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RESEARCH REVIEW

# Land management: data availability and process understanding for global change studies

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## Abstract

In the light of daunting global sustainability challenges such as climate change, biodiversity loss and food security, improving our understanding of the complex dynamics of the Earth system is crucial. However, large knowledge gaps related to the effects of land management persist, in particular those human-induced changes in terrestrial ecosystems that do not result in land-cover conversions. Here, we review the current state of knowledge of ten common land management activities for their biogeochemical and biophysical impacts, the level of process understanding and data availability. Our review shows that ca. one-tenth of the ice-free land surface is under intense human management, half under medium and one-fifth under extensive management. Based on our review, we cluster these ten management activities into three groups: (i) management activities for which data sets are available, and for which a good knowledge base exists (cropland harvest and irrigation); (ii) management activities for which sufficient knowledge on biogeochemical and biophysical effects exists but robust global data sets are lacking (forest harvest, tree species selection, grazing and mowing harvest, N fertilization); and (iii) land management practices with severe data gaps concomitant with an unsatisfactory level of process understanding (crop species selection, artificial wetland drainage, tillage and fire management and crop residue management, an element of crop harvest). Although we identify multiple impediments to progress, we conclude that the current status of process understanding and data availability is sufficient to advance with incorporating management in, for example, Earth system or dynamic vegetation models in order to provide a systematic assessment of their role in the Earth system. This review contributes to a strategic prioritization of research efforts across multiple disciplines, including land system research, ecological research and Earth system modelling.

Keywords: data availability, earth system models, global land-use data sets, land management, land-cover modification, process understanding

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## Introduction

We have entered a proposed new geologic epoch, the Anthropocene, characterized by a surging human population and the accumulation of human-made artefacts resulting in grand sustainability challenges such as climate change, biodiversity loss and threats to food security (Steffen et al., 2015). Finding solutions to these challenges is a central task for policymakers and scientists (Reid et al., 2010; Foley et al., 2011). A central prerequisite to overcome these sustainability challenges is an improved understanding of the complex and dynamic interactions between the various Earth system components, including humans and their activities. However, many unknowns relate to the extent and degree of human impacts on the natural components of the Earth system. While a relatively robust body of knowledge exists on the effect of land-cover conversions, for example change in forest cover (Brovkin et al., 2004; Feddema et al., 2005; Pongratz et al., 2009), land-use activities that result in 'land modifications', that is changes that occur within the same land-cover type, remain much less studied (Erb, 2012; Rounsevell et al., 2012; Campioli et al., 2015; McGrath et al., 2015). Changes in land-use intensity are a prominent example for such effects (Erb et al., 2013a; Kuemmerle et al., 2013; Verburg et al., 2016). These land-use activities, which we here summarize under the term 'land management', are the focus of our review.

Evidence suggests that the effects of land management on key Earth system parameters are considerable (Mueller et al., 2015; Erb et al., 2016; Naudts et al., 2016) and can be of comparable magnitude as land-cover conversions (Lindenmayer et al., 2012; Luyssaert et al., 2014). Furthermore, management-induced land modifications cover larger areas than those affected by land conversions (Luyssaert et al., 2014). Omitting land management in assessing the role of land use in the Earth system may hence result in a substantial underestimation of human impacts on the Earth system, or difficulties to elucidate spatiotemporal dynamics and patterns of crucial Earth System parameters (e.g. Bai et al., 2008; Forkel et al., 2015; Pugh et al., 2015). This calls for the development of strategies that allow us to comprehensively and systematically quantify management effects (Arneeth et al., 2012).

However, two distinct – albeit interrelated – barriers hinder our current ability to fully assess land management impacts. First, major knowledge gaps exist in our qualitative and quantitative understanding of the biogeochemical and biophysical impacts of land management. Second, serious data gaps exist on the extent as well as intensity of various management practices. Here, we review the current state of knowledge of ten

common land management activities for their global impact, the level of process understanding and data availability to improve both analytical and modelling capacities as well as to prioritize future modelling and data generation activities.

## Key land management activities

During an interdisciplinary workshop cycle (see Acknowledgements), we identified ten important land management activities that may impact the Earth system profoundly (Table S1 in the Appendix S1), namely (i) forest harvesting; (ii) tree species selection; (iii) grazing and mowing harvest; (iv) crop harvest and crop residue management; (v) crop species selection; (vi) nitrogen (N) fertilization of cropland and grazing land; (vii) tillage; (viii) crop irrigation (including paddy rice irrigation); (ix) artificial drainage of wetlands for agricultural purposes; and (x) fire as a management tool (Fig. 1). These ten management practices were selected based on their global prevalence across a diversity of biomes and based on their strong biophysical and biogeochemical effects, as described in the literature. Table S1 provides definitions and lists ecosystems in which the management practices prevail and which are in the focus of our review. The provision of bioenergy, for example biofuels from plant oil, starch or sugar, or wood fuel, is not classified as own management type. Rather, it is subsumed under items i) and iv). It is important to note that this list represents a subjective, consensus-oriented group opinion and is thus neither exhaustive nor representative. For instance, many management activities have not been considered here, for example litter raking, peat harvest, phosphate or potassium fertilization, crop protection, forest fertilization or mechanization. Such activities can be of central importance, for example, in specific contexts, and advancing the understanding of their divers and impacts is equally important.

For each management activity, we compiled information on the current global extent; past, ongoing and anticipated dynamics; data availability; and state of knowledge on biogeochemical and biophysical effects. Biogeochemical effects include changes in greenhouse gas (GHG) and aerosol concentrations caused by changes in surface emissions (CO, CO<sub>2</sub>, H<sub>2</sub>O, N<sub>2</sub>O, NO<sub>x</sub>, NH<sub>3</sub>, CH<sub>4</sub>) or by changes in atmospheric chemistry (CH<sub>4</sub>, O<sub>3</sub>, H<sub>2</sub>O, SO<sub>2</sub>, biogenic secondary organic aerosols). Biophysical effects include changes in surface reflectivity (i.e. albedo) and changing surface fluxes of energy and moisture through sensible heat fluxes and evapotranspiration. The combined information is then used to suggest prioritizations of future research efforts.

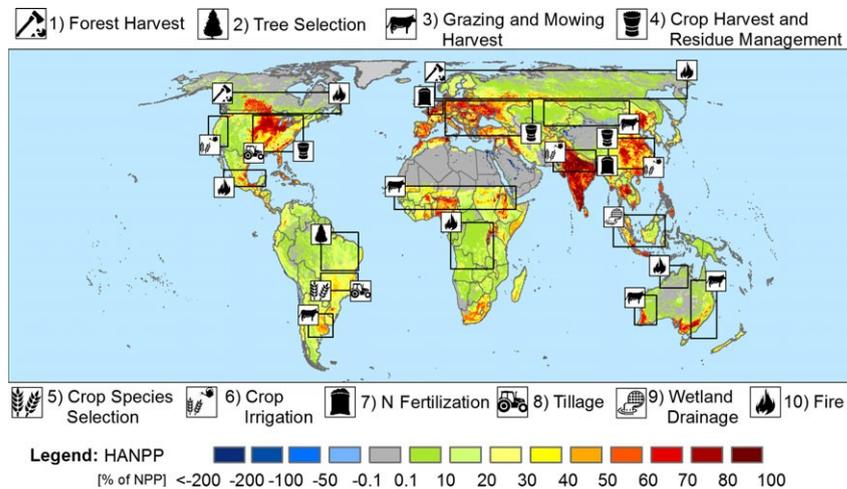


Fig. 1 The ten selected management activities and a selection of geographic regions where these activities play an important role. The background map displays the human appropriation of net primary production (Haberl et al., 2007; Copyright 2007 National Academy of Sciences, USA), that is the ratio between annual potential net primary production (NPP) and NPP remaining in ecosystems after harvest. Negative values indicate areas where due to management NPP remaining in ecosystems surmounts the hypothetical potential NPP.

### Forestry harvest

Extent and data availability. Forests cover 32.7–40.8 Mkm<sup>2</sup> or 30% of the ice-free land surface and 2/3–3/4 of global forests (26.5–29.4 Mkm<sup>2</sup>) are under some form of management (Erb et al., 2007; FAO, 2010; Pan et al., 2013; Luysaert et al., 2014; Birdsey & Pan, 2015). Forest use reaches back to the cradle of civilization (Perlin, 2005; Hosonuma et al., 2012), while scientific forest management, that is management schemes that involve careful planning based on empirical observations and forest ecological process understanding (Moral et al., 2016), originated in the late 18th century (Farrell et al., 2000). The share of managed forests and management intensity are expected to increase further along with global demand for wood products (Eggers et al., 2008; Meyfroidt & Lambin, 2011; Levers et al., 2014). Virtually all temperate and southern boreal forests in the Northern Hemisphere are already managed for wood production (Farrell et al., 2000). Northern boreal forests are at present largely unused for wood production (Erb et al., 2007) and could become increasingly managed in the future due to growing global demand for wood products and comparative advantages in boreal forestry compared to other regions (Westholm et al., 2015). Temperate forests are mostly under some version of age class-based management. In contrast, wood extraction from tropical forest often targets selected species, resulting in forest degradation. Significant parts of tropical forest (5.5 Mkm<sup>2</sup>) are in different stages of recovery from prior logging and/or agricultural use (Pan et al., 2011). The use of tropical forests is also predicted to

increase, both in extent and intensity, mainly to supply international markets (Hosonuma et al., 2012; Kissinger et al., 2012). 7% of managed forests are intensive plantations, 65% subject to regular harvest schemes, and 28% under other (e.g. sporadic) uses (Appendix S1). Data on wood harvest are surprisingly scarce (Table 1), given the importance of forests and forestry in the Earth system as well as a socio-economic resource. Time series of national-level data exist, but are uncertain, particularly regarding fuelwood harvest (Bais et al., 2015). This uncertainty is, among others, the result of differences in reporting schemes, induced by semantic discrepancies, or oversimplified approaches for creating gridded time series (Erb et al., 2013b; Birdsey & Pan, 2015).

Effects of forestry harvest. The knowledge on biogeochemical effects of wood harvest is relatively advanced, although considerable uncertainties still persist, and biogeochemical as well as biophysical effects are strong. Around 2000, forest harvest amounted to 1 Pg C (carbon) yr<sup>-1</sup> consisting of around 0.5 Pg C yr<sup>-1</sup> for wood fuel and another 0.5 Pg C yr<sup>-1</sup> as timber (Krausmann et al., 2008; FAOSTAT, 2015). Forest harvest mobilizes annually <0.5% of the global standing biomass (Saugier et al., 2001; Pan et al., 2011), but the flux represents around 7% of the global forest net primary production (NPP) (Haberl et al., 2007), reaching 15% in highly managed regions such as Europe (Luysaert et al., 2010). Uncertainty ranges in wood flows are large (Krausmann et al., 2008; Bais et al., 2015). In general, harvest reduces standing biomass compared to intact forest (Harmon et al., 1990; McGarvey et al., 2014), with the

Table 1 Overview of data availability for the ten land management activities reviewed in this study

Management activity	National statistics (based) w. global coverage*		Gridded spatial data, continental or global			Comments
	Static	Time series	Continental or ecozone, static	Global, static	Global, time series	
Forestry harvest	FAOSTAT (2015); FAO (2015a)	FAOSTAT (2015); FAO (2015a); Krausmann et al. (2013)	Europe: McGrath et al., (2015); Levers et al., (2014); Verkerk et al. (2015)	Haberl et al. (2007) – forest system approach	Hurt et al. (2011); [Europe: Vil�n et al. (2012): age class info. could be used for reconstructions]	Spatially explicit Information on used/unused forests lacking, but data on wilderness Sanderson et al. (2002) or intact forests (Potapov et al. 2008) might provide proxies (Erb et al. 2007). Oversimplified
Tree species selection	FAO (2015a)	FAO (2015a)	Europe: Brus et al. (2011); Hengeveld et al. (2012); McGrath et al. (2015)			FAO FRA only discerns the total area of planted forest. Other sources usually only discern coniferous from deciduous trees. Spatially explicit data on plantations lacking
Grazing and mowing harvest	Bouwman et al. (2005); Herrero et al. (2013); Krausmann et al. (2008); Wirsenius (2003);	Krausmann et al. (2013)	Petz et al. (2014), relying on Wint & Robinson (2007); Chang et al. (2015), based on ORCHIDEE-GM	Herrero et al. (2013) relying on Wint & Robinson (2007); Haberl et al. (2007)		Extreme uncertainty level – estimates on the global extent vary strongly ( $\pm 40\%$ ), and data on grazing volumes are not statistically reported but modelled only
Crop harvest + residue management	FAOSTAT (2015); Krausmann et al. (2008); Wirsenius (2003);	FAOSTAT (2015); Krausmann et al. (2013)		Haberl et al. (2007); Monfreda et al. (2008); Ray & Foley (2013); You et al. (2014);	Ray et al. (2012); Iizumi et al. (2014); Iizumi & Ramankutty (2016);	Intricacies relate to the difference between harvest yields (harvested biomass per harvest event) and physical yields (total harvest per land-use areas, including fallows)
Crop species selection	FAOSTAT (2015); FAO (2010);	FAOSTAT (2015)		Monfreda et al. (2008); You et al. (2014); Portmann et al. (2010);		No information on interannual dynamics, such as rotational schemes, available
N Fertilization	FAOSTAT (2015);	FAOSTAT (2015)		Potter et al. (2010); Mueller et al. (2012); Liu et al. (2010);		Spatially explicit data are modelling derived and show large discrepancies, in particular livestock manure is error prone No data on fertilization outside croplands

Table 1 (continued)

Management activity	National statistics (based) w. global coverage*		Gridded spatial data, continental or global			Comments
	Static	Time series	Continental or ecozone, static	Global, static	Global, time series	
Tillage						No data on tillage, but presumable all croplands are tilled with two exceptions: permanent crops and zero tillage agriculture. For the latter, no data are available
Irrigation (including paddy rice)	FAOSTAT (2015);	FAOSTAT (2015);	Parry rice: Frohking et al. (2006);	Portmann et al. (2010) Salmon et al. (2015) Wisser et al. (2008)	Freydank & Siebert (2008); Siebert et al. (2015);	Many data, for example those by FAO, relate to area equipped for irrigation, while the amount of water actually used is difficult to assess. Higher quality for paddy rice
Artificial wetland drainage				Feick et al. (2005);		Poor data availability. Gridded assessments cover all drainage, not only wetlands
Fire as management tool	Human-induced fires: Lauk & Erb (2009);		All fires: for example, Africa: Liousse et al. (2010); Canada: Stocks et al. (2002)	All fires: for example Giglio et al. (2013); Alonso-Canas & Chuvieco (2015);	All fires: for example Giglio et al. (2013);	Problems relate to discerning natural from human-induced fires as well as agricultural fires. Scarce data for prescribed fires and no data on fire prevention available

\*Statistical or statistical data derived sources with global coverage only. Please note that at the continental or subcontinental level, many more data sets are available. Prominent data providers (nonexhaustive) are Eurostat for European countries (<http://ec.europa.eu/eurostat>) or the United States Department of Agriculture (<http://www.ers.usda.gov/topics.aspx>).



notable exception of coppices (Luyssaert et al., 2011). Soil and litter carbon pools generally decrease only slightly, but deadwood decreases in managed forests by 95% compared to old-growth forests (McGarvey et al., 2014). Nevertheless, the net effect of forest management on carbon stock reductions on the one hand and wood use for fossil fuel substitution on the other remain unclear, due to complex legacy effects (Marland & Schlamadinger, 1997; Lippke et al., 2011; Holtmark, 2012). The effects of forest management on CH<sub>4</sub> and N<sub>2</sub>O emissions are considered negligible, with the exception of fertilized short-rotation coppices (Robertson et al., 2000; Zona et al., 2013). Predicted intensification of forest management by means of short-rotation coppicing or total tree harvest may require frequent fertilization, potentially resulting in increased N<sub>2</sub>O emissions (Schulze et al., 2012).

