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The Nature and Longevity of Agricultural Impacts on Soil Carbon and Nutrients: A Review

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Abstract

Since the domestication of plant and animal species around 10,000 years ago, cultivation and animal husbandry have been major components of global change. Agricultural activities such as tillage, fertilization, and biomass alteration lead to fundamental changes in the pools and fluxes of carbon (C), nitrogen (N), and phosphorus (P) that originally existed in native ecosystems. Land is often taken out of agricultural production for economic, social, or biological reasons, and the ability to predict the biogeochemical trajectory of this land is important to our understanding of ecosystem development and our projections of food security for the future. Tillage generally decreases soil organic matter (SOM) due to erosion and disruption of the physical, biochemical, and chemical mechanisms of SOM stabilization, but SOM can generally reaccumulate after the cessation of cultivation. The use of organic amendments causes increases in SOM on agricultural fields that can last for centuries to millennia after the termination of applications, although the locations that provide the organic amendments are concurrently depleted. The legacy of agriculture is therefore highly variable on decadal to millennial time scales and depends on the specific management practices that are followed during the agricultural period. State factors such as climate and parent material (particularly clay content and mineralogy) modify ecosystem processes such that they may be useful predictors of rates of postagricultural biogeochemical change. In addition to accurate biogeochemical budgets of postagricultural systems, ecosystem models that more explicitly incorporate mechanisms of SOM loss and formation with agricultural practices will be helpful. Developing this predictive capacity will aid in ecological restoration efforts and improve the management of modern agroecosystems as demands on agriculture become more pressing.

Key words: agriculture; clay content; manure; nitrogen; persistence; phosphorus; reversibility; soil organic carbon; sustainability; tillage.

Introduction

Cultivated areas currently comprise 12% of the total land surface of Earth, making agriculture a pervasive feature of the planet (Ramankutty and Foley 1998; Leff and others 2004). Cropping systems and tightly integrated livestock husbandry systems have been providing grain and meat for human consumption since plants and animals were domesticated 10,000 years ago (Smith 1995; Diamond 2002). Ancient and modern agricultural systems strongly influence ecosystem processes such as carbon (C) and nutrient cycling as well as disturbance regimes, plant cover, primary productivity, and water use and therefore are of direct interest to ecosystem ecologists (Matson and others 1997). Hans Jenny, who founded modern

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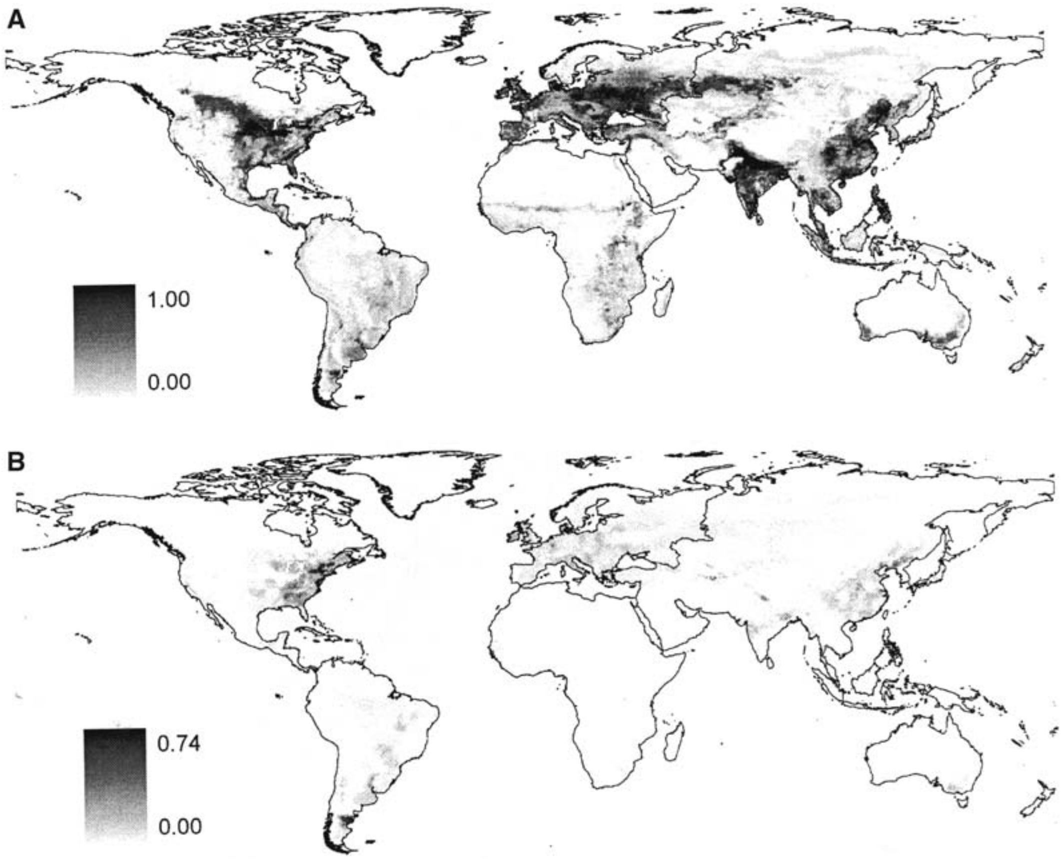

Figure 1. A: Global distribution of maximum cropland extent, expressed as the fraction of land in a 0.5° grid cell. This was calculated as the greatest fraction of land occupied by cultivation during the period 1700–1992. **B**: Global distribution of abandoned cropland area, expressed as the fraction of land in a 0.5° grid cell. This was calculated as the difference between the maximum cropland extent during 1700–1992. and the cropland extent in 1992, with data from Ramankutty and Foley (1999).

pedology, considered agricultural activity to be so influential for the development of soil that he included humans as a soil-forming factor (Jenny 1980). The International Society of Soil Science recognizes Anthrosols as a reference soil group characterized by the presence of an anthropedogenic horizon that ''comprise[s] a variety of surface and subsurface horizons which result from long-continued cultivation'' (FAO/ISSS/ISRIC 1998).

Agriculture, defined as crop plant production systems, has enormous consequences for soils and ecosystems that outlast the duration of agricultural activity. The effect of tillage on soil organic C levels has been well defined, but not much is known about either the legacy of this effect or the concurrent effects on soil nutrients. A legacy in this context is a characteristic or property that is attributable to prior agriculture. Soils can be described as being ''resilient'' or ''resistant'' to agricultural perturbation, where a resilient system changes in response to agriculture but quickly returns to its initial condition, and a resistant soil requires a severe perturbation to exhibit change but may not return to its initial condition (Schimel and others 1985). Despite ubiquitous agricultural legacies, we have limited capabilities for incorporating land-use history into modern ecological studies in a quantitative way (Foster and others 2003) and for determining the future consequences of modern land use.

The cessation of agriculture provides an opportunity to study agricultural legacies. Although some areas or farms are cultivated for centuries, others are removed from agriculture for a variety of reasons, including economics, degradation the of land, political and social changes, technological changes, biological changes in crop varieties, and local depopulation (Burgi and Turner 2002). The area of cultivated land has recently been expanding on a global scale, but, agricultural abandonment has occurred in the past, is occurring now, and will continue to occur in the future (Figure 1). For example, cropland area is currently decreasing in most European countries (Smith and others 2005b).

