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Status of the World's Soil Resources

Main Report

Soil change:
impacts and
responses
Chapter 7
The impact of
soil change on
ecosystem services



GLOBAL SOIL
PARTNERSHIP



2015
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Soil change: impacts and responses



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7 | The impact of soil change on ecosystem services

7.1 | Introduction

Soils are now recognized to be in the 'front line' of global environmental change and we need to be able to predict how they will respond to changing climate, vegetation, erosion and pollution. This requires a better understanding of the role of soils in the Earth system to ensure that they continue to provide for humanity and the natural world (Schmidt *et al.*, 2011). Although only a thin layer of material at the Earth's surface, soils like many interfaces play a pivotal role in regulating the flow and transfer of mass and energy between the atmosphere, biosphere, hydrosphere and lithosphere. Moreover, the structure and organization of soils leaves an important imprint on the Earth's surface in terms of how land is used and how ecosystems develop. Soils help regulate the Earth's physical processes such as water and energy balances, and act as the biogeochemical engine at the heart of many of the Earth system cycles and processes on which life depends. Some soil processes contribute directly to the delivery of ecosystem goods and services, while other soil processes influence the delivery of goods and services. This section examines how soil processes affect soil and ecosystem function and the production of goods and services of benefit to humanity.

Humanity has had an indelible impact on the Earth's surface, so much so that it has been proposed that the planet has entered a new geological epoch, the Anthropocene (Crutzen, 2002). A population of ca. 7 billion people that will likely grow to 9.6 billion by 2050 is stressing Earth's resources. Maintaining the planet in an equitable state for human life is perhaps our greatest challenge. Currently, humans have adapted 38 percent of the earth's ice-free land surface to agriculture, crops and pasture (Foley *et al.*, 2011). Agricultural production, driven by the need to produce food for a growing population, has had a tremendous impact on our ecosystems and resources, especially through the abstraction of water and the leaving of residues. Rockström *et al.* (2009) proposed that we need a 'safe operating space for humanity with respect to the Earth system'. They argue that that there exist biophysical planetary boundaries (or thresholds) which it is inadvisable to cross if we are to maintain the needed balance. Vince and Raworth (2012) adapted these concepts to include social goals (1). This presentation underlines the fact that we live in a coupled human earth system. The ecosystem services analytic approach has been developed in order to bridge the science/policy divide. The approach aims to make the concepts clear for all and to set out what needs to be considered in order for humanity to live within sustainable boundaries.



Soils and soil security are at the heart of this effort. Soil security is defined in McBratney, Field and Koch (2014) as “maintaining and improving the world’s soil resources to produce food, fibre and freshwater, to contribute to energy and climate sustainability, and to maintain the biodiversity and the overall protection of the ecosystem”. Soils perform important ecosystem services (e.g. functions for humanity) including: biomass production; storing, filtering and transforming nutrients and water; maintaining a gene pool; providing a source of raw material for products such as bricks and tiles; regulating climate and hydrology; and providing an archive of cultural heritage. Soils provide ecosystem goods and services directly but some soil processes can have an adverse impact on the delivery of ecosystem goods and services. The ability of soils to function can be threatened by human activity (on this, see the Soil Thematic Strategy, SEC, 2006). A growing population, resource extraction, agricultural production, land use change and climate change all contribute to this threat.

As population increases, food security is becoming more important in the global agenda. Our historical solution to producing more food has been to mechanize, cultivate more land, and increase the application of plant nutrients and water. This has led to an almost linear increase in production over time (Pretty, 2008). However, the rate of increase is likely to plateau, as has already been seen with wheat in Northern Europe and with rice in Korea and China (Cassman, Grassini and Wart, 2010). In addition, agricultural growth comes with environmental costs or externalities, which are costs not accounted for in the cost of production. The degradation caused can adversely affect everyone, and even the production systems themselves - for instance, declines in pollinators can threaten future production (Deguines *et al.*, 2014).

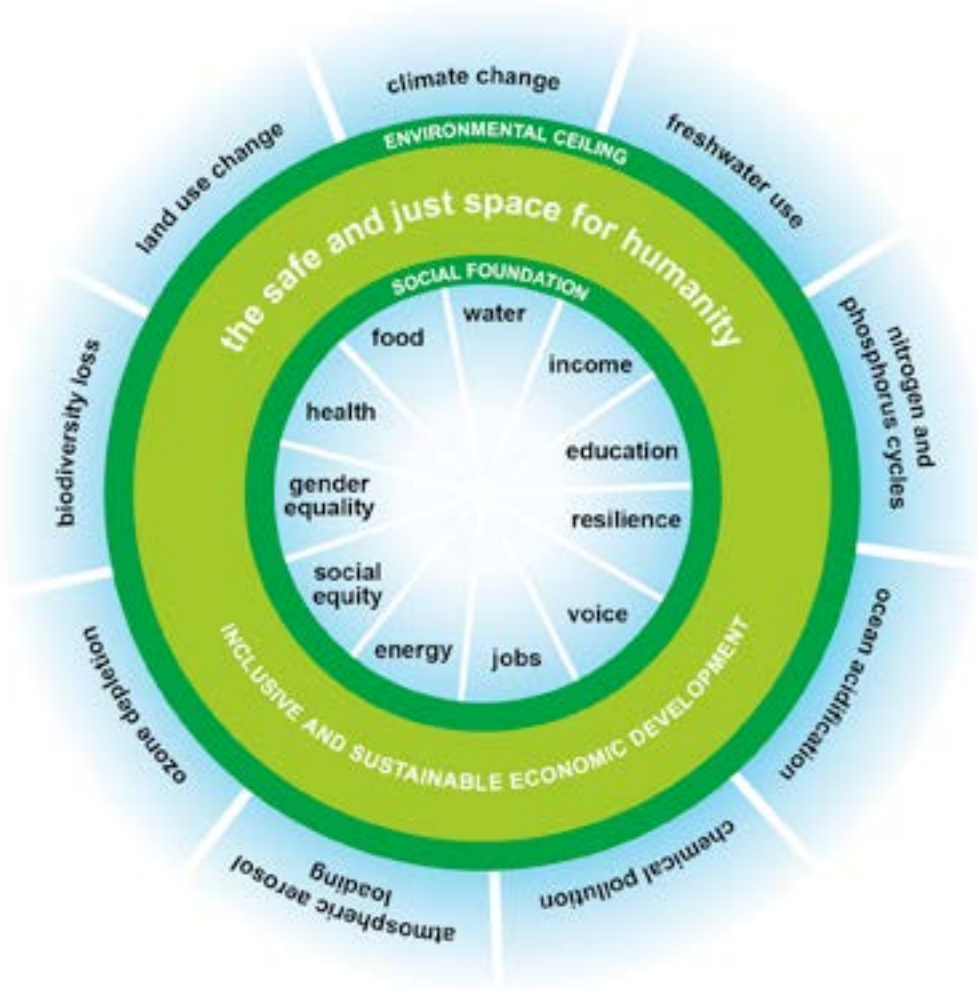


Figure 7.1 | The 11 dimensions of society’s ‘social foundation’ and the nine dimensions of the ‘environmental ceiling’ of the planet. Source: Vince and Raworth, 2012.



Rockström *et al.* (2009) suggest we are approaching the limits of the planet’s cultivatable land, while the addition of nutrients, especially nitrogen, continues to overload many terrestrial-aquatic systems (Diaz and Rosenberg, 2008). At the same time, arable production has seen declines in the carbon content of soils - the largest terrestrial carbon reservoir – and these declines are affecting other soil functions, including water and nutrient retention (Reynolds *et al.*, 2013). Food production systems will need to change to create multifunctional agro-ecosystems capable of maintaining a balance between yields, soil functions and biological diversity. Within the field of ecology, this challenge has led to a rigorous debate concerning the loss of natural species from agricultural lands - often termed, the ‘land sparing, land sharing’ debate (Green *et al.*, 2005). This debate has now been integrated within the broad ecosystem services discussion whose central ten also focuses on human interaction with ecosystems and their long-term sustainability and continued functionality (MA, 2005).

Conceptually, Foley *et al.* (2005) proposed that a natural ecosystem provides a range of goods and services (2) while on intensively farmed agricultural land, crop production dominates at the expense of all other goods and services. They proposed that an ideal situation would be one of balance, with the system producing a range of goods and services including food – the ‘sharing’ side of the land sparing/ land sharing debate. Organic agriculture has been seen as a model of this sharing or balance. However, organic agriculture has so far generally failed to maintain productivity levels in either crop or livestock systems (Pretty, 2008). The implication is that organic agriculture does not yet promise balance, because it requires more land and more use of natural capital to maintain production levels. Determining if there are viable ‘sharing’ systems should continue to be an important research goal but for the moment ‘sparing’ appears to have the upper hand in the debate (Phalan *et al.*, 2011). But how do we achieve sustainable intensification? While the viability of sharing remains in question, should we focus on a narrow-minded, single service supply management strategy, e.g. arable soils for crop production or peat soils for carbon storage? Sustainable intensification research, which seeks to find ways of optimizing production while blending in new strategies for multifunctional ecosystem service management, is being championed as a way forward (Firbank *et al.*, 2013).

There is no single solution. Foley’s conceptual diagram (2) highlights the challenges and possible trade-offs: a natural ecosystem delivers a wide range of ecosystem services but scant production; and an intensive cropland system delivers royally on production but precious little on ecosystem services. A balanced system of cropland with restored ecosystem services would deliver on all services, including production. A recent synthesis and analysis of data from the Countryside Survey, a national survey of Great Britain, suggests that Foley’s conceptual diagram of intensive cropland (2) is the current situation. Different services reach optimums at different points along the productivity gradient, but we cannot have everything (3). The ecosystem service indicators alter, often in a non-linear way with the proportion of intensive land use – but with exception of production, they all decline with intensification. 3 b and c go on to show that changes in moisture inputs or moisture regime, or alteration of soil pH would change the service delivery balance. At no point do we get everything, so we will need to choose priorities with our current systems.

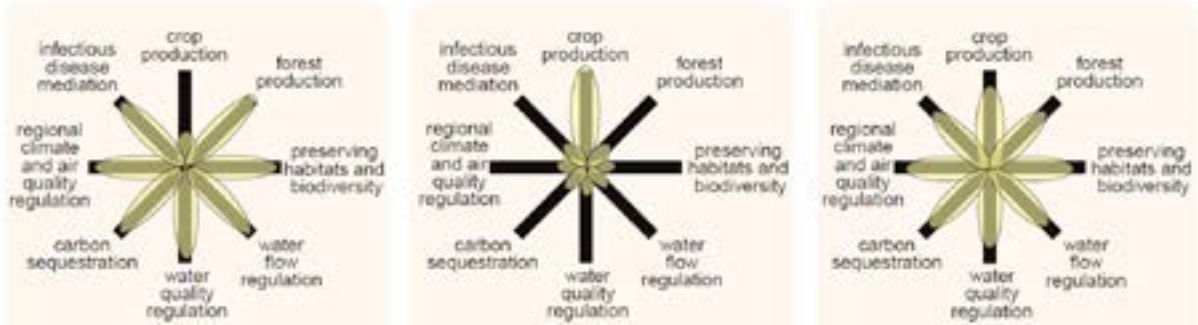


Figure 7.2 | Conceptual framework for comparing land use and trade-offs of ecosystem services. Source: Foley *et al.*, 2005.



The land sparing, land sharing approach can also be framed in terms of resilience (sharing) and efficiency (sparing). Efficient systems by their very nature will prioritise the performance of one function over that of others. The degree to which others are affected will depend on whether they perform well under similar management or not. Currently, the data in 3 indicate that the choice lies between efficiency and redundancy. We can have an efficient carbon storage system, e.g. peat development, which may also perform well as a climate thermal buffer because the conditions for peat accumulation require lots of water, but it will not be productive for crops in that state, nor will the arable system have high biodiversity as this is inefficient.

Choices need to be made as to what types of systems we wish to promote. In light of this, the focus of this chapter is to assess the global scientific literature and understand how soil change discussed in Chapters 5 and 6 is likely to impact soil functions and the likely consequences for ecosystem service delivery. Each section of this chapter outlines key soil processes involved with the delivery of goods and services and how these are changing or - where evidence permits - may change. Each section then reviews how this change impacts soil function and affects ecosystem service delivery. Some soil change does not produce an ecosystem service, but does impact it; these impacts are considered when assessed as important and adverse. The focus is on the local, regional and global scales and follows the general reporting categories of the MA (2005) modified by TEEB (2014) to provisioning, regulating and cultural services. Towards the end of the section there is a focus on the links with policy, institutions and management.

7.2 | Soil change and food security

Keating *et al.* (2014) provide a useful frame for examining the main roles of soils in food supply through their development of the food wedge concept. The food wedge is the triangular area between the level of food demand in 2010 and the upper bound of food demand in 2050 (suggested by Keating and Carberry (2010) as a wedge equal to approximately 127×10^{15} kcal). The food wedge presented by Keating *et al.* (2014) assumes that food supply and demand were broadly in balance in 2010. Increases in food supply (through, for example, the strategies suggested by Foley *et al.*, 2011) would increase the supply to meet the rising demand for food.

Either the incremental loss of productivity from current agricultural land or the total loss of agricultural land due to degradation in the future would cause the lower boundary of the wedge to decrease and hence increase the gap between food supply and demand (Figure 7.4). This decrease (or total loss) could occur if the services for plant production supplied by the soil decreased due to a significant impairment of one or more of the soil functions. Alternatively the restoration of productivity to previously degraded land would increase plant production in addition to addressing the yield gap or increases in food delivery. Therefore a key soil-focused strategy is to reduce future productivity loss from agricultural soils due to degradation to a minimum and to restore productivity to soils that have previously experienced productivity losses.



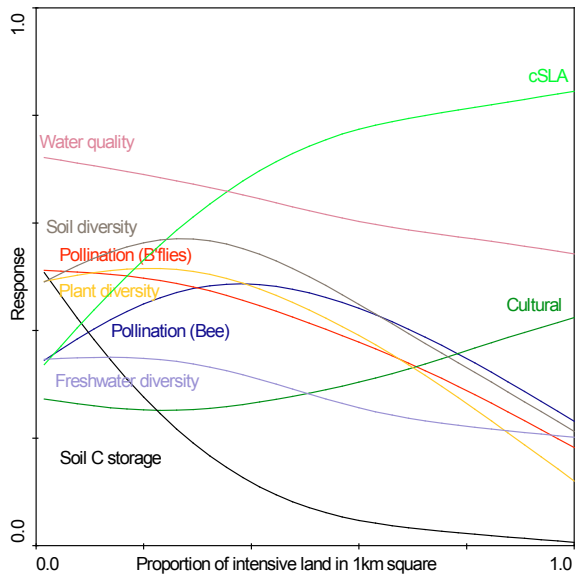
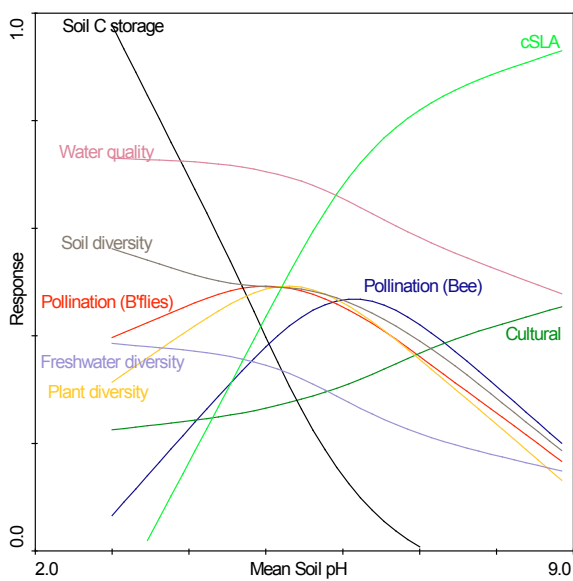
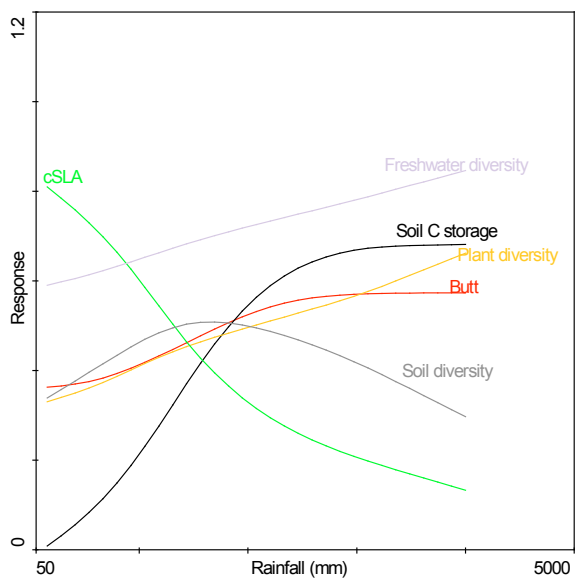


Figure 7.3 | Response curves of mean ecosystem service indicators per 1-km² across Great Britain.

