Chapter 4
Review of the Impacts on Soils of Land-Use Changes Induced by Non-food Biomass Production

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Abstract Over the past decade, the exponential growth in the production of biomass for energy use has raised concerns as to the environmental impacts of this type of land use, as well as the potential land-use changes (LUC) associated with an extension of agricultural land areas. Determining the environmental impacts of an expanding bioenergy sector requires reconstructing the chains of cause and effect from the determinants of land-use change (both direct and indirect) and land-use practices through to the impacts of those practices. Conducting an exhaustive literature review from 1975 to 2014, we identified 241 articles relevant to this causal chain, thus enabling an analysis of the environmental impacts of LUC for bioenergy. This chapter presents the results of a detailed literature analysis and literature review of the 52 articles within this corpus specifically addressing impacts on soils. The variation in soil organic carbon (SOC) is the most commonly used impact indicator, followed by soil loss to erosion and, to a lesser extent, the potential for environmental acidification as determined by life-cycle assessments. Background and transitional SOC levels during LUC affect the predictive value of estimated final SOC variations but are not generally accounted for in default static stock-difference approaches. Perennial crops tend to be better at maintaining or even improving SOC levels, but results vary according to pedoclimatic and agronomic conditions. The mechanisms involved notably include protection of the soil surface with a dense perennial cover and the limitation of tillage operations, especially deep plowing; accumulation of organic matter and SOC linked to biomass production, especially belowground production of rhizomes and deep, dense root systems; associated reductions in nutrient loss *via* runoff and erosion. Nevertheless, additional research is needed to improve our understanding of and ability to model the full range of processes underlying soil quality and LUC impacts on soil quality.

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4.1 Introduction

The production of biomass for bioenergy and biomaterials has expanded considerably in recent years. This expansion is likely to continue given a context in which substitutes must be found for diminishing fossil resources (Chum et al. 2011). Increases in biomass production present challenges linked to the expansion of agricultural land area and the potential impacts of land-use change (LUC) (Searchinger et al. 2008). These concerns have prompted a sharp increase in the number of scientific publications on this topic over the past 10 years . Assessing the environmental impacts of bioenergy development requires reconstructing the chains of cause and effect from the direct and indirect determinants of land-use change and land-use practices through to their various impacts, all along the value chain from biomass production to the final product.

Recent studies have surveyed these issues and documented emerging research trends (e.g., Broch et al. 2013), but no published work to date has conducted a systematic literature review corresponding to the three steps of this causal sequence: from the determinants of increased land use for bioenergy, through changes in land allocation, up to the environmental impacts of biomass production; i.e., the "reorganization-LUC-impact" causal sequence. Indeed, a recent review of methodologies for analysis and meta-analysis of this causal sequence identified a disconnect between research examining the drivers of LUC, on the one hand, and work on environmental impacts, on the other hand (van Vliet et al. 2016). Environmental impacts can be diverse in nature, affecting soils, air quality, biodiversity, etc., but these various types of impacts are rarely considered together in the studies reviewed in the present chapter.

The aim of the overarching study, whose outputs are detailed throughout this volume, was to provide quantitative data, based on an exhaustive literature review, for the analysis of these causal sequences. The results are broken down into a meta-analysis accompanied by focused literature reviews of each stage in the causal chain: from the analysis of LUC drivers, to analyses of LUC, to assessments of the various categories of identified impacts. In the first step of the literature review, 5730 articles (from 1975 to 2014) relating to LUC in general were extracted from the Web of Science and CAB databases. The second step consisted of an automated textual search procedure (see El Akkari et al., Chap. 2 in this volume) to identify articles allowing for an analysis of the causal sequence reorganization-LUC-impact, including at least one impact category from the following list: climate change (greenhouse gas emissions), depletion of fossil/non-renewable resources or water resources, impacts on biodiversity or soil quality, atmospheric pollution, human health and ecotoxicity. This reduced the corpus to 1785 articles, which were then examined in more detail by a dozen scientific experts, seeking to identify articles addressing the full causal sequence as well as those featuring datasets available for the meta-analysis. This third step reduced the corpus to 241 articles. The present chapter describes the results of the focused literature review of the studies examining impacts on soil quality.

In the following section, a qualitative analysis of the causal sequences in this focused corpus enables us to appreciate the representativeness of these results in terms of geographic coverage, sectors examined, and the robustness of the methodologies and data employed. The subsequent section engages in a more detailed analysis of methodologies, impacts, and the mechanisms underlying those impacts.

4.2 Bibliometric Analysis

The corpus analyzed in this chapter consists of 52 of the 241 articles identified. Three-quarters of these articles were published in the last 4 years of the study period (i.e., 2011–2014) (Fig. 4.1).

4.2.1 Areas of Historical Importance More Strongly Represented Than Emerging Areas

The majority of the land-use changes (LUC) examined in the corpus are located in the United States (30%) or Brazil (14%) (Fig. 4.2). At the continental level, the Americas account for 54% of locations and Europe for 30%, far ahead of Africa (5%), Asia (5%), and Oceania (2%). The remaining 4% correspond to two studies focused on the global level. The predominance of research focusing on the United

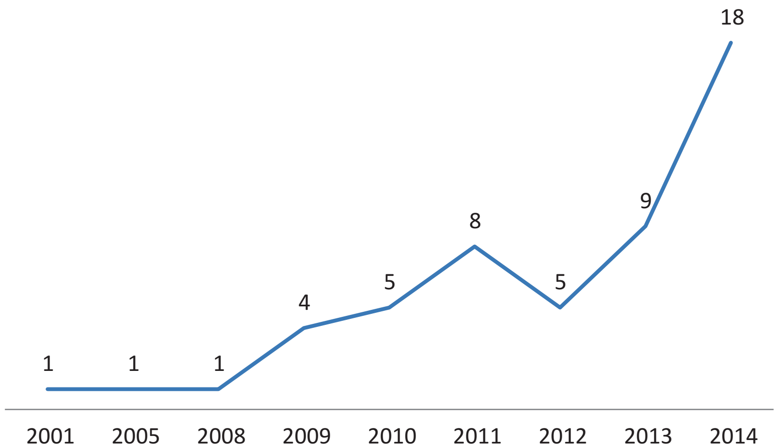


Fig. 4.1 Number of publications on impacts of land-use change on soils by year (2001–2014)

Fig. 4.2 Case study locations

States and Brazil is likely due to a greater accumulation of research efforts given the longer history of the biofuel sectors in these countries. More recent developments in the European countries are notable at the aggregated continental level.

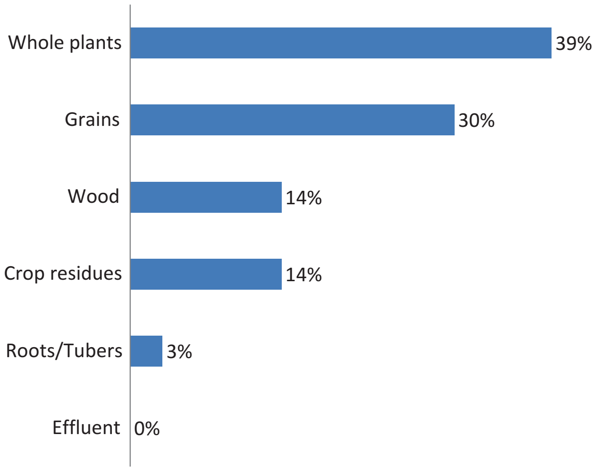
The information recorded during the experts' examination of the full corpus did not allow for the identification of all production areas, since some research articles consider multiple origins for plant products but not all of these origins were necessarily listed in the reading grid, notably in the case of imports of bioenergy feedstocks (e.g., palm oil from Malaysia is used in one scenario, but this country is not listed in the "location" field). We thus find only a few studies addressing emerging tropical regions for bioenergy production where land-use change for agricultural development is taking place most rapidly, such as in Indonesia, Malaysia, or Congo.

The scale of the research described in the articles is generally large. Approximately 60% of those studies for which the scale of spatial analysis was recorded were conducted at a level equal to or greater than a region or county. Similarly, where this information was recorded, land area considered for biomass provision exceeded 1000 ha in 70% of the articles, and exceeded 1,000,000 ha and 25% of the articles.

4.2.2 *Crops Dedicated for Biofuels Predominate*

The principal types of biomass represented are whole plants (all aboveground biomass harvested) or grains (Fig. 4.3). These trends appear robust despite the fact that biomass type was not systematically recorded (for 17% of the articles, the experts' review did not indicate biomass type). Double counting may also skew this breakdown (e.g., "Entire plant + wood," recorded for plantings of species used for short-term coppice rotations).

Fig. 4.3 Breakdown of the subcorpus by biomass type (not noted in 17% of studies)



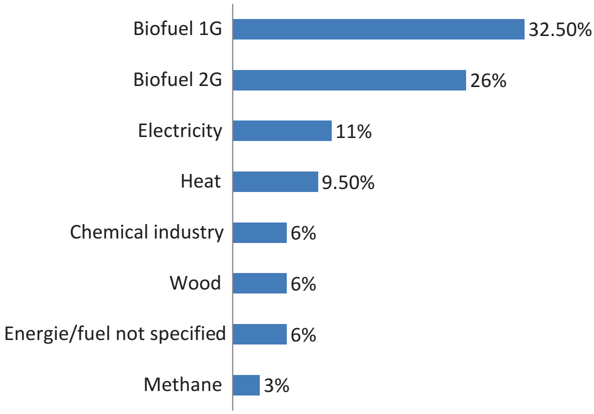


Fig. 4.4 Breakdown of the subcorpus by final product type (not noted in 15% of studies)

An analysis of species distribution by biomass type could not be completed, since the list of species recorded was not exhaustive, notably as a result of the global studies reviewing numerous species without a fine level of detail. Nevertheless, the results suggest a prevalence of miscanthus, switchgrass and sugar cane (whole plants); soybean, rape, and maize (grains); poplar (wood); wheat (crop residue); and sugarbeet (roots/tubers). The principal final products examined were first- and second-generation biofuels (Fig. 4.4).

Agricultural practices for these biomass types were not systematically recorded in the studies, and so could not be analyzed: there were only 17 entries out of a total of 238 scenarios. Most studies focus primarily on LUC scenarios rather than on changes in agricultural practices scenarios that are unrelated to LUC in the strict sense of a change in land allocation. LUC are not necessarily associated with a change in practices defined as "crop diversification". Changes in "crop diversification" practices may be understood as a diversification at the level of the farm system rather than at the level of the land allocation mosaic. The "short-term coppice rotation" practice represented 40% of the 17 practice types recorded, which is understandable given the clearer correlation between this practice and an LUC type, i.e., the establishment of dedicated plantings as part of a new agricultural system.

4.2.3 Poorly Characterized Aspects of Land-Use Changes

No clear trends appear in the corpus analyzed here as to the regulatory context of LUC (regulated = 33%, not regulated = 19%, not recorded = 48%), or as to the LUC timeframe (retrospective = 14%, prospective = 44%, both = 40%, does not apply = 2%). Years or time periods assessed with respect to LUC were also poorly recorded (in 62% and 77% of articles, respectively, these details were not provided).

In correlation with the major biomass types recorded (Fig. 4.3), the most important types of direct LUC involve conversions of forests, annual crops, or grasslands into perennial energy crops, which are mostly harvested as whole plants or grains (34%). Next in importance are direct LUC in which forests or grasslands were converted into annual crops (16%). Approximately 18% of the LUC examined relate, *a priori,* to a change in agricultural practices rather than to an LUC in the strict sense. Indirect LUC were examined 4 times less often than direct LUC, but trends are similar in terms of the types of land use involved.

