



## 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.

## Global Change Pressures on Soils from Land Use and Management

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57

58

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59 **Abstract**

60

61 Soils are subject to varying degrees of direct or indirect human disturbance, constituting a  
62 major global change driver. Factoring out natural from direct and indirect human influence is  
63 not always straightforward, but some human activities have clear impacts. These include land  
64 use change, land management, and land degradation (erosion, compaction, sealing and  
65 salinization). The intensity of land use also exerts a great impact on soils, and soils are also  
66 subject to indirect impacts arising from human activity, such as acid deposition (sulphur and  
67 nitrogen) and heavy metal pollution. In this critical review, we report the state-of-the-art  
68 understanding of these global change pressures on soils, identify knowledge gaps and  
69 research challenges, and highlight actions and policies to minimise adverse environmental  
70 impacts arising from these global change drivers.

71

72 Soils are central to considerations of what constitutes sustainable intensification. Therefore,  
73 ensuring that vulnerable and high environmental value soils are considered when protecting  
74 important habitats and ecosystems, will help to reduce the pressure on land from global  
75 change drivers. To ensure that soils are protected as part of wider environmental efforts, a  
76 global soil resilience programme should be considered, to monitor, recover or sustain soil  
77 fertility and function, and to enhance the ecosystem services provided by soils. Soils cannot,  
78 and should not, be considered in isolation of the ecosystems that they underpin and vice  
79 versa. The role of soils in supporting ecosystems and natural capital needs greater  
80 recognition. The lasting legacy of the International Year of Soils in 2015 should be to put  
81 soils at the centre of policy supporting environmental protection and sustainable  
82 development.

## 83 1. Introduction

84

85 2015 is the International Year of Soil. This represents an ideal time to take stock of scientific  
86 knowledge about the changing global pressures that humans are exerting on soils. 2015 is  
87 also the year when policy makers will adopt a new legally-binding climate agreement under  
88 the United Nations Framework Convention on Climate Change (UNFCCC), with individual  
89 countries and businesses making policies and targets on greenhouse gas emissions and  
90 removals. Soils storage and cycling of carbon and nitrogen are part of emissions and  
91 removals from the land sector. Furthermore, 2015 is the year when countries will shape and  
92 adopt a new development agenda that will build on the Millennium Development Goals  
93 (MDGs). With increasing population, issues such as food security, water security, energy  
94 security (including bioenergy production) and sustainable integrated land and resource  
95 management are central to many development research and policy agendas. Soils underpin  
96 the provision of many ecosystem services related to development.

97

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

107

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

115

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

124

125 Land use change has been accelerated by population increases and migration as food, shelter,  
126 and materials are sought and acquired. It is estimated that humans have directly modified at  
127 least 70 Mkm<sup>2</sup>, or >50 percent of Earth's ice-free land area (Hooke et al. 2012). The new  
128 Global Land Cover Share-database (Latham et al., 2014) represents the major land cover  
129 classes defined by the FAO. Croplands and grasslands (including both natural grasslands and  
130 managed grazing lands) each covered 13.0 %. "Tree-covered areas" (i.e. both natural and  
131 managed forests) covered 28%, shrub-covered areas 9.5 %. Artificial surfaces (including  
132 urbanised areas) occupy 1 %. Land degradation can be found in all land cover types.  
133 Degraded land covers approximately 24% of the global land area (35 Mkm<sup>2</sup>). 23% of  
134 degrading land is under broadleaved forest, 19% under needle-leaved forests and 20-25% on  
135 rangeland (Bai *et al.*, 2008).

136

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

143

## 144 **2. Land use/land cover change**

145

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

150 well as soil erosion.

151

152 Deforestation affected an estimated 13 million hectares per year between 2000 and 2010; net  
153 forest loss was 5.2 million hectares per year (FAO, 2010). Most of this recent deforestation  
154 has taken place in tropical countries (FAO, 2010; Hansen et al., 2013). Over 50% of tropical  
155 forest loss occurred in Brazil and Indonesia, largely driven by a few commodities: timber,  
156 soy, beef, and oil palm (West *et al.*, 2014). There has been a reduced rate of deforestation in  
157 some regions over the last decade, most notably Brazil (INPE, 2014), largely because of land  
158 use conservation policies (Soares-Filho *et al.*, 2014; Nolte *et al.*, 2013) as well as economics.  
159 Most developed countries with temperate and boreal forest ecosystems – and more recently,  
160 countries in the Near East and Asia – are experiencing stable or increasing forest areas in  
161 contrast to the large scale historic deforestation in these regions, with afforestation reported in  
162 Europe, USA, China, Vietnam and India (FAO 2013).

163

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

175

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

181

182 ***2.1. Observations of impacts of land cover change***

183



184 Table 1 presents the results of different meta-analysis studies across different climatic zones  
185 that compared the impacts of land use changes on SOC (Guo & Gifford 2002; Don *et al.*  
186 2011; Poeplau *et al.* 2011; Bárcena *et al.* 2014; Murty *et al.* 2002; Wei *et al.* 2014a). Changes  
187 in SOC after the conversion of forests to croplands ranged from -24 to -52% without marked  
188 differences between climatic regions. The conversion of pastures to other uses (tree  
189 plantations and particularly, croplands) also induced decreases in SOC (-10% and -59%,  
190 respectively). On the other hand, the substitution of croplands by other land uses (forest  
191 regrowth, tree plantation, grassland, pasture) resulted in an increase of SOC (+18 to +53%).  
192 In the case of afforestation, soil C increase with time after afforestation, and C sequestration  
193 depends on prior land use, climate and the tree species planted.

194

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

203

204 [Table 1 here]

205

## 206 ***2.2. Modelled impacts of land cover change***

207

208 Dynamic Global Vegetation Models (DGVMs) are used to look at the combined effects of  
209 land use change, climate, CO<sub>2</sub>, and in some cases N deposition, on vegetation and soil  
210 properties over time. A few global models include some aspects of forest, grassland or  
211 cropland management (Bondeau *et al.* 2007; Lindeskog *et al.* 2013; Drewniak *et al.* 2013;  
212 Jain *et al.* 2005). Most DGVMs do not currently model peatland soils. In Tables 1 and 2, and  
213 Figures 1 and 2, we show impacts of past land cover and management change on soil carbon  
214 and nitrogen as calculated by three DGVMs: ISAM (Jain *et al.* 2013; El-Masri *et al.* 2013;  
215 Barman *et al.* 2014 a,b); LPJ-GUESS (Smith *et al.* 2001; Lindeskog *et al.* 2013); and LPJmL  
216 (Bondeau *et al.* 2007). The ISAM and LPJ-GUESS models were run with the HYDE  
217 historical land use change data set (History Database of the Global Environment; Klein

218 Goldewijk *et al.* 2011). ISAM included wood harvest following (Hurtt *et al.* 2011). The  
219 LPJmL group combined 3 land use change data sets with the geographic distribution of  
220 global agricultural lands in the year 2000. All models were run with historical climate and  
221 CO<sub>2</sub>, and additionally N deposition in the ISAM model only as it includes a nitrogen cycle.  
222 The effects of land cover change were isolated by comparing model runs with and without  
223 land use/management (Le Quéré *et al.* 2014). Table 2 and Figure 1 show the loss of soil  
224 carbon due to historical land use change from 1860 to 2010 (note there was land use change  
225 causing soil carbon loss prior to 1860 particularly in Europe and central Asia, but there  
226 results are not shown as they were not available for all three models). As with the observed  
227 data (Table 1) high carbon losses are associated with the conversion of forests to croplands.  
228 Figure 2 shows the mineral soil C and N concentration of different land cover types in  
229 different geographic ranges.

230

231 [Figure 1 & 2; Table 2 here]

232

233 Differences between the models are large for some systems and regions due to different land  
234 use change data, different land cover definitions, and different processes included in the  
235 models. For example, soil carbon losses are higher in the LPJmL model (Table 2, Figure 1) in  
236 part due to greater land cover change in their land cover reconstructions, while their boreal  
237 grassland soil carbon is high due to the inclusion of permafrost slowing soil carbon  
238 decomposition (Figure 2). Treatment of management processes turns out to be an important  
239 differentiator. ISAM shows strong decreases of soil carbon in some regions e.g. the southern  
240 Boreal zone (Figure 1) where the inclusion of wood harvest removes carbon and nutrients  
241 from the soil, while increases in soil carbon in parts of the mid.-latitudes are due to regrowth  
242 of forest following abandonment of agricultural land.

