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of the United Nations

Status of the World's Soil Resources

Main Report

Chapter 4
Soils and Humans

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PARTNERSHIP



INTERGOVERNMENTAL
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4.1 | Current land cover and land use

The new Global Land Cover Share database (Latham *et al.*, 2014) includes eleven global land cover layers, each representing the major land cover classes defined by the FAO and SEEA legend (Weber, 2010).

Analysis of the database indicates that of the global land mass, artificial surfaces occupy 0.6 percent, croplands 12.6 percent, grasslands 13.0 percent, tree-covered areas 27.7 percent, shrub-covered areas 9.5 percent, herbaceous vegetation 1.3 percent, mangroves 0.1 percent, sparse vegetation 7.7 percent, bare soils 15.2 percent, snow and glaciers 9.7 percent and inland water bodies 2.6 percent.

The intensity of each land-cover type varies substantially across the globe according to numerous factors, including soils, altitude, climatic conditions and anthropogenic influences. For example, while cultivated land is less than 10 percent in most African regions, it accounts for more than 25 percent of the land in the Asia region. A land cover map is given in Figure 4.1. Summary statistics by region, derived from the respective GIS layers are given in Figure 4.2. In the following discussion, attention is focused on three main land cover classes: cropland, grasslands/grazing lands and forests. The management of these three classes has large impacts on soils and ecosystem services. The presence of artificial surfaces is treated in more detail in Section 6.7. More than 25 percent of the land mass carries almost no vegetation because of climatic factors (glaciers, deserts) or topographic or soil conditions.

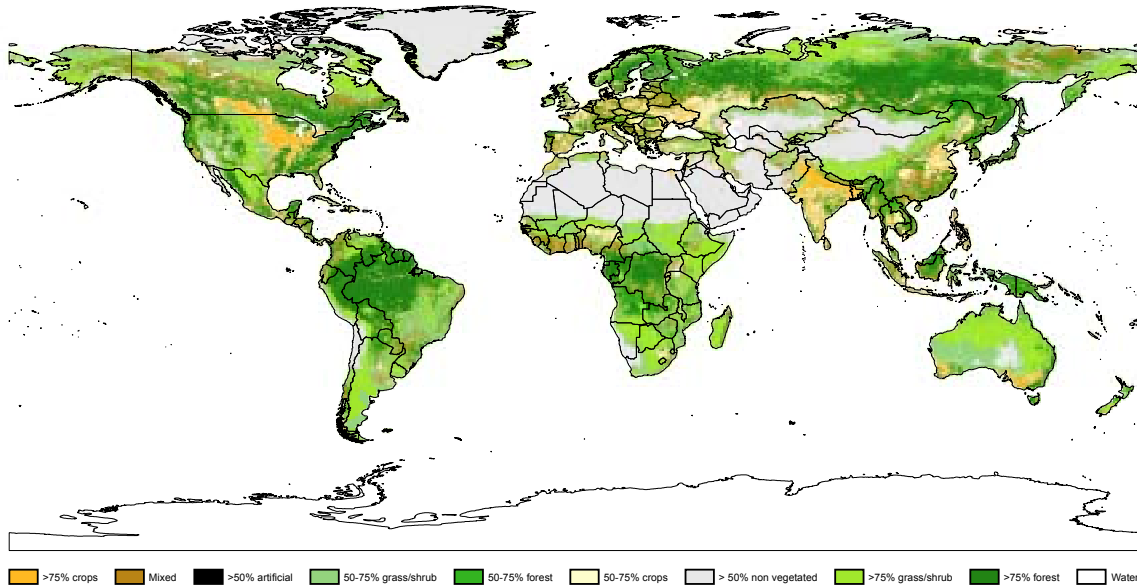


Figure 4.1 | Global Land Cover.
Source: Latham et al., 2014.

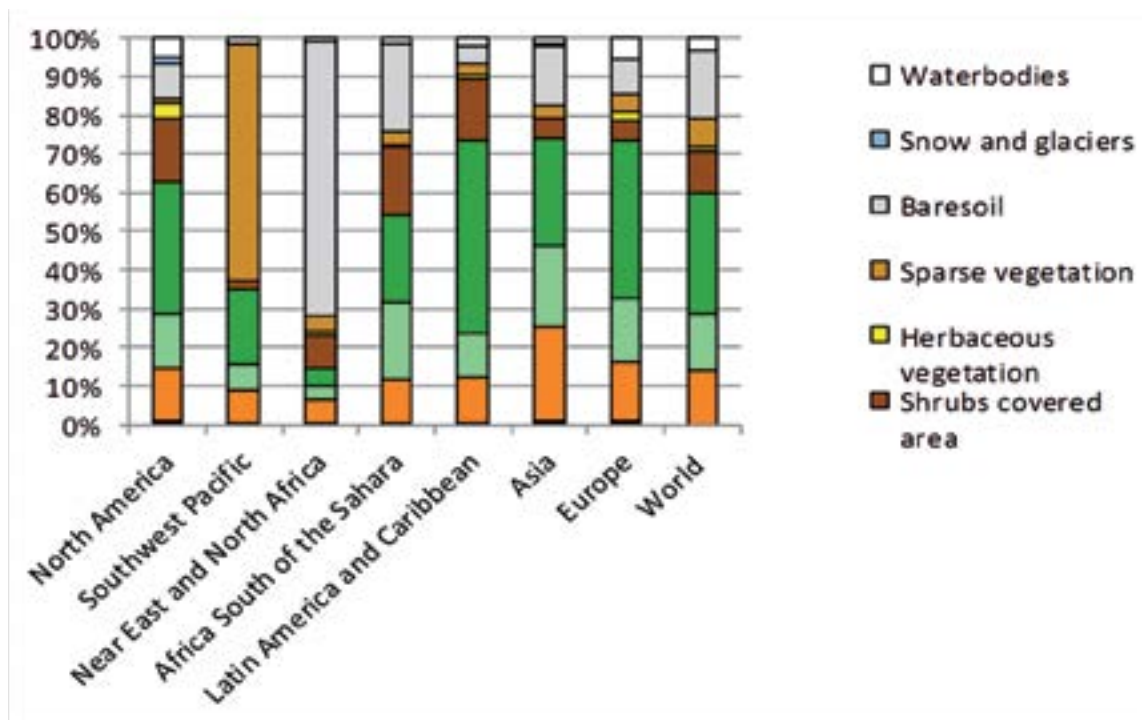


Figure 4.2 | Distribution of land cover in different regions.
Source: Latham et al., 2014.

Cropland

SOLAW (FAO, 2011) established that the cultivated land area in terms of per capita use in 2000 was highest in Australia (more than 2.2 ha per person), followed by North America and Eastern Europe and Russia (about 0.7 ha per person). In contrast, current cultivated land used per capita is only 0.2 ha in Western Europe and in most less developed countries.

By dividing the current cultivated land by the projected populations, the anticipated cultivated land area per capita in 2050 can be estimated. In the more developed countries, the cultivated area per capita would change little. In less developed countries, the cultivated area per capita is expected to halve to 0.1 ha by 2050, unless there is further expansion of the cultivated area.

Further characterization of cropland and land use at a global scale by remote sensing is difficult because:

1. The spatial extent of croplands is highly variable between and within nations. Cropland characteristics such as field size can be highly variable, even for the same crop type. Spatial extent of cropland depends on a host of factors, including the historical, political, social and technological context of agricultural development as well as natural factors such as landscape patterns.
2. Patterns of agricultural intensification – for example, the use of fertilizer – vary greatly, especially between developed and developing nations.
3. Each crop type has a specific growth phenology and structure, with significant seasonal variation between and even within individual crop types.
4. Cropland can be confused with natural vegetation cover types – for example, surveys may confuse cereal grains with tall-grass prairie (Pittman *et al.*, 2010). Better cropland information – in terms of both its extent and the purpose and intensity of its use – is vital to understanding soil change and to formulating adequate responses. Special attention should be paid to irrigated agriculture in developing countries, which covers about one-fifth of all arable land, and accounts for 47 percent of all crop production and almost 60 percent of cereal production (Nachtergaele *et al.*, 2011).

Grazing lands

Grazing lands, including sown pasture and rangeland with various coverage (grasslands, bush/shrublands), are among the largest ecosystems in the world and contribute to the livelihoods of more than 800 million people. They are a source of goods and services such as food and forage, energy and wildlife habitat, and also provide carbon and water storage and watershed protection for many major river systems. Grasslands are also important for *in situ* conservation of genetic resources. Of a total of 10 000 species, only 100 to 150 forage species have been cultivated, but many more hold potential for sustainable agriculture. Estimates of the proportion of the Earth's land area covered by grasslands vary between 20 and 40 percent, depending on the definition. Those differences are due to a lack of harmonization in the definition of grasslands.

There has been a significant reduction of pasture in Eastern Africa, partially because large grassland areas have been destroyed or converted to agricultural land. In South America, pastures have been lost because of conversion to soybean cultivation. In Europe there has been a gain in grazing lands because European policies such as the 'set-aside' measures oblige farmers to leave a portion of their agricultural land in fallow as a condition for benefiting from direct payments (Suttie, Reynolds and Batello, 2005).

Forests

In 2010, forests covered about 28 percent of the world's total land area. Deforestation affected an estimated 13 million ha per year between 2000 and 2010. Net forest loss was, however, considerably less – 5.2 million ha per year – as losses were compensated by afforestation and some natural expansion (FAO, 2014a). Most

deforestation takes place in tropical countries, whereas most developed countries with temperate and boreal forest ecosystems – and more recently, countries in the Near East and Asia – are experiencing stable or increasing forest areas. Between 1990 and 2010, the amount of forest land designated primarily for the conservation of biological diversity increased by 35 percent, indicating a political commitment to conserve forests. These forests now account for 12 percent of the world's forests.

Approximately 13.2 million people worldwide are formally employed in the forestry sector. Many more depend directly on forests and forest products for their living. In developing countries, wood-based fuels are the dominant source of energy for more than two billion mostly poor people. In Africa, over 90 percent of harvested wood is used for energy. Wood accounts for 27 percent of total primary energy supply in Africa, 13 percent in Latin America and the Caribbean and five percent in Asia and Oceania. However, it is also increasingly used in developed countries with the aim of reducing dependence on fossil fuels. For example, about 90 million people in Europe and North America now use wood energy as the main source of domestic heating (FAO, 2014a).

Conclusion

Land cover and land use are essential factors to understand soil change. In particular, better cropland information, in terms of extent, purpose and intensity of use, is vital to understanding soil change and to formulating adequate responses.

4.2 | Historical land cover and land use change

Since the early days of agriculture, human activity has altered vegetation cover and soil properties. 'Land use change' or 'land cover change' typically refers to changing from one type of vegetation cover to another (e.g forest to pasture, natural grassland to cropland). Although the terms land use change and land cover change are often used interchangeably, 'land use' is more typically used to refer to management within a land cover type. Land use is thus "characterised by the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it" (FAO/UNEP, 1999). Land use change has been accelerated by migration and population increase as food, shelter, and materials are sought and acquired. It is estimated that humans have directly modified at least 70 million km², or >50 percent of Earth's ice-free land area (Hooke, Martín-Duque and Pedraza, 2012).

For a long period of human activity, until about a thousand years ago, cropland and pasture occupied less than one to two percent each of the global ice-free land area (based on a range of data sources in Klein Goldewijk *et al.*, 2011 and depicted in Figure 4.3; also see Ramankutty, Foley and Olejniczak, 2002). Subsequently, as the population centres of Europe, South Central Asia and Eastern Asia expanded, more land was converted from natural vegetation to cultivated lands. Cover of croplands and pastures was about two to four percent each by 1700 (Klein Goldewijk *et al.*, 2011). By 1900, agriculture had further expanded in these areas, and spread to North America. Since 1900 rapid expansion has continued, including the arable areas of South America, Africa and Australia. As a result, today nearly all soils and climates suitable for cultivation in industrialized countries are in use for crop production. In some of these countries, cropland expansion has been reversed in recent years, as with the EU set-aside programme. South America and Africa continue to convert land use to crop production. By 2000, global cropland cover had reached 11 percent and pasture cover 24 percent, according to Klein Goldewijk *et al.* (2011) based on FAO statistics.

