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Mitigation of Greenhouse Gases in Agricultural Ecosystems

Ilya Gelfand and G. Philip Robertson

Modern cropping systems use substantial amounts of fossil energy in the form of fertilizers, pesticides, and fuel for field operations. An important environmental consequence of this use is the emission of greenhouse gases (GHGs) to the atmosphere, from sources both direct and indirect. Direct sources include fossil fuel used for tillage and other field operations as well as GHGs produced and consumed by microbes in cropped soils. Indirect sources include fossil energy used off-site to produce fertilizers and other agronomic inputs, as well as GHGs produced by microbes in noncropped sites that receive nutrients escaped from cropped fields. Row-crop agriculture can thus be either a net source or sink of GHGs, with the balance (net emission or uptake) influenced greatly by management practices.

All three of the major biogenic GHGs are affected by agriculture: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Not including postharvest activities or land-use conversion caused by agricultural expansion, agriculture is responsible for 10–14% of total global anthropogenic GHG emissions (Barker et al. 2007, Smith et al. 2007). This includes ~84% of anthropogenic N_2O emissions and ~53% of anthropogenic CH_4 emissions (Robertson 2004). The manufacture of agrochemicals adds another 0.6–1.5% to the global total (Vermeulen et al. 2012).

Most agricultural CO_2 emissions are from land conversion and fossil fuel use. Methane emissions associated with agriculture are from enteric fermentation by ruminant animals such as cattle, cultivated rice soils, animal wastes, and agricultural biomass burning. In addition, land conversion to agriculture substantially reduces microbial CH_4 oxidation in soil, thereby attenuating an important CH_4 sink and effectively increasing CH_4 in the atmosphere. Nitrous oxide emissions from agriculture are produced mostly from nitrogenous fertilizers, with lesser contributions from animal wastes and biomass burning.

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The global importance of GHG fluxes from established cropping systems and their sensitivity to management make agriculture an attractive sector for mitigation measures. And because many of these fluxes are interdependent and sensitive to the same management practices (though often differentially sensitive), there are many opportunities to manage them together. In fact, because management practices

Table 12.1. Description of the KBS LTER Main Cropping System Experiment (MCSE).[a]

Cropping System/Community	Dominant Growth Form	Management
Annual Cropping Systems		
Conventional (T1)	Herbaceous annual	Prevailing norm for tilled corn–soybean–winter wheat (c–s–w) rotation; standard chemical inputs, chisel-plowed, no cover crops, no manure or compost
No-till (T2)	Herbaceous annual	Prevailing norm for no-till c–s–w rotation; standard chemical inputs, permanent no-till, no cover crops, no manure or compost
Reduced Input (T3)	Herbaceous annual	Biologically based c–s–w rotation managed to reduce synthetic chemical inputs; chisel-plowed, winter cover crop of red clover or annual rye, no manure or compost
Biologically Based (T4)	Herbaceous annual	Biologically based c–s–w rotation managed without synthetic chemical inputs; chisel-plowed, mechanical weed control, winter cover crop of red clover or annual rye, no manure or compost; certified organic
Perennial Cropping Systems		
Alfalfa (T6)	Herbaceous perennial	5- to 6-year rotation with winter wheat as a 1-year break crop
Poplar (T5)	Woody perennial	Hybrid poplar trees on a ca. 10-year harvest cycle, either replanted or coppiced after harvest
Coniferous Forest (CF)	Woody perennial	Planted conifers periodically thinned
Successional and Reference Communities		
Early Successional (T7)	Herbaceous perennial	Historically tilled cropland abandoned in 1988; unmanaged but for annual spring burn to control woody species
Mown Grassland (never tilled) (T8)	Herbaceous perennial	Cleared woodlot (late 1950s) never tilled, unmanaged but for annual fall mowing to control woody species
Mid-successional (SF)	Herbaceous annual + woody perennial	Historically tilled cropland abandoned ca. 1955; unmanaged, with regrowth in transition to forest
Deciduous Forest (DF)	Woody perennial	Late successional native forest never cleared (two sites) or logged once ca. 1900 (one site); unmanaged

[a]Site codes that have been used throughout the project's history are given in parentheses. Systems T1–T7 are replicated within the LTER main site; others are replicated in the surrounding landscape. For further details, see Robertson and Hamilton (2015, Chapter 1 in this volume).

produce different effects on GHG fluxes, it is especially important to consider them together, that is, to take a systems approach toward their understanding and management (Robertson 2014).

In this chapter, we describe an ecosystems approach to documenting changes in GHG fluxes in intensive row-crop agriculture. We draw, in particular, on results from the Kellogg Biological Station Long-Term Ecological Research site (KBS LTER), where GHG fluxes have been studied in the Main Cropping System Experiment (MCSE; Table 12.1; Robertson and Hamilton 2015, Chapter 1 in this volume) since 1989. We discuss the value of long-term comparisons of different cropping systems in determining the potential for management practices to contribute to or mitigate GHG fluxes. We end with consideration of the GHG implications of crop production not only for grain but also for cellulosic biomass, which is anticipated to become increasingly important in a future that includes cellulosic biofuels.

Row-Crop Agriculture and GHG Mitigation

Historically, agricultural impacts on atmospheric chemistry have been dominated by land-use change. Since the late eighteenth century, conversion of forests and grasslands to cropland has resulted in emissions of CO_2 to the atmosphere on the order of 130 to 170 Pg C (Wilson 1978, Sauerbeck 2001), mostly due to immediate biomass burning and subsequent soil carbon (C) oxidation. Global CO_2 emissions from deforestation today amount to ~1.5 Pg C yr^{-1} (Canadell et al. 2007).

In few established croplands today are GHG emissions dominated by soil C oxidation. Rather, emissions now are dominated by CO_2 from fossil fuel combustion during farm operations; CO_2 produced during the manufacture and transport of fertilizers, pesticides, and other agricultural inputs; N_2O emitted when nitrogen (N) fertilizers are applied to soil; and CH_4 emitted during flooded conditions in lowland rice. In most of the world's established agricultural soils (except drained wetlands), soil C is either stable or, if managed appropriately, increasing, though this trend could be reversed by a warming climate (Senthilkumar et al. 2009; Paul et al. 2015, Chapter 5 in this volume).

The need for mitigation of agricultural GHG emissions becomes especially important in light of the agricultural intensification yet required to feed an increasing and more affluent world population (Tilman et al. 2011, Mueller et al. 2012). Although intensification to date has improved yields on existing farmland and thereby fed more people at a lower per-capita GHG cost (i.e., at a lower GHG cost per unit yield) (Burney et al. 2010), the efficiency gained has not been sufficient to halt the increase in GHG emissions from agriculture.

Growing demands for biofuel feedstocks could further increase agriculture's GHG footprint: over the next several decades, millions of hectares will likely be converted to biofuel cropping systems that will consume fuel and fertilizer and could—if not carefully managed—exacerbate rather than alleviate atmospheric GHG loading (Melillo et al. 2009). Bioenergy cropping systems correctly

implemented, on the other hand, provide a substantial opportunity for mitigating anthropogenic GHG contributions as well as providing other environmental benefits (Robertson et al. 2008, NRC 2009, Tilman et al. 2009).

In light of the expectation for worldwide expansion and intensification of agriculture in the coming decades, it seems crucial to pursue opportunities for reducing the GHG contributions of agricultural crop production. Many such opportunities are available, particularly in the areas of soil C conservation (CAST 2011) and better N management (Robertson and Vitousek 2009). Through the strategic adoption of agronomic practices known to attenuate GHG emissions (e.g., Millar et al. 2010), agriculture could contribute significantly to climate change mitigation.

Long-term research such as that conducted at the KBS LTER has a particularly important contribution to make in climate change mitigation because of the variable nature and slow rate of change for many agricultural GHG fluxes. While some emissions are sudden, such as biomass burning during land clearing, and others are episodic but easily quantified, such as fuel used during agronomic operations, others can be difficult to reliably estimate based on short-term observations because they change very slowly or are temporally variable. Changes in soil C sequestration, for example, are normally too gradual to detect on an annual basis: a change of 50 g C m^{-2} (a typical annual gain after conversion to no-till management) cannot be distinguished in 1 year against a spatially variable background pool of 5000 g C m^{-2}. Long-term research provides the time necessary to document such changes; detecting an increase of 500 g C m^{-2} over 10 years is much more tractable (Kravchenko and Robertson 2011).

Similarly, changes in N_2O emissions are difficult to detect against a background of high temporal variability. Nitrous oxide emissions from soils are notoriously variable and unpredictable: fluxes can change an order of magnitude within a single day (e.g., Ambus and Robertson 1998, Barton et al. 2008) in response to a variety of environmental drivers. Long-term N_2O research provides the large set of measurements and hence the statistical power needed to assess differences among agronomic systems and practices against an otherwise confusing backdrop of short-term variability.

Providing a Common Basis for Systemwide Comparisons

The Concept of Global Warming Impact (GWI)

Greenhouse gases vary greatly in radiative forcing and residence time in the atmosphere, so it is not enough to know that one system stores more soil C but liberates more N_2O than another system that oxidizes more CH_4: a reference is needed to appropriately weight the effect of different gases on the atmosphere's capacity to hold heat. The Global Warming Potential (GWP; IPCC 2001) index satisfies this need. The GWP is a combined measure of the radiative forcing of a given GHG based on its physical capacity to absorb infrared radiation, its current concentration in the atmosphere, and its atmospheric lifetime.

By convention, CO_2 has a GWP of 1; the GWPs of all other gases are expressed relative to this. Because GHGs have different atmospheric lifetimes, their GWPs change differentially after emission—for example, the GWP for a quantity of N_2O emitted today is higher than it will be a century from now, when less of it will remain in the atmosphere. Methane, with its briefer atmospheric lifetime (~12 years vs. 114 years for N_2O), will have a correspondingly smaller impact a century after emission. To provide a common means for comparison, the IPCC has identified 100 years as an appropriate standard time horizon for comparing mitigation options (Forster et al. 2007). Methane has a 100-year GWP of 23 and N_2O, 298. Manufactured halocarbons with atmospheric lifetimes of millennia can have 100-year GWPs greater than 10,000 (Prinn 2004).

We use the term GWI to refer to the effect of a given activity or group of activities on the atmosphere's heat-trapping capacity. Both GWP and GWI are measured in CO_2 equivalents (CO_2e). By way of example, a cropping practice that releases 1 g m^{-2} of CO_2 has a GWI of 1 g CO_2e m^{-2}, and a practice that releases 1 g m^{-2} of N_2O has a GWI of 298 g CO_2e m^{-2}; the GWI of both practices combined would be 299 g CO_2e m^{-2}. Thus, management practices that affect N_2O fluxes can disproportionately influence climate forcing relative to practices that affect fluxes of CO_2.

GWI in Practice

The literature is rich with estimates for GWIs of individual cropping activities. These include the effects of tillage on soil C sequestration (e.g., no-till management increases soil organic C; Paul et al. 2015, Chapter 5 in this volume); the amount of CO_2 emitted by the manufacture, transport, and application of agrochemicals; and the amount of N_2O emitted from fertilized fields as a function of the rate, timing, and formulation of N fertilizer (Millar and Robertson 2015, Chapter 9 in this volume). Still rare, however, are full-cost accountings of entire cropping systems or farms, in which GWIs from all significant sources are tallied to provide a systemwide net GWI.

