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# The Role of Simulation Models in Monitoring Soil Organic Carbon Storage and Greenhouse Gas Mitigation Potential in Bioenergy Cropping Systems

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Manyowa N. Meki, James R. Kiniry,  
Kathrine D. Behrman, Meghan N. Pawlowski and  
Susan E. Crow

Additional information is available at the end of the chapter

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## 1. Introduction

There is an increased demand on agricultural systems worldwide to provide food, fiber, and feedstock for the emerging bioenergy industry, raising legitimate concerns about the associated impacts of such intensification on the environment [1, 2]. In the US, the revised National Renewable Fuel Standard program has mandated 30 billion gallons of renewable fuel by 2020 [3]. This aggressive promotion of the bioenergy industry in the US, and elsewhere in the world, is being fueled by several factors: the opportunity to reduce dependence on fossil fuels through renewable energy; the search for energy independence or security; its potential to reduce net greenhouse gas (GHG) emissions and hence provide mitigation options to combat climate change; and the possibility to improve farmers' incomes and rural economies, reduce national budget deficits and trade imbalances [4]. There is thus immense pressure on agricultural systems to provide the much needed feedstocks for this fast emerging industry. Of critical concern, however, is the fact that bioenergy feedstock production depends on finite resources such as land and water, and if not sustainably managed could have huge negative impacts on ecosystem services.

Of the many ecosystem services that could be impacted by the large-scale production of bioenergy feedstocks, soil organic carbon (SOC) plays a pivotal role in maintaining and enhancing the natural soil resource base and the GHG emission balance. According to the [5], periodic checks of SOC should be conducted to ensure that soil quality is not sacrificed for renewable bioenergy. Soil degradation causes carbon (C) loss and release of GHGs as a result

of accelerated decomposition from land use change or unsustainable land management practices [6]. To date, there is little or no information that can be used to assess the long-term impacts of bioenergy production on SOC sequestration and storage.

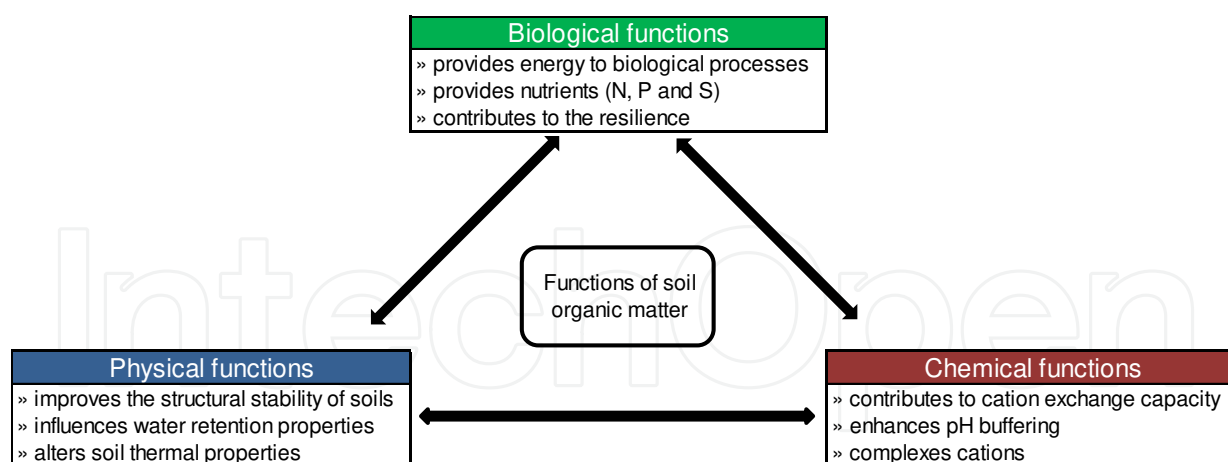
In this chapter we discuss and examine the potential application of process-based simulation models in monitoring SOC and its GHG mitigation potential. The first section highlights the critical functions and benefits of SOC in maintaining soil quality, and hence productivity and sustainability of cropping systems. In addition, we examine the ability of bioenergy cropping systems to sequester SOC and hence mitigate climate change. In the second part, we focus on the need for periodic checks and long-term monitoring of SOC. We point out the challenges of measuring and quantifying SOC at the field level and review selected modeling studies that illustrate how process-based simulation models can be applied to monitor SOC and GHG emissions. A discussion of the importance of life cycle assessment (LCA) in evaluating a bioenergy cropping system's effect on the GHG emission balance or global warming potential (GWP) is presented. Finally, we offer our perspectives on how simulation models could provide the tools to track SOC for the Carbon Credit Markets (CCMs), and briefly examine government policies that are necessary to ensure success.

## **2. Soil organic matter and soil organic carbon – Functions and benefits**

Soils contain C in both organic and inorganic forms. In most soils the majority of C is held as SOC. The term soil organic matter (SOM) is used to describe the organic constituents in the soil, while SOC refers to the C occurring in the SOM [7]. The terms SOM and SOC are often used interchangeably because SOC is the main constituent of SOM, which is typically around 57% C. Besides carbon, SOM also contains other important elements such as oxygen (O, ~40%), nitrogen (N, ~3%), as well as smaller amounts of phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), sulfur (S) and micronutrients [8].

Soil organic matter is composed of dead plant and animal material at various stages of decomposition. These materials are mainly plant residues and leaf litter, animal wastes and remains, plant roots and exudates, and soil biota, e.g., microbes and earthworms. Soil biota are continually breaking down the SOM through physical and biochemical reactions, releasing C and nutrients back to the soil, and GHGs such as carbon dioxide (CO<sub>2</sub>), nitrous oxide (N<sub>2</sub>O) and methane (CH<sub>4</sub>) to the atmosphere. Soils also contain inorganic C in the form of biochar and carbonate minerals, such as calcite, dolomite and gypsum.

Soil organic carbon plays a very important function in the growth of plants through maintenance as well as improvement of many soil properties. These functions of SOC can be classified into three broad categories: biological, physical, and chemical. According to [9], these categories are not static entities; instead, a change in one property will likely affect the other soil properties as well. An overview of the principal functions and benefits of SOM in soils is given in Figure 1.



