1   **Global Change Pressures on Soils from Land Use and Management**

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59　**Abstract**
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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^2, 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^2). 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_2, 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_2, 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 results are

226   not shown as they were not available for all three models). As with the observed data (Table

227   1) high carbon losses are associated with the conversion of forests to croplands. Figure 2

228   shows the mineral soil C and N concentration of different land cover types in different

229   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_2 emissions (Friedlingstein *et al.* 2006). A recent intercomparison of

258   11 ESMs used in the IPCC 5th 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_2 and N_2O 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^2 of drained organic soils under cropland and grassland, with total GHG

285   emissions (N_2O plus CO_2) of 0.9 Pg CO_2eq yr^{-1} in 2010, with the largest contributions from

286   Asia (0.44 Pg CO_2eq yr^{-1}) and Europe (0.18 Pg CO_2eq yr^{-1}; FAOSTAT, 2013; Tubiello *et al.*,

287   2015). Joosten (2010) estimated that there are >500 000 km^2 of drained peatlands in the

288   world, including under forests, with CO_2 emissions having increased from 1.06 Pg CO_2 yr^{-1}

289   in 1990 to 1.30 Pg CO_2 yr^{-1} in 2008, despite a decreasing trend in developed countries, from

290   0.65 to 0.49 Pg CO_2 yr^{-1}, primarily due to natural and artificial rewetting of peatlands. In

291   Southeast Asia, CO_2 emissions from drained peatlands in 2006 were 0.61 ± 0.25 Pg CO_2 yr^{-1}

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^2 of wetland soils were shifted to cultivated arable lands, which led to a SOC loss of

296   5.5 Pg CO_2, 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_2 ha^{-1} 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_2eq. 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_2O) to the atmosphere when both direct

345   (from soils) and indirect (e.g. downstream from nitrate leaching) emissions are considered

346   (Galloway *et al*., 2004), and N_2O has ~300 times the radiative forcing of carbon dioxide

347   (IPCC, 2007). Recent research indicates that the relationship between nitrogen application

348   and N_2O 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_2O 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_2O

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). Soil

365   organic matter content of soils can be increased through use of improved crop varieties or

366   grassland species mixtures with greater root mass or deeper roots (Kell, 2012), improved crop

367   rotations in which C inputs are increased over a rotation (Burney *et al.*, 2010), greater residue

368   retention (Wilhelm *et al.*, 2004), and use of cover crops during fallow periods to provide

369   year-round C inputs (Burney *et al.*, 2010; Poeplau & Don, 2015). Several studies report that

370   soil carbon increases in croplands under no-till management (West & Post, 2002; Ogle *et al*.,

371   2005). However, the carbon benefits of no-till may be limited to the top 30cm of soil

372   (Blanco-Canqui & Lal 2008; Powlson *et al*., 2014). Baker *et al*. (2007) found that total soil

373   carbon was similar in non-till and conventional systems, suggesting that carbon accumulation

374   is occurring at different depths in the soil profile under different management schemes. Given

375   the larger variability in sub-surface horizons and lack of statistical power in most studies,

376   more research is needed on soil carbon accumulation at depth under different tillage regimes

377   (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_2O emissions from N fertilizers (Liu *et al.*, 2012; Huang *et al.*, 2012), as well as

386   enhancing soil fertility (Woolf *et al.*, 2010).

387

388   *3.3 Water management*

389

390   The amount of irrigated croplands has doubled in the last 50 years and now accounts for 70%

391   of all water use on the planet (Gleick, 2003). While irrigated crops cover 24% of all cropland

392   area, they account for 34% of all production (Siebert & Döll, 2010). Irrigation is concentrated

393   in precipitation-limited areas such as India, China, Pakistan, and the USA, which account for

394   72% of irrigation water use (West *et al.*, 2014). Agricultural water-use competes with uses

395   for human and natural ecosystems exacerbating water stress in dry regions. Increased

396   irrigation has occurred in many areas of world agriculture due to the increasing frequency of

397   drought under the climate change (West *et al.*, 2014). Where irrigation increases productivity

398   (e.g. in drought prone areas), organic carbon inputs to the soils would be expected to

399   increase, increasing soil organic matter content (section 3.2).

400

401   Irrigation can increase soil salinity in dry regions with high salt content in the subsoil

402   (Ghassemi *et al.*, 1995; Setia *et al.*, 2011). Where salinization occurs, additional irrigation is

403   needed to "flush" the salts beyond the root zone of the crops, which can further exacerbate

404   stress on water resources, particularly when using underground water sources. Saline soils,

405   which have a high concentration of soluble salts, occupy approximately 3.1% (397 Mha) of

406   the world's land area (FAO, 1995). Climate change (need for more frequent irrigation) and

407   increases in human population (increasing demand for more production) are likely to increase

408   the extent of saline soils (Rengasamy, 2008). The energy required by plants or soil organisms

409   to withdraw water from the soil or retain it in cells increases with decreasing osmotic

410   potential. As soils dry out, the salt concentration in the soil solution increases (decreasing

411   osmotic potential), so two soils of different texture may have the same electrical conductivity,

412   but the osmotic potential is lower in the soil with low water content (Setia *et al.*, 2011a;

413   Chowdhury *et al.*, 2011; Ben-Gal *et al.*, 2009). The accumulation of salts in the root zone has

414   adverse effects on plant growth activity, not only due to negative osmotic potential of the soil

415   solution resulting in decreased availability of water to plants, but also ion imbalance and

416   specific ion toxicity (Chowdhury *et al.*, 2011). Salinity affects microorganisms mainly by

417   decreasing osmotic potential, which affects a wide variety of metabolic activities and alters

418   the composition and activity of the microbial community (Chowdhury *et al.*, 2011) and

419   thereby soil organic matter decomposition.

420

421   In saline soils, SOC content is influenced by two opposing factors: reduced plant inputs

422   which may decrease SOC, and reduced rates of decomposition (and associated mineralisation

423   of organic C to CO_2) which could increase SOC content if the C input were unchanged.

424   Using a modified Rothamsted Carbon model (RothC) with a newly-introduced salinity

425   decomposition rate modifier and a plant input modifier (Setia *et al.*, 2011b, 2012), Setia *et al.*

426   (2013) estimated that, historically, world soils that are currently saline have lost an average of

427   3.47 t SOC ha^{-1} since they became saline. With the extent of saline soils predicted to increase

428   under the future climate, Setia et al. (2013) estimated that world soils may lose 6.8 Pg SOC

429   due to salinity by the year 2100. Soil salinization is difficult to reverse, but salt tolerant plant

430   species could be used to rehabilitate salt affected soils (Setia et al., 2013).

431

432   Water efficiency can be improved through management practices that reduce water

433   requirement and evaporation from the soil (such as adding mulch as groundcover), more

434   precise irrigation scheduling and rates, fixing leaks in dryland irrigation systems, improved

435   application technology (e.g., drip irrigation) and use of intermittent irrigation in rice paddies.

