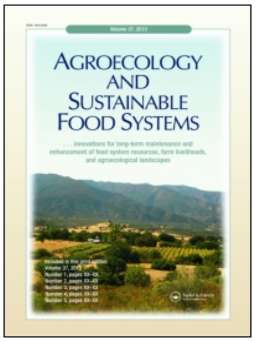

Taylor & Francis
Taylor & Francis Group

Agroecology and Sustainable Food Systems

ISSN: 2168-3565 (Print) 2168-3573 (Online) Journal homepage: www.tandfonline.com/journals/wjsa21

Nutrient cycling in agroecosystems: Balancing food and environmental objectives

Kate Tully & Rebecca Ryals

To cite this article: Kate Tully & Rebecca Ryals (2017) Nutrient cycling in agroecosystems: Balancing food and environmental objectives, Agroecology and Sustainable Food Systems, 41:7, 761-798, DOI: 10.1080/21683565.2017.1336149

To link to this article: https://doi.org/10.1080/21683565.2017.1336149

Published online: 17 Jul 2017.

AGROECOLOGY AND SUSTAINABLE FOOD SYSTEMS
2017, VOL. 41, NO. 7, 761–798
https://doi.org/10.1080/21683565.2017.1336149

Taylor & Francis
Taylor & Francis Group

Nutrient cycling in agroecosystems: Balancing food and environmental objectives

Kate Tully [a] and Rebecca Ryals [b]

[a]Plant Science and Landscape Architecture, University of Maryland, College Park, Maryland, USA; [b]Department of Natural Resources and Environmental Management, University of Hawaii, Honolulu, Hawaii, USA

ABSTRACT

As our rapidly growing human population puts great demands on our agricultural production systems, we must promote management practices that balance both food and environmental objectives. We focus this literature review on farm management strategies that tighten nutrient cycles and maintain yields. We examined six metrics for efficient nutrient cycling in agroecosystems: reduced runoff and erosion, reduced leaching, improved soil carbon storage, enhanced microbial biomass, low greenhouse gas emissions, improved water holding capacity, and high yields. We evaluated these metrics in six farm management practices: intercrops, agroforestry, cover crops, organic amendments, integrated crop-livestock, and conservation tillage. Agroforestry and cover crops consistently reduced runoff and erosion and improved carbon sequestration compared to conventional systems. Agroforestry was the only practice that consistently reduced nutrient leaching over conventional practices. Organic amendments and conservation tillage improve water holding capacity. There exists uncertainty in the effectiveness of these practices to reduce nitrous oxide emissions. Finally, although agroforestry tends to suppress yields, all of the practices had either a neutral or positive effect on yields. Evaluating cropping systems in terms of multiple services gives us insight into how to match practices to environmental goals, where the uncertainties lie, and where opportunities exist for improved agroecosystem management.

KEYWORDS
Agroecology; ecosystem services; nutrient cycling

Introduction

Nutrient cycles serve as the link between agricultural systems and the broader environment. These cycles may be localized to an agricultural ditch, a stream, or a watershed. At the regional- and global-scales, nutrient cycles connect agricultural practices to climate change, pollution of water and air, and extraction of geologic reserves. Over the past 50 years, nutrient cycles have been drastically altered by the rapid increase in synthetic fertilizer applications. Heavy fertilizer loads lead to a host of environmental consequences

CONTACT Kate Tully kltully@umd.edu Plant Science and Landscape Architecture, University of Maryland, 2108 Plant Sciences Building, College Park, MD 20742, USA.

Part of the special issue "Agroecology: building an ecological knowledge-base for food system sustainability."

(Carpenter et al. 1998; Daoji and Daler 2004; Galloway et al. 2003; Tilman 2002) many of which have associated social and economic costs from diminished ecosystem services. These challenges have arisen in part because most agricultural production systems that rely on increased fertilizer are designed to optimize harvestable yields over short timescales. However, when considered within their broader environmental matrix, or *agroecosystem*, it is clear that agricultural nutrient cycles can be improved through farm management to also provide additional ecosystem services, such as water quality and runoff control, soil fertility maintenance, carbon storage, climate regulation, and biodiversity (Power 2010; Tomich et al. 2011).