4.2.4 Overview of Methods and Data Used

A survey of the methods used in the articles shows that efforts to model final impacts on soils are overall more common than characterizations of earlier stages in the causal sequence, i.e., modeling of the causes and types of LUC (Table 4.1). Specifically, the reorganization of land-use types is mostly either not recorded or is estimated according to basic calculations based on observations or suppositions of direct changes without a global modeling. Economic models, although widely used in the modeling of agricultural reorganizations and LUC, are not strongly present here. The most frequently used method for analyzing soil impacts is life-cycle

Table 4.1 Overview of methodologies and data types utilized

Steps in the impact chain		***Reorganization***	***LUC***	***Soil impact***
Method	Not stated/other	**43/2%**	17/3%	4/12%
	Basic calculation	23%	**33%**	20%
	Statistical model	5%	7%	7%
	Life Cycle Analysis	8%	9%	**31%**
	Biophysical/process-based/ecological model	5/3/2%	11/7%	10/10/1%
	Economic model	7%	9%	4%
	Meta-analysis	2%	1%	1%
	Qualitative	-	1%	1%
Type of data	Not stated/other	25%	12%	4/2%
	Scientific reference	24%	29%	**37%**
	Statistics/land use	9/11%	12/19%	12/9%
	Field data (observations/measurements/interviews/expert opinion/climate/satellite imagery/soil data)	8/6/1/13/1/1%	5/5/-/12/4/-/2%	**10/10/1/11/4/-/- %**
	Global economic models	1/- %	-	-/1%
Accessibility of results	Not stated/no information	**38/- %**	25/1%	5/- %
	Tables/figures/maps	**32/21/4%**	**33/25/4%**	**44/43/3%**
	Raw data/text	1%	7/3%	2/2%
Precision of results	Not stated/no information/other	**58/35%**	**44/44%**	**20/35/11%**
	Standard error/standard deviation/confidence interval	2/2/- %	2/2/4%	4/2/11%
	Sensitivity analysis	4%	4%	18%

NB: Totals ≠ 100% due to rounding

Cell shading corresponds to percentage totals: Light blue: >15–30%; Gray: >30–45%; Dark gray: >45%

LUC Land use change

assessment (LCA), followed by basic calculations relying notably on changes in quantities of biomass or carbon, followed by more mechanistic models.

Data types are better recorded for all stages of the "reorganization-LUC-impact" causal sequence. Literature references and statistical data are the most frequently found data types, particularly for land use. Nevertheless, data gathered in the field – including experimental data, climate data, and survey data – account for almost a third of the data used. Again, the final link in the causal chain, that is, the modeling of impacts, attracts most of the work of characterization, with the largest number of both literature and field data.

Finally, the results overall are not highly detailed. Data are mainly accessible in the form of tables, which do not include all stages of the causal chain; or in figures or maps presenting results in a more or less aggregated form. The statistical robustness of the results is not always noted, nor is the validity domain of the results always discussed.

4.3 Analysis of Soil Impacts

4.3.1 Few Impacts Addressed

Soils are a complex resource, supporting many functions (Doran and Parkin 1994; Karlen et al. 2003; Patzel et al. 2000). These functions are enabled and may be affected by a variety of interacting physicochemical and biological soil properties and conditions. Impacts on soil quality, i.e., the capacity of a soil to support diverse functions, are as potentially numerous as all the possible combinations of modifications that may occur for these diverse soil properties. While some processes are broadly understood (e.g., erosion, acidification), the impact mechanisms connecting environmental conditions and agricultural practices to variations in soil properties, and their consequences for soil functions, have only been partially described (Karlen et al. 2003; Kibblewhite et al. 2008).

In the corpus considered here, only a few types of impacts on soils are described in detail. The impacts most often addressed are the levels of soil organic matter (SOM) and soil organic carbon (SOC), acidification, and erosion (Fig. 4.5). The preponderance of the impacts on SOM and SOC levels can be related to the climate change challenge, a primary driver for bioenergy development, since soil C sequestration and/or release plays a part in the greenhouse gas balance. Hence, in most studies a variety of more or less complex methods are applied to estimate, at least, variations in carbon stocks, including soil carbon. The climate change impact is a standard "midpoint" impact in LCA, which explains why this method is so widely represented in the corpus. Studies seeking to establish bioenergy sector impacts on climate change frequently rely on impact characterizations from LCA or from a carbon footprint assessment, which is a partial LCA. The use of LCA to characterize impacts in approximately a third of the cases is thus explained by the logical

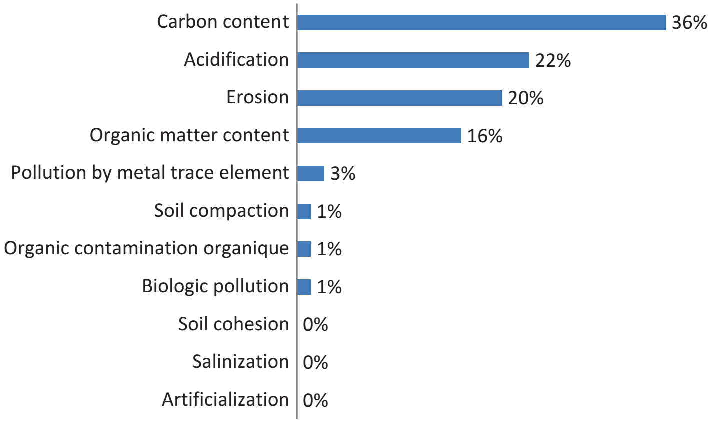


Fig. 4.5 Breakdown of the subcorpus by type of soil impact considered

connection that may be made between impacts on SOC and the climate change impact. The same is true for the "depletion of fossil resources" impact, another key impact for bioenergy sectors, and to a lesser extent for the impacts of acidification and eutrophication, which are also "midpoint" impacts in LCA that are commonly studied in cases of agricultural production, given the impact contributions of fertilizers. The imbalance in the number of studies across these various impacts arises from the fact that most published LCAs (particularly those related to bioenergy) are partial LCAs, examining only 1–3 impact categories, usually impacts on climate change and the depletion of fossil resources (Bessou et al. 2013). Within the corpus examined here, 22% of LCAs examine only 1 or 2 LCA impact categories, with the climate change impact being the only impact common to all studies.

Erosion, on the other hand, is not a standard impact category in LCA. It can be found in some LCA characterization methods, e.g., LANCA© (Bos et al. 2016) and ACV-SOL (Garrigues et al. 2013), but none of these were used in the articles in the corpus, which is already somewhat dated with respect to recent developments in this subfield of LCA. Erosion is generally regarded as a sensitive impact type for soils, linked primarily to cultivation (conversion of forest into arable land, etc.) or to a change in agricultural practices (change in soil cover, reduced tillage, etc.). It is among the most significant risks for soils. According to the Food and Agriculture Organization (FAO) and the Intergovernmental Technical Panel on Soils (FAO, ITPS 2016), if erosion continues at current levels, it will result in yield losses equivalent to the removal from production of 150 million hectares of agricultural land by 2050. It is thus unsurprising to find erosion among the most studied impacts. Erosion is moreover primarily a physical or physicochemical impact, and one for which a variety of more or less complex models are available. By comparison, our understanding of and the availability of models for assessing other environmental impact mechanisms, especially those involving complex biogeochemical cycles and soil biodiversity, remain a limiting factor in characterizing the impacts of land use and land-use change (LULUC) on soils.

4.3.2 Critical Review of Methods Used for Quantifying Impacts

4.3.2.1 Differences in Carbon Stocks

Calculating impacts on soils with respect to levels of SOM or SOC usually involves evaluating a difference in stored amounts between two or several successive states. Other approaches include *in situ* measurement of fluxes or the use of modeling (see Sect. 4.3.2.3). The difference in stored amounts or "stock-difference" approach is one of two calculation methods recommended by the Intergovernmental Panel on Climate Change (IPCC) guidelines for establishing greenhouse gas emissions at the national level according to the Tier 1 and Tier 2 frameworks (IPCC 2006). The stock-difference approach is thus generally used to calculate greenhouse gas emissions levels and their contribution to climate change, rather than specifically for the assessment of soil impacts. The second method recommended by the IPCC, known as "Gain-Loss," uses a different temporal basis for its calculations but likewise relies on a calculation of differences in carbon stocks. The IPCC stock-difference calculation method is the most widely applied both in the literature and in international standards (British Standards Institute 2011; European Commission 2014; WRI/WBCSD 2011; Bernoux et al. 2010; Colomb et al. 2013; Peter et al. 2016), and is notably that used in Annex V of the European Directive on Renewable Energy (EU 2009/28/EC; Decision 2010/335/EU).

The stock-difference is calculated between two soil uses, assuming that those uses are in place for long enough for organic matter levels to have reached equilibrium. This "necessary and sufficient" duration is set at 20 years as a default, and gains and losses linked to changes in use are linearly amortized over that time period. The net annual change thus ignores both temporary effects and irreversible effects, notably those occurring at or immediately following land conversion.

Stocks are defined for several compartments (aboveground plant material, belowground plant material, SOC, etc.), and depend on initial pedoclimatic conditions as well as on weighting factors linked to the soil-use type and to broad soil management categories describing soil tillage and input levels (low, intermediate, high; with or without organic manures). Soil depths used for measuring soil carbon in the articles in the corpus range from 20 to 360 cm. The median depth is 30 cm, which is the standard depth used in the framework of values provided by the IPCC Tier 1 and the European Directive on Renewable Energies. For Tier 1, default stock values are supplied by the IPCC (IPCC 2006). These default values are used in 11 articles (21% of the total corpus, or 30% of those articles considering an SOC impact, whether or not this is specified in the results), including 10 LCAs (43% of LCA). For Tier 2, measured stock values or values derived from more specific references may be used. In 10 articles (19% of the corpus, or 27% of articles considering the SOC impact), including 3 LCAs (13%), SOC levels are directly measured or come from other references besides the IPCC Tier 1. The other studies considering SOC impact use data from models or do not specify references for the stock values used.

The default amortization period of 20 years to allocate a stock difference to each year of cropping is explicitly or apparently applied in 13 articles (25% of the corpus, or 35% of articles considering SOC impacts). This period can vary in some situations. The rationale for adjusting this parameter may be based on a longer period for returning to equilibrium, e.g., 100 years (Cocco et al. 2014), or on a dynamic specific to a particular type of land use, e.g., a linear amortization over the full length of perennial crop cycle (Mello et al. 2014). Variations in the amortization period may be justified, notably by using context-specific stock values. In some cases, the amortization period is defined by socioeconomic or political criteria, independent of the ecological or agronomic basis, e.g. Kauffman and Hayes (2013). When removed from their original study context, such variations can lead to biases in comparing studies or in seeking to analyze historical LUC impacts.

The advantage of the stock-difference method used in the IPCC Tier 1 lies in its global applicability, with values and coefficients that make it possible to calculate and compare soil carbon levels worldwide or across different types of land use and land management. The disadvantage is the lack of sensitivity to specific management conditions or geographic particularities. A key issue is that the IPCC land use categories do not allow for a precise differentiation of different crop types or rotation types, and the weighting factors only broadly account for the effects of different agricultural management practices, with no way to adjust for the full range of practices constituting a cropping system.