Cropping systems with a net positive GWI are net emitters of GHGs and thus drivers of anthropogenic climate change, whereas systems with a net negative GWI mitigate climate change. Important to realize, however, is that any system or practice with a GWI lower than that which is currently the norm will represent mitigation relative to business as usual—even if the GWI of the new system or practice remains positive. Equally important is the notion that only by placing GWIs for different practices in an ecosystem context can the net benefits of any change be assessed. No-till practices, for example, will save fuel and store more soil C relative to conventional tillage, but the need for additional herbicide use has a C cost that will offset some of the fuel savings and soil C gain, and in some soils no-till practices may increase N_2O emissions (van Kessel et al. 2013).

Results from a full-cost analysis of GWI in the MCSE (Table 12.2) illustrate both tradeoffs and synergies. In one of the first whole-system analyses of the contribution of different GHGs to agriculture's GWI, Robertson et al. (2000) showed that the GWI of MCSE cropping systems differed markedly—and for different reasons.

Table 12.2. Global Warming Impacts for the first decade (1989–1999) of the MCSE.[a]

System	Global Warming Impact (GWI)[b] (g CO_2e m^{-2} yr^{-1})						
	Soil C	N Fertilizer	Lime	Fuel	N_2O	CH_4	Net GWI
Annual Crops (corn–soybean–wheat rotation)							
Conventional	0	27	23	16	52	–4	114
No-till	–110	27	34	12	56	–5	14
Reduced Input	–40	9	19	20	60	–5	63
Biologically Based	–29	0	0	19	56	–5	41
Perennial Crops							
Alfalfa	–161	0	80	8	59	–6	–20
Poplar	–117	5	0	2	10	–5	–105
Successional Communities							
Early Successional	–220	0	0	0	15	–6	–211
Mid-successional	–32	0	0	0	16	–15	–31
Mown Grassland (never tilled)	0	0	0	0	18	–17	1
Deciduous Forest	0	0	0	0	21	–25	–4

[a]See Table 12.1 for a description of systems. All systems are replicated (n = 3–6).
[b]Net GWI is determined as the sum of GWI components: soil carbon (C) sequestration, agronomic inputs of nitrogen (N) fertilizer, lime and fuel, and GHG exchanges of nitrous oxide (N_2O) and methane (CH_4) with the atmosphere. Units are carbon dioxide equivalents (CO_2e; g m^{-1} yr^{-1}) based on IPCC conversion factors (IPCC 2007). Negative values indicate net climate change mitigation potential.
Source: Robertson et al. (2000).

Net GWIs over a 9-year period (Table 12.2) ranged from 114 g CO_2e m^{-2} yr^{-1} (net emission) in the conventionally managed corn–soybean–wheat rotation to –211 g CO_2e m^{-2} yr^{-1} (net mitigation) in the Early Successional community abandoned from agriculture 9 years earlier. Net GWIs also differed substantially among the annual cropping systems: net GWI was low in the No-till system (14 g CO_2e m^{-2} yr^{-1}) and intermediate in the Reduced Input and Biologically Based systems (63 and 41 g CO_2e m^{-2} yr^{-1}, respectively), suggesting the potential for substantial mitigation relative to the Conventional management.

Close analysis shows the source of these differences. While in all the annual crops, N_2O production was the largest single source of GWI, in the No-till system soil C storage more than offset the GWI of N_2O emissions, although additional contributions from N fertilizer manufacture, lime (calcium and magnesium carbonate) application, and fuel use kept GWI in the No-till system positive (Table 12.2). And although not enough C was stored in the Reduced Input and Biologically Based systems to offset N_2O production, savings from lower N fertilizer and lime use helped to reduce their net GWI to about half that of the Conventional system.

The hybrid Poplar system's combination of low N_2O emissions and enhanced soil C accumulation over 9 years resulted in a substantial mitigation capacity of –105 g CO_2e m^{-2} yr^{-1} (Table 12.2). Although Alfalfa, the other perennial system

evaluated, also had substantial soil C accumulation, much of this was offset by agricultural lime applications and high N_2O emissions—both related to alfalfa's N fixation capacity. Nitrogen fixation provides inorganic N to nitrifying bacteria, which in turn provide nitrate (NO_3^-) to denitrifiers, and both nitrifiers and denitrifiers produce N_2O (Ostrom et al. 2010). Nitrifiers also produce acidity, increasing the need for lime application. As a result, alfalfa possessed only a modest net mitigation capacity in spite of high rates of soil C sequestration.

Results from the full-cost analysis of GWI in the MCSE (Table 12.2) also suggest an eventual diminution of the Early Successional community's strong mitigation potential. Older successional communities had a substantially higher net GWI (though still negative), primarily because of lower soil C accumulation (Table 12.2). For example, in the late successional Deciduous Forest net soil C accumulation was nil, and although CH_4 oxidation was significant, it was largely offset by N_2O emissions, leading to an overall GWI close to zero.

Gelfand et al. (2013) extended the GWI analysis of the MCSE by an additional decade, and although results showed similar trends, there were two important differences (Table 12.3). First, Hamilton et al. (2007) found that lime contributions to GWI are likely far less than calculated earlier due to how lime is dissolved in these soils. Dissolution by strong acids such as nitric (HNO_3) leads to immediate CO_2 release—as was assumed in the earlier analysis. However, dissolution by carbonic acid (H_2CO_3)—a weak acid existing in equilibrium with dissolved CO_2—leads to net CO_2 capture by the soil solution and its hydrologic export as bicarbonate (HCO_3^-), which resides in the groundwater system for long periods. Thus, the net GWI in KBS soils, where dissolution by the two reactions tends to occur in about equal proportions, is likely nil. And second, a more recent and deeper soil C sampling (Syswerda et al. 2011) showed that soil C sequestered by the hybrid Poplar system was largely lost during reestablishment after harvesting, when for ~2 years soils were warmer and moister as a result of greater insolation and reduced transpiration due to lack of canopy cover. These results revise but do not substantially alter the original study's conclusion that different cropping practices contribute differentially to a given cropping system's GWI, and they illustrate how a long-term systems approach is necessary to fully partition the benefits and liabilities of specific management systems.

Biofuel and Energy Flux Considerations

Neither Robertson et al. (2000) nor others (e.g., Mosier et al. 2004) considered the end use of the biomass produced by cropping systems in their calculations of GWI—all harvested biomass was assumed to be oxidized to CO_2, thereby providing no further mitigation capacity. If, on the other hand, harvested biomass is used for energy that would otherwise be provided by fossil fuel, then an additional mitigation credit must be added to the GWI of the cropping system that produced it, so long as additional GHGs are not produced elsewhere by land cleared to offset a potential loss of food production (Searchinger et al. 2008, 2009). For example, the MCSE Poplar system discussed above would gain an additional mitigation credit of ~319 g CO_2e m^{-2} yr^{-1} were those trees grown on previously unforested land not

Table 12.3. GWIs over two decades of the MCSE.[a]

System	Global Warming Impact (GWI, g CO_2e m^{-2} yr^{-1})										
	Soil C	N Fertilizer	Lime	P	K	Seed	Pest	Fuel	N_2O	CH_4	Net GWI Revised
Annual Crops (corn–soybean–wheat rotation)											
Conventional	0 (31)	33	3	0.4	1.3	7.3	7.1	13	37 (6)	–1 (0)	101 (32)
No-till	–122 (31)	33	4	0.3	1.3	7.0	15.5	9	39 (3)	–1 (0)	–14 (31)
Reduced Input	–92 (122)	11	2	0.2	1.0	7.9	4.3	21	35 (2)	–1 (0)	–11 (122)
Biologically Based	–183 (31)	0	0	0	0	7.9	0	20	32 (3)	–1 (0)	–124 (31)
Perennial Crops											
Alfalfa	–122 (92)	0	14	0.3	4	6	3	11	46 (4)	–1 (0)	–39 (92)
Poplar	61 (53)	3	0	0	0	0	2	1	6 (1)	–1 (0)	73 (53)
Successional Communities											
Early Successional	–397 (31)	0	0	0	0	0	0	0	11 (1)	–1 (0)	–387 (31)
Mid-successional	–214 (275)	0	0	0	0	0	0	0	16 (3)	–3 (1)	–201 (275)
Mown Grassland (never tilled)	0	0	0	0	0	0	0	0	11 (2)	–4 (0)	7 (2)
Deciduous Forest	0	0	0	0	0	0	0	0	12 (2)	–5 (1)	7 (2)

[a]See Table 12.2 for further explanation. Mean values (±SE) are based on data from 1989 to 2009, except mean values of N_2O and CH_4 are based on data from 1991 to 2011.
Source: Revised from data in Gelfand et al. (2013).

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now used for food crops and were the harvested biomass used to make biofuel that offsets fossil fuel use (Gelfand et al. 2013).

Energy balance can also provide a common basis for comparing the GWIs of different cropping systems because of the interconnection between GHG emissions and energy use (West and Marland 2002). An accurate estimate of agricultural energy efficiency—the ratio of useable energy in the end products to the energy used for production (Gelfand et al. 2010)—can, in addition to illustrating GWI differences, provide insights into how society can meet food and fuel security needs most energy efficiently. Energy efficiency can be calculated using energy balance tools (e.g., Kim and Dale 2003), and becomes especially useful when assessing the potential for bioenergy crops to offset fossil fuel use.

Table 12.4 shows the large range in annual energy inputs and food energy outputs for the MCSE annual cropping systems (Gelfand et al. 2010). While the Conventional system produced more than 10 times the energy in food than was used in farming (72.7 vs. 7.1 GJ ha^{-1} yr^{-1}), the No-till system produced even more energy (78.5 GJ ha^{-1} yr^{-1}) and at two-thirds of the energy input (4.9 GJ ha^{-1} yr^{-1}), for a net energy efficiency (energy output:input ratio) of 16, far higher than that of the Conventional (10). High energy costs of tillage account for most of the difference. Gelfand et al. (2010) also showed that the energy efficiency for food production was always higher than for liquid fuel production from the same crops, even when crop residues were to be used for fuel. However, this analysis assumes that food is produced for direct human consumption; allocating a portion of food crops to support livestock for meat and dairy production would change the energy balance because of the inherent inefficiency of energy transfer through food chains.

