Sustainable Bioenergy Feedstock Production Systems: Integrating Carbon Dynamics, Erosion, Water Quality, and Greenhouse Gas Production

Jane M. F. Johnson[1], Jeffrey M. Novak[2]
[1]USDA-Agricultural Research Service, North Central Soil Conservation Research Lab, Morris, MN
[2]USDA-Agricultural Research Service, Coastal Plains Research Center, Florence, SC

CHAPTER OUTLINE

NB: The U.S. Department of Agriculture offers its programs to all eligible persons regardless of race, color, age, sex, or national origin, and is an equal opportunity employer.

Abbreviations: ARS, Agricultural Research Service; CO_2, carbon dioxide; CRP, Conservation Reserve Program; GRACEnet, Greenhouse gas Reduction through Agricultural Carbon Enhancement network; GHG, greenhouse gas; HPEC, herbaceous perennial energy crops; CH_4, methane; N_2O, nitrous oxide; REAP, Renewable Energy Assessment Project; SOC, soil organic carbon; SOM, soil organic matter; WPEC, woody perennial energy crops

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INTRODUCTION

There are multiple forces driving the paradigm shift toward using more renewable energy, although justifications are shrouded in debate and controversy. First, energy, national security, and rural development are toted frequently as drivers for expanding the use of biofuels; extensive socio-political and economic implication have been discussed (Charles et al., 2007; Steenhof and McInnis, 2008; Demirbas, 2009; Hoekman, 2009; Londo et al., 2010). Second, fossil fuels are a finite resource, yet fuel consumption and prices are increasing (Duncan and Youngquist, 1999; Pimentel et al., 2004; Charles et al., 2007; Nehring, 2009; OCHA, 2010; U.S. EIA, 2010). Third, burning fossil fuel is currently a major contributor to GHG emission, which is linked to global warming and climate change (IPCC, 2007a). In response to these interrelated drivers, multiple renewable energy sources are being developed and expanded to help meet the growing energy demands.

Hydropower, wind, solar, geothermal, biomass, and hydrogen are renewable energy sources available for power and/or liquid fuel production. Unlike some of the renewable energy forms, biomass can be used for production of multiple fuel sources, including heat, power, electricity, and liquid fuels. From 2009 to 2025, the demand for power and liquid fuels is predicted to increase by 25% and 10%, respectively (U.S. EIA, 2011a), with renewable energy sources expected to provide a larger percentage of the overall energy consumed (BRDB, 2008; Hoekman, 2009). Renewable biomass feedstocks include crop biomass (grain and crop residues), dedicated woody and herbaceous energy crops, and organic waste (e.g. manure) products (Wilhelm et al., 2004; Perlack et al., 2005; Hill et al., 2006; Johnson et al., 2007c; Lal and Pimentel, 2007; BRDB, 2008; Lal, 2009). Although biomass fuel currently represents only a small percentage of the total U.S. energy consumption, future demand might represent a substantial (>500 Tg yr^{-1}) amount of biomass (Perlack et al., 2005; BRDB, 2008). Thus, a serious effort to expand dedicated bioenergy feedstock production and/or integrate with traditional food crops is expected to meet the growing feedstock demand. Potentially, this could place stress on the environment due to land-use changes, increased tillage, expanded fertilizer use, and increased (direct or indirect) GHG production.

Ideally, renewable energy sources provide C-neutral or C-negative energy; thereby, mitigating GHG emission (IPCC, 2007b; Johnson et al., 2007c; BRDB, 2008; Hattori and Morita, 2010). However, the extent that bioenergy exacerbates or mitigates GHG emission is controversial and highly dependent upon feedstock (e.g. corn grain or perennial grass), production systems (e.g. tillage and nutrient management), energy conversion platforms (e.g. fermentation or thermochemical), and assumptions used in life-cycle analysis (Farrell et al., 2006; Fargione et al., 2008, 2010; Steenhof and McInnis, 2008; Borjesson, 2009; Malça and Freire, 2011). For example, life-cycle analyses vary in how they treat co-products such as dry distiller's grain, conversion efficiency of feedstock to ethanol and assumptions concerning potential indirect land-use changes.

