

INTERRELATIONSHIP OF THE NITROGEN CYCLE AND CARBON SEQUESTRATION: GREENHOUSE GAS MITIGATION IN ROW CROPS

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Abstract

Agriculture's contribution to global warming is principally through its historical release of carbon in soil and vegetation to the atmosphere and through its contemporary release of nitrous oxide and methane. The sequestration of soil carbon in soils now depleted in soil organic matter is a well-known strategy for mitigating the buildup of CO₂ in the atmosphere. Less well-recognized are other mitigation potentials such as better management of nitrogen fertility. A full-cost accounting of the effects of agriculture on greenhouse gas emissions can show the relative importance of all mitigation options. Such an analysis shows nitrogen fertilizer, agricultural liming, fuel use, nitrous oxide emissions, and methane fluxes to have additional significant potential for mitigation. By evaluating all sources in terms of their global warming potential (GWP) it is possible to directly evaluate greenhouse policy options for agriculture.

Introduction

Potentials for reducing the buildup of greenhouse gases in the atmosphere via soil carbon sequestration have received widespread attention in the past 5 years, and have recently led to the initiation of carbon credit markets (e.g. McCarl and Schneider, 2001; CAST, 2004). Most attention has focused on no-till agriculture (e.g. Lal, 1999) because of its capacity in many cropping systems to build soil carbon (C) towards levels that existed prior to agricultural conversion (Paul et al., 1997). Other means for sequestering soil carbon have also been suggested, however, including cover crops, natural fallows that remove land from cultivation for a period of time, and fertilization to increase residue inputs.

While the focus on soil carbon and in particular on no-till cultivation has been useful for stimulating policy discussions, in some respects it is short-sighted. First, there are other potentials for mitigating greenhouse gas emissions that are commonly overlooked in discussions of policy options. These other potentials can be as or more effective than soil carbon capture in many systems, and may be especially suitable for regions and cropping systems for which no-till agriculture is agronomically unsuitable or economically prohibitive. For example, no-till is obviously unsuitable for root crops such as potatoes and sugar beets, and where soil pathogens persist in the absence of soil disturbance. Moreover, very little U.S. cropland is in permanent no-till, and it is not yet clear whether "partial no till" sequesters significant carbon.

Second, changes in tillage practices may have unanticipated and unwanted effects on other sources or sinks of greenhouse gases. If, for example, increased soil moisture associated with no-till were to stimulate nitrifying and denitrifying bacteria as well as plants, then production of the greenhouse gas nitrous oxide (N₂O) might increase, offsetting some or all of the mitigation potential of carbon storage (Robertson, 1999).

Third and finally, managing systems specifically for soil carbon storage by boosting the production of crop residues to enhance soil organic matter inputs can be counterproductive. In particular, if greenhouse-gas generating inputs are used to stimulate residue production (if yield increases are not the primary goal), then the mitigation gained with such production can be more than offset by the greenhouse costs of that production (Schlesinger, 1999). Carbon dioxide released during fertilizer manufacture and during the electric power generation for irrigation pumps are examples of such offsetting practices (cf. Izurralde et al., 2000).

The need to include all sources of greenhouse warming potential in cropping systems is acute – without a complete analysis of a cropping system’s capacity to affect the radiative forcing of the atmosphere, it is difficult to judge whether one mitigation strategy is better than another (or better than none at all). It is also very easy to overlook additional mitigation options that may be particularly well suited to specific cropping systems or regions, especially for those in the developing tropics.

Global Warming Potential

Global Warming Potential (GWP) provides a means for comparing the relative impact of one source or sink of greenhouse gas against another. By placing all fluxes in common terms it is possible to compare the relative cost of, for example, increased carbon storage due to residue production (GWP mitigation) against increased N₂O from additional fertilizer application (GWP source).