**Figure 1.** The biological, physical, and chemical functions of soil organic matter (SOM). Source: adapted from [10].

*Biological functions* – Organic matter is a source of energy for many soil biota. Soil fauna play an important role in the initial breakdown of complex and large pieces of organic matter, making it easier for soil microorganisms to release C and plant nutrients during the process of decomposition. While microorganisms only make up less than 5% of SOM [11], they are the key driver of important processes such as decomposition, nutrient cycling, N fixation, and symbiotic plant relationships. These processes have direct and indirect effects on plant nutrient availability. It is estimated that 90-95% of the soil N, and about 2% of available P and S in natural ecosystems, come from organic matter.

*Chemical functions* – Humus is finely divided stable organic matter. Due to the large exposed surface area, humus contributes 30-70% of the cation exchange sites that adsorb plant nutrients that are eventually taken up by plants. Humus chelates micronutrients (usually iron, zinc, copper, or manganese) into soluble compounds, hence increasing their general mobility in soils and availability to plants. Humus also plays an important role in buffering the soil against any changes in acidity, salinity, and damage by pesticides. The exchange sites are also believed to help in cleaning up contaminated water by temporarily adsorbing heavy metal pollutants such as lead, cadmium, copper and zinc.

*Physical functions* – Plant roots, microbially-derived polysaccharides, and fungal hyphae cement soil particles into aggregates, thereby improving soil aeration, rate of water infiltration, and hence water holding capacity [12]. Build-up of organic matter lowers the soil bulk density, which allows the soil to be tilled with less horsepower, and improves plant rooting. Organic matter left on the soil surface, such as plant residues, also enhances rainfall capture, which reduces sediment and nutrient losses by runoff. Furthermore, organic matter shades the soil, which reduces excessive evapotranspiration and soil temperature in the summer, while keeping the soil warmer in winter.

A review of the literature shows that the productivity and sustainability of cropping systems are positively related to SOC content [13]. The ability to increase SOC in agricultural ecosystems is of interest both for sequestering atmospheric CO<sub>2</sub> and for restoring organic-matter

pools important to soil health [14]. The challenge is to identify soil management practices that promote SOC sequestration while ensuring productivity and profitability for farmers.

## 2.1. Bioenergy crop yields, impacts on soil organic carbon storage and greenhouse gas mitigation potential

Although most current bioenergy crop production is from annual row crops such as corn (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.], we do not discuss these here because they offer few or no environmental benefits over current agricultural practices [16]. Instead, we focus on second-generation bioenergy crops (Table 1) as identified by the US Department of Energy (DOE) [15, 16]: (1) the highly productive perennial warm-season grasses (WSGs) as emerging biofuel feedstocks for cellulosic ethanol production [e.g. switchgrass (*Panicum virgatum* L.), miscanthus (*Miscanthus giganteus*), Napiergrass (elephantgrass) (*Pennisetum purpureum* Schum.), and energy cane (*Saccharum officinarum* L.)]; and (2) the short-rotation woody crops (SRWCs) [e.g. hybrid poplar (*Populus* spp.) and willow (*Salix* spp.)].

In the US, average yields of selected WSGs ranged from 4.0 to 14.0 Mg ha<sup>-1</sup> yr<sup>-1</sup> [15]. More recent results indicate the potential to reach 20.0 Mg ha<sup>-1</sup> yr<sup>-1</sup> in several locations. Energy cane and Napiergrass can produce yields from 12.2 to 32.4 Mg ha<sup>-1</sup> yr<sup>-1</sup>. However, these species are very sensitive to frost and are limited to the southern regions of the US. Short rotation woody crop yields range from 10.0 to 17.0 Mg ha<sup>-1</sup> yr<sup>-1</sup>. This is 2-3 times the yields normally achieved by traditional forest management. Observed positive attributes of these bioenergy crops include: being perennials, they require less tillage than conventional row crops (tillage is costly and is detrimental to maintenance of SOC stocks); their deep and extensive rooting patterns effectively exploit and extract water and soil nutrients deep-down the soil profile; they require little or no external nutrient inputs due to nutrient retention and efficient nutrient cycling between growing years or harvests; most are adapted to low-quality land (nutrient-depleted, compacted, poorly drained, and eroded); and they are tolerant to a host of abiotic stresses, such as drought, heat, cold, low pH, metal toxicity and salinity. The perennial grasses switchgrass and miscanthus have the additional advantage of the C<sub>4</sub> pathway of photosynthesis, which is associated with high water use efficiency (WUE).

To date, there are a few empirical studies of SOC sequestration under both WSGs and SRWCs cropping systems. Therefore, the following discussion is based on data from studies carried out mainly in the US and Europe. More data is needed to provide a comprehensive understanding for these cropping systems.

### 2.1.1. Impacts of Warm Season Grasses

Growing WSGs offers promise to displace fossil fuels and reduce net CO<sub>2</sub> emissions through SOC sequestration [17]. When compared to row crops, the greater SOC storage under WSGs is attributed to greater aboveground biomass and root biomass concentration [18, 19]. Switchgrass produces about 6.7 Mg ha<sup>-1</sup> yr<sup>-1</sup> of total root biomass and half of this is found below the 30-cm soil depth [19], while about 10% is found below the 60-cm soil depth. Warm-season grass root systems, which often extend to nearly 3 m in the soil profile, enhance translocation

Common name	Scientific name	Observed yields (Mg ha <sup>-1</sup> )
<b>Perennial Warm Season Grasses (WSGs)</b>		
Switchgrass	<i>Panicum virgatum</i>	4 to 14
Miscanthus	<i>Miscanthus giganteus</i>	
Napiergrass (elephantgrass) (ratoon)	<i>Pennisetum purpureum</i>	12 to 32
Energy cane (ratoon)	<i>Saccharum spp.</i>	
<b>Short rotation woody crops (SRWCs)</b>		
Poplars, cottonwoods & hybrids	<i>Populus spp.</i>	
Willows	<i>Salix spp.</i>	
Eucalypts	<i>Eucalyptus spp.</i>	
Sycamore	<i>Platanus occidentalis</i>	10 to 17
Silver maple	<i>Acer saccharinum</i>	
Sweetgum	<i>Liquidambar styraciflw</i>	
Black locust	<i>Robinia pseudoacacia</i>	

Source: Modified from [15].

