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Soil carbon sequestration and land use change associated with biofuel production: empirical evidence

ZHANGCAI QIN[1], JENNIFER B. DUNN[1], HOYOUNG KWON[2], STEFFEN MUELLER[3] and MICHELLE M. WANDER[4]

[1]*Energy Systems Division, Argonne National Laboratory, 9700 South Cass Avenue, Argonne, IL 60439, USA,* [2]*Environment and Production Technology Division, International Food Policy Research Institute, 2033K St. NW, Washington, DC 20006, USA,* [3]*Energy Resources Center, University of Illinois at Chicago, 1309 South Halsted Street, Chicago, IL 60607, USA,* [4]*Department of Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, 1102 South Goodwin Avenue, Urbana, IL 61801, USA*

Abstract

Soil organic carbon (SOC) change can be a major impact of land use change (LUC) associated with biofuel feedstock production. By collecting and analyzing data from worldwide field observations of major LUCs from cropland, grassland, and forest to lands producing biofuel crops (i.e. corn, switchgrass, *Miscanthus*, poplar, and willow), we were able to estimate SOC response ratios and sequestration rates and evaluate the effects of soil depth and time scale on SOC change. Both the amount and rate of SOC change were highly dependent on the specific land transition. Irrespective of soil depth or time horizon, cropland conversions resulted in an overall SOC gain of 6–14% relative to initial SOC level, while conversion from grassland or forest to corn (without residue removal) or poplar caused significant carbon loss (9–35%). No significant SOC changes were observed in land converted from grasslands or forests to switchgrass, *Miscanthus,* or willow. The SOC response ratios were similar in both 0–30 and 0–100 cm soil depths in most cases, suggesting SOC changes in deep soil and that use of top soil only for SOC accounting in biofuel life cycle analysis (LCA) might underestimate total SOC changes. Soil carbon sequestration rates varied greatly among studies and land transition types. Generally, the rates of SOC change tended to be the greatest during the 10 years following land conversion and had declined to approach 0 within about 20 years for most LUCs. Observed trends in SOC change were generally consistent with previous reports. Soil depth and duration of study significantly influence SOC change rates and so should be considered in carbon emission accounting in biofuel LCA. High uncertainty remains for many perennial systems and forest transitions, additional field trials, and modeling efforts are needed to draw conclusions about the site- and system-specific rates and direction of change.

Keywords: corn, cropland, emission factor, forest, grassland, life cycle analysis, *Miscanthus*, poplar, switchgrass, willow

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Introduction

Current cropland, grassland, forest, or other land use types may be converted to grow bioenergy crops to meet growing demand for energy, and this will inevitably change land cover and consequently soil carbon dynamics (Fargione *et al.*, 2010; Qin *et al.*, 2012). Soil organic carbon (SOC) change associated with this type of land use change (LUC) is a major concern. SOC is one of the largest carbon pools in the earth system (Post & Kwon, 2000; Guo & Gifford, 2002; Don *et al.*, 2011), storing three to four times as much carbon as atmospheric or biotic carbon pools (Batjes, 1996; Lal, 2004). Land use shifts prompt immediate soil disturbance (e.g. tillage, cropping) and ambient environmental changes (e.g. micro-scale climate conditions, vegetation type) that can fundamentally alter both carbon inputs and decomposition rates and eventually affect soil carbon content (Tolbert *et al.*, 2002; Zatta *et al.*, 2013). Even though soil carbon is sensitive to, and changes with, environmental conditions such as temperature, moisture, and biota, it comes to equilibrium with time when carbon inputs and environmental factors are relatively stable (Stewart *et al.*, 2007; Qin *et al.*, 2013). This contributes to uncertainty about the direction, amount, and rate of SOC change following LUC that presents a challenge for scientists and policy makers charged to manage soil, energy, and climate (Anderson-Teixeira *et al.*, 2009; Don *et al.*, 2011). Efforts to predict carbon emissions following LUC are being improved by accounting

Correspondence: Zhangcai Qin, tel. + 1 630 252 6428, e-mail: zqin@anl.gov; Jennifer B. Dunn, tel. + 1 630 252 4667, fax + 1 630 252 3443, e-mail: jdunn@anl.gov

for factors already known to influence the direction and magnitude of SOC change. At present, many attempts to refine carbon emissions factors are using process models to increase site and system specificity (Kwon *et al.*, 2013; Van der Hilst *et al.*, 2014). Data-driven estimates of SOC change are needed to inform and evaluate such modeling efforts (e.g. Liska *et al.*, 2014) and carbon emissions factors (e.g. Murphy & Kendall, 2014). These emission factors are increasingly incorporated into biofuel life cycle analysis (LCA) and can drive LCA carbon emission results.

Earlier field-based studies suggest that LUC can either increase or decrease SOC depending on the type of LUC and specific management practices applied (Post & Kwon, 2000; Guo & Gifford, 2002). Converting natural ecosystems, such as forest and grassland, to managed agriculture has been reported to result in a 10–59% decrease in soil carbon stocks while replacing crops with pasture or woody plantation tends to increase SOC (Guo & Gifford, 2002). Don *et al.* (2011) explored LUC in the tropics and reported the highest SOC loss (25–30%) occurred when primary forests were converted to crops. This is consistent with Anderson-Teixeira *et al.* (2009) who synthesized site-level SOC observations to find that conversion of uncultivated land to biofuel feedstock agriculture could cause significant SOC losses. Of the transitions they examined, the most pronounced carbon loss occurred when native land (e.g. forest, grass) was converted to sugarcane agriculture. However, they also observed that under certain perennial grasses, SOC accumulated with an average annual rate of over 1 t C ha^{-1}. Using modeling experiments examining shifts from food crops to biofuel crops in the U.S., Qin *et al.* (2012) and Mishra *et al.* (2013) predicted noticeable carbon gains in the soils of cellulosic ecosystems (e.g. switchgrass, *Miscanthus*) that suggested potential soil carbon sequestration. Kwon *et al.* (2013) examined SOC change due to various LUCs and predicted small SOC losses or even modest SOC gains for land transitions (particularly cropland) to corn-based systems. Collectively, these works show that historical and current land use types largely determine the direction and magnitude of SOC change.

Most studies of LUC have not adequately considered how soil depth affects SOC change. Many studies have focused on SOC change in top soil (generally soil depths from 0 to 20 or 30 cm) because surface soils contain most of a crop's roots and residues, and so, it is assumed that the majority of SOC change will occur in this zone. SOC changes in deeper soil (e.g. 30–100 cm) are often neglected (Knops & Bradley, 2009; Harrison *et al.*, 2011). This simplification may be defensible for shallow-rooted food crops, for which most management practices disturb only the top soil, but may be inadequate for many energy crops with roots that penetrate deeper than 30 cm. Perennial crops such as switchgrass and *Miscanthus* can develop an extensive root system with roots extending to a depth of over 200 cm (Neukirchen *et al.*, 1999; USDA (U.S. Department of Agriculture), 2011);. The top 0–30 cm soil contains <30% of root biomass in the *Miscanthus* system (Neukirchen *et al.*, 1999). Recent studies of annual (e.g. corn) and perennial crops (e.g. switchgrass) indicate that over half of all soil carbon accumulation may occur in deep soil depths (Knops & Bradley, 2009; Follett *et al.*, 2012). Some have recommended that SOC stocks be determined for deeper soil depths (e.g. 100 cm) or even the entire soil profile, to more accurately evaluate soil carbon change and carbon sequestration potential (Fontaine *et al.*, 2007; Knops & Bradley, 2009; Poeplau *et al.*, 2011; Wiesmeier *et al.*, 2012). Examination of SOC changes in deeper soil depths is needed to determine whether inventories or trading protocols can rely on surface-based SOC accounting (Harrison *et al.*, 2011; Wiesmeier *et al.*, 2012).