Agricultural scientists trying to improve the efficiency of nutrient cycling often focus on the "4 R's" of fertilizer application—right rate, right time, right place, and right kind. This framework is designed to help producers make more precise decisions about fertilizer applications and to reduce overall nutrient use at the field-level. Precision agricultural techniques employ technologies (e.g., variable rate fertilizer applications) and diagnostic soil tests (e.g., post side-dress nitrate test) to optimize nutrient use efficiency (NUE) of their crops. To truly tighten nutrient cycling, we need to consider nutrient management beyond the edge of the farm field within the broader agroecosystem context and scale. Nutrient cycles in agroecosystems can be managed at the field-, farm- and landscape-level and at seasonal and long-term timescales to achieve dual objectives of food production and environmental stewardship.

Nutrient use efficiency is the primary metric used to evaluate how well plants take up available nutrients in a cropping system. There are at least 18 different ways to calculate NUE, depending on the output focus (e.g., nitrogen, calories, protein), which makes it difficult to compare studies (Ladha et al. 2005). Typically, there are two main components of NUE calculations: (1) the recovery of the applied nutrient either in the harvestable biomass of the plant or in the total aboveground biomass and (2) the quantity of the applied nutrient that was neither incorporated into plant biomass nor made available to subsequent crops. Due to the difficulty involved in quantifying residual nutrient mineralization and uptake, most studies focus on fertilizer recovery in crops within one growing season (Ladha et al. 2005). The distribution of nutrient sources and sinks is not uniform across a landscape, and therefore, there is a growing need to consider NUE at larger, agroecosystem scales. At the plant-level, demands for water and light, temperature, impacts of pests, and stand structure may influence plant nutrient uptake and use efficiency (Baligar, Fageria, and He 2001; Clark and Duncan 1991; Gerloff and Gabelman 1983). At the field-level, nutrients that are not taken up by plants or stored in stable soil reserves are lost to the environment, where they can contribute to localized and global pollution (Dobermann and Cassman 2004; Tilman 2002).

Human alteration of nutrient cycles has greatly benefited our global capacity to feed humanity, but at considerable cost to stability of earth system functions (Steffen et al. 2015). There is a critical need to invest in field and landscape management practices that can improve yields per unit nutrient input and minimize losses to the environment, effectively tightening nutrient cycles in agroecosystems. Luckily, many strategies exist, and here, we review the state of knowledge of the potential for agroecological practices to tighten nutrient cycles. The specific objectives of this review are to summarize the fundamental biological role of plant nutrients and the biological, physical, and chemical properties that influence their cycling. We also synthesize the evidence from and agreement of peer-reviewed research articles on the effects of agroecological practices on nutrient cycles, with a particular focus on agroforestry, cover cropping, intercropping, integrating livestock and cropping systems, organic matter amendments, and conservation tillage. Finally, we conclude by identifying research and management opportunities that will help improve understanding and application of sustainable nutrient management of agroecosystems.

The role of macronutrients for plant growth and management considerations

Plants derive most of their essential nutrients from the soil to create the structures of plant tissues, nucleic acids, and enzymes as well as regulate their osmotic balance. To maintain healthy, productive crops, plants must maintain an adequate balance of 15 essential nutrients, 12 of which are supplied by the soil and managed by farmers (e.g., nitrogen, phosphorus, potassium, sulfur, magnesium, calcium, iron, boron, manganese, zinc, molybdenum, copper). Fertilizers are added to agroecosystems to prevent nutrient limitation and maximize yields. Nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), and sulfur (S) are considered crop macronutrients as plants require them in higher concentrations (0.5–5.0%) than they require micronutrients, such as molybdenum (Mo) and copper (Cu). Nitrogen, P, and K are the primary components of most commercial fertilizers as these macronutrients are often limiting factors to plant growth. Fertilizers containing other macro- or micronutrients are used to rectify observed plant nutrient deficiencies or soil types that are known to be nutrient-poor. Below, we discuss the biological roles and management considerations associated with N, P, and K (Table 1).

Nitrogen

Nitrogen limits primary productivity across terrestrial ecosystems and is involved in one way or another with 50% of dry plant tissue. It is an

Table 1. Table of the roles, sources, sinks, and management of nitrogen, phosphorus, and potassium in agroecosystems.