By convention, GWP is measured in CO₂- equivalents (IPCC, 1996, 2001). Conversions from other gases to CO₂ are based on the effect of a particular gas on the radiative forcing of the atmosphere relative to CO₂’s effect. GWP is largely a function of a molecule’s ability to capture infrared radiation, its current concentration in the atmosphere, the concentration of other greenhouse gases, and its atmospheric lifetime. All else being equal, a gas molecule with a greater atmospheric lifetime will have a higher GWP than one that cycles rapidly. For example, N₂O is long-lived relative to CH₄. Thus the 100 y N₂O GWP (296 CO₂-equivalents) is not much different from its 20 y GWP (275 CO₂-equivalents), whereas the GWP for methane falls off rapidly over this period, from 62 to 23 CO₂-equivalents. Likewise, relatively novel molecules with high IR capture capacities will have higher GWPs. Sulfur hexafluoride (SF₆), for example, has a 100-year GWP that is 22,200 times that of CO₂ owing to its radiative properties, its novelty in the atmosphere, and an atmospheric lifetime of 3200 y (Table 1).

Table 1. Global warming potentials (GWPs) of greenhouse gases in agriculture (IPCC, 2001).

Greenhouse Gas	Atmospheric Lifetime (years)	20-Year GWP	100-Year GWP
Carbon dioxide (CO ₂)		1	1
Methane (CH ₄)	12	62	23
Nitrous oxide (N ₂ O)	114	275	296

In general, only three greenhouse gases are affected by agriculture: CO₂, N₂O, and methane (CH₄). Although CH₄ and especially N₂O are at far lower atmospheric concentrations than CO₂, their GWPs are sufficiently high that small changes have a disproportionate effect on radiative forcing (Table 1). Over a 100-year time horizon, the time period that is used for national greenhouse gas inventories, the GWP of methane is 23 while that of nitrous oxide is 296; this means that a molecule of contemporary N₂O released to the atmosphere will have about 300 times the radiative impact of a molecule of CO₂ released at the same time. Thus, an agronomic activity that reduces N₂O emissions by 1 kg ha⁻¹ is equivalent to an activity that sequesters 296 kg ha⁻¹ CO₂ as soil carbon.

Sources of Global Warming Potential in Row Crops

Sources of GWP arise from a number of agronomic practices. Some, such as soil CO₂ emission following clearing and plowing and such as CO₂ emitted by diesel farm machinery, are direct sources of CO₂. Others, such as CO₂ emitted during fertilizer and pesticide manufacture, are indirect. Still others, such as CH₄ emitted by livestock and N₂O produced by soil bacteria, are non-CO₂ based. All must be considered when calculating the total contribution of agriculture to global warming.

Mitigation occurs when existing sources of GWP are reduced. A GWP of zero means that no net GWP is attributable to a particular cropping system or agronomic practice. A negative GWP implies mitigation, but mitigation only occurs when GWP is less than the GWP of the pre-existing cropping condition – regardless of whether the pre-existing, business-as-usual condition was net positive or net negative.

Specific sources of GWP in modern cropping systems include soil carbon change, nitrogen fertilizer, lime, irrigation, fuel use, nitrous oxide, and methane.

Soil Carbon Change

Conversion of natural ecosystems to agriculture releases substantial CO₂ to the atmosphere. The release of CO₂ from cleared vegetation that is burned or left to decompose is one of the most well-documented and important sources of the atmospheric CO₂ increase (e.g. IPCC, 2002). Historically, land clearing (biomass burning) has been a major contributor to atmospheric CO₂ loading; today it still accounts for about 25% (1.6 Gt C y⁻¹) of the total global CO₂ loading, which includes another 6.3 Gt C y⁻¹ from fossil fuel use and cement production (IPCC 2002). Almost all of the contemporary flux is from land clearing in tropical regions.

Soil carbon is also lost upon agricultural conversion. Forests and savannahs newly cultivated usually lose a substantial fraction of their original carbon content in the decades following initial cultivation (Figure 1). This occurs for a number of reasons: reduced plant residue inputs, tillage-induced soil disturbance, erosion, and the creation of more favorable conditions for microbial decomposition (CAST, 2004). Generally soil carbon contents stabilize at 40-60% of original pre-cultivation values: the new equilibrium state is a function of climate, soil physical and chemical characteristics, and agronomic management factors such as tillage, crop types and cover, and residue management (Robertson and Paul, 2000).