Using the static, non-mechanistic approach underlying the stock-difference method, some studies seek to compare soil carbon levels resulting from different soil use categories, in different locations, but in comparable conditions in order to determine potential LUC impacts at a given moment in time. Levels are compared using a stock-difference approach without necessarily going so far as a full implementation of Tier 2 and application of an amortization period. These synchronic sequences make it possible, in some cases, to construct virtual LUC based on plausible references (Zimmermann et al. 2013; Mello et al. 2014). However, this approach is limited by the availability of such references in comparable conditions for soils sharing the same inherent properties (Bailis and McCarthy 2011; Rasmussen et al. 2012), as well as by the failure to account for hysteresis effects linked to the history of the soil (i.e., the site effect).

The comparison of carbon stocks between natural levels and a given land use type can be expanded across different pedoclimatic and agronomic conditions *via* meta-analyses or statistical studies. Expansion to varied contexts and the inclusion of numerous parameters can potentially give rise to a wide variability of observations and requires multiple datasets for an analysis of determining factors. In the case of eucalyptus in Brazil, for example, a meta-analysis of 89 datasets showed that on average, eucalyptus did not lead to significant changes in SOC compared to natural vegetation, despite non-negligible gains and losses of SOC in certain cases (Fialho and Zinn 2014). By contrast, a statistical analysis of an experimental study based on 135 sites in the South-Central Region of Brazil (~6000 soil samples) found significant average effects from LUC involving the conversion of arable fields, grassland, or *cerrado* into sugarcane (Mello et al. 2014). Depending on the previous use and the number of observations, however, effects were not significant for all soil depths: e.g., after conversion from *cerrado* (5 sites), variations in carbon levels below 30 cm were not significant (Mello et al. 2014). Using a large dataset including LUC over longer timeframes, changes in carbon stocks were analyzed[1] and assembled over multiple time scales in five-year increments and then converted into a "land-use change factor" by soil depth and time period. Values obtained at 30 cm after 20 years corresponded to a complete implementation of the IPCC Tier 2. The different findings of these two studies may arise from real differences between the study contexts, or it may result from a lack of robustness emerging from insufficient sample sizes given the variability of the contexts, practices, and impacts over time and space.

4.3.2.2 Life-Cycle Assessment (LCA)

Life-cycle assessment (LCA) is a standardized methodological framework (ISO 14040 and 14,044, 2006) for multi-criteria assessment of the environmental impacts of a product or service. LCA makes it possible to quantify more than a dozen potential environmental impacts across the entire life cycle of a product, from the extraction of the primary materials through disposal of the product and its residues. This holistic life-cycle assessment approach has become essential, notably for the evaluation of bioenergy sectors, as a way of verifying that the environmental gains relative to fossil fuel use – in terms of carbon emitted into the atmosphere through combustion – are not cancelled out by other impacts, such as increased emissions of other gases during combustion or other emissions and impacts elsewhere in the commodity chain.

To include the whole commodity chain, the LCA must quantify the impacts of all resource uses and emissions at all stages and locations. These contributions are summed up independently of their various origins *via* linear models characterizing a potential final impact, without strong specificity to local circumstances (e.g., environmental sensitivity of the site, threshold effects, etc.). Some models make it possible to weight these different categories based on regionalized factors so as to better account for localized impacts, e.g., an index of water scarcity (Pfister et al. 2009). Nevertheless, LCA impact analyses indicate aggregated impacts calculated in parallel, providing an estimate of potential impacts at the global level.

Acidification impact (terrestrial)[2] as calculated via LCA represents a non-local potential impact for airborne emissions of ammonia, sulfur oxides and nitrous oxides. The relative contribution of the different gases varies according to characterization methods (ReCiPe, ILCD, CML, etc.), and not all methods necessarily

[1] The statistical approach applied was a linear model with mixed effects.

[2] Some methods also characterize aquatic acidification (e.g. IMPACT+2002), in which other substances are involved (e.g. phosphorous). Due to a lack of precision in some cases, acidification is commonly understood to mean "terrestrial acidification," and was correlated as a soil impact in the experts' analysis of the corpus.

include the fate of the substances in the air. Thus, the acidification impacts reported in the corpus correspond to theoretical impacts linked to the potential for acidification of the different emissions inventoried along the commodity chain. In the case of bioenergy sectors, inventoried emissions relate primarily to the use of nitrogen fertilizers and the combustion of diesel fuel for machinery (Cocco et al. 2014). Impacts are calculated linearly, regardless of the location or timing of emissions, and thus indicate the potential, overall, non-localized impact. The impact pathway leading to actual acidification of a soil takes time, and to date there is no model that allows for quantifying this impact at any given moment or location with the ability to highlight the contributions of a specific production system or activity. Thus, the potential impacts of a bioenergy production chain as determined via LCA provide only a minimal indication of the impacts of biomass crops on the soils and the overall environment directly hosting the crop under study.

The LCA land use impact category with the indicator "Biotic Production Potential" (Brandão and Milà i Canals 2013) is a partial exception to this disconnect from local conditions. The conceptual background to this impact was developed within the framework of thesis research on agricultural LCA (Milà i Canals 2003; Milà i Canals et al. 2007) and in response to a growing awareness, since the early 2000s, of the need to adapt better LCA for agricultural products (the LCA concept was initially developed for industrial products). The importance of soils and soil quality within the analysis of agricultural production drove the scientific community to develop new models for characterizing soil impacts (e.g., Cowell and Clift 2000; Lindeijer 2000; Weidema and Lindeijer 2001). Other methodologies have been developed both within and beyond this conceptual framework for land use impacts, so that today they are more or less complete and accessible (Nuñez et al. 2013; Saad et al. 2013; Garrigues et al. 2013). The LANCA© method (Bos et al. 2016), which is particularly complete, was recently recommended within the context of European harmonization of LCA characterization methods (Vidal Legaz et al. 2013).

These most recent developments are not reflected in the corpus (2001–2014). Of the studies in the corpus, 46% use LCA, 31% consider a soil impact or consider soils as an aspect of an LCA climate change impact, 6%, or just 3 articles, include the land use impact category, and 4%, or 2 articles, include various recent developments related to soil use and soil quality (Saad et al. 2013; Brandão and Milà i Canals 2013; de Baan et al. 2013 *In* Munoz et al. 2014; Helin et al. 2014). The "land-use impact" category is recommended in the European Union Research Center's ILCD[3]'s directives (JRC 2011), albeit with the caveat "to be used with caution". Reservations with regard to this impact category were twofold. The first concern related to the difficulty of implementation due to a lack of specific data on carbon stocks and the need to develop characterization factors on an *ad hoc* basis. These challenges explain the lack of results with respect to this impact category in the literature up to that point. LCA software now includes characterization factors based on default levels from the IPCC (Brandão and Milà i Canals 2013). A second

[3] International Reference Life Cycle Data System

concern related to the sole focus on soil carbon levels as a way of characterizing impacts on soil quality. Indeed, the land-use impact was originally defined as a proxy for the impact on soil quality of an agricultural or other type of land use. Soil quality is a broad concept that cannot be defined in a single way. Nevertheless, authors agree on the need to consider soil quality with regard to the expected functions to be provided by soils, and the connections between the physicochemical and the biological properties of a soil and its capacity to supply those functions (Doran et al. 2002; Karlen et al. 2003; Kibblewhite et al. 2008). The land-use impact is based on this reasoning and relies on the quantification of changes in soil carbon as an indicator of changes in soil organic matter, itself indicative of significant changes in a soil's capacity to supply various functions, particularly those relating to life and biological development (Milà i Canals et al. 2007). Variations in carbon stocks are thus expressed in terms of the "Biotic Production Potential" indicator. Authors point to the fact that organic matter levels have been shown to be a dynamic soil attribute indicative of various aspects of soil quality, including cation exchange capacity and biological activity (Reeves 1997; Brady and Weil 2002), and are thus the most useful way of evaluating impacts on the life-supporting capacity of soils for agricultural or forestry production, even if other aspects of soil quality also play a role (Milà i Canals et al. 2007).

As currently used in LCA softwares such as Simapro and OpenLCA, the land use impact category uses characterization factors that quantify variations in soil carbon levels based on the values and coefficients proposed by the IPCC[4] (IPCC 2006 Tier 1). As in the use of the IPCC Tier 2, these stock values may be modified by manually adjusting the characterization factors within the LCA software. On the other hand, the conceptual background for the land use impact category is not limited to a strict application of the stock-difference approach. The impact is calculated using two principal reference fluxes, "land transformation" and "land occupation." The first may be included in a "classic" land-use change impact, using a stock difference allocated over 20 years. The second, by contrast, quantifies a theoretical difference in quality relative to a reference state, which will not naturally rebuild itself so long as the land is in use. The definition of initial and reference quality states, which will critically influence fluxes, varies according to the objectives of the study and thus the LCA approach put in place. Initially, the complete conceptual framework also allowed one to take into account additional irreversible impacts or impacts linked to a change in quality directly during the land occupation. In practice, these impacts are not implemented. On the one hand, the use of the IPCC stock values assumes an equilibrium state tied to each type of land use, which does not fit with the calculation of quality-sensitive variations around equilibria during land occupation. On the other hand, irreversible impacts are difficult to identify *a priori* and are generally not considered due to a lack of data and a lack of consensus.

In theory, the land use impact's conceptual framework allows for a more complete characterization of soil quality impacts based on other fluxes connecting changes in soil properties to changes in soil functions. Nevertheless, at present LUC

[4] According to the stock-difference approach detailed in Sect. 4.3.2.1.

impacts are embedded in land transformation and occupation impacts based on IPCC data, and thus reflect the soil quality impact in terms of soil carbon stock difference only. This stock difference is also used to quantify inventory fluxes for the climate change impact category, although the alignment of these inventory fluxes and impact categories is not always clear.

4.3.2.3 Biophysical Modeling

Approximately 20% of the articles in the corpus make use of mechanistic models to characterize soil impacts. Most of these models are one of two types: those oriented toward the modeling of physicochemical and hydric processes in the soil, with a focus on erosion, water, and SOC levels (including USLE,[5] SWAT,[6] GORCAM, RothC, ICBM, C-Tool, and the Matthews and Grogan model); or more integrative models, including some that aim to provide a full agroecosystem simulation (CROPWAT, MISCANMOD, CENTURY,[7] CERES-EGC, EPIC,[8] SECRETS[9]) or others that attempt to integrate a sector (GREET[10]). These different models can interact, e.g., USLE is used in EPIC, CENTURY is used in GREET, etc. The CROPWAT[11] and MISCANMOD[12] models are not full agroecosystem models because they do not allow for a simulation of losses to the environment.

Modeling Specific to Soils

Various models for soil function simulations are used in the corpus. Most are so-called mechanistic or process-based models, although some may also include some empirical correlations. Two principal models are used for modeling the physical and hydric processes in soils, especially erosion risks: SWAT, used in Garcia-Quijano et al. (2005), Babel et al. (2011) Wu et al. (2012), and Hoque et al. (2014); and USLE/MUSLE/RUSLE, used in van Dam et al. (2009), Smeets and Faaij (2010),

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[5] Universal Soil Loss Equation (USLE) http://www.omafra.gov.on.ca/english/engineer/facts/12-051.htm, last consulted January 15, 2017.

[6] Soil and Water Assessment Tool (SWAT) Arnold et al. 1998. A description of this model's parameters (calibration, validation, and performance) can be found in Cibin et al. (2012); a sensitivity analysis can be found in Heuvelmans et al. (2005).

[7] Metherell et al. 1993.