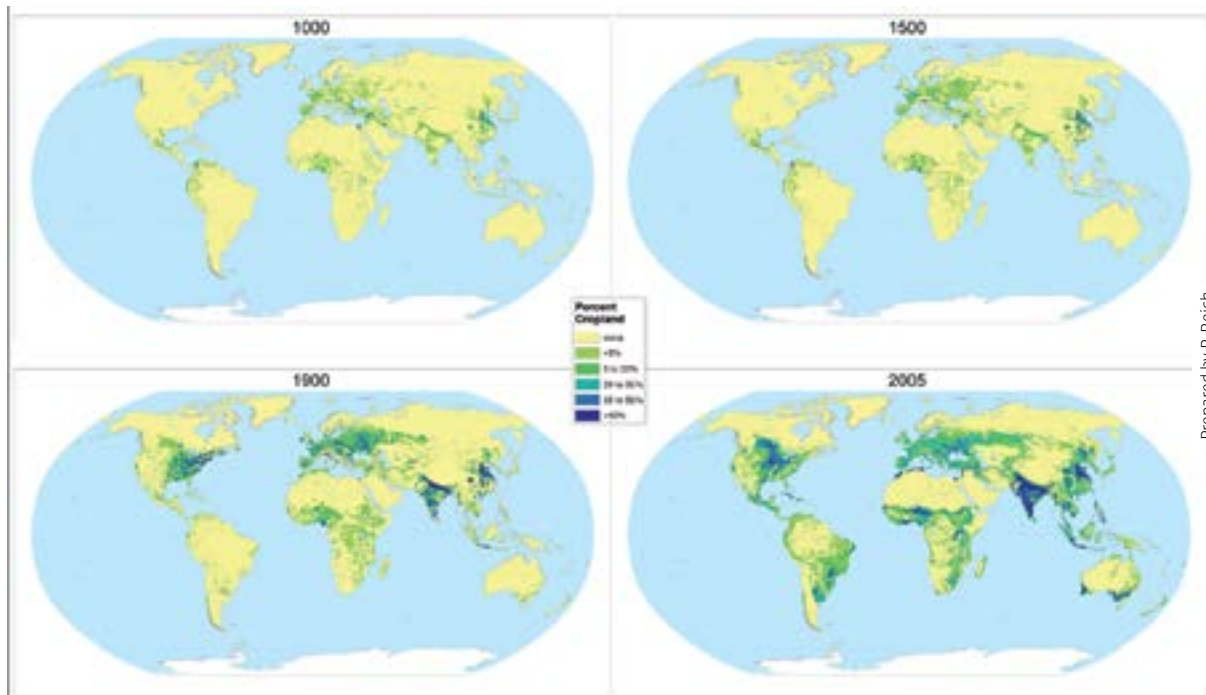


Figure 4.3 | Historical land use change 1000 – 2005.
Source: Klein Goldewijk et al., 2011.

The net loss of natural land has been dominated by loss of tropical forests (3.3 million km²), tropical grasslands (6.8 million km²) and temperate grasslands (5.5 million km²). Quantification from satellite imagery of global forest change over the period 2000–2012 shows that tropical deforestation remains the predominant source of losses (Hansen *et al.*, 2013). However, there has been a reduced rate of deforestation in some regions over the last decade, most notably in Brazil. This is coupled with a rising rate of afforestation in some areas in recent decades, notably in Europe and the United States, and more recently in China, Vietnam and India (FAO, 2013).

4.3 | Interactions between soils, land use and management

Many soils are subject to some degree of direct or indirect human disturbance. However, distinguishing natural from direct and indirect human influence is not always straightforward (Smith, 2005). Nonetheless, some human activities have clear direct impacts. These include land use change, land management, land degradation, soil sealing, and mining. The intensity of land use also has a great impact on soils. Soils are also subject to indirect impacts arising from human activity, such as acid deposition (for example, sulphur and nitrogen) and heavy metal pollution. In this section, we report the state-of-the-art understanding and the knowledge gaps concerning these impacts on soils.

4.3.1 | Land use change and soil degradation

Land cover change (Section 4.2), for example from forest or natural grassland to pasture or cropland, removes biomass and disturbs soils. This in turn leads to loss of soil carbon and other nutrients and to changes in soil properties and in soil biodiversity. Some land cover conversions – for example, afforestation after abandonment of cropland – can result in increases of soil carbon and nutrients. Land use that does not result in a change of cover, such as forest harvest and regrowth, or increasing grazing intensity, can nonetheless result in degradation of soil properties.

Degrading land covers approximately 24 percent of the global land area (35 million km²). 23 percent of degrading land is broadleaved forest, 19 percent needle-leaved forests, 20–25 percent rangeland (Bai *et al.*, 2008). The scale and nature of the changes are highly variable with type of land cover change, climate, and method of vegetation removal (e.g. land clearing fires, mechanical harvest). This section focuses on meta-analyses of field data and global model results. The effects of land use changes within agricultural lands are dealt with in Section 4.3.2.

Impacts of land cover change

Wei *et al.* (2014) collated observations from 119 publications of 453 paired or chrono-sequential sites in 36 countries where tropical, temperate, and boreal forests were converted to agricultural land. The SOC stocks were corrected for changes in soil bulk density after land-use change and only SOC in the upper 0–30 cm was considered. The SOC stocks decreased at 98 percent of the sites by an average of 52 percent in temperate regions 41 percent in tropical regions and 31 percent in boreal regions. The decrease in SOC stocks and the turnover rate constants both varied significantly according to forest type, cultivation stage, climate and soil factors. A meta-analysis (Guo and Gifford, 2002) of 74 publications across tropical and temperate zones showed a decline in soil C stocks after conversion from pasture to plantation (-10 percent), native forest to plantation (-13 percent), native forest to crop (-42 percent), and pasture to crop (-59 percent). Soil C stocks increased after conversions from native forest to pasture (+8 percent), crop to pasture (+19 percent), crop to plantation (+18 percent), and crop to secondary forest (+53 percent). Broadleaf tree plantations placed onto prior native forest or pastures did not affect soil C stocks whereas pine plantations reduced soil C stocks by -12 to -15 percent. In this study, soil depth varied from less than 30 cm to more than 100 cm and was not adjusted to account for changes in bulk density with land use change.

In a meta-analysis of 385 studies on land use changes in the tropics (Don, Schumacher and Freibauer, 2011), SOC decreased when primary forest was converted to cropland (-25 percent), perennial crops (-30 percent) and grassland (-12 percent). SOC increased when cropland was afforested (+29 percent) or under cropland fallow (+32 percent) or converted to grassland (+26 percent). Secondary forests stored 9 percent less SOC than primary forests. Relative changes were equally high in the subsoil as in the surface soil (Don, Schumacher and Freibauer, 2011). In this study, SOC stocks were corrected to an equivalent soil mass and sampling depth was on average 32 cm.

The response of soil organic carbon (SOC) to afforestation in deep soil layers is still poorly understood. Shi *et al.* (2013) compiled information on changes in deep SOC (defined as at least 10 cm deeper than the 0–10 cm layer) after afforestation of croplands and grasslands (total 63 sites from 56 literature). The responses of SOC to afforestation were slightly negative for grassland, and significantly positive for cropland. The SOC in soil depth layers (up to 80 cm) was reduced after afforestation of grassland but not significantly. By contrast, conversion of cropland to forests (trees or shrubs) increased SOC significantly for each soil layer up to 60 cm depth.

Poepflau *et al.* (2011) compiled 95 studies conducted on conversion in temperate climates. One finding was that topsoil (0–30 cm) SOC decreases quickly (~20 years) when cropland is established on grassland (-32 percent) or forest (-36 percent). By contrast, long lasting (> 120 years) sinks are created through conversion of cropland to forest (+16 percent) or grassland (+28 percent). Afforestation of grassland did not result in significant long term SOC stock trends in mineral soils, but did cause a net carbon accumulation in the labile forest floor (e.g. 38 Mg ha⁻¹ over 100 years). However, this carbon accumulation cannot be considered as an intermediate or long-term C storage since it may be lost easily after disruptions such as fire, windthrow or clear cut (Poepflau *et al.*, 2011).

Peatlands (organic soils) store a large amount of carbon which is rapidly lost when these peatlands are drained for agriculture and commercial forestry (Hooijer *et al.*, 2010). A rapid increase in decomposition rates leads to increased emissions of CO₂, and N₂O, and vulnerability to further impacts through fire. The FAO emissions database estimates globally there are 250 000 km² of drained organic soils under cropland and grassland, with total GHG emissions of 0.9 Gt CO₂eq yr⁻¹ in 2010. The largest contributions are from Asia (0.44 Gt CO₂eq yr⁻¹) and Europe (0.18 Gt CO₂eq yr⁻¹; FAO, 2013). Joosten (2010) estimated that there are >500 000 km² of drained peatlands in the world including under forests, with CO₂ emissions having increased from 1.06 Gt CO₂ yr⁻¹ in 1990 to 1.30 Gt CO₂ yr⁻¹ in 2008. This is despite a decreasing trend in Annex I countries, from 0.65 to 0.49 Gt CO₂ yr⁻¹, primarily due to natural and artificial rewetting of peatlands. In Southeast Asia, CO₂ emissions from drained peatlands in 2006 were 0.61 ± 0.25 Gt CO₂ yr⁻¹ (Hooijer *et al.*, 2010).

Soil drainage also affects mineral soils. Meersmans *et al.* (2009) showed that initially poorly drained valley soils in Belgium have lost significant amount of topsoil SOC (e.g. between – 2 and – 4 kg C m⁻² for the 1960–2006 period). The cause is most probably intensified soil drainage in these environment for cultivation purposes.

A serious consequence of deforestation is extensive loss of carbon from the soil, a process regulated by microbial diversity. Crowther *et al.* (2014) assessed the effects of deforestation on soil microbial communities across multiple biomes, drawing on data from eleven regions ranging from Hawaii to Northern Alaska. The magnitude of the vegetation effect varied between sites. Deforestation dramatically altered the microbial communities in sandy soils, while the effects were minimal in clay-rich soils, even after extensive tree removal. Fine soil particles have a larger surface area to bind nutrients and water. This capacity might buffer soil microbes in clay-rich soils against the disturbance of deforestation. Sandy soils, by contrast, have larger particles with less surface area and so retain fewer nutrients and less organic matter. Microbial community changes were associated with distinct changes in the microbial catabolic profile.

Dynamic Global Vegetation Models (DGVMs) can be used to look at the combined effects of land use change, climate, CO₂, and in some cases N deposition, on vegetation and soil properties over time. In Table 4.1, Figure 4.4 and Figure 4.5 we show results from three vegetation models: *ISAM* (Jain *et al.*, 2013; El-Masri *et al.*, 2013; Barman *et al.*, 2014 a, b), *LPJ-GUESS* (Smith *et al.*, 2001; Pugh *et al.*, 2015) and *LPJmL* (Bondeau *et al.*, 2007; Schaphoff *et al.*, 2013). The *ISAM* model includes a nitrogen cycle, N deposition and changes in soil N. The *ISAM* and *LPJ-GUESS* models were run with the HYDE historical land use change data set (*History Database of the Global Environment*, Klein Goldewijk *et al.*, 2011). The *LPJmL* group combined three land use change data sets (Klein Goldewijk and Drecht, 2006; Ramankutty *et al.*, 2008; Portmann, Siebert and Döll, 2010) with the global geographic distribution of agricultural lands in the year 2000 (Fader *et al.*, 2010). The models were also run with historical climate and CO₂ (and N deposition in the case of *ISAM*). Figure 4.4 shows the mineral soil C and N concentration of different land cover types in different geographic ranges while Table 4.1 and Figure 4.5 show the loss of carbon due to historical land use change from 1860 to 2010.