The Importance of System Boundaries for GWI Comparisons

A full accounting of the GWI or energy balance of an agricultural ecosystem requires a clear definition of the boundaries that meet the purpose and needs of the analysis. Inclusion of solar energy inputs, for example, would make fossil fuel

Table 12.4. Crop yields and energy balances for the annual cropping systems of the MCSE from 1989 to 2007.[a]

Annual Cropping System	Crop Yield (Mg ha^{-1} yr^{-1})			Crop Rotation Energy Balance (GJ ha^{-1} yr^{-1})		
	Corn	Wheat	Soybean	Farming Energy Inputs	Food Energy Output[b]	Net Energy Efficiency[c]
Conventional	5.9	3.5	2.3	7.1	72.7	10
No-till	6.3	3.7	2.7	4.9	78.5	16
Reduced Input	5.2	3.1	2.6	5.2	66.9	13
Biologically Based	4.1	2.1	2.4	4.8	53.1	11

[a]See Table 12.1 for a description of systems.
[b]Food produced for direct human consumption (i.e., not via livestock production for human consumption).
[c]Net Energy Efficiency calculated as output to input ratio.
Source: Gelfand et al. (2010).

energy inputs insignificant even though they are often the component of greatest interest. Energy in human labor and machinery manufacture (Hülsbergen et al. 2001) could similarly be included, but these inputs do not significantly differ among production-scale farming systems regardless of the final products (Pimentel and Patzek 2005). Thus, if one is calculating an energy balance to determine the most energy-efficient system, only significant sources of manageable energy need be included, that is, energy inputs that differ and are affected by various management options. Such comparisons assume that differences outside the farm gate are negligible, that is, that the energy costs of labor inputs, farm implements, and storing or transporting crop yields are identical or sufficiently similar to be an insignificant part of the overall system budget. This makes analyses more tractable, as measurements of fluxes and pools at the farm scale are relatively straightforward.

Thus, the choice of system boundaries should be explicit and based on the needs of the study. As for nutrient budgets or biogeochemical cycles (Robertson 1982), boundaries should be expanded only as far as necessary to encompass the fluxes relevant to the question under study. In a comparative analysis of biofuel cropping systems, for example, it make sense to expand the boundary to include the cost of transporting harvested grain and cellulosic biomass, as does inclusion of the fate of grain ethanol end-products such as dry distillers grain.

Components of GHG Balances in Cropping Systems

The primary purpose of an agricultural GHG balance is to track the exchanges of GHGs between cropping systems and the atmosphere. Figure 12.1 summarizes major fluxes between these two pools. The cropping system contains three main compartments: agricultural inputs that cost CO_2e to manufacture and transport, GHG production and consumption by soil microbes, and CO_2 captured by the cropping system and ultimately emitted in consumption of the harvested biomass. All three compartments are interrelated and influenced by management decisions.

The GWI of a given system can be studied using a mass-balance approach, which accounts for fluxes into and out of the system and provides estimates of change in the pool of interest—ultimately resulting in GHG exchanges (expressed as CO_2e) with the atmosphere:

$$\frac{dX}{dt} = Flux\,In\,X(t) - Flux\,Out\,X(t)$$

where X is the pool of interest, and *Flux In* and *Flux Out* are the sum of all measured and estimated fluxes into or out of the studied system over a given time period t. Although t is usually annualized, when processes involve different time scales, it is important that t be appropriately normalized, such as over the length of a rotation. A comparison of a 1-year continuous corn rotation to a 3-year corn–soybean–wheat rotation, for example, should be performed over at least one 3-year period to capture different crop effects, and preferably more in order to capture climatic variation. The same is true for other periodic management practices as well; for example, if

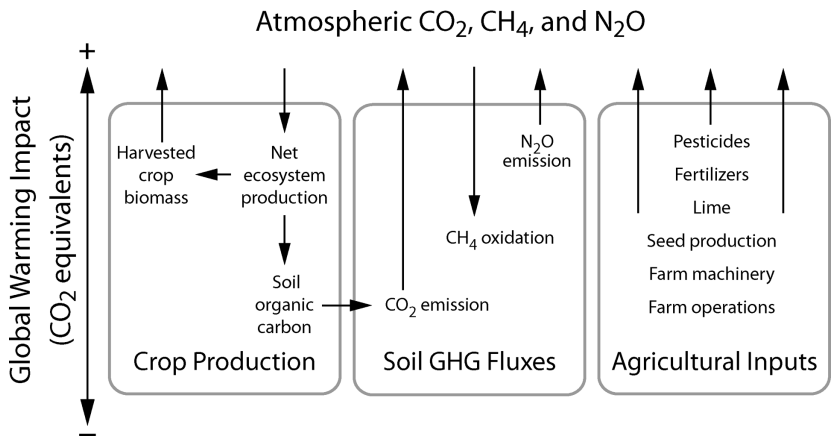


Figure 12.1. Conceptual diagram of Global Warming Impact (GWI) components in agricultural cropping systems. Arrows indicate the flux of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) between cropping systems and the atmosphere. Atmospheric exchanges are CO_2 unless noted otherwise.

a no-till system is plowed every few years to solve a management problem or reap mineralizable N benefits, then *t* needs to span one or more of these tillage cycles.

Agricultural Inputs

Management decisions have a strong influence on the magnitude of CO_2e emissions associated with agricultural inputs including seed production, agrochemicals, and fuel use in farm operations. For example, the MCSE Conventional system, which is tilled, emits 35% more CO_2 from fuel use than does the No-till system (Fig. 12.2). Although the No-till system lacks soil preparation, the additional herbicides and energy required at planting (because the soil is more resistant than had it been plowed) partly offset the CO_2e savings associated with reduced fuel use by not tilling (Fig. 12.2). Likewise, synthetic N fertilizer can be a large source of CO_2e because of CO_2 emitted during its manufacture (Table 12.2), but this cost is avoided in alfalfa, which acquires its N from the atmosphere through biological N fixation. However, this savings is almost entirely offset by the CO_2e costs of alfalfa's increased agricultural lime and potassium (K) requirements. Thus, overall CO_2e emissions of the Alfalfa system are ~60% of those of the No-till and Conventional systems, despite the absence of N fertilizer use (Fig. 12.2).

The CO_2e cost of producing agricultural lime (0.04 g CO_2e kg^{-1}; West and Marland 2002) is independent of its fate. As noted earlier, Hamilton et al. (2007) estimate that CO_2 emissions from agricultural lime applied to KBS soils are fully offset by CO_2 capture when at least 50% of the lime is dissolved by carbonic acid rather than by a strong mineral acid. Nitric acid in agricultural soils is largely produced by nitrifying bacteria that produce 2 moles of H^+ for every mole of ammonium oxidized to NO_3^- (Robertson and Groffman 2015), and this can be

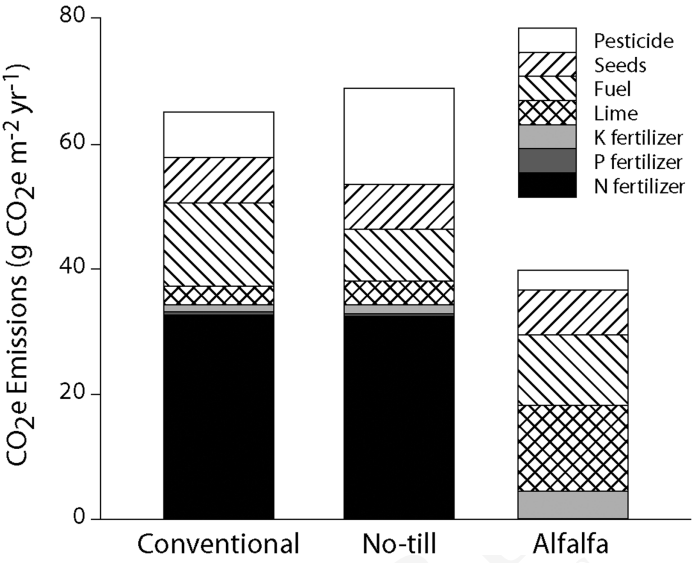


Figure 12.2. Emissions of carbon dioxide equivalents (CO_2e) from agricultural fuel and chemical inputs in the Conventional and No-till corn-soybean-wheat and the Alfalfa systems of the Main Cropping System Experiment (MCSE). Emissions from N fertilizer represent production costs only, not the resultant emission of nitrous oxide after fertilizer application.

an important source of strong acid in fertilized soils. The relative significance of these reactions with liming materials in most soils and environmental settings is not well known.

Pesticides have high CO_2e production costs (4–5 kg CO_2e kg^{-1}) but a disproportionately low impact on ecosystem CO_2e fluxes because usually only a few grams of active ingredient are applied per hectare; they thus represent only ~10% of total agricultural inputs, except in the No-till system, where they represent ~20% of total inputs (Fig. 12.2). Seeds have a larger impact on GWI due to their high production costs and seeding rates of ~20, 70, and 170 kg ha^{-1} for corn, soybean, and wheat, respectively (Gelfand et al. 2013). Estimates of GWI for seeds vary widely, however, depending on how seed production costs are estimated: West and Marland (2002) used a dollar value method to estimate a cost of 0.25 kg CO_2e kg^{-1} soybean seed (Table 12.5); and Sheehan et al. (1998) estimate a cost of 2.62 kg CO_2e kg^{-1} soybean seed based on 150% of the soybean energy content of 23.8 MJ kg^{-1} (Rathke et al. 2007). Based on average actual production costs for irrigated soybean, we estimate a cost of 0.31 kg CO_2e kg^{-1} soybean seed (Table 12.5; West and Marland 2002).

Other inputs not common to KBS cropping systems can also have significant GHG costs. Most notable among these is irrigation. Pumped irrigation uses energy to move water from lower landscape positions or groundwater to the crop, and the electricity or diesel used for this can readily become the dominant component of the GWI of irrigated systems (Mosier et al. 2005). Irrigation scheduling

Table 12.5. Estimation of the GHG cost of producing 1 kg of soybean seeds using three different approaches.

Approach	GHG Cost ($kg\ CO_2e\ kg^{-1}$ seeds)
U.S. dollar value[a]	0.25
Energy content[b]	2.62
Direct estimation[c]	0.31

[a]From West and Marland (2002).
[b]Assumes all energy for soybean production is derived from fossil diesel with an energy content of 36.4 MJ L^{-1}; the energy content of soybean seeds is 23.8 MJ kg^{-1}; CO_2 emission from burning fossil diesel is 2.67 kg CO_2 L^{-1}.
[c]Based on CO_2e emissions from irrigated soybean production (239.9 kg C ha^{-1} yr^{-1}; West and Marland 2002) and average U.S. soybean yield (2.8 Mg ha^{-1}; http://www.nass.usda.gov/).

can also affect the amount of NO_3^- driven from the root zone into surface water and groundwater systems (Gehl et al. 2005), where it can be denitrified to N_2O (Beaulieu et al. 2011).

Nitrous Oxide and Methane Fluxes

Soil N_2O emissions are directly related to soil N availability and therefore to N fertilization and N fixation. In KBS LTER systems, those with high soil N availability—either from fertilizer inputs (e.g., in the Conventional and No-till systems) or from N fixed by leguminous cover crops (e.g., by red clover in the Reduced Input and Biologically Based systems) or by the primary crops themselves (e.g., soybean and alfalfa)—showed higher N_2O emissions than did systems with lower N inputs and availability (Fig. 12.3). This is a common finding in the N_2O literature (see Robertson and Vitousek 2009, Millar et al. 2010); in fact, global GHG inventories for agricultural N_2O emissions are largely based on a simple percentage of national fertilizer N inputs (IPCC 2006). Higher N_2O emissions in crops with lower N availability (i.e., wheat vs. soybeans, Fig. 12.3) suggest, however, that not only N availability but also specific crop (i.e., rotation type) may have an effect.

