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

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Abstract

In 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: (1) management activities for which datasets are available, and for which a good knowledge base exists (cropland harvest and irrigation); (2) management activities for which sufficient knowledge on biogeochemical and biophysical effects exists but robust global datasets are lacking (forest harvest, tree species selection, grazing and mowing harvest, N-fertilization); and (3) 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 e.g. 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: Land management, global land use datasets, data availability, land-cover modification, process understanding, Earth system models

1. 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 policy makers 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, e.g. change in forest cover (Brovkin *et al.*, 2004; Feddema *et al.*, 2005; Pongratz *et al.*, 2009), land-use activities that result in 'land modifications', i.e. 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 spatio-temporal 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 (Arneth *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.

2. Key land management activities

During an interdisciplinary workshop cycle (see Acknoweldgements), we identified ten important land management activities that may impact the Earth system profoundly (Table S1 in the Supplementary Information, SI), namely 1) forest harvesting; 2) tree species selection; 3) grazing and mowing harvest; 4) crop harvest and crop residue management; 5) crop species selection; 6) nitrogen (N) fertilization of cropland and grazing land; 7) tillage; 8) crop irrigation (including paddy rice irrigation); 9) artificial drainage of wetlands for agricultural purposes; and 10) fire as a management tool (Figure 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 these management practices prevail. The provision of bioenergy, e.g. biofuels from plant oil, starch or sugar, or wood fuel, is not classified as own management type. Rather, it is subsumed under items 1) and 4). It is important to note that this list represents a subjective, consensus-oriented group opinion and is thus not exhaustive nor representative. For instance, many management activities have not been considered here e.g., litter raking, peat harvest, phosphate or potassium fertilization, crop protection, forest fertilization, or mechanization. Such activities can be of central importance, e.g. in specific contexts, and advancing the understanding of their divers and impacts is equally important.

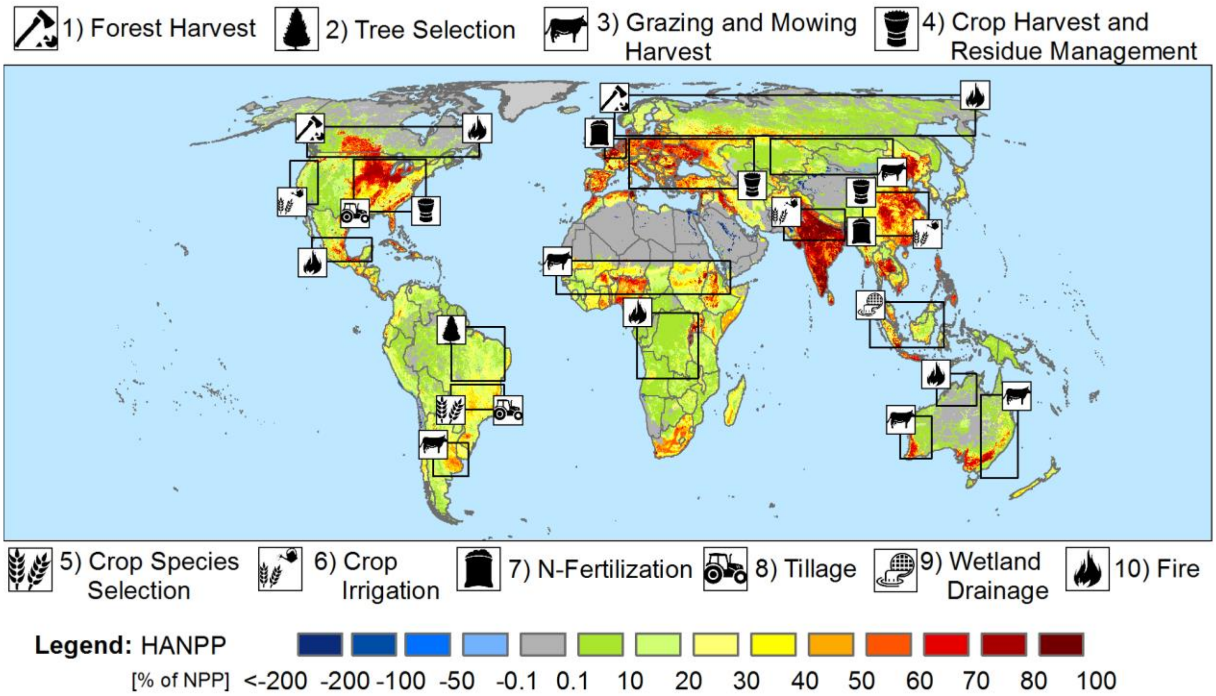

Figure 1. The ten selected management activities and a selection of geographic regions where these activities play an important role. The background map displays the human appropriation of net primary production (Haberl *et al.*, 2007; Copyright 2007 National Academy of Sciences, USA), i.e. the ratio 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.