Robust empirical evidence exists on multiple interactions between forest harvest and biophysical processes. Thinning practices affect the albedo by up to 0.02 in the visible range and 0.05 in the near infrared, with intensive thinning having the largest effect (Otto et al., 2014). The albedo of forests could decrease with age, and thus longer rotations, due to changes in canopy structure (Amiro et al., 2006; Hollinger et al., 2010; Rautiainen et al., 2011; Otto et al., 2013). The length of rotations substantially affects tree height, which affects surface roughness (Raupach, 1994; Nakai et al., 2008). Through removal of leaf mass, harvest can reduce evapotranspiration by 50% (Kowalski et al., 2003). At the stand level in tropical forests, gaps resulting from selective cutting could modify local circulation resulting in a drier sub-canopy (Miller et al., 2007) which in turn could increase fire susceptibility. In temperate and boreal sites, biophysical effects of forest management on surface temperature were shown to be of a similar magnitude (e.g. around 2K at the vegetation surface) as the effects of land-cover changes (Luyssaert et al., 2014).

### Tree species selection

Extent and data availability. Forest plantations cover 2.2 Mkm<sup>2</sup>, being particularly important in, for example, in China, Brazil, Chile, New Zealand and South Africa (FAO, 2015a). Species composition is also affected by management in less intensively managed forests on up to 18 Mkm<sup>2</sup> (Luyssaert et al., 2014). In Europe, for instance, species selection has resulted in an increase of 0.5 Mkm<sup>2</sup> of conifers since 1750, largely at the expense of deciduous species (McGrath et al., 2015). Although species selection has become more salient in the last century, this practice dates back 4k to 5k years (Bengtsson et al., 2000). Planted forests, mainly with conifer species, cover 9% of total

forest area in the United States (Oswalt et al., 2014) and 7% of the global used forests (Appendix S1). Whether the tendency of species selection will continue depends on climate-driven changes in tree species occurrence (Hanewinkel et al., 2013). Data on tree species selection are particularly scarce (Table 1; Appendix S1) and prone to large uncertainties. Spatially explicit information on present-day species distribution (Brus et al., 2011) could inform reconstructions of past species selection (McGrath et al., 2015). For industrial plantations of typically fast-growing tree exotic species, the most extreme form of species selection, data are only available in short time series from FAO Forest Resources Assessments (FAO, 2015a).

Effects of tree species selection. The biogeochemical and biophysical effects of tree species selection are well documented, and in particular, biophysical parameters are strongly affected. Species selection affects carbon allocation between above- and belowground pools, nitrogen cycling, evapotranspiration rates and surface albedo (Farley et al., 2005; Kirschbaum et al., 2011). Species composition can affect the fate of soil carbon, with larger stocks under hardwoods or nitrogen-fixing tree species (Paul et al., 2002; Resh et al., 2002; Bårçena et al., 2014). Pine plantations are commonly reported to lead to soil carbon losses, compared to broadleaf species including Eucalyptus (Paul et al., 2002; Farley et al., 2005; Berthrong et al., 2009). Also, tree mixes, especially with nitrogen-fixing species, store at least as much, if not more, carbon as monocultures of the most productive species of the mixture (Hulvey et al., 2013). These effects are, however, location dependent. For the boreal zone in Europe, soil carbon stocks were larger on sites afforested with conifers compared to those where deciduous species prevailed (Bårçena et al., 2014). Tree species selection and species mixtures can be used to prevent spread of disease and pests that cause large releases of carbon through tree mortality or to improve the recovery after damages have occurred (Boyd et al., 2013). For the boreal and temperate zones, information about the emission potential of biogenic volatile organic compounds (BVOCs) for different species is now available (Kesselmeier & Staudt, 1999). Uncertainty, however, is large concerning the evolution of emission potentials of different species under climate change and their feedback on the climate itself. The uncertainty on whether the climate effect of BVOCs is dominated by its direct warming or its indirect cooling due to its role as condensation nuclei (Penuelas & Llusià, 2003) suggests that BVOCs might be one of the remaining key uncertainties in

quantifying the climate effect of tree species selection.

Forest composition affects albedo through canopy height, canopy density and leaf phenology. Over a 100 year long rotation, tree species was found to explain 50–90% of the variation in short wave albedo (Otto et al., 2014). In absolute terms, summer albedo ranges between 0.06–0.10 and 0.12–0.18 for evergreen coniferous and broadleaved deciduous forest, respectively (Hollinger et al., 2010). As different tree species grow to different heights, differing by up to several metres under the same environmental conditions, roughness length is also affected. Changes in roughness and thus turbulent exchange as well as different efficiencies of evapotranspiration of tree species can alter the water balance. Species conversion from pine to hardwood forest resulted in a sustained decrease in streamflow of about 200 mm yr<sup>-1</sup> for sites experiencing similar precipitation (Ford et al., 2011). Similar decreases were observed where Eucalyptus replaced pines, with the effect increasing with forest age (Farley et al., 2005). At a single site in the south-eastern United States, the radiative temperature of deciduous forest was 0.3K higher than that of coniferous forest (Stoy et al., 2006; Juang et al., 2007). Over Europe, a massive conversion of deciduous to coniferous forests has warmed the lower boundary layer by 0.08K between 1750 and 2010 (Naudts et al., 2016).

### Grazing and mowing harvest

Extent and data availability. Grazing and mowing harvest is the most spatially extensive land management activity worldwide, covering 28–56 Mkm<sup>2</sup> or 21–40% of the terrestrial, ice-free surface, with a wide range of grazing intensity (Herrero et al., 2013; Luyssaert et al., 2014; Petz et al., 2014; FAOSTAT, 2015). Grazing is one of the oldest land management activities, reaching back 7–10k years (Blondel, 2006; Dunne et al., 2012), and occurs across practically all biomes: from arid to wet climates and over soils with varying fertility (Asner et al., 2004; Steinfeld et al., 2006; Erb et al., 2007). Livestock fulfils many functions beyond the provision of food (FAO, 2011), but animal-based food production almost increased exponentially since the 1950s, due to increasing population and more meat- and dairy-rich diets (Naylor et al., 2005; Kastner et al., 2012; Tilman & Clark, 2014). These trends are expected to continue, but depending on the degree of intensification of livestock production systems, the uncertainties on future net changes in grazing lands area are very large (Alexandratos & Bruinsma, 2012). Data on the extent of grazing areas show large discrepancies (Erb et al., 2007), and grazing intensity is high on <10%, medium on around

two-thirds and low on one-fourth of the grazing lands (Appendix S1). Existing national and gridded data on grazing usually refer to recent time periods, do not separate grazing and mowing and are subject to severe uncertainties (Table 1), exacerbated by problems with conflicting definitions (Erb et al., 2007; Ramankutty et al., 2008).

Effects of grazing and mowing harvest. While large knowledge gaps relate to the extent and intensity of grazing, the biogeochemical and biophysical impacts of grazing are well documented. While biophysical effects are found to be relatively low, strong biogeochemical effects relate to this activity. Estimates on the amount of grazed and mowed biomass show a large range from 1.2 to 1.8 Pg C yr<sup>-1</sup> in 2000 (Wirsenius, 2003; Bouwman et al., 2005; Krausmann et al., 2008; Herrero et al., 2013), which is up to one-third of the total global socio-economic biomass harvest (Krausmann et al., 2008). Grazing is a key factor for many ecosystem properties, including plant biomass and diversity. Grazing can both deplete and enhance soil C stocks, depending on grazing intensity. For example, in arid lands, overgrazing is a pervasive driver of loss of soil function (Bridges & Oldeman, 1999), resulting in reductions in soil organic carbon (SOC) and aboveground biomass (Gallardo & Schlesinger, 1992; Asner et al., 2004). In semi-arid regions, high grazing pressures could lead to woody encroachment (Eldridge et al., 2011; Anadón et al., 2014) and thus to an increase in both above- and belowground carbon stocks. A global meta-analysis of grazing effects on belowground C revealed large differences in the response of C3- and C4-dominated grasslands under different rainfall regimes (McSherry & Ritchie, 2013). Globally, the response of plant traits to grazing is influenced by climate and herbivore history (Díaz et al., 2007). At the same time, grazing can influence ecosystem C uptake in the Arctic tundra, with implications for response to a warming climate (Vassilainen et al., 2014). Incorporation of current grazing and grazing history into climate models will improve predictions of terrestrial C sinks and sources.

Forest grazing (e.g. reindeer grazing in the boreal zone) directly affects the understorey and indirectly forest growth through nutrient export, recruitment and the promotion of grazing tolerant species (Adams, 1975; Erb et al., 2013b), but comprehensive assessments are lacking. The production of methane is an important biogeochemical effect of ruminant grazers, strongly determined by the fraction of roughage (grass biomass) in feedstuff (Steinfeld et al., 2006; Thornton & Herrero, 2010; Herrero et al., 2013), but large uncertainties related to quantities remain (Lassey, 2007). Soil compaction, induced, for example, by trampling, can

contribute to anaerobic microsites, reducing the CH<sub>4</sub> oxidation potential of the soil (Luo et al., 1999). Nitrogen cycling is strongly affected by the addition of manure and urine (Allard et al., 2007). The effect of animal waste N inputs interacts with poor drainage, influenced also by topography, to result in localized greater N<sub>2</sub>O fluxes (Saggar et al., 2015). Biogeochemical effects of grazing are influenced by livestock density. Some modelling and site-specific studies have found that a reduction of livestock densities results in increased soil C storage and decreased N<sub>2</sub>O and CH<sub>4</sub> (Baron et al., 2002; Chang et al., 2015). A study of year-round measurements of N<sub>2</sub>O in the Mongolian steppe found that while animal stocking rate was positively correlated with growing-season emissions, grazing decreased overall annual N<sub>2</sub>O emissions (Wolf et al., 2010). Sites with little and no grazing showed large pulses of N<sub>2</sub>O release during spring snowmelt compared to high grazing sites, suggesting that grazing may influence N cycling response to changes in climate in high-altitude ecosystems. Biophysical effects of grazing mainly depend on ecosystem type and soil properties. In local contexts, grazing has been reported to reduce plant biomass, thus increasing albedo by about 0.04 compared to unmanaged grassland (Rosset et al., 2001; Hammerle et al., 2008). However, the effect of soil exposure resulting from canopy decreases is ambiguous, resulting in an albedo reduction on dark soils (Rosset et al., 1997; Fan et al., 2010) and in an albedo increase on bright soils (Li et al., 2000). Reindeer grazing has been reported to reduce albedo due to a reduction of the light-coloured lichen layer (Cohen et al., 2013). Reductions in roughness length due to grazing are expected to have a small affect on turbulent fluxes (i.e. surface fluxes of energy, moisture and momentum), but can lead to enhanced soil erosion (Li et al., 2000). The observed effect of mowing on the cumulative evapotranspiration was small (10% increase, about 40 mm), although sufficient to decrease soil water content in a managed field (Rosset et al., 2001). The integrated climate effect from excluding grazing by bison in the Great Plains was modelled to be a 0.7K decrease in maximum temperatures and a small increase in minimum temperatures (Eastman et al., 2001).

#### Crop harvest and residue management

Extent and data availability. Approximately 15 Mkm<sup>2</sup> or 12% of the global terrestrial, ice-free surface is currently used as cropland (Ramankutty et al., 2008; FAOSTAT, 2015). Of these, 1.4 Mkm<sup>2</sup> are permanent cultures, including perennial, woody vegetation (e.g. fruit trees, vineyards). Approximately two-thirds of the arable land is harvested annually, with cropping season

extending over approximately six months, while one-third of cropland remains temporarily idle on average (Siebert et al., 2010). On one-quarter of the global cropland multicropping (i.e. more than one harvest per year) occurs (Appendix S1). Cropping activities are closely tied to the sedentary lifestyle that emerged with the Neolithic revolution some 12k years ago, marking the beginning of the Holocene. Since then, cropland has significantly expanded at the expense of grasslands, forests and wetlands. Sedentary cropland management origins from shifting cultivation (Boserup, 1965), that is the alteration of short cultivation and long fallow periods, which was a particularly widespread form of cropland management in many regions of the world (Emanuelsson, 2009) and illustrates the highly interconnected nature of management and land-cover change. Today, this form of land use is declining at the global scale, although it remains important in many frontier areas characterized by, for example, unequal or insecure access to investment and market opportunities or in areas with low incentives to intensify cropland production (van Vliet et al., 2012). Cropland expansion is tied to human population growth, but moderated by technological development that allowed for substantial yield increases per cropland area, in particular after 1950 (Pongratz et al., 2008; Kaplan et al., 2010; Ellis et al., 2013; Krausmann et al., 2013). The dynamics of cropland expansion and contraction in different regions of the world are caused by complex interactions between endogenous factors such as population dynamics, consumption patterns, technologies and political decisions, and exogenous forces related to international trade and other manifestations of globalization, in interplay with intensification dynamics (Krausmann et al., 2008, 2013; Meyfroidt & Lambin, 2011; Kastner et al., 2012; Kissinger et al., 2012; Ray et al., 2012; Ray & Foley, 2013). Cropland shows the highest land-use intensity, compared to grazing land or forest, in terms of inputs to land (capital, energy, material) as well as outputs from land (Kuemmerle et al., 2013; Niedertscheider et al., 2016). The spatial extent of cropland is probably the best-described land-use feature at the global scale, with many data sets existing (see Table 1). Nevertheless, major uncertainties remain related to cropland patterns in some world regions, particularly across large swaths of Central, Southern and Northern Africa, Brazil and Papua New Guinea (Ramankutty et al., 2008; Fritz et al., 2011, 2015; Anderson et al., 2015; See et al., 2015). In these regions, land-cover maps are often the only source of land management data. These errors propagate into estimates of cropland harvest flows and harvest intensity, for which much less data are available. Data on crop residues are scarce, as they are not reported in official statistics (e.g.