**Table 1.** Model bioenergy crops identified by the US Department of Energy (DOE).

of SOC to deeper layers, thereby reducing decomposition of SOC and promoting long-term C storage [19]. Data from a regional study across 42 paired sites in Minnesota, North Dakota and South Dakota showed that switchgrass stored about 2.0 Mg ha<sup>-1</sup> of SOC in the 0- to 5-cm depth, 7.7 Mg ha<sup>-1</sup> in the 30- to 60-cm depth, and 4.4 Mg ha<sup>-1</sup> in the 60- to 90-cm depth [20]. The fraction of SOC stored in deeper (>30 cm) soil profiles is critical for long-term SOC sequestration because this fraction has longer residence times and slower turnover due to reduced microbial processes and fluctuations in soil water content and temperature [21]

In central and northwest Europe, annual increases of SOC for the conversion of croplands to miscanthus differed widely among studies and among sites within one study, ranging from 6.9 to 7.7 Mg C ha<sup>-1</sup> yr<sup>-1</sup> [22 - 24]. Data from soil samples of miscanthus fields and adjacent reference croplands at four different locations showed SOC change rates ranging from 2.6 to 2.8 Mg C ha<sup>-1</sup> yr<sup>-1</sup> [22], while [23] found change rates of 0.8 to 2.2 Mg C ha<sup>-1</sup> yr<sup>-1</sup>. Since all the above studies were conducted in central and northwest Europe, this scatter emphasizes the impact of site-specific factors on SOC sequestration. Carbon sequestration rates in miscanthus fields were determined at 0.1-0.2 Mg C ha<sup>-1</sup>yr<sup>-1</sup> [25], while a comparative study of C sequestration in miscanthus and willow in Poland reported sequestration rates of 0.6 and 0.3 Mg C ha<sup>-1</sup> yr<sup>-1</sup>, respectively [26]. Observations by [27] show that the establishment of miscanthus on grasslands may lead to SOC losses and that the co-benefit of SOC sequestration mainly occurs under miscanthus established on former croplands.



The amount of SOC sequestered by WSGs is a function of site-specific factors; soil texture, management practices, initial SOC levels, and climate [28, 29]. The sequestration rate can be greater in soils with initial low SOC levels than in those that are about to reach their C saturation level. On average, perennial WSGs can sequester between 0 and 3.0 Mg C ha<sup>-1</sup> yr<sup>-1</sup> in the 0- to 5-cm soil depth [28]. In a clay loam in Iowa, switchgrass stored 0.8 Mg C ha<sup>-1</sup> yr<sup>-1</sup> in plant and litter biomass, which was higher than that in cool-season grass and row crops [30]. The SOC content under switchgrass was 45% more than under row crops planted in a sandy loam in Alabama [31]. A comparative evaluation of SOC sequestration in croplands, and under 5 to 8-yr-old WSGs across 10 sites in Indiana, showed only moderate increases in SOC storage: WSGs - 22.4 g C kg<sup>-1</sup>, compared to 19.8 g C kg<sup>-1</sup> under croplands [32]. A review of 115 studies of grassland SOC sequestration from 17 countries including Australia, Brazil, Canada, New Zealand, the UK, and the US reported SOC sequestration rates ranging from 0.1 to 3.0 Mg C ha<sup>-1</sup> yr<sup>-1</sup> [33]. Site-specific rates depended on biome type and climate.

Carbon sequestration rates during early grassland establishment on disturbed lands ranged from 0.3 to 0.5 Mg C ha<sup>-1</sup> yr<sup>-1</sup> [33, 34]. For the short term, perennial grasses may offer better economic returns due to the quicker establishment and the annual harvests. Data on SOC sequestration under Napiergrass and energy cane is scarce. One study on Napiergrass in the southeast US showed an average total soil C increase of 3.2 Mg ha<sup>-1</sup> after four years of cultivation [35].

While growing WSGs increases SOC concentration by 0.3 to 0.5 Mg ha<sup>-1</sup> yr<sup>-1</sup> [33, 34], crop residue removal (e.g., corn stover) reduces SOC sequestration by 1 to 1.5 Mg ha<sup>-1</sup> yr<sup>-1</sup> in the 0- to 30-cm soil depth [12]. Growing WSGs thus increases SOC concentration while providing alternative feedstocks for cellulosic ethanol production, unlike crop residue removal, which results in large losses of SOC. The increase in SOC with WSGs not only plays a major role in mitigating GHG emissions, it also enhances ecosystem functions.

### 2.1.2. Impacts of Short Rotation Woody Crops

Information on SOC storage has not been widely reported for SRWCs. The limited data available so far comes from some few studies in the US, Canada and Europe. Two US studies were established in Minnesota to specifically address the issue of SOC sequestration under SRWCs [36, 37]. In the study by [36], soil C content was monitored in poplar plantations established on previously tilled agricultural prairie land, and compared to adjacent control grass and arable fields, while in the study by [37], poplar plantations were established on grass and arable lands. In both studies, there was a net loss of SOC over the first 6–12 years of initial establishment of the plantations. The SOC loss was largely from the surface 30 cm of soil, suggesting that it was due to enhanced decomposition. While [37] observed no increase in SOC under the poplar plantations, [36] reported an average rate of SOC increase of 1.6 Mg C ha<sup>-1</sup> yr<sup>-1</sup>. Other research also indicates that SRWCs may result in high erosion and runoff rates during the first year(s) of establishment.

As with perennial WSGs, SRWCs have a huge potential to increase SOC due to the abundant above- and belowground biomass input. For example, one study by [38] showed that willow trees produce finer root biomass than corn (5.8 Mg ha<sup>-1</sup> versus 0.9 Mg ha<sup>-1</sup> for corn). On average,

SRWCs can store between 0 and 1.6 Mg C ha<sup>-1</sup> yr<sup>-1</sup> in the 0- to 100-cm soil depth [39]. On relatively fertile soils in Canada, willow trees stored more SOC than either corn or switchgrass after 4 yr of establishment [40]. A determination of C sequestration in forests and willow cultivations showed sequestration rates at about 0.2 Mg C ha<sup>-1</sup>yr<sup>-1</sup> [25]. In the US Midwest region, gains in SOC under SRWCs were greater than in croplands when the initial SOC concentration was low [41].

In the UK, estimates of SOC sequestration rates under SRWCs were based on willow and poplar plantations that were coppiced for varying time periods at three sites [42]. The average annual increase in SOC under SRWCs was estimated at between 1.2% and 2.2% [43]. In a study of three mixed poplar, aspen and willow plantation sites across Germany, substantial increases in SOC (0.1 to 0.6 Mg C ha<sup>-1</sup> yr<sup>-1</sup>) in the upper surface soil were observed after seven to nine years [44]. For much of Europe, research investigating potential SOC sequestration under SRWCs is still in its infancy.