Another critical factor is the time period considered in the estimation. The time horizon over which SOC change is considered has a significant influence on LCAs that include SOC changes from LUC. SOC changes quantified in LCA are often amortized over a period of time that represents the implementation of a biofuels policy, such as 30 years (USEPA (U.S. Environmental Protection Agency), 2011; Dunn *et al.*, 2013; Plevin & Kammen, 2013). Many studies address SOC level changes and soil carbon sequestration rates as a result of LUC, but do not substantially address the influence of the study duration. Very rapid rates of SOC change have been reported immediately following land use changes between natural and managed ecosystems (Guo & Gifford, 2002; Anderson-Teixeira *et al.*, 2009; Zimmermann *et al.*, 2012). Rates of SOC change typically decrease with time and become minimal as the soil reaches a new SOC equilibrium (Guo & Gifford, 2002; Stewart *et al.*, 2007). Short-term experiments designed to understand the impact of the establishment phase of a new land use may provide a poor basis for long-term estimates of SOC change and soil carbon sequestration rate associated with bioenergy feedstock cropping. Rates of SOC change developed over a brief time horizon could potentially overestimate total ecosystem SOC fluxes when extended to a longer time period. To accurately estimate SOC change and sequestration rates, one must consider how long the production system is likely to be maintained under a consistent suite of management practices.

The objective of this study is to identify general patterns in the literature concerning the direction and order of magnitude of SOC change and the sequestration rate observed under important biofuel feedstock land conversion scenarios. We draw on the many individual field experiments that have studied SOC changes under various potential biofuel crops, producing a variety of site-based results that are variable and difficult to extrapolate to the larger scale (Guo & Gifford, 2002; Anderson-Teixeira *et al.*, 2009). A limited number of sites in a single study do not provide sufficient data to broadly predict SOC change patterns among different land use types, soil depths, or time periods (Walter *et al.*, 2014). In this work, we reviewed SOC change results from experiments distributed across the full range of existing climate, soil, and vegetation backgrounds where biofuel feedstocks are being produced to improve generalized estimates of C change resulting from LUC. The observational data were collected from available field experiments to study SOC changes when one of three original land types: cropland, grassland, and forest were converted to five biofuel feedstock cropping systems (corn, switchgrass, *Miscanthus*, poplar, and willow). To improve confidence in, and reduce variability associated with, estimates of the direction, magnitude, and rate of SOC change, we summarize LUC data to account for land use history, soil depth, and duration under feedstock production.

Materials and methods

Data collection and soil carbon calculations

We collected data for corn, energy grasses, and short rotation woody crops that are recognized as current or potential biomass feedstocks in the U.S. (USDOE (U.S. Department of Energy), 2011);. The only grain crop included is corn (*Zea Mays* L.), which is widely grown in the Corn Belt with and without residue harvest as a cellulosic feedstock. The energy grasses we considered were switchgrass (*Panicum virgatum* L.), a perennial, warm-season cellulosic crop native to North America, with high biomass production (McLaughlin & Adams Kszos, 2005). *Miscanthus*, a genus of several species native to Asia, mostly used in Europe but also in the U.S. (e.g. *Miscanthus* × *giganteus*) (Heaton *et al.*, 2008) is also included. Switchgrass and *Miscanthus* are considered potential cellulosic biomass feedstocks as both have high biomass production, environmental resistance (e.g. drought tolerance), and potential to contribute to soil carbon sequestration (Heaton *et al.*, 2008; Fargione *et al.*, 2010; Qin *et al.*, 2012; Kwon *et al.*, 2013). Finally, this study incorporates two types of woody crops, poplar, and willow, into the database. Both are high-yield, short rotation coppice crops used to produce woody solid biomass (Aust *et al.*, 2013; Walter *et al.*, 2014). Poplar (*Populus* L.) is a genus of over 20 deciduous species native to mostly the Northern Hemisphere and is widely distributed in the U.S. (Sannigrahi *et al.*, 2010). Willow (*Salix* L.) has been promoted as a bioenergy crop in the U.S. as well as in Europe (Pacaldo *et al.*, 2013). Data were primarily compiled from published peer-reviewed literature on LUC impacts on soil carbon stocks that rely on field-based studies where the change in SOC was documented or could be computed using data reported for an individual site. Multiple data sources (e.g. multiple publications, reports, or personal communication) were often referenced to acquire sufficient information to complete single-site observations within the dataset. Occasionally, we requested additional information or clarification from study authors. Site observations were separated according to location, time, land use history, crop type, and treatment. Single studies may have multiple-site observations considered within depth and study duration comparisons (e.g. Ceotto & Candilo, 2011; Bonin & Lal, 2014; Walter *et al.*, 2014).

About 100 references were included in the database (Table S1). The studies were conducted worldwide but the majority are from North America and Europe (Fig. 1). The studies used repeated sampling, paired-site, or chronosequence approaches to determine SOC change. For the repeated sampling technique, researchers measure SOC levels at the same site over time. These experiments are typically long term, to encompass different experimental phases (e.g. land use transitions) and allow for sufficient time to past to produce measurable SOC change. For our meta-analysis, the control for studies sampled over time was the initial SOC level determined for the site. The paired-site experimental design compares multiple replicated treatments side-by-side. The experimental plots are normally adjacent with similar soil, slope, elevation, and drainage (Novara *et al.*, 2012). For paired-site designs, the control is the SOC level in plots maintained under the initial or dominant land use. Chronosequence studies, the third type of experimental design, use a ‘space-for-time’ design. In this technique, SOC measurements are taken at a set of sites with similar land use that are of different ages where all other attributes (e.g. climate, soil) are similar. Chronosequence studies are typically used for very long-term assessments where investigators are interested in years or decades of monitoring (Knops & Bradley, 2009; Novara *et al.*, 2012). In our analysis, studies were first grouped by biofuel feedstock type (corn, switchgrass, *Miscanthus*, poplar, and willow). For corn agriculture identified as current land use, the datasets that reported corn residue removal were first excluded from the database when estimating SOC change for the cropland to corn transition. They were later used to estimate possible impacts of residue removal on SOC. Studies for each feedstock were categorized by historical (or referenced) land use (cropland, grassland, or forest). All agricultural lands occupied by crops, mostly annual food crops (e.g. corn, soy, and wheat) were classified as cropland. Pasture and native grassland were classified as grassland. Forest in this study refers to primary or secondary forest. We did not find studies documenting SOC changes upon forest conversion to *Miscanthus* or switchgrass and therefore excluded these transitions. In total, 13 LUC scenarios were studied. It is important to note renewable fuel standard (RFS) provisions limit the likelihood of forest-to-feedstock production conversions. For example, under the U.S. Environmental Protection Agency’s aggregate compliance approach (Code of Federal Regulations, 2014), the total agricultural land in the U.S. is compared to the 2007 baseline of 402 million eligible crop acres to assess whether the area of this land has increased. If it has, biofuel producers must show that the land

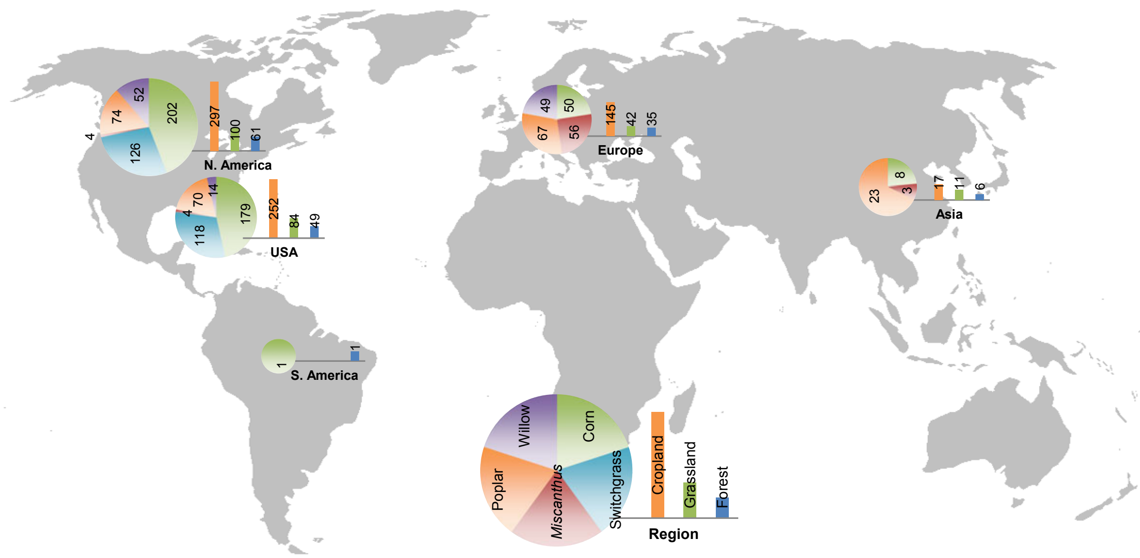

Fig. 1 Summary of datasets collected from field studies reporting SOC change under biofuel feedstock production. The legend is at the bottom: the biofuel crop types and historical land use types are colored in the pie chart and bar chart, respectively, designated by region. The values in the charts indicate the number of studies for each land type. The studies in USA were also included in the North America charts.

from which feedstock is produced was cleared or cultivated prior to December 19, 2007. Additionally, EPA requires that the land be managed, fallow, and nonforested on December 19, 2007. Short rotation woody crops such as willow or poplar would need to be produced on either agricultural land or land actively managed as a tree plantation prior to December 19, 2007. It is therefore possible that a tree plantation in operation before December 19, 2007 could convert from producing a species like pine to produce short rotation woody crops while meeting RFS requirements. The data we have compiled regarding forest-to-poplar and forest-to-willow transitions could apply for this type of land transition.