Because soil represents about 80% of the carbon stocks in terrestrial ecosystems (ranging from 50% in tropical forests to 95% in tundra; IPCC 2002), the global impact of soil carbon loss due to agriculture is considerable. Recent estimates suggest that 50-100 Gt C (CAST, 2004; Smith, 2004) have been lost from soils in the past few hundred years, although higher estimates range to 142 Gt C (Lal, 1999).

Soils can also gain carbon. The soil carbon balance is the net difference between carbon inputs from plant roots and aboveground litter (that remaining after harvest or fire), and carbon loss from microbial respiration and erosion. In agricultural systems, manure and compost can represent additional inputs. Because erosion repositions carbon in the landscape rather than converts it to CO₂, erosion is not in itself a source of GWP. Microbial respiration, on the other hand, is a major source of GWP – where respiration is slowed, as in no-till systems, carbon can accumulate at slow but significant rates to some new equilibrium (Paustian et al., 1997).

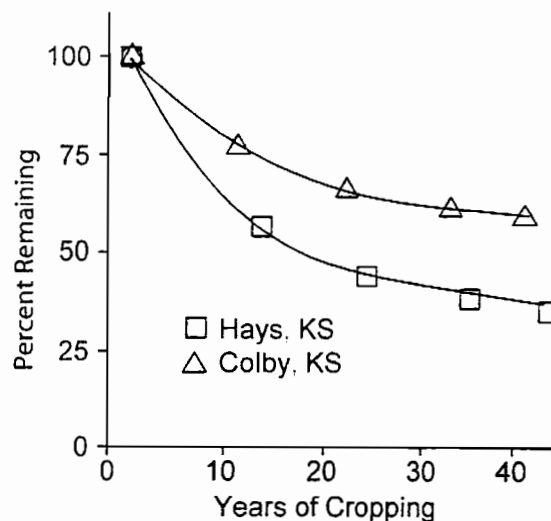


Figure 1. Soil organic matter loss following cultivation at two sites (Hays and Colby, Kansas) in the Midwestern U.S. Redrawn from Haas et al. (1957).

Estimates of historic soil C loss provide a reference point for carbon sequestration potentials. Models suggest that 60-80% of the soil carbon lost as CO₂ could be regained under no-till conditions over a period of 50 years (IPCC, 1996); if this is the case, then as much as 60-85 Gt C could be regained by agricultural soils at a rate of about 1.1 to 1.7 Gt C y⁻¹. In soils of the U.S. Midwest, the median rate of annual carbon gain under no-till is 30 g C m⁻² (Franzleubbers and Steiner, 2002), which is equivalent to a GWP of -110 g CO₂-equivalents m⁻² y⁻¹.

Nitrogen Fertilizer

The Haber-Bosch process for producing fertilizer nitrogen results in the production of 0.375 moles of CO₂ per mole of N produced at 100% efficiency (Schlesinger, 1999); at normal efficiencies a mole of N is manufactured at a cost of about 0.58 moles of CO₂ (IPCC 1997). Additional CO₂ produced during the processing, transport, and application of N fertilizer pushes this value to around 1.4 moles of CO₂ released per mole of N applied (Schlesinger, 1999, Izaurralde et al., 1999).

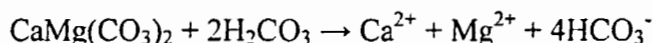
Nitrogen fertilizer is thus a significant source of GWP in cropping systems fertilized with synthetic nitrogen. Worldwide, nitrogen fertilizer use is increasing rapidly, especially in developing regions except for subsaharan Africa). Rates of nitrogen application vary widely, largely as a function of market availability, crop value, and national subsidies; typical rates in developed regions range from 50 kg N ha⁻¹ for wheat to 200 kg N ha⁻¹ for maize. For every 100 kg N ha⁻¹ that is applied, the GWP cost is 45 g CO₂-equivalents m⁻².

