Global change pressures on soils from land use and management

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Abstract

Soils are subject to varying degrees of direct or indirect human disturbance, constituting a major global change driver. Factoring out natural from direct and indirect human influence is not always straightforward, but some human activities have clear impacts. These include land-use change, land management and land degradation (erosion, compaction, sealing and salinization). The intensity of land use also exerts a great impact on soils, and soils are also subject to indirect impacts arising from human activity, such as acid deposition (sulphur and nitrogen) and heavy metal pollution. In this critical review, we report the state-of-the-art understanding of these global change pressures on soils, identify knowledge gaps and research challenges and highlight actions and policies to minimize adverse environmental impacts arising from these global change drivers. Soils are central to considerations of what constitutes sustainable intensification. Therefore, ensuring that vulnerable and high environmental value soils are considered when protecting important habitats and ecosystems, will help to reduce the pressure on land from global change drivers. To ensure that soils are protected as part of wider environmental efforts, a global soil resilience programme should be considered, to monitor, recover or sustain soil fertility and function, and to enhance the ecosystem services provided by soils. Soils cannot, and should not, be considered in isolation of the ecosystems that they underpin and vice versa. The role of soils in supporting ecosystems and natural capital needs greater recognition.

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The lasting legacy of the International Year of Soils in 2015 should be to put soils at the centre of policy supporting environmental protection and sustainable development.

Keywords: heavy metal deposition, land-use change, land-use intensity, nitrogen deposition, soil, sulphur deposition

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Introduction

2015 is the International Year of Soil. This represents an ideal time to take stock of scientific knowledge about the changing global pressures that humans are exerting on soils. 2015 is also the year when policymakers will adopt a new legally binding climate agreement under the United Nations Framework Convention on Climate Change (UNFCCC), with individual countries and businesses making policies and targets on greenhouse gas emissions and removals. Soils storage and cycling of carbon and nitrogen are part of emissions and removals from the land sector. Furthermore, 2015 is the year when countries will shape and adopt a new development agenda that will build on the Millennium Development Goals (MDGs). With increasing population, issues such as food security, water security, energy security (including bioenergy production) and sustainable integrated land and resource management are central to many development research and policy agendas. Soils underpin the provision of many ecosystem services related to development.

Soils provide multiple ecosystem services, allowing sustained food and fibre production, and delivering climate regulation, flood regulation, improved air and water quality, reducing soil erosion, and provide a reservoir for biodiversity (Smith et al., 2015). All soils are subject to some degree of human disturbance, either directly through land use and land management, or indirectly through responses to human-induced global change such as pollution and climate change. Distinguishing natural from direct and indirect human influence is not always straightforward (Smith, 2005), but some human activities and their consequences have clear impacts, and despite large heterogeneity in soil properties and responses, robust scientific knowledge exists.

Human impacts on soils largely emerge from the need to meet the food, fibre and fuel demands of a growing population including an increase in meat consumption as developing nations become wealthier, the production of biofuels, and increasing areas of urbanization. This has led to conversion of natural land to managed land (extensification) and intensification of agricultural and other management practices on existing land such as increasing nutrient and water inputs and increasing harvest frequency to increase yields per hectare.

Land-cover or land-use change (e.g. from forest or natural grassland to pasture or cropland) removes biomass, changes vegetation and disturbs soils, leading to loss of soil carbon and other nutrients, changes in soil properties and changes to above- and below-ground biodiversity. Some land-cover conversions, for example reforestation after abandonment of cropland, can increase both above- and below-ground carbon and nutrients. Land use or land management that does not result in a change of cover (e.g. forest harvest and regrowth, increased grazing intensity and intensification of crop production) can potentially result in degradation of soil properties, depending on the characteristics of the management practices.

Land-use change has been accelerated by population increases and migration as food, shelter and materials are sought and acquired. It is estimated that humans have directly modified at least 70 Mkm² or >50% of Earth’s ice-free land area (Hooke & Martín-Duque 2012). The new Global Land Cover Share database (Latham et al., 2014) represents the major land-cover classes defined by the FAO. Croplands and grasslands (including both natural grasslands and managed grazing lands) each covered 13.0%. ‘Tree-covered areas’ (i.e. both natural and managed forests) covered 28% and shrub-covered areas 9.5%. Artificial surfaces (including urbanized areas) occupy 1%. Land degradation can be found in all land-cover types. Degraded land covers approximately 24% of the global land area (35 Mkm²). Twenty-three per cent of degrading land is under broadleaved forest, 19% under needle-leaved forests and 20–25% on rangeland (Bai et al., 2008).

In this review, we report the state-of-the-art understanding, and knowledge gaps concerning impacts of changes in anthropogenic land use and land management on soils, including interactions with other anthropogenic global change pressures. We also review actions and policies that limit the adverse impacts arising from these global change drivers. We make the case to put soils at the centre of research strategy and policy actions as a legacy of the International Year of Soils.

Land-use/land-cover change

Land-cover change has been dominated by deforestation, but also conversion of grasslands to cropland and grazing land. Deforestation has had the greatest impact on historical soil carbon change, causing on average around 25% of soil carbon to be lost (Guo & Gifford, 2002; Murty et al., 2002). Soil carbon losses largely stem
from oxidation of the organic matter as well as soil erosion.

Deforestation affected an estimated 13 million hectares per year between 2000 and 2010; net forest loss was 5.2 million hectares per year (FAO, 2010). Most of this recent deforestation has taken place in tropical countries (FAO, 2010; Hansen et al., 2013). Over 50% of tropical forest loss occurred in Brazil and Indonesia, largely driven by a few commodities: timber, soy, beef and oil palm (West et al., 2014). There has been a reduced rate of deforestation in some regions over the last decade, most notably Brazil (INPE, 2014), largely because of land-use conservation policies (Nolte et al., 2013; Soares-Filho et al., 2014) as well as economics. 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 in contrast to the large-scale historic deforestation in these regions, with afforestation reported in Europe, USA, China, Vietnam and India (FAO 2013).

Changes in soil properties can vary markedly with type of land-cover change, climate, and method, extent of vegetation removal (e.g. land clearing, fires, mechanical harvest) and management postharvest. For example, West et al. (2010) estimated that clearing land in the tropics generally emits three times the amount of carbon per ton of annual crop production compared to clearing land in temperate areas. Emissions are particularly high when organic peatland/wetland soils are drained to enable agriculture as the initial soil carbon is higher, and drainage results in large losses of carbon as previously anaerobic soils become aerobic, allowing the organic matter to oxidize. For example, clearing forest on organic soils for palm oil production in Kalimantan emits nine times more carbon than clearing on neighbouring mineral soils (Carlson & Curran, 2013). Impacts of deforestation can be reduced by avoiding deforestation on organic soils and on steep slopes prone to erosion.

There is large heterogeneity in soil measurements of carbon, nitrogen, microbes, etc., and still many areas of the world with poor data coverage. Models can be used to fill gaps in spatial coverage and look at past and future time periods (Smith et al., 2012), but these too give very variable results. Nevertheless, there are some clear signals that can be obtained from meta-analyses of field data and global model results of land-use/land-cover change with respect to soil carbon.