Differences between the models are large for some systems and regions due to different land use change data, different land cover definitions, different processes included in the models, etc. For example, soil carbon losses are higher in the *LPJmL* model in part due to greater land cover change in their land cover reconstructions. The highest carbon losses are associated with the conversion of forests to croplands (Figures 4.4 and 4.5). While Table 4.1 shows the global mean soil carbon loss, the effects are not the same everywhere (Figure 4.5). This may be the case, for example, when forests are converted to pastures in regions where pastures strongly favour soil C accumulation.

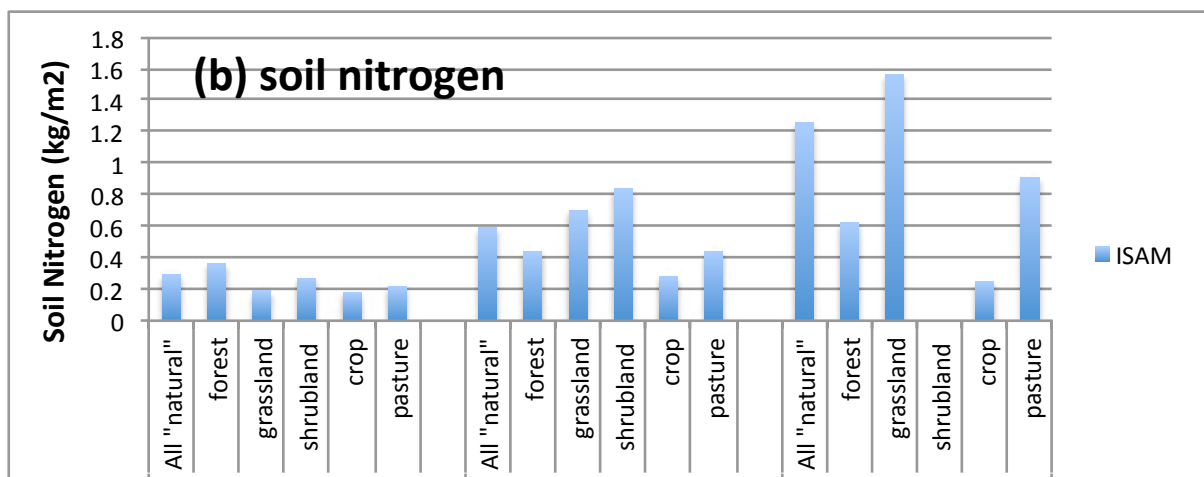
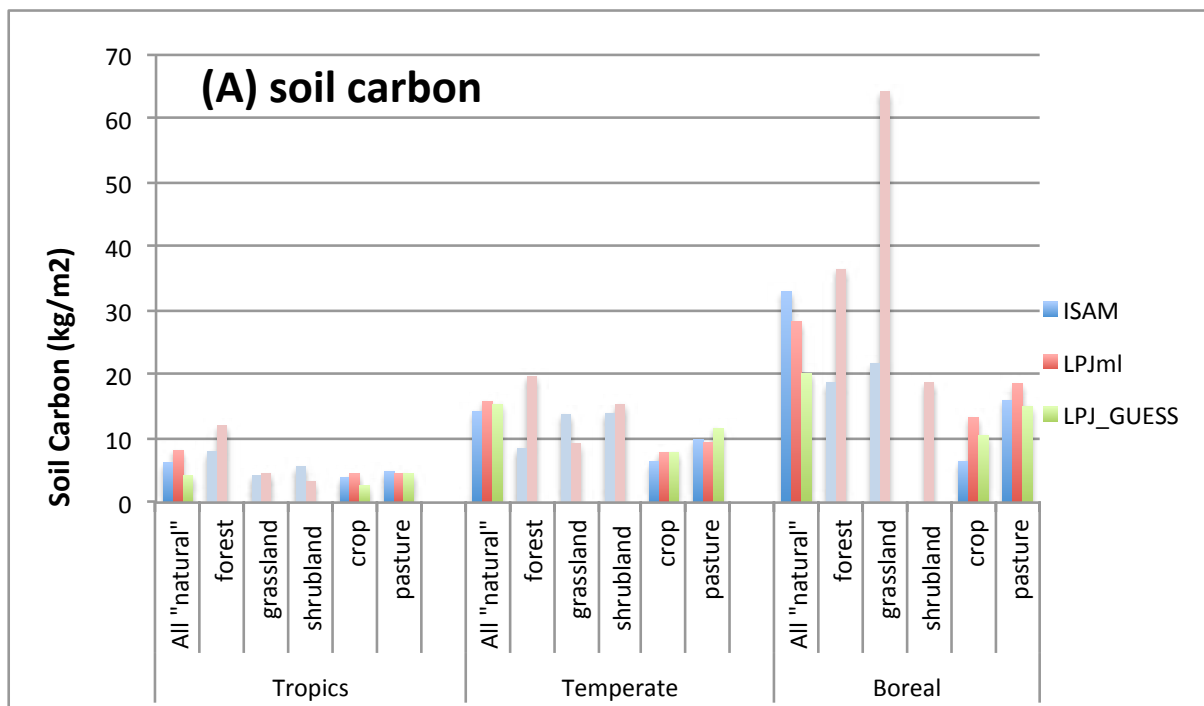


Figure 4.4 | Soil carbon and nitrogen under different land cover types.
Source: Smith et al. (in press).

Panel (a) shows mean soil carbon stocks; Panel (b) shows mean soil nitrogen stocks. Based on three vegetation models ISAM (Jain *et al.*, 2013; El-Masri *et al.*, 2013; Barman, Jain and Liang, 2014 a, b), LPJ-GUESS (Smith *et al.*, 2001; Pugh *et al.*, 2014); and LPJmL (Bondeau *et al.*, 2007; Schaphoff *et al.*, 2013). The soil carbon and soil nitrogen are the average over the period 2001 to 2010 (2003 for LPJmL) in model simulations with historical land-use change, climate, and CO₂ (and N₂ for the ISAM model). All 'natural' land is the mean of all lands without pasture or crop land cover. It includes 'un-managed' forest, grassland and shrubland categories and may include other land cover types depending on the models e.g. bare soil.



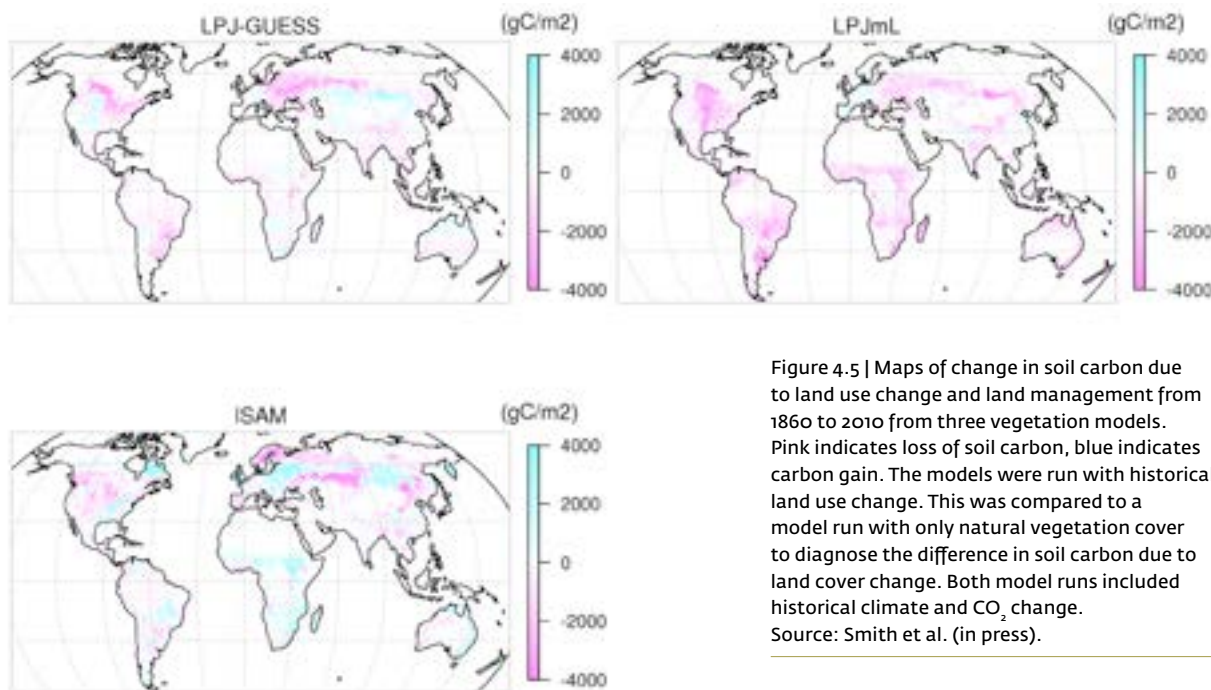


Figure 4.5 | Maps of change in soil carbon due to land use change and land management from 1860 to 2010 from three vegetation models. Pink indicates loss of soil carbon, blue indicates carbon gain. The models were run with historical land use change. This was compared to a model run with only natural vegetation cover to diagnose the difference in soil carbon due to land cover change. Both model runs included historical climate and CO₂ change. Source: Smith et al. (in press).

Panel (a) of Figure 4.5 shows cropland and pasture coverage in 2003. The model was run with historical land use change. This was compared to a model run with only natural vegetation cover to diagnose the difference in soil carbon due to land cover change up to year 2003 as shown in Panel (b). Both model runs included historical climate and CO₂ change. Pink indicates loss of carbon due to land use, blue indicates areas of carbon gain.

Table 4.1 | Soil carbon lost globally due to land use change over the period 1860 to 2010 (PgC)

Model	Tropical	Temperate	Boreal	Global
LPJ-GUESS	12.63	15.01	0.37	29.85
LPJmL	34.86	25.99	0.05	61.86
ISAM	17.24	37.83	5.28	60.35
Mean	21.57666667	26.27666667	1.9	50.68666667

Data are from three vegetation models ISAM (Jain *et al.*, 2013; El-Masri *et al.*, 2013; Barman, Jain and Liang, 2014 a, b); LPJ-GUESS (Smith *et al.*, 2001; Pugh *et al.*, 2015); and LPJmL (Bondeau *et al.*, 2007; Schaphoff *et al.*, 2013). Each model is run with and without historical land use change data and the difference between the 'with land use change' and 'no land use change' runs gives the loss due to land use change. The runs also included historical climate and CO₂ and cover the period from 1900 to 2010.

Impacts of land management and degradation

Logging and fire are the major causes of forest degradation in the tropics (Bryan *et al.*, 2013). Logging removes nutrients. Logging operations also cause soil disturbance affecting soil physical properties and nutrient levels (soil and litter) in tropical (e.g. Olander *et al.*, 2005; Villela *et al.*, 2006; Alexander, 2012) and temperate forests (Perez *et al.*, 2009). Many physical, chemical, mineralogical, and biological soil properties can be affected by forest fires depending on fire regime (Certini, 2005). Increased frequency of fires contributes to degradation and reduces the resilience of the biomes to natural disturbances

A meta-analysis of 57 publications (Nave *et al.*, 2011) showed that fire had significant overall effects on soil C (-26 percent) and soil N (-22 percent). Fires reduced forest floor storage (pool sizes only) by an average of 59 percent (C) and 50 percent (N), but the concentrations of these two elements did not change. Prescribed fires caused smaller reductions in C and N storage (-46 percent and -35 percent) than wildfires (-67 percent and -69 percent). Burned forest floors recovered their C and N pools in an average of 128 and 103 years, respectively. Among mineral soil layers, there were no significant changes in C or N storage, but C and N concentrations declined significantly (-1 percent and -12 percent, respectively). Mineral soil C and N concentrations were significantly reduced in response to wildfires but not after prescribed burning.