[8] Environmental Policy Integrated Climatic (EPIC) model (Williams 1990), previously known as Erosion Productivity Impact Calculator: http://epicapex.tamu.edu/files/2013/02/epic0509usermanualupdated.pdf

[9] Stand to Ecosystem CaRbon and EvapoTranspiration Simulator (SECRETS) is a mechanistic model for the simulation of forest cover (Sampson and Ceulemans 1999; Sampson et al. 2001).

[10] Argonne National Laboratory's Greenhouse Gases, Regulated Emissions, and Energy use in Transportation (GREET™): GREET1_2012. http://greet.es.anl.gov/main

[11] FAO: http://www.fao.org/nr/water/infores_databases_cropwat.html

[12] MISCANMOD by Clifton-Brown et al. 2000; Jain et al. 2010.

Secchi et al. (2011), and Debnath et al. (2014). Next in importance are various more or less mechanistic models used specifically to simulate SOC dynamics, i.e. RothC, which is used in two studies (Cherubini and Ulgiati 2010; and Brovkin et al. 2013); and some less widely known models, including the Matthews and Grogan's model (2001), developed specifically for energy crops and used in Styles and Jones (2008) and Mishra et al. (2013); ICBM, used in Tidaker et al. (2014); C-Tool, used in Hamelin et al. (2014); and GORCAM, used in Garcia-Quijano et al. (2005).

USLE and SWAT are the most frequently used models. MUSLE and RUSLE are the "modified" and "revised" versions of USLE, respectively. SWAT is a model designed to assess the long-term effects of land use via the aggregation of units of hydrological response within a watershed. This spatial approach is useful for analyzing the impact of alternative land-use scenarios (Garcia-Quijano et al. 2005). SWAT is widely used,[13] notably because of its flexibility in the choice of calculation methods for evapo-transpiration; the ability to select climate data or have it be generated automatically; the availability of a land use database including a wide range of plant species; and the ability to select different time periods for model outputs for the movement of sediments, nutrients (including four types of nitrogen, total nitrogen, two forms of phosphorous, and total phosphorous), and pesticides (Heuvelmans et al. 2005; Hoque et al. 2014). SWAT is based in part on empirical relationships, some of them from MUSLE, mainly derived from experiments conducted in the United States. Its use outside this area of validation is to be considered with caution (Heuvelmans et al. 2005).

USLE and its later versions are likewise widely used, notably as sub-models within more integrative models such as EPIC. USLE relies on a rudimentary equation to determine waterborne erosion as a product of the erosion risk factors of precipitation (R), soil erodibility (K), length (L) and degree (S) of slope, management of soil cover (C), and anti-erosion practices (P). Parameters R and K nevertheless require datasets based on extended time periods (at least 20 years of continuous climate data) for application in pedoclimatic conditions distant from the initial areas of validity (Devatha et al. 2015). The modified equation MUSLE also takes into account the volume and maximum rate of run-off as well as a factor linked to large soil fragments (Zhang et al. 2010).

Agroecosystem Modeling

Agroecosystem models such as CERES-EGC (Gabrielle et al. 1998; Goglio et al. 2013; Gabrielle et al. 2014) generally combine several sub-models or modules to enable a modeling of the principal processes acting within and at the interface of the soil-plant-atmosphere compartments. These modules thus allow for the modeling of physicochemical and hydric soil processes, microbial processes and variations in

[13] SWAT literature database, https://www.card.iastate.edu/swat_articles/, last consulted January 15, 2017.

SOC levels, the development of vegetative cover, and emissions to the environment.

CERES-EGC, used in one article in the corpus (Gabrielle et al. 2014), makes it possible to simulate losses of reactive nitrogen (N_2O, NO, NH_3, NO_3^-) in addition to yields and carbon dynamics. The EPIC models, used in 3 articles (Zhang et al. 2010; Secchi et al. 2011; Debnath et al. 2014), and CENTURY, used in 2 articles (Rasmussen et al. 2012; Dunn et al. 2013), also allow for simulations of agroecosystem functioning, including emissions of nitrogen and phosphorous into the environment. The latter two articles actually only make use of a sub-model within CENTURY relating to SOC dynamics. Besides, the latest versions of the EPIC model also contain routines for simulating SOC that come from CENTURY.

The mechanistic approach requires detailed datasets reporting on the full range of relevant parameters at a sufficiently detailed timescale (e.g., daily). These may be input data (e.g. temperature, precipitation) or fixed parameters determining system properties (e.g., field capacity, variety characteristics) or initial conditions (e.g., level of mineral nitrogen in the soil). Availability of all the necessary data is often a limiting factor in the use of a mechanistic model. In particular, the lack of data for calibrating fixed parameters strongly restricts the use of a model outside its validity domain as initially calibrated. In an example with switchgrass production in Oklahoma, in the United States, the model used (EPIC) could not be calibrated for SOC, with the result that the findings with respect to SOC were not readily useable and, as the authors admit, were not consistent with the literature (Debnath et al. 2014). Similarly, in a study conducted in Iowa on LUC linked to increases in maize acreage based on modeling with EPIC, some data for the soil parameters could not be compiled, leading to a potential underestimation of environmental risks in the five counties considered (Secchi et al. 2011). In another example, in Mozambique, the physiological parameters could not be calibrated for jatropha and thus limited the scope of the CENTURY model for approximating the temporal dynamics of SOC losses (Rasmussen et al. 2012).

Where data are available, mechanistic models make it possible to simulate crop cycles over long timeframes, enabling one to assess the variability and robustness of the results while accounting for inter-annual variations. The corpus includes studies with simulations over 20 years (Gabrielle et al. 2014), 30 years (Secchi et al. 2011; Tidaker et al. 2014), 50 years with 10 climate scenarios (Debnath et al. 2014), and 150 years (Garcia-Quijano et al. 2005). Long-term simulations are particularly important when considering long-term dynamics such as SOC, or in the study of perennial crops. Comparing results from the first and the second crop cycles of eucalyptus in Brazil, for example, revealed notable differences (Fialho and Zinn 2014).

4.3.3 Results

4.3.3.1 Overview of Impacts Examined

Considering the full range of potential impacts, we can observe that impacts are poorly reported overall (Table 4.2). SOC impacts apart, over 67% of impacts are not reported. It is thus impossible to draw broad conclusions for a given impact type. Nor is there an observable trend with regard to impacts across all commodity chains and LUC types. Of particular note is the fact that levels of SOC and SOM do not show strongly correlated trends, despite the fact that they are intrinsically connected. The lack of information on impacts is potentially at the root of this disjuncture, with (for example) 81% of potential impacts on organic matter levels not studied or not reported. Nevertheless, a more detailed analysis with regard to commodity chains and LUC types is needed to interpret better the slight downward trend of organic carbon levels, and the more heterogeneous results observed for other impacts: acidification, erosion, and organic matter levels.

4.3.3.2 Impacts Quantified

Impacts quantified in terms of variations in SOM or SOC (Table 4.3) and erosion (Table 4.4) are reported by commodity chain. Only those scenarios explicitly addressing a bioenergy chain were examined. Among these scenarios, those considering only a change in practice (e.g., export of crop residues) and not a change in land allocation (in the strict sense of a direct or indirect LUC) were excluded from

Table 4.2 Overview of land use change (LUC) impacts on soils

Impact	*Not reported/not studied/studied but not reported*	*Decrease*	*Stable*	*Increase*	*Variable*
Level of organic carbon	**54%**	23%	4%	6%	13%
Acidification	**67%**	10%	2%	17%	4%
Erosion	**77%**	12%	-	8%	4%
Level of organic matter	**81%**	10%	2%	4%	4%
Trace metallic elements	**96%**	-	-	2%	2%
Compaction	**98%**	-	-	-	2%
Organic contamination	**98%**	-	-	-	2%
Biological pollution	**98%**	2%	-	-	-

NB: Totals ≠ 100% due to rounding

Cell shading corresponds to percentage totals: Light blue: >15–30%; Gray: >30–45%; Dark gray: >45%

Table 4.3 Impacts on soil carbon reported in case studies of biomass production for bioenergy (sector × scenario)

Crop resulting in an LUC	Country species	Level of soil organic carbon (SOC) or soil organic matter	References
Cardoon	Italy *Cynara cardunculus* var. *altilis* DC	Direct LUC from marginal abandoned lands (grasslands: 53.7 t C/ha) to cardoon cultivation (50.3 t C/ha) leads to a loss of SOC. This loss is recovered after 100 years. Soil depth considered: 20 cm.	Cocco et al. (2014)
Corn	United States *Zea mays* L.	Domestic LUC primarily involve changes from arable land to corn (50%), from grassland to corn (35%) or from woodland to corn (15%). Orders of magnitude for land area affected by LUC domestically and internationally are similar. International LUC involve conversions of grassland (~ 82%) and reforestation (~ 8%), primarily in Russia. Conversion from forests into corn production (with removal of stover) results in SOC losses of 0.2–0.6 t C/ha/year; from grasslands into corn, in SOC losses of 0.02–0.22 t C/ha/year, and from arable lands into corn in no SOC loss or even a potential gain of 0.12 t C/ha/year. Soil depth considered: 30 cm.	Dunn et al. (2013)
		Direct LUC from conservation grassland (CRP[a]) into corn-soybean or corn-corn-soybean rotations result in average losses of approximately 18.6 t C/ha over 30 years; LUC from CRP into continuous corn result in average losses of approximately 32.2 t C/ha over 30 years.	Secchi et al. (2011)
		Corn cultivation results in a loss of SOC, land use transformation and land occupation impacts combined, of 7 t C/ha/year, with or without removal of stover (theoretical calculations based on the "Land Use" approach and "BPP" indicator of Milà i Canals et al. 2007; Brandão and i Canals 2013). According to FAO statistics and the approach of Milà i Canals et al. 2012, this crop does not lead to LUC. Soil depth considered: 30 cm.	Munoz et al. (2014)
Eucalyptus	Brazil *Eucalyptus* spp.	Compared to natural vegetation (forest, grassland, and savanna) across three biomes (cerrado, pampa, Mata Atlântica), eucalyptus plantations result, on average, in a loss of SOC in the top 20 cm of 1.5 t C/ha, but a gain of SOC in the top 40 cm of 0.3 t c/ha. These averages, taken over 50 studies and 39 studies, respectively, are not statistically significant. Average losses over the first plantation cycle are balanced by gains in the second plantation cycle (on average over the 2nd cycle, +2.3 t C/ha at 0–20 cm, and +3.1 t C/ha at 0–40 cm), but again these results are not statistically significant.	Fialho and Zinn (2014)

(continued)

Table 4.3 (continued)

Crop resulting in an LUC	Country species	Level of soil organic carbon (SOC) or soil organic matter	References
Jatropha	India *Jatropha curcas*	Conversion of *Prosopis juliflora* (mesquite) groves into jatropha plantations results in a loss of SOC of 0.5 t C/ha after 4 years. This reduction is not statistically significant. Reduced leaf litter relative to the preceding land use may lead to a more significant loss of SOC over the long term. Soil depth considered: 30 cm.	Bailis and McCarthy (2011)
	Mozambique *Jatropha curcas*	Direct LUC from corn (85±14 t C/ha at 0–60 cm) to jatropha (92±7 t C/ha at 0–60 cm) leads to no significant change in SOC. LUC from forest (210±17 t C/ha at 0–60 cm) to jatropha, however, leads to definite losses (modeled with CENTURY at 0.7%/year based on a corn-fallow proxy). Nevertheless, according to the authors, variations in measured SOC levels under forest may be more strongly influenced by soil type than by the LUC. Soil depths considered: 20, 40, 60 cm.	Rasmussen et al. (2012)
Miscanthus	Several European countries *Miscanthus* x *giganteus*	Conversion from grassland into miscanthus leads to limited initial losses of SOC (not significant according to the statistical model), followed by an increase, resulting in a net gain already in the first years. Maximum soil depth considered: 60–100 cm.	Anderson-Teixeira et al. (2009)
	United States *Miscanthus* x *giganteus*	Domestic LUC primarily involve changes from arable land to miscanthus (96%) or forest to miscanthus (4%). International LUC involve 17 times less land area and are exclusively changes from grassland. A small amount of reforestation is simulated (< 2%). Domestic conversion into miscanthus from forests leads to gains in SOC from 0.1 to 0.18 t C/ha/year, from grassland to gains of 0.38–0.48 t C/ha/year, from arable land to gains of 0.55–0.65 t C/ha/year. Soil depth considered: 30 cm.	Dunn et al. (2013)
		LUC of cultivated lands (all cultivated land in the US without differentiation) into miscanthus results in a potential SOC sequestration of 0.16–0.81 t C/ha/year; or 0.6 t C/ha/year on average across the United States. This result is not compared to initial SOC levels for potentially converted cultivated lands, since this LUC is not explicitly analyzed.	Mishra et al. (2013)

