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SOIL QUALITY AND BIOFUEL PRODUCTION

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INTRODUCTION

The United States is interested in expanding renewable energy resources to address the interrelated problems of finite fossil fuels and global climate change by changing the energy paradigm from one based almost solely on fossil fuels to another that integrates multiple renewable energy platforms (Johnson et al., 2007d). Plant biomass feedstocks will be among the sources of renewable energy. Ethanol from corn (maize; *Zea mays.* L.), grain, and sugarcane (*Saccharum officinarum* L.) and biodiesel fuel from soybeans (*Glycine max* L.) and other oilseed crops are already used for transportation fuels. However, alone they are insufficient to replace petroleum (Perlack et al., 2005). Interest in using non-grain, cellulosic biomass has increased recently (Perlack et al., 2005). Agricultural and forest products represent potential non-grain biomass feedstocks for thermochemical (pyrolysis and gasification) and sugar (fermentation) platforms. Thermochemical technologies can substitute biomass for natural gas or coal and can also be used for producing liquid (pyrolysis oil) and solid (biochar) fuels (Islam and Ani, 2000; Gercel, 2002; Yaman, 2004).

Potential cellulosic biomass feedstocks are numerous and include woody and herbaceous perennial species, lumber industry wastes, forage crops, industrial and municipal wastes, animal manure, crop residues, and agricultural wastes or co-products such as bagasses and cannery wastes (FAO, 2004; Perlack et al., 2005; Johnson et al., 2007d). Corn stover and wheat (*Triticum aestivum* L.) straw are grown in sufficient quantities to support commercial-sized cellulosic ethanol production (Dipardo, 2000; Hettenhaus et al., 2000; Nelson, 2002; Graham et al., 2007). The sources and importance of individual feedstocks vary with location. Regional sources such as sugarcane bagasse and rice (*Oryza sativa* L.) may individually make only local contributions, but collectively they significantly help satisfy United States energy needs (Dipardo, 2000).

Several demands compete for non-grain crop biomass (a term used interchangeably with *crop residue* in this chapter). Small grain straw and corn stover are used for animal bedding and high-fiber feed. Burning corn cobs and other cellulosic materials for heating or cooking was a relatively common practice less than a century ago, and still occurs in some locations. Straw is considered viable as a low-cost building or insulation material (Bainbridge, 1986; Yang et al., 2003). From a soil perspective, keeping non-grain biomass in the field returns essential nutrients for subsequent crops, maintains soil organic matter (SOM), promotes soil aggregate stability, and provides groundcover to reduce erosion (Johnson et al., 2006a).

Most estimates of the amounts of crop residues available for harvest are based on the sole constraint of minimized soil erosion (Lindstrom, 1986; Nelson, 2002; Perlack et al., 2005; Graham et al., 2007). Soil loss tolerance (T) was defined in 1997 as the average annual erosion rate (mass per area per year) that can occur and still permit a high level of crop productivity to be sustained economically and indefinitely by the United States Department of Agriculture (USDA) Natural Resource Conservation Service (NRCS). Nelson (2002) completed a three-year (1995–1997) county-level evaluation of residue removal rates that would provide soil erosion rates less than T. This analysis suggests that an average of 43 million Mg of corn stover and 8 million Mg of wheat could be removed annually for biofuel production (Nelson, 2002). Graham et al. (2007) estimated that sufficient stover was available in central Illinois, northern Iowa, southern Minnesota, and along the Platte River to support large biorefineries. Harvest rates were limited to amounts that maintained erosion rates less than T, and the study assumed all lands included were managed without tillage (Graham et al., 2007). However, Wilhelm et al. (2007) noted that residue requirements for maintaining soil organic carbon (SOC) exceeded those needed to limit erosion at or below T for a similar geographic area. The assessments conducted by Nelson (2002) and Graham et al. (2007) constrained harvest rates only to limit erosion; however, they provide a basis for more detailed analyses including the impact of residue removal on C cycling, future crop productivity, and other important considerations raised by Wilhelm et al. (2004; 2007).

Harvest of non-grain biomass has the potential to directly and indirectly affect many soil physical, chemical, and biological processes. Similar issues are raised concerning soil quality and sustainability for all proposed bioenergy platforms. Understanding the impacts of non-grain biomass harvest on soil processes will aid in developing harvest management systems including utilization of by-products to offset harvest impacts, thus optimizing potential benefits and reducing risks. Harvest guidelines and BMPs are necessary to protect soil from degradation. In this chapter, we discuss soil processes impacted by residue management with emphasis on non-grain biomass harvest. Soil processes reviewed include temperature, moisture and energy balance, C cycling, soil biology, nutrient cycling, soil aggregation, soil erosion, and watershed hydrology.

TEMPERATURE, MOISTURE, AND ENERGY BALANCE

The impacts of residue management on temperature (McCalla, 1943) and moisture (Duley and Russel, 1939) have been researched for more than 60 years, especially in the context of tillage. Surface residues modify the soil microclimate (moisture and temperature) primarily by altering the surface energy balance (Enz et al., 1988; Steiner, 1994; Horton et al., 1996; Wilhelm et al., 2004). Specifically, residue adds a boundary layer between the soil and atmosphere (Enz et al., 1988) that changes the corresponding energy inputs into the soil system (Horton et al., 1996). The net radiation for a bare soil is represented by:

$$R_{net} = S_{sky} - \alpha(S_{sky}) + L_{sky} - L_{soil} \tag{1.1}$$

where R_{net} is the net radiation; S_{sky} is the incident short-wave (solar) radiation; α is the surface albedo (fraction of radiation reflected from the surface); L_{sky} is the incident long-wave sky radiation; and L_{soil} is the emitted long-wave radiation from the soil (Horton et al., 1996; Hillel, 1998). As indicated by Hillel (1998), day and night energy balances exhibit a major difference (Figure 1.1). At night, S_{sky} is negligible and the soil long-wave radiation is typically larger than the long-wave sky radiation, resulting in a negative net radiation flux at night and as a result, net energy movement is from the soil to the atmosphere. For a bare soil, the net radiation is the difference between the energy absorbed and lost by the soil. The net radiation on a bare soil can be apportioned as (1) sensible heat, (2) energy to heat the soil, or (3) energy used to evaporate soil moisture (Figure 1.1). However, the addition of a residue layer provides additional sources and sinks of energy (Ross et al., 1985; Bristow et al., 1986; Chung and Horton, 1987; Enz et al., 1988).

In addition to the processes described for a bare soil, the residue layer can (1) reflect more or less of the incoming radiation, depending on the residue surface albedo (Table 1.1); (2) utilize some of the incoming radiation to heat the residue layer; (3) use energy to evaporate water from the residue; (4) add increased resistance to water vapor fluxes from the soil, thereby reducing soil evaporation flux, and (5) transmit remaining energy to the soil surface (Shen and Tanner, 1990; Horton et al., 1996). Typically residues are lighter in color than soils, thereby increasing the albedo of a residue-covered surface compared to bare soil (Sharratt and Campbell, 1994; Table 1.1). Residues also trap a significant amount of air within the residue layer, thus significantly reducing the effective thermal conductivity of the material layer and reducing the amount of heat transmitted through the residue (Pratt, 1969). Thus, the amount of energy incoming to the soil surface will be less with a residue layer present compared to bare soil. Residues can intercept 50% to 80% of incoming radiation

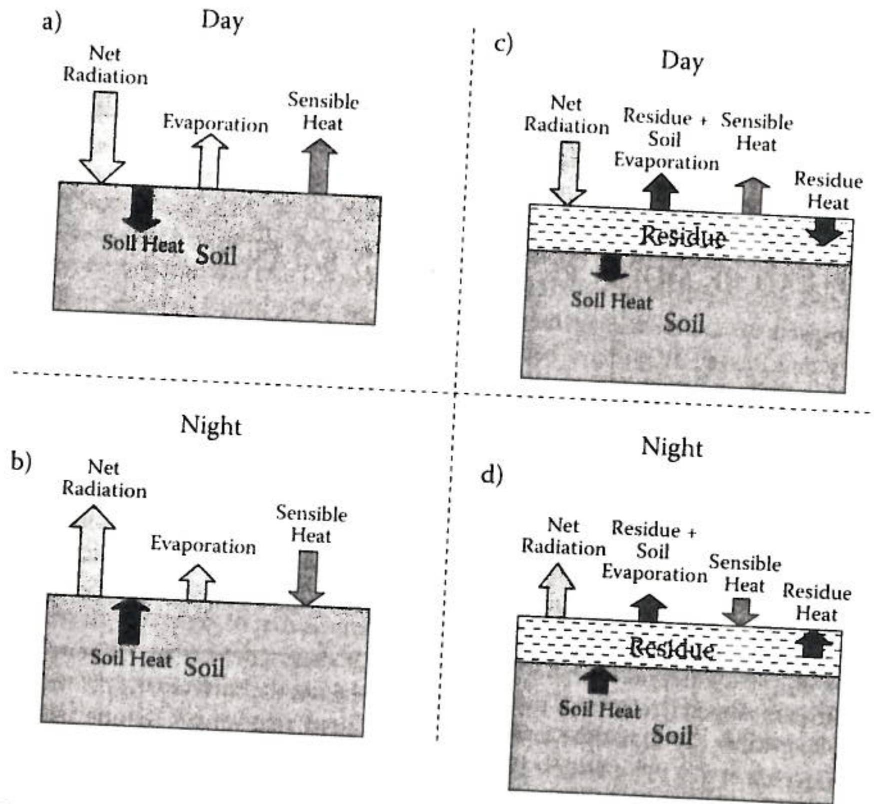

FIGURE 1.1 Energy fluxes for (a) bare soil during daytime, (b) bare soil during nighttime, (c) residue-covered soil during daytime, and (d) residue-covered soil during nighttime conditions. Arrows indicate directions of energy movement. Relative sizes of arrows approximate one potential scenario of dynamics of energy fluxes. Residue cover reduces energy gained (during daytime) and lost (at night) by dry soil surface. (*Sources:* Hillel, D. 1998. *Environmental Soil Physics*. Academic Press, San Diego, CA; Horton, R., K.L. Bristow, G.J. Kluitenberg, and T.J. Sauder. 1996. *Theor. Appl. Clim.* 54:27–37.)

($S_{sk\downarrow}$), keeping the surface soil temperatures within 20°C of ambient, whereas bare soil temperatures may rise 30°C or more above ambient (Ross et al., 1985).

The residue layer also impacts the aerodynamic boundary layer conditions of the soil surface (van Bavel and Hillel, 1976; Hagen, 1996). Residues typically increase surface roughness and correspondingly impact surface exchanges of heat and water and reduce soil loss by wind erosion. Residues increase infiltration and decrease evaporation, generally resulting in a net increase in soil moisture (Smika and Unger, 1986; Blevins and Frye, 1993; Wells et al., 2003; Govaerts et al., 2007a). In regions that experience significant amounts of wind-blown snow, surface residues trap snow, reducing frost penetration depth due to the insulating properties of the snow pack (Benoit et al., 1986). In addition, the snow surface is typically smooth due to low winter evaporation, producing additional soil moisture in the spring (Sauer et al.,

TABLE 1.1
Albedo Comparisons for Several Crop Residues and Materials

Material	Albedo*	Citations
Bare soil	0.04 to 0.40	Lobell and Asner, 2002; Markvart and Castañer, 2003
Green grass	0.25	Markvart and Castañer, 2003
Growing crops	0.10 to 0.40	Stanhill et al., 1966; Al-Yemeni and Grace, 1995
Maize (*Zea Mays* L.) residue	0.31 to 0.46	Tanner and Shen, 1990
Barley (*Hordeum vulgare* L.) straw	0.42 to 0.50	Novak et al., 2000
Sugarcane (*Saccharum officinarum* L.) residue	0.31	Bussière and Cellier, 1994
Wheat (*Triticum aestivum* L.) Straw	0.48 to 0.70	Major et al., 1990
Snow	0.70 to 0.90	Markvart and Castañer, 2003

* Albedo values depend on residue moisture content; typically, the higher the moisture content, the lower the albedo.

1998). Furthermore, trapped snow provides additional soil water recharge during spring thaws (Benoit et al., 1986).

As shown in Figure 1.1, residues impact the surface energy balance by reducing diurnal energy gain and loss at the soil surface. The resulting changes in soil temperature and moisture are functions of the physical properties of the residues and the conditions of the soil (Bristow et al., 1986; Steiner, 1994). In general, most studies agree that with increased residue cover (i.e., decreased crop residue removal), soil moisture content is increased (Russel, 1940); soil temperature maximums decrease, and minimums increase (Blanco-Canqui et al., 2006a). Consequences of these temperature and moisture impacts from residue coverage are less clear. All these factors depend on the interactions of altered soil microclimate conditions with other factors (soil type, climate, and crop type). Crop emergence has been shown to be sensitive to alterations in soil microclimate (Ford and Hicks, 1992; Drury et al., 2003). These resulting effects can be beneficial (Linden et al., 2000; Dam et al., 2005; Blanco-Canqui et al., 2006a), detrimental (Munawar et al., 1990; Liu et al., 2004), or negligible (Bristow, 1988; Swan et al., 1994) for crop emergence, development, and yield. Lower yields observed with high residue covers are hypothesized to result from slower soil warming during seed germination, lower pH, nutrient immobilization, and higher incidence of weeds and pests under residues (Cox et al., 1990; Mann et al., 2002; Drury et al., 2003; Jiang and Thelen, 2004; Liu et al., 2004). Delayed soil warming may delay planting, thereby offsetting gains of soil moisture retention (Nafziger et al., 1991). However, in drought-stressed areas, increased soil moisture can be vital (Power et al., 1986; Power et al., 1998; Jalota et al., 2001; Kato et al., 2007).

Surface residues insulate the soil surface, reducing diurnal temperature fluctuations in a residue-covered soil compared to a bare soil (Buerkert, 2000). In Minnesota, tall (0.6 m) corn stubble reduced frost penetration by 0.5 m and increased the minimum soil temperature by 2°C compared to soil with no residue, leading to a 25-day decrease to begin spring thaw (Sharratt, 2002). However, these beneficial properties of reducing frost depth and increasing soil moisture (Sharratt et al., 1998) resulted in lower spring soil temperatures. This is problematic because these factors may delay spring field operations and significantly impede early germination (Liu et al., 2004). Crop residue impacts on soil microclimate affect both crop emergence and growth and also the timing of the biological production of N_2O (Wagner-Riddle et al., 2008), weed pressure (Garcia-Huidobro et al., 1982; Shafii and Price, 2001; Duppong et al., 2004; Dhima et al., 2006), and C and nutrient cycling (Bayer et al., 2006).

SOIL ORGANIC MATTER (SOM)

Many physical, chemical, and biological characteristics of high quality soils are related to SOM (Doran and Parkin, 1994). Soil biota, nutrient cycling, residue decomposition, humification, and SOM cycling are interrelated. Soils tend to be more productive when organic matter is added regularly and allowed to decompose, thus stimulating nutrient and C cycling and maintaining or enhancing soil structure (Albright, 1938; Kumar and Goh, 2000). Soil organic matter enhances aeration, permeability, water retention, cation exchange, and buffer capacity (Stevenson, 1994; Kumar and Goh, 2000) and reduces soil compactability (Guérif, 1990; Soane, 1990; Diaz-Zorita and Grosso, 2000; Krzic et al., 2004).

Soil compactability is likely to increase if biomass harvest lowers SOM. Within one year, soil bulk density in the surface 6 cm was related inversely to the amount of corn stover returned on silt loam and clay loam soils in Ohio (Blanco-Canqui et al., 2006b). The highest bulk density was 1.45 Mg m^{-3} when all harvestable stover was removed compared to 1.24 Mg m^{-3} with 10 Mg ha^{-1} stover returned. Similarly, cone index and shear strength measurements also decreased with increasing surface stover.

