# Global Change Pressures on Soils from Land Use and Management

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

**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 minimise 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. 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.
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 minimise 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. 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.
1. 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 policy makers 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 e.g. 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 Mkm$^2$, or >50 percent of Earth’s ice-free land area (Hooke et al. 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%, shrub-covered areas 9.5 %. Artificial surfaces (including urbanised areas) occupy 1 %. Land degradation can be found in all land cover types. Degraded land covers approximately 24% of the global land area (35 Mkm$^2$). 23% 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.

2. 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
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 (Soares-Filho et al., 2014; Nolte et al., 2013) 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 post-harvest. 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 oxidise. 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, 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.

2.1. 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; Don et al. 2011; Poeplau et al. 2011; Bárcena et al. 2014; Murty et al. 2002; 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 increase 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).

2.2. 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 (Bondeau et al. 2007; Lindeskog et al. 2013; Drewniak et al. 2013; Jain et al. 2005). Most DGVMs do not currently model peatland soils. In Tables 1 and 2, and Figures 1 and 2, we show impacts of past land cover and management change on soil carbon and nitrogen as calculated by three DGVMs: ISAM (Jain et al. 2013; El-Masri et al. 2013; Barman et al. 2014 a,b); LPJ-GUESS (Smith et al. 2001; Lindeskog et al. 2013); and 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 (Hurtt et al. 2011). The LPJmL group combined 3 land use change data sets with the geographic distribution of global agricultural lands in the year 2000. All models were run with historical climate and CO$_2$, 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 et al. 2014). Table 2 and Figure 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 there 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 processes included in the models. For example, soil carbon losses are higher in the LPJmL model (Table 2, Figure 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 slowing soil carbon decomposition (Figure 2). Treatment of management processes turns out to be an important differentiator. ISAM shows strong decreases of soil carbon in some regions e.g. the southern Boreal zone (Figure 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 (Figure 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 Section 4.3), and may also be partly be 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 modeling soil carbon, in particular, the strength of the relationship between increasing temperatures and the increasing rate of soil carbon decomposition ($Q_{10}$) causing climate-carbon feedbacks via CO$_2$ 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 890-1600PgC (95% confidence interval) from the Harmonized World Soil Data Base (FAO/IIASA/ISRIC/ISSCAS/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 NPP and temperature strongly influenced soil carbon stocks, much more so than in the observational data, and differences between models was found to be largely due to the representation of NPP and the parameterization of soil decomposition sub-models. 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).

2.3 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 Southeast 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 km² of drained organic soils under cropland and grassland, with total GHG
emissions (N₂O plus CO₂) of 0.9 Pg CO₂eq yr⁻¹ in 2010, with the largest contributions from
Asia (0.44 Pg CO₂eq yr⁻¹) and Europe (0.18 Pg CO₂eq yr⁻¹; FAOSTAT, 2013; Tubiello et al.,
2015). 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 Pg CO₂ yr⁻¹
in 1990 to 1.30 Pg CO₂ yr⁻¹ in 2008, despite a decreasing trend in developed countries, from
0.65 to 0.49 Pg 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 Pg CO₂ yr⁻¹
(Hooijer et al., 2010). Conversion of peatlands in Southeast 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 km² of wetland soils were shifted to cultivated arable lands, which led to a SOC loss of
5.5 Pg CO₂, 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 CO₂ ha⁻¹ over the 1960 – 2006 period), most probably as a consequence of
intensified soil drainage practices for cultivation purposes.

3. 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 summarised in Table 3.

[Table 3 here]

3.1 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 section 3.5), 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 2000’s was twice that in 1980’s (State Bureau of Statistics-China, 2005).

Between 50-60% 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, over-use of synthetic nitrogen fertilizers is causing soil acidification and increased decomposition of soil organic matter, leading to loss of soil function in over-fertilized soils (Ju et al., 2009; Tian et al., 2012).