Agricultural Lime

Agricultural lime in the form of calcium carbonate (CaCO_3) and dolomite ($\text{CaMg}(\text{CO}_3)_2$) are commonly applied to agricultural soils to counteract soil acidity. These carbonate minerals are mined from geologic reservoirs, ground, and applied to agricultural soils in humid regions at several-year intervals.

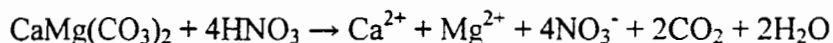
In regions where lime is inexpensive and readily available, lime requirements are estimated based on simple soil tests and generalized relationships, and generally strive to maintain soil pH in the range of 5.5 – 6.5 (Coleman and Thomas, 1967; van Lierop 1990). Most varieties of alfalfa (*Medicago sativa*), for example, perform best at pH >6.5. In addition to raising soil pH, liming also supplies Ca^{2+} and Mg^{2+} for plant uptake. Additionally, in highly weathered soils the precipitation of reactive Al^{3+} by reaction with lime ($2\text{Al}^{3+} + 3\text{CaCO}_3 + 6\text{H}_2\text{O} \rightarrow 3\text{Ca}^{2+} + 2\text{Al}(\text{OH})_3 + 3\text{H}_2\text{CO}_3$) can be as or more beneficial than raising soil pH per se.

Lime applications to neutralize undesirable acidity are commonly in the range of 5-10 Mg ha⁻¹, and re-application is usually required every few years as the pH drops following fertilizer additions, nitrification, plant harvest, and leaching. As lime dissolves, the fate of its carbon is difficult to predict. Carbonic acid formed in the presence of CO_2 from root and microbial respiration reacts with solid carbonates to consume H^+ . For dolomite:



In this case lime weathers to bicarbonate, which is then leached out of the soil profile.

If, on the other hand, carbonate comes into contact with a strong mineral acid such as HNO_3 (nitric acid), the end product will be CO_2 rather than bicarbonate:



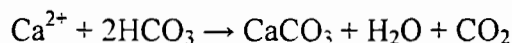
Nitric acid is formed by nitrifying bacteria in most soils (Robertson, 1982) including acid tropical soils (Sollins et al., 1988). Added lime thus seems likely to be a source of CO_2 where it is applied, but no data is available today to verify this.

The GWP of every metric ton of CaCO_3 added to soil is thus -44 to +44 g CO_2 -equivalents m^{-2} ; likewise, the GWP for MgCO_3 is -52 to +52 g CO_2 -equivalents m^{-2} .

Irrigation

The GWP of irrigation is the result of fuel use during pumping and of carbonate reactions when calcium-saturated groundwater is sprayed on calcareous surface soils (Schlesinger, 1999). For irrigated land in the United States, the fossil fuel cost of pumping totals 22 to 83 g C $\text{m}^{-2} \text{y}^{-1}$ (Maddigan et al., 1982), equivalent to a GWP of 81 to 304 g CO_2 -equivalents m^{-2} .

In arid regions groundwater often contains as much as 1% Ca and CO_2 . When this water reaches the surface, the CO_2 (at 10,000 ppm_v) equilibrates with atmospheric CO_2 (365 ppm_v): CaCO_3 precipitates and CO_2 is released to the atmosphere:



Schlesinger (1999) uses the average water use efficiency of arid-land plants to estimate that the net CO₂ released from the formation of soil carbonate due to irrigation would be 8.4 g C m⁻² annually. This represents a GWP of 31 g CO₂-equivalents m⁻² y⁻¹.

Fuel Use

Diesel (C₁₆H₃₄) is 85% C, almost all of which is oxidized to CO₂ when burned. Agronomic activities that are mechanized, including plowing, cultivating, hoeing, spraying, planting, baling, chopping, and harvesting, exact a CO₂ cost. Every liter of fuel (at a density of 832 g L⁻¹) releases 706 g C. Annual fuel use of 100 L ha⁻¹ would thus have a GWP of 26 g CO₂-equivalents m⁻².