436   Given that water limitation is projected to become even more limiting in several semi-arid

437   regions, e.g. Sub-Saharan Africa, where the human population will probably increase most in

438   the future, and climate change impacts are projected to be severe, improved water harvesting

439   methods, e.g. storage systems, terracing and other methods for collecting and storing runoff,

440   are required to make best use of the limited water resource.

441

442   *3.4 Harvest frequency*

443

444   Approximately 9% of crop production increases from 1961-2007 was from increasing the

445   harvest frequency (Alexandratos & Bruinsma, 2012). The global harvested area (i.e. counting

446   each time an area is harvested) increased four times faster than total cropland area between

447   2000 and 2011 (Ray & Foley, 2013). The fraction of net primary production (NPP) extracted

448   by humans is increasing (Haberl *et al.*, 2007). Global warming is increasing the total area

449   suitable for double or even triple cropping in subtropical and warm temperate regions (Liu *et*

450   *al.*, 2013). The increase results from fewer crop failures, fewer fallow years, and an increase

451   in multi-cropping.

452

453   Increasing harvest frequency can reduce soil quality by e.g. continuously removing soil

454   nutrients and increasing soil compaction through greater soil traffic, but if legumes are

455   included in rotations as harvest frequency increases, soil quality could be improved.

456   Increasing harvest frequency may require increasing pesticide and herbicide use, and

457   increased use of fertilisers contributing to pollution (section 3.1). The net effect will depend

458   on the effectiveness of the management practices followed.

459

460   *3.5 Soil compaction*

461

462   Soil compaction causes degradation of soil structure by increasing soil bulk density or

463   decreasing porosity through externally or internally applied loads, as air is displaced from the

464   pores between the soil grains (McCarthy, 2007; Alakukku, 2012). It is the most important

465   subtype of physical soil deterioration, covering 68 Mha globally when first mapped in the

466   1990s (Oldeman et al., 1991). Compaction of agricultural soils often results from heavy

467   machinery or from animal trampling, so is more likely to occur in intensive agricultural

468   systems (machinery use and high stocking densities), and affects physical, chemical and

469   biological properties of soil. Top soil compaction can be reversed and controlled, but when

470   compaction creates impermeable layers in the subsoil, this is less easily reversed.

471

472   Subsoil compaction can disrupt nutrient water flows, which in turn can lead to reduced crop

473   yields, poorer crop quality and can give rise to increased GHG emissions, water and nutrient

474   run-off, erosion, reduced biodiversity and reduced groundwater recharge (Batey, 2009).

475   Where compaction cannot be avoided, mitigation is necessary. Biological approaches to

476   mitigation include planting deep rooted plants such as agroforestry; chemical methods

477   include fertilization (to overcome yield penalty, though not to remedy compaction); and

478   technical measures include machinery in which planting does not coincide with wheel tracks,

479   wide tyres / reduced tyre pressures to reduce pressure per unit area, and precision farming to

480   retain the same wheel tracks each year (Hamza & Anderson, 2005).

481

482   *3.6 Livestock density*

483

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

485   demand from a growing population and increase in per-capita meat consumption, with total

486   demand for meat expected to grow by more than 200 Mt by 2050 (Alexandratos & Bruinsma,

487   2012). The greatest increases in per-capita consumption are projected to be in developing and

488   transition countries (Alexandratos & Bruinsma, 2012). Since the 1970s, most increased

489   livestock production has resulted from intensification: increasing livestock density and

490   shifting to a greater fraction of livestock raised in industrial conditions (Bouwmann *et al*.,

491   2006). For example, 76-79% of pork and poultry production is industrialized (Herrero &

492   Thornton, 2013). Manure, inputs for growing feed, and soil loss from intensively managed

493   areas can be major sources of water pollution to local and downstream freshwater

494   ecosystems. Clearing natural ecosystems for new pastures, particularly in arid and semi-arid

495   regions, typically occurs on low-productivity lands with a much higher risk of soil erosion

496   and soil carbon/nutrient depletion (Alexandratos & Bruinsma, 2012), and negatively impacts

497   water storage and biodiversity. The impacts of livestock production are particularly prevalent

498   for beef production, which has a least an order of magnitude greater impact on land, water,

499   GHGs, and reactive nitrogen compared to other livestock (Eshel *et al*., 2014; Ripple *et al*.,

500   2014). Moreover, industrial livestock production had led to an increased use of veterinary

501   medicines, antibiotics and hormones, posing potential risks to soil, water, ecosystems and

502   human health. Improved grazing management (e.g. optimised stocking density) can reduce

503   soil degradation, and thereby maintain and enhance organic matter content (McSherry &

504   Ritchie, 2013; see sections 3.2 and 4.3), and can reduce soil compaction, thereby increasing

505   infiltration and water storage and reduce risk of runoff and flooding downstream (Marshall *et*

506   *al.*, 2009).

507

508   **4.   Other land management**

509

510   *4.1 Forest management*

511

512   Logging and fire are the major causes of forest degradation in the tropics (Bryan *et al*., 2013).

513   Logging removes nutrients and negatively affects soil physical properties and nutrient levels

514   (soil and litter) in tropical (e.g. Olander *et al*., 2005; Villela *et al*., 2006; Alexander, 2012)

515   and temperate forests (Perez *et al*., 2009). Forest Fires affect many physical, chemical,

516   mineralogical, and biological soil properties, depending on fire regime (Certini, 2005).

517   Increased frequency of fires contributes to degradation, and reduces the resilience of the

518   biomes to natural disturbances. A meta-analysis of 57 publications (Nave *et al*., 2011)

519   showed that fire caused a significant decrease in soil C (-26%) and N (-22%). Fires reduced

520   forest floor storage (pool sizes only) by an average of 59% (C) and 50% (N), but the relative

521   concentrations of these two elements did not change. Prescribed fires caused smaller

522   reductions in C and N storage (-46% and -35%) than wildfires (-67% and -69%). These

523   differences are likely because of lower fuel loads or less extreme weather conditions in

524   prescribed fires, both factors that result in lower fire intensity. Burned forest floors recovered

525   their C and N pools in an average of 128 and 103 years, respectively. Among mineral soil

526   layers, there were no significant changes in C or N storage, but C and N concentrations

527   declined significantly (-11% and -12%, respectively). Mineral soil C and N concentrations

528   were significantly reduced in response to wildfires, but not after prescribed burning.

529

530   Forest fires produce charcoal, or black carbon, some of which can be preserved over centuries

531   and millennia in soils. Dissolved black carbon (DBC) from burning of the Brazilian Atlantic

532   forest continued to be mobilized from the watershed each year in the rainy season, despite the

533   fact that widespread forest burning ceased in 1973 (Dittmar *et al.*, 2012).

