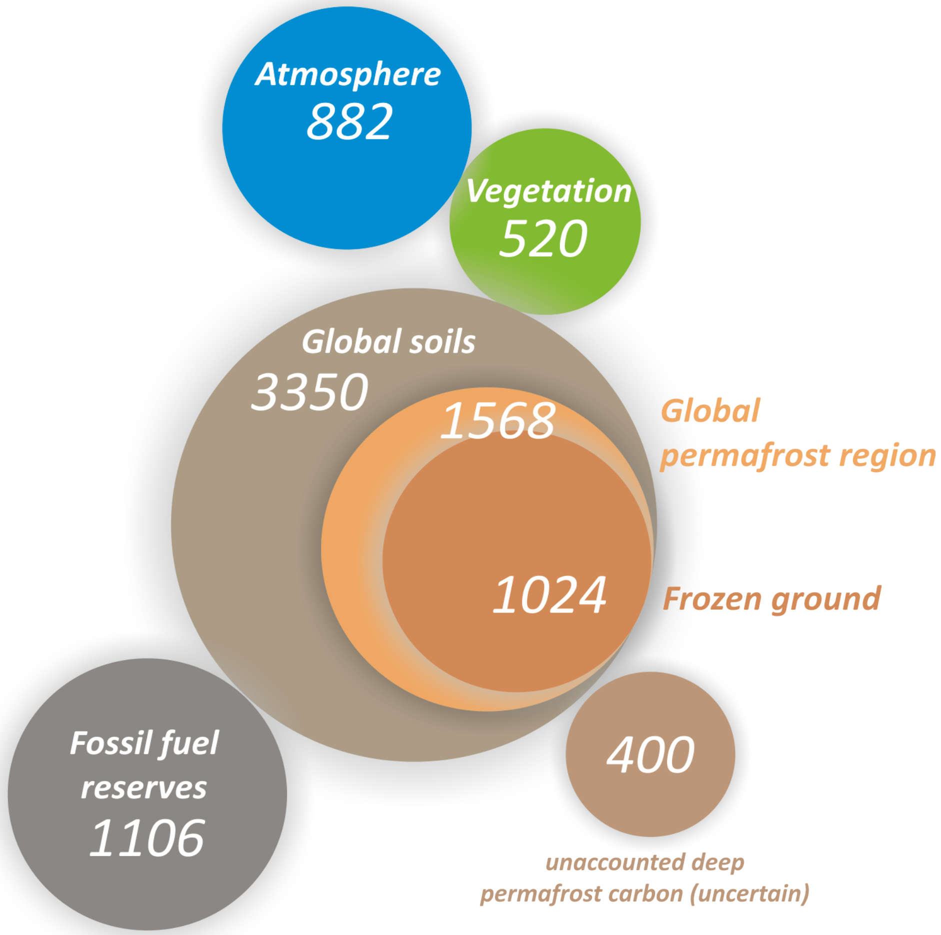


Figure 22. Terrestrial carbon stocks and atmospheric carbon in relation to the carbon stored in the permafrost region

The size of the circles is proportional to the size of the carbon stock. The stocks are given in gigatons (Gt)

The global soil estimate (3350 Gt) is based on soils to 3 m (2800 Gt) as well as other pools in deep permafrost (500 Gt) and tropical peatlands (50 Gt; Jackson *et al.*, 2017)

(Adapted and updated from Strauss *et al.*, 2017). Based on data from different International Panel on Climate Change (IPCC) reports (e.g. IPCC, 2019) and Hugelius *et al.* (2014; 2020)

Following IPCC 2013, the ocean stocks (not visualized) contain 900 Gt in the surface ocean, 37100 Gt in the intermediate and deep sea, 3 Gt in the marine biota and 700 Gt as dissolved organic carbon. For the ocean floor sediments 1750 Gt are estimated.

3.1. Potential mechanisms for additional C storage

While permafrost ecosystems typically support relatively low net primary productivity and total living biomass compared to temperate and tropical ecosystems (Abbott *et al.*, 2016), permafrost soils have sequestered C over tens of millennia through different natural mechanisms. The active layer of permafrost soils is exposed to seasonal cycles of freeze and thaw, which cause complex soil mixing processes called cryoturbation. Over time, cryoturbation incorporates SOC from the surface into deeper soil, where SOC is protected from decomposition, eventually becoming part of the permafrost. This is a key mechanism leading to the large SOC stocks in the soil sub-order Turbels (Table 20). Peat accumulation, both with and without permafrost, has also led to large C stocks in both Histels and Histosols (Table 20). While permafrost peatlands lose C to the atmosphere when they thaw, there is also potential for increased rates of C accumulation in existing peatlands associated with vegetation changes and the formation of new peatlands. The latter would require additional areas with suitable conditions, such as drained thermokarst lakes (Walter Anthony *et al.*, 2014) or newly exposed, poorly drained surfaces such as areas of coastal uplift following glacial recession (Treat *et al.*, 2019), or changes in environmental conditions that promote widespread peat formation. However, given that the formation of peat is a slow process, current projections suggest that C loss from thawing and draining peatlands will likely be larger than the gains for several centuries (Hugelius *et al.*, 2020).

In addition to cryoturbation and peat formation, substantial SOC accumulation occurred during the Pleistocene and Holocene from wind, water, and colluvial transport. These processes buried SOC in deep sediments, such as ice-rich yedoma deposition in the Late Pleistocene (Strauss *et al.*, 2017; Treat *et al.*, 2019). Solifluction (flow of soil downslope (Figure 24) buried and continues to bury surface C in valley bottoms. However, it is unclear how important this mechanism will be for organic matter preservation because solifluction areas are most prominent on moderately steep slopes where SOC density is often lower. Permafrost C can also be eroded, transported, and sequestered in river, delta, and ocean sediments (Figure 24), although the relative stability and residence time of this C is poorly constrained.

Increased vegetation growth in the permafrost region due to increasing air temperature and CO_2 fertilization may increase ecosystem C storage, but the uncertainty about this potential C sink is large. Stock observations show that the upper active layer of Tibetan alpine permafrost currently functions as a substantial regional C sink, implying that C losses of deeper and older permafrost C might be offset by increases in upper-active-layer SOC stocks. Other studies in Alaska found a net C loss due to losses from deep soils, despite enhanced vegetation growth with permafrost thaw, suggesting that there may be limits to vegetation C uptake in Arctic and Boreal regions (e.g. Schuur *et al.*, 2009). A simple C budget based on a complete biome shift suggests that vegetation could take up 11 Gt total, assuming a complete shift of all Arctic Tundra becoming Boreal Forest and all Boreal Forest becoming Temperate Forest (Abbott *et al.*, 2016), which is substantially less than projections from current models. Overall, whether increased vegetation growth is enough to compensate for the potential C losses with increased soil warming and permafrost thaw is an open question. The absolute size of the permafrost soil C pool versus the size of the current global vegetation C pool (Figure 21) suggests that a vegetation C sink may only provide a limited capacity to counter permafrost C losses.

3.2. Soil organic carbon loss potential

Ground temperature is increasing rapidly in all of the permafrost regions, particularly since the early 1980s. There has been a global mean increase of 0.3 ± 0.1 °C per decade at the depth of no seasonal temperature fluctuation (Figure 20) (Biskaborn *et al.*, 2019; IPCC, 2019). The mean warming of global permafrost has also been 0.3 °C per decade since 2007, based on a global network of permafrost boreholes, with the rate of increase varying regionally (IPCC, 2019). The warming and thawing of permafrost is projected to lead to widespread disturbance and disappearance of Boreal, Subarctic, and alpine permafrost during this century and large decreases of near-surface permafrost in the Arctic (Figure 23). This could have substantial consequences for the global climate. By 2100, the near-surface (0–3 m) permafrost area may decrease by 2–66 percent for the International Panel on Climate Change (IPCC) mitigation scenario (RCP2.6) and 30–99 percent for the high-emission scenario (RCP8.5) (IPCC, 2019). Between 2010 and 2300, simulations indicate a decrease of 6 to 16 million km^2 in permafrost area for the high-emission scenario (RCP8.5).

Projections of SOC stability are substantially more uncertain than projections of permafrost degradation. For the high warming scenario (RCP8.5), projected losses in SOC vary between 74 and 652 Gt C (mean loss of 341 Gt C; McGuire *et al.*, 2018). For this scenario, the C uptake by vegetation C is likely not large enough to compensate for the losses of permafrost C, with net changes in ecosystem C ranging from a 641 Gt C loss to a 167 Gt C gain (mean, 208 Gt C loss) (McGuire *et al.*, 2018). Under moderate warming (RCP4.5), gains in vegetation C across the circumarctic could result in overall net gains in ecosystem C by the year 2300 (-8 to 244 Gt C gains; RCP4.5 scenario; McGuire *et al.*, 2018). It is important to note that the spread between model results is very large and that many current models have only rudimentary representation of permafrost C and mechanisms of its mobilization across depths. This introduces uncertainty and potential underestimation of SOC mineralization.

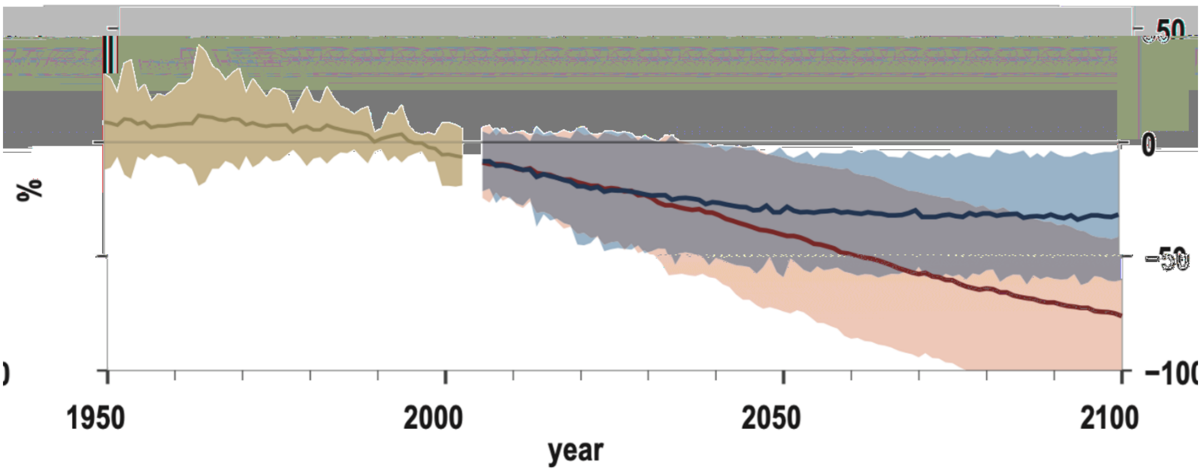


Figure 23. Projected permafrost areal change (x-axis) of the topmost 3 m until 2100

The high-emission scenario is illustrated in red (RCP8.5), the low-emission scenario (RCP 2.5) in blue. The greyish areas represent the overlap in the ranges

A reduction of up to 75 percent of the permafrost area, meaning a loss of more than 10 million km^2, is possible. (Adapted from IPCC, 2019).

One of the specific limitations of current modelling approaches is that models only simulate gradual, top-down thaw via a deepening of the active layer from the surface. Observations now show that permafrost containing high and moderate amounts of ground ice is affected by abrupt thaw events, such as thermokarst and thermo-erosion. These events can be triggered gradual warming, wildfires, excess rainfall, shore and hillslope erosion, human disturbance or other factors (Grosse *et al.*, 2011; Turetsky *et al.*, 2020). Abrupt permafrost disturbances are widespread across the permafrost distribution classes (i.e. continuous, discontinuous, etc.; Figure 21), including relatively warm and very cold permafrost regions (Nitze *et al.*, 2018). Thermokarst and thermo-erosion processes alter surface topography, hydrology, vegetation, soils, and C cycling. Thermokarst formation can create lakes (Figure 20, left side), mobilizing SOC previously stored in surrounding and underlying soil, but also acting as a C sink on centennial to millennial timescales (Turetsky *et al.*, 2020). Hydrological reorganization can cause inundation of surface soils, releasing CH_4 and CO2. Regions vulnerable to abrupt thaw include ice-wedge polygons in tundra lowlands (IPCC, 2019), ice-rich yedoma regions (Strauss *et al.*, 2017), and northern peatlands (Hugelius *et al.*, 2020). C loss from permafrost and thawed permafrost can also occur along rivers and coasts. Here the transport of dissolved and particulate C takes place with up to 20 m of lateral erosion per year (Fuchs *et al.*, 2020). Peatlands impacted by thermokarst also have high potential for N_2O emissions (Voigt *et al.*, 2020).

Given projections of increasing permafrost degradation during the 21st century, a corresponding loss of freeze-locked SOC together with increases in greenhouse gas emissions is anticipated (Schuur *et al.*, 2015; McGuire *et al.*, 2018; Hugelius *et al.*, 2020). Observations have shown that the magnitude of C loss and pathways (aerobic and anaerobic) is strongly related to the hydrology, and whether sites become wetter or drying upon thaw (Schuur *et al.*, 2015). To predict the moisture regime following thaw is complex and more progress is needed in the mapping of ground ice as well as model development to better project future changes.

Land use change and human impacts in permafrost regions may also alter soil C stocks. The degradation of permafrost can occur directly as a result of wildfire or land use in which the upper permafrost layer is disturbed. The construction of buildings, traffic routes and pipelines as well as agricultural activities can trigger gradual and abrupt permafrost degradation. As shown by Iwasaki *et al.* (2018), when forest was converted to arable land in Central Yakutia, a significant decrease in the total C content of the soil was observed, mainly due to mechanical disruption, decomposition, and removal of plant residues. As a result, there was only 41 percent of the SOC content in the cultivated soil compared to the original forest. After cessation of agricultural activity, vegetation recovery gradually restored some of the SOC. Pioneer species such as grasses and shrubs reestablished SOC over a 20-year period. However, new forest growth on some abandoned arable land follows the tendency of decreasing total C content due to a low level of productivity and a suppressive effect on grass vegetation. Yet, there is no data on the impact of land use change and human impacts on soil N stocks in permafrost regions.

Better integration of direct human disturbances, such as land use change, needs to occur to improve model estimates of the permafrost climate feedback. Widespread human activity in areas such as the Siberian boreal regions is rarely taken into account in predictions of SOC response (Crate *et al.*, 2017). Human activity reduces SOC via (I) use of thermokarst basins as pastures and hay making areas, (II) increased emission of CO_2 from moderately humid and humid grasslands in hot summers, and (III) significant CH_4 emissions from temporary flooded grasslands and thaw processes beneath thermokarst lakes and ponds that formed following deforestation or intensive agriculture in areas of ice-rich permafrost.

4. Importance of permafrost conservation for the provision of specific ecosystem services

The Arctic may seem remote and disconnected from current events, but the unprecedented environmental changes occurring there have important consequences for our global society. The loss of permafrost and associated greenhouse gas release could weaken the permafrost zone's service as a long-term C storage and sink (Schuur and Mack, 2018; IPCC, 2019). Thaw and release of just a fraction of this frozen C in the form of greenhouse gases into the atmosphere would accelerate and magnify global climate warming. This destabilizing feedback could cause further degradation of permafrost in both polar and mountain areas (Schuur *et al.*, 2015). It is unlikely that such large thaw induced losses could be compensated by increased plant growth or northward shifts in biomes. Because these permafrost feedbacks are still not incorporated into IPCC projections, current climate policy may not achieve desired targets.

In addition to the global consequences of GHGs emissions, permafrost thaw and degradation affects local habitats, degrading some of the last pristine areas on Earth. These local dynamics affect human communities living on permafrost through water quality and quantity, natural hazards, and stability of infrastructure and land loss. Changes in ground stability and weather patterns are altering travel routes, impeding access to culturally significant hunting and gathering areas and travel to other communities. Reliable transportation and timing of resources are fundamental to northern indigenous livelihoods.

Another ecosystem service that could be threatened by climate change is freshwater storage. Ground ice in the permafrost zone contains a globally-significant volume of freshwater: 22 to 300 × 10^3 km^3, which represents up to 90 cm sea level rise (Abbott *et al.*, 2019). While complete ground ice melt is not a realistic scenario for the 21st century, the projected widespread loss of near-surface permafrost, where most of the ground ice is located, suggests that this is a factor to be accounted for over the next few centuries.

In summary, permafrost is no longer permanent. Climate change and human disruption of the soil are causing irreversible changes to circumpolar and alpine permafrost areas.

4.1. Minimization of threats to soil functions

The only viable way to reduce permafrost soil threats is to reduce anthropogenic climate change. It appears that much of the SOC of the permafrost zone can be protected if human emissions are actively reduced. Specifically, greenhouse gas release, lateral C export, and disturbance such as wildfire and thermokarst are all reduced when human emissions are rapidly reduced (Abbott *et al.*, 2016; Turetsky *et al.*, 2020). Otherwise, because of its vast size and remote location, on-the-ground interventions are not feasible for most of the permafrost zone. Ice-rich permafrost, like the yedoma region, and steep mountain permafrost areas are particularly prone to hazards because permafrost and ground ice exert strong controls on ground stability (Krautblatter *et al.*, 2013, IPCC, 2019; Strauss *et al.*, 2017; Turetsky *et al.*, 2020). Projected permafrost thaw will affect Arctic hydrology and wildfire, with impacts on vegetation and soil. About 20 percent of Arctic land permafrost is vulnerable to abrupt permafrost thaw and ground subsidence, which is expected to increase small lake area by over 50 percent by 2100 for RCP8.5 (Turetsky *et al.*, 2020). Even as the overall regional water cycle intensifies, including increased precipitation, evapotranspiration, and river discharge to the Arctic Ocean, decreases in permafrost may lead to soil drying (IPCC, 2019) as the landscape loses its frozen underpinning. In mountain permafrost regions, permafrost degradation has changed some alpine ecosystems through altered soil temperature and permeability, decreasing the climate regulating service of a vast region and leading to lowered groundwater and new and shrinking lakes on the Tibetan Plateau. Minimizing these threats requires coordinated global action to limit anthropogenic warming as much as possible (IPCC, 2019).

4.2. Increases in production and food security

Food and water security have been and will be negatively impacted by changes in snow cover, lake and river ice, and permafrost in many Arctic regions. These changes have disrupted access to herding, hunting and fishing grounds, and caused the instability of agricultural land (IPCC, 2019).

Lowland permafrost is expected to contain a significant amount of natural mercury, which may be released into the environment after thaw, affecting drinking water and ecosystem food webs (IPCC, 2019). In some high mountain areas, water quality has been affected by contaminants, particularly mercury, released from melting glaciers and thawing permafrost already (IPCC, 2019). The release of heavy metals and other legacy contaminants currently stored in glaciers and permafrost, is projected to reduce water quality for freshwater biota as well as human household and agricultural use. Additionally, permafrost degradation can enhance the release of other elements (e.g., aluminum, manganese and nickel) (IPCC, 2019). Permafrost degradation is also a major and increasing source of bioavailable dissolved organic C, which can degrade drinking water and affect food webs in aquatic and marine ecosystems. The release of metals, C, and nutrients could consequently affect the food security of humans living in the permafrost zone.

4.3. Improvement of human well-being

The combination of thawing permafrost, loss of sea ice, extreme weather events, and rising sea level has multiple negative impacts on Arctic livelihoods Climate-driven environmental change harms the livelihoods, wellbeing, and cultural identity of all Arctic residents (AMAP, 2017; IPCC, 2019). In some Arctic regions, tipping points may have already been reached such that adaptive practices can no longer insulate local peoples from the worst effects of climate change. People displaced by the collapsing ground and eroding coastlines of the permafrost zone are among the first climate refugees. Coastal erosion and thawing permafrost forced entire villages to relocate at enormous economic and cultural cost (Welch, 2019).

Another risk from permafrost soil is the potential for thawing permafrost to release ancient pathogens (Legendre *et al.*, 2015, non-pathogenic in this case). A 2016 outbreak of anthrax likely from frozen ground on the Yamal Peninsula in Siberia led to the culling of more than 200 000 reindeer and the death of one human (Hueffer *et al.*, 2020). The potential for viruses and diseases to be revived from permafrost should be of concern in the context of global warming, though it is unclear how widespread or common such events could be.

Wildfire frequency and intensity are projected to increase during this century across most tundra and boreal regions (Abbott *et al.*, 2016), and also in some mountain regions. Interactions between climate and shifting vegetation will influence future fire intensity and frequency (Schuur and Mack, 2018; IPCC, 2019; Holloway *et al.*, 2020). The years 2019 and 2020 were characterized by extraordinary intense wildfire seasons in Siberia (NASA, 2020), as well as extreme heat waves in northern high latitudes. In Verkhoyansk, located in the northern part of the Republic of Sakha (Yakutia), a record temperature of 38 °C was measured in June 2020 (WMO, 2020). Fires endanger infrastructure and human well-being by reducing air quality. They also burn surface soil organic matter, causing an immediate release of soil C to the atmosphere. On longer timescales, wildfire can remove the insulating layer on top of permafrost soils, degrading permafrost and enhancing soil organic C decomposition (Holloway *et al.*, 2020).

Another challenge is that permafrost decline alters the frequency, magnitude and location of most of the natural hazards. Exposure of people and infrastructure to natural hazards has increased due to growing population, tourism and socio-economic development. Seventy percent of Arctic infrastructure is located in regions at risk from permafrost thaw and subsidence by the year 2050 (IPCC, 2019). Even cold Arctic permafrost in northern Siberia is projected to be affected by thaw subsidence by the end of the 21st century (Nitzbon *et al.*, 2020). In May 2020 the largest reported diesel spill to date in the Arctic region from a tank facility at a power plant in Norilsk was linked to infrastructure damage furthered by permafrost thaw likely caused by human disturbance.

Permafrost thaw also has negative impacts on infrastructure in high mountain areas (IPCC, 2019). Cable cars, mountain huts, power lines, and rockfall or avalanche protections built on permafrost in the European Alps, mostly found in the high mountain region above 2.500 m, have been destabilized by permafrost thaw (Krautblatter *et al.*, 2013). On the Tibetan Plateau, deformation or damage has been found on roads, power lines and an oil pipeline. Tourism and recreation activities such as hiking, skiing and mountaineering have been negatively affected by permafrost thawing. In several regions, worsening trail safety has reduced mountaineering opportunities and will further endanger subsistence and recreational activities in mountainous areas.