Nitrous Oxide (N₂O)

Nitrous oxide is produced during nitrification and denitrification in agricultural soils. During nitrification ammonium is converted to nitrite (NO₂⁻) and then to nitrate (NO₃⁻) by aerobic autotrophic bacteria collectively known as nitrifiers; N₂O is a minor byproduct. Denitrification is a soil microbial process in which nitrate is converted to dinitrogen gas (N₂) by heterotrophic, facultatively anaerobic bacteria collectively known as denitrifiers; N₂O is a requisite intermediate that under some environmental conditions and for some denitrifier taxa is the end product (Cavigelli and Robertson, 2000).

Nitrification occurs whenever soil ammonium is available and environmental conditions such as temperature and moisture are favorable for nitrifier activity, which in many agronomic situations prevail most of the time (Robertson, 1982). Denitrification occurs whenever soil carbon and nitrate are available and oxygen is in short supply – denitrifiers can use nitrate rather than oxygen as a terminal electron acceptor if oxygen is unavailable (Robertson, 2000). This occurs in wet soils when diffusion of oxygen to microsites is slowed by saturated conditions, and inside soil aggregates in even well-drained soils. In the center of aggregates oxygen demand is often greater than can be provided by diffusion through the aggregate from the surrounding soil atmosphere (Sexstone et al., 1985).

Nitrous oxide can also be produced from livestock waste, though only when stored under relatively aerobic conditions such as in compost heaps. Under anaerobic conditions, as in waste lagoons, nitrification is inhibited by lack of oxygen and denitrification by the consequent lack of nitrate; further, any nitrate that is available tends to be denitrified all the way to N₂ rather than stop at N₂O (CAST, 2004) due to the low availability of electron acceptors.

Of all the sources of GWP in cropping systems, none are more poorly quantified than N₂O production. This is mainly because of the difficulty with which N₂O fluxes are measured. Unlike for CO₂ and methane, N₂O flux is not suited to micrometeorological measurement (Holland et al., 1999); rather fluxes must be measured using small chambers placed on the soil surface for 1-2 hour intervals. High temporal and spatial variability means that many chambers must be deployed simultaneously at weekly or more frequent intervals in a given cropping system; sampling and analysis costs are thus high.

However, for the few cropping systems for which we have reliable N_2O fluxes, N_2O loss is frequently the major source of GWP. Robertson et al. (2000), for example, found for a 9-year measurement campaign in several annual and perennial cropping systems in Michigan that N_2O was the single greatest source of GWP in all four of their annual crop systems, ranging from 50 to 60 g CO_2 -equivalents $m^{-2} y^{-1}$. IPCC methodology assumes that 1.25% of nitrogen inputs to most cropping systems is subsequently emitted as N_2O -N; if true, then for every 100 kg N ha^{-1} applied as fertilizer, about 1.25 kg N will be emitted as N_2O , for a GWP (over a 100-year time horizon) of 58 g CO_2 -equivalents $m^{-2} y^{-1}$.

Soil nitrogen availability appears to be the single best predictor of N_2O flux in most terrestrial ecosystems including agricultural (Figure 2). Any activity or process that acts to keep available soil nitrogen low should thus lead to smaller N_2O flux. Plant demand for nitrogen is therefore one of the most important determinants of N_2O flux, and more precise application of N-fertilizer — to maximize plant uptake of added N both spatially and temporally — may be one of the best means available for mitigating current N_2O fluxes from agriculture.

Methane

Methane is produced by anaerobic bacteria in soil, animal waste, and ruminant stomachs, and agricultural sources of methane are a significant fraction of the global methane budget. About 15% of the 598 Tg global CH_4 flux is from lowland rice systems, and another 15% is from enteric fermentation during livestock digestion (Hein et al., 1997; IPCC 2001). Because methanogenesis is a strictly anaerobic process, under normal conditions upland cropping systems are not a direct source of methane, and methane flux in paddy rice can be partly mitigated through water level and residue management and cultivar selection (Mosier et al., 1998).