Across the wide range of the U.S.'s climatic regions, there is a vast array of native and agronomic ecosystems for production of biomass feedstocks. Recently, a regional biomass feedstock roadmap for the U.S. was developed, based on feedstocks, land use and environmental concerns (Braun et al., 2010).The eastern region is dominated by forest and special uses (urban), the central region has the largest percentage of cropland, while the western region is mixed but has the largest acreage of grassland, as well as substantial forest and special-use acreages (http://www.ers.usda.gov/Data/MajorLandUses/map.htm) (Figure 8.1). A combination of bioenergy feedstocks may be available in any of the three regions. Based on existing land-use patterns, woody perennial energy crops (WPEC) would be the expected major feedstock in eastern and portions of the western regions; whereas, herbaceous perennial energy crops (HPEC) would be a major feedstock in other areas of the western and portions of the central regions. Due to the high acreage of cropland, crop residues are expected to be a major feedstock in the central region. Local specialty crops such as hay from grass-seed production in the Pacific Northwest or rice bagasse could provide local or regional scale feedstocks for biofuel

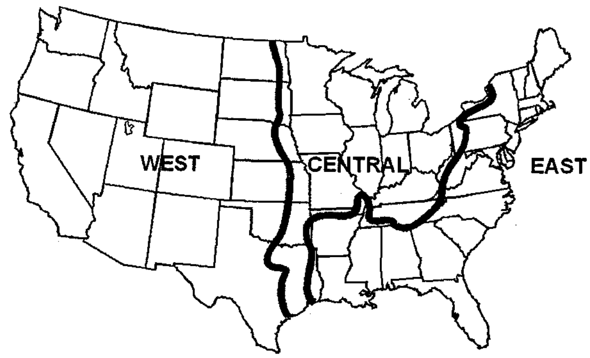

FIGURE 8.1
Delineation of three broad regions in the U.S.

production (Johnson et al., 2010c). However, across all regions and feedstock types there is a need to safeguard against soil organic carbon (SOC) loss and soil movement via wind and water erosion (Johnson et al., 2010c).

If biomass harvest decreases SOC content (Cowie et al., 2006) or increases GHG emission, then the benefit of displacing fossil fuel would be reduced or lost (Adler et al., 2007). Unfortunately, there is a dearth of data on the direct response to harvesting crop residues on GHG emission compared to returning residues to the soil. Jacinthe and Lal (2003) compared N_2O flux from no-till plots amended with 0, 8, and 16 $Mg\,ha^{-1}$ wheat (*Triticum aestivum* L.) straw, and with or without N fertilization. The presence of straw and N fertilizer on bare silt loam soil had highest N_2O emission. In contrast, in a multi-year field study on a sandy clay loam, the addition of wheat, barley (*Hordeum vulgare* L.), canola (*Brassica napus* L.) or pea (*Pisum sativum* L.) residue had negligible impact on N_2O flux, but there was an interaction with N application (Malhi and Lemke, 2007). Likewise, two cycles of returning or harvesting corn (*Zea mays* L.) stover residue on a loam soil had negligible impact on N_2O flux (Johnson and Barbour, 2010). Clearly, the diversity of responses points to a need for research to examine the impact of harvesting biomass feedstocks on GHG emission. Fortunately, we are aware of several recently initiated studies across the U.S. that will help assess biomass harvest on GHG emission and SOC content. For example, experiments in Minnesota, Iowa, and South Dakota are assessing the impact of corn stover removal on direct GHG emission in conjunction with the USDA-Agricultural Research Service (ARS) Renewable Energy Assessment Project (REAP) (http://www.ars.usda.gov/research/programs/programs.htm?np_code=212&docid=21224) and a broad integrated study funded through USDA and DOE (Karlen, 2010; Wilhelm et al., 2010).

Historically, maintaining SOC was not an important consideration since typical agricultural practices (e.g. land clearing, tillage, monoculture, replacing perennials with annuals, etc.) have contributed to its decline (Reicosky et al., 1995; Lal et al., 1998b; Balesdent et al., 2000; West and Post, 2002; Franzluebbers and Follett, 2005; Novak et al., 2009a). As a result of land-use change, SOC content declined between 20 and 75% compared to pre-agriculture levels (Bruce et al., 1999; Slobodian et al., 2002; Johnson et al., 2005; Liebig et al., 2005b). The magnitude of SOC decline varied greatly in these studies due to differences in regional climate, years under cultivation, and slope position. Substantial SOC decline has raised concerns that harvesting bioenergy feedstocks, especially over-harvesting crop residues, could further exacerbate SOC losses (Wilhelm et al., 2004; Lal, 2008b) and cause other agronomic or environmental damage (Wilhelm et al., 2004, 2010; Blanco-Canqui et al., 2006; Blanco-Canqui and Lal, 2007a, b, 2009a, b; Johnson et al., 2007c, 2010a, b; Blanco-Canqui, 2010). Only a small portion of C from plant biomass is converted into stable SOC, none-the-less reducing biomass inputs can reduce SOC (Wilhelm et al., 2004; Johnson et al., 2006). A loss of SOC undermines the agricultural sector's ability to mitigate climate change through C sequestration (IPCC, 2007b).