Source: Maskell et al., 2013.

The curves are fitted using generalized additive models to ordination axes constrained by; (a) proportion of intensive land (arable and improved grassland habitats) within each 1-km square from CS field survey data; (b) mean long-term annual average rainfall (1978–2005); and (c) mean soil pH from five random sampling locations in each 1-km square. All X axes are scaled to the units of each constraining variable



The restoration of productivity on degraded soils can be complex insofar as soils may have been degraded to the point where they cannot readily respond to fertility-improving management techniques. These complex interactions among inherent soil properties, management history and the response to inputs is well illustrated in the work of Rusinamhodzi *et al.* (2013) on maize production intensification on smallholder farms in Zimbabwe. In this region two major controls of productivity exist – significant differences in yield between sandy and clay soils (e.g. inherent soil properties); and pronounced fertility gradients between more productive fields close to the homestead and more degraded soils in outfields further from the homestead (a management-induced fertility gradient common in many areas of Africa). The sandy soils required long-term additions of manure to restore soil functions before the benefit of the mineral fertilizer additions could begin to be realized; however even after nine years of substantial organic inputs, the highly degraded sandy outfields did not recover their productivity. The authors speculate that the initial soil organic carbon levels in the sandy outfields were too low for yields to recover. Moreover at the village scale, the overall amount of manure produced is insufficient to apply the required amounts of manure in all fields.

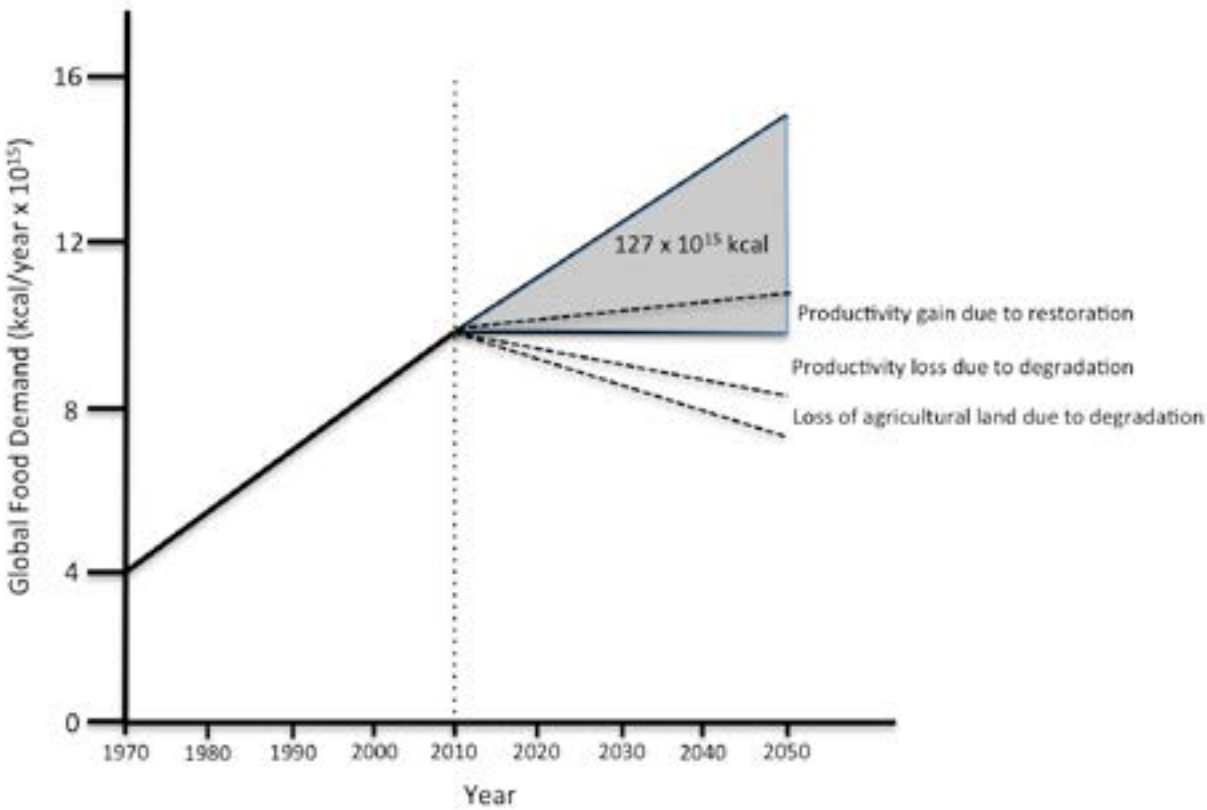


Figure 7.4 | The food wedge and the effect of soil change on the area of the wedge. Source: Keating *et al.*, 2014.

The relative sizes of the effects of soil change on the food wedge are not drawn to scale.



One approach to maintaining soil health is 'conservation agriculture', which comprises a range of agricultural practices that include reduced tillage and no-till, greater retention of crop residues, and crop rotations. However, the lack of organic inputs which constrained productivity in the Zimbabwe example above also limits the ability of conservation agriculture to restore fertility in sub-Saharan Africa generally. Palm *et al.* (2014) found that the greatest obstacle to improving soil functions and other ecosystem services in Sub-Saharan Africa region is the lack of residues produced due to the low productivity of the soils. The limited supply of crop residues also highlights the need to make optimum use of all sources of organic inputs, such as animal manure and properly processed human wastes.

These studies emphasize the inability of mineral fertilizers alone to significantly increase food production in regions where the yield gap is greatest. Removing the nutrient limitations through additions of mineral fertilizers alone will also exacerbate the range of environmental issues (e.g. N₂O emission from N-fertilizer, surface and groundwater contamination) in all food-producing regions unless the efficiency of crop use of agricultural inputs can be increased. Additionally the fraction of P available as mineable phosphate rock is finite. Recent concerns that the world's supply of phosphorus was being rapidly depleted and that 'peak phosphorus' was only a few decades away (Cordell and White, 2010) have been dispelled, due to recent upward revisions of world phosphate rock reserves and resources (Van Kauwenbergh, 2010). However, the world supply of phosphorus is limited, and rising prices and market volatility are inevitable. More efficient use of phosphorus is therefore essential. This overall issue is termed the 'Goldilocks' problem by Foley *et al.* (2011) – there are many regions with too much or too little fertilizer but few that are 'just right'.

A final strategy is to minimize diversion of agricultural soils to production of non-food crops. Recent large-scale bioenergy production on land previously used for food production has driven a significant land use change and represents a major shift of agricultural soils away from food production. Demand for soybean, maize and oil palm for biofuel has been a driver of agricultural land conversion in recent years particularly in Latin America. Conversion of existing cropland or the development of new cropland for bioethanol and biodiesel production competes with food production and carbon returns to the soil (Foley *et al.*, 2011) and thus constitutes a threat to soil and food security. Biofuels produced from crops using conventional agricultural practices will exacerbate stresses on water supplies, water quality and land use. In any case, biofuels are not expected to mitigate the impact of climate change as compared with petroleum (Delucchi, 2011).

Threats to the food security dimension 'availability' are mainly (but not only) caused by soil and land degradation and associated water resources (Khan *et al.*, 2009). This is particularly the situation when the degradation is irreversible or very hard to reverse. This may, for example, be the case with severe topsoil losses caused by wind or water erosion, terrain deformation by gully erosion or mass movement, acidification, alkalization/salinization, soil sealing, or contamination with toxic substances (Scherr, 1999; Palm *et al.*, 2007; Mullan, 2013). The resulting loss of productivity will reduce yields from a site, leading to reduced returns to producers and, in some cases, abandonment of production at the site. Productivity may be restored, but economic considerations may limit the adoption of restorative measures.

The impact of each threat on specific soil functions relevant to crop production has been covered in Chapter 6 and is summarized in Figure 7.5. The present chapter will focus on the implications for food security of the trends in each threat.

7.2.1 | Soil erosion

A summary by den Biggelaar *et al.* (2003) suggests that global mean rates of erosion are between 12 to 15 tonnes ha⁻¹ yr⁻¹ (Table 7.1). The mid-point of this range yields a soil loss of 0.9 mm yr⁻¹ (see Table 7.1), very similar to the mean soil loss of 0.95 mm yr⁻¹ calculated by Montgomery (2007). Overall these rates are substantially higher than rates of soil formation, and hence pose a long-term global threat to soils (Montgomery, 2007; see also Section 6.1 above).



Our understanding of the rates for the three erosion agents (wind, water and tillage) is uneven. Erosion rates due to water erosion remain very high (> ca. 20 tonnes ha⁻¹ yr⁻¹) in cropland in many agricultural regions (Figure 7.2); essentially any cropped area with hilly land and sufficient precipitation is at risk. No reliable global estimates for current wind erosion rates exist, and the estimates of the human contribution to current dust emissions range from only 8 percent in North Africa to approximately 75 percent in Australia (see also Section 6.1 above). Tillage erosion primarily results in in-field redistribution of soil, and decreases the productivity of soils in convex slope elements and near-upslope field or terrace borders. Global-scale summaries also require consideration of the fate of eroded soil – in some regions deposition of eroded soil in river floodplains and deltas creates areas of very high and enduring fertility.

The effect of soil erosion on individual soil properties related to crop production is well documented, but the aggregate effect of soil loss on crop yields themselves is less firmly established. The four integrative studies summarized in Table 7.1 are based on data sources which range from experimental plot data to re-interpretation of GLASOD data. The range of estimates of annual crop loss due to erosion ranges from 0.1 percent to 0.4 percent, with two studies estimating 0.3 percent yield reduction.

If the median value of 0.3 percent annual crop loss is valid for the period from 2015 to 2050, a total reduction of 10.25 percent could be projected to 2050 (assuming no other changes such as the adoption of additional conservation measures by farmers). Foley *et al.* (2011) cite a value of 1.53 billion ha for cropland globally; the loss of 10.25 percent of yield due to erosion would be equivalent to the removal of 150 million ha from crop production or 4.5 million ha per year.

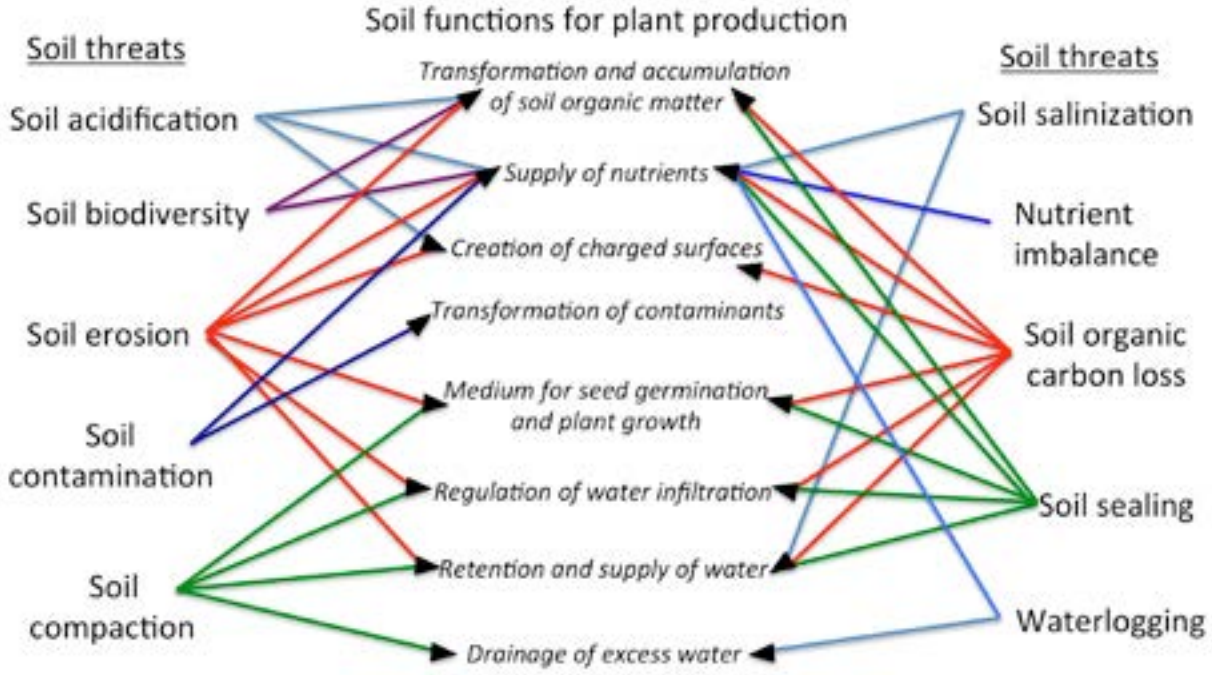


Figure 7.5 | Direct impacts of soil threats on specific soil functions of relevance to plant production.



Table 7.1 | Erosion and crop yield reduction estimates from post-2000 review articles

¹ Calculated using average bulk density of 1.5 tonnes m⁻³ (den Biggelaar *et al.*, 2003) and average erosion rate of 13.5 tonnes ha⁻¹ yr⁻¹ (mid-point of den Biggelaar *et al.*, 2003 range)

| Author | Database Used | Extent | Estimates |
|--------------------------------------|---|----------------------------------|---|
| Den Biggelaar <i>et al.</i> (2003) | Erosion: 179 plot-level studies Crop yield-erosion: 362 | Global (37 countries) | Erosion: Average rates between 12 – 15 t ha ⁻¹ yr ⁻¹ (0.8 to 1.0 mm per year) ¹ Relative annual crop yield reduction due to erosion: 0.3 percent per year (for six major crops) |
| Bakker, Govers and Rounsevell (2004) | Erosion-yield: 24 experimental studies | Primarily North America + Europe | Yield reductions of approximately 4 percent per 10 cm soil loss (= 0.36 percent per year) ¹ |
| Scherr (2003) | 28 regional studies and 54 national or sub-national studies on soil degradation (many GLASOD based, primarily soil erosion) | Global | Productivity losses since WWII of about 0.3 percent per year for cropland and 0.1–0.2 percent for pasture. |
| Crosson (2003) | Re-analysis of GLASOD and Dregne and Chou (1992) | Global | Cumulative loss of 5 percent of productivity on 4.7 billion ha of cropland and permanent pasture in 1945–1990 period; average annual rate of loss of 0.1 percent |

The regional differences in crop response to erosion are, however, major. There are great disparities in the sensitivity of soils to erosion – soils with growth-limiting sub-soil layers (e.g. shallow soils over bedrock, soils with high sodium and/or dense B horizons) are inherently more susceptible to yield reductions due to soil loss (Bakker *et al.*, 2007). In a study modelling the impact of erosion in Europe over the next century, Bakker *et al.* (2007) predicted yield reductions on the order of 6 to 12 percent in southern Europe and reductions of only 0 to 1 percent in much of northern Europe. The overall impact on European food production is, however, relatively small as the yields from southern Europe are lower to begin with. In addition, increases in climatic extremes associated with human-induced climate change may lead to enhanced levels of wind and water erosion, but the impact of these changes will differ greatly among regions.

Finally, the crop yield/soil erosion relationship may be a less critical reason to reduce soil erosion than the off-site impacts of erosion, especially the transport of agricultural inputs such as N and P to waterways (Steffen *et al.*, 2015).



7.2.2 | Soil sealing

Soil sealing is most commonly associated with the expansion of urban areas and leads to a permanent, non-reversible loss of agricultural land. Yields are eliminated, not just reduced and the soil, if completely sealed, becomes effectively non-soil. Urbanization of agricultural land should thus be considered as a threat to future food production, not only for the loss of good quality agricultural land but also because of the risk of soil pollution through waste disposal and acid deposition from urban air pollution (Chen, 2007; Hubacek *et al.*, 2009; Clavero, Villero and Brotons, 2011). Blum and Nortcliff (2013) provide a very rough estimate of daily losses of soil due to sealing at the global scale of 250–300 km², and suggest this rate could increase due to continuing migration of rural dwellers to urban areas. Thus, new policies that favour sustainable rural development, oriented to avoid rural-urban migration as well as to support the return to rural areas of people living in the cities, could avoid soil degradation and promote food security.