243 In semi-arid to arid regions, LPJ-GUESS and LPJmL show opposite signs of soil carbon  
244 change after conversion of natural land to pastures (Figure 1), primarily because LPJ-GUESS  
245 simulates a greater fraction of woody vegetation than LPJmL in these regions under potential  
246 natural vegetation. Conversion of woody vegetation to pasture slightly increases soil carbon  
247 (see the meta analysis of Guo & Gifford 2002), partly because of boosted productivity and  
248 higher turnover rates adding more C to the soil, while the change from potential natural  
249 grassland to managed pasture (for which the literature is sparse) results in a soil carbon

250 decrease in LPJmL. Pasture management strategies can have a large influence on the soil  
251 carbon storage (see Section 4.3), and may also be partly be responsible for differences.

252 Vegetation models are embedded in Earth System Models (ESMs) used to project future  
253 climates under different human activity including different land management. Some  
254 significant differences between future model climate projections stem from the differences in  
255 modeling soil carbon, in particular, the strength of the relationship between increasing  
256 temperatures and the increasing rate of soil carbon decomposition ( $Q_{10}$ ) causing climate-  
257 carbon feedbacks *via* CO<sub>2</sub> emissions (Friedlingstein *et al.* 2006). A recent intercomparison of  
258 11 ESMs used in the IPCC 5<sup>th</sup> Assessment Report (Todd-Brown *et al.* 2013), found the  
259 estimate of global soil carbon from ESMs ranged from 510 to 3040 PgC across 11 ESMs  
260 compared to an estimate of 890-1600 PgC (95% confidence interval) from the Harmonized  
261 World Soil Data Base (FAO/IIASA/ISRIC/ISSCAS/JRC, 2012), with all models having  
262 difficulty representing the spatial variability of soil carbon at smaller (1 degree) scales  
263 compared to empirical data. In all models NPP and temperature strongly influenced soil  
264 carbon stocks, much more so than in the observational data, and differences between models  
265 was found to be largely due to the representation of NPP and the parameterization of soil  
266 decomposition sub-models. A similar, systematic analysis of DGVMs including  
267 benchmarking with observational data, and careful testing of assumptions and process  
268 representations in these models, making use of the very large number of observations that  
269 have become available in the years since these algorithms were formulated (e.g. Medlyn *et al.*  
270 2015), could significantly improve model performance. This, along with better representation  
271 of critical biological and geochemical mechanisms would improve model capability (Todd-  
272 Brown *et al.* 2013).

273

### 274 ***2.3 Drainage and conversion of peatlands/wetlands for agriculture***

275

276 The organic soils in peatlands/wetlands store vast quantities of carbon which decomposes  
277 rapidly when they are drained for agriculture or commercial forestry, resulting in emissions  
278 of CO<sub>2</sub> and N<sub>2</sub>O to the atmosphere (Hooijer *et al.*, 2010). Other services, in particular water  
279 storage and biodiversity, are negatively impacted. Drainage increases vulnerability to further  
280 losses through fire. The majority of soil carbon is concentrated in peatlands in the boreal zone  
281 and tropical peatland forests in Southeast Asia. These areas, along with wetlands along the  
282 banks of rivers, lakes and estuaries have increasingly been developed for croplands/bioenergy

283 production over recent decades. The FAO emissions database estimates that globally there  
284 are 250 000 km<sup>2</sup> of drained organic soils under cropland and grassland, with total GHG  
285 emissions (N<sub>2</sub>O plus CO<sub>2</sub>) of 0.9 Pg CO<sub>2</sub>eq yr<sup>-1</sup> in 2010, with the largest contributions from  
286 Asia (0.44 Pg CO<sub>2</sub>eq yr<sup>-1</sup>) and Europe (0.18 Pg CO<sub>2</sub>eq yr<sup>-1</sup>; FAOSTAT, 2013; Tubiello *et al.*,  
287 2015). Joosten (2010) estimated that there are >500 000 km<sup>2</sup> of drained peatlands in the  
288 world, including under forests, with CO<sub>2</sub> emissions having increased from 1.06 Pg CO<sub>2</sub> yr<sup>-1</sup>  
289 in 1990 to 1.30 Pg CO<sub>2</sub> yr<sup>-1</sup> in 2008, despite a decreasing trend in developed countries, from  
290 0.65 to 0.49 Pg CO<sub>2</sub> yr<sup>-1</sup>, primarily due to natural and artificial rewetting of peatlands. In  
291 Southeast Asia, CO<sub>2</sub> emissions from drained peatlands in 2006 were 0.61 ± 0.25 Pg CO<sub>2</sub> yr<sup>-1</sup>  
292 (Hooijer *et al.*, 2010). Conversion of peatlands in Southeast Asia is increasing, particularly  
293 for oil palm expansion, where cleared peatlands typically emit ~9 times more carbon than  
294 neighbouring mineral soils (Carlson & Curran 2013). In China, between 1950 and 2000, 13  
295 000 km<sup>2</sup> of wetland soils were shifted to cultivated arable lands, which led to a SOC loss of  
296 5.5 Pg CO<sub>2</sub>, mostly from peatlands in Northeast China and Tibet (Zhang *et al.*, 2008).

297

298 Soil drainage also affects mineral soils. Meersmans *et al.* (2009) showed that initially poorly  
299 drained valley soils in Belgium have lost significant amount of topsoil SOC (i.e. between ~70  
300 and 150 t CO<sub>2</sub> ha<sup>-1</sup> over the 1960 – 2006 period), most probably as a consequence of  
301 intensified soil drainage practices for cultivation purposes.

302

### 303 3. Agricultural management

304

305 To meet projected increases in food demand, crop production will need to increase by 70-  
306 110% by 2050 (World Bank, 2008; Royal Society of London, 2009; Tilman *et al.*, 2011).  
307 This can be achieved either through further expansion of agricultural land (extensification),  
308 or through intensification of production on existing land. Intensification is widely promoted  
309 as the more sustainable option because of the negative environmental consequences of land  
310 expansion through deforestation and wetland cultivation (Foley *et al.*, 2011). For example,  
311 Burney *et al.* (2010) estimate that intensification of production on croplands between 1961  
312 and 2010 avoided the release of 590 PgCO<sub>2</sub>eq. Increased productivity per unit land area can  
313 be achieved through a variety of management practices, such as fertilization, irrigation and  
314 increased livestock density, but these can lead to adverse consequences for the soil and wider  
315 environment (Tilman *et al.*, 2002). Intensifying land use can potentially reduce soil fertility  
316 (without additional inputs) and its ability to sustain high production, as well as soil resilience

317 to extreme weather under climate change, pests and biological invasion, environmental  
318 pollutants and other pressures. Some key management practices and consequences are  
319 highlighted below and summarised in Table 3.

320

321 [Table 3 here]

322

### 323 **3.1 Nutrient management**

324

325 Cultivation of soils results in a decline in soil nutrients (nutrient mining). Nutrient inputs,  
326 from both natural and synthetic sources, are needed to sustain soil fertility and supply nutrient  
327 requirements for crop production. Nutrient supply can improve plant growth which increases  
328 organic matter returns to the soil, which in turn can improve soil quality (see section 3.5), so  
329 balanced nutrient supply has a positive impact on soils (Smith *et al.*, 2015). Overuse,  
330 however, has negative environmental consequences. Annual global flows of nitrogen and  
331 phosphorus are now more than double natural levels (Matson *et al.*, 1997; Smil, 2000; Tilman  
332 *et al.*, 2002). In China, for example, N input in agriculture in the 2000's was twice that in  
333 1980's (State Bureau of Statistics-China, 2005).

334

335 Between 50-60% of nutrient inputs remain in agricultural soils after harvest (West *et al.*,  
336 2014) and can enter local, regional, and coastal waters becoming a major source of pollution  
337 such as eutrophication leading to algal blooms (Carpenter *et al.*, 1998). In many places  
338 around the world, over-use of synthetic nitrogen fertilizers is causing soil acidification and  
339 increased decomposition of soil organic matter, leading to loss of soil function in over-  
340 fertilized soils (Ju *et al.*, 2009; Tian *et al.*, 2012).