Four components of agricultural land use have the potential to alter soil and ecosystem processes: biomass alterations, tillage, fertilization, and altered hydrology. These four types of management have been practiced since close to the beginning of crop plant domestication. Agriculture alters plant biomass both by replacing natural vegetation with monocultures of annual crop plants and by removing biomass through harvest, and sometimes even in fallow periods. Tillage is a unique type of belowground disturbance that has immense physical effects via the inversion and pulverization of the topsoil. Fertilization increases inputs of various nutrients, either in organic or inorganic states. Organic sources are characterized by the presence of C in addition to the nutrients, whereas inorganic sources lack C. Finally, altered hydrological regimes such as irrigation or drainage are common agricultural practices but will not be considered in detail here. Irrigation can cause irreversible soil damage through siltation and salinization, both of which occurred in ancient Mesopotamia and continue to occur today (Jacobsen and Adams 1958; Hillel 1991), whereas drainage exposes organic soils to oxidation. These four types of agricultural management are very different mechanisms, but all of them cause long-term changes to soils and ecosystems.

There are two time periods that must be considered when we examine the long-term consequences of agriculture: the period during agriculture and the period after agriculture. First, the type of agricultural management that is practiced plays a large role in determining the content and type of soil organic matter (SOM), rates of nutrient cycling, and primary productivity in agricultural versus native ecosystems. Second, after the cessation of agriculture, some systems or properties attain preagricultural characteristics within decades, whereas others remain altered for millennia. Thus, the longevity of agricultural impacts can be measured.

Two types of studies are particularly useful for of gaining insight into the duration of agricultural effects on ecosystems: (a) long-term monitoring of agricultural and postagricultural fields, and (b) space-for-time substitution, where a reference system is chosen to represent preagricultural conditions. Thus, the studies chosen for this review generally describe the vegetation or ecosystem functioning of postagricultural systems, investigate the capacity of agroecosystems for continuous long-term cultivation, or provide insight from modern agroecosystems into recovery processes. These studies have been scattered geographically around the globe and intellectually across disciplines including paleoecology, ecosystem ecology, soil science, archaeology, and agronomy. Most studies adopt a time frame of decades to millennia and follow an empirical approach; there are few theoretical studies that have attempted to provide a conceptual framework. Although a complete understanding of ecosystem change during and after agriculture is currently lacking, a synthesis of some interesting patterns is nonetheless possible.

The first objective of this review was to determine how past agricultural activities alter ecosystem processes, particularly the direction and magnitude of postagricultural soil change. A few ecosystem characteristics, specifically soil organic C (SOC), soil nitrogen (N), and soil phosphorus (P), were chosen to highlight particular changes in soil properties due to agriculture. The second objective was to summarize the longevity of past agricultural impacts from an empirical perspective. Several factors that could explain the observed variation in longevity are considered, including differences in agricultural management, climate, and soil texture. The final objective was to build a biogeochemical conceptual framework for predicting rates of change due to the commencement and cessation of agriculture. If we understand the mechanisms by which past and present agriculture influence elemental pools and fluxes, and the magnitude of this influence, we should be able to predict the future consequences of current agricultural activity.

Types of Ecosystem Responses to Agricultural Activity

Soil Organic Carbon Content

Soil organic C content in agricultural soils has been of interest for several reasons. Carbon can be transferred from the soil to the atmosphere as a result of agricultural processes. These emissions have contributed approximately 55 Pg to anthropogenic increases in atmospheric (carbon dioxide) and changes in the global C balance (Amundson 2001; Post and others 2004). Net C flux budgets for agriculture that incorporate a full accounting of emissions and sequestration are beginning to show that modern agricultural land can function either as a net source or a sink of C, and it is important to clarify these budgets further (West and Marland 2003). Additionally, SOC is an important component of soil quality, and it regulates soil moisture and structure, nutrient supply rates, and microbial activity.

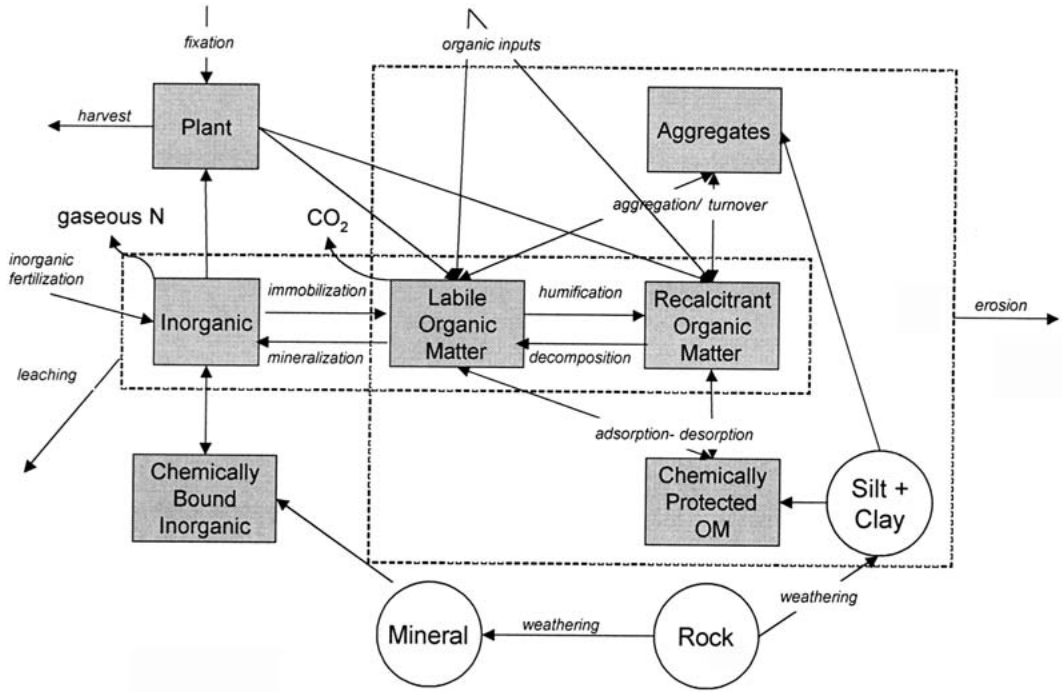

Figure 2. Conceptual framework of the mechanisms that determine agricultural influence on carbon (C), nitrogen (N), and phosphorus (P) in ecosystems. The diagram is general enough to encompass the C, N, and P cycles, with the understanding that the magnitude and existence of some pathways will be different for each element. Boxes with solid lines indicate pools that contain C, N, and P. Circles indicate other pools that influence the function of C, N, and P. Consequences of agricultural management practices described in the text include erosion, addition of inorganic and organic fertilizers, leaching, harvest, and increases in respiration of carbon dioxide (CO_2). For a more detailed discussion of mechanisms of organic matter protection, see Six and others (2002).

Soil organic C is a heterogeneous mixture of organic compounds in the soil that has an array of turnover times. It can generally be grouped into ''labile'' and ''recalcitrant'' forms. Theories of SOC formation emphasize the difference between C inputs and outputs, with several mechanisms providing SOC stabilization (Oades 1988; Ekschmitt and others 2005). Carbon-containing compounds that are particularly resistant to microbial decomposition, such as charcoal, lignin, or highly humified substances, are stabilized because of their biochemical recalcitrance (Krull and others 2003). Clay minerals directly stabilize SOC onto or inside their structures, although other compounds such as iron and aluminum oxides also chemically stabilize SOC (Percival and others 2000; Osher and others 2003). Some SOC is protected from decomposition by the physical structure of soil, particularly aggregates (Tisdall and Oades 1982). The relative magnitude of these mechanisms in different soils provides a framework for determining the rates of loss and accumulation of SOC with different agricultural practices (Figure 2).