Nitrous oxide emissions appear to be especially high where N fertilization exceeds crop N requirements. McSwiney and Robertson (2005) found a nonlinear, exponentially increasing N_2O emission rate from KBS soils in continuous corn as fertilization levels increased beyond the point required for maximum yield. Others have since found similar responses (Grant et al. 2006, Ma et al. 2010, Millar et al. 2010, Hoben et al. 2011), suggesting that mitigation efforts directed toward more precise fertilizer use may have greater payoffs than those estimated by inventory methods based on a simple percentage of inputs. Millar et al. (2010, 2012, 2013) incorporated this relationship into C market incentives that can compensate farmers for more conservative N fertilizer use, which in theory is a promising way to promote fertilizer conservation in general, with both climate and water quality (Hamilton 2015, Chapter 11 in this volume) benefits.

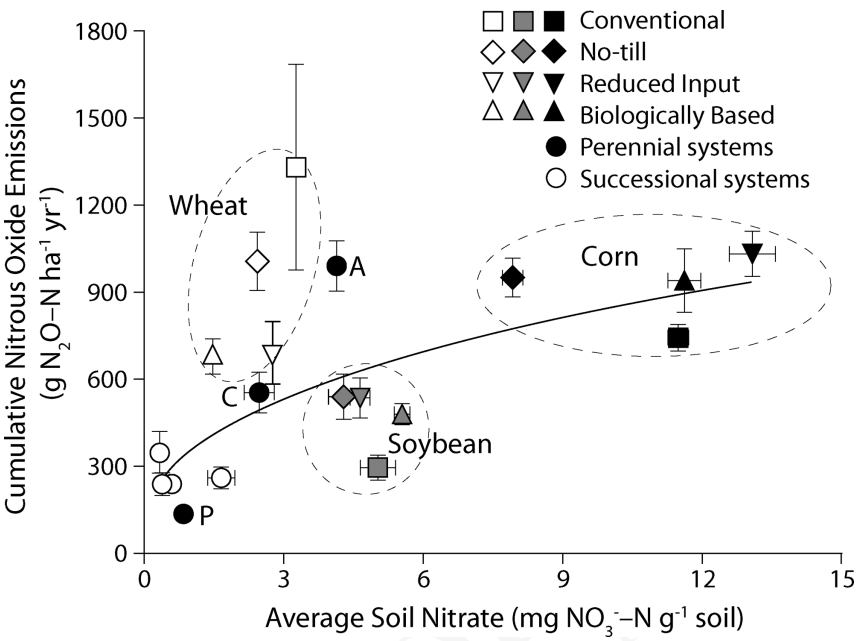


Figure 12.3. Relationship between soil nitrous oxide (N_2O) emissions and KCl-extractable nitrate (NO_3^-) in near-surface soils, fit with a power regression ($y = 358.9 \times (1 - e^{(0.37)})$, $R^2 = 0.58$, $p < 0.001$; the wheat portions of Conventional and No-till systems are not included in the regression). Annual systems are circled by crop; perennial systems are labeled as A = Alfalfa, P = Poplar, and C = Coniferous Forest.

Nitrous oxide is also emitted from aquatic systems that drain agricultural watersheds. Considerable NO_3^- is lost from intensively fertilized fields (Syswerda et al. 2012, Hamilton 2015, Chapter 11 in this volume), and based on watershed mass balances, most of this NO_3^- appears to be denitrified to N_2O and N_2. A recent cross-site study of stream N cycling that includes the broader watershed around KBS (Beaulieu et al. 2008, 2011) suggests that streams and rivers play a particularly important role in N transformations, and may be responsible for a surprising proportion of global anthropogenic N_2O emissions.

Methane is consumed by—rather than emitted from—most field crop systems other than flooded rice. In most well-aerated soils, more CH_4 is oxidized to CO_2 by methanotrophic bacteria than is produced by methanogenic bacteria. This means that soil methanotrophs also consume atmospheric CH_4, helping to attenuate atmospheric concentrations that would otherwise build at even higher rates than are occurring today. Methane oxidation by soil methanotrophs is estimated to consume around 30 Tg yr^{-1}. Although this is only ~5% of the total global CH_4 flux (Forster et al. 2007), it is close to the rate at which CH_4 is accumulating in the atmosphere (37 Tg yr^{-1}), suggesting that were consumption reduced—or intensified—atmospheric concentrations might be likewise affected.

Conversion of forest and grassland soils to agriculture reduces rates of soil CH_4 oxidation by 80–90% (Mosier et al. 1991, Smith et al. 2000, Del Grosso et al. 2000). In the MCSE, CH_4 oxidation in the Conventional system is about 20% of the rate in the Deciduous Forest (Robertson et al. 2000, Suwanwaree and Robertson 2005). Much of this suppression appears to stem from greater N availability in cropped soils rather than N fertilizer per se or tillage-induced changes in soil structure: oxidation was equally low in the unfertilized Biologically Based system, and fertilizing Deciduous Forest plots immediately reduces oxidation rates for the period that inorganic N pools are elevated, while tilling them has no discernible effect (Fig. 12.4; Suwanwaree and Robertson 2005). Gulledge and Schimel (1998) showed that much of the effect of N appears related to the competitive inhibition of CH_4 oxidation enzymes by ammonium ions. A longer time period of measurements of GHG fluxes from KBS soils shows, however, some recovery of CH_4 oxidation in the Biologically Based, Alfalfa, and Early successional systems 20 years after establishment (Table 12.6), despite relatively high N availability.

Nitrogen availability alone also does not explain the very slow recovery of CH_4 oxidation rates in abandoned cropland or in cropland converted to unfertilized perennial crops. After 10 years, there was no recovery of oxidation rates in either the Poplar system or in the Early Successional community (Robertson et al. 2000)—two systems in which soil NO_3^- levels and NO_3^- leaching rates are vanishingly low

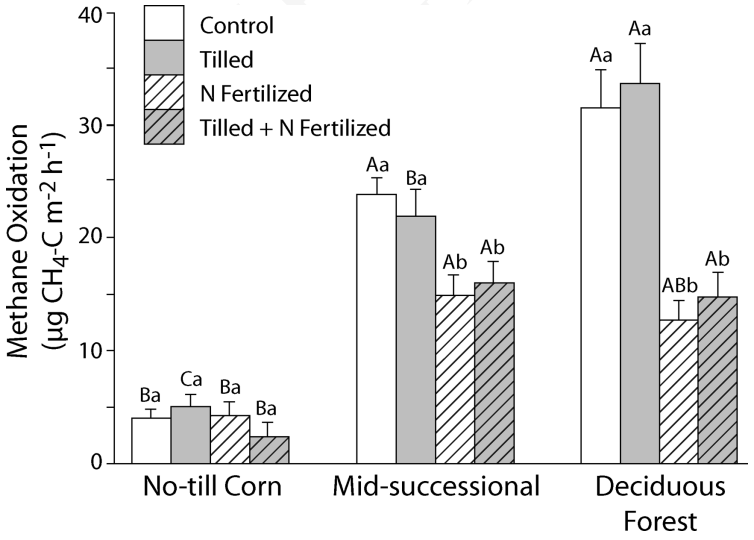


Figure 12.4. The reduction of methane (CH_4) oxidation upon soil disturbance and ammonium nitrate fertilization (100 kg N ha^{-1}) in the No-till (planted in corn), Mid-successional, and Deciduous Forest systems of the MCSE. Vertical bars are standard errors of the mean (SE, n = 3 sites × 7 sampling dates). Different uppercase and lowercase letters represent significant treatment differences ($p < 0.05$) among and within sites, respectively. Modified from Suwanwaree and Robertson (2005).

Table 12.6. Nitrous oxide (N_2O) and methane (CH_4) fluxes and GWIs from 1991 to 2010 of the MCSE.*

System	GHG Flux†		GWI	
	N_2O-N	CH_4-C	N_2O	CH_4
	(g ha^{-1} d^{-1})		(g CO_2e m^{-2} y^{-1})	
Annual Crops (corn–soybean–wheat rotation)				
Conventional	2.15 (0.33)[a]	–0.69 (0.09)[a]	36.6 (5.6)	–0.8 (0.1)
No-till	2.27 (0.15)[a]	–0.65 (0.06)[a]	38.6 (2.5)	–0.7 (0.1)
Reduced Input	2.06 (0.13)[a]	–0.57 (0.05)[a]	35.0 (2.3)	–0.6 (0.1)
Biologically Based	1.91 (0.15)[a]	–0.85 (0.03)[b]	32.5 (2.6)	–1.0 (0.0)
Perennial Crops				
Alfalfa	2.72 (0.24)[a]	–0.87 (0.08)[b]	46.16 (4.2)	–1.0 (0.1)
Poplar	0.38 (0.04)[b]	–0.81 (0.05)[a,b]	6.4 (0.6)	–0.9 (0.1)
Successional Community				
Early Successional	0.66 (0.05)[c]	–0.89 (0.05)[b]	11.2 (0.9)	–1.0 (0.1)

*Statistically significant differences (ANOVA repeated measures, $p < 0.05$) are indicated by different letters within columns. GHG fluxes are based on untransformed values and GWIs are carbon dioxide equivalents (CO_2e), calculated using a 100-year time horizon (IPCC 2007), and all are expressed as mean (±SE, $n = 4$ replicates).
†Soil GHG fluxes were sampled April–December, 1991–2010. Positive values indicate emission to the atmosphere; negative values are uptake.
Source: Gelfand et al. (2013).

(Syswerda et al. 2012). After 20 years of abandonment, however, CH_4 oxidation does begin to recover slightly in these systems (Table 12.6; Gelfand et al. 2013). Measurements in our Mid-successional community suggest that it takes 50 years or more for CH_4 oxidation to exceed 50% of preconversion rates (Robertson et al. 2000, Suwanwaree and Robertson 2005).

Why does CH_4 consumption take so long to recover to preconversion levels? Part of the explanation may be related to methanotroph community composition and, in particular, methanotroph diversity (Gulledge et al. 1997). Levine et al. (2011) found substantially higher methanotroph diversity in MCSE systems with higher oxidation rates, suggesting that microbial community composition (see Schmidt and Waldron 2015, Chapter 6 in this volume) may matter for CH_4 oxidation in the same way that it matters for N_2O production via denitrification (Cavigelli and Robertson 2000, 2001; Schmidt and Waldron 2015, Chapter 6 in this volume).

Soil CH_4 oxidation is not known to be affected by any existing agronomic practice; it is as low in the MCSE No-till and Reduced Input systems and in various organic systems of the Living Field Lab Experiment (Robertson and Hamilton 2015, Chapter 1 in this volume; Snapp et al. 2015, Chapter 15 in this volume) as it is in the fertilized Conventional system (Suwanwaree 2003). Alternative agronomic practices that increase the capacity for CH_4 oxidation could have the potential for significant GWI mitigation (Gelfand et al. 2013).