341

342 Use of fertilisers and manures contributes to climate change through their energy intensive  
343 production and inefficient use (Tubiello *et al.*, 2015). Globally, approximately 3-5% of  
344 nitrogen additions are released as nitrous oxide (N<sub>2</sub>O) to atmosphere when both direct (from  
345 soils) and indirect (e.g. downstream from nitrate leaching) emissions are considered  
346 (Galloway *et al.*, 2004), and N<sub>2</sub>O has ~300 times the radiative forcing of carbon dioxide  
347 (IPCC, 2007). Recent research indicates that the relationship between nitrogen application  
348 and N<sub>2</sub>O emissions is non-linear, resulting in an increasing proportion of added N being  
349 emitted, as application rate increases (Philibert *et al.*, 2013; Shcherbak *et al.*, 2014). China,

350 India, and the United States account for ~56% of all N<sub>2</sub>O emissions from croplands, with  
351 28% from China alone (West *et al.*, 2014). Overuse of nitrogen and phosphorus fertilizer can  
352 contribute to eutrophication of water bodies, adversely affecting water quality and  
353 biodiversity (Galloway *et al.*, 2003, 2004, 2008).

354

355 Nutrient use-efficiency can be significantly increased, and nitrate losses to water and N<sub>2</sub>O  
356 emissions can be reduced, through changes in rate, timing, placement, and type of  
357 application, as well as balancing fertilization (Venterea *et al.*, 2012; Snyder *et al.*, 2014). It  
358 has been estimated that current levels of global cereal production could be maintained while  
359 decreasing global nitrogen application by 50% (Mueller *et al.*, 2014).

360

### 361 ***3.2 Carbon management: reduced disturbance and organic matter additions***

362

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

378

379 Adding plant-derived carbon from external sources such as composts and biochar can  
380 increase soil carbon stocks. Composts and biochars are more slowly decomposed compared  
381 to fresh plant residues, with mean residence times several (composts) to 10-100 (biochars)  
382 longer than un-composted organic materials (Ryals *et al.*, 2015; Lehmann *et al.*, 2015).  
383 Recent developments suggest that biochar, from the pyrolysis of crop residues or other

384 biomass, can consistently increase crop N use efficiency while greatly (over 25%) reducing  
385 direct N<sub>2</sub>O emissions from N fertilizers (Liu *et al.*, 2012; Huang *et al.*, 2012), as well as  
386 enhancing soil fertility (Woolf *et al.*, 2010). Paustian *et al.* (2015) provide a recent review of  
387 soil sequestration measures.

388

### 389 **3.3 Water management**

390

391 The amount of irrigated croplands has doubled in the last 50 years and now accounts for 70%  
392 of all water use on the planet (Gleick, 2003). While irrigated crops cover 24% of all cropland  
393 area, they account for 34% of all production (Siebert & Döll, 2010). Irrigation is concentrated  
394 in precipitation-limited areas such as India, China, Pakistan, and the USA, which account for  
395 72% of irrigation water use (West *et al.*, 2014). Agricultural water-use competes with uses  
396 for human and natural ecosystems exacerbating water stress in dry regions. Increased  
397 irrigation has occurred in many areas of world agriculture due to the increasing frequency of  
398 drought under the climate change (West *et al.*, 2014). Where irrigation increases productivity  
399 (e.g. in drought prone areas), organic carbon inputs to the soils would be expected to  
400 increase, increasing soil organic matter content (section 3.2).

401

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

418 decreasing osmotic potential, which affects a wide variety of metabolic activities and alters  
419 the composition and activity of the microbial community (Chowdhury *et al.*, 2011) and  
420 thereby soil organic matter decomposition.

421

422 In saline soils, SOC content is influenced by two opposing factors: reduced plant inputs  
423 which may decrease SOC, and reduced rates of decomposition (and associated mineralisation  
424 of organic C to CO<sub>2</sub>) which could increase SOC content if the C input were unchanged.

425 Using a modified Rothamsted Carbon model (RothC) with a newly-introduced salinity  
426 decomposition rate modifier and a plant input modifier (Setia *et al.*, 2011b, 2012), Setia *et al.*  
427 (2013) estimated that, historically, world soils that are currently saline have lost an average of  
428 3.47 t SOC ha<sup>-1</sup> since they became saline. With the extent of saline soils predicted to increase  
429 under the future climate, Setia *et al.* (2013) estimated that world soils may lose 6.8 Pg SOC  
430 due to salinity by the year 2100. Soil salinization is difficult to reverse, but salt tolerant plant  
431 species could be used to rehabilitate salt affected soils (Setia *et al.*, 2013).

432

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

442

### 443 **3.4 Harvest frequency**

444

445 Approximately 9% of crop production increases from 1961-2007 was from increasing the  
446 harvest frequency (Alexandratos & Bruinsma, 2012). The global harvested area (i.e. counting  
447 each time an area is harvested) increased four times faster than total cropland area between  
448 2000 and 2011 (Ray & Foley, 2013). The fraction of net primary production (NPP) extracted  
449 by humans is increasing (Haberl *et al.*, 2007). Global warming is increasing the total area  
450 suitable for double or even triple cropping in subtropical and warm temperate regions (Liu *et*  
451 *al.*, 2013). The increase results from fewer crop failures, fewer fallow years, and an increase



452 in multi-cropping.

453

454 Increasing harvest frequency can reduce soil quality by e.g. continuously removing soil  
455 nutrients and increasing soil compaction through greater soil traffic, but if legumes are  
456 included in rotations as harvest frequency increases, soil quality could be improved.  
457 Increasing harvest frequency may require increasing pesticide and herbicide use, and  
458 increased use of fertilisers contributing to pollution (section 3.1). The net effect will depend  
459 on the effectiveness of the management practices followed.

460

### 461 ***3.5 Soil compaction***

462

463 Soil compaction causes degradation of soil structure by increasing soil bulk density or  
464 decreasing porosity through externally or internally applied loads, as air is displaced from the  
465 pores between the soil grains (McCarthy, 2007; Alakukku, 2012). It is the most important  
466 subtype of physical soil deterioration, covering 68 Mha globally when first mapped in the  
467 1990s (Oldeman et al., 1991). Compaction of agricultural soils often results from heavy  
468 machinery or from animal trampling, so is more likely to occur in intensive agricultural  
469 systems (machinery use and high stocking densities), and affects physical, chemical and  
470 biological properties of soil. Top soil compaction can be reversed and controlled, but when  
471 compaction creates impermeable layers in the subsoil, this is less easily reversed.

472

473 Subsoil compaction can disrupt nutrient water flows, which in turn can lead to reduced crop  
474 yields, poorer crop quality and can give rise to increased GHG emissions, water and nutrient  
475 run-off, erosion, reduced biodiversity and reduced groundwater recharge (Batey, 2009).

476 Where compaction cannot be avoided, mitigation is necessary. Biological approaches to  
477 mitigation include planting deep rooted plants such as agroforestry; chemical methods  
478 include fertilization (to overcome yield penalty, though not to remedy compaction); and  
479 technical measures include machinery in which planting does not coincide with wheel tracks,  
480 wide tyres / reduced tyre pressures to reduce pressure per unit area, and precision farming to  
481 retain the same wheel tracks each year (Hamza & Anderson, 2005).

482

### 483 ***3.6 Livestock density***

484

485 Livestock production is projected to increase significantly in order to meet the growing

486 demand from a growing population and increase in per-capita meat consumption, with total  
487 demand for meat expected to grow by more than 200 Mt by 2050 (Alexandratos & Bruinsma,  
488 2012). The greatest increases in per-capita consumption are projected to be in developing and  
489 transition countries (Alexandratos & Bruinsma, 2012). Since the 1970s, most increased  
490 livestock production has resulted from intensification: increasing livestock density and  
491 shifting to a greater fraction of livestock raised in industrial conditions (Bouwmann *et al.*,  
492 2006). For example, 76-79% of pork and poultry production is industrialized (Herrero &  
493 Thornton, 2013). Manure, inputs for growing feed, and soil loss from intensively managed  
494 areas can be major sources of water pollution to local and downstream freshwater  
495 ecosystems. Clearing natural ecosystems for new pastures, particularly in arid and semi-arid  
496 regions, typically occurs on low-productivity lands with a much higher risk of soil erosion  
497 and soil carbon/nutrient depletion (Alexandratos & Bruinsma, 2012), and negatively impacts  
498 water storage and biodiversity. The impacts of livestock production are particularly prevalent  
499 for beef production, which has a least an order of magnitude greater impact on land, water,  
500 GHGs, and reactive nitrogen compared to other livestock (Eshel *et al.*, 2014; Ripple *et al.*,  
501 2014). Moreover, industrial livestock production had led to an increased use of veterinary  
502 medicines, antibiotics and hormones, posing potential risks to soil, water, ecosystems and  
503 human health. Improved grazing management (e.g. optimised stocking density) can reduce  
504 soil degradation, and thereby maintain and enhance organic matter content (McSherry &  
505 Ritchie, 2013; see sections 3.2 and 4.3), and can reduce soil compaction, thereby increasing  
506 infiltration and water storage and reduce risk of runoff and flooding downstream (Marshall *et*  
507 *al.*, 2009).