According to [41], SRWCs should be grown in marginal and degraded lands where there will be greater accumulation of SOC rather than in croplands or natural forests. However, [45] argue that perennial WSGs may be more suitable for C sequestration since SRWCs take time for canopy closure, making soil more prone to SOC loss. If a management strategy increases the above- and belowground biomass, then a net gain in SOC sequestration can result.

### *2.1.3. Greenhouse gas mitigation potential*

In contrast to row crops, evidence shows that biofuels from perennial WSGs and SRWCs have a positive GHG mitigation potential. Carbon sequestration associated with bioenergy-based cropping systems could potentially offset 1 to 2 Pg C yr<sup>-1</sup> globally [46]. Three GHGs: CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub> contribute directly to the GHG emission balance of cropping systems [47]. Nitrous oxide is produced in soils mostly through nitrification and denitrification. The majority of agricultural soils are minor sinks or sources of CH<sub>4</sub> [48]. Carbon dioxide emissions come from diesel fuel used in farm operations and during the manufacture of agricultural machinery, seed, and agrochemicals. Accounting for the GHG emission balance of bioenergy cropping systems entails evaluating their net impact on all emissions associated directly and indirectly to bioenergy feedstock production [1].

In the US, 60 million (M) ha of land is potentially available for conversion to bioenergy plantations. The conversion to bioenergy crops could mitigate C emissions at a rate 5.4 Mg C ha<sup>-1</sup> yr<sup>-1</sup>. On 60 M ha, that would represent 324 Tg C yr<sup>-1</sup>, which is equivalent to a 20% reduction from current fossil-fuel CO<sub>2</sub> emissions [39]. In one study it was estimated that SRWCs sequester soil C at an average rate of 0.7 Mg C ha<sup>-1</sup> yr<sup>-1</sup> (Table 2). In addition, SRWCs could also generate substantial reductions in fertilizer and fuel use, reducing CO<sub>2</sub> and N<sub>2</sub>O emissions, for a net GHG mitigation potential of 1.1 Mg C ha<sup>-1</sup> yr<sup>-1</sup>. Accordingly, as much as 40 M ha of highly eroded, degraded or mined lands in the US could be planted with SRWCs with limited negative impact on the production of key food and fiber crops [49].



GHG category	Switch to short rotation woody crops
Number of observations	<b>35</b>
	Mg C ha <sup>-1</sup> yr <sup>-1</sup>
Soil carbon	<b>0.7</b> (-2.1 to 3.6)
Nitrous oxide	<b>0.2</b>
Methane	<b>no data</b>
Process and upstream emissions	<b>0.2</b>
Sum of GHGs	<b>1.2</b> (-1.6 to 4.0)
Maximum U.S. applicable area, M ha	<b>40</b>

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Source: [49].

**Table 2.** Carbon sequestration and greenhouse gas (GHG) mitigation potential in short rotation woody crops (SRWCs) – based on 35 field-based observations.

Fluxes of N<sub>2</sub>O and CH<sub>4</sub> of SRWCs remain unknown. Uncertainties regarding N<sub>2</sub>O emissions from nitrification and denitrification complicate the estimation of a system's GHG mitigation potential or GWP [1]. However, estimates of CH<sub>4</sub> consumption by soils are small, generally found to have little impact on net GWP [48]. One study across Europe noted much lower N<sub>2</sub>O emissions in forested versus agricultural lands [50]. A review of the potential of SOC sequestration and CO<sub>2</sub> offset by SRWCs in the US [51] reported C sequestration rates ranging from 0 to 1.6 Mg C ha<sup>-1</sup> yr<sup>-1</sup>, and a GHG mitigation potential of 5.4 Mg C ha<sup>-1</sup> yr<sup>-1</sup>. In contrast, little difference was found in the N<sub>2</sub>O emissions of annual crops versus those of poplar plantations [52]. Other researchers have estimated the bioenergy displacement of fossil fuels from SRWCs being as much as 4.9 to 5.5 Mg C ha<sup>-1</sup> yr<sup>-1</sup> [49]. In the majority of studies, N<sub>2</sub>O emissions are the main source or driver of the cropping system's GWP. The assumption is that C sequestration by bioenergy crops of around 0.25 Mg C ha<sup>-1</sup> yr<sup>-1</sup> makes bioenergy neutral in terms of the GHG emission balance [53]. Quantifying these factors is necessary to determine the net effect of bioenergy cropping systems on soil C sequestration and GHG emissions.

## 2.2. Monitoring soil organic carbon storage and greenhouse gas fluxes

Soil organic carbon is the single most important indicator of soil quality [54, 55]. In addition to the importance of SOC as an indicator of soil quality, accurate measurements of soil C are also crucial in the assessment of the GHG emission balance of bioenergy systems. In general, methods for the determination of soil C involve direct and indirect approaches. Direct methods employ field and laboratory measurements of soil C stocks or CO<sub>2</sub> fluxes over time, while indirect methods combine data collected in direct soil C measurements and process-based simulation models to predict soil C changes temporally and spatially.

### 2.2.1. Field monitoring

The International Climate Change Treaty (Kyoto Protocol) recognized removal of CO<sub>2</sub> from the atmosphere by plants as a valid approach to mitigating climate change [56] and identified

the need to conduct long-term monitoring of C stocks under various land uses. While the methods for analyzing the SOC concentration of a given soil sample are well established and easily carried out with high precision and negligible analytical error [57], field documentation of SOC changes faces many challenges because of the heterogeneity of soils, environmental conditions, land use history, sampling methods and analytical errors [58]. Here we highlight some of the challenges.

Variability in soil C estimates can be due to a number of factors or conditions: uncertainties in bulk density measurements as impacted by above and below-ground biomass inputs; rock and stone contents; compaction; sediment loss; and deposition by wind and water erosion. These conditions can drastically affect the accuracy of soil C sequestration estimates, implying that a relatively large number of soil samples will be required to obtain more reliable estimates. For example, [59] suggest that 20 to 30 soil samples (250 cm<sup>3</sup>) will be needed for estimates of mean SOC within  $\pm 10\%$  at the 95% confidence level, while [60] estimate that more than 100 samples (2.54 cm cores) would be needed to detect a 2–3% change in soil C under switchgrass.