Crop Carbon Dioxide Capture

Crop production goals, that is, crop productivity and how its biomass is used, have great influence on the GWI of agricultural systems. Net Ecosystem Productivity (NEP; Fig. 12.1) is the net annual uptake of CO_2 from the atmosphere by the plant–soil system, defined as gross primary production (total CO_2 uptake) less ecosystem respiration (total CO_2 produced) (Randerson et al. 2002). Net Ecosystem Productivity thus represents the overall C balance, with a few caveats (Chapin et al. 2006). In long-established annual cropping systems, NEP is typically zero—as much CO_2 is respired as is captured annually. Although not all the biomass may be consumed in the year produced—a portion of the crop residue, for example, may persist as soil organic matter (SOM) C for decades or centuries (see Paul et al. 2015, Chapter 5 in this volume)—for ecosystems at C-balance equilibrium an equivalent amount of older SOM C may be respired. Thus on an annual basis, as much CO_2 will leave the ecosystem as enters.

Cropping systems recently converted from forests or grasslands have a negative NEP, annually releasing more CO_2 to the atmosphere than they capture. More CO_2 is respired than fixed because the original vegetation left on-site, including roots, will decompose and long-stored SOM will be rapidly oxidized when tillage breaks up soil aggregates and exposes protected C to microbial attack (Grandy and Robertson 2006, 2007; Paul et al. 2015, Chapter 5 in this volume). In most temperate regions, the SOM content of recently converted soils will approach a new steady-state equilibrium at 40–60% of original levels in 40–60 years (Paul et al. 1997).

Conversely, cropping systems that are gaining C have a positive NEP. In annual cropping systems, this occurs when SOM accumulates with the adoption of no-till cultivation or cover crops. When left untilled, soil aggregates that form around small particles of organic matter are more stable and protect the organic matter from microbial oxidation (Six et al. 2000, Grandy et al. 2006)—thereby allowing soil C pools to rebuild to some proportion of their original C content (West and Post 2002). At KBS, rates of soil C gain in the No-till system are typical of gains elsewhere in the Midwest (West and Post 2002): ~33 g C m^{-2} yr^{-1} in the Ap horizon, with no significant change in deeper layers to 1 m (Syswerda et al. 2011).

Even in tilled soils, cover crops can build SOM quickly—in the unfertilized Biologically Based system, C was sequestered in the surface soil (A/Ap horizon) at ~50 g C m^{-2} yr^{-1} over the first 12 years of establishment (Syswerda et al. 2011). The mechanisms underlying cover crop gains are not yet clear, but may be related to the greater polyphenolic content in legume residue that could slow its decomposition (Palm and Sánchez 1991). Chemical protection may also be occurring in the Early Successional community, where in addition to the cessation of tillage, plant residue diversity and perennial roots help to explain C sequestration rates in the surface soil of >100 g C m^{-2} yr^{-1} over the first 12 years of abandonment (Syswerda et al. 2011).

Perennial crops provide an additional soil C advantage by having permanent deep roots, which both sequester C in long-lived belowground tissue and produce

exudates that microbes at depth can transform into recalcitrant organic matter (Wickings et al. 2012). In the Early Successional community, soils below the Ap horizon accumulated ~33 g C m^{-2} yr^{-1} over the first 12 years of establishment, and SOM also increased at depth in the Alfalfa and Poplar systems during this time (Syswerda et al. 2011).

Climate Change Mitigation through Sustainable Biofuel Production

Projections of reduced fossil fuel availability and growing concerns about the environmental impacts of fossil fuel use have stimulated interest in renewable energy sources from agricultural crops (Robertson et al. 2008, Tilman et al. 2009), which would result in the concomitant expansion of cropland to satisfy new production demands (Field et al. 2008, Feng and Babcock 2010). Biofuels produced from crops could provide climate benefits by offsetting fossil fuel use. Offsets are created when fuels produced from harvested crop biomass are used instead of fossil fuels. A fossil fuel CO_2 offset credit is equivalent to the amount of CO_2 in the avoided fossil fuel use. A full cost accounting or life cycle analysis is necessary to determine the net amount of fossil fuel CO_2 actually avoided: feedstocks can greatly differ in their net C balance (Fargione et al 2010, Gelfand et al. 2013), and calculations must include both the direct C debt accrued from creating a new biofuel cropping system (Fargione et al. 2008, Gelfand et al. 2011) as well as the indirect debt created by the need to clear land elsewhere to replace lost food production (Searchinger et al. 2008). Moreover, crop residue removed to produce biofuel is residue that would otherwise have contributed to maintain or build soil C (Wilhelm et al. 2007), such that its removal can be a net GWI cost as foregone soil C sequestration. Although the mitigation impact of a biofuel cropping system can be substantial, benefits depend entirely on where and how and which crops are grown. Two examples from KBS serve to illustrate this point: one based on the use of existing cropping systems for biofuel production, and the other on the conversion of former cropland enrolled for 22 years in the USDA's Conservation Reserve Program (CRP).

The GWI of Established Biofuel Crops

Currently, most U.S. biofuel production is ethanol made from corn grain. A small amount of biodiesel is produced from soybean and other oil seed crops. In other parts of the world, sugarcane and oil seed crops such as palm oil are used to produce biofuels, and in the future cellulosic ethanol will likely be produced from agricultural wastes and residues, perennial grasses, and woody vegetation (NRC 2009). Future fuels will likely also include butanol, alkanes, and other so-called drop-in hydrocarbons, and biomass is also likely to be combusted directly to produce electricity and heat, avoiding some of the energy loss associated with biorefining and with burning fuel in internal combustion engines of individual automobiles.

Over the next several decades, then, agricultural biomass will increasingly be used as feedstocks to produce a variety of energy sources. This will place

unprecedented demands on croplands globally; in the United States, agricultural biomass needs are expected to approach 700 Tg (Perlack et al. 2005, NRC 2009), which could take as much as 90 million ha (222 million acres) of additional cropland (CAST 2011, Robertson et al. 2011)—about half as much U.S. land as we use today for all annual crops. Impacts on the biogeochemistry (Robertson et al. 2011) and biodiversity (Fletcher et al. 2011) of agricultural landscapes are likely to be correspondingly high. The climate change implications of these impacts make it all the more important that policy and landowner decisions be based on accurate GWI assessments.

Gelfand et al. (2013) used 20 years of observations from the MCSE to analyze the life-cycle C balances of systems that could potentially be harvested for use as biofuel feedstocks. For the two annual crop systems evaluated—the Conventional and No-till systems—they assumed grain was used for grain-based ethanol (corn, wheat) or biodiesel (soybean), and that 60% of wheat straw was used for cellulosic ethanol. No residue was removed from the corn or soybean portions of the rotations in order to protect existing SOM stores (NRC 2009). Three perennial cropping systems provided biomass for cellulosic ethanol—Alfalfa, Poplar, and the Early Successional community, which was either fertilized or unfertilized.

Resulting GHG balances (Fig. 12.5B) show a negative (net mitigating) GWI for all biofuel cropping systems. Fossil fuel offset credits were greatest in the Alfalfa and fertilized Early Successional communities and lowest in the more intensively managed systems. The differences were related to both yield and management. For an example, high yields of the No-till system were balanced by relatively high management inputs, which decreased total fossil fuel offset credits. On the other hand, cellulosic biomass produced in the less productive Early Successional system lacked significant management inputs and therefore provided more fossil fuel offset credits (Fig. 12.5A). Credits for the Early Successional community would be substantially higher were technology developed to improve harvest efficiency for perennial grasses, now only 55% (Monti et al. 2009). Nevertheless, the Early Successional community still exhibited the highest net mitigation potential with a GWI of about –851 g CO_2e m^{-2} yr^{-1}, while the more productive No-till system was only fourth, with a net GWI of –397 g CO_2e m^{-2} yr^{-1}. Alfalfa was intermediate to these with a mitigation potential of about –605 g CO_2e m^{-2} y^{-1} because of the high GWI cost of increased N_2O emissions and lower SOC accumulation (Fig. 12.5B). The net mitigation potential of the Poplar system was low, owing to the lack of net soil organic C gain over its rotation including the subsequent break period. Fertilizing the Early Successional community increased its productivity and thus its fossil fuel offset by ~35%, though net GWI remained basically unchanged due to the greater CO_2e cost of the fertilizer N and increased soil N_2O emissions associated with fertilization. Nevertheless, by increasing productivity with no net change in GWI, N fertilization would reduce the amount of land needed to produce a given amount of biofuel feedstock.

The boundary of this analysis includes the full life cycle of biofuel and fossil fuel production. Expanding the boundary to include indirect land-use effects could change GWIs significantly for the worse. More specifically, the GWI of these systems will be significantly less mitigating if biofuel crops were to displace

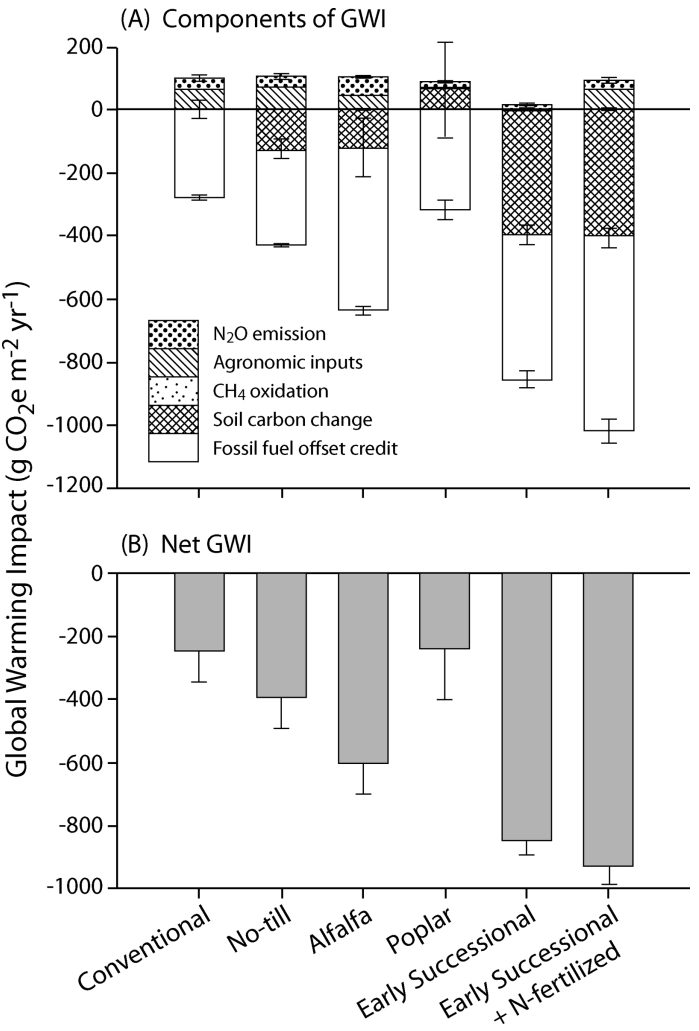


Figure 12.5. Components of Global Warming Impact (GWI, A) and the net impact (B) for agricultural and successional ecosystems in the MCSE, if harvested for cellulosic biofuel feedstock production. Error bars represent standard error (n = 6). Conventional and No-till are in a corn-soybean-wheat rotation. Redrawn from Gelfand et al. (2013).

food crops that must then be grown elsewhere on land not otherwise in agricultural production—what C markets call leakage. To avoid leakage, biofuel crops could be grown on marginal land, that is, land not now used for food crops or grazing. This could also avoid the ethical issue of food vs. fuel when feedstocks are grown on arable cropland.