**Observations of impacts of land-cover change**

Table 1 presents the results of different meta-analysis studies across different climatic zones that compared the impacts of land-use changes on SOC (Guo & Gifford, 2002; Murty et al., 2002; Don et al., 2011; Poeplau et al., 2011; Bárcena et al., 2014; Wei et al., 2014a). Changes in SOC after the conversion of forests to croplands ranged from −24% to −52% without marked differences between climatic regions. The conversion of pastures to other uses (tree plantations and particularly, croplands) also induced decreases in SOC (−10% and −59%, respectively). On the other hand, the substitution of croplands by other land uses (forest regrowth, tree plantation, grassland, pasture) resulted in an increase of SOC (+18% to +53%). In the case of afforestation, soil C increases with time after afforestation, and C sequestration depends on prior land use, climate and the tree species planted.

Fewer meta-analysis studies are available for changes in soil N with changes in land uses. A compilation with predominance of data from tropical sites indicated that average loss of 15% of soil N after conversion of forests to croplands (Murty et al., 2002). In Australia, N losses after conversion of native vegetation to perennial pasture and cropland were more than 20% and 38%, respectively (Dalal et al., 2013), while in China, N loss (0–10 cm depth) was 21% and 31% after 4 and 50 years after conversion of forests to cropland (Wei et al., 2014b). Similarly to what was described for SOC, afforestation in subtropical zone results in a significant increase of N stocks 50 years after conversion (Li et al., 2012).

**Modelled impacts of land-cover change**

Dynamic Global Vegetation Models (DGVMs) are 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. A few global models include some aspects of forest, grassland or cropland management (Jain et al., 2005; Bondeau et al., 2007; Drewski et al., 2013; Lindeskog et al., 2013). Most DGVMs do not currently model peatland soils. In Tables 1 and 2 and Figs 1 and 2, we show impacts of past land-cover and management change on soil carbon and nitrogen as calculated by three DGVMs: Integrated Science Assessment Model (ISAM) (El-Masri et al., 2013; Jain et al., 2013; Barman et al., 2014a,b), Lund-Potsdam-Jena General Ecosystem Simulator (LPJ-Guess) (Smith et al., 2001; Lindeskog et al., 2013) and Lund-Potsdam-Jena managed Land (LPJmL) (Bondeau et al., 2007). 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). ISAM included wood harvest following (Hurt et al., 2011). The LPJmL group combined three land-use change data sets with the geo-
graphic distribution of global agricultural lands in the
year 2000. All models were run with historical climate
and CO2, and additionally N deposition in the ISAM
model only as it includes a nitrogen cycle. The effects of
land-cover change were isolated by comparing model
runs with and without land use/management (Le
Quéré/Cer et al., 2014). Table 2 and Fig. 1 show the loss of
soil carbon due to historical land-use change from 1860
to 2010 (note there was land-use change causing soil
carbon loss prior to 1860 particularly in Europe and
central Asia, but results are not shown as they were not
available for all three models). As with the observed
data (Table 1), high carbon losses are associated with
the conversion of forests to croplands. Figure 2 shows
the mineral soil C and N concentration of different
land-cover types in different geographic ranges.
Differences between the models are large for some
systems and regions due to different land-use change
data, different land-cover definitions and different pro-
cesses included in the models. For example, soil carbon
losses are higher in the LPJmL model (Table 2, Fig. 1)
in part due to greater land-cover change in their land-
cover reconstructions, while their boreal grassland soil
carbon is high due to the inclusion of permafrost slow-
ing soil carbon decomposition (Fig. 2). Treatment of

<table>
<thead>
<tr>
<th>Model</th>
<th>Tropical</th>
<th>Temperate</th>
<th>Boreal</th>
<th>Global</th>
</tr>
</thead>
<tbody>
<tr>
<td>LPJ-GUESS</td>
<td>46</td>
<td>55</td>
<td>1</td>
<td>109</td>
</tr>
<tr>
<td>LPJmL</td>
<td>128</td>
<td>95</td>
<td>0</td>
<td>227</td>
</tr>
<tr>
<td>ISAM</td>
<td>63</td>
<td>139</td>
<td>19</td>
<td>221</td>
</tr>
<tr>
<td>Mean</td>
<td>79</td>
<td>96</td>
<td>7</td>
<td>186</td>
</tr>
</tbody>
</table>

Table 2 Soil carbon loss due to land-use change 1860–2010 (PgCO2)
management processes turns out to be an important differentiator. ISAM shows strong decreases of soil carbon in some regions, for example the southern Boreal zone (Fig. 1), where the inclusion of wood harvest removes carbon and nutrients from the soil, while increases in soil carbon in parts of the mid-latitudes are due to regrowth of forest following abandonment of agricultural land.

In semi-arid to arid regions, LPJ-GUESS and LPJmL show opposite signs of soil carbon change after conversion of natural land to pastures (Fig. 1), primarily because LPJ-GUESS simulates a greater fraction of woody vegetation than LPJmL in these regions under potential natural vegetation. Conversion of woody vegetation to pasture slightly increases soil carbon (see the meta-analysis of Guo & Gifford, 2002), partly because of boosted productivity and higher turnover rates adding more C to the soil, while the change from potential natural grassland to managed pasture (for which the literature is sparse) results in a soil carbon decrease in LPJmL. Pasture management strategies can have a large influence on the soil carbon storage (see Grassland management and dryland degradation) and may also be partly responsible for differences.

Vegetation models are embedded in Earth System Models (ESMs) used to project future climates under different human activity including different land management. Some significant differences between future model climate projections stem from the differences in modelling soil carbon, in particular, the strength of the relationship between increasing temperatures and the increasing rate of soil carbon decomposition (Q10) causing climate–carbon feedbacks via CO2 emissions (Friedlingstein et al., 2006). A recent intercomparison of 11 ESMs used in the IPCC 5th Assessment Report (Todd-Brown et al., 2013) found the estimate of global soil carbon from ESMs ranged from 510 to 3040 PgC across 11 ESMs compared to an estimate of 890–1600 PgC (95% confidence interval) from the Harmonized World Soil Data Base (FAO/IIASA/ISRIC/ISS-CAS/JRC, 2012), with all models having difficulty representing the spatial variability of soil carbon at smaller (1 degree) scales compared to empirical data. In all models, net primary production (NPP) and temperature strongly influenced soil carbon stocks, much more so than in the observational data, and differences between models were found to be largely due to the representation of NPP and the parameterization of soil decomposition submodels. A similar, systematic analysis of DGVMs including benchmarking with observational data, and careful testing of assumptions and process representations in these models, making use of the very large number of observations that have become available in the years since these algorithms were formulated (e.g. Medlyn et al., 2015), could significantly improve model performance. This, along with better representation of critical biological and geochemical mechanisms would improve model capability (Todd-Brown et al., 2013).
Drainage and conversion of peatlands/wetlands for agriculture