	United Kingdom *Miscanthus* x *giganteus*	Miscanthus cultivation results in an increase in SOC during land occupation relative to the average initial level (+0.62 t C/ha/year) and an opposite virtual impact attributable to the delay in reestablishment of natural vegetation. The total annual impact amounts to a loss of 40.3 t C/ha/year (i.e. the LCA characterization factor within the "Land Use" impact category (Milà i Canals et al. 2007), with 80 t C/ha in the initial state (i.e. arable land), 150 t C/ha at the potential natural state (i.e. warm-temperate forest), and a potential regeneration rate of 0.32 t C/ha/year. Soil depth considered: 30 cm.	Brandao et al. (2011)
	France *Miscanthus* x *giganteus*	Conversion of arable land or fallow into miscanthus results in an average increase in SOC of 0.58 t c/ha/year according to a simulation over 21 years with CERES-EGC.	Gabrielle et al. (2014)
	Belgium *Miscanthus* x *giganteus*	Conversion of arable land (72% annual crops + 26% permanent grassland + infrastructure) into miscanthus results in a total average increase in SOC of 45.4 t C/ha after a simulation of 150 years.	Garcia-Quijano et al. (2005)
	Ireland *Miscanthus* x *giganteus*	Short-rotation miscanthus cultivation results in an increase in SOC of 11.6 t C/ha/year in the case of an LUC from plowed land coming out of fallow, but no change in the case of an LUC from grassland.	Styles and Jones (2008)
		Conversion of arable plowed land or grassland into miscanthus resulted in no significant change in SOC on the observed sites (2 for each type of LUC) 3–4 years after the planting of miscanthus. Following grassland, SOC under miscanthus was sometimes higher than the control (~114 t C/ha > ~81 t C/ha), sometimes lower (~99 t C/ha < ~113 t C/ha). Following plowed land, SOC under miscanthus was slightly higher, i.e. 44–48 t C/ha > 36–39 t C/ha. Plowing history had a significant influence on contrasting SOC levels across the different sites. Soil depth considered: 30 cm.	Zimmermann et al. (2013)
Mustard	Italy *Brassica carinata* A. Braun	Direct LUC from marginal abandoned land (grassland: 53.7 t C/ha) into mustard cultivation (44.1 t C/ha) results in a loss of SOC. This loss is restored over 100 years rather than over 20 years as per the default IPCC recommendations (2006). Soil depth considered: 20 cm	Cocco et al. (2014)
Poplar	Belgium *Populus spp.*	Conversion of arable land (72% annual crops + 26 % permanent grassland + infrastructure) into poplar plantations results in an average total increase in SOC of 59.7 t C/ha after simulation of 150 years.	Garcia-Quijano et al. (2005)

(continued)

Table 4.3 (continued)

Crop resulting in an LUC	Country species	Level of soil organic carbon (SOC) or soil organic matter	References
Rapeseed	United Kingdom *Brassica napus*	Rapeseed production (with straw returned to the soil) results in a loss of SOC for the duration of land occupation relative to an average initial level (–0.24 t C/ha/year; –0.40 t C/ha/year if straw is removed), and an estimated additional "virtual" impact resulting from the delay in potentially recovering the initial natural vegetation. The total annual impact is equal to a loss of 122.7 t C/ha/year (i.e. the LCA characterization factor within the "Land Use" impact category [Milà i Canals et al. 2007]), with 80 t C/ha at the initial state and 150 t C/ha at the natural potential state (i.e. warm-temperate forest), and a potential regeneration rate of 0.32 t C/ha/year. Soil depth considered: 30 cm.	Brandao et al. (2011)
	Italy *Brassica napus* L. var. *oleifera* DC	Direct LUC from marginal abandoned lands (grasslands: 53.7 t C/ha) to rapeseed cultivation (44.7 t C/ha) leads to a loss of SOC. This loss is restored over 100 years. Soil depth considered: 20 cm.	Cocco et al. (2014)
	Chile *Brassica napus* L.	Direct LUC from highly degraded grasslands into no-till rapeseed production results in an increase in SOC of 50 kg C/ha/year. This value is based on the use of IPCC Tier 1, adding parts of the gains allocated to the four crops in the rotation. Soil depth considered: 30 cm.	Iriarte et al. (2012)
Reed canary grass	Scandinavia *Phalaris arundinacea* spp.	Cultivation of reed canary grass results in a virtual reduction in SOC of 263 t C/ha/year due to the absence of natural vegetation during the land occupation period. The impact from conversion of forest adds a loss of 132 t C/ha/year, whereas conversion of a previously developed area restored to agricultural use results in a restocking of 1432 t C/ha/year, occupation impact included (theoretical calculations based on the "Land Use" approach and "BPP" indicator of Milà i Canals et al. 2007; Brandão and i Canals 2013). Soil depth considered: 30 cm.	Helin et al. (2014)
Sitka spruce	United Kingdom *Picea sitchensis*	Use of forestry residues results in no change in SOC during the period of land occupation relative to an average initial level (130 t C/ha) but the total annual impact, including the impact of the delay in reestablishment of natural vegetation, amounts to a loss of 20 t C/ha/year (i.e. the LCA characterization factor within the "Land Use" impact category (Milà i Canals et al. 2007) with 130 t C/ha in the initial state (wood) and 150 t C/ha in the natural potential state (i.e. warm-temperate forest) and a potential regeneration rate of 0.32 t C/ha/year). Soil depth considered: 30 cm.	Brandao et al. (2011)

Soybean	United States *Glycine max* L. Merr.	Conversion of conservation grassland (CRP[a]) dominated by *Bromus inermis* (smooth brome) into no-till soybean production (GMO cultivar 92M91) has no significant effect on carbon levels after one year. However, comparison with lands in corn-soybean production for more than 10 years suggests that these soils hold less carbon and generally show a lower quality index relative to conservation grassland soils. Soil depth considered: 100 cm.	Bhardwaj et al. (2011)
		Conversion of conservation grassland (CRP[a]) dominated by *Bromus inermis* (smooth brome) into no-till soybean production results in a soil carbon deficit of 13.7 t C/ha compared to the potential additional sequestration of such grasslands after 22 years (assuming they are not converted). Soil depth considered: 29 cm.	Gelfand et al. (2011)
	Argentina *Glycine max* L. Merr.	Direct LUC from degraded or non-degraded grasslands into reduced-tillage soybean production results in SOC losses of approximately 0.65 and 0.7 t C/ha/year, respectively. Change of species on arable lands is not considered an LUC; nevertheless, according to the IPCC method, shifting to no-till for the same crop results in an increase in SOC (+0.09 t C/ha/year in this case).	van Dam et al. (2009)
		Indirect LUC is supposed to lead to a displacement of the replaced crops to the detriment of natural grasslands. Two displacement scenarios (25% and 50% of the total area involved in the direct LUC) and two calculation methods (total crop or oil-content equivalent) are considered. Indirect LUC thus results in an SOC loss of 0.04–0.35 t C/ha/year. Soil depth considered: 30 cm.	
Sugarbeet	France *Beta vulgaris* ssp.	Sugar beet production results in a loss of SOC, land use transformation and land occupation impacts combined, of 7.1 t C/ha/year (theoretical calculations based on the “Land Use” “BPP” indicator of Milà i Canals et al. 2007; Brandão and Milà i Canals 2013). According to FAO statistics and the approach used by Milà i Canals et al. (2012), this crop does not lead to LUC. Soil depth used: 30 cm.	Munoz et al. (2014)

(continued)

Table 4.3 (continued)

Crop resulting in an LUC	Country species	Level of soil organic carbon (SOC) or soil organic matter	References
Sugarcane	Brazil *Saccharum officinarum*	Direct LUC of grasslands into sugar cane production results in a loss of SOC according to IPCC (2006) Tier 1 coefficients. Indirect LUC results in conversion of Amazonian forest into grassland, again resulting in a loss of SOC according to IPCC Tier 1 coefficients.	Alvarenga et al. (2013)
		The authors conclude that direct LUC of arable lands (including grasslands) into sugar cane cultivation has a net zero effect on SOC. However, an LUC from natural cerrado vegetation leads to emissions of 1040 kg CO_{2eq}/m^3 ethanol (data are not detailed enough to calculate the impact on SOC). Indirect LUC is included in this study but is based on emissions factors per unit of biofuel as drawn from the literature, and are not specified according to the preceding land use.	Liptow and Tillman (2012)
		Direct LUC of native cerrado vegetation into sugar cane cultivation leads to a significant reduction of SOC at depths of 0–30 cm, on the order of 21 t C/ha over 20 years. The impact below 30 cm is presumed to be not significant.	Mello et al. (2014)
		Direct LUC from grasslands to sugar cane cultivation leads to a significant reduction of SOC, on the order of 6 t C/ha at depths of 0–30 cm over 20 years and 8.7 t C/ha at depths of 0–100 cm over 20 years.	
		Direct LUC from annual crops to sugar cane cultivation leads to a significant increase in SOC, on the order of 10 t C/ha at depths of 0–30 cm over 20 years and 24.6 t C/ha at depths of 0–100 cm over 20 years.	
		Sugar cane production results in a loss of SOC, land use transformation and land occupation impacts combined, of 6.9 t C/ha/year in the North-East and 10.5 t C/ha/year in the South-Central region (theoretical calculations based on the "Land Use" "BPP" indicator of Milà i Canals et al. 2007; Brandão and i Canals 2013). According to FAO statistics and the approach of Milà i Canals et al. 2012, for each hectare of sugar cane, 0.032 ha of forest is indirectly converted. Soil depth considered: 30 cm.	Munoz et al. (2014)
	Various countries *Saccharum* spp. L.	Direct LUC from native ecosystems (forest and grassland) to sugar cane production leads to a sharp initial drop in SOC, which will be recovered 60 years after the change in use (using an interval of 100 years). Soil depths considered: 70–100 cm.	Anderson-Teixeira et al. (2009)