534

535   A large field study in the Amazon (225 forest plots) on the effects of anthropogenic forest

536   disturbance (selective logging, fire, and fragmentation) on soil carbon pools showed that the

537   first 30 cm of the soil pool did not differ between disturbed primary forests and undisturbed

538   areas of forest, suggesting a resistance to impacts from selective logging and understory fires

539   (Berenguer *et al.*, 2014). As with deforestation, impacts of human disturbances on the soil

540   carbon are of particular concern in tropical forests located on organic soils and on steep

541   easily-eroded slopes.

542

543   *4.2 Shifting cultivation*

544

545   Shifting cultivation practices, where land is cleared through fire, have been practiced for

546   thousands of years, but recent increasing demographic pressure has reduced the duration of

547   the fallow period, affecting the system sustainability. Moreover, especially in Southeast Asia

548   where urbanisation is expanding in fertile planes, shifting cultivation is practiced in sloping

549   uplands, which are prone to soil and carbon loss by erosion (Chaplot *et al.*, 2005). A review

550   by Ribeiro Filho *et al.* (2013) reported negative impact on SOC associated with the

551   conversion stage, modified by the characteristics of the burning. Chop-and-mulch of enriched

552   fallows appears to be a promising alternative to slash-and-burn, conserving soil bulk density,

553   and significantly increasing nutrient concentrations and organic matter content compared to

554   burnt cropland, and a control forest in a study in the Amazon (Comtea *et al.*, 2012).

555

556   *4.3 Grassland management and dryland degradation*

557

558   Grasslands, including rangelands, shrublands, pastureland, and cropland sown with pasture

559   and fodder crops, cover 26% of the global ice-free land area and 70% of the agricultural area,

560   and contain about 20% of the world's soil organic carbon (C) stocks. Grasslands on every

561   continent have been degraded due to human activities, with about 7.5% of grassland having

562   been degraded because of overgrazing (Conant, 2012). A meta-analysis (McSherry & Ritchie,

563   2013) of grazer effects on SOC density (17 studies that include grazed and un-grazed plots)

564   found higher grazing intensity was associated with increased SOC in grasslands dominated

565   by C4 grasses (increase of SOC by 6–7%), but with lower SOC in grasslands dominated by

566   C3 grasses (decrease of SOC by an average 18%). An increase in mean annual precipitation

567   of 600 mm resulted in a 24% *decrease* in the magnitude of the grazer effect on finer textured

568   soils, but on sandy soils the same increase in precipitation produced a 22% *increase* in the

569   grazer effect on SOC (McSherry & Ritchie, 2013).

570

571   Land use dynamics and climate change are the major drivers of dryland degradation with

572   important feedbacks, with changes in plant community composition (e.g. shrub encroachment

573   and decrease in vegetation cover; D'Odorico *et al.*, 2013). A review by Ravi *et al.* (2010)

574   indicated soil erosion as the most widespread form of land degradation in drylands, with wind

575   and water erosion contributing to 87% of the degraded land. Grazing pressure, loss of

576   vegetation cover, and the lack of adequate soil conservation practices increase the

577   susceptibility of these soils to erosion. The degree of plant cover is negatively related to

578   aridity, and an analysis of 224 dryland sites (Delgado-Baquerizo *et al.*, 2013) highlighted a

579   negative effect of aridity on the concentration of soil organic C and total N, but a positive

580   effect on the concentration of inorganic P, possibly indicating the dominance of physical

581   processes such as rock weathering, a major source of P to ecosystems, over biological

582   processes that provide more C and N through litter decomposition (Delgado-Baquerizo *et al.*,

583   2013).

584

585   Soil carbon dynamics in pastures strongly depend on management, with soil carbon increases

586   or decreases observed for different combinations of animal densities and grazing frequency

587   (Conant 2012; Machmuller *et al.* 2015). Different grazing strategies, especially in the semi-

588   natural dryland biomes, have large implications for vegetation and the carbon balance (Yates

589   *et al.* 2000). Under certain conditions, grazing can lead to increased annual net primary

590   production over un-grazed areas, particularly with moderate grazing in areas with a long

591   evolutionary history of grazing and low primary production but this does not always lead to

592   an increase in soil carbon (e.g. Badini *et al.* 2007); grazing, like crop harvest, fundamentally

593   leads to the rapid oxidation of carbon that would otherwise be eventually transferred to the

594   soil. It has long been recognised that the potential effects of management on carbon storage in

595   grassland and dryland soils are substantially greater than that of climate change or CO_2

596   enhancement (Ojima *et al.* 1993), and Henderson *et al.* (2015) estimated that the optimization

597   of grazing pressure could sequester 148 Tg CO_2 yr^{-1}.

598

599   *4.4 Artificial surfaces, urbanisation and soil sealing*

600

601   In 2014, 54% of the world's population was urban, and by 2050, two thirds of the global

602   population will be urban. Many regions in the world, (such as Europe and Asia) are affected

603   by migration of populations from rural area to large megacities. Africa and Asia have more

604   rural populations, but are urbanizing faster than the other regions (World Urbanization

605   Prospects, 2014). With urbanization comes land take (development of scattered settlements in

606   rural areas, the expansion of urban areas around an urban nucleus, and densification on land

607   within an urban area) and soil sealing. Soil sealing refers to the permanent covering of an

608   area of land and its soil by impermeable artificial material (e.g. asphalt and concrete), for

609   example through buildings and roads. The area actually sealed is only part of a settlement

610   area, and gardens, urban parks and other green spaces are not covered by an impervious

611   surface (Prokop *et al*., 2011).

612

613   Sealing by its nature has a major effect on soil, diminishing many of its benefits (Tóth *et al*.,

614   2007). It is normal practice to remove the upper layer of topsoil, which delivers most of the

615   soil-related ecosystem services, and to develop a strong foundation in the subsoil and/or

616   underlying rock to support the building or infrastructure. Loss of ecosystem and social

617   services (mainly on high-quality soils) include impacts on water resources (e.g. reduction of

618   rainfall absorbed by the soil, reduction of soil water holding capacity affecting flooding), on

619   soil biodiversity when sealing prevents recycling of dead organic material (Marfenina *et al.*

620   2008), on the carbon cycle due to topsoil and vegetation removal (Davies *et al*., 2011).

621   Sealing through expansion of urban areas can also lead to agricultural land becoming more

622   marginal since the best agricultural land around settlements is lost as they expand, with

623   agricultural land displaced to more marginal land.