508

#### 509 **4. Other land management**

510

##### 511 **4.1 Forest management**

512

513 Logging and fire are the major causes of forest degradation in the tropics (Bryan *et al.*, 2013).  
514 Logging removes nutrients and negatively affects soil physical properties and nutrient levels  
515 (soil and litter) in tropical (e.g. Olander *et al.*, 2005; Villeda *et al.*, 2006; Alexander, 2012)  
516 and temperate forests (Perez *et al.*, 2009). Forest Fires affect many physical, chemical,  
517 mineralogical, and biological soil properties, depending on fire regime (Certini, 2005).  
518 Increased frequency of fires contributes to degradation, and reduces the resilience of the  
519 biomes to natural disturbances. A meta-analysis of 57 publications (Nave *et al.*, 2011)

520 showed that fire caused a significant decrease in soil C (-26%) and N (-22%). Fires reduced  
521 forest floor storage (pool sizes only) by an average of 59% (C) and 50% (N), but the relative  
522 concentrations of these two elements did not change. Prescribed fires caused smaller  
523 reductions in C and N storage (-46% and -35%) than wildfires (-67% and -69%). These  
524 differences are likely because of lower fuel loads or less extreme weather conditions in  
525 prescribed fires, both factors that result in lower fire intensity. Burned forest floors recovered  
526 their C and N pools in an average of 128 and 103 years, respectively. Among mineral soil  
527 layers, there were no significant changes in C or N storage, but C and N concentrations  
528 declined significantly (-11% and -12%, respectively). Mineral soil C and N concentrations  
529 were significantly reduced in response to wildfires, but not after prescribed burning.

530

531 Forest fires produce charcoal, or black carbon, some of which can be preserved over centuries  
532 and millennia in soils. Dissolved black carbon (DBC) from burning of the Brazilian Atlantic  
533 forest continued to be mobilized from the watershed each year in the rainy season, despite the  
534 fact that widespread forest burning ceased in 1973 (Dittmar *et al.*, 2012).

535

536 A large field study in the Amazon (225 forest plots) on the effects of anthropogenic forest  
537 disturbance (selective logging, fire, and fragmentation) on soil carbon pools showed that the  
538 first 30 cm of the soil pool did not differ between disturbed primary forests and undisturbed  
539 areas of forest, suggesting a resistance to impacts from selective logging and understory fires  
540 (Berenguer *et al.*, 2014). As with deforestation, impacts of human disturbances on the soil  
541 carbon are of particular concern in tropical forests located on organic soils and on steep  
542 easily-eroded slopes.

543

#### 544 **4.2 Shifting cultivation**

545

546 Shifting cultivation practices, where land is cleared through fire, have been practiced for  
547 thousands of years, but recent increasing demographic pressure has reduced the duration of  
548 the fallow period, affecting the system sustainability. Moreover, especially in Southeast Asia  
549 where urbanisation is expanding in fertile plains, shifting cultivation is practiced in sloping  
550 uplands, which are prone to soil and carbon loss by erosion (Chaplot *et al.*, 2005). A review  
551 by Ribeiro Filho *et al.* (2013) reported negative impact on SOC associated with the  
552 conversion stage, modified by the characteristics of the burning. Chop-and-mulch of enriched  
553 fallows appears to be a promising alternative to slash-and-burn, conserving soil bulk density,

554 and significantly increasing nutrient concentrations and organic matter content compared to  
555 burnt cropland, and a control forest in a study in the Amazon (Comtea *et al.*, 2012).

556

#### 557 **4.3 Grassland management and dryland degradation**

558

559 Grasslands, including rangelands, shrublands, pastureland, and cropland sown with pasture  
560 and fodder crops, cover 26% of the global ice-free land area and 70% of the agricultural area,  
561 and contain about 20% of the world's soil organic carbon (C) stocks. Grasslands on every  
562 continent have been degraded due to human activities, with about 7.5% of grassland having  
563 been degraded because of overgrazing (Conant, 2012). A meta-analysis (McSherry & Ritchie,  
564 2013) of grazer effects on SOC density (17 studies that include grazed and un-grazed plots)  
565 found higher grazing intensity was associated with increased SOC in grasslands dominated  
566 by C4 grasses (increase of SOC by 6–7%), but with lower SOC in grasslands dominated by  
567 C3 grasses (decrease of SOC by an average 18%). An increase in mean annual precipitation  
568 of 600 mm resulted in a 24% decrease in the magnitude of the grazer effect on finer textured  
569 soils, but on sandy soils the same increase in precipitation produced a 22% increase in the  
570 grazer effect on SOC (McSherry & Ritchie, 2013).

571

572 Land use dynamics and climate change are the major drivers of dryland degradation with  
573 important feedbacks, with changes in plant community composition (e.g. shrub encroachment  
574 and decrease in vegetation cover; D'Odorico *et al.*, 2013). A review by Ravi *et al.* (2010)  
575 indicated soil erosion as the most widespread form of land degradation in drylands, with wind  
576 and water erosion contributing to 87% of the degraded land. Grazing pressure, loss of  
577 vegetation cover, and the lack of adequate soil conservation practices increase the  
578 susceptibility of these soils to erosion. The degree of plant cover is negatively related to  
579 aridity, and an analysis of 224 dryland sites (Delgado-Baquerizo *et al.*, 2013) highlighted a  
580 negative effect of aridity on the concentration of soil organic C and total N, but a positive  
581 effect on the concentration of inorganic P, possibly indicating the dominance of physical  
582 processes such as rock weathering, a major source of P to ecosystems, over biological  
583 processes that provide more C and N through litter decomposition (Delgado-Baquerizo *et al.*,  
584 2013).

585

586 Soil carbon dynamics in pastures strongly depend on management, with soil carbon increases  
587 or decreases observed for different combinations of animal densities and grazing frequency

588 (Conant 2012; Machmuller *et al.* 2015). Different grazing strategies, especially in the semi-  
589 natural dryland biomes, have large implications for vegetation and the carbon balance (Yates  
590 *et al.* 2000). Under certain conditions, grazing can lead to increased annual net primary  
591 production over un-grazed areas, particularly with moderate grazing in areas with a long  
592 evolutionary history of grazing and low primary production but this does not always lead to  
593 an increase in soil carbon (e.g. Badini *et al.* 2007); grazing, like crop harvest, fundamentally  
594 leads to the rapid oxidation of carbon that would otherwise be eventually transferred to the  
595 soil. It has long been recognised that the potential effects of management on carbon storage in  
596 grassland and dryland soils are substantially greater than that of climate change or CO<sub>2</sub>  
597 enhancement (Ojima *et al.* 1993), and Henderson *et al.* (2015) estimated that the optimization  
598 of grazing pressure could sequester 148 Tg CO<sub>2</sub> yr<sup>-1</sup>.

599

#### 600 **4.4 Artificial surfaces, urbanisation and soil sealing**

601

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

613

614 Sealing by its nature has a major effect on soil, diminishing many of its benefits (Tóth *et al.*,  
615 2007). It is normal practice to remove the upper layer of topsoil, which delivers most of the  
616 soil-related ecosystem services, and to develop a strong foundation in the subsoil and/or  
617 underlying rock to support the building or infrastructure. Loss of ecosystem and social  
618 services (mainly on high-quality soils) include impacts on water resources (e.g. reduction of  
619 rainfall absorbed by the soil, reduction of soil water holding capacity affecting flooding), on  
620 soil biodiversity when sealing prevents recycling of dead organic material (Marfenina *et al.*  
621 2008), on the carbon cycle due to topsoil and vegetation removal (Davies *et al.*, 2011).