A large field study in the Amazon (225 forest plots) examined the effects of anthropogenic forest disturbance (selective logging, fire, and fragmentation) on soil carbon pools. Results showed that the first 30 cm of the soil pool did not differ between disturbed primary forests and undisturbed areas of forest, suggesting a resistance to impacts from selective logging and understory fires (Berenguer *et al.*, 2014). However, impacts of human disturbances on the soil carbon are of particular concern in tropical forests growing on organic soils.

Forest fires produce pyrogenic carbonaceous matter (PCM), which can contain significant amounts of fused aromatic pyrogenic C (often also called black C), some of which can be preserved in soils over centuries and even millennia. This was found to be the reason for similar soil organic C contents modelled for scenarios with and without burning in Australia: the loss in litter C input by fire was compensated by the greater persistence of the pyrogenic C (Lehmann *et al.*, 2008). Dissolved pyrogenic carbon (DPyC) from burning of the Brazilian Atlantic forest continued to be mobilized from the watershed each year in the rainy season, despite the fact that widespread forest burning ceased in 1973 (Dittmar *et al.*, 2012). Fire events are a source of carbonaceous aerosol emissions, and these are considered a major source of global warming (Kaufman, Tanre and Boucher, 2002)

Shifting cultivation practices of clearing land through fire have been used for thousands of years but in recent years increasing demographic pressure has often reduced the duration of the fallow period and so affected system sustainability. A review by Ribeiro Filho, Adams and Sereni Murrieta (2013) reported negative impact on SOC associated with the conversion stage, although impacts depended on the characteristics of the burning. Chop-and-mulch of enriched fallows appears to be a promising alternative to slash-and-burn. A study in the Amazon (Comtea *et al.*, 2012) found that this technique conserves soil bulk density and significantly increases nutrient concentrations and organic matter content compared to burnt cropland and to a control forest.

Climate change and land use dynamics are the major drivers of dryland degradation with important feedbacks through changes in plant community composition – for example shrub encroachment or decrease in vegetation cover (D’Odorico *et al.*, 2013). A review conducted by Ravi *et al.* (2010) indicated soil erosion as the most widespread form of land degradation in drylands, with wind and water erosion of dryland soils accounting for 87 percent of the land degradation. Grazing pressure, loss of vegetation cover, and the lack of adequate soil conservation practices increase the susceptibility of these soils to erosion. An analysis of 224 dryland sites highlighted a negative effect of aridity on the concentration of soil organic C and total N, but a positive effect on the concentration of inorganic P (Delgado-Baquerizo *et al.*, 2013). Because aridity is negatively related to plant cover, the authors argue that these effects might be related to the dominance in arid areas of physical processes such as rock weathering, a major source of P to ecosystems, over biological processes that provide more C and N, such as litter decomposition.

Grasslands, including rangelands, shrublands, pastureland, and cropland sown with pasture and fodder crops, covered approximately 3.5 billion ha in 2000. This represented 26 percent of the global ice-free land area and 70 percent of the agricultural area, and contained about 20 percent of the world’s soil organic carbon (C) stocks. Portions of the grasslands on every continent have been degraded due to human activities – about



7.5 percent of grassland worldwide has been degraded because of overgrazing (Conant, 2012). Grassland management and grazing intensity can affect the stock of SOC. A multifactorial meta-analysis of grazer effects on SOC density (17 studies that include grazed and ungrazed plots) found a significant interaction between grazing intensity and grass type. Specifically, higher grazing intensity was associated with increased SOC in grasslands dominated by C_4 grasses (increase of SOC by 6–7 percent), but with lower SOC in grasslands dominated by C_3 grasses (decrease of SOC by an average 18 percent). Impacts of grazing were also influenced by precipitation. An increase in mean annual precipitation of 600 mm resulted in a 24 percent decrease in grazer effect size on finer textured soils, while on sandy soils the same increase in precipitation produced a 22 percent increase in grazer effect on SOC (McSherry and Ritchie, 2013).

4.3.2 | Land use intensity change

Land use intensity has increased in recent decades, largely driven by the need to feed a growing population, by shifts in dietary patterns towards more meat consumption, and by the growing production of biofuels. At the same time, fast urbanization has occupied more of the land, reducing the stock available for agricultural production. Intensification has been widely advocated because of the many negative environmental consequences of clearing natural ecosystems to expand agricultural areas.

However, intensifying management practices, such as fertilization, irrigation, tillage and increased livestock density, can have negative environmental impacts (Tilman *et al.*, 2002). Intensifying land use can potentially reduce soil fertility. Intensification can also reduce soil resilience to extreme weather under climate change, to pests and biological invasion, to environmental pollutants and to other disasters. This section provides an overview of the benefits and consequences of intensifying use of agricultural lands. The section also highlights examples of how negative consequences can be minimized.

Several factors influence the increase in land use intensity during the recent decades. On the demand side, three main factors are at play: (I) the need to meet the food, fibre, and fuel demands of a growing population; (II) an increase in meat consumption as developing nations become wealthier and tastes change; and (III) rising demand for crops for biofuels. On the supply side, settlements are occupying more land and so reducing the land available for agriculture.

To meet the increased demand, it is estimated that food production will need to increase by 70–100 percent by 2050 (World Bank, 2008; Royal Society of London, 2009; Keating *et al.*, 2014). Of the two pathways of increasing production—intensification and expansion—intensification is widely promoted as the more sustainable option because of the negative environmental consequences of land expansion through deforestation and conversion of wetlands to cultivation (Foley *et al.*, 2011; MA, 2005). However, the current increase in land use intensity is generally not sustainable. In order to give a clear picture of the effects of increased land use intensity, this section is organized according to the primary management practices that characterize intensification of agricultural lands (see Table 4.2 for summary).

Nutrient management

Nutrient inputs, from both natural and synthetic sources, are needed to sustain soil fertility and to supply the nutrient needs of higher yielding crop production. Intensification in recent years has led to the annual global flows of nitrogen and phosphorus now being more than double the natural levels (Matson *et al.*, 1997; Smil, 2000; Tilman, 2002). The trend is still increasing – in China, for example, N input in agriculture in the 2000s was more than double the levels of the 1980s (State Bureau of Statistics-China, 2005). Nutrient management is particularly intensive in greenhouse production systems. In some parts of Asia, for example, up to six tons of chemical nutrient and hundreds tons of organic fertilizers are applied per hectare each year in order to achieve high yielding multiple cropping of vegetables (Liu *et al.*, 2008). Between 50–60 percent of



the nutrient inputs remain in the croplands after harvest (West *et al.*, 2014). When these nutrients are later mobilized, they become a major source of pollution to local, regional and coastal waters (Carpenter *et al.*, 1998). Intensive nutrient input in agriculture has been shown to be a major cause of eutrophication and algae blooming in lakes and inshore waters. In addition, over-use of nitrogen chemical fertilizers has been found in many locations globally to be a cause of acidification and accelerated decomposition of soil organic matter, leading to further soil degradation in over-fertilized soils (Ju *et al.*, 2009; Tian *et al.*, 2012).

Nutrient inputs also affect the earth's climate. Globally, approximately one percent of nitrogen additions are released to the atmosphere as nitrous oxide (N₂O), a gas which has 300 times the warming power of carbon dioxide (Klein Goldewijk and van Drecht, 2006). China, India, and the United States account for ~56 percent of all N₂O emissions from croplands, with 28 percent originating from China alone (West *et al.*, 2014).

One remedy is to increase the efficiency of nutrient use. Nutrient efficiency can be significantly increased – and N₂O emissions can be reduced – through changes in the rate, timing, placement, and type of application of nutrients, and by improving the balance amongst nutrients applied (Venterea *et al.*, 2011). In addition, if best management practices are used, agricultural soils have the potential to be carbon storage areas (Paustian *et al.*, 2004; Smith, 2004). Technological improvements are being made to the production of biochar which converts a fraction of the C present in the original material into a more persistent form through carbonisation. Biochar can then be used as a soil amendment to provide agronomic and environmental benefits (Lehmann and Joseph, 2015). In many cases, the presence of biochar has caused a reduction in N₂O emissions, especially when these originate from denitrification. However, the mechanics of the process are not yet fully understood (Cayuela *et al.*, 2013; 2014).

The effect of pesticides on soil biodiversity

The large-scale use of pesticides may have direct or indirect effects on soil biodiversity. With the intensification of agriculture, the use of pesticides has increased worldwide to approximately two million tonnes per year (herbicides 47.5 percent, insecticides 29.5 percent, fungicides 17.5 percent, other 5.5 percent by De *et al.* (2014)). Studies of the effect that pesticides have on soil biodiversity have shown contradictory results. Effects are dependent on a variety of factors including the chemical composition, the rate applied, the buffering capacity of the soil, the soil organisms in question, and the time-scale. For example, Boldt and Jacobsen (1998) tested the effects of sulfonylurea herbicides on strains of fluorescent pseudomonads cultured from agricultural field soils. They found that the herbicide Metsulfuron methyl was toxic to the majority of fluorescent pseudomonads (77 strains) in low concentrations, while Chlorsulfuron was only toxic at high concentrations, and Thifensulfuron methyl was toxic only to a few strains, even at high concentrations.

In a review by Bünemann, Schwenke and Van Zwieten (2006) of the effects of pesticide application on soil organisms, there were no data available for 325 of 380 active constituent pesticides registered for use in Australia. The review thus effectively highlighted the huge gap in knowledge. A synthesis of the impact of herbicides on non-target organisms concluded that herbicides did not have a major effect on soil organisms (Bünemann, Schwenke and Van Zwieten, 2006) with the exception of butachlor, which was toxic to earthworms when applied at typical agricultural rates (Panda and Sahu, 2004). In addition, the application of bromoxynil herbicides caused a shift in the communities of four out of five targeted bacterial taxa even after degradation of the herbicide (Baxter and Cummings, 2008). Avoidance behaviour to phendimethipam has also been observed for collembola (Heupel, 2002) and earthworms (Amorim, Rombke and Soares, 2005).

Insecticide application, however, has a much greater effect on soil biota, including changes in microbial community composition (Pandey and Singh, 2004), lower collembolan abundance (Endlweber, Schädler and Scheu, 2005) and earthworm reproduction. Because some species of earthworm such as *Eisenia Fetida* can be easily bred and because they ingest large quantities of organic matter in the soil, earthworms have often been

used as bioindicators of chemical toxicity in soils (Yasmin and D'Souza, 2010). A variety of studies have reported changes in earthworm reproductive rates, growth rates and weight loss when the pesticides Malathion (Espinoza-Navarro and Bustos-Obregon, 2005), Chlorpyrifos (Zhou *et al.*, 2007; De Silva *et al.*, 2010), Benomyl (Römbke, Garcia and Scheffczyk, 2007), Carbofuran (De Silva *et al.*, 2010) were applied to soil in laboratory experiments. Non-target effects of insecticide applications may be highly dependent on the organism since field application of Chlorpyrifos did not affect the abundance of soil predatory mites (Navarro-Campos *et al.*, 2012). Fungicides have also demonstrated significant negative effects on earthworms (Eijsackers *et al.*, 2005). In particular, copper-based fungicides that are resistant to degradation have caused long-term reductions in earthworm populations (Van Zwieten *et al.*, 2004).