C CYCLING

Plant roots and unharvested above-ground biomass provide the raw materials for building SOM. Photosynthate (organic C) enters the below-ground food web and traverses through multiple trophic levels before returning to the atmosphere with only a small fraction humified into stable SOM. As reviewed by Wilhelm et al. (2004), the amount of plant residue C in soil decreases over time through decomposition; within two years, less than 20% remains in the soil. These authors suggested that the small amount of new C converted to stable SOM implied that a large biomass influx was needed to provide substrate in excess of the respiratory demand of soil fauna. Soil organic matter is about 56% organic C (Stevenson, 1994). Soil organic C (SOC) is frequently used as a proxy to estimate SOM. A simple one-component model of SOC turnover using first-order kinetics (Equation 1.2) is useful where input for more complex models is lacking (Bayer et al., 2006):

$$C_t = C_0 e^{-k_2 t} + \frac{k_1 A}{k_2}(1 - e^{-k_2 t}) \tag{1.2}$$

where C_t is the SOC stock at time t; C_0 is the initial SOC at time $t = 0$; k_2 is the annual rate of SOC loss by mineralization and erosion; k_1 is the annual rate of added C humified into SOM; and A is annual rate of C addition. The first derivative of Equation 1.2 can be expressed as Equation 1.3 (Bayer et al., 2006; Huggins et al., 2007):

$$\frac{dC}{dt} = k_1A - k_2C \tag{1.3}$$

Simply stated, the change in SOC over time is a function of the rate of humification minus the rate of mineralization (inputs minus outputs). At equilibrium, dC/dt goes to zero and $k_1A = k_2C$ and SOC content reaches dynamic equilibrium C_e, as noted by Bayer et al. (2006):

$$C_e = \frac{k_1A}{k_2} \tag{1.4}$$

Conversely, it is possible to use this simple model to solve for the annual rate of C addition (Huggins et al., 2007) at C_e:

$$A = \frac{k_2C_e}{k_1} \tag{1.5}$$

If k_1A exceeds k_2C, then SOC should increase; if not, SOC will decrease. If k_1 and k_2 remain constant, the change in soil C is proportional to inputs for a given management system. Several studies indicate that the amounts of both SOC and C inputs increased linearly (Larson et al., 1972; Paustian et al., 1997; Wilhelm et al., 2004; Follett et al., 2005; Kong et al., 2005; Bayer et al., 2006; Johnson et al., 2006a). However, other results reveal that SOC sequestration did not correlate with the amounts of organic matter inputs (Dexter et al., 1982; Campbell et al., 1991; Johnson and Chamber, 1996; Nicholson et al., 1997), implying that k_2C exceeded k_1A or that the rate coefficients were not constant over the duration of these experiments. The k_1 coefficient is a function of C input quality (Franck et al., 1997; Heal et al., 1997; Wang et al., 2004; Johnson et al., 2007a). The k_2 coefficient is affected by temperature, rainfall, soil texture, mineralogy, and residue management, especially tillage (Bayer et al., 2006). Although first-order kinetics can provide preliminary information, the rates of decomposition and humification slow as more labile materials are decomposed (Wieder and Lang, 1982; Johnson et al., 2007a).

When above-ground biomass is harvested, the quality and quantity of C inputs change because roots and other plant organs may have different chemical compositions (Johnson et al., 2007a). This has the potential to shift the rate of decomposition and subsequent humification (k_1). Carbon originating from root biomass and rhizodeposition contributes 1.5 to 3.0 times more C to stable SOM compared to C originating from above-ground biomass (Balesdent and Balabane, 1996; Allmaras et al., 2004; Wilts et al., 2004; Hooker et al., 2005). The higher values correspond to systems with little or no incorporation of shoot material. Unincorporated residues decompose more slowly (Ghidey and Alberts, 1993) and have fewer opportunities to enter the soil. Roots of corn, alfalfa (*Medicago sativa* L.), and switchgrass (*Panicum virgatum* L.) decompose more slowly than corresponding leaves or stems, but this is not the case for soybean or cuphea (*Cuphea viscisissima* Jacq. and *Cuphea lanceolata* W.T. Aiton) (Johnson et al., 2007a). Although roots contribute more C to SOC, they comprise less plant biomass than above-ground biomass for most annual species (Amos and Walters, 2006; Johnson et al., 2006a).

Using empirical data and linear regression of C inputs and SOC, Johnson et al. (2006a) proposed *minimum source C* (MSC) as a term to describe annual C inputs necessary for dC/dt (Equation 1.3) to equal zero, implying no net change in SOC content. For many agronomic crops, grain is harvested and not returned to the soil, and thus is not included in calculating MSC. Since the Johnson et al. (2006a) review, several other studies allowing MSC estimates revealed similar above-ground MSC estimates (Table 1.2). Using above-ground non-grain C inputs, MSC was 2.5 ± 1.7 Mg C ha^{-1} yr^{-1} (n = 28) for different crops and tillage practices at several experimental sites—slightly higher than the mean MSC of 2.2 ± 1.1 Mg C ha^{-1} yr^{-1} (n = 21) cited by Johnson et al. (2006a).

Moldboard plowed systems had higher MSC requirements than those with no tillage; this result was also reported by Bayer et al. (2006). In general, wheat systems have lower MSC than corn-based systems (Kong et al., 2005; Sainju et al., 2006; Kundu et al., 2007). When rhizodeposition is included, MSC values are larger (Clay et al., 2006; Huggins et al., 2007).

Herbaceous perennial species (e.g., switchgrass) have extensive and deep rooting systems (Ma et al., 2000), and thus may exhibit low above-ground MSC relative to annual species so long as sufficient cover is provided to minimize erosion. Several authors reported increases in SOC under perennial grasses. After six years, SOC under tall fescue (*Festuca arundinacea*) was 3 Mg ha^{-1} greater than under corn in Ohio (Lal et al., 1998). After four years, SOC under switchgrass stands in southwestern Quebec increased by 3 Mg ha^{-1} compared to corn (Zan et al., 2001). In a three-year study, SOC increased at 10 Mg C ha^{-1} yr^{-1} (0 to 0.9 m depth) in central North Dakota under switchgrass harvested annually (Frank et al., 2004). The very low initial soil C content of the North Dakota soil was thought to contribute to the very high SOC accrual rate.

The MSC is a useful guideline for determining the amount of allowable biomass harvest for a management system (Johnson et al., 2006a; Johnson et al., 2006b; Wilhelm et al., 2007). Clearly, given the range of MSC values reported, using an average value is unlikely to provide accurate local harvest rates. Improved understanding of SOM dynamics is critical to developing sustainable biomass harvest guidelines. In the short term, use of process or mechanistic models such as CENTURY (Parton, 1996) or CQSTR (Rickman et al., 2002) may be useful to estimate site- and system-specific biomass harvest rates.

TABLE 1.2
Empirical Estimates of Annual Above-Ground Non-Grain Carbon Inputs Required to Maintain Soil Organic Carbon Levels

Location	Crop*	Primary Tillage**	Soil Type§	MSC (Mg C ha^{-1} yr^{-1})	Citation
MN	M	CP	SiL	2.6	Allmaras et al., 2004
SD	M	CP	L	3.21	Pikul et al., 2008
WI	M	NT	SiL	2.0	Kucharik et al., 2001
MI	M	MBP	SaL	1.6	Vitosh et al., 1997
WI	M	MBP	SiL	2.3	Vanotti et al., 1997
IN	M	MBP	SiL	>4.0	Barber, 1979
IA	M	MBP	CL	2.4	Larson et al., 1972
MN	M	MBP	CL	3.0	Crookston et al., 1991 Huggins et al., 1998
MN	M	MBP	CL, SiCL, SiL	3.3	Reicosky et al., 2002 Wilts et al., 2004
MN	M, S	MBP	CL	3.0	Crookston et al., 1991 Huggins et al., 1998
NE	M, S	D	SiL	2.4	Varvel and Wilhelm, 2008
MN	M, S	NT	CL	8.7	Huggins et al., 2007
SD	M, S	NT or ST	CL, L, SiCL	1.6	Clay et al., 2001; 2006
KS	S, Sr	CP	SiL	1.7	Havlin and Kissel, 1997
KS	S, Sr	NT	SiL	1.2	Havlin and Kissel, 1997
WA	W	NR	—	1.2	Horner et al., 1960 Rasmussen et al., 1980
KS	W	NR	—	2.0	Hobbs and Brown, 1965; Rasmussen, 1980
OR	W	MBP	SiL	2.1	Rasmussen, 1980
WA	W	NR	SiL	2.0	Paustian et al., 1997
WA	W	NR	SiL	4.0	Paustian et al., 1997
MT	W	NT	CL	0.82	Sainju et al., 2006
MT	W	V	SaL	0.3	Black, 1973
CA	W, M, T	CT	SiL, SiCL	2.6	Kong et al., 2005
Sweden	W, Ba	HT	SaCL	1.5	Paustian et al., 1992
Mexico	W, M	MBP, NT	C	1.5	Follett et al., 2005
India	W, S	NR	SaL	0.032	Kundu et al., 2007
Brazil	O, M, V, C	CT	SaCL	6.2	Bayer et al., 2006
Brazil	O, M, V, C	NT	SaCL	2.7	Bayer et al., 2006
Average				2.5 ± 1.7	N = 28

* Crops: Ba = barley (*Hordeum vulgare* L.). C = cowpea (*Vigna unguiculta* (L.) Walp.). M = maize (*Zea Mays* L.). O = oat (*Avena strigoas* Schreb.). S = soybean (*Glycine max* (L.) Merr.). Sr = sorghum (*Sorghum bicolor* L.). T = tomato (*Lycopersicon esculentum* Mill.). V = vetch (*Vicia sativa* L.). W = wheat (*Triticum aestivum* L.).

** Primary tillage: CP = chisel plow. CT = conventional tillage, details not provided. D = disk. HT = hand tillage. MBP = moldboard plow. NR = not reported. RT = ridge till. ST = strip tillage. V = V-blade, 9 to12 cm.

§ Soil type: Si = silt. Sa = sandy. L = loam. C = clay.

SOIL BIOLOGY AND NUTRIENT CYCLING

Plant biomass provides a complex matrix of organic materials that interact with SOM. This complexity can influence the diversity of the soil microbial community and related physiological and enzymatic processes, thus affecting nutrient mineralization and N availability (Bending et al., 2002). In general, harvesting crop residues reduces indicators of soil biology activity. Based on a compendium of worldwide studies, harvesting residues reduced the concentrations of microbial biomass C by 25% and microbial biomass N by 29% (Tables 1.3 and 1.4). In some cases, the impact of residue harvest was measured as early as two years after biomass harvest. Of the 25 observations, only three cases indicated that residue removal had no effect or exerted a positive impact on microbial biomass C concentration relative to treatments that retained residue (Table 1.3). Microbial biomass C increased proportionally to the amount of biomass returned when harvest rates were varied (Karlen et al., 1994; Cookson et al., 1998; Debosz et al., 1999; Salinas-Garcia et al., 2001; Limon-Ortega et al., 2006).

Earthworms are macroscopic indicators of a healthy soil and provide beneficial functions related to nutrient cycling, soil structure, hydrology, and root growth. A reduction in earthworm activity caused a decrease in saturated hydraulic conductivity to a depth of 20 cm (Blanco-Canqui et al., 2007). Numerous studies have noted that reducing or eliminating tillage increases earthworm biomass (Nuutinen, 1992) and abundance (Edwards et al., 1990; Nuutinen, 1992; Kladivko, 2001). Elimination of burning crop residue also increased earthworm abundance (Fraser et al., 1996; Wuest et al., 2005). Therefore, it was expected that retaining non-grain biomass would reveal greater earthworm populations than areas from which the biomass was removed (Table 1.5). Similar to microbial biomass C, earthworm abundance increased with the amount of biomass returned when harvest rates varied such that at least 25% of corn stover decreased midden numbers.

Crop residue management may influence plant diseases. Retention of residues can result in net changes in soil microbiota by retaining inoculum or creating an environment more conducive to pathogens (Cook et al., 1978). For example, corn stover and small grain straws are the principal inoculum sources of *Fusarium* spp. that cause head blight in wheat, especially in no-till systems (Maiorano et al., 2008). Population counts of *Fusarium* were highest in continuous corn with residue retention and lowest under continuous wheat or corn–wheat rotation, also with residue retained (Govaerts et al., 2008). The same study indicated that residue retention increased populations of disease-suppressing microorganisms including fluorescent *Pseudomonas* that provide biological control of *Fusarium* and other fungal pathogens.

Greater incidence of root rot in corn was associated with stover retention relative to stover harvest without tillage; however, root rot did not reduce yield (Govaerts et al., 2007a). Govaerts et al. (2008) proposed that no tillage and residue retention have potential for biological control by promoting plant growth and suppressing disease, but eliminating tillage alone did not improve soil health. Disease response to repeated burning of wheat stubble was variable, depending on precipitation and N management (Smiley et al., 1996). Crown rot incidents were positively correlated with SOC, while root rot incidents were negatively correlated to microbial biomass

TABLE 1.3

Microbial Biomass C Concentration ($\mu g\ g^{-1}$) or Content ($kg\ ha^{-1}$) Response to Non-Grain Biomass Harvest Treatment

Location	Soil*	Crop**	Tillage§	Years	Depth (cm)	Biomass Treatment: Removed	Biomass Treatment: Retained	Comment	Citation¶
							μg Microbial biomass C g^{-1}		
Lancaster, WI	SiL	M, M	NT	10	0 to 8	330	696	Normal retained	b
Lancaster, WI	SiL	M, M	NT	10	0 to 8	330	1060	Double biomass	b
Ste-Anne-de-Bellevue, Que, Canada	Sa-LSa	M, M	MBP, RT, NT	9	0 to 10	124	200	Average across tillage	h
Ste-Anne-de-Bellevue, Que, Canada	S-LSa	M, M	MBP, RT, NT	9	10 to 20	128	158	Average across tillage	h
El Batán, Mexico	L	M, M	NT	14	0 to 15	122	453		j
El Batán, Mexico	L	M, M	CT	14	0 to 15	291	322		j
El Batán, Mexico	L	W, W	NT	14	0 to 15	329	374		j
El Batán, Mexico	L	W, W	CT	14	0 to 15	288	544		j
Central Mexico	SaL	M, M	NT	5	0 to 15	303	654	7 Mg ha^{-1} retained	g
Central Mexico	SaL	M, M	NT	5	0 to 15	303	426	5 Mg ha^{-1} retained	g
Central Mexico	SaL	M, M	NT	5	0 to 15	303	345	3 Mg ha^{-1} retained	g
Central Mexico	SaL	M, M	NT	5	0 to 15	303	495	3 Mg ha^{-1} retained + vicia cover	g
Central Mexico	SaL	M, M	NT	5	0 to 15	303	488	3 Mg ha^{-1} retained + *P. vulgaris* cover	g
Central Mexico	SaL	M, M	CT	5	0 to 15	NR•	264	All biomass retained	g
Yaqui Valley, Mexico	SaL	W, M	CT	3	0 to 7	NR	425	All retained	f
Yaqui Valley, Mexico	SaL	W, M	CT	3	7 to 15	NR	292	All retained	f

TABLE 1.3 (CONTINUED)
Microbial Biomass C Concentration (μg g⁻¹) or Content (kg ha⁻¹) Response to Non-Grain Biomass Harvest Treatment

Location	Soil*	Crop**	Tillage§	Years	Depth (cm)	Biomass Treatment Removed	Biomass Treatment Retained	Comment	Citation¶
						μg Microbial biomass C g⁻¹			
Yaqui Valley, Mexico	SaL	W, M	NT	3	0 to 7	321	350	Only stover harvested, wheat straw retained	f
Yaqui Valley, Mexico	SaL	W, M	NT	3	7 to 15	218	184	Only stover harvested, wheat straw retained	f
Yaqui Valley, Mexico	SaL	W, M	NT	3	0 to 7	321	348	Stover and straw harvested	f
Yaqui Valley, Mexico	SaL	W, M	NT	3	7 to 15	218	210	Stover and straw harvested	f
Yaqui Valley, Mexico	SaL	W, M	CT	10	0 to 7	NR	461	All retained	f
Yaqui Valley, Mexico	SaL	W, M	NT	10	0 to 7	584	642	Only wheat straw retained	f
Yaqui Valley, Mexico	SaL	W, M	NT	10	0 to 7	584	533	Both retained	f
Yaqui Valley, Mexico	SaL	W, M	CT	12	0 to 7	NR	596	All retained	f
Yaqui Valley, Mexico	SaL	W, M	CT	12	0 to 7	617	681	Only stover harvested, wheat straw retained	f
Yaqui Valley, Mexico	SaL	W, M	NT	12	0 to 7	617	687	Stover and straw harvested	f
Nairobi, Kenya	Humic nitisol	C, B	CU	18	0 to 15	54	72		c
Varanasi, India	SaL	R, Ba	D*2, CU	2	0 to 10	235	347	Cultivated to 20 cm	d
Varanasi, India	SaL	R, Ba	D, CU	2	0 to 10	271	427	Cultivated to 10 cm	d
Varanasi, India	SaL	R, Ba	NT	2	0 to 10	283	320	Averaged across crop periods	d
						kg Microbial Biomass C ha⁻¹			
Apatzingan, Mexico	C	M, M	NT	6	0 to 5	830	970	100% stover retained	e