Use of fertilisers 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 nitrous oxide (N$_2$O) to atmosphere when both direct (from soils) and indirect (e.g. downstream from nitrate leaching) emissions are considered (Galloway et al., 2004), and N$_2$O has ~300 times the radiative forcing of carbon dioxide (IPCC, 2007). Recent research indicates that the relationship between nitrogen application and N$_2$O emissions is non-linear, resulting in an increasing proportion of added N being emitted, as application rate increases (Philibert et al., 2013; Shcherbak et al., 2014).
India, and the United States account for ~56% of all N\textsubscript{2}O emissions from croplands, with 28% from China alone (West \textit{et al.}, 2014). Overuse of nitrogen and phosphorus fertilizer can contribute to eutrophication of water bodies, adversely affecting water quality and biodiversity (Galloway \textit{et al.}, 2003, 2004, 2008).

Nutrient use-efficiency can be significantly increased, and nitrate losses to water and N\textsubscript{2}O emissions can be reduced, through changes in rate, timing, placement, and type of application, as well as balancing fertilization (Venterea \textit{et al.}, 2012; Snyder \textit{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 \textit{et al.}, 2014).

3.2 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 (Paustian \textit{et al.}, 1997; Smith, 2008; Smith, 2012). As recently reviewed by Paustian \textit{et al.} (2015), 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 \textit{et al.}, 2010), greater residue retention (Wilhelm \textit{et al.}, 2004), and use of cover crops during fallow periods to provide year-round C inputs (Burney \textit{et al.}, 2010; Poeplau & Don 2015). Several studies report that soil carbon increases in croplands under no-till management (West & Post, 2002; Ogle \textit{et al.}, 2005). However, the carbon benefits of no-till may be limited to the top 30cm of soil (Powelson \textit{et al.}, 2014). Baker \textit{et al.} (2007) found that total soil carbon was similar in non-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 sub-surface 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 un-composted organic materials (Ryals \textit{et al.}, 2015; Lehmann \textit{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 N$_2$O emissions from N fertilizers (Liu et al., 2012; Huang et al., 2012), as well as
enhancing soil fertility (Woolf et al., 2010). Paustian et al. (2015) provide a recent review of
soil sequestration measures.

### 3.3 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 (section 3.2).

Irrigation can increase soil salinity in dry regions with high salt content in the subsoil
(Ghassemi et al., 1995; Setia et al., 2011). 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 (Setia et al., 2011a;
Chowdhury et al., 2011; Ben-Gal et al., 2009). 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 mineralisation
of organic C to CO$_2$) which could increase SOC content if the C input were unchanged.
Using a modified Rothamsted Carbon model (RothC) 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, e.g. 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, e.g. storage systems, terracing and other methods for collecting and storing runoff,
are required to make best use of the limited water resource.

3.4 Harvest frequency

Approximately 9% of crop production increases from 1961-2007 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 net primary production (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., 2013). The increase results from fewer crop failures, fewer fallow years, and an increase
Increasing harvest frequency can reduce soil quality by e.g. 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 fertilisers contributing to pollution (section 3.1). The net effect will depend on the effectiveness of the management practices followed.

### 3.5 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 (Batey, 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, though 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).

### 3.6 Livestock density

Livestock production is projected to increase significantly in order 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 (Bouwmann 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 (Eshel 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. optimised stocking density) can reduce soil degradation, and thereby maintain and enhance organic matter content (McSherry & Ritchie, 2013; see sections 3.2 and 4.3), and can reduce soil compaction, thereby increasing infiltration and water storage and reduce risk of runoff and flooding downstream (Marshall et al., 2009).

4. Other land management

4.1 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 (DBC) 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.

**4.2 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 Southeast Asia where urbanisation 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).

4.3 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 un-grazed 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
Different grazing strategies, especially in the semi-natural dryland biomes, have large implications for vegetation and the carbon balance (Yates et al. 2000). Under certain conditions, grazing can lead to increased annual net primary production 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 recognised that the potential effects of management on carbon storage in grassland and dryland soils are substantially greater than that of climate change or CO$_2$ enhancement (Ojima et al. 1993), and Henderson et al. (2015) estimated that the optimization of grazing pressure could sequester 148 Tg CO$_2$ yr$^{-1}$.