Perennial grasses are particularly well suited for such marginal lands—after establishment, they require no agronomic attention other than harvest and perhaps low rates of fertilization, and thus should have few environmental liabilities.

Moreover, the right mixture of grasses could provide habitat for beneficial insects as well as for birds and other wildlife, providing additional environmental benefits especially if the marginal land were otherwise degraded due to prior management. Using land-use databases and the EPIC model (Zhang et al. 2010) to scale KBS results for the fertilized Early Successional community to a 10-state U.S. North Central region, Gelfand et al. (2013) estimated that marginal lands could produce at least 21×10^9 L of biofuel annually, or about 25% of the 80 billion L 2022 target legislated for advanced biofuels by the U.S. Energy Independence and Security Act of 2007.

The GWI of Land-Use Conversion for Biofuel Production

In 2014, about 10 million ha of former U.S. cropland were enrolled in the USDA Conservation Reserve Program (CRP) (USDA-FSA 2014). Converting these conservation plantings—most commonly in grassland vegetation—back to cropland risks the release of substantial amounts of stored soil organic C, effectively creating a C debt that models suggest could wipe out the benefits of up to 48 years of subsequent grain-based feedstock production (Fargione et al. 2008). Actual measurements of C debt following conversion, however, are not yet available, and theory suggests that the debt could be significantly less than this with careful management.

In 2009 three KBS fields enrolled in the CRP program since 1987 were converted from long-term brome grass (*Bromus inermis*) to no-till soybean as a recommended break crop prior to growing various cellulosic feedstocks. The advantage of soybeans as a break crop is that glyphosate-tolerant soybeans can be sprayed multiple times during the growing season to kill any remnants of the preexisting vegetation (brome grass, in this case). A CO_2 eddy covariance tower was placed in each field and in an unconverted CRP reference field (Zenone et al. 2011). Eddy covariance towers measure net ecosystem CO_2 flux by observing CO_2 concentrations and the movement of air between the atmosphere and the plant canopy at intervals of one-tenth of a second, allowing estimation of CO_2 fluxes that are then summed over a 30-minute period to provide half-hour snapshots of net ecosystem C gain and loss. Summing the half-hour snapshots over days and weeks provides, ultimately, the annual NEP of the studied ecosystem. In this way, total soil C change can be inferred long before it can be measured directly with soil sampling.

Figure 12.6A shows seasonal patterns of NEP in the converted and reference CRP systems during the year of conversion. Net Ecosystem Productivity was negative in both systems at the beginning of the year, reflecting net emissions of CO_2 as soil respiration exceeded wintertime photosynthesis by brome grass, which was nil. The negative fluxes turned positive beginning in the spring (around Day 100) as brome grass CO_2 fixation began to exceed total respiration. The CRP reference system continued to gain CO_2 until ca. Day 220, when brome grass senescence in the fall led to reduced photosynthesis, and respiration again dominated the CO_2 flux. By the end of the year, however, the cumulative NEP was still positive (above the origin in Fig. 12.6A), indicating net sequestration of CO_2 within the ecosystem. In the CRP converted system, on the other hand, an herbicide application around Day 120 interrupted CO_2 fixation by the brome grass, and the system continued to lose more

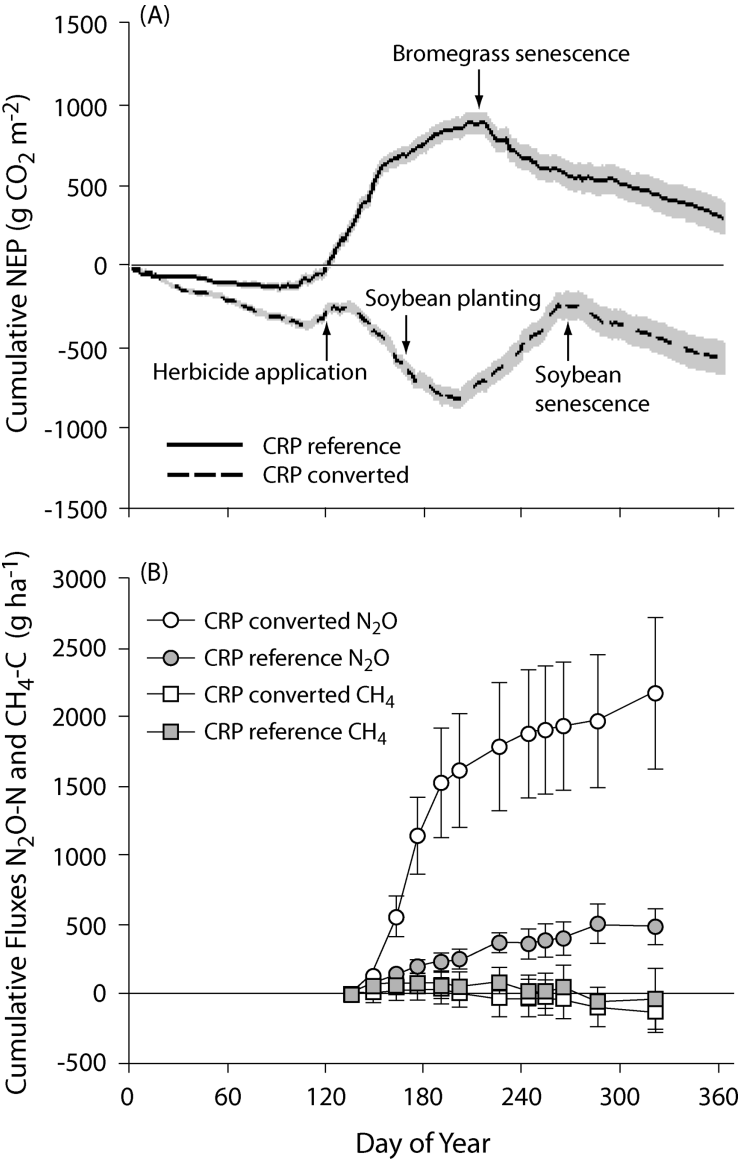


Figure 12.6. Cumulative fluxes of greenhouse gases from Conservation Reserve Program (CRP) grasslands converted to no-till soybean crops. A) Average cumulative net ecosystem productivity (NEP) during 2009 for the CRP reference field (top solid line) and converted field (bottom dashed line). Positive values indicate net CO_2 sequestration. Shaded area represents the standard deviation of cumulative NEP. B) Average net cumulative fluxes of N_2O (circles) and CH_4 (squares) at the study sites over the same period. Error bars represent standard errors (n = 3 replicate fields for CRP converted and n = 4 replicates within one field for CRP reference). Redrawn from Gelfand et al. (2011).

CO_2 than it gained until around Day 200, when net photosynthesis by the recently planted soybeans exceeded the respiration of the herbicide-treated grasses. Once the soybeans senesced around Day 260, respiration again dominated the system's CO_2 flux, and the cumulative NEP remained negative (i.e., net CO_2 release) until the end of the year, by which time some 500 g CO_2 m^{-2} had been emitted by the system.

Overall, during the first year of the conversion study, converted fields lost ~520 g CO_2 m^{-2}, mostly from the decomposition of killed grasses and soil C oxidation. This compares to a gain of ~300 g CO_2 m^{-2} by the reference field, which sequestered C into belowground biomass and SOM (Zenone et al. 2011, Gelfand et al. 2011).

Combining eddy covariance results with the other major sources of GWI in the system—farming inputs and N_2O and CH_4 fluxes, in particular—provides a measure of net GWI analogous to other, less continuous methods. N_2O emissions were also substantially higher in the converted sites (Fig. 12.6B), contributing to a total GWI or C debt of 68±7 Mg CO_2e m^{-2} (Gelfand et al. 2011). This measured C debt (from no-till conversion of CRP fields to agricultural production) is substantial but stands at the lower end of previously modeled estimates of 75–305 Mg CO_2e m^{-2} (Fargione et al. 2006, Searchinger 2008). No-till continuous corn or corn–soybean rotations, when used for grain ethanol production, could repay this C debt in 29–40 years, which is somewhat shorter than previously estimated (Fargione et al. 2008).

Summary

Intensively managed crop production systems contribute substantially to anthropogenic climate change, but changing how systems are managed could mitigate their impact. GWI analyses provide a measure for comparing the climate benefits and costs of different management practices and, by summation, of entire cropping systems. Major components of GWI include land-use change (where appropriate), farming inputs (fuel, fertilizers, pesticides), soil C change, and fluxes of the non-CO_2 GHGs N_2O and CH_4. Nitrous oxide emissions represent the largest GWI in the MCSE annual cropping systems, mainly stemming from high fertilizer inputs but also from the cultivation of N-fixing crops. Improved N management thus represents one of the largest potentials for the mitigation of agricultural GHG emissions. Soil organic C gain represents an equally large mitigation potential where soils could be managed to sequester C via no-till management, cover crops, and the cultivation of perennial crops. Perennial, cellulosic biofuel crops offer substantial climate change mitigation potential so long as their production does not cause food crops with a higher GWI to be planted elsewhere.

References

Ambus, P., and G. P. Robertson. 1998. Automated near-continuous measurement of carbon dioxide and nitrous oxide fluxes from soil. Soil Science Society of America Journal 62:394–400.

Barker, T., I. Bashmakov, L. Bernstein, J. E. Bogner, P. R. Bosch, R. Dave, O. R. Davidson, B. S. Fisher, S. Gupta, K. Halsnæs, G. J. Geij, S. Kahn Riveiro, S. Kobayashi,

M. D. Levine, D. L. Martino, O. Masera, B. Metz, L. A. Meyer, G. J. Nabuurs, A. Najam, N. Nakicenovic, H.-H. Rogner, J. Roy, J. Sathaye, R. Schock, P. Shukla, R. E. H. Sims, P. Smith, D. A. Tirpak, D. Urge-Vorsatz, and D. Zhou. 2007. Technical Summary. Pages 25–93 in B. Metz, O. R. Davidson, P. R. Bosch, R. Dave, and L. A. Meyer, editors. Climate change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, New York, New York, USA.

Barton, L., R. Kiese, D. Gatter, K. Butterbach-Bahl, R. Buck, C. Hinz, and D. V. Murphy. 2008. Nitrous oxide emissions from a cropped soil in a semi-arid climate. Global Change Biology 14:177–192.

Beaulieu, J. J., C. P. Arango, S. K. Hamilton, and J. L. Tank. 2008. The production and emission of nitrous oxide from headwater streams in the Midwestern United States. Global Change Biology 14:878–894.

Beaulieu, J. J., J. L. Tank, S. K. Hamilton, W. M. Wollheim, R. O. Hall, Jr., P. J. Mulholland, B. J. Peterson, L. R. Ashkenas, L. W. Cooper, C. N. Dahm, W. K. Dodds, N. B. Grimm, S. L. Johnson, W. H. McDowell, G. C. Poole, H. M. Valett, C. P. Arango, M. J. Bernot, A. J. Burgin, C. L. Crenshaw, A. M. Helton, L. T. Johnson, J. M. O'Brien, J. D. Potter, R. W. Sheibley, D. J. Sobota, and S. M. Thomas. 2011. Nitrous oxide emission from denitrification in stream and river networks. Proceedings of the National Academy of Sciences USA 108:214–219.