Agriculture Depletes Soil Organic Carbon. According to the dominant paradigm, cultivation causes an immediate and rapid loss of SOC (Davidson and Ackerman 1993). The loss of SOC continues for several decades after the implementation of cultivation, reducing SOC pools on agricultural soils to 70% of their original levels on average (Mann 1986). These patterns of reduced SOC occur regardless of climate, soil type, and original vegetation, although they have been observed most commonly for temperate North American agriculture in the 20th century. Generally, the rate of loss slows as SOC levels reach a new equilibrium that depends on tillage practice (West and Post 2002) and the level of C inputs returned to the soil as crop residue or animal manures (Kirchmann and others 2004).

There are several mechanisms of agricultural reduction in SOC due to alteration of C inputs to and outputs from soil. First, the removal of annual crop biomass through harvest reduces the quantity of C inputs to soil compared to perennial native vegetation (Imhoff and others 2004). Crop biomass may also differ from native vegetation in terms of

C quality. Soil organic C content in agricultural systems is often a linear function of C inputs from crop residue; changes in C quality alter this function, with a high proportion of recalcitrant plant inputs becoming SOC (Kong and others 2005). Second, the physical disturbance of tillage increases SOC decomposition rates, which has been shown to occur both by measuring soil respiration immediately after tillage (Reicosky and others 1997) and by comparing of soil C levels in agricultural fields with and without tillage (Collins and others 2000). Labile C is considered to be more susceptible than recalcitrant C to loss through tillage, with association with clay minerals leading to C recalcitrance (Tiessen and Stewart 1983). Third, tillage destroys aggregates and physical structure, exposing previously physically protected SOC to decomposition (Six and others 2000). Finally, tillage and the lack of plant cover on agricultural fields enable soil erosion to occur by water and wind; fields cultivated for 100 years with maize have up to 56% less topsoil than perennial grassland (Gantzer and others 1991). Soil erosion is one of the largest environmental problems caused by agriculture (Trimble and Crosson 2000; Owens and others 2002). In general, SOC is lost through erosion, although the quantity of eroded C that is lost from the soil system to the atmosphere or water versus transferred downslope is variable and often unknown (Smith and others 2005c).

Loss of SOC due to agriculture is not restricted to modern times or the use of plows. In semi-arid New Mexico, prehistorically cultivated soils were shown to have 40% lower SOC than uncultivated soils approximately 1,000 years after cessation of agriculture (Sandor and others 1986a). Natural landscape processes such as the downhill flow of runoff provided moisture and nutrients for this agricultural system. Although these Native Americans did not practice tillage, the location of their terraced fields on slopes lacking grass cover caused SOC loss by erosion (Muenchrath and others 2002).

Are Agriculture-induced Losses of Soil Organic Carbon Reversible? Agricultural fields abandoned in North America during the 19th and 20th centuries show that SOC accumulates after the cessation of agriculture and the establishment of perennial vegetation in both forests and grasslands, although the rate of accumulation is variable (Table 1). In a classic study of primary succession (Schlesinger 1990), However, no such pattern is shown in Table 1, at least on the century scale after agricultural abandonment, because there is no relationship between the accumulation interval and the rate of SOC accumulation. It could be that SOC accumulation is more variable during secondary succession than during primary succession, with variation in initial conditions causing the lack a of consistent pattern. Other reviews of postagricultural SOC levels have shown variability in the rate of SOC accumulation; this variability could be due to the difficulty of detecting small changes in SOC the fact and different studies measured SOC to different depths (Post and Kwon 2000; Paul and others 2002).

Because of variability in the rate of accumulation, such that estimates on the US Great Plains range from 0 to 100 g C $m^{-2}y^{-1}$, it has been difficult to generalize about the longevity of the effect of agriculture on SOC (Table 1). Several recent conceptual advances in out understanding of SOC formation have improved predictions of the length of time required for preagricultural SOC levels to be attained. Labile SOC pools, with short turnover times, may increase faster than recalcitrant or total SOC pools (Robles and Burke 1998). At a single tallgrass prairie site, aggregate structure that protects labile C has been shown to recover within 10 years after the cessation of agriculture, which is more quickly than total SOC (Jastrow 1996). The universality of these findings has not been established. It is possible that there is a characteristic maximum SOC content for each soil that determines the length of time of postagricultural increases in SOC (Six and others 2002). Presumably SOC would stop increasing once this maximum level was reached and the soil became ''saturated'' with C, although SOC content may be in a dynamic equilibrium that changes in response to plant inputs. The magnitude of difference between agricultural SOC levels and native ecosystem SOC levels will also play a role in determining the longevity of reduced SOC. Determining these initial and final SOC levels for a particular system will enable prediction of the longevity of SOC depletion. Because many postagricultural studies sample only a few time points, it is not known if the empirical rate of accumulation changes over time, as predicted by mathematical equations (Olson 1963).

Attempts to determine that factors that control the trajectory of SOC decline and recovery have met with varied success. State factors are often postulated to be the cause of differences in rates among different studies, but these assertions are rarely tested because it would require the isolation of state factors across very large areas. Whole-ecosystem C budgets, which provided a full accounting of primary productivity and respiration, may be a fruitful way to predict the rate of SOC accumulation after the cession of agriculture (Odum 1960). An alternative approach is to use the mechanisms

Table 1. Rates of SOC Accumulation on Former Agricultural Fields in Temperate Grasslands and Forests

Biome Type and Location	Accumulation Interval (y)	SOC Accumulation Rate (g C m^{-2} y^{-1})	Sample Depth (cm)	Reference
Mixed grass prairie, Wyoming	4	15.9	10	Reeder and others (1998)
Grasslands of the Great Plains	5	74.0	10	Follett (2001)
Shortgrass steppe and northern mixed prairie, southeastern Wyoming	6	0	5	Robles and Burke (1998)
Shortgrass prairie, north-central South Dakota	8	21	7	White and others (1976)
Grasslands of the southern Great Plains	12	110.0	300	Gebhart and others (1994)
Tallgrass prairie, central Minnesota	12	19.7	10	Knops and Tilman (2000)
Tallgrass prairie, Nebraska	12	29.5	10	Baer and others (2002)
Tallgrass prairie, Illinois	15	78	10	Jastrow (1996)
Eastern deciduous forest, Ontario	30	5.6	100	Paul and others (2003)
Mixed-grass prairie, western Minnesota	40	62	10	McLauchlan and others (2006)
Southeastern pine forests, South Carolina	40	3.6	60	Richter and others (1999)
Shortgrass prairie, Colorado	50	3.1	10	Burke and others (1995)
Eastern deciduous forest, Ohio	50	1.5	100	Paul and others (2003)
Eastern deciduous forest, Ohio	50	5.8	100	Paul and others (2003)
Mixed-grass prairie, central Texas	60	44.7	60	Potter and others (1999)
Northeastern temperate forest, Rhode Island	115	3	20	Hooker and Compton (2003)

SOC, soil organic carbon
Studies consider conversion or secondary succession of cultivated land to permanent vegetation in North America. Accumulation interval is the length of time since cessation of agriculture (often only one sampling point in time).

of SOC stabilization, such as biochemical recalcitrance, chemical stabilization, and physical protection, to generate hypotheses about the mechanisms of recovery (Figure 2).