### 4.4 Artificial surfaces, urbanisation 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 (Tóth 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) include impacts on water resources (e.g. reduction of rainfall absorbed by the soil, reduction of soil water holding capacity affecting flooding), on soil biodiversity when sealing prevents recycling of dead organic material (Marfenina et al. 2008), on the carbon cycle due to topsoil and vegetation removal (Davies et al., 2011).
Appropriate mitigation measures can be taken in order to maintain some of the soil functions. In urban planning management, objectives to reduce the impact of soil sealing include: i) preventing the conversion of green areas, ii) re-use of already built-up areas (e.g. brownfield sites Meuser, 2010; Hester & Harrison, 2001 – though 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. In order to deliver this mitigation a number of actions are necessary, e.g. 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 insufficient incentives to re-use 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, re-use of extracted topsoil, de-sealing and is incentivised by land take fees and application of environmental cost calculations.

5. 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 into surface waters (Galloway et al., 2008) and/or transported to other areas by soil erosion (Ravi et al., 2010). Pollutants disrupt natural biogeochemical 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 into the atmosphere by fossil fuel combustion (e.g. power generation, industry and transport) and non-combustion 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, 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 (Figure 3). For instance, global emissions of sulphur and nitrogen have increased 3-10 fold 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 (LRTAP) 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).
5.1 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\(^{-1}\) where China and Republic of Korea, western Europe and eastern North America (Vet et al., 2014; Figure 3a). Deposition in un-impacted areas is <1 kg S ha\(^{-1}\) yr\(^{-1}\) (Figure 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 (SO\(_x\)) react with water to form sulphuric acid (H\(_2\)SO\(_4\)). Excess inputs of acidity (H\(^+\)) displace soil base cations (e.g. calcium (Ca\(^{2+}\)) and magnesium (Mg\(^{2+}\))) 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, the 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; Figure 3c). Wetland 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 mineral acidity in naturally acidic organic soils (Krug and Frink, 1983).

Decreased soil fertility or ‘sterilisation’ due to loss of nutrients and mobilisation of toxic metals, particularly Al, are caused by acidification. Impacts in Scandinavia over the 1960s-80s 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, non-agricultural land (Dentener et al., 2006). Application of lime to naturally acidic forest soils can cause further acidification of deep soil layers whilst 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 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 (CH$_4$) 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).

5.2 Nitrogen deposition

Nitrogen deposition covers a wider geographical area than sulphur, as the sources are more varied, including extensive agriculture fertilizer application, ammonia derived from livestock operations, and biomass burning in addition to fossil fuel combustion (Figure 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 un-impacted 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), whilst 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., 2013).

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\(^{-1}\) (Dentener et al., 2006; Bouwman et al. 2002; Figure 3d). In Europe, eutrophication has and will continue to impact a larger area than acidification (EEA, 2014).

Nitrogen fertilisation 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 into 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 N\(_2\)O and CH\(_4\) (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).

5.3 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 are helping to reduce the environmental load of heavy metals in many regions, though 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 to 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 mobilised to bioavailable form following drought-induced acidification, drainage and soil erosion (Tipping et al., 2003; Rothwell et al., 2005).
Whilst 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 semi-natural systems (Greaver et al., 2012). Process understanding 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 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 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).

6. 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, re-vegetation 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:

a) Economic incentives, e.g., 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,
b) Regulatory approaches, e.g. enforcement of environmental law to protect natural areas, set-aside policies,
c) Research, development and diffusion investments, e.g. increase of resource use-efficiency, livestock improvement,
d) Information and certification schemes, e.g. 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 United Nations Framework Convention on Climate Change (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, e.g. 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.
6.1 United Nations Framework Convention on Climate Change (UNFCCC) and other climate specific policies

Soil carbon storage and nutrient cycling as climate services are being increasingly recognised e.g. 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 incentivising 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. 400000 km$^2$ in China). Under the Clean Development Mechanism (CDM) developed countries can fund projects in developing countries that generate certified emission reduction credits (CERs). China, for example, has the largest number of CERs in the world (IFPRI, 2011). Brazil also has 180 CDM projects, the third largest number of CERs after China and India (Cole & Liverman, 2011). Paustian et al. (2015) list several projects in Africa, North America and South Asia that 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 INDCs (Intended Nationally determined Contributions). 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.
6.2 United Nations Convention to Combat Desertification (CCD)

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.