Although an assessment of soil food webs across Europe did not specifically focus on pesticide application, the study demonstrated that land-use intensification was related to decreased diversity of soil fauna and resulted in less diversity among functional groups. Larger soil animals showed the most sensitivity (Tsiafouli *et al.*, 2015). However, there have been no such comprehensive studies to quantify the effects of pesticides on soil organisms at multiple trophic levels across regions. Such studies need to consider also the indirect effect of pesticides, including interactions between pesticides and biotic factors. Since below-ground biodiversity is intimately linked to above-ground vegetation patterns (De Deyn and van der Putten, 2005) and vice versa (Bardgett and van der Putten, 2014), changes in plant diversity resulting from herbicide may cause indirect effects of herbicide application.

Water management

The area of irrigated croplands has doubled in the last 50 years and irrigation now accounts for 70 percent of all water diversions on the planet (Gleick, 2003). Irrigated areas account for 34 percent of crop production, yet only cover 24 percent of all cropland area (Siebert and Doll, 2010). With the increased frequency of drought under climate change, demand for agricultural water is rising in many locations. Not surprisingly, irrigation is most commonly used in more arid areas. Where a high proportion of available water is used for agriculture, this can cause water stress for both people and nature. Water efficiency can be improved through infrastructure and through better management practices. Irrigation can potentially increase soil salinity in dry regions (Ghassemi, Jakeman and Nix, 1995). Where salinization occurs, additional irrigation is needed to 'flush' the salts beyond the root zone of the crops. This additional water requirement can further exacerbate water stress.

Harvest frequency

Land use intensity can also be increased by harvesting a parcel of farmland more frequently (double cropping, triple cropping). Approximately 9 percent of crop production increases from 1961-2007 came from increases in the harvest frequency (Alexandratos and Bruinsma, 2012). As more land was double cropped, the global harvested area increased four times faster than total cropland between 2000 and 2011 (Ray and Foley, 2013). In addition, with global warming, the areas suited for double or even triple cropping are extending into subtropical and warm temperate regions (Liu *et al.*, 2013a). The factors involved in this fast rate of increase include: fewer crop failures; fewer fallow years; and an increase in multi-cropping.

Greenhouse production has allowed multiple cropping around the world. For fruit and vegetable crops, world greenhouse cultivated area reached a total area of 408 890 ha in 2013, which includes as many as five harvests in a single year. This increasing harvest frequency has reduced soil quality through soil compaction and has increased the risk of pathogen diseases. The intensive use of pesticides and herbicides in greenhouses not only affects soil quality but creates risks to human health. In some greenhouse systems, long term multiple cropping has led to soil acidification, salinization and biological deterioration, especially where large amounts of fertilizer and pesticide/herbicide have been used. In these situations, there is a need to improve management practices, using organic matter, balancing nutrient additions and adopting intermittent fallow.



Livestock density

Livestock production is projected to increase to meet the growing demand for livestock products from a rising population and from an increase in per capita consumption. The greatest increases in per capita demand are projected to be in developing and transition countries (Bouwman *et al.*, 2006). Since the 1970s, most increases in livestock production have resulted from intensification, with a shift to a greater fraction of livestock raised in industrial conditions (Bouwmann *et al.*, 2006). For example, 76-79 percent of pork and poultry production is now industrialized (Herrero *et al.*, 2013).

Industrial livestock production systems can be highly polluting. The manure from animals, the inputs for growing animal feed, and the soil loss from intensively managed areas can all be major sources of water pollution to local and downstream freshwater ecosystems. Where natural ecosystems are cleared and converted to pasture, particularly in arid and semi-arid regions, the lands are typically low potential and have a high risk of soil erosion and soil carbon/nutrient depletion (Delgado *et al.*, 1999; Seré and Steinfeld, 1996). The soils capacity for water storage and their biodiversity are also at risk. Moreover, intensified livestock production requires an increased use of veterinary medicines, sulfa-antibiotics and hormones, all of which carry risks of pollution to soil, water and the livestock products themselves, with risks to biological and human health.

Forestry harvest and wetland draining

Forests and wetlands and their soils are massive reservoirs of carbon. In fact, forest soils store approximately the same amount of carbon as the living biomass of the forest itself (FAO, 2010). Wetlands are important not only for the huge carbon pool they contain but also for their role in the hydrological cycle. However, wetlands along big river banks, lakes and estuaries have been increasingly developed for croplands/bioenergy production in recent decades, particularly in Asia. The majority of soil carbon is concentrated in peatlands within the boreal forest as well as tropical forests in Southeast Asia. Around the world, deforestation causes ~25 percent of the total loss of soil carbon (Guo and Gifford, 2002; Murty *et al.*, 2002). This loss largely stems from oxidation of the organic matter and from soil erosion. In China over the last four decades, almost 1.3 million ha of wetlands have been converted to crop production, causing the loss of about 1.5 Pg C of soil carbon (Zhang *et al.*, 2009). Deforestation continues through conversion to agriculture and through extraction of forest products. Between 2000 and 2012, there was a net loss of 1.5 million square kilometres of forests, with the most pronounced trend in the tropics (Hansen *et al.*, 2013). Soil erosion and organic matter oxidation can be reduced through selective tree harvesting rather than clear felling, and by avoiding deforestation on steep slopes. Draining and cultivating wetlands can also affect local and regional water storage.



Research needs

It will be evident from the discussion in this section that much remains to be learned. Amongst the priority research questions are the following:

1. *Sustainable intensification* – How can we get the benefits from intensification while minimizing the associated environmental and social costs?
2. *Trade-offs between soils and efficiency* – How can we manage for resilient soil and related ecosystem services while continuing to maximize efficiency? To what extent can we have both?
3. *Soil degradation and intensification* – What is the extent of degraded soils? There are currently no sound estimates. What portion of degraded soils can be attributed to un-sustainable intensification?
4. *Options and trade-offs for improved soil management* – What can we learn from management practices used in intensification areas to help restore degraded soils? Are there any options that can integrate best management practice for sustainable intensification? What are the short – and long-term trade-offs of resource use and sustainability? What are the environmental and social costs and economic benefits of land use intensification?
5. *Farming practices and soil health* – How do changes in harvest frequency and crop rotation affect soil resilience? How much change is needed to restore degraded soils?

Table 4.2 | Threats to soil resource quality and functioning under agricultural intensification

Land intensification	Sector	Distribution	Major environmental consequence	Knowledge gap
Cropping intensification	Harvest frequency	Globally	Soil quality and resilience	Ecosystem service
	Continuing monoculture	Developing and transition countries	Soil health, pesticide residue	Biological resilience
Nutrient intensification	Over fertilization	Developing countries	Soil acidification, water pollution, N ₂ O emission and nitrate accumulation	Rate reducing versus balancing?
Irrigation	Submerged Rice	Developing countries, Asia	Water scarcity, methane emission	Trade-offs C and water,
	Dry crops	Arid/semi-arid regions	Secondary salinization, water scarcity	Competition for water
Livestock intensification	Over grazing	Developing countries	Soil degradation, water storage, C loss	Forage versus feed crops?
	Industrial breeding	Industrialized countries	Waste, water pollution, residue of veterinary medicine and antibiotics	Safe waste treatment and recycling
Forest clearance, wetlands drainage	Deforestation, wetland shrink	Developing and transition countries	Biodiversity, natural wealth, C loss	Agro-benefit versus natural value



4.3.3 | Land use change resulting in irreversible soil change

In this section we deal with soil sealing and mining, which have been identified as two important soil degradation processes occurring around the world. The current extent and rate of growth of soil sealing and mining are significant, and create considerable risks to essential ecosystem services. These changes in land use nearly always require a trade-off between various social, economic and environmental needs.

Sealing and land take

The ongoing urbanization and conversion of the landscape with settlements, infrastructure and services is occurring in many regions. Europe and Asia, in particular, are experiencing high rates of urban expansion and urban sprawl, and there are often insufficient incentives to re-use brownfield sites. These factors are causing an increase in land take and soil sealing. The drivers are essentially economic and demographic growth. In Europe, America and Oceania, at least 70 to 80 percent of the population currently lives in urban areas. The rate of urbanization is expected to continue to increase, particularly in Asia and Africa.

The concept of **land take** covers all forms of conversion for the purpose of settlement, including: the development of scattered settlements in rural areas; the expansion of urban areas around an urban nucleus; the conversion of land within an urban area (densification); and the expansion of transport infrastructure such as roads, highways and railways. Broadly, this discussion considers as land take any conversion of agricultural, natural or semi-natural land cover to an 'artificial' (e.g. human-made) area. Artificial land cover classes are categorized in the Corine Land Cover system – see Table 4.3.

A greater or smaller part of land take will result in soil sealing. **Soil sealing** means the permanent covering of an area of land and its soil by impermeable artificial material such as asphalt or concrete, for example through buildings and roads. As shown in Figure 4.6, the sealed area is only part of a settlement area. Gardens, urban parks, leisure areas and other green spaces within the boundaries of settlements are not covered by an impervious surface or are only partially covered. They thus form part of a land take but do not contribute to soil sealing (Prokop, Jobstmann and Schöbauer, 2011.) The ratio between sealed area and total area for a given land use class is measured by the *soil sealing* index. An example of this index, calculated for the Italian region of Emilia-Romagna, is shown in Table 4.4.

Table 4.3 | Artificial areas in Corine Land Cover Legend

Corine CODE	LABEL1	LABEL2	LABEL3
111	Artificial surfaces	Urban fabric	Continuous urban fabric
112	Artificial surfaces	Urban fabric	Discontinuous urban fabric
121	Artificial surfaces	Industrial, commercial and transport units	Industrial or commercial units
122	Artificial surfaces	Industrial, commercial and transport units	Road and rail networks and associated land
123	Artificial surfaces	Industrial, commercial and transport units	Port areas
124	Artificial surfaces	Industrial, commercial and transport units	Airports
131	Artificial surfaces	Mine, dump and construction sites	Mineral extraction sites
132	Artificial surfaces	Mine, dump and construction sites	Dump sites
133	Artificial surfaces	Mine, dump and construction sites	Construction sites
141	Artificial surfaces	Artificial, non-agricultural vegetated areas	Green urban areas
142	Artificial surfaces	Artificial, non-agricultural vegetated areas	Sport and leisure facilities





A) Typical structure of settlement

B) Sealed areas about 70 percent (black color)

Figure 4.6 | Schematic diagram showing areas sealed (B) as a result of infrastructure development for a settlement (A).
Source: European Union, 2012.

Table 4.4 | Artificial areas in Emilia Romagna according to the Corine Land Cover Legend and sealing index.

Classes of artificial areas	Sealing Index
1.1.1.1 Dense residential areas	0.95
1.1.1.2 Loose residential areas	0.7
1.1.2.0 Discontinuous urban areas	0.3
1.2.1.1 Industrial and agro-industrial productive districts	0.75
1.2.1.2 Commercial districts	0.75
1.2.1.3 Service and tertiary districts	0.75
1.2.1.4 Hospitals	0.75
1.2.1.5 Large technological plants	0.75
1.2.2.1 Roads and accessory areas	0.75
1.2.2.2 Railroads and accessory areas	0.25
1.2.2.3 Logistic centers	0.9
1.2.2.4 Telecommunication plants	0.25
1.2.2.5 Areas for production, transformation and transport of energy	0.75
1.2.2.6 Areas for water treatment and distribution	0.25
1.2.3.1 Commercial harbours	0.2
1.2.3.2 Touristic harbours	0.2
1.2.3.3 Fishing harbours	0.15
1.2.4.1 Commercial airports	0.3
1.2.4.2 Leisure airports	0.15
1.2.4.3 Military airports	0.2
1.3.1.1 Active mining areas	0.1
1.3.1.2 Inactive mining areas	0.1
1.3.2.1 Mining and industrial dumping sites	0.1
1.3.2.2 Urban waste disposals	0.1
1.3.2.3 Car cemeteries and scra yards	0.3
1.3.3.1 Building and escavation sites	0.05
1.3.3.2 Derelict parcels	0.1
1.4.1.1 Parks and villas	0.1
1.4.1.2 Uncultivated urban areas	0.1
1.4.2.1 Campsites	0.2
1.4.2.2 Sportive areas	0.22
1.4.2.3 Leisure areas	0.3
1.4.2.4 Golf courses	0.05
1.4.2.5 Racecourses	0.05
1.4.2.6 Racetracks	0.2
1.4.2.7 Archeological areas	0.1
1.4.2.8 Bathing areas	0.15
1.4.3.0 Cemeteries	0.4

Impact of land take

Land take, by its definition, is the subtraction of an area from a previous agricultural, natural or semi-natural land use. According to this definition, the most obvious impact on the ecosystem services that can be provided by soil is on the production of biomass, and in particular of food. To clarify the concept, we may imagine that a city expands its urbanized area by a new allotment of 100 ha created at the expense of agricultural land. This area will be covered by buildings, private and public gardens, commercial centres, roads, etc. The entire area will clearly lose most of its capacity to produce food, with the possible minor exception of family horticulture in unsealed areas such as gardens or allotments. Had the entire area been previously cultivated with, say, winter wheat with an average yield of 5 tonnes ha⁻¹, the total loss in terms of food production potential will be equal to 500 tonnes of winter wheat per year.