Switchgrass	United States *Panicum virgatum* L.	Direct LUC from cultivated land, fallow, or grassland into switchgrass shows a tendency to increase SOC, but data are insufficient for this species to test for statistical robustness. Maximum soil depth considered: 90–360 cm.	Anderson-Teixeira et al. (2009)
		Direct LUC from arable lands into switchgrass leads to SOC storage of 4.42 t C/ha/year; from marginal lands (arid or degraded lands in CRP[a]) into switchgrass, to storage of 3.2 t C/ha/year; and from grasslands, to storage of 0.32 t C/ha/year (values drawn from the literature).	Cobuloglu and Buyuktahtakin (2014)
		Direct LUC from no-till wheat into switchgrass (5 cycles of 10 years of production modeled with stochastic climate data) results in an increase in SOC in all counties modeled (including 3 "land capability classes," with slopes from 0.5% to 4%), i.e. +112.1–531.4 kg C/ha/year. The EPIC model could not be calibrated for SOC, however.	Debnath et al. (2014)
		Domestic LUC primarily involves changes from arable land to switchgrass (92%) followed by woodland into switchgrass (8%). International LUC involves 12 times less land area and are primarily conversions from grassland (~ 90%) and forest (~ 10%). Domestic conversion from forest into switchgrass results in SOC losses of 0.01–0.1 t C/ha/year; from grasslands, in SOC gains of 0.2–0.25 t C/ha/year; from arable lands, to gains of 0.35–0.45 t C/ha/year. Soil depth considered: 30 cm.	Dunn et al. (2013)
	Argentina *Panicum virgatum* L.	Direct LUC from arable lands into switchgrass result in an increase in SOC of approximately 0.7 t C/ha/year; from degraded grasslands, the increase is only 0.04 t C/ha/year. LUC from non-degraded grasslands into switchgrass is considered to have no impact on SOC.	van Dam et al. (2009)
		Indirect LUC is thought to result in a displacement of the replaced crops to the detriment of natural grasslands. Two displacement scenarios were considered, corresponding to 25% and 50% of the total area affected by the direct LUC. The indirect LUC thus results in an SOC release of 0.2 and 0.36 t C/ha/year respectively. Soil depth considered: 30 cm.	
Temporary grassland	Sweden mixed grasses and clover	Introduction into a cereals rotation of 2 consecutive years of temporary grassland, cut to produce biogas, leads to an annual average accumulation of 0.19 t C/ha compared to 0.02 t c/ha in the initial rotation, according to simulations over 30 years. Soil depth considered: 25 cm.	Tidaker et al. (2014)

(continued)

Table 4.3 (continued)

Crop resulting in an LUC	Country species	Level of soil organic carbon (SOC) or soil organic matter	References
Wheat	France *Triticum* spp.	Wheat production leads to a loss of SOC, land use transformation and land occupation impacts combined, of 9.4 t C/ha/year (theoretical calculations based on the "Land Use" "BPP" indicator of Milà i Canals et al. 2007; Brandão and i Canals 2013). According to FAO statistics and the approach used by Milà i Canals et al. 2012, for each hectare of wheat, 0.016 ha of grassland is directly converted. Soil depth considered: 30 cm.	Munoz et al. (2014)
Willow	United Kingdom *Salix* spp.	Cultivation of short-rotation willow results in an increase of SOC relative to an average initial level (+0.14 t C/ha/year) during land occupation and an opposite virtual impact due to the delay in reestablishment of natural vegetation. The total annual impact amounts to a gain of 65.3 t C/ha/year (i.e. the LCA characterization factor within the "Land Use" impact category [Milà i Canals et al. 2007]), with 80 t C/ha in the initial state (i.e. arable lands), 150 t C/ha in the potential natural state (i.e. warm-temperate forest), and a potential regeneration rate of 0.32 t C/ha/year. Soil depth considered: 30 cm.	Brandao et al. (2011)
	Ireland *Salix* spp.	Cultivation of short-rotation willow results in an increase in SOC of 0.5 t C/ha/year in the case of LUC from soils formerly plowed and then put in and out of fallow, but no change in the case of LUC from grasslands.	Styles and Jones (2008)

Quantified results, notably impacts from life-cycle analysis case studies, depend on the boundary of system studied and cannot be repurposed without first verifying their area of validity. Soil depths are only mentioned when explicitly stated in the article or deducible from the methodological approach utilized. The term "grassland" is generally considered to mean "permanent grassland," as opposed to "pasture," which can be understood as temporary grassland where it is associated with the land-use type "cropland," i.e. "arable land"

[a]*CRP* Conservation Reserve Program in the United States

Table 4.4 Impacts on soil erosion reported in case studies of biomass production for bioenergy (sector × scenario)

Crop resulting in an LUC	Country species	Erosion	References
Corn	United States *Zea mays* L.	Corn production leads to erosion risks of 13 t C/ha/year with or without removal of stover (theoretical calculations based on the "Land Use" framework [Milà i Canals et al. 2007; Saad et al. 2013]).	Munoz et al. (2014)
Eucalyptus	Brazil *Eucalyptus* spp.	Eucalyptus plantations in Brazil (precipitation 1000–2000 mm/year) results in losses by erosion 1–47 t/ha/year for finely or medium-textured soils on slopes of 2–10%; e.g. losses of 8–47 t/ha/year on medium-textured soils on slopes of 6–10% (modeling with USLE). According to data drawn from the literature, eucalyptus plantations have 0.06–0.14 times the erosion risk of seasonal horticultural crops or cereals, and 1–7.14 times the erosion risk of preceding grasslands or forests.	Smeets and Faaij (2010)
Manioc	Thailand *Manihot esculenta* Crantz	All conversions into manioc plantations (from forest, rubber trees, orchards, mixed) result in an increase in surface erosion (+9 to 75 %).	Babel et al. (2011)
Oil palm	Thailand *Elais guineensis* Jacq.	Conversion of forest or of a mixture of forest/orchard/rubber trees into oil palm plantations result in an increase in surface erosion (+13%). By contrast, conversion of orchard, rubber trees, or a mixture of orchard/rubber trees into oil palm plantations results in a slight decrease in erosion (−1%).	Babel et al. (2011)
Poplar	Ukraine *Populus spp.*	Poplar plantations in the Ukraine (precipitation 400–600 mm/year) result in losses by erosion of 0–11 t/ha/year for finely or medium-textured soils on slopes of 2–10%; e.g. losses of 3–15 t/ha/year on medium-textured soils with slopes of 6–10% (modeling with USLE). According to data drawn from the literature, poplar plantations have 0.16–0.22 times the erosion risk of seasonal horticultural crops or cereals and 1.55–11.09 times the erosion risk of preceding grasslands or forests.	Smeets and Faaij (2010)
Reed canary grass	Scandinavia *Phalaris arundinacea* ssp.	Production of reed canary grass leads to erosion risks during occupation of 26 t C/ha/year relative to natural vegetation. Impacts linked to the conversion of forest add an erosion potential of 56 t C/ha/year, whereas conversion of developed land restored to agricultural use leads to a reduction in erosion risks of 447 t C/ha/year (theoretical calculations based on the "Land Use" framework [Milà i Canals et al. 2007; Saad et al. 2013]). Soil depth considered: 30 cm.	Helin et al. (2014)

(continued)

Table 4.4 (continued)

Crop resulting in an LUC	Country species	Erosion	References
Soybean	United States *Glycine max* L. Merr.	Conversion of grasslands into no-till soybean production has no significant effect on soil erodibility after one year. The effect is more important over the long term, however (same conclusion as for soil carbon).	Bhardwaj et al. (2011)
	Argentina *Panicum virgatum* L.	Direct LUC of arable lands and non-degraded grasslands into reduced-tillage soybeans results in an increase in soil losses to erosion of 2.3 and 2 t C/ha/year, respectively; the same LUC from degraded grasslands results in an increase of 3.2 t C/ha/year.	van Dam et al. (2009)
Sugarbeet	France *Beta vulgaris* ssp.	Sugar beet cultivation results in erosion risks of 15.5 t C/ha/year (theoretical calculations based on the "Land Use" framework [Milà i Canals et al. 2007; Saad et al. 2013]).	Munoz et al. (2014)
Sugarcane	Thailand *Saccharum officinarum*	All conversions into sugar cane plantations (from forest, rubber trees, orchards, mixed) result in an increase in surface erosion (+10 to 74%).	Babel et al. (2011)
	Brazil *Saccharum officinarum*	Sugar cane production results in erosion risks of 3.4 t C/ha/year in the North-East region and 8.3 t C/ha/year in the South-Central region (theoretical calculations based on the "Land Use" framework [Milà i Canals et al. 2007; Saad et al. 2013]).	Munoz et al. (2014)
Switchgrass	United States *Panicum virgatum* L.	Direct LUC from arable lands, grasslands, or marginal lands into switchgrass is believed to entirely stop the erosion observed with these land use types.	Cobuloglu and Buyuktahtakin (2014)
		Direct LUC from no-till wheat into switchgrass (5 cycles of 10 years of cropping, modeled with stochastic climate data) results in a reduction of erosion in all counties modeled (with 3 soil classes distributed according to their productive potential and erosion risks; i.e. "land capability classes" with slopes from 0.5% to 4%), equivalent −0.4 to −5.5 t soil/ha/year.	Debnath et al. (2014)
	Argentina *Panicum virgatum* L.	Direct LUC of arable lands and degraded grasslands into switchgrass results in a reduction in soil losses to erosion of 4 and 5.3 t C/ha/year, respectively; from non-degraded grasslands, in an increase of 0.5 t C/ha/year.	van Dam et al. (2009)
Wheat	France *Triticum* spp.	Wheat cultivation results in erosion risks of 21.9 t C/ha/year (theoretical calculations based on the "Land Use" framework [Milà i Canals et al. 2007; Saad et al. 2013]).	Munoz et al. (2014)

Quantified results, notably impacts from life-cycle analysis case studies, depend on the boundary of system studied and cannot be repurposed without first verifying their area of validity. Soil depths are only mentioned when explicitly stated in the article or deducible from the methodological approach utilized. The term "grassland" is generally considered to mean "permanent grassland," as opposed to "pasture," which can be understood as temporary grassland where it is associated with the land-used type "cropland," i.e. "arable land" the table. Scenarios not allowing for the individual quantification of soil impacts (e.g., soil carbon separated from total biomass) were likewise excluded.

Out the 52 articles in the corpus, 37 consider a potential impact on SOC, with 26 presenting numerical results that make it possible to identify an LUC and its potential SOC impacts (corresponding to 50% of the corpus or 70% of those articles addressing an SOC impact). Erosion impacts are addressed in an explicitly quantified way by commodity chain in 8 articles (15% of the corpus). The commodity chains most often considered, and in the most detail, with respect to SOC impacts are sugarcane in Brazil, maize and switchgrass in the United States, and miscanthus, primarily in Europe. The most widely considered final product is first- and second-generation ethanol. Results vary considerably by chain, and even within a given chain, depending on the study context and the methods used to characterize the impact, especially with respect to initial stocks and the timeframes considered. Because of the prevalence of the SOC impacts within the corpus as a whole, the mechanisms analyzed (detailed in the following Sect. 4.3.3.3) likewise primarily relate to SOC.

Impacts quantified in terms of acidification are also reported, and are extracted from 5 articles, or barely 10% of the corpus, all of them using LCA (Table 4.5). Nevertheless, contributions to this impact come from various points in the commodity chain, including fertilization for biomass production, transport, transformation, etc., to the extent that it is not always possible to distinguish the specific contribution of the LUC, or even of any step directly linked to the agricultural or forestry phases. Thus, acidification impacts not calculated per hectare are not reported in the table. Besides, although partly linked to earlier phases in the commodity chain (e.g., the production of biomass), the characterization of the acidification impact barely makes it possible to identify impacts on the soil directly where the biomass is grown (see Sect. 4.3.2.2). Biomass crops grown for bioenergy give rise to emissions that can potentially cause acidification, principally by means of the loss of volatile nitrogenous compounds in the production and application of fertilizers and from fuel use for mechanical field operations. At the same time, these emissions may result in an acidification impact elsewhere than on the soil where the biomass is grown.