Carbon stored in SOM pools is not released quickly to the atmosphere as CO_2; therefore, increasing SOM content is a viable GHG mitigation strategy (Paustian et al., 1997). Thus, limiting harvest rates of annual or perennial feedstocks and/or adding compensatory management are needed to maintain SOC.

Soil organic C is explicitly linked to C inputs (e.g. plant biomass); thus a prerequisite for sustainable bioenergy is establishing harvest rates that avoid SOC reduction. Soil organic C content is also a function of decomposition. Reducing biomass inputs or increasing decomposition can shift the equilibrium toward declining SOC contents (Lal, 2009; Johnson et al., 2010b). Based on empirical literature from several regions, crops, and soil types in the U.S., Johnson et al. (2010b) estimated that 2.5 ± 1.7 Mg C ha^{-1} (6.2 ± 4.35 Mg dry biomass ha^{-1}) needs to be returned annually to maintain SOC content. Conservation practices that protect SOC also reduce the risk of soil erosion and avoid related water quality degradation (Johnson et al., 2010a). Soil organic C influences many soil biological, chemical and physical properties and processes that are important to many soil functions (Johnson et al., 2007c, 2010b; Lal et al., 2007a, b; Lal, 2008a, 2009; Cruse et al., 2009). Therefore, sustainable biofuel crop production systems must also be managed to protect the soil resource and its functions to serve as effective media for crop production.

CHALLENGES OF HISTORICAL BIOENERGY FEEDSTOCKS

In the U.S., corn-grain ethanol has been a long-term reliable feedstock for industrial ethanol production (Farrell et al., 2006; Hattori and Morita, 2010). However, there has been extensive and unpleasant societal discourse on producing corn-grain ethanol, primarily due to the economic cost of its production (Pimentel and Patzek, 2005; Pimentel and Pimentel, 2008) and the highly publicized and controversial competition with human food (Hattori and Morita, 2010; Meyer, 2010). Other concerns about corn-grain ethanol have been raised about water quality and its consumption for ethanol industrial production (Committee on Water Implications of Biofuels Production in the United States National Research Council, 2008; Singh and Kumar, 2011), land-use change (CAST, 2007; Fargione et al., 2008; Searchinger et al., 2008) and GHG emission (Borjesson, 2009; Fargione et al., 2010). Much of the controversy on water use has focused on industrial production, even though the amount of water used during conversion is minor compared to the water needed to grow corn (Wilhelm et al., 2010). Many of the mentioned environmental risks from annual cropping systems are related to management (e.g. tillage, fertilizer management) that would still exist if corn were grown for food. Furthermore, it is noteworthy that corn-grain ethanol production only represents a small fraction of the total U.S.'s current and projected liquid fuel consumption (Perlack et al., 2005; Committee on Water Implications of Biofuels Production in the United States National Research Council, 2008; Hoekman, 2009; U.S. EIA, 2011b, a). The Renewable Fuels Standard mandate in the Energy Independence and Security Act of 2007 capped corn-grain ethanol at 56.8 billion L of the desired annual 136.3 billion L of biofuel by 2022 and 227 billion L by 2030 (U.S. Congress, 2007). Recently, this mandate was reduced to 24.6 billion L, which is more aligned with current projected production capacity (Coyle, 2010). Substantial amounts of non-grain biomass will be needed to meet feedstock demand, not just for cellulosic ethanol (BRDB, 2008), but also for renewable heat and power demands.