	Role in plants	Major sources into agroecosystems	Major losses from agroecosystems	Management challenges	Management opportunities
Nitrogen	Component of chlorophyll, amino acids, and nucleic acids	N-fixing plants, plant residue organic fertilizers, inorganic fertilizers, manure	Crop harvest Runoff & leaching Gaseous emissions	~50% N use efficiency Eutrophication GHGs	Conservation tillage, Covers and residues, Mixed cropping-livestock, Intercropping, Recycling organic matter Landscape design and management
Phosphorus	Component of DNA, RNA, ATP, and cell membranes	Organic fertilizers, inorganic fertilizers, manure	Crop harvest Erosion Runoff & leaching	P fixation in soil Peak P Soil loss Eutrophication	Conservation tillage, mycorrhizal interactions, Covers and residues, Mixed cropping-livestock, Recycling organic matter, pH management
Potassium	Facilitates osmoregulation, protein synthesis, enzyme activation, and photosynthate translocation	Organic fertilizers, inorganic fertilizers, manure	Crop harvest Erosion Runoff & leaching	Soil loss K fixation in soil	Covers and residues Recycling organic matter

important component of chlorophyll, amino acids and nucleic acids, and therefore crucial to the productivity of agroecosystems. Most intensive agricultural production systems depend on high application rates of synthetic N fertilizer to maintain high yields, dramatically altering the global N cycle (Galloway et al. 2003). Nitrogen is needed to offset losses from crop harvests, which removes large portions of N-rich organic material from agroecosystems. The intensive cultivation of N-fixing crops has also increased the quantity of reactive N in cropping systems by converting atmospheric N_2 into biologically available reactive N forms. In fact, the incorporation of legume biomass into soils can provide 23–176 kg N ha^{-1} (Herridge, Peoples, and Boddey 2008), which is comparable to inorganic additions.

Tightening up N cycling is a primary goal of sustainable agroecosystems. It involves management strategies at the field-level to improve NUE and promote a balance between soil inorganic and organic nutrient pools (Table 1). It also requires understanding how the spatial arrangement of fields within a landscape can affect nutrient flows between and among natural and cultivated systems. For example, the recently released California Nitrogen Assessment uses a state-wide mass-balance approach to determine overall nitrogen use efficiency and by sector. They found that greater adoption of soil management practices and increased NUE could result in nearly 40,000 tons less fertilizer N use per year across California (Tomich et al. 2016). The European Nitrogen Assessment uses a similar approach and reports the overall NUE in European agriculture (food and feed) is only about 30% (Sutton and Van Grinsven 2011). In almost, no other industry would 30–50% input efficiency be tolerated. In this review, we identify strategies that can improve efficiency.

Phosphorus

Phosphorus is an essential component of DNA, RNA, and ATP and the phospholipids that comprise cell membranes and can also limit productivity in agroecosystems. Phosphorus plays a critical role in root growth, flowering, fruiting, and seed formation (Smil 2000). In order to deal with P-limitation, many plant species form symbiotic associations with mycorrhizal fungi, which scavenge P from the soil and deliver it directly to root cells (Smith and Read 2008). Unfortunately many agricultural practices such as high P fertilizer applications, cultivation of non-host crops (e.g., Brassicaceae), and tillage disrupt the hyphal networks of mycorrhizae in the soil (Jansa, Weimken, and Frossard 2006).

P in fertilizers is mined from unevenly distributed phosphate rock reserves concentrated in a few regions of the world for use in others, leading to local imbalances of this critical nutrient (Cordell, Drangert, and White 2009). Based on consumption rates and estimates of global reserves, it is expected that increasing P scarcity will place greater limitation on agricultural production in the future, a phenomenon known as "Peak Phosphorus" (Cordell and White 2014). These P imbalances can have dramatic effects on ecosystem services, especially in aquatic systems. Inland lakes receiving P inputs from agriculture rapidly undergo eutrophication, leading to algal blooms, low dissolved oxygen, and fish kills.

Potassium

Potassium plays an important regulatory role in plants and is involved in osmoregulation, protein synthesis, enzyme activation, and photosynthate translocation (Epstein and Bloom 2005; Marschner 1995). In fact, the cells of plants, animals, and humans all use K^+ in the cytosol to regulate their ionic or electrolyte balance. Leaf edges start to yellow and dry out if a plant is K-deficient as it cannot properly regulate water movement. Plants also tend to be more susceptible to frost and drought damage and certain diseases if they are K-deficient.

Mineral fertilizers contain potash (K_2O) to compensate for harvest K removal. Historically, K^+ was produced by leaching wood ashes and evaporating the solution in large iron pots, hence the name "pot-ash" (Mikkelsen and Bruulsema 2005). On a larger scale, K is released when forests were cut and burned to prepare fields (e.g., slash-and-burn or shifting cultivation). Clearly, none of these practices are scalable or sustainable. Currently, most K fertilizer (whether organic or inorganic) comes from evaporated marine salt deposited as inland seas (Mikkelsen 2007).