The organic soils in peatlands/wetlands store vast quantities of carbon which decomposes rapidly when they are drained for agriculture or commercial forestry, resulting in emissions of CO$_2$ and N$_2$O to the atmosphere (Hooijer et al., 2010). Other services, in particular water storage and biodiversity, are negatively impacted. Drainage increases vulnerability to further losses through fire. The majority of soil carbon is concentrated in peatlands in the boreal zone and tropical peatland forests in South-East Asia. These areas, along with wetlands along the banks of rivers, lakes and estuaries, have increasingly been developed for croplands/bioenergy production over recent decades. The FAO emissions database estimates that globally, there are 250 000 km$^2$ of drained organic soils under cropland and grassland, with total GHG emissions (N$_2$O plus CO$_2$) of 0.9 Pg CO$_2$eq yr$^{-1}$ in 2010, with the largest contributions from Asia (0.44 Pg CO$_2$eq yr$^{-1}$) and Europe (0.18 Pg CO$_2$eq yr$^{-1}$; FAOSTAT, 2013; Tubiello et al., 2015). Joosten (2010) estimated that there are >500 000 km$^2$ of drained peatlands in the world, including under forests, with CO$_2$ emissions having increased from 1.06 Pg CO$_2$ yr$^{-1}$ in 1990 to 1.30 Pg CO$_2$ yr$^{-1}$ in 2008, despite a decreasing trend in developed countries, from 0.65 to 0.49 Pg CO$_2$ yr$^{-1}$, primarily due to natural and artificial rewetting of peatlands. In South-East Asia, CO$_2$ emissions from drained peatlands in 2006 were 0.61 ± 0.25 Pg CO$_2$ yr$^{-1}$ (Hooijer et al., 2010). Conversion of peatlands in South-East Asia is increasing, particularly for oil palm expansion, where cleared peatlands typically emit ~9 times more carbon than neighbouring mineral soils (Carlson & Curran, 2013). In China, between 1950 and 2000, 13 000 km$^2$ of wetland soils were shifted to cultivated arable lands, which led to a SOC loss of 5.5 Pg CO$_2$, mostly from peatlands in Northeast China and Tibet (Zhang et al., 2008).

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 (i.e. between ~70 and 150 t CO$_2$ ha$^{-1}$ over the

Fig. 2 Soil carbon and nitrogen under different land-cover types in three different vegetation models (values are the annual average over the period 2001–2010).
1960–2006 period), most probably as a consequence of intensified soil drainage practices for cultivation purposes.

### Agricultural management

To meet projected increases in food demand, crop production will need to increase by 70–110% by 2050 (World Bank, 2008; Royal Society of London, 2009; Tilman et al., 2011). This can be achieved either through further expansion of agricultural land (extensification) or through intensification of production on existing land. Intensification is widely promoted as the more sustainable option because of the negative environmental consequences of land expansion through deforestation and wetland cultivation (Foley et al., 2011). For example, Burney et al. (2010) estimate that intensification of production on croplands between 1961 and 2010 avoided the release of 590 PgCO₂eq. Increased productivity per unit land area can be achieved through a variety of management practices, such as fertilization, irrigation and increased livestock density, but these can lead to adverse consequences for the soil and wider environment (Tilman et al., 2002). Intensifying land use can potentially reduce soil fertility (without additional inputs) and its ability to sustain high production, as well as soil resilience to extreme weather under climate change, pests and biological invasion, environmental pollutants and other pressures. Some key management practices and consequences are highlighted below and summarized in Table 3.

### Nutrient management

Cultivation of soils results in a decline in soil nutrients (nutrient mining). Nutrient inputs, from both natural and synthetic sources, are needed to sustain soil fertility and supply nutrient requirements for crop production. Nutrient supply can improve plant growth which increases organic matter returns to the soil, which in turn can improve soil quality (see Soil compaction), so balanced nutrient supply has a positive impact on soils (Smith et al., 2015). Overuse, however, has negative environmental consequences. Annual global flows of nitrogen and phosphorus are now more than double natural levels (Matson et al., 1997; Smil, 2000; Tilman et al., 2002). In China, for example, N input in agriculture in the 2000s was twice that in 1980s (State Bureau of Statistics-China, 2005).

Between 50% and 60% of nutrient inputs remain in agricultural soils after harvest (West et al., 2014) and can enter local, regional and coastal waters becoming a major source of pollution such as eutrophication leading to algal blooms (Carpenter et al., 1998). In many places around the world, overuse of synthetic nitrogen fertilizers is causing soil acidification and increased decomposition of soil organic matter, leading to loss of soil function in overfertilized soils (Ju et al., 2009; Tian et al., 2012).

Use of fertilizers and manures contributes to climate change through their energy intensive production and inefficient use (Tubiello et al., 2015). Globally, approximately 3–5% of nitrogen additions are released as

### Table 3 Threats to soil resource quality and functioning under increasing intensity of agricultural management

<table>
<thead>
<tr>
<th>Agricultural management practice</th>
<th>Specific issue</th>
<th>Distribution</th>
<th>Major environmental consequence</th>
<th>Knowledge gap</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cropping practice</td>
<td>Harvest frequency</td>
<td>Global</td>
<td>Soil quality and resilience</td>
<td>Impact on total C and nutrient cycles</td>
</tr>
<tr>
<td></td>
<td>Monoculture</td>
<td>Global but particularly in developing and transition countries</td>
<td>Soil health, pesticide residue in intensively managed monocultures</td>
<td>Biological resilience</td>
</tr>
<tr>
<td>Use of agrochemicals</td>
<td>Over fertilization</td>
<td>Particularly in some developing countries, Asian</td>
<td>Soil acidification, water pollution, N₂O emission and nitrate accumulation</td>
<td>Rate reducing vs. balancing</td>
</tr>
<tr>
<td>Irrigation</td>
<td>Submerged Rice</td>
<td>Developing countries, Asian</td>
<td>Water scarcity, methane emission</td>
<td>Trade-offs C and water</td>
</tr>
<tr>
<td>Livestock management</td>
<td>Overgrazing</td>
<td>Largely in developing countries</td>
<td>Soil degradation, water storage, C loss</td>
<td>Competition use of water</td>
</tr>
<tr>
<td></td>
<td>Industrial breeding</td>
<td>Largely in industrialized and transition countries</td>
<td>Waste pressure, water pollution, residue of veterinary medicine and antibiotics</td>
<td>Forage vs. feed crops?</td>
</tr>
<tr>
<td>Agriculture in wetlands</td>
<td>Wetland drainage</td>
<td>Developing and transition countries</td>
<td>C loss</td>
<td>Safe waste treatment and recycling</td>
</tr>
</tbody>
</table>

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nitrous oxide (N₂O) to the atmosphere when both direct (from soils) and indirect (e.g. downstream from nitrate leaching) emissions are considered (Galloway et al., 2004), and N₂O has ~300 times the radiative forcing of carbon dioxide (IPCC, 2007). Recent research indicates that the relationship between nitrogen application and N₂O emissions is nonlinear, resulting in an increasing proportion of added N being emitted, as application rate increases (Philibert et al., 2012; Scherbak et al., 2014). China, India and the United States account for ~56% of all N₂O emissions from croplands, with 28% from China alone (West et al., 2014). Overuse of nitrogen and phosphorus fertilizer can contribute to eutrophication of water bodies, adversely affecting water quality and biodiversity (Galloway et al., 2003, 2004, 2008).

Nutrient use efficiency can be significantly increased, and nitrate losses to water and N₂O emissions can be reduced, through changes in rate, timing, placement and type of application, as well as balancing fertilization (Venterea et al., 2011; Snyder et al., 2014). It has been estimated that current levels of global cereal production could be maintained while decreasing global nitrogen application by 50% (Mueller et al., 2014).

**Carbon management: reduced disturbance and organic matter additions**

Agricultural soils have the potential to store additional carbon than at present if best management practices are used (Paula et al., 1997; Smith, 2008, 2012). Soil organic matter content of soils can be increased through use of improved crop varieties or grassland species mixtures with greater root mass or deeper roots (Kell, 2012), improved crop rotations in which C inputs are increased over a rotation (Burney et al., 2010), greater residue retention (Wilhelm et al., 2004) and use of cover crops during fallow periods to provide year-round C inputs (Burney et al., 2010; Poelplau & Don, 2015). Several studies report that soil carbon increases in croplands under no-till management (West & Post, 2002; Ogle et al., 2005). However, the carbon benefits of no-till may be limited to the top 30 cm of soil (Blanco-Canqui & Lal, 2008; Powlson et al., 2014). Baker et al. (2007) found that total soil carbon was similar in no-till and conventional systems, suggesting that carbon accumulation is occurring at different depths in the soil profile under different management schemes. Given the larger variability in subsurface horizons and lack of statistical power in most studies, more research is needed on soil carbon accumulation at depth under different tillage regimes (Kravchenko & Robertson, 2010).