The small annual incremental changes of soil C are difficult if not impossible to detect by present analytical methods. According to [61], noticeable soil C changes require a longer time period (e.g., at least 20 years). Statistically significant differences in SOC were observed after 5 to 10 yr [34, 62 - 64]. Coleman et al. [41] collected soil samples from 27 study sites across north central US to compare the soil C of short rotation poplar plantations to adjacent agricultural crops and woodlots. Soil organic carbon ranged from 20 to more than 160 Mg ha<sup>-1</sup> across the sampled sites. Woodlots consistently contained greater SOC than the other crops, especially at depth. Surprisingly, [41] observed little difference between paired poplar and switchgrass. Furthermore, there was no evidence of changes in poplar SOC relative to adjacent agricultural soils when considered for stand ages up to 12 years.

A review of the different approaches to measurement and monitoring of SOC stocks shows that the main challenge is in designing an efficient, cost-effective sampling and SOC stock estimation system [65]. Current field sampling methods rely on a set of measurements that are extrapolated to represent a given geographic area. However, such intensive sampling is time consuming and costly. Furthermore, quantifying SOC changes at national or regional scales requires even higher sampling densities [66]. Conant et al. [65] are however optimistic that alternative measurement techniques, such as the infrared probe used by [67] or gamma-ray spectroscopy [68], could allow field-wide measurements of SOC stocks, even though the technologies currently require extensive site-specific instrument calibration. In addition, eddy covariance methods have improved greatly over the last two decades, and if coupled with appropriate and thoroughly tested terrestrial biosphere models, could be an alternative method to identify portions of the net ecosystem exchange associated with SOC [69]. Until remotely sensed measurements are calibrated to subsurface information, remotely sensed SOC inventories should only be considered as minimum estimates [70].

Given the many benefits and functions of SOC, a number of soil monitoring networks (SMN) that target soil C and other related data have been implemented or are under development in a number of countries throughout the world. For example, the European SMN – ENVASSO (ENVironmental ASsessment of Soil for mOnitoring; [71] has partners from 25 countries and

a total of 33 334 soil C monitoring sites. The density of the European networks is high, with a median coverage of 300 km<sup>2</sup> for each monitoring site [58]. Although most SMNs are designed primarily for documenting soil C stocks and related data, their objectives do vary from country to country. For a more detailed review of the concepts, design and objectives of some of the SMNs across the world, see [72].

In the US there has been considerable investment in soil C surveys in accordance with the Energy Independence and Security Act (EISA) [73]. Two such programs are (1) the U.S. Geological Survey (USGS) carbon sequestration program ([www.usgs.gov/climate\\_landuse/carbon\\_seq/](http://www.usgs.gov/climate_landuse/carbon_seq/)). Two USGS national-scale assessments (subsurface and terrestrial) covering all 50 states and primary ecosystems are currently under way, and will provide the most comprehensive accounting of C storage potential in the US. The second program (2) is the USDA Natural Resources Conservation Service (NRCS) Rapid Assessment of US Soil Carbon (RaCA) (<http://soils.usda.gov/survey/raca/>). In general, the two programs were initiated to develop statistically reliable quantitative estimates of amounts and distribution of C stocks for US soils under various lands, differing agricultural management; to provide data to support model simulations of soil C change related to land use change, agricultural management, conservation practices, and climate change; and to provide a scientifically and statistically defensible inventory of soil C sequestration and GHG emissions for the US.

In Canada, a group of energy industries formed the GEMCo (Greenhouse Gas Emissions Management Consortium) in recognition of the potential of soil C sequestration as a possible GHG offset mechanism, together with the need to develop soil C measuring and verification protocols for C trading [74, 75]. The GEMCo, in partnership with Canadian soil conservation associations and government agencies, went on to initiate work aimed at monitoring and documenting soil C sequestration at the field level [69].

Some national [76, 77] and global [78, 79, 80] scale assessments of soil C stocks have been made. The World Bank-Global Environmental Fund's sustainable development project is supporting similar soil C monitoring initiatives in developing countries, e.g., Mexico [69]. In the US, the USDA NRCS soil databases; (1) State Soil Geographic (STATSGO) [81], provides generalized soil data for the entire US, and (2) the Soil Survey Geographic (SSURGO) provides detailed soil information for selected US counties [82]. Both these databases can be used to estimate SOC stocks [69]. Although at a lower resolution, the FAO's Soil Map of the World [79, 83] provides global estimates of SOC stocks.

Globally, a major issue with SMNs or any other soil C monitoring program is that there is no consensus on soil C measurement, monitoring, and verification methods. Post et al. [69] stress that there is an urgent need to develop robust, science-based, flexible, and agreed-upon practical protocols for monitoring and verifying temporal and spatial changes in soil C.

### *2.2.2. Application of simulation models*

Computer simulation models can complement and extend the applicability of information collected in field trials [84] and can be applied to obtain realistic assessments of bioenergy feedstock production effects on SOC storage and the GHG emission balance. Combining

measurement of SOC with models provides more flexibility and reliability because the model can be continually updated with new field sample data, and will be able to account for both C sequestration and GHG mitigation potential through LCA.

Although several soil C and GHGs emission models have been developed for conventional agricultural and forest systems, most of them have not been fully parameterized and effectively tested for application on WSGs and SRWCs. Detailed documentation on the main SOC models, their main features and comparative performance is widely available in the literature [85, 86, 87, 88, 89]. We focus on the applicability of a select number of process-based models that have been, or are currently being applied and hence could be adopted or adapted to monitor SOC sequestration and the GHG emission balance of bioenergy crops.

The amount of SOC in a given soil can increase or decrease depending on numerous factors, including climate, vegetation type, nutrient availability, disturbance, land use, and management practice. Most common models that describe this dynamic behavior or turnover of SOC follow the approach by [90], who proposed that soil C dynamics can be represented by two pools comprising one C pool and one residue pool, and can be described as follows:

$$dC_s/dt = hC_i - kC_s \quad (1)$$

where  $C_s$  is soil carbon ( $\text{Mg ha}^{-1}$ ),  $t$  is time (yr),  $C_i$  is carbon inputs from plant residues and roots ( $\text{Mg ha}^{-1}$ ),  $h$  is a humification factor and  $k$  is the apparent soil decomposition rate ( $\text{yr}^{-1}$ ). This model, while simple, is adequate for hypothesis development and testing, and for field-level data interpretation [91]. However, as more information on SOC dynamics became available, new multi-compartment models accounting for variations in turnover rates for different SOC pools were developed [92]. For example, the most widely applied soil C simulation models, CENTURY [93] and RothC [94], divide SOM into different pools with varying turnover rates (Table 3). This partitioning is supposed to reflect the average biochemical and physical properties of SOM in the pools (e.g., metabolic, structural and recalcitrant) or the accessibility of the SOM to soil microbes or catalytic enzymes, or constraints imposed on decomposition by environmental conditions [95].