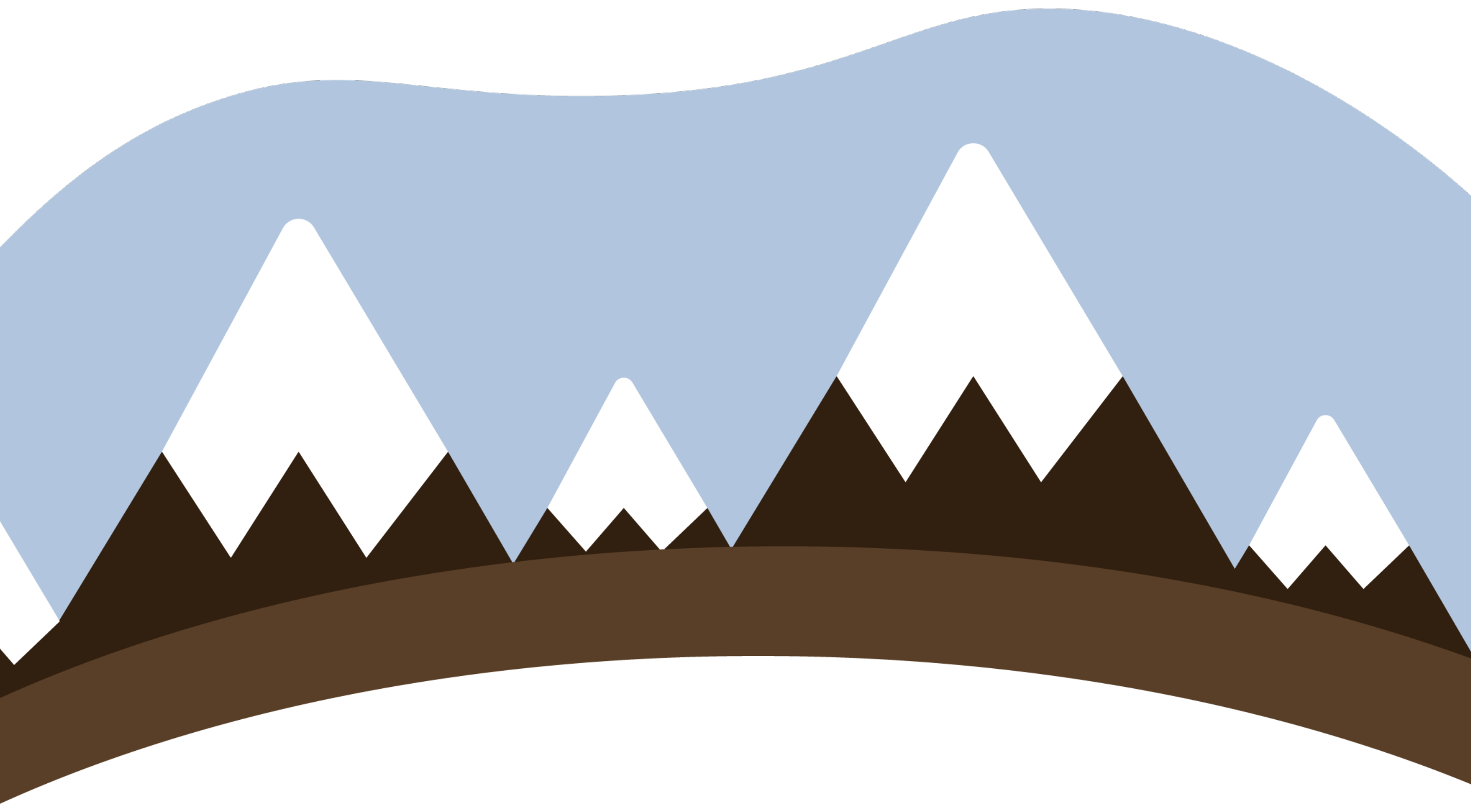
4.4. Mitigation of and adaptation to climate change

Arctic residents, especially indigenous peoples, have adjusted the timing of important activities and practices to respond to changes in seasonality and safety of land, ice, and snow travel conditions. Municipalities and industry are beginning to address infrastructure failures associated with flooding and thawing permafrost and some coastal communities are planning village relocations. Retrofitting and redesigning infrastructure has the potential to halve the costs arising from permafrost thaw and related climate-change impacts by 2100. For infrastructure on permafrost, engineering practices suitable for polar and high mountain environments have been developed to support adaptation (Doré *et al.*, 2016). It is suggested that effective mitigation efforts during the remainder of this century could attenuate the negative consequences of the permafrost climate feedback.

5. General challenges and trends

Permafrost thaw is expected to be irreversible on time scales relevant to human societies and current ecosystems. Long response times of decades to millennia mean that the permafrost region is committed to long-term change even after anthropogenic greenhouse gas and radiative forcing stabilize. Thawing of permafrost involves thresholds that allow for abrupt responses to ongoing climate warming. These characteristics pose risks and challenges to adaptation. The cryosphere also amplifies climate changes through snow, ice and permafrost feedbacks. The permafrost C feedback is a self-reinforcing one (Schuur *et al.*, 2015).

Global-scale permafrost thaw is projected to continue in the near-term (2031–2050) due to surface air temperature increases, ocean water temperature increases, and the ice-free season extension, with unavoidable consequences for river runoff and local hazards such as surface subsidence or coastal erosion. This leads to loss of soil stability, threatens livelihoods and potentially release of additional C into the atmosphere. By 2100, projected near surface (within 3–4 m) permafrost area shows a decrease of 24 ± 16 percent for the mitigation scenario (RCP2.5) and 69 ± 20 percent for higher emission (RCP 8.5) scenarios (IPCC, 2019). This last scenario leads to the cumulative release of substantial permafrost C as CO_2 and CH_4 to the atmosphere by 2100 with the potential to exacerbate climate change. Even larger emissions are projected from processes not yet included to models, such as abrupt thaw (Nitzbon *et al.*, 2020; Turetsky *et al.*, 2020) and fine-scale ecological interactions (Keuper *et al.*, 2020). Lower emissions scenarios dampen the response of C emissions from the permafrost region. CH_4 contributes a small fraction of the total additional C release but is significant because of its higher warming potential (28–36-fold warming potential compared to CO_2 over 100 years, Schuur *et al.* 2015). Increased plant growth is projected to replenish or partly offset soil C losses in the short-term, but will not match C releases over the long term or at high rates of C loss. The present-day N2O emissions of permafrost soils are estimated at up to 7 percent of the total N2O emissions from natural soils (Voigt, 2020), but the future release is yet poorly constrained. It has been shown, however, that climate-change related disturbances favor N2O production and release (Elberling *et al.*, 2010; Voigt *et al.*, 2017)



Future climate-induced changes in permafrost will drive habitat and biome shifts (Schuur and Mack, 2018), with associated changes in the ranges and abundance of many species. Even as the overall regional water cycle is projected to intensify, including increased precipitation, evapotranspiration, and river discharge to the Arctic Ocean, decreases in permafrost may lead to soil drying with consequences for ecosystem productivity.

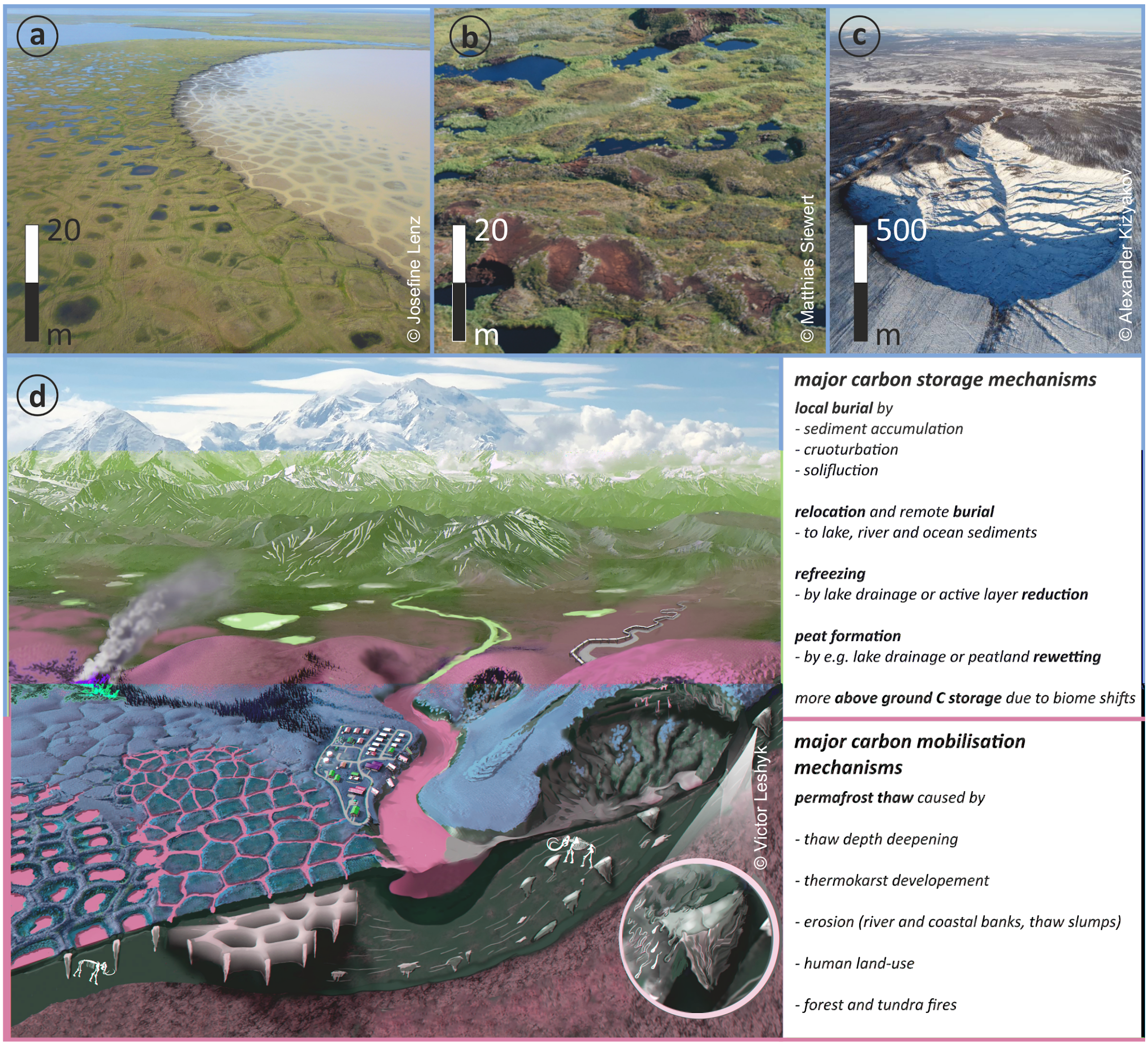


Figure 24. Permafrost degrades as the ice in the ground melts in response to e.g. climate warming, human disturbance, or more wildfires

The resulting ground collapse causes permafrost ecosystems to subside and erode. Previously frozen permafrost soil carbon can escape to the atmosphere via microbial action or be carried away by water. This image depicts features of a permafrost landscape with a focus on lowland permafrost of the Northern Hemisphere

a) Thermokarst degradation by lake expansion in northern Alaska

b) Palsa peatland complex in Tavvavuoma, Sweden

c) Batagai thaw slump in the boreal zone of Yakutia, Russia. The slump is more than 900 meters wide

d) Cartoon of major processes and landscape features in Schuur and Mack (2018)

6. Related terminology

6.1. Soil related terminology

Turbels: cryoturbated permafrost soils

Orthels: non-cryoturbated permafrost-affected mineral soils

Histels: organic permafrost soils

6.2. Permafrost specific terms

Simplified from van Everdingen *et al.* 2005

Active layer: top layer of ground subject to seasonal thawing and freezing in areas underlain by permafrost

Cryoturbation: soil movements causes by to freeze-thaw cycles, including expansion and contraction due to temperature changes and the growth and disappearance of ground-ice bodies,

Ice wedge: A massive, generally wedge-shaped body with its apex pointing downward. Ice wedges occur in thermal contraction cracks in which water from melting snow penetrates in the spring. Repeated annual contraction cracking of the ice in the wedge, followed by freezing of water in the crack, gradually increases the width and depth of the wedge

Lowland permafrost: Permafrost existing in high latitudes and outside alpine areas

Mountain permafrost (also alpine permafrost): Permafrost existing at high altitudes, also occurring in middle and low latitudes

Permafrost: Ground (including soil or rock) that remains at or below 0 °C for at least two following years

(Ice wedge) polygons: A type of patterned ground consisting of a closed, roughly equidimensional figure bounded by more or less straight sides. Causes by soil shrinking, water infiltration and thick wedged shape ice bodies (ice wedges) in the ground.

Solifluction (also frost creep): Slow downslope flow of saturated unfrozen earth materials

Talik: A layer or body of unfrozen ground within or through permafrost

Thaw subsidence: Drop in elevation of the ground surface due to ice volume loss caused by thaw

Thermo-erosion: The erosion of ice-rich permafrost by the combined thermal and mechanical action of moving water

Thermokarst: Process: melting of excess ground ice and subsequent thaw settlement, often caused by a water body (thermokarst lake); Landform: topography resulting from the melting of excess ground ice and subsequent thaw settlement. Thermokarst terrain is so named because of its superficial resemblance to the karst topography typical of limestone regions

Yedoma: Pleistocene ice-rich permafrost with syngenetic ice-wedges. Widespread in Siberia, Alaska, and Yukon (Canada) and prone to rapid-thaw processes.

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Drylands

10. Drylands

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1. Definition and description

The expectation has been raised that carbon sequestration in soils can provide a bridge to reduce the impacts of increased carbon emissions until sufficiently clean and efficient technologies are available to replace fossil fuel burning (Lal, 2004; Smith, 2004).

Recommended management practices for soil restoration and vegetation cover can have a positive impact on the sequestration of soil organic carbon (SOC) and the provision of ecosystem services, especially for the population living in drylands, and particularly for small rural producers, who are the most vulnerable to degradation (Thomas, 2008). Capturing SOC through the restoration of degraded drylands is paramount, especially in regions where it is technically and socioeconomically a viable option (FAO, 2002). The capacity for carbon sequestration in drylands is potentially high because their historical losses due to degradation are still far from the maximum point of restoration (United Nations, 2011).

In order to determine the contribution of drylands to the potential for SOC sequestration, a meta-analysis is presented with bottom-up scaling, based on information from 33 local studies and nine regional studies, with measurement periods or projection long enough (34±23 years) to reflect annual variations in SOC stores, in response to land use under different management practices and diverse environmental conditions. For this study, the potential for sequestration of SOC was quantified based on the dryland surface of UNEP-WCMC (Sörensen, 2007) comparing other available sources with a high level of agreement (geographic area with intersection) but with different delimitation approaches (Trabucco and Zomer, 2019; Sayre *et al.*, 2020). These approaches use (with different precision and breadth) the various intervals of the aridity index (AI), which was established by the United Nations Environment Program, and which represents the dimensionless relationship between annual precipitation (P) and potential evapotranspiration (PET) (Trabucco and Zomer, 2019). The AI = 0.65 divides the planet's lands into two classes: arid and humid, referred to as dry and humid domains. There are four subclasses of the dry domain: hyper-arid, arid, semi-arid, and sub-humid, that correspond to the upper intervals of the AI 0.05, 0.2, 0.5, and 0.65, respectively (Cherlet *et al.*, 2018).

2. Global distribution of hotspot

There is no absolute consensus on the distribution of drylands on a global scale. The area can vary from 26.3 percent by Köppen (1931) to 52.3 percent by UNCBD (2007). Divergences occur due to the different concepts and objectives on the part of the organizations in charge of promoting their knowledge and conservation. The United Nations Convention to Combat Desertification (UNCCD) excludes hyper-arid zones (AI <0.05) as they are regions not prone to desertification (Zdruli, Kapur and Celik 2010). At the same time, the United Nations Convention on Biological Diversity (UNCBD) considers not only hyper-arid ecosystems but also other ecosystems with higher humidity (AI> 0.65) that are functionally connected and that in some cases are difficult to separate from drylands (Sörensen, 2007).

In 1977 the United Nations Conference on Desertification (UNCOD) adopted a Plan of Action to Combat Desertification (PACD). Despite this and other efforts, the United Nations Environment Programme (UNEP) promoted the production of the world's aridity maps. One of the first maps was created by the Climate Research Unit (CRU) of the University of East Anglia (UEA) and the Global Resource Information Database (GRID) of the United Nations Development Program (UNDP) for the first edition of the World Desertification Atlas (Harris *et al.*, 2014). Based on this information, the drylands limits were defined for the Millennium Ecosystem Assessment in 2000. Subsequently, some authors updated this information to improve the precision of the limits of tropical dry and sub-humid forests and deserts (Miles *et al.*, 2006) or include additional information on drylands under an ecoregions approach (Sörensen, 2007).

There are currently three maps that are comparable concerning the global distribution of drylands. The first is the map updated by the World Conservation Monitoring Center, based on the aridity index and functionality criteria between the ecoregions (map UNEP-WCMC, produced by Sörensen in 2007). The second is the map generated by the Consortium for Spatial Information (CGIAR-CSI), based exclusively on the aridity index (produced by Trabucco and Zomer, 2009 and updated in 2019). This second map is in the third edition of the World Atlas of Desertification (map UNEP-EC in 2018). A third global map on drylands is part of the World Terrestrial Ecosystems platform produced by ESRI, Nature Conservancy and USGS (Sayre *et al.*, 2020), however, this aridity map represents only two main intervals of the aridity index (upper intervals of 0.2 and 0.65) (Figure 25).

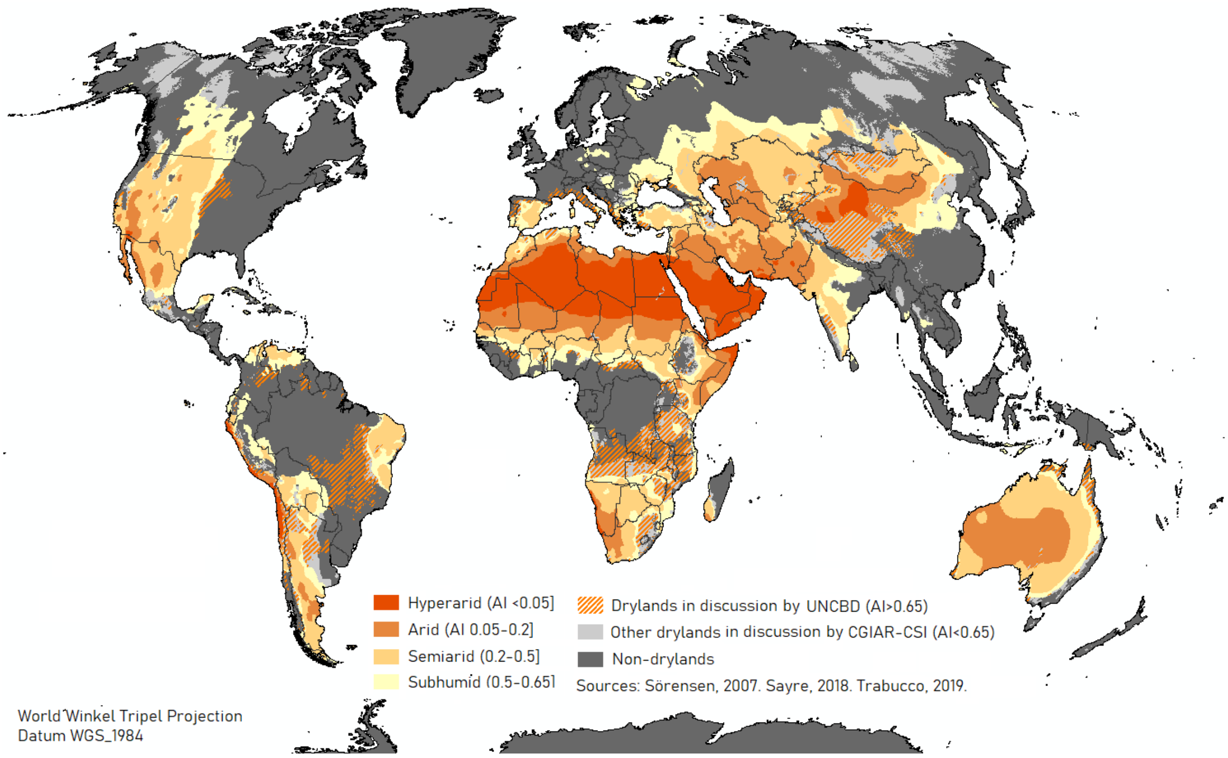


Figure 25. World distribution of drylands according to the level of aridity and other criteria used by sources of information

According to the UNEP-WCMC global aridity map (Sorensen, 2007) the area of drylands is 61.2 Mkm^2 which represents 41.7percent of the land surface. Other sources indicate an area of 65.5 Mkm^2 (Sayre *et al.*, 2020) and 66.4 Mkm^2 (Trabucco and Zomer, 2019) (Table 21).