Adding plant-derived carbon from external sources such as composts and biochar can increase soil carbon stocks. Composts and biochars are more slowly decomposed compared to fresh plant residues, with mean residence times several (composts) to 10–100 (biochars) longer than uncomposted organic materials (Lehmann et al., 2015; Ryals et al., 2015). Recent developments suggest that biochar, from the pyrolysis of crop residues or other biomass, can consistently increase crop N use efficiency while greatly (over 25%) reducing direct N₂O emissions from N fertilizers (Liu et al., 2012; Huang et al., 2013), as well as enhancing soil fertility (Wolff et al., 2010).

**Water management**

The amount of irrigated croplands has doubled in the last 50 years and now accounts for 70% of all water use on the planet (Gleick, 2003). While irrigated crops cover 24% of all cropland area, they account for 34% of all production (Siebert & Döll, 2010). Irrigation is concentrated in precipitation-limited areas such as India, China, Pakistan and the USA, which account for 72% of irrigation water use (West et al., 2014). Agricultural water-use competes with uses for human and natural ecosystems exacerbating water stress in dry regions. Increased irrigation has occurred in many areas of world agriculture due to the increasing frequency of drought under the climate change (West et al., 2014). Where irrigation increases productivity (e.g. in drought prone areas), organic carbon inputs to the soils would be expected to increase, increasing soil organic matter content (Carbon management: reduced disturbance and organic matter additions).

Irrigation can increase soil salinity in dry regions with high salt content in the subsoil (Ghassemi et al., 1995; Setia et al., 2011a,b). Where salinization occurs, additional irrigation is needed to ‘flush’ the salts beyond the root zone of the crops, which can further exacerbate stress on water resources, particularly when using underground water sources. Saline soils, which have a high concentration of soluble salts, occupy approximately 3.1% (397 Mha) of the world’s land area (FAO, 1995). Climate change (need for more frequent irrigation) and increases in human population (increasing demand for more production) are likely to increase the extent of saline soils (Rengasamy, 2008). The energy required by plants or soil organisms to withdraw water from the soil or retain it in cells increases with decreasing osmotic potential. As soils dry out, the salt concentration in the soil solution increases (decreasing osmotic potential), so two soils of different texture may have the same electrical conductivity, but the osmotic potential is lower in the soil with low water content (Ben-Gal et al., 2009; Chowdhury et al., 2011; Setia et al., 2011a). The accumulation of salts in the root zone has adverse effects on plant growth activity, not only due
to negative osmotic potential of the soil solution resulting in decreased availability of water to plants, but also ion imbalance and specific ion toxicity (Chowdhury et al., 2011). Salinity affects microorganisms mainly by decreasing osmotic potential, which affects a wide variety of metabolic activities and alters the composition and activity of the microbial community (Chowdhury et al., 2011) and thereby soil organic matter decomposition.

In saline soils, SOC content is influenced by two opposing factors: reduced plant inputs which may decrease SOC, and reduced rates of decomposition (and associated mineralization of organic C to CO₂) which could increase SOC content if the C input was unchanged. Using a modified Rothamsted carbon model with a newly introduced salinity decomposition rate modifier and a plant input modifier (Setia et al., 2011b, 2012), Setia et al. (2013) estimated that, historically, world soils that are currently saline have lost an average of 3.47 t SOC ha⁻¹ since they became saline. With the extent of saline soils predicted to increase under the future climate, Setia et al. (2013) estimated that world soils may lose 6.8 Pg SOC due to salinity by the year 2100. Soil salinization is difficult to reverse, but salt-tolerant plant species could be used to rehabilitate salt-affected soils (Setia et al., 2013).

Water efficiency can be improved through management practices that reduce water requirement and evaporation from the soil (such as adding mulch as groundcover), more precise irrigation scheduling and rates, fixing leaks in dryland irrigation systems, improved application technology (e.g. drip irrigation) and use of intermittent irrigation in rice paddies. Given that water limitation is projected to become even more limiting in several semi-arid regions, for example sub-Saharan Africa, where the human population will probably increase most in the future, and climate change impacts are projected to be severe, improved water harvesting methods, for example storage systems, terracing and other methods for collecting and storing runoff, are required to make best use of the limited water resource.

Harvest frequency

Approximately 9% of crop production increases from 1961 to 2007 was from increasing the harvest frequency (Alexandratos & Bruinsma, 2012). The global harvested area (i.e. counting each time an area is harvested) increased four times faster than total cropland area between 2000 and 2011 (Ray & Foley, 2013). The fraction of NPP extracted by humans is increasing (Haberl et al., 2007). Global warming is increasing the total area suitable for double or even triple cropping in subtropical and warm temperate regions (Liu et al., 2013a). The increase results from fewer crop failures, fewer fallow years and an increase in multicropping.

Increasing harvest frequency can reduce soil quality by, for example continuously removing soil nutrients and increasing soil compaction through greater soil traffic, but if legumes are included in rotations as harvest frequency increases, soil quality could be improved. Increasing harvest frequency may require increasing pesticide and herbicide use, and increased use of fertilizers contributing to pollution (Nutrient management). The net effect will depend on the effectiveness of the management practices followed.

Soil compaction

Soil compaction causes degradation of soil structure by increasing soil bulk density or decreasing porosity through externally or internally applied loads, as air is displaced from the pores between the soil grains (McCarthy, 2007; Alakukku, 2012). It is the most important subtype of physical soil deterioration, covering 68 Mha globally when first mapped in the 1990s (Oldeman et al., 1991). Compaction of agricultural soils often results from heavy machinery or from animal trampling, so is more likely to occur in intensive agricultural systems (machinery use and high stocking densities), and affects physical, chemical and biological properties of soil. Top soil compaction can be reversed and controlled, but when compaction creates impermeable layers in the subsoil, this is less easily reversed.

Subsoil compaction can disrupt nutrient water flows, which in turn can lead to reduced crop yields, poorer crop quality and can give rise to increased GHG emissions, water and nutrient run-off, erosion, reduced biodiversity and reduced groundwater recharge (Batley, 2009). Where compaction cannot be avoided, mitigation is necessary. Biological approaches to mitigation include planting deep rooted plants such as agroforestry; chemical methods include fertilization (to overcome yield penalty, although not to remedy compaction); and technical measures include machinery in which planting does not coincide with wheel tracks, wide tyres/reduced tyre pressures to reduce pressure per unit area, and precision farming to retain the same wheel tracks each year (Hamza & Anderson, 2005).