624

625   Appropriate mitigation measures can be taken in order to maintain some of the soil functions.

626   In urban planning management, objectives to reduce the impact of soil sealing include: i)

627   preventing the conversion of green areas, ii) re-use of already built-up areas (e.g. brownfield

628   sites Meuser, 2010; Hester & Harrison, 2001 – though remediation of contaminated sites can

629   be costly; Maderova & Paton, 2013), iii) using (where appropriate) permeable cover materials

630   instead of concrete or asphalt supporting green infrastructure, and iv) implementation of

631   compensation measures. In order to deliver this mitigation a number of actions are necessary,

632   e.g. reduction of subsidies that act as drivers for unsustainable land take and soil sealing

633   (Prokop *et al.*, 2011), and strong collaboration between relevant public authorities and

634   governance entities (Siebielec *et al.*, 2010). Development impacts can be reduced by

635   inclusion of green infrastructure, a network of high-quality green spaces and other

636   environmental features that have a positive effect on well-being (Gill *et al.,* 2007) as well as

637   soils. In some regions, urban sprawl is exacerbated by insufficient incentives to re-use

638   brownfield (derelict, underused or abandoned former industrial or commercial) sites, putting

639   increasing pressure on greenfield land take.

640

641   Actions to alleviate pressures on soils driven by sealing fall into three categories: limiting,

642   mitigating and compensating. Actions to limit soil sealing centre around reduction of land

643   take through development of spatial urban planning and environmental protection. Mitigation

644   of soil sealing entails use of strategic environmental assessment for plans and programmes,

645   use of permeable materials and surfaces, green infrastructure within built and urban

646   environments, and natural water harvesting. Compensating soil sealing entails reclamation of

647   degraded land, re-use of extracted topsoil, de-sealing and is incentivised by land take fees and

648   application of environmental cost calculations.

649

650   **5. Anthropogenic environmental change pressures that interact with land**

651   **management pressures on soils**

652

653   In addition to the direct impacts of humans on soils *via* land use change and land

654   management, anthropogenic activity has indirect impacts through human-induced

655   environmental change, such as pollution and climate change. These interact with land

656   management. Soils provide a temporary but labile store for pollutants (Meuser, 2010).

657   Natural processes can release pollutants back to the atmosphere, make them available to be

658   taken up by plants and organisms, leached in to surface waters (Galloway *et al.*, 2008) and/or

659   transported to other areas by soil erosion (Ravi *et al.*, 2010). Pollutants disrupt natural

660   biogeochemical cycles by altering both soil quality and function through direct changes to the

661   nutrient status, acidity and bioavailability of toxic substances and also by indirect changes to

662   soil biodiversity, plant uptake and litter inputs (EEA, 2014). Soil sensitivity to atmospheric

663   pollution varies with respect to key properties influenced by geology (cation exchange

664   capacity, soil base saturation, aluminium), organic matter, carbon to nitrogen ratio (C:N) and

665   water table elevation (EEA, 2014).

666

667

668   Atmospheric pollutant deposition impacts on soils vary with respect to soil sensitivity to a

669   specific pollutant and the actual pollutant load. Sulphur, nitrogen and heavy metals are

670   released in to the atmosphere by fossil fuel combustion (e.g. power generation, industry and

671   transport) and non-combustion processes (e.g. agricultural fertilizers, waste). These pollutants

672   are transported off-site and deposited as either dry or wet deposition, which can cross

673   national borders. Deposition is enhanced in forests and with altitude because of reduced wind

674   speeds and greater precipitation, respectively, impacting remote areas. Harmful effects to soil

675   function and structure occur where deposition exceeds the 'critical load' that a particular soil

676   can buffer (Nilsson & Grennfelt, 1988). Spatial differences in soil sensitivity (commonly

677   defined by the 'crucial load') and pollutant deposition result in an uneven global distribution

678   of impacted soils (Figure 3). For instance, global emissions of sulphur and nitrogen have

679   increased 3-10 fold since the pre-industrial period (van Aardenne *et al.*, 2001), yet only 7-

680   17% of the global land area sensitive to acidification is in a region where deposition exceeds

681   the critical load (Bouwman *et al.*, 2002).

682

683   Emissions of pollutants, notably sulphur, across Europe and North America have declined

684   since the 1980s following protocols established under the 1979 Convention on Long-range

685   Transboundary Air Pollution (LRTAP) and the 1990 US Clean Air Act Amendments

686   (CAAA) (Greaver *et al.*, 2012; Reis *et al.*, 2012; EEA, 2014). Conversely, emissions are

687   likely to increase in response to industrial and agricultural development in south and east

688   Asia, sub-Saharan Africa and South America (Kuylenstierna *et al.*, 2001; Dentener *et al.*,

689 2006). Further emission increases are occurring in remote areas due to mining activity, such

690 as oil sand extraction in Canada (Kelly *et al.*, 2010; Whitfield *et al.*, 2010).

691

692 *5.1 Sulphur deposition*

693

694 Sulphur emissions are primarily from combustion of coal and oil, typically associated with

695 power generation and heavy industry. In 2001, regions with deposition in excess of 20 kg S

696 ha^{-1} yr^{-1} where China and Republic of Korea, western Europe and eastern North America

697 (Vet *et al.*, 2014; Figure 3a). Deposition in un-impacted areas is <1 kg S ha^{-1} yr^{-1} (Figure 3a).

698 Pollution control measures have seen an 80% reduction in pollutant sulphur deposition across

699 Europe between 1990 and 2010 (Reis *et al.*, 2012), and emissions in China have declined

700 since 2005 (Fang *et al.*, 2013).

701

702 Soil acidification is a natural process that is altered and accelerated by sulphur and nitrogen

703 deposition (Greaver *et al.*, 2012). Sulphur oxides (SO$_x$) react with water to form sulphuric

704 acid (H$_2$SO$_4$). Excess inputs of acidity (H$^+$) displace soil base cations (e.g. calcium (Ca^{2+}) and

705 magnesium (Mg^{2+})) from soil surfaces into solution, which are subsequently lost by leaching

706 (Reuss & Johnson, 1986). Mineral soils can buffer base cation losses if inputs from rock

707 weathering and/or atmospheric dust deposition exceed the amount lost. Therefore, the global

708 distribution of acid sensitive soils is associated with conditions that favour development of

709 soils with low cation exchange capacity and base saturation (Bouwman *et al.*, 2002; Figure

710 3c). Wetland can also buffer inputs of acidity through biological sulphate reduction, although

711 acidity can be mobilised again following drought and drainage (Tipping *et al.*, 2003; Laudon

712 *et al.*, 2004; Daniels *et al.*, 2008). Organic acids can also buffer mineral acidity in naturally

713 acidic organic soils (Krug and Frink, 1983).

714

715 Decreased soil fertility or 'sterilisation' due to loss of nutrients and mobilisation of toxic

716 metals, particularly Al, are caused by acidification. Impacts in Scandinavia over the 1960s-

717 80s included declines in freshwater fish populations and damage to forests (EEA, 2014).