7.2.3 | Soil contamination

Soil contamination reduces food security both by reducing yields of crops due to toxic levels of contaminants and by causing the crops that are produced to be unsafe to consume. As summarized in Chapter 6 (Section 6.3), there are worldwide tens of thousands of known contaminated sites due to local or point-source contamination. In regions with a long-standing industrial base, the expansion of contamination is limited, but in countries undergoing rapid industrialization or resource development the potential for the further spread of contamination is great. The tremendous expansion of industry in China is one example of this: 20 million ha of China's farmland (approximately one fifth of China's total farmland) is estimated to be contaminated by heavy metals, and this may lead to a significant reduction in food availability (see also Section 6.3 above). Contamination is also severe due to point sources such as Cs pollution from the Fukushima Dai-ichi nuclear power plant and the Chernobyl disaster of 1983. Diffuse soil contamination occurs in many regions (Blum and Nortcliff, 2013), but is more commonly linked with concerns about food safety rather than significant decreases in crop yields.

7.2.4 | Acidification

Acidification of agricultural soils is primarily associated with the net removal of base cations (e.g. product removal without replacement with ameliorants such as lime) and the direct addition of acidifying inputs (e.g. ammonium-based N fertilizer) to inherently low-pH soils, which have a low capacity to buffer added acidity. It is most prevalent on ancient, highly weathered soils. Acidification is a significant regional threat in countries such as Australia and Vietnam (see Chapters 10 and 15). Liming is an effective response to control acidity of surface horizons, but rates of lime addition lag behind required levels even in developed countries like Australia (SOE, 2011) and continuing loss of yield occurs.

7.2.5 | Salinization

Salinization in a soil progressively reduces crop yields; beyond a certain crop-specific threshold, growth of a given crop may fail entirely. The regional summaries in Chapters 9 to 16 illustrate how difficult it often is to separate the causes of salinization: whether the saline soils are naturally occurring (primary salinization) or the salinization has been caused by inappropriate management, which is often the case with poorly executed irrigation programmes (secondary salinization). Estimates from the 1990s place the land area affected by primary salinization at approximately one billion ha, and the area of land with secondary salinization at 77 million ha (Ghassemi, Jakeman and Nix, 1995).

Salinization is typically associated with arid and semi-arid areas, and may be exacerbated by climate change (see also Section 6.5). An increase in irrigated land is commonly suggested as a means to increase food production, but poorly designed and implemented irrigation schemes can readily cause an increase in



salinization. Safe design and operation of irrigation systems requires a high level of managerial expertise. Irrigation expansion can contribute to increases in food production but great care is needed in planning and design to avoid negative effects such as salinization.

7.2.6 | Compaction

Compaction impairs soil functions by impeding root penetration and limiting water and gas exchange. In soils where it occurs, it can reduce crop yields but it rarely eliminates plant growth entirely. The susceptibility of different crops to compaction differs greatly (see also Section 6.9 above). Good soil management requires care in minimizing soil compaction and the adoption of management practices which alleviate existing compaction. The effect of soil compaction on output and hence on food security is, however, difficult to assess, especially in tropical areas (Lal, 2003).

7.2.7 | Nutrient imbalance

The problems associated with under-supply of nutrients in regions such as Sub-Saharan Africa will be discussed in Chapter 8 in the context of closing the yield gap. Foley *et al.* (2011) and Steffen *et al.* (2015) clearly indicates the regions where over-supply of nutrients is occurring: mid-west United States, western Europe, northern India, and the coastal areas of China. Foley *et al.* (2011) emphasize the need to address the economic and environmental issues in nutrient over-supply by increasing the efficiency of nutrient uptake by plants. This, coupled with reductions in transport of nutrients to waterways by minimizing erosion, would substantially reduce eutrophication. It would also allow the redistribution of N and P to areas of nutrient-poor soils without exceeding the planetary boundaries for the elements (Steffen *et al.*, 2015).

7.2.8 | Changes to soil organic carbon and soil biodiversity

Soil organic carbon (SOC) and soil biodiversity are commonly linked to three dimensions of food security: increases in food availability, restoration of productivity in degraded soils, and the resilience of food production systems.

Soil C is not itself a direct control on food production but is a proxy for soil organic matter (SOM), which is one of the key attributes associated with many soil functions. Soil microbial C is normally included in aggregate measures of SOC, and soil microbes are a component of the soil organic matter; hence in terms of mass, SOC/SOM and soil microorganisms are directly related. The focus on SOC, rather than SOM, occurs because of the ease of measurement of C as a proxy for SOM, and because of the direct connection between SOC and atmospheric C.

The roles of SOC and soil biodiversity in increasing food availability are also inextricably bound together. Increases in SOC and in soil biodiversity are believed to be beneficial for crop production, and decreases in both are equally believed to be deleterious for crops; however providing evidence for these qualitative statements and establishing predictive relationships has been difficult (Naeem *et al.*, 2009; Bommarco, Kleijn and Potts, 2013; Palm *et al.*, 2014).

The more readily understood relationship between soil C storage and atmospheric C levels has driven much of the work in the past 15 years on soil carbon dynamics, but the secondary benefit of increasing SOC levels for crop production is commonly cited, if rarely quantified. Efforts to determine a threshold SOC value for maximum crop production in temperate soils have not been successful as it depends on management and on other factors such as soil limitations and precipitation (Loveland and Webb, 2003). Lal (2006) estimates yield gains associated with a 1 Mg ha⁻¹ gain in SOC in the tropics and sub-tropics ranging from 20-70 kg ha⁻¹ yr⁻¹ for wheat to 30-300 kg ha⁻¹ yr⁻¹ for maize. However, the study acknowledges that the data are meagre and that functional relationships between SOC pool and crop yield are not available, especially for degraded soils in the tropics and subtropics.



Research in tropical and semi-tropical lands has established that inputs of organic material through the return of residues and manure to the soil are essential for fertility restoration in degraded soils, but that low residue production and competing uses for residues and manure limit the adoption of these SOC-aggrading approaches (e.g. Lal, 2006; Rusinamhodzi *et al.*, 2013; Palm *et al.* 2014). Sustainable soil management that increases SOM levels will assist in maintaining productivity, but the specific measures taken to increase SOM must be locally developed.

Establishing a direct, quantitative link between soil biodiversity and increasing food production is even more elusive. Sylvain and Wall (2011) observe that “the total invertebrates found in a soil will interact to provide many services and participate in several ecosystem functions, but it is unlikely that a single species will influence all services and functions that influence plant growth or composition at the same time or in the same manner”. Biodiversity beyond the soil plays an important role in regulating services such as biological pest control and crop pollination (Bommarco, Kleijn and Potts, 2013), and public concerns about the effects of pesticides on key species continues to grow.

A final role for SOC enhancement and maintenance of soil biodiversity is to increase the resilience of the soil for food production, especially its ability to withstand disruption due to human-induced climate change. SOC buffers the impact of climate extremes on soils and crops by: (I) regulating water supply by reducing runoff and increasing soil-water holding capacity; (II) reducing erosion through runoff reductions and improved aggregation; and (III) providing sites for nutrient retention and release (Loveland and Webb, 2003; Lal, 2006). The combined role of soil organic matter and biodiversity in nutrient cycling ensures a continuing supply of nutrients for crop growth. It is difficult to quantify this relationship, especially in the light of the uncertainties associated with human-induced climate change, but the existing qualitative understanding is sufficient to establish the importance of SOC and biodiversity in sustainable soil management.

Summary

The importance of soil degradation and soil rehabilitation are highlighted in principles eight and nine of the proposed World Soil Charter:

Soil degradation inherently reduces or eliminates soil functions and their ability to support ecosystem services essential for human well-being. Minimizing or eliminating significant soil degradation is essential to maintain the services provided by all soils and is substantially more cost-effective than rehabilitating soils after degradation has occurred.

Soils that have experienced degradation can, in some cases, have their core functions and their contributions to ecosystem services restored through the application of appropriate rehabilitation techniques. This increases the area available for the provision of services without necessitating land use conversion.

Our ability to predict the effect of soil degradation on food security is very limited for two main reasons. First, there is a lack of up-to-date knowledge both on the area affected by degradation and on the linkages between degradation and soil functions (and ultimately plant production). The research community continues to cite research summaries on the effects of soil degradation on crop yields from the 1990s based on data gathered in the 1980s. Yet crop production in many regions has undergone profound change since the 1980s – for example, the widespread adoption of conservation tillage in many regions occurred during the 1990s and 2000s. There is a pressing need for meta-analyses on all of the soil threats discussed here. This in-depth review of existing work needs to be complemented by new research to address major information gaps, and in particular to prove more conclusively the functional relationships between soil attributes and plant production.



The second limitation to predictions is that farmers are not simply passive observers of inexorable degradation processes – farmers in all regions, including the tropics, are willing to invest in the future to protect soils and the essential services that they provide (Stocking, 2003). For example, the adoption of conservation tillage in heavily mechanized systems such as those in North America has substantially lowered erosion rates (Montgomery, 2007). The general applicability of conservation tillage in other regions may be limited (Palm *et al.*, 2014) but the principle that farmers are active participants in soil change is essential to recognize and encourage.

7.3 | Soil change and climate regulation

Soils play a fundamental role in the maintenance of a climate favourable to life. A range of soil processes helps regulate climate, including the thermal and moisture balance, greenhouse gases (H₂O, CO₂, CH₄ and N₂O) and particulates in the atmosphere. Soils can also adversely impact the maintenance of air quality.

7.3.1 | Soil carbon

Although it is hard to estimate quantities, it is certain that soils contain vast reserves of carbon. Recent estimates range between 1200 and 3000 Pg C depending on the depth to which estimates extend, and on the way in which wetland soils are counted (Hiederer and Köchy, 2012). Roughly 1670 Pg of C is stored in peatlands and permafrost soils in high northern latitudes (Tarnocai *et al.*, 2008). Hence soil organic matter is a large pool. Consequently, only small changes in soil C storage can have a large effect on atmospheric CO₂. Soils also contain approx. 950 PgC in the form of pedogenic carbonates to 2 m depth (Batjes, 1996).

Carbon respired from soils and derived from decomposition of organic matter in soils approximately balances annual net primary production of carbon by biomass. Carbon dioxide derived from plant roots and their symbionts below ground adds to the total flux of CO₂ from soils to the atmosphere, which in total is ~10 times larger than the current release of CO₂ to the atmosphere by fossil fuel burning (Schimel, 1995). Hence relatively small changes in the cycling of soil C can lead to large changes in atmospheric CO₂.

Management that changes C inputs or tillage that alters the stability of soil organic matter through changes in soil aeration or structure measurably alter soil C storage. Historically, the expansion of agriculture has led to losses of soil C to the atmosphere, estimated globally to be of order 40-90 PgC, some of which has remained in the atmosphere (Smith, 2004, 2012).

In terms of climate change, most projections suggest soil carbon changes driven by future climate change will range from small losses to moderate gains, but these global trends show considerable regional variation (Smith, 2012). The response of soil C in future will be determined by two factors: (I) the impacts of increased temperature and altered soil moisture on decomposition rates; and (II) the balance between increases in C losses resulting from accelerated decomposition and predicted C gains through enhanced productivity under elevated CO₂ and nutrient deposition (Smith, 2012).

Soil organic matter (SOM) is considered dynamic and has importance beyond its climate role. Plant residues added to soils provide energy for a cascade of heterotrophic organisms. A key outcome of organic matter (OM) breakdown is the release of essential nutrients into the soil. If the breakdown of OM exceeds the supply of O₂, e.g. under high moisture conditions, the degradation of OM using other electron acceptors drives the production and consumption of other important greenhouse gases such as methane and nitrous oxide. The degradation of OM also indirectly affects greenhouse gases like troposphere ozone by altering the emission of reactive trace gases. In addition to climate effects through regulation of greenhouse gases, SOM determines properties such as nutrient retention, water retention, and the structure and size of the microbial community in soils.



SOM feedbacks to climate change (Figure 7.6) include direct responses such as: (I) the alteration of microbial activity with temperature (Conant *et al.*, 2011); and (II) moisture-related changes in the supply of O₂ relative to other electron acceptors that reflect precipitation change. In this context, probably the biggest concern is the thawing of large stores of C in permafrost at high northern latitudes, which will make organic C that has been frozen for millennia available for decomposition. This response is predicted to create a significant positive feedback to climate change (Schuur *et al.*, 2008).

Another direct response of soil organic carbon pools is predicted in response to elevated CO₂ and its effect on ecosystem productivity. Free Air Carbon Dioxide Enrichment (FACE) studies have shown that belowground productivity can be strongly affected, with cascading and mixed consequences for SOM storage. However indirect effects are such as altered stabilization of older C associated with the increased inputs of fresh plant inputs ('priming') add uncertainty to the prediction of future soil C responses. As with CO₂ fertilization, increased deposition of reactive N associated with regional air pollution affects production, quality and spatial distribution of plant inputs (e.g. above- versus belowground) and can alter the decomposition rates through changes in the soil microbial community (Berg and Matzner, 1997). Hence the net effects on soil C storage are difficult to predict, though the combined effects of climate change and fertilization are expected to result in net losses of soil C overall from temperate forest soils (Hopkins, Torn and Trumbore, 2012)

Many of the processes affecting SOM over the past century have been dominated by human management of vegetation, which in turn affects the inputs and status of SOM. Changes in vegetation cover, including those occurring in response to climate as well as to land use or management, influence soil organic matter by altering the rates, quality and location of plant litter inputs to soils. In turn, litter inputs influence the amount and composition of the decomposer organisms, including soil fauna, as well as the soil microbial community. Studies of a number of vegetation transitions – for example the replacement of forests with agriculture or pasture – have shown that these transitions have led to a loss of soil C to the atmosphere. However, the trajectory of vegetation change in response to climate, and the consequences for atmospheric CO₂, are not well known, as soils will in turn determine what kind of vegetation will take over. For example, C from thawing permafrost soils may eventually be sequestered in the biomass of forests that can grow in the warmer climate. Evidently the time lags required for these transitions are an important part of understanding the net effect of soil C on the carbon cycle.

In addition to direct effects of changes in plant litter addition to soils, management or vegetation change also alters the chemical and physical framework of soil and thereby the organisms inhabiting it. For example, ploughing can break up soil aggregates and make organic matter that was previously protected available to decomposers. Changes in evapotranspiration can change local and regional water resources. Addition of fertilizers increases plant productivity but also alters soil microbial communities and can stimulate production of reactive N gases and N₂O.

Large-scale soil erosion is thought to slow decomposition of buried, eroded organic matter, while growth of vegetation on the remaining soil will tend to increase soil C storage. However, these effects have been shown to be relatively small (Van Oost *et al.*, 2007). By removing topsoil that is generally high in organic matter, erosion can have profound effects on physical and chemical soil properties such as water retention and cation exchange capacity.