For political purposes, the variation in measured rates of SOC accumulation among postagricultural systems may not be significant. Most reported rates of SOC accumulation are within an order of magnitude, 33 g C $m^{-2}y^{-1}$ on average, for cropland converted to grassland or forest (Post and Kwon 2000) (Table 1). Most predictions of the longevity of SOC depletions caused by agriculture provide estimates for North America and Europe that extend less than a century into the future. Despite uncertainty in C budgets, it has been possible to estimate changes in C stocks due to of conversion of arable land to grassland or woodland that extend up to the year 2080 for ares of the European Union (Smith and others 2005a).

Agriculture Elevates Soil Organic Carbon. Several studies suggest that agricultural activities can increase SOC content relative to natural systems (Bakels 1997; Buyanovsky and Wagner 1998). Somewhat predictably, management that includes organic amendments is implicated in this increase in SOC; however, the type of organic amendment, method of incorporation, and magnitude and length of time of application required for the elevation of SOC are still poorly understood. The spatial extent of this enrichment tends to be local in scale, and it is unclear if there is actually a net increase in SOC due to agriculture, or if there is a transfer of C to concentrated agricultural areas on a landscape level. For example, over 3,000 years it has taken 5–10 ha of grassland to support the organic fertilization of 1 ha of arable land in the Netherlands the past (Blume and Leinweber 2004). Regardless of spatial scale, the addition of biochemically recalcitrant C sources such as manure and charcoal to soil, in conjunction with agriculture, can create persistently elevated levels of SOC.

The increase in SOC due to organic amendments can be substantial. Results from the Rothamsted long-term agricultural experiment in Great Britain

Table 2. Legacies and Longevity of Different Agricultural Practices

Type of Agriculture	Location	Legacy Effect on SOC	Longevity (y)	Component Processes	Reference
Continuous barley	Southern England	Elevated	130	Manure addition	Johnston (1986)
Runoff agriculture	New Mexico	Reduced	1,000	Erosion	Sandor and others (1986a)
Terraced fields	Peru	Elevated	400	Organic amendments	Sandor and Eash (1995)
Plaggen-livestock-crop systems	Germany	Elevated	200–1,000	Manure and sod additions	Springob and Kirchmann (2002)
Tropical swidden	Brazil	Elevated	1,000	Charcoal additions	Glaser and others (2000)

SOC, soil organic carbon
Reduced and elevated SOC are relative to nearby reference ecosystems.

show that continuous application of farmyard manure almost tripled SOC content in 100 years (Johnston 1986). Similarly, agricultural terraces in Peru that have been cultivated for at least 15 centuries have 30% more C than uncultivated soils, probably due to the traditional practice of fertilizing the soil with manure and ashes (Sandor and Eash 1995).

In general, the length of time for which organic amendments such as manure are applied to the soil is much shorter than the duration of elevated SOC. At Rothamsted, one treatment of manure application was performed for only 20 years, yet this soil still had elevated SOC levels 100 years after the cessation of manure application (Johnston 1986). At the site in Peru, fields that had once been cultivated but were abandoned approximately 400 years ago still showed elevated levels of SOC (Sandor and Eash 1995). Both of these findings indicate that organic amendments have a persistent influence on soil properties that lasts for several centuries. The longevity of the agricultural legacy depends on the magnitude of the agricultural alteration of SOC levels, which is often a function of the type of agricultural management that is practiced (Table 2).

In places where the history of organic amendment is unclear, or more information about the quantity and type of manure applied would be helpful, it may be nonetheless possible to reconstruct this history. New techniques such as lipid biomarkers and micromorphology hold promise for reconstructing the details of past amendments that can persist for hundreds to thousands of years in soil (Bull and others 1998). For example, lipid biomarkers were able to distinguish three types of organic amendments on archaeological soils from the 12th century in Britain: composted turf, ruminant animal manure, and omnivorous animal manure (Simpson and others 1999).

Plaggen soils are another example of agricultural soils with elevated levels of SOC. They were formed when composted sod and animal bedding was spread on cultivated fields over a large area for an extended time period (Giani and others 2004). They are locally common in several areas of Europe, but because this type of soil amendment is no longer practiced, plaggen soils are relicts of previous land use. The use of plaggen practices began approximately 3,000 years ago; over time, resources were depleted from the areas where sod was harvested (Blume and Leinweber 2004). Plaggen soils in Germany have elevated SOC levels (approximately double) and a high proportion of unhydrolyzable C (which is biochemically recalcitrant to microbial degradation) compared to nearby reference soils that had no plaggen influence (Springob and Kirchmann 2002). Plaggen soils generally have an extremely sandy texture, (over 75% sand on average) so the stabilization of SOC onto clay mineral surfaces is not responsible for the persistent elevated SOC. It is more likely that the biochemical properties of the organic amendments make SOC in plaggen soils resistant to microbial degradation.

Charcoal contains C that is extremely resistant to microbial breakdown, and it may play an important role in maintaining or raising SOC levels in agricultural soils. In the Brazilian Amazon, prehistorically cultivated soils called *terra preta* (''black earth'') have extremely high SOC content compared to nearby uncultivated Oxisols, probably due to the incorporation of wood charcoal deep into the soil profile (Glaser and others 2000; Lima and others 2002). Modern experimental charcoal additions to soil have shown increased cation exchange capacity due to the large surface area and exchange properties of charcoal (Glaser and others 2002; Lehmann and others 2003a). Terra preta soils were likely formed in conjunction with pre-Columbian settlements along the Amazon River dated to 2,500–1,500 years bp (Lehmann and others 2003b; Glaser and Woods 2004). Although adding charcoal to soil is not currently an intentional management goal in temperate systems, large quantities of charcoal have been found in agricultural soils in the United States (Skjemstad and others 2002). Up to 30% of total SOC consists of charcoal C in some agricultural soils in Australia (Skjemstad and others 1996). Neither the origin nor the fate of this charcoal are currently known, but they have important implications for soil C content and soil function.

Nitrogen Cycling

Agriculture Depletes Soil Nitrogen. Nitrogen is frequently limiting to plant productivity in temperate agroecosystems; therefore, it is of major importance for crop yield (Vitousek and Howarth 1991). Similar to the trends observed with SOC, the dominant paradigm holds that agriculture causes sustained and large losses of soil N (Tiessen and others 1982). The mechanisms of N loss are similar to those of C loss: increased decomposition rates from tillage, erosion, gaseous losses from microbial processes, and decreased plant inputs due to crop removal (Figure 2). Nitrogen has the additional potential for large losses from agricultural systems in soluble inorganic forms, particularly nitrate.

Where tillage is practiced with no supplementary sources of N, N loss can be large and prolonged. An influential study of soils from a cultivated Canadian grassland showed that the rate of N loss was constant for 60–70 years after the beginning of cultivation (Tiessen and others 1982). In long-term agricultural plots in Oklahoma, where wheat has been grown continuously for 109 years, only 35% of the original soil N remains in plots that receive no additional nutrient inputs (Davis and others 2004). Similarly, over 50% of soil N was lost within 25 years of the implementation of cultivation in Nigeria where the native vegetation is savanna woodland (Jaiyeoba 2003). Coarse-textured grassland soils in South Africa lost 55% of total soil N after 98 years of cultivation (Lobe and others 2001). In all of these cases, reductions of N occur in conjunction with reductions in C. A review of N content after land-use change showed that the conversion of forest soils to agricultural use resulted in an average loss of 15% N, less than the average loss of SOC, indicating a that the C:N ratio narrows when agriculture is introduced (Murty and others 2002).