718 Sulphur can also stimulate microbial processes that make mercury bioavailable, leading to

719 bioaccumulation in the food chain (Greaver *et al.*, 2012). In agricultural soils in Europe,

720 however, fertilizer inputs of sulphur have increased to combat crop sulphur deficiencies as a

721 result of sulphur emission controls (Bender & Weigel, 2011).

722

723   Acidification is reversible, evident by increases in soil pH following decreased sulphur

724   emissions, although the recovery time varies; some areas with organic soils where deposition

725   has declined are showing either slow or no recovery (Greaver *et al.*, 2012; Lawrence *et al.*,

726   2012; RoTAP, 2012). On agricultural soils, lime can be applied to increase soil pH.

727   However, 50-80% of sulphur deposition on land is on natural, non-agricultural land

728   (Dentener *et al.*, 2006). Application of lime to naturally acidic forest soils can cause further

729   acidification of deep soil layers whilst increasing decomposition in surface litter, with no

730   improvement in tree growth (Lundström *et al.*, 2003).

731

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

733   Decreased organic matter decomposition due to acidification has increased soil carbon

734   storage in tropical forests (Lu *et al.*, 2014). However, in temperate forest soils acidification

735   can lead to reduced C:N ratios in soil, which in turn increases nitrification (Aber *et al.*, 2003),

736   but on already acidic soils reduces nitrification. In wetland soils, methane (CH_4) emissions

737   have also been suppressed by sulphur deposition (Gauci *et al.*, 2004). Conversely, declining

738   sulphur deposition has been associated with increased dissolved organic carbon fluxes from

739   organic soils (Monteith *et al.*, 2007), and decreased soil carbon stocks in temperate forest

740   soils (Oulehle *et al.*, 2011; Lawrence *et al.*, 2012).

741

742   *5.2 Nitrogen deposition*

743

744   Nitrogen deposition covers a wider geographical area than sulphur, as the sources are more

745   varied, including extensive agriculture fertilizer application, ammonia derived from livestock

746   operations, and biomass burning in addition to fossil fuel combustion (Figure 3b). Regions

747   with deposition in excess of 20 kg N ha^{-1} yr^{-1} in 2001 were western Europe, South Asia

748   (Pakistan, India, Bangladesh) and eastern China (Vet *et al.*, 2014); although extensive areas

749   with 4 kg N ha^{-1} yr^{-1} were found across North, Central and South America, Europe and Sub-

750   Saharan Africa. By contrast, 'natural' deposition in un-impacted areas is around 0.5 kg N ha^{-1}

751   yr^{-1} (Dentener *et al.*, 2006). While emissions related to fossil fuel combustion have declined

752   along with sulphur across Europe, agricultural sources of nitrogen are likely to stay constant

753   in the near future across Europe (EEA, 2014), whilst overall global emissions are likely to

754   increase (Galloway *et al.*, 2008). Nitrogen deposition in China's industrialized and

755   intensively managed agricultural areas in the 2000s was similar to peaks in Western Europe

756   during the 1980s before mitigation (Liu *et al.*, 2013).

757

758   Deposition of nitrogen induces a 'cascade' of environmental problems, including both

759   acidification and eutrophication that can have both positive and negative effects on ecosystem

760   services (Galloway *et al.*, 2003). Excluding agricultural areas where nitrogen is beneficial,

761   11% of land surface received nitrogen deposition above 10 kg N ha^{-1} yr^{-1} (Dentener *et al.*,

762   2006; Bouwman *et al*. 2002; Figure 3d). In Europe, eutrophication has and will continue to

763   impact a larger area than acidification (EEA, 2014).

764

765   Nitrogen fertilisation can increase tree growth (Magnani *et al.*, 2007) and cause changes in

766   plant species and diversity (Bobbink *et al.*, 2010), which in turn will alter the amount and

767   quality of litter inputs in to soils, notably the C:N ratio and soil-root interactions (RoTAP,

768   2012). However, increased carbon sequestration (Reay *et al*., 2008) may be offset by

769   increased emissions of the greenhouse gases N_2O and CH_4 (Liu & Greaver, 2009). Long-term

770   changes caused by nitrogen deposition are uncertain as transport times vary between

771   environmental systems; and the only way to remove excess nitrogen is to convert it to an

772   unreactive gas (Galloway *et al.*, 2008).

773

774   [Figure 3 here]

775

776   *5.3 Heavy metal deposition*

777

778   Heavy metal emissions are associated with coal combustion and heavy industry. In the UK,

779   deposition is responsible for 25-85% of inputs to UK soils (Nicholson *et al.*, 2003). In

780   Europe, the areas at risk from cadmium, mercury and lead deposition in 2000 were 0.34%,

781   77% and 42% respectively, although emissions are declining (Hettelingh *et al.*, 2006).

782   Tighter legislation to control industrial emissions of heavy metals are helping to reduce the

783   environmental load of heavy metals in many regions, though rapid industrial growth in some

784   regions such as East Asia is increasing pressures on soil from heavy metal deposition. Global

785   heavy metal emissions and deposition are poorly understood in comparison to sulphur and

786   nitrogen; although the on-site impact of heavy metal contamination has been well studied

787   (Guo *et al.*, 2014). Metals in bioavailable form have toxic effects on soil organisms and

788   plants, influencing the quality and quantity of plant inputs to soils, rate of decomposition and,

789   importantly, can bio-accumulate in the food chain. Some heavy metals will persist for

790   centuries as they are strongly bound to soil organic matter (RoTAP, 2012), although they can

791   be mobilised to bioavailable form following drought-induced acidification, drainage and soil

792   erosion (Tipping *et al.*, 2003; Rothwell *et al.*, 2005).

793

794   Whilst the direct impacts of sulphur, nitrogen and heavy metals on inorganic soil chemical

795   processes are generally well understood, many uncertainties still exist about pollutant impacts

796   on biogeochemical cycling, particularly interactions between organic matter, plants and

797   organisms in natural and semi-natural systems (Greaver *et al.*, 2012). Process understanding

798   is dominated by research in Europe and North America (e.g. Bobbink *et al.*, 2010). Research

799   is needed across Asia, Africa and South and Central America where soil properties and

800   environmental conditions differ. Models need to be developed to examine the combined

801   effects of air pollutants and their interactions with climate change impacts and feedbacks on

802   greenhouse gas balances and carbon storage (Spranger *et al.*, 2008; RoTAP, 2012). Air

803   quality, biodiversity and climate change polices all impact on soils. A more holistic approach

804   to protecting the environment is needed, particularly as some climate change policies (e.g.

805   biomass burning, carbon capture and storage) have potential to impact air quality and,

806   therefore, soil quality (Reis *et al.*, 2012; RoTAP, 2012; Aherne & Posch, 2013).

807

808   Indirect impacts on soils can be addressed largely by preventing the pollution at source, or by

809   mitigating the adverse effects where these have already occurred. Air pollution control on

810   coal burning and increased car and fleet efficiency standards has been effective in reducing

811   sulphur deposition in many areas of the world, particularly in Europe since the 1970s.