Global increases in carbon stocks have a large, cost-competitive potential for climate change mitigation (Smith *et al.*, 2008). Mechanisms include reduced soil disturbance, improved rotations and residue/organic input management, and restoration of degraded soils. Nevertheless, limitations on soil C sequestration include time limitation, non-permanence, displacement and difficulties in verification (Smith, 2012). Despite these limitations, soil C sequestration can be useful to meet short- to medium-term targets. In addition, soil C sequestration confers a number of co-benefits on soils. It is thus a viable option for reducing the atmospheric



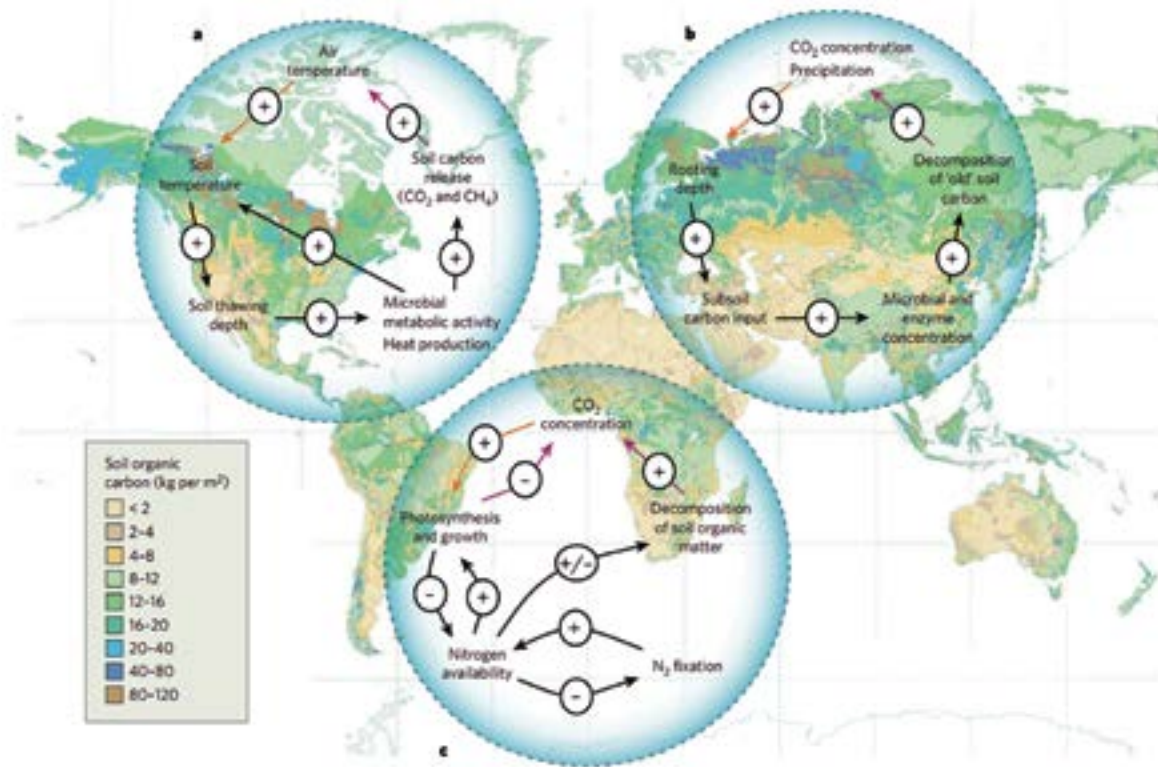


Figure 7.6 | Some soil-related feedbacks to global climate change to illustrate the complexity and potential number of response pathways. Source: Heimann and Reichstein, 2008.

CO₂ concentration in the shorter term, buying time to develop longer term emission reduction solutions across all sectors of the global economy (Smith, 2012).

Just as reductions in soil C stocks are associated with negative consequences for soil function, increased soil carbon stocks are associated with increased soil fertility, workability, water holding capacity, reductions in greenhouse gas emissions and reduced erosion risk (Lal, 2004). Increasing soil carbon stocks can thus reduce the vulnerability of managed soils to future global warming (Smith and Olesen, 2010). Management practices effective in increasing SOC stocks include: (I) improved plant productivity through nutrient management, rotations and improved farming practices; (II) reduced or conservation tillage and residue management; (III) more effective use of organic amendments; (IV) land use change, for example from crops to grass or trees; (V) set-aside; (VI) agroforestry; (VII) optimizing livestock densities; and (VIII) planting legumes or improving the crop mix (Smith *et al.*, 2008). While these measures have the technical potential to increase SOC stocks by about 1 – 1.3 Pg C yr⁻¹ (Smith *et al.*, 2007a, 2008), they are dependent on economics: the economic potential for SOC sequestration was estimated to be 0.4, 0.6 and 0.7 Pg C yr⁻¹ at carbon prices of up to US\$20, \$50 and \$100 per tonnes CO₂-eq. yr⁻¹, respectively (Smith *et al.*, 2008). In addition, the size of the potential sequestration is relatively small in comparison to the threats: only a small loss of C from permafrost or peatlands could offset this potential sequestration (Joosten *et al.*, 2014). However, an increase in SOC through improved management is expected to also reduce vulnerability of the soils to future SOC loss under global warming. As such, soil carbon sequestration can, in many respects, be regarded as a ‘win-win’ and a ‘no regrets’ option (Smith *et al.*, 2007b).

7.3.2 | Nitrous oxide emissions

Soils emit nitrous oxide (N₂O), a greenhouse gas that is around 300 times more potent for radiative forcing (climate warming) over 100 years than CO₂. Of the approximately 16 Tg N₂O-N yr⁻¹ emitted globally in the 1990s, between 40 and 50 percent was a result of human activities (Reay *et al.*, 2012). The main sources were

agriculture, industry, biomass burning and indirect emissions from reactive nitrogen, such as leaching, runoff and atmospheric deposition (Reay *et al.*, 2012). Of these sources, agricultural soils are the dominant source, contributing over 80 percent of global anthropogenic N₂O emissions during the 1990s (Smith *et al.*, 2007a). N₂O emissions from agricultural soils have increased from just under 4 Tg N₂O-N yr⁻¹ in 1990, to over 4 Tg N₂O-N yr⁻¹ in 2010. Emissions are projected to increase to over 5 Tg N₂O-N yr⁻¹ by 2030 (Reay *et al.*, 2012).

Nitrous oxide is emitted from soils through two processes, nitrification and denitrification. Any mineral N available in the soil is subject to loss through one of these processes. The processes depend on soil environmental conditions such as the availability of mineral N, soil temperature and soil water content, soil pH, organic matter content and soil type. Nitrification tends to be favoured under aerobic conditions and denitrification under anaerobic conditions (Galloway *et al.*, 2003). Subject to mineral N being available, any soil can emit N₂O through mineralisation of soil organic matter. However, the majority of emissions are driven by sources of N added to the soil as fertiliser, either as synthetic fertilizer, or as organic amendments (e.g. manures, slurries, composts). So close is the relationship between N addition and emission, that N₂O emissions are often calculated as a direct function of N added to the soil (Reay *et al.*, 2012). Emissions of N₂O from agricultural soils driven by addition of synthetic fertilizers have increased from 67 MtCO₂-eq. yr⁻¹ in 1961, to 683 MtCO₂-eq. yr⁻¹ in 2010 (Tubiello *et al.*, 2013).

Given the close association between N inputs and N₂O emissions, soil management strategies to reduce N₂O emissions, and thereby improve this aspect of their climate regulation function, are mostly centred on removing surplus N in the soil. This is mainly accomplished by improving N-use efficiency to reduce the N surplus, either by reducing inputs or by better matching applications (timing and amount) to plant demand (Snyder *et al.*, 2014). In a recent review, Snyder *et al.* (2014) noted that soil N₂O emissions can be reduced by selecting the right source, rate, time and place of N application and that new technologies and greater farmer/adviser skills can improve N input management. They estimate that crop N recovery could be increased by >20 percent, reducing risks of N₂O emissions by >20–30 percent (Snyder *et al.*, 2014).

Beyond these technical measures, N₂O emissions could also be reduced through demand-side management, for example through reduced food waste. Another demand-side measure could be to encourage dietary change away from less efficiently produced food products such as meat and other livestock products, or foods with very high energy inputs, such as heated glasshouses during winter (Reay *et al.*, 2012).

In summary, managed soils can play a key role in climate regulation via N₂O emissions, and a number of options exist to improve the soil's delivery of its climate regulation service both by enhanced N management and by wider systemic changes in agriculture (Flynn and Smith, 2010; Reay *et al.*, 2012; Snyder *et al.*, 2014).

7.3.3 | Methane emissions

Methane (CH₄) is a greenhouse gas that is around 20–35 times more potent for radiative forcing (climate warming) over 100 years than CO₂. Soils often emit methane through methanogenesis when decomposition of organic matter occurs in anaerobic soil layers. Methane is also oxidised by methanotrophy in aerobic layers, so the emission is a balance between methanogenesis and methanotrophy (Le Mer and Roger, 2001). About 30 percent of total global CH₄ emissions are natural (including the natural wetland flux), and around 70 percent anthropogenic (Le Mer and Roger, 2001). Given that methanogenesis occurs under anaerobic conditions, waterlogged soils, particularly wetlands, peatlands and rice paddies, are the largest source of methane emissions (Le Mer and Roger, 2001). Since much of the methane flux from wetland and peatland

soils occurs on largely unmanaged areas, the emissions are not considered anthropogenic, so are not routinely included in greenhouse gas inventories. This means that the quantification of soil methane emissions over time from peatlands and wetlands is not as well documented as for N₂O. Nonetheless, some global estimates of CH₄ emissions from wetlands do exist: in 1998, total global emissions of CH₄ from wetlands were estimated to be 145 Tg yr⁻¹, of which 92 Tg yr⁻¹ came from natural wetlands and 53 Tg yr⁻¹ from rice paddies (Cao, Gregson and Marshall, 1998), with some estimates a little higher (Le Mer and Roger, 2001). Emissions from rice paddies, however, are included in inventories: CH₄ emissions from rice paddies were estimated to have increased from 366 MtCO₂-eq. yr⁻¹ in 1961 to 499 MtCO₂-eq. yr⁻¹ in 2010 (Tubiello *et al.*, 2013).

By contrast, aerobic soils tend to act as sinks for CH₄, thereby having a positive impact on climate regulation. Temperate and tropical aerobic soils that are exposed to atmospheric concentrations of CH₄ usually exhibit low levels of atmospheric CH₄ oxidation but, since they cover large areas, they are estimated to consume ~10 percent of the atmospheric CH₄ (Le Mer and Roger, 2001). Forest soils are the strongest CH₄ sink, followed by grasslands, with the sink capacity of cultivated land much lower than that of undisturbed soils (Steudler *et al.*, 1996; Priemé *et al.*, 1997). Atmospheric CH₄ oxidation also occurs in extreme environments such as deserts and glaciers, in the floodwater of submerged soils and in river waters (Le Mer and Roger, 2001). Potter, Davidson and Verchot (1996) estimated global soil CH₄ consumption to be 17–23 Tg yr⁻¹.

Soil management strategies to reduce CH₄ emissions or enhance CH₄ uptake can improve this aspect of the soil's climate regulation function. Enhancing uptake in managed soils is difficult, so most mitigation options occur for CH₄ emission reduction, and since wetlands are often unmanaged, most mitigation options have been developed for rice paddies. These include draining the wetland rice once or several times during the growing season, selection of rice cultivars with low exudation rates, off-rice season water management, fertilizer management and the timing and composting of organic residue additions (Smith *et al.*, 2008). For managed peatlands and wetlands (e.g. those used for forestry or agriculture), methane emissions can be reduced by fertilizer, water and tillage management (Le Mer and Roger, 2001). Rewetting of drained or cultivated peatlands to restore wetland function and maintain carbon stocks is likely to increase CH₄ emissions, but the overall impact on climate will vary between systems and depending on the time horizon considered (Joosten *et al.*, 2014).

7.3.4 | Heat and moisture transfer

Soils play an essential role in storage of water. Soil moisture strongly affects water, energy and carbon exchanges, leading to major forcings and feedbacks within the climate system (Seneviratne *et al.*, 2010). Soil moisture generally refers to the amount of water stored in the unsaturated soil zone. The most important soil moisture storage is that affecting plant transpiration, e.g. the water available within the root zone. Land evapotranspiration is an essential component of the continental water cycle, since it returns as much as 60 percent of precipitated land water back to the atmosphere (e.g. Dirmeyer *et al.*, 2006; Oki and Kanae, 2006; van der Ent *et al.*, 2010). Soil moisture is the main water source for this process, through plant transpiration and bare soil evaporation. Plant transpiration contributes about 60 percent of all land evapotranspiration (Schlesinger and Jasechko, 2014).

Evapotranspiration is itself a function of soil moisture (Koster *et al.*, 2004; Seneviratne *et al.*, 2010). This dependency is conceptually illustrated in Figure 7.7, which builds upon the classical Budyko framework (Budyko, 1956, 1974). It shows that three main soil moisture regimes can be distinguished: (I) a wet soil moisture regime in which evapotranspiration is solely limited by the availability of energy; (II) a transitional soil moisture regime in which evapotranspiration is strongly sensitive to the availability of soil moisture; and (III) a dry soil moisture regime in which soil moisture is at or below the wilting point and for which evapotranspiration is negligible.

The geographical distribution of these soil moisture and evapotranspiration regimes can be estimated with various methods, as discussed in Seneviratne *et al.* (2010). As an illustration, Figure 7.8 displays the correlation of annual mean evapotranspiration with radiation and precipitation in an observation-driven land surface model using a two-dimensional colour map. This analysis illustrates the existence of distinct evapotranspiration regimes, with most regions clearly displaying either the characteristics of a soil moisture- or energy-limited evapotranspiration regime.

One should note that the relationship displayed in Figure 7.7 is qualitative, and is affected (both in space and time) by variations in soil parameters, land cover characteristics, and other factors (e.g. Teuling *et al.*, 2010; Koster and Mahanama, 2012; Guillod *et al.*, 2013).

The water and energy balances of land are tightly connected through the process of evapotranspiration. It follows that the soil moisture effects on evapotranspiration (illustrated in Figure 7.8) are also highly relevant for land energy exchanges at the land surface. This link makes soil moisture a strong control of temperature variability and temperature extremes on land (e.g. Seneviratne *et al.*, 2006; Fischer *et al.*, 2007; Vautard *et al.*, 2007; Mueller and Seneviratne, 2012). Modelling estimates suggest that soil moisture feedbacks affect about 60 percent of temperature variability in the present Mediterranean climate in summer (Seneviratne *et al.*, 2006) and that they induced additional temperature anomalies of the order of 2°C in Central Europe during the 2003 European summer heat wave (Fischer *et al.*, 2007). Observation-based analyses also confirm the existence of strong correlations between the occurrence of hot extremes in regional hottest months and prior precipitation deficits in regions with soil moisture-limited evapotranspiration regimes (Hirschi *et al.*, 2011; Quesada *et al.*, 2012; Mueller and Seneviratne, 2012). The example of the European summer heat wave shows, moreover, that these feedbacks can be relevant in extreme years even in regions like Central Europe which have a dominant energy-limited evapotranspiration regime under the present climate.

For present climate conditions, the relationship between soil moisture deficits and hot extremes implies that information on soil moisture deficits could be used for improved forecasting of temperature mean and

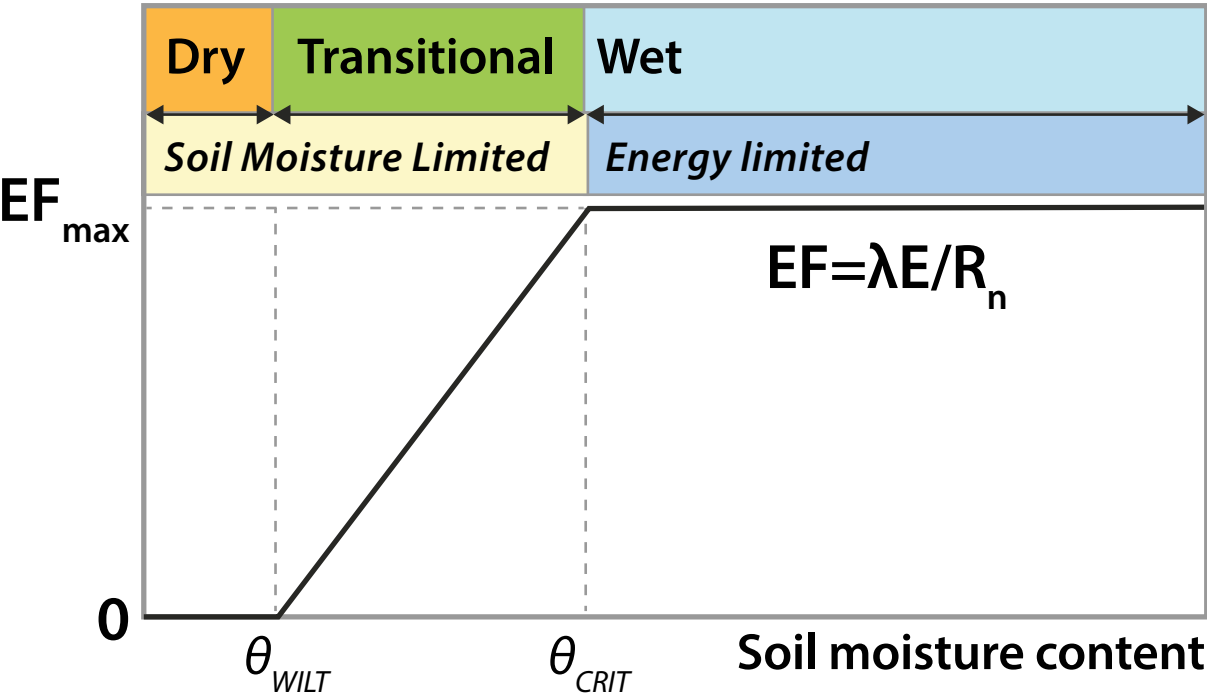


Figure 7.7 | Definition of soil moisture regimes and corresponding evapotranspiration regimes.