Apatzingan, Mexico	C	M, M	NT	6	5 to 10	705	760	100% stover retained	e
Apatzingan, Mexico	C	M, M	NT	6	10 to 20	640	705	100% stover retained	e
Apatzingan, Mexico	C	M, M	NT	6	0 to 5	830	830	66% stover retained	e
Apatzingan, Mexico	C	M, M	NT	6	5 to 10	705	760	66% stover retained	e
Apatzingan, Mexico	C	M, M	NT	6	10 to 20	640	708	66% stover retained	e
Apatzingan, Mexico	C	M, M	NT	6	0 to 5	830	805	33% stover retained	e
Apatzingan, Mexico	C	M, M	NT	6	5 to 10	705	750	33% stover retained	e
Apatzingan, Mexico	C	M, M	NT	6	10 to 20	640	680	33% stover retained	e
Apatzingan, Mexico	C	M, M	MBP	6	0 to 5	NR	710	100% stover retained	e
Apatzingan, Mexico	C	M, M	MBP	6	5 to 10	NR	815	100% stover retained	e
Apatzingan, Mexico	C	M, M	MBP	6	10 to 20	NR	640	100% stover retained	e
Apatzingan, Mexico	C	M, M	D	6	0 to 5	NR	810	100% stover retained	e
Apatzingan, Mexico	C	M, M	D	6	5 to 10	NR	760	100% stover retained	e
Apatzingan, Mexico	C	M, M	D	6	10 to 20	NR	645	100% stover retained	e
Casas Blancas, Mexico	SiC	M, M	NT	6	0 to 5	350	710	100% stover retained	e
Casas Blancas, Mexico	SiC	M, M	NT	6	5 to 10	310	500	100% stover retained	e
Casas Blancas, Mexico	SiC	M, M	NT	6	10 to 20	300	360	100% stover retained	e
Casas Blancas, Mexico	SiC	M, M	NT	6	0 to 5	350	650	66% stover retained	e
Casas Blancas, Mexico	SiC	M, M	NT	6	5 to 10	310	490	66% stover retained	e
Casas Blancas, Mexico	SiC	M, M	NT	6	10 to 20	300	300	66% stover retained	e
Casas Blancas, Mexico	SiC	M, M	NT	6	0 to 5	350	500	33% stover retained	e
Casas Blancas, Mexico	SiC	M, M	NT	6	5 to 10	310	480	33% stover retained	e
Casas Blancas, Mexico	SiC	M, M	NT	6	10 to 20	300	280	33% stover retained	e
Casas Blancas, Mexico	SiC	M, M	MBP	6	0 to 5	NR	300	100% stover retained	e
Casas Blancas, Mexico	SiC	M, M	MBP	6	5 to 10	NR	280	100% stover retained	e
Casas Blancas, Mexico	SiC	M, M	MBP	6	10 to 20	NR	240	100% stover retained	e
Casas Blancas, Mexico	SiC	M, M	D	6	0 to 5	NR	475	100% stover retained	e
Casas Blancas, Mexico	SiC	M, M	D	6	5 to 10	NR	440	100% stover retained	e

TABLE 1.3 (CONTINUED)
Microbial Biomass C Concentration (μg g^{-1}) or Content (kg ha^{-1}) Response to Non-Grain Biomass Harvest Treatment

Location	Soil*	Crop**	Tillage$	Years	Depth (cm)	Biomass Treatment Removed	Biomass Treatment Retained	Comment	Citation¶
							kg Microbial Biomass C ha^{-1}		
Casas Blancas, Mexico	SiC	M, M	D	6	10 to 20	NR	350	100% stover retained	e
Biloela, Australia	SaC	Sr	D, CU	6	12	273	315		a
Biloela, Australia	SaC	Sr	NT	6	10	313	347		a

* Soils: Si = silt, silty, C = clay, L = loam, Sa = sand, sandy, V = vertisol, H = haplustol.

** Crops: Ba = barley (*Hordeum vulgare* L.), B = bean (*Phaseolus*), G = grass, M = maize (*Zea Mays* L.), R = rice (*Oryza sativa* L.), Sr = sorghum (*Sorghum bicolor* L.), W = wheat (*Triticum aestivum* L.).

$ Primary tillage: CT = conventional tillage, implement not designated, CU = cultivated, D = disk, MBP = moldboard plow, NT = no tillage, RT = ridge tillage.

¶ Citations: a = Saffigna et al., 1989; b = Karlen et al., 1994; c = Kapkiyai et al., 1999; d = Kushwaha et al., 2000; e = Salinas-Garcia et al., 2001; f = Limon-Ortega et al., 2002; g = Roldan et al., 2003; h = Spedding et al., 2004; i = Limon-Ortega et al., 2006; j = Govaerts et al., 2007b.

NR = not reported or treatment not measured.

TABLE 1.4
Microbial Biomass N Concentration (μg g⁻¹) or Content (kg ha⁻¹) Response to Non-Grain Biomass Harvest

Location	Soil*	Crop**	Tillage§	Years	Depth (cm)	Biomass Treatment: Removed	Biomass Treatment: Retained	Comment	Citation¶
						μg Microbial Biomass N g⁻¹			
Canterbury, New Zealand	NR	W-W-Ba-Ba	CU	4	0 to 5	38	47		b
Canterbury, New Zealand	NR	W-W-Ba-Ba	CU	4	5 to 10	34	49		b
Canterbury, New Zealand	NR	W-W-Ba-Ba	CU	4	10 to 25	18	38		b
Ste-Anne-de-Bellevue, Que, Canada	Sa-LSa	M-M	MBP, RT, NT	9	0 to 10	13	26		f
Ste-Anne-de-Bellevue, Que, Canada	Sa-LSa	M-M	MBP, RT, NT	9	10 to 20	19	19		f
El Batán, Mexico	L	M-M	NT	14	0 to 15	17	39		g
El Batán, Mexico	L	M-M	CT	14	0 to 15	12	24		g
El Batán, Mexico	L	W-W	NT	14	0 to 15	21	29		g
El Batán, Mexico	L	W-W	CT	14	0 to 15	25	27		g
Nairobi, Kenya	Humic nitisol	M-B	CU	18	0 to 15	9.4	11.2		c
Varanasi, India	SaL	R-Ba	D*2, CU	2	0 to 10	24	38	Cultivated to 20 cm	d
Varanasi, India	SaL	R-Ba	D, CU	2	0 to 10	28	49	Cultivated to 10 cm	d
Varanasi, India	SaL	R-Ba	NT	2	0 to 10	27	31		d
Biloela, Australia	SaC	Sr	D, CU	5	12	39	48		a
Biloela, Australia	SaC	Sr	NT	5	10	37	46		a
						kg Microbial Biomass N ha⁻¹			
Apatzingan, Mexico	C	M-M	NT	6	0 to 5	33	53	100% stover retained	e
Apatzingan, Mexico	C	M-M	NT	6	5 to 10	34	43	100% stover retained	e

TABLE 1.4 (CONTINUED)

Microbial Biomass N Concentration (μg g^{-1}) or Content (kg ha^{-1}) Response to Non-Grain Biomass Harvest

Location	Soil*	Crop**	Tillage§	Years	Depth (cm)	Biomass Treatment		Comment	Citation¶
						Removed	Retained		
						kg Microbial Biomass N ha^{-1}			
Apatzingan, Mexico	C	M-M	NT	6	10 to 20	28	34	100% stover retained	e
Apatzingan, Mexico	C	M-M	NT	6	0 to 5	33	44	66% stover retained	e
Apatzingan, Mexico	C	M-M	NT	6	5 to 10	34	40	66% stover retained	e
Apatzingan, Mexico	C	M-M	NT	6	10 to 20	28	33	66% stover retained	e
Apatzingan, Mexico	C	M-M	NT	6	0 to 5	33	39	33% stover retained	e
Apatzingan, Mexico	C	M-M	NT	6	5 to 10	34	37	33% stover retained	e
Apatzingan, Mexico	C	M-M	NT	6	10 to 20	28	32	33% stover retained	e
Apatzingan, Mexico	C	M-M	MBP	6	0 to 5	NR*	24	100% stover retained	e
Apatzingan, Mexico	C	M-M	MBP	6	5 to 10	NR	25	100% stover retained	e
Apatzingan, Mexico	C	M-M	MBP	6	10 to 20	NR	24	100% stover retained	e
Apatzingan, Mexico	C	M-M	D	6	0 to 5	NR	46	100% stover retained	e
Apatzingan, Mexico	C	M-M	D	6	5 to 10	NR	35	100% stover retained	e
Apatzingan, Mexico	C	M-M	D	6	10 to 20	NR	29	100% stover retained	e
Casas Blancas, Mexico	SiC	M-M	NT	6	0 to 5	41	72	100% stover retained	e
Casas Blancas, Mexico	SiC	M-M	NT	6	5 to 10	38	42	100% stover retained	e
Casas Blancas, Mexico	SiC	M-M	NT	6	10 to 20	34	35	100% stover retained	e
Casas Blancas, Mexico	SiC	M-M	NT	6	0 to 5	41	60	66% stover retained	e
Casas Blancas, Mexico	SiC	M-M	NT	6	5 to 10	38	41	66% stover retained	e
Casas Blancas, Mexico	SiC	M-M	NT	6	10 to 20	34	35	66% stover retained	e
Casas Blancas, Mexico	SiC	M-M	NT	6	0 to 5	41	51	33% stover retained	e
Casas Blancas, Mexico	SiC	M-M	NT	6	5 to 10	38	39	33% stover retained	e

Casas Blancas, Mexico	SiC	M-M	NT	6	10 to 20	34	35	33% stover retained	e
Casas Blancas, Mexico	SiC	M-M	MBP	6	0 to 5	NR	40	100% stover retained	e
Casas Blancas, Mexico	SiC	M-M	MBP	6	5 to 10	NR	32	100% stover retained	e
Casas Blancas, Mexico	SiC	M-M	MBP	6	10 to 20	NR	33	100% stover retained	e
Casas Blancas, Mexico	SiC	M-M	D	6	0 to 5	NR	50	100% stover retained	e
Casas Blancas, Mexico	SiC	M-M	D	6	5 to 10	NR	33	100% stover retained	e
Casas Blancas, Mexico	SiC	M-M	D	6	10 to 20	NR	31	100% stover retained	e

* Soils: Si = silt, silty. C = clay. L = loam. Sa = sand, sandy. V = vertisol. H = haplustol.

** Crops: Ba = barley (*Hordeum vulgare* L.). B = bean (*Phaseolus*). G = grass. M = maize (*Zea mays* L.). R = rice (*Oryza sativa* L.). Sr = sorghum (*Sorghum bicolor* L.). W = wheat (*Triticum aestivum* L.).

§ Tillage: CT = conventional tillage, implement not designated. CU = cultivated. D = disk. MBP = moldboard plow. NT = no tillage. RT = ridge tillage.

¶ Citations: a = Saffigna et al., 1989; b = Fraser and Piercy, 1998; c = Kapkiyai et al., 1999; d = Kushwaha et al., 2000; e = Salinas-Garcia et al., 2001; f = Spedding et al., 2004; g = Govaerts et al., 2007b.

NR = not reported or treatment not measured.

TABLE 1.5
Earthworm Response to Non-Grain Biomass Harvest Treatment

Location	Soil*	Crop**	Tillage§	Years	Depth (cm)	Earthworm Abundance: Biomass Removed	Earthworm Abundance: Biomass Retained	Comment	Citation¶
						(Number m⁻²)			
Jokioinen, Finland	SiCL	SW	MBP, CU	8/9	NA	14	17	Average tillage and years	a
Mouhijarvi, Finland	SaCL	SW	MBP, CU	8/9	NA	18	19	Average across tillage	a
Palkane, Finland	L	SW	MBP, CU	8/9	NA	16	20	Average across tillage	a
Lancaster, WI	SiL	M-M	NT	10	NA	53	78		b
Shanxi, China	SiL	WiW	MBP	6	30	0	NR		e
Shanxi, China	SiL	WiW	NT	6	30	NR	5		e
Shanxi, China	SiL	WiW	MBP	14	30	0	NR		e
Shanxi, China	SiL	WiW	NT	14	30	NR	19		e
Canterbury, New Zealand	NR	W-W-Ba-Ba	M	4	25	NR	400		c
Canterbury, New Zealand	NR	W-W-Ba-Ba	CU	4	25	125	NR		c
						Number middens m⁻²			
Coshocton, OH	SiL	M	NT	1	Surface	3	19	10 Mg ha⁻¹ stover retained	d
Coshocton, OH	SiL	M	NT	1	Surface	3	10	5 Mg ha⁻¹ stover retained	d
Coshocton, OH	SiL	M	NT	1	Surface	3	5	3.75 Mg ha⁻¹ stover retained	d
Coshocton, OH	SiL	M	NT	1	Surface	3	5	2.5 Mg ha⁻¹ stover retained	d
Coshocton, OH	SiL	M	NT	1	Surface	3	5	1.25 Mg ha⁻¹ stover retained	d
South Charleston, OH	SiL	M	NT	1	Surface	1	17	10 Mg ha⁻¹ stover retained	d
South Charleston, OH	SiL	M	NT	1	Surface	1	17	5 Mg ha⁻¹ stover retained	d
South Charleston, OH	SiL	M	NT	1	Surface	1	12	3.75 Mg ha⁻¹ stover retained	d

South Charleston, OH	SiL	M	NT	1	Surface	1	4	2.5 Mg ha^{-1} stover retained	d
South Charleston, OH	SiL	M	NT	1	Surface	1	3	1.25 Mg ha^{-1} stover retained	d
Hoytville, OH	CL	M	NT	1	Surface	2	24	10 Mg ha^{-1} stover retained	d
Hoytville, OH	CL	M	NT	1	Surface	2	18	5 Mg ha^{-1} stover retained	d
Hoytville, OH	CL	M	NT	1	Surface	2	15	3.75 Mg ha^{-1} stover retained	d
Hoytville, OH	CL	M	NT	1	Surface	2	13	2.5 Mg ha^{-1} stover retained	d
Hoytville, OH	CL	M	NT	1	Surface	2	10	1.25 Mg ha^{-1} stover retained	d

* Soils: Si = silt, silty. C = clay. L = loam. Sa = sand, sandy. V = vertisol. H = haplustol.

** Crops: Ba = barley (*Hordeum vulgare* L.). B = bean (*Phaseolus*). G = grass. M = maize (*Zea mays* L.). R = rice (*Oryza sativa* L.). S = sorghum (*Sorghum bicolor* L.). W = wheat (*Triticum aestivum* L.).

§ Tillage: CT = conventional tillage, implement not designated. CU = cultivated. D = disk. MBP = moldboard plow. NT = no tillage. RT = ridge tillage.

¶ Citations: a = Nuutinen, 1992; b = Karlen et al., 1994; c = Fraser and Piercy, 1998; d = Blanco-Canqui et al., 2007; e = Li et al., 2007.

NA = not applicable, soil treated to draw worms to soil surface.

NR = not reported or treatment not measured.

(Smiley et al., 1996). Effects of residue harvest on microbial species are difficul to predict; negative, neutral, and positive responses have been reported along witl interactions of tillage, climate, and nutrient management.

Harvesting crop non-grain biomass affects nutrient cycling by removing plant macronutrients (N, P, K, Ca, and Mg) (Mubarak et al., 2002) and micronutrients (Fageria, 2004). The concentration of nutrient in non-grain biomass averaged 9.0 ± 5.1 g N kg^{-1}, 1.1 ± 0.52 g P kg^{-1}, and 5.0 ± 1.3 g K kg^{-1} based on results from several common annual crops and likely perennial biomass crops (Table 1.6). The amounts of nutrients removed vary among plant species, organs harvested (cob versus entire stover), physiological stage, and amount of biomass harvested (Lindstrom, 1986. Burgess et al., 2002; Mubarak et al., 2002; Fageria, 2004; Johnson et al., 2007a).

The amount of nutrient removed can be calculated from the concentration and biomass harvest rates. Bransby et al. (1998) indicated that harvest of above-ground switchgrass biomass has the potential to remove 126 to 281 kg N ha^{-1} or more under fertilized conditions and 38 kg N ha^{-1} under unfertilized conditions. From a nutrient management view, harvesting biomass after senescence removes the least amount of mineral nutrient. In Washington state, the amount of N removed by harvesting switchgrass varied by cultivar and harvest date more than by the amount of N fertilizer applied; early harvesting prior to N translocation below ground removed more N than harvesting in October (personal communication, Hal Collins, USDA ARS, Prosser, WA). Continued removal of nutrient without replacement by applying fertilizer, manure, or compost depletes soil fertility, in turn reducing soil productivity.

Harvesting non-crop biomass affects soil microbial processes that impact N availability. For example, the activity of N-acetyl-b-D-glucosaminidase was reduced by harvesting corn stover for 10 years in a continuous corn system, suggesting a reduction in N mineralization (Ekenler and Tabatabai, 2003). In Kenya, corn stover harvest for 18 years in a corn–bean (*Phaseolus vulgaris* L.) rotation reduced N stocks (0 to 15 cm depth) that corresponded to declines in total N, particulate matter N, mineral N, microbial biomass N, and potentially mineralizable N (Kapkiyai et al., 1999). Corn and soybeans took up more N where stover was retained than where stover was removed, possibly because stover maintained a soil environment more conducive to biological activity that increased N availability (Power et al., 1986). In India, more N was available (0 to 30 cm depth) with residue retained and incorporated compared to residue removal in wheat–groundnut (*Archis hypogea* L.) rotation (Bhatnagar et al., 1983). These studies indicate increased plant-available N with stover retention and suggest that harvesting non-grain biomass may impact soil fertility adversely.