When studies did not directly report SOC density (t C ha^{-1}), we calculated it from SOC concentration (% or g kg^{-1}) (Eqn 1):

$$\begin{aligned}\text{SOC density (t C ha}^{-1}) = {} & \text{SOC concentration (g kg}^{-1})\\ & \times \text{Bulk density (g cm}^{-3})\\ & \times \text{Soil depth (cm)} \times 10^{-1}\end{aligned} \tag{1}$$

When studies did not report bulk density, we calculated it from soil organic matter (SOM) concentration (Adams, 1973; Rawls, 1983) using Eqn (2) after estimating SOM (%) from SOC (%) with the *Van Bemmelen* factor of 0.58 (Guo & Gifford, 2002). SOC stocks at various soil depths were converted to either 0–30 cm or 0–100 cm soil volume of carbon content. Unless specified in the literature, we assumed even distribution of carbon between separate soil intervals and used the vertical distribution of SOC provided by Jobbágy & Jackson (2000) in their global summary for each land type.

$$\begin{aligned}&\text{Bulk density (g cm}^{-3})\\ &\quad = [\text{SOM concentration (\%)}/0.244\\ &\qquad + (100 - \text{SOM concentration (\%)})/1.64]^{-1} \times 100\end{aligned} \tag{2}$$

SOC change and soil carbon sequestration rate

Meta-analysis, which synthesizes and contrasts results from different studies, has been extensively used to assess the LUC impacts on soil carbon (Post & Kwon, 2000; Guo & Gifford, 2002; Don *et al*., 2011). A common measure of effect size in meta-analysis is the magnitude of treatment mean relative to the control or reference mean. In this study, the difference would be between SOC of current land use (μ_c) and historical land use (μ_h). Following Johnson & Curtis (2001) and Guo & Gifford (2002), we used the response ratio, $\gamma = \mu_c/\mu_h$, to describe the relative SOC change due to LUC. The γ was then log-transformed (Eqn 3) such that $\ln(\gamma)$ can be approximately normally distributed if μ_c and μ_h are distributed normally.

$$\ln(\gamma) = \ln(\mu_c) - \ln(\mu_h) \tag{3}$$

Under each of the 13 LUC scenarios, observations were also categorized according to soil depth and time horizon of the study. For soil depth, we grouped observations that considered only depths of 30 cm or less and studies that examined SOC changes in depths below 30 cm. If a study reported separate results for soil depth to 30 cm and soil depths exceeding 30 cm, we included both sets of results. As for time, data from the studies included herein were grouped into three time periods: 0–5 years, 5–10 years, and more than 10 years. The heterogeneity partitioning between within-class and between-class was examined against Chi-square (χ^2) distribution with $(k - 1)$ degrees of freedom (Johnson & Curtis, 2001). The mean effect size of each category/group was computed, and 95% confidence intervals (CI) were reported. In the following descriptions, the SOC changes were presented as response ratio (reported as % of initial control), with zero suggesting no SOC change, and positive and negative values indicating SOC increase and decrease, respectively. The changes were considered to be statistically different if their 95% CIs were not overlapping. For a SOC change to be nonzero, its 95% CI cannot include zero.

The soil carbon sequestration rate, determined as the SOC change over a defined time period, was also obtained from studies. If a study did not explicitly report sequestration rate but did report the time horizon of the data collection, we calculated the sequestration rate with Eqn 4:

$$\text{SOC rate (t C ha}^{-1}\text{yr}^{-1}) = [\text{SOC}_c(\text{t C ha}^{-1}) - \text{SOC}_h(\text{t C ha}^{-1})]/\text{time (yr)} \quad (4)$$

where SOC_c and SOC_h are current and historical SOC, respectively. Most studies reported SOC content and elapsed time between land transition and SOC measurements allowing direct calculation of the soil carbon sequestration rate. Rate assessment also included a couple of results from the few studies that report the soil carbon sequestration rate (e.g. Nafziger & Dunker, 2011).

To identify soil change patterns in different soil depths and time periods, the LUC scenarios were further categorized into two soil depths (0–30 cm and 0–100 cm) or three time periods (0–5 year, 5–10 year, and 10+ year). Separate analyses were conducted for soil depth categories and time categories using the same datasets. Results were grouped by biofuel feedstock (i.e. current land use): corn, switchgrass, *Miscanthus*, poplar, and willow.

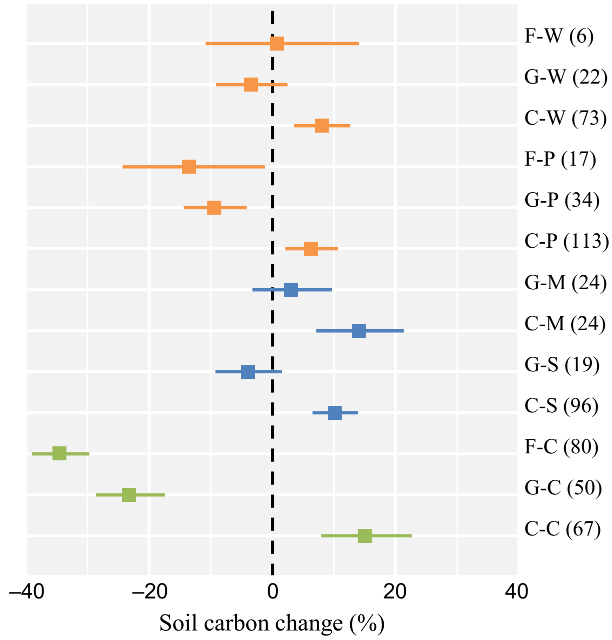

Fig. 2 Overall soil carbon change responding to land use changes. The estimates show response ratio (% change of initial control) with 95% confidence interval (error bars) for land use changes from cropland (C), grassland (G), and forest (F) to corn (C), switchgrass (S), *Miscanthus* (M), poplar (P), and willow (W), irrespective of soil depth and time horizon. Studies reporting corn residue removals were not included. Number of datasets is shown in parenthesis.

Results

SOC response to LUC

The overall response ratio of SOC change, regardless of soil depth and time, varies among different land transition scenarios (Fig. 2). The conversion from cropland generally resulted in increasing soil carbon stocks, with the average response ranging from 6% in poplar (n = 113) to 14% in *Miscanthus* (n = 24), although there were no significant differences in the magnitude of SOC change between cropland transitions. For conversions from grassland and forest, negative responses in soil carbon stocks were observed in corn and poplar, but not in switchgrass, *Miscanthus,* or willow.

When results for LUC to corn were separated to allow evaluation of depth (Fig. 3a) and study duration (Fig. 3b) on SOC response, we found the conversions to corn resulted in a SOC increase if the land was previously cropped and decreased if land was previously occupied by grass or forest (Fig. 3a). For different depth categories, all response ratios were significantly different from zero, except for the comparison for 0–100 cm depth describing the conversion of cropland to corn (C-C) (Fig. 3a). The magnitude of change was not altered by varying depths compared but variance increased for comparisons made to 100 cm because the dataset was smaller. The influence of study duration on SOC change was more notable (Fig. 3b). For the C-C scenarios, no data were available for the 0–5 years category. Median SOC responses for C-C conversions suggest an increase of 20% (n = 67) between 5–10 years and 14% beyond 10 years, but the difference in response ratios between the two study lengths is not significant. There were large 95% CI ranges in the 5–10 years category, suggesting substantial variances among different studies. Conversions from grassland (G-C) and forest to corn (F-C) exhibited a large SOC loss and the extent of loss increased with study length (Fig. 3b). For the G-C scenario, the SOC change was not significant during the first five years after conversion, but then reached −28% (n = 37) after 10 years. The F-C transition also caused SOC losses but these were apparent in short duration studies and were greatest (−36%) for comparisons made 10+ years after land conversion.