Table 21. Global Area of drylands according to Aridity Index (AI) by source

Aridity	AI	Sorensen, 2007 (UNEP-WCMC)		Sayre, 2018 (USGS-ESRI)		Trabucco, 2019 (CGIAR-CSI)	
		Mkm^2	%Area	Mkm^2	%Area	Mkm^2	%Area
Hyperarid	<0.05	9.8	6.7	11.6	7.9	11.0	7.5
Arid	0.05–0.2	15.7	10.7			18.0	12.3
Semiarid	0.2–0.5	22.7	15.4	53.9	36.7	24.2	16.5
Dry subhumid	0.5–0.65	13.1	8.9			13.2	9.0
Total Drylands		61.2	41.7	65.5	44.6	66.4	45.3

Source: Sorensen (2007), Sayre et al. (2020) and Trabucco and Zomer (2019)

Note: Proportion of the area is based on the total world land area **(146.7 Mkm^2).**

3. Global carbon stocks and additional carbon storage potential

3.1. Global SOC stocks

The drylands are incredibly diverse. Even in the most inhospitable deserts such as the Atacama or Namibia, the processes of respiration and evaporation in the soil occur differently between riparian zones, springs, beaches, and areas of bare soil. This diversity complicates estimating the magnitude of SOC stores and the rates of loss or absorption (Goudie, 2013). Some factors determining the accumulation level of SOC depend on current land use (e.g. the micro-relief and plant structure). In contrast, other factors depend on the geological history (e.g. the texture or the mineralogical composition of clays) (Hong *et al.*, 2020) (Table 22). According to the world map of drylands (UNEP-WSCS) and the SOC world map (FAO and ITPS, 2020), drylands represent a store of 216.1 PgC in the first 30 cm of depth (35.3 ± 18.3 tC/ha as a world average). This store represents 28.6 percent of the world's SOC content (755 Pg C) for the same depth (Batjes, 2016). Of the cumulative total, 38.8 PgC is by tropical drylands, 31.9 PgC by subtropical drylands, 27.1 PgC by warm temperate drylands, 83.9 PgC by cool temperate drylands, 31.4 PgC by boreal drylands and 3.1 PgC by polar drylands (Table 22)

Table 22. Total area and soil organic carbon content of drylands, by aridity level, temperature regime, and landform

Aridity	Climate zone	Mkm²	SOC			Landform (t SOC/ha)			
			t/ha	SD	Pg	Plains	Hills	Mount	Tablelands
Hyperarid	Tropical	5.0	13.1	4.1	6.6	3.4	1.0	0.5	0.1
	Subtropical	3.9	13.5	5.2	5.3	2.3	0.9	0.5	0.3
	Warm Temperate	0.5	22.9	6.8	1.1	0.2	0.1	0.2	0.0
	Cool Temperate	0.3	29.0	8.1	0.9	0.1	0.1	0.1	0.0
	Boreal	0.0	46.0	7.0	0.1	0.0	0.0	0.0	0.0
	Polar	0.0	53.3	21.2	0.1	0.0	0.0	0.0	0.0
	Subtotal	9.8			14.2	5.9	2.1	1.3	0.5
Arid	Tropical	4.5	17.6	8.6	7.9	3.6	0.5	0.4	0.1
	Subtropical	5.5	17.4	8.1	9.6	3.5	0.8	1.0	0.2
	Warm Temperate	2.8	18.6	11.3	5.3	1.5	0.4	0.9	0.1
	Cool Temperate	2.7	32.3	12.6	8.7	1.3	0.7	0.5	0.1
	Boreal	0.1	51.5	14.9	0.6	0.0	0.0	0.1	0.0
	Polar	0.1	68.4	28.4	0.6	0.0	0.0	0.1	0.0
	Subtotal	15.7			32.5	9.9	2.4	2.9	0.5

Aridity	Climate zone	Mkm²	SOC			Landform (t SOC/ha)			
			t/ha	SD	Pg	Plains	Hills	Mount	Tablelands
Semiarid	Tropical	5.1	25.0	12.3	12.8	3.6	0.9	0.4	0.2
	Subtropical	3.8	26.9	14.1	10.3	2.1	0.7	0.8	0.2
	Warm Temperate	4.7	29.6	18.3	14.0	1.6	0.9	1.9	0.3
	Cool Temperate	7.6	53.6	34.2	41.0	3.3	1.7	2.2	0.5
	Boreal	1.2	68.9	31.5	8.1	0.1	0.3	0.6	0.1
	Polar	0.1	95.8	50.2	1.3	0.0	0.0	0.1	0.0
	Subtotal	22.7			87.5	10.8	4.6	6.0	1.3
Dry Subhumid	Tropical	3.1	37.3	19.7	11.5	1.6	0.9	0.4	0.1
	Subtropical	1.7	40.6	19.9	6.7	0.6	0.3	0.6	0.1
	Warm Temperate	1.7	40.2	18.7	6.7	0.7	0.2	0.6	0.1
	Cool Temperate	4.3	78.0	42.8	33.3	2.0	0.9	0.9	0.4
	Boreal	2.3	99.4	50.2	22.6	1.2	0.5	0.4	0.2
	Polar	0.1	88.4	73.0	1.1	0.0	0.0	0.1	0.0
	Subtotal	13.1			81.9	6.2	2.9	3.0	1.0
Total Drylands		61.2	35.3		216.1	32.8	11.9	13.2	3.3

Source: Sorensen (2007), Sayre et al. (2020) and Trabucco and Zomer (2019)

SD refers to Standard Deviation

Climate, altitude, the slope of the land, and the type of soil are some of the factors that determine the magnitude and distribution of SOC stores in drylands. Usually, there is a higher SOC density in landscapes with higher altitude and slope percentage. The SOC increases due to the higher availability of moisture are more significant and soils are less accessible to humans and cattle. Sandy soils (e.g. Arenosols), underdeveloped soils (e.g. Aridic Regosols), and heavily degraded soils (e.g. some Densic Planosols, or Nudi-natric Solonetz) have lower SOC densities. Strongly organic soils (e.g. Humic Andosols), soils formed by re-washed aeolian sediments (loess) or by the accumulation of illuvial clays in flat or slightly undulating grasslands, in environments with cold winters and hot summers (for example, Luvic Chernozems) constitute the densest SOC reserves (Figure 26).

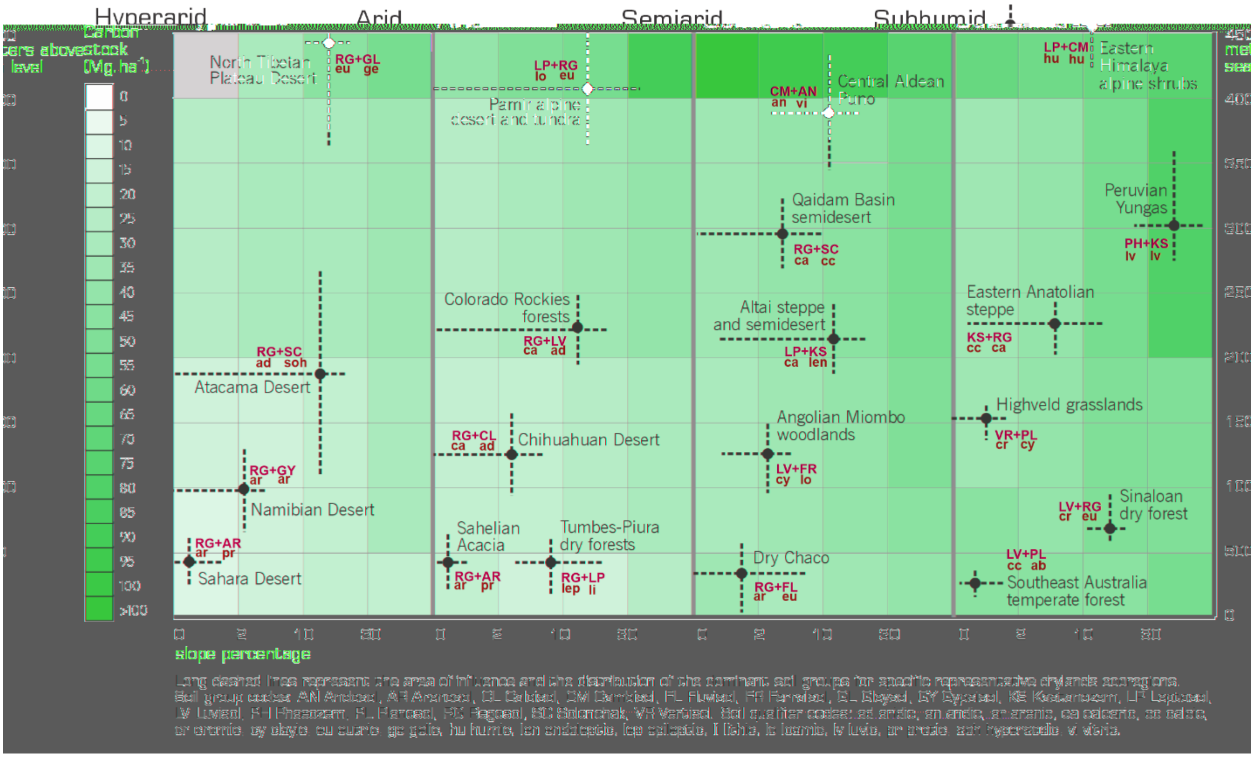


Figure 26. General view of the magnitude and distribution of global SOC stores in drylands (to 30 cm depth)

This by aridity level, showing transitions in the relief (altitude, slope) and the dominant groups and qualifiers soil (according with World Reference Base, 2014), in some representative ecoregions. (Data sets sources: Sörensen, 2007; FAO and ITPS, 2020).

Tibet's steppe organic soils have 12 times more SOC intensity (194 gC/kg soil) than the iron-rich mineral soils of the Atlantic Forest of Brazil (16 gC/kg soil). However, Atlantic Forest bulk density soils are 12 times heavier (1.65 t/m^3) than the soils of Tibet (0.14 t/m^3), indicating equality in both stores' SOC. Tibet's soils have a completely different accumulation process than the Brazilian soils formed millions of years ago (Driessen *et al.*, 2001; FAO and ITPS, 2020).

On the other hand, the most extensive soil inorganic carbon deposits are in drylands due to secondary carbonate translocation's systematic processes (Sombroek, Nachtergaele and Hebel, 1993; Batjes, 1999). Inorganic carbon fixation occurs through the movement of HCO_3^- into groundwater and closed systems (Driessen *et al.*, 2001). Some authors indicate that the carbonate binding potential can contribute to capture up to 0.26 PgC/yr in drylands (Dumanski *et al.*, 2006).

Table 23. Total area and SOC stocks of drylands by aridity index, temperature class for land use classes

Aridity	Climate zone	Total		Forest		Croplands		Grasslands		Shrublands		Sparse/None		Snow-Ice		Settlements	
		Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha
Hyperarid	Tropical	500.5	13.1	0.5	21.6	4.8	16.9	1.0	25.3	4.0	25.1	489.5	12.9	0.0	ns	0.8	13.9
	Subtropical	393.9	13.5	0.5	37.2	5.5	20.2	0.7	37.3	3.9	20.2	382.3	13.3	0.0	ns	1.1	22.3
	Warm Temperate	49.8	22.9	0.1	16.4	0.4	35.0	1.8	28.1	2.2	17.7	45.1	22.8	0.0	ns	0.1	19.3
	Cool Temperate	32.6	29.0	0.0	ns	0.2	41.7	2.9	35.6	0.3	26.2	29.2	28.3	0.0	ns	0.1	37.6
	Boreal	2.2	46.0	0.0	ns	0.0	ns	0.7	50.2	0.0	ns	1.5	44.0	0.0	ns	0.0	ns
	Polar	1.6	53.3	0.0	ns	0.0	ns	0.4	57.9	0.1	28.8	1.1	53.1	0.0	ns	0.0	ns
	Average		14.4		28.5		19.7		35.2		21.8		14.1		ns		19.4
	Subtotal	980.5		1.1		10.9		7.4		10.4		948.6		0.0		2.0	
Arid	Tropical	448.6	17.6	10.0	26.5	47.1	21.5	61.8	17.2	60.6	24.2	268.7	15.1	0.0	ns	0.3	23.3
	Subtropical	548.3	17.4	11.2	34.8	18.0	24.9	82.4	20.5	62.8	15.6	373.3	16.2	0.0	ns	0.6	18.9
	Warm Temperate	283.9	18.6	5.1	38.3	24.2	26.0	22.5	24.0	63.2	18.4	168.5	16.3	0.0	ns	0.4	18.2
	Cool Temperate	268.1	32.3	1.8	28.2	14.2	46.1	55.4	38.1	15.8	24.3	180.7	30.1	0.0	ns	0.2	49.1
	Boreal	11.2	51.5	0.3	78.2	0.5	72.2	3.3	56.0	0.7	62.4	6.6	45.5	0.0	ns	0.0	ns
	Polar	8.8	68.4	0.2	106.7	0.1	78.0	2.7	58.6	1.5	76.1	4.0	66.7	0.3	117.9	0.0	ns
	Average		20.7		33.0		26.8		25.2		20.3		18.8		117.9		23.6
	Subtotal	1569.0		28.6		104.1		228.0		204.6		1001.9		0.3		1.5	
Semiarid	Tropical	514.6	25.0	52.0	37.1	195.3	21.8	47.6	21.0	137.0	29.0	81.6	20.3	0.0	ns	1.2	24.8
	Subtropical	382.0	26.9	53.7	37.8	67.1	28.4	48.9	24.6	153.4	25.7	57.4	20.4	0.0	ns	1.4	25.0
	Warm Temperate	474.5	29.6	47.2	42.9	139.3	34.1	54.8	29.6	145.6	24.9	83.4	22.4	0.0	ns	4.3	32.7
	Cool Temperate	764.4	53.6	82.2	64.7	249.2	70.5	216.9	47.0	85.3	29.9	128.6	40.4	0.0	ns	2.2	75.4

Aridity	Climate zone	Total		Forest		Croplands		Grasslands		Shrublands		Sparse/None		Snow-Ice		Settlements	
		Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha	Mha	t/ha
	Boreal	117.6	68.9	50.7	73.8	14.1	73.6	28.4	68.5	2.3	71.1	22.0	54.9	0.0	ns	0.1	103.0
	Polar	13.4	95.8	2.2	158.5	1.0	87.5	4.9	84.2	1.5	77.8	2.8	84.7	1.0	81.5	0.0	ns
	Average		38.6		53.4		44.5		40.9		27.4		30.2		81.5		41.3
	Subtotal	2266.5		287.9		666.1		401.5		525.0		375.7		1.0		9.1	
Dry Subhumid	Tropical	308.0	37.3	118.0	46.4	115.9	30.9	10.4	32.5	61.3	33.0	1.4	35.6	0.0	ns	1.1	30.7
	Subtropical	166.2	40.6	59.7	47.1	39.4	36.8	14.8	33.8	50.5	38.0	1.1	33.9	0.0	ns	0.8	36.1
	Warm Temperate	165.6	40.2	22.3	49.3	100.1	39.4	15.5	39.8	16.5	37.8	7.0	28.7	0.0	ns	4.2	38.2
	Cool Temperate	427.0	78.0	127.6	78.9	225.6	82.1	46.1	62.8	11.8	77.8	10.7	50.9	0.1	122.4	5.0	70.1
	Boreal	226.9	99.4	182.1	95.9	12.3	94.6	5.4	102.5	21.2	131.3	5.7	102.8	0.1	42.6	0.2	54.3
	Polar	12.3	88.4	1.0	191.4	0.1	15.0	3.0	85.6	1.1	117.5	6.0	38.9	1.1	160.8	0.0	ns
	Average		62.7		72.7		58.1		54.2		51.7		51.7		147.2		51.8
	Subtotal	1306.0		510.8		493.3		95.0		162.4		31.9		1.4		11.2	
Average			35.3		48.3		38.9		38.8		29.9		29.3		91.8		35.5
Total		6122.0		828.4		1274.5		732.0		902.5		2358.1		2.7		23.9	

Sources (data sets): Sorensen (2017), Sayre et al. (2020), FAO and ITPS (2020), Trabucco and Zomer (2019)

NS = not significant

3.2. Additional SOC storage potential

The depletion of SOC stocks has created a world-historical deficit (78 ± 12 PgC) that currently represents, paradoxically, an opportunity for restoration (Lemus and Lal, 2005). This reduction in SOC storage is mainly due to soil degradation and inadequate management.

From the monitoring of various vegetation indices (NDVI) observed between 1999 and 2013 by the Copernicus Global Land Spot satellite, at intervals of every ten days and 1 km resolution, it is confirmed that

14.8 percent of the drylands surface maintains a persistent decline in productivity, that is, a severe stagnation in their ability to sustain stable productivity (Cherlet *et al.*, 2018).

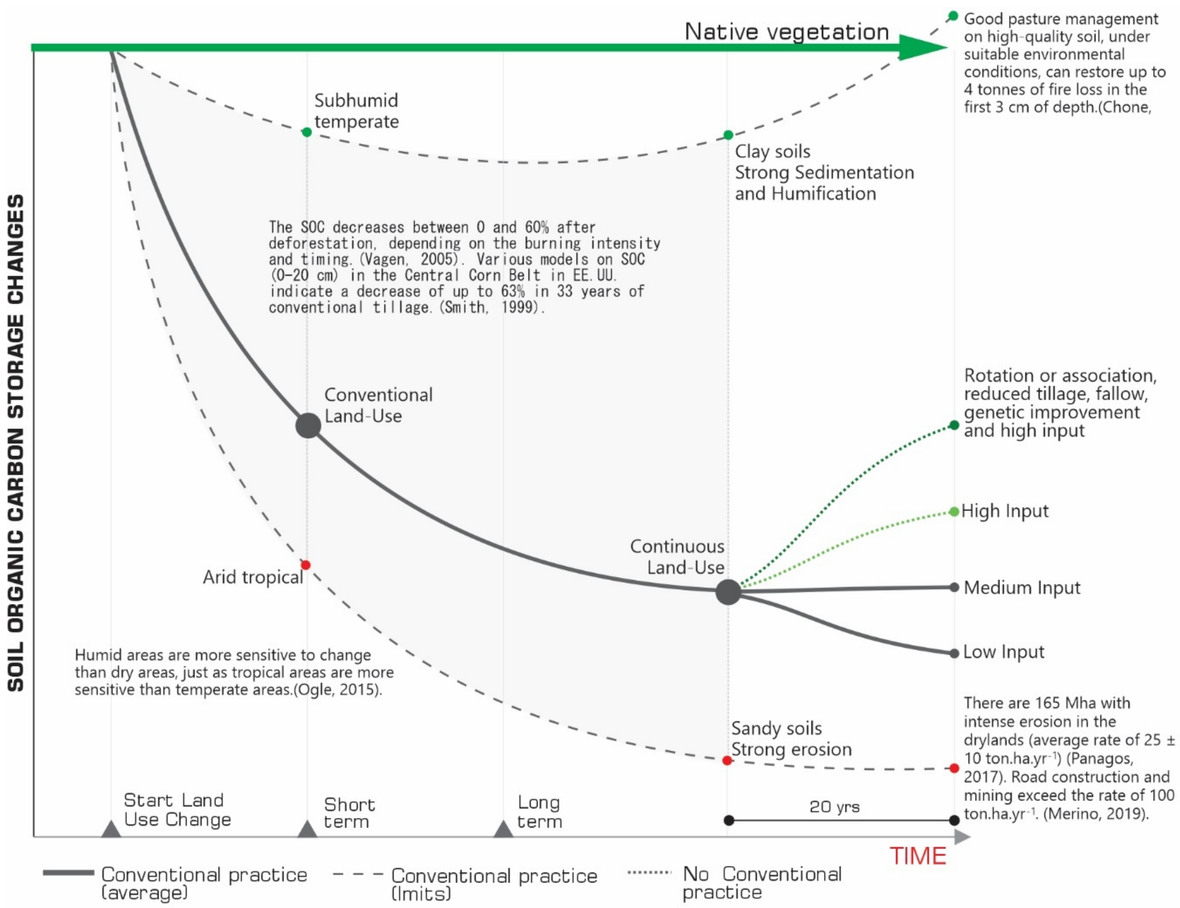
A strategy to reverse this negative trend can be the annual global sequestration of 0.8 ± 0.4 PgC in the soil (Lal, 2004), which means 8.5 percent of the total fossil emissions (9.46 ± 0.5 PgC/yr), or more than half of the emissions from land-use change and deforestation (1.5 ± 0.71 PgC/yr) during the period 2007–2016 (World Meteorological Organization, 2020).

This scenario will only be possible if the soils are managed through better production systems: optimization of tillage and fallow time, responsible and ethical use of irrigation and fertilizers, use of crop residues and cover crops, better prevention measures against hydrometeorological and market risks, as well as a growing rehabilitation of tree and shrub ground cover, for instance. These conservation and restoration practices will be particularly notable in the most degraded soils due to their high restoration threshold (historical loss of 58 ± 8 percent SOC due to degradation) (Lal, 2004). The effectiveness of restoration comes from of a combination of measures, according to the situation leading to the existing degradation (Carvalho and Lourenço, 2014).

The largest agricultural drylands area is in the semi-arid region; however, degraded agricultural lands in the sub-humid region have the most significant potential for capture. One reason is that the litter decomposition rate is higher in sub-humid soils (0.45–2.00 gC/yr) than in dry soils (0.001–0.44 gC/yr), due to the more microbial activity and significant amount of edaphic fauna and other favorable abiotic factors (García-Oliva *et al.*, 2006) such as the average tree dossal coverage, which in the sub-humid zone is almost four times higher than in the semi-arid zone.

Another reason is that the sub-humid lands maintain a greater distance towards the restoration threshold than semi-arid lands. This distance is due to the greater intensity and frequency of fires, and higher deforestation and erosion rates in this region (Table 24).

Various factors will impact the potential for sequestration in the short and long term: the degradation baseline (distance to the restoration threshold), the intensity and direction of climate change (Pataki *et al.*, 2003), the complex land management (Zdruli *et al.*, 2017), and the natural heterogeneity of the ecosystem (altitude, latitude, shape, and roughness of the relief) (Thompson and Kolka, 2005), including physical and biochemical processes that occur at the first meter of soil depth (Driessen *et al.*, 2001) (Figure 27).



Sources: Conceptual framework for the IPCC method supported with data of Althof, 2015, Alvarez et al., 1998, Ardo, 2004, Baez et al, 2017, Bessam y Mrabet, 2003, Chan, 1992, Dalal et al, 1995, FAO, 2004, FAO, 2020, Farage et al., 2007, Gomez et al., 1995, Gonzalez, 2020, Helyar et al, 1997, Hong, 2020, Horner et al, 1960, Madeira et al., 2012, Merino, 2019, Pinter, 2008, Reeder and Shuman, 2002, Shuman, 2009, Singh et al., 1998, Tarre et al, 2001 and Tschakert, 2004.

Figure 27. The conceptual framework for the IPCC (2019) method

Long-term cultivation generally causes substantial losses of SOC from the amounts found under native vegetation. Non-conventional practices and high inputs can alter the long-term loss of SOC in drylands. This scheme shows higher SOC stability in humid, clay, and healthy soils than in dry, sandy, and eroded soils (McSherry and Ritchie, 2013).

Estimation method

For global assessment purposes SOC sequestration potential (ΔC) can be obtained by estimating historical SOC stocks under conditions of minimal anthropogenic intervention (SOC_0) and estimating current SOC stocks due to land management or change of land-use ($SOC_{0\text{-}T}$) for various reference conditions (H) (based on Equation 2.24 of IPCC, 2019).

$$\Delta C = \Sigma H_{A»F} \left[(SOC_0 (h_n) - SOC_{0\text{-}T} (h_1)) \right] / D$$

Where the SOC_0 value represents the organic carbon store in the last year of measurement, $SOC_{0\text{-}T}$ is the value of the SOC storage in the first year of measurement, from the introduction of non-conventional practices (h=1»n) (Ogle, Breidt and Paustian, 2005). D is the time dependence of mineral SOC stock change factors, which is the default period for a transition between equilibrium SOC values, commonly 20 years.

The H value represents the reference conditions: altitude, landform, aridity index, and soil type (Eve *et al.*, 2002), for which correction factors can be obtained for unconventional land management (Ogle, Breidt and Paustian, 2005). This value includes four aridity indices (H_A), ten altitude ranges (H_B), five temperature regions (H_C), and four class landforms (H_D) (Table 21). It also includes seven types of land cover (H_E) (Table 22) and twenty-one dominant soil groups (H_F). For this evaluation, all data sets in raster format were vectorized in their original spatial resolution (30 to 1000 meters). The tabular results were generated by simple intersections and by scaling "up" using extraction subsets (342k systematic points).

Large-scale evidence

The implementation of national and transnational megaprojects of ecological restoration (with an area greater than 1 Mha), accompanied by a series of management practices aimed at sequestering SOC, have improved ecosystem services such as soil erosion control, retention of water, drought mitigation, and biodiversity conservation (Wang *et al.*, 2016; Xu *et al.*, 2017).

One of the first successful experiences was the program of tree planting, grass-field crop rotations, and alkaline structureless soils rehabilitation (1949–1965) in the former Soviet Union. For that, about 4 million hectares of forests were planted in drought-affected regions (5,300 km of forest windbreaks), which helped to improve the climatic and soil-forming conditions for agricultural development on 120 million hectares (Kovda, 1952). To expert estimations, this program helped to store about 100 Gg C in woody biomass and soil. The effect of this program is still evident in southern regions of former Soviet republics (Ukraine, the Republic of Moldova, the Russian Federation, Kazakhstan, and Central Asia).