Source: Seneviratne *et al.*, 2010.

EF denotes the evaporative fraction, and EF_{max} its maximal value.



extremes several weeks in advance (e.g. Koster *et al.*, 2010a; Mueller and Seneviratne, 2010). Such early soil moisture information can be either provided by an offline land surface model driven with observation-based forcing (e.g. Dirmeyer *et al.*, 2006), by remote sensing products (e.g. Wagner *et al.*, 2007; De Jeu *et al.*, 2008), or by the assimilation of remote sensing products in land surface models (e.g. Reichle, 2008). However, the scarcity of precipitation and soil moisture observations still limits the derivation of reliable soil moisture estimates and the evaluation of satellite approaches on most continents (e.g. Koster *et al.*, 2010b; Dorigo *et al.*, 2013).

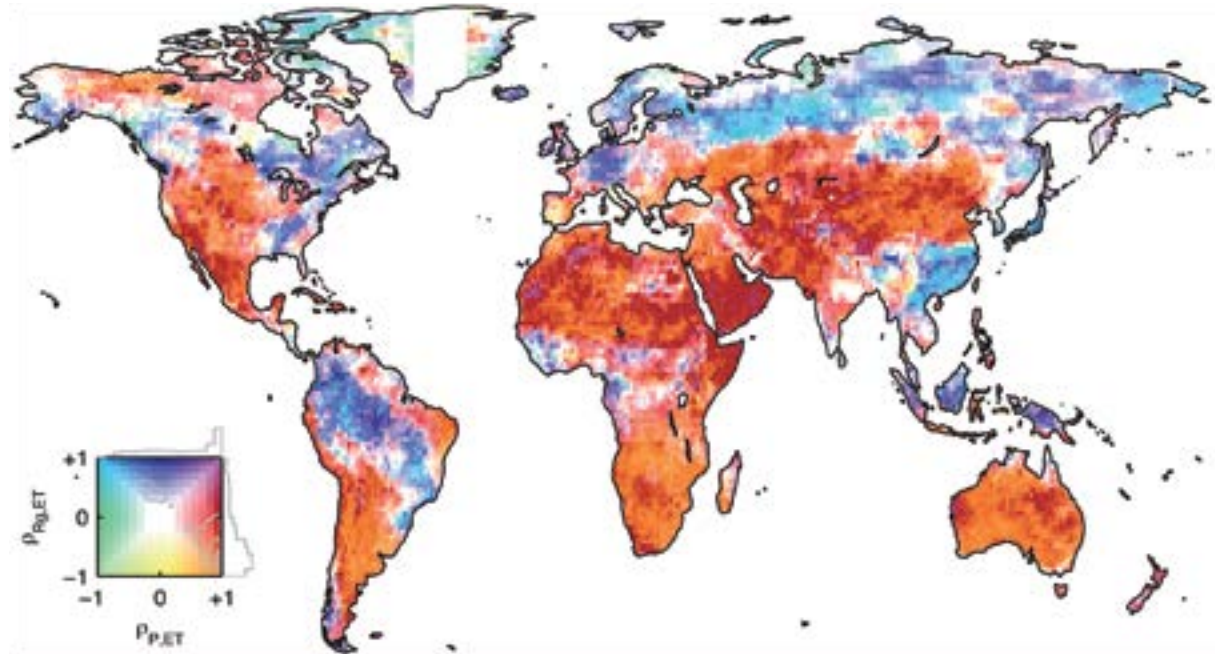


Figure 7.8 | Estimation of evapotranspiration drivers (moisture and radiation) based on observation-driven land surface model simulation.

Source: Seneviratne *et al.*, 2010.

The figure displays yearly correlations of evapotranspiration with global radiation R_g and precipitation P in simulations from the 2nd phase of the Global Soil Wetness Project (GSWP, Dirmeyer *et al.*, 2006) using a two-dimensional color map, based on Teuling *et al.* 2009, redrawn for the whole globe. (Seneviratne *et al.*, 2010)

Climate models project that several regions will be affected by more frequent drought conditions in the future as a consequence of enhanced greenhouse gas concentrations (e.g. Wang, 2005; Sheffield and Wood, 2007; Seneviratne *et al.*, 2012). This implies shifts in climate and soil moisture regimes, with important impacts on temperature projections (e.g. Seneviratne *et al.*, 2006; Dirmeyer *et al.*, 2012), in particular for temperature extremes (Seneviratne *et al.*, 2013).

Another feedback of soil moisture on climate is the possible impact of droughts on plant carbon uptake and a resulting decreased sink for CO_2 emissions (Ciais *et al.*, 2005; Friedlingstein *et al.*, 2006; Sitch *et al.*, 2008; Reichstein *et al.*, 2013). One particularly important region for this feedback is the Amazon rainforest, which is projected in some models to dry substantially (e.g. Mahli *et al.*, 2008). However, these projections are associated with high uncertainty in current climate models (Orlowsky and Seneviratne, 2013), and the resulting effects on carbon uptake could also be affected by the representation of plant physiology in the land surface schemes (Huntingford *et al.*, 2012).

Finally, the combined effects of soil moisture on near-surface humidity and temperature are also relevant for boundary layer development and precipitation occurrence (e.g. Betts, 2004; Koster *et al.*, 2004; Taylor *et al.*, 2012). More details on these feedbacks are provided in Sections 7.5 and 7.6 below.

7.4 | Air quality regulation

According to the World Health Organisation¹, air pollution is “contamination of the indoor or outdoor environment by any chemical, physical or biological agent that modifies the natural characteristics of the atmosphere”. The status of air pollution is often referred to as *air quality* (Monks *et al.*, 2009). *Air quality* affects human health through exposure to toxic inorganic compounds (e.g. HBr, elemental Hg vapour), toxic organic compounds (e.g. organic pesticides), and particulate matter (PM). Air quality also affects the climate system through changes in greenhouse gas concentrations (CO₂, CH₄, N₂O) – as discussed in Section 7.3 – and through aerosols (e.g. mineral particles, black carbon or ‘BC’). After deposition of atmospheric pollutants (e.g. N and S compounds, or compounds containing trace elements) on land or water, acidification, eutrophication, and contamination might occur (see Section 4.4), which can have harmful effects on ecosystem function and structure, particularly where deposition exceeds the ‘critical load’ that a particular soil can buffer (Nilsson and Grennfelt, 1988). Specific compounds in the atmosphere, such as ammonia (NH₃), can result in a host of environmental problems (e.g. impacts on human health, odour, climate change, soil acidification, eutrophication, biodiversity). The magnitude of the problems would depend on interactions with other compounds (Aneja, Schlesinger and Erisman, 2009).

7.4.2 | Ammonia emissions

Agriculture accounts for 80–99 percent of all NH₃ emissions (FAO, 2014). In Europe, agriculture accounts for 94 percent (EEA, 2012). These emissions mainly come from animal manure and fertiliser application (Olivier *et al.*, 1998). In the United States, NH₃ reductions are voluntary and there are neither federal nor national regulations controlling its emission (Aneja, Schlesinger and Erisman, 2009; Greaver *et al.*, 2012). In Europe, however, NH₃ emissions have been an important policy issue (van der Hoek, 1998) and regulation has led to an overall reduction in NH₃ emissions. Between 1990 and 2010, NH₃ emissions decreased in the EU-27 by 28 percent (EEA, 2012), with especially large reductions in Poland, the Netherlands and Germany. Ammonia emission reductions have been associated with a reduction in the number of livestock (especially cattle), improvement of manure management, and the lower input of nitrogenous fertilisers to soils (EEA, 2011, 2012). The effectiveness of manure injection to decrease emissions is under debate, as a result of its effect on pollutant swapping, as there may be a reduction in NH₃ but an increase in N₂O emissions and/or NO₃ leaching (Erisman *et al.*, 2008). A better understanding is needed on the contribution of NH₃ as a precursor of PM concentrations, both emissions of primary PM₁₀ (particulate matter with a size < 10 µm) and secondary formation of PM_{2.5} (particulate matter with a size < 2.5 µm) (Aneja, Schlesinger and Erisman, 2009). It is worth mentioning that interactions occur with other compounds in the atmosphere, to the extent that reductions in SO₂ and NO_x are only effective in the reduction of PM_{2.5} if carried out simultaneously with NH₃ reductions (Erisman and Schaap, 2004).

7.4.3 | Aerosols

Mineral dust, sulphate aerosols, and organic C and black C (BC) aerosols from fossil fuel and biomass burning have a significant effect on radiative forcing (Forster *et al.*, 2007). Mineral dust is mainly emitted from deep and extensive alluvial flood deposits emplaced during the Pleistocene, for example in the Sahara, East Asia, the Arabian deserts, and Central Australia (Prospero *et al.*, 2002). The largest sources are located in the Northern Hemisphere, in the so-called ‘global dust belt’ that extends from the west coast of North Africa, through the Middle East, into Central Asia. Outside this belt, areas with remarkable persistent dust activity include the Great Basin in south-western North America, the Lake Eyre Basin in Australia, some areas of South America (predominantly in Argentina), and southern Africa (Prospero *et al.*, 2002) (Figure 6.2). The ‘Red Dawn’ dust storm that affected Sydney, Australia in September 2009 is described in Chapter 15. Mineral dust originating in the Sahel has been reported to be regularly carried over large areas of the Atlantic and the Caribbean; the

1 http://www.who.int/topics/air_pollution/en/



largest export occurs during years of low rainfall in the source region (Prospero and Lamb, 2003). Although this process might have been exacerbated by anthropogenic activities (Prospero and Nees, 1978), recent evidence indicates that vegetation cover in the region has not changed substantially in the past 20 years and that, on a global scale, dust mobilisation is probably mostly driven by natural events (Prospero *et al.*, 2002).

The direct effect of aerosols on the climatic system is mainly through the reflection and absorption of solar radiation (Miller and Tegen, 1998). The indirect effect involves the modification of cloud properties (Kaufman, Tanre and Boucher, 2002). Greenhouse gases, in contrast, reduce the outgoing thermal radiation to space. Differences in lifetime and spatial distribution between greenhouse gases and aerosols are also considerable: greenhouse gases have a lifetime of more than 100 years and a homogeneous distribution (Forster *et al.*, 2007), whereas aerosols have a lifetime of about a week and a rather heterogeneous distribution (Andreae *et al.*, 1986). Soil dust aerosols have also been reported to modify the lifetime of some greenhouse gases (Dentener *et al.*, 1996). They also provide essential nutrients to ocean ecosystems that may increase the efficiency of the ocean's biological pump and help sequester CO₂ in the deep ocean (Martin, 1990). This is specially the case of iron, which is an important micronutrient for phytoplankton (Falkowski, Barber and Smetacek, 1998).

Most aerosols are highly reflective, thus raising the albedo of our planet and having a cooling effect. However, aerosols containing BC are dark and strongly absorb the incoming sunlight (Kaufman, Tanre and Boucher, 2002). This warms the atmosphere and cools the Earth's surface before a redistribution of the energy occurs in the atmosphere column (Ramanathan and Carmichael, 2008). Black C alters the radiative forcing through different processes: (I) the presence of BC in the atmosphere above surfaces with high albedo such as snow or clouds may cause a significant positive radiative forcing (Ramaswamy *et al.*, 2001); (II) BC aerosols deposited on snow may promote melting (Warren and Wiscombe, 1980; Hansen and Nazarenko, 2004); and (III) BC influences evaporation and cloud formation by modifying the atmosphere's vertical temperature gradient (Ackerman *et al.*, 2000; Raufman and Fraser, 1997). However, the exact radiative forcing depends on how BC is mixed with other aerosol constituents (Jacobson, 2001).

Carbonaceous aerosol emission inventories suggest that approximately 34-38 percent of these emissions come from biomass burning sources, the remainder from fossil fuel burning sources (Forster *et al.*, 2007). Fossil-fuel-dominated BC emissions are approximately 100 percent more efficient warming agents than biomass-burning-dominated plumes (Ramana *et al.*, 2010). The type of smoke is also largely influenced by the type of biomass being burned (Takemura *et al.*, 2002). In savannah ecosystems, about 85 percent of the biomass (mostly grasses) is consumed by flaming during fire events. In forest fires this value decreases to 50 percent or less, as the flaming stage is followed by a long, cooler smouldering stage in which the thicker wood, not completely consumed, emits smoke composed of organic particles without BC (Takemura *et al.*, 2002). Black C is thus mostly emitted during the hot, flaming stage of the fire (Kaufman *et al.*, 2002). The intense surface heating caused by fires can further cause a rapid uplift of heated air, known as pyro-convection, which can considerably disturb the chemical conditions in the free and upper troposphere and, in some cases, in the stratosphere (Monks *et al.*, 2009). Aerosols from fires are more likely to be injected at higher altitudes and are likely to experience long-range transport. Aerosol emissions from large boreal fires in Alaska and Russia have been shown to be transported very efficiently over long distances (Damoah *et al.*, 2006; Petzold *et al.*, 2007).

7.5 | Soil change and water quality regulation

Soils provide a biogeochemically activated filtration and cleaning service that transforms or retains materials deposited at the land surface. These materials include not only nitrogen and phosphorous, elements from grey

water used for irrigation, and acidic compounds, but also inorganic and organic toxins. If the capacity of the soil to retain, transform or filter these materials is exceeded, there can be severe environmental consequences for water quality. Soils also adversely impact the provision of clean water through erosion into water courses, through salinization and through redox cycling and the release of metals such as arsenic.

7.5.1 | Nitrogen and phosphorous retention and transformation

By increasing fertilizer production and crop N fixation, human activities have doubled nitrogen (N) fixation from the atmosphere during the last century. Half (210 Tg N yr⁻¹) of global nitrogen fixation (413 Tg N yr⁻¹) is human-driven (Fowler *et al.*, 2013). Mining and erosion have increased the phosphorus (P) flow from land into the ocean by at least ten-fold (preindustrial value of 1 to current estimate of 9-32 Tg P yr⁻¹; Carpenter and Bennett, 2011). A recent inventory indicates that approximately 60 percent of the nitrogen fixed by human activities is released back into the environment without being incorporated into food or products (Houlton *et al.*, 2013). Increases in the release of reactive nitrogen (N) and phosphorus (P) to the environment are associated with many significant environmental concerns, including surface water contamination, harmful algal blooms, hypoxia, air pollution, nitrogen saturation in forests, drinking water contamination, stratospheric ozone depletion and climate change (Bennett, Carpenter and Caraco, 2001; Sutton *et al.*, 2011; Davidson *et al.*, 2012).

Soils serve as an important regulator of the leakage of this anthropogenic N and P back into the air or to surface and ground water, since much of the release occurs from fertilizers or atmospheric deposition. Soil is the largest pool of N and P within terrestrial ecosystems (Cole and Rapp, 1981), illustrating the magnitude and stability of soil N and P storage. Review of ¹⁵N tracer studies reinforces that idea that soils are the strongest sink for nitrogen in the short and medium term (Fenn *et al.*, 1998; Templer *et al.*, 2012). Flows of N through the landscape and the consequences of excess N can be represented by the N cascade (Galloway *et al.*, 2003). Nitrogen and phosphorus removal occurs through plant or microbial uptake, storage in soil organic matter, by complexation, and sorption or exchange. Nitrogen is cycled biologically through plant uptake, litterfall and microbial cycling, and is stored in organic forms except in areas with substantial rock-derived N (Morford, Houlton and Dahlgren, 2011). By contrast, soil P is mainly found in an inorganic form, sorbed or complexed by soil minerals and the exchanger. Organic P is a smaller pool in most soils, found in a review of global soil P to range from 5-40 percent (Yang and Post, 2011). For N, there are also significant gaseous losses via NO_x or NH₃ and through denitrification as N₂ or N₂O. Storage in soils or perennial plants and conversion into other inert forms (N₂ for N or stable inorganic complexes for P) represent stable sinks that remove N and P from flowpaths and the N cascade for a period of time determined by the residence time of those sinks.

An important service provided by soils is to remove N and P along flowpaths, preventing mobile nitrate and phosphate from moving from terrestrial ecosystems into surface waters and groundwater. Global models indicate that soils are responsible for the largest portion of landscape N removal - 22 percent of global N removal as denitrification - second only to coastal ocean sediments (Seitzinger *et al.*, 2006). Riparian soils or wetlands can remove N that has leaked from forests, farms, rangelands or the built environment (Peterjohn and Correll, 1984), as long as riparian zones are downgradient of the N source (Weller and Baker, 2014). One study indicates that replacement of 10 percent of historical riparian buffers could substantially reduce N loading to the Gulf of Mexico (Mitsch *et al.*, 2001).