FAOSTAT, 2015), and estimates usually rely on crude factors (Lal, 2004, 2005; FAO, 2015b)

Effects of crop harvest. A mixed picture emerges with regard to biogeochemical and biophysical effects of crop harvest, but impacts on both dimensions appear to be strong. For instance, the inclusion of crop harvest and residue removal into a dynamic vegetation model significantly increased the amount of historical land-use change based C emissions estimated by the most common agricultural scenarios, which do not include management information (Pugh et al., 2015). Cropland harvest amounted to 3.2 PgC yr<sup>-1</sup> in 2000, around half of total biomass harvest or around 5% of global terrestrial NPP (Wirseniuss, 2003; Krausmann et al., 2008). Primary products (e.g. grains) cover 45%, secondary products (e.g. straw, stover and roots) 46% and 9% are fodder crops. The majority of cropland produce is used directly as food, but a non-negligible amount of around 1.3 PgC yr<sup>-1</sup> is used as feed for livestock (fodder crops and concentrates). In 2004, crop harvest for bioenergy amounted to 1.6 EJ yr<sup>-1</sup> from agricultural by-products and 1.1 EJ yr<sup>-1</sup> from fuel crops, which is roughly equivalent to 0.043 and 0.03 PgC yr<sup>-1</sup>, respectively (Sims et al., 2007). 0.7 PgC yr<sup>-1</sup> of secondary products remain on site, possibly ploughed to the soil or burned subsequently (Wirseniuss, 2003; Krausmann et al., 2008). Cropland systems, mainly consisting of annual, herbaceous plants, usually contain little carbon in vegetation and soil per m<sup>2</sup> (Saugier et al., 2001). Thus, crop residues left on field add only small amounts of carbon to soil pools (Bolinder et al., 2007; Anderson-Teixeira et al., 2012). Information on local impact of crop residue removal (or retention) on GHG emissions, soil carbon and yields is available (Bationo & Mokwunye, 1991; Lal, 2004, 2005; Lehtinen et al., 2014; Pittelkow et al., 2015). Also national data on emissions from crop residues are available (FAOSTAT, 2015). However, the lack of primary data such as from long-term field studies and the use of crude factor introduce large uncertainties related to estimates of crop residue management effects. Large uncertainties also relate to the contribution of crop residue, including roots and exudates, to the build-up of soil organic carbon (Bolinder et al., 2007; Kätterer et al., 2012). This limits our ability to assess its impact at the global scale. With current policies for increasing biomass use for bioenergy, crop residue harvest can result in additional SOC losses, proportional to residue removal (Gollany et al., 2011). Synergistic effects are also frequent: negative effects of crop residue removal on soil carbon are enhanced with N fertilization (Smith et al., 2012).

Biophysical effects of crop harvest are well documented, in particular related to changes in albedo,

roughness and evapotranspiration. When crops are harvested, soil becomes exposed and albedo (Davin et al., 2014) as well as roughness drop (Oke, 1987). Evapotranspiration was estimated to decrease by 23% in a Belgium experiment (Verstraeten et al., 2005). The magnitude and persistence of these changes depend on the presence and intensity of postharvest management practices, for example ploughing, tillage, after cropping or mulching. Evapotranspiration partly depends on soil water holding capacity, which in turn is affected by tillage (Cresswell et al., 1993) and crop residue management (Horton et al., 1996). Crop residue management is an important factor, but information is scarce. Compared to bare soil, crop residues reduce extremes of heat and water fluxes at the soil surface when crops residues are left on-site (Horton et al., 1996; Davin et al., 2014).

### Crop species selection

Extent and data availability. On almost all cropland, single crops form monocultures while other plants are excluded via weeding, herbicides or by other means. Prominent exceptions include agroforestry (i.e. systems where tree species and annual crops are cultivated together, Nair & Garrity, 2012). Crop species selection is as old as agriculture, with species selected according to human needs (e.g. food, health, stimulants, fibre). Recently, biomass energy production from dedicated oil, starch or sugar plants, but also fast-growing grasses, has increased rapidly and is anticipated to accelerate in the future (Beringer et al., 2011; Haberl et al., 2013). Data availability for recent crop type distribution is similar to that on cropland harvest; however, spatially explicit time series and global data on interannual dynamics, such as rotational schemes, are lacking (Table 1; Appendix S1).

Effects of crop species selection. While information on biophysical effects of crop species selection is available, much less is available on biogeochemical effects. Both effects seem to be relatively weak in comparison to other management types, probably also owing to the comparatively small knowledge base. In particular, effects of species selection on individual carbon pools are largely unknown. Crop type is known to affect SOC accumulation and decomposition rates, and the allocation of carbon to shoots or roots. For example, shoot-to-root ratios were found to increase in the order natural grasses < forages < soya bean < corn (Bolinder et al., 2007). A shift from annual to perennial crops and the introduction of cover crops can significantly increase SOC stocks (Poepflau & Don, 2014, 2015). Anderson-Teixeira et al. (2013) found a 400–750% increase in

belowground biomass under perennial bioenergy grasses (switchgrass, *Miscanthus*, native prairie mix) compared to a corn–corn–soya rotation agricultural system. Increasing crop rotational diversity can also positively influence SOC storage (McDaniel et al., 2013; Tiemann et al., 2015). Strong difficulties to assess species selection effects arise from legacy effects, which render systematic long-term studies necessary. For instance, in a 22-year experiment, comparing maize, wheat and soya bean cultivation, SOC content was found to be about 7% higher under soya bean as compared to wheat and maize. Other GHG emissions are also crop specific. For example,  $N_2O$  emission factors from fertilization vary from 0.77% of added nitrogen for rice to 2.76% for maize (Stehfest, 2005). Effects of crop species on  $CH_4$  balances are less clear, except for paddy rice, where high emissions occur.

Cropland albedo varies significantly among crops, ranging between 0.15 for sugarcane and 0.26 for sugar beet, with significant variations even among related species, for example 0.04 higher for wheat compared to barley (Piggin & Schwerdtfeger, 1973; Monteith & Unsworth, 2013). Even within a species, cultivars show differences in albedo of up to 0.03 units. Differences in planting and harvesting dates for different crop species and cultivars, and associated changes in leaf phenology, also affect biophysical conditions. More productive cultivars and earlier planting dates lead, for example, to an earlier harvest and to enhanced exposure of dark soil in the fall, resulting in lower end-of-season albedo and an increase in net radiation (Sacks & Kucharik, 2011). Whether the end-of-season albedo increases or decreases depends on the ratio between soil and vegetation albedo. In many regions of the world, soil albedo is lower than plant albedo, but not in some (semi-)arid regions where soils may have a similar or even higher albedo than the vegetation. Similarly, water-use efficiency and evapotranspiration between crop species differ widely (Yoo et al., 2009), even for the same cultivars (Anda & Løke, 2005). Although crop heights are limited, roughness can be expected to vary similarly as for grasslands (Li et al., 2000).

#### N fertilization of cropland and grazing land

Extent and data availability. Fertilizers are used to enhance plant growth by controlling the level of nutrients in soils. Nitrogen (N) plays a prominent role as one of the most important plant nutrients which is often limited in agriculture (LeBauer & Treseder, 2008). N fertilizers are either organic fertilizer derived from manure (livestock faeces), sewage sludge or mineral fertilizer. Reactive nitrogen was a scarce resource in preindustrial agriculture, mainly available only in the

form of animal manure, leading to sophisticated management schemes to balance the N withdrawals associated with harvest (Sutton et al., 2011). The invention of the Haber–Bosch process and the availability of fossil energy triggered a process of innovation in agriculture with surging levels of N fertilization. Today, the transformation of N to reactive forms and its use as fertilizer on agricultural lands represent one of the most important human-induced environmental changes (Gruber & Galloway, 2008; Davidson, 2009). The use of synthetic fertilizers is projected to increase in response to growing human population, increases in food consumption and crop-based biofuel production (IFA, 2007). Practically all croplands are under N fertilization schemes, with strong regional variations in intensity of input volumes and composition (Gruber & Galloway, 2008; Vitousek et al., 2009), but also grasslands and forests (the latter not discussed here) can be under N fertilization schemes. The highest cropland fertilization levels surpass  $200 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ , for example, in the Nile delta and  $90 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  in New Zealand (Potter et al., 2010; Mueller et al., 2012), and 14% of croplands are fertilized with levels above  $100 \text{ kgN ha}^{-1} \text{ yr}^{-1}$ . Globally, much lower intensity level prevails; 59% of the global cropland area show application rates below  $50 \text{ kgN ha}^{-1} \text{ yr}^{-1}$ , and around one-quarter of global croplands below  $10 \text{ kgN ha}^{-1} \text{ yr}^{-1}$  (Appendix S1). Grasslands often do not receive any N fertilization (except for manure inputs from grazing animals), but some grasslands are also heavily fertilized with rates put to 100 (Haas et al., 2001) and even  $300 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  (Flechar et al., 2007). Globally, animal manure makes up approximately 65% of N inputs to cropland (Potter et al., 2010) and is the dominant N source in the Southern Hemisphere. Regionally, mainly in concentrated industrial livestock production, manure availability can exceed local fertilizer demand, resulting in substantial environmental problems such as groundwater pollution (IAASTD, 2009). The status of data availability is intermediate. National time series data as well as spatially explicit assessments are available (Table 1), but characterized by large gaps and uncertainties, particularly relating to spatial patterns and livestock manure. Global data on N fertilization of grasslands, albeit a widespread activity in many region, are scarce and crude model-derived (Appendix S1).

Effects of N fertilization. The biogeochemical effects of N fertilization, of both cropland and grazing land, are strong and relatively well documented and understood. Cropland fertilization is a strong driver of anthropogenic GHG emissions, in particular of nitrous oxide ( $N_2O$ ), nitric oxide (NO) and ammonia ( $NH_3$ ). A typical fertilized cropland emits 2–3 times more nitrogen than

the approximately  $0.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  emitted under nonfertilized conditions (Stehfest & Bouwman, 2006), while fertilized grasslands emit 3–4 times more  $\text{N}_2\text{O}$  than unfertilized ones (Flechard et al., 2007). The global  $\text{N}_2\text{O}$  emissions on fertilized croplands and grazing lands sum to  $4.1\text{--}5.3 \text{ Tg N yr}^{-1}$  in the beginning of the century (Stehfest & Bouwman, 2006; Syakila & Kroeze, 2011), one-fifth of it occurring on grazing lands (Stehfest & Bouwman, 2006). Beyond N application rates,  $\text{N}_2\text{O}$  emissions are determined by crop type, fertilizer type, soil water content, SOC content, soil pH and texture, soil mineral N content and climate.  $\text{NH}_3$  emissions are determined by fertilizer type, temperature, wind speed, rain and pH (Sommer et al., 2004). Acidification from N fertilizers can lead to increased abiotic  $\text{CO}_2$  emissions from calcareous soils (Matocha et al., 2016). Fertilization also affects ecological processes, including productivity, C inputs to the soil and SOC storage in croplands by affecting the shoot-to-root ratio (Müller et al., 2000), influences the efficiency of photosynthesis, and ultimately the exchange of C between land and the atmosphere, as fertilization studies in forests reveal (Vicca et al., 2012; Fernández-Martínez et al., 2014). Long-term studies from Sweden suggest that each kg N fertilizer increased SOC stocks by 1 to 2 kg (Kätterer et al., 2012). Fertilization effects on SOC were particularly strong with organic fertilization (Kerschens et al., 2013). Fertilization also increases atmospheric N and thus deposition (Ciais et al., 2013a) and results in N leakage (Galloway et al., 2003). Fluxes of total anthropogenic N from land to the ocean via leaching from soils and riverine transport have been estimated at  $40\text{--}70 \text{ Tg N yr}^{-1}$  (Boyer et al., 2006; Fowler et al., 2013). Increased nutrient input to rivers and freshwater systems impacts on water quality and biodiversity (Settele et al., 2014) and the subsequent increased nutrient loading of coastal oceans is believed to be the primary cause of hypoxia (Wong et al., 2014).

Few direct effects of fertilization on biophysical properties – besides indirect effects of changes in crop biomass or height due to altered productivity – have been documented, and the magnitude of impacts is probably not strong. Forest site studies suggest that enhanced leaf nitrogen concentrations increase canopy albedo (Ollinger et al., 2008), presumably through changes in canopy structure rather than in leaf-level albedo (Wicklein et al., 2012). Also, nitrogen fertilization improved grassland water-use efficiency but simultaneously increased absolute evapotranspiration and thus the latent heat flux, from 280 to 310 mm (Brown, 1971; Rose et al., 2012). N-driven increases in plant height and leaf mass will be reflected in increasing roughness length.

## Tillage

Extent and data availability. With the mechanization of agriculture, arable land became regularly tilled to suppress weeds and enhance soil structure and nutrient availability. Archaeological findings suggest that humans manipulated soil structure through some form of tillage with ards and hoes already some 4.5k years ago (Postan et al., 1987). From the 1950s, with the advent of modern herbicides no-till systems became more prominent, mainly in the United States (IAASTD, 2009). To date, continental or global data on the area, distribution or intensity of tillage is sparse. It can be assumed, however, that all croplands that are permanently used are regularly tilled, except for (i) perennial crops, which cover approximately 10% of cropland area or  $1.5 \text{ Mkm}^2$  (FAOSTAT, 2015) and (ii) no-till agriculture (or reduced tillage) on  $1.11 \text{ million km}^2$  (Derpsch et al., 2010), which is around 8% of the global arable land. No-tillage systems are particularly widespread in Brazil and the United States, where 70% and 30%, respectively, of the total cultivated area is under no-tillage management. However, most of these lands are not permanently under zero tillage but are still ploughed from time to time. Global maps of zero tillage are missing, as do maps on qualitative aspects of tillage, such as type and depth of tillage.

Effects of tillage. Tillage effects remain weakly understood. Ploughing of native grassland upon conversion to croplands drastically depleted SOC (Mann, 1986). Such ploughing disrupts aggregate structure, aerating the soil and activating microbial decomposition (Rovira & Greacen, 1957). No-tillage practices promised to significantly mitigate carbon emissions from SOC (IAASTD, 2009). However, some evidence is available indicating that on most soil types and in most climate regimes adoption of no-tillage practices after tillage-based management does not significantly increase SOC stocks (Baker et al., 2007; Hermle et al., 2008; Govaerts et al., 2009), but there is still controversy on this aspect of the adaption of no-tillage (Powlson et al., 2014, 2015; Neufeldt et al., 2015). These findings and studies looking deeper into the soil profile suggest that conventional tillage may not result in net losses of soil C, but rather result in a redistribution of carbon in the soil profile. Other findings are inconclusive, for example, on the impacts of conservation tillage on productivity of cropland. While no-tillage is often reducing crop yields, other activities such as crop residue management of crop rotations play a decisive role for the overall effects (Pittelkow et al., 2015). Other key factors are the depth and type of tillage, which vary worldwide.

Evidence on the effects of no-tillage on N<sub>2</sub>O emissions is site specific and inconclusive (Rochette, 2008). A recent meta-analysis reported that no-till reduced N<sub>2</sub>O emissions after 10 years of adoption and when fertilizer was added below the soil surface, especially in humid climates (van Kessel et al., 2013). No-tillage generally reduces soil erosion, but regional- to global-scale effects are uncertain, because most eroded soil carbon is deposited in nearby ecosystems (Van Oost et al., 2007).

Tillage has small biophysical effects. Through a decreased soil water holding capacity, excess tillage increased the shortwave albedo from 0.12 under minimum tillage to 0.15 under excess tillage (Cresswell et al., 1993). Furthermore, soil water holding capacity, which is affected by tillage (Cresswell et al., 1993) and crop residue management (Horton et al., 1996), also controls evapotranspiration. Soils covered with crop residues after harvest evaporate less than tilled soils (Horton et al., 1996) and show a higher albedo (Davin et al., 2014). When only part of the site is tilled, the effects become less straightforward. Strip-tillage, leaving three-fourths of the surface covered, can increase evapotranspiration within the tilled strips while maintaining the same soil temperature compared to a bare site (Hares & Novak, 1992), thus providing protection against wind and water erosion without affecting seed germination (Hares & Novak, 1992). The direct effects of tillage on surface roughness are likely negligible for the surface climate.