Currently, China is developing six mega-restoration projects in 16 percent of its territory, which induces 56 percent of the national C sink (132 TgC/yr) (Lu *et al.*, 2018). One of these projects is developed almost entirely in drylands: the North Shelter Forest (from 1978 to 2070), better known as the Green Wall of North China, which aims to contain the expansion of the Gobi desert and storms of sand gradually invade arable land, and the urban areas in this region (Gelken, 2009). Until 2010, 5.4 million hectares of native trees were planted, representing an increase of 19.3 tC/ha/yr in biomass and of 3.2 tC/ha/yr in the mineral soil during the last ten years (Lu *et al.*, 2018).

Another example is the Great Green Wall African initiative, which has restored 20 million hectares and generated 2 million green jobs, mainly in Ethiopia, Nigeria, Senegal, and Sudan. If the countries involved in this project maintain their current restoration rate, organic carbon sequestration could increase to 70 Gg C in woody biomass and up to 15 Gg C in soil by 2030 (UNCCD, 2020).

Large anthropogenic phenomena derived from economic or social conflicts in drylands can generate large volumes of SOC under suitable temperature and humidity conditions. The abandonment of 60 million hectares (±14.2 Mha) of sub-humid and semi-arid agricultural lands during the economic collapse of the Union of Soviet Socialist Republics (USSR), between 1993 and 2010 (Lesiv *et al.*, 2018) indirectly caused the restoration of more than 100 million tons SOC (2.9 ± 1.7 t/ha/yr), according to the Orchidee-Stics-C-N Cycle simulation model (Vuichard *et al.*, 2008).

Table 24. Area of degraded drylands by aridity index and degradation factor

Aridity	Area	Tree coverage 2000-2019			Fires 2018-2019	Water stress		Loss of Fertility	Erosion	Strong erosion[1]
	Mkm^2	Dossal (%)	Area (Mha)	Loss Forest (%)	Area (%)	Area (%)	Area (Mha)	Area (%)	t/ha/yr	Area (Mha)
Hyperarid	9.8	0.0	0.0	0.0	1.1	44.8	21.9	2.2	nd[2]	6.1
Arid	15.7	0.4	1.8	0.1	6.0	29.1	301.3	19.2	1.7	25.7
Semiarid	22.7	6.9	27.5	1.2	20.1	34.6	523.5	23.1	1.9	65.0
Dry Subhumid	13.1	25.8	43.7	3.3	32.9	23.0	240.2	18.4	2.6	66.1

Sources (data sets): Hansen et al. (2013), NASA (2020), Cherlet et al. (2018), Sorensen (2007), JRC (2019)

[1]Strong erosion represents an erosion rate greater than 10 t/ha/yr;

[2] Non-determined.

Specific evidence

The measurements currently available in 33 scientific publications are presented in Appendix 1, representing sufficiently long measurement or projection periods (34 ± 23 years) of the change in SOC due to the adoption-conversion of unconventional practices. Only the evidence on SOC changes from the first to the fourth quintile of information (sequestration rates less than 2.0 tC/ha/yr) was considered.

In order to optimize the calculation processes, the data on 54 unconventional practices were coded, resulting from 10 categories of agricultural and livestock management: cropping system (h_1), fallow (h_2), tillage (h_3), organic fertilization (h_4), inorganic fertilization (h_5), crop residues (h_6), irrigation management (h_7), risks control (h_8), grazing management (h_9) and the rehabilitation of arboreal and shrub cover (h_{10}).

Analysis

There are two ways to compare changes in the SOC content of stores. The first way is by comparing the exchange rates (expressed in tC/ha/yr), which allows quantifying the total potential for SOC sequestration (ΔC) and allows identifying the most profitable environmental practices and conditions for the SOC sequestration (Figure 28). The second way is comparing the annual SOC increase factors (dimensionless values), which allows the comparison of unconventional practices' effectiveness, regardless of the size of the initial stores (Figure 29).

Effect of environmental practices and conditions on SOC stocks

According to the available data and considering the vast diversity of environmental conditions and forms of agricultural management, the potential for SOC sequestration of "unconventional" practices ranges between 0.04 and 0.59 tC/ha/yr. The use of inorganic fertilizers constitutes the most variable management strategy for SOC (0.21 ± 0.33 tC/ha/yr).

According to the data obtained for this analysis, the most promising SOC sequestration practices in agriculture are organic manures, plant residues management, and fallow. In livestock, the most efficient practices are grazing management and improving pastures (Gallardo, 2016). Enhancing biodiversity through the use of mixed species is an example of pasture improvement (Sebastia *et al.*, 2018). Under some circumstances, the accumulation of SOC in grasslands and croplands can exceed initial levels.

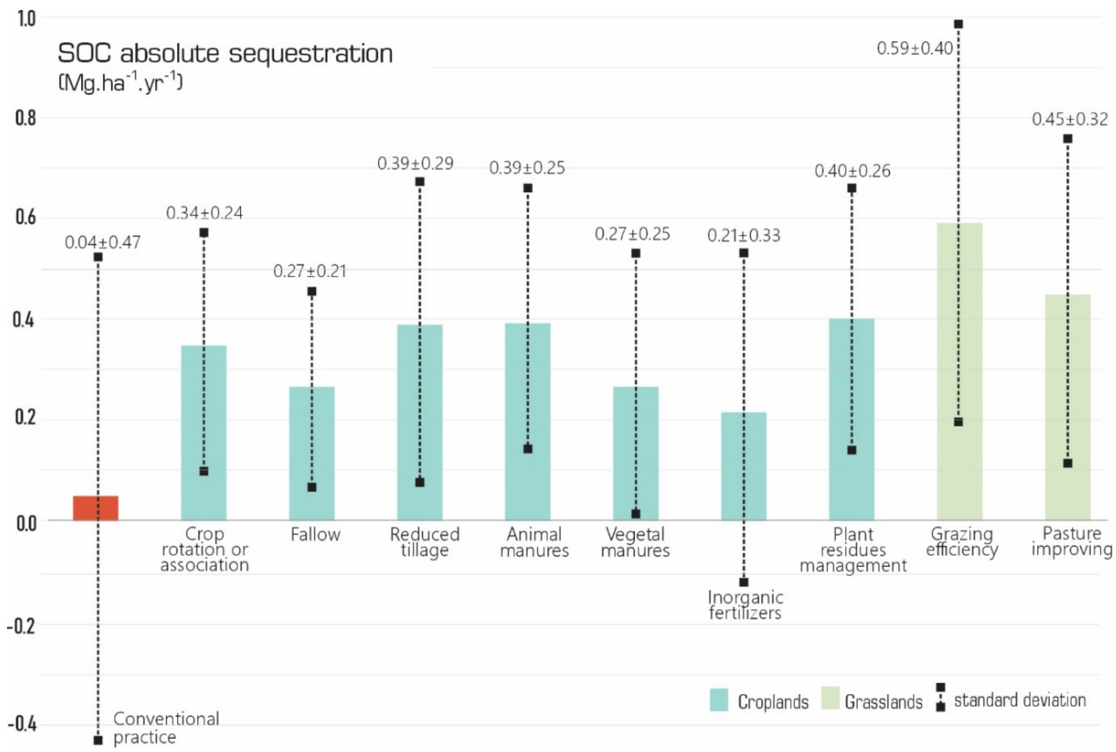


Figure 28. SOC absolute increase (t/ha) in soils with non-conventional treatments and reference soils with conventional treatment over 20 years, based on the data in Appendix 1

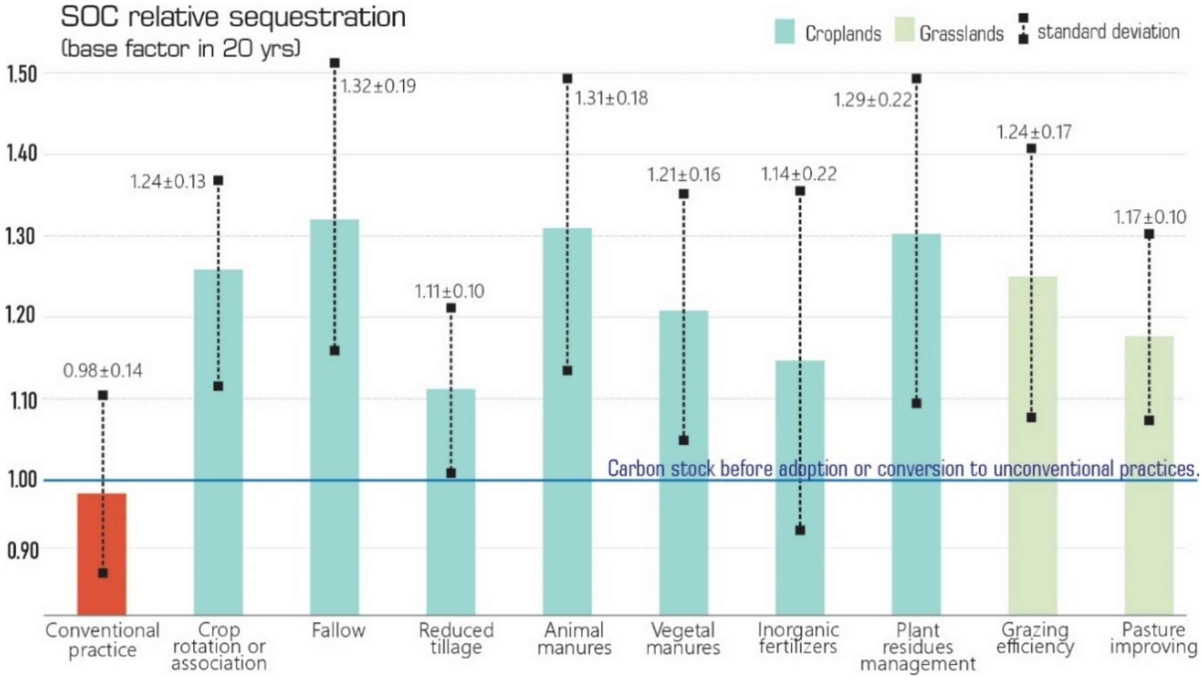


Figure 29. SOC relative increase (Base Factor) in soils with non-conventional treatments and reference soils with conventional treatment over 20 years, based on the data in Appendix 1

The Base factor is the relative soil carbon storage compared to the native or traditional system (Ogle, Breidt and Paustian, 2005) and is used to compute the relative change in storage from the reference native condition following long-term cultivation (i.e. 20 years)

Global potential of Drylands

Soil organic carbon recovery is typically a slow process, lasting decades to centuries, depending on the system's carbon balance (Bai *et al.*, 2008). Considering that a fifth of the drylands have maintained, adopted, or improved their soil management practices during the last 20 years (IPCC, 2000 in FAO, 2002), the global sequestration may reach 5.2 Gt C (2020–2040, in the first 30 cm of depth). This amount means an average potential of 0.26 tC/ha/yr for global drylands, if we consider available data sets and the average of all reports with rates lower than 2 tC/ha/yr (Table 25, Table 26 and Figure 30).

Table 25. Global Potential Restoration SOC (30 cm of depth) under unconventional agricultural and livestock systems, annually and after 20 years, by Aridity Index

Land-use	Aridity Index	Area	SOC average	Factor change	Obs	Potential Sequestration/1	Adoption-Conversion/2	Global Potential Sequestration	
	PT/EPT	Mha	t/ha		n	t/ha/yr	%Estimated Area	Mha	Pg 20 yr
Croplands	Less 0.05	10.9	19.7	1.001	0	0.02	20	2.2	0.0
(C,CG)	0.05–0.2	104.1	26.8	1.008	8	0.21	20	20.8	0.1
	0.2–0.5	666.1	44.5	1.013	34	0.58	20	133.2	1.5
	0.5–0.65	493.3	58.1	1.016	1	0.93	20	98.7	1.8
	Subtotal	1274.5						254.9	3.4
Grasslands	Less 0.05	7.4	35.2	1.001	0	0.04	10	0.7	0.0
(G,CG,GC)	0.05–0.2	228.0	25.2	1.003	1	0.08	10	22.8	0.1
	0.2–0.5	401.5	40.9	1.012	18	0.49	10	40.2	0.4
	0.5–0.65	95.0	54.2	1.010	10	0.54	10	9.5	0.1
	Subtotal	732.0						73.2	0.6
Total		2006.5						328.1	4.0

/1 Factors from sources in appendix 1.

/2 Estimators proposed by IPCC, 2000 in FAO, 2002.

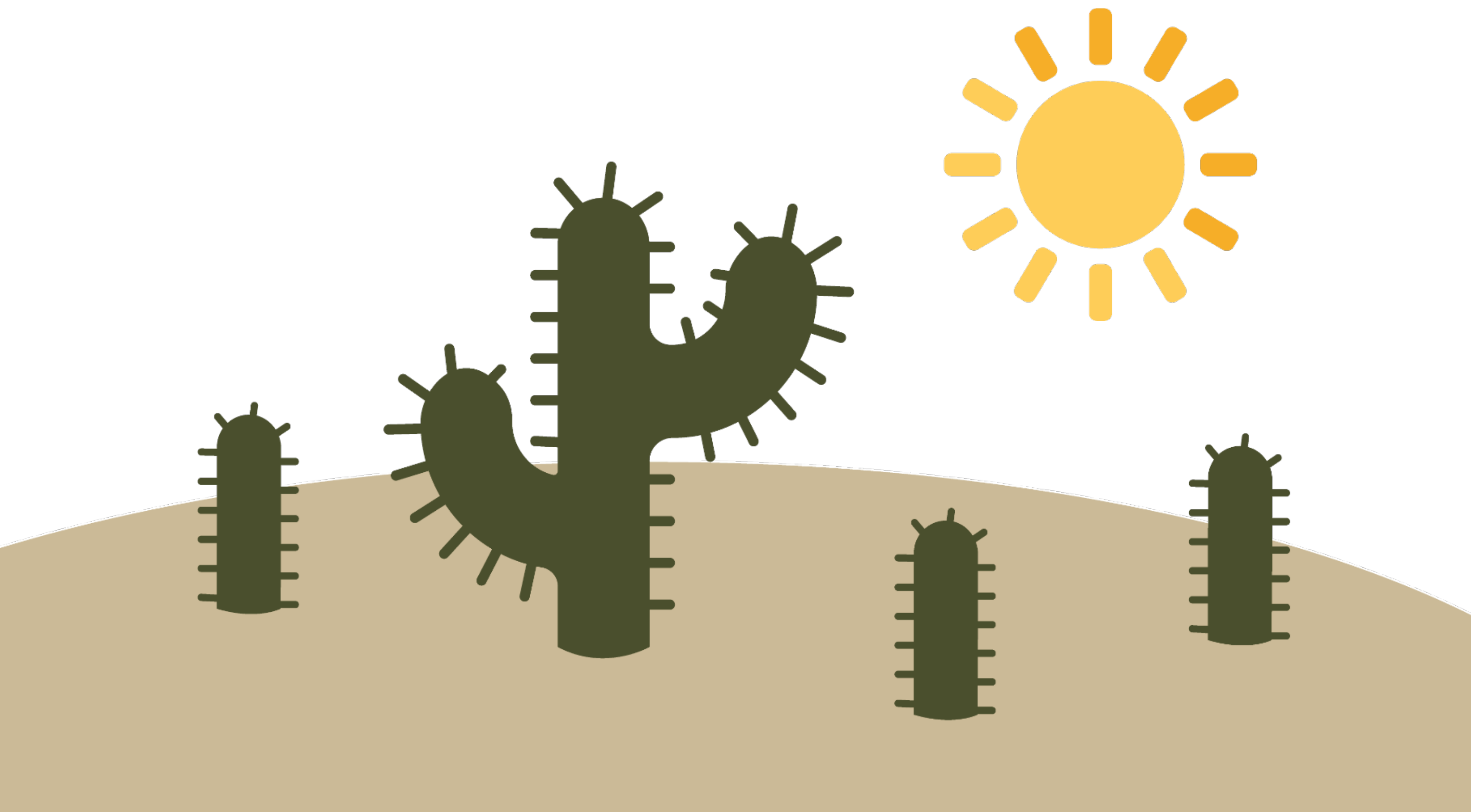


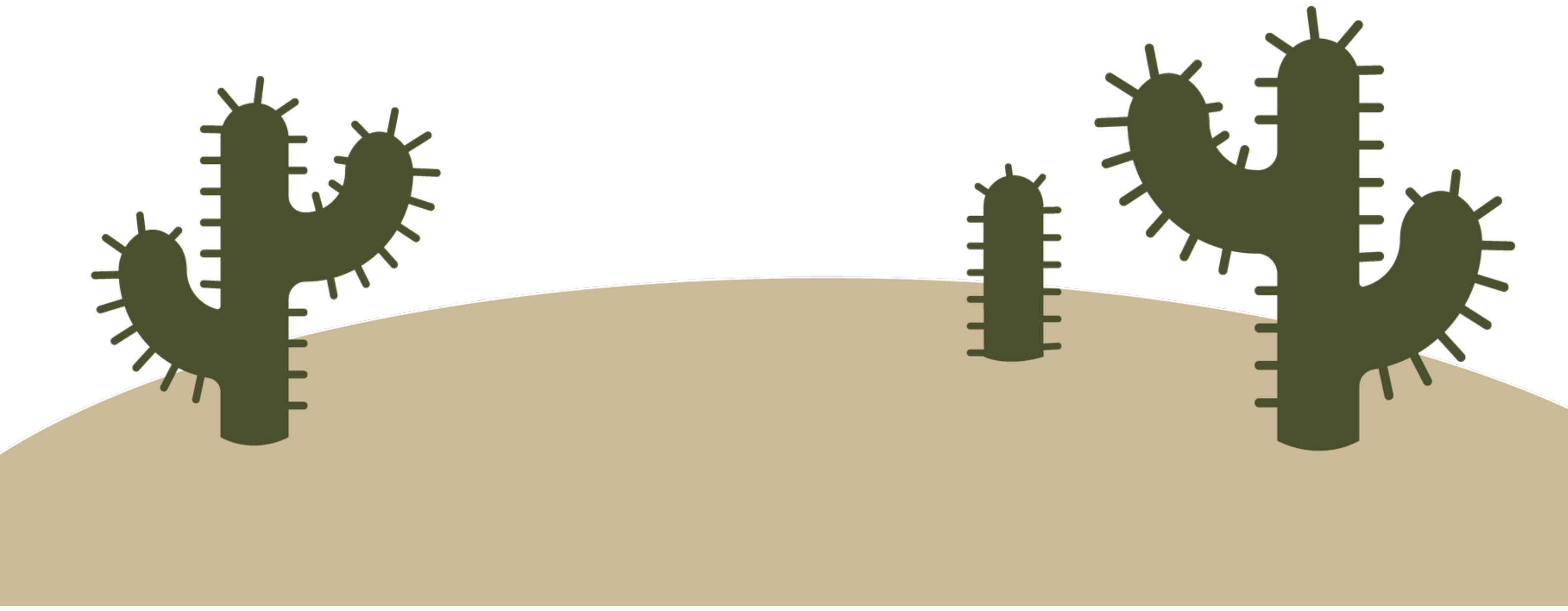
Table 26. Global Potential Restoration SOC (30 cm of depth) in drylands, annually and after 20 years

Land-use	Area	SOC average	Factor change	Potential Sequestration	Adoption-Conversion[/2]	Global Potential Sequestration	
	Mha	t/ha		t/ha/yr	%Area	Mha	Pg C/yr
Croplands	1274.5	38.9	1.014	0.68[/1]	20	254.9	0.17
Grasslands	732.0	38.8	1.009	0.36[/1]	10	73.2	0.03
Forest	828.4	32.8	1.010	0.25[/2]	10	82.8	0.02
Shrublands	902.5	29.9	1.010	0.25[/2]	5	45.1	0.01
Sparse/non-...vegetated	2358.1	29.3	1.010	0.25[/2]	5	117.9	0.03
Snow and Ice	2.7	91.8	1.000	0.00[/2]	0	0.0	0.00
Settlements	23.9	35.5	1.010	0.30[/2]	2	0.5	0.00
Total	6122.0					574.4	0.26

Sources: Sayre et al. (2020), FAO and ITPS (2020), Cherlet et al. (2018), Trabucco and Zomer (2009), and studies listed in Appendix 3

[/1] Factors of Croplands and Grasslands from sources in Appendix 1.

[/2] Factors and Estimators proposed by IPCC, 2000 in FAO, 2002.



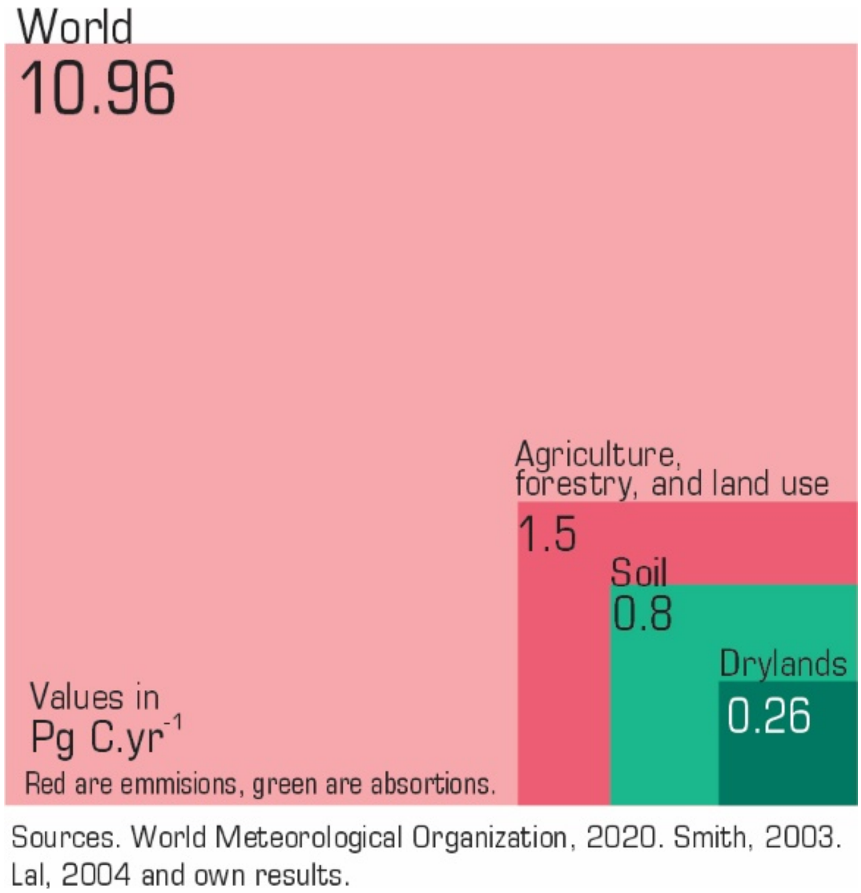


Figure 30. SOC sequestration scenarios from drylands within the context of the global emissions

Considerations in the estimation

Sequestration estimates should include emissions incorporated into manufacturing processes (fertilizers, herbicides, pumping water) or transportation (fodder, cattle, inputs) (Schuma, Janzen and Herrick, 2002; IPCC, 2006). The management of nutrients through manures and fertilizers improves the SOC storage; however, a study showed that the SOC sequestration rate was reduced from 0.16 to 0.06 tC/ha/yr after considering the emission of 1.4 kgC for each kg of nitrogen manufactured, in addition to the emissions from fertilization and other associated practices (Lee and Dotson, 1996).