Conversion from cropland to switchgrass and *Miscanthus* increased carbon stocks, and the magnitude was slightly greater for comparisons in the 0–30 cm depth than the 0–100 cm depth but these differences were not significant (Fig. 4a and c). Land transitions from grassland to either of these energy grasses, however, did not

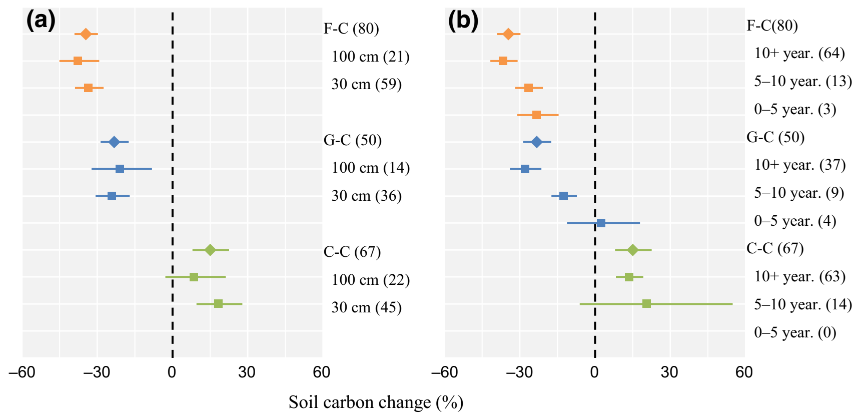

Fig. 3 Soil carbon change of corn ecosystem responding to land use changes with respect to soil depth and time horizon. The estimates show response ratio (% change of initial control) with 95% confidence interval (error bars) for land use changes from cropland (C), grassland (G), and forest (F) to corn (C) at different soil depths and time periods. Studies reporting corn residue removals were not included. Number of datasets is shown in parenthesis. Diamond symbols indicate overall response regardless of soil depth and time horizon.

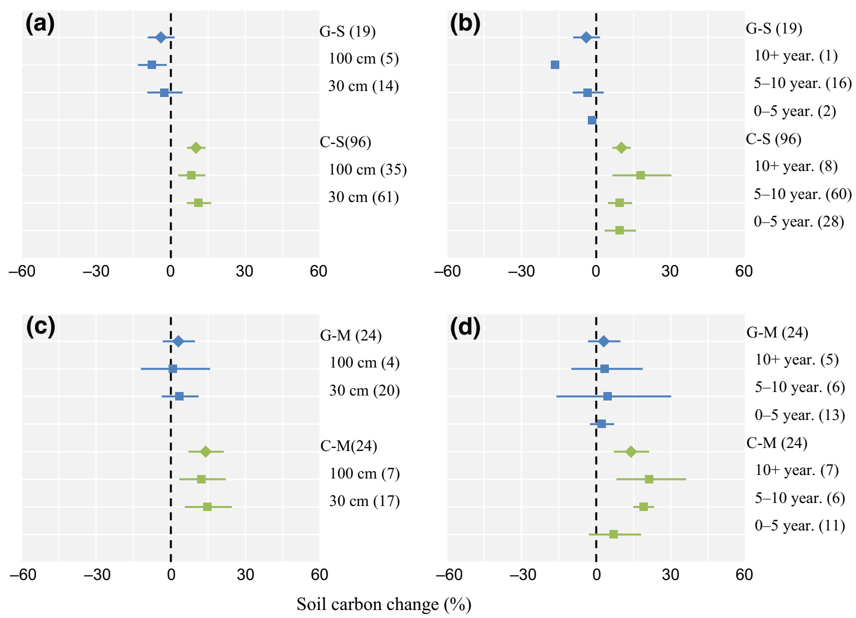

Fig. 4 Soil carbon change of switchgrass and *Miscanthus* ecosystems responding to land use changes with respect to soil depth and time horizon. The estimates show response ratio (% change of initial control) with 95% confidence interval (error bars) for land use changes from cropland (C) and grassland (G) to switchgrass (S) and *Miscanthus* (M) at different soil depths and time periods. Number of datasets is shown in parenthesis. Diamond symbols indicate overall response regardless of soil depth and time.

increase SOC under *Miscanthus* and slightly decreased SOC under switchgrass but only the comparison for the 0–100 cm depth was significantly different from zero (G-S) (Fig. 4a). An examination of SOC change in studies of different durations suggests that effects are slightly greater for longer duration trials comparing cropland transition to switchgrass (Fig. 4b) and *Miscanthus* (Fig. 4d) but differences were not significant. Study duration had no effect on SOC change for comparisons of grassland conversions to energy grass feedstock production. No data were available for transitions from forest to either switchgrass or *Miscanthus*.

The SOC response under poplar and willow increased when land had been converted from cropland, with slightly higher average change shown in shallow than deep soil (Fig. 5a and c). Installing poplar plantations on previous grassland caused a SOC loss of −6% (n = 20) in the 0–30 cm soil depth and −14% (n = 14) in the 0–100 soil depth. For converted forests, the change was only observed in the 0–30 cm soil (Fig. 5a). No

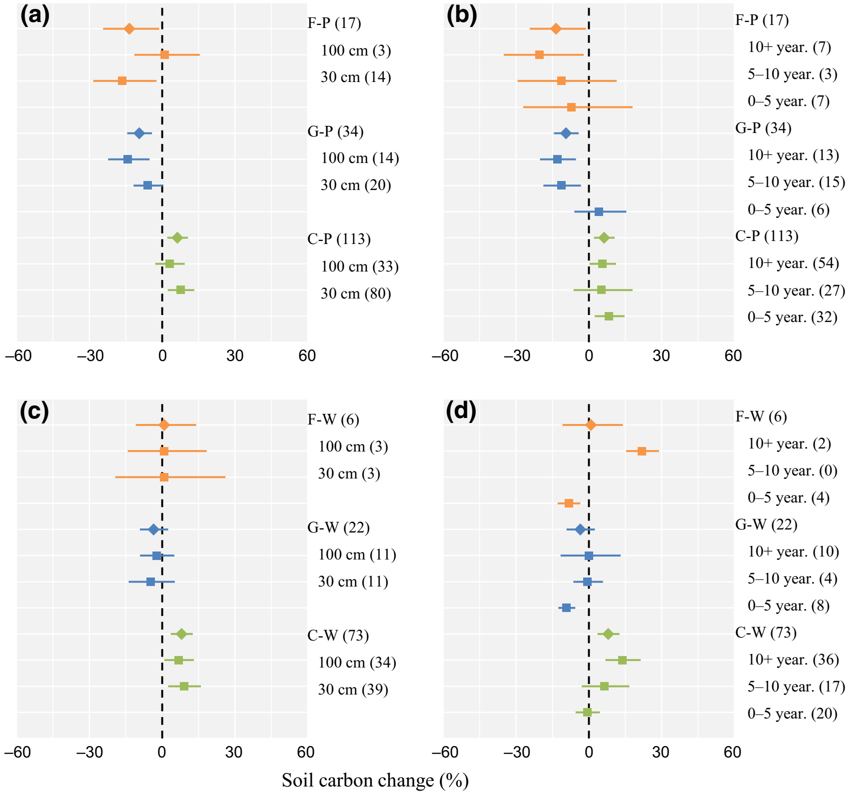

Fig. 5 Soil carbon change of poplar and willow ecosystems responding to land use changes with respect to soil depth and time horizon. The estimates show response ratio (% change of initial control) with 95% confidence interval (error bars) for land use changes from cropland (C), grassland (G), and forest (F) to poplar (P) and willow (W) at different soil depths and time periods. Number of datasets is shown in parenthesis. Diamond symbols indicate overall response regardless of soil depth and time.

significant changes were detected at any depth in grassland- or forest-to-willow transitions (Fig. 5c). SOC changes were of similar magnitude for studies of different length for cropland to poplar (C-P) scenarios (Fig. 5b). Duration of study did influence the SOC change for poplar produced on land converted from grassland (over 5 years) or forest (over 10 years) (Fig. 5b). For willow transitions, SOC changes tend to be more positive with increasing time but effects were only significant a decade after cropland conversion and 0–5 years after grassland or forest conversion (Fig. 5d). In the case of forest to willow (F-W), a very small sample set limits interpretation of the data in Fig. 5d. While four studies indicate that SOC decreases initially after the land transition, a couple of studies indicate SOC could increase a decade after the conversion. More data and information about initial site conditions are needed to clarify the observed temporal trends in SOC for forest-to-willow transitions.