## Irrigation

Extent and data availability. Globally 2.3–4.0 Mkm<sup>2</sup> or 15–26% of the global croplands are equipped for irrigation (Portmann et al., 2010; Salmon et al., 2015), with hotspots in the Near East, Northern Africa, Central, South and South-East Asia and western North America. Paddy rice, the largest single crop species cultivated with irrigation, covers 0.7–1.0 Mkm<sup>2</sup> (Portmann et al., 2010; Salmon et al., 2015), or 5–7% of the global cropland area. Paddy rice cultivation is particularly important in East, South and South-East Asia where its history reaches back at least 6k years, originating probably in China (Cao et al., 2006; Fuller, 2012; Kalbitz et al., 2013). Small-scale crop irrigation dates back to the origins of agriculture (Postel, 2001), while large-scale irrigation is a recent outcome of the green revolution. Nowadays, 30% of the global wheat fields (0.7 Mkm<sup>2</sup>), 20% of the maize fields (0.3 Mkm<sup>2</sup>) and half of the global citrus, sugar cane and cotton crops are irrigated (Portmann et al., 2010). Moreover, cropland irrigation accounts for approximately 70% of global freshwater consumption (Wisser et al., 2008). Rice cultivation requires a particularly intensive form of irrigation,

involving regular flooding of fields for longer periods (Salmon et al., 2015). Irrigation data sets exist and are relatively robust, in particular for rice, but large similar problems of uncertainties prevail as with cropland maps (see above; Salmon et al., 2015). Furthermore, Earth system effects depend on actually applied irrigation, which is much less documented than area equipped for irrigation.

Effects of cropland irrigation. Strong biogeochemical and biophysical effects of irrigation are documented. Knowledge gaps exist related to synergistic effects with other management practices. Irrigation significantly enhances NPP where water is limiting plant growth, in particular in semi-arid and arid regions. Irrigation affects soil moisture, temperature and N availability, which are all drivers for the production and evolution of GHG emissions from soils (Dobbie et al., 1999; Dobbie & Smith, 2003). Accelerated soil carbon decomposition under irrigation is typically offset by higher NPP and greater carbon inputs into the soil (Liebig et al., 2005; Smith et al., 2008). A global review of irrigation effects concluded that irrigated cropping systems in arid and semi-arid regions typically realize SOC increases of 11% to 35% compared to nonirrigated systems, but the size of the effect is highly dependent on climate and initial SOC content (Liebig et al., 2005; Trost et al., 2013). Furthermore, irrigated soils are more often affected by anoxic soil conditions which in turn favour denitrification and N<sub>2</sub>O production, especially when fertilized (Verma et al., 2006). This is particularly the case in paddy fields, where emission factors range between 341 and 993 g N ha<sup>-1</sup>, depending on the length of the irrigation scheme, corresponding to irrigation-induced emission factors of 0.22–0.37% of the added nitrogen (Akiyama et al., 2005). Soil texture and climate can mediate these effects of irrigation on biogeochemical processes, but the statistical evidence is weak (Scheer et al., 2012; Trost et al., 2013; Jamali et al., 2015). According to the review by Trost et al. (2013), there is no consistent effect of irrigation on N<sub>2</sub>O emissions. The capacity of soils to oxidize atmospheric CH<sub>4</sub> may be reduced under irrigation (Ellert & Janzen, 1999; Sainju et al., 2012). Irrigated rice fields alone are emitting approximately 30–40 Tg CH<sub>4</sub> per year (Kirschke et al., 2013).

Changes in ecosystem water availability significantly alter the surface albedo and roughness through their impact on plant growth and ecosystem conditions (Cresswell et al., 1993; Wang & Davidson, 2007). Because water surfaces have lower reflectance, flooding reduces the albedo of dry soil of about 0.2 to a level of 0.03 – 0.1 (Kozłowski, 1984). A modelling study over the Great Plains in the USA has shown that irrigation

can alter atmospheric circulation and precipitation patterns (Huber et al., 2014). Despite its surface cooling effect (about 0.8 K), irrigation was simulated to increase global radiative forcing in the range of 0.03 to 0.1 W m<sup>-2</sup> (Boucher et al., 2004).

#### Artificial drainage of wetlands

Extent and data availability. Drainage aims at improving soil characteristics for agriculture and at facilitating the use of machinery. While historically drainage relied on channels and sewers, currently prevailing drainage systems often also use subsurface hollow pipes or similar technologies (FAO, 1985). Approximately 11% of global croplands, or 1.6 Mkm<sup>2</sup>, are subject to artificial drainage (Feick et al., 2005), but the strongest biogeochemical and biophysical effects of drainage are expected when wetlands are drained, for example peatlands, inland flood plains, coastal wetlands or lakes. Wetlands are estimated to cover 5.3–26.9 Mkm<sup>2</sup> (Melton et al., 2013), of which 0.18 Mkm<sup>2</sup> are probably drained (Appendix S1), but data are scarce. Wetland drainage dates back for millennia, for example, in lowland Europe (Emanuelsson, 2009), but accelerated especially between 1830 and 1950 with the drainage of over 30% of the Scandinavian peatlands and large-scale drainage projects in Russia, Canada and the United States (Brinson & Malvárez, 2002). Despite attempts for wetland conservation (see, e.g., Dugan, 1990), or the international RAMSAR treaty ([www.ramsar.org](http://www.ramsar.org)), large-scale new drainage installation is still ongoing (Brinson & Malvárez, 2002; Laitinen et al., 2009), in particular in Asia, for instance in relation to palm oil expansion (Davidson, 2014). Consistent data on wetland drainage are practically inexistent.

Effects of wetland drainage. The biogeochemical and biophysical effects of drainage are not well documented, partly because most studies aim at assessing the effects of associated land-use and land-cover changes, rather than the effects of drainage itself. While the sparse evidence suggests that biogeochemical effects are strong, biophysical effects are probably only of medium size. On forest sites, drainage can increase biomass through increased NPP (Trettin & Jurgensen, 2003). Drained peatlands are, however, hotspots of GHG emissions (Hiraishi et al., 2014). When expressed in units of radiative forcing, the soil emissions of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O in drained forested peatlands decrease or even offset the carbon sink in aboveground biomass (Schils et al., 2008). The cultivation of drained wetlands leads to rapid losses of large stocks of soil carbon accumulated over thousands of years (Dressler et al., 2013). A 50%

increase in fluvial carbon losses (particulate and dissolved organic carbon) was observed from degraded tropical swamp forest (Moore et al., 2013). Drainage-related increases in fluvial carbon loss may add up to approximately 10% of the South-East Asian land-use emissions (Abrams et al., 2016). Drainage increases vulnerability to surface fires by drying the top soil. Drainage and fire associated with oil palm and other plantations in Indonesia, for example, released an amount of CO<sub>2</sub> equal to 19–60% of the global carbon emissions from fossil fuels between 1997 and 2006 (Jaenicke et al., 2008).

The biophysical effects of drainage are also poorly documented. Regional model simulations in Finland, where drainage allowed for the afforestation of treeless peatlands, suggested early season warming of 0.2 to 0.43 K and late season cooling (Gao et al., 2014). Drainage decreases evapotranspiration (Lafleur et al., 2005) which in turn results in lower minimum night-time temperatures (Marshall et al., 2003). The relationship between evapotranspiration and night-time temperatures has been modelled (Venäläinen et al., 1999; Marshall et al., 2003), suggesting considerable temperature drops of up to 10 K. Although the direct effect of drainage on albedo and roughness length is not clear, increasing plant growth is likely to increase the surface roughness and decrease springtime albedo (Lohila et al., 2010).

#### Fire management

Extent and data availability. Fire began to be used by humans around 50k to 100k years ago (James, 1989; Bar-Yosef, 2002), and while it is unclear when it was first employed to shape ecosystems, today is a versatile land management tool (Lauk & Erb, 2009; Bowman et al., 2011), for example, for plant selection or agricultural waste removal. Note that fire use for land clearing, including swidden agriculture, represents a land-cover change and is thus not discussed here. Fire occurs naturally in most ecosystems, while in many regions natural fires today are suppressed (Hurt et al., 2002; Andela & van der Werf, 2014), population density playing an important role (Archibald et al., 2009). Yet, prescribed fires are, next to mechanical thinning, a widespread practice to reduce or retard wildfire spread and intensity (Fernandes & Botelho, 2003). As fire frequency is expected to increase in the future due to climate change, fire prevention might increase in importance. Globally, the annual area burned through human-induced and natural fires is estimated at 3.0–5.1 Mkm<sup>2</sup> in the last decades (Wiedinmyer et al., 2011; Giglio et al., 2013). The proportion of human-induced fires is difficult to assess (van der Werf et al., 2008), and

in particular the ratio between fires that lead to land-cover change and fires used to manage ecosystems is unknown. No specific global, spatially explicit information on fire as a management tool (including fire prevention and prescribed fires) exists (Table 1).

Effects of fire management. The effects of fire management on biogeochemical and biophysical properties of ecosystems are well documented and mainly biogeochemical. However, these studies do not systematically separate natural from anthropogenic fires. Globally, fire-induced carbon emissions are estimated to range from 1.6 to 2.8 PgC yr<sup>-1</sup> (van der Werf et al., 2010), while human-induced fires range from 1.7 to 2.0 PgC yr<sup>-1</sup> (Lauk & Erb, 2009). The large uncertainties owe to large differences in the assumptions of fuel loads (Granier et al., 2011) and the difficulty to assess smaller fires. Fire emissions also include aerosols and trace gases (Akagi et al., 2011), which impact atmospheric chemistry and significantly contribute to overall aerosol direct and indirect radiative forcing (Ward et al., 2012). Fires result in short-term carbon losses from the direct combustion of biomass and lagged losses from the decomposition of dead biomass (Hurteau & Brooks, 2011). Fires affect nutrient supply (Mahowald et al., 2005) and soil carbon dynamics (Knicker, 2007). The storage of carbon in long-lived pools such as SOC is influenced by fires through the accumulation of char or pyrogenic carbon (Santín et al., 2008). Repeated burning in the process of agricultural land management (e.g. residue burning) reduces carbon accumulation rates (Zarin et al., 2005). The effects of fire suppression (Archibald et al., 2009; Wang et al., 2010) or management activities that indirectly alter fire regimes (van Wilgen et al., 2014), however, represent a knowledge gap. Despite the direct carbon stock increases resulting from fire prevention and similar measures (Bond-Lamberty et al., 2007), such activities can lead to greater future ecosystem carbon losses through the accumulation of large fuel loads that potentially increase the risk of severe fires (Hurteau & Brooks, 2011; O'Connor et al., 2014). Indirect biogeochemical effects of fire, for example postfire degradation, are not systematically quantified.

Various observational studies scrutinized the effects of specific fires on surface energy fluxes. Immediately after a boreal forest fire, albedo decreased to 0.05, increasing to 0.12 over a period of 30 years and then averaging to 0.08 similar to a prefire state (Amiro et al., 2006). Effects of fire aerosols might also be important, although uncertainty is high (Landry et al., 2015). Also latent heat energy fluxes and overall radiative forcing are affected (Randerson et al., 2006). Randerson et al. (2006) estimated a radiative forcing of  $-5 \text{ W m}^{-2}$

immediately after a boreal forest fire, which remained high at  $-4 \text{ W m}^{-2}$  over 80 years after the fire. In a savannah, a halving of the albedo (0.12–0.07) was observed, followed by a recovery to a prefire state after several weeks (Scholes & Walker, 1993; Beringer et al., 2003).

## Discussion and conclusions

The ten land management practices selected for this review affect a considerable proportion of the global terrestrial surface (Fig. 2). Grazing and forest harvest and tree species selection are largest in terms of extent, covering almost 60% of the terrestrial, ice-free global land surface. However, the importance of a management practice depends not only on its spatial extent and effects on the Earth system, but also on the intensity of management, which differs markedly in extent across management practice (Fig. 2). Management intensity has shown pronounced increases at the global scale in recent decades, yet is currently largely overlooked (Rounsevell et al., 2012; Erb et al., 2013a; Luysaert et al., 2014). According to our review, around 10% of the ice-free land surface is under intense human management, half of it under medium and one-fifth under extensive management (Appendix S1; Fig. 2).

The level of understanding of management effects on biogeochemical and biophysical patterns and processes varies strongly between management activities. Some of the direct impacts of activities such as wood harvest and tree species selection, grazing, N fertilization, irrigation and crop harvest are well documented. Considerable uncertainty of knowledge prevails for crop species selection, artificial wetland drainage, tillage, crop residue management and fire as management tool. Furthermore, how these processes vary across heterogeneous soils, how they affect plant diversity or how they depend on climate conditions are questions that have not been rigorously explored. Here, continuing efforts are needed to systematically combine local ground observations with assessments at coarser spatial and temporal scales along with model implementation. These efforts require increased information exchange between research communities in land system science, Earth system modelling, and experiment-based ecological and agronomic research.

Despite these knowledge gaps, some insights in the relative weight of biogeochemical and biophysical impacts of individual management activities emerged from our review. For instance, while grazing is associated with strong biogeochemical, but relatively small biophysical effects, tree species selection is characterized by strong biophysical, but limited biogeochemical effects. In contrast, forest harvest is important in both

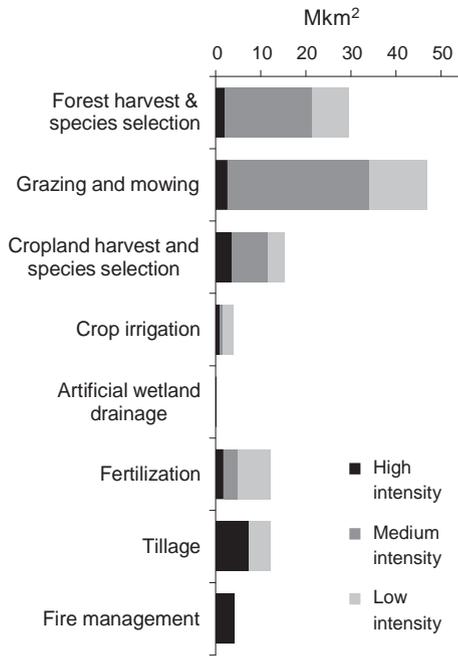


Fig. 2 Global extent and intensity of land management activities. Globally, approximately 80% of the 130 Mkm<sup>2</sup> of ice-free land is under managed schemes at varying intensity. Note that the bars are not additive, as, for example, crop irrigation, fertilization and tillage all occur on cropland. For data and assumptions, see Appendix S1.

respects (Fig. 3). Similarly, strong biophysical as well as biogeochemical effects originate from irrigation, cropland harvest and wetland drainage, although affecting much smaller areas. Other agricultural activities, such as fertilization, tillage, residue management, are associated mainly with biogeochemical impacts. Crop species selection, in contrast, ranks low with regard to biogeochemical and biophysical effects. But, as most land management activities are not isolated from each other, but intricately linked (e.g. crop harvest, irrigation and fertilization), robust assessment on their relative significance requires the application of Earth system models and, as our review reveals, improved databases.