Livestock density

Livestock production is projected to increase significantly to meet the growing demand from a growing population and increase in per capita meat consumption, with total demand for meat expected to grow by
more than 200 Mt by 2050 (Alexandratos & Bruinsma, 2012). The greatest increases in per-capita consumption are projected to be in developing and transition countries (Alexandratos & Bruinsma, 2012). Since the 1970s, most increased livestock production has resulted from intensification: increasing livestock density and shifting to a greater fraction of livestock raised in industrial conditions (Bouwman et al., 2006). For example, 76–79% of pork and poultry production is industrialized (Herrero & Thornton, 2013). Manure, inputs for growing feed, and soil loss from intensively managed areas can be major sources of water pollution to local and downstream freshwater ecosystems. Clearing natural ecosystems for new pastures, particularly in arid and semi-arid regions, typically occurs on low-productivity lands with a much higher risk of soil erosion and soil carbon/nutrient depletion (Alexandratos & Bruinsma, 2012) and negatively impacts water storage and biodiversity. The impacts of livestock production are particularly prevalent for beef production, which has a least an order of magnitude greater impact on land, water, GHGs and reactive nitrogen compared to other livestock (Eschel et al., 2014; Ripple et al., 2014). Moreover, industrial livestock production had led to an increased use of veterinary medicines, antibiotics and hormones, posing potential risks to soil, water, ecosystems and human health. Improved grazing management (e.g. optimized stocking density) can reduce soil degradation, and thereby maintain and enhance organic matter content (McSherry & Ritchie, 2013; see Carbon management: reduced disturbance and organic matter additions and Grassland management and dryland degradation), and can reduce soil compaction, thereby increasing infiltration and water storage and reduce risk of runoff and flooding downstream (Marshall et al., 2009).

Other land management

Forest management

Logging and fire are the major causes of forest degradation in the tropics (Bryan et al., 2013). Logging removes nutrients and negatively affects 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). Forest fires affect many physical, chemical, mineralogical and biological soil properties, 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 caused a significant decrease in soil C (−26%) and N (−22%). Fires reduced forest floor storage (pool sizes only) by an average of 59% (C) and 50% (N), but the relative concentrations of these two elements did not change. Prescribed fires caused smaller reductions in C and N storage (−46% and −35%) than wildfires (−67% and −69%). These differences are likely because of lower fuel loads or less extreme weather conditions in prescribed fires, both factors that result in lower fire intensity. 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 (−11% and −12%, respectively). Mineral soil C and N concentrations were significantly reduced in response to wildfires, but not after prescribed burning.

Forest fires produce charcoal, or black carbon, some of which can be preserved over centuries and millennia in soils. Dissolved black carbon 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). A large field study in the Amazon (225 forest plots) on the effects of anthropogenic forest disturbance (selective logging, fire and fragmentation) on soil carbon pools 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). As with deforestation, impacts of human disturbances on the soil carbon are of particular concern in tropical forests located on organic soils and on steep easily eroded slopes.

Shifting cultivation

Shifting cultivation practices, where land is cleared through fire, have been practiced for thousands of years, but recent increasing demographic pressure has reduced the duration of the fallow period, affecting the system sustainability. Moreover, especially in South-East Asia where urbanization is expanding in fertile planes, shifting cultivation is practiced in sloping uplands, which are prone to soil and carbon loss by erosion (Chaplot et al., 2005). A review by Ribeiro-Filho et al. (2013) reported negative impact on SOC associated with the conversion stage, modified by the characteristics of the burning. Chop-and-mulch of enriched fallows appears to be a promising alternative to slash-and-burn, conserving soil bulk density, and significantly increasing nutrient concentrations and organic matter content compared to burnt cropland, and a control forest in a study in the Amazon (Comtea et al., 2012).
Grassland management and dryland degradation

Grasslands, including rangelands, shrublands, pastureland and cropland sown with pasture and fodder crops, cover 26% of the global ice-free land area and 70% of the agricultural area, and contain about 20% of the world’s soil organic carbon (C) stocks. Grasslands on every continent have been degraded due to human activities, with about 7.5% of grassland having been degraded because of overgrazing (Conant, 2012). A meta-analysis (McSherry & Ritchie, 2013) of grazer effects on SOC density (17 studies that include grazed and ungrazed plots) found higher grazing intensity was associated with increased SOC in grasslands dominated by C4 grasses (increase of SOC by 6–7%), but with lower SOC in grasslands dominated by C3 grasses (decrease of SOC by an average 18%). An increase in mean annual precipitation of 600 mm resulted in a 24% decrease in the magnitude of the grazer effect on finer textured soils, but on sandy soils, the same increase in precipitation produced a 22% increase in the grazer effect on SOC (McSherry & Ritchie, 2013).

Land-use dynamics and climate change are the major drivers of dryland degradation with important feedbacks, with changes in plant community composition (e.g. shrub encroachment and decrease in vegetation cover; D’Odorico et al., 2013). A review by Ravi et al. (2010) indicated soil erosion as the most widespread form of land degradation in drylands, with wind and water erosion contributing to 87% of the degraded land. Grazing pressure, loss of vegetation cover and the lack of adequate soil conservation practices increase the susceptibility of these soils to erosion. The degree of plant cover is negatively related to aridity, and an analysis of 224 dryland sites (Delgado-Baquerizo et al., 2013) 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, possibly indicating the dominance of physical processes such as rock weathering, a major source of P to ecosystems, over biological processes that provide more C and N through litter decomposition (Delgado-Baquerizo et al., 2013).

Soil carbon dynamics in pastures strongly depend on management, with soil carbon increases or decreases observed for different combinations of animal densities and grazing frequency (Conant, 2012; Machmuller et al., 2015). Different grazing strategies, especially in the seminatural dryland biomes, have large implications for vegetation and the carbon balance (Yates et al., 2000). Under certain conditions, grazing can lead to increased annual NPP over ungrazed areas, particularly with moderate grazing in areas with a long evolutionary history of grazing and low primary production, but this does not always lead to an increase in soil carbon (e.g. Badini et al., 2007); grazing, like crop harvest, fundamentally leads to the rapid oxidation of carbon that would otherwise be eventually transferred to the soil. It has long been recognized that the potential effects of management on carbon storage in grassland and dryland soils are substantially greater than that of climate change or CO2 enhancement (Ojima et al., 1993), and Henderson et al. (2015) estimated that the optimization of grazing pressure could sequester 148 Tg CO2 yr−1.

Artificial surfaces, urbanization and soil sealing

In 2014, 54% of the world’s population was urban, and by 2050, two-thirds of the global population will be urban. Many regions in the world (such as Europe and Asia) are affected by migration of populations from rural area to large megacities. Africa and Asia have more rural populations, but are urbanizing faster than the other regions (World Urbanization Prospects, 2014). With urbanization comes land take (development of scattered settlements in rural areas, the expansion of urban areas around an urban nucleus and densification on land within an urban area) and soil sealing. Soil sealing refers to the permanent covering of an area of land and its soil by impermeable artificial material (e.g. asphalt and concrete), for example through buildings and roads. The area actually sealed is only part of a settlement area, and gardens, urban parks and other green spaces are not covered by an impervious surface (Prokop et al., 2011).

Sealing by its nature has a major effect on soil, diminishing many of its benefits (Toth et al., 2007). It is normal practice to remove the upper layer of topsoil, which delivers most of the soil-related ecosystem services, and to develop a strong foundation in the subsoil and/or underlying rock to support the building or infrastructure. Loss of ecosystem and social services (mainly on high-quality soils) includes impacts on water resources (e.g. reduction in rainfall absorbed by the soil, reduction in soil water holding capacity affecting flooding), on soil biodiversity when sealing prevents recycling of dead organic material (Marienina et al., 2008) and on the carbon cycle due to topsoil and vegetation removal (Davies et al., 2011). Sealing through expansion of urban areas can also lead to agricultural land becoming more marginal because the best agricultural land around settlements is lost as they expand, with agricultural land displaced to more marginal land.