Phosphorus cycling has important distinctions from N cycling. In particular, the dominant inorganic form of phosphorus, orthophosphate, binds strongly to soil particles via sorption or complexation as inorganic P, in contrast to nitrate, which is quite mobile. Phosphorus can be displaced under reducing conditions, and thus efforts to target N removal may in fact cause unanticipated increases in dissolved P concentrations (Ardón *et al.*, 2010). While we do not have a parallel conceptual P cascade, P availability can drive the formation of harmful algal blooms, and recent work indicates that joint management of N and P is critical (Conley *et al.*, 2009). In efforts to reduce effects on ecosystems and water quality, it is important to consider the soil processes involved in removal of both elements and their interactions.



Perturbations that increase the mobility of N and P may saturate the retention capacity of soils such that the ability to remove these elements declines as inputs increase. Disturbances that affect soil structure, rooting patterns and organic matter also decrease N and P retention capacity. At the ecosystem scale, N removal capacity declines as N loads increase above a point where N can be taken up by plants and soil processes (Aber *et al.*, 1989). While studies illustrate that the rate of N removal does generally decline with increasing N inputs (e.g. Perakis, Compton and Hedin, 2005), there are still questions about the ability of soils to retain N over time. The saturation point may vary by ecosystem and soil type. For example, wetland ecosystems have a tremendous capacity to retain N – a recent meta-analysis indicates that wetland N removal is linear with N loading, removing about 47 percent of N inputs even at very high loads (Jordan, Stoffer and Nestlerode, 2011). However, recent work on agricultural soils found that N₂O production increases with increased N loading (Shcherbak, Millar and Robertson, 2014). This reinforces the pattern of decline in capacity of soils to serve as a stable N sink under high N inputs, and suggests that efforts to reduce N₂O production should target areas of high N loads where larger benefits will be seen per unit N.

The connection between ecosystem services and soil processes is sometimes distant. The benefit of N uptake in a riparian soil in Iowa might be most appreciated in distant coastal fisheries. In addition, ecosystem services do not turn on or off with the flick of a switch; for example, it may take decades to recover water quality after a widespread land use change (Hart, 2003; Howden *et al.*, 2010). Our perspectives about soils and ecosystem services should include these distant connections and time lags.

Removal of N from the cascade has implications for many aspects of human health and well-being (Figure 1; Brauman *et al.*, 2007; Compton *et al.*, 2011), and an increasing number of studies are including soil processes in ecosystem service assessments and valuation frameworks (De Groot, Wilson and Boumans, 2002; Robinson *et al.*, 2013). Soil N and P removal is generally seen as an intermediate service or a supporting or regulating service in current ecosystem services classification schemes, as it affects a number of final ecosystem goods and services (Boyd and Banzhaf, 2007).

Impacts of nitrogen on ecosystem services (ES), on the economy and on human well-being have been examined in a number of studies (Birch *et al.*, 2010; Compton *et al.*, 2011; van Grinsven *et al.*, 2013). Soil N and P storage could have implications for many benefits, including the following: (I) avoidance of consequences to ecosystem services provided by freshwater, groundwater and coastal waters from reduced quality for swimming, drinking, recreation or fishing; (II) avoidance of air quality problems associated with N such as those affecting human respiratory health or visibility (NO_x, NH₃); (III) avoidance of damage from climate change and stratospheric ozone depletion (N₂O); and (IV) maintenance of soil fertility and ecosystem production (both N and P). Eutrophication of coastal areas and associated hypoxia can result in physiological and behavioural impacts on important coastal organisms, populations and ecosystems that result in lowered fitness and productivity. However, there is a good deal of uncertainty about the economic damages associated with coastal eutrophication in many areas (Rabotyagov *et al.*, 2014). Efforts to inform policy should bring together ecologists and economists to study the impacts of N and P on ecosystem services all along the cascade.

7.5.2 | Acidification buffering

Soil acidity is controlled by both biota (plant roots and microorganisms) and particles (soil minerals and organic matter). Production of carbon dioxide, organic matter decomposition, and the excretion of acidic compounds by biota increase soil acidity, while binding of acidic compounds to root and particle surfaces, as well as mineral weathering, decrease it (Sposito, 2008). Over periods ranging from centuries to millennia, while most of the less resistant minerals become depleted through weathering reactions with rainwater and subsequent leaching, highly acidic soils are produced naturally. They now occupy about one-third of the ice-free land area on Earth (Guo *et al.*, 2010), mainly in the humid tropics and in the forested regions of temperate zones.

Industrial effluents (for example, sulphur and nitrogen oxide gases dissolved in atmospheric precipitation or transformed to particles, or acidic wastewaters) and nitrogenous fertilizers, such as urea, are typical anthropogenic inputs of acidity to soils. If these two acidic inputs exceed about 15 percent of the capacity of soil to neutralize them, acidification increases markedly, with a variety of serious problems arising for both plant and microbial growth. The potential for generating polluted runoff or drainage water also increases markedly. Over a 20 year period Guo *et al.* (2010) documented such increases of acidity in Chinese topsoils, caused by nitrogen fertilization and acidic deposition. The topsoils investigated showed an average pH decrease of 0.50, which is quite serious. Other long-term studies document decadal changes in soil acidity that are even larger (Richter and Markewitz, 2001). Acidic deposition is an important problem in China, but the acidification caused by nitrogen fertilization was found to be 10 to 100 times greater than that caused by acid rain. In the principal double-cropping cereal systems of China (wheat-maize, rice-wheat, and rice-rice), nitrogen fertilizer use efficiencies are only 30 to 50 percent. The progressive acidification of topsoil – as well as nitrogen pollution of agricultural runoff and drainage – will remain unchecked as long as this low nitrogen use efficiency is not addressed. Guo *et al.* (2010) noted that optimal nutrient-management strategies can significantly reduce nitrogen fertilization rates without decreasing crop yield, thus providing benefits to both agriculture and water quality.

7.5.3 | Filtering of reused grey water

Nearly 80 percent of urban 'blue water' becomes wastewater. At about 100 m³yr⁻¹ per household in the developed world, wastewater thus represents a rapidly expanding environmental and health challenge, particularly in urban centres. The ecological footprint of untreated wastewater is unsustainable even in regions where water is plentiful (e.g. South East Asia), as it may either increase nutrient loads in rivers and coastal regions or represent a direct hazard to human health. By contrast, arid regions increasingly rely on treated wastewater for irrigation, often practiced with little consideration of long-term impacts on the soil, hydrology and ecology of the producing area. The sustainability of this coupled agro-urban hydrological cycle hinges on proper management to mitigate adverse impacts of long-term wastewater use and avoid potential collapse of soil ecological functions. Various studies (e.g. Bond, 1998; Assouline and Narkis, 2013) have shown that, over the long term, even irrigation with wastewater results in significantly increased soil ESP that can adversely impact soil structure and hydraulic properties. In the absence of proper regulation, irrigation with wastewater may pose a range of human health and other ecological risks associated with introduction of pathogenic microorganism into the soil and crop (del Mar *et al.*, 2012). The sustainable management of wastewater irrigation requires new management strategies including water source mixing, proper selection and rotation of crops, and avoidance of sensitive soils.

7.5.4 | Processes impacting service provision

Trace elements

Elevated concentrations of potentially toxic trace elements can affect provision of the services that depend on soils. Trace elements – such as arsenic, cadmium, chromium, lead, mercury, and selenium – naturally occur in low quantities within soils. They may also be introduced and concentrated through anthropogenic activities like waste disposal, fertilizer and pesticide application, and atmospheric particulate emission and deposition (Sparks, 2003; Pierzynski, Vance and Sims, 2005). Even when at low concentrations in soils, they can have pronounced impacts on water quality. This is particularly the case where the capacity of soils to store trace elements is exceeded or where there are changes in the soil chemical, physical and/or biological environment that influence the partitioning of trace elements between the solid and aqueous phases.

The concept of the critical load of a specific trace element enables a precautionary assessment of the risks its input causes to food quality and of the eco-toxicological effects on organisms in soils and surface waters (Lofts *et al.*, 2007; de Vries *et al.*, 2013b). The critical load of trace elements is defined as “the load resulting at steady state in a concentration in a compartment (e.g. soil solution, plant, fish) that equals the critical limit for that compartment” (Lofts *et al.*, 2007; de Vries *et al.*, 2013b). The critical limit is a receptor-specific concentration below which significant effect on the receptor is assumed not to occur (Lofts *et al.*, 2007). The concept of critical loads – specifically the critical loads of acidity – was key in gaining acceptance of the need for reduction of atmospheric deposition of N and S (Section 4.4.1 above). However, the usefulness of the concept of critical loads of trace elements in international negotiations aimed at reducing trace element deposition is not equally evident. This is mainly owing to two factors that distinguish trace elements from the case of acidity and acid rain: (I) the time needed for a specific trace element in a specific scenario to attain steady state is much longer than for N and S; and (II) other changes in the environment, notably acidification, may have a greater influence on the exposure and effects of a specific trace element than the particular amount entering the system (de Vries *et al.*, 2013b). In fact, problems associated with trace elements in soils are commonly exacerbated by changes in land use that alter environmental conditions and increase the potential for exposure to trace elements through food and water consumption. Because of this, in addition to applying the concept of critical loads, the assessment of the future risks of trace elements needs to employ dynamic models (de Vries *et al.*, 2013b).

Salinity

Salinization of soil and water resources remains a chronic problem in many parts of the world, mostly in arid regions where evapotranspiration exceeds rainfall. The increased frequency of extreme climate events (droughts, intense rainfall events) together with the expansion of irrigated agriculture are expected to increase the range of soils affected by salinity.

In addition to the effects of hotter and drier climate patterns, the primary causes of salinity risk include: (I) increasing salt loads due to use of marginal water sources such as waste water; (II) over exploitation of coastal aquifers and related sea water intrusion (Várallyay, 1994); (III) overpumping and degradation of slowly replenishing inland aquifers (Ogallala); (IV) sea level rise impacting coastal wetlands (e.g. Mexico pacific coastline); (V) mismanagement of rapidly expanding irrigation in arid regions, particularly inadequate leaching and drainage and (VI) clearing of perennial vegetation in landscapes with significant salt stores in soils and deeper regolith.

One solution is to reduce the salt content of irrigation water through desalination. Recent advances in desalination techniques have resulted in a dramatic reduction in costs. Irrigation experiments with desalinated water show substantial increase in yield with less water used and less salt leaching to groundwater resources. However, the use of desalinated water requires careful management to avoid soil and ecological damage (e.g. clay dispersion) due to irrigation with extremely pure water (Yermiyahu *et al.*, 2007; Tal, 2006).

Erosion

Intensification of agriculture, changes in rainfall patterns with more intense rain events, and potentially more compacted soil surfaces may all contribute to increased rates of surface soil erosion. In addition to the removal of the top layer of productive soil and the incision of stream channels, the potential increase in soil transport to surface water may cause a cascade of adverse effects downstream. Pimentel *et al.* (1995) list impacts on stream and lake ecology, dam siltation and effects on waterways, and of course, potential for enhanced pollution by agrochemicals and colloid-facilitated transport of phosphorous and carbon. Soil erosion is also linked to climate change as it mobilizes large amounts of soil organic carbon (SOC). Since the industrial revolution and associated land use changes, SOC has been estimated to contribute 78±12 Gt of C to the atmosphere, of which about one-third is due to accelerated erosion and two-thirds to mineralization (WMO, 2005).



The WMO (2005) report estimates that 25 percent of African soils are prone to risk of water erosion (excluding deserts that comprise about 46 percent of the African land surface), and that 50 percent of cropland in Australia is susceptible to water erosion. Drier conditions associated with future climate extremes (droughts) may limit rates of soil carbon accumulation and reduce soil aggregation, thereby enhancing vulnerability to wind erosion. WMO (2005) estimate that about 22 percent of the African land surface is prone to wind erosion, and 15 percent of the cropland in Australia. A host of soil conservation strategies for combating land degradation due to soil erosion also offer co-benefits such as enhanced water storage in the soil profile (Pimentel *et al.*, 1995; Troeh, Hobbs and Donahue, 1991). Eroded landscapes may take centuries to millennia before their abilities to provide quality ecosystem services are restored.

7.6 | Soil change and water quantity regulation

Soil moisture regulation of precipitation

Soil moisture acts as a buffer for precipitation anomalies. As long as the soil is not saturated, it can reduce the direct impact of flooding. Similarly, soil moisture acts as a buffer against dry anomalies in the onset of meteorological droughts, before soil moisture or streamflow droughts are noticeable. However, if pre-event soil moisture is anomalously wet or dry, these same properties can also lead to significant flooding and droughts even where precipitation is not abnormally high or low. For these reasons, the monitoring of soil moisture conditions (as well as of snow and groundwater) is valuable for the forecasting of floods and droughts (e.g. Koster *et al.*, 2010b; Fundel, Jörg-Hess and Zappa, 2013; Orth and Seneviratne, 2013; Reager, Thomas and Famiglietti, 2014).

In addition to effects related to the buffering or persistence of soil moisture, several studies suggest that soil moisture also affects the regional water cycle through impacts of evapotranspiration on precipitation (e.g. Beljaars *et al.*, 1996; Koster *et al.*, 2004; Seneviratne *et al.*, 2010; Taylor *et al.*, 2012). However, the underlying feedbacks, including their sign, are strongly model-dependent (e.g. Koster *et al.*, 2004; Hohenegger *et al.*, 2008). Also observational studies diverge with respect to inferred soil moisture-precipitation feedbacks. Some suggest the presence of positive (temporal) feedbacks while others identify mostly negative (spatial) feedbacks (Findell *et al.*, 2011; Taylor *et al.*, 2012). In addition, causality is very difficult to establish based on observations (e.g. Salvucci, Saleem and Kaufmann, 2002). Precipitation persistence could, for example, lead to some confounding effects (Guillod *et al.*, 2014). Overall, effects of soil moisture on precipitation are still uncertain.

Human land and water use strongly affects soil moisture variations and the resulting land water balance, for instance through irrigation (Wisser *et al.*, 2010; Wei *et al.*, 2013) or other changes in agricultural practices (Davin *et al.*, 2014; Jeong *et al.*, 2014). These effects are generally not considered in present day climate models, although they could substantially affect soil moisture and hydrological drought projections, including feedbacks to the atmosphere.

7.6.2 | Precipitation interception by soils

Together with vegetation, soils help to regulate water quantity by intercepting water, reducing floods and maintaining the soil moisture buffer. Precipitation arriving at the Earth's surface can be intercepted by vegetation canopies and returned directly to the atmosphere through evaporation, never reaching the soil moisture pool. Typically, trees can intercept 25-50 percent of precipitation and shrubs 10-25 percent, while interception by grass is significantly less (Calder, 1999). The rest of the precipitation arrives at the soil surface, the characteristics of which control the partitioning between what infiltrates and what runs off into surface water. In a recent meta-analysis, Jarvis *et al.* (2013) have shown that K is largely dependent on bulk density, organic carbon content and land use. This has important consequences for ecosystem service delivery by soils,



as it indicates that management and land-use change will affect the soil infiltration service temporally as well as spatially. This analysis by Jarvis *et al.* (2013) corresponds to an increasing number of studies that show the importance of vegetation in determining soil K values on similar soils.

Because of their large root systems, trees in particular create conduits for conducting water into soil. Both dead and living roots can create flow networks. Beven and Germann (1982) cited work suggesting that as much as 35 percent of the volume of a forest soil may contain macropores formed by roots. Chandler and Chappell (2008) demonstrated that K was highest near the trunk of single oak trees and decreased toward the edge of the canopy. The ratio of K geometric mean values under the tree at 3 metres from the trunk to the adjacent pasture was 3.4 times higher, similar to results compiled from the literature in the same paper. Gonzalez-Sosa *et al.* (2010) presented conductivity data for a range of land use types in France, with trees being generally higher, and crops and pasture lower for the same soil. In the tropics, deforestation results in a major reduction in infiltration, whether the forest is recently cleared or has been turned into pasture (Zimmermann, Elsenbeer and De Moraes, 2006).