The estimation of SOC's absorption potential (ΔC) in ecosystems, especially in drylands, has high uncertainty (Schrumpf *et al.*, 2011). It happens because organic matter (SOM) has a complex composition, association, and distribution, and therefore is very sensitive to natural or anthropogenic changes. Therefore, there are multiple interactions between the processes of photosynthesis, decomposition, and soil respiration, which can determine the potential of each store (Hungate *et al.*, 1997). In the end, the rapid population growth and the increasing demand for food and energy will generate intense pressure on land use in the coming decades and with it a loss of stored SOC that is difficult to quantify (Cruz *et al.*, 2017).

This study shows that drylands have a high potential to sequester SOC as long as the most appropriate management practices are carried out and precise and exhaustive monitoring and evaluation to achieve this objective.

4. Importance of conservation of drylands for the provision of specific ecosystem services

4.1. Minimization of threats to soil functions

Erosion

The baseline of the water erosion model (RUSLE) shows an increase in the average erosion (2.5 percent in global erosion) between 2001 and 2012, as a consequence of more significant anthropogenic pressure on the soil and a historic irregularity in the intensity and duration of rainfall (Borrelli *et al.*, 2017; Panagos *et al.*, 2017).

According to the Global Soil Erosion Modelling platform (GloSEM), a total of 165 Mha of drylands have a substantial rate of erosion, with losses between 24 and 31 t soil/ha/yr (Joint Research Centre, 2019). Practices to maintain or protect soils for SOC sequestration, as implementing a mulch cover, help reduce or reverse the impact of erosion, especially in heavily eroded agricultural areas (Faroda, 1998; Li, Yu and Geng, 2011). The use of crop residue mulch or cover crops is essential to reduce soil erosion and improve soil moisture and temperature regimes (Carvalho and Lourenço, 2014).

Increasing fertility

Typically, SOC sequestration means improving soil fertility (Gallardo, 2016). The area with declining fertility is 14.8 percent for drylands in general, and 19.2 percent for arid, 23.1 percent for semi-arid and 18.4 percent for sub-humid lands in particular (Cherlet *et al.*, 2018). Afforestation and reforestation in arid, shallow, and low fertility soils, with better-adapted species such as *Prosopis juliflora, Salvadora persica, Acacia tortilis*, or *Albizzia amara* can be a viable strategy in the medium term to improve fertility (Saxena, Sharma and Sharma, 1997).

A possibly faster measure is inorganic fertilization, provided a comprehensive management plan that accompanies it. The application of nitrogen fertilizers (90 kg urea/ha/yr) can increase biomass production in tall grasses, which contributes to an increase in the rate of SOC content in the soil (up to 1.6 t/ha/yr) (Rice, 2000 in Schuma, Janzen and Herrick, 2002). Planting legume species in degraded pastures can improve production and underground nitrogen fixation, which improves soil fertility (Conant, Paustian and Elliott, 2001). Practices leading to decreased organic matter mineralization will contribute to the sequestration of organic carbon in the soil. Higher SOC in the soil improves soil structure and facilitates root growth. It will also allow better water infiltration and retention and increase cation-exchange capacity. All these aspects relate to increasing soil fertility. Some conservation practices, such as zero-tillage, reduce soil compaction and crop production costs, desirable for increased food production potential in drylands.

Correcting salinization

Rainfed agriculture and irrigation largely determine groundwater quality and quantity, the first case through infiltration and the second through the extraction (Mortimore *et al.*, 2009). Salinity affects SOC stocks and biodiversity (Lal, 2003). However, the irrigation can improve water and nitrogen balances in soil, which leads to a correction in plant productivity and carbon inputs to the soil (Ghosh and Mahanta, 2014). When this practice is improper it can increase salinity levels in soils, especially in regions with higher evapotranspiration (generally lower aridity index). The introduction of species such as *Prosopis juliflora* in salinity affected soils in northeast India has increased SOC amount by 10 t/ha over five years (Garg, 1998).

4.2. Increases in production and food security

Food security in drylands has high risks because the frequent climatic variations affect expected production systems more strongly, especially those for self-consumption (Photo 16). Increasing the SOC pool of degraded agricultural land by one ton can increase crop yields from 20 to 40 kg/ha for wheat, 10 to 20 kg/ha for maize, and 0.5 to 1 kg/ha for cowpea (Lal, 2004).

4.3. Mitigation of and adaptation to climate change

Scientists and politicians have warned that environmental degradation and climate change cause massive displacements of people every day (Cruz *et al.*, 2017). These phenomena affect the soil's ability to produce healthy and nutritious food for humans and other organisms (Parr *et al.*, 1992). SOC sequestration could lessen the impact of climate change. The Brazilian semi-arid region (Caatinga) projects long-term losses (80 years) of 650 Tg C in a surface of 50 Mha due to global warming (Althoff *et al.*, 2015). This damage can be reduced by increasing the soil rest period from 20 to 50 years and extending the Caatinga cutting intervals from 10 to 20 years. With this, both firewood and carbon's sustainable production can achieve in this region (Choné *et al.*, 1991; Araújo-Filho *et al.*, 2013).

5. General challenges and trends

None of the environment's problems can be rationally addressed until its dimensions in space and time are known. The economic costs of knowing the profit and loss history of the soils and their producers will be significant, true; however, they would reflect a more transparent and consistent strategy to achieve the conservation of the soil and other natural resources that we can still use (Trimble and Crosson, 2000).

Photo 15. Poor water management leads to desiccation or salinization of the soil. Lucerne irrigation in Cuatro Ciénegas, Mexico

Photo 16. Use of native grasslands for feeding sheep and camelids in Central Andean Dry Puna. Oruro, the Plurinational State of Bolivia

Table 27. Related cases studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Short-time effects of no-tillage in olive orchards in Lebanon	NENA	5	4	1
Irrigation and SOC sequestration in the region of Navarre in Spain	Europe	6 to 20	4	19
Application of mulching in subtropical orchards in Granada, Spain	Europe	5	4	20
Reduced tillage frequency and no-till to allow ground covers and seeding cover crops in rainfed almond fields, Spain	Europe	10	4	21
Biochar and compost application in an olive orchard, Spain	Europe	4	4	22
Syntropic Agriculture in a Mediterranean Context	Europe	2	4	23
Pickle Melon (Cucumis melo) production in Karapınar, Central Turkey	Eurasia	60	4	24
Irrigated Wheat-Maize-Cotton in the Harran Plain, Southeast Turkey	Eurasia	30	4	25
Management of ornamental lawns and athletic fields in California, United States	North America	2, 10, 20 and 33	6	29

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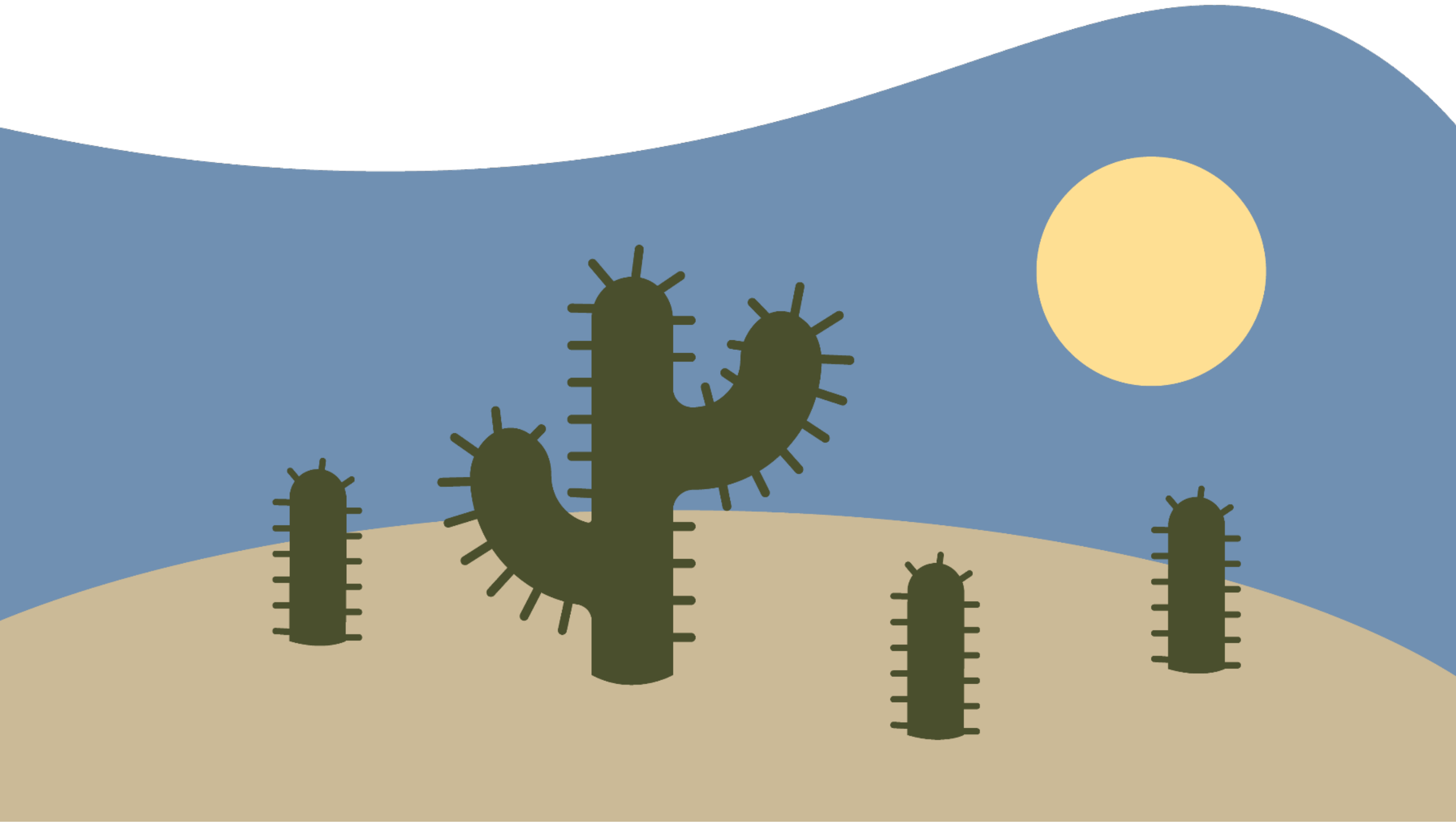

Technosols and urban soils

11. Technosols and urban soils

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1. Definition and description

Urban areas constitute less than 2 percent of world's land area (Yang and Zhang, 2015) but have been found to contain, on average, 1.5 to 3 times more carbon than natural soils. Urban soils function as "hotspots," or a concentrated medium, of carbon storage (Edmondson *et al.,* 2012). Urban soils are commonly human-made and include Technosols constructed with various organic and inorganic materials. Urban areas are hotspots of carbon stocks on a global scale because they may be subject to rapid gain or loss of carbon. Constructed soils commonly contain high amounts of organic materials because they are well managed and consequently, they contain higher amounts of soil organic carbon (SOC) than their rural counterparts. Soil organic (SOC) and soil inorganic carbon (SIC) vary greatly within a city. Their content is mostly controlled by historical and current land use. Buried fill materials (e.g. fly ash) or a cultural layer (e.g. carbonate materials) can contain extremely high levels of black carbon (BC) or inorganic carbon with minimal loss over long periods. Rapid weathering of concrete materials (Ca-silicate) also contributes to the rapid increase of SIC content in the soil. High SOC content is mostly the result of active management, which introduces large amounts of carbon and nitrogen into the soil and allows the buildup of biomass over time. This process is particularly dominant in arid climates that have minimal biological productivity. SOC content in buried soils is largely preserved due to low respiration activities and mineralization rates.

2. Global distribution of hotspot

The image below shows the global distribution of urban lights from space and therefore potential locations for hotspots of soil carbon in urban areas. As world population grows and urban areas expand, the significance of soil carbon stored in urban areas increases at the global scale (Figure 31).

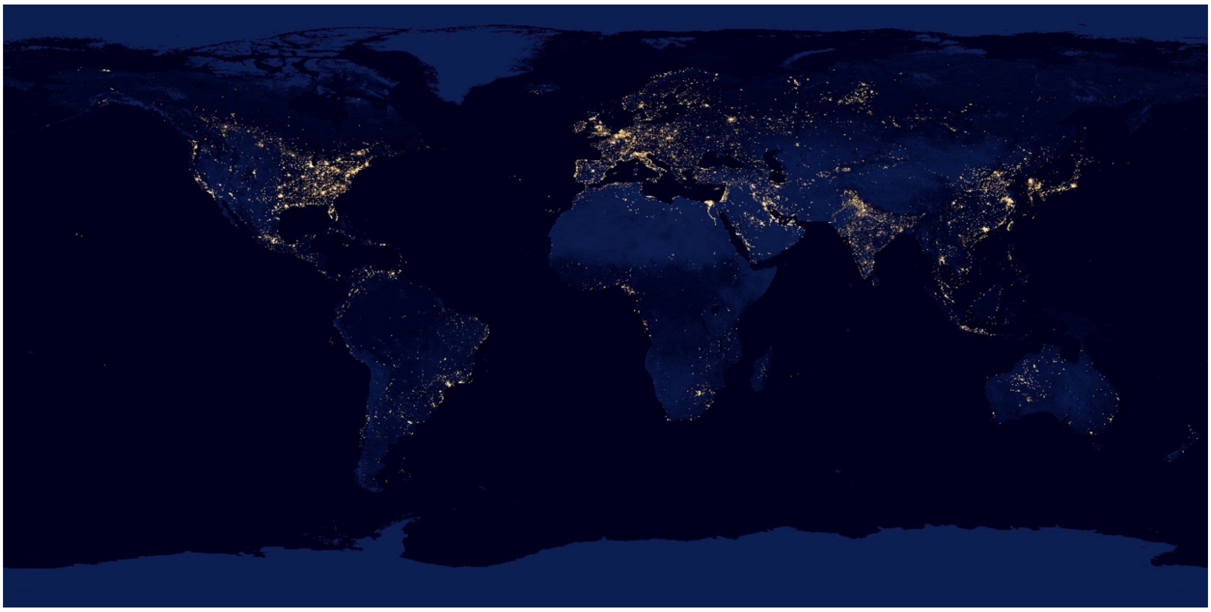

Figure 31. Worldwide image of Earth at night showing the extent of city lights from space, NASA Earth Observatory (NASA, 2012)

3. Urban carbon-stock hotspots (example of New York City)

3.1. Combustic and Artifactic soils

Soil types that formed in human-transported materials were differentiated and mapped by the USDA-NRCS, Soil Survey of New York City. Among these types were soils that formed in coal combustion and fly ash (Combustic material class) and in construction debris (Artifactic material class). These soils contain a considerable amount of human artifacts (greater than 10 percent by volume) and are enriched in black carbon (BC), which is a byproduct of the incomplete combustion of plant material and fossil fuels. Black carbon in these soils includes soot, charcoal, various types of ash, and asphalt. The carbon figures in the table below represent the fine-earth fraction only (Table 28). Artifacts larger than 2 mm were sieved out before analyses but can contain BC.

Table 28. Soil organic carbon stocks in the fine-earth fraction of Combustic and Artifactic soils sampled in New York City, United States of America at 0–100 cm depth

Location	Soil Type (U.S. Soil Taxonomy)	Cseq (tC/ha)	Material class	Pedon ID
Staten Island	Combustic, mixed, mesic Typic Udipsamments	3781.46	Coal combustion bottom ash	S1995NY085007
		2860.56		S2009085002
Camden		1180.60		S2010NJ007001
Bronx	Sandy, combustic, mixed, mesic Anthroportic Udorthents	508.93[9]		S2011NY005001
Queens	Loamy-skeletal, artifactic, mixed, superactive, nonacid, mesic Anthropic Udorthents	612.22[10]	Urban construction debris	S2011NY081001
Bronx		364.04		S2009NY005001
		292.22		S2009NY005002
Queens		286.44		S2011NY081002

Source: National Cooperative Soil Survey Characterization Database[11]

The Combustic soils cover only about 0.1 percent of the mapped land area of New York City. In these soils, stocks of soil organic carbon (SOC) to a depth of 100 cm average 2 207.36 tC/ha. The Artifactic soils comprise almost 6 percent of the city's mapped land area. In these soils, stocks of SOC to a depth of 100 cm average 306.22 t/ha. Given that SOC stocks for native soils in areas of woodland average 109.65 t/ha both types of high-artifact soil represent a significant increase in carbon sequestration. In the soil sealing or pavement process, the burial of large amounts of ash, charcoal, asphalt fragments, biochar, and similar materials beneath the impervious surface would provide a good method to sequester carbon (Lorenz and Lal, 2015).

Black carbon (BC) is more resistant to degradation than thermally unaltered organic carbon (from plant and animal residues). BC stability in soils is promoted by: (1) biological or physical mixing that removes the BC from the soil surface and redistributes it to subsurface horizons; (2) the formation of organo-mineral associations

[9] Photo 17
[10] Photo 18
[11] http://ncsslabdatamart.sc.egov.usda.gov/

with iron and aluminum oxides and other clay minerals; and (3) high levels of calcium. Fire, ozone, and ultraviolet radiation can degrade BC (Czimczik and Masiello, 2007). Coal ash and construction debris can have elevated levels of trace metals and polycyclic aromatic hydrocarbons (PAHs). Black carbon has a high affinity for persistent organic pollutants, such as PAHs and polychlorinated biphenyls (PCBs). The strong sorption of these compounds by BC can lower the exposure potential and toxicological risks (Koelmans *et al.*, 2006).

3.2. Spolic soils

Spolic soils formed in human-transported materials with low amounts of human artifacts (less than 10 percent). In the soil survey of New York City, soils that were in the Spolic material class and were in well-established turfgrass cover were also found to have considerable amounts of SOC (Table 29).

Table 29. Soil organic carbon stocks in Spolic soils sampled in New York City, United States of America at 0-100 cm depth

Location	Soil Type (U.S. Soil Taxonomy)	Cseq (tC/ha)	Material type	Pedon ID
New York	Coarse-loamy, spolic, mixed, active, acid, mesic Anthropic Udorthents	364.91[12]	Human-transported material; <10 percent artifacts	S1998NY061010
Bronx		205.54		S2000NY005006
Queens	Coarse-loamy over sandy, spolic, mixed, superactive, nonacid, mesic Anthropic Udorthents	204.58		S1998NY081001
Brooklyn	Coarse-loamy, spolic, mixed, active, acid, mesic Anthropic Udorthents	191.69		S2011NY047001

Source: National Cooperative Soil Survey Characterization Database[13]

Spolic soils, formed in human-transported materials, make up about 16 percent of the mapped land area of New York City. In these soils, stocks of soil organic carbon (SOC) to a depth of 100 cm average 151.51 tC/ha. This figure is slightly higher than the average for native soils in areas of woodlands in the city (109.65 t/ha). Pouyat *et al.* (2009) proposed that turfgrass soils in Baltimore reached maximum SOC levels in 40 years and mentioned

[12] Photo 19

[13] http://ncsslabdatamart.sc.egov.usda.gov/

the maintenance budget, i.e., the carbon costs of mowing and applying lime and fertilizer, needs to be considered. In most of these areas of New York City, the grass clippings are returned to the soil surface and applications of lime and fertilizer are infrequent. At least one cemetery in New York City, Green-Wood in Brooklyn, is investigating alternative, lower maintenance ground covers and their associated soil ecosystem services, including carbon sequestration rates.

3.3. Inorganic Carbon

The Artifactic soils are also enriched in soil inorganic carbon (SIC). Fragments of concrete are common in these construction-debris soils. The calcium weathered from silicate and hydroxide minerals in concrete reprecipitate with atmospheric carbon dioxide as calcium carbonate (Table 30).

Table 30. Soil inorganic carbon stocks in Artifactic soils sampled in New York City, United States of America at 0-100 cm depth

Location	Soil Type (U.S. Soil Taxonomy)	Cseq (tC/ ha)	Material type	Pedon ID
Bronx	Loamy-skeletal, artifactic, mixed, superactive, nonacid, mesic Anthropic Udorthents	66.15	Urban construction debris	S2009NY005002
Queens		32.05[14]		S2011NY081001
New York		30.18		S2011NY061001
Bronx		29.43		S2009NY005001
Queens		11.29		S2011NY081002

Source: National Cooperative Soil Survey Characterization Database[15]

Like most soils in the humid temperate zone, native soils in New York City have little or no calcium carbonate in the soil profile, representing a novel method for these urban soils to sequester carbon. Because the manufacture of cement is currently responsible for about 5 percent of global carbon dioxide (CO_2) emissions, this form of carbon sequestration is not really a net gain. Still, it can be a quick and efficient way to remove some atmospheric CO_2.

[14] Photo 18
[15] http://ncsslabdatamart.sc.egov.usda.gov/

Washbourne *et al.* (2015) estimated the removal of 85 tons of CO_2 /ha per hectare by calcium carbonate precipitation in the top 100 mm of soil in just 18 months at a demolition site in Newcastle, England. At the site, fine materials from onsite production of secondary concrete aggregates were incorporated into the soil.

4. Importance of urban soil conservation for the provision of specific ecosystem services

4.1. Minimization of threats to soil functions

Table 31. Soil threats

Soil threats	
Soil erosion	Maintaining groundcover, establishing vegetation after completion of construction activities (Lorenz and Lal, 2015), land leveling, and terracing of slopes for residential or commercial development reduce soil erosion (Vasenev and Kuzyakov, 2018). SOC improves aggregation, structure stability, and infiltration.
Nutrient imbalance and cycles	Soil organic matter (SOM) improves nutrient cycling. Adding SOM, such as compost, is an effective and common practice used by urban gardeners (Brown *et al.* 2016). Benefits include increased organic matter, slow-release of nutrients, increased cation exchange capacity (CEC), increased water-holding capacity, and dilution of trace elements or other contaminants in the soil (NRCS Urban Technical Note 4).
Soil salinization and alkalinization	Urban soils have higher pH than soils in native areas (Craul, 1999) related to mixing of construction debris (Morel, Chenu and Lorenz, 2014) and weathering of artifacts. This higher pH affects the solubility of nutrients and trace metals in soil and can reduce the need for liming in some areas.
Soil contamination / pollution	Additions of SOC, such as compost, can dilute contaminant concentrations in soil. SOC can also immobilize and reduce the bioavailability of several contaminants (NRCS Urban Technical Note 4; Ge *et al.*, 2000), such as soil lead related to strong bonds at soil particle surface (exchange sites). It may, however, increase the bioavailability of arsenic (Fleming *et al.*, 2013).