Burney, J. A., S. J. Davis, and D. B. Lobell. 2010. Greenhouse gas mitigation by agricultural intensification. Proceedings of the National Academy of Sciences USA 107:12052–12057.

Canadell, J. G., C. Le Quere, M. R. Raupach, C. B. Field, E. T. Buitenhuis, P. Ciais, T. J. Conway, R. A. Gillett, R. A. Houghton, and G. Marland. 2007. Contributions to accelerating atmospheric CO_2 growth from economic activity, carbon intensity, and efficiency of natural sinks. Proceedings of the National Academy of Sciences USA 104:18866–18870.

CAST (Council for Agricultural Science and Technology). 2011. Carbon sequestration and greenhouse gas fluxes in agriculture: challenges and opportunities. Task Force Report No.142, CAST, Ames, Iowa, USA.

Cavigelli, M. A., and G. P. Robertson. 2000. The functional significance of denitrifier community composition in a terrestrial ecosystem. Ecology 81:1402–1414.

Cavigelli, M. A., and G. P. Robertson. 2001. Role of denitrifier diversity in rates of nitrous oxide consumption in a terrestrial ecosystem. Soil Biology & Biochemistry 33:297–310.

Chapin, F. S., G. M. Woodwell, J. T. Randerson, E. B. Rastetter, G. M. Lovett, D. D. Baldocchi, D. A. Clark, M. E. Harmon, D. S. Schimel, R. Valentini, C. Wirth, J. D. Aber, J. J. Cole, M. L. Goulden, J. W. Harden, M. Heimann, R. W. Howarth, P. A. Matson, A. D. McGuire, J. M. Melillo, H. A. Mooney, J. C. Neff, R. A. Houghton, M. L. Pace, M. G. Ryan, S. W. Running, O. E. Sala, W. H. Schlesinger, and E. D. Schulze. 2006. Reconciling carbon-cycle concepts, terminology, and methods. Ecosystems 9:1041–1050.

Del Grosso, S. J., W. J. Parton, A. R. Mosier, D. S. Ojima, C. S. Potter, W. Borken, R. Brumme, K. Butterbach-Bahl, P. M. Crill, K. E. Dobbie, and K. A. Smith. 2000. General CH_4 oxidation model and comparisons of CH_4 oxidation in natural and managed systems. Global Biogeochemical Cycles 14:999–1019.

Fargione, J., J. Hill, D. Tilman, S. Polasky, and P. Hawthorne. 2008. Land clearing and the biofuel carbon debt. Science 319:1235–1237.

Fargione, J. E., R. J. Plevin, and J. D. Hill. 2010. The ecological impact of biofuels. Annual Review Ecology, Evolution, and Systematics 41:351–377.

Feng, H., and B. A. Babcock. 2010. Impacts of ethanol on planted acreage in market equilibrium. American Journal of Agricultural Economics 92:789–802.

Field, C. B., J. E. Campbell, and D. B. Lobell. 2008. Biomass energy: the scale of the potential resource. Trends in Ecology & Evolution 23:65–72.

Fletcher Jr., R. J., B. A. Robertson, J. Evans, J. R. R. Alavalapati, P. J. Doran, and D. J. Schemske. 2011. Biodiversity conservation in the era of biofuels: risks and opportunities. Frontiers in Ecology and the Environment 9:161–168.

Forster, P., P. Ramsaswamy, T. Artaxo, T. Bernsten, R. Betts, D. W. Fahey, J. Haywood, J. Lean, D. C. Lowe, G. Myhre, J. Nganga, R. Prinn, G. Raga, M. Schultz, and R. Van Dorland. 2007. Changes in atmospheric constituents and in radiative forcing. Pages 129–234 in D. Solomon, D. Qin, M. Manning, Z. Chen, K. B. Marquis, M. Averyt, M. Tignor, and H. L. Miller, editors. Climate change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.

Gehl, R. J., J. P. Schmidt, L. R. Stone, A. J. Schlegel, and G. A. Clark. 2005. In situ measurements of nitrate leaching implicate poor nitrogen and irrigation management on sandy soils. Journal of Environmental Quality 34:2243–2254.

Gelfand, I., R. Sahajpal, X. Zhang, R. C. Izaurralde, K. L. Gross, and G. P. Robertson. 2013. Sustainable bioenergy production from marginal lands in the US Midwest. Nature 493:514–517.

Gelfand, I., S. S. Snapp, and G. P. Robertson. 2010. Energy efficiency of conventional, organic, and alternative cropping systems for food and fuel at a site in the U.S. Midwest. Environmental Science and Technology 44:4006–4011.

Gelfand, I., T. Zenone, P. Jasrotia, J. Chen, S. K. Hamilton, and G. P. Robertson. 2011. Carbon debt of Conservation Reserve Program (CRP) grasslands converted to bioenergy production. Proceedings of the National Academy of Sciences USA 108:13864–13869.

Grandy, A. S., and G. P. Robertson. 2006. Aggregation and organic matter protection following tillage of a previously uncultivated soil. Soil Science Society of America Journal 70:1398–1406.

Grandy, A. S., and G. P. Robertson. 2007. Land-use intensity effects on soil organic carbon accumulation rates and mechanisms. Ecosystems 10:58–73.

Grandy, A. S., G. P. Robertson, and K. D. Thelen. 2006. Do productivity and environmental trade-offs justify periodically cultivating no-till cropping systems? Agronomy Journal 98:1377–1383.

Grant, R. F., E. Pattey, T. W. Goddard, L. M. Kryzanowski, and H. Puurveen. 2006. Modeling the effects of fertilizer application rate on nitrous oxide emissions. Soil Science Society of America Journal 70:235–248.

Gulledge, J., A. P. Doyle, and J. P. Schimel. 1997. Different NH4+-inhibition patterns of soil CH_4 consumption: a result of distinct CH_4-oxidizer populations across sites? Soil Biology and Biochemistry 29:13–21.

Gulledge, J., and J. P. Schimel. 1998. Low-concentration kinetics of atmospheric CH_4 oxidation in soil and mechanism of NH_4^+ inhibition. Applied and Environmental Microbiology 64:4291–4298.

Hamilton, S. K. 2015. Water quality and movement in agricultural landscapes. Pages 275–309 in S. K. Hamilton, J. E. Doll, and G. P. Robertson, editors. The ecology of agricultural ecosystems: long-term research on the path to sustainability. Oxford University Press, New York, New York, USA.

Hamilton, S. K., A. L. Kurzman, C. Arango, L. Jin, and G. P. Robertson. 2007. Evidence for carbon sequestration by agricultural liming. Global Biogeochemical Cycles 21:GB2021.

Hoben, J. P., R. J. Gehl, N. Millar, P. R. Grace, and G. P. Robertson. 2011. Nonlinear nitrous oxide (N_2O) response to nitrogen fertilizer in on-farm corn crops of the US Midwest. Global Change Biology 17:1140–1152.

Hülsbergen, K.-J., B. Feil, S. Biermann, G.-W. Rathke, W.-D. Kalk, and W. Diepenbrock. 2001. A method of energy balancing in crop production and its application in a long-term fertilizer trial. Agriculture, Ecosystems & Environment 86:303–321.

IPCC (Intergovernmental Panel on Climate Change). 2001. Climate change 2001: the scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change, J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. Van der Linden, X. Dai, K. Maskell, and C. A. Johnson, editors. Cambridge University Press, New York, New York, USA.

IPCC (Intergovernmental Panel on Climate Change). 2006. N_2O emissions from managed soils, and CO_2 emissions from lime and urea application Pages 11.11–11.54 in H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe, editors. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. IGES (Institute for Global Environmental Strategies), Hayama, Japan.

Kim, S., and B. E. Dale. 2003. Cumulative energy and global warming impact from the production of biomass for biobased products. Journal of Industrial Ecology 7:147–162.

Kravchenko, A. N., and G. P. Robertson. 2011. Whole-profile soil carbon stocks: the danger of assuming too much from analyses of too little. Soil Science Society of America Journal 75:235–240.

Levine, U., T. K. Teal, G. P. Robertson, and T. M. Schmidt. 2011. Agriculture's impact on microbial diversity and associated fluxes of carbon dioxide and methane. The ISME Journal 5:1683–1691.

Ma, B. L., T. Y. Wu, N. Tremblay, W. Deen, M. J. Morrison, N. B. McLaughlin, E. G. Gregorich, and G. Stewart. 2010. Nitrous oxide fluxes from corn fields: on-farm assessment of the amount and timing of nitrogen fertilizer. Global Change Biology 16:156–170.

McSwiney, C. P., and G. P. Robertson. 2005. Nonlinear response of N_2O flux to incremental fertilizer addition in a continuous maize (*Zea mays* L.) cropping system. Global Change Biology 11:1712–1719.

Melillo, J. M., J. M. Reilly, D. W. Kicklighter, A. C. Gurge, T. W. Cronin, S. Paltsev, B. S. Felzer, X. Wang, A. P. Sokolov, and C. A. Schlosser. 2009. Indirect emissions from biofuels: how important? Science 326:1397–1399.

Millar, N., and G. P. Robertson. 2015. Agricultural nitrogen: boon and bane. Pages 213–251 in S. K. Hamilton, J. E. Doll, and G. P. Robertson, editors. The ecology of agricultural ecosystems: long-term research on the path to sustainability. Oxford University Press, New York, New York, USA.

Millar, N., G. P. Robertson, A. Diamant, R. J. Gehl, P. R. Grace, and J. P. Hoben. 2012. Methodology for quantifying nitrous oxide (N_2O) emissions reductions by reducing nitrogen fertilizer use on agricultural crops. American Carbon Registry, Winrock International, Little Rock, Arkansas, USA.

Millar, N., G. P. Robertson, A. Diamant, R. J. Gehl, P. R. Grace, and J. P. Hoben. 2013. Quantifying N_2O emissions reductions in US agricultural crops through N fertilizer rate reduction. Verified Carbon Standard, Washington, DC, USA.

Millar, N., G. P. Robertson, P. R. Grace, R. J. Gehl, and J. P. Hoben. 2010. Nitrogen fertilizer management for nitrous oxide (N_2O) mitigation in intensive corn (Maize) production: an emissions reduction protocol for US Midwest agriculture. Mitigation and Adaptation Strategies for Global Change 15:185–204.

Monti, A., S. Fazio, and G. Venturi. 2009. The discrepancy between plot and field yields: harvest and storage losses of switchgrass. Biomass & Bioenergy 33:841–847.

Mosier, A., R. Wassmann, L. Verchot, J. King, and C. Palm. 2004. Methane and nitrogen oxide fluxes in tropical agricultural soils: sources, sinks and mechanisms. Environment, Development and Sustainability 6:11–49.