Methane is also consumed, but by a different class of soil bacteria called methanotrophs, and methane consumption in soils is a small but significant part of the global methane budget, comparable in magnitude to the annual atmospheric increase in methane. In rice paddies and wetlands the total methane flux is the net difference between methogenesis in submerged anaerobic horizons and methane consumption at or above the soil-water interface. In upland soils including field crops the net flux appears to be largely a function of methane consumption. Agricultural conversion tends to reduce natural rates of methane consumption in soils by a factor of 5-10 (Bronson and Mosier, 1993; Smith et al., 2000), and at our current state of knowledge there is no known way to restore consumption other than allowing natural revegetation.

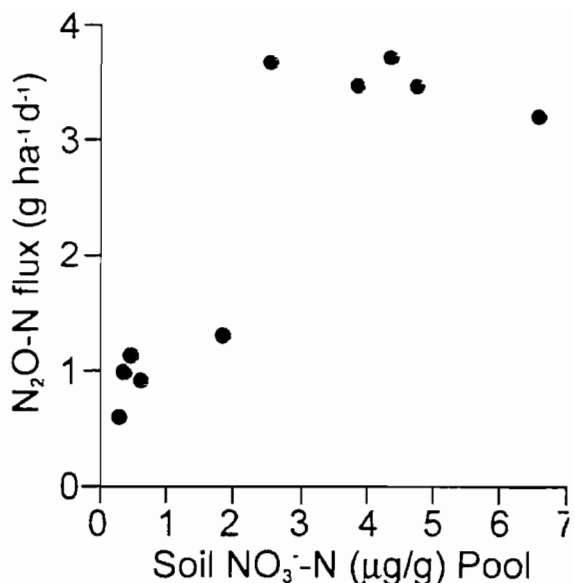


Figure 2. Annual average nitrous oxide fluxes vs. soil nitrate pools in different crop and unmanaged ecosystems on the same soil series in Michigan (calculated from Robertson et al., 2002).

Consumption rates are not much affected by fertilization, organic management, or tillage. By reducing a natural source of mitigation, agriculture thus creates an indirect source of GWP.

Robertson et al. (2000) found for a U.S. Midwest landscape that the GWP of methane oxidation in old-growth forest was $-25 \text{ g CO}_2\text{-equivalents m}^{-2} \text{ y}^{-1}$; for various cropping systems on the same soil type they found GWPs ranging from -4 to $-6 \text{ g CO}_2\text{-equivalents m}^{-2} \text{ y}^{-1}$. Similar changes have been documented for a variety of soil and climates (e.g. Smith et al., 2000), including tropical (Keller and Reiners, 1994).

A Cropping System Example

There are as yet very few cropping systems for which all significant sources of GWP have been measured. Table 2 presents GWP values for three Midwest U.S. cropping systems and a late successional forest on the same Michigan soil type (Robertson et al., 2000). In the conventional tillage system N_2O is the principal source of GWP, accounting for 52 of the system's total of $114 \text{ g CO}_2\text{-equivalents m}^{-2} \text{ y}^{-1}$. Contributions of N-fertilizer ($60 \text{ kg N ha}^{-1} \text{ y}^{-1}$ on average) and agricultural lime were each about half of N_2O 's contribution to total GWP (23 to $27 \text{ g CO}_2\text{-equivalents m}^{-2} \text{ y}^{-1}$), and fuel use accounted for about half again of this ($16 \text{ g CO}_2\text{-equivalents m}^{-2} \text{ y}^{-1}$). Because soil C in this system was equilibrated (at about 1% C), soil carbon did not contribute to GWP; likewise, methane oxidation contributed very little mitigation capacity, about $-4 \text{ g CO}_2\text{-equivalents m}^{-2} \text{ y}^{-1}$.