Soil threats	
Soil biodiversity loss	Green spaces and infrastructure improve urban biodiversity and may provide critical habitat for native, rare, or protected species (Morel, Chenu and Lorenz, 2014). Healthy urban soils support these species on various levels by providing habitat for faunal (Joimel *et al.*, 2018), microbial, and vegetative communities and by providing various ecological services, including nutrient cycling, soil hydrologic functions, and immobilization of soil contaminants.
Soil sealing	Encapsulation promotes retention of SOC by limiting mineralization (Wei *et al.*, 2014) and associated microbial respiration; however, Lu *et al.* (2020) described the removal and redistribution of SOC-rich topsoil during construction prior to sealing. Such removal may reduce the carbon stock at the time of sealing. The existing research indicates lower contents of SOC and nitrogen in sealed areas than in adjacent unsealed soils (Raciti *et al.*, 2012; Piotrowska-Dlugosz and Charzynski, 2015).
Soil compaction	A benefit of increasing SOC content in urban soils is lowering the bulk density. Although compaction is common in urban areas, Scharenbroch *et al.* (2005) observed a reduction in bulk density in vegetated urban areas over time following site construction. Some management practices may alleviate surface compaction over time. Examples include initial tillage to loosen compacted soils followed by planting of strong or deep-rooted vegetation and additions of organic residue (Lorenz and Lal, 2015).
Soil water management	Increasing the SOC content in urban soils greatly improves soil structure, thus increasing water infiltration (Beniston, Lal and Mercer, 2014) and thereby reducing runoff during rain events. Additions of organic matter also decrease the evaporation rate of Technosols (Robin *et al.*, 2018).
Soil temperature	Soil organic matter improves plant productivity on urban soils and thereby delivers ecological benefits from urban greening. Improvements to productivity result in improvements to the regulating services for local climate, carbon storage, and soil temperature (Morel, Chenu and Lorenz, 2014).

4.2. Increases in production and food security

Soil organic carbon supports food production in urban soils by enhancing soil functions, such as nutrient cycling, while improving soil properties, such as cation exchange capacity (CEC) and water holding capacity and lowering the bulk density. Urban soils that support vegetation can be used to produce food where conditions permit. Examples include in-ground planting, constructed planting, raised beds, and rooftop gardens (Grard *et al.,* 2017, 2020). Food produced in urban soils and constructed soils can help mitigate food security issues in urban communities where accessibility to fresh produce is otherwise limited.

4.3. Improvement of human well-being

Healthy urban soils that are rich in organic carbon provide a variety of social, environmental, and economic benefits for urban residents. Benefits are related to green spaces and other pervious areas. Zhou and Parves Rana (2012) describe social benefits related to urban green spaces, including recreational opportunities, aesthetic enjoyment, physical health, psychological well-being, enhanced social ties, and educational opportunities (Sanyé-Mengual *et al.,* 2018).

Other ecosystem services provided by urban and constructed soils include food production, stormwater management, groundwater recharge, and mitigation of urban heat island effects. Vegetation is also a significant regulating factor of climate and air quality (Morel, Chenu and Lorenz, 2014). Economic benefits may result from improved well-being of residents and revenue from community supported agriculture (CSA), urban farming, gardening, and horticultural plants. Food production is also supported in urban gardens, urban farms, and green roofs (Morel, Chenu and Lorenz, 2014). Urban soils also provide a medium to create pollinator habitats that increase butterfly and bee populations. These species are essential to food production, and it's important to create favorable habitats in which they can flourish.

4.4. Mitigation of and adaptation to climate change

Soil carbon stocks and sequestration in urban soils have the potential to mitigate climate change (Morel, Chenu and Lorenz, 2014). Ecological services are provided by soils, including those that are significantly impacted by urban activity. Examples include constructed Technosols and other soils formed in human-transported materials. Soils offer the potential to serve as a carbon sink that may support future efforts to off-set atmospheric carbon additions. Urban soil carbon stocks accumulate over time following site establishment. For example, over a 10-year period in France, carbon stocks in constructed Technosols have been measured with 50 percent more total carbon in the upper 30 cm and up to 5 times more in the upper 100 cm (Rees *et al.*, 2019) than reference sites. Soil carbon storage in urban soils also includes soil inorganic carbon (SIC), such as carbonate and black carbon (including biochar). Calcium-carbonate minerals may also be retained from the source of transported materials (Lorenz and Lal, 2015) or added through anthropogenic processes. Other sources include xenobiotic carbon, such as artificial polymers, polycyclic aromatic hydrocarbons (PAH), and other organic pollutants that sequester carbon in urban soils where they are concentrated (Vasenev and Kuzyakov, 2018). In addition to their inherent properties, urban soils have ability to capture and store atmospheric carbon through mineral weathering of silicate minerals. Such minerals originate from crushed concrete and result in secondary precipitation of carbonates, a SIC source in urban soils (Washbourne *et al.*, 2015).

SOC accumulation in urban soils also increases their adaption to climate change, in particular such extreme events as drought and thunderstorms with high precipitation (Robin *et al.*, 2018).

5. General challenges and trends

Creation of constructed Technosols (human-transported materials)

The process of soil construction creates greater carbon storage within subsoil layers (Rees *et al.,* 2019). Soil carbon content may occur in higher concentrations in subsurface horizons due to material source and development practices. Some human-transported and human-altered materials can have elevated contaminant levels.

Possible greenhouse gas (GHG) emissions

On-going maintenance of turfgrass and green areas requires the use of equipment that contributes to carbon dioxide emissions (Selhorst and Lal, 2013).

Soil sealing

Sealing promotes the retention of soil carbon stocks at the time of encapsulation; however, soils encapsulated by impervious surfaces potentially have lower soil carbon stocks due to the removal of topsoil that has high SOC content. Topsoil can be removed during construction activities, and the soil below the sealed surface has only a limited ability to accumulate more soil carbon (Lu *et al.,* 2020).

Wetlands

Due to construction activities, a loss of wetlands caused by draining or filling an area reduces the ability of the wetland to sequester SOC. The natural primary carbon collection environments are diminished, and existing carbon stocks in the wetland system are subject to volatilization.

Soil compaction

Compaction resulting from construction activities during site development can delay the successful establishment of plants, restrict root growth and depth, and increase soil erosion and runoff.

Green Infrastructure

Urban trees and lawns contribute to higher SOC stocks in urban soils (Nowak and Crane, 2002; Zirkle, Lal and Augustin, 2011). Organic soil amendments, such as compost and various mulches, also increase SOC stocks in urban green infrastructure. Additionally, street trees can provide SOC in dominantly impervious areas and other urban areas with high-density development; however, there are high installation and management costs (Kovacs *et al.*, 2013). Green roofs are constructed soils that have high SOC content. They have increasing significance and growing extent in urban areas (Oberndorfer *et al.,* 2007).

Photo 17. Soil profile, Combustic human-altered and human-transported (HAHT) material class. Mosholu soil series, Van Cortlandt Park, Bronx, NY, United States of America, containing 508.93 t SOC/ha

Photo 18. Soil profile, Artifactic HAHT Material Class. Laguardia soil series, Flushing Meadows, Queens, NY, United States of America, containing 612.2 t SOC/ha and 32.05 t SIC/ha

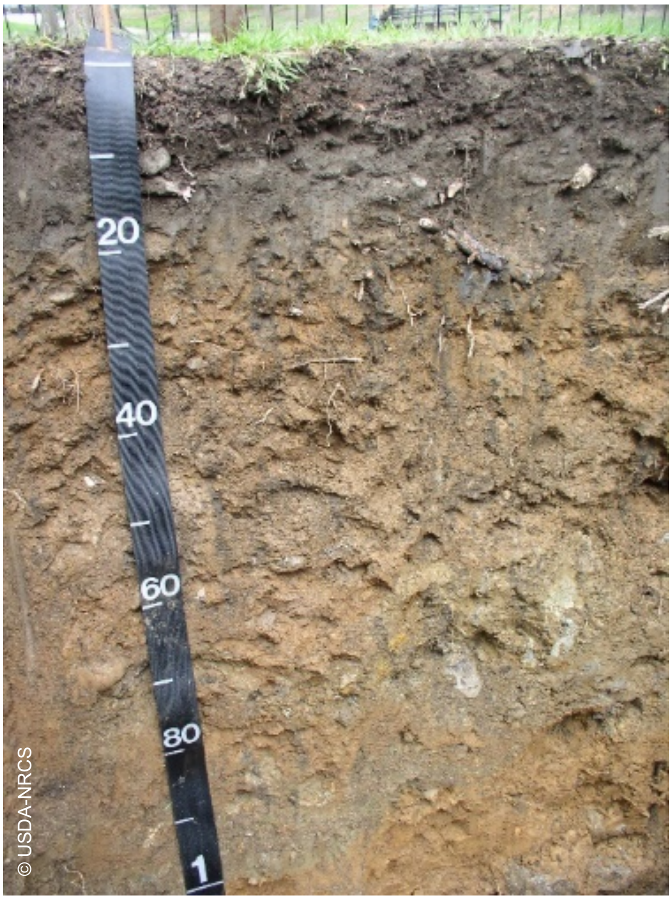

Photo 19. Soil profile, Spolic HAHT material class. North Meadow soil series, Central Park, New York, NY, United States of America, containing 364.91 t SOC/ha

Table 32. Related cases studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study nº
Carbon storage in soils built from waste for tree plantation in Angers, France	Europe	3	6	22
Urban agriculture on rooftops in Paris, France - the T4P research project (Pilot Project of Parisian Productive Rooftops)	Europe	5	6	23
Organic amendments for soils rehabilitation of open-pit mines in Spain	Europe	6, 10 and 18	6	24
Urban Forestry effects on soil carbon in Leicester, United Kingdom of Great Britain and Northern Ireland	Europe	20 to 100	6	25
Urban Agriculture in Tacoma, Washington, United States of America	North America	1	6	26
Soil Organic Carbon in forested and non-forested urban plots in the Chicagoland Region, United States of America	North America	Various	6	27
Compost application to restore post-disturbance soil health in Montgomery County, Virginia, United States	North America	4	6	28
Management of ornamental lawns and athletic fields in California, United States	North America	2, 10, 20 and 33	6	29
Water and residues management on a golf course, Nebraska, United States	North America	4	6	30
Maintenance of Marshlands in Urban Tidal Wetlands in New York City, United States	North America	100	6	31

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Annexes

Annex 1. Tropical Moist Forest – Supplementary documents

Annex 1A. Data on Soil Organic Carbon in moist tropical forests, including references

Site	Country	Continent	SOC (tC/ha)	Soil depth (cm)	Reference
Bimbia- Bonadikombo Community Forest	Cameroon	Africa	17.7	30	Longonje *et al.* (2018)
Doume Communal Forest	Cameroon	Africa	39.39	20	Zekeng *et al.* (2020)
Eastern Cameroon (old Forest)	Cameroon	Africa	106.4	100	Sugihara *et al.* (2019)
Mount Cameroon	Cameroon	Africa	65.4	30	Tegha *et al.* (2016)
Technical Operational Unit (Campo-Ma'an)	Cameroon	Africa	78.5	50	Lontsi *et al.* (2019)
Kisangani- Biosphere Reserve (YOKO)	Democratic Republic of the Congo	Africa	44.2	90	Doetterl *et al.* (2015)
Kisangani- Biosphere Reserve (Yangambi)	Democratic Republic of the Congo	Africa	109.5	90	Doetterl *et al.* (2015)
Belete Forest	Ethiopia	Africa	88.3	50	Lemma *et al.* (2006)
Gera	Ethiopia	Africa	47.88	30	Mohammed *et al.* (2006)
Shashamane Forest	Ethiopia	Africa	89.13	80	Lemenih *et al.* (2005)
Upper Gacheb Catchment	Ethiopia	Africa	183.5	80	Kasa *et al.* (2017)
Wushwush	Ethiopia	Africa	9.9	10	Solomon *et al.* (2002)
Mount. Birougou National Park	Gabon	Africa	93	100	Guatam *et al.* (2018)

Site	Country	Continent	SOC (tC/ha)	Soil depth (cm)	Reference
Ankasa Forest	Ghana	Africa	82.95	100	Chiti *et al.* (2010)
Jomoro District (Primary Forest)	Ghana	Africa	168.87	100	Chiti *et al.* (2013)
Jomoro District (Secondary Forest)	Ghana	Africa	137.2	100	Chiti *et al.* (2013)
Kakum National Park	Ghana	Africa	349.38	40	Adu-Bredu *et al.* (Unpublished data)*
Bobiri Forest Reserve (Old-growth forest)	Ghana	Africa	259	100	Addo-Danso (Unpublished data)*
Semi-deciduous Forest (Atwima Nwabiagya)	Ghana	Africa	45.6	20	Dawoe *et al.* (2014)
Banco plateau	Ivory Coast	Africa	107.5	50	Reversat *et al.* (1978)
Eastern Mau Forest Reserve	Kenya	Africa	110	30	Were *et al.* (2016)
Kakamega National Forest (Old trees)	Kenya	Africa	100	60	Glenday (2006)
Kakamega National Forest (Young trees)	Kenya	Africa	63	60	Glenday (2006)
Mau Forest Complex (Montane Forest)	Kenya	Africa	185	100	Chiti *et al.* (2017)
Remnant Indigenous forest, Taiti Hills	Kenya	Africa	305	50	Omoro *et al.* (2013)
Ankeniheny-Zahamena Corridor	Madagascar	Africa	136.2	100	Andriamananjara *et al.* (2016)
Tropical Humid Forest	Nigeria	Africa	117.6	100	Akpa *et al.* (2016)
Rukarara catchment	Rwanda	Africa	310	50	Wasige *et al.* (2014)
Eastern Usambra Mountain (Primary Forest)	United Republic of Tanzania	Africa	97.5	100	Kirsten *et al.* (2016)
Eastern Usambra Mountain (Secondary Forest)	United Republic of Tanzania	Africa	101.5	100	Kirsten *et al.* (2016)
Hanang Forest Reserve	United Republic of Tanzania	Africa	22.85	45	Swai *et al.* (2014)
Bwindi Impenetrable National Park	Uganda	Africa	34.2	30	Twongyirwe *et al.* (2013)
Burahya County	Uganda	Africa	20.36	15	Majaliwa *et al.* (2010)
Xishuangbanna, Yunnan Province	China	Asia	50	100	Tang *et al.* (2016)
Moist Deciduous Forests	India	Asia	85.52	100	Chhabra *et al.* (2003)

Site	Country	Continent	SOC (tC/ha)	Soil depth (cm)	Reference
Evergreen Forest, Western Ghats	India	Asia	75.1	30	Subashree *et al.* (2019)
Semi-evergreen Forest, Western Ghats	India	Asia	68.9	30	Subashree *et al.* (2019)
Lowland Tropical Rain Forest	Indonesia	Asia	27.5	100	Yonekura *et al.* (2010)
Pasoh	Malaysia	Asia	70	100	DeAngelis *et al.* (2003)
Sabah (Mount. Kinabalu)	Malaysia	Asia	54.83	100	Kitayama and Aiba (2002)
Berembun Forest Reserve (Unlogged Forest)	Malaysia	Asia	87.86	100	Abdullahi *et al.* (2018)
Berembun Forest Reserve (Logged Forest)	Malaysia	Asia	65.66	100	Abdullahi *et al.* (2018)
Borneo Island	Malaysia	Asia	39.6	100	Saner *et al.* (2012)
Sarawaka	Malaysia	Asia	39.5	70	Rahman *et al.* (2018)
Bukit Timah Nature Reserve (Primary Forest)	Singapore	Asia	110.8	300	Ngo *et al.* (2013)
Khao Chong	Thailand	Asia	37.5	100	Yoda and Kira (1969)
Mor Ridge	Jamaica	Caribbean	125	45	Tanner (1985)
El Verde (Liquillo Experimental Forest)	Puerto Rico	Caribbean	57	100	Brown *et al.* (1983)
Colorado Forest	Puerto Rico	Caribbean	95	50	Weaver and Murphy (1990)
Barro Colorado Island	Panama	Central America	22.4	5	Shwendemann *et al.* (2007)
Lowland forest, Isthmus	Panama	Central America	133	100	Cusack *et al.* (2018)
Laupohoehoe Forest Reserve	Hawaii	North America	76.8	20	Townsend *et al.* (1995)
Saruwaged Mountain Range	Papua New Guinea	Pacific	194	100	Dieleman *et al.* (2013)
Montane Rain Forest	Papua New Guinea	Pacific	300	100	Edwards and Grubb (1977)
Igarape-Acu and Peixe-Boi	Brazil	South America	186	600	Sommer *et al.* (2000)
Terra Firme Forest, Manaus	Brazil	South America	20.16	30	Santos *et al.* (2016)

Site	Country	Continent	SOC (tC/ha)	Soil depth (cm)	Reference
Primary Forest, eastern Amazonia, Pará State	Brazil	South America	280	800	Trumbore *et al.* (1995)
Mato Grosso and Pará State	Brazil	South America	52.05	100	Strey *et al.* (2016)
Porce II	Colombia	South America	48.3	30	Sierra *et al.* (2007)
Primary forest, Central Cordillera	Colombia	South America	114	40	Moreno and Oberbauer (2008)
La Selva	Costa Rica	South America	330	300	VeldKamp *et al.* (2003)
Eastern Andes	Ecuador	South America	219.33	50	Leuschner *et al.* (2013)
Lower Montane Forest	Ecuador	South America	23.88	100	Rhoades *et al.* (2000)
Montane Forest, Zamora-chinchipe and Loja	Ecuador	South America	70.3	30	Moser *et al.* (2011)
Manu National Park	Peru	South America	118	90	Zimmermann *et al.* (2010)
Manu National Park- Montane Forest (Primary forest)	Peru	South America	158.65	90	Oliveras *et al.* (2017)
Manu National Park- Montane Forest (Burned forest)	Peru	South America	99.1	90	Oliveras *et al.* (2017)
Tropical Rainforest (Mid-Altitude)	Peru	South America	41.5	30	Segnini *et al.* (2011)
Montano Forest, Wayqecha Biological Station	Peru	South America	182.5	30	Segnini *et al.* (2011)
Lowland Moist Forest	Venezuela (Bolivarian Republic of)	South America	111.67	100	Delaney *et al.* (1997; 1998)

SOC = Soil Organic Carbon;

*Data are available when needed

Annex 1B: Data on Soil Organic Carbon changes due to conversion of tropical moist forests to other land uses, including references