In the CENTURY model, SOC is divided into the active, slow and passive pools, with mean residence times (MRTs, i.e., the time required for half of the SOM to decompose) of 1.5, 25, and about 1000 yr, respectively. The organic inputs are separated into metabolic (readily decomposable; MRT of 0.1–1 yr) and structural (difficult to decompose; MRT of 1–5 yr) pools as a function of the lignin:N ratio [96]. A more detailed discussion of the SOC pools is given in [95]. It is, however, important to note that the C pool compartments are only conceptual, and have not been verified experimentally [96]. Despite this drawback, compartmental models have proved to be reasonably good in simulating changes in SOC stocks.

Although the CENTURY model was originally developed for grasslands [97], versions of the model have successfully been developed to simulate crops [98] and forest systems [99]. Among the nine SOC models that were evaluated against long-term datasets at a NATO Advanced Research Workshop held at Rothamsted, UK, in 1995, the CENTURY model had the best

potential for adaptation to bioenergy cropping systems because of its integrated plant-soil approach and the availability of specific forestry subroutines [88]. A similar comparative study of six simulation models; RothC, CENTURY, EPIC [100], DNDC (DeNitrification DeComposition) [101], ECOSYS [102] and SOCRATES (Soil Organic Carbon Reserves And Transformations in agro-EcoSystems) [103] was conducted by [87]. Based on the model simulated outputs, the SOCTRATES model was best able to describe temporal (soil C changes observed in long-term experiments) and spatial SOC dynamics, and hence could be applied to scale-up soil C sequestration at the regional scale [87]. In another study, the model successfully predicted SOC change at eighteen long-term crop, pasture and forestry trials from North America, Europe and Australia [104]. A major advantage of the SOCRATES model is that it requires minimal data inputs, and is specifically designed to examine the impact of land use and land use change on SOC storage.

Despite sharing several basic concepts with the CENTURY model, the RothC model's applicability in monitoring SOC stocks in bioenergy cropping systems is rather limited due to the fact that it is solely concerned with soil processes, and does not contain a sub-model for plant production. In addition, it does not calculate annual returns of plant C to the soil from above-ground yields [88]. The model has however been successful applied to describe SOC evolution under several long-term experiments, across a variety of land use and climate types [105].

Residue type	Century	RothC	Residence time (yr)	C:N	Compounds
Litter	Metabolic	DPM	0.1–0.5	10–25	Simple sugars Amino acids Starch
	Structural		2–5	100–200	Polysaccharides
SOM	Active	BIO	1–2	15–30	Living biomass
		DPM			POM Polysaccharides
	Slow	RPM	15–100	10–25	Lignified tissues Waxes Polyphenols
		Passive	HUM	500–5000	7–10
		IOM			Clay: OM complexes Biochar

DPM, decomposable plant material; BIO, microbial biomass; RPM, resistant plant material; HUM, humified organic matter; IOM, inert organic matter; POM, particulate organic matter; OM, organic matter. Source: [95].

**Table 3.** The pools of SOM defined according to their mean residence times (MRTs) and corresponding compound classes.



Due to the general applicability of the CENTURY model, a number of process-based simulation models such as EPIC, CropSyst [106] and DAYCENT [107], either adopted or adapted the SOC modeling algorithms from the CENTURY model. EPIC has since evolved into a comprehensive agro-ecosystem model with a SOC module capable of simulating SOC dynamics in a wide range of plant species, including crops, native grasses and trees [100]. Recently, [108] successfully applied the EPIC model to simulate the production and environmental effects of perennial biomass feedstocks grown on marginal lands. The comparative assessment of six alternative cropping systems over 20 years showed that once well established, perennial herbaceous grasses have a direct GHG emissions mitigation capacity ( $-2.3 \pm 0.1 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ ) which is higher than that of conventional crops. Meki et al. [84] applied the sister model, ALMANAC [109], to evaluate sustainable energy sorghum biomass harvest thresholds and tillage effects on SOC and bulk density. Model results showed that 75% of energy sorghum biomass can be removed from a continuous no till system without any detrimental effects on SOC storage. This level of biomass removal, however, significantly increased soil bulk density, a critical indicator of soil compaction.

By integrating outputs from the REAP model [111] with environmental impact estimates derived from the EPIC model, [110] were able to evaluate the environmental and economic impacts of increased ethanol production from switchgrass. Highlights from this study indicated that commercial-scale switchgrass production could be associated with reduced erosion and reduced nutrient losses, but the projected increases in N fertilizer application and the associated  $\text{N}_2\text{O}$  emissions could offset soil C sequestration benefits and result in substantial increases in total GHGs emissions from the agricultural sector.

In many European countries, soil C and GHG simulation relies on the Yasso07 soil C model [112]. This model describes litter decomposition and soil C cycle based on the chemical quality of organic matter and the climatic conditions. Total soil C stocks are estimated for the top-most 1 m soil layer. Since Yasso07 is a reasonably new model [113], not many studies on the model's applicability to GHG gas inventory purposes are available. The strategy when developing the Yasso07 model was to achieve a soil C model that had fewer input data requirements, thus ensuring wider applicability [114]. A recent study in southern Finland by [112] showed the Yasso07 model accurately predicting the long-term soil C stocks of a coniferous forest soil ( $67.0 \text{ Mg ha}^{-1}$ ), which accumulated at an average rate of  $0.12 \pm 0.06 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ . Previously, [115] evaluated the performance of the Yasso07 model in predicting the soil C stock changes in afforestation and deforestation sites. Overall, there was a good agreement between simulated and measured soil C stocks at most sites.

Garten and Wullschleger [60] used a simple, parameterized two-compartment model for SOC dynamics to predict the potential for soil C sequestration under switchgrass in the southeastern US. Model simulations indicated a measurable and verifiable recovery of soil C ( $\sim 12\%$  increase) on degraded lands through one decade of switchgrass production. A sensitivity analysis of the model indicated that switchgrass potential to sequester C depends on initial soil C inventories, prevailing climate, soil type, and site management.