Our review focused on documented Earth system effects of land management that have occurred over the past decades. Yet land management plays an increasing role in discussions on mitigating future climate change (Foley et al., 2005). This makes it particularly important to consider that management effects act on a range of timescales: while changes in land surface properties impose immediate effects on the atmosphere, changes in carbon and nitrogen fluxes invokes counter-fluxes in the coupled land–atmosphere–ocean system, causing a distinct temporal evolution and a delayed response of the Earth system (Ciais et al., 2013b). The emergence of

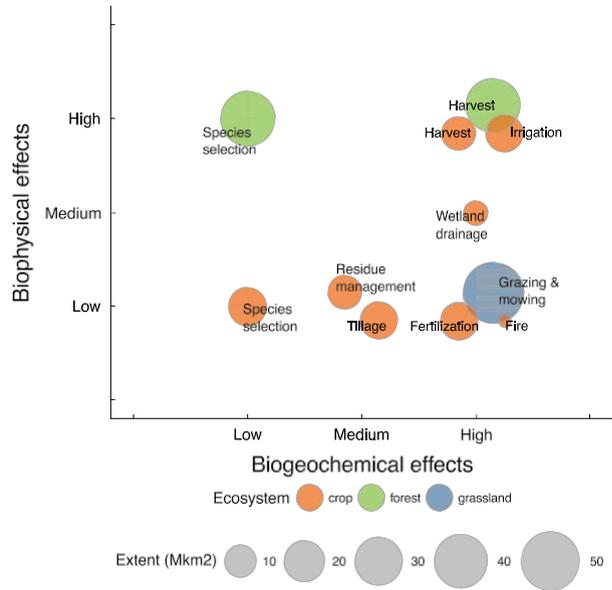


Fig. 3 Extent and biogeochemical and biophysical effects of management activities. The classification (see Appendix S1) is based on expert judgement and hence contains a certain degree of subjectivity and ambiguity.

biogeochemical effects can also typically include longer timescales than that of biogeophysical effects, as they can alter slow-responding system components such as SOC. While biogeophysical effects and greenhouse gas fluxes due to management are persistent once the new management system is in equilibrium, changes in carbon stocks cease to cause fluxes over time. Assessment of a land-use activity in the mitigation context thus depends not just on the spatial scale, with fluxes of the well-mixed greenhouse gases causing a global signal, while biogeophysical effects act predominantly on the local scale, but crucially also on an integrated assessment of the various effects and their different timescales in relation to the time horizon of interest (Cherubini et al., 2012).

A mixed picture emerges regarding data availability and robustness of global, long-term land management information (Table 1). This is a consequence of the history of research and past investments in generating the data sets. Remote sensing, while particularly well suited to assess certain land uses at the global level (e.g. cropping, irrigation, or the outbreak of fires), encounters severe difficulties in depicting other uses such as grazing (Erb et al., 2007; Kuemmerle et al., 2013). Furthermore, statistical reporting schemes focus mainly on management activities of economic interest, such as crop and forest harvest and ignore others, for example crop residue management. In addition, inconsistent definitions affect data robustness (FAOSTAT, 2015; See et al., 2015).

While a comprehensive assessment of Earth system impacts induced by management requires more data and ultimately their integration in a modelling environment, as well as the inclusion of other management activities not discussed here, we conclude that management is a key factor in the Earth system, severely influencing many biogeochemical and biophysical processes and parameters. We also conclude that the current status of process understanding and data availability is sufficient to advance with the integration of land management in Earth system models in order to assess their overall impacts. Hence, we are able to classify the ten land management activities into groups along the two dimensions data availability and process understanding (Table 2), and thus identify the most pressing research priorities.

A first group is characterized by relatively advanced data availability and process understanding. This group contains irrigation and cropland harvest. For these activities, the state of knowledge is sufficient for implementing these activities in integrative assessment environments such as Earth system models.

The second group is characterized by severe data gaps, but relatively advanced process understanding. This includes wood harvest, tree species selection, grazing and N fertilization, motivating calls for fostered research efforts from the global land-use data community (e.g. Verburg et al., 2016) to develop improved data sets, for example, by taking advantage of the increasingly available data from satellite observations (Kuemerle et al., 2013; Joshi et al., 2016), or crowdsourcing (See et al., 2015), but also alternative approaches that exploit existing databases. These management activities could be included in Earth system models, but global parameterization and validation may be difficult for now. A third group is characterized by concomitant data and knowledge gaps. The management types in

this group require an intensification of efforts of both the data and the ecological communities, in order to advance the understanding of the impact of these management practices on the Earth system. No activity was classified as a combination ‘advanced data’ and ‘poor understanding’.

Advancing the current state of process understanding and data availability on land management is a central undertaking to improve the understanding of land-use induced impacts on the Earth system and their feedbacks in the coupled socio-ecological system, central for, for example, the recently published sustainability development goals (Costanza et al., 2016). In addition to enhancing data availability and process understanding, data access, usability and quality control will become essential for transferring these achievements into beneficial information across multiple disciplines to tackle the grand sustainability challenges relate to land management.

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## References

- Abrams JF, Hohn S, Rixen T, Baum A, Merico A (2016) The impact of Indonesian peatland degradation on downstream marine ecosystems and the global carbon cycle. *Global Change Biology*, 22, 325–337.
- Adams SN (1975) Sheep and cattle grazing in forests: a review. *The Journal of Applied Ecology*, 12, 143.
- Akagi SK, Yokelson RJ, Wiedinmyer C et al. (2011) Emission factors for open and domestic biomass burning for use in atmospheric models. *Atmospheric Chemistry and Physics*, 11, 4039–4072.
- Akiyama H, Yagi K, Yan X (2005) Direct N<sub>2</sub>O emissions from rice paddy fields: summary of available data. *Global Biogeochemical Cycles*, 19, GB1005.
- Alexandratos N, Bruinsma J (2012) World agriculture towards 2030/2050: the 2012 revision, Vol. 3.
- Allard V, Soussana J-F, Falcimagne R et al. (2007) The role of grazing management for the net biome productivity and greenhouse gas budget (CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub>) of semi-natural grassland. *Agriculture, Ecosystems and Environment*, 121, 47–58.
- Alonso-Canas I, Chuvieco E (2015) Global burned area mapping from ENVISAT-MERIS and MODIS active fire data. *Remote Sensing of Environment*, 163, 140–152.

Table 2 Classification of management activities according to current process understanding and data availability

	Data advanced	Data poor
Understanding advanced	<ul style="list-style-type: none"> <li>• Crop harvest</li> <li>• Irrigation</li> </ul>	<ul style="list-style-type: none"> <li>• Forestry harvest</li> <li>• Tree species selection</li> <li>• Grazing and mowing harvest</li> <li>• N fertilization</li> </ul>
Understanding poor	...	<ul style="list-style-type: none"> <li>• Crop species selection</li> <li>• Artificial wetland drainage</li> <li>• Tillage</li> <li>• Fire management</li> <li>• Crop residue management*</li> </ul>

\*Separated here from crop harvest.

- Amiro BD, Orchansky AL, Barr AG et al. (2006) The effect of post-fire stand age on the boreal forest energy balance. *Agricultural and Forest Meteorology*, 140, 41–50.
- Anadón JD, Sala OE, Turner BL, Bennett EM (2014) Effect of woody-plant encroachment on livestock production in North and South America. *Proceedings of the National Academy of Sciences*, 111, 12984–12953.
- Anda A, Løke Z (2005) Radiation balance components of maize hybrids grown at various plant densities. *Journal of Agronomy and Crop Science*, 191, 202–209.
- Andela N, van der Werf GR (2014) Recent trends in African fires driven by cropland expansion and El Niño to La Niña transition. *Nature Climate Change*, 4, 791–795.
- Anderson W, You L, Wood S, Wood-Sichra U, Wu W (2015) An analysis of methodological and spatial differences in global cropping systems models and maps. *Global Ecology and Biogeography*, 24, 180–191.
- Anderson-Teixeira KJ, Snyder PK, Twine TE, Cuadra SV, Costa MH, DeLucia EH (2012) Climate-regulation services of natural and agricultural ecoregions of the Americas. *Nature Climate Change*, 2, 177–181.
- Anderson-Teixeira KJ, Masters MD, Black CK, Zeri M, Hussain MZ, Bernacchi CJ, DeLucia EH (2013) Altered belowground carbon cycling following land-use change to perennial bioenergy crops. *Ecosystems*, 16, 508–520.
- Archibald S, Roy DP, Van WILGENBW, Scholes RJ (2009) What limits fire? An examination of drivers of burnt area in Southern Africa. *Global Change Biology*, 15, 613–630.
- Arnell A, Mercado L, Kattge J, Booth BBB (2012) Future challenges of representing land-processes in studies on land-atmosphere interactions. *Biogeosciences*, 9, 3587–3599.
- Asner GP, Elmore AJ, Olander LP, Martin RE, Harris AT (2004) Grazing systems, ecosystem responses, and global change. *Annual Review of Environment and Resources*, 29, 261–299.
- Bai ZG, Dent DL, Olsson L, Schaepman ME (2008) Proxy global assessment of land degradation. *Soil Use and Management*, 24, 223–234.
- Bais ALS, Lauk C, Kastner T, Erb K (2015) Global patterns and trends of wood harvest and use between 1990 and 2010. *Ecological Economics*, 119, 326–337.
- Baker JM, Ochsner TE, Venterea RT, Griffis TJ (2007) Tillage and soil carbon sequestration—What do we really know? *Agriculture, Ecosystems and Environment*, 118, 1–5.
- Bárceña TG, Kiær LP, Vesterdal L, Stefánsdóttir HM, Gundersen P, Sigurdsson BD (2014) Soil carbon stock change following afforestation in Northern Europe: a meta-analysis. *Global Change Biology*, 20, 2393–2405.
- Baron VS, Mapfumo E, Dick AC, Naeth MA, Okine EK, Chanasyk DS (2002) Grazing intensity impacts on pasture carbon and nitrogen flow. *Journal of Range Management*, 55, 535–541.
- Bar-Yosef O (2002) The upper paleolithic revolution. *Annual Review of Anthropology*, 31, 363–393.
- Bationo A, Mokwunye AU (1991) Role of manures and crop residue in alleviating soil fertility constraints to crop production: with special reference to the Sahelian and Sudanian zones of West Africa. In: *Alleviating Soil Fertility Constraints to Increased Crop Production in West Africa* (ed. Mokwunye AU), pp. 217–225. Springer, Netherlands.
- Bengtsson J, Nilsson SG, Franc A, Menozzi P (2000) Biodiversity, disturbances, ecosystem function and management of European forests. *Forest Ecology and Management*, 132, 39–50.
- Beringer J, Hutley LB, Tapper NJ, Coutts A, Kerley A, O'Grady AP (2003) Fire impacts on surface heat, moisture and carbon fluxes from a tropical savanna in northern Australia. *International Journal of Wildland Fire*, 12, 333–340.
- Beringer T, Lucht W, Schaphoff S (2011) Bioenergy production potential of global biomass plantations under environmental and agricultural constraints. *GCB Bioenergy*, 3, 299–312.
- Berthrong ST, Jobbágy EG, Jackson RB (2009) A global meta-analysis of soil exchangeable cations, pH, carbon, and nitrogen with afforestation. *Ecological Applications: A Publication of the Ecological Society of America*, 19, 2228–2241.
- Birdsey R, Pan Y (2015) Trends in management of the world's forests and impacts on carbon stocks. *Forest Ecology and Management*, 355, 83–90.
- Blondel J (2006) The 'Design' of Mediterranean landscapes: a millennial story of humans and ecological systems during the historic period. *Human Ecology*, 34, 713–729.
- Bolinder MA, Janzen HH, Gregorich EG, Angers DA, VandenBygaart AJ (2007) An approach for estimating net primary productivity and annual carbon inputs to soil for common agricultural crops in Canada. *Agriculture, Ecosystems and Environment*, 118, 29–42.
- Bond-Lamberty B, Peckham SD, Ahl DE, Gower ST (2007) Fire as the dominant driver of central Canadian boreal forest carbon balance. *Nature*, 450, 89–92.
- Boserup E (1965) *The Conditions of Agricultural Growth: The Economics of Agrarian Change under Population Pressure*, vol. 4. Earthscan, London.
- Boucher O, Myhre G, Myhre A (2004) Direct human influence of irrigation on atmospheric water vapour and climate. *Climate Dynamics*, 22, 597–603.
- Bouwman AF, Van der Hoek KW, Eickhout B, Soenari I (2005) Exploring changes in world ruminant production systems. *Agricultural Systems*, 84, 121–153.
- Bowman DMJS, Balch J, Artaxo P et al. (2011) The human dimension of fire regimes on Earth. *Journal of Biogeography*, 38, 2223–2236.
- Boyd IL, Freer-Smith PH, Gilligan CA, Godfray HCJ (2013) The consequence of tree pests and diseases for ecosystem services. *Science*, 342, 1235773.
- Boyer EW, Howarth RW, Galloway JN, Dentener FJ, Green PA, Vörsmarty CJ (2006) Riverine nitrogen export from the continents to the coasts. *Global Biogeochemical Cycles*, 20, GB1591.
- Bridges EM, Oldeman LR (1999) Global assessment of human-induced soil degradation. *Arid Soil Research and Rehabilitation*, 13, 319–325.
- Brinson MM, Malvárez AI (2002) Temperate freshwater wetlands: types, status, and threats. *Environmental Conservation*, 29, 115–133.
- Brovkin V, Sitch S, Von Bloh W, Claussen M, Bauer E, Cramer W (2004) Role of land cover changes for atmospheric CO<sub>2</sub> increase and climate change during the last 150 years. *Global Change Biology*, 10, 1253–1266.
- Brown PL (1971) Water use and soil water depletion by dryland winter wheat as affected by nitrogen fertilization. *Agronomy Journal*, 63, 43.
- Brus DJ, Hengeveld GM, Walvoort DJJ, Goedhart PW, Heidema AH, Nabuurs GJ, Gunia K (2011) Statistical mapping of tree species over Europe. *European Journal of Forest Research*, 131, 145–157.
- Campioli M, Vicca S, Luysaert S et al. (2015) Biomass production efficiency controlled by management in temperate and boreal ecosystems. *Nature Geoscience*, 8, 843–846.
- Cao ZH, Ding JL, Hu ZY et al. (2006) Ancient paddy soils from the Neolithic age in China's Yangtze River Delta. *Naturwissenschaften*, 93, 232–236.
- Chang J, Ciaia P, Viovy N, Vuichard N, Sultan B, Soussana J-F (2015) The greenhouse gas balance of European grasslands. *Global Change Biology*, 21, 3748–3761.
- Cherubini F, Bright RM, Strömman AH (2012) Site-specific global warming potentials of biogenic CO<sub>2</sub> for bioenergy: contributions from carbon fluxes and albedo dynamics. *Environmental Research Letters*, 7, 45902.
- Ciaia P, Gasser T, Paris JD et al. (2013a) Attributing the increase in atmospheric CO<sub>2</sub> to emitters and absorbers. *Nature Climate Change*, 3, 926–930.
- Ciaia P, Sabine C, Bala G et al. (2013b) Carbon and other biogeochemical cycles. In: *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, (eds Stocker TF, Qin D, Plattner G-K, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM), pp. 465–570. Cambridge University Press, Cambridge, UK.
- Cohen J, Pulliainen J, Ménard CB, Johansen B, Oksanen L, Luojus K, Ikonen J (2013) Effect of reindeer grazing on snowmelt, albedo and energy balance based on satellite data analyses. *Remote Sensing of Environment*, 135, 107–117.
- Costanza R, Fioramonti L, Kubiszewski I (2016) The UN sustainable development goals and the dynamics of well-being. *Frontiers in Ecology and the Environment*, 14, 59.
- Cresswell HP, Painter DJ, Cameron KC (1993) Tillage and water content effects on surface soil hydraulic properties and shortwave Albedo. *Soil Science Society of America Journal*, 57, 816.
- Davidson EA (2009) The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860. *Nature Geoscience*, 2, 659–662.
- Davidson NC (2014) How much wetland has the world lost? Long-term and recent trends in global wetland area. *Marine and Freshwater Research*, 65, 934.
- Davin EL, Seneviratne SI, Ciaia P, Olliso A, Wang T (2014) Preferential cooling of hot extremes from cropland albedo management. *Proceedings of the National Academy of Sciences*, 111, 9757–9761.
- Derpsch R, Friedrich T, Kassam A, Li H (2010) Current status of adoption of no-till farming in the world and some of its main benefits. *International Journal of Agricultural and Biological Engineering*, 3, 1–25.
- Dlaz S, Lavorel S, McIntyre S et al. (2007) Plant trait responses to grazing – a global synthesis. *Global Change Biology*, 13, 313–341.
- Dobbie KE, Smith KA (2003) Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water-filled pore space and other controlling variables. *Global Change Biology*, 9, 204–218.
- Dobbie KE, McTaggart IP, Smith KA (1999) Nitrous oxide emissions from intensive agricultural systems: variations between crops and seasons, key driving variables, and mean emission factors. *Journal of Geophysical Research: Atmospheres*, 104, 26891–26899.