In the U.S. by 2030, potentially >500 Tg of dry biomass could be harvested annually from agriculture for cellulosic ethanol production (Perlack et al., 2005). Corn stover represents a majority of the identified annual non-grain biomass supply (Nelson, 2002; Perlack et al., 2005). Additional biomass sources are agricultural waste (e.g. hulls, shells), forestry waste products, and dedicated lignocellulosic biofuel crops (Perlack et al., 2005; Graham et al., 2007; Lal and Pimentel, 2007; BRDB, 2008). Dedicated biofuel crops or so-named second-generation feedstocks include sweet sorghum (*Sorghum vulgare* Pers.), HPEC [i.e. switchgrass (*Panicum Virgatum* L.), and giant miscanthus (*Miscanthus giganteus*)] and WPEC [i.e. hybrid poplar (*Populas* spp.), eucalyptus (*Eucalyptus* spp.) and willow (*Salix* spp.)] (Perlack et al., 2005; Graham et al., 2007; BRDB, 2008). As will be discussed, there are limited land and water resources available to produce these feedstocks, due to competition from societal and environmental needs.

Arable land is a limited resource; therefore, efficient land use and management are needed to avoid deleterious impacts on our natural resources or undue competition among demands for agricultural products (food, feed, fiber, and fuel) (Karlen et al., 2009; Wilhelm et al., 2010). By definition, a sustainable bioenergy system is managed to minimize or avoid potential risks (e.g. loss of SOC, water quality, etc.) and optimize benefits such as mitigating GHG emission (CAST, 2007; Johnson et al., 2007c, 2010a; Cruse and Herndl, 2009; Hattori and Morita, 2010; Wilhelm et al., 2010). In the U.S. there is about 374 Mha of total farm land, of which about 30 Mha could be shifted for dedicated biofuel crop production (Dicks et al., 2009). Indeed, availability of this additional land for biofuel production is a key determinant for biofuel sustainability and its growth (Gopalakrishnan et al., 2009). Therefore, current production lands must be efficiently utilized. The most efficient use of land area varies among and within geographic regions (Braun et al., 2010). For example, it has been proposed in the Midwest to utilize double or relay cropping to grow both a food [e.g. soybean (*Glycine max* L.) Merr.] and energy crop [e.g. camelina (*Camelina sativa* L.)] on the same parcel of land in a single growing season (Russ Gesch, personal communication, 2010).

Over the past two centuries, cropland in the U.S. has experienced severe soil erosion (Bennett, 1939; USDA, 1989). While the rate of soil loss has declined in the past few decades, cropland still is losing 4.7 to 7.4 Mg ha^{-1} yr^{-1} from wind or water erosion (USDA-NRCS, 2009), which in many cases exceeds the soil tolerance level "T" for U.S. soils (Hudson, 1982; USDA-NRCS, 1997, 2009). As noted, the overzealous harvest of annual crop residues will likely cause increased soil erosion and sediment and nutrient loading resulting in decreased quality of recipient surface water systems (Pimentel and Kounang, 1998; Blanco-Canqui et al., 2009; Cruse et al., 2009; Cruse and Herndl, 2009). Erosion not only displaces valuable topsoil, but the off-site environmental impacts and losses in crop productivity (CAST, 1982; Cruse et al., 2009) are estimated to cost $17 to 27 billion yr^{-1} (Pimentel et al., 1995).

MANAGEMENT OF ROW CROPS FOR SUSTAINABILITY—AVOIDING/MITIGATING RISKS

Soil cover and residue management have long been recognized as means to prevent or reduce erosion risk (Oschwald et al., 1978). No-tillage management alone is insufficient to minimize run-off and soil erosion, especially if sufficient residue is not available (Blanco-Canqui et al., 2009; Karlen et al., 2009). The risk of increased soil erosion due to residue harvest is widely recognized. Therefore, predictions of harvestable crop residues typically exclude highly erodible lands and are constrained to maintain erosion at or below tolerable soil loss (T) (Nelson, 2002; Nelson et al., 2004; Perlack et al., 2005; Graham et al., 2007; BRDB, 2008). Erosion control is a critical factor when considering how much residue can be harvested from row crops. A limited amount of crop residue might be available for removal without risking erosional soil losses (Blanco-Canqui et al., 2009; Johnson et al., 2010a). However, basing harvest limits solely on avoiding erosion may not provide sufficient biomass input to maintain SOC (Wilhelm et al., 2007, 2010). Therefore, applying conservation management approaches that coincidentally build SOM while avoiding erosion and provide other environmental services may be needed to offset risks associated with crop residue harvest (Johnson et al., 2010a; Wilhelm et al., 2010).