Other ecosystem services are at risk also. Water infiltration and purification and carbon storage are mainly reduced by the effective sealed area, and not by the entire land taken. Support to biodiversity is clearly affected, although the degree depends on the different groups of organisms and also on the design of the urbanized area. In this context, a positive mitigation role can be played by 'Urban Green Infrastructure' – the incorporation of a network of high-quality green spaces and other environmental features. Green Infrastructure can include natural areas as well as human-made rural and urban elements such as urban green spaces, reforestation zones, green bridges, green roofs, eco-ducts to allow crossing of linear barriers, corridors, parks, restored floodplains, biodiverse farmland.

Regulation of land take and mitigation of its impacts

Where policy aims to minimize land take, measures can be implemented to encourage re-use of existing urban areas such as derelict areas, brownfields and upgrading of degraded neighborhoods. Measures promoting densification of existing urban areas can also contribute to the reduction of land take.

Fiscal measures can prevent speculative urban sprawl. A number of municipalities, and regional governments, especially in Europe, have already adopted policies designed to achieve zero net urban expansion. However, zero expansion becomes more problematic when there is significant demographic pressure and a high rate of rural to urban migration.

Rational and efficient urban planning and intelligent building and infrastructure design can also help reduce land take. In the past, urban planners, architects and civil engineers too often considered soil as a raw material, abundantly available and of limited value. Examples of efficient consideration of the value of soil in urban development include: the construction of parking lots in the basement of buildings; and 'green' covering of areas that are only occasionally used, such as parking lots for exhibitions and fairs etc.

Where expansion of urban and built-up areas is a policy and planning imperative, intelligent urban planning needs to take account of the soil dimension to mitigate the impact of land take. An education process is needed to make urban planners aware of the value of soil quality and land capability and of the options for mitigating negative impacts of land take.

Impacts of soil sealing

Sealing by its nature has a major effect on soil, diminishing many of its benefits. Normal construction practice is to remove the upper layer of topsoil, which delivers most of the soil-related ecosystem services, in order to be able to develop strong foundations in the subsoil or underlying rock to support the building or infrastructure. Where strong foundations are not required, only a thin layer of topsoil is generally excavated and the surfaces are simply covered by a layer of impervious material, such as asphalt or concrete. Both techniques impair or eliminate the soil's capacity to deliver ecosystem services.

The main impacts include the following.

1. Water infiltration and purification are lost, and regulation of the water cycle is completely altered. The concentration time of water flow is shortened, promoting flood events.
2. Soil biodiversity is impaired, as sealing prevents the production, release and recycling of organic material, so affecting the soil biological communities (Marfenina *et al.*, 2008). In addition, the alteration of soil water regimes, soil structure and redox potential have a strong impact on soil biodiversity.
3. Soil carbon storage potential is fundamentally altered (Jones *et al.*, 2005), particularly where topsoil, which normally contains about half of the organic carbon in mineral soils, is stripped off.
4. The urban microclimate is altered. The reduction of evapotranspiration in urban areas due to the loss of vegetation and through alteration of albedo strengthens the 'urban heat island' effect (Früh *et al.*, 2011).

Prevention of soil sealing and mitigation of its impacts

Appropriate mitigation measures can be taken in order to maintain some of the ecosystem functions of soils and to reduce negative effects on the environment and human well-being. Key options available to urban planners and managers include: (I) minimizing conversion of green areas; (II) re-use of already built-up areas, such as brownfield sites; (III) using permeable cover materials instead of concrete or asphalt; (IV) supporting Green Infrastructure (see above); and (V) providing incentives to developers to minimize soil sealing.

In practice, planners need to be able to evaluate the tradeoffs and ensure that policy instruments are used to ensure optimal outcomes which consider both human needs for urbanization and the preservation of the integrity of the soil and its services:

1. Existing policies for development of settlements and infrastructure should be reviewed and adapted to take account of the value of soils, particularly where subsidies or other incentives are driving unplanned land take and soil sealing (Prokop, Jobstmann and Schöbauer, 2011).
2. An integrated approach to urban planning should be followed. Existing best practice has demonstrated that soil sealing can be limited, mitigated and compensated. This requires that spatial planning follow an integrated approach and involve the full commitment of all relevant public authorities and governance entities responsible for land management, such as municipalities, counties and regions (Siebielec *et al.*, 2010).
3. Specific regional and local approaches can be developed. These could, for example, take into account unused resources at the local level such as a particularly large number of empty buildings or brownfield sites.

Mining

Ancient mining

Mining is the extraction from the Earth of rocks, valuable minerals, and other geological materials of economic interest. It is one of the most ancient activities in human history (Mighall *et al.*, 2002; Shotyk *et al.*, 1998). Mining for specific materials such as quartz, silex and clays began as far back as the Palaeolithic – the Old Stone Age – when the first stone tools were developed. In the Neolithic era – the New Stone Age – flint mines existed in Belgium, Britain and elsewhere. Landscape records and evidence from bogs show that mining activities became more intense with the development of metal tools in the Bronze Age, and subsequently in the Iron Age (Martínez-Cortizas *et al.*, 2002; Shotyk *et al.*, 1998). Examples of the environmental impact of ancient mining are numerous (Figure 4.7) (López – Merino *et al.*, 2010; Grattan, Huxley and Pyatt, 2003; Fernández Caliani, 2008).



Figure 4.7 | (A) Panoramic view of Las Medulas opencast gold mine (NW Spain). The Roman extractive technique – known as ‘ruina montis’ – involved the massive use of water that resulted in important geomorphological changes; (B) Weathered gossan of the Rio Tinto Cu mine, considered the birthplace of the Copper and Bronze Ages; (C) typical colour of Rio Tinto (‘red river’ in Spanish), one of the best known examples of formation of acid mine waters. These are inhabited by extremophile organisms.

Impact of mining

The impact of mining on the environment differs greatly depending on the type of extraction, the ore or material exploited, and the method used to process the material extracted (Moore and Luoma, 1990). Traditional underground mining, which follows profitable veins beneath the earth’s surface, has less impact than open cast mining activities – also referred to as strip mining – which grew very rapidly in the last hundred years (Salomons, 1995). In some instances, entire mountains have been literally blasted apart to reach thin ore vein seams within, leaving permanent scars on the landscape. Nonetheless, mining operations themselves affect relatively small areas. By contrast, significant environmental problems are caused by tailing and waste rock deposits and by subsequent smelting operations. Pollutants can be transferred to surrounding areas by acid mine drainage or by atmospheric deposition of wind-blown dust. The incidence of these problems depends on local climatic and hydrologic conditions (Aslibekian and Moles 2003; Batista, Abreu and Serrano, 2007; López, González and Romero, 2008). Other environmental effects, in addition to those caused by pollutants, include deforestation, erosion and formation of sinkholes (Meuser, 2010; Hester and Harrison, 2001).

Only a small fraction of the material extracted is valuable ore. The ore needs to be separated by milling and flotation from the large volume of other material discarded as tailings. When the remaining concentrate is refined by processes such as smelting, flue dust and slag are produced (Hutchinson, 1979). Atmospheric contamination has commonly occurred throughout the world during smelting operations, leading to contaminated soils and risks to livestock (Down and Stocks, 1977; Munshower, 1977). Mining for coal, gold, uranium, wolfram, tin, platinoids and, in particular, poly-metallic sulphides has created large environmental impacts on soil, water and biota. Sulphide minerals include iron sulphides such as pyrite and pyrrhothite, and other poly-metallic sulphides, such as those containing Cu, Pb, Zn, Hg, Cd, Tl, Sb, Bi etc. These sulphides can also in some instances combine with arsenides or selenides to form sulfoarsenides or sulfoselenides (Evangelou, 1995; Abreu *et al.*, 2010).

Sulphide minerals oxidise when brought to surface conditions (Nordstrom and Southam, 1997; Nordstrom and Alpers, 1999). The sulphide oxidation can cause extreme changes in Eh and pH (Figure 4.8) – negative pH values (as low as -3.6) have been measured in the acid mine waters of the Richmond mine in California (Nordstrom and Alpers, 1999). Depending on the local geochemical and hydrological conditions, sulphide oxidation can also affect the electrical conductivity of the system and may lead to elevated concentrations of many toxic elements in soils and waters nearby. Waters downstream of these mine systems (Figure 4.7C) are frequently hyperacid, hyperoxidant and hyperconductive. These waters may exhibit high activities of: (I) various metal species such as Al^{+3} , $Al-SO_4$; (II) heavy metal species, for example Cu^{+2} , Cd^{+2} , Zn^{+2} , Hg^{+2} y Hg^0 ; and (III) metalloids, including arseniates, arsenites and seleniates (Sengupta, 1993; Macías, 1996; Monterroso, Alvarez and Macías, 1994; Monterroso *et al.*, 1998, 1999; Azcue, 1999). Smelting operations of sulphide minerals also generate SO_2 , which, if not recovered, is released into the atmosphere and thus contributes to acid deposition (described in Section 4.4).

The mining of gold deserves special attention given its contribution to Hg emissions (Drude de Lacerda, 2003). Mercury is used to concentrate the fine gold particles through amalgamation and then the gold is separated from the amalgam by applying heat. When this process is carried out under uncontrolled conditions – as in small-scale gold mining (Drude de Lacerda, 2003) – Hg volatilises to the atmosphere. Tailings from Hg amalgamation are then leached with cyanide, and waste contaminated with metals and cyanide is released into the environment (Veiga *et al.*, 2009). Arsenic exposure has also been recorded in many gold and base metal producing countries (Williams, 2001). However, arsenate and arsenite mobilisation can be controlled with soil colloidal compounds such as reactive Fe and Al (Goldberg, 2002).

As materials from mining are exposed to the environmental conditions of the Earth's surface, these minesoils develop through weathering (Sencindiver and Ammons, 2000). However, their properties differ considerably from the original soil. They contain a high percentage of rock fragments, a low nutrient content, and elevated levels of potentially harmful trace elements. They also usually lack a distinct horizonation. These soils are in fact very young soils characterised by properties that limit their functions and their capability to support vegetation (Macias, 1996; Vega *et al.*, 2004; Abreu and Magalhães, 2009). When the overburden contains sulphidic material such as pyritic mine waste, the major weathering process is the oxidative dissolution of pyrite. Here the rate of soil formation is mainly controlled by the sulphide content and its particle-size distribution, causing strongly acidic conditions, as described above (Neel *et al.*, 2003; Haering, Daniels and Galbraith, 2004). Quite often, restoration of mine soils requires the addition of exogenous material to correct the extreme pH, Eh and/or EC values and the anomalous concentrations of toxic elements common in these systems which are generally bioavailable and susceptible to mobilisation.