6.3 Millennium Development Goals (MDGs)

Of the eight MDGs (UNDP, 2014a), soil protection is most relevant to the goal to ensure environmental sustainability, since 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 plays 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 localising 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.

6.4 Protected areas and the Convention on Biological Diversity (CBD)

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 percent, indicating a political commitment to conserve forests. These forests now account for 12 percent 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.

6.5 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 programs have contributed to global efforts to reduce deforestation in addition to other sustainable natural resource management programs 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 (RSPO) 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 programs 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).

6.6 Agricultural policies and practices

The pressures on soils imposed by land use intensity change can be mitigated by regulation of over-grazing and reduction of over-stock ing 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).

7. 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 policy makers, 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 Millennium Development Goals, keeping soil carbon and nitrogen central to land based greenhouse gas monitoring and reporting will maintain awareness with policy makers 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 on-going 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 on-going 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.

Acknowledgements

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(NNX14AD94G).

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For Review Only


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Tables

**Table 1.** Observed and modelled soil carbon change (%) when converting from land cover classes in the left hand column to land cover classes listed across the top. Results are from meta-analysis of observations from the sources listed below. Model results (range across three models) are shown for comparison in square brackets, range across the ISAM, LPJml, and LPJ_GUESS models (see text), although note this calculated as difference in soil carbon under the different land classes in 2010 and is thus not modelled loss/gain after a conversion. Negative numbers represent loss of soil carbon.

<table>
<thead>
<tr>
<th></th>
<th>Regrowth Forest</th>
<th>Tree plantation</th>
<th>Grassland</th>
<th>Pasture</th>
<th>Cropland</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Forest</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Global Trop.</td>
<td>-9% (2)</td>
<td>-13% (3)</td>
<td>+8% (3)</td>
<td>-12% (2)</td>
<td>-42% (3)</td>
</tr>
<tr>
<td>Temp.</td>
<td></td>
<td></td>
<td>[-40 to -63%]</td>
<td></td>
<td>-41% (1)</td>
</tr>
<tr>
<td>Boreal</td>
<td></td>
<td></td>
<td>[-30% (2)]</td>
<td></td>
<td>-25% (2)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>-24% (5)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>[-51 to -62%]</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>-52% (1)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>-36% (4)</td>
<td></td>
<td></td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>-24 to -60%</td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td>-31% (1)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>[-63 to -65%]</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Grassland</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Global Trop.</td>
<td></td>
<td></td>
<td>[-1 to +15%]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Temp.</td>
<td></td>
<td></td>
<td>[-2 to -6%]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Boreal</td>
<td></td>
<td></td>
<td>[-15 to -53%]</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>[-70 to -79%]</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Pasture</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Global Trop.</td>
<td>-10% (3)</td>
<td></td>
<td></td>
<td></td>
<td>-59% (3)</td>
</tr>
<tr>
<td>Temp.</td>
<td></td>
<td></td>
<td>[-19 to +0.5%]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Boreal</td>
<td></td>
<td></td>
<td>[-17 to -55%]</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>[-28 to -55%]</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Cropland</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Global Trop.</td>
<td>+53% (3)</td>
<td>+18% (3)</td>
<td>+19% (3)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Temp.</td>
<td>+16% (4)</td>
<td>+29% (2)</td>
<td>+26% (2)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Boreal</td>
<td></td>
<td>+20% (6)</td>
<td>+28% (4)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Footnotes:  