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to plantation	Ghana	Africa	-27.8	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-20.7	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-9.2	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-61.0	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-57.0	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-56.1	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-35.8	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-34.5	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-32.6	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-55.8	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-60.0	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-59.3	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-23.5	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-28.6	Chiti *et al.* (2014)
Forest to plantation	Ghana	Africa	-30.0	Chiti *et al.* (2014)
Forest to plantation	Nigeria	Africa	-0.7	Aborisade and Aweto (1990)
Forest to plantation	Nigeria	Africa	19.4	Aborisade and Aweto (1990)
Forest to plantation	India	Asia	-45.1	Saha *et al.* (2010)
Forest to plantation	India	Asia	-37.9	Saha *et al.* (2010)
Forest to plantation	India	Asia	-62.5	Saha *et al.* (2010)
Forest to plantation	India	Asia	-54.5	Saha *et al.* (2010)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to plantation	India	Asia	-43.1	Saha *et al.* (2010)
Forest to plantation	India	Asia	-41.4	Saha *et al.* (2010)
Forest to plantation	India	Asia	-41.7	Saha *et al.* (2010)
Forest to plantation	India	Asia	-22.7	Saha *et al.* (2010)
Forest to plantation	India	Asia	-31.4	Saha *et al.* (2010)
Forest to plantation	India	Asia	-34.5	Saha *et al.* (2010)
Forest to plantation	India	Asia	-39.6	Saha *et al.* (2010)
Forest to plantation	India	Asia	-22.7	Saha *et al.* (2010)
Forest to plantation	India	Asia	-25.5	Saha *et al.* (2010)
Forest to plantation	India	Asia	-37.9	Saha *et al.* (2010)
Forest to plantation	India	Asia	-39.6	Saha *et al.* (2010)
Forest to plantation	India	Asia	-27.3	Saha *et al.* (2010)
Forest to plantation	Indonesia	Asia	-33.9	Hertl *et al.* (2009)
Forest to plantation	Indonesia	Asia	-14.0	Hertl *et al.* (2009)
Forest to plantation	Indonesia	Asia	-6.1	Ishizuka *et al.* (2005)
Forest to plantation	Indonesia	Asia	-10.9	Ishizuka *et al.* (2005)
Forest to plantation	Indonesia	Asia	-13.9	Ishizuka *et al.* (2005)
Forest to plantation	Indonesia	Asia	-7.6	Ishizuka *et al.* (2005)
Forest to plantation	Indonesia	Asia	28.3	Ishizuka *et al.* (2005)
Forest to plantation	Indonesia	Asia	-0.7	Ishizuka *et al.* (2005)
Forest to plantation	Indonesia	Asia	-49.4	Ishizuka *et al.* (2005)
Forest to plantation	Indonesia	Asia	-60.7	Ishizuka *et al.* (2005)
Forest to plantation	Indonesia	Asia	-22.3	Ishizuka *et al.* (2005)
Forest to plantation	Indonesia	Asia	17.6	Ishizuka *et al.* (2005)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to plantation	Indonesia	Asia	2.0	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	68.4	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	-35.2	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	47.0	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	23.3	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	10.9	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	158.5	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	208.0	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	14.2	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	6.9	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	155.7	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	-53.6	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	-26.4	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	6.7	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	20.7	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	1.2	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	-34.0	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	-2.3	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	-6.3	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	-3.9	Smiley and Kroschel (2008)
Forest to plantation	Indonesia	Asia	-24.3	Guillaume *et al.* (2015)
Forest to plantation	Indonesia	Asia	-13.4	Guillaume *et al.* (2015)
Forest to plantation	Indonesia	Asia	-39.7	Guillaume *et al.* (2015)
Forest to plantation	Indonesia	Asia	-18.2	Guillaume *et al.* (2015)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to plantation	Indonesia	Asia	-42.2	Guillaume *et al.* (2015)
Forest to plantation	Indonesia	Asia	-23.9	Guillaume *et al.* (2015)
Forest to plantation	Indonesia	Asia	18.4	Frazao *et al.* (2013)
Forest to plantation	Indonesia	Asia	-13.8	Guillaume *et al.* (2018)
Forest to plantation	Indonesia	Asia	7.3	Guillaume *et al.* (2018)
Forest to plantation	Indonesia	Asia	-10.0	Guillaume *et al.* (2018)
Forest to plantation	Indonesia	Asia	8.3	Guillaume *et al.* (2018)
Forest to plantation	Indonesia	Asia	-31.2	Guillaume *et al.* (2018)
Forest to plantation	Indonesia	Asia	-14.2	Guillaume *et al.* (2018)
Forest to plantation	Argentina	South America	-42.3	Piccolo *et al.* (2008)
Forest to plantation	Brazil	South America	-26.0	Smith *et al.* (2002)
Forest to plantation	Brazil	South America	-18.8	Smith *et al.* (2002)
Forest to plantation	Brazil	South America	-12.5	Smith *et al.* (2002)
Forest to plantation	Brazil	South America	18.8	Smith *et al.* (2002)
Forest to pasture	Bangladesh	Asia	52.3	Islam *et al.* (2000)
Forest to pasture	Indonesia	Asia	-33.8	Ishizuka *et al.* (2005)
Forest to pasture	Indonesia	Asia	-18.1	Ishizuka *et al.* (2005)
Forest to pasture	Indonesia	Asia	-0.9	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	17.5	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	23.9	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	22.1	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	20.7	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	22.0	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	22.9	Yonekura *et al.* (2010)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Indonesia	Asia	-15.9	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	5.0	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	20.6	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	23.4	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	24.8	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	26.7	Yonekura *et al.* (2010)
Forest to pasture	Indonesia	Asia	27.5	Yonekura *et al.* (2010)
Forest to pasture	Dominican Republic	Caribbean	-36.4	Templer *et al.* (2005)
Forest to pasture	Puerto Rico	Caribbean	12.7	Brown and Lugo (1990)
Forest to pasture	Puerto Rico	Caribbean	-71.3	Brown and Lugo (1990)
Forest to pasture	Puerto Rico	Caribbean	56.8	Brown and Lugo (1990)
Forest to pasture	Puerto Rico	Caribbean	-53.4	Brown and Lugo (1990)
Forest to pasture	Puerto Rico	Caribbean	64.0	Brown and Lugo (1990)
Forest to pasture	Puerto Rico	Caribbean	2.9	Marin-Spiotta *et al.* (2009)
Forest to pasture	Puerto Rico	Caribbean	-6.1	Marin-Spiotta *et al.* (2009)
Forest to pasture	Puerto Rico	Caribbean	1.2	Marin-Spiotta *et al.* (2009)
Forest to pasture	Puerto Rico	Caribbean	-2.2	Marin-Spiotta *et al.* (2009)
Forest to pasture	Costa Rica	Central America	22.2	Cleveland *et al.* (2003)
Forest to pasture	Costa Rica	Central America	8.8	Cleveland *et al.* (2003)
Forest to pasture	Costa Rica	Central America	-19.4	Guggenberger and Zech (1999)
Forest to pasture	Costa Rica	Central America	-20.1	Krishnaswamy and Richter (2002)
Forest to pasture	Costa Rica	Central America	-4.8	Powers and Veldkamp (2005)
Forest to pasture	Costa Rica	Central America	-19.9	Reiners *et al.* (1994)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Costa Rica	Central America	-21.2	Reiners *et al.* (1994)
Forest to pasture	Costa Rica	Central America	-24.5	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-31.7	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-10.3	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	11.7	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-17.1	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	14.8	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	56.2	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	31.8	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-19.9	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	10.0	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	19.6	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	19.9	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	4.5	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	15.6	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	11.0	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	24.7	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	22.8	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	32.3	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-21.5	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-19.7	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-19.2	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-18.3	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-12.9	Van Dam *et al.* (1997)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Costa Rica	Central America	-1.1	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-4.9	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	7.2	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	13.6	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-10.5	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-17.2	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-31.0	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-24.8	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-28.1	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-13.7	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	10.1	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-15.6	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-28.3	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-36.4	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-28.9	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-28.4	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	-13.5	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	8.0	Van Dam *et al.* (1997)
Forest to pasture	Costa Rica	Central America	35.5	Veldkamp *et al.* (2003)
Forest to pasture	Costa Rica	Central America	-24.1	Veldkamp *et al.* (2003)
Forest to pasture	Costa Rica	Central America	-24.2	Veldkamp *et al.* (2003)
Forest to pasture	Costa Rica	Central America	56.6	Veldkamp *et al.* (2003)
Forest to pasture	Costa Rica	Central America	14.0	Veldkamp *et al.* (2003)
Forest to pasture	Costa Rica	Central America	-5.9	Veldkamp *et al.* (2003)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Costa Rica	Central America	-29.2	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-3.7	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-4.5	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-25.9	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-26.7	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-30.0	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	11.1	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-20.0	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-10.0	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	7.4	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-13.3	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-30.0	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-40.0	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-30.0	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	14.8	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	6.7	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	20.0	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	61.5	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	20.0	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-7.7	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	7.7	Veldkamp (1994)
Forest to pasture	Costa Rica	Central America	-7.7	Veldkamp (1994)
Forest to pasture	Mexico	Central America	55.2	Campos *et al.* (2007)
Forest to pasture	Mexico	Central America	12.7	Garcia-Oliva *et al.* (1994)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Mexico	Central America	31.6	Garcia-Oliva *et al.* (1994)
Forest to pasture	Mexico	Central America	-12.7	Garcia-Oliva *et al.* (1994)
Forest to pasture	Mexico	Central America	-6.6	Garcia-Oliva *et al.* (1994)
Forest to pasture	Mexico	Central America	-24.6	Hughes *et al.* (2000)
Forest to pasture	Mexico	Central America	23.1	Jaramillo *et al.* (2003)
Forest to pasture	Mexico	Central America	29.9	Jaramillo *et al.* (2003)
Forest to pasture	Panama	Central America	-31.7	Schwendenmann and Pendall (2006)
Forest to pasture	Panama	Central America	-14.4	Schwendenmann and Pendall (2006)
Forest to pasture	Panama	Central America	-25.2	Schwendenmann and Pendall (2006)
Forest to pasture	Panama	Central America	-27.9	Schwendenmann and Pendall (2006)
Forest to pasture	Panama	Central America	-18.8	Schwendenmann and Pendall (2006)
Forest to pasture	Panama	Central America	-25.3	Schwendenmann and Pendall (2006)
Forest to pasture	Panama	Central America	-16.7	Schwendenmann and Pendall (2006)
Forest to pasture	Hawaii	North America	-21.4	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	20.5	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	27.4	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	0.3	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	21.1	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	33.8	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	33.3	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	22.8	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	16.4	Osher *et al.* (2003)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Hawaii	North America	-5.2	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	-5.9	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	1.1	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	26.4	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	61.8	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	38.5	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	-4.5	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	6.1	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	-0.5	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	-0.5	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	-11.5	Osher *et al.* (2003)
Forest to pasture	Hawaii	North America	19.6	Townsend *et al.* (1995)
Forest to pasture	Hawaii	North America	-7.7	Townsend *et al.* (1995)
Forest to pasture	Australia	Oceania	27.0	Mendham *et al.* (2003)
Forest to pasture	Brazil	South America	26.0	da Silva *et al.* (2009)
Forest to pasture	Brazil	South America	50.0	da Silva *et al.* (2009)
Forest to pasture	Brazil	South America	60.0	da Silva *et al.* (2009)
Forest to pasture	Brazil	South America	10.3	da Silva *et al.* (2009)
Forest to pasture	Brazil	South America	32.0	da Silva *et al.* (2009)
Forest to pasture	Brazil	South America	33.0	da Silva *et al.* (2009)
Forest to pasture	Brazil	South America	-3.2	da Silva *et al.* (2009)
Forest to pasture	Brazil	South America	4.8	da Silva *et al.* (2009)
Forest to pasture	Brazil	South America	11.7	da Silva *et al.* (2009)
Forest to pasture	Brazil	South America	17.7	Desjardins *et al.* (1994)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Brazil	South America	-4.0	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	-10.1	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	38.8	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	-3.8	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	-8.8	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	45.2	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	1.7	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	-10.8	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	22.9	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	11.1	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	3.5	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	37.2	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	11.3	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	3.5	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	64.9	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	4.7	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	-6.3	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	-2.7	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	-12.5	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	-5.7	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	-4.7	Desjardins *et al.* (1994)
Forest to pasture	Brazil	South America	34.6	Fernandes *et al.* (2002)
Forest to pasture	Brazil	South America	40.0	Fernandes *et al.* (2002)
Forest to pasture	Brazil	South America	92.3	Fernandes *et al.* (2002)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Brazil	South America	30.0	Fernandes *et al.* (2002)
Forest to pasture	Brazil	South America	23.1	Fernandes *et al.* (2002)
Forest to pasture	Brazil	South America	56.3	Kainer *et al.* (1998)
Forest to pasture	Brazil	South America	5.5	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	-7.3	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	-20.2	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	31.0	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	22.6	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	9.8	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	9.9	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	-11.1	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	-20.0	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	6.9	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	5.9	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	8.1	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	30.2	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	-4.4	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	-13.2	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	10.4	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	20.7	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	23.7	Koutika *et al.* (1997)
Forest to pasture	Brazil	South America	-8.3	Luizao *et al.* (1992)
Forest to pasture	Brazil	South America	-10.8	Luizao *et al.* (1992)
Forest to pasture	Brazil	South America	-40.8	Macedo *et al.* (2008)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Brazil	South America	-33.9	Macedo *et al.* (2008)
Forest to pasture	Brazil	South America	-38.7	Macedo *et al.* (2008)
Forest to pasture	Brazil	South America	-34.5	Macedo *et al.* (2008)
Forest to pasture	Brazil	South America	-43.6	Macedo *et al.* (2008)
Forest to pasture	Brazil	South America	-41.9	Macedo *et al.* (2008)
Forest to pasture	Brazil	South America	7.7	Markewitz *et al.* (2004)
Forest to pasture	Brazil	South America	6.1	Markewitz *et al.* (2004)
Forest to pasture	Brazil	South America	-24.8	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	-5.2	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	2.0	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	6.2	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	22.7	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	-1.6	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	-1.8	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	-18.3	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	9.0	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	7.1	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	52.9	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	35.6	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	49.1	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	23.2	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	32.3	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	9.9	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	37.8	Neill *et al.* (1997)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Brazil	South America	20.1	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	59.2	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	45.2	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	79.1	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	54.8	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	37.6	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	44.9	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	60.6	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	33.2	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	64.6	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	43.1	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	64.1	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	29.6	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	54.4	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	50.2	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	24.2	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	-5.6	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	50.6	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	7.7	Neill *et al.* (1997)
Forest to pasture	Brazil	South America	70.8	Salimon *et al.* (2004)
Forest to pasture	Brazil	South America	28.4	Salimon *et al.* (2004)
Forest to pasture	Brazil	South America	27.1	Salimon *et al.* (2004)
Forest to pasture	Brazil	South America	23.8	Salimon *et al.* (2004)
Forest to pasture	Brazil	South America	44.4	Salimon *et al.* (2004)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Brazil	South America	59.7	Salimon *et al.* (2004)
Forest to pasture	Brazil	South America	47.2	Salimon *et al.* (2004)
Forest to pasture	Brazil	South America	48.1	Salimon *et al.* (2004)
Forest to pasture	Brazil	South America	11.9	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	25.0	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	62.9	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	11.4	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	13.9	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	38.1	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	11.6	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	-1.8	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	-6.8	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	13.6	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	-23.2	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	9.3	Wick *et al.* (2005)
Forest to pasture	Brazil	South America	3.4	Wick *et al.* (2005)
Forest to pasture	Ecuador	South America	38.0	Paul *et al.* (2008)
Forest to pasture	Ecuador	South America	0.7	Paul *et al.* (2008)
Forest to pasture	Ecuador	South America	-14.8	Paul *et al.* (2008)
Forest to pasture	Ecuador	South America	-14.8	Paul *et al.* (2008)
Forest to pasture	Ecuador	South America	-11.2	Paul *et al.* (2008)
Forest to pasture	Ecuador	South America	9.3	Paul *et al.* (2008)
Forest to pasture	Ecuador	South America	-1.8	Paul *et al.* (2008)
Forest to pasture	Ecuador	South America	-3.2	Paul *et al.* (2008)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to pasture	Ecuador	South America	-17.2	Rhoades *et al.* (2000)
Forest to pasture	Ecuador	South America	-9.8	Rhoades *et al.* (2000)
Forest to pasture	Ecuador	South America	-22.9	Rhoades *et al.* (2000)
Forest to pasture	Ecuador	South America	-15.1	Rhoades *et al.* (2000)
Forest to pasture	Ecuador	South America	2.3	Rhoades *et al.* (2000)
Forest to pasture	Ecuador	South America	-2.7	Rhoades *et al.* (2000)
Forest to crop	Ethiopia	Africa	-16.6	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	96.3	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	-34.0	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	85.5	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	-37.0	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	78.7	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	-46.6	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	58.1	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	-50.4	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	35.5	Lemenih *et al.* (2005)
Forest to crop	Ethiopia	Africa	-76.6	Lemma *et al.* (2006)
Forest to crop	Ethiopia	Africa	-56.2	Lemma *et al.* (2006)
Forest to crop	Ethiopia	Africa	-33.4	Lemma *et al.* (2006)
Forest to crop	Ethiopia	Africa	-22.2	Lemma *et al.* (2006)
Forest to crop	Ethiopia	Africa	-29.1	Lemma *et al.* (2006)
Forest to crop	Ethiopia	Africa	-54.9	Solomon *et al.* (2002)
Forest to crop	Ethiopia	Africa	-63.1	Solomon *et al.* (2002)
Forest to crop	Ethiopia	Africa	45.0	Yimer *et al.* (2007)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to crop	Madagascar	Africa	-42.9	Vågen *et al.* (2006)
Forest to crop	Nigeria	Africa	-75.0	Aina, (1979)
Forest to crop	Nigeria	Africa	-14.6	Ghuman and Lal (1991)
Forest to crop	Nigeria	Africa	-41.6	Ghuman and Lal (1991)
Forest to crop	Nigeria	Africa	-24.2	Ghuman and Lal (1991)
Forest to crop	Nigeria	Africa	-37.6	Ghuman and Lal (1991)
Forest to crop	Nigeria	Africa	-27.0	Ghuman and Lal (1991)
Forest to crop	Nigeria	Africa	-40.4	Ghuman and Lal (1991)
Forest to crop	Nigeria	Africa	-28.7	Ghuman and Lal (1991)
Forest to crop	Nigeria	Africa	-42.7	Ghuman and Lal (1991)
Forest to crop	India	Asia	-57.8	Chandran *et al.* (2009)
Forest to crop	India	Asia	-58.4	Chandran *et al.* (2009)
Forest to crop	India	Asia	-58.3	Chandran *et al.* (2009)
Forest to crop	India	Asia	-62.8	Chandran *et al.* (2009)
Forest to crop	Indonesia	Asia	-32.9	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-28.2	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-40.5	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-32.5	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-21.0	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-42.6	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-25.0	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-28.5	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-5.7	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-14.3	Dechert *et al.* (2004)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to crop	Indonesia	Asia	-8.3	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-9.4	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-38.1	Dechert *et al.* (2004)
Forest to crop	Indonesia	Asia	-34.8	Dechert *et al.* (2004)
Forest to crop	Thailand	Asia	-28.4	Jaiarree *et al.* (2011)
Forest to crop	Thailand	Asia	-52.3	Jaiarree *et al.* (2011)
Forest to crop	Thailand	Asia	-58.6	Jaiarree *et al.* (2011)
Forest to crop	Martinique	Caribbean	-27.8	Feller *et al.* (2001)
Forest to crop	Martinique	Caribbean	-42.7	Feller *et al.* (2001)
Forest to crop	Martinique	Caribbean	-33.9	Feller *et al.* (2001)
Forest to crop	Martinique	Caribbean	-54.8	Feller *et al.* (2001)
Forest to crop	Martinique	Caribbean	-60.3	Feller *et al.* (2001)
Forest to crop	Martinique	Caribbean	-16.7	Feller *et al.* (2001)
Forest to crop	Puerto Rico	Caribbean	-34.5	Brown and Lugo (1990)
Forest to crop	Puerto Rico	Caribbean	-70.4	Brown and Lugo (1990)
Forest to crop	Puerto Rico	Caribbean	1.1	Brown and Lugo (1990)
Forest to crop	Costa Rica	Central America	-15.1	Powers (2004)
Forest to crop	Costa Rica	Central America	-4.2	Powers (2004)
Forest to crop	Costa Rica	Central America	28.4	Powers (2004)
Forest to crop	Mexico	Central America	-38.2	Hughes *et al.* (2000)
Forest to crop	Mexico	Central America	38.4	Hughes *et al.* (2000)
Forest to crop	Mexico	Central America	-14.4	Hughes *et al.* (2000)
Forest to crop	Mexico	Central America	-38.2	Hughes *et al.* (2000)
Forest to crop	Hawaii	North America	-37.0	Bashkin and Binkley (1998)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to crop	Hawaii	North America	-48.9	Bashkin and Binkley (1998)
Forest to crop	Hawaii	North America	-26.8	Bashkin and Binkley (1998)
Forest to crop	Hawaii	North America	-44.2	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-31.1	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-35.7	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-43.5	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-27.5	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-4.6	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-22.8	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-25.2	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-13.2	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-32.6	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-23.5	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	6.1	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	65.0	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	31.0	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-10.0	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-13.9	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-30.8	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-13.0	Osher *et al.* (2003)
Forest to crop	Hawaii	North America	-30.0	Osher *et al.* (2003)
Forest to crop	Solomon Islands	Oceania	-59.3	Wairu *et al.* (2003)
Forest to crop	Solomon Islands	Oceania	-26.8	Wairu *et al.* (2003)
Forest to crop	Brazil	South America	-20.3	Denef *et al.* (2007)

Land use transition	Country	Region	Change in SOC (%)	Reference
Forest to crop	Brazil	South America	1.5	Denef *et al.* (2007)
Forest to crop	Brazil	South America	-39.2	Denef *et al.* (2007)
Forest to crop	Brazil	South America	-6.5	Denef *et al.* (2007)
Forest to crop	Brazil	South America	-34.9	Denef *et al.* (2007)
Forest to crop	Brazil	South America	-0.2	Denef *et al.* (2007)
Forest to crop	Brazil	South America	-38.6	Denef *et al.* (2007)
Forest to crop	Brazil	South America	-5.0	Denef *et al.* (2007)
Forest to crop	Brazil	South America	65.0	Eden *et al.* (1990)
Forest to crop	Ecuador	South America	-28.7	Rhoades *et al.* (2000)
Forest to crop	Ecuador	South America	-14.7	Rhoades *et al.* (2000)
Forest to crop	Ecuador	South America	-7.6	Rhoades *et al.* (2000)

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Annex 2. Mountain soils – Supplementary documents

Table 33. Soil organic C stock data in the considered mountain ranges and massifs, with the corresponding number of articles, studied soil depth and consideration or not of O horizons in the C stock calculations

Mountain range	N° of articles	Countries	Land use/cover	Reference	C stock (t/ha)	Depths (cm)	O horizons	LUC
Alps	16	Austria	Mixed (forest, grassland)	Djukic *et al.* (2010)	130–380	Profile	Yes	No
		Germany	Forest	Prietzel and Christophel (2014)	63–190	Profile	Yes	No
				Prietzel and Wiesmeier (2019)	63–190	Profile	Yes	No
				Wiesmeier *et al.* (2014)	92	Profile	Yes	No
		Italy	Mixed (forest, grassland, alpine tundra)	Canedoli *et al.* (2020)	38–79	0–40	Yes	No
			Alpine tundra	Freppaz *et al.* (2010)	27–132	Profile	No	No
			Mixed (forest, grassland)	Garlato *et al.* (2009a)	88–102	0–30	Yes	No
			Mixed (forest, grassland)	Garlato *et al.* (2009b)	50–165	0–30; 0–100	Yes	No
			Forest	Bonifacio, Falsone and Petrillo (2011)	70–150	Profile	Yes	No
			Forest	Pellis *et al.* (2019)	153–228	Profile	Yes	Yes
			Forest, grassland	Thuille, Buchmann and Schulze (2000)	20–50	A horizon	No	Yes
		Switzerland	Mixed (forest, grassland, alpine tundra)	Hoffmann *et al.* (2014a)	86*	0–10; 0–30	Yes	No
			Grassland	Leifeld *et al.* (2009)	53–116	0–30	No	No
			Alpine tundra	Budge *et al.* (2010)	55–102	0–30	No	No
			Forest	Perruchoud *et al.* (2000)	62–98	Profile		No
			Mixed (forest, alpine tundra)	Zollinger *et al.* (2013)	100–150	Profile	Yes	No
Altun	2	China	Mixed (forest, grassland)	Yan *et al.* (2019)	~0–60	Depth intervals	No	No

Mountain range	N° of articles	Countries	Land use/cover	Reference	C stock (t/ha)	Depths (cm)	O horizons	LUC
			Mixed (grassland, alpine tundra)	Zhao *et al.* (2019a)	26–80	0–15	No	No
Andes	3	Colombia	Forest (tropical montane rainforest)	De la Cruz-Amo *et al.* (2020)	31–170	Profile	Yes	No
		Ecuador	Forest (tropical montane rainforest)	Ließ, Schmidt and Glaser (2016)	10–250	0–50	Yes	No
		Colombia	Forest (tropical montane rainforest)	Phillips *et al.* (2019)	50–260	0–100	Yes	No
Appalachians	1	United States of America	Forest	Garten and Hanson (2006)	44–122	0–30	Yes	No
Apennines	5	Italy	Forests	Chiti *et al.* (2007)	34–43	0–20	No	Yes
			Forest	Conforti *et al.* (2016)	132*	Profile	Yes	No
			Forest	Conforti *et al.* (2020)	12–137	Profile	No	No
			Forest	De Feudis *et al.* (2020)	15–25	Profile	Yes	Yes
			Forest	Pellis *et al.* (2019)	135–279	Profile	Yes	Yes
Aravally	1	India	Forest	Kumar *et al.* (2010)	108–173	0–30	No	No
Atlas	2	Algeria	Mixed (forest, crops)	Bounouara *et al.* (2017)	100–170	Profile	No	No
		Morocco	Forest	Zaher *et al.* (2020)	71–213	0–30	No	Yes
Balkan	1	Bulgaria	Grassland	Karatoteva and Malinova (2017)	70–340	Profile	No	No
Carpathian	8	Poland	Forest, grassland	Bojko and Kabala (2017)	107.4	Profile	Yes	Yes
			Forest	Galka *et al.* (2014)	152–202	0–100	Yes	Yes
			Forest	Gruba *et al.* (2019)	84–110	Profile	Yes	No
			Forest	Reyna-Bowen *et al.* (2019)	50–905	0–100	Yes	No
			Forest, grassland	Sokolowska *et al.* (2020)	17.1–36	0–20	No	Yes
			Forest	Szopka (2016)	100–120	0–20	No	No
		Romania	Forest	Dincă *et al.* (2015)	111–253	0–100	No	No
			Forest	Valtera and Šamoni (2018)	182–320	0–20	No	No
Mt. Cameroon	1	Cameroon	Mixed (forest, agro-forestry)	Tsozuè *et al.* (2019)	150–300	0–75	No	No