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_2, H_2O, N_2O, NOx, NH_3, CH_4) or by changes in atmospheric chemistry (CH_4, O_3, H_2O, SO_2, 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.

2.1. Forestry harvest

2.1.1. Extent and data availability

Forests cover 32.7-40.8 Mkm^2 or 30% of the ice-free land surface and 2/3 – 3/4 of global forests (26,5-29,4 Mkm^2) are under some form of management (Erb *et al.*, 2007; FAO, 2010; Pan *et al.*, 2013; Luyssaert *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, i.e. management schemes that involve careful planning based on empirical observations and forest-ecological process understanding (Mårald *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 forest 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^2) 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). 29-34 Mkm^2 forests are under harvest, of which 7% are intensive plantations, 65% subject to regular harvest schemes, and 28% under other (e.g. sporadic) uses (SI). Data on wood harvest is 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, and oversimplified approaches for creating gridded time series (Erb *et al.*, 2013b; Birdsey & Pan, 2015).

2.1.2. 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^{-1} consisting of around 0.5 Pg C yr^{-1} for wood fuel and another 0.5 Pg C yr^{-1} as timber (Krausmann *et al.*, 2008; FAOSTAT, 2015). Forest harvest mobilizes annually less than 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 (Luyssaert *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 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; Holtsmark, 2012). The effects of forest management on CH_4 and N_2O 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_2O 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 subcanopy (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).

2.2. Tree species selection

2.2.1. Extent and data availability

Forest plantations cover 2.9 Mkm^2, or 7% of the world's forest areas, e.g. 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^2 (Luyssaert *et al.*, 2014). In Europe, for instance, species selection has resulted in an increase of 0.5 Mkm^2 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 U.S (Oswalt *et al.*, 2014), and 7% of the global used forests (SI). 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 is particularly scarce (Table 1; SI) 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 is only available in short time series from FAO Forest Resources Assessments (FAO, 2015a).

2.2.2. 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; Bárcena *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 (Bárcena 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 (Peñuelas & 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 to 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 meters 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/year 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 southeastern US, 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).

2.3. Grazing and mowing harvest

2.3.1. Extent and data availability

Grazing and mowing harvest is the most spatially extensive land management activity worldwide, covering 28-56 Mkm^2 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 7k-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), grazing intensity is high on less than 10%, medium on around two thirds and low on one fourth of the grazing lands (SI). 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).

2.3.2. 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 – 1.8 PgC yr^{-1} 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 semiarid 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 (Väisänen *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, e.g., by trampling, can contribute to anaerobic microsites, reducing the CH4 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_2O 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_2O and CH_4 (Baron *et al.*, 2002; Chang *et al.*, 2015). A study of year-round measurements of N_2O in the Mongolian steppe found that while animal stocking rate was positively correlated with growing-season emissions, grazing decreased overall annual

N_2O emissions (Wolf *et al.*, 2010). Sites with little and no grazing showed large pulses of N_2O 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-colored 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).

2.4. Crop harvest and residue management

2.4.1. Extent and data availability

Approximately 15 Mkm^2 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^2 are permanent cultures, including perennial, woody vegetation (e.g. fruit trees, vineyards). Approximately two thirds of the arable land are 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 multi-cropping (i.e. more than one harvest per year) occurs (SI). Cropping activities are closely tied to the sedentary lifestyle that emerged with the Neolithic revolution some 12 k 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), i.e. 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) which 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 e.g. 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 datasets existing (see Table 2).).. Nevertheless, major uncertainties remain related to cropland patterns in some world regions, particularly across large swaths of Central, Southern and Northern Africa, Brazil and Papau 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 is available. Data on crop residues is 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)

2.4.2. 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^{-1} in 2000, around half of total biomass harvest, or around 5% of global terrestrial NPP (Wirsenius, 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^{-1} is used as feed for livestock (fodder crops and concentrates). In 2004, crop harvest for bioenergy amounted to 1.6 EJ yr^{-1} from agricultural by-products and 1.1 EJ yr^{-1} from fuel crops, which is roughly equivalent to 0.043 and 0.03 PgC yr^{-1}, respectively (Sims *et al.*, 2007). 0.7 PgC yr^{-1} of secondary products remain on site, possibly ploughed to the soil or burned subsequently (Wirsenius, 2003; Krausmann *et al.*, 2008). Cropland systems, mainly consisting of annual, herbaceous plants, usually contain little carbon in vegetation and soil per m^2 (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 is available (FAOSTAT, 2015). However, the lack of primary data such as from long-term field studies and the use of crude factor introduces 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 post-harvest management practices, e.g. 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).