812   Substitution of coal with bioenergy might also reduce sulphur emissions, but unless burned

813   cleanly in a controlled way, can also release pollutants to the air. In terms of nitrogen,

814   ammonia abatement techniques when fertilizers are spread (e.g. slurry injection) are helping

815   to reduce N deposition (Sutton *et al.*, 2007).

816

817   **6. Existing policies and practices that alleviate global change pressures on soils**

818   **from land use and management**

819

820   The previous text has highlighted specific anthropogenic activities that exert or alleviate

821   pressures on soils. Actions that alleviate pressures on soils driven by land use change and

822   land management can be broadly divided into three categories, those that:

823

824   1) Prevent conversion of natural ecosystems to other uses (e.g. protected areas, reduced

825   deforestation, prevention of wetland drainage, intensification rather than extensification);

826   2) Prevent soil degradation (erosion control, fire management, reduced tillage / conservation

827   agriculture, long term fallows, flood protection, use of organic amendments, intercropping,

828   improved rotations); and

829   3) Result in soil / ecosystem restoration (e.g. peatland rewetting, afforestation, re-vegetation

830   on degraded lands, improved grass varieties, appropriate animal stocking densities,

831   bioremediation).

832

833   Policies to encourage such actions were recently reviewed by Bustamante *et al.* (2014) and

834   include:

835

836   a) Economic incentives, e.g., special credit lines for low carbon agriculture and forestry

837   practices and projects, payment for ecosystem services (such as carbon storage) and tradable

838   credits such as carbon,

839   b) Regulatory approaches, e.g. enforcement of environmental law to protect natural areas, set-

840   aside policies,

841   c) Research, development and diffusion investments, e.g. increase of resource use-efficiency,

842   livestock improvement,

843   d) Information and certification schemes, e.g. in China, forest certification to promote

844   sustainable forest management, state regulation for protecting mandatory arable lands,

845   protection projects on Tibetan grasslands, a national wetland protection programme, and the

846   "grain for green" programme.

847

848   Many of these actions and policies are not directed at soil conservation, but nevertheless have

849   an effect on soil quality. Two of the main pieces of international policy that have served to

850   reduce pressures on soils, directly and indirectly, are the United Nations Convention to

851   Combat Desertification (CCD) and the United Nations Framework Convention on Climate

852   Change (UNFCCC). In general, policies and actions are important at all scales from

853   international conventions to local action, and local activity is encouraged by policies at

854   regional, national and global level. Policies to sustainably increase land productivity, for

855   example, can prevent land use change, and there are various other supporting actions that can

856   help deliver improvements, e.g. agricultural research, technology transfer, knowledge transfer

857   and improved rural infrastructure. Some examples of policies that impact on land

858   management and soil quality are given below.

859

860   *6.1 United Nations Framework Convention on Climate Change (UNFCCC) and other*

861   *climate specific policies*

862

863   Soil carbon storage and nutrient cycling as climate services are being increasingly recognised

864   e.g. under UNFCCC as part of national reporting and accounting, as part of life-cycle

865   greenhouse gas assessments for biofuels, in various regional initiatives and national efforts.

866   The UNFCCC is an international treaty, which came into force in 1994, setting an overall

867   framework for intergovernmental efforts to tackle the challenge posed by climate change.

868   The requirements for the 196 country Signatories (or 'Parties') to the UNFCCC include

869   adopting national mitigation policies and publishing national inventories of anthropogenic

870   emissions and sinks of greenhouse gases including activities on the land such as afforestation,

871   deforestation, agricultural management and wetland drainage and rewetting. Developed

872   country signatories have legally binding targets under the Kyoto Protocol and can count land

873   based emissions or sinks towards meeting these targets, thus incentivising activities that

874   protect soil carbon. Developing countries currently have voluntary targets and several

875   countries have made pledges that include reduced deforestation (e.g. Brazil and Indonesia) or

876   afforestation (e.g. 400000 km^2 in China). Under the Clean Development Mechanism (CDM)

877   developed countries can fund projects in developing countries that generate certified emission

878   reduction credits (CERCs). China, for example, has the largest number of CERCs in the

879   world (IFPRI, 2011). Brazil also has 180 CDM projects, the third largest number of CERCs

880   after China and India (Cole & Liverman, 2011). A number of projects in Africa, North

881   America and South Asia have a significant component for soil greenhouse gas emission

882   reduction of soil carbon sequestration, financed through the Verified Carbon Standard or the

883   American Carbon Registry.

884

885   As part of negotiations leading to the new climate treaty in Paris in December 2015, all

886   parties will be required to submit INDCs (Intended Nationally determined Contributions).

887   The new treaty will also include provision for REDD+ (reduced Emissions from

888   Deforestation and Degradation, including management of forests and enhancement of forest

889   carbon stocks). This could go some way to protecting forest soils, and negotiations have

890   been intense around methods for monitoring reporting and verification, with key issues such

891   as permanence (the risk the forest may be lost at a later date due to management or

892   environmental change) and leakage (displacement of land use change to other areas), and

893   how to finance such activities.

894

895   *6.2 United Nations Convention to Combat Desertification (CCD)*

896

897   The CCD entered into force in December 1996; today 179 countries acknowledge it as a

898   legally binding framework to tackle land degradation and promote sustainable development

899   in fragile ecosystems. The Global Mechanism was established under the convention to

900   "promote actions leading to the mobilization and channelling of substantial financial

901   resources, including for the transfer of technology, on a grant basis, and/or on concessional or

902   other terms, to affected developing country Parties". In September 2011 the United Nations

903   General Assembly declared a goal of building a world with no land degradation. In October

904   2011 parties to the CCD issued a declaration calling for zero land degradation and for

905   adopting sustainable land management as a way to achieve sustainable development.

906

907   *6.3 Millennium Development Goals (MDGs)*

908

909   Of the eight MDGs (UNDP, 2014a), soil protection is most relevant to the goal to ensure

910   environmental sustainability, since soils are critical in underpinning environmental

911   sustainability (Smith *et al.*, 2015). A complementary MDG, to develop a global partnership

912   for development, will improve the governance structure to deliver soil security. The other

913   MDG to which soils plays a critical contribution is the goal to eradicate extreme poverty and

914   hunger, with the role of soils in supporting food provision critical for the latter part of this

915   MDG (Smith et al., 2015). The MDGs are currently being revisited to set a post-2015

916   development agenda (UNDP, 2014b), with discussion around the themes of localising the

917   post-2015 development agenda, helping to strengthen capacities and build effective

918   institutions, participatory monitoring for accountability, partnerships with civil society,

919   engaging with the private sector, and culture and development. The key emerging principles

920   from the dialogue are participation, inclusion, and the need for strengthened capacities and

921   partnerships (UNDP, 2014b). It is important that soils play their role in delivering this post-

922   2015 agenda.