Appropriate mitigation measures can be taken to maintain some of the soil functions. In urban planning management, objectives to reduce the impact of soil sealing include the following: (i) preventing the conversion of green areas, (ii) reuse of already built-up areas...
(e.g. brownfield sites Meuser, 2010; Hester & Harrison, 2001 – although remediation of contaminated sites can be costly; Maderova & Paton, 2013), (iii) using (where appropriate) permeable cover materials instead of concrete or asphalt supporting green infrastructure and (iv) implementation of compensation measures. To deliver this mitigation, a number of actions are necessary, for example reduction of subsidies that act as drivers for unsustainable land take and soil sealing (Prokop et al., 2011), and strong collaboration between relevant public authorities and governance entities (Siebielec et al., 2010). Development impacts can be reduced by inclusion of green infrastructure, a network of high-quality green spaces and other environmental features that have a positive effect on well-being (Gill et al., 2007) as well as soils. In some regions, urban sprawl is exacerbated by insufficient incentives to reuse brownfield (derelict, underused or abandoned former industrial or commercial) sites, putting increasing pressure on greenfield land take.

Actions to alleviate pressures on soils driven by sealing fall into three categories: limiting, mitigating and compensating. Actions to limit soil sealing centre around reduction of land take through development of spatial urban planning and environmental protection. Mitigation of soil sealing entails use of strategic environmental assessment for plans and programmes, use of permeable materials and surfaces, green infrastructure within built and urban environments, and natural water harvesting. Compensating soil sealing entails reclamation of degraded land, reuse of extracted topsoil, desalting and is incentivized by land take fees and application of environmental cost calculations.

**Anthropogenic environmental change pressures that interact with land management pressures on soils**

In addition to the direct impacts of humans on soils via land-use change and land management, anthropogenic activity has indirect impacts through human-induced environmental change, such as pollution and climate change. These interact with land management. Soils provide a temporary but labile store for pollutants (Meuser, 2010). Natural processes can release pollutants back to the atmosphere, make them available to be taken up by plants and organisms, leached in to surface waters (Galloway et al., 2008) and/or transported to other areas by soil erosion (Ravi et al., 2010). Pollutants disrupt natural biochemical cycles by altering both soil quality and function through direct changes to the nutrient status, acidity and bioavailability of toxic substances and also by indirect changes to soil biodiversity, plant uptake and litter inputs (EEA, 2014). Soil sensitivity to atmospheric pollution varies with respect to key properties influenced by geology (cation exchange capacity, soil base saturation, aluminium), organic matter, carbon to nitrogen ratio (C : N) and water table elevation (EEA, 2014).

Atmospheric pollutant deposition impacts on soils vary with respect to soil sensitivity to a specific pollutant and the actual pollutant load. Sulphur, nitrogen and heavy metals are released in to the atmosphere by fossil fuel combustion (e.g. power generation, industry and transport) and noncombustion processes (e.g. agricultural fertilizers, waste). These pollutants are transported off-site and deposited as either dry or wet deposition, which can cross national borders. Deposition is enhanced in forests and with altitude because of reduced wind speeds and greater precipitation, respectively, thereby impacting remote areas. Harmful effects to soil function and structure occur where deposition exceeds the ‘critical load’ that a particular soil can buffer (Nilsson & Grennfelt, 1988). Spatial differences in soil sensitivity (commonly defined by the ‘crucial load’) and pollutant deposition result in an uneven global distribution of impacted soils (Fig. 3). For instance, global emissions of sulphur and nitrogen have increased threefold to tenfold since the pre-industrial period (Van Aardenne et al., 2001), yet only 7–17% of the global land area sensitive to acidification is in a region where deposition exceeds the critical load (Bouwman et al., 2002).

Emissions of pollutants, notably sulphur, across Europe and North America have declined since the 1980s following protocols established under the 1979 Convention on Long-range Transboundary Air Pollution and the 1990 US Clean Air Act Amendments (CAAA) (Greaver et al., 2012; Reis et al., 2012; EEA, 2014). Conversely, emissions are likely to increase in response to industrial and agricultural development in south and east Asia, sub-Saharan Africa and South America (Kuylenstierna et al., 2001; Dentener et al., 2006). Further emission increases are occurring in remote areas due to mining activity, such as oil sand extraction in Canada (Kelly et al., 2010; Whitfield et al., 2010).

**Sulphur deposition**

Sulphur emissions are primarily from combustion of coal and oil, typically associated with power generation and heavy industry. In 2001, regions with deposition in excess of 20 kg S ha⁻¹ yr⁻¹ were China and Republic of Korea, Western Europe and eastern North America (Vet et al., 2014; Fig. 3a). Deposition in unimpacted areas is <1 kg S ha⁻¹ yr⁻¹ (Fig. 3a). Pollution control measures have seen an 80% reduction in pollutant sulphur deposition across Europe between 1990 and 2010 (Reis et al., 2012), and emissions in China have declined since 2005 (Fang et al., 2013).
Soil acidification is a natural process that is altered and accelerated by sulphur and nitrogen deposition (Greaver et al., 2012). Sulphur oxides (SOx) react with water to form sulphuric acid (H₂SO₄). Excess inputs of acidity (H⁺) displace soil base cations [e.g. calcium (Ca²⁺) and magnesium (Mg²⁺)] from soil surfaces into solution, which are subsequently lost by leaching (Reuss & Johnson, 1986). Mineral soils can buffer base cation losses if inputs from rock weathering and/or atmospheric dust deposition exceed the amount lost. Therefore, global distribution of acid sensitive soils is associated with conditions that favour development of soils with low cation exchange capacity and base saturation (Bouwman et al., 2002; Fig. 3c). Wetlands can also buffer inputs of acidity through biological sulphate reduction, although acidity can be mobilized again following drought and drainage (Tipping et al., 2003; Laudon et al., 2004; Daniels et al., 2008). Organic acids can also buffer mineral acidity in naturally acidic organic soils (Krug & Frink, 1983).

Decreased soil fertility or ‘sterilization’ due to loss of nutrients and mobilization of toxic metals, particularly Al, is caused by acidification. Impacts in Scandinavia over the 1960s–1980s included declines in freshwater fish populations and damage to forests (EEA, 2014). Sulphur can also stimulate microbial processes that make mercury bioavailable, leading to bioaccumulation in the food chain (Greaver et al., 2012). In agricultural soils in Europe, however, fertilizer inputs of sulphur have increased to combat crop sulphur deficiencies as a result of sulphur emission controls (Bender & Weigel, 2011). Acidification is reversible, evident by increases in soil pH following decreased sulphur emissions, although the recovery time varies; 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% of sulphur deposition on land is on natural, nonagricultural land (Dentener et al., 2006). Application of lime to naturally acidic forest soils can cause further acidification of deep soil layers while increasing decomposition in surface litter, with no improvement in tree growth (Lundström et al., 2003).