2.5. Crop species selection

2.5.1. 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 sedentary subsistence, with species selected according to human needs (e.g. food, health, stimulants, fiber). 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 inter-annual dynamics, such as rotational schemes, are lacking (Table 1; SI).

2.5.2. 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 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 < soybean < corn (Bolinder *et al.*, 2007). A shift from annual to perennial crops and the introduction of cover crops can significantly increase SOC stocks (Poeplau & 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-soy 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 soybean cultivation, SOC content was found to be about 7% higher under soybean as compared to wheat and maize. Other GHG emissions are also crop-specific. For example, N_2O emissions 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, e.g. 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 the 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 differs 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).

2.6. N-Fertilization of cropland and grazing land

2.6.1. 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 feces), sewage sludge or mineral fertilizer. Reactive nitrogen was a scarce resource in preindustrial agriculture, mainly 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 kg N $ha^{-1}yr^{-1}$e.g. in the Nile delta and 90 kg N $ha^{-1}yr^{-1}$ in New Zealand (Potter et al., 2010; Mueller et al., 2012), and 14% of cropland are fertilized with levels above 100kgN ha^{-1} yr^{-1}. Globally, much lower intensity level prevail, 59% of the global cropland area show application rates below 5010kgN ha^{-1} yr^{-1}, and around one quarter of global croplands show fertilization rates below 10kgN ha^{-1} yr^{-1} (SI). 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 kg N ha-1 yr-1 (Flechard *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 wide-spread activity in many region, is scarce and crude-model derived (SI).

2.6.2. 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 (N2O), nitric oxide (NO) and ammonia (NH3). A typical fertilized cropland emits 2-3 times more nitrogen than the approximately 0.5 kg N ha^{-1} yr^{-1} emitted under non-fertilized conditions (Stehfest & Bouwman, 2006), while fertilized grasslands emit 3-4 times more N_2O than unfertilized ones (Flechard *et al.*, 2007). The global N2O emissions on fertilized croplands and grazing lands sum to 4.1 to 5.3 Tg N yr 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, N2O emissions are determined by crop type, fertilizer type, soil water content, SOC content, soil pH and texture, soil mineral N content and climate. 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 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 (Körschens 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 40–70 Tg N yr^{-1} (Boyer *et al.*, 2006; Fowler *et al.*, 2013). Increased nutrient input to rivers and freshwater systems impact 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.

2.7. Tillage

2.7.1. Extent and data availability

With the mechanization of agriculture, arable land became regularly tilled to suppress weeds and enhance soil structure and nutrient availability. Archeological findings suggest that humans manipulated soil structure through some form of tillage with ards and hoes already some 4500 years ago (Postan *et al.*, 1987). From the 1950s, with the advent of modern herbicides no-till systems became more prominent, mainly in the U.S. (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 (1) perennial crops, which cover approximately 10% of cropland area or 1.5 Mkm^2 (FAOSTAT, 2014) and (2) no-till agriculture (or reduced tillage) on 1.11 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 U.S., where 70% respectively 30% 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.

2.7.2. 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 results in a redistribution of carbon in the soil profile. Other findings are inconclusive, e.g. 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_2O emissions is site-specific and inconclusive (Rochette, 2008). A recent meta-analysis reported that no-till reduced N_2O 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 whilst maintaining the same soil temperature compared to a bare site (Hares and Novak, 1992), thus providing protection against wind and water erosion without affecting seed germination (Hares and Novak, 1992). The direct effects of tillage on surface roughness are likely negligible for the surface climate.

2.8. Irrigation

2.8.1. Extent and data availability

Globally 2.3–4.0 Mkm^2 or 15 to 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 to 1.0 Mkm^2 (Salmon et al., 2015, Portmann et al., 2010), or 5-7% of the global cropland area. Paddy rice cultivation is particularly important in East, South and Southeast 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^2), 20% of the maize fields (0.3 Mkm^2), 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 datasets 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.