A case study based on Iowa soils and yields (10.2 Mg ha^{-1} corn grain yield) using RUSLE2 and CQESTR simulation models demonstrated that converting conventionally tilled fields to no tillage allowed about 50% of the corn stover to be harvested without increasing erosion or losing SOC (Wilhelm et al., 2010). This case study also modeled using cover crops in conjunction with no tillage; the results suggested that the maximum harvest rate could be sustained while concomitantly protecting soil resource from erosion and SOC loss. Since corn cobs represent about 20% of the stover mass, harvesting grain and cobs instead of total stover is another strategy to harvest both a food and an energy crop from the same parcel of land (Varvel and Wilhelm, 2008; Halvorson and Johnson, 2009; Wilhelm et al., 2011). Cob and grain harvest did not increase erosion or nutrient run-off, since sufficient stover remained on the soil surface (Wienhold and Gilley, 2010). Although the type and capacity of the conversion facility, storage, transportation, and other economic issues are relevant, they are beyond the scope of this review to recommend best row-crop harvest strategies in a given locale.

Strategies to reduce erosion in row crops can also maintain or build SOC. For example, reducing or eliminating tillage can reduce erosion losses while also rebuilding SOC (Reicosky et al., 1995; Hunt et al., 1996; Wander et al., 1998; Deen and Kataki, 2003; Novak et al., 2009a). Unincorporated crop residues decompose slower than when buried by inversion tillage, such as disking or moldboard plowing (Reicosky and Lindstrom, 1993; Lal and Kimble, 1997; Paustian et al., 2000; Burgess et al., 2002). Surface residue placement commonly results in accumulation of SOC at the immediate soil surface (Hunt et al., 1996; Dolan et al., 2006; Novak et al., 2007; Blanco-Canqui and Lal, 2008) with some studies reporting SOC declines at deeper topsoil depths due to reduced residue incorporation (Wander et al., 1998; Deen and Kataki, 2003; Novak et al., 2009a). Other strategies for rebuilding SOC contents include adding mulches, compost, and manures to soils (Tiessen et al., 1994; Gregorich et al., 1996; Johnson et al., 2007b; Viaud et al., 2010). The combination of animal manure and no-till management increased SOC, especially in the soil surface compared to using conventional tillage practices (Bissonnette et al., 2001; Jiao et al., 2006). Unfortunately, studies have also reported that the SOC increases using reduced tillage and mulches are not long-lasting, but must be continually resupplied with fresh residue to overcome decomposition (Parton et al., 1987; Wang et al., 2000). Some have questioned the ability of reduced or no tillage to sequester SOC (Baker et al., 2007) and studies that measured C in horizons below the depth of tillage in row crops have not necessarily shown an increase in SOC through the profile (Venterea et al., 2006; Blanco-Canqui and Lal, 2008). In contrast, others have reported increases in SOC in a 1 m profile comparing no tillage to moldboard plowing (Gal et al., 2007). Furthermore, Kravchenko and Robertson (2011) delineated that differences in SOC may be erroneously missed due to inadequate replication to compensate for the inherent variability in SOC. These authors went on to provide a strong argument for using power analysis and making comparison by horizon to improve sensitivity of statistical analysis. A rigorous assessment of SOC throughout the profile is important for understanding the ability of an agroecosystem to mitigate GHG emissions through soil C sequestration.

MANAGING FOR SUSTAINABLE BIOENERGY WITH PERENNIALS

Row crops are essential for food production, but opportunities exist to diversify the landscape by integrating HPEC and WPEC into the agroecosystem. Compared to annual row crops, HPEC and WPEC have a greater potential to increase SOC, reduce soil erosion and improve off-site water quality. These beneficial outcomes strengthen the rationale for considering them as bioenergy feedstock (Brown et al., 2000; Nelson et al., 2006; Johnson et al., 2007c; Evans and Cohen, 2009). Perennial grasses and trees provide year-round cover, extensive rooting systems and an increased level of raindrop interception, which collectively contributes to reduced erosion and run-off losses (Thompson and Luckman, 1993; Meyer et al., 1995; Kort et al., 1998; Pimentel and Kounang, 1998; Dabney et al., 1999; Self-Davis et al., 2003). Strips of trees and switchgrass planted in various landscape positions (field edges, wetlands, riparian buffers) can effectively reduce 80 to 97% of sediment, N and P loads (Lee et al., 2003). Taking advantage of the environmental benefits with perennial species is consistent with a "sustainable landscape vision" proposed by Karlen et al. (2009) and further described by Wilhelm et al. (2010). This vision integrates economic, environmental, and social aspects of agriculture and selectively allows harvests only from those areas where crop residue exceeds that needed to maintain soil resource condition. Integrating food and fuel production systems, therefore, could mitigate surface and groundwater quality issues through nutrient capture and reduced run-off, and provide wildlife habitat for pollinators (Karlen et al., 2009; Wilhelm et al., 2010). For example, Mitchell et al. (2010) estimated that harvesting switchgrass grown only in the corners of center pivots without irrigation could provide enough feedstock within a 40 km radius to operate a 190 million L cellulosic ethanol plant. Such a strategic approach converts underutilized areas into viable parcels for HPEC production without major land-use shifts, which is consistent with the landscape vision for providing food, feed, fiber, and fuel.