622

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

638

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

647

#### 648 **5. Anthropogenic environmental change pressures that interact with land** 649 **management pressures on soils**

650

651 In addition to the direct impacts of humans on soils *via* land use change and land  
652 management, anthropogenic activity has indirect impacts through human-induced  
653 environmental change, such as pollution and climate change. These interact with land  
654 management. Soils provide a temporary but labile store for pollutants (Meuser, 2010).  
655 Natural processes can release pollutants back to the atmosphere, make them available to be

656 taken up by plants and organisms, leached in to surface waters (Galloway *et al.*, 2008) and/or  
657 transported to other areas by soil erosion (Ravi *et al.*, 2010). Pollutants disrupt natural  
658 biogeochemical cycles by altering both soil quality and function through direct changes to the  
659 nutrient status, acidity and bioavailability of toxic substances and also by indirect changes to  
660 soil biodiversity, plant uptake and litter inputs (EEA, 2014). Soil sensitivity to atmospheric  
661 pollution varies with respect to key properties influenced by geology (cation exchange  
662 capacity, soil base saturation, aluminium), organic matter, carbon to nitrogen ratio (C:N) and  
663 water table elevation (EEA, 2014).

664

665

666 Atmospheric pollutant deposition impacts on soils vary with respect to soil sensitivity to a  
667 specific pollutant and the actual pollutant load. Sulphur, nitrogen and heavy metals are  
668 released in to the atmosphere by fossil fuel combustion (e.g. power generation, industry and  
669 transport) and non-combustion processes (e.g. agricultural fertilizers, waste). These pollutants  
670 are transported off-site and deposited as either dry or wet deposition, which can cross  
671 national borders. Deposition is enhanced in forests and with altitude because of reduced wind  
672 speeds and greater precipitation, respectively, impacting remote areas. Harmful effects to soil  
673 function and structure occur where deposition exceeds the ‘critical load’ that a particular soil  
674 can buffer (Nilsson & Grennfelt, 1988). Spatial differences in soil sensitivity (commonly  
675 defined by the ‘critical load’) and pollutant deposition result in an uneven global distribution  
676 of impacted soils (Figure 3). For instance, global emissions of sulphur and nitrogen have  
677 increased 3-10 fold since the pre-industrial period (van Aardenne *et al.*, 2001), yet only 7-  
678 17% of the global land area sensitive to acidification is in a region where deposition exceeds  
679 the critical load (Bouwman *et al.*, 2002).

680

681 Emissions of pollutants, notably sulphur, across Europe and North America have declined  
682 since the 1980s following protocols established under the 1979 Convention on Long-range  
683 Transboundary Air Pollution (LRTAP) and the 1990 US Clean Air Act Amendments  
684 (CAAA) (Greaver *et al.*, 2012; Reis *et al.*, 2012; EEA, 2014). Conversely, emissions are  
685 likely to increase in response to industrial and agricultural development in south and east  
686 Asia, sub-Saharan Africa and South America (Kuylenstierna *et al.*, 2001; Dentener *et al.*,  
687 2006). Further emission increases are occurring in remote areas due to mining activity, such  
688 as oil sand extraction in Canada (Kelly *et al.*, 2010; Whitfield *et al.*, 2010).

689

### 690 **5.1 Sulphur deposition**

691

692 Sulphur emissions are primarily from combustion of coal and oil, typically associated with  
693 power generation and heavy industry. In 2001, regions with deposition in excess of 20 kg S  
694 ha<sup>-1</sup> yr<sup>-1</sup> where China and Republic of Korea, western Europe and eastern North America  
695 (Vet *et al.*, 2014; Figure 3a). Deposition in un-impacted areas is <1 kg S ha<sup>-1</sup> yr<sup>-1</sup> (Figure 3a).  
696 Pollution control measures have seen an 80% reduction in pollutant sulphur deposition across  
697 Europe between 1990 and 2010 (Reis *et al.*, 2012), and emissions in China have declined  
698 since 2005 (Fang *et al.*, 2013).

699

700 Soil acidification is a natural process that is altered and accelerated by sulphur and nitrogen  
701 deposition (Greaver *et al.*, 2012). Sulphur oxides (SO<sub>x</sub>) react with water to form sulphuric  
702 acid (H<sub>2</sub>SO<sub>4</sub>). Excess inputs of acidity (H<sup>+</sup>) displace soil base cations (e.g. calcium (Ca<sup>2+</sup>) and  
703 magnesium (Mg<sup>2+</sup>)) from soil surfaces into solution, which are subsequently lost by leaching  
704 (Reuss & Johnson, 1986). Mineral soils can buffer base cation losses if inputs from rock  
705 weathering and/or atmospheric dust deposition exceed the amount lost. Therefore, the global  
706 distribution of acid sensitive soils is associated with conditions that favour development of  
707 soils with low cation exchange capacity and base saturation (Bouwman *et al.*, 2002; Figure  
708 3c). Wetland can also buffer inputs of acidity through biological sulphate reduction, although  
709 acidity can be mobilised again following drought and drainage (Tipping *et al.*, 2003; Laudon  
710 *et al.*, 2004; Daniels *et al.*, 2008). Organic acids can also buffer mineral acidity in naturally  
711 acidic organic soils (Krug and Frink, 1983).

712

713 Decreased soil fertility or 'sterilisation' due to loss of nutrients and mobilisation of toxic  
714 metals, particularly Al, are caused by acidification. Impacts in Scandinavia over the 1960s-  
715 80s included declines in freshwater fish populations and damage to forests (EEA, 2014).  
716 Sulphur can also stimulate microbial processes that make mercury bioavailable, leading to  
717 bioaccumulation in the food chain (Greaver *et al.*, 2012). In agricultural soils in Europe,  
718 however, fertilizer inputs of sulphur have increased to combat crop sulphur deficiencies as a  
719 result of sulphur emission controls (Bender & Weigel, 2011).

720

721 Acidification is reversible, evident by increases in soil pH following decreased sulphur  
722 emissions, although the recovery time varies; some areas with organic soils where deposition  
723 has declined are showing either slow or no recovery (Greaver *et al.*, 2012; Lawrence *et al.*,



724 2012; RoTAP, 2012). On agricultural soils, lime can be applied to increase soil pH.  
725 However, 50-80% of sulphur deposition on land is on natural, non-agricultural land  
726 (Dentener *et al.*, 2006). Application of lime to naturally acidic forest soils can cause further  
727 acidification of deep soil layers whilst increasing decomposition in surface litter, with no  
728 improvement in tree growth (Lundström *et al.*, 2003).

729

730 Wider effects of acidification are starting to be understood through long-term monitoring.  
731 Decreased organic matter decomposition due to acidification has increased soil carbon  
732 storage in tropical forests (Lu *et al.*, 2014). However, in temperate forest soils acidification  
733 can lead to reduced C:N ratios in soil, which in turn increases nitrification (Aber *et al.*, 2003),  
734 but on already acidic soils reduces nitrification. In wetland soils, methane (CH<sub>4</sub>) emissions  
735 have also been suppressed by sulphur deposition (Gauci *et al.*, 2004). Conversely, declining  
736 sulphur deposition has been associated with increased dissolved organic carbon fluxes from  
737 organic soils (Monteith *et al.*, 2007), and decreased soil carbon stocks in temperate forest  
738 soils (Oulehle *et al.*, 2011; Lawrence *et al.*, 2012).

739

## 740 **5.2 Nitrogen deposition**

741

742 Nitrogen deposition covers a wider geographical area than sulphur, as the sources are more  
743 varied, including extensive agriculture fertilizer application, ammonia derived from livestock  
744 operations, and biomass burning in addition to fossil fuel combustion (Figure 3b). Regions  
745 with deposition in excess of 20 kg N ha<sup>-1</sup> yr<sup>-1</sup> in 2001 were western Europe, South Asia  
746 (Pakistan, India, Bangladesh) and eastern China (Vet *et al.*, 2014); although extensive areas  
747 with 4 kg N ha<sup>-1</sup> yr<sup>-1</sup> were found across North, Central and South America, Europe and Sub-  
748 Saharan Africa. By contrast, 'natural' deposition in un-impacted areas is around 0.5 kg N ha<sup>-1</sup>  
749 yr<sup>-1</sup> (Dentener *et al.*, 2006). While emissions related to fossil fuel combustion have declined  
750 along with sulphur across Europe, agricultural sources of nitrogen are likely to stay constant  
751 in the near future across Europe (EEA, 2014), whilst overall global emissions are likely to  
752 increase (Galloway *et al.*, 2008). Nitrogen deposition in China's industrialized and  
753 intensively managed agricultural areas in the 2000s was similar to peaks in Western Europe  
754 during the 1980s before mitigation (Liu *et al.*, 2013).