- Broadleaf tree plantations onto prior native forest or pasture did not affect soil C stocks whereas pine plantations reduced soil C stocks by -12 to -15%;  
- Annual crops;  
- Perennial crops;  
- Wei *et al.* (2014a);  
- Don *et al.* (2011);  
- Guo & Gifford (2002; tropical and temperate zones compiled);  
- Poeplau *et al.* (2011);  
- Murty *et al.* (2014);  
- Barcena *et al.* (2014).
**Table 2.** Soil carbon loss due to land use change 1860 to 2010 (PgCO₂)

<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><strong>Mean</strong></td>
<td><strong>79</strong></td>
<td><strong>96</strong></td>
<td><strong>7</strong></td>
<td><strong>186</strong></td>
</tr>
</tbody>
</table>
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>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>
<td></td>
</tr>
<tr>
<td>Use of agrochemicals</td>
<td>Over fertilization</td>
<td>Particularly in some developing countries</td>
<td>Soil acidification, water pollution, N₂O emission and nitrate accumulation</td>
<td>Rate reducing versus 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>Arid/semi-arid regions</td>
<td>Arid/semi-arid regions</td>
<td>Secondary salinization, water scarcity</td>
<td>Competition use of water</td>
<td></td>
</tr>
<tr>
<td>Livestock management</td>
<td>Over-grazing</td>
<td>Largely in developing countries</td>
<td>Soil degradation, water storage, C loss</td>
<td>Forage versus feed crops?</td>
</tr>
<tr>
<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>Safe waste treatment and recycling</td>
<td></td>
</tr>
<tr>
<td>Agriculture in wetlands</td>
<td>Wetland drainage</td>
<td>Developing and transition countries</td>
<td>C loss</td>
<td>Agro-benefit versus natural value</td>
</tr>
</tbody>
</table>
**Figure Legends**

**Figure 1.** Maps of change in soil carbon due to land use change land and land management from 1860 to 2010 from three vegetation models. Pink indicates loss of soil carbon, blue indicates carbon gain.

**Figure 2.** Soil carbon and nitrogen under different land cover types in three different vegetation models (values are the annual average over the period 2001 to 2010).

**Figure 3.** Uneven global distribution of soils sensitive to pollution by (a) acidification and (b) eutrophication (measured by soil C:N) compared to 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.
Response to editor’s and reviewer’s comments on GCB-15-0248

Subject Editor’s Comments to Authors:

Comment: Both reviewers found this review appropriate for GCB, yet the reviews pointed out weaknesses in the manuscript that would require revisiting the structure and scope of the manuscript. I hope that you find these comments helpful if you decide to revise and resubmit this as a new manuscript.
Response: Thank you for these comments. The comments from the editor and the two reviewers have significantly improved the manuscript, so we thank the reviewers / editor for their comments. We have addressed all of the comments in a very substantial revision, as described below.

Reviewer(s)' Comments to Author:

Reviewer: 1

Comments to the Author
General comments

Comment: The objective of this paper is to review the major global pressures on soils, to identify knowledge gaps and putting soils at the centre of policy actions during the International Year of Soils. The authors highlight the importance of soils as an integrated ecosystem property and their role in supporting ecosystem services. A global soil resilience programme is proposed.
In general, I share the view regarding the pressures on soils that are highlighted and reviewed – but I think that several major pressures, especially salinization and compaction are missing and should be included in the review.
Response: Thank you for these comments. We have used them in our revision. We have totally restructured the manuscript and have added sections on salinization (under water management – new section 3.3) and compaction (new section 3.5).

Comment: In several chapters, “intensification” is mentioned as a potential risk for soil degradation. It should be specified what is meant with intensification in different context (especially in the abstract) – land use intensification or crop management intensification. It has been shown in many studies that intensification of cropland (more inputs of fertilizer, lime, amendments etc.) can increase soil fertility. In contrary, intensification in terms of changing land use or change in crop rotations including perennial crops to monocultures with only annual crops will lead to decreased soil fertility.
Response: Thank you. We have removed the section entitled intensification, and have now clarified what we mean by intensification at each usage (now mentioned only 8 times).