Nitrogen loss pathways responsible for these declines are numerous. Although nitrate leaching is considered the main pathway for the loss of N from agricultural systems, leaching of dissolved organic N, erosion of organic N, and losses of N-containing gases resulting from nitrification, denitrification, and ammonia volatilization have all been documented in agroecosystems (Mosier 2001; Christou and others 2005). Each of these processes has different environmental controls (Figure 2). Soil organic N, the largest pool of soil N, can be depleted if it is mineralized and inorganic N is lost. However, N mineralization rates are generally lower in temperate agricultural systems than native grasslands or woodlands (Booth and others 2005). Similarly, N mineralization rates are 50–60% lower in soils cultivated for at least 30 years in Oregon than in similar uncultivated soils (Rasmussen and others 1998a).

Two aspects of agricultural management, tillage and biomass alterations, tend to lower soil N content and alter N cycling. Tilled soils contain less soil N than untilled soils if all other aspects of management are the same (Reicosky and others 1995). Compared to no-till systems and restored prairies, tillage increases net N mineralization, which can lead to a depletion of soil N over time, depending on the fate of the mineralized N (Brye and others 2003). However, a comparison of different tillage systems indicated that the magnitude of the increase in N mineralization due to tillage may be very small (Kristensen and others 2003). Crop demand for and uptake of nitrate are assumed to control the magnitude of N losses, but annual crop plants take up nutrients for only a portion of each year. The conversion of perennial to annual crop plants in the US Upper Midwest significantly and immediately increased nitrate runoff (Huggins and others 2001).

Soil N depleted by agriculture can accumulate in abandoned croplands, but the accumulation rate for N has not been measured as frequently as the rate for C. There is variation in the C:N ratio of plant litter and SOM, so it is not inevitable that soil N accumulates at a rate similar to that of SOC (Murty and others 2002). Agricultural soils in New Zealand are accumulating soil organic N while maintaining levels of SOC (Schipper and others 2004). The rate of N accumulation is generally lower than the rate of C accumulation, but the rate of N accumulation may determine the rate of C accumulation in systems where N limits primary productivity (Johnson 1992). Features of the N cycle maintain N limitation (Vitousek and others 2002), and N limitation is not relieved in postagricultural ecosystems such as temperate forests (Richter and others 2000).

The mechanisms that control the postagricultural rate of accumulation are different for C and N. Different organisms fix C and N, with different controls on the rates of fixation. Vegetation composition during or after agriculture may have the potential to alter the duration of N depletion, because leguminous plants have the potential to add N to the soil through symbiotic N fixation. The presence of a leguminous crop, alfalfa (*Medicago sativa* L.), slowed the N loss rate to half the rate due to tillage in Canadian agricultural fields (Tiessen and others 1982). Similarly, the rate of N fixation by either symbiotic or free-living bacteria may determine the rate of accumulation of SOC on postagricultural fields, with more SOC accumulating on sites with N-fixing organisms (Johnson 1992; Resh and others 2002).

Agriculture Elevates Soil Nitrogen. Just as organic amendments increase SOC, fertilizer applications in agricultural systems increase soil N relative to nonagricultural systems (Haynes and Naidu 1998). The composition of N inputs can be either organic or inorganic, easily decomposable or recalcitrant, and derived from either plant or animal sources. Nitrogen applied as inorganic fertilizer can be retained in agroecosystems by crop plant uptake, conversion to organic N, and subsequent residue return to the soil. Nitrogen applied as organic fertilizer can also be directly retained as soil N in association with SOC. The content and composition of vary, and these factors manure C and determine the fate of organic N (Griffin and others 2005) (Figure 2).

Although a few studies indicate that postagricultural soil N levels may remain elevated for several hundred years as a consequence of previous manuring practices, much better quantification of this phenomenon is needed before any kind of pattern can be discerned. In forested sites in Massachusetts where agriculture ceased 100 years ago, previously cultivated soils have higher N content and greater nitrification rates than uncultivated soils, a condition anecdotally attributed to manure amendments (Compton and Boone 2000). On fields that were abandoned in the middle of the 19th century in France, δ^{15}N rose as past land-use intensity increased (Koerner and others 1999), probably due to the spread of animal manure on cultivated fields (Koerner and others 1997).

Better quantification of the N cycle on postagricultural land would also help to clarify the spatial extent of these apparent agricultural increases in N. It is possible that a more detailed spatial analysis would reveal a net transfer of nutrients from certain portions of the landscape to agricultural fields. For example, in the 1800s, colonial farmers in Concord, Massachusetts, relied on annual inputs of N from river sediments and developed a reliable system to move organic N from lowlands to tilled fields using hay and manure transfer (Donahue 2004). This type of nutrient transfer results in patchy concentrations of soil N (Fraterrigo and others 2005). To determine the long-term effect of fertilizer application, it is important to clarify whether the ultimate source of N for agricultural fields is located on site or off site.

Detailed analyses of inorganic and organic applications of N at similar rates indicate that manured soils generally have a higher N content than soils fertilized with N in mineral form (Edmeades 2003). As compared to conventional systems that rely on inorganic N fertilizer, modern agricultural management practices that use leguminous crops as a nutrient source can increase soil N content and reduce nitrate leaching within 5 years of establishment (Drinkwater and others 1998). This suggests that the composition of N inputs, as well as the quantity of N inputs, affects soil N levels. However, the duration or longevity of these effects as cultivation continues and after the cessation of agriculture is almost completely unknown. A recent review that compared agricultural N cycling under a regime of legumes versus one of inorganic fertilizer found insignificant differences between the two management systems in soil N and nitrate leaching (Crews and Peoples 2004).

Much attention has been paid to nitrate loss after fertilizer application, but this practice does not necessarily deplete soil organic N (Riley and others 2001). Large amounts of inorganic N fertilizer do leave agroecosystems due to its overapplication and the asynchrony of fertilizer relative to crop demand, as well as to the notoriously low nutrient-use efficiency of crop plants (Janzen and others 2003; Fageria and Baligar 2005). On average, less than 50% of the N applied in the form of inorganic fertilizer on Earth is recovered in crop biomass during the growing season for maize, rice, and wheat, indicating that agroecosystems are susceptible to N leakage (Cassman and others 2002). Surprisingly, a meta-analysis showed that the application of fertilizer does not significantly affect N mineralization or nitrification rates; however although, individual studies indicate that organic N additions stimulate N mineralization, whereas inorganic N additions do not (Booth and others 2005). Inorganic N fertilization can have a long-term effect on N cycling. Nitrification rates have been shown to be higher on soils with a history of high levels of N-fertilizer application than those receiving to low levels of N-fertilizer application (Watson and Mills 1998).