923

924   *6.4 Protected areas and the Convention on Biological Diversity (CBD)*

925

926   Many measures to protect biodiversity and vulnerable habitats also protect the soils

927   underpinning them, so numerous conservation actions around the world serve to protect soils,

928   even if this was not the primary aim (Smith *et al.*, 2013). Between 1990 and 2010, the

929   amount of forest land designated primarily for the conservation of biological diversity

930   increased by 35 percent, indicating a political commitment to conserve forests. These forests

931   now account for 12 percent of the world's forests (FAO, 2010). In India, a Supreme Court

932   ruling in 2011 on effective self-governance of "common" or communal land by local

933   communities may help to protect these valuable resources, and thereby the soils that underpin

934   them. Soil biodiversity is known to be important for soil function (Bodelier, 2011), yet it

935   rarely receives the attention enjoyed by larger flora and fauna within the ecosystem.

936

937   *6.5 Reduced deforestation and forest management*

938

939   Various actions have been implemented to reduce deforestation (Bustamante *et al.*, 2014),

940   and to reduce the impact of forestry activities, such as reduced impact logging. UNFCCC,

941   carbon markets and other international environmental programs have contributed to global

942   efforts to reduce deforestation in addition to other sustainable natural resource management

943   programs in countries and by industry. For example, zero deforestation commitments made

944   by several companies (many made in 2014), and activities from bodies such as the

945   Roundtable for Sustainable Palm Oil (RSPO) and the Forest Stewardship Council (FCO)

946   certification scheme. Land improvement has increased in East Asia between 1981 and 2006

947   despite population increase, attributed largely to policies promoting tree planting and forest

948   plantation programs in China and Korea. In Brazil, deforestation was rapidly reduced after

949   national laws and regulations were enacted to protect forests in the 1990s and early 2000s

950   (including the soy moratorium and the forest code), followed up by state and municipal

951   governments setting further by-laws enforcing the deforestation moratorium (Bustamante *et*

952   *al.*, 2014).

953

954   *6.6 Agricultural policies and practices*

955

956   The pressures on soils imposed by land use intensity change can be mitigated by regulation of

957   over-grazing and reduction of over-stocking on grazed grasslands, return of crop residues to

958   the soil, reduced tillage, best management practices, targeted nutrient management and

959   precision farming on croplands, and wetland / floodplain restoration. These actions have been

960   encouraged by various policies. Some examples include: The EU set-aside programme of the

961   1990s encouraged less intensive use of agricultural land where production is low and

962   environmental impacts are high. The EU Common Agricultural Policy ties agricultural

963   subsidies to implementation of best management practices and environmental protection, for

964   example through pillar 2 (rural development programmes) providing crop insurance for lower

965   fertilizer application rates; in Africa, policies for integrated land management to help protect

966   vulnerable soils; China's conservation tillage program (2012-2030); the USA Conservation

967   Reserve Program (set aside marginal lands, steep slopes).

968

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

970

971   The International Year of Soils in 2015 is an excellent opportunity to raise the profile of soils

972   in the minds of national and international policy makers, land managers, timber and agro-

973   industries, and the public. Ensuring that vulnerable and high environmental value soils (e.g.

974   peatlands) are considered when making policies and decisions about which habitats and

975   ecosystems to convert or to protect, will help to reduce the pressure on soils particularly

976   vulnerable to global change drivers such as land use and land management, and maintain

977   important ecosystem services. This is in part happening with agendas around valuation of

978   ecosystem services and life-cycle assessments of impacts of land use change that include soil

979   carbon. At a time when governments are negotiating a legally binding climate change treaty

980   and making national targets for greenhouse gas reduction, and revisiting the Millennium

981   Development Goals, keeping soil carbon and nitrogen central to land based greenhouse gas

982   monitoring and reporting will maintain awareness with policy makers and industries with

983   emissions reduction targets. Both science and policy agendas are increasingly concerned with

984   long-term food security, ensuring that soils are central to considerations of how to achieve

985   on-going increases in production will enable those increases to be more sustainable into the

986   future.

987

988   Research and policy regarding soil quality and sustainability is abundant, but patchy and

989   disjointed. To ensure that soils are protected as part of on-going wider environmental and

990   sustainable production efforts, soils cannot, and should not, be considered in isolation of the

991   ecosystems that they underpin, but the role of soils in supporting ecosystems and natural

992   capital needs greater recognition (Robinson *et al.*, 2013, 2014). This can, in part, be enhanced

993　through education and awareness-raising which has started during the International Year of

994　the Soils in 2015. The time is ripe to consider a global soil resilience programme, under the

995　auspices of a global body such as the UN or one of its delivery agencies such as the FAO to

996　monitor, recover or sustain soil fertility and function, and to enhance the ecosystem services

997　provided by soils. The lasting legacy of the International Year of Soils in 2015 should be to

998　bring together robust scientific knowledge on the role of soils, and to put soils at the centre of

999　policy supporting environmental protection and sustainable development.

1000

1009

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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)[a]		+8% (3) -12% (2)	-42% (3) -41% (1) -25% (2)[b] -30% (2)[c] -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 Temp				[-1 to +15%]	[-2 to -6%] -32% (4)
	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 Temp	+53% (3) +16% (4)	+18% (3) +29% (2) +20% (6)	+28% (4)	+19% (3) +26% (2)	
	Boreal					

Footnotes: [a] 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%; [b] Annual crops; [c] Perennial crops; 1 Wei *et al.* (2014a); 2 Don *et al.* (2011); 3 Guo & Gifford (2002; tropical and temperate zones compiled); 4 Poeplau *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 ($PgCO_2$)

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

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

| Agricultural management practice	Specific issue	Distribution	Major environmental consequence	Knowledge gap
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_2O 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 land 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.