SOIL AGGREGATION

In agricultural systems, maintenance of SOM has long been recognized as a strategy to improve soil structure and reduce soil degradation. Soil structure is an important property that mediates many physical and biological processes and controls SOM and residue decomposition (Van Veen and Kuikman, 1990). Soil aggregates are the basic units of soil structure and consist of primary particles and binding agents (Figure 1.2; Edwards and Bremner, 1967; Tisdall and Oades, 1982; Tisdall, 1996; Jastrow and Miller, 1997). Water stability of soil aggregates depends on organic

TABLE 1.6

Plant Concentration ± Standard Deviations Based on Literature Reports of N, P, and K in Non-Grain Above-Ground Portions of Potential Non-Grain Biomass Feedstocks

Crop	N (g/kg)	P (g/kg)	K (g/kg)	Citations*
	Annuals			
Barley (*Hordeum vulgare* L.)	6.5 ± 1.1	1.1 ± NA†	12.5 ± NA	a
N	6	1	1	
Maize (*Zea mays* L.)	7.5 ± 3.0	1.3 ± 0.5	11.8 ± 6.2	b
N	16	6	5	
Millet (*Panicum miliaceaum* L.)	8.9 ± 5.6	0.85 ± 0.21	12.8 ± NA	c
N	3	2	1	
Rice (*Oryza sativa* L.)	9.3 ± 7.2	0.71 ± 0.24	19.3 ± 14.1	d
N	9	4	3	
Sorghum (*Sorghum bicolor* L.)	12.0 ± 9.4	0.5 ± NA	NR‡	e
N	4	1		
Soybean (*Glycine max* (L.)	17.6 ± 12.1	1.85 ± 0.5	12.8 ± 3.3	f
N	11	2	2	
Wheat (*Triticum aestivum* L.).	6.8 ± 2.2	1.1 ± 0.6	7.8 ± 2.7	g
N	10	2	2	
	Perennials			
Miscanthus (*Miscanthus × giganteus*)	8.0 ± 5.3	0.25 ± 0.21	3.45 ± 0.21	h
N	6	2	2	
Switchgrass (*Panicum virgatum* L.)	6.8 ± 4.2	0.62 ± 0.23	3.2 ± 3.3	i
N	17	7	6	
Other grass¶	9.0 ± 5.1	1.08 ± 0.52	5.0 ± 1.3	j
N	5	5	5	
Annuals	9.8 ± 7.5	1.1 ± 0.5	13.1 ± 7.6	
N	60	18	14	
Perennials	7.5 ± 4.5	0.73 ± 0.5	3.9 ± 2.4	
N	28	14	13	

*Citations:

a = Christensen, 1986; Lindstrom, 1986; Andren and Paustian, 1987; Cookson et al., 1998; Mitchell et al., 2001; Velthof et al., 2002; Halvorson and Reule, 2007.

b = Lindstrom, 1986; Breakwell and Turco, 1989; Tian et al., 1992; Burgess et al., 2002; Manlay et al., 2002; Velthof et al., 2002; Fageria, 2004; Al-Kaisi et al., 2005; Hoskinson et al., 2007; Yu et al., 2008; Halvorson and Johnson, in press.

c = Manlay et al., 2002; Fatondji et al., 2006; Sarr et al., 2008.

d = Tian et al., 1992; Ying et al., 1998; Manlay et al., 2002; Abiven et al., 2005; Tirol-Padre et al., 2005; Linquist et al., 2007; Kaewpradit et al., 2008.

e = Saffigna et al., 1989; Franzluebbers et al., 1995; Abiven et al., 2005; Monti et al., 2008.

f = Lindstrom, 1986; Franzluebbers et al., 1995; Fageria, 2004; Abiven et al., 2005; Al-Kaisi et al., 2005; Rao et al., 2005; Johnson et al., 2007a.

TABLE 1.6 (CONTINUED)
Plant Concentration ± Standard Deviations Based on Literature Reports of N, P, and K in Non-Grain Above-Ground Portions of Potential Non-Grain Biomass Feedstocks

'Citations (continued):

g = Jawson and Elliott, 1986; Lindstrom, 1986; Franzluebbers et al., 1995; Cookson et al., 1998; Mitchell et al., 2001; Borie et al., 2002; Velthof et al., 2002; Abiven et al., 2005; Tirol-Padre et al., 2005.

h = Clifton-Brown and Lewandowski, 2002; Monti et al., 2008.

i = Bransby et al., 1998; Madakadze et al., 1999; Reynolds et al., 2000; Duffy and Nanhou, 2001; Lemus et al., 2002; Vogel et al., 2002; Cassida et al., 2005; Adler et al., 2006; Lemus et al., 2008; Monti et al., 2008.

j = Katterer et al., 1998; Monti et al., 2008.

NA = not appropriate.

NR = not reported.

¶ Other grasses include cardoon (*Cynara Cardunculus* L.), giant reed (*Arundo donax* L.), and reed canary grass (*Phalaris arundinacea* L.).

materials such as polysaccharides, roots, fungal hyphae, and aromatic compounds (Tisdall and Oades, 1982).

SOM is considered a major bonding agent responsible for the formation and stabilization of soil aggregates (Tisdall and Oades, 1982; Dormaar, 1983; Chaney and Swift, 1984; Miller and Jastrow, 1990; Haynes et al., 1991; Degens, 1997; Angers, 1998). In addition, improvement of soil aggregate stability results from the microbial utilization of carbohydrates and from plant phenolics released during decomposition

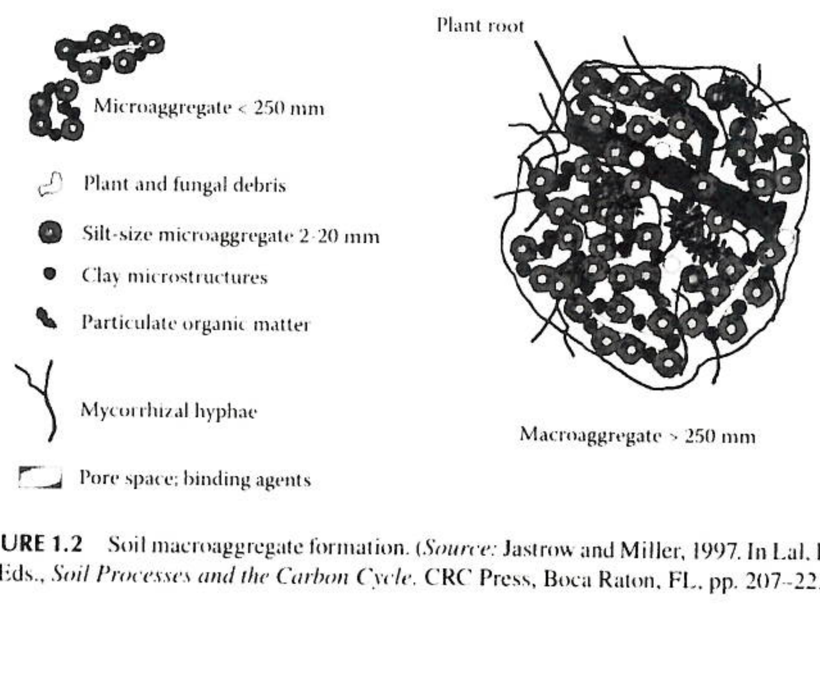

FIGURE 1.2 Soil macroaggregate formation. (*Source:* Jastrow and Miller, 1997. In Lal, R. et al., Eds., *Soil Processes and the Carbon Cycle*. CRC Press, Boca Raton, FL, pp. 207–223.)

of structural components such as lignin (Martens, 2000). Plant root and fungal hyphae form a network in soil that entangles microaggregates to form macroaggregates that are then stabilized by extracellular polysaccharides that confer increased resilience to aggregates in the presence of water (Elliott and Coleman, 1988; Tisdall, 1994). The length of fungal hyphae can be reduced by harvesting residue (Cookson et al., 1998), which may contribute to a reduction in aggregate stability.

The addition of fresh organic residue induces the formation and stabilization of macroaggregates by the addition of a C source for microbial activity (Golchin et al., 1994b; Jastrow, 1996; Six et al., 1999; Mikha and Rice, 2004; Johnson et al., 2007c). In a conceptual model proposed by Golchin et al. (1994a), plant residues are colonized by microorganisms as they enter the soil. Plant fragments also can be encrusted by mineral particles that become the centers of water-stable aggregates (Figure 1.2). Since these plant fragments are rich in readily decomposable carbohydrates, microbial metabolites permeate the coatings of mineral particles and stabilize the aggregates (Golchin et al., 1994a). In addition, soil conditions can cause increased solubility of some polyvalent cations such as Fe and Mn, thereby contributing to the formation of soil microaggregates and the stabilization of SOM (Figure 1.3) through formation of cation bridges (Elliott and Coleman, 1988). Thus, the addition of organic residues high in available C can promote the stabilization of soil aggregates; conversely, insufficient C inputs can lead to losses of stable aggregates.

Different management practices affect formation and stabilization of soil aggregates through their effects on SOM level and soil biota (Tisdall and Oades, 1982; O'Halloran et al., 1986; Beare and Bruce, 1993; Edwards et al., 1993; Frey et al., 1999; Six et al., 2000a). Cultivation affects soil structure due to the destruction of soil aggregates and the loss of SOM (Low, 1972; Van Veen and Paul, 1981; Tisdall

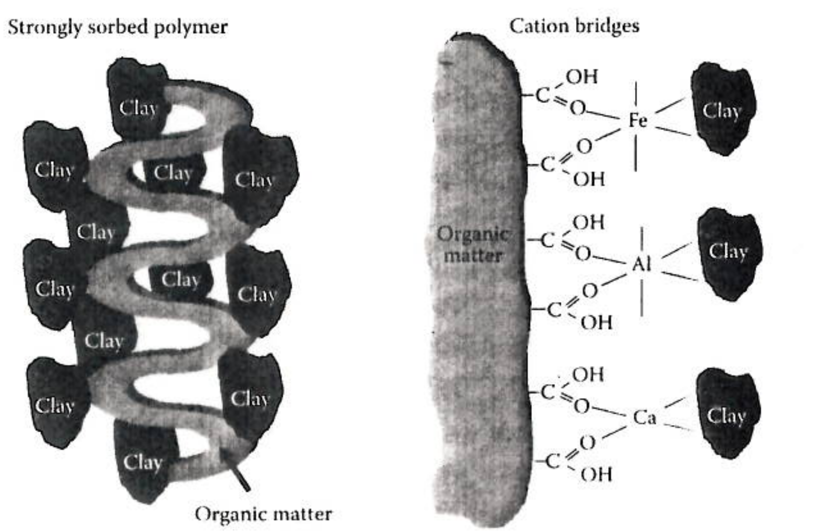

FIGURE 1.3 Soil microaggregate formation (<250 μm) and SOM stabilization. Note cation bridges that connect SOM and clay particles. (*Source:* Tisdall, J.M. and J.M. Oades. 1982. *J. Soil Sci.* 33:141–163.)

and Oades, 1982; Elliott, 1986; Angers et al., 1992; Six et al., 1998; Six et al., 1999). Losses of SOM from cultivation of grassland have been attributed at least partly to the mineralization of organic materials that bind microaggregates into macroaggregates (Elliott, 1986; Gupta and Germida, 1988).

McVay et al. (2006) observed that aggregate stability was greatest in treatments with the highest SOC in several long-term studies in Kansas. De Gryze et al. (2005) observed that aggregate formation increased linearly with increasing residue amounts at a rate of 12.0 ± 1.24 g aggregate g^{-1} residue added. They also observed that macroaggregates (> 2000 μm) increased from 3% to 40% as the amount of residue added was increased from 0 g to 3 g per 100 g soil. Water-stable aggregation index was significantly greater in a tillage-plus-straw-retained treatment (0.97) compared to tillage without straw treatment (0.68) (Singh et al., 1994). In another study, addition of a high-lignin organic material and corn stover increased water-stable aggregates (Johnson et al., 2007c). These results are indicative of the beneficial effects of organic matter addition on the aggregation process.

Soil aggregation affects soil water and aeration, which are important factors in crop production. The size, shape, and stability of soil aggregates impact pore size distribution (Lynch and Bragg, 1985). Soil structural stability depends on the ability of aggregates to remain intact when subjected to stress such as rapid wetting (Tisdall, 1996). Lynch and Bragg (1985) reported that unstable aggregates slake when wetted. Slaking occurs when aggregates are too unstable to withstand pressures resulting from entrapped air inside air-dried aggregates during rapid rewetting (Elliott, 1986; Gäth and Frede, 1995; Six et al., 2000b). Resistance to slaking is associated with large pieces of organic debris from plant roots, surface litter, and fungal hyphae (Oades, 1984). When air-dried soils are slowly rewetted, changes in aggregates are minimal (Six et al., 2000b). Under field conditions, aggregates near the surface are subjected to more slaking compared to aggregates below the surface layer that are protected from air drying and rapid wetting (Lynch and Bragg, 1985).

Soil aggregation is important for increasing water infiltration. Residue cover protects the soil surface from direct raindrop impact and minimizes aggregate slaking from fast rewetting, thus maintaining soil aggregates and reducing surface crusting compared with bare soil. Unstable aggregates at the surface can lead to the formation of crusts that inhibit water infiltration and air movement into the soil (Tisdall and Oades, 1982; Lynch and Bragg, 1985). Within 24 hours of the formation of surface crusts, the O_2 diffusion rate is reduced by 50% (Rathore et al., 1982). Not tilling and retaining crop stubble increased infiltration rate 3.7-fold compared with conventional tillage (three cultivation passes) and burnt stubble in a 24-year study (Zhang et al., 2007). Water-stable macroaggregates were positively correlated to hydraulic conductivity and negatively correlated to bulk density under dryland crop production in eastern Colorado (Benjamin et al., 2008). Govaerts et al. (2007a) reported that retaining wheat and maize residue improved water infiltration dramatically in both no-till and conventionally tilled plots. Harvest of non-grain biomass has the potential to increase water runoff and soil erosion by impairing soil structure through decreased aggregate stability and macroporosity of the soil surface.

SOIL EROSION

Soil erosion is a two-step process in which soil particles are detached in response to an energy input and then transported. Eroding soil can be moved by wind, water, ice, or gravity. These erosion processes redistribute soil within a landscape and can remove soil under some conditions. Soil loss by erosion removes topsoil from a soil profile. Areas subject to soil loss by erosion are typically less productive because of lower water-holding capacity and decreased fertility. Blowing soil can damage crop plants and soil deposition can bury small plants. From 1994 through 1996, soil erosion caused an annual productivity loss of at least \$40 km^{-2} throughout most of the United States Corn Belt; large areas experienced annual losses in excess of \$380 km^{-2} (Magleby, 2003). In addition to decreasing productivity, erosion causes off-site impairment of surface water and air quality, property damage, and detrimental effects on human and animal health. Damage caused by soil erosion in the United States has been estimated at \$2 billion to \$8 billion annually (Magleby, 2003).

The agricultural community recognizes that returning crop residues to the soil is essential to avoid large declines in SOM that adversely impact soil fertility, soil strength, aggregation, and other properties and thus negatively influence crop production. While crop residues affect determinants of soil erodibility (e.g., water-stable aggregates), this section will focus on the physical role of crop residues in reducing soil lost by erosion.

Years of water erosion research consistently indicate for a given soil type, loss rates by water erosion increase with increasing slope gradient, slope length, and rainfall intensity. Soil cover and production practices also influence soil loss via erosion. Crop residue decreases the detachment and transport of soil by water by intercepting raindrops before they impact the soil, by slowing the flow of water over the soil surface, increasing the depth of water on the surface, and by providing small areas of ponded water where sediment can be deposited (Cogo et al., 1982). Soil cover decreases soil loss by water erosion, but the relationship is not linear (Figure 1.4; Lindstrom, 1986; Erenstein, 2002; Merrill et al., 2006). Typically, the ratio of soil loss with a groundcover relative to loss incurred without groundcover decreases exponentially with increasing cover (Figure 1.4). For example, on the same soil type, the amount of soil lost due to water erosion was similar with 100% groundcover and with 60% corn stover or wheat straw cover (Cogo et al., 1982).