Wider effects of acidification are starting to be understood through long-term monitoring. Decreased

Fig. 3 Uneven global distribution of soils sensitive to pollution by (a) acidification and (b) eutrophication (measured by soil C : N) compared to the uneven distribution of atmospheric (c) sulphur and (d) nitrogen pollution. Soils most sensitive to acidification have low base saturation and cation exchange capacity, as defined by Kuylenstierna et al. (2001). Acidification is caused by both sulphur and nitrogen. Eutrophication is caused by nitrogen. Soil data in (a) and (b) were produced using the ISRIC-WISE derived soil properties (ver 1.2) (Batjes, 2012) and the FAO Digital Soil Map of the World. Atmospheric deposition data in (c) and (d) 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.
organic matter decomposition due to acidification has increased soil carbon storage in tropical forests (Lu et al., 2014). However, in temperate forest soils, acidification can lead to reduced C : N ratios in soil, which in turn increases nitrification (Aber et al., 2003), but on already acidic soils reduces nitrification. In wetland soils, methane (CH₄) emissions have also been suppressed by sulphur deposition (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 as the sources are more varied, and include extensive agriculture fertilizer application, ammonia derived from livestock operations and biomass burning in addition to fossil fuel combustion (Fig. 3b). Regions with deposition in excess of 20 kg N ha⁻¹ yr⁻¹ in 2001 were western Europe, South Asia (Pakistan, India, Bangladesh) and eastern China (Vet et al., 2014), although extensive areas with 4 kg N ha⁻¹ yr⁻¹ were found across North, Central and South America, Europe and sub-Saharan Africa. By contrast, ‘natural’ deposition in unimpacted areas is around 0.5 kg N ha⁻¹ yr⁻¹ (Dentener et al., 2006). While emissions related to fossil fuel combustion have declined along with sulphur across Europe, agricultural sources of nitrogen are likely to stay constant in the near future across Europe (EEA, 2014), while overall global emissions are likely to increase (Galloway et al., 2008). Nitrogen deposition in China’s industrialized and intensively managed agricultural areas in the 2000s was similar to peaks in Western Europe during the 1980s before mitigation (Liu et al., 2013a,b).

Deposition of nitrogen induces a ‘cascade’ of environmental problems, including both acidification and eutrophication that can have both positive and negative effects on ecosystem services (Galloway et al., 2003). Excluding agricultural areas where nitrogen is beneficial, 11% of land surface received nitrogen deposition above 10 kg N ha⁻¹ yr⁻¹ (Bouwman et al., 2002; Dentener et al., 2006; Fig. 3d). In Europe, eutrophication has and will continue to impact a larger area than acidification (EEA, 2014).

Nitrogen fertilization can increase tree growth (Magnani et al., 2007) and cause changes in plant species and diversity (Bobbink et al., 2010), which in turn will alter the amount and quality of litter inputs in to soils, notably the C : N ratio and soil–root interactions (RoTAP, 2012). However, increased carbon sequestration (Reay et al., 2008) may be offset by increased emissions of the greenhouse gases N₂O and CH₄ (Liu & Greaver, 2009). Long-term changes caused by nitrogen deposition are uncertain as transport times vary between environmental systems; and the only way to remove excess nitrogen is to convert it to an unreactive gas (Galloway et al., 2008).

**Heavy metal deposition**

Heavy metal emissions are associated with coal combustion and heavy industry. In the UK, deposition is responsible for 25–85% of inputs to UK soils (Nicholson et al., 2003). In Europe, the areas at risk from cadmium, mercury and lead deposition in 2000 were 0.34%, 77% and 42%, respectively, although emissions are declining (Hettelingh et al., 2006). Tighter legislation to control industrial emissions of heavy metals is helping to reduce the environmental load of heavy metals in many regions, although rapid industrial growth in some regions such as East Asia is increasing pressures on soil from heavy metal deposition. Global heavy metal emissions and deposition are poorly understood in comparison with sulphur and nitrogen, although the on-site impact of heavy metal contamination has been well studied (Guo et al., 2014). Metals in bioavailable form have toxic effects on soil organisms and plants, influencing the quality and quantity of plant inputs to soils, rate of decomposition and, importantly, can bio-accumulate in the food chain. Some heavy metals will persist for centuries as they are strongly bound to soil organic matter (RoTAP, 2012), although they can be mobilized to bioavailable form following drought-induced acidification, drainage and soil erosion (Tipping et al., 2003; Rothwell et al., 2005).

While the direct impacts of sulphur, nitrogen and heavy metals 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 seminatural 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 across Asia, Africa and South and Central America where soil properties and environmental conditions differ. Models need to be developed to examine the combined effects of air pollutants and their interactions with climate change impacts and feedbacks on greenhouse gas balances and carbon storage (Spranger et al., 2008; RoTAP, 2012). Air quality, biodiversity and climate change polices 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 quality (Reis et al., 2012; RoTAP, 2012; Aherne & Posch, 2013). Indirect impacts on soils can be addressed largely by preventing the pollution at source, or by mitigating the adverse effects where these have already occurred. Air pollution control on coal burning and increased car and fleet efficiency standards has been effective in reducing sulphur deposition in many areas of the world, particularly in Europe since the 1970s. Substitution of coal with bioenergy might also reduce sulphur emissions, but unless burned cleanly in a controlled way, can also release pollutants to the air. In terms of nitrogen, ammonia abatement techniques when fertilizers are spread (e.g. slurry injection) are helping to reduce N deposition (Sutton et al., 2007).

Existing policies and practices that alleviate global change pressures on soils from land use and management

The previous text has highlighted specific anthropogenic activities that exert or alleviate pressures on soils. Actions that alleviate pressures on soils driven by land-use change and land management can be broadly divided into three categories, those that:

1. Prevent conversion of natural ecosystems to other uses (e.g. protected areas, reduced deforestation, prevention of wetland drainage, intensification rather than extensification);
2. Prevent soil degradation (erosion control, fire management, reduced tillage/conservation agriculture, long-term fallows, flood protection, use of organic amendments, intercropping, improved rotations); and
3. Result in soil/ecosystem restoration (e.g. peatland rewetting, afforestation, revegetation on degraded lands, improved grass varieties, appropriate animal stocking densities, bioremediation).

Policies to encourage such actions were recently reviewed by Bustamante et al. (2014) and include the following:

1. Economic incentives, for example, special credit lines for low carbon agriculture and forestry practices and projects, payment for ecosystem services (such as carbon storage) and tradable credits such as carbon,
2. Regulatory approaches, for example enforcement of environmental law to protect natural areas, set-aside policies,
3. Research, development and diffusion investments, for example increase of resource use efficiency, livestock improvement,
4. Information and certification schemes, for example in China, forest certification to promote sustainable forest management, state regulation for protecting mandatory arable lands, protection projects on Tibetan grasslands, a national wetland protection programme, and the ‘grain for green’ programme.

Many of these actions and policies are not directed at soil conservation, but nevertheless have an effect on soil quality. Two of the main pieces of international policy that have served to reduce pressures on soils, directly and indirectly, are the United Nations Convention to Combat Desertification (CCD) and the UNFCCC. In general, policies and actions are important at all scales from international conventions to local action, and local activity is encouraged by policies at regional, national and global level. Policies to sustainably increase land productivity, for example, can prevent land-use change, and there are various other supporting actions that can help deliver improvements, for example agricultural research, technology transfer, knowledge transfer and improved rural infrastructure. Some examples of policies that impact on land management and soil quality are given below.

**United Nations Framework Convention on Climate Change and other climate specific policies**

Soil carbon storage and nutrient cycling as climate services are being increasingly recognized for example under UNFCCC as part of national reporting and accounting, as part of life cycle greenhouse gas assessments for biofuels, in various regional initiatives and national efforts.