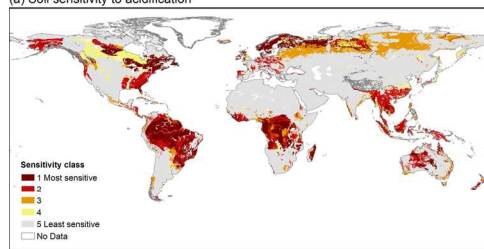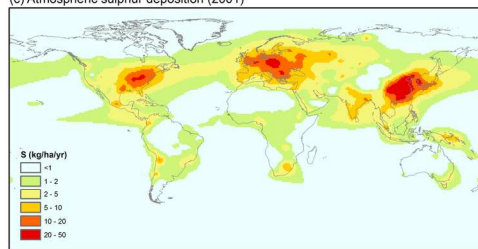254x190mm (96 x 96 DPI)
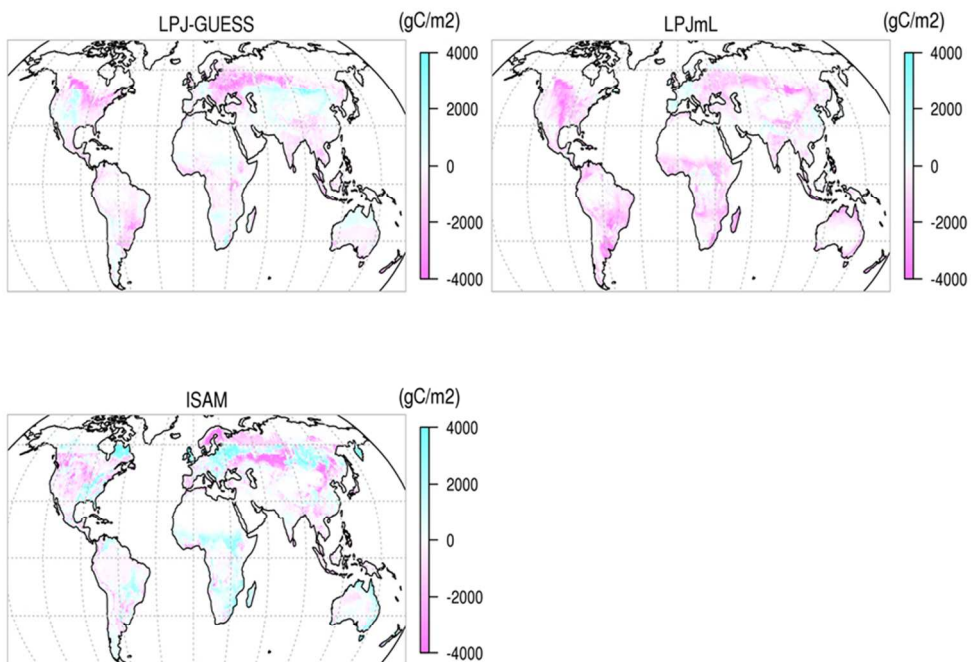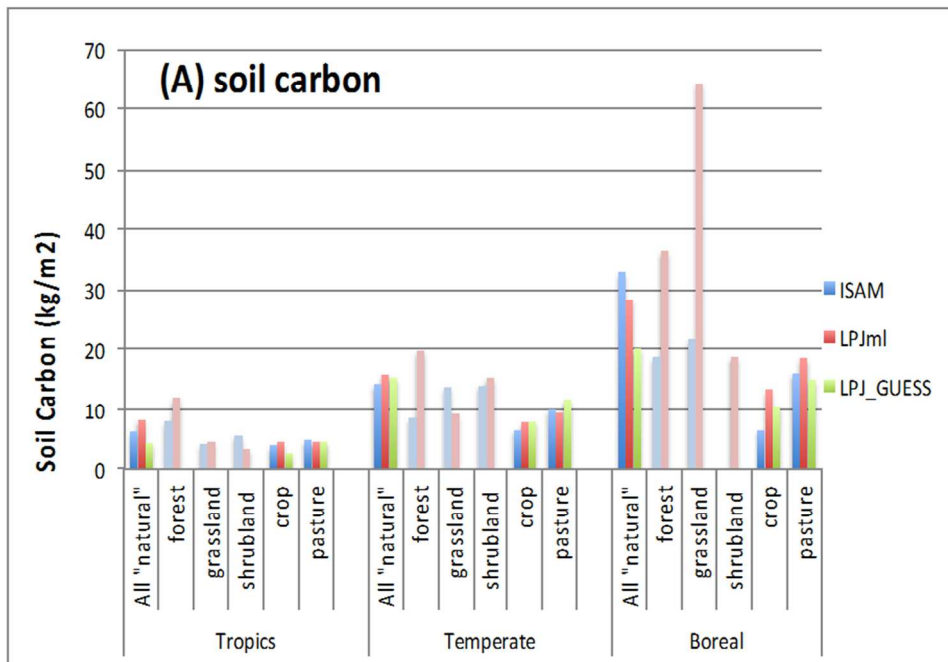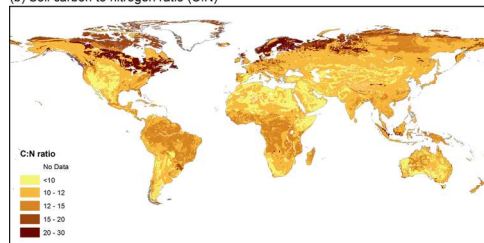
(A) soil carbon

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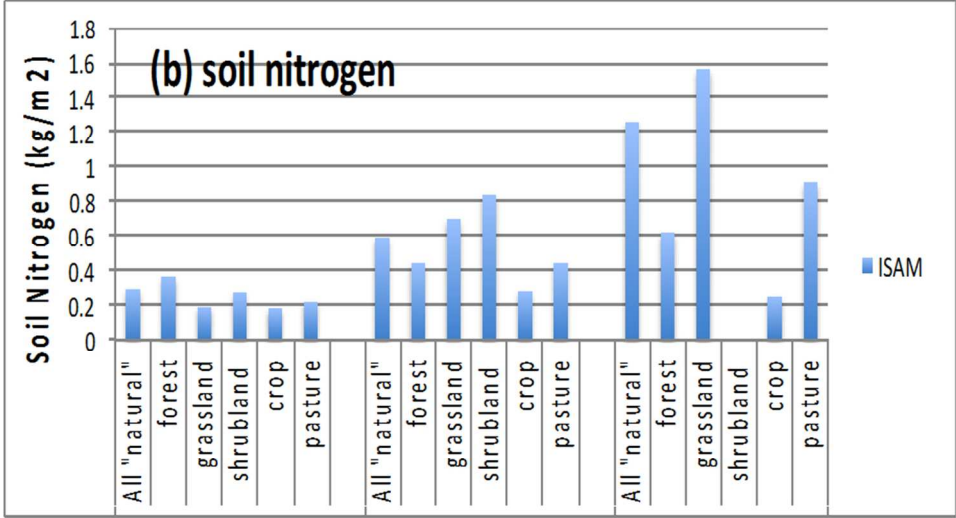
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(a) Soil sensitivity to acidification

Sensitivity class
- 1 Most sensitive
- 2
- 3
- 4
- 5 Least sensitive
- No Data

(c) Atmospheric sulphur deposition (2001)

S (kg/ha/yr)
- <1
- 1 - 2
- 2 - 5
- 5 - 10
- 10 - 20
- 20 - 50

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

C:N ratio
- No Data
- <10
- 10 - 12
- 12 - 15
- 15 - 20
- 20 - 30

(d) Atmospheric nitrogen deposition (2001)

N (kg/ha/yr)
- <1
- 1 - 2
- 2 - 5
- 5 - 10
- 10 - 20
- 20 - 50

168x101mm (300 x 300 DPI)