4.3.3.3 Mechanisms Involved

LUC impacts are the result of interactions between two sets of processes: those attributable to the change in soil cover and associated effects at the soil surface (e.g., erosion, run-off) or below the soil surface (e.g., rooting, infiltration, absorption); and those attributable to management practices associated with the change in land use (e.g. drainage, soil tillage, fertilizer applications, etc.). Some soil impacts are thus intrinsically related to the type of land-use (e.g. a more or less dense vegetation cover, strongly rooted or weakly rooted, annual or perennial, etc.), while other impacts are determined by interactions between the type of land use and management practices (e.g. crop production with or without tillage, crops requiring

Table 4.5 Impacts on terrestrial acidification reported in case studies of biomass production for bioenergy (sector × scenario)

Crop resulting in an LUC	Country species	Acidification potential	References
Cardoon	Italy *Cynara cardunculus* var. *altilis* DC	Cardoon production involves emissions of volatile compounds primarily from the production and application of fertilizers (93%), with the remainder coming from fuel combustion for mechanical field operations. These emissions contribute to a potential environmental acidification impact (~37.85 kg SO_{2eq}/ha/year).	Cocco et al. (2014)
Miscanthus	United Kingdom *Miscanthus* x *giganteus*	Cultivation of miscanthus results in emissions of volatile compounds resulting from the production and application of fertilizers. These emissions, particularly ammonia, contribute to an environmental acidification impact (~6 kg SO_{2eq}/ha/year).	Brandao et al. (2011)
Mustard	Italy *Brassica carinata* A. Braun	Mustard cultivation results in emissions of volatile compounds from the production and application of fertilizers (89%), with the remainder coming from fuel combustion for mechanical field operations. These emissions contribute to a potential environmental acidification impact (~33.34 kg SO_{2eq}/ha/year).	Cocco et al. (2014)
Oil palm	Malaysia *Elais guineensis* J.	Oil palm cultivation results in emissions of volatile compounds primarily from the production and application of fertilizers; these emissions contribute to an acidification potential of 4.2 kg SO_{2eq}/ha/year.	Silalertruksa and Gheewala (2012)
Rapeseed	United Kingdom *Brassica napus* ssp.	Rapeseed production results in emissions of volatile compounds, primarily from the production and application of fertilizers. These emissions contribute to the potential environmental acidification impact (~28 kg SO_{2eq}/ha/year).	Brandao et al. (2011)
	Italy *Brassica napus* L. var. *oleifera* DC	Rapeseed production results in emissions of volatile compounds from the production and application of fertilizers (89%), with the remainder from fuel combustion for mechanical field operations. These emissions contribute to the potential environmental acidification impact (~30.12 kg SO_{2eq}/ha/year).	Cocco et al. (2014)
	Chile *Brassica napus* L.	Rapeseed production results in emissions of volatile compounds from the production and application of fertilizers (> 95% of the impact). These emissions contribute to the potential environmental acidification impact (1.9 kg SO_{2eq}/metric ton of seed at 92% DM, 49% of which is oil).	Iriarte et al. (2010)
Sitka spruce	United Kingdom *Picea sitchensis*	The use of residues results in NO_x emissions from diesel use for machinery. These emissions contribute to a small acidification impact (< 1 kg SO_{2eq}/ha/year).	Brandao et al. (2011)

Sunflower	Chile *Helianthus annuus* ssp.	Sunflower cultivation results in emissions of volatile compounds from the production and application of fertilizers (> 95% of the impact amount). These emissions, especially ammonia, contribute to an environmental acidification impact (2.3 kg SO_{2eq}/metric ton of seed at 92% DM, 49% of which is oil).	Iriarte et al. (2010)
Temporary grassland	Sweden mixed grasses and clover	The introduction into a cereal rotation of 2 consecutive years of temporary grassland, cut to produce biogas, results in a potential acidification impact of 14 kg SO_{2eq}/ha/year. This is higher than that of the initial rotation (5 kg SO_{2eq}/ha/year) due to higher NH_3 emissions resulting from the spreading of digestate from biogas production, notwithstanding reductions in SO_2 and NO_x achieved by substituting biogas for fossil fuels.	Tidaker et al. (2014)
Willow	United Kingdom *Salix* spp.	Cultivation of short-rotation willow results in emissions of volatile compounds from the use of diesel-powered machinery. These emissions, especially ammonia, contribute to an environmental acidification impact (~2 kg SO_{2eq}/ha/year).	Brandao et al. (2011)

Quantified results, notably impacts from life-cycle analysis case studies, depend on the boundary of system studied and cannot be repurposed without first verifying their area of validity. Soil depths are only mentioned when explicitly stated in the article or deducible from the methodological approach utilized. The term “grassland” is generally considered to mean “permanent grassland,” as opposed to “pasture,” which can be understood as temporary grassland where it is associated with the land-used type “cropland,” i.e. “arable land” different amounts of water or other inputs, etc). The net impact of these complex processes and interactions will depend first of all on the soil type and its properties (see Sect. 4.3.3.3.3), thus limiting the applicability of the observed results to various soils and contexts.

LUC impacts concern first of all the soil resource, but also touch upon other environmental compartments directly or indirectly impacted by soil processes, including water cycling, nutrient cycling, and the movement and transformation of other elements. The mechanisms analyzed in the articles in the corpus primarily consider impacts on SOC.

Processes Affecting the Soil Resource

Mechanisms for soil impacts include first of all the physicochemical processes contributing to the loss of soil or of its constituent elements, notably organic matter and nutrients. The most important processes in this regard are erosion and runoff (Brady and Weil 2002). Burning also contributes to the loss of soil organic matter in the case of some LUC. Erosion and leaching of dissolved SOC can account for a large percentage of SOC losses in agriculture, up to 20–30% of changes in SOC (Izaurralde et al. 2007 *in* Zhang et al. 2010). Nevertheless, in the context of a modeling with CENTURY of SOC emissions linked to LUC in the United States, the addition of erosion to the model did not affect emissions (Dunn et al. 2013).

These processes of loss are influenced by changes in land use *via* changes in the vegetation cover, which can play a mechanical role in protecting the soil. The greater average soil cover of perennial crops compared to annual crops reduced erosion risks (Smeets and Faaij 2010). Conversion of grasslands into switchgrass in the lower Mississippi watershed achieved a reduction of erosion and runoff; this reduction was notably correlated with an increase in evapotranspiration and a reduction in the water charge in the watershed (Wu et al. 2012). Forests and perennial crops such as oil palm or rubber tree resulted in less surface runoff thanks to their more extensive root systems and higher evapotranspiration rates compared to manioc and sugarcane (Babel et al. 2011). Losses were also exacerbated in manioc fields because of its limited soil cover (even at maturity), low planting density, and a leaf architecture that accentuated the mechanical action of rain (Babel et al. 2011). These effects were also observed in sugarcane fields compared to grasslands or forests due to periods of bare soil at planting and between harvest and re-growth (Babel et al. 2011). Erosion risks can thus be significant during the initial phase of development following the LUC, until canopy closure; for example in the case of warm-season grasses, like switchgrass, which is slow to establish. These risks can be limited with an appropriate choice of species and the use of improved planting systems (van Dam et al. 2009).

Cultivation activities can also contribute to the physical degradation of the soil, on the one hand, via soil tillage and the destruction of soil aggregates, which can create a soil more susceptible to erosion (Zimmermann et al. 2013), and, on the other hand, via soil compaction, which leads to reduced water infiltration and thus increased erosion potential (van Dam et al. 2009; Smeets and Faaij 2010). Such effects were observed in sugarcane as a result of compaction linked to cultivation operations and harvest (Fiorio et al. 2000; Prado and Centurion 2001 *in* Babel et al. 2011). Increased apparent density and reduced water infiltration, and thus the risk of losses to runoff and erosion were higher where SOC levels were low (Wu et al. 2012). On top of these physicochemical factors are biological processes, which vary depending on soil aeration. Tillage can stimulate processes of decomposition and loss of organic matter linked to increased soil aeration deeper in the soil (Solomon et al. 2001; Zimmermann et al. 2013). Compaction, on the other hand, can lead to anoxic areas in reduced soil pore space, favoring denitrification and hence nitrous oxide emissions.

Changes in land use or land management practices allowing for a protection of the soil surface and an increase in biomass can help maintain or even improve SOM levels. Reduced tillage, notably with the shift from an annual to a perennial crop, and returning crop residues to the soil (including leaf litter with perennial crops) can have a positive effect on the accumulation of SOC (Anderson-Teixeira et al. 2009; Mishra et al. 2013; Zimmermann et al. 2013; Gabrielle et al. 2014; Mello et al. 2014). In the case of conversions to perennial crops, however, reductions in tillage and the maintenance of SOC will vary depending on the type of crop and the type of LUC. In the case of sugarcane in Brazil, for example, the land is usually plowed every 5 years and the sugarcane is replanted. This time span does not necessarily preserve all the carbon stored during the first crop cycle, resulting in net gains where sugarcane followed an annual crop but net losses where it followed grasslands (Mello et al. 2014). On the other hand, the benefits of switchgrass in terms of SOC storage following conversion of arable land or degraded grasslands would persist after 100 years of simulation, although the annual rate was divided by 10 (van Dam et al. 2009).

Influence of Plant Type on SOC Storage

Increases in SOC are potentially greater with a higher productivity of the soil-plant system. Higher yields and higher above- and belowground biomass production are thus correlated with increases in SOC levels in CENTURY (Dunn et al. 2013). System productivity depends on the land use type and on the match between pedo-climatic conditions and optimum crop conditions. The photosynthetic type (e.g. C3 or C4 plants) is key to the scaling of these optima and of the impact variations across different LUC (van Dam et al. 2009). Overall and relative performances obviously vary depending on location (Dunn et al. 2013; Debnath et al. 2014). In a comparison of two agroecological zones in the United States, various LUC involving the conversion of forests, grasslands, or arable land into maize, switchgrass, and miscanthus resulted in changes in SOC that followed consistent trends but were greater in temperate humid zones (“temperate sub-humid agroecological zone,” AEZ10) than in temperate arid zones (“temperate arid agroecological zone” AE2710) (Dunn et al. 2013). Mmetaiscanthus showed highly spatially variable environmental impacts at the level of a region in the Netherlands (Elbersen et al. 2014). Similarly, the potential for SOC storage increased from west to east across the United States as a function of increased soil moisture and associated productivity levels (Mishra et al. 2013). By contrast, in a study of eucalyptus in Brazil, SOC levels did not vary significantly as a function of biome from the cerrado in the center of the country to the pampas in the south or the forests of the eastern coastal region (Mata Atlântica). Annual average precipitation is similar across these biomes (1200–1500 mm), but the length of the dry season differs (Fialho and Zinn 2014). For switchgrass in the United States, studies do not agree on the correlation between biomass production and SOC accumulation (Follett et al. 2012; Mondzozo et al. 2013 *in* Debnath et al. 2014). This example illustrates the complexity of the underlying processes and the need to explore both multiple contexts and other correlated mechanisms.