Soil carbon sequestration rate

To assess the amount of SOC change rather than just the percent change (response ratio) based on initial samples, we computed soil carbon sequestration rates (Eqn 4) unless rates of change were reported. The rates were synthesized for both 0–30 cm and 0–100 cm soil depths. We only included results for the 30–100 cm soil depth when studies reported SOC changes in both shallow and deep soils. The sequestration rate averaged for the 0–100 cm depth was not necessarily the summation of the average rates for the 0–30 cm and 30–100 cm depths, because the datasets used for each summary do not come from entirely the same data sources. Generally, the results showed large variances of soil carbon sequestration rates across studies and LUC scenarios; variance was greatest for estimates of the entire 0–100 cm depth for all scenarios except for *Miscanthus* (Fig. 6a–j).

For scenarios with LUC from cropland, the SOC change was generally positive which was in accordance with the change detected in Fig. 2, but the magnitude of sequestration rates varied among soil depths and crop types (Fig. 6a, d, f, h and k). Rates in converted croplands tended to be positive for the 0–30 cm soil depth; the median soil carbon sequestration rate ranged from 0.29 t C ha^{-1} yr^{-1} (n = 39) in willow (Fig. 6k) to 1.22 t C ha^{-1} yr^{-1} (n = 23) in *Miscanthus* (Fig. 6f).

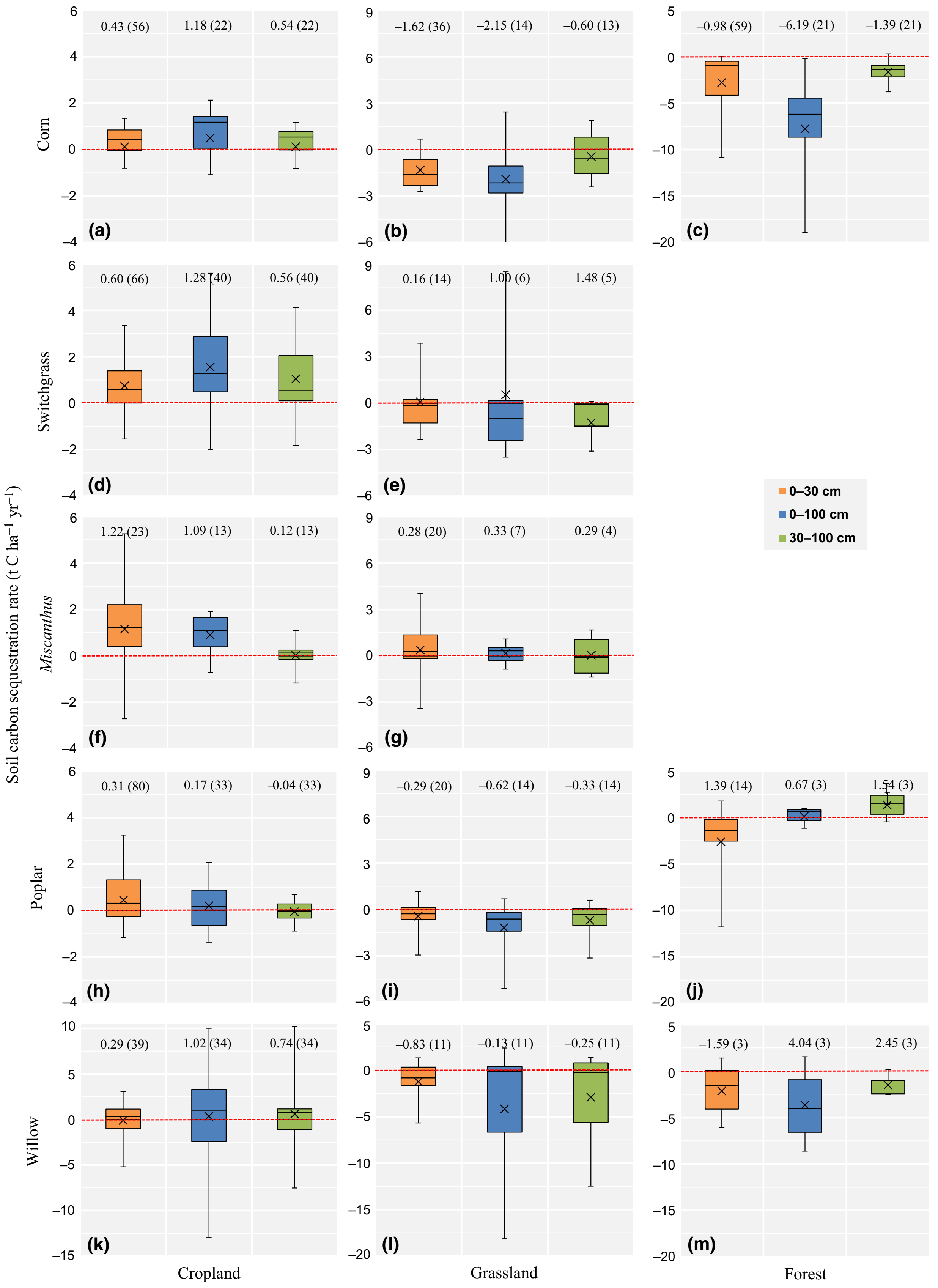
Soil carbon sequestration rate (t C ha−1 yr−1)
Corn
Switchgrass
Miscanthus
Poplar
Willow
Cropland
Grassland
Forest
0–30 cm
0–100 cm
30–100 cm
(a)
0.43 (56)
1.18 (22)
0.54 (22)
(b)
−1.62 (36)
−2.15 (14)
−0.60 (13)
(c)
−0.98 (59)
−6.19 (21)
−1.39 (21)
(d)
0.60 (66)
1.28 (40)
0.56 (40)
(e)
−0.16 (14)
−1.00 (6)
−1.48 (5)
(f)
1.22 (23)
1.09 (13)
0.12 (13)
(g)
0.28 (20)
0.33 (7)
−0.29 (4)
(h)
0.31 (80)
0.17 (33)
-0.04 (33)
(i)
−0.29 (20)
−0.62 (14)
−0.33 (14)
(j)
−1.39 (14)
0.67 (3)
1.54 (3)
(k)
0.29 (39)
1.02 (34)
0.74 (34)
(l)
−0.83 (11)
−0.13 (11)
−0.25 (11)
(m)
−1.59 (3)
−4.04 (3)
−2.45 (3)

Fig. 6 Soil carbon sequestration rate within three soil depths (0–30, 0–100, and 30–100 cm) under different land use change scenarios. The rates (t C ha^{-1} yr^{-1}) were estimated for land use changes from cropland, grassland, and forest to corn, switchgrass, *Miscanthus*, poplar, and willow, regardless of time horizon. The rates of 30–100 cm were estimated from soils with rates of both 0–30 cm and 0–100 cm. The nonparametric box plot: the bottom and top of the box are the first and third quartiles, and the band inside the box is the second quartile (the median). The cross is the arithmetic mean. The ends of the whiskers represent 5th (lower) and 95th (upper) percentile. The values show the median with number of datasets in parenthesis. No significance test was performed.

Median sequestration rates were also positive, and tended to be higher in estimates for the 0–100 cm depth, which is consistent with evidence that carbon change also occurs in deep soil for many LUC scenarios. This observation held for cropland converted to corn (Fig. 6a), switchgrass (Fig. 6d) and willow (Fig. 6k) where sequestration rates in the 0–100 cm soil depth exceeded those in the 0–30 cm depth. For those scenarios, median sequestration rates observed for the 0–100 cm depth were in general agreement with estimates based on the sum of rates estimated for the 0–30 and 30–100 cm depth increments. This was not the case for *Miscanthus* (Fig. 6f) and poplar (Fig. 6h) where the median sequestration rates were lower for the 0–100 cm depth and very low sequestration rates were suggested for 30–100 cm depths. For land conversions from grassland, all transitions except those to *Miscanthus* produced negative median soil carbon sequestration rates for all three soil depth-based comparisons. Lower median values and more negative observations suggested that 0–100 cm soil lost more carbon than 0–30 cm soil in most cases, for example, corn (Fig. 6b), switchgrass (Fig. 6e), poplar (Fig. 6i), and willow (Fig. 6l). The largest median rate of SOC loss [2.15 t C ha^{-1} yr^{-1} (n = 14)] was in the 0–100 cm soil depth after grassland conversion to corn. The smallest rate [0.13 t C ha^{-1} yr^{-1} (n = 11)] observed was for studies including change in deeper depths after grassland conversion to willow. Conversion to *Miscanthus* was the only grassland transition that produced positive median sequestration rates (Fig. 6g).