1.1.1. 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 non-irrigated 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_2O production, especially when fertilized (Verma *et al.*, 2006). This is particularly the case in paddy fields, where emission factors range between 341 and 993 gN ha^{-1}, 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 N2O emissions. The capacity of soils to oxidize atmospheric CH_4 may be reduced under irrigation (Ellert & Janzen, 1999; Sainju *et al.*, 2012). Irrigated rice fields alone are emitting approximately 30-40 $TgCH_4$ yr^{-1} (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 (Kozlowski, 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 Wm^{-2} (Boucher *et al.*, 2004).

2.9. Artificial drainage of wetlands

2.9.1. 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^2, are subject to artificial drainage (Feick *et al.*, 2005), but the strongest biogeochemical and biophysical effects of drainage are expected when wetlands are drained, e.g., peatlands, inland flood plains, coastal wetlands, or lakes. Wetlands are estimated to cover 5.3-26.9 Mkm^2 (Melton *et al.*, 2013), of which 0.18 Mkm^2 are probably drained (SI), but data are scarce. Wetland drainage dates back for millennia, e.g., 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 US (Brinson & Malvárez, 2002). Despite attempts for wetland conservation (see e.g. (Dugan, 1990), or the international RAMSAR treaty (www.ramsar.org), large-scale new drainage installation is still ongoing (Brinson & Malvárez, 2002; Lähteenoja *et al.*, 2009), in particular in Asia , for instance in relation with palm oil expansion (Davidson, 2014). Consistent data on wetland drainage are practically inexistent.

2.9.2. 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 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_2, CH_4 and N_2O 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 (Drösler *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_2 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 spring-time albedo (Lohila *et al.*, 2010).

2.10. Fire management

2.10.1. 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), e.g., 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 (Hurtt *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^2 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).

2.10.2. 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^{-1} (van der Werf et al., 2010), while human-induced fires range from 1.7-2.0 PgC yr^{-1} (Lauk and 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, e.g. post-fire 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 pre-fire 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 W/m^2 immediately after a boreal forest fire, which remained high at -4 W/m^2 over 80 years after the fire. In a savannah, a halving of the albedo (0.12 to 0.07) was observed, followed by a recovery to a pre-fire state after several weeks (Scholes & Walker, 1993; Beringer *et al.*, 2003).

3. 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; Luyssaert *et al.*, 2014). According to our review, around 10% of the ice free land surface are under intense human management, half of it under medium and one fifth under extensive management (Supplementary information; Fig. 2).

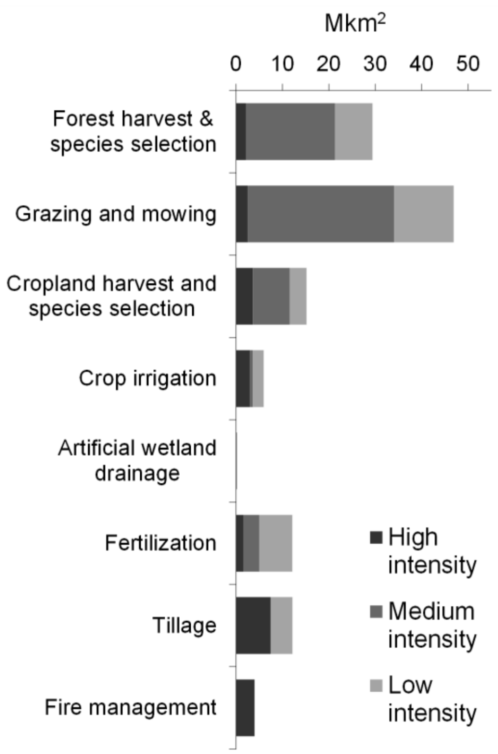

Figure 2. Global extent and intensity of land management activities. Globally, approximately 80% of the 130 Mkm² of ice-free land is under managed schemes at varying intensity. Note that the bars are not additive, as e.g. crop irrigation, fertilization and tillage all occur on cropland. For data and assumptions, see SI.

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.