Soil macrofauna - worms, ants and termites etc. - also play an important role in determining infiltration at local scales (Beven and Germann, 1982; Lal, 1988), and perhaps also regionally and globally given the prevalence of these organisms. There are typically two modes of macrofauna action impacting hydraulic properties. The first is the creation of burrows forming macropores; the other is the turnover of soil and aggregation which impacts infiltration and water retention, generally increasing both. Soils might offer potential for slowing water movement across landscapes under certain precipitation conditions (Marshall *et al.*, 2009). However, once runoff is generated and large quantities of precipitation fall, the role of soils is likely to be less important. Above a certain threshold, massive floods can occur in almost any landscape

Although often cited as an important ecosystem service, the impact of land use on altering flood risk remains hard to quantify with any precision (Pattison and Lane, 2011). The link between land management and flood risk is complex and scale dependent as conceptualized by Bloschl *et al.* (2007). Many studies have demonstrated how land or soil management impact infiltration and runoff generation at the plot to hillslope scale (Wheater and Evans, 2009). These tend to be local effects in temperate zones, but can be large scale in the tropics. Beven *et al.* (2008) found a distinct land use signal hard to detect, and also pointed out that "adequate information about past land management changes and soil conditions is not readily available but will need to be collected and made available in future for different land use categories if improved understanding of the links between runoff and land management is to be gained and used at catchment scales."

7.6.3 | Surface water regulation

Soils provide a maintenance service that contributes to the regulation of base flow and water supply in rivers. Groundwater, lakes and soil drainage all play a role in setting base flow in surface waters (Price, 2011). Groundwater dominates in the lowlands, but soil drainage dominates upland catchments. Changes to the hydraulic characteristics of upland catchments and to the quantity of water stored by soils will have distinct implications for water supply downstream. Again, the soil water retention characteristics and hydraulic conductivity play a crucial role in the regulation of drainage.

7.7 | Soil change and natural hazard regulation

Soil and its characteristics (depth, hydro-mechanical properties, mineralogy, ecological function, and position in the landscape) play an important role in several natural hazards including: landslides, debris flows, floods, dam failure, droughts, shrink and swell damage to roads and infrastructure, and more. The United Nations International Strategy for Disaster Reduction (UNISDR, 2009) defines a natural hazard as a "natural

process or phenomenon that may cause loss of life, injury or other health impacts, property damage, loss of livelihoods and services, social and economic disruption, or environmental damage". Projected human population expansion, agricultural intensification, and greater human presence and infrastructure in mountainous regions combined with projected changes in climate extremes (IPCC, 2012) are expected to jointly contribute to enhanced vulnerability to soil-mediated natural hazards (Figure 7.9). The extent of the vulnerability and exposure to a particular type of hazard vary considerably among regions (ESPON, 2013). For example, floods may increase in flat terrains with increasing mean precipitation or rapid snowmelt, and landslides may become more common in mountainous areas with changes in the seasonality and intensity of rainfall (Huggel, Clague and Korup, 2012).

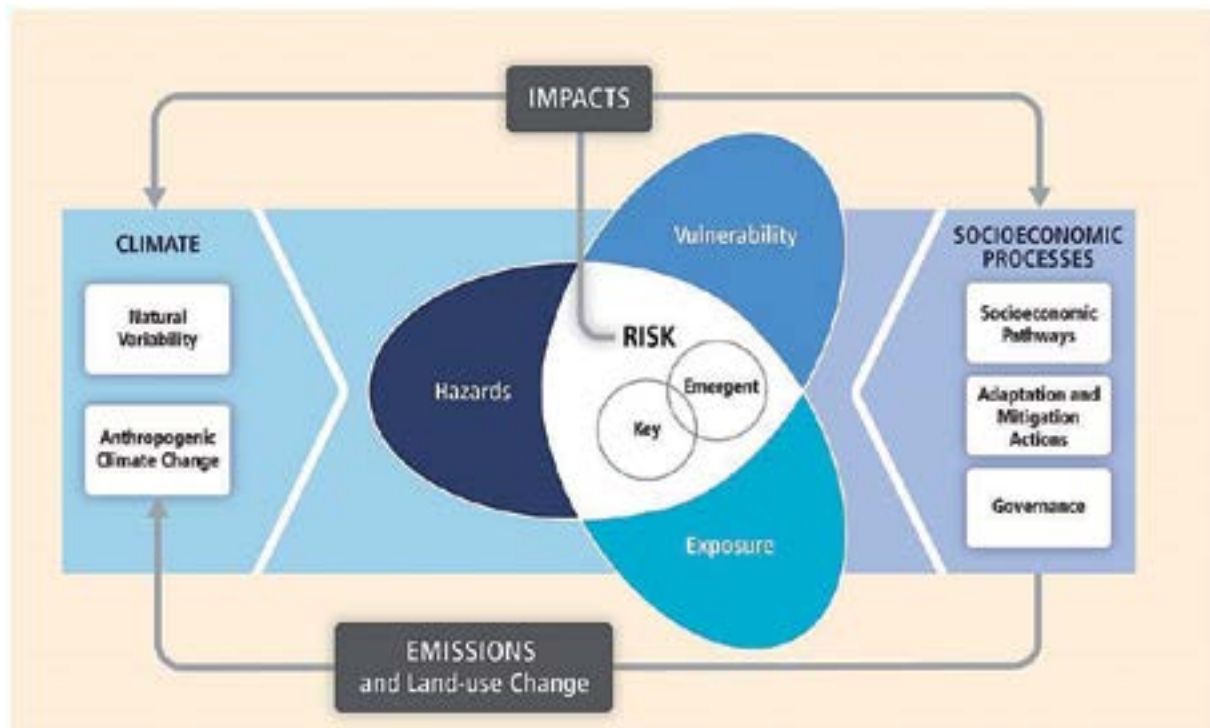


Figure 7.9 | A conceptual sketch of how vulnerability, exposure and external events (climate, weather, geophysical) contribute to the risk of a natural hazard.
Source: IPCC, 2012.

The past few decades have been marked by an increase in the frequency and magnitude of damages caused by soil-climate related hazards such as landslides (Figure 7.10, FAO 2011). In part this increase may be simply attributed to more timely and accurate reporting, and also to deeper human penetration into soil-hazard prone regions, facilitated by increases in mobility and personal wealth (Keiler, 2013; Papathoma-Köhle *et al.*, 2015). The reports of EM-DAT (<http://www.emdat.be/publications>) provide a global perspective of all aspects of natural disasters and their human and economic impacts. The 2013 EM-DAT² report estimates global damages by natural hazard attributed to hydrological and geophysical causes (most closely related to soil) in excess of US\$ 60 billion, with impacts on the lives of 40 million people in 2013 alone. It is instructive to place the various natural hazards in their soil-human-climate context to enable general inferences and detection of future trends with global change (population growth, land use, and climate change).

2 <http://www.emdat.be/publications>

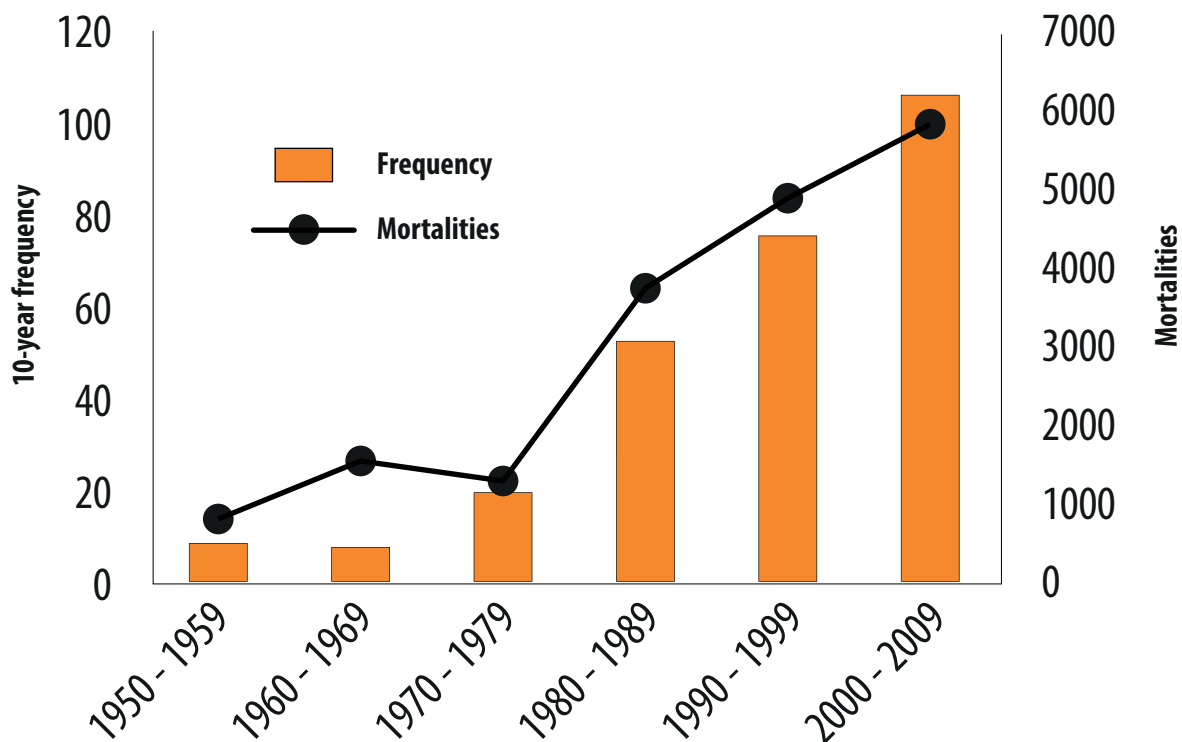


Figure 7.10 Trends in landslide frequency and mortality on Asia.
Source: FAO, 2011; EM-DAT, 2010.

7.7.1 | Soil landslide hazard

The depth of the soil mantle forming over mountainous topography reflects a natural balance between soil production and soil erosion processes (Trustrum and De Rose, 1988; Heimsath *et al.*, 1997). The primary soil removal process in mountainous regions is landsliding, driven by the topographic relief and triggered by climatic forcing such as rainfall or snowmelt (Iverson, 2000; Larsen, Montgomery and Korup, 2010; Kawagoe, Kazama and Sarukkalige, 2009) or by earthquakes (Huang and Fan, 2013). Landslide damage is costly: Sidle and Ochiai (2006) estimated the direct costs associated with rebuilding or replacing infrastructure at several billion dollars per year, even without considering indirect costs related to construction and temporary loss of site functionality. Similar estimates have been made just for Europe (Papathoma-Köhle *et al.*, 2015).

Rainfall is the most common trigger for shallow landslides (Iverson, 2000). The strong relationship between rainfall intensity-duration and landslide triggering conditions has prompted the use of rainfall characteristics for early warning (Guzzetti *et al.*, 2008; von Ruettele, Lehmann and Or, 2014). The observed increase in precipitation variability and in extreme events attributed to climate change has been linked to the observed increase in landslide frequency in mountainous regions (Huggel, Clague and Korup, 2012). The recent IPCC report (IPCC, 2012) lists evidence for the contiguous United States confirming statistically significant increases in heavy (upper 5 percent) and very heavy (upper 1 percent) precipitation of 14 and 20 percent, respectively. Moreover, evidence from Europe and the United States suggests that the relative increase in precipitation extremes is larger than the increase in mean precipitation.

Schmidt and Dikau (2004) found that climatic scenarios representing unstable conditions of transition from more humid to a dryer climate produced the highest slope instabilities. Soil hydraulic properties play an important role in imparting mechanical sensitivity. Indeed, the soil plays multiple roles in the landslide hazard, not only as the mass that slides down the slope, but also through its own mechanical strength and through its modulation of local hydrology via infiltration capacity, base flow, macropore flow and ground cover (Iverson, 2000; Sidle and Ochiai, 2006; Lehmann and Or, 2012). The partitioning of precipitation between infiltration, overland flows and base flows is critical to the loading of the soil and to the ultimate soil failure. The mechanical reinforcement by plant roots helps to stabilize the soil mantle (Abe and Ziemer, 1991; Schwarz, Cohen and Or, 2012), and bulk soil mechanical and hydraulic properties affect the susceptibility to failure.



Recent widespread drought-induced forest die-offs highlight how climate change could accelerate forest mortality. This has potential consequences for the carbon cycle and for ecosystem services (Anderegg *et al.*, 2013). Through loss of root reinforcement, die-off may also increase landslide hazard. Rapid landslide processes have also been observed in Southeast Asia and the Western Pacific where large tropical cyclones induce numerous landslides and remove significant amounts of soil and particular carbon through the river systems to the ocean (Hilton *et al.*, 2008). These extreme tropical precipitation events are likely to increase in frequency and magnitude (Huang *et al.*, 2013).

7.7.2 | Soil hazard due to earthquakes

Keefer (2002) provides a historical overview of the study of earthquake-induced landslides. These are often extensive in their size and occurrence and cause more significant damage than hydrologically-induced shallow landslides. For example, the 2008 Wenchuan earthquake in Sichuan province in China triggered more than 60 000 landslides over an area of 35 000 km² causing about one-third of the total number of fatalities in the earthquake disaster (Huang and Fan, 2013). In addition to the direct damages, the Wenchuan earthquake induced an unprecedented number of secondary geohazards such as heightened subsequent landslide frequency, causing river damming and consequent floods as well as debris flows. The links between seismic activity and landslide characteristics were systematically investigated by Malamud *et al.* (2004) based on landslide inventory data of landslide size-frequency distribution in the affected landscape. These analyses are useful for deriving large-scale soil erosion rates enhanced by seismic activity. Erosion rates in active subduction zones are around 0.2–7 mm yr⁻¹. Hazard schemes often classify earthquake-induced landslides as 'geophysical' or 'dry' events to indicate they do not require water for mass movement initiation, unlike hydrological 'wet' landslides.

On March 11, 2011, a seaquake followed by an enormous tsunami and by the destruction of the Fukushima Atomic Power Plant, Japan, brought about additional soil changes such as liquefaction, tsunami sedimentation and radio isotope contamination, all of which affected the local population. Liquefaction brought about by the earthquake occurred mainly on soil-banked lands or soil-dressed lands, causing extreme damage to housing and structural facilities. The tsunami carried massive deposits from the bottom of the sea onto farmlands along the seashore. This sedimentation contained considerable quantities of arsenic (Kozak and Niedzielski, 2013). The explosion of the atomic power plant resulted in soil contamination (mainly with Cesium 137) of an area as large as 800 square kilometres (Steinhauser, 2014; Itoha *et al.*, 2014). Cleaning these contaminants is vital before the population can return. More broadly, although a variety of soil hazard regulation techniques have been developed (Gasso *et al.*, 2013; Delgado *et al.*, 2011; Esteves *et al.*, 2012) there is a need for both more research and more regulation related to soil hazards than hitherto.

7.7.3 | Soil and drought hazard

Droughts limit primary production and thus the accumulation rates of organic matter. Reduced accumulation rates contribute to soil vulnerability to water and wind erosion. Recent meso-scale strategies for combating drought damage and reducing risk in agro ecosystems have proposed landscape-scale vegetation management. This can, for example, take the form of patches or bands of perennial vegetation to promote feedbacks that are conducive to recycling of water vapour, soil moisture and nutrients (Ryan, McAlpine and Ludwig, 2010). An often ignored consequence of prolonged drought and soil water depletion is soil subsidence and related damage to buildings and infrastructure (Corti *et al.*, 2011). Corti *et al.* (2011) presented a systematic study of damage costs from drought-induced soil subsidence applicable across different climate regimes. The primary variables include drought severity, soil type (shrink/swell properties), land use, and vegetation.

Prolonged droughts and drier climate patterns accentuate damages due to soil *shrink/swell* properties. The insurance industry reports that damage to infrastructure often peaks following extreme drought events, especially in the densely built up regions of Europe and United States. Dry climate also induces other

phenomena such as the onset of massive *dust storms*. Dust storms can arise either from destabilization of vulnerable surface soils (the Dust Bowl), or from the drying of lake beds, or from desertification and loss of vegetation and similar soil destabilizing activities over large scales. The rates of wind erosion associated with sand storms may exceed 100 mm topsoil yr⁻¹ in sensitive regions in the Sahel. Prolonged exposure is known to pose respiratory health hazards to human population.