Soil loss rates through wind erosion are affected by surface roughness, field dimensions, and wind characteristics. Similar to water erosion, soil cover can decrease soil loss by wind erosion. The effect of soil cover on loss from wind erosion is a complex function of crop type, residue orientation, and other factors. Generally, standing stubble is more effective at reducing wind erosion than flattened residue, and stubble oriented in rows perpendicular to the wind direction is more effective than stubble in rows parallel to the wind direction (Skidmore, 1988). Bilbro and Fryrear (1994) summarized the relationship of soil cover to soil loss by wind erosion. The ratio of the amount of soil lost from protected soil to that lost from flat, bare soil decreased exponentially with increasing groundcover (Soil loss ratio = $e^{(-0.04380 * \% \text{ soil cover})}$). This relationship indicates that when soil is at least 50% covered with residue, loss by wind erosion is expected to be 10% or less of losses from flat, bare soil. Wind tunnel

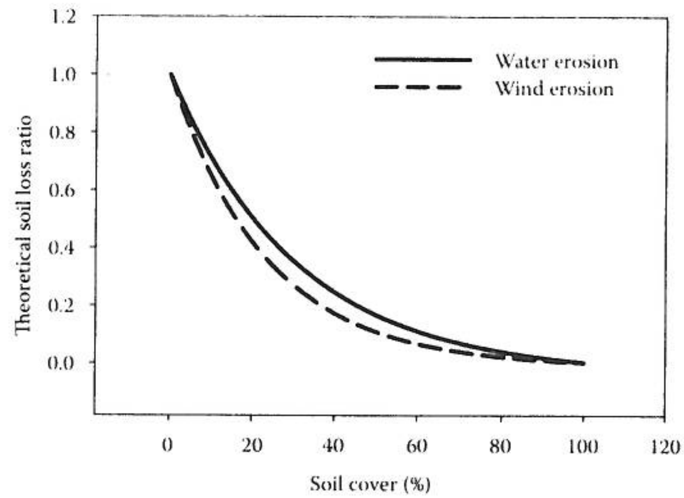

FIGURE 1.4 Soil loss ratios predicted by the revised universal soil loss equation (RUSLE) and revised wind erosion equation (RWEQ) with varying amounts of soil cover. Soil loss ratios include specified amount of residue cover for bare soil under high erodibility conditions. (*Source:* Merrill, S.D. et al. 2006. *J. Soil Water Conserv.* 61:7–13.)

and field studies suggest even low fractions of residue cover can drastically decrease wind erosion (Bilbro and Fryrear, 1994).

For both wind and water erosion, the relationship between soil cover and loss is a non-linear function where >50% groundcover can virtually eliminate erosion (Figure 1.4). Generally, researchers have observed that the amount of crop residue can be estimated from grain yield. For example, linear relationships between the amount of crop residue and grain yield have been reported for small grains (McCool et al., 2006) and corn (Linden et al., 2000), such that grain yield has been used to predict residue yield (Johnson et al., 2006a).

The relationship between residue amount and groundcover varies with crop. Gregory (1982) presents a method to estimate the fraction of groundcover from the mass of residue per area of ground, using coefficients determined from field studies. For each crop, Gregory reported that the fraction of soil covered increased exponentially with increasing residue amount, with the lowest rate of increase for cotton (*Gossypium* sp.) and the highest rate of increase for oats (*Avena sativa* L.). Thus, the amount of soil cover generally increases exponentially with grain yield. This exponential relationship indicates that harvesting residue will not result in a proportional decrease in groundcover. For example, under most circumstances, harvesting 25% of the crop residue mass will decrease the amount of groundcover by less than 25%.

Roots and growing plant materials also effectively reduce wind and water erosion and show trends similar to crop residues. Soil loss rates by water erosion decrease exponentially with increasing vegetative cover; soil loss is approximately the same for 60% and 100% vegetation cover (Stocking, 1988). Research also has shown exponential decreases in soil loss rates by water erosion with increasing root mass (Gyssels et al., 2005).

Cover crops and perennials also may reduce soil erosion drastically by keeping the soil covered. The amount of soil cover provided by growing plants, their height, structure, orientation, rooting characteristics, position, and other factors are important in determining their effectiveness at reducing erosion. For example, erosion from land planted to woody species can be comparable to traditional row-cropped land if no cover crop is present (Malik et al., 2000), and erosion rates can be very high in mature forests (Stocking, 1988). Positioning of plants plays a role in their ability to reduce erosion. In a field study, measured soil loss exceeded the predicted wind erosion soil loss because plants were sited between soil ridges, limiting their ability to reduce erosion (Van Donk and Skidmore, 2003).

For residue to be effective for decreasing erosion, it must cover the soil surface during the erosive event. Thus, residue must remain on the soil surface until the next crop is established. Tillage, seeding, and other soil disruptions decrease crop residue present on soil surface. Extensive research has been conducted to determine soil erosion rates under differing conditions of cover following tillage. Eck et al. (2001) reported that each tillage operation can decrease the crop residue cover by 10% to 20% (for mild disturbance caused by some drills or planters) to 95% or more (for aggressive tillage such as a moldboard plow). Tillage incorporates residue into the soil, where it can still contribute to C cycling and nutrient cycling, but harvesting residue removes soil cover along with C and other nutrients.

Soils with poor aggregate structures exhibit less resilience against erosive forces such as wind and water. In both tilled and not-tilled soils, residue harvest increased the number of small aggregates susceptible to wind erosion (Singh et al., 1994; Malhi et al., 2006; Singh and Malhi, 2006; Malhi and Kutcher, 2007; Malhi and Lemke, 2007). Blanco-Canqui and Lal (2007) also reported that removing wheat straw reduced soil aggregate strength compared to mulching. Tillage can increase erosion through decreased aggregate stability and increased soil detachment. Conversely, under some conditions, tillage can decrease soil loss by wind and water due to increased water infiltration and soil surface roughness (Dabney et al., 2004).

The effect of interaction between tillage and crop residue on soil loss by erosion is complex and varies with soil properties such as moisture (Cogo et al., 1982; Dabney et al., 2004). Reduced tillage can provide more soil cover (Guy and Cox, 2002), but residue removal can negate some of the benefits of reduced tillage. In both the northern and southern U.S., removing corn residue from reduced-till or no-till plots can result in soil loss rates by water erosion similar to those for conventionally tilled soil with no residue removed (Lindstrom, 1986; Dabney et al., 2004). In no-till soils, the portion of standing residue relative to flat residue changes with time (Steiner et al., 2000), and this is expected to alter the effectiveness of remaining residues in reducing wind and water erosion. McCool et al. (2006) noted that for small grains, stems are the most important components for reducing erosion because they are more resistant to degradation and relocation than leaves. Some studies suggest that decomposition of corn residue over winter reduces cover by 20% to 30% (Van Donk and Skidmore, 2003; Wilson et al., 2008). Residue decomposition rates vary with the chemical composition of plant materials, temperature, moisture, soil characteristics, and placement (Paul, 1991).

The highly non-linear nature of the erosion process and the simultaneous interactions of numerous processes make soil loss by erosion very difficult to predict. The amount of soil lost by erosion is a complex function of macro- and microtopography, the energy of the wind or water impacting the soil, erodibility, and other factors. Current models based on years of research indicate soil cover is a critical determinant of soil loss by wind and water erosion (Figure 1.4) that varies with soil types (Lindstrom, 1986; Erenstein, 2002). Additional research is needed to more completely characterize the implications of biomass harvest on long-term and episodic soil erosion in relation to biofuel production.

WATERSHED HYDROLOGICAL IMPACTS

Generally, much of the preceding material is based on studies of residue cover as affected by tillage practices (i.e., incorporation) rather than removal. Most of the work was done at plot scale. Extending this knowledge to include effects on watershed hydrology is difficult. Uhlenbrook (2007) stated that no research had been published on the impacts of biofuel development on watershed hydrology. It is critical to develop an understanding of these impacts as soon as possible and more clearly appreciate the differences between residue incorporation and removal in terms of their relative effects on interacting C, nutrient, and water cycles.

As more land becomes dedicated to producing biofuel crops, environmental impacts of associated land use conversions will depend on the nature and extent of changes in land cover and vegetation management. Any change in land use will influence the partitioning of precipitation into canopy interception, overland flow, evaporation, transpiration, and deep percolation, along with accompanying hydrological consequences. In the tropics, land use may shift toward clearing of forests and expansion of agricultural areas with hydrological consequences that are difficult to model due to limited datasets covering hydrology of tropical watersheds (Uhlenbrook, 2007).

In temperate zones, land use conversion for biofuel crops may expand perennial cover at the expense of annual crop cover. Short-rotation tree crops (*Poplar* or *Salix*) or tall prairie grass species (switchgrass) can be highly productive in semi-arid to humid temperate climates, and probably require fewer nutrient inputs than annual crops (Johnson et al., 2007d). Land cover conversions to these perennial crops would increase transpiration and reduce overland flow (Rachman et al., 2004; Updegraffa et al., 2004), and have been shown to sequester more soil C than corn in highly fertile soils (Zan et al., 2001). This would benefit the hydrologic regimens of Midwestern streams and rivers based on recent trends of increasing precipitation amounts and intensities that are predicted to continue (Nearing et al., 2004; Hodgkins et al., 2007).

To the extent that bioenergy feedstocks are derived from residues of annual crops such as corn, potential hydrologic impacts may lean toward greater fractions of precipitation lost via overland flow and less near-surface soil moisture (Rhoton et al., 2002; Montgomery, 2007). Tillage practices that incorporate residue were shown to increase the overland flow component of stream discharge from small watersheds by nearly 50% in one long-term (25-year) study (Tomer et al., 2005). The differences occurred during large rainfall events (up to 100 mm d^{-1}) and intermediate and small runoff-producing events. At the same time, watersheds with conservation tillage systems like ridge tills yielded greater baseflow and total discharge, and showed more rapid recovery from drought. The increase in discharge was about 3% of the total hydrologic budget, accompanied by less variation in streamflow. Conservation tillage resulted in lower bulk densities, greater SOM, and under wet soil conditions, greater water contents than conventional tillage (Tomer et al., 2006). Differences in hydrology likely result from the direct impacts of soil cover on water flow and changes in aggregate stability and infiltration capacity. Stover removal reduced saturated conductivity on three Ohio soils (Blanco-Canqui et al., 2007), and is expected to result in greater overland flow and decreased baseflow. Ensuring adequate ground cover following residue harvest may be critical to increase infiltration and reduce surface runoff, especially if climate change increases amounts and intensities of precipitation.

INTEGRATION OF CARBON, NUTRIENT, AND WATER CYCLES

Removal of non-grain biomass simultaneously interacts with C, nutrient, microclimate, and hydrological cycles (Figure 1.5). Harvesting residue in excess of MSC will reduce SOC. Excess harvesting limits the organic material needed for soil aggregation, making soil more susceptible to erosive forces. Removal of biomass can lead to surface sealing of soil, reducing infiltration and increasing surface runoff. Surface runoff across unprotected soils removes top soil and the nutrients it contains. An influx of P and K into surface water promotes algal blooms, eutrophication, and hypoxia (Kim and Dale, 2005). Over time, soil erosion can result in exposed subsoil that typically is less fertile with less SOM

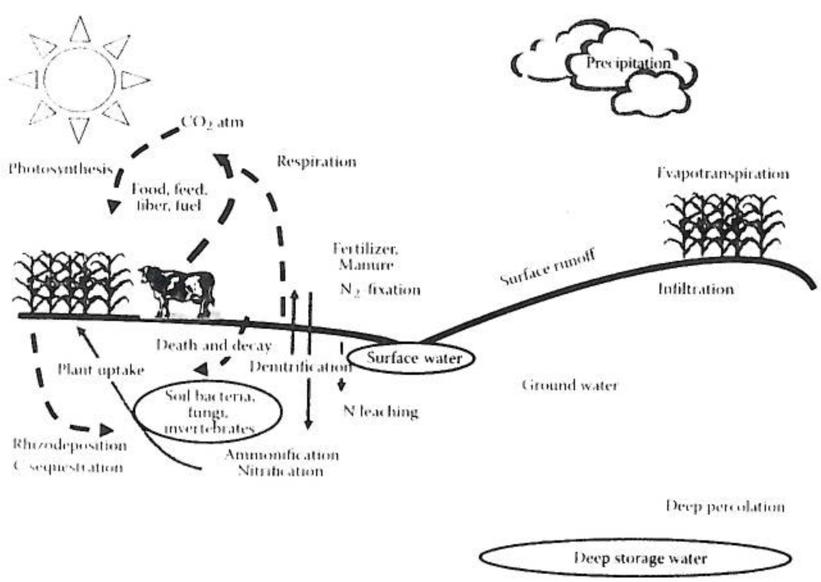

FIGURE 1.5 Interconnection of carbon, nitrogen, and water cycles.

compared to surface soil, thus impeding soil nutrient cycling and decreasing nutrient-holding capacity. As a result, additional nutrients are required to support production.

The lack of biomass inputs decreases soil fauna (Tables 1.3 through 1.5) and can interfere with nutrient cycling. Retaining biomass on the surface can promote infiltration, reducing surface runoff, but can increase the potential for leaching nutrients such as nitrates into ground water. Surface residue keeps the soil surface cooler, slowing evaporation and promoting denitrifying conditions by extending the duration of anaerobic soil conditions (Ball et al., 1999; Aulakh et al., 2001). Cooler surface soil can delay germination and retard early season growth in regions with cool, wet springs (Swan et al., 1994). In warmer drier climates, the lack of surface cover promotes water stress and decreases yield (Power et al., 1986; Wilhelm et al., 1986). The interactions of residue management with biological, chemical and physical processes are complicated by climatic factors and management practices. The key is finding a balanced non-grain biomass harvest approach that supports soil processes and controls erosion to minimize potential negative effects of non-grain biomass removal.

BIOMASS HARVEST: COMPENSATION STRATEGIES

Several strategies can avoid or reduce loss of SOM and solve related problems arising from biomass harvest. Harvest rates should be limited to those that maintain SOM and do not exacerbate erosion. Reducing or eliminating tillage utilizes remaining residue as ground cover to reduce erosion. If harvest rates exceed the amount needed to provide adequate inputs for SOM, alternative inputs such as manure should be applied. In general, manures tend to increase SOC under a wide range of management and climatic conditions (Johnson et al., 2007b). Animal manures contain 40% to 60% C on a dry weight basis and can promote SOC sequestration and provide nutrient inputs (CAST, 1992). Another strategy is planting cover crops and living mulches where crop residues are harvested to prevent erosion and replace C and N removed the residues (Zemenchik et al., 2000; Drinkwater and Snapp, 2007).

Other amendments such as application of by-products of cellulosic fermentation containing high lignin concentrations improved soil quality characteristics in laboratory studies (Johnson et al., 2004; Johnson et al., 2007c). Another by-product is biochar from pyrolysis or gasification. Biochar has the potential to enhance plant growth by supplying and retaining nutrients and improving soil physical and chemical properties. Biochar may also remove pesticides or other pollutants from soil water (Glaser et al., 2002; Lehmann et al., 2003; Lehmann et al., 2006; Lehmann and Rondon, 2006). Compensation strategies will vary by management system, climatic regime, and suitability of strategy to farming systems.

SUMMARY

Harvesting crop non-grain biomass initiates a cascade of interrelated biological, chemical, and physical soil events. Biomass harvest has the potential to disrupt soil nutrient dynamics, water relations, and other important soil processes. Considerable knowledge exists about ways to minimize the risks of harvesting non-grain biomass.

It is essential that regional and site-specific guidelines be developed as quickly as possible. Clearly, non-grain biomass harvest must be limited to avoid loss of SOM and prevent excessive soil erosion.

Management strategies that enhance soil quality such as reduced or no tillage, inclusion of perennial species, use of cover crops and living mulches, and applying amendments such as biochar or manure may compensate for non-grain biomass removal. Soil quality building strategies reduce the risk of erosion, improve C and nutrient cycling, and increase aggregate stabilization. In developing a biofuel economy, it is paramount that soil resources be protected to secure our nation's ability to provide adequate food, feed, fiber, and fuel for a growing world.

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REFERENCES

Abiven, S. et al. 2005. Mineralisation of C and N from root, stem and leaf residues in soil and role of their biochemical quality. *Biol. Fertil. Soils* 42: 119–128.

Adler, P.R. et al. 2006. Biomass yield and biofuel quality of switchgrass harvested in fall or spring. *Agron. J.* 98: 1518–1525.

Al-Kaisi, M.M., X. Yin, and M.A. Licht. 2005. Soil carbon and nitrogen changes as affected by tillage system and crop biomass in a corn–soybean rotation. *Appl. Soil Ecol.* 30: 174–191.

Al-Yemeni, M.N. and J. Grace. 1995. Radiation balance of an alfalfa crop in Saudi Arabia. *J. Arid Environ.* 29: 447–454.

Albright, W.A. 1938. Loss of soil organic matter and its restoration. In *Soils and Men: Yearbook of Agriculture*. Washington, D.C., U.S. Department of Agriculture, pp. 347–360.

Allmaras, R.R., D.R. Linden, and C.E. Clapp. 2004. Corn residue transformations into root and soil carbon as related to nitrogen, tillage, and stover management. *Soil Sci. Soc. Am. J.* 68: 1366–1375.

Amos, B. and D.T. Walters. 2006. Maize root biomass and net rhizodeposited carbon: An analysis of the literature. *Soil Sci. Soc. Am. J.* 70: 1489–1503.

Andren, O. and K. Paustian. 1987. Barley straw decomposition in the field: A comparison of models. *Ecology* 68: 163–210.

Angers, D.A. 1998. Water-stable aggregation in Quebec silty clay soil: Some factors controlling its dynamics. *Soil Tillage Res.* 47: 91–96.