Agriculture Does not Affect Soil Nitrogen after a Century. Sometimes N content and cycling are not perceptibly affected by former agricultural practices on time scales of a century after the cessation of agriculture. There was no change in total ecosystem N content 115 years after agricultural abandonment and forest regrowth in Rhode Island; however, over time, less N was found in soil and more N was found in woody biomass (Hooker and Compton 2003). Neither soil N content nor net N mineralization rate was affected by plowing history on reforested lands removed from agriculture at different times over the previous 99 years in New England (Compton and others 1998).

In these study areas, it is possible that the earlier regime of agriculture never caused any changes to the N cycle, that the changes did not persist for a century, or that there are limitations to the categorical approach to past land use such that changes did occur exist but cannot be detected due to variation and the influence of other factors. If the first scenario is true, it implies that the N cycle is not as susceptible to agricultural alteration as the C cycle, but nutrient management and manipulation are features of most agricultural systems. It is more likely that, in these cases, the effects of colonial agricultural on both C and N content were S small, because of the short duration of agriculture, that they have disappeared after a century.

Phosphorus Content

Several properties of P lead to long-lasting residual effects of P fertilization. Phosphorus occurs in the soil in several different chemical forms, most of which are relatively insoluble in water, sensitive to pH changes, and immobile. Soil P can be grouped into inorganic and organic forms. The potential is much greater for ecosystem P retention than for N retention due to strong P sorption by many soils and the fact that there are fewer loss pathways for P than for N. Mineralization of organic P tends to be important for crop production in the first decades to century after the implementation of agriculture, whereas the supply of inorganic P, either from weathering inputs or fertilizer import, is what sustains P levels in agricultural systems for the long term (Newman 1997).

Agriculture Depletes Soil Phosphorous. Mechanisms that lower soil P on agricultural soils include the removal of plant biomass, relatively high uptake of P by leguminous crops compared to nonllegumes, and erosion. Interestingly, burning, which is often associated with agriculture, especially in tropical forest systems, tends to conserve soil P but remove C and N in the long term (Johnston 2003).

Crop removal can be a major loss of P from agroecosystems. In modern British wheat farming, P input in the form of inorganic fertilizer its and output by crop removal exceed all natural inputs and outputs by an order of magnitude (Newman 1997). In several early agricultural systems that relied on obtaining a natural supply of P from weathering reactions in the soil, crop yields could not be sustained because the rates of P removal in crop biomass exceeded the rate of P supply (Newman 1997). Similarly, the finding that P export in agricultural products was greater than P import and supply from soil weathering was correlated with declines in wheat yield on a medieval farm in Great Britain (Newman and Harvey 1997).

Perennial legumes, such as the alfalfa grown in traditional Mexican agroecosystems with no additions of P fertilizer, deplete labile inorganic soil P (Crews 1996). The major source of plant-available P in these systems is inorganic P weathered from parent material, rather than that mineralized from soil organic P. Therefore, concentrations of mineral P in soil calcium phosphate are an important factor in determining long-term agricultural fertility. Despite the inexorable depletion of soil P by alfalfa, these Mexican agro ecosystems have been cultivated successfully for a minimum of 200 years and in some cases for millennia. Yield has not yet been compromised, indicating that the mineral P pool has been sufficient to maintain fertility.

Because of its association with soil particles, P is easily lost through the soil erosion that accompanies tillage. Many agronomic studies support the idea that tillage results in P losses. The total P content of a cultivated soil in Canada was found to be 29% lower than that of an adjacent permanent pasture, and most of the P that was lost was in organic forms (Hedley and others 1982). Moreover, soils under conventional row-crop agriculture contained 79% less extractable organic P than those at an adjacent forested site (Daroub and others 2001).

Soil erosion may have been the mechanism for soil P loss on abandoned agricultural fields on hillslope sites in New Mexico. Prehistoric agricultural fields from 1,000 years ago show depleted P content relative to uncultivated sites, with about 40% lower available P and 15% lower total P in the formerly cultivated sites (Sandor and others 1986b). These cultivated sites also have lower SOM levels, which could be causing the lower available P.

Agriculture Elevates Soil Phosphorus. Just as with C and N, P content can be elevated in agricultural soils relative to adjacent nonagricultural soils. There are two possible explanations for these observations of trends in increased P: either amendment, similar to what we have seen with C and N, or an interaction between the placement of the agricultural crops and site characteristics that also correlate with P availability. Distinguishing between these two possibilities requires good P budgets and some knowledge of preagricultural conditions, which are rarely available or easy to obtain.

The first explanation for agriculturally elevated P, amendment, is fairly common. Fertilizer amendments increase soil P content when inputs exceed outputs. For example, a 25-year agronomic experiment in the US Midwest showed that extractable P levels in soil were linearly related to the level of P fertilizer application minus P removed in crop harvest (Barber 1979). The treatment with the highest level of phosphate fertilizer application, 54 kg P ha^{-1} y^{-1}, had extractable soil P levels 1.5 times higher than the treatment with no fertilizer added. The elevation in soil P levels in this experiment appears to be reversible, because when fertilizer inputs ceased, extractable P decreased within 8 years. Similarly, extractable soil P had accumulated due to both single-year and 8-year fertilizer applications at a site in North Carolina (McCollum 1991). Then, extractable P levels declined exponentially for 26 years after the cessation of fertilization due to harvest removal and reversion to unextractable forms of soil P (McCollum 1991).

If the application rate is high enough, P levels in agricultural fields can exceed those of native grassland. For example, extractable P is an average of 14% higher on agricultural fields where manure has been applied than on grasslands of the US Great Plains (Haas and others 1961). Many areas of the United States and Europe have extremely high soil P levels due to prolonged P fertilizer application (Gough and Marrs 1990). When manure is applied at the correct rate for optimal N supply to crop plants, P is often in excess (Edmeades 2003). Soil type may influence the rate of P accumulation during the amendment period. A loamy sand in North Carolina showed increases in extractable P when annual P application exceeded crop removal, whereas extractable P levels in a nearby clay loam declined regardless of P application and removal rate (Schmidt and others 1996).

Although soil P levels often decline when P fertilization stops, there is some evidence that agriculturally elevated soil P levels can remain high for centuries to millennia after P inputs cease if P is not continually removed in harvest. Former farm sites may return higher levels of P than nearby to nonfarmed sites a century (Compton and Boone 2000) to a millennium after the abandonment of agriculture (Sandor and Eash 1995). Archaeologists frequently use soil phosphate levels to locate sites of past human occupation, because soil phosphate levels remain relatively elevated for at least several hundred years (Eidt 1977). The legacy of P additions can therefore be persistent.

Just as with N, the physical origin of the P amendments must be considered. Because of its relative immobility, transportability in both organic and inorganic forms, and lack of volatility, P can easily be concentrated from a large area to a relatively localized agricultural field. Soils in France that were farmed during Roman times (50–250 ad) had soil P levels that correlated positively with intensity of human land use (Dupouey and others 2002). A global P budget shows that P is accumulating in agricultural soils, but that this rate of accumulation is currently higher in developing than in developed nations due to recent increases in P-fertilizer use in developing nations (Bennett and others 2001).

Long-term P content may also be influenced by whether the fertilizer amendment is applied in an organic or inorganic form. Nature of input (inorganic versus organic) did not seem to affect the quantity of soil P in the long-term experiments at Rothamsted, but soil texture may (Blake and others 2003). However, a comparison of different European agricultural trials indicated that the form in which P is applied, as well as differences in the soils physicochemical properties, the climate, and the availability of other major nutrients, all influence the effectiveness of P fertilization and the P balance (Blake and others 2000).