The Terrestrial Ecosystem Model (TEM) is a process-based model that describes C and N dynamics of plants and soils for terrestrial ecosystems (<http://ecosystems.mbl.edu/tem/>). The

TEM uses spatially referenced information on climate, elevation, soils, vegetation and water availability as well as soil- and vegetation-specific parameters to make monthly estimates of C and N fluxes and pool sizes of terrestrial ecosystems. Conversion of agricultural lands to bioenergy crops consequently affects the ecosystem C balance. Qin et al. [116] applied the TEM model to evaluate the impacts of this change on the C balance, assuming several land use change scenarios from corn, soybean, and wheat to bioenergy crops of switchgrass and miscanthus. Overall, SOC increased significantly when land use changed from corn, soybean or wheat, to either switchgrass or miscanthus, reaching an average of  $\sim 100 \text{ Mg C ha}^{-1}$  in the two bioenergy crops.

Relatively simple statistical models can also be useful in estimating SOC losses associated with land conversion. Anderson-Teixeira, et al. [117] used reduced ANOVA and the forced- and free-intercept regression statistical models to estimate and compare SOC losses associated with land conversion and SOC sequestration rates under corn, sugarcane, switchgrass, miscanthus and mixed native grass. Their predicted SOC accumulation rates under switchgrass, miscanthus and mixed native grass ranged from  $0.1\text{--}1.0 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$  in the top 30 cm, while SOC losses under corn and sugarcane ranged from  $3.0\text{--}8.0 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ . These results are consistent with data from numerous field studies reported elsewhere in the literature [20, 28, 31, 40].

Life-cycle assessments are commonly used to determine whether a biomass biofuel system results in a net reduction in GHG emissions or an improved energy source. In LCAs, all input and output data in all phases of the product's life cycle including biomass production, feedstock storage, feedstock transportation, biofuel production, biofuel transportation and final use are required. Because field measurements of bioenergy LCA components are difficult to carry out, simulation models have been applied effectively to obtain realistic estimates of the bioenergy crop's GWP or GHG mitigation potential.

Standard and useful tools for LCA are multidimensional spreadsheet models such as the GREET [118] and GHGenius [119] software models, which were specifically designed to address full LCA of biofuels. These spreadsheet models have advantages in that they are user-friendly, publicly available, straightforward, and relatively transparent. In a study commissioned by the US Department of Energy, Wang [120] applied the GREET model to characterize the soil C sequestration and GHG mitigation potential for the bioenergy crops switchgrass, poplars and willows. Model estimates indicated that there is a 70-85% reduction in GHGs emission when agricultural lands are converted to grasslands and forests.

DAYCENT is the daily time-step version of the CENTURY biogeochemical model [121]. DAYCENT simulates fluxes of C and N among the atmosphere, vegetation and soil [107]. Adler et al. [122] routinely applied the DAYCENT model to assess and compare GHG fluxes and biomass yields for corn, soybean, alfalfa, hybrid poplar, reed canarygrass, and switchgrass as bioenergy crops in Pennsylvania, US. All simulated cropping systems provided net GHG sinks compared with the fossil fuel life cycle, even in the long term when there were no further increases in soil C sequestration due to SOC storage reaching a new steady state. Switchgrass and hybrid poplar (*Populus* spp.) provided the largest net GHG sinks, ( $> 0.5 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ ) for biomass conversion to ethanol, and  $> 1.1 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$  for biomass gasification for electricity generation. Compared with the LCA of fossil fuels, switchgrass-derived biofuels

reduced GHG emissions by more than 115%. Similar results were reported by [123], who also applied the DAYCENT model to calculate the net GHG fluxes, when converting from cotton and unmanaged grasslands to a switchgrass system in the southern U.S. Long-term simulations predicted a decrease in GHG emissions ( $0.3\text{--}1.0\text{ Mg C ha}^{-1}\text{ yr}^{-1}$ ) for conversions from cotton to switchgrass at N application rates of  $0\text{--}135\text{ kg N ha}^{-1}$ . Conversely, conversion from unmanaged grasses to switchgrass resulted in annual increases of net GHG emissions ( $0.1\text{--}0.4\text{ Mg C ha}^{-1}\text{ yr}^{-1}$ ) for switchgrass at no and low ( $45\text{ kg N ha}^{-1}$ ) fertilization rates. Using the DAYCENT model, Davis et al. [124] estimated the effects of replacing corn ethanol with switchgrass and miscanthus on GHG emissions in Central US. Overall, there was a 29 to 473% reduction in GHG emissions even after accounting for emissions associated with indirect land-use change. Conversion from a high-input annual crop to a low-input perennial crop for biofuel production can thus transition the central US from a net source to a net sink for GHGs. A simulation study with the DAYCENT model showed that cultivation of energy cane on former pasture on Spodosol soils in the southeast US has the potential for high biomass yield and a positive GHG emission balance [35].

The USGS General Ensemble Biogeochemical Modeling System (GEMS) [125] was developed to integrate well-established ecosystem biogeochemical models, such as CENTURY, with various spatial databases for the simulation of biogeochemical cycles over large areas. GEMS accounts for the impacts of atmospheric and climatic changes (e.g., changes in precipitation and temperature,  $\text{CO}_2$  enrichment, N deposition), land use change (e.g., biofuel crops, tillage, forest logging, reforestation, urbanization, fire fuel treatment), natural disturbances and extreme events (land fire, insect outbreak, drought, flooding, hurricane) on ecosystem C balances and GHG emissions.

Even with all the uncertainties associated with LCAs, most suggest that WSGs and SRWCs have higher net energy balances and positive GHG emission balances than conventional crops [122, 126, 127]. Sarker [128] concludes that the use of computational models to quantify GHG fluxes in LCAs of bioenergy crops gives a more accurate representation of the system than using constant IPCC [129] emission factors. The USDA-ARS GRACEnet (Greenhouse gas Reduction through Agricultural Carbon Enhancement network: [www.ars.usda.gov/research/GRACEnet](http://www.ars.usda.gov/research/GRACEnet)) research initiative compares various GHG mitigation strategies based on field studies in several agroecoregions of the US. It is hoped that results from this environmental research program will be important to inform public policy on valuing the benefits of bioenergy crop production, and in addition, provide the much needed data for model GHG emission calibration and testing.