Of all the land transitions examined, conversion of forest to corn exhibited the greatest rate of SOC loss [−6.19 t C ha^{-1} yr^{-1} (n = 21)] in the 0–100 cm depth but estimates also had a very high degree of variance (Fig. 6c). The negative rates estimated for the 0–30 and 30–100 cm depths suggest that losses from forest converted to corn are derived from losses from both surface and subsurface depths. A limited number of observations showed a similar pattern of carbon loss upon forest conversion to willow plantations (Fig. 6m). Conversion from forest to poplar seemed to cause carbon loss in 0–30 cm soil at 1.39 t C ha^{-1} yr^{-1} (n = 14), but some carbon gains on average (n = 3) in the 30–100 cm (0.67 t C ha^{-1} yr^{-1}) (Fig. 6j). More studies are needed to verify changes in the poplar deep soil.

By evaluating the change in sequestration rate observed as a function of study duration (dSOC/dt), a substantial number of studies showed that soil carbon sequestration rates in many cases decline as soils reach a new equilibrium (Fig. 7a–c). This has been shown in previous studies (Stewart *et al.*, 2007; Qin *et al.*, 2013). The most noticeable changes occurred within about 10 years following land conversion, in many cases with SOC change rate of over ±5 t C ha^{-1} yr^{-1} for 0–30 cm soil (Fig. 7a) and ±10 t C ha^{-1} yr^{-1} for 0–100 cm soil (Fig. 7b). With longer time horizon postland transition, the sequestration rates decreased and approached zero in most instances as SOC change became minimal after 10 years. For example, the rates converged at zero after 20 years for C-C scenario in all soil depths (Fig. 7a–c). The only exception to this was for forests converted to corn, where high negative sequestration rates were sustained in many cases for longer time periods (Fig. 7a–c).

Discussion

The intent of this review was to infer the direction and order of magnitude of SOC change and sequestration rate observed under land conversion scenarios that are important for biofuel feedstock production and associated LCA. We note again that, although conversion of forests to feedstock production is concerning because of the associated carbon stock change, RFS provisions limit the likelihood of this land transition. Our synthesis of soil carbon sequestration rate results from field observations echoes other independent estimates, including modeling projections (Table 1). Freibauer *et al.* (2004) summarized potential soil carbon sequestration in Europe (mostly 30 cm) and found that, in cropland, reduced tillage and elevated residue return could augment soil carbon by up to 0.4 and 0.6 t C ha^{-1} yr^{-1}, respectively. Cropping of perennial grasses, permanent crops, and bioenergy crops could sequester soil carbon at the rate of 0.6 t C ha^{-1} yr^{-1}. Converting arable lands to grassland or woodland may provide an opportunity to sequester 0.3–1.7 t C ha^{-1} each year. Reversion of grassland, woodland, and permanent crops back to arable lands could cause soil carbon to drop by 0.6–1.7 t C ha^{-1} yr^{-1} (Freibauer *et al.*, 2004). With the terrestrial ecosystem model, Qin *et al.* (2012) estimated that (Table 1), grown on current croplands with corn, soy, and wheat, switchgrass and *Miscanthus* could sequester an additional 0.67–0.95 t C ha^{-1} of soil carbon (100 cm) annually within a 50-year time frame, which

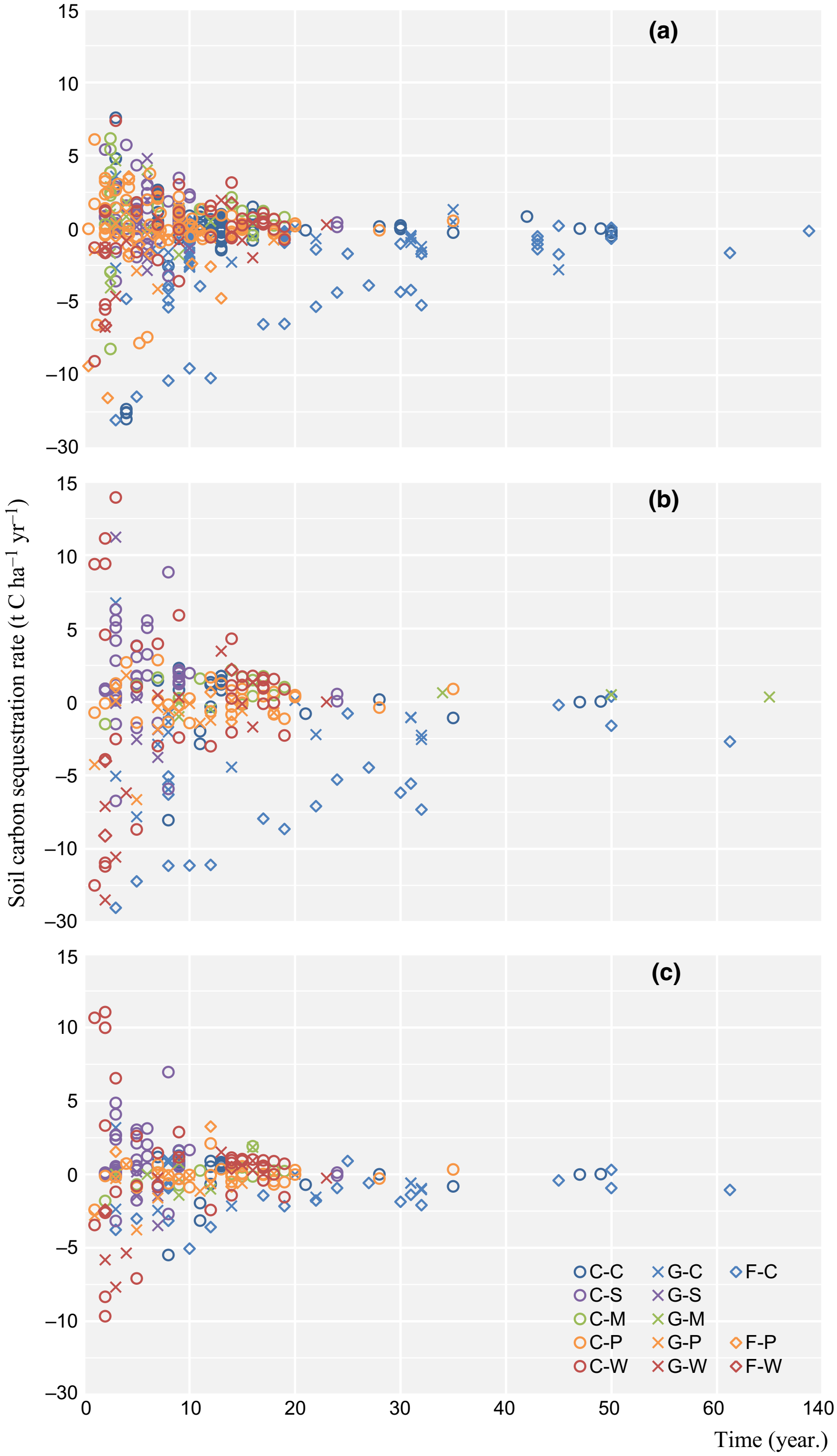

Fig. 7 Soil carbon sequestration rate changes with time. The rates were estimated for three soil depths: (a) 0–30 cm, (b) 0–100 cm, and (c) 30–100 cm, corresponding to Fig. 6. Land use types changed from cropland (C), grassland (G), and forest (F) to corn (C), switchgrass (S), *Miscanthus* (M), poplar (P), and willow (W).

can be roughly translated into 1.1–1.6 t C ha^{-1} yr^{-1} for a 30-year time period. Similarly, Mishra *et al.* (2013) used process-based and geospatial models and predicted a net soil carbon gain at the rate of 0.16–0.82 t C ha^{-1} yr^{-1} with growth of *Miscanthus* within US croplands (Table 1). Kwon *et al.* (2013) used a surrogate CENTURY model to estimate state-level soil carbon emission factors (0–30 cm) under various land

Table 1 Modeled soil carbon sequestration rates for biofuel land use change (LUC) in the conterminous U.S.