Mosier, A. R., A. D. Halvorson, G. A. Peterson, G. P. Robertson, and L. Sherrod. 2005. Measurement of net global warming potential in three agroecosystems. Nutrient Cycling in Agroecosystems 72:67–76.

Mosier, A. R., D. Schimel, D. Valentine, K. Bronson, and W. Parton. 1991. Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. Nature 350:330–332.

Mueller, N. D., J. S. Gerber, M. Johnston, D. K. Ray, N. Ramankutty, and J. A. Foley. 2012. Closing yield gaps through nutrient and water management. Nature 490:254–257.

NRC (National Research Council). 2009. Liquid transportation fuels from coal and biomass: technological status, costs, and environmental impacts. National Academies Press, Washington, DC, USA.

Ostrom, N. E., R. Sutka, P. H. Ostrom, A. S. Grandy, K. H. Huizinga, H. Gandhi, J. C. von Fisher, and G. P. Robertson. 2010. Isotopologue data reveal bacterial denitrification as the primary source of N_2O during a high flux event following cultivation of a native temperate grassland. Soil Biology & Biochemistry 42:499–506.

Palm, C. A., and P. A. Sánchez. 1991. Nitrogen release from the leaves of some tropical legumes as affected by their lignin and polyphenolic contents. Soil Biology and Biochemistry 23:83–88.

Paul, E. A., A. Kravchenko, A. S. Grandy, and S. Morris. 2015. Soil organic matter dynamics: controls and management for ecosystem functioning. Pages 104–134 in S. K. Hamilton, J. E. Doll, and G. P. Robertson, editors. The ecology of agricultural ecosystems: long-term research on the path to sustainability. Oxford University Press, New York, New York, USA.

Paul, E. A., K. Paustian, E. T. Elliott, and C. V. Cole, editors. 1997. Soil organic matter in temperate agroecosystems: long-term experiments in North America. CRC Press, Boca Raton, Florida, USA.

Perlack, R. D., L. L. Wright, A. F. Turhollow, R. L. Graham, B. J. Stokes, and D. C. Erbach. 2005. Biomass as feedstock for a bioenergy and bioproducts industry: the technical feasibility of a billion-ton annual supply. Technical Report DOE/GO-102005–2135, U.S. Department of Energy, Washington, DC, USA.

Pimentel, D., and T. Patzek. 2005. Ethanol production using corn, switchgrass, and wood; biodiesel production using soybean and sunflower. Natural Resources Research 14:65–76.

Prinn, R. G. 2004. Non-CO_2 greenhouse gases. Pages 205–216 in C. B. Field and M. R. Raupach, editors. The global carbon cycle. Island Press, Washington, DC, USA.

Randerson, J. T., F. S. Chapin III, J. W. Harden, J. C. Neff, and M. E. Harmon. 2002. Net ecosystem production: a comprehensive measure of net carbon accumulation by ecosystems. Ecological Applications 12:937–947.

Rathke, G.-W., B. J. Wienhold, W. W. Wilhelm, and W. Diepenbrock. 2007. Tillage and rotation effect on corn-soybean energy balances in eastern Nebraska. Soil & Tillage Research 97:60–70.

Robertson, G. P. 1982. Regional nitrogen budgets: approaches and problems. Plant and Soil 67:73–79.

Robertson, G. P. 2004. Abatement of nitrous oxide, methane, and the other non-CO_2 greenhouse gases: the need for a systems approach. Pages 493–506 in C. B. Field and M. R. Raupach, editors. The global carbon cycle. Island Press, Washington, DC, USA.

Robertson, G. P. 2014. Soil greenhouse gas emissions and their mitigation. Pages 185–196 in N. K. Van Alfen, editor. The encyclopedia of agriculture and food systems. Academic Press, San Diego, California, USA.

Robertson, G. P., V. H. Dale, O. C. Doering, S. P. Hamburg, J. M. Melillo, M. M. Wander, W. J. Parton, P. R. Adler, J. N. Barney, R. M. Cruse, C. S. Duke, P. M. Fearnside, R. F. Follett, H. K. Gibbs, J. Goldemberg, D. J. Miadenoff, D. Ojima, M. W. Palmer, A. Sharpley, L. Wallace, K. C. Weathers, J. A. Wiens, and W. W. Wilhelm. 2008. Sustainable biofuels redux. Science 322:49–50.

Robertson, G. P., and P. M. Groffman. 2015. Nitrogen transformations. Pages 421–426 in E. A. Paul, editor. Soil microbiology, ecology, and biochemistry. Fourth edition. Academic Press, Burlington, Massachusetts, USA.

Robertson, G. P., and S. K. Hamilton. 2015. Long-term ecological research at the Kellogg Biological Station LTER Site: conceptual and experimental framework. Pages 1–32 in S. K. Hamilton, J. E. Doll, and G. P. Robertson, editors. The ecology of agricultural ecosystems: long-term research on the path to sustainability. Oxford University Press, New York, New York, USA.

Robertson, G. P., S. K. Hamilton, S. J. Del Grosso, and W. J. Parton. 2011. The biogeochemistry of bioenergy landscapes: carbon, nitrogen, and water considerations. Ecological Applications 21:1055–1067.

Robertson, G. P., E. A. Paul, and R. R. Harwood. 2000. Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. Science 289:1922–1925.

Robertson, G. P., and P. M. Vitousek. 2009. Nitrogen in agriculture: balancing the cost of an essential resource. Annual Review of Environment and Resources 34:97–125.

Sauerbeck, D. R. 2001. CO_2 emissions and C sequestration by agriculture perspectives and limitations. Nutrient Cycling in Agroecosystems 60:253–266.

Schmidt, T. M., and C. Waldron. 2015. Microbial diversity in agricultural soils and its relation to ecosystem function. Pages 135–157 in S. K. Hamilton, J. E. Doll, and G. P. Robertson, editors. The ecology of agricultural ecosystems: long-term research on the path to sustainability. Oxford University Press, New York, New York, USA.

Searchinger, T., R. Heimlich, R. A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes, and T.-H. Yu. 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319:1238–1240.

Searchinger, T. D., S. P. Hamburg, J. Melillo, W. L. Chameides, P. Havlik, D. M. Kammen, G. E. Likens, R. N. Lubowski, M. Obersteiner, M. Oppenheimer, G. P. Robertson, W. H. Schlesinger, and G. D. Tilman. 2009. Fixing a critical climate accounting error. Science 326:527–528.

Senthilkumar, S., B. Basso, A. N. Kravchenko, and G. P. Robertson. 2009. Contemporary evidence for soil carbon loss in the U.S. corn belt. Soil Science Society of America Journal 73:2078–2086.

Sheehan, J., V. Camobreco, J. Duffield, M. Graboski, and H. Shapouri. 1998. Life cycle inventory of biodiesel and petroleum diesel for use in an urban bus. National Renewable Energy Laboratory, Golden, Colorado, USA.

Six, J., E. T. Elliott, and K. Paustian. 2000. Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. Soil Biology & Biochemistry 32:2099–2103.

Smith, K. A., K. E. Dobbie, B. C. Ball, L. R. Bakken, B. K. Situala, S. Hansen, and R. Brumme. 2000. Oxidation of atmospheric methane in Northern European soils, comparison with other ecosystems, and uncertainties in the global terrestrial sink. Global Change Biology 6:791–803.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, and O. Sirotenko. 2007. Agriculture. Pages 498–540 in B. Metz, O. R. Davidson, P. R. Bosch, R. Dave, and L. A. Meyer, editors. Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK, and New York, New York, USA.

Snapp, S. S., R. G. Smith, and G. P. Robertson. 2015. Designing cropping systems for ecosystem services. Pages 378–408 in S. K. Hamilton, J. E. Doll, and G. P. Robertson, editors. The ecology of agricultural ecosystems: long-term research on the path to sustainability. Oxford University Press, New York, New York, USA.

Suwanwaree, P. 2003. Methane oxidation in terrestrial ecosystems: patterns and effects of disturbance. Dissertation, Michigan State University, East Lansing, Michigan, USA.

Suwanwaree, P., and G. P. Robertson. 2005. Methane oxidation in forest, successional, and no-till agricultural ecosystems: effects of nitrogen and soil disturbance. Soil Science Society of America Journal 69:1722–1729.

Syswerda, S. P., B. Basso, S. K. Hamilton, J. B. Tausig, and G. P. Robertson. 2012. Long-term nitrate loss along an agricultural intensity gradient in the Upper Midwest USA. Agriculture, Ecosystems & Environment 149:10–19.

Syswerda, S. P., A. T. Corbin, D. L. Mokma, A. N. Kravchenko, and G. P. Robertson. 2011. Agricultural management and soil carbon storage in surface vs. deep layers. Soil Science Society of America Journal 75:92–101.

Tilman, D., R. Socolow, J. A. Foley, J. Hill, E. Larson, L. Lynd, S. Pacala, J. Reilly, T. Searchinger, and C. Somerville. 2009. Beneficial biofuels—the food, energy, and environment trilemma. Science 325:270.

Tilman, D., C. Balzer, J. Hill, and B. L. Befort. 2011. Global food demand and the sustainable intensification of agriculture. Proceedings of the National Academy of Sciences USA 108:20260–20264.

USDA-FSA (U.S. Department of Agriculture-Farm Service Agency). 2014. Conservation programs. Statistics, CRP contract summary and statistics. Washington, DC, USA. http://www.fsa.usda.gov/FSA/webapp?area=home&subject=copr&topic=crp-st

van Kessel, C., R. Venterea, J. Six, M. A. Adviento-Borbe, B. Linquist, and K. J. van Groenigen. 2013. Climate, duration, and N placement determine N_2O emissions in reduced tillage systems: a meta-analysis. Global Change Biology 19:33–44.

Vermeulen, S. J., B. M. Campbell, and J. S. I. Ingram. 2012. Climate change and food systems. Annual Review of Environment and Resources 37:195–222.

West, T. O., and G. Marland. 2002. A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: comparing tillage practices in the United States. Agriculture, Ecosystems & Environment 91:217–232.

West, T. O., and W. M. Post. 2002. Soil organic carbon sequestration rates by tillage and crop rotation: a global data analysis. Soil Science Society of America Journal 66:1930–1946.

Wickings, K., A. S. Grandy, S. C. Reed, and C. C. Cleveland. 2012. The origin of litter chemical complexity during decomposition. Ecology Letters. 15:1180–1188.

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. Agronomy Journal 99:1665–1667.

Wilson, A. T. 1978. The explosion of pioneer agriculture: contribution to the global CO_2 increase. Nature 273:40–41.

Zenone, T., J. Chen, M. W. Deal, B. Wilske, P. Jasrotia, J. Xu, A. K. Bhardwaj, S. K. Hamilton, and G. P. Robertson. 2011. CO_2 fluxes of transitional bioenergy crops:

effect of land conversion during the first year of cultivation. Global Change Biology-Bioenergy 3:401–412.

Zhang, X., R. C. Izaurralde, D. Manowitz, T. O. West, W. M. Post, A. M. Thompson, V. P. Bandaru, J. Nichols, and J. R. Williams. 2010. An integrative modeling framework to evaluate the productivity and sustainability of biofuel crop production systems. Global Change Biology Bioenergy 2:258–277.

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