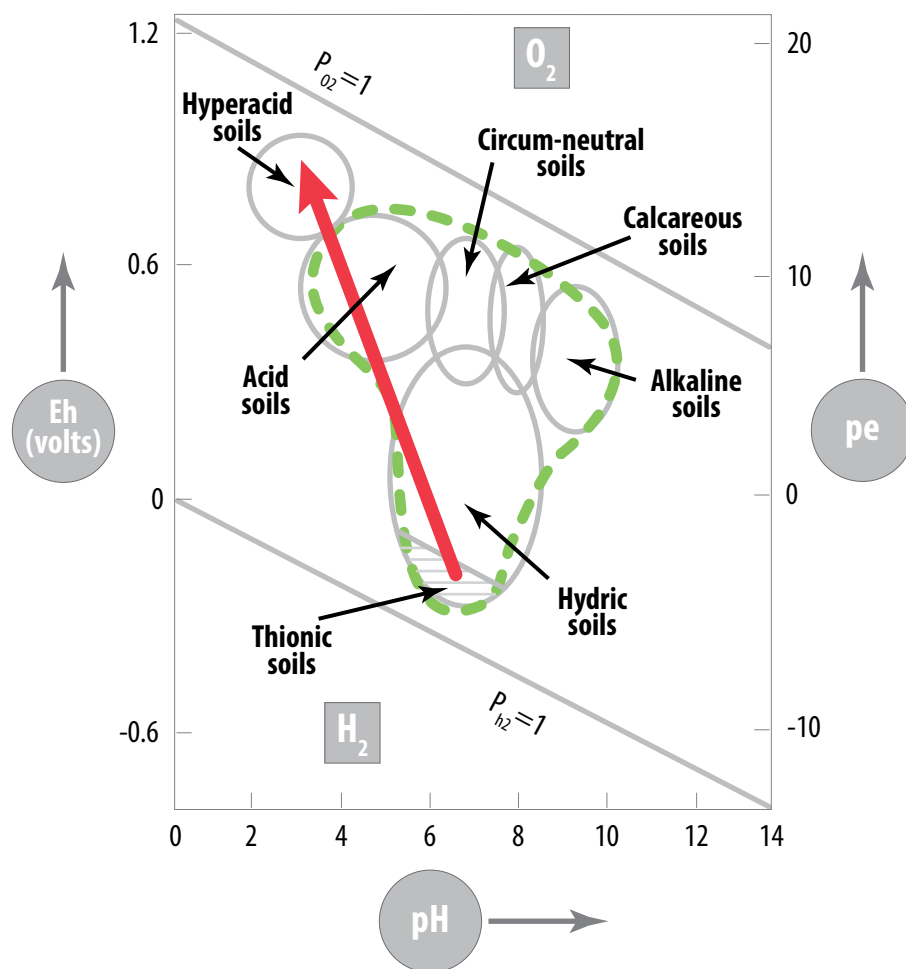


Figure 4.8 | Eh-pH conditions of thionic/sulfidic soils and of hyperacidic soils.
Source: Otero *et al.*, 2008.

The formation of sulfidic material requires strongly reducing conditions and slight acidity. Once these are oxidised, and in the absence of minerals with high acid buffering capacity, extremely acid and oxidising conditions are generated. The dashed envelope in Figure 4.8 is the approximate extent of redox-pH conditions of mineral soils (with the exception of hyper-acid soils).

Preventing impacts from mining

The rehabilitation of abandoned mines is a difficult and costly task. In fact, in many instances, the landscape cannot be repaired. Some mining methods may have significant environmental and public health effects. The Aznalcollar pyritic sludge spill (SW Spain) (López-Pamo *et al.*, 1999; Grimalt, Ferrer and McPherson, 1999; Aguilar *et al.*, 2004; Calvo de Anta and Macías, 2009) is such an example. It occurred in 1998 in the surroundings of Doñana Park – the largest reserve of bird species in Europe – as a result of the failure of a tailings dam which contained several million tons of pyrite stockpile, flotation tailings and acid waters. The toxic spill contaminated ca. 26 km² of riverbanks and adjacent farmlands, extending 45 km downstream, with an estimated quantity of 16 000 tonnes of Zn and Pb, 10 000 tonnes of As, 4 000 tonnes of Cu, 1 000 tonnes of Sb, 120 tonnes of Co, 100 tonnes of Tl and Bi, 50 tonnes of Cd and Ag, 30 tonnes of Hg, and 20 tonnes of Se.

Mining operations have a responsibility to protect the environment: air, water, soils, ecosystems and landscape. Many countries require reclamation plans for mining sites to follow environmental and rehabilitation codes. Nonetheless, mine restoration is still problematic, mainly because the environmental impacts were only recently understood or appreciated (Azcue, 1999; Sengupta, 1993). In addition, the technology available has not always been adequate to prevent or control environmental damage. Restoration of such systems requires a thorough understanding of material properties and their geochemistry. Only through such an understanding can the current and future behaviour of such systems be predicted and appropriate decisions taken to ensure their restoration (Gil *et al.*, 1990; Macías-García, Camps Arbestain and Macías, 2009; Macías-García *et al.*, 2009).

Development of tailor-made Technosols to restore mine soils

Technosols are defined by the FAO (2014b) as those soils with recent human activities in industrial and urban environments which have resulted in the presence of artificial and human-made objects. Technosols often result from the abandonment of urban, mining or industrial waste. These soils tend to have a large content of artefacts – that is objects that are either human-made, strongly transformed by human activity, or excavated (e.g. mine spoils, rubbles, cinders) (FAO, 2014b).

Throughout history, humans have formed soils – ‘anthropogenic soils’ – and in certain cases these soils have proved more fertile than natural soils nearby (Sombroek, Nachtergaele and Hebel, 1993). Thus, it is feasible to produce specific Technosols which can fulfil the environmental and productive functions of natural soils – essentially, tailor-made Technosols. This may require the formulation and mixing of artefacts and other waste materials such as manure and biosolids. The production of these Technosols could be a feasible technique through which waste products are reused and the elements they contain are returned to their biogeochemical cycles, while restoring degraded areas and contributing to the sequestration of C in soils and biomass (Macías and Camps Arbestain, 2010).

Environmental problems associated with this use of Technosols may be prevented if: (I) the characteristics of the materials used provide the soil with adequate buffering properties against contaminants, pH and/or redox changes; and (II) there is a good understanding of how the constituent mixtures will evolve over time under the pedoclimatic conditions of the area to be restored. Figure 4.9 illustrates the benefits of the use of tailor-made Technosols in the restoration of an abandoned Cu mine rich in pyrite (Macías-García, Camps Arbestain and Macías, 2009; Macías-García *et al.*, 2009; Macías and Camps Arbestain, 2010).



Figure 4.9 | Use of different Technosols derived from wastes in the recovery of hyperacid soils and waters in the restored mine of Touro (Galicia, NW Spain).

4.4 | Atmospheric deposition

4.4.1 | Atmospheric deposition

The impacts of the deposition of atmospheric pollutants on soils vary with respect to soil sensitivity to a specific pollutant and to the total pollutant load. Anthropogenic emissions of sulphur, nitrogen and trace elements to the atmosphere mainly derive from fossil fuel and waste combustion in, for example, power generation, incineration, industry and transport. Emissions may also derive from non-combustion processes such as agricultural fertilizers or waste amendments. Mining activities may also contribute, for example Hg mining. Once in the atmosphere, these pollutants can be transported off-site and even cross national borders before being deposited either as dry or wet deposition. Deposition is more accentuated in forests, especially in coniferous forests (because of reduced wind speeds) and in areas of high elevation because of high precipitation rates.

Once in the soil, pollutants can be mobilised by being: (I) released back to the atmosphere; (II) made available to biota; (III) leached out to surface waters; or (IV) transported to other areas by soil erosion. Pollutants disrupt natural biogeochemical cycles by altering soil functions. This disruption may come about through direct changes to the nutrient status, acidity, and bioavailability of toxic substances, or through indirect changes to soil biodiversity, plant uptake and litter inputs. Soil sensitivity to atmospheric pollution varies with respect to: (I) key properties influenced by geology and associated pedogenesis such as cation exchange capacity, soil base saturation, aluminium, or rate of base cation supply by mineral weathering); (II) organic matter content and carbon to nitrogen ratio (C:N); and (III) position of the water table. When atmospheric pollution is associated with sulphate deposition, the capacity of soils to adsorb sulphate (e.g. soils with a dominance of short-range ordered constituents) plays a key role in buffering the acidification process (Camps Arbostain, Barreal and Macías, 1999; Rodríguez-Lado, Montanarella and Macías, 2007). Harmful effects on soil function and structure occur where deposition exceeds the 'critical load' - the specific amount of one or more pollutants that a particular soil can buffer (Nilsson and Grennfelt, 1988). Estimates and mapping of critical loads of acidity

are however strongly dependent on the neutralisation mechanisms considered in the analysis, for example, the inclusion or exclusion of sulphate adsorption (Rodríguez-Lado, Montanarella and Macías, 2007). Spatial differences in soil sensitivity – commonly defined by the ‘critical load’ – and in pollutant deposition result in an uneven global distribution of impacted soils (Figure 4.10). For instance, global emissions of sulphur and nitrogen have increased 3–10 fold since the pre-industrial period (van Aardenne *et al.*, 2001), yet critical loads for acidification are only exceeded in 7–17 percent of the global natural terrestrial ecosystems area (Bouwman *et al.*, 2002).

4.4.2 | Main atmospheric pollutants: Synopsis of current state of knowledge

Since the 1980s, emissions of pollutants, notably sulphur, across Europe and North America have declined. The decline is due to the establishment of protocols under the 1979 *Convention on Long-range Transboundary Air Pollution* (LRTAP) and the 1990 United States Clean Air Act Amendments (CAAA) (Greaver *et al.*, 2012; Reis *et al.*, 2012; EEA, 2014). Conversely, emissions in South and East Asia, sub-Saharan Africa and South America are likely to increase in response to industrial and agricultural development (Kuylenstierna *et al.*, 2001; Dentener *et al.*, 2006). Further emission increases are also occurring in remote areas due to mining activity, such as oil sands extraction in Canada (Kelly *et al.*, 2010; Whitfield *et al.*, 2010).

Sulphur deposition

Sulphur emissions primarily result from combustion of coal and oil and are typically associated with power generation and heavy industry. In 2001, deposition exceedances of 20 kg S ha⁻¹ yr⁻¹ were detected in regions of China and Republic of Korea, Western Europe and eastern North America (Vet *et al.*, 2014; Figure 4.10.(a)). Deposition in unaffected ecosystems is <1 kg S ha⁻¹ yr⁻¹ (Figure 4.10a). The deployment of sulphur emission protocols led to the reduction of approximately 80 percent in the deposition levels of sulphur across Europe between 1990 and 2010 (Reis *et al.*, 2012). This reduction led to an increase in the use of sulphur fertilizer to combat crop sulphur deficiencies in agricultural soils in Europe (Bender and Weigel, 2011). Sulphur emissions in China also declined between 2005 and 2010 (Fang *et al.*, 2013).