Perennials for erosion control have been utilized by the Conservation Reserve Program (CRP). Conversion of marginal croplands to CRP played a dramatic role in reducing soil erosion from >45 Mg ha^{-1} yr^{-1} to about 2.5 Mg ha^{-1} yr^{-1} (Taylor and Lacewell, 2009). In the U.S. approximately 15 Mha of land are enrolled under the CRP program. Land under CRP protection is considered marginal land that is unfit for prime agricultural use because of poor quality characteristics such as high slopes, shallow soil, or poor physical and chemical characteristics that result in low crop yields (USDA, 1989). Taylor and Lacewell (2009) estimated that if only 20% of CRP lands were brought back into row crop production, annual soil erosion could increase by nearly 190 Tg. In contrast, converting CRP or abandoned agricultural land into large-scale production of WPEC or HPEC (Campbell et al., 2008) would improve nutrient depleted soils (Frank et al., 2004), and reduce GHG emission due to higher amounts of C sequestration (Coleman et al., 2004; Johnson et al., 2007b, c). Perennials have a greater potential to sequester soil C for several reasons, even if a portion is harvested as bioenergy feedstock. Typically, perennials compared to annual crop species have a greater root biomass (Zan et al., 2001; Bolinder et al., 2002). Biochemically, roots tend to be more recalcitrant than shoots (Johnson et al., 2007a). Perennials can also utilize more photosynthetically available days resulting in more atmospheric CO_2 converted into plant C (Baker and Griffis, 2009). Thus, more C enters the soil under perennials compared to annual crops. As an example, switchgrass is capable of adding significant amounts of root C to about 90 cm depth (Ma et al., 2000; Garten et al., 2010). In fact, the top 30 cm of soil below a >20-year-old switchgrass stand had about 15 Mg ha^{-1} root biomass (Al-Kaisi and Grote, 2007). Belowground deposition from a switchgrass root system can add up to 1.2 Mg C ha^{-1} yr^{-1} to the top 30 cm of soil (Liebig et al., 2005a, 2008). An unpublished study on a Norfolk sandy loam in the Coastal Plain region of SC showed dramatic increases in profile SOC content in as few as 2 years of switchgrass growth (Figure 8.2). Likewise, Hansen et al. (2004) attributed the observed change in SOC of 0.71 to 1.03 Mg C ha^{-1} yr^{-1} in the top 100 cm of soil to deep miscanthus roots. Changing from annual crops to perennial species such as miscanthus, switchgrass, and tall fescue (*Festuca arundinacea*) has been reported to increase SOC content by 0.49 to 0.75 Mg C ha^{-1} yr^{-1} (Lal et al., 1998a; Zan et al., 2001; Heaton et al., 2004; King et al., 2004).

Research monitoring changes in SOC contents under WPEC production systems has shown mixed results. Both Grigal and Berguson (1998) and Hansen (1993) reported that SOC content initially declined after establishment of a WPEC, but as the trees matured (10 to 18 years), SOC increased 1 to 1.6 Mg C ha^{-1} yr^{-1}. In contrast, Jug et al. (1999) reported an SOC increase at one of four sites in Germany under willow, poplar, and aspen (*P. tremula* × *P. tremuloides* cv. *Astria*—AS) production. Makeschin (1994) also reported mixed SOC responses under WPEC. The review by Johnson et al. (2007c) discussed additional environmental considerations related to establishment and management of HPEC and WPEC. While crop biomass sources and activities can rebuild SOC levels, researchers are also examining direct and novel approaches to increase SOC levels.