- Drešler M, Verchot LV, Freibauer A et al. (2013) Drained inland organic soils. In: 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands (eds Hiraishi T, Krug T, Tanabe K, Srivastava N, Baasansuren J, Fukuda M, Troxler T), pp. 2.1–2.79. Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, UK.
- Dugan PJ (1990) Wetland conservation: A review of current issues and required action. IUCN.
- Dunne J, Evershed RP, Salque M et al. (2012) First dairying in green Saharan Africa in the fifth millennium BC. *Nature*, 486, 390–394.
- Eastman JL, Coughenour MB, Pielke RA (2001) Does grazing affect regional climate? *Journal of Hydrometeorology*, 2, 243–253.
- Eggers J, Lindner M, Zudin S, Zaehle S, Liski J (2008) Impact of changing wood demand, climate and land use on European forest resources and carbon stocks during the 21st century. *Global Change Biology*, 14, 2288–2303.
- Eldridge DJ, Bowker MA, Maestre FT, Roger E, Reynolds JF, Whitford WG (2011) Impacts of shrub encroachment on ecosystem structure and functioning: towards a global synthesis. *Ecology Letters*, 14, 709–722.
- Ellert BH, Janzen HH (1999) Short-term influence of tillage on CO<sub>2</sub> fluxes from a semi-arid soil on the Canadian Prairies. *Soil and Tillage Research*, 50, 21–32.
- Ellis EC, Kaplan JO, Fuller DQ, Vavrus S, Goldewijk KK, Verburg PH (2013) Used planet: a global history. *Proceedings of the National Academy of Sciences*, 110, 7978–7985.
- Emanuelsson U (2009) The Rural Landscapes of Europe: How Man has Shaped European Nature. Swedish Research Council Formas, Stockholm.
- Erb K-H (2012) How a socio-ecological metabolism approach can help to advance our understanding of changes in land-use intensity. *Ecological Economics*, 76, 8–14.
- Erb K-H, Gaube V, Krausmann F, Plutzer C, Bondeau A, Haberl H (2007) A comprehensive global 5 min resolution land-use data set for the year 2000 consistent with national census data. *Journal of Land Use Science*, 2, 191–224.
- Erb K-H, Haberl H, Jepsen MR et al. (2013a) A conceptual framework for analysing and measuring land-use intensity. *Current Opinion in Environmental Sustainability*, 5, 464–470.
- Erb K-H, Kastner T, Luysaert S, Houghton RA, Kuemmerle T, Olofsson P, Haberl H (2013b) Bias in the attribution of forest carbon sinks. *Nature Climate Change*, 3, 854–856.
- Erb K-H, Fetzel T, Plutzer C et al. (2016) Biomass turnover time in terrestrial ecosystems halved by land use. *Nature Geoscience*, (in press). doi: 10.1038/ngeo2782
- Fan L, Kutzer B, Liu H, Bernhofer C (2010) Grazing effects on seasonal dynamics and interannual variabilities of spectral reflectance in semi-arid grassland in Inner Mongolia. *Plant and Soil*, 340, 169–180.
- FAO (1985) Irrigation Water Management: Training Manual No. 1 - Introduction to Irrigation.
- FAO (2010) Global Forest Resources Assessment 2010. Main Report. FAO, Rome, 378 pp.
- FAO (2011) World Livestock 2011. Livestock in food security. Food and Agriculture Organization of the United Nations.
- FAO (2015a) Global Forest Resources Assessments 2015. Food and Agriculture Organization of the United Nations, Rome.
- FAO (2015b) Methods and Standards.
- FAOSTAT (2015) Statistical Databases. Available at: <http://faostat.fao.org> (accessed 11 February 2016).
- Farley KA, Jobbágy EG, Jackson RB (2005) Effects of afforestation on water yield: a global synthesis with implications for policy. *Global Change Biology*, 11, 1565–1576.
- Farrell EP, Furrer E, Ryan D, Andersson F, Hüttl R, Piussi P (2000) European forest ecosystems: building the future on the legacy of the past. *Forest Ecology and Management*, 132, 5–20.
- Feddema JJ, Oleson KW, Bonan GB, Mearns LO, Buja LE, Meehl GA, Washington WM (2005) The importance of land-cover change in simulating future climates. *Science*, 310, 1674–1678.
- Feick S, Siebert S, Döll P (2005) A Digital Global Map of Artificially Drained Agricultural Areas.
- Fernandes PM, Botelho HS (2003) A review of prescribed burning effectiveness in fire hazard reduction. *International Journal of Wildland Fire*, 12, 117–128.
- Fernández-Martínez M, Vicca S, Janssens IA et al. (2014) Nutrient availability as the key regulator of global forest carbon balance. *Nature Climate Change*, 4, 471–476.
- Flechard CR, Ambus P, Skiba U et al. (2007) Effects of climate and management intensity on nitrous oxide emissions in grassland systems across Europe. *Agriculture, Ecosystems and Environment*, 121, 135–152.
- Foley JA, DeFries R, Asner GP et al. (2005) Global consequences of land use. *Science*, 309, 570.
- Foley JA, Ramankutty N, Brauman KA et al. (2011) Solutions for a cultivated planet. *Nature*, 478, 337–342.
- Ford CR, Laseter SH, Swank WT, Vose JM (2011) Can forest management be used to sustain water-based ecosystem services in the face of climate change? *Ecological Applications*, 21, 2049–2067.
- Forkel M, Migliavacca M, Thonicke K, Reichstein M, Schaphoff S, Weber U, Carvalhais N (2015) Codominant water control on global interannual variability and trends in land surface phenology and greenness. *Global Change Biology*, 21, 3414–3435.
- Fowler D, Coyle M, Skiba U et al. (2013) The global nitrogen cycle in the twenty-first century. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 368, 20130164.
- Freydank K, Siebert S (2008) Towards mapping the extent of irrigation in the last century: time series of irrigated area per country. Research report. Institute of Physical Geography, University of Frankfurt (Main), 46 pp.
- Fritz S, See L, McCallum I et al. (2011) Highlighting continued uncertainty in global land cover maps for the user community. *Environmental Research Letters*, 6, 44005.
- Fritz S, See L, McCallum I et al. (2015) Mapping global cropland and field size. *Global Change Biology*, 21, 1980–1992.
- Frolking S, Yeluripati JB, Douglas E (2006) New district-level maps of rice cropping in India: a foundation for scientific input into policy assessment. *Field Crops Research*, 98, 164–177.
- Fuller DQ (2012) Pathways to Asian civilizations: tracing the origins and spread of rice and rice cultures. *Rice*, 4, 78–92.
- Gallardo A, Schlesinger WH (1992) Carbon and nitrogen limitations of soil microbial biomass in desert ecosystems. *Biogeochemistry*, 18, 1–17.
- Galloway JN, Aber JD, Erisman JW, Seitzinger SP, Howarth RW, Cowling EB, Cosby BJ (2003) The nitrogen cascade. *BioScience*, 53, 341–356.
- Gao Y, Markkanen T, Backman L, Henttonen HM, Pietikäinen J-P, Mäkelä HM, Laaksonen A (2014) Biogeophysical impacts of peatland forestation on regional climate changes in Finland. *Biogeosciences*, 11, 7251–7267.
- Giglio L, Randerson JT, van der Werf GR (2013) Analysis of daily, monthly, and annual burned area using the fourth-generation global fire emissions database (GFED4). *Journal of Geophysical Research: Biogeosciences*, 118, 317–328.
- Gollany HT, Rickman RW, Liang Y, Albrecht SL, Machado S, Kang S (2011) Predicting agricultural management influence on long-term soil organic carbon dynamics: implications for biofuel production. *Agronomy Journal*, 103, 234.
- Govaerts B, Verhulst N, Castellanos-Navarrete A, Sayre KD, Dixon J, Dendooven L (2009) Conservation agriculture and soil carbon sequestration: between myth and farmer reality. *Critical Reviews in Plant Sciences*, 28, 97–122.
- Granic C, Bessagnet B, Bond T et al. (2011) Evolution of anthropogenic and biomass burning emissions of air pollutants at global and regional scales during the 1980–2010 period. *Climatic Change*, 109, 163–190.
- Gruber N, Galloway JN (2008) An Earth-system perspective of the global nitrogen cycle. *Nature*, 451, 293–296.
- Haas G, Wetterich F, Köpke U (2001) Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. *Agriculture, Ecosystems and Environment*, 83, 43–53.
- Haberl H, Erb KH, Krausmann F et al. (2007) Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. *Proceedings of the National Academy of Sciences*, 104, 12942–12947.
- Haberl H, Erb K-H, Krausmann F, Running S, Searchinger TD, Smith WK (2013) Bioenergy: how much can we expect for 2050? *Environmental Research Letters*, 8, 31004.
- Hammerle A, Haslwanter A, Tappeiner U, Cernusca A, Wohlfahrt G (2008) Leaf area controls on energy partitioning of a temperate mountain grassland. *Biogeosciences*, 5, 421–431.
- Hanewinkel M, Cullmann DA, Schelhaas M-J, Nabuurs G-J, Zimmermann NE (2013) Climate change may cause severe loss in the economic value of European forest land. *Nature Climate Change*, 3, 203–207.
- Hares MA, Novak MD (1992) Simulation of surface energy balance and soil temperature under strip tillage: II. Field test. *Soil Science Society of America Journal*, 56, 29.
- Harmon ME, Ferrell WK, Franklin JF (1990) Effects on carbon storage of conversion of old-growth forests to young forests. *Science*, 247, 699–702.
- Hengeveld GM, Nabuurs G-J, Didion M, van den Wyngaert I, Clerckx APPM, Schelhaas M-J (2012) A forest management map of European forests. *Ecology and Society*, 17, 53.
- Hermle S, Anken T, Leifeld J, Weisskopf P (2008) The effect of the tillage system on soil organic carbon content under moist, cold-temperate conditions. *Soil and Tillage Research*, 98, 94–105.

- Herrero M, Havlik P, Valin H et al. (2013) Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proceedings of the National Academy of Sciences*, 110, 20888–20893.
- Hiraiishi T, Krug T, Tanabe K, Srivastava N, Jamsranjav B, Fukuda M, Troxler T (2014) 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands.
- Hollinger DY, Ollinger SV, Richardson AD et al. (2010) Albedo estimates for land surface models and support for a new paradigm based on foliage nitrogen concentration. *Global Change Biology*, 16, 696–710.
- Holtmark B (2012) The outcome is in the assumptions: analyzing the effects on atmospheric CO<sub>2</sub> levels of increased use of bioenergy from forest biomass. *GCB Bioenergy*, 5, 467–473.
- Horton R, Bristow KL, Kluitenberg GJ, Sauer TJ (1996) Crop residue effects on surface radiation and energy balance — review. *Theoretical and Applied Climatology*, 54, 27–37.
- Hosonuma N, Herold M, De Sy V et al. (2012) An assessment of deforestation and forest degradation drivers in developing countries. *Environmental Research Letters*, 7, 44009.
- Huber DB, Mechem DB, Brunsell NA (2014) The effects of great plains irrigation on the surface energy balance, regional circulation, and precipitation. *Climate*, 2, 103–128.
- Hulvey KB, Hobbs RJ, Standish RJ, Lindenmayer DB, Lach L, Perring MP (2013) Benefits of tree mixes in carbon plantings. *Nature Climate Change*, 3, 869–874.
- Hurteau MD, Brooks ML (2011) Short- and long-term effects of fire on carbon in US dry temperate forest systems. *BioScience*, 61, 139–146.
- Hurtt GC, Pacala SW, Moorcroft PR, Caspersen J, Shevliakova E, Houghton RA, Moore B (2002) Projecting the future of the U.S. carbon sink. *Proceedings of the National Academy of Sciences*, 99, 1389–1394.
- Hurtt G, Chini L, Froliking S et al. (2011) Harmonization of land-use scenarios for the period 1500–2100: 600 years of global gridded annual land-use transitions, wood harvest, and resulting secondary lands. *Climatic Change*, 109, 117–161.
- IAASTD (2009) Agriculture at a Crossroad. International Assessment of Agricultural Knowledge, Science and Technology for Development. Global Report. Island Press, Washington, D.C.
- IFA (2007) Sustainable management of the nitrogen cycle in agriculture and mitigation of reactive nitrogen side effects. International Fertilizer Industry Association, Paris.
- Izumi T, Ramankutty N (2016) Changes in yield variability of major crops for 1981–2010 explained by climate change. *Environmental Research Letters*, 11, 34003.
- Izumi T, Yokozawa M, Sakurai G et al. (2014) Historical changes in global yields: major cereal and legume crops from 1982 to 2006. *Global Ecology and Biogeography*, 23, 346–357.
- Jaenicke J, Rieley JO, Mott C, Kimman P, Siebert F (2008) Determination of the amount of carbon stored in Indonesian peatlands. *Geoderma*, 147, 151–158.
- Jamali H, Quayle WC, Baldock J (2015) Reducing nitrous oxide emissions and nitrogen leaching losses from irrigated arable cropping in Australia through optimized irrigation scheduling. *Agricultural and Forest Meteorology*, 208, 32–39.
- James SR (1989) Hominid use of fire in the lower and middle pleistocene. *Current Anthropology*, 30, 1–26.
- Joshi N, Baumann M, Ehammer A et al. (2016) A review of the application of optical and radar remote sensing data fusion to land use mapping and monitoring. *Remote Sensing*, 8, 70.
- Juang J-Y, Katul G, Siqueira M, Stoy P, Novick K (2007) Separating the effects of albedo from eco-physiological changes on surface temperature along a successional chronosequence in the southeastern United States. *Geophysical Research Letters*, 34, L21408.
- Kalbitz K, Kaiser K, Fiedler S et al. (2013) The carbon count of 2000 years of rice cultivation. *Global Change Biology*, 19, 1107–1113.
- Kaplan JO, Krumhardt KM, Ellis EC, Ruddiman WF, Lemmen C, Goldewijk KK (2010) Holocene carbon emissions as a result of anthropogenic land cover change. *Holocene*, 21, 775–791.
- Kastner T, Rivas MJI, Koch W, Nonhebel S (2012) Global changes in diets and the consequences for land requirements for food. *Proceedings of the National Academy of Sciences*, 109, 6868–6872.
- Kätterer T, Bolinder MA, Berglund K, Kirchmann H (2012) Strategies for carbon sequestration in agricultural soils in northern Europe. *Acta Agriculturae Scandinavica, Section A—Animal Science*, 62, 181–198.
- van Kessel C, Venterea R, Six J, Adviento-Borbe MA, Linquist B, van Groenigen KJ (2013) Climate, duration, and N placement determine N<sub>2</sub>O emissions in reduced tillage systems: a meta-analysis. *Global Change Biology*, 19, 33–44.
- Kesselmeier J, Staudt M (1999) Biogenic Volatile Organic Compounds (VOC): an overview on emission, physiology and ecology. *Journal of Atmospheric Chemistry*, 33, 23–88.
- Kirschbaum MUF, Whitehead D, Dean SM, Beets PN, Shepherd JD, Ausseil A-GE (2011) Implications of albedo changes following afforestation on the benefits of forests as carbon sinks. *Biogeosciences Discussions*, 8, 8563–8589.
- Kirschke S, Bousquet P, Ciais P et al. (2013) Three decades of global methane sources and sinks. *Nature Geoscience*, 6, 813–823.
- Kissinger G, Herold M, de Sy V (2012) Drivers of deforestation and forest degradation: A synthesis report for REDD+ policymakers. Center for International Forestry Research.
- Knicker H (2007) How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. *Biogeochemistry*, 85, 91–118.
- Körner M, Albert E, Armbruster M et al. (2013) Effect of mineral and organic fertilization on crop yield, nitrogen uptake, carbon and nitrogen balances, as well as soil organic carbon content and dynamics: results from 20 European long-term field experiments of the twenty-first century. *Archives of Agronomy and Soil Science*, 59, 1017–1040.
- Kowalski S, Sartore M, Buret R, Berbigier P, Loustau D (2003) The annual carbon budget of a French pine forest (*Pinus pinaster*) following harvest. *Global Change Biology*, 9, 1051–1065.
- Kozlowski TT (1984) Flooding and Plant Growth. Academic Press, Orlando.
- Krausmann F, Erb K-H, Gingrich S, Lauk C, Haberl H (2008) Global patterns of socioeconomic biomass flows in the year 2000: a comprehensive assessment of supply, consumption and constraints. *Ecological Economics*, 65, 471–487.
- Krausmann F, Erb K-H, Gingrich S et al. (2013) Global human appropriation of net primary production doubled in the 20th century. *Proceedings of the National Academy of Sciences*, 110, 10324–10329.
- Kuemmerle T, Erb K, Meyfroidt P et al. (2013) Challenges and opportunities in mapping land use intensity globally. *Current Opinion in Environmental Sustainability*, 5, 484–493.
- Lafleur PM, Hember RA, Admiral SW, Roulet NT (2005) Annual and seasonal variability in evapotranspiration and water table at a shrub-covered bog in southern Ontario, Canada. *Hydrological Processes*, 19, 3533–3550.
- Lehtonen O, Ruokolainen K, Schulman L, Oinonen M (2009) Amazonian peatlands: an ignored C sink and potential source. *Global Change Biology*, 15, 2311–2320.
- Lal R (2004) Soil carbon sequestration impacts on global climate change and food security. *Science*, 304, 1623–1627.
- Lal R (2005) World crop residues production and implications of its use as a biofuel. *Environment International*, 31, 575–584.
- Landry J-S, Matthews HD, Ramankutty N (2015) A global assessment of the carbon cycle and temperature responses to major changes in future fire regime. *Climatic Change*, 133, 179–192.
- Lassey KR (2007) Livestock methane emission: from the individual grazing animal through national inventories to the global methane cycle. *Agricultural and Forest Meteorology*, 142, 120–132.
- Lauk C, Erb K-H (2009) Biomass consumed in anthropogenic vegetation fires: global patterns and processes. *Ecological Economics*, 69, 301–309.
- LeBauer DS, Treseder KK (2008) Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. *Ecology*, 89, 371–379.
- Lehtinen T, Schlatter N, Baumgarten A et al. (2014) Effect of crop residue incorporation on soil organic carbon and greenhouse gas emissions in European agricultural soils. *Soil Use and Management*, 30, 524–538.
- Leyers C, Verkerk PJ, Meier D et al. (2014) Drivers of forest harvesting intensity patterns in Europe. *Forest Ecology and Management*, 315, 160–172.
- Li S-G, Harazono Y, Oikawa T, Zhao HL, Ying He Z, Chang XL (2000) Grassland desertification by grazing and the resulting micrometeorological changes in Inner Mongolia. *Agricultural and Forest Meteorology*, 102, 125–137.
- Liebig MA, Morgan JA, Reeder JD, Ellert BH, Gollany HT, Schuman GE (2005) Greenhouse gas contributions and mitigation potential of agricultural practices in northwestern USA and western Canada. *Soil and Tillage Research*, 83, 25–52.
- Lindenmayer D, Cunningham S, Young A (2012) Land Use Intensification: Effects on Agriculture, Biodiversity and Ecological Processes. CSIRO Publishing, Collingwood, Australia.
- Lioussé C, Guillaume B, Grégoire JM et al. (2010) Updated African biomass burning emission inventories in the framework of the AMMA-IDAF program, with an evaluation of combustion aerosols. *Atmospheric Chemistry and Physics*, 10, 9631–9646.
- Lipke B, Oneil E, Harrison R, Skog K, Gustavsson L, Sathre R (2011) Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. *Carbon*, 2, 303–333.