755

756 Deposition of nitrogen induces a 'cascade' of environmental problems, including both  
757 acidification and eutrophication that can have both positive and negative effects on ecosystem

758 services (Galloway *et al.*, 2003). Excluding agricultural areas where nitrogen is beneficial,  
759 11% of land surface received nitrogen deposition above 10 kg N ha<sup>-1</sup> yr<sup>-1</sup> (Dentener *et al.*,  
760 2006; Bouwman *et al.* 2002; Figure 3d). In Europe, eutrophication has and will continue to  
761 impact a larger area than acidification (EEA, 2014).

762

763 Nitrogen fertilisation can increase tree growth (Magnani *et al.*, 2007) and cause changes in  
764 plant species and diversity (Bobbink *et al.*, 2010), which in turn will alter the amount and  
765 quality of litter inputs in to soils, notably the C:N ratio and soil-root interactions (RoTAP,  
766 2012). However, increased carbon sequestration (Reay *et al.*, 2008) may be offset by  
767 increased emissions of the greenhouse gases N<sub>2</sub>O and CH<sub>4</sub> (Liu & Greaver, 2009). Long-term  
768 changes caused by nitrogen deposition are uncertain as transport times vary between  
769 environmental systems; and the only way to remove excess nitrogen is to convert it to an  
770 unreactive gas (Galloway *et al.*, 2008).

771

772 [Figure 3 here]

773

### 774 **5.3 Heavy metal deposition**

775

776 Heavy metal emissions are associated with coal combustion and heavy industry. In the UK,  
777 deposition is responsible for 25-85% of inputs to UK soils (Nicholson *et al.*, 2003). In  
778 Europe, the areas at risk from cadmium, mercury and lead deposition in 2000 were 0.34%,  
779 77% and 42% respectively, although emissions are declining (Hettelingh *et al.*, 2006).  
780 Tighter legislation to control industrial emissions of heavy metals are helping to reduce the  
781 environmental load of heavy metals in many regions, though rapid industrial growth in some  
782 regions such as East Asia is increasing pressures on soil from heavy metal deposition. Global  
783 heavy metal emissions and deposition are poorly understood in comparison to sulphur and  
784 nitrogen; although the on-site impact of heavy metal contamination has been well studied  
785 (Guo *et al.*, 2014). Metals in bioavailable form have toxic effects on soil organisms and  
786 plants, influencing the quality and quantity of plant inputs to soils, rate of decomposition and,  
787 importantly, can bio-accumulate in the food chain. Some heavy metals will persist for  
788 centuries as they are strongly bound to soil organic matter (RoTAP, 2012), although they can  
789 be mobilised to bioavailable form following drought-induced acidification, drainage and soil  
790 erosion (Tipping *et al.*, 2003; Rothwell *et al.*, 2005).

791

792 Whilst the direct impacts of sulphur, nitrogen and heavy metals on inorganic soil chemical  
793 processes are generally well understood, many uncertainties still exist about pollutant impacts  
794 on biogeochemical cycling, particularly interactions between organic matter, plants and  
795 organisms in natural and semi-natural systems (Greaver *et al.*, 2012). Process understanding  
796 is dominated by research in Europe and North America (e.g. Bobbink *et al.*, 2010). Research  
797 is needed across Asia, Africa and South and Central America where soil properties and  
798 environmental conditions differ. Models need to be developed to examine the combined  
799 effects of air pollutants and their interactions with climate change impacts and feedbacks on  
800 greenhouse gas balances and carbon storage (Spranger *et al.*, 2008; RoTAP, 2012). Air  
801 quality, biodiversity and climate change policies all impact on soils. A more holistic approach  
802 to protecting the environment is needed, particularly as some climate change policies (e.g.  
803 biomass burning, carbon capture and storage) have potential to impact air quality and,  
804 therefore, soil quality (Reis *et al.*, 2012; RoTAP, 2012; Aherne & Posch, 2013).

805

806 Indirect impacts on soils can be addressed largely by preventing the pollution at source, or by  
807 mitigating the adverse effects where these have already occurred. Air pollution control on  
808 coal burning and increased car and fleet efficiency standards has been effective in reducing  
809 sulphur deposition in many areas of the world, particularly in Europe since the 1970s.  
810 Substitution of coal with bioenergy might also reduce sulphur emissions, but unless burned  
811 cleanly in a controlled way, can also release pollutants to the air. In terms of nitrogen,  
812 ammonia abatement techniques when fertilizers are spread (e.g. slurry injection) are helping  
813 to reduce N deposition (Sutton *et al.*, 2007).

814

## 815 **6. Existing policies and practices that alleviate global change pressures on soils** 816 **from land use and management**

817

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

821

822 1) Prevent conversion of natural ecosystems to other uses (e.g. protected areas, reduced  
823 deforestation, prevention of wetland drainage, intensification rather than extensification);

824 2) Prevent soil degradation (erosion control, fire management, reduced tillage / conservation  
825 agriculture, long term fallows, flood protection, use of organic amendments, intercropping,  
826 improved rotations); and  
827 3) Result in soil / ecosystem restoration (e.g. peatland rewetting, afforestation, re-vegetation  
828 on degraded lands, improved grass varieties, appropriate animal stocking densities,  
829 bioremediation).

830

831 Policies to encourage such actions were recently reviewed by Bustamante *et al.* (2014) and  
832 include:

833

- 834 a) Economic incentives, e.g., special credit lines for low carbon agriculture and forestry  
835 practices and projects, payment for ecosystem services (such as carbon storage) and tradable  
836 credits such as carbon,
- 837 b) Regulatory approaches, e.g. enforcement of environmental law to protect natural areas, set-  
838 aside policies,
- 839 c) Research, development and diffusion investments, e.g. increase of resource use-efficiency,  
840 livestock improvement,
- 841 d) Information and certification schemes, e.g. in China, forest certification to promote  
842 sustainable forest management, state regulation for protecting mandatory arable lands,  
843 protection projects on Tibetan grasslands, a national wetland protection programme, and the  
844 “grain for green” programme.

845

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

857

858 **6.1 United Nations Framework Convention on Climate Change (UNFCCC) and other**  
859 **climate specific policies**

860

861 Soil carbon storage and nutrient cycling as climate services are being increasingly recognised  
862 e.g. under UNFCCC as part of national reporting and accounting, as part of life-cycle  
863 greenhouse gas assessments for biofuels, in various regional initiatives and national efforts.

864 The UNFCCC is an international treaty, which came into force in 1994, setting an overall  
865 framework for intergovernmental efforts to tackle the challenge posed by climate change.

866 The requirements for the 196 country Signatories (or 'Parties') to the UNFCCC include  
867 adopting national mitigation policies and publishing national inventories of anthropogenic  
868 emissions and sinks of greenhouse gases including activities on the land such as afforestation,  
869 deforestation, agricultural management and wetland drainage and rewetting. Developed  
870 country signatories have legally binding targets under the Kyoto Protocol and can count land  
871 based emissions or sinks towards meeting these targets, thus incentivising activities that  
872 protect soil carbon. Developing countries currently have voluntary targets and several  
873 countries have made pledges that include reduced deforestation (e.g. Brazil and Indonesia) or  
874 afforestation (e.g. 400000 km<sup>2</sup> in China). Under the Clean Development Mechanism (CDM)  
875 developed countries can fund projects in developing countries that generate certified emission  
876 reduction credits (CERCs). China, for example, has the largest number of CERCs in the  
877 world (IFPRI, 2011). Brazil also has 180 CDM projects, the third largest number of CERCs  
878 after China and India (Cole & Liverman, 2011). Paustian *et al.* (2015) list several projects in  
879 Africa, North America and South Asia that have a significant component for soil greenhouse  
880 gas emission reduction of soil carbon sequestration, financed through the Verified Carbon  
881 Standard or the American Carbon Registry.

882

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

892

893 **6.2 United Nations Convention to Combat Desertification (CCD)**

894

895 The CCD entered into force in December 1996; today 179 countries acknowledge it as a  
896 legally binding framework to tackle land degradation and promote sustainable development  
897 in fragile ecosystems. The Global Mechanism was established under the convention to  
898 "promote actions leading to the mobilization and channelling of substantial financial  
899 resources, including for the transfer of technology, on a grant basis, and/or on concessional or  
900 other terms, to affected developing country Parties". In September 2011 the United Nations  
901 General Assembly declared a goal of building a world with no land degradation. In October  
902 2011 parties to the CCD issued a declaration calling for zero land degradation and for  
903 adopting sustainable land management as a way to achieve sustainable development.