Angers, D.A., A. Pesant, and J. Vigneux. 1992. Early cropping-induced changes in soil aggregation, organic matter, and microbial biomass. *Soil Sci. Soc. Am. J.* 56: 115–119.

Aulakh, M.S. et al. 2001. Denitrification, N_2O and CO_2 fluxes in rice–wheat cropping system as affected by crop residues, fertilizer N and legume green manure. *Biol. Fertil. Soils* 34: 675–689.

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Bainbridge, D.A. 1986. High performance low cost buildings of straw. *Agric. Ecosyst. Environ* 16: 281–284.

Balesdent, J. and M. Balabane. 1996. Major contribution of roots to soil carbon storage inferre from maize cultivated soils. *Soil Biol. Biochem.* 28: 1261–1263.

Ball, B.C., J.P. Parker, and A. Scott. 1999. Soil and residue management effects on croppin conditions and nitrous oxide fluxes under controlled traffic in Scotland 2. Nitrous oxide soil N status and weather. *Soil Tillage Res.* 52: 191–201.

Barber, S.A. 1979. Corn residue management and soil organic matter. *Agron. J.* 71: 625–627.

Bayer, C. et al. 2006. A method for estimating coefficients of soil organic matter dynamics based on long-term experiments. *Soil Tillage Res.* 91: 217–226.

Beare, M.H. and R.R. Bruce. 1993. A comparison of methods for measuring water-stable aggregates: Implications for determining environmental effects on soil structure. *Geoderma* 56: 87–104.

Bending, G.D., M.K. Turner, and J.E. Jones. 2002. Interactions between crop residue and soil organic matter quality and the functional diversity of soil microbial communities. *Soil Biol. Biochem.* 34: 1073–1082.

Benjamin, J.G., M.M. Mikha, and M.F. Vigil. 2008. Organic carbon effects on soil physical and hydraulic properties in a semiarid climate. *Soil Sci. Soc. Am. J.* 72: 1357–1362.

Benoit, G.R. et al. 1986. Tillage-residue effects on snow cover, soil water, temperature and frost. *Trans. ASAE* 29: 473–479.

Bhatnagar, V.K., T.N. Chaudhary, and B.R. Sharma. 1983. Effect of tillage and residue management on properties of two coarse-textured soils and on yield of irrigated wheat and groundnut. *Soil Tillage Res.* 3: 27–37.

Bilbro, J.D. and D.W. Fryrear. 1994. Wind erosion losses as related to plant silhouette and soil cover. *Agron. J.* 86: 550–553.

Black, A.L. 1973. Soil property changes associated with crop residue management in a wheat-fallow rotation. *Soil Sci. Soc. Am. J.* 37: 943–946.

Blanco-Canqui, H. and R. Lal. 2007. Soil structure and organic carbon relationships following 10 years of wheat straw management in no-till. *Soil Tillage Res.* 95: 240–254.

Blanco-Canqui, H. et al. 2006a. Changes in long-term no-till corn growth and yield under different rates of stover mulch. *Agron. J.* 98: 1128–1136.

Blanco-Canqui, H. et al. 2006b. Corn stover impacts on near-surface soil properties of no-till corn in Ohio. *Soil Sci. Soc. Am. J.* 70: 266–278.

Blanco-Canqui, H. et al. 2007. Soil hydraulic properties influenced by corn stover removal from no-till corn in Ohio. *Soil Tillage Res.* 92: 144–155.

Blevins, R.L. and W.W. Frye. 1993. Conservation tillage: An ecological approach to soil management. *Adv. Agron.* 51: 33–78.

Borie, F. et al. 2002. Interactions between crop residues application and mycorrhizal developments and some soil-root interface properties and mineral acquisition by plants in an acidic soil. *Biol. Fertil. Soils* 36: 151–160.

Bransby, D.I., S.B. McLaughlin, and D.J. Parrish. 1998. A review of carbon and nitrogen balances in switchgrass grown for energy. *Biom. Bioenergy* 14: 379–384.

Breakwell, D.P., and R.F. Turco. 1989. Nutrient and phytotoxic contributions of residue to soil in no-till continuous-corn ecosystems. *Biol. Fertil. Soils* 8: 328–334.

Bristow, K.L. 1988. The role of mulch and its architecture in modifying soil temperature. *Aust. J. Soil Res* 26: 269–280.

Bristow, K.L. et al. 1986. Simulation of heat and moisture transfer through a surface residue soil system. *Agric. For. Meteorol.* 36: 193–214.

Buerkert, A., A. Bationo, and K. Dossa. 2000. Mechanisms of residue mulch-induced cereal growth increases in West Africa. *Soil Sci. Soc. Am. J.* 64: 346–358.

Burgess, M.S., G.R. Mehuys, and C.A. Madramootoo. 2002. Decomposition of grain corn residues (*Zea mays* L.): Litterbag study under three tillage systems. *Can. J. Soil Sci.* 82: 127–138.

Bussière, F. and P. Cellier. 1994. Modification of the soil temperature and water content regimes by a crop residue mulch: Experiment and modeling. *Agric. For. Meteorol.* 68: 1–28.

Campbell, C.A. et al. 1991. Effect of crop rotations and fertilization on soil organic matter and some biochemical properties of a thick black chernozem. *Can. J. Soil Sci.* 71: 377–387.

Cassida, K.A. et al. 2005. Biofuel component concentrations and yields of switchgrass in South Central U.S. environments. *Crop Sci.* 45: 682–692.

CAST. 1992. Preparing U.S. agriculture for global climate change. Task Force Report 119. Council for Agricultural Science and Technology, Ames, Iowa.

Chaney, K. and R.S. Swift. 1984. The influence of organic matter on aggregate stability in some British soils. *J. Soil Sci.* 35: 223–230.

Christensen, B.T. 1986. Barley straw decomposition under field conditions: Effects of placement and initial nitrogen content on weight loss and nitrogen dynamics. *Soil Biol. Biochem.* 18: 523–529.

Chung, S.O. and R. Horton. 1987. Soil heat and water flow with a partial surface mulch. *Water Resour. Res.* 23: 2175–2186.

Clay, D.E. et al. 2006. Theoretical derivation of stable and nonisotopic approaches for assessing soil organic carbon turnover. *Agron. J.* 98: 443–450.

Clay, D.E. et al. 2001. Factors influencing spatial variability of apparent electrical conductivity. *Commun. Soil Sci. Plt. Anal.* 32: 2993–3008.

Clifton-Brown, J.C. and I. Lewandowski. 2002. Screening miscanthus genotypes in field trials to optimise biomass yield and quality in Southern Germany. *Eur. J. Agron.* 16: 97–110.

Cogo, N.P., W.C. Moldenhauer, and G.R. Foster. 1982. Soil loss reductions from conservation tillage practices. *Soil Sci. Soc. Am. J.* 48: 368–373.

Cook, R.J., M.G. Boosalis, and B. Doupnik. 1978. Influence of crop residue on plant diseases. In Oschwald, W.R. et al., Eds., *Crop Residue Management Systems*, Special Publication 31. SSSA, Madison, WI, pp. 147–163.

Cookson, W.R., M.H. Beare, and P.E. Wilson. 1998. Effects of prior crop residue management on microbial properties and crop residue decomposition. *Appl. Soil Ecol.* 7: 179–188.

Cox, W.J. et al. 1990. Tillage effects on some soil physical and corn physiological characteristics. *Agron. J.* 82: 806–812.

Crookston, R.K. et al. 1991. Rotational cropping sequence affects yield of corn and soybeans. *Agron. J.* 83: 108–113.

Dabney, S.M. et al. 2004. History, residue, and tillage effects on erosion of loessial soil. *Trans. ASAE* 47: 767–775.

Dam, R.F. et al. 2005. Soil bulk density and crop yield under eleven consecutive years of corn with different tillage and residue practices in a sandy loam soil in central Canada. *Soil Tillage Res.* 84: 41–53.

De Gryze, S., J. Six, C. Brits, and R. Merckx. 2005. A quantification of short-term macroaggregate dynamics: Influences of wheat residue input and texture. *Soil Biol. Biochem.* 37: 55–66.

Debosz, K., P.H. Rasmussen, and A.R. Pedersen. 1999. Temporal variations in microbial biomass C and cellulolytic enzyme activity in arable soils: Effects of organic matter input. *Appl. Soil Ecol.* 13: 209–218.

Degens, B.P. 1997. Macroaggregation of soil by biological bonding and binding mechanisms and the factors affecting these: A review. *Aust. J. Soil Res.* 35: 431–459.

Dexter, A.R., D. Hein, and J.S. Hewett. 1982. Macrostructure of the surface layer of self-mulching clay in relation to cereal stubble management. *Soil Tillage Res.* 2: 251–264.

Dhima, K.V. et al. 2006. Allelopathic potential of winter cereals and their cover crop mulch effect on grass weed suppression and corn development. *Crop Sci.* 46: 345–352.

Diaz-Zorita, M. and G.A. Grosso. 2000. Effect of soil texture, organic carbon and water retention on the compactability of soils from the Argentinean pampas. *Soil Tillage Res.* 54: 121–126.

Dipardo, J. 2000. Outlook for biomass ethanol production and demand. U.S. Energy Information Agency, Washington, D.C. http://www.eia.doe.gov/oiaf/analysispaper/pdf/biomass.pdf (posted March 10, 2002; verified Sept. 4, 2008).

Doran, J.W. and T.B. Parkin. 1994. Defining and assessing soil quality. In Doran, J.W. et al., Eds., *Defining Soil Quality for a Sustainable Environment.* Special Publication 35. SSSA, Madison, WI, pp. 3–21.

Dormaar, J.R. 1983. Chemical properties of soil and water-stable aggregation after 67 years of cropping to spring wheat. *Plant Soil* 75: 51–61.

Drinkwater, L.E. and S.S. Snapp. 2007. Nutrients in agroecosystems: Rethinking the management paradigm. *Adv. Agron.* 92: 63–186.

Drury, C.F. et al. 2003. Impacts of zone tillage and red clover on corn performance and soil physical quality. *Soil Sci. Soc. Am. J.* 67: 867–877.

Duffy, M.D. and V.Y. Nanhou. 2001. Costs of producing switchgrass for biomass in Iowa. <http://www.extension.iastate.edu/Publications/PM1866.pdf> (posted April 2001; verified Sept. 5, 2008).

Duley, F.L. and J.C. Russel. 1939. The use of crop residues for soil and moisture conservation. *J. Am. Soc. Agron.* 31: 703–709.

Duppong, L.M. et al. 2004. The effect of natural mulches on crop performance, weed suppression and biochemical constituents of catnip and St. John's wort. *Crop Sci.* 44: 861–869.

Eck, K.J., D.E. Brown, and A.B. Brown. 2001. Estimating corn and soybean residue cover. Purdue University Cooperative Extension Service. http://www.agry.purdue.edu/ext/pubs/AY-269-W.pdf (verified Sept. 4, 2008).

Edwards, A.P. and J.M. Bremner. 1967. Microaggregates in soils. *J. Soil Sci.* 18: 64–73.

Edwards, W.M. et al. 1993. Factors affecting preferential flow of water and atrazine through earthworm burrows under continuous no-tillage corn. *J. Environ. Qual.* 22: 453–457.

Edwards, W.M. et al. 1990. Effect of *Lumbricus terrestris* L. burrows on hydrology of continuous no-till corn fields. *Geoderma* 46: 73–84.

Ekenler, M. and M.A. Tabatabai. 2003. Tillage and residue management effects on β-glucosaminidase activity in soils. *Soil Biol. Biochem.* 35: 871–874.

Elliott, E.T. 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. *Soil Sci. Soc. Am. J.* 50: 627–633.

Elliott, E.T. and D.C. Coleman. 1988. Let the soil work for us. *Ecol. Bull* 39: 23–32.

Enz, J.W., L.J. Brun, and J.K. Larsen. 1988. Evaporation and energy balance for bare and stubble covered soil *Agric. Forest Meteor.* 43: 59–70.

Erenstein, O. 2002. Crop residue mulching in tropical and semi-tropical countries: Evaluation of residue availability and other technological implications. *Soil Tillage Res.* 67: 115–133.

Fageria, N.K. 2004. Dry matter yield and shoot nutrient concentrations of upland rice, common bean, corn, and soybean grown in rotation on an oxisol. *Commun. Soil Sci. Plant Anal.* 35: 961–974.

FAO. 2004. Unified bioenergy terminology. Food and Agriculture Organization of the United Nations, Forestry Department, Wood Energy Programme. ftp://ftp.fao.org/docrep/fao/007/j4504e/j4504e00.pdf (posted 2004; verified Sept. 4, 2008).

Fatondji, D. et al. 2006. Effect of planting technique and amendment type on pearl millet yield, nutrient uptake, and water use on degraded land in Niger. *Nutr. Cycl. Agroecosys.* 76: 203–217.

Follett, R.F., J.Z. Castellanos, and E.D. Buenger. 2005. Carbon dynamics and sequestration in an irrigated vertisol in Central Mexico. *Soil Tillage Res.* 83: 148–158.

Ford, J.H. and D.R. Hicks. 1992. Corn growth and yield in uneven emerging stands. *J. Prod. Agric.* 5: 185–188.

Franck, W.M. et al. 1997. Decomposition of litter produced under elevated CO_2: Dependence on plant species and nutrient supply. *Biogeochemistry* 36: 223–237.

Frank, A.B. et al. 2004. Biomass and carbon partitioning in switchgrass. *Crop Sci.* 44: 1391–1396.

Franzluebbers, A.J., F.M. Hons, and V.A. Saladino. 1995. Sorghum, wheat and soybean production as affected by long-term tillage, crop sequence and N fertilization. *Plant Soil* 173: 55–65.

Fraser, P.M. and J.E. Piercy. 1998. The effects of cereal straw management practices on lumbricid earthworm populations. *Appl. Soil Ecol.* 9: 369–373.

Fraser, P.M., P.H. Williams, and R.J. Haynes. 1996. Earthworm species, population size and biomass under different cropping systems across the Canterbury Plains, New Zealand. *Appl. Soil Ecol.* 3: 49–57.

Frey, S.D., E.T. Elliott, and K. Paustian. 1999. Bacterial and fungal abundance and biomass in conventional and no-tillage agroecosystems along two climatic gradients. *Soil Biol. Biochem.* 31: 573–585.

Garcia-Huidobro, J., J. Monteith, and G. Squire. 1982. Time, temperature, and germination of pearl millet (*Pennisetum typhoides* S&H). *J. Exp. Bot.* 33: 288–296.

Gäth, S. and H.G. Frede. 1995. Mechanisms of air slaking. In Hartge, K.H. and Stewart, B.A., Eds., *Soil Structure: Its Development and Function*. Advances in Soil Science Series. CRC Press, Boca Raton, FL, pp. 159–173.

Gercel, H.F. 2002. Production and characterization of pyrolysis liquids from sunflower-pressed bagasse. *Bioresour. Technol.* 85: 113–117.

Ghidey, F. and E. Alberts. 1993. Residue type and placement effects on decomposition: Field study at model evaluation. *Trans. ASAE* 36: 1611–1617.

Glaser, B., J. Lehman, and W. Zech. 2002. Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal: Review. *Biol. Fertil. Soils* 35: 219–230.

Golchin, A., J.M. Oades, J.O. Skjemstad, and P. Clark. 1994a. Soil structure and carbon cycling. *Aust. J. Soil Res.* 32: 1043–1068.

Golchin, A. et al. 1994b. Study of free and occluded particulate organic matter in soils by solid state ^{13}C CP/MAS NMR spectroscopy and scanning electron microscopy. *Aust. J. Agric. Res.* 32: 285–309.

Govaerts, B. et al. 2007a. Infiltration, soil moisture, root rot and nematode populations after 12 years of different tillage, residue and crop rotation managements. *Soil Tillage Res.* 94: 209–219.

Govaerts, B. et al. 2007b. Influence of tillage, residue management, and crop rotation on soil microbial biomass and catabolic diversity. *Appl. Soil Ecol.* 37: 18–30.

Govaerts, B. et al. 2008. Long-term consequences of tillage, residue management, and crop rotation on selected soil micro-flora groups in the subtropical highlands. *Appl. Soil Ecol.* 38: 197–210.

Graham, R.L. et al. 2007. Current and potential U.S. corn stover supplies. *Agron. J.* 99: 1–11.

Gregory, J.M. 1982. Soil cover prediction with various amounts and types of crop residue. *Trans. ASAE* 25: 1333–1337.

Guérif, J. 1990. Factors influencing compaction-induced increases in soil strength. *Soil Tillage Res.* 16: 167–178.

Gupta, V.V.S.R. and J.J. Germida. 1988. Distribution of microbial biomass and its activity in different soil aggregate size classes as affected by cultivation. *Soil Biol. Biochem.* 20: 777–786.

Guy, S.O. and D.B. Cox. 2002. Reduced tillage increases residue groundcover in subsequent dry pea and winter wheat crops in the Palouse region of Idaho. *Soil Tillage Res.* 66: 69–77.