In contrast to amendments, the second (and less common) explanation of elevated P content in former agricultural soils is correlation between soil properties and agricultural land use. According to this scenario, agricultural management does not have a direct effect on soil P, but sites with high levels of soil P are selected for agriculture. Prehistoric farmers in western New Mexico placed their agricultural systems on alluvial fans in the mesa–canyon landscape where they cultivated maize (Norton and others 2003). These alluvial fans, defined by topographic position, receive nutrient inputs from higher topographic positions. Similarly, native Hawai'ian farmers selected sites with high P content and base saturation and adequate rainfall for their most productive dryland agricultural systems (Kirch and others 2004; Vitousek and others 2004). The high P content of the Hawai'ian agricultural soils is caused by the transportation of P to surface soils via preagricultural vegetation, the presence of relatively young parent material, and the clever placement of the fields in an optimal spot along the precipitation gradient that controls P weathering rates; thus, it is not caused by mulching or other agricultural practices. Although this agricultural system was abandoned around the time of European contact in 1778, these differences in soil properties have persisted to the present day. Therefore, inherent soil properties can be equally or even more important than agricultural management techniques as an explanation of P legacies.

Mitigation of Responses by State Factors

There are two major classes of mechanisms that affect the response of ecosystems to agriculture. First, as has been described, agricultural management techniques, especially tillage and fertilization, affect long-term ecosystem processes. Second, state factors also influence the response of ecosystems to agriculture, either by the susceptibility of a system to perturbation or change (resistance) or by the rate of change of the system after the cessation of agriculture (resilience).

To improve our predictions of ecosystem response to agriculture, it is important to identify the system properties that confer resilience to environmental change (*sensu* Carpenter and others 2005). Two candidate state factors, climate and parent material, are particularly important for determining SOM levels in nonagricultural systems and therefore may also be particularly important for determining SOM response to agricultural land use (Jenny 1941; Burke and others 1989). In addition to management indicators such as N-fertilizer application rate and cropping intensity, rainfall, temperature, and soil texture are important predictors of SOC levels in tropical and temperate agricultural soils (Alvarez 2005).

Climate

Two components of climate, mean annual precipitation and mean annual temperature, have been used to explain spatial variation in global SOM levels (Post and others 1982). Soil organic C content increases with increasing precipitation; at a fixed precipitation level, it also increases with decreasing temperature. Spatial patterns of SOC content are due to the effect of climate on inputs (primary productivity) and outputs (decomposition) of SOC, assuming-steady state conditions. If climate can explain SOC content at a given point in time, perhaps it can also explain responses to the commencement and cessation of agriculture.

We are only beginning to gain synthetic understanding of how rates of soil C, N, and P loss induced by agriculture vary with climate. There is a perception that drier areas, particularly arid ecosystems, may be environmentally sensitive such that perturbations from agriculture cause degradation in soil properties that may be irreversible (Tiessen and others 1998). An analysis of paired cultivated and uncultivated soils in two different regions of the globe (India and the Great Plains of North America) showed that the greatest absolute losses of SOC and soil N occurred in regions of low temperature and high precipitation in India (Miller and others 2004). A recent global meta-analysis revealed that long-term cultivation causes the greatest losses of SOC in tropical moist climates, with cultivated soils in those regions containing 58% on average of the SOC in uncultivated soils (Ogle and others 2005). Temperate dry systems seemed to be the least pervious to SOC losses from cultivation, with 82% of the original SOC remaining in cultivated soils (Ogle and others 2005). Although temperature and precipitation influence the rates of SOC loss from agriculture, the exact mechanisms responsible for these patterns have not yet been tested.

The effect of climate on rates of change after the cessation of agriculture is even less well described. Because net primary productivity (NPP) and decomposition rates are determined partially by precipitation, it is possible that former agricultural land in wetter areas, where NPP is not limited by moisture, may increase SOC faster than dry areas, where C inputs to soil are smaller because of moisture limitation (Epstein and others 2002). In a global review, there was some evidence that C sequestration rates in agricultural land increase with increasing mean annual precipitation (Paustian and others 1997a). However, this pattern, in which climate influences the rates of postagricultural change, should be considered preliminary.

Parent Material

Parent material partially determines the composition and content of clay minerals factors that are considered to be important in determining soil C and N dynamics. Clay minerals stabilize SOM, generally resulting in a positive relationship between SOC and clay content, as demonstrated by studies on the US Great Plains (Nichols 1984). However, there is some evidence from New Zealand that clay content is not a good predictor of SOM content; therefore, clay content and SOC may not be correlated in all systems (Percival and others 2000). Overall, bonds with clay minerals certainly contribute to SOM stabilization.

Several long-term tillage trials, found no relationship between soil texture and the effects of tillage on SOC content (Paustian and others 1997b). Nonetheless, clayey soils are considered to be more resistant, or less susceptible to degradation from tillage, than sandy soils (Tiessen and others 1982). An analysis of cultivated soils from the US Great Plains showed that the relative loss of SOC decreases with increasing clay content (Burke and others 1989). Similarly, parent material determined the degree of resistance and the mechanism for resistance at two sites in North Dakota, where soils derived from shale were resistant to erosion, whereas soils derived from sandstone showed erosion but could nonetheless maintain SOC levels despite erosion (Schimel and others 1985). Sand content significantly predicted SOC content in the surface soils of agricultural fields in Illinois, where there was a negative relationship between sand content and SOC for tilled soils (Needelman and others 1999). Soil organic C has declined linearly from 1.5% to under 1.0% during the past century of cultivation at the site of the Woburn Experiment in Great Britain, where soils contain 10% clay; whereas SOC has remained stable with similar management at Rothamsted, with 20–25% clay (Johnston 1986). Because of these mixed results, controlled studies or gradients designed to isolate the role of soil texture in determining SOC in tilled systems would be especially desirable.

As with climate, the role of parent material in determining rates of SOM increase after the cessation of agriculture is just beginning to be considered systematically. The same mechanisms of stabilization of SOM by clay minerals may also promote SOM formation. Distinct types of clay minerals characterized by different activity and structure—crystalline versus noncrystalline—are likely to have different effects on SOM formation, so this topic deserves should be further investigation (Torn and others 1997; Parfitt and others 2002). Alternatively, other mechanisms of SOM stabilization such as the biochemical recalcitrance of C inputs (Krull and others 2003), may be more important factors than clay minerals in determining the consequences of agriculture on SOM content. Empirical studies of the recovery of SOM after the cessation of agriculture that explicitly address the mechanisms of SOM formation, including the role of soil texture and clay mineralogy in stabilizing SOM, are needed (Jastrow 1996).

Conclusions

The main findings from this analysis are: (a) the nature and longevity of the agricultural effect on soil C, N, and P depend on the type and duration of agricultural management; (b) good elemental budgets and a biogeochemical framework can help to quantify the effect; and (3) climate and soil texture affect both the rate of loss and the rate of accumulation of soil C, N, and P during and after agriculture. As a means of understanding the relative magnitude of state factors and management effects, a reasonable description is that climate and the characteristics of parent material define an envelope of possible soil C, N, and P levels for an ecosystem, whereas agricultural management determines the levels within that envelope. The longevity of agricultural effects depends on how different the C, N, and P levels in the agricultural system were from those of the native ecosystem; the most severe alterations are caused by cropping intensity, tillage, fertilization, or duration of agriculture.