904

905 **6.3 Millennium Development Goals (MDGs)**

906

907 Of the eight MDGs (UNDP, 2014a), soil protection is most relevant to the goal to ensure  
908 environmental sustainability, since soils are critical in underpinning environmental  
909 sustainability (Smith *et al.*, 2015). A complementary MDG, to develop a global partnership  
910 for development, will improve the governance structure to deliver soil security. The other  
911 MDG to which soils plays a critical contribution is the goal to eradicate extreme poverty and  
912 hunger, with the role of soils in supporting food provision critical for the latter part of this  
913 MDG (Smith *et al.*, 2015). The MDGs are currently being revisited to set a post-2015  
914 development agenda (UNDP, 2014b), with discussion around the themes of localising the  
915 post-2015 development agenda, helping to strengthen capacities and build effective  
916 institutions, participatory monitoring for accountability, partnerships with civil society,  
917 engaging with the private sector, and culture and development. The key emerging principles  
918 from the dialogue are participation, inclusion, and the need for strengthened capacities and  
919 partnerships (UNDP, 2014b). It is important that soils play their role in delivering this post-  
920 2015 agenda.

921

922 **6.4 Protected areas and the Convention on Biological Diversity (CBD)**

923

924 Many measures to protect biodiversity and vulnerable habitats also protect the soils  
925 underpinning them, so numerous conservation actions around the world serve to protect soils,

926 even if this was not the primary aim (Smith *et al.*, 2013). Between 1990 and 2010, the  
927 amount of forest land designated primarily for the conservation of biological diversity  
928 increased by 35 percent, indicating a political commitment to conserve forests. These forests  
929 now account for 12 percent of the world's forests (FAO, 2010). In India, a Supreme Court  
930 ruling in 2011 on effective self-governance of "common" or communal land by local  
931 communities may help to protect these valuable resources, and thereby the soils that underpin  
932 them. Soil biodiversity is known to be important for soil function (Bodelier, 2011), yet it  
933 rarely receives the attention enjoyed by larger flora and fauna within the ecosystem.

934

### 935 ***6.5 Reduced deforestation and forest management***

936

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

951

### 952 ***6.6 Agricultural policies and practices***

953

954 The pressures on soils imposed by land use intensity change can be mitigated by regulation of  
955 over-grazing and reduction of over-stocking on grazed grasslands, return of crop residues to  
956 the soil, reduced tillage, best management practices, targeted nutrient management and  
957 precision farming on croplands, and wetland / floodplain restoration. These actions have been  
958 encouraged by various policies. Some examples include: The EU set-aside programme of the  
959 1990s encouraged less intensive use of agricultural land where production is low and

960 environmental impacts are high. The EU Common Agricultural Policy ties agricultural  
961 subsidies to implementation of best management practices and environmental protection, for  
962 example through pillar 2 (rural development programmes) providing crop insurance for lower  
963 fertilizer application rates; in Africa, policies for integrated land management to help protect  
964 vulnerable soils; China's conservation tillage program (2012-2030); the USA Conservation  
965 Reserve Program (set aside marginal lands, steep slopes).

966

## 967 **7. Conclusion: Keeping soils central to the science and policy agendas**

968

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

985

986 Research and policy regarding soil quality and sustainability is abundant, but patchy and  
987 disjointed. To ensure that soils are protected as part of on-going wider environmental and  
988 sustainable production efforts, soils cannot, and should not, be considered in isolation of the  
989 ecosystems that they underpin, but the role of soils in supporting ecosystems and natural  
990 capital needs greater recognition (Robinson *et al.*, 2013, 2014). This can, in part, be enhanced  
991 through education and awareness-raising which has started during the International Year of  
992 the Soils in 2015. The time is ripe to consider a global soil resilience programme, under the  
993 auspices of a global body such as the UN or one of its delivery agencies such as the FAO to



994 monitor, recover or sustain soil fertility and function, and to enhance the ecosystem services  
995 provided by soils. The lasting legacy of the International Year of Soils in 2015 should be to  
996 bring together robust scientific knowledge on the role of soils, and to put soils at the centre of  
997 policy supporting environmental protection and sustainable development.

998

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1000

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1007

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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.

		Regrowth Forest	Tree plantation	Grassland	Pasture	Cropland
Forest	Global Trop.	-9% (2)	-13% (3) <sup>a</sup>		+8% (3) -12% (2)	-42% (3) -41% (1) -25% (2) <sup>b</sup> -30% (2) <sup>c</sup> -24% (5)
	Temp.				[-40 to -63%]	[-51 to -62%] -52% (1) -36% (4)
	Boreal				[-52% to +17] [-14 to -49%]	[-24 to -60%] -31% (1) [-63 to -65%]
Grassland	Global Trop				[-1 to +15%]	[-2 to -6%] -32% (4)
	Temp Boreal				[-28 to +3%] [-26 to -71%]	[-15 to -53%] [-70 to -79%]
Pasture	Global Trop Temp Boreal		-10% (3)			-59% (3) [-19 to +0.5%] [-17 to -35%] [-28 to -59%]
Cropland	Global Trop	+53% (3)	+18% (3)		+19% (3)	
	Temp Boreal	+16% (4)	+29% (2) +20% (6)	+28% (4)	+26% (2)	

**Footnotes:** <sup>a</sup> 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%; <sup>b</sup> Annual crops; <sup>c</sup> Perennial crops; 1 Wei *et al.* (2014a); 2 Don *et al.* (2011); 3 Guo & Gifford (2002; tropical and temperate zones compiled); 4 Poepflau *et al.* (2011); 5 Murty *et al.* (2014); 6 Barcena *et al.* (2014).

**Table 2.** Soil carbon loss due to land use change 1860 to 2010 (PgCO<sub>2</sub>)

Model	Tropical	Temperate	Boreal	Global
LPJ-GUESS	46	55	1	109
LPJmL	128	95	0	227
ISAM	63	139	19	221
Mean	79	96	7	186

For Review Only

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

<b>Agricultural management practice</b>	<b>Specific issue</b>	<b>Distribution</b>	<b>Major environmental consequence</b>	<b>Knowledge gap</b>
Cropping practice	Harvest frequency	Global	Soil quality and resilience	Impact on total C and nutrient cycles
	Monoculture	Global but particularly in developing and transition countries	Soil health, pesticide residue in intensively managed monocultures	Biological resilience
Use of agrochemicals	Over fertilization	Particularly in some developing countries	Soil acidification, water pollution, N <sub>2</sub> O emission and nitrate accumulation	Rate reducing versus balancing
Irrigation	Submerged Rice	Developing countries, Asian	Water scarcity, methane emission	Trade-offs C and water,
	Arid/semi-arid regions	Arid/semi-arid regions	Secondary salinization, water scarcity	Competition use of water
Livestock management	Over-grazing	Largely in developing countries	Soil degradation, water storage, C loss	Forage versus feed crops?
	Industrial breeding	Largely in industrialized and transition countries	Waste pressure, water pollution, residue of veterinary medicine and antibiotics	Safe waste treatment and recycling
Agriculture in wetlands	Wetland drainage	Developing and transition countries	C loss	Agro-benefit versus natural value