- Liu J, You L, Amini M, Obersteiner M, Herrero M, Zehnder AJB, Yang H (2010) A high-resolution assessment on global nitrogen flows in cropland. *Proceedings of the National Academy of Sciences*, 107, 8035–8040.
- Lohila A, Minkkinen K, Laine J et al. (2010) Forestation of boreal peatlands: impacts of changing albedo and greenhouse gas fluxes on radiative forcing. *Journal of Geophysical Research: Biogeosciences*, 115, G04011.
- Luo J, Tillman RW, Ball PR (1999) Grazing effects on denitrification in a soil under pasture during two contrasting seasons. *Soil Biology and Biochemistry*, 31, 903–912.
- Luyssaert S, Ciais P, Piao SL et al. (2010) The European carbon balance. Part 3: forests. *Global Change Biology*, 16, 1429–1450.
- Luyssaert S, Hoesung M, de la Torre J, Geronzi M, Obersteiner M, Schulze ED (2011) Quantifying land-use and disturbance intensity in forestry, based on the self-thinning relationship. *Ecological Applications*, 8, 3272–3284.
- Luyssaert S, Jammot M, Stoy PC et al. (2014) Land management and land-cover change have impacts of similar magnitude on surface temperature. *Nature Climate Change*, 4, 389–393.
- Mahowald NM, Artaxo P, Baker AR, Jickells TD, Okin GS, Randerson JT, Townsend AR (2005) Impacts of biomass burning emissions and land use change on Amazonian atmospheric phosphorus cycling and deposition. *Global Biogeochemical Cycles*, 19, GB4030.
- Mann LK (1986) Changes in soil carbon storage after cultivation. *Soil Science*, 142, 279–288.
- Mörwald E, Langston N, Stens A, Moen J (2016) Changing ideas in forestry: a comparison of concepts in Swedish and American forestry journals during the early twentieth and twenty-first centuries. *Ambio*, 45, 74–86.
- Marland G, Schlamadinger B (1997) Forests for carbon sequestration or fossil fuel substitution? A sensitivity analysis. *Biomass and Bioenergy*, 13, 389–397.
- Marshall CH, Pielke RA, Steyaert LT (2003) Wetlands: crop freezes and land-use change in Florida. *Nature*, 426, 29–30.
- Matocha CJ, Grove JH, Karathanasis TD, Vandiviere M (2016) Changes in soil mineralogy due to nitrogen fertilization in an agroecosystem. *Geoderma*, 263, 176–184.
- McDaniel MD, Tiemann LK, Grandy AS (2013) Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. *Ecological Applications*, 24, 560–570.
- McGarvey JC, Thompson JR, Epstein HE, Shugart HH (2014) Carbon storage in old-growth forests of the Mid-Atlantic: toward better understanding the eastern forest carbon sink. *Ecology*, 96, 311–317.
- McGrath MJ, Luyssaert S, Meyfroidt P et al. (2015) Reconstructing European forest management from 1600 to 2010. *Biogeosciences Discussions*, 12, 5365–5433.
- McSherry ME, Ritchie ME (2013) Effects of grazing on grassland soil carbon: a global review. *Global Change Biology*, 19, 1347–1357.
- Melton JR, Wania R, Hodson EL et al. (2013) Present state of global wetland extent and wetland methane modelling: conclusions from a model inter-comparison project (WETCHIMP). *Biogeosciences*, 10, 753–788.
- Meyfroidt P, Lambin EF (2011) Global forest transition: prospects for an end to deforestation. *Annual Review of Environment and Resources*, 36, 343–371.
- Miller SD, Goulden ML, da Rocha HR (2007) The effect of canopy gaps on subcanopy ventilation and scalar fluxes in a tropical forest. *Agricultural and Forest Meteorology*, 142, 25–34.
- Monfreda C, Ramankutty N, Foley JA (2008) Farming the planet: 2. Geographic distribution of crop areas, yields, physiological types, and net primary production in the year 2000. *Global Biogeochemical Cycles*, 22, 1–19.
- Monteith J, Unsworth M (2013) *Principles of Environmental Physics: Plants, Animals, and the Atmosphere*. Academic Press, Oxford, UK.
- Moore S, Evans CD, Page SE et al. (2013) Deep instability of deforested tropical peatlands revealed by fluvial organic carbon fluxes. *Nature*, 493, 660–663.
- Mueller ND, Gerber JS, Johnston M, Ray DK, Ramankutty N, Foley JA (2012) Closing yield gaps through nutrient and water management. *Nature*, 490, 254–257.
- Mueller ND, Butler EE, McKinnon KA, Rhines A, Tingley M, Holbrook NM, Huybers P (2015) Cooling of US Midwest summer temperature extremes from cropland intensification. *Nature Climate Change*, 6, 317–322.
- Müller I, Schmid B, Weiner J (2000) The effect of nutrient availability on biomass allocation patterns in 27 species of herbaceous plants. *Perspectives in Plant Ecology, Evolution and Systematics*, 3, 115–127.
- Nair PKR, Garrity D (eds.) (2012) *Agroforestry - The Future of Global Land Use*, vol. 9. Springer, Dordrecht, Netherlands.
- Nakai T, Sumida A, Daikoku K et al. (2008) Parameterisation of aerodynamic roughness over boreal, cool- and warm-temperate forests. *Agricultural and Forest Meteorology*, 148, 1916–1925.
- Naudts K, Chen Y, McGrath MJ, Ryder J, Valade A, Otto J, Luyssaert S (2016) Europe's forest management did not mitigate climate warming. *Science*, 351, 597–600.
- Naylor R, Steinfeld H, Falcon W et al. (2005) Losing the links between livestock and land. *Science*, 310, 1621–1622.
- Neufeldt H, Kissing G, Alcamo J (2015) No-till agriculture and climate change mitigation. *Nature Climate Change*, 5, 488–489.
- Niedertscheider M, Kastner T, Fetzel T, Haberl H, Kroisleitner C, Plutzer C, Erb K-H (2016) Mapping and analysing cropland use intensity from a NPP perspective. *Environmental Research Letters*, 11, 14008.
- O'Connor CD, Falk DA, Lynch AM, Swetnam TW (2014) Fire severity, size, and climate associations diverge from historical precedent along an ecological gradient in the Pinaleno Mountains, Arizona, USA. *Forest Ecology and Management*, 329, 264–278.
- Oke TR (1987) *Boundary Layer Climates*, 2nd edn. Routledge, London.
- Ollinger SV, Richardson AD, Martin ME et al. (2008) Canopy nitrogen, carbon assimilation, and albedo in temperate and boreal forests: functional relations and potential climate feedbacks. *Proceedings of the National Academy of Sciences*, 105, 19336–19341.
- Oswalt SN, Smith WB, Miles PD, Pugh SA (2014) *Forest Resources of the United States, 2012: a technical document supporting the Forest Service 2015 update of the RPA Assessment*. Gen. Tech. Rep. WO-91. U.S. Department of Agriculture Forest Service, Washington Office, Washington, DC, 218 pp.
- Otto J, Berveiller D, Bréon F-M et al. (2013) Summertime canopy albedo is sensitive to forest thinning. *Biogeosciences Discussions*, 10, 15373–15414.
- Otto J, Berveiller D, Bréon F-M et al. (2014) Forest summer albedo is sensitive to species and thinning: how should we account for this in Earth system models? *Biogeosciences*, 11, 2411–2427.
- Pan Y, Birdsey RA, Fang J et al. (2011) A large and persistent carbon sink in the world's forests. *Science*, 333, 988–993.
- Pan Y, Birdsey RA, Phillips OL, Jackson RB (2013) The structure, distribution, and biomass of the world's forests. *Annual Review of Ecology, Evolution, and Systematics*, 44, 593–622.
- Paul KL, Polglase PJ, Nyakuengama JG, Khanna PK (2002) Change in soil carbon following afforestation. *Forest Ecology and Management*, 168, 241–257.
- Penuelas J, Llusià J (2003) BVOCs: plant defense against climate warming? *Trends in Plant Science*, 8, 105–109.
- Perlin J (2005) *A Forest Journey: The Story of Wood and Civilization*. The Countryman Press, Woodstock, Vermont.
- Petz K, Alkemade R, Bakkenes M, Schulp CJE, van der Velde M, Leemans R (2014) Mapping and modelling trade-offs and synergies between grazing intensity and ecosystem services in rangelands using global-scale datasets and models. *Global Environmental Change*, 29, 223–234.
- Piggin I, Schwerdtfeger DP (1973) Variations in the albedo of wheat and barley crops. *Archiv für Meteorologie, Geophysik und Bioklimatologie, Serie B*, 21, 365–391.
- Pittelkow CM, Liang X, Linquist BA et al. (2015) Productivity limits and potentials of the principles of conservation agriculture. *Nature*, 517, 365–368.
- Poelau C, Don A (2014) Soil carbon changes under *Miscanthus* driven by C4 accumulation and C3 decomposition – toward a default sequestration function. *GCB Bioenergy*, 6, 327–338.
- Poelau C, Don A (2015) Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis. *Agriculture, Ecosystems and Environment*, 200, 33–41.
- Pongratz J, Reick C, Raddatz T, Claussen M (2008) A reconstruction of global agricultural areas and land cover for the last millennium. *Global Biogeochemical Cycles*, 22, GB3018.
- Pongratz J, Reick C, Raddatz T, Claussen M (2009) Effects of anthropogenic land cover change on the carbon cycle of the last millennium. *Global Biogeochemical Cycles*, 23, GB4001.
- Portmann FT, Siebert S, Döll P (2010) MIRCA2000–Global monthly irrigated and rainfed crop areas around the year 2000: a new high-resolution data set for agricultural and hydrological modeling. *Global Biogeochemical Cycles*, 24, GB1011.
- Postan MM, Miller E, Postan C (1987) *The Cambridge Economic History of Europe from the Decline of the Roman Empire: Volume 2, Trade and Industry in the Middle Ages*, 2nd edn. Cambridge University Press, Cambridge, UK.
- Potapov P, Yaroshenko A, Turubanova S et al. (2008) Mapping the world's intact forest landscapes by remote sensing. *Ecology and Society*, 13, 51.
- Postel S (2001) Growing more food with less water. *Scientific American*, 284, 46–50.
- Potter P, Ramankutty N, Bennett EM, Donner SD (2010) Characterizing the spatial patterns of global fertilizer application and manure production. *Earth Interactions*, 14, 1–22.
- Powlson DS, Stirling CM, Jat ML, Gerard BG, Palm CA, Sanchez PA, Cassman KG (2014) Limited potential of no-till agriculture for climate change mitigation. *Nature Climate Change*, 4, 678–683.