Comment: The text in not very focused or concise. The topics are piled up one after one and the reader gets wondering “what is novel with this?”. I miss concluding remarks at the end of each chapter or in a concluding section. Similar reviews have been published before. You should guide the reader by providing a read threat introduced in the introduction.
Response: Thank you – we have added context to the introduction and have restructured the paper to make it more coherent – and tied the concluding remarks together in the final sections (instead of at the end of each section).

Comment: Inconsistent use of units: Different units are used for SOC stocks and changes (C, CO2 and CO2eq). I suggest using the same units throughout. Regarding the use of metric tonne: Although the metric tonne is accepted as a SI-unit it is not a SI-unit per se. Often “t” or Gt are used in the text
but also Pg (line 425). This should be consistent – I prefer the real SI-unit – but this up to the editor
to decide. Also prefixes such as Mega are not used consistently. E.g. in line 166 it is written 500 000
km2, whereas above in the text the M-prefix is used. I would suggest 0.5 Mkm2 here.
**Response:** All units have been harmonised throughout the manuscript. All Gt have been converted
to Pg and all values are expressed in CO\textsubscript{2}-eq.

**Specific comments**

- **Comment:** Line 52: You should specify what you mean with “mining” – in the text you focused on the
  mobilisation of metals from mines. Nutrient mining is also considered to be major threat leading to
  soil degradation.
  **Response:** Nutrient mining is dealt with (briefly) under nutrient management (section 3.1 in the new
  structure). The section on mining (the process of extraction of minerals) has been removed.

- **Comment:** Line 128: A decline of -10% is actually an increase. Either use change “decline” to
  “change” or remove the minus from the figures. Check this for the whole manuscript (e.g. line 137).
  **Response:** It is useful to the reader to indicate plus or minus signs, and also to indicate in the text
  whether it is increasing or decreasing. However we accept the reviewers point so have added
  “(change of – x%)” to the first number in the bracket in each section to make it clear.

- **Comment:** Line 169: Is everybody aware of what the “Annex I” countries are? Please explain.
  **Response:** Have changed to “developed countries”

- **Comment:** Line 181: Table 2 only shows estimated changes in soil carbon stocks – “mineral soil C and
  N concentrations” are not shown in Table 2. The models that were used to derived table 2 are not
  explained and references are not provided. Moreover, the huge differences in model output are
  commented in the text.
  **Response:** We accept these comments. The text, table and figure captions have been extensively re-
  written, models explained, references provided and differences between models commented on.

- **Comment:** Line 241: Delete “land is”
  **Response:** Done

- **Comment:** Line 282: Since the effect of tillage on soil quality has been studied and discussed
  excessively in the literature during the recent 2 decades, I think this would deserve more than 5 lines
  in a review like this.
  **Response:** The short section on tillage has been moved to section on carbon management (new
  section 3.2) and discussed under the broad driver of “reduced disturbance”. We have expanded the
  text but do not attempt a thorough review here as recent reviews dedicated to this topic have done
  so comprehensively. We refer the reader to these recent reviews.

- **Comment:** Line 338: Explain why over-use of N fertilizers should cause soil compaction and increased
decomposition of SOM. Soil compaction is caused by heavy machinery and not by N fertilization.
  Decomposition of nutrient-poor litter may be stimulated by N fertilization – but for SOM it is rather
  the other way round.
  **Response:** This was an editing error and has been removed.

- **Comment:** Chapter 3.2. Water will probably become even more limiting production in several semi-
  arid regions e.g. Sub-Saharan Africa where the human population will probably increase most in the
  future. Due to the severity of water limitation in the future, I suggest to elaborate more on different
  water harvesting methods here, e.g. storage systems, terracing and other methods for collecting and
  storing runoff.
Response: We have added these suggestions in the new section on water management (section 3.3) and have used the reviewer’s suggestions in the closing sentences of this section. Thank you.

Comment: Line 386: yes, but increased harvest frequency can also result in increased soil quality through higher C inputs or N inputs if legumes are used. The net effect will depend on the prevailing alternative management regime.
Response: We have added these points in the revised section on harvest frequency (now section 3.4)

Comment: Line 430: peatands should read peatlands
Response: done

Comment: Line 478: Remediation of contaminated sites is an issue that should be discussed in this context. The problems associated with using “broundfield sites” as mentioned in the text, should be elaborated on.
Response: We have added the issue of remediation to our mention of use of brownfield sites – and added three references.