### 2.3. Carbon Credit Markets and Enabling Policies

It has been demonstrated conclusively that bioenergy crops can sequester soil C and mitigate GHG emissions. Accreditation of bioenergy production to CCMs will provide the much needed incentives for land managers and bioenergy feedstock producers to expand production, both for GHG mitigation benefits and the resulting improvements to soil health. Pioneered by the European Union's (EU) Emissions Trading Schemes (ETSs) launched in 2005, several CCMs are now operating across the world - see [130]. For agriculture, the schemes or

programs allow producers and landowners to earn income by storing C in their soil through GHG capture projects. Soil C credits are then traded on CCMs, much like any other agricultural commodity. Large companies and other entities purchase C credits on the CCMs to offset their own GHG emissions into the atmosphere.

Carbon credits are allocated to an enterprise or project based on verifiable GHG emission reductions. For bioenergy feedstock producers to participate in CCMs there is need for low-cost, simple, but yet reliable methods for verifying soil C change in bioenergy cropping systems under different soil types, climates and management practices. As discussed earlier, field monitoring of soil C to document soil C changes is not only cost prohibitive but is generally not feasible due to the spatial variability in C stocks, and for most systems, the small annual incremental changes are not detectable by present analytical methods. According to [131], quantifying soil C may be even more costly than C stored in above-ground biomass, considering that soil C cannot be observed in the ways that biomass can. It is apparent that fully parameterized process-based models could be adopted or adapted to provide the much needed tools to track soil C sequestration and GHG emissions for the CCMs. As pointed out by [132], a major reason for developing soil C or agroecosystem models is to be able to use them to predict SOC changes under climate-soil management combinations lacking in soil C measurements.

Models such as Century, RothC, EPIC, SOCRATES, and more recently, C-Farm [92] can be used to assess and verify C sequestration rates and GHG mitigation potential for the CCMs. For most models, applicability is often limited because they require numerous data inputs. More user-friendly tools with minimum calibration requirements are needed. The Comet 2.0 tool (<http://www.cometvr.colostate.edu/>) is an easy to use Carbon Management Tool and Greenhouse Gas Accounting system for farm-ranch-orchard operations. Growers in the US can apply the Comet tool to obtain farm level mean estimates and uncertainty of C sequestration and GHG emissions from annual crops, hay, pasture and range, perennial woody crops (orchards, vineyards), agroforestry practices, and fossil fuel usage. The tool links a large set of databases containing information on soils, climate and management practices to dynamically run the CENTURY model as well as empirical models for soil N<sub>2</sub>O emissions and CO<sub>2</sub> from fuel usage for field operations. Similarly, C-FARM is a user-friendly cropping systems model that allows calculating rates of C storage for the soil profile on a layer-by-layer basis. For simplicity, the model is built around a robust single-pool C model and can provide estimates of short- and long-term on-farm C storage rates. According to [92], C-FARM is a useful tool for growers, consultants, and state and federal agencies that need to develop C storage estimates for local conditions.

The energy crisis of 1973, and more recent concerns about global climate change and energy security have led to significant policy support for bioenergy throughout the world. The United States, European Union (EU), Australia, Canada and Switzerland spent at least \$11 billion on biofuel subsidies in 2006 [133]. Here we briefly examine policies that could stimulate increased C sequestration by promoting bioenergy feedstocks production. Some suggest that, policy changes must focus on finding new avenues to divert public support to producers and



landowners in the form of stewardship or green payments as well as making progress in encouraging the success of private C trading systems [28].

Most producers are reluctant to take their land out of conventional crop production into bioenergy crops for a long period of time. Governments should introduce policies that include programs that help farmers make the transition to growing bioenergy crops. Although only partially implemented, the USDA Biomass Crop Assistance Program (BCAP) is meant to provide cost-share payments for establishing dedicated energy crops and annual payments to cover the foregone income from alternative uses of the land [28]. Such payments give producers the opportunity to preserve their land by increasing the SOC pool, while at the same time producing a crop that helps to meet future energy needs.

It is however important to note that while subsidies do create incentives for a transition to biofuels, they may also encourage increased fuel consumption by lowering the price of blended fuel, thus mitigating the benefits of land conversion to biofuels through increased GHG emissions. Many environmental economists recognize that a tax or fee on CO<sub>2</sub> emission from fossil fuel sources is the most efficient system to reduce emissions and spread the burden equitably across all sources: industrial and personal [134]. A CO<sub>2</sub> tax is currently under discussion in a number of countries, including the US and EU. In general, the tax would be applied to all fossil fuels before combustion according to the C content of the respective fuel.

Although CCMs keep developing throughout the world, their sustained future existence is uncertain as they face many challenges: the global economic crisis and its impacts on trading markets; failure by governments to pass cap-and-trade legislation, especially the US, one of the world's largest GHG emitters; great uncertainty over future investments in some developing countries as CCM investors do not know where demand will come from; and finally, the international community's inability to agree on a post-2012 Kyoto framework in Copenhagen (COP15) and Cancun (COP16), which has seriously damaged the confidence of the private sector in the long-term viability of existing CCMs [135].

### 3. Concluding remarks

The potential for bioenergy landscapes under WSGs and SRWCs to sequester soil C and mitigate GHG emissions is indisputable. The functions and benefits of SOC in the maintenance of soil quality, productivity and sustainability of cropping systems are well documented. It is imperative to monitor the SOC integrity of bioenergy crops through the establishment of stringent SOC measurement and quantification methodologies. Available information suggests the best approach would be to combine field measurements with process-based simulation modeling. While models can capture complex C dynamics as impacted by site-specific factors or conditions, field measurements are crucial for the verification of model outputs. Besides promoting participation in CCMs by producers, accurate monitoring of SOC stocks will also allow assessment of whether a given bioenergy cropping system meets government-stipulated life cycle GHG emission reductions. Whether or not countries imple-



ment policies that stimulate increased C sequestration through bioenergy cropping, the importance of enhancing the soil resource base should be increasingly highlighted.

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## Author details

Manyowa N. Meki<sup>1\*</sup>, James R. Kiniry<sup>2</sup>, Kathrine D. Behrman<sup>2</sup>, Meghan N. Pawlowski<sup>3</sup> and Susan E. Crow<sup>3</sup>

\*Address all correspondence to: nmeki@brc.tamus.edu, jim.kiniry@ars.usda.gov

1 Texas A&M AgriLife Blackland Research and Extension Center, Temple, US.

2 USDA, Agricultural Research Service, Grassland Soil and Water Research Laboratory, Temple, US.

3 Dept. of Natural Resources and Environmental Management, University of Hawaii at Manoa, Honolulu, US.

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