- Powelson DS, Stirling CM, Jat ML, Gerard BG, Palm CA, Sanchez PA, Cassman KG (2015) Reply to 'No-till agriculture and climate change mitigation'. *Nature Climate Change*, 5, 489.
- Pugh TAM, Arneth A, Olin S et al. (2015) Simulated carbon emissions from land-use change are substantially enhanced by accounting for agricultural management. *Environmental Research Letters*, 10, 124008.
- Ramankutty N, Evan AT, Monfreda C, Foley JA (2008) Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. *Global Biogeochemical Cycles*, 22, GB1003.
- Randerson JT, Liu H, Flanner MG et al. (2006) The impact of boreal forest fire on climate warming. *Science*, 314, 1130–1132.
- Raupach MR (1994) Simplified expressions for vegetation roughness length and zero-plane displacement as functions of canopy height and area index. *Boundary-Layer Meteorology*, 71, 211–216.
- Rautiainen M, Stenberg P, Mottus M, Manninen T (2011) Radiative transfer simulations link boreal forest structure and shortwave albedo. *Boreal Environment Research*, 16, 91–100.
- Ray DK, Foley JA (2013) Increasing global crop harvest frequency: recent trends and future directions. *Environmental Research Letters*, 8, 44041.
- Ray DK, Ramankutty N, Mueller ND, West PC, Foley JA (2012) Recent patterns of crop yield growth and stagnation. *Nature Communications*, 3, 1293.
- Reid WV, Chen D, Goldfarb L et al. (2010) Earth system science for global sustainability: grand challenges. *Science*, 330, 916–917.
- Resh SC, Binkley D, Parrotta JA (2002) Greater soil carbon sequestration under nitrogen-fixing trees compared with eucalyptus species. *Ecosystems*, 5, 217–231.
- Robertson GP, Paul EA, Harwood RR (2000) Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. *Science*, 289, 1922–1925.
- Rochette P (2008) No-till only increases N<sub>2</sub>O emissions in poorly-aerated soils. *Soil and Tillage Research*, 101, 97–100.
- Rose L, Coners H, Leuschner C (2012) Effects of fertilization and cutting frequency on the water balance of a temperate grassland. *Ecology*, 93, 64–72.
- Rosset M, Riedo M, Grub A, Geissmann M, Fuhrer J (1997) Seasonal variation in radiation and energy balances of permanent pastures at different altitudes. *Agricultural and Forest Meteorology*, 86, 245–258.
- Rosset M, Montani M, Tanner M, Fuhrer J (2001) Effects of abandonment on the energy balance and evapotranspiration of wet subalpine grassland. *Agriculture, Ecosystems and Environment*, 86, 277–286.
- Rounsevell MDA, Pedrolí B, Erb K-H et al. (2012) Challenges for land system science. *Land Use Policy*, 29, 899–910.
- Rovira A, Greacen E (1957) The effect of aggregate disruption on the activity of microorganisms in the soil. *Australian Journal of Agricultural Research*, 8, 659–673.
- Sacks WJ, Kucharik CJ (2011) Crop management and phenology trends in the U.S. Corn Belt: impacts on yields, evapotranspiration and energy balance. *Agricultural and Forest Meteorology*, 151, 882–894.
- Saggar S, Giltrap DL, Davison R et al. (2015) Estimating direct N<sub>2</sub>O emissions from sheep, beef, and deer grazed pastures in New Zealand hill country: accounting for the effect of land slope on the N<sub>2</sub>O emission factors from urine and dung. *Agriculture, Ecosystems and Environment*, 205, 70–78.
- Sainju UM, Stevens WB, Caesar-TonThat T, Liebig MA (2012) Soil greenhouse gas emissions affected by irrigation, tillage, crop rotation, and nitrogen fertilization. *Journal of Environment Quality*, 41, 1774.
- Salmon JM, Friedl MA, Frohling S, Wissler D, Douglas EM (2015) Global rain-fed, irrigated, and paddy croplands: a new high resolution map derived from remote sensing, crop inventories and climate data. *International Journal of Applied Earth Observation and Geoinformation*, 38, 321–334.
- Sanderson EW, Jaiteh M, Levy MA, Redford KH, Wannebo AV, Woolmer G (2002) The human footprint and the last of the wild. *BioScience*, 52, 891–904.
- Santín C, Knicker H, Fernández S, Menéndez-Duarte R, Álvarez MÁ (2008) Wildfires influence on soil organic matter in an Atlantic mountainous region (NW of Spain). *Catena*, 74, 286–295.
- Saugier B, Roy J, Mooney HA (2001) Estimations of global terrestrial productivity: converging toward a single number? In: *Terrestrial Global Productivity* (eds Roy J, Saugier B, Mooney HA), pp. 543–557. Academic Press, San Diego.
- Scheer C, Grace PR, Rowlings DW, Payer J (2012) Soil N<sub>2</sub>O and CO<sub>2</sub> emissions from cotton in Australia under varying irrigation management. *Nutrient Cycling in Agroecosystems*, 95, 43–56.
- Schils R, Kuikman P, Liski J et al. (2008) Review of existing information on the interrelations between soil and climate change. (ClimSoil). Final report. European Commission, Brussels, Belgium, 208 pp.
- Scholes RJ, Walker BH (1993) *An African Savanna: Synthesis of the Nylsvley Study*. Cambridge University Press, Cambridge, UK.
- Schulze E-D, Körner C, Law BE, Haberl H, Luysaert S (2012) Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral. *GCB Bioenergy*, 4, 611–616.
- See L, Fritz S, You L et al. (2015) Improved global cropland data as an essential ingredient for food security. *Global Food Security*, 4, 37–45.
- Settele J, Scholes R, Betts R et al. (2014) Terrestrial and inland water systems. In: *Climate Change 2014: Impacts, Adaptations, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (eds Field C, Barros VR, Dokken DJ, Mach KJ, Mastrandrea MD, Bilir TE, Chatterjee M, Ebi KL, Estrada YO, Genova RC, Girma B, Kissel ES, Levy AN, MacCracken S, Mastrandrea PR, White LL), pp. 271–359. Cambridge University Press, Cambridge, UK.
- Siebert S, Portmann FT, Dell P (2010) Global patterns of cropland use intensity. *Remote Sensing*, 2, 1625–1643.
- Siebert S, Kummu M, Porkka M, Dell P, Ramankutty N, Scanlon BR (2015) A global data set of the extent of irrigated land from 1900 to 2005. *Hydrology and Earth System Sciences*, 19, 1521–1545.
- Sims REH, Schock RN, Adegbulugbe A et al. (2007) Energy supply. In: *Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, (eds Metz B, Davidson OR, Bosch PR, Dave R, Meyer LA), pp. 251–322. Cambridge University Press, Cambridge, UK.
- Smith P, Martino D, Cai Z et al. (2008) Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363, 789–813.
- Smith WN, Grant BB, Campbell CA, McConkey BG, Desjardins RL, Kröbel R, Malhi SS (2012) Crop residue removal effects on soil carbon: measured and inter-model comparisons. *Agriculture, Ecosystems and Environment*, 161, 27–38.
- Sommer SG, Schjoerring JK, Denmead OT (2004) Ammonia emission from mineral fertilizers and fertilized crops. *Advances in Agronomy*, 82, 557–622.
- Steffen W, Richardson K, Rockström J et al. (2015) Planetary boundaries: guiding human development on a changing planet. *Science*, 347, 1259855.
- Stehfest E (2005) Modelling of global crop production and resulting N<sub>2</sub>O emissions. Dissertation. University Kassel, Kassel.
- Stehfest E, Bouwman L (2006) N<sub>2</sub>O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. *Nutrient Cycling in Agroecosystems*, 74, 207–228.
- Steinfeld H, Gerber P, Wassenaar T, Castel V, deHaan C (2006) *Livestock's long shadow: environmental issues and options*. FAO.
- Stocks BJ, Mason JA, Todd JB et al. (2002) Large forest fires in Canada, 1959–1997. *Journal of Geophysical Research: Atmospheres*, 107, 8149.
- Stoy PC, Katul GG, Siqueira MBS et al. (2006) Separating the effects of climate and vegetation on evapotranspiration along a successional chronosequence in the southeastern US. *Global Change Biology*, 12, 2115–2135.
- Sutton MA, Howard CM, Erisman JW et al. (2011) *The European Nitrogen Assessment: Sources, Effects and Policy Perspectives*. Cambridge University Press, Cambridge, UK.
- Syakila A, Kroeze C (2011) The global nitrous oxide budget revisited. *Greenhouse Gas Measurement and Management*, 1, 17–26.
- Thornton PK, Herrero M (2010) Potential for reduced methane and carbon dioxide emissions from livestock and pasture management in the tropics. *Proceedings of the National Academy of Sciences*, 107, 19667–19672.
- Tiemann LK, Grandy AS, Atkinson EE, Marin-Spiotta E, McDaniel MD (2015) Crop rotational diversity enhances belowground communities and functions in an agroecosystem. *Ecology Letters*, 18, 761–771.
- Tilman D, Clark M (2014) Global diets link environmental sustainability and human health. *Nature*, 515, 518–522.
- Trettin CC, Jurgensen MF (2003) Carbon cycling in wetland forest soils. In: *The Potential of U.S. Forest Soils to Sequester Carbon and Mitigate the Greenhouse Effect* (eds Kimble JM, Heath LS, Birdsey RA, Lal R), pp. 311–331. Lewis Publishers, Boca Raton, FL.
- Trost B, Prochnow A, Drastig K, Meyer-Aurich A, Ellmer F, Baumecker M (2013) Irrigation, soil organic carbon and N<sub>2</sub>O emissions. A review. *Agronomy for Sustainable Development*, 33, 733–749.
- Vasunen M, Ylianne H, Kaarlejärvi E, Sjögren S, Olofsson J, Crout N, Stark S (2014) Consequences of warming on tundra carbon balance determined by reindeer grazing history. *Nature Climate Change*, 4, 384–388.
- Van Oost K, Quine TA, Govers G et al. (2007) The impact of agricultural soil erosion on the global carbon cycle. *Science*, 318, 626–629.

- Venäläinen A, Rontu L, Solantie R (1999) On the influence of peatland draining on local climate. *Boreal Environment Research*, 4, 89–100.
- Verburg PH, Crossman N, Ellis EC et al. (2016) Land system science and sustainable development of the earth system: a global land project perspective. *Anthropocene*, 12, 29–41.
- Verkerk PJ, Levers C, Kuemmerle T, Lindner M, Valbuena R, Verburg PH, Zudin S (2015) Mapping wood production in European forests. *Forest Ecology and Management*, 357, 228–238.
- Verma A, Tyagi L, Yadav S, Singh SN (2006) Temporal changes in N<sub>2</sub>O efflux from cropped and fallow agricultural fields. *Agriculture, Ecosystems and Environment*, 116, 209–215.
- Verstraeten WW, Muys B, Feyen J, Veroustraete F, Minnaert M, Meiresonne L, De Schrijver A (2005) Comparative analysis of the actual evapotranspiration of Flemish forest and cropland, using the soil water balance model WAVE. *Hydrology and Earth System Sciences*, 9, 225–241.
- Vicca S, Luyssaert S, Peñuelas J et al. (2012) Fertile forests produce biomass more efficiently. *Ecology Letters*, 15, 520–526.
- Vilén T, Gunia K, Verkerk PJ, Seidl R, Schelhaas M-J, Lindner M, Bellassen V (2012) Reconstructed forest age structure in Europe 1950–2010. *Forest Ecology and Management*, 286, 203–218.
- Vitousek PM, Naylor R, Crews T et al. (2009) Nutrient imbalances in agricultural development. *Science*, 324, 1519–1520.
- van Vliet N, Mertz O, Heinemann A et al. (2012) Trends, drivers and impacts of changes in swidden cultivation in tropical forest-agriculture frontiers: a global assessment. *Global Environmental Change*, 22, 418–429.
- Wang S, Davidson A (2007) Impact of climate variations on surface albedo of a temperate grassland. *Agricultural and Forest Meteorology*, 142, 133–142.
- Wang Z, Chappellaz J, Park K, Mak JE (2010) Large variations in southern hemisphere biomass burning during the last 650 years. *Science*, 330, 1663–1666.
- Ward DS, Kloster S, Mahowald NM, Rogers BM, Randerson JT, Hess PG (2012) The changing radiative forcing of fires: global model estimates for past, present and future. *Atmospheric Chemistry and Physics*, 12, 10857–10886.
- van der Werf GR, Randerson JT, Giglio L, Gobron N, Dolman AJ (2008) Climate controls on the variability of fires in the tropics and subtropics. *Global Biogeochemical Cycles*, 22, GB3028.
- van der Werf GR, Randerson JT, Giglio L et al. (2010) Global fire emissions and the contribution of deforestation, savanna, forest, agricultural, and peat fires (1997–2009). *Atmospheric Chemistry and Physics*, 10, 11707–11735.
- Westholm E, Lindahl KB, Kraxner F (2015) *The Future Use of Nordic Forests: A Global Perspective*. Springer, Heidelberg, Germany.
- Wicklein HF, Ollinger SV, Martin ME et al. (2012) Variation in foliar nitrogen and albedo in response to nitrogen fertilization and elevated CO<sub>2</sub>. *Oecologia*, 169, 915–925.
- Wiedinmyer C, Akagi S, Yokelson R, Emmons L, Al-Saadi J, Orlando J, Soja A (2011) The Fire INventory from NCAR (FINN): a high resolution global model to estimate the emissions from open burning. *Geoscientific Model Development*, 4, 625–641.
- van Wilgen BW, Govender N, Smit IPI, MacFadyen S (2014) The ongoing development of a pragmatic and adaptive fire management policy in a large African savanna protected area. *Journal of Environmental Management*, 132, 358–368.
- Wint W, Robinson T (2007) *Gridded livestock of the world 2007*. Food and Agriculture Organization of the United Nations, Washington, D.C., 138 pp.
- Wirsenius S (2003) The biomass metabolism of the food system: a model-based survey of the global and regional turnover of food biomass. *Journal of Industrial Ecology*, 7, 47–80.
- Wisser D, Froking S, Douglas EM, Fekete BM, Vermeir SM, Schumann AH (2008) Global irrigation water demand: variability and uncertainties arising from agricultural and climate data sets. *Geophysical Research Letters*, 35, L24408.
- Wolf B, Zheng X, Brüggemann N et al. (2010) Grazing-induced reduction of natural nitrous oxide release from continental steppe. *Nature*, 464, 881–884.
- Wong PP, Losada IJ, Gattuso JP et al. (2014) Coastal systems and low-lying areas. In: *Climate change 2014: Impacts, Adaptations, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (eds Field C, Barros VR, Dokken DJ, Mach KJ, Mastrandrea MD, Bilir TE, Chatterjee M, Ebi KL, Estrada YO, Genova RC, Girma B, Kissel ES, Levy AN, MacCracken S, Mastrandrea PR, White LL), pp. 361–409. Cambridge University Press, Cambridge, UK.
- Yoo CY, Pence HE, Hasegawa PM, Mickelbart MV (2009) Regulation of transpiration to improve crop water use. *Critical Reviews in Plant Sciences*, 28, 410–431.
- You L, Wood S, Wood-Sichra U, Wu W (2014) Generating global crop distribution maps: from census to grid. *Agricultural Systems*, 127, 53–60.
- Zarin DJ, Davidson EA, Brondizio E et al. (2005) Legacy of fire slows carbon accumulation in Amazonian forest regrowth. *Frontiers in Ecology and the Environment*, 3, 365–369.
- Zona D, Janssens IA, Gioli B, Jungkunst HF, Serrano MC, Ceulemans R (2013) N<sub>2</sub>O fluxes of a bio-energy poplar plantation during a two years rotation period. *GCB Bioenergy*, 5, 536–547.

## Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Summary of data status and methods.

Table S1. Selected management activities.

Table S2. Estimates of management intensities.

Table S3. Evaluation of the biophysical and biogeochemical effects of each of the reviewed management activities with the reference ecosystem considered and the processes taken into account for the evaluation.