Carbons, N, and P cycles each have unique properties that determine their resistance and resilience to change during agriculture. Relative to C, N is easily and rapidly lost from ecosystems due to multiple loss pathways and it is slow to accumulate in natural systems due to ecological constraints on N fixation. Phosphorus is the element that is least susceptible to depletion by agriculture, unless there are high erosion rates, but it is also the slowest to accumulate without off-site amendments because the ultimate source is weathering reactions in the soil. Fertilizer amendments can increase C, N, and P levels relative to natural systems, and these increases have been persist shown as for millennia (Table 2).

Different types of agricultural management practices lead to different outcomes for SOC and nutrients (Table 2). Tillage and harvesting tend to decrease concentrations of SOC and nutrients, whereas fertilization tends to increase these concentrations. To further complicate the picture, the same type of agricultural management can lead to different outcomes depending on the relative magnitude of different fluxes. For example, on the Sanborn Field in Missouri, long-term plots with

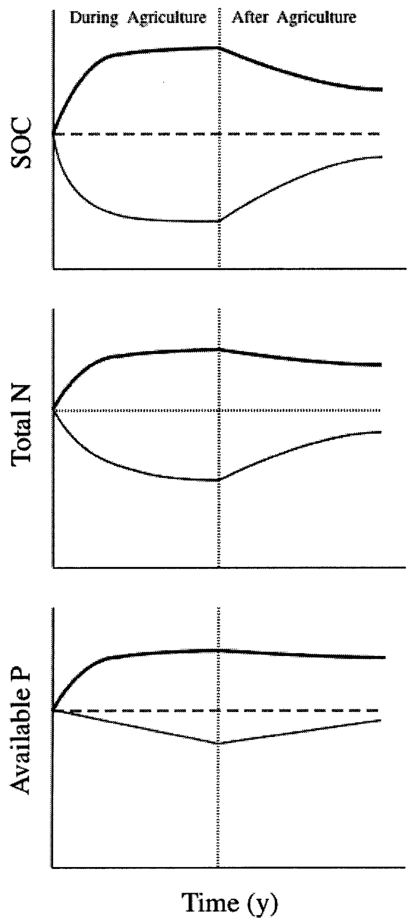

Figure 3. Hypothetical trajectories of changes in soil properties during and after two different types of agriculture. Characteristics of scenario 1 (*thin line*) and scenario 2 (*thick line*) are given in Table 3. SOC, soil organic carbon; N, nitrogen; P, phosphorus.

manure applications showed net losses of SOC during the first part of the 20th century but net gains of SOC during the second part of the century due to an increase in the quantity of crop residue returned to the soil (Buyanovsky and Wagner 1998). Although this degree of variability is frustrating for ecosystem ecologists striving to predict when agricultural legacies may be expected to exert an influence on ecosystem properties and the magnitude or even the direction of postagricultural change, we also have the intellectual tools and analytical techniques to make progress in this area.

Although the difficult to predict changes in soil C, N, and P when agriculture ceases, there are a few useful approaches. Accurate budgets for these elements for the agricultural and postagricultural periods, accounting for the magnitude of inputs and outputs, are essential (Richter and others 2000; Richter and Markewitz 2001). Simply put, C, N, and P pools will increase in both agricultural and postagricultural systems when inputs exceed outputs and they will decrease when outputs exceed inputs. These types of calculations are very familiar to ecosystem ecologists, and this information is often available for agroecosystems. Adding a spatially explicit component to these calculations, such as the farm-based nutrient budgets used by agronomists, would help determine the spatial extent of agriculture's effect on soil C, N, and P (Rotz and others 2005). In conjunction, it is useful to have a good grasp of the biogeochemical processes that govern elemental transformations during and after agriculture (Figure 2). This conceptual framework provides a way to predict the longevity of agricultural impacts by quantifying the pools and fluxes of C, N, and P while explicitly considering agricultural management. The degree to which agriculture alters the magnitude of fluxes, and therefore the pool sizes of C, N, and P, depends on site-specific conditions and practices.

An example of the application of a biogeochemical approach can generate testable, quantitative hypotheses about the patterns of change in C, N, and P levels during and after agriculture (Figure 3) and the mechanisms responsible for these patterns (Table 3). In this example, two different types of agriculture—moldboard tillage on the Great Plains of North America and ancient plaggen agriculture in Europe—cause different biogeochemical effects, which lead to different postagricultural trajectories that depend on the rate of dominant ecosystem processes. Although these scenarios should be considered speculative, they provide an example of how variability in agricultural management can be mechanistically incorporated into an ecosystem framework. Further refinements of this approach would improve our predictions of the consequences of future land-use change.

Climate and soil texture, because they affect input and output processes, influence both the rate of loss and the rate of accumulation of SOC, N, and P in agroecosystems. In fact, all five state factors affect soil formation and therefore must play a role in post-agricultural soil development. The challenge is to quantify the magnitude of their roles.

Models of SOM, such as Century, have been successful at predicting C and nutrient content in agricultural soils and can be used to test the relative importance of specific mechanisms, such as the temperature sensitivity of SOC decomposition and the role of clay concentration in stabilizing newly -formed SOC (Parton and others 1987).

Table 3. Characteristics and Hypothetical Mechanisms of Changes in Surface Soil Properties during and after Two Different Types of Agriculture

	Scenario 1	Scenario 2
Location	Great Plains of North America	Sandy lowlands of northern Germany
Agricultural management (cropping, tillage, fertilization)	Continuous cultivation of wheat; moldboard plow; no fertilizer amendments	Continuous cultivation of rye; ard plow; plaggen fertilization
Mechanism of SOC change during agriculture	Aggregate destruction; reduced plant inputs via harvest	Addition of plaggen material (composted heather sod and animal bedding)
Postagricultural vegetation	Grassland	Forest
Mechanism of SOC change after agriculture	Aggregate formation; increased plant inputs	Microbial respiration
Mechanism of N change during agriculture	Nitrate leaching	Plaggen addition
Mechanism of N change after agriculture	Biological N fixation	Leaching
Mechanism of P change during agriculture	Harvest	Plaggen addition
Mechanism of P change after agriculture	Weathering	Occlusion

SOC, soil organic carbon; N, nitrogen; P, phosphorus.

Based on the results of long-term experiments that have endured for close to a century, it is clear that the level of the nutrients applied to agricultural systems is more important in determining levels of soil N and P than the form in which the nutrients are applied (Rasmussen and Parton 1994). In situations where nutrients in organic form are less susceptible to loss (because of their association with C molecules) than inorganic nutrients, there will be differences between organic and inorganic fertilizer treatments in nutrient content (Edmeades 2003). However, depending on management practices, there may not be a difference.

The state factor approach and ecological gradients that separate the effects of different predictors on a response variable such as SOC have been very useful in determining trajectories of change after agriculture, and they will be useful in the future. When direct attribution of changes due to agriculture is important, a key element of study design is to find a reference system or a control that adequately represents nonagricultural conditions. In some systems, this may be very difficult. Another option is to directly observe changes due to agriculture over time, although the time scale for such studies, is usually limited to a few decades (Rasmussen and others 1998b). Synthesis of large data sets and soil surveys may also be fruitful. Although empirical approaches may be the most useful means of answering questions about the longevity and reversibility of agricultural impacts, theoretical approaches should be developed to supplement these studies.

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