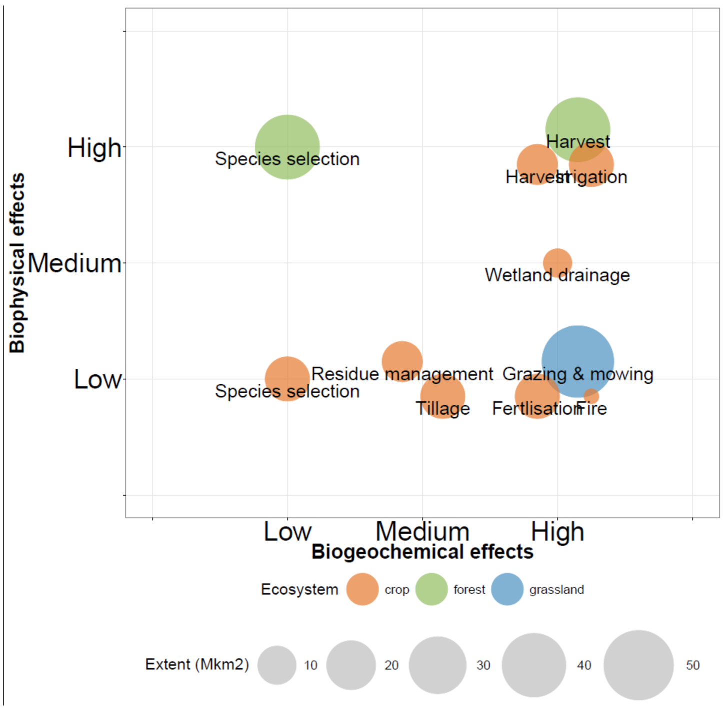

Figure 3. Extent and biogeochemical and biophysical effects of management activities. The classification (see SI) is based on expert judgement and hence contains a certain degree of subjectivity and ambiguity.

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 respects (Figure 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 require 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 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).

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

	National Statistics (based) w. global coverage*		Gridded Spatial Data, continental or global			
Management activity	**Static**	**Time Series**	**Continental or Ecozone, Static**	**Global, Static**	**Global, Time Series**	**Comments**
Forestsry 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	(Hurtt et al., 2011) [Europe: (Vilén 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, 2015	Europe: (Brus *et al.*, 2011) (Hengeveld *et al.*, 2012)-system approach (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)* (Chang *et al.*, 2015)** *relying on (Wint & Robinson, 2007) *based on ORCHIDEE-GM	(Herrero *et al.*, 2013)* (Haberl *et al.*, 2007) *relying on (Wint & Robinson, 2007)		Extreme uncertainty level - estimates on the global extent vary strongly (+/-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) (Iizumi *et al.*, 2014)	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 inter-annual 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 modeling derived and show large discrepancies, in particular livestock manure is error prone No data on fertilization outside croplands
Tillage						No data on tillage, but presumable all cropland is tilled with two exceptions: permanent crops and zero-tillage agriculture. For the latter, no data is available
Irrigation (including paddy rice)	(FAOSTAT, 2015)	(FAOSTAT, 2015)	Parry rice: (Frolking *et al.*, 2006)	(Portmann *et al.*, 2010) (Salmon *et al.*, 2015) (Wisser *et al.*, 2008)	(Freydank & Siebert, 2008) (Siebert *et al.*, 2015)	Many data, e.g. 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: e.g. Africa: (Liousse *et al.*, 2010) Canada: (Stocks *et al.*, 2002)	all fires: e.g. (Giglio *et al.*, 2013); (Alonso-Canas & Chuvieco, 2015)	all fires: e.g. (. (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 datasets are available. Prominent data providers (non-exhaustive) are Eurostat for European countries (http://ec.europa.eu/eurostat) or the United States Department of Agriculture (http://www.ers.usda.gov/topics.aspx).

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 datasets. 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, e.g. 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 activites 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, i.e. 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 the state of knowledge is sufficient for implementing these activities in integrative assessment environments such as Earth System Models.

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

	Data advanced	Data poor
Understanding advanced	• Crop harvest • Irrigation	• Forestry harvest • Tree species selection • Grazing and mowing harvest • N-fertilization
Understanding poor	…	• Crop species selection • Artificial wetland drainage • Tillage • Fire management • Crop residue management[1]

[1] Separated here from crop harvest

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 datasets, e.g. by taking advantage of the increasingly available data from satellite observations (Kuemmerle *et al.*, 2013; Joshi *et al.*, 2016), or crowd sourcing (See *et al.*, 2015), but also alternative approaches that exploit existing databases. These management activities could be included in Earth system models but global parameterisation 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 e.g. 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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