1 **Figure Legends**

2

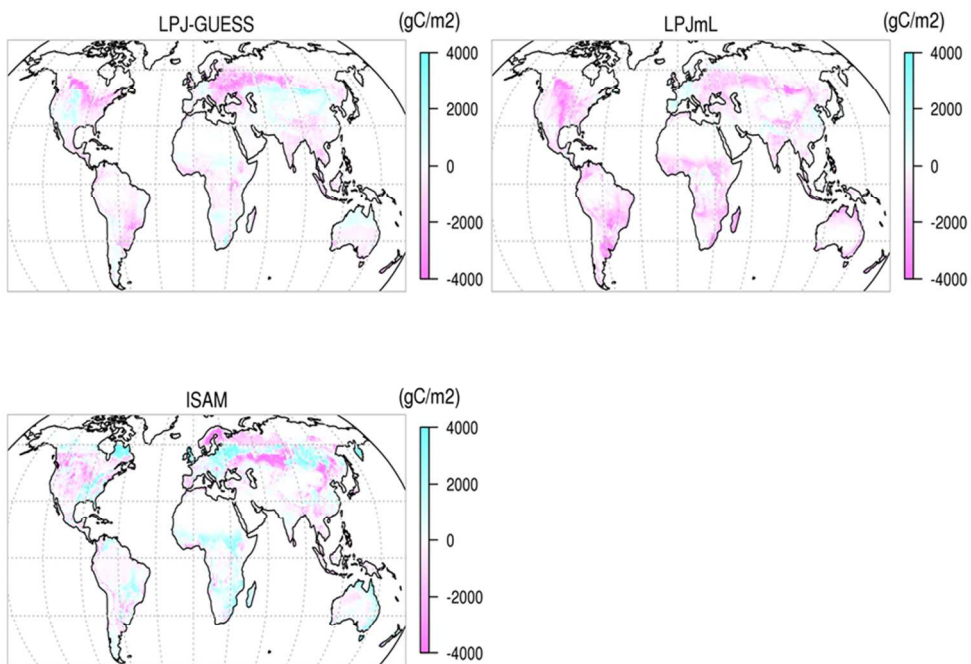
3 **Figure 1.** Maps of change in soil carbon due to land use change and land management  
4 from 1860 to 2010 from three vegetation models. Pink indicates loss of soil carbon, blue  
5 indicates carbon gain.

6

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

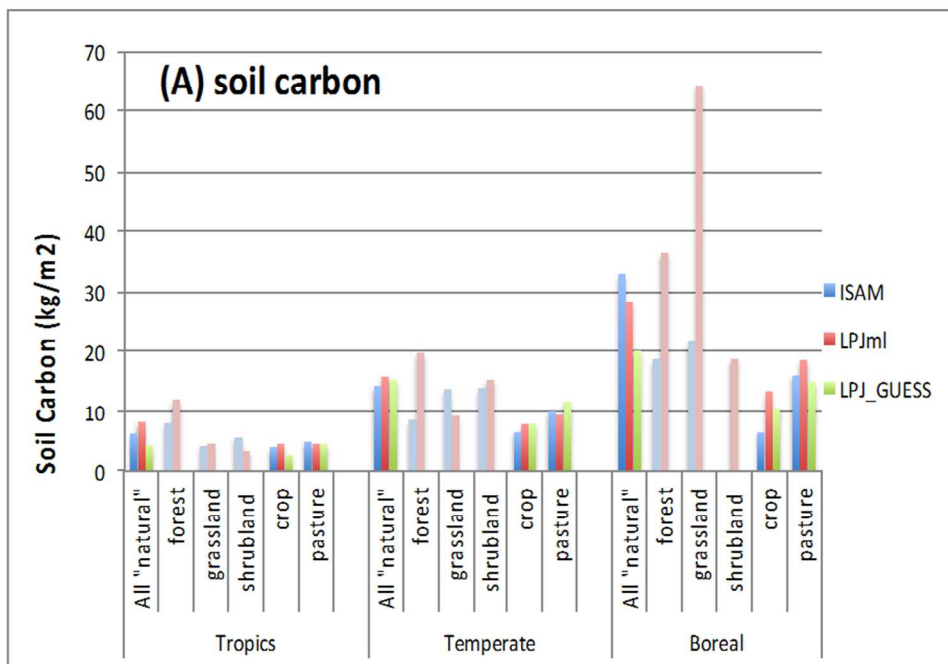
9

10 **Figure 3.** Uneven global distribution of soils sensitive to pollution by (a) acidification and (b)  
11 eutrophication (measured by soil C:N) compared to uneven distribution of atmospheric (c)  
12 sulphur and (d) nitrogen pollution. Soils most sensitive to acidification have low base  
13 saturation and cation exchange capacity, as defined by (Kuylenstierna *et al.*, 2001).  
14 Acidification is caused by both sulphur and nitrogen. Eutrophication is caused by nitrogen.  
15 Soil data in (a) and (b) were produced using the ISRIC-WISE derived soil properties (ver 1.2)  
16 (Batjes, 2012) and the FAO Digital Soil Map of the World. Atmospheric deposition data in  
17 (c) and (d) were provided by the World Data Centre for Precipitation Chemistry  
18 (<http://wdcpc.org>, 2014) and are also available in Vet *et al.* (2014). Data show the ensemble-  
19 mean values from the 21 global chemical transport models used by the Task Force on  
20 Hemispheric Transport of Air Pollution (HTAP) (Dentener *et al.*, 2006). Total wet and dry  
21 deposition values are presented for sulphur, oxidized and reduced nitrogen.



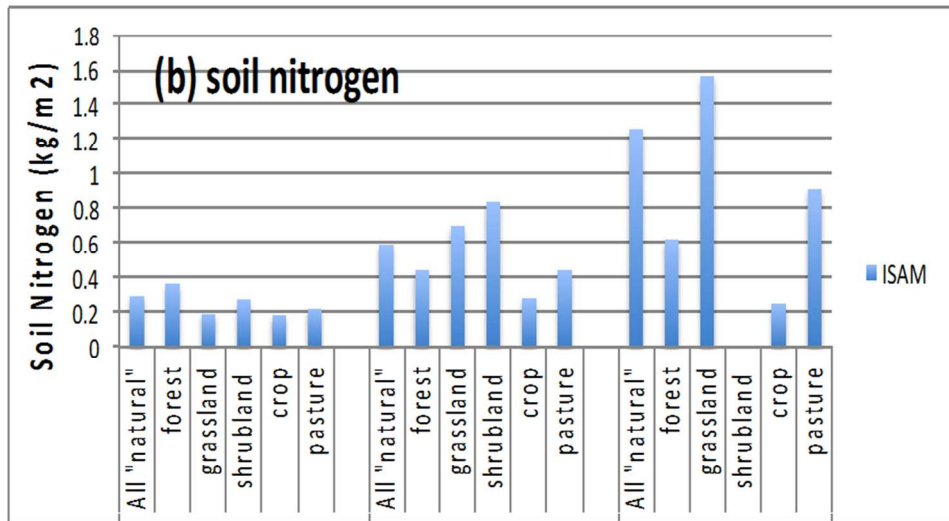
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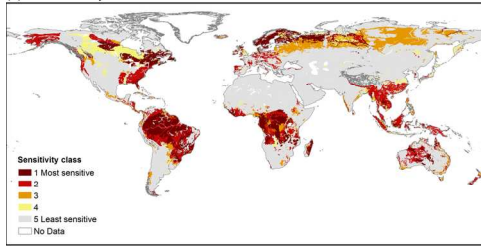
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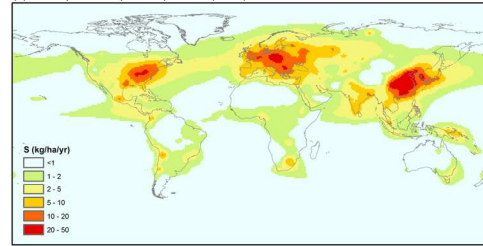
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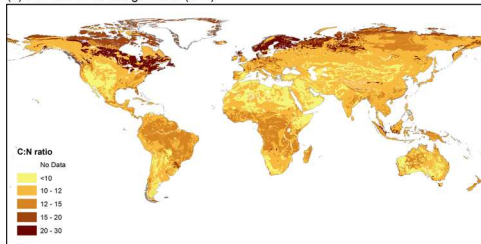
(a) Soil sensitivity to acidification



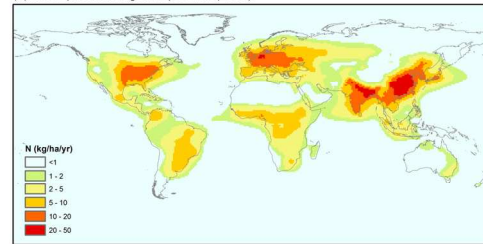
(c) Atmospheric sulphur deposition (2001)



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



(d) Atmospheric nitrogen deposition (2001)



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**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 is 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, CO<sub>2</sub> and CO<sub>2</sub>eq). 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 km<sup>2</sup>, whereas above in the text the M-prefix is used. I would suggest 0.5 Mkm<sup>2</sup> here.

Response: All units have been harmonised throughout the manuscript. All Gt have been converted to Pg and all values are expressed in CO<sub>2</sub>-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 "brounfield 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 CO<sub>2</sub> 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 Kashmir. 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 that 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

synthesise 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 of 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 where 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.

For Review Only