Reference	Land use: Historical	Land use: Current	Time (year)	Soil depth (cm)	Rate (t C ha^{-1} yr^{-1})	Notes
Qin *et al.* (2012)	Croplands with corn, soy, and wheat	Switchgrass and *Miscanthus*	50	100	0.67–0.95	The terrestrial ecosystem model (TEM) was used, 3 × 2 LUC scenarios were reported
Mishra *et al.* (2013)	Croplands	*Miscanthus*	N/S*	N/S	0.16–0.82	The Miscanmod model was used for biomass production, and mass-balance approach used for SOC
Kwon *et al.* (2013)	Croplands, grasslands, croplands/reserves and forests	Corn with various tillage and residue removal, switchgrass and *Miscanthus*	30	30	−0.35–0.54‡	A surrogate CENTURY model was used, 4 × 8 LUC scenarios were modeled at state level

*N/S, not specified.
‡Median value.

change scenarios from croplands, grasslands, and forest to corn, corn with stover removal, switchgrass, and *Miscanthus* (Table 1). Results showed large variances among states, depending on inherent climate and soil characteristics and anticipated productivity of feedstock. In cropland conversions, the median emission factors (opposite to soil carbon sequestration rate) ranged from −0.54 t C ha^{-1} yr^{-1} in *Miscanthus* to −0.05 t C ha^{-1} yr^{-1} in corn with 30% residue harvest under conventional tillage, which suggested net soil carbon sequestration under all cropland LUC scenarios. Grassland conversions produced net soil carbon sequestration in *Miscanthus* systems, neutral emissions in switchgrass and about 0.1–0.3 t C ha^{-1} yr^{-1} SOC loss in corn under different agricultural practices. Converting forest to *Miscanthus* was nearly carbon neutral in terms of median emission factor (−0.02 t C ha^{-1} yr^{-1}), but to corn and switchgrass boosted the median emission factor to about 0.15–0.35 t C ha^{-1} yr^{-1} (Kwon *et al.*, 2013). The rates predicted in the modeling were generally consistent with the wide range of soil carbon sequestration rates observed in our analyses (Fig. 6).

We encountered common data limitations (e.g. scope of coverage, data abundance, and lack of detail) that constrain meta-analysis. Gaps in data coverage included forest transitions to switchgrass and *Miscanthus*. The paucity of the data may be due to the fact that that transition is unlikely to occur in the initial establishment of a *Miscanthus* and switchgrass bioenergy program where land areas are anticipated to be sourced from current agricultural land and/or grasslands. Future efforts could overcome the absence of published replicated trials or field experiments for forest to perennial grass transitions using paired sides or the chronosequence method (Guo & Gifford, 2002; Don *et al.*, 2011). We acknowledge the potential for bias to be introduced for scenarios that rely on very small datasets like the forest-to-willow transition. Small datasets are problematic when they rely on studies that use methodologies skew results or have suffered from measurement difficulties, or high environmental variability (Guo & Gifford, 2002; Anderson-Teixeira *et al.*, 2009). Additionally and as previously noted, results may be biased in certain circumstances if database is dominated by data from certain regions, authors, or time periods, especially when the database is small (Guo & Gifford, 2002; Don *et al.*, 2011). Also, a more robust meta-analysis might be possible if reporting of means, sampling size and measures of variance were more universal (Johnson & Curtis, 2001). Given the range of spatial, temporal, climate, management, and land use history considered in this work, it is not possible to produce quantitative estimates of carbon dynamics for all land transitions considered in this study.

The results produced by this work are sufficient to contribute to ongoing efforts to refine estimates of direct SOC change to improve the accuracy and precision of biofuel LCA. Results from our meta-analysis suggest there is a positive response of 6–14% following conversions from arable crops (mostly annual crops, e.g. corn, soy, and wheat) to perennial (i.e. switchgrass and *Miscanthus*) and woody crops (i.e. poplar and willow). This effect is smaller than the 18–53% increase over initial SOC stocks reported by Guo & Gifford (2002) for pasture and forest lands converted from croplands based on meta-analysis of data from 74 studies. Their work and ours relied most heavily on data for converted croplands that came from North America and Europe.

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Trends for converted croplands are consistent with their results and other published summaries of field-based data noting SOC content generally declines with increasing frequency of disturbance and decreasing residue return and that, C stocks of croplands, especially those with annual crops, are generally lower than that stocks found in less disturbed ecosystems including those supporting perennial crops such as cellulosic biofuel crops (Anderson-Teixeira *et al.*, 2009), grasslands, or forests (Guo & Gifford, 2002; Don *et al.*, 2011). Past reviews of the literature have also shown that the extent of SOC change at any site will vary with the quantity and decomposability of organic matter inputs including crop residues, and changes in physical protection of SOC that result following LUC (Post & Kwon, 2000; Jastrow *et al.*, 2007). We did not account for soil texture or the ability of fine particles to provide physico–chemical protection of SOC (Six *et al.*, 2002; Jastrow *et al.*, 2007). Fine particles are positively related to SOC content and soil carbon sequestration potential (Angers *et al.*, 2011; Qin *et al.*, 2013) and may dampen the effects of LUC on SOC change. Anderson-Teixeira *et al.* (2009) noticed that clay content reduced the net change of SOC due to LUC, moderating losses upon conversion to corn or gains in switchgrass and *Miscanthus*. The extent of SOC change has also been shown to vary with climate (Don *et al.*, 2011). In Guo & Gifford's study (2002), the conversion of pasture to woody plantation had little impact on soil carbon when precipitation was low (<1200 mm), but significantly reduced soil carbon in high-precipitation (>1200 mm) areas, with about 23% carbon loss when precipitation was over 1500 mm. We did not explicitly consider geography or climatological factors in our analyses but note that conclusions about SOC change under various perennials that generally draws on data from forests in North America, Europe, and Asia are disproportionally allocated to specific bioenergy crops with poplar being studied in all regions, *Miscanthus* mostly in Europe with a few results from Asia, and switchgrass results coming exclusively from North America. Data for corn-based bioenergy production came from all regions including South America. The relative importance of edaphic factors of SOC change will become more apparent as new data become available.

As previously noted, LUC frequently alters SOC by varying inputs; accordingly, studies that consider residue harvest and return rates are particularly important for biofuel LCA. Many studies note crop residue removal for biofuel production can reduce organic matter inputs and therefore SOC, but suggest partial removal may be viable as long as return rates maintain SOC levels (Lal, 2005; Blanco-Canqui, 2010; Johnson *et al.*, 2014). Residue removal from biofuel production systems results in SOC loss when smaller amounts of C are returned to the soil than has been added by previous land use (Anderson-Teixeira *et al.*, 2009; Don *et al.*, 2011). Johnson *et al.* (2014) suggested that return of 6 t ha^{-1} yr^{-1} of residue may be a heuristic for minimum residue return. Conversion of croplands with low residue return rates to corn-based production for biofuels could result in net sequestration as most annual crops in the U.S. produce 4–8 t ha^{-1} of residue while corn can produce about 10 t ha^{-1} (Lal, 2005). If corn were grown on previous barley cropland, a 30% residue return would equal 70% of total barley residue (4.3 t ha^{-1}). Further, corn has a more extensive rooting system than some annual crops (e.g. soy, barley) and this can also contribute more to organic matter inputs (Dwyer *et al.*, 1988). Using 94 observations reporting residue return rates for croplands used for biofuel production, we did not find a consistent relationship between SOC change and corn residue return (Fig. 8). Our results suggested that, for land use transitioned from general crops to corn, SOC change was not significant when residue removal exceeded 70% and that SOC actually increased 15–23% when residue removal was <70%. Our finding is substantially different from results found by Anderson-Teixeira *et al.* (2009), who reported soil carbon loss in the 0–30 cm depth in corn-based systems with residue removal rates ranging between 25 and 100%. Using regression, they found SOC loss increased 0.2% (or 0.06–0.09 t C ha^{-1}) for every 1% increase in residue removal. Previous crop type, tillage, fertilization, and other soil and environmental factors that vary among studies can affect SOC change as much as, if not more than, residue returns (Dick *et al.*, 1998; Clapp *et al.*, 2000; Gale & Cambardella, 2000). Differences in our results may stem from a variety of factors including our use of a larger dataset, baseline characteristics, and specifics of the sites used for study. Our data represent trends observed over a greater geographic range. SOC change is typically calculated as the difference between current SOC (i.e. after LUC) and either the historical (e.g. based on the repeating sampling technique) or referenced SOC (e.g. based on paired-site or chronosequence techniques). For repeated sampling, the historical SOC is assumed to be at equilibrium at the time of land transition and resulting SOC change is attributed solely to LUC. Both paired-site and chronosequence techniques that sample baseline and new land use concurrently permit assessment of 'foregone' SOC change that might occur if land was maintained under historical practice. In this study (Table S1), the repeated sampling approach was frequently used in corn, switchgrass, and *Miscanthus*-based experiments, which are relatively easy to establish and monitor over a short time period. Willow and poplar experiments, however, were mostly based on paired-site and chronosequence designs. The referenced soils in such comparisons normally have a stable, nearly constant land use history. The difference between repeated measurement and paired-site designs may be minimal if the historical or referenced soils are at equilibrium, but can be significant if the historical or referenced land use is not well established, and SOC is highly unstable (Don *et al.*, 2011). Differences can also result if the productivity and thus residue return rates of the baseline system are changing or erosion occurs. While not a major source of uncertainty in this meta-analysis, changes in baseline productivity within annual crop systems might be a factor in our assessment if reliance on concurrent sampling of reference and new land use permitted us to credit improvements in the performance of the baseline system. Paired-site approaches do not effectively track net foregone changes in SOC because the baseline can change. As more data become available, baseline and site-specific factors affecting SOC change will be quantified. Future meta-analyses can help us determine when and where it is essential to account for these.