Factors influencing nutrient cycles in agroecosystems

Nutrient cycles are a function of biophysical factors, landscape arrangement, and farm management. Together, these factors govern the size of nutrient pools and the magnitude of nutrient fluxes within an agroecosystem (Figure 1). Important biophysical factors that regulate agroecosystem nutrient cycling include: soil texture, soil mineralogy, soil structure topography, depth to water table, local climate, and plant diversity. Soil mineralogy, especially below the plough-layer (~50 cm), can influence long-term nutrient supply and retention due to the presence or absence of exchange sites for mobile ions (Sollins, Robertson, and Uehara 1988; Wong, Hughes, and Rowell 1990). Soil texture and structure regulate the residence time of fertilizer in agricultural soils, as well as the movement of water and soluble nutrients. Soils with sandy textures are more prone to leaching losses than clayey textured soils (Weil and Brady 2015). Well-aggregated soils tend to have better nutrient retention compared to soils with poor physical structure. Topography and depth to the water table can affect movement of nutrients into waterways and sediment into surface waters. Current and future rainfall

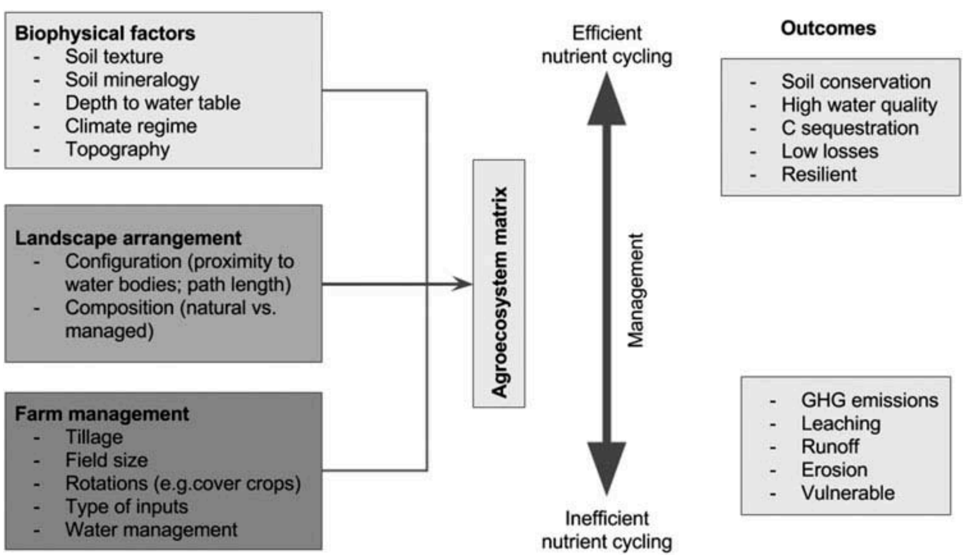

Figure 1. Impacts of biophysical, landscape arrangement, and farm management on nutrient cycling in agroecosystems.

and temperature regimes play a key role in the regulation of nutrient cycling in agroecosystems as rainfall tends to mobilize nutrients and temperature controls establishment, growth, and associated nutrient uptake by plants.

About 80% of agriculture worldwide is rainfed (Molden et al. 2011), while only 20% of agriculture is irrigated. However, most farmers employ some form or another of soil water management on their farms. Construction of ridges and furrows, stone bunds, terraces, contour furrowing, tied-ridging, and mulching is all forms of soil water conservation (FAO 2002; Rockström, Barron, and Fox 2002). The type and diversity of plant species present affect how and when nutrients are utilized across seasons. Perennial species, like fruit trees, tea, coffee, and perennial grasses, maintain biomass throughout the year and constantly uptake nutrients from the soil. Annual crops, like most grains, legumes, and vegetables, tend to take up the majority of their nutrients during a short time window that coincides with the vegetative growth stage of the plant (Russelle et al. 1981; Welch et al. 1971).

The configuration and composition of landscapes also alter the fate of agricultural nutrients. The diversity of land use types connected to a zone of agricultural production determines landscape composition. For example, a landscape dominated by agriculture with little or no natural habitat will pose a much larger threat to local water bodies than a forested landscape (Lamy et al. 2016; Thompson et al. 2016). However, most landscapes are complex mosaics, and thus, the composition of this mosaic will determine the rate and magnitude of nutrient losses based on the proportion of sources (e.g., farm fields) and sinks (e.g., natural areas). Riparian buffers and native habitats, like forest fragments, may reduce nutrient losses to surface waters by serving as nutrient and sediment sinks. However, the effectiveness of these buffers is highly variable across slopes