The UNFCCC is an international treaty, which came into force in 1994, setting an overall framework for intergovernmental efforts to tackle the challenge posed by climate change. The requirements for the 196 country Signatories (or ‘Parties’) to the UNFCCC include adopting national mitigation policies and publishing national inventories of anthropogenic emissions and sinks of greenhouse gases including activities on the land such as afforestation, deforestation, agricultural management and wetland drainage and rewetting. Developed country signatories have legally binding targets under the Kyoto Protocol and can count land based emissions or sinks towards meeting these targets, thus incentivizing activities that protect soil carbon. Developing countries currently have voluntary targets and several countries have made pledges that include reduced deforestation (e.g. Brazil and Indonesia) or afforestation (e.g. 400 000 km² in China). Under the Clean Development Mechanism (CDM), developed countries can fund projects in developing countries that
generate certified emission reduction credits (CERCs). China, for example, has the largest number of CERCs in the world (IFPRI, 2011). Brazil also has 180 CDM projects, the third largest number of CERCs after China and India (Cole & Liverman, 2011). A number of projects in Africa, North America and South Asia have a significant component for soil greenhouse gas emission reduction of soil carbon sequestration, financed through the Verified Carbon Standard or the American Carbon Registry.

As part of negotiations leading to the new climate treaty in Paris in December 2015, all parties will be required to submit Intended Nationally determined Contributions (INDCs). The new treaty will also include provision for REDD+ (reduced Emissions from Deforestation and Degradation, including management of forests and enhancement of forest carbon stocks). This could go some way to protecting forest soils, and negotiations have been intense around methods for monitoring reporting and verification, with key issues such as permanence (the risk the forest may be lost at a later date due to management or environmental change) and leakage (displacement of land-use change to other areas), and how to finance such activities.

**United Nations Convention to Combat Desertification**

The CCD entered into force in December 1996; today 179 countries acknowledge it as a legally binding framework to tackle land degradation and promote sustainable development in fragile ecosystems. The Global Mechanism was established under the convention to ‘promote actions leading to the mobilization and channelling of substantial financial resources, including for the transfer of technology, on a grant basis, and/or on concessional or other terms, to affected developing country Parties’. In September 2011, the United Nations General Assembly declared a goal of building a world with no land degradation. In October 2011, parties to the CCD issued a declaration calling for zero land degradation and for adopting sustainable land management as a way to achieve sustainable development.

**Millennium Development Goals**

Of the eight MDGs (UNDP, 2014a), soil protection is most relevant to the goal to ensure environmental sustainability, because soils are critical in underpinning environmental sustainability (Smith et al., 2015). A complementary MDG, to develop a global partnership for development, will improve the governance structure to deliver soil security. The other MDG to which soils play a critical contribution is the goal to eradicate extreme poverty and hunger, with the role of soils in supporting food provision critical for the latter part of this MDG (Smith et al., 2015). The MDGs are currently being revisited to set a post-2015 development agenda (UNDP, 2014b), with discussion around the themes of localizing the post-2015 development agenda, helping to strengthen capacities and build effective institutions, participatory monitoring for accountability, partnerships with civil society, engaging with the private sector, and culture and development. The key emerging principles from the dialogue are participation, inclusion, and the need for strengthened capacities and partnerships (UNDP, 2014b). It is important that soils play their role in delivering this post-2015 agenda.

**Protected areas and the Convention on Biological Diversity**

Many measures to protect biodiversity and vulnerable habitats also protect the soils underpinning them, so numerous conservation actions around the world serve to protect soils, even if this was not the primary aim (Smith et al., 2013). Between 1990 and 2010, the amount of forest land designated primarily for the conservation of biological diversity increased by 35%, indicating a political commitment to conserve forests. These forests now account for 12% of the world’s forests (FAO, 2010). In India, a Supreme Court ruling in 2011 on effective self-governance of ‘common’ or communal land by local communities may help to protect these valuable resources, and thereby the soils that underpin them. Soil biodiversity is known to be important for soil function (Bodelier, 2011), yet it rarely receives the attention enjoyed by larger flora and fauna within the ecosystem.

**Reduced deforestation and forest management**

Various actions have been implemented to reduce deforestation (Bustamante et al., 2014) and to reduce the impact of forestry activities, such as reduced impact logging. UNFCCC, carbon markets and other international environmental programmes have contributed to global efforts to reduce deforestation in addition to other sustainable natural resource management programmes in countries and by industry. For example, zero deforestation commitments made by several companies (many made in 2014), and activities from bodies such as the Roundtable for Sustainable Palm Oil and the Forest Stewardship Council (FCO) certification scheme. Land improvement has increased in East Asia between 1981 and 2006 despite population increase, attributed largely to policies promoting tree planting and forest plantation programmes in China and Korea.
In Brazil, deforestation was rapidly reduced after national laws and regulations were enacted to protect forests in the 1990s and early 2000s (including the soy moratorium and the forest code), followed up by state and municipal governments setting further by-laws enforcing the deforestation moratorium (Bustamante et al., 2014).

**Agricultural policies and practices**

The pressures on soils imposed by land-use intensity change can be mitigated by regulation of overgrazing and reduction of overstocking on grazed grasslands, return of crop residues to the soil, reduced tillage, best management practices, targeted nutrient management and precision farming on croplands, and wetland/floodplain restoration. These actions have been encouraged by various policies. Some examples include The EU set-aside programme of the 1990s encouraged less intensive use of agricultural land where production is low and environmental impacts are high. The EU Common Agricultural Policy ties agricultural subsidies to implementation of best management practices and environmental protection, for example through pillar 2 (rural development programmes) providing crop insurance for lower fertilizer application rates; in Africa, policies for integrated land management to help protect vulnerable soils; China’s conservation tillage program (2012–2030); the USA Conservation Reserve Program (set aside marginal lands, steep slopes).

**Conclusion: Keeping soils central to the science and policy agendas**

The International Year of Soils in 2015 is an excellent opportunity to raise the profile of soils in the minds of national and international policymakers, land managers, timber and agro-industries, and the public. Ensuring that vulnerable and high environmental value soils (e.g. peatlands) are considered when making policies and decisions about which habitats and ecosystems to convert or to protect, will help to reduce the pressure on soils particularly vulnerable to global change drivers such as land use and land management, and maintain important ecosystem services. This is in part happening with agendas around valuation of ecosystem services and life cycle assessments of impacts of land-use change that include soil carbon. At a time when governments are negotiating a legally binding climate change treaty and making national targets for greenhouse gas reduction, and revisiting the MDGs, keeping soil carbon and nitrogen central to land based greenhouse gas monitoring and reporting will maintain awareness with policymakers and industries with emissions reduction targets. Both science and policy agendas are increasingly concerned with long-term food security, ensuring that soils are central to considerations of how to achieve ongoing increases in production will enable those increases to be more sustainable into the future.

Research and policy regarding soil quality and sustainability is abundant, but patchy and disjointed. To ensure that soils are protected as part of ongoing wider environmental and sustainable production efforts, soils cannot, and should not, be considered in isolation of the ecosystems that they underpin, but the role of soils in supporting ecosystems and natural capital needs greater recognition (Robinson et al., 2013, 2014). This can, in part, be enhanced through education and awareness raising which has started during the International Year of the Soils in 2015. The time is ripe to consider a global soil resilience programme, under the auspices of a global body such as the UN or one of its delivery agencies such as the FAO to monitor, recover or sustain soil fertility and function, and to enhance the ecosystem services provided by soils. The lasting legacy of the International Year of Soils in 2015 should be to bring together robust scientific knowledge on the role of soils and to put soils at the centre of policy supporting environmental protection and sustainable development.

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