7.7.4 | Soil and flood hazard

Agricultural intensification has been linked to alteration of runoff mechanisms and to increased risk and burden of floods (Marshall *et al.*, 2014). Some of the primary changes in land management documented in the United Kingdom and elsewhere that affect soils include: heavy traffic contributing to soil compaction, tillage operation and consequent loss of soil structure, the formation of larger fields, choice of cover crops in rainy seasons, and increased livestock densities (O'Connell *et al.*, 2007). However, establishing rigorous causal links between changes in land management practices, local runoff generation and catchment scale flood behaviour remains a challenge (Ewen *et al.*, 2013). Nevertheless, mounting evidence suggests that soil and land management contribution to flood risk is not limited to management of lowland agricultural regions. Management of upland soils and related impacts on runoff generation mechanisms cascade and also have impacts on flood risk downstream (Wheater and Evans, 2009; Marshall *et al.*, 2009). A recent review by Hall *et al.* (2014) on flood trends in Europe (including climatic effects) confirms the important role of land use changes (urbanization, afforestation, etc.) as key factors in modifying large scale flood risk. Some of the strategies for reducing flood risk include afforestation in upland catchments (Ewen *et al.*, 2013), creation of retention basins, and adding floodplains by lowering levees (Hall *et al.*, 2014).

7.7.5 | Hazards induced by thawing of permafrost soil

Permafrost is perennially frozen soil remaining at or below 0°C for at least two consecutive years (Brown *et al.*, 1998). Permafrost regions occupy about 24 percent of the exposed land area in the Northern Hemisphere and in some high mountainous regions (UNEP, 2012). Expected thawing of permafrost is projected to induce alterations in soil hydrology and biological activity, and to have an impact on the global carbon cycle (Schuur *et al.*, 2008). In addition, the thawing of permafrost is expected to change vegetation species and reshape many ecosystem functions. The mechanical weakening of the previously frozen soil is likely to result in foundation settling, with damage to buildings, roads, pipelines, railways and power lines (Nelson, Anisimov and Shiklomonov, 2001; Jorgenson, Shur and Pullman, 2006). Estimates of infrastructure repair in Alaska up to 2030 are in the range of US\$ 6 billion (UNEP, 2012). Changes in mean temperature and snow cover also affect sensitive permafrost in high mountains, and contribute to a higher risk of landslides and avalanches (Gruber and Haeberli, 2007; Harris *et al.*, 2009). Schoeneich *et al.* (2011) present an extensive report and case studies, largely from the European Alps, on various slope movement hazards (landslides, rock fall, and debris flow initiation) associated with degrading permafrost. Evidence suggests accelerated erosion rates of the thawed permafrost, especially along coastlines and rivers banks as documented by Schreiner, Bianchi and Rosenheim (2014) and Vonk *et al.* (2012), with subsequent transport of the carbon-rich sediment through river systems to the ocean.

7.8 | Soil biota regulation

Soil biodiversity is vulnerable to many anthropogenic disturbances, including land use and climate change, nitrogen enrichment, soil pollution, invasive species and the sealing of soil. A recent sensitivity analysis revealed that increasing land use intensity and associated soil organic matter loss are placing the greatest pressure on soil biodiversity (Gardi, Jeffery and Saltelli, 2013). Numerous studies report soil biodiversity declines as result of the conversion of natural lands to agriculture (Bloemers *et al.*, 1997; Eggleton *et al.*, 2002; Dlamini



and Haynes, 2004), and as a result of agricultural intensification (Mulder *et al.*, 2005; Postma-Blaauw *et al.*, 2010; De Vries *et al.*, 2013a). In particular, studies show larger bodied soil animals, such as earthworms and termites are especially vulnerable, but intensive land use can also reduce the abundance and variety of species of nematodes, mites and collembolans.

Climate change also poses a considerable threat to soil biodiversity through direct effects of warming and altered precipitation (e.g. drought and flooding) on the availability of moisture in soil (Bardgett *et al.*, 2008). Indirect climate change effects of warming and elevated atmospheric carbon dioxide may also have an impact on the quantity and quality of organic matter in soil (Blankinship, Niklaus and Hungate, 2011; van Groenigen *et al.*, 2014). Although poorly understood, predicted increases in the frequency of erosive rainfall events (Nearing *et al.*, 2005) and climate-induced shifts in land use (Mullan, 2013) could pose a considerable future threat to soil biodiversity. Other threats to soil biodiversity include nitrogen enrichment, which negatively impacts soil fungi (Treseder, 2008), soil sealing, which effectively stops the natural functioning of soil (Gardi, Jeffery and Saltelli, 2013), and invasive species, which affect native soil biodiversity through a range of mechanisms, including altered resource supply, competitive interactions and predation, and physical and chemical modification of the soil environment (Wardle *et al.*, 2011).

Although it is well known that soil organisms play key roles in many ecosystem processes, our understanding of the functional consequences of belowground diversity loss is limited, at least compared to what is known about aboveground losses (Cardinale *et al.*, 2012). Recent synthesis of experimental studies on soil diversity-function relationships indicate that diversity effects on processes of nutrient and carbon cycling are highly variable, but effects of species loss are most pronounced at the low end of the diversity spectrum (Nielsen *et al.*, 2011). There is also a general consensus that changes in the functional composition of belowground communities, rather than species diversity per se, are of most importance for ecosystem functioning (Nielsen *et al.*, 2011). Consistent with this, laboratory studies with low numbers of species have shown the functional composition of soil macrofauna communities to be a better predictor of litter decomposition than species richness (Heemsbergen *et al.*, 2004). The selective removal of different groups of soil organisms has been shown to impair soil functioning (Wagg *et al.*, 2014). Likewise, a recent cross-biome field experiment showed that the loss of key components of the decomposer communities consistently slowed rates of litter decomposition and carbon and nitrogen cycling, indicating negative effects of diversity loss on soil functions (Handa *et al.*, 2014). A field-based study of different sites across Europe also showed that changes in soil food web composition resulting from intensive agriculture consistently strongly affected processes of carbon and nitrogen cycling (De Vries *et al.*, 2013a). At one site, high intensity management reduced the resistance and resilience of the soil food web to drought, increasing soil carbon and nitrogen loss as greenhouse gases and in leachates (De Vries *et al.*, 2012a, 2012b, 2012c).

Changes in soil biodiversity can also modify vegetation dynamics, both directly through associations of symbionts and pathogens with plant roots, and indirectly, by modifying nutrient availability to plants (van der Putten *et al.*, 2013). For example, mycorrhizal fungi, which form symbiotic associations with roots of most plant species and are very vulnerable to soil disturbances, can enhance plant species diversity by relaxing plant competition intensity and promoting more equitable distribution of resources within the plant community (van der Heijden, Bardgett and van Straalen, 2008). Also, plant diversity and productivity have been shown, in some situations, to be positively related to arbuscular mycorrhizal fungal diversity due to more efficient use of soil phosphorus (van der Heijden *et al.*, 1998). Soil pathogens, which cause considerable problems for agricultural crops, have also been shown to impact vegetation dynamics in natural settings, by suppressing the growth of their host plant species more than their neighbours, thereby contributing to vegetation change (Bever, Westover and Antonovics, 1997; Packer and Clay, 2000; Klironomos, 2002). The spread of invasive plant species has also been linked to release from their natural soil enemies in their new territories, giving the invasive plant a competitive edge over native species. This often leads to declines in plant diversity and to shifts in the functioning of the soil (Wardle *et al.*, 2011; van der Putten *et al.*, 2013).



Although poorly explored, diversity changes in soil are likely to impact soil physical properties, with consequences for ecosystem services related to soil formation and water regulation. Diversity effects on soil physical properties have not been explicitly studied, but they are likely to be important given the potential for different groups of soil organisms to differentially impact soil structure through different routes. For example, fungi promote soil aggregate stability through the physical enmeshment of soil particles by their extensive networks of mycelia, whereas bacteria produce metabolic products, mainly polysaccharides, which bind soil particles together (Hallett *et al.*, 2009). Mycorrhizal infection can also influence soil aggregate stability through physical enmeshment of soil particles by their extensive networks of mycelium, but also through the binding of soil particles via the production of extracellular polysaccharides and proteins, including the protein glomalin, which alters the wetting behaviour of soil (Rillig and Mulley, 2006). Finally, soil animals, especially ecosystem engineers such as earthworms and termites, impact soil structure by creating macropores and channels, thereby improving water movement through soil (Bardgett, 2005).

While evidence is mounting that shifts in soil biodiversity resulting from human activities have significant consequences for ecosystem functions and the services that they underpin, there is still much to be learned. The mechanisms by which soil biodiversity change can impact ecosystem are enormous, involving a range of ecological and evolutionary processes at different spatial and temporal scales, and links between aboveground and soil communities. Moreover, impacts of soil biodiversity change on soil functions are likely to be context dependent, varying with soil abiotic properties and vegetation type. Unravelling this complexity in order to make better predictions about the consequences of soil biodiversity change for the services that ecosystems provide is a major challenge.

7.9 | Soils and human health regulation

The linkage between soils and human health is increasingly being recognized (Abrahams, 2006, 2013; Baumgardner, 2012; Brevik and Burgess, 2013; Jeffrey and van der Putten, 2011; Oliver, 1997). A central understanding is that soils form an integral link in a holistic view of human health that includes physical, mental and social dimensions. The soil acts as a natural filter, it can kill off pathogens, it can biodegrade organics and, in general, it does a wonderful job of protecting us from human health threats. However, soil is not able to protect itself against all the insults it is subject to on a regular basis.

Soils aid in the regulation of human health. They do this by keeping in check, or balancing, the beneficial versus deleterious concentrations of elements and moderating disease-causing organisms. For example, soils regulate human health by impacting the nutrient quality or nutrient density of foods. Too little of an essential nutrient in soil can lead to human diseases such as Keshan disease caused by selenium deficiencies in the human diet (Chen, 2012). Conversely, health problems can be caused by an excess amount of organics or trace elements such as the arsenic released by soils into the drinking and irrigation waters of Bangladesh (Khan, Hamra and Mu, 2009) (Section 7.3).

Soil is a natural source of radiation that can adversely affect human health, and soil can also affect human health by directly interacting with people. One example is the disease of podoconiosis or Mossy Foot disease (Mossy Foot Project, 2014). Mossy Foot disease affects about 5 percent of the population in highland tropical areas with volcanic soils and lots of rainfall. These soils are rich in silicates that can penetrate the skin of susceptible people as they go barefoot about their daily business. Soils can also act as a reservoir of all kinds of introduced materials that can impact human health. The dioxin at Love Canal in New York, United States is a classic example (Silkworth, Culter and Sack, 1989). There are large quantities of industrial and agricultural products and by-products added to soil every year that have the potential to impact human health.

Vast numbers of people, primarily women, infants and children, are afflicted with trace element deficiencies (notably Fe, I, Se, and Zn), mostly in the resource-poor countries of the developing world. A diet with low boron (B) has been found to lead to a number of general health problems and to increase cancer risk. The most common symptoms of B deficiency include arthritis, memory loss, osteoporosis, degenerative and soft cartilage diseases, hormonal disequilibria and a drop in libido (Scoreⁱ and Popa, 2010). According to one hypothesis, the low cervical cancer incidence in Turkey correlates with its B-enriched soil (Simsek *et al.*, 2003). Indeed, the ingestion of B via drinking water prevents cervical cancer risk (Korkmaz *et al.*, 2007). In a survey in northern France, exposure to high levels of boron (>0.3 mg L⁻¹) in the drinking water was associated with a significantly lower mortality rate as compared to that of a low-boron reference area (Yazbeck *et al.*, 2005).

Silicon is the second most abundant element in the Earth's crust. Dietary silicon intake is positively associated with bone mineral density in men and premenopausal women of the Framingham Offspring cohort (Jugdaohsingh *et al.*, 2004). Silicon is bound to glycosaminoglycans and has an important role in the formation of cross-links between collagen and proteoglycans (Carlisle, 1976). In vitro studies have demonstrated that silicon stimulates type 1 collagen synthesis and osteoblast differentiation (Reffitt *et al.*, 2003).

Many physicians have believed that zinc deficiency is a rare occurrence in Japan. Nevertheless, One study found many zinc-deficient patients at a clinic in Japan since 2002 (Kurasawa, Kubori and Okuizumi, 2010). Their complaints were anorexia, general fatigue, impaired sense of taste, burning mouth, various types of skin lesion, delayed wound healing and emotional instability.

Based on dietary intake recommendations, subclinical or marginal Mg deficiency (50 percent to <100 percent of requirement) commonly occurs throughout the world (Nielsen, 2010). Yet, pathological conditions attributed specifically to dietary Mg deficiency alone are considered rare. However, epidemiological and correlation studies indicate that a low Mg status is linked to numerous pathological conditions associated with aging, including atherosclerosis and hypertension (Ma *et al.*, 1995), osteoporosis (Rude, Singer and Gruber, 2009), diabetes mellitus (Barbagallo *et al.*, 2003), and some cancers (Dai *et al.*, 2007). Magnesium (Mg) deficiency increases genomic instability and Mg intake has been reported to be inversely associated with a risk of colorectal cancer (CRC). An experiment designed to determine whether Mg in drinking water suppresses inflammation-associated colon carcinogenesis in mice showed Mg at all doses caused a significant inhibition of CRC development (Kuno *et al.*, 2012).

The role of soil in contributing to human health is considerable. Soils rich in biodiversity produce healthier and more nutritious foods and control the proliferation of any pathogenic microorganisms that affect both plant and human health. Biodiversity is a result of a highly functioning, high quality soil with a good balance of nutrients and good water infiltration and aeration.

An example of how imbalance in a soil, for example improper soil water balance, causes human and animal disease comes from Australia (Hampton *et al.*, 2011; Creswell, 2012). Birds and people were being infected and some died due to a disease, melioidosis, caused by the bacteria *Burkholderia pseudomallei*. This microorganism is normally found only at low levels in soil, mostly in subsoils. However, after heavy rains, pools or puddles of water provide a suitable habitat for the bacteria to proliferate. It enters the body via a cut or graze or through the lungs by inhalation. Soil that is properly drained and aerated regulates the prevalence of this bacteria and keep it in check.

Soils also serve as a source of many medicines. For example, soil microorganisms still account for many of the current clinically relevant antibiotics (D'Costa *et al.*, 2006; Pepper *et al.*, 2009). It is therefore important to maintain the vast diversity of microorganisms in soil in order preserve the untapped potential for future discoveries important to human health.



We are just now beginning to understand how the chemical, physical and biological properties of soil can affect the health of humans and animals and entire ecosystems. The same techniques that are available to map the human microbiome can also be applied to map the soil microbiome. We are thus on the verge of understanding what constitutes a healthy soil microbiome and how a degraded or unhealthy soil microbiome may affect our food production and overall human health.

7.10 | Soil and cultural services

The soil is one of the main sources of information on the prehistoric culture of humankind. Indeed, soil is an excellent medium for preserving artefacts. Different soil types have particular characteristics to preserve remains. For instance, in permanently or seasonally wet soils, the lack of oxygen slows down the decomposition of organic matters. Sometimes the remains of animals can be found with hunting marks from arrows or spears. Well-preserved human bodies have been excavated from moors and bogs. The anaerobic conditions preserve the bodies very well and several thousands of years later they are excavated with skin, flesh and clothes still present. Wooden constructions, such as poles for bridges, boats and wooden tools may also be preserved, giving us valuable information on the level of technology at that time.

Past farming practices can also be recognized in the soil profile, particularly in Anthrosols. For example, in northwest Europe, notably in the Netherlands and Germany, a human-made soil type, known as plaggen soil, has developed as a result of a specialized agricultural system. On the strongly leached, acid sandy outwash plains and moraines, Podzols developed underneath a vegetation cover of heather. Farmers used the heather and the uppermost level of the soil as bedding in the stables. The droppings from the animals, mixed with the bedding, were later used as manure on the nearby fields, slowly building up a thick soil layer rich in organic matter and high in nutrients and with a good soil water retention. These fields provide a relatively high and stable crop production compared to the surrounding land (European Soil Bureau Network, 2005).

In the Amazon basin, the Terra Preta soil owes its name to its very high charcoal content. It was created by the addition of a mixture of charcoal, bone and manure to the otherwise relatively infertile Amazonian soil. Terra Preta soils were created by indigenous peoples in the pre-Columbian era between 450 BC and AD 950 (Sombroek *et al.*, 2002). Technosols are modern examples of soils that store artefacts or are strongly influenced by (modern) humankind. They include soils from wastes (landfills, sludge, cinders, mine spoils and ashes), pavements with their underlying unconsolidated materials, and constructed soils in human-made materials (FAO, 2014).

Soils provide aesthetic and recreational value through the landscape, particularly in Globally Important Agricultural Heritage Systems (Koochak and Altieri, 2011). They have also been used as an aesthetic approach to raise soil awareness in contemporary art (Toland and Wessolek, 2014). Churchman and Landa (2014) provide a comprehensive treatment of the topic.

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