Mountain range	N° of articles	Countries	Land use/cover	Reference	C stock (t/ha)	Depths (cm)	O horizons	LUC
Cascades	2	United States of America	Forest	Shaw, Boyle and Omule (2008)	23–114	0–5	No	No
			Forest	Sun *et al.* (2004)	70–360	0–100	Yes	No
Central Chain	3	Spain	Forest	Chiti, Díaz-Pinés and Rubio (2012)	34–59	0–30; 0–100	No	No
			Forest	Díaz-Pinés *et al.* (2011)	40–140	0–50	Yes	No
			Grassland, shrubs	Montané, Rovira and Casals (2007)	45–321	Profile	Yes	Yes
Datian Mtn	1	Taiwan province of China	Forest	Tsui, Tsai, and Chen (2013)	60–360	0–100	No	No
Daxing'an	1	China	Forest	Xiao, Man and Duan (2020)	99–239	0–100	No	No
Eastern Arc	4	Kenya	Forest, Tea plantation	Chiti *et al.* (2018)	350–400	0–100	Yes	Yes
		Mozambique	Forest	Guedes *et al.* (2018)	87–139	0–50	Yes	Yes
		United Republic of Tanzania	Mixed (forest, agroforestry)	Kirsten *et al.* (2016)	169–224	0–100	No	No
			Forest	Kirsten *et al.* (2019)	168–200	0–100	No	Yes
Elbutz	2	Iran (Islamic Republic of)	Forest	Kooch *et al.* (2012)	102–163	0–40	No	Yes
			Forest	Motlagh *et al.* (2020)	54–90	0–30	Yes	No
Mt. Elgon	1	Uganda	Agroforestry	Mugagga *et al.* (2015)	17–121	0–30	No	No
Ethiopian Massif	3	Ethiopia	Forest	Eshetu and Hailu (2020)	195–266	0–30	Yes	No
			Mixed (forest, grassland, crops)	Girmay and Singh (2012)	159–516	0–80	No	No
			Alpine tundra	Yimer, Ledin and Abdelkadir (2006)	350–450	0–100	No	No
Himalaya	9	India	Forest	Bangroo, Najar and Rasool (2017)	202–272	Profile	No	No
			Forest	Bhat *et al.* (2012)	19.2–35.4	0–30	No	No
			Grassland	Dad (2019)	29–95	0–50		No
			Mixed (forest, grassland, crops)	Dinakaran *et al.* (2018)	70–281	0–100		No
			Forest, grassland, crops	Martin *et al.* (2010)	135–424	0–150	No	Yes
			Mixed (forest, grassland, crops)	Singh *et al.* (2011)	107–227	0–100	No	No
		Nepal	Mixed (forest, crops)	Shrestha and Singh (2008)	62–157	0–70		No

Mountain range	N° of articles	Countries	Land use/cover	Reference	C stock (t/ha)	Depths (cm)	O horizons	LUC
			Forest, crops	Shrestha *et al.* (2009)	30–50	0–30	No	Yes
		Bhutan	Forest	Simon *et al.* (2018)	57–338	Profile		No
Japanese Alps	2	Japan	Forest	Li *et al.* (2010)	120–200	0–30	Yes	No
			Grassland	Toma *et al.* (2013)	21–416	0–100	No	No
Karakoram	1	Pakistan	Mixed (forest, grassland, crops)	Ali *et al.* (2017)	40–140	0–60	No	No
Kilimanjaro	1	United Republic of Tanzania	Mixed (forest, grassland, crops)	Pabst *et al.* (2016)	30–80	Profile	Yes	No
Lushan	1	China	Forest	Du *et al.* (2014)	70–150	0–60	Yes	No
Pyrenees	6	Spain	Forest	Campo *et al.* (2019)	15–125	0–50	Yes	Yes
			Grassland	Garcia-Pausas *et al.* (2007)	59–300	Profile	No	No
			Grassland, shrubs	Montané, Rovira and Casals (2007)	45–321	Profile	Yes	Yes
			Forests, grasslands	Nadal-Romero *et al.* (2016)	13–137	Profile	Yes	Yes
			Forest, grassland, shrubs	Nadal-Romero *et al.* (2018)	91–148	0–50		Yes
			Forest, grassland, shrubs	Urbina *et al.* (2020)	36–90	0–10	Yes	Yes
Qinhai Plateau - Tibet	3	China	Grassland	Liu *et al.* (2016)	164	0–100	No	No
			Alpine tundra	Ohtsuka *et al.* (2008)	10–137	0–30	No	No
			Grassland	Yang *et al.* (2008)	44–91	0–100	No	No
Rocky Mountains	2	Canada	Forest	Hoffmann *et al.* (2014b)	10–45	0–30	No	No
		United States of America	Mixed (forest, alpine tundra)	Scott and Wohl (2020)	50–200	Profile	No	No
Sayan	1	Mongolia	Forest	Tungalag, Gerelbaatar and Lobanov (2020)	94	0–30	No	No
Sierra Madre	1	Mexico	Forest	Santini *et al.* (2019)	78–109	0–100	No	No
Sierra Nevada	2	Spain	Mixed (dry habitats)	Román-Sánchez *et al.* (2018)	17–94.1	0–30	No	No
			Mixed (forest, grassland)	Willaarts *et al.* (2016)	9–66	Profile	No	No

Mountain range	N° of articles	Countries	Land use/cover	Reference	C stock (t/ha)	Depths (cm)	O horizons	LUC
SW Uganda	1	Uganda	Mixed (forest, agroforestry, crops)	Twongyirwe *et al.* (2013)	69–80	0–30	No	No
Yunnan	1	China	Mixed (forest, grassland, crops)	Duan *et al.* (2014)	40–760	profile	No	No
others	1	Arctic Areas	Alpine tundra	Palmtag *et al.* (2015)	83–300	0–100	Yes	No
others	1	Portugal	Forest	Fonseca *et al.* (2019)	140–200	0–30	Yes	Yes
others	5	Europe	Forest	Baritz *et al.* (2010)	11–126	0–20	Yes	No
			Forest	Bečvářová *et al.* (2018)	6–58	A horizon	Yes	No
			Forest	De Vos *et al.* (2015)	<50–400	Profile	Yes	No
			Forest, grassland, cop	Poeplau and Don (2013)	10–24	0–30	Yes	Yes
			Forest	Vanguelova *et al.* (2016)				No
others	1	Europe, United States of America	Mixed (forest, alpine tundra)	Egli *et al.* (2012)	~10–250	profile	No	No
others	2	World	Alpine tundra	Bockheim and Munroe (2014)	152	0–30	No	No
			Forest	Lal *et al.* (2005)	96–723			No
others	1	China	Alpine tundra	Chen *et al.* (2016)	95–311	0–50	No	No

In the first column, **"others"** include works dealing with C stocks in many mountain ranges, in whole countries or continents.

The category **"Mixed"** in the column **"Land use/cover"** refers to the presence of more than one land use/cover.

*****: median values.

LUC indicates the focus on land-use change in the paper, as reported in Figure 19 of the main chapter's document.

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Annex 3. Drylands – Supplementary documents

Table 34. Rates of change (ΔSOC) and SOC increase factors (ΔF) by conventional and unconventional practices (h) in croplands (C), grasslands (G), and forest (F) of representative reference conditions (HA»F), according to equation 2.24 of IPCC (2019) in diverse drylands in the world

Site	$H_{A»F}$	H_F	SOC_O t/ha	SOC_{O-T} t/ha	O-T yr	ΔSOC t/ha/yr	ΔF	h (not encoded) Practice(s)	Reference
Santa Maria, Argentina	AI 0.2, 1916 m a.s.l., temperate, mountain, Aridic Fluvisol, 0.3 depth	C	36.4	32.1	50	-0.08	0.998	Conventional tillage + use of organic fertilizer	FAO (2004)
		G	36.4	36.9	50	0.01	1.000	Meadow maintenance	
		C	36.4	56.6	50	0.40	1.011	No-till + animal manure 1.5 t/ha/yr, with organic fertilizer	
Tucuman, Argentina	AI 0.5, 360-420 m a.s.l., subtropic, plain, Haplic Kastanozem, 0.3 depth	C	65.0	67.6	3	0.04	1.013	No-tillage + use of inorganic fertilizers	Farage *et al.* (2007); FAO (2004)
		C	65.0	77.4	3	0.19	1.063	Green manures + compost	
		C	43.3	39.4	50	0.73	0.998	Continuous cultivation	
		C	43.3	41.5	50	0.73	0.999	No-till	
		C	43.3	48.3	50	0.73	1.002	No-till, No inorganic fertilizers, Manure 3.3 t/ha/yr.	
		G	43.3	42.6	50	0.73	1.000	Meadow maintenance	
Big Jacks, Australia	AI 0.65, 446 m a.s.l., temperate, hill, Luvic Phaeozem, 0.3 depth	G	47.4	45.7	8	-0.22	0.995	Continuous cutting with inorganic fertilizer 115 kgN/ha/yr	Young *et al.* (2009)
		G	73.6	79.9	8	0.79	1.011	Native vegetation	
Book Book, Australia	AI 0.65, 336 m a.s.l., temperate, hill, Subnatric yellow Sodosol, 0.3 depth	G	41.0	41.0	13	0.00	1.000	Periodic application of P, K, Mb; Annual pasture without liming	Chan, Roberts, and Heenan (1992)
		G	41.0	42.4	13	0.11	1.003	Periodic application of P, K, Mb. Conversion	

Site	$H_{A»F}$	H_F	SOC_O t/ha	SOC_{O-T} t/ha	O-T yr	ΔSOC t/ha/yr	ΔF	h (not encoded) Practice(s)	Reference
								to perennial grass without liming	
		G	41.0	44.1	13	0.24	1.006	Periodic application of P, K, Mb. Conversion to perennial grass with liming	
Hudson, Australia	AI 0.65, 411 m a.s.l., temperate, hill, Luvic Phaozem, 0.3 depth	G	38.6	39.6	8	0.12	1.003	Continuous cutting with inorganic fertilizer 90 kgN/ha/yr	Young *et al.* (2009)
		G	38.2	42.3	8	0.52	1.014	Conversion to perennial grasses. Use of inorganic fertilizer for pastures C3-C4 50 kg/ha/yr	
Wagga Wagga, Australia	AI 0.65, 198 m a.s.l., temperate, hill, Kandosol , 0.2 depth	CG	43.0 0	33.6	25	-0.38	0.991	3 pass tillage, stubble burnt, wheat/wheat	Chan, Roberts, and Heenan (1992)
		C	43.0 0	48.0	25	0.20	1.005	No tillage, stubble retained, wheat/clover rotation, keep mown	
Yarramanbah, Australia	AI 0.65, 147 m a.s.l., temperate, plain, Luvic Phaeozem, 0.3 depth	G	27.8	29.3	8	0.19	1.007	Continuous cutting with inorganic fertilizer 100 kg/ha/yr	Young *et al.* (2009)
Santa Teresinha, Brazil	AI 0.5, Eutric Leptosol, 281 m a.s.l., tropic, hill, 0.5 depth	F	32.5	26.0	100	-0.07	0.998	Cut every 20 years with controlled fire	Althof *et al.* (2015)
		F	32.5	28.3	100	-0.04	0.999	Cut every 10 years with controlled fire	
		F	32.5	32.5	100	0.00	1.000	Only one court of Caatinga, without fire	
Bugacpuszta, Hungary	AI 0.65, 109 m a.s.l., temperate, plain, Eutric Regosol, 0.3 depth	G	20.4	22.2	1	1.86	1.091	The highest precipitation and lowest temperature. Positive extreme case (highest capture value in the normal case studies).	Pinter *et al.* (2008)

Site	$H_{A»F}$	H_F	SOC_O t/ha	SOC_{O-T} t/ha	O-T yr	ΔSOC t/ha/yr	ΔF	h (not encoded) Practice(s)	Reference
Szurdokpospoki Hungary	Al 0.65, 205 m a.s.l., temperate, mountain, Haplic Phaeozem, 0.3 depth	F	38.9	38.2	1	-0.73	0.981	The lowest precipitation and highest temperature. Extreme negative case	
Gawalpahari, India	Al 0.5, 259 m a.s.l., tropic, hill, Aridic Luvisol, 0.6 depth	SC	16.6	18.3	2	0.81	1.049	Incorporation of 0.81 tC/ha/yr of liter and total sequestration estimate of 2.94 tC/ha/yr	Bhojvaid and Timmer (1998)
Lingampally, India	Al 0.5, 568 m a.s.l., tropic, plain, Chromic Luvisol, 0.3 depth	C	19.3	18.4	50	-0.02	0.999	Conventional practice	FAO (2004)
		C	19.3	30.9	50	0.23	1.012	Incorporation of vegetal waste into the soil	
		C	19.3	35.6	50	0.33	1.017	Manure 3 t/ha/yr. Plant residues, green manures, worm compost	
Malligere, India	Al 0.5, 756 m a.s.l., subtropic, hill, Chromic Luvisol, 0.3 depth	C	18.5	22.4	50	0.73	1.004	Current practice. Waste grazing. Inorganic fertilizers urea and diammonium phosphate (75 kg/ha/yr)	FAO (2004)
		C	18.5	29.8	50	0.73	1.012	Manure. Waste grazing. Inorganic fertilizers 75 kg/ha/yr diammonium phosphate and 75 kg urea/ha/yr	
Metalkunta, India	Al 0.65, 619 m a.s.l., tropic, plain, Plinthic Luvisol, 0.3 depth	C	25.0	27.7	50	0.73	1.002	Conventional practice	FAO (2004)
		C	25.0	45.2	50	0.73	1.016	Incorporation of plant residues	
Ojuelos, Mexico	Al 0.5, 2219 m a.s.l., temperate, hill, Epileptic Cambisol, 0.3 depth	G	44.0	44.1	28	0.00	1.000	Pasture with moderate grazing	Delgado - Balbuena *et al.* (2013)
		G	44.0	51.2	28	0.26	1.006	Scrubbing	

Site	$H_{A»F}$	H_F	SOC_O t/ha	SOC_{O-T} t/ha	O-T yr	ΔSOC t/ha/yr	ΔF	h (not encoded) Practice(s)	Reference
Athi Kamunyuni, Kenya	AI 0.5, 900 m a.s.l., subtropic, mountain, Epileptic Regosol, 0.3 depth	C	26.7	25.90	50	-0.02	0.999	Conventional practice	FAO (2004)
		G	26.7	30.48	50	0.08	1.003	Conventional grazing	
		C	26.7	35.96	50	0.19	1.007	Manure 1.25 t/ha/yr	
Darajani, Kenya	AI 0.5, 656 m a.s.l., tropic, table, Eutric Regosol, 0.3 depth	CG	33.7	32.55	50	-0.02	0.999	Intercropping corn-millet crops	FAO (2004)
		CG	33.7	61.15	50	0.55	1.016	Manure 4.5 t/ha/yr, burning of residues and fallow. Alternate corn-millet crops	
		CG	33.7	67.53	50	0.68	1.020	Manure 4.5 t/ha/yr, without burning of residues, no fallow. Intercroping corn-millet crops	
Kaiani, Kenya	AI 0.5, 1039 m a.s.l., subtropic, mountain, Aridic Regosol, 0.3 depth	C	35.2	35.09	50	0.00	1.000	Conventional practices	FAO (2004)
		C	35.2	52.15	50	0.34	1.010	Manure 2 t/ha/yr, Millet-cowpea intercrop.	
		C	35.2	53.94	50	0.37	1.011	Manure 2 t/ha/yr. Millet-cowpea intercrop. Vegetable waste 0.3 t/ha/yr	
Kymausoi, Kenya	AI 0.5, 894 m a.s.l., subtropic, mountain, Ferric Luvisol, 0.3 depth	F	38.4	38.4	50	-0.02	1.000	Native vegetation	FAO (2004)
		CG	33.5	32.2	50	0.10	0.999	Conventional practice	
		CG	33.5	39.4	50	0.33	1.004	Vegetable waste 0.3 t/ha/yr	
		CG	33.5	53.8	50	0.73	1.012	Manure 1.5 t/ha/yr; Plant waste 3 t/ha/yr	
Calpulalpan, Mexico	AI 0.5, 2648 m a.s.l., temperate,	C	0.1	11.1	50	0.22	3.200	Corn monoculture and harvest residue removal	Báez Pérez *et al.* (2021)

Site	$H_{A»F}$	H_F	SOC_O t/ha	SOC_{O-T} t/ha	O-T yr	ΔSOC t/ha/yr	ΔF	h (not encoded) Practice(s)	Reference
	mountain, Andic Cambisol, 0.3 depth	C	0.1	24.1	50	0.48	5.800	Rotation of corn with other crops with permanent application of manure	
Sidi El Aidi, Morroco	AI 0.5, 269 m a.s.l., temperate, plain, Vertic Calcixeroll Clay, 0.5 depth	C	41.1	45.6	11	0.41	1.010	Zero tillage	Bessam and Mrabet (2003)
		C	41.1	48.3	11	0.66	1.016	Conventional tillage	
Dagaceri, Nigeria	AI 0.5, 399 m a.s.l., tropic, plain, Luvic Arenosol, 0.3 depth	CG	13.6	17.67	50	0.08	1.006	Conventional practice	FAO (2004)
		CG	13.6	25.84	50	0.24	1.018	Manure 1.29 t/ha/yr, short fallow, waste grazing	
Futchimiram, Nigeria	AI 0.2, 318 m a.s.l., tropic, plain, Chromic Arenosol, 0.3 depth	C	5.5	3.85	50	-0.03	0.994	Continuous cultivation	FAO (2004)
		C	5.5	5.10	50	-0.01	0.999	Conventional practice	
		C	5.5	5.90	50	0.01	1.001	Vegetal waste 0.5 t/ha/yr	
		C	5.5	11.61	50	0.12	1.022	5-year fallow, 5-year cultivation two applications of organic fertilizer of 3 t/ha/yr, and waste grazing	
Kasha, Nigeria	AI 0.2, 343 m a.s.l., tropic, plain, Eutric Gleysol, 0.3 depth	C	7.5	7.7	50	0.00	1.001	Conventional practice	FAO (2004)
		C	7.5	16.2	50	0.17	1.023	Fallow, Manure 3 t/ha	
Tumbau, Nigeria	AI 0.5, 420-460 m a.s.l., tropic, plain, Eutric Regosol, 0.3 depth	C	14.5	12.0	50	-0.05	0.997	Conventional tillage	Farage *et al.* (2007); FAO (2004)
		C	14.5	12.3	50	-0.05	0.997	Inorganic fertilizers only	
		C	14.5	18.5	50	0.08	1.006	Compost, legumes, retained plant residues	
		C	14.5	19.5	50	0.10	1.007	Additional afforestation	

Site	$H_{A»F}$	H_F	SOC_O t/ha	SOC_{O-T} t/ha	O-T yr	ΔSOC t/ha/yr	ΔF	h (not encoded) Practice(s)	Reference
		C	14.5	15.7	50	0.73	1.002	Conventional practice. Waste grazing	
		C	14.5	38.0	50	0.73	1.032	Manure 6.75 t/ha/yr. Plant waste 2 t/ha/yr	
		C	14.5	18.1	50	0.73	1.005	Manure 3.75 t/ha/yr. Waste grazing	
Aldeia di Biscaia, Portugal	AI 0.65, 265 m a.s.l., temperate, hill, Aridic Regosol, 0.05 depth	FG	35.1	35.9	5	0.19	1.005	Incorporation of shrub residues 23.4 t/ha, for two years, with complementary applications of N (80 kg/ha) and P (35 kg/ha)	Madeira *et al.* (2012); Merino *et al.* (2019)
Évora, Portugal	AI 0.5, Mediterranean plain, Luvisol, 0.3 depth, 38°28'N, 7°28'W Rotation lupin-wheat-forage oat – barley	C	61.0	61.0	11	0.00	1.000	Conventional tillage	Carvalho *et al.* (2012)
		C	65.0	75.0	11	10.00	1.153	Reduced tillage	
		C	66.0	82.0	11	16.00	1.242	No-till + cereal straw removal	
		C	66.0	120.0	11	54.00	1.818	No-till without straw removal	
Old Peanut, Senegal	AI 0.5, 9 m a.s.l., tropic, plain, Sandy Dior and Hydromorphic Deck, 0.3 depth	C	11.9	12.4	25	0.02	1.002	Compost 2 t/ha/yr	Tschakert, Khouma and Sène (2004); FAO (2004)
		G	11.9	13.4	25	0.06	1.005	Converting from crop to pasture with grazing	
		C	11.9	14.4	25	0.1	1.008	Manure 4 t/ha/yr	
		C	11.9	15.4	25	0.14	1.012	Fallow 3 yr and manure 2 t/ha/yr. Rotation 4 yr of cultivation	
		G	11.9	16.2	25	0.17	1.014	Converting from crop to pasture without grazing.	

Site	$H_{A»F}$	H_F	SOC_O t/ha	$SOC_{O\text{-}T}$ t/ha	O-T yr	ΔSOC t/ha/yr	ΔF	h (not encoded) Practice(s)	Reference
		C	11.9	16.4	25	0.18	1.015	Fallow 3 yr, 2 t Leucaena, rotation with cultivation every 4 yr	
		G	11.9	17.7	25	0.23	1.019	Conversion from crop to pasture without grazing with tree plantations of Faidherbia albida	
		C	11.9	18.2	25	0.25	1.021	Fallow 10 yr, manure 2 t/ha/yr, rotation with 6 year of cultivation	
		C	11.9	22.7	25	0.43	1.036	Improved millet, manure, incorporation of Laucaena, inorganic fertilization, animal traction and fallow 1 yr	
		C	8.5	7.1	27	-0.17	0.994	Long continuous crop of sorghum	
Kordofan-Kaba, Senegal	AI 0.2, 580-640 m a.s.l., tropic, plain, Chromic Vertisol, 0.2 depth	C	8.5	8.9	27	0.04	1.001	Conversion from only cultivation to cultivation: fallow (5:20)	Ardö and Olsson (2004); Farage *et al.* (2007)
		C	4	4.6	27	0.15	1.006	Tree sowing. Conversion from cultivation to cultivation: fallow (5:20)	
Kordofan-UmmHiglig, Senegal	AI 0.2, 559 m a.s.l., tropic, plain, Arenic Cambisol, 0.3 depth	C	10.1	8.6	27	-0.15	0.994	Long continuous crop of sorghum	Ardö and Olsson (2004)
		C	10.1	10.5	27	0.04	1.001	Conversion from only cultivation to cultivation: fallow (5:20)	
Cheyenne, United States of America	AI 0.5, 1917 m a.s.l., temperate, plain, Aridic	G	47.9	58	12	0.84	1.018	Continuous light grazing	Schuman (2009)

Site	$H_{A»F}$	H_F	SOC_O t/ha	SOC_{O-T} t/ha	O-T yr	ΔSOC t/ha/yr	ΔF	h (not encoded) Practice(s)	Reference
	Argiustolls , 0.3 depth	G	47.9	58.3	12	0.86	1.018	Continuous heavy grazing	
Fort Collins, United States of America	Al 0.5, 1930 m a.s.l., temperate, plain, Fine-loamy, mixed, mesic, Ustollic Haplargid, 0.3 depth	G	37.4	38.2	55	0.01	1	Short grasses	Reeder and Shuman (2002)
		G	37.3	42.3	55	0.09	1.002	Short grasses	
		G	58.3	66.5	12	0.68	1.012	Mix grasses	
		G	58.3	67.4	12	0.76	1.013	Mix grasses	
Pacific Northwest, United States of America	Al 0.5, 795 m a.s.l., temperate, hill, Luvic Phaeozem, 0.3 depth	C	61	41.8	55	-0.35	0.994	Native vegetation to continuous cultivation	Brown and Huggins (2012)
		C	61	60.6	12	-0.03	1	Burning waste	Horner *et al.* (1960)
		C	61.0	63.1	10	0.21	1.003	No-till	Brown and Huggins (2012)
		C	61.0	69.3	12	0.69	1.011	Mixed-perennial crops + Crop rotation	

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