Soil acidification is a natural process that is altered and accelerated by anthropogenic sulphur and nitrogen deposition (Greaver *et al.*, 2012). Sulphur oxide (SO_x) gases react with water vapour in the atmosphere to form sulphuric acid (H₂SO₄). Once in the soil, excess inputs of acidity (H⁺) displace base cations (e.g. calcium (Ca₂⁺) and magnesium (Mg₂⁺)) from soil surfaces to the soil solution, and the base cations are subsequently lost from the soil profile by leaching (Reuss and Johnson, 1986). In mineral soils, these base cation losses can be balanced by rock weathering or atmospheric dust deposition. Thus, the global distribution of acid sensitive soils is mainly associated with conditions that favour development of soils with low cation exchange capacity and base saturation (Bouwman *et al.*, 2002; Figure 4.10b). The exception is where soils are dominated by variable-charge constituents, as in the case of Acrisols, Ferrasols, Nitosols and Andosols. On these soils, sulphate adsorption may become the most important acid-buffering mechanism (Rodríguez-Lado, Montanarella and Macías, 2007). Wetlands can also buffer inputs of acidity through biological sulphate reduction, although acidity can be mobilised again following drought and drainage (Tipping *et al.*, 2003; Laudon *et al.*, 2004; Daniels *et al.*, 2008). Organic acids can also buffer acid deposition in naturally acidic organic soils (Krug and Frink, 1983; Monteith *et al.*, 2007).

Acidification decreases soil fertility due to loss of nutrients and increases the mobilisation of toxic metals, particularly Al and heavy metals. The negative effect of Al species on crop yield is particularly strong in soils with a dominance of 2:1 clay minerals with high CEC and low organic matter content. The atmospheric deposition of acid compounds had a huge impact on Scandinavian ecosystems over the 1960s–80s, including declines in freshwater fish populations and damage to forests (EEA, 2014). Sulphur inputs can also stimulate microbial processes that increase Hg bioavailability, leading to bioaccumulation of Hg in the food chain (Greaver *et al.*, 2012).

The increase in soil pH following the reduction of sulphur emissions shows that the acidification process is reversible, although the recovery time is highly variable and dependent on soil properties. Some areas with organic soils where deposition has declined are showing either slow or no recovery (Greaver *et al.*, 2012; Lawrence *et al.*, 2012; RoTAP, 2012). On agricultural soils, lime can be applied to increase soil pH. However, 50–80 percent of sulphur deposition on land is on natural land (Dentener *et al.*, 2006). Application of lime to naturally acidic forest soils can cause further acidification of deep soil layers by increasing the decomposition in surface litter (Lundström *et al.*, 2003). In acid waters, the addition of liming material may favour the formation of polymeric Al hydroxides (e.g. $Al_3(OH)_7^{+12}$), which are highly toxic to aquatic species (Monterroso, Alvarez and Macías, 1994).

Wider effects of acidification are starting to be understood through long-term monitoring. Decreased organic matter decomposition due to acidification has increased soil carbon storage in tropical forests (Lu *et al.*, 2014). In wetland soils, methane (CH_4) emissions have also been suppressed. This is because sulphate-reducing bacteria have a higher affinity for substrate (H_2 and acetate) than methanogenic microbes (Gauci *et al.*, 2004). Conversely, declining sulphur deposition has been associated with increased dissolved organic carbon fluxes from organic soils (Monteith *et al.*, 2007) and decreased soil carbon stocks in temperate forest soils (Oulehle *et al.*, 2011; Lawrence *et al.*, 2012).

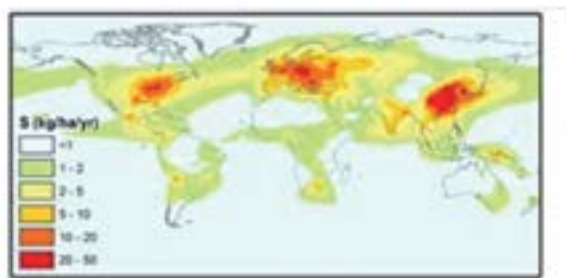
Nitrogen deposition

Nitrogen deposition covers a wider geographical area than sulphur deposition. This is because the sources are more varied, including extensive agriculture fertilizer and animal waste application, biomass burning, and fossil fuel combustion (Figure 4.10c). Regions with deposition in excess of $20 \text{ kg N ha}^{-1}\text{yr}^{-1}$ in 2001 include Western Europe, South Asia (Pakistan, India, Bangladesh) and eastern China (Vet *et al.*, 2014). In addition, extensive areas with deposits of $4 \text{ kg N ha}^{-1}\text{yr}^{-1}$ or more were found across North, Central and South America and parts of Europe and Sub-Saharan Africa. By contrast, 'natural' deposition in un-impacted areas is as little as $0.5 \text{ kg N ha}^{-1}\text{yr}^{-1}$ (Dentener *et al.*, 2006). While both nitrogen and sulphur emissions related to fossil fuel combustion have declined across Europe, agricultural sources of nitrogen in the region are likely to stay constant in the near future (EEA, 2014). At the same time, overall global emissions are likely to increase (Galloway *et al.*, 2008). Nitrogen deposition in China in the 2000s was similar to peaks in Europe during the 1980s before Europe embarked on mitigation measures (Liu *et al.*, 2013b).

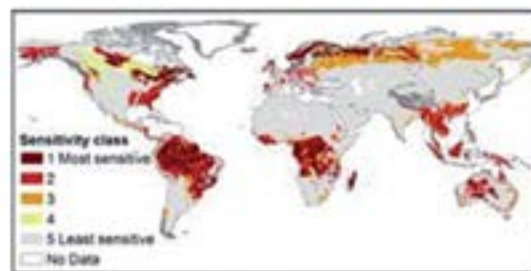
Deposition of nitrogen induces a 'cascade' of environmental effects, including acidification and eutrophication that can have both positive and negative effects on ecosystem services (Galloway *et al.*, 2003). Soils with low nitrogen content are most sensitive to eutrophication – typically Histosols, Cryosols and Podzols located in cold areas in northern countries such as northern Canada, Scandinavia and northern Russia (Bouwman *et al.*, 2002; Rodríguez-Lado, Montanarella and Macías, 2007; Figure 4.10d). Excluding agricultural areas where nitrogen deposition is beneficial, 11 percent of the world's natural land experiences nitrogen exceedances above $10 \text{ kg N ha}^{-1}\text{yr}^{-1}$ (Dentener *et al.*, 2006). In Europe, eutrophication has and will continue to impact a larger area than acidification (Rodríguez-Lado and Macías, 2005; EEA, 2014).

Nitrogen fertilisation can increase tree growth (Magnani *et al.*, 2007) and cause changes in plant species and diversity (Bobbink *et al.*, 2010). This can in turn alter the amount and quality of litter inputs to soils, notably the C:N ratio and soil-root interactions (RoTAP, 2012). However, increased global terrestrial carbon sink can be largely offset by increased emissions of the greenhouse gases N_2O and CH_4 (Liu and Greaver, 2009). Long-term changes caused by nitrogen deposition are uncertain as transport times vary between environmental systems. The only way to remove excess nitrogen is to convert it to an unreactive gas (Galloway *et al.*, 2008).

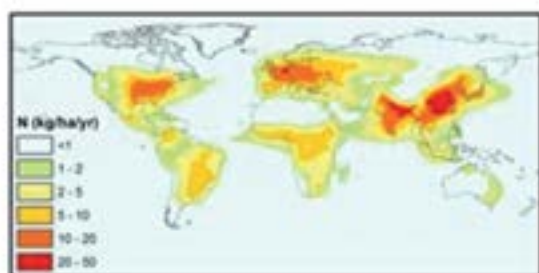
a) Atmospheric sulphur deposition (2001)



b) Soil sensitivity to acidification



c) Atmospheric nitrogen deposition (2001)



d) Soil carbon to nitrogen ratio (C:N)

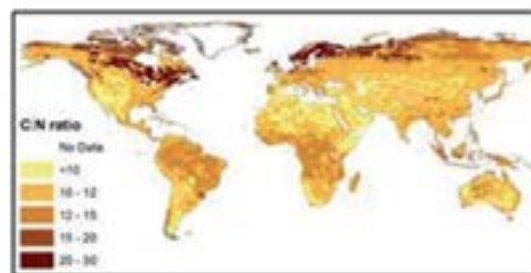


Figure 4.10 | Global distribution of (a) atmospheric S deposition, (b) soil sensitivity to acidification, (c) atmospheric N deposition, and (d) soil carbon to nitrogen ratio (soils most sensitive to eutrophication have a high C:N ratio; eutrophication is caused by N). Source: Vet et al., 2014; Batjes, 2012; FAO, 2007.

Atmospheric deposition data in (a) and (c) were provided by the World Data Centre for Precipitation Chemistry (<http://wdcpc.org>, 2014) and are also available in Vet *et al.* (2014). Data show the ensemble-mean values from the 21 global chemical transport models used by the Task Force on Hemispheric Transport of Air Pollution (HTAP) (Dentener *et al.*, 2006). Total wet and dry deposition values are presented for sulphur, oxidized and reduced nitrogen. Soil data in (b) and (d) were produced using the ISRIC-WISE derived soil properties (ver 1.2) (Batjes, 2012) and the FAO Digital Soil Map of the World.

Trace element deposition

Global trace element emissions and deposition are poorly understood in comparison to our understanding of emissions of sulphur and nitrogen. Emissions of trace elements are associated with combustion of fossil fuel (V, Ni, Hg, Se, Sn), traffic (Pb), insecticides (As), steel manufacture (Mn, Cr), and mining and smelting (As, Cu, Zn, Hg) (Mohammed, Kapri and Goel, 2011). In the United Kingdom, trace element deposition is responsible for 25-85 percent of total trace element inputs to soils (Nicholson *et al.*, 2003). In Europe, the area at risk from Cd, Hg and Pb deposition in 2000 was 0.34 percent, 77 percent and 42 percent respectively, although emissions are declining (Hettelingh *et al.*, 2006). In China, 43-85 percent of total As, Cr, Hg, Ni and Pb inputs to agricultural soils originate from atmospheric deposition (Luo *et al.*, 2009). In bioavailable form these elements have a toxic effect on soil organisms and plants, influencing the quality and quantity of plant inputs to soils and the rate of decomposition. Significantly, they can also bioaccumulate in the food chain. Activity of trace elements in soils will depend on the specific mobility of the element and this will be influenced by pH, Eh and the concentration of dissolved organic matter with complexing properties (Blaser *et al.*, 2000). Some trace elements will persist for centuries as they are strongly bound to soil particles. However, they can become bioavailable, as observed in peatlands following drought-induced acidification, drainage and soil erosion (Tipping *et al.*, 2003; Rothwell *et al.*, 2005).

4.4.3 | Knowledge gaps and research needs

Atmospheric pollution is a global phenomenon impacting large areas of the land surface. Regional and global scale assessment relies on the use of simple models to: (I) upscale site-specific soil data, in some instances using soil databases collected as long ago as the 1970s; and (II) estimate where soil sensitivity – the ‘critical load’ – of a single pollutant is exceeded. There are few locations with long-term soil monitoring data, particularly in comparison to the data available on air, rain and surface water quality. Therefore, the actual global extent and magnitude of polluted soils are unclear. Essentially, we lack data at adequate scales to check the model outputs. A long-term global soil monitoring network is needed.

While the direct impacts of sulphur, nitrogen and trace elements on inorganic soil chemical processes are generally well understood, many uncertainties still exist about pollutant impacts on biogeochemical cycling, particularly interactions between organic matter, plants and organisms in natural and semi-natural systems (Greaver *et al.*, 2012). Process understanding is dominated by research in Europe and North America (e.g. Bobbink *et al.*, 2010). Research is needed in other regions where soil properties and environmental conditions differ from the empirically studied areas in Europe and North America. Models need to be developed to examine the combined effects of air pollutants and their interactions with climate change and feedbacks on greenhouse gas balances and carbon storage (Spranger *et al.*, 2008; RoTAP, 2012). Air quality, biodiversity and climate change policies all impact on soils. A more holistic approach to protecting the environment is needed, particularly as some climate change policies (e.g. biomass burning, carbon capture and storage) have potential to impact air quality and, therefore, soil functions (Reis *et al.*, 2012; RoTAP, 2012; Aherne and Posch, 2013).

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