Comment: Line 531: Most parts of the text are support by appropriate references but not all. In this chapter e.g. there are no references. I would expect at least one for the last sentence in this chapter.
Response: We have added references to all under-referenced sections, and have removed some references in sections were fewer were required – giving a more even distribution of citations between sections in the revision.

Comment: Line 594: yes, but acidification of soil which already have low pH can reduce nitrification.
Response: We have added this point. Thank you.

Comment: Line 595: Is this sentence correct? As I understand – the microbes using sulphate as electron acceptor are more competitive than those using CO2 or acetic acid as terminal electron acceptor since they gain more energy from the oxidation of SOM than methanogens. Sulphate is not the substrate – rather the electron acceptor in the respiration chain.
Response: We have removed this statement.

Comment: Line 619: Please explain why soils with low nitrogen content are most sensitive to eutrophication. I don’t understand this statement. In figure 3, the statement is the reverse – i.e. soil with high C:N-ratio are most sensitive to eutrophication. Why should soils be sensitive to eutrophication at all? Eutrophication is a problem in water bodies – but why should it be a problem in soil?
Response: We agree. We have removed this statement.

Comment: Tables 1. This table is not connected to the text. The models (ISAM and LPFmL) are not explained. Where do these estimates derive from? References are not provided.
Response: The text and table titles have been extensively re-written including more explanation of the models and references.

Comment: Tables 2. This table is not connected to the text. The models (ISAM, ISAM and LPFmL) are not explained. References are not provided.
Response: The text and table titles have been extensively re-written including more explanation of the models and references.

Comment: Fig. 1. The only blue areas that I can see on the map are in northern India or Kashimir. This deserves some explanation in the text. Why did SOC increase in this area?
Response: The maps have been redrawn with results from other models added for comparison, and the text extensively re-written including more complete explanations.

Comment: Fig. 2. Does this figure add anything to our understanding? I think it is redundant. Response: Agreed; figure deleted.

Comment: Fig. 3, legend line 17: Soil may cause eutrophication but soils are not sensitive to eutrophication. Line 18: Eutrophication of fresh-water is often caused by P rather than by N. Please explain why high CN-ratio in SOM should be an indicator for eutrophication. This would mean than forest soils, which usually have higher CN-ratios, contribute more to N-leaching than arable soils. This is not the case. Wetlands with high CN-ratios are reducing N leaching. In general, eutrophication is not a threat to soil and outside the scope of this review. Response: We have removed the statement in the text and in the figure legend.

Reviewer: 2

Comments to the Author

Comment: I appreciate that good reviews are a big task however the (lack of) structure in this review would appear to have made the task even harder. I found the selection of topics quite diverse and lacking in focus – land use/degradation, land use intensity, irreversible change (urban/mining), off site pressures (pollution) have diffuse connections - especially the last two. Response: We agree. We have rationalised the order and focused more on soil management issues, removed the text on mining and pollution, and focused on how the remaining drivers interact with land management pressures on soils for the indirect drivers (which we have retained). We have put the focus more on integrated management for multiple ecosystem services and integrated land use policy.

Comment: In some sections there has been an excellent synthesis to include the latest knowledge in a concise manner (e.g. 3.1. Nutrient management) whilst on the other hand, some sections have been literally thrown together (e.g. 2.2. Impacts of land management resulting in soil degradation). In general, I found it quite difficult to read at times because of its lack of continuity and readability in many cases just throwing a paragraph from a few innocuous references together. It is obvious all of the authors have provided input, but some better than others. Response: We agree, and thank the reviewer for their insights. In a significant restructuring and re-write, we have tried to make each of the section more consistent and synthetic.