Gyssels, G. et al. 2005. Impact of plant roots on the resistance of soil to erosion by water: Review. *Progr. Phys. Geogr.* 29: 189–221.

Hagen, L.J. 1996. Crop residue effects on aerodynamic processes and wind erosion *J. Theor. Appl. Clim.* 54: 39–46.

Halvorson, A.D. and J.M.F. Johnson. 2009. Irrigated corn cob production in the Central Great Plains. *Agron. J.*, in press.

Halvorson, A.D. and C.A. Reule. 2007. Irrigated, no-till corn and barley response to nitrogen in Northern Colorado. *Agron. J.* 99: 1521–1529.

Havlin, J.L. and D.E. Kissel. 1997. Management effects on soil organic carbon and nitrogen in the east-central Great Plains of Kansas. In Paul, E.A. et al., Eds., *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*. CRC Press, Boca Raton, FL, pp. 381–386.

Haynes, R.J., R.S. Swift, and R.C. Stephen. 1991. Influence of mixed cropping rotations (pasture-arable) on organic matter content, stable aggregation and clod porosity in a group of soils. *Soil Tillage Res.* 19: 77–87.

Heal, O.W., J.M. Anderson, and M.J. Swift. 1997. Plant litter quality and decomposition: Historical overview. In Cadish, G. and Giller, K.E., Eds., *Driven by Nature: Plant Litter Quality and Decomposition*. CAB International, Wallingford, U.K., pp. 47–66.

Hettenhaus, J.R., R. Wooley, and A. Wiselogel. 2000. Biomass commercialization prospects in the next 2 to 5 years: Biomass colloquies 2000 NREL/ACO-9-29-039-01. National Renewable Energy Laboratory, Golden, CO. http://www.nrel.gov/docs/fy01osti/28886.pdf (posted Oct. 10, 2000; verified March 1, 2007).

Hillel, D. 1998. *Environmental Soil Physics*. Academic Press, San Diego, CA.

Hobbs, J.A. and P.I. Brown. 1965. Effects of cropping and management on nitrogen and organic carbon contents of a western Kansas soil. *Agric. Exp. Stn. Tech. Bull.* 144. Kansas State University, Manhattan.

Hodgkins, G.A., R.W. Dudley, and S.S. Aichele, Eds. 2007. Historical changes in precipitation and streamflow in the U.S. Great Lakes Basin, 1915–2004. U.S. Geological Survey Scientific Investigations Report 5118. Reston, VA, p. 37.

Hooker, B.A. et al. 2005. Long-term effects of tillage and corn stalk return on soil carbon dynamics. *Soil Sci. Soc. Am. J.* 69: 188–196.

Horner, G.M. et al. 1960. Effect of cropping practices on yield, soil organic matter, and erosion in the Pacific Northwest wheat region. USDA ARS Coop. Bull. 1, Pullman, WA.

Horton, R. et al. 1996. Crop residue effects on surface radiation and energy balance: Review. *Theor. Appl. Clim.* 54: 27–37.

Hoskinson, R.L. et al. 2007. Engineering, nutrient removal, and feedstock conversion evaluations of four corn stover harvest scenarios. *Biom. Bioenergy* 31: 126–136.

Huggins, D.R. et al. 2007. Corn-soybean sequence and tillage effects on soil carbon dynamics and storage. *Soil Sci. Soc. Am. J.* 71: 145–154.

Huggins, D.R. et al. 1998. Carbon dynamics in corn–soybean sequences as estimated from natural carbon-13 abundance. *Soil Sci. Soc. Am. J.* 62: 195–203.

Islam, M.N. and F.N. Ani. 2000. Techno-economics of rice husk pyrolysis, conversion with catalytic treatment to produce liquid fuel. *Bioresourc. Technol.* 73: 67–75.

Jalota, S.K., R. Khera, and S.S. Chahal. 2001. Straw management and tillage effects on soil water storage under field conditions. *Soil Use Mgt.* 17: 282–287.

Jastrow, J.D. 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. *Soil Biol. Biochem.* 28: 656–676.

Jastrow, J.D. and R.M. Miller. 1997. Soil aggregate stabilization and carbon sequestration: Feedbacks through organomineral associations. In Lal, R. et al., Eds., *Soil Processes and the Carbon Cycle*. CRC Press, Boca Raton, FL, pp. 207–223.

Jawson, M.D. and L.F. Elliott. 1986. Carbon and nitrogen transformation wheat straw and root decomposition. *Soil Biol. Biochem.* 18: 15–22.

Jiang, P. and K.D. Thelen. 2004. Effect of soil and topographic properties on crop yield in a north central corn–soybean cropping system. *Agron. J.* 96: 252–258.

Johnson, J.M.F., R.R. Allmaras, and D.C. Reicosky. 2006a. Estimating source carbon from crop residues, roots and rhizodeposits using the national grain-yield database. *Agron. J.* 98: 622–636.

Johnson, J.M.F. et al. 2006b. A matter of balance: Conservation and renewable energy. *J. Soil Water Conserv.* 61: 120A–125A.

Johnson, J.M.F., N.W. Barbour, and S.L. Weyers. 2007a. Chemical composition of crop biomass impacts its decomposition. *Soil Sci. Soc. Am. J.* 71: 155–162.

Johnson, J.M.F. et al. 2007b. Agricultural opportunities to mitigate greenhouse gas emissions. *Environ. Pollut.* 150: 107–124.

Johnson, J.M.F. et al. 2007c. Impact of high lignin fermentation by-product on soils with contrasting soil organic carbon. *Soil Sci. Soc. Am. J.* 71: 1151–1159.

Johnson, J.M.F. et al. 2007d. Biomass-bioenergy crops in the United States: Changing paradigm. *Am. J. Plant Sci. Biotechnol.* 1: 1–28.

Johnson, J.M.F. et al. 2004. Characterization of soil amended with the by-product of corn stover fermentation. *Soil Sci. Soc. Am. J.* 68: 139–147.

Johnson, P.A. and B.J. Chamber. 1996. Effects of husbandry on soil organic matter. *Soil Use Mgt.* 13: 102–103.

Kaewpradit, W. et al. 2008. Regulating mineral N release and greenhouse gas emissions by mixing groundnut residues and rice straw under field conditions. *Eur. J. Soil Sci.* 59: 640–652.

Kapkiyai, J.J. et al. 1999. Soil organic matter and nutrient dynamics in a Kenyan nitisol under long-term fertilizer and organic input management. *Soil Biol. Biochem.* 31: 1773–1782.

Karlen, D.L. et al. 1994. Crop residue effects on soil quality following 10 years of no-till corn. *Soil Tillage Res.* 31: 149–167.

Kato, Y. et al. 2007. Improvement of rice (*Oryza sativa* L.) growth in upland conditions with deep tillage and mulch. *Soil Tillage Res.* 92: 30–44.

Katterer, T., O. Andren, and R. Pettersson. 1998. Growth and nitrogen dynamics of reed canary grass (*Phalaris arundinacea* L.) subjected to daily fertilization and irrigation in the field. *Field Crops Res.* 55: 153–164.

Kim, S. and B.E. Dale. 2005. Life cycle assessment of various cropping systems utilized for producing biofuels: Bioethanol and biodiesel. *Biom. Bioenergy* 29: 426–439.

Kladivko, E.J. 2001. Tillage systems and soil ecology. *Soil Tillage Res.* 61: 61–76.

Kong, A.Y.Y. et al. 2005. The relationship between carbon input, aggregation, and soil organic carbon stabilization in sustainable cropping systems. *Soil Sci. Soc. Am. J.* 69: 1078–1085.

Krzic, M. et al. 2004. Soil properties influencing compactability of forest soils in British Columbia. *Can. J. Soil Sci.* 84: 219–226.

Kucharik, C.J. et al. 2001. Measurements and modeling of carbon and nitrogen cycling in agroecosystems of southern Wisconsin: Potential for SOC sequestration during the next 50 years. *Ecosystems* 4: 237–258.

Kumar, K. and K.M. Goh. 2000. Crop residues and management practices: effects on soil quality, soil nitrogen dynamics, crop yield and nitrogen recovery. *Adv. Agron.* 68: 197–219.

Kundu, S. et al. 2007. Carbon sequestration and relationship between carbon addition and storage under rainfed soybean–wheat rotation in a sandy loam soil of the Indian Himalayas. *Soil Tillage Res.* 92: 87–95.

Kushwaha, C.P., S.K. Tripathi, and K.P. Singh. 2000. Variations in soil microbial biomass and N availability due to residue and tillage management in a dryland rice agroecosystem. *Soil Tillage Res.* 56: 153–166.

Lal, R., P. Henderlong, and M. Flowers. 1998. Forages and row cropping effects on soil organic carbon and nitrogen contents. In Lal, R. et al., Eds., *Management of C Sequestration in Soil*. CRC Press, Boca Roca, FL, pp. 365–379.

Larson, W.E. et al. 1972. Effects of increasing amounts of organic residues on continuous corn II. Organic carbon, nitrogen, phosphorus and sulfur. *Agron. J.* 64: 204–208.

Lehmann, J. and M. Rondon. 2006. Biochar soil management on highly weathered soils in the humid tropics. In Uphoff, N. et al., Eds., *Biological Approaches to Sustainable Soil Systems*. CRC Press, Boca Raton , FL, pp. 517–530.

Lehmann, J., J. Gaunt, and M. Rondon. 2006. Biochar sequestration in terrestrial ecosystems: Review. *Mitigation Adapt. Strat. Glob. Clim. Change* 11: 403–427.

Lehmann, J. et al. 2003. Nutrient availability and leaching in an archaeological anthrosol and a aerralsol of the Central Amazon Basin: Fertilizer, manure and charcoal amendments. *Plant Soil* 249: 343–357.

Lemus, R. et al. 2008. Effects of nitrogen fertilization on biomass yield and quality in large fields of established switchgrass in southern Iowa, USA. *Biom. Bioenergy* 32: 1187–1194.

Lemus, R. et al. 2002. Biomass yield and quality of 20 switchgrass populations in southern Iowa, USA. *Biom. Bioenergy* 23: 433–442.

Li, H. et al. 2007. Effects of 15 years of conservation tillage on soil structure and productivity of wheat cultivation in northern China. *Aust. J. Soil Res.* 45: 344–350.

Limon-Ortega, A. et al. 2006. Soil aggregate and microbial biomass in a permanent bed wheat–maize planting system after 12 years. *Field Crops Res.* 97: 302–309.

Limon-Ortega, A. et al. 2002. Soil attributes in a furrow-irrigated bed planting system in northwest Mexico. *Soil Tillage Res.* 63: 123–132.

Linden, D.R., C.E. Clapp, and R.H. Dowdy. 2000. Long-term corn grain and stover yields as a function of tillage and residue removed in east central Minnesota. *Soil Tillage Res.* 56: 167–174.

Lindstrom, M.J. 1986. Effects of residue harvesting on water runoff, soil erosion and nutrient loss. *Agric. Ecosyst. Environ.* 16: 103–112.

Linquist, B.A., V. Phengsouvanna, and P. Sengxue. 2007. Benefits of organic residues and chemical fertilizer to productivity of rain-fed lowland rice and to soil nutrient balances. *Nutr. Cycling Agroecosyst.* 79: 59–72.

Liu, W.D. et al. 2004. Response of corn grain yield to spatial and temporal variability in emergence *Crop Sci.* 44: 847–854.

Lobell, D.B. and G.P. Asner. 2002. Moisture effects on soil reflectance. *Soil Sci. Soc. Am. J.* 66: 722–727.

Low, A.J. 1972. The effect of cultivation on the structure and other physical characteristics of grassland and arable soils. *J. Soil Sci.* 23: 363–380.

Lynch, J. and E. Bragg. 1985. Microorganisms and soil aggregate stability. *Adv. Soil Sci.* 2: 133–171.

Ma, Z., C.W. Wood, and D.I. Bransby. 2000. Impacts of soil management on root characteristics of switchgrass. *Biom. Bioenergy* 18: 105–112.

Madakadze, I.C. et al. 1999. Cutting frequency and nitrogen fertilization effects on yield and nitrogen concentration of switchgrass in a short season area. *Crop Sci.* 39: 552–557.

Magleby, R. 2003. Soil Management and Conservation. USDA-ARS. http://www.ers.usda.gov/publications/arei/ah722/dbgen.htm (verified June 25, 2008).

Maiorano, A. et al. 2008. Effects of maize residues on the *Fusarium* spp. infection and deoxynivalenol (DON) contamination of wheat grain. *Crop Prot.* 27: 182–188.

Major, D.J., F.J. Larney, and C.W. Lilndwall. 1990. Spectral reflectance characteristics of wheat residues. In *Tenth Annual International Geoscience and Remote Sensing Symposium: Remote Sensing for the Nineties*. College Park, MD, May 1990, pp. 603–607.

Malhi, S.S. and H.R. Kutcher. 2007. Small grains stubble burning and tillage effects on soil organic C and N, and aggregation in northeastern Saskatchewan. *Soil Tillage Res.* 94: 353–361.

Malhi, S.S. and R. Lemke. 2007. Tillage, crop residue and N fertilizer effects on crop yield, nutrient uptake, soil quality and nitrous oxide gas emissions in a second 4-year rotation cycle. *Soil Tillage Res.* 96: 269–283.

Malhi, S.S. et al. 2006. Tillage, nitrogen and crop residue effects on crop yield, nutrient uptake, soil quality, and greenhouse gas emissions. *Soil Tillage Res.* 90: 171–183.

Malik, R.K. et al. 2000. Use of cover crops in short rotation hardwood plantations to control erosion. *Biom. Bioenergy* 18: 479–487.

Manlay, R.J. et al. 2002. Carbon, nitrogen and phosphorus allocation in agro-ecosystems of a West African savanna III. Plant and soil components under continuous cultivation. *Agric. Ecosyst. Environ.* 88: 249–269.

Mann, L., V. Tolbert, and J. Cushman. 2002. Potential environmental effects of corn (*Zea mays* L.) stover removal with emphasis on soil organic matter and erosion. *Agric. Ecosyst. Environ.* 89: 149–166.

Markvart, T. and L. Castañer, Eds. 2003. *Practical Handbook of Photovoltaics: Fundamentals and Applications*. Elsevier, New York.

Martens, D.A. 2000. Plant residue biochemistry regulates soil carbon cycling and carbon sequestration. *Soil Biol. Biochem.* 32: 361–369.

McCalla, T.M. 1943. Microbiological studies of the effect of straw used as a mulch. *Trans. Kan. Acad. Sci.* 46: 52–56.

McCool, D.K. et al. 2006. Residue characteristics for wind and water erosion control. 14th International Soil Conservation Organization Conference, Marrakech, Morocco, May 2006, p. 416. http://www.tucson.ars.ag.gov/isco/index_files/Page416.htm (verified Dec. 4, 2008).

McVay, K.A. et al. 2006. Management effects on soil physical properties in long-term tillage studies in Kansas. *Soil Sci. Soc. Am. J.* 70: 434–438.

Merrill, S.D. et al. 2006. Soil coverage by residue as affected by ten crop species under no-till in the northern Great Plains. *J. Soil Water Conserv.* 61: 7–13.

Mikha, M.M. and C.W. Rice. 2004. Tillage and manure effects on soil and aggregate-associated carbon and nitrogen. *Soil Sci. Soc. Am. J.* 68: 809–816.

Miller, R.M. and J.D. Jastrow. 1990. Hierarchy of root and mycorrhizal fungal interactions with soil aggregation. *Soil Biol. Biochem.* 22: 570–584.

Mitchell, R., J. Webb, and R. Harrison. 2001. Crop residues can affect N leaching over at least two winters. *Eur. J. Agron.* 15: 17–29.

Montgomery, D.R. 2007. Soil erosion and agricultural sustainability. *Proc. Natl. Acad. Sci. USA* 104: 13268–13272.

Monti, A., N. Di Virgilio, and G. Venturi. 2008. Mineral composition and ash content of six major energy crops. *Biom. Bioenergy* 32: 216–223.

Mubarak, A.R. et al. 2002. Decomposition and nutrient release of maize stover and groundnut haulm under tropical field conditions of Malaysia. *Commun. Soil Sci. Plant Anal.* 33: 609–622.

Munawar, A. et al. 1990. Tillage and cover crop management for soil water conservation. *Agron. J.* 82:773-777.

Nafziger, E.D., P.R. Carter, and E.E. Graham. 1991. Response of corn to uneven emergence. *Crop Sci.* 31: 811–815.