SOC accumulation most likely depends on both biomass productivity and plant eco-physiology, which determines the allocation of carbon into roots and rhizomes (Anderson-Teixeira et al. 2009; Zimmermann et al. 2013). Hence, the different components of yield, exported biomass, and recycled biomass are not sufficient to understand their influence on SOC. Harvest dates can be chosen to favor leaf senescence and the reallocation of plant reserves into storage organs, and thus potential SOC accumulation; for example by delaying harvest of miscanthus after winter senescence (Mishra et al. 2013; Zimmermann et al. 2013). Root depth also plays a role, as seen in comparisons between sugarcane and miscanthus or switchgrass. SOC accumulation with the latter two grasses was more even, regardless of rooting depth, whereas for sugarcane, SOC accumulation took place primarily at the surface level (Anderson-Teixeira et al. 2009). Fifty percent of the root biomass for miscanthus was found below 90 cm deep (Neukirchen et al. 1999 *in* Mishra et al. 2013). Switchgrass also made it possible to store considerable quantities of SOC by favoring the production of humus, and through the production of a large quantity of rhizomes and root biomass deep in the soil (Lewandowski and Elbersen 2000; Liebig et al. 2005 *in* van Dam et al. 2009).

The impact of recycled biomass depends on interactions between plant type, notably the amount of lignin in the residues, and microclimatic conditions for decomposition. The chemical composition of residues does not fully explain carbon longevity in the soil, which in fact depends on total ecosystem functioning (Schmidt et al. 2011 *in* Fialho and Zinn 2014). In North America, net changes in SOC under switchgrass increased along a positive temperature gradient (Anderson-Teixeira et al. 2009). In Sweden, decomposition dynamics were lower when artificial or temporary grasslands were introduced into annual crop rotations, as the result of a combined effect of the reduction in tillage with drier and colder average conditions during plant growth (Bolinder et al. 2012 *in* Tidaker et al. 2014). Nitrogen fertilizer practices can also alter soil C:N ratios and thus modify decomposition dynamics, as it was observed in the case of LUC toward miscanthus (Schneckenberger and Kuzyakov 2007 *in* Mishra et al. 2013).

Influence of Soil Type on Variations in SOC and Risk of Losses

Soil properties necessarily play a major role in the capacity of a crop or a vegetation management practice to influence SOC storage or loss. In some cases, inherent soil properties may even mask the LUC effect, as suggested by the authors of a study in Mozambique, in which large variations in SOC under forest outweighed measured variations in SOC under forest versus in maize or in jatropha (Rasmussen et al. 2012).

In the first place, the original SOC level is critical for characterizing the LUC impact. An LUC from grassland to miscanthus can result in a net increase in SOC on mineral soils (Anderson-Teixeira et al. 2009), but a net loss of SOC on organic soils (Elbersen et al. 2014). The magnitude of change in soil carbon levels will thus depend on the reference used to define a soil initial SOC: e.g., a potential level relative to a theoretical natural reference amount (Sect. 4.3.2.2), an initial prior level in the case of an LUC, or a potential theoretical prior level, obtained either by synchronous observation or by comparison with the literature (e.g. Helin et al. 2014). In the case of modeling with CENTURY, the choice of the parameter value for the increase in the rate of SOC degradation associated with cultivation, i.e. the "clteff" (cultivation effect parameter), either as default value or as calibrated for land in maize resulted in increased emissions under maize production but reduced emissions upon conversion to switchgrass or miscanthus, due to the reduced SOC levels prior to the LUC (Dunn et al. 2013).

At the same time, soil texture and structure also play a role in mechanisms of SOC storage and loss. Soil clay content, for example, was shown to moderate SOC gains under perennial crops and losses under maize in a study covering several countries (Anderson-Teixeira et al. 2009). In the case of eucalyptus in Brazil, clayey soils tended to store more SOC than sandy soils, particularly in the top 20 cm of the soil profile, although the results were not statistically significant (Fialho and Zinn 2014). Retention of SOC in sandy soils in Brazil was mainly linked to unstable debris, which may be maximized and stabilized with specific practices in order to increase SOC over the long term (Fialho and Zinn 2014). This type of large-particle organic matter, distinct from soil organo-mineral complex, is more sensitive to mineralization than organic matter bonded to silt and clay. Thus, a study in Tanzania found that soil texture influenced the change in the type of organic matter present in the soil following an LUC of degraded woodland into annual crops, notably via amino sugar signatures. Microbial sugar metabolites are more stable and were less affected by this LUC in more finely textured soils (Solomon et al. 2001). Other properties, notably soil aggregates and levels of iron and aluminum oxides, were shown to strongly influence SOC dynamics and retention in Brazilian Oxisols (*in* Fialho and Zinn 2014).

A group of soil parameters relating to the productive capacity of soils was used in the United States to define classes of soil capacity, or "land capability classes." In this system, soils with a similar, sustainable productive capacity with respect to a specific pedoclimatic and agronomic context are classed together. Criteria include parameters for the morphology, structure, texture, and mineral composition of soils.

In the case of an LUC from no-till wheat to switchgrass, modeling studies suggested an increase in SOC and a reduction in losses of soil, nitrogen, and phosphorous across the three soil classes considered (land capability classes I, II, and III). The higher the initial loss risk through erosion and runoff, the larger the potential reduction in losses: thus, type I soils having the lowest risk of erosion showed the smallest reduction in losses; type III soils having the highest risk of erosion showed the greatest reduction in losses; type II soils were intermediate in both respects. SOC storage was more significant for type I soils, however, given their higher productive potential and a greater associated SOC storage compared to the type II and type III soils as modeled (Debnath et al. 2014).

Indirect Impacts

Erosion, runoff, and leaching can result in losses of nutrients into the environment (Babel et al. 2011), potentially leading to a eutrophication impact on wetland environments. Such losses are influenced by both LUC and practices. The expansion of manioc and sugarcane production in the Khlong Phlo watershed in Thailand might lead to greater losses of sediment, nitrate, and phosphorous than those caused by oil palm production due to increased risks of erosion and runoff. On the other hand, at a similar erosion and runoff risk level, oil palm could lead to greater losses than rubber plantations or orchards due to more intensive fertilization (Babel et al. 2011). Miscanthus, in another context, resulted in less nitrate loss compared to annual crops of oilseed rape, sugar beet, and wheat in the region of the Ile de France by a factor of 1.05–4 (Gabrielle et al. 2014). Switchgrass was observed to lead to lower losses via runoff of nitrogen (up to 68.5 kg $N.ha^{-1}.year^{-1}$ less) and phosphorous (up to 1.5 kg $P.ha^{-1}.year^{-1}$ less) when compared to wheat on various sites in Oklahoma, in the United States. Again in the United States, in Indiana, miscanthus and switchgrass were found, through modeling work, to result in lower losses of sediments (up to 30% less), total nitrogen (up to 16% less), and total phosphorous (up to 33% less) when compared to a previous land use of grassland, maize-soybean rotation, or a mixture of the two. Loss reductions were comparable between miscanthus and switchgrass with the exception of nitrogen losses, which were slightly higher in miscanthus than switchgrass despite equivalent inputs of nitrogen fertilizer (Hoque et al. 2014). In Iowa, the conversion of grassland areas from Conservation Reserve Program into maize monoculture or maize in rotation led in all scenarios to increased losses of sediments, phosphorous, and nitrogen into the environment (Secchi et al. 2011).

Nitrogen fertilizer inputs are likewise accompanied by direct and indirect emissions of volatile nitrogen compounds into the atmosphere, which can contribute notably to climate change and acidification impacts. These impacts, as described above in particular for acidification (see Sect. 4.3.2.2), can also have indirect impacts on the soil. In the northern part of the Netherlands, the replacement of crop rotations with miscanthus would lead to a reduction of such emissions and subsequent impacts due to the relatively low levels of inputs (Elbersen et al. 2014). In

Ile-de-France, the same LUC of annual crops to miscanthus would result in similar reductions in these emissions. Emissions of N_2O would be 2–6 times lower compared to those for crops of oilseed rape, wheat, and sugar beet; those of NO would be divided by 2, and those of NH_3 by 14–32 (Gabrielle et al. 2014).

Finally, compared to annual crops, the higher evapotranspiration rates of perennial crops – as observed for eucalyptus, poplar, and switchgrass – could reduce runoff and associated nutrient losses (see Sect. 4.3.3.3.4), although this also implies a potential reduction in available water resources (van Dam et al. 2009; Smeets and Faaij 2010; Wu et al. 2012).

4.4 Discussion and Conclusion

Although soils are the first resource impacted by land use and land-use changes, the characterization of the impacts on soils and soil quality remains limited, both in terms of the number of articles addressing the subject and in terms of the properties and parameters that have been explored to assess effects on soil quality. Barely 20% of the articles in the corpus addressed soil impacts (52 out of 241) and only 15% quantified these impacts (37 out of 241). Within this limited sample of 37 articles, 70% detailed impacts on SOC; 22% also addressed erosion impacts or erosion only (1 article). The dominant focus on SOC impacts is explained by the critical role played by SOM in the capacity of a soil to fulfill various functions. SOC levels, which are directly correlated with SOM levels, thus act as an indicator reflecting a variety of potential changes in the properties and functions of soils. Besides, SOC is also a relatively easy parameter to measure.

Overall, the studies in the corpus show that perennial plants tend to be better at maintaining and even improving SOC levels compared to annual crops. Nevertheless, quantified results are highly variable and depend on the pedoclimatic and agronomic context. Some results show variations in SOC that are more or less sensitive to soil type. Detailed analysis of the influence of soil type would require large meta-analyses or studies based on exhaustive measurement protocols across numerous field sites. This type of study is poorly represented in the corpus, which for this reason offers relative little robust information as to the influence of soil type on soil impact mechanisms resulting from LUC.

Experimental and modeling results have shown the importance of history and evolution in LUC – that is, the importance of considering change over time from a point of equilibrium prior to the LUC and depending on the plot history up to a new point of equilibrium. Nevertheless, many studies rely on default reference values; e.g., 30% of studies quantifying LUC make use of coefficients from the IPCC Tier 1 (2006). The significance of a change in SOC in terms of impacts on soil quality is only demonstrable if that change does result from modifications in soil processes. The use of the static stock-difference method based on default coefficients is an uncertain proxy for soil impacts, particularly if the land uses to be compared are not at equilibrium in terms of soil properties and functions. Taking into account the dynamics of change in SOC using a modeling approach based on these processes seems unavoidable. Analysis of these dynamics requires a more holistic approach both to SOC itself (i.e., at different soil depths, in various soil organic matter fractions, etc.) and to the interactions between SOC and other soil parameters and soil properties, notably connections to the carbon and nitrogen cycles, biological activity, etc. Much research is needed to improve these models: on the one hand, some models still use default parameters that are not necessarily calibrated for all pedoclimatic conditions, potential uses, and LUC (e.g. Goglio et al. 2015); on the other hand, not all the processes involved in variations in soil quality are fully understood (Brady and Weil 2002).

LUC impacts on SOC are overall more thoroughly researched than other impacts relating to soil quality. Although this indicator could be still much improved, other impacts also need to be examined more closely and potentially better integrated into existing models. The study of some perennial crops has suggested a potentially antagonistic effect between the maintenance of soil quality (via increases in SOC and the reduction of erosion and runoff risks) on the one hand, and impacts on water resources on the other hand. This dilemma hints at the necessary tradeoffs and compromises that can be involved in multi-criteria assessments. The various impacts on soils must be analyzed in parallel with other impacts, e.g. on water (Bispo, Chap. 5, this volume) and biodiversity (Gaba, Chap. 8, this volume) The processes at work within the different environmental compartments influence a full suite of resources and are interconnected via geochemical cycles (carbon, nitrogen, water) and via changes in biological habitats. LUC for energy crops can also displace food crops. In these situations in particular, but also in the broader context of sustainable development, tradeoffs between different crops must be considered also with respect to their socioeconomic impacts.

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