More specific comments:

Comment: The preamble of Section 2 provides a good lead in, but section 2.1 is a disjointed collection of meta-analyses. The peatlands section is quite detailed but perhaps out of place, and some of it is replicated in Section 3.5. It is obvious some information has been gleaned from the IPCC Agriculture chapter (as per the respective authorship) but the distinction should be made (e.g. remove reference to Annex I countries), also the tables and graphics relevant to this section do not provide detail of the models except abbreviations. This adds to my comment above that some sections were thrown together, in this case using other documents. I am also curious why in fact there is a need to show three vastly different model outcomes (Table 2) and then provide little detail of why these large differences have occurred. In this section, the paragraph on microbial communities adds little to the review, with minimal key references. Response: In section 2.1 we have retained the findings from the meta-analyses, as these are powerful strands of evidence, but we have summarised in a new table and have added text to
synthesize these findings. The peatland sections have been combined and reference to Annex I (from the Joosten report) has been removed. The text on the models has been extensively re-written with model descriptions, references and explanations of differences. In a time when models are relied on heavily to predict outcomes for ecosystems under different land use and climate, and impacts of ecosystem change on climate, it is worth discussing and understanding the suitability of state of the art models to do this. However some of the large differences were due to different protocols being followed by the models, this has been rationalised making a discussion of the differences more focused. The paragraph on microbial communities has been deleted.

Comment: In Section 2.2, the majority of the information is based on a couple of meta-analyses which could quite easily have been condensed. The section on shifting cultivation needs to be re-written. In the dryland degradation paragraph there is a large slice of text which is nearly word for word from the Delgado-B et al 2013 paper. The grassland section looks to be based on a large slice of information taken straight out McSherry and Ritchie’s analysis and the section on no-till management is scant to say the least.

Response: The short section on tillage has been moved to section on carbon management (new section 3.2) and discussed under the broad driver of “reduced disturbance”. We have expanded the text but do not attempt a thorough review here as recent reviews dedicated to this topic have done so comprehensively. We refer the reader to these recent reviews. We have combined the sections on grassland management and dryland degradation (in a new section 4.3) but have retained the key findings from these two excellent and powerful meta-analyses. We have improved the referencing (now citing the source at the start and end of the findings presented) to ensure that the provenance of the values presented are clear.

Comment: Section 3 on land use change is well written but only captures a few key references. Other than Nutrient management (see above), the other sections do not say much with scant referencing. Greenhouse should be excluded from the section on harvest frequency. The section on forest harvest and wetland drainage needs to be totally revised as it just reads like a number of one liners and disjointed topics.

Response: All sections have been improved with regards to quantitative information on how these managements affect soils. We have added references to all under-referenced sections, and have removed some references in sections were fewer were required – giving a more even distribution of citations between sections in the revision. The text on greenhouse growing has been deleted. The section on forest harvest and wetland drainage have been rewritten, and combined with other sections on forests and peatlands in our restructuring of the manuscript.

Comment: The sections on sealing and offsite pressure look out of place in this specific review. These could be replaced by sections on soil chemical and physical changes. Sections 6 and 7 do not say much that has not already been said in earlier sections and are large sections from other documents. Section 7 is very much focused on specific topics e.g. REDD and CDM.

Response: The section on sealing has been merged into a new section entitled “Artificial surfaces, urbanisation and soil sealing” (section 4.4), but the section on mining has been deleted. The “offsite pressures” section has been retained, but reduced and tied in with how they interact with integrated land management pressures on soils in a section now called “Anthropogenic environmental change pressures that interact with land management pressures on soils” (section 5). Section 6 has been removed and any insights woven into earlier sections. Section 7 (now section 6) has been further developed to relate better to specific policy actions, but new sections have been added to make this more comprehensive and the whole section has been re-organised.
Comment: I appreciate the time the authors have spent putting this together but it needs a different structure altogether and exacting reviews. At the moment it is far too disjointed and inconsistent in style and lacks readability.

Response:
The structure has been completely revised, largely following the advice of the reviewer – thank you for these suggestions. The individual sections have been improved, and we have revised the whole document to make it more consistent. Thank you for your comments – you will see that we have used them to structure our revision.