Nearing, M.A., F.F. Pruski, and M.R. O'Neal. 2004. Expected climate change impacts on soi erosion rate: Review. *J. Soil Water Conserv.* 59: 43–50.
Nelson, R.G. 2002. Resource assessment and removal analysis for corn stover and wheat strav in the Eastern and Midwestern United States: Rainfall- and wind-induced soil erosior methodology. *Biom. Bioenergy* 22: 349–363.
Nicholson, F.A. et al. 1997. Effects of repeated straw incorporation on crop fertiliser nitrogen requirements, soil mineral nitrogen and nitrate leaching losses. *Soil Use Mgt.* 13: 136–142.
Novak, M.D., W. Chen, and M.A. Hares. 2000. Simulating the radiation distribution within a barley–straw mulch. *Agric. For. Meteorol.* 102: 173–186.
Nuutinen, V. 1992. Earthworm community response to tillage and residue management on different soil types in southern Finland. *Soil Tillage Res.* 23: 221–239.
O'Halloran, I.P., M.H. Miller, and G. Arnold. 1986. Absorption of P by corn (*Zea mays* L.) as influenced by soil disturbance. *Can. J. Soil Sci.* 66: 287–302.
Oades, J.M. 1984. Soil organic matter and structural stability: Mechanisms and implications for management. *Plant Soil* 76: 319–337.
Parton, W.J. 1996. The CENTURY model. In Dowlson, D.S. et al., Eds., *Evaluation of Soil Organic Matter Models*. NATO ASI Series I, Vol. 38. Springer, Berlin, pp. 283–291.
Paul, E.A. 1991. Decomposition of organic matter. In Lederburg, J., Ed., *Encyclopedia of Microbiology*, Vol. 3. Academic Press, San Diego, CA, pp. 289–304.
Paustian, K., H.P. Collins, and E.A. Paul. 1997. Management controls on soil carbon. In Paul, E.A. et al., Eds., *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*. CRC Press, Boca Raton, FL, pp. 15–49.
Paustian, K., W.J. Parton, and J. Persson. 1992. Modeling soil organic matter in organic-amended and nitrogen-fertilized long-term plots. *Soil Sci. Soc. Am. J.* 56: 476–488.
Perlack, R.D. et al. 2005. Biomass as feedstock for a bioenergy and bioproducts industry: Technical feasibility of a billion-ton annual supply. U.S. Department of Energy and U.S. Department of Agriculture. http://www.eere.energy.gov/biomass/pdfs/final_billionton_vision_report2.pdf (posted July 15, 2005; verified Sept. 4, 2008).
Pikul, J.L.J. et al. 2008. Change in surface soil carbon under rotated corn in eastern South Dakota. *Soil Sci. Soc. Am. J.* 72: 1738–1744.
Power, J.F., W.W. Wilhelm, and J.W. Doran. 1986. Crop residue effects on soil environment and dryland maize and soybean production. *Soil Tillage Res.* 8: 101–0111.
Power, J.F. et al. 1998. Residual effects of crop residues on grain production and selected soil properties. *Soil Sci. Soc. Am. J.* 62: 1393–1397.
Pratt, A.W. 1969. Heat transmission in low conductivity materials. In Tye, R.P., Ed., *Thermal Conductivity*, Vol. 1. Academic Press, New York, pp. 301–405.
Rachman, A. et al. 2004. Soil hydraulic properties influenced by stiff-stemmed grass hedge systems. *Soil Sci. Soc. Am. J.* 68: 1386–1393.
Rao, S.C., H.S. Mayeux, and B.K. Northup. 2005. Performance of forage soybean in the southern Great Plains. *Crop Sci.* 45: 1973–1977.
Rasmussen, P.E. et al. 1980. Crop residue influences on soil carbon and nitrogen in a wheat-fallow system. *Soil Sci. Soc. Am. J.* 44: 596–600.
Rathore, T.R., B.P. Ghildyal, and R.S. Sachan. 1982. Germination and emergence of soybean under crusted soil conditions II. Seed environment and varietal differences. *Plant Soil* 65: 73–77.
Reicosky, D.C. et al. 2002. Continuous corn with moldboard tillage: Residue and fertility effects on soil carbon. *J. Soil Water Conserv.* 57: 277–284.
Reynolds, J.H., C.L. Walker, and M.J. Kirchner. 2000. Nitrogen removal in switchgrass biomass under two harvest systems. *Biom. Bioenergy* 19: 281–286.
Rhoton, F.E., M.J. Shipitalo, and D.L. Lindbo. 2002. Runoff and soil loss from midwestern and southeastern U.S. silt loam soils as affected by tillage practice and soil organic matter content. *Soil Tillage Res.* 66: 1–11.

Rickman, R. et al. 2002. Tillage, crop rotation, and organic amendment effect on changes in soil organic matter. *Environ. Pollut.* 116: 405–411.

Roldan, A. et al. 2003. No-tillage, crop residue additions, and legume cover cropping effects on soil quality characteristics under maize in Patzcuaro watershed. *Soil Tillage Res.* 72: 65–73.

Ross, P.J., J. Williams, and R.L. McCown. 1985. Soil temperature and the energy balance of vegetative mulch in the semi-arid tropics II. Dynamic analysis of the total energy balance. *Aust. J. Soil Res.* 23: 515–532.

Russel, J.C. 1940. The effect of surface cover on soil moisture losses by evaporation. *Soil Sci. Soc. Am. J.* 4: 65–70.

Saffigna, P.G. et al. 1989. Influence of sorghum residues and tillage on soil organic matter and soil microbial biomass in an Australian vertisol. *Soil Biol. Biochem.* 21: 759–765.

Sainju, U.M. et al. 2006. Carbon sequestration in dryland soils and plant residue as influenced by tillage and crop rotation. *J. Environ. Qual.* 35: 1341–1347.

Salinas Garcia, J.R. et al. 2001. Residue removal and tillage interaction effects on soil properties under rain-fed corn production in Central Mexico. *Soil Tillage Res.* 59: 67–79.

Sarr, P.S. et al. 2008. Effect of pearl millet–cowpea cropping systems on nitrogen recovery, nitrogen use efficiency and biological fixation using the ^{15}N tracer technique. *Soil Sci. Plant Nutr.* 54: 142–147.

Sauer, T.J. et al. 1998. Surface energy balance of a corn residue-covered field. *Agric. For. Meteorol.* 89: 155–168.

Shafii, B. and W. Price. 2001. Estimation of cardinal temperatures in germination data analysis. *J. Agric. Biol. Environ. Stat.* 6: 356–366.

Sharratt, B.S. 2002. Corn stubble height and residue placement in the northern U.S. Corn Belt I. Soil physical environment during winter. *Soil Tillage Res.* 64: 243–252.

Sharratt, B.S. and G.S. Campbell. 1994. Radiation balance of a soil-straw surface modified by straw color. *Agron. J.* 86: 200–203.

Sharratt, B.S., G.R. Benoit, and W.B. Voorhees. 1998. Winter soil microclimate altered by corn residue management in the northern Corn Belt of the USA. *Soil Tillage Res.* 49: 243–248.

Shen, Y.J. and C.B. Tanner. 1990. Radiative and conductive transport of heat through flail-chopped corn residue. *Soil Sci. Soc. Am. J.* 54: 653–658.

Singh, B. and S.S. Malhi. 2006. Response of soil physical properties to tillage and residue management on two soils in a cool temperate environment. *Soil Tillage Res.* 85: 143–153.

Singh, B. et al. 1994. Residue and tillage management effect on soil properties of a typic cryoboroll under continuous barley. *Soil Tillage Res.* 32: 117–133.

Six, J., E.T. Elliott, and K. Paustian. 1999. Aggregate and soil organic matter dynamics under conventional and no-tillage systems. *Soil Sci. Soc. Am. J.* 63: 1350–1358.

Six, J., E.T. Elliott, and K. Paustian. 2000a. Soil macroaggregate turnover and microaggregate formation: Mechanism for C sequestration under no-tillage management. *Soil Biol. Biochem.* 32: 2099–2103.

Six, J. et al. 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Sci. Soc. Am. J.* 62: 1367–1377.

Six, J. et al. 2000b. Soil structure and organic matter I. Distribution of aggregate-size classes and aggregate-associated carbon. *Soil Sci. Soc. Am. J.* 64: 681–689.

Skidmore, E.L. 1988. Wind erosion. In Lal, R., Ed., *Soil Erosion Research Methods*. Soil Water Conservation Society, Ankeny, IA, pp. 203–233.

Smika, D.E. and P.W. Unger. 1986. Effect of surface residues on soil water storage. *Adv. Soil Sci.* 5: 111–138.

Smiley, R.W., H.P. Collins, and P.E. Rasmussen. 1996. Diseases of wheat in long-term agronomic experiments at Pendleton, Oregon. *Plant Dis.* 80: 813–820.

Soane, B.D. 1990. The role of organic matter in soil compactibility: Review of some practical aspects. *Soil Tillage Res.* 16: 179–201.

Spedding, T.A. et al. 2004. Soil microbial dynamics in maize-growing soil under different tillage and residue management systems. *Soil Biol. Biochem.* 36: 499–512.

Stanhill, G., G.J. Hofstede, and J.D. Kalma. 1966. Radiation balance of natural agricultural vegetation. *Q. J. Roy. Meteorol. Soc.* 92.

Steiner, J. 1994. Crop residue effects on water conservation. In Unger, P.W., Ed., *Managing Agricultural Residues.* CRC Press, Boca Raton, FL, pp. 41–76.

Steiner, J.L. et al. 2000. Biomass and residue cover relationships of fresh and decomposing small grain residue. *Soil Sci. Soc. Am. J.* 64: 2109–2114.

Stevenson, F.J. 1994. *Humic Chemistry: Genesis, Composition, Reactions*, 2nd ed. John Wiley & Sons, New York.

Stocking, M.A. 1988. Assessing vegetative cover and management effects. In Lal, R., Ed., *Soil Erosion Research Methods.* Soil Water Conservation Society, Ankeny, IA, pp. 211–234.

Swan, J.B. et al. 1994. Surface residue and in-row treatment on long-term no-tillage continuous corn. *Agron. J.* 86: 711–718.

Tanner, C.B. and Y. Shen. 1990. Solar-radiation transmittance of flail-chopped corn residue layers. *Soil Sci. Soc. Am. J.* 54: 650–652.

Tian, G., B.T. Kang, and L. Brussand. 1992. Biological effects of plant residues with contrasting chemical composition under humid tropical conditions: Decomposition and nutrient release. *Soil Biol. Biochem.* 24: 1051–1060.

Tirol-Padre, A., K. Tsuchiya, K. Inubushi, and J.K. Ladha. 2005. Enhancing soil quality through residue management in a rice–wheat system in Fukuoka, Japan. *Soil Sci. Plant Nutr.* 51: 849–860.

Tisdall, J.M. 1994. Possible role of soil microorganisms in aggregation in soil. *Plant Soil* 159: 115–121.

Tisdall, J.M. 1996. Formation of soil aggregates and accumulation of soil organic matter. In Carter, M.R. and B.A. Stewart, Eds., *Structure and Organic Matter Storage in Agricultural Soils.* CRC Press, Boca Raton, FL, pp. 57–96.

Tisdall, J.M. and J.M. Oades. 1982. Organic matter and water-stable aggregates in soils. *J. Soil Sci.* 33: 141–163.

Tomer, M.D., D.W. Meek, and L.A. Kramer. 2005. Agricultural practices influence flow regimes of headwater streams in western Iowa. *J. Environ. Qual.* 34: 1547–1558.

Tomer, M.D. et al. 2006. Surface soil properties and water contents across two watersheds with contrasting tillage histories. *Soil Sci. Soc. Am. J.* 70: 620–630.

Uhlenbrook, S. 2007. Biofuel and water cycle dynamics: What are the related challenges for hydrological processes research? *Hydrol. Proc.* 21: 3647–3650.

Updegraffa, K., P. Gowdaa, and D.J. Mullaa. 2004. Watershed-scale modeling of the water quality effects of cropland conversion to short-rotation woody crops. *Renew. Agricult. Food Syst.* 19: 118–127.

USDA NRCS. 1997. Tolerable Erosion Levels. USDA-NRCS. http://www.mn.nrcs.usda.gov/technical/nri/findings/erosion_t.htm (verified Dec. 1, 2008).

van Bavel, C.H.M. and D.I. Hillel. 1976. Calculating potential and actual evaporation from a bare soil surface by simulation of concurrent flow of water and heat. *Agric. Meteorol.* 17: 453–476.

Van Donk, S.J. and E.L. Skidmore. 2003. Measurement and simulation of wind erosion, roughness degradation and residue decomposition on an agricultural field. *Earth Surf. Proc. Landforms* 28: 1243–1258.

Van Veen, J.A. and P.J. Kuikman. 1990. Soil structural aspects of decomposition of organic matter by microorganisms. *Biogeochemistry* 11: 213–223.

Van Veen, J.A. and E.A. Paul. 1981. Organic carbon dynamics in grassland soils 1: Background information and computer simulation. *Can. J. Soil Sci.* 61: 185–201.

Vanotti, M.B., L.G. Bundy, and A.E. Peterson. 1997. Nitrogen fertilizer and legume–cereal rotation effects on soil productivity and organic matter dynamics in Wisconsin. In Paul, E.A. et al., Eds., *Soil Organic Matter in Temperate Agroecosystems*. CRC Press, Boca Raton, FL, pp. 105–119.

Varvel, G.E., and W.W. Wilhelm. 2008. Soil carbon levels in irrigated Western Corn Belt rotations. *Agron. J.* 100: 1180–1184.

Velthof, G.L., P.J. Kuikman, and O. Oenema. 2002. Nitrous oxide emission from soils amended with crop residues. *Nutr. Cycl. Agroecosys.* 62: 249–261.

Vitosh, M.L., R.E. Lucas, and G.H. Silva. 1997. Long-term effects of fertilizer and manure on corn yield, soil carbon, and other soil chemical properties in Michigan. In Paul, E.A. et al., Eds., *Soil Organic Matter in Temperate Egroecosystems*. CRC Press, Boca Raton, FL, pp. 129–139.

Vogel, K.P. et al. 2002. Switchgrass biomass production in the midwest USA: Harvest and nitrogen management. *Agron. J.* 94: 413–420.

Wagner-Riddle, C. et al. 2008. Linking nitrous oxide flux during spring thaw to nitrate denitrification in the soil profile. *Soil Sci. Soc. Am. J.* 72: 908–916.

Wang, W.J. et al. 2004. Decomposition dynamics of plant materials in relation to nitrogen availability and biochemistry determined by NMR and wet-chemical analysis. *Soil Biol. Biochem.* 36: 2045–2058.

Wells, R.R. et al. 2003. Infiltration and surface geometry features of a swelling soil following successive simulated rainstorms. *Soil Sci. Soc. Am. J.* 67: 1344–1351.

Wieder, R.K. and G.E. Lang. 1982. A critique of analytical methods used in examining decomposition data obtained from litter bags. *Ecology* 63: 1636–1642.

Wilhelm, W.W., J.W. Doran, and J.F. Power. 1986. Corn and soybean yield response to crop management under no-tillage production systems. *Agron. J.* 7: 184–189.

Wilhelm, W.W., J.M.F. Johnson, D.L. Karlen, and D.T. Lightle. 2007. Corn stover to sustain soil organic carbon further constrains biomass supply. *Agron. J.* 99: 1665–1667.

Wilhelm, W.W. et al. 2004. Crop and soil productivity response to corn residue removal: A literature review. *Agron. J.* 96: 1–17.

Wilson, G.V., K.C. McGregor, and D. Boykin. 2008. Residue impacts on runoff and soil erosion for different corn plant populations. *Soil Tillage Res.* 99: 300–307.

Wilts, A.R., D.C. Reicosky, R.R. Allmaras, and C.E. Clapp. 2004. Long-term corn residue effects: Harvest alternatives, soil carbon turnover, and root-derived carbon. *Soil Sci. Soc. Am. J.* 68: 1342–1351.

Wuest, S.B. et al. 2005. Organic matter addition, N, and residue burning effects on infiltration, biological, and physical properties of an intensively tilled silt-loam soil. *Soil Tillage Res.* 84: 154–167.

Yaman, S. 2004. Pyrolysis of biomass to produce fuels and chemical feedstocks. *Energy Conv. Mgt.* 45: 651–671.

Yang, H.S., D.J. Kim, and H.J. Kim. 2003. Rice straw–wood particle composite for sound absorbing wooden construction materials. *Bioresour. Technol.* 86: 117–121.

Ying, J. et al. 1998. Comparison of high-yield rice in tropical and subtropical environments II. Nitrogen accumulation and utilization efficiency. *Field Crops Res.* 57: 85–93.

Yu, F., R. Ruan, and P. Steele. 2008. Consecutive reaction model for the pyrolysis of corn cob. *Trans. ASABE* 51: 1023–1028.

Zan, C.S. et al. 2001. Carbon sequestration in perennial bioenergy, annual corn and uncultivated systems in southern Quebec. *Agric. Ecosyst. Environ.* 86: 135–144.

Zemenchik, R.A. et al. 2000. Corn production with Kura clover as a living mulch. *Agron. J* 92: 698–705.

Zhang, G.S. et al. 2007. Relationship between soil structure and runoff/soil loss after 24 years of conservation tillage. *Soil Tillage Res.* 92: 122–128.