Table 2. Sources of global warming potential (GWP) in a maize – soybean – wheat cropping system of the U.S. Midwest based on 9 years of measurements (Robertson et al., 2000). The system had been cropped for decades earlier, depleting soil organic matter to 1 % C. N_2O and CH_4 fluxes are measured. N fertilizer was added only to maize (120 kg N ha^{-1}) and wheat (60 kg N ha^{-1}) crops. Only wheat residue was removed. GWPs for N_2O and CH_4 are based on 20-year time horizons using IPCC (1996) values of 280 and 56, respectively (cf. Table 1).

System	CO ₂						Net GWP
	Δ Soil C	N Fertilizer	Lime	Fuel	N ₂ O	CH ₄	
	GWP (g CO ₂ -equivalents m ⁻² y ⁻¹)						
Conv. till	0	27	23	16	52	-4	114
No-till	-110	27	34	12	56	-5	14
Organic	-29	0	0	19	56	-5	41
Forest	0	0	0	0	21	-25	-4

GWP values in the no-till system were equivalent to those in the conventional system for most sources of GWP except soil C, lime, and fuel. In the no-till system soil carbon had accumulated at $30 \text{ g m}^{-2} \text{ y}^{-1}$, providing a GWP mitigation of $-110 \text{ g CO}_2\text{-equivalents m}^{-2} \text{ y}^{-1}$. Slightly lower fuel costs ($12 \text{ g CO}_2\text{-equivalents m}^{-2} \text{ y}^{-1}$) were offset by somewhat higher lime requirements (34

g CO₂-equivalents m⁻² y⁻¹), providing a net GWP for this system of 14 g CO₂-equivalents m⁻² y⁻¹, substantially lower than the 114 g CO₂-equivalents m⁻² y⁻¹ GWP of the conventional tillage system.

The organic system, which used legumes rather than synthetic fertilizer for nitrogen inputs, was midway between the conventional and no-till systems, having a net systemwide GWP of 41 g CO₂-equivalents m⁻² y⁻¹. N₂O flux was the same as in the other systems, but there were neither N fertilizer nor lime inputs in this system, and carbon accumulated at a moderate rate (8 g m⁻² y⁻¹) owing to cover crop residue. The forested system, in contrast, had a net GWP of -4 g CO₂-equivalents m⁻² y⁻¹ – soil carbon neither accumulated nor disappeared, there were no agronomic inputs, and the GWP of N₂O flux (1.3 g N₂O-N ha⁻¹ d⁻¹ on average for a GWP of 21 g CO₂-equivalents m⁻² y⁻¹) was counterbalanced by the GWP of CH₄ oxidation (-9.2 g CH₄-C ha⁻¹ d⁻¹ on average for a GWP of -25 g CO₂-equivalents m⁻² y⁻¹).

This type of analysis is extremely valuable from a policy and management perspective because it shows that in these systems mitigation could achieve even greater GWP savings. Cropping systems could benefit substantially from efforts to mitigate N₂O production, for example; these efforts could take the form of better nitrogen conservation by basing fertilizer rates on seasonal soil and plant tests, by applying fertilizers closer to the time of crop uptake, by planting cover crops to remove nitrogen from the soil solution during the non-growing season, and possibly by using nitrification inhibitors (CAST, 2004). Significant savings could also result from using nonsynthetic N fertilizers such as leguminous cover crops or manure; by reducing diesel use with either biogas production or better mechanical efficiencies; by managing soil acidity to reduce the need for lime applications; and by using cover crop and other residue management strategies in addition to no-till to increase soil carbon.

In all of these cases a whole-system analysis also serves to identify the value and costs of trade-offs. Such analysis might show, for example, that soil C gained by adding manure or compost to soil might be offset by a concomitant increase in soil N₂O flux. Only a full-cost, whole system GWP accounting can fully calculate the true net value of various cropping strategies for greenhouse gas mitigation.

Acknowledgements

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