Soil depth has already been identified as a factor that should be considered in meta-analysis to improve estimates of SOC change driven by LUC (Ugarte *et al.*, 2014). It is well known that stocks of SOC below the rooting zone are as large, or larger, than those contained in the surface depths <30 cm typically considered by most LCAs. In many of our LUC scenarios, the magnitude of SOC change (response ratio) was similar and significantly different from zero in the 0–30 and 0–100 cm comparisons. Exclusive use of SOC changes from 0–30 cm in carbon stock accounting might not be appropriate if SOC change in deep soil is as important as change in surface depths. It is well known that about 50% SOC is stored in deeper soil depths (30–100 cm) in most soils (Jobbágy & Jackson, 2000). Following this reasoning, and assuming the global average SOC in croplands is about 110 t C ha^{-1} in the 0–100 cm soil depth (Jobbágy & Jackson, 2000), a 10% SOC change after land conversion would result in a 5.5 t C ha^{-1} increase in SOC within the 0–30 cm depth and an 11 t C ha^{-1} increase in the entire 0–100 cm soil if rates of change were equivalent. Increases or losses from the 30–100 cm depth could contribute greatly to changes in SOC stocks in LCA if trends could be reliably predicted. However, deep-soil comparisons suffer from data limitations; our meta-analysis cannot guarantee pairwise comparisons between surface and subsoil samples for all single-site observations. Many single-site studies fail to report statistically significant changes in SOC within deeper depths following LUC and so have been reluctant to advocate for comparisons based on change in deeper depths due to the costs for comparison (e.g. Yang *et al.*, 2008; Syswerda *et al.*, 2011). The significance and magnitude of SOC change in deep soils depends on LUC scenarios where in some LUC cases, the median response ratio is larger for 0–30 cm soil (e.g. Fig. 3a). This suggests that the subsoil may not affect the total SOC stock change as much as the top soil in systems like the crop-based (C-C; Fig. 3a) and crop-to-pasture (C-P; Fig. 5a) LUC scenarios. Note, that there are a very limited number of studies evaluating change in the deeper soil depths that allow us to separate the contributions of change in the top and subsoil for forest to poplar (Fig. 5a) or willow transitions (Fig. 5c). Future LCA studies will need to consider depth in soil SOC accounting by applying site- or region-specific observations with modeling efforts and report uncertainty along with estimates of change.

To accurately assess SOC stock change, the time duration of land transition in LCA would ideally match the time period used to evaluate LUC (e.g. change in SOC observed within 30 years by modeling or field observation). The most dramatic changes in SOC stocks occurred during the 5–10 years following land transitions for all expect F-C scenarios (Fig. 7). If an LCA with a 30-year time horizon adopted the SOC difference between current and historical land use types observed 10 years after LUC, the rate would likely capture the bulk of expected SOC change. If the LCA were to apply that rate to the entire 30-year time horizon, it would then overestimate SOC change because, for most land transitions, the soil carbon sequestration rate declines significantly after 10 years, approaching zero by

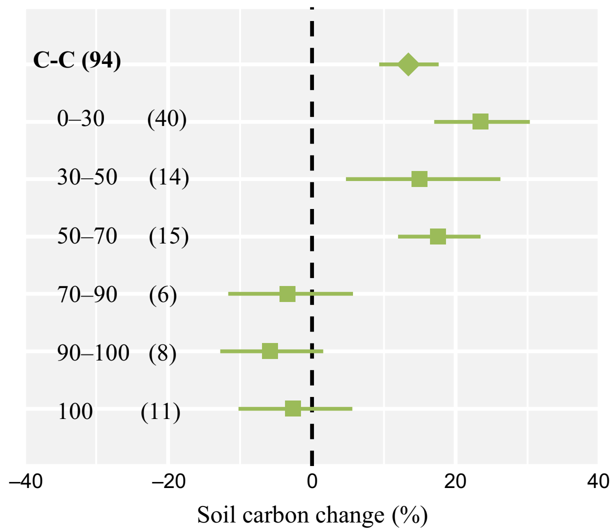

Fig. 8 Soil carbon change responding to land use change from cropland to corn with residue removal. The estimates show response ratio (% change of initial control) with 95% confidence interval (error bars) for different ranges of corn residue removal rate (%). Number of datasets is shown in parenthesis. Diamond symbols indicate overall response regardless of residue return rate.

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20 years. LCA practitioners should adjust rates to reflect SOC equilibrium status. It is important to note that in the F-C scenario soil carbon sequestration rates are higher than observed in other scenarios and approach a new equilibrium more slowly. More data are needed to determine whether LCAs examining this land transition should consider extending the time boundary over which SOC changes are considered to account for SOC changes after 30 years.

Considering uncertainties and spatial heterogeneity in field observations, well-validated process models may serve as a better alternative to quantify soil carbon dynamics and regional soil carbon change accounting on a large scale (Bondeau *et al*., 2007; Qin *et al*., 2012; Smith *et al*., 2012; Wang *et al*., 2013). Unlike most meta-analyses, model simulations are able to capture site- or region-specific soil and climate conditions and quantify their impacts on local SOC dynamics. Additionally, various agricultural practices (e.g. residue removal, tillage) can be incorporated in modeling experiments to identify possible influences from human disturbances in addition to natural environmental impacts. Also, the factors affecting SOC are easier to control in a systematic matter in models than in field studies established across variable field backgrounds (Bondeau *et al*., 2007; Kwon *et al*., 2013; Qin *et al*., 2014). Models are very well suited to studies considering different time horizons (10 vs. 30 years) and could be used to adjust soil carbon sequestration rates based on SOC saturation. Modeling-based studies can but do not always consider soil depth (e.g. shallow vs. deep soil) in estimates of soil carbon change or soil carbon sequestration rates. This study and others (Knops & Bradley, 2009; Don *et al*., 2011; Harrison *et al*., 2011) note deep soil plays a critical role in soil carbon dynamics and should therefore be further investigated to allow us to reliably predict the rate and direction of change under different scenarios.

This review of about 100 studies of SOC change related to biofuel feedstock production has raised two important points about the incorporation of SOC changes into biofuel LCA. First, SOC changes should incorporate deep soil carbon changes, which at present is not a common practice. More observational data are needed to reliably predict the rate and direction of SOC change in deeper soil depths. Second, the time horizon over which soil carbon sequestration rates (or emission factors) are calculated should be consistent with the LCA time horizon. Otherwise, the SOC change may be overestimated or underestimated. The results of this meta-analysis identify general patterns of SOC changes in biofuel-related LUCs. Because of the limitations of meta-analysis and our dataset, it is not advisable to incorporate values reported here directly into biofuel LCAs. In the absence of robust SOC change field data for the many LUC scenarios that are possible as biofuel feedstock production expands, SOC modeling is a key resource for developing SOC change estimates that can be used in biofuel LCA.

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Supporting Information

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

Table S1. References included in the land use change database.