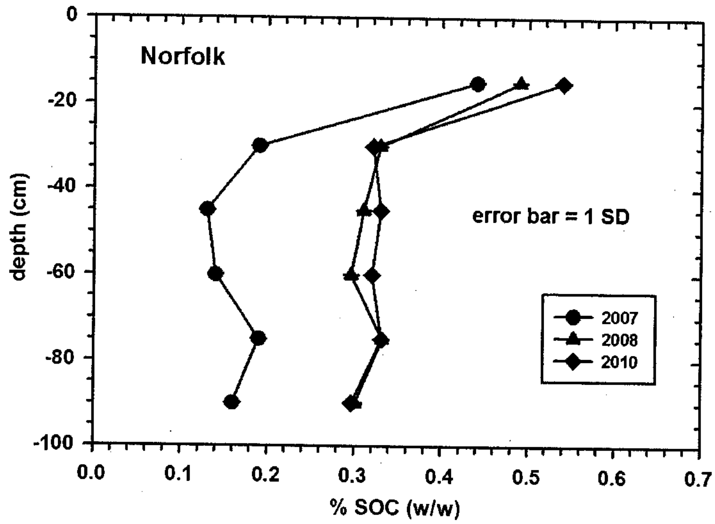

FIGURE 8.2
Soil profile SOC concentration of a Norfolk loamy sand measured before planting (2007) and after 1 (2008) and 2 years (2009) of switchgrass production (Novak and Frederick, unpublished data, 2011). Error bars represent one standard deviation of the mean (SD). In some cases the error bars are obscured by the symbols.

MANAGING FOR SUSTAINABLE BIOENERGY WITH A NOVEL AMENDMENT

Rebuilding SOC levels using reduced tillage, mulches, or by higher residue input and even perennials may take many years to have a measurable impact. Only a fraction of the organic C inputs are converted into stable SOC (Wilhelm et al., 2004; Johnson et al., 2006; Novak et al., 2009a). In contrast to fresh organic C inputs that cycle relatively quickly through the soil, biochars may provide a more stable soil amendment (Lehmann, 2007; Laird, 2008; Busscher et al., 2010). Thus, biochars are a novel approach for rebuilding SOC.

Biochar is a charcoal-like product produced through thermochemical conversion under low oxygen conditions (Antal and Gronli, 2003; Laird, 2008; Brown, 2009; Spokas, 2010). Biochars have a highly recalcitrant structure (Cheng et al., 2008; Kuzyakov et al., 2009), which stems from its highly aromatic composition (Glaser et al., 2002; Novak et al., 2009b) and a low O:C molar ratio (0.25 to 0.6) (Spokas, 2010). Although, pyrogenic C sources like biochar should not be considered humic substances, they do contribute to the organic material within the soil phase defined as SOC (Laird et al., 2008). Because biochars can have a very long residence time in soils (Laird, 2008), they may be used to reduce atmospheric CO_2 concentrations by sequestering C in soils (Sombroek et al., 2003; Lehmann et al., 2006; Liang et al., 2006; Fowles, 2007; Gaunt and Lehmann, 2008; Spokas, 2010; Woolf et al., 2010). A best-case model scenario predicted a maximum sustainable potential for C mitigation from biochar systems at 1.6 Pg yr^{-1}, which is equivalent to 12% of global CO_2 emissions (Woolf et al., 2010). This model assumed that it took about 40 years to reach maximize biomass pyrolysis and biochar production. This abatement strategy also has the potential to improve soil quality (see below), making biochar application a provocative consideration among future candidates of mitigation strategies.

There are reports that in addition to sequestering C, biochars can improve soil fertility (Glaser et al., 2002; Lehmann et al., 2003; Steiner et al., 2007; Novak et al., 2009b), increase soil moisture storage (Glaser et al., 2002; Novak et al., 2009b), and boost crop yields (Day et al., 2005; Steiner et al., 2007; Chan et al., 2008). Biochar properties are related to feedstock and pyrolysis conditions (Chan and Xu, 2009; Sohi et al., 2009), which can influence their quality as a soil amendment (Novak et al., 2009c; Spokas, 2010). For example, several biochars produced from different feedstocks and at different pyrolysis temperatures were laboratory incubated for 127 days in the Norfolk loamy sand, which is an extremely weathered Ultisol

TABLE 8.1 Mean Soil Organic Carbon (SOC) Concentration in a Norfolk Loamy Sand (Ap Horizon) After a 127-day Incubation at Ambient Laboratory Conditions Following the Addition of 2% biochar (w w^{-1}) (Novak et al., Unpublished data)†

		SOC	
		g kg^{-1}	
Feedstock	**Pyrolysis (°C)**	**Mean**	**SD**
Control	–	2.81e‡	0.08
Switchgrass	250	12.91c	0.34
	500	19.64a	0.07
Poultry litter	350	11.07d	0.75
	700	10.28d	0.41
Hardwood	450–600	17.18b	0.63

†Biochar was added at 20 g kg^{-1} into Norfolk Ap; incubated at 10% (w w^{-1}) soil water content, and then every 30 d leached (4× total) with 1.2–1.3 pore volumes of deionized H_2O.
‡Means followed by a different letter differed based on Fisher LSD pair-wise multiple comparison procedures at $P = 0.05$.

from the SC middle Coastal Plain region (Table 8.1). Adding 2% biochars (w w^{-1}) increased SOC compared to the control by as much as six-fold (Table 8.1) (Novak et al., 2009c). Similarly, other laboratory incubation experiments found that amending with biochars increased SOC content (Kimetu and Lehmann, 2010; Laird et al., 2010).

While biochars may increase SOC content, they should be applied discriminately to soil. Biochars produced at high pyrolysis temperatures (500 to 700°C) can be alkaline. Although alkaline biochars may be suitable for use in buffered acidic soils, their application to a poorly buffered loamy sand dramatically raised soil pH levels from 5.9 in the control to as high as 10 (Novak et al., 2009c). In addition, there are few studies that assayed soil microbial and macro-invertebrate communities' response to biochar application. A high temperate poultry-litter biochar impeded earthworm survival and growth (Liesch et al., 2010). Furthermore, biochars are expensive to apply, especially at high application rates (\$300 at 112 Mg ha^{-1}) (Williams and Arnott, 2010). Therefore, it may be more financially prudent to invest in a biochar with definite chemical and physical properties that can target specific soil problems (Steinbeiss et al., 2009; Atkinson et al., 2010; Novak and Busscher, 2011). Applying appropriate biochars to soil would avoid pernicious biological, chemical and physical legacies.

SUMMARY AND RESEARCH NEEDS

Corn grain is the historical ethanol feedstock. However, to meet existing production mandates, next generation biofuel dedicated feedstocks and/or crop residue are under strong consideration. A regionally specific, balanced, and integrated landscape approach is critical for sustainable biofuel production that protects soil resources and crop productivity, and reduces GHG emissions. However, there are many questions that need to be addressed if this lofty goal is to be achieved. For example, harvest rates for crop residue and perennials alike are needed that avoid exacerbating erosion and related risks, such as loss of SOM or direct or indirect acceleration of GHG emission. A broad integrated study funded through USDA and DOE is under way to answer some of these questions and to provide data for simulation and predictive models (Karlen, 2010; Wilhelm et al., 2010). Empirical data on direct impacts of harvesting non-grain biomass (including WPEC and HPEC) on GHG emission is sorely needed. Fortunately, research is under way through projects such as the USDA-ARS-REAP and GRACEnet (Greenhouse gas Reduction through Agricultural Carbon Enhancement net; http://www.ars.usda.gov/research/programs/programs.htm?np_code=212&docid=21223) as well as university projects. Viable and sustainable bioenergy systems will be highly diverse across the country, fitted to the feedstocks available and the size and scope of the conversion platform. On a national scale, large-scale cellulosic ethanol production is of interest; however, there is also interest on a local scale for using feedstocks in an institutional or medium-size industrial facility (http://renewables.morris.umn.edu/biomass/). Research is also needed to assess potential benefits and risks of thermochemical co-products and their use as soil amendments. The recalcitrance of biochar makes land application a strong strategy for long-term C sequestration and reducing atmospheric CO_2 concentrations. However, additional information is needed so biochars can be designed with properties that will safely infuse fertility into depleted soils.

A simple one-size-fits-all sustainable bioenergy system does not exist. However, conservation-based management and forethought will promote biofuel development and utilization, all of which can mitigate GHG emission, provide environmental services beneficial to natural resources, and provide domestic and renewable energy. Efficient land use will be needed to provide sufficient food, feed, fiber, and fuel.

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Managing Agricultural Greenhouse Gases

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Mark A. Liebig

USDA-ARS
Northern Great Plains Research Laboratory
Mandan, ND

Alan J. Franzluebbers

USDA-ARS
Watkinsville, GA

Ronald F. Follett

USDA-ARS
Soil-Plant-Nutrient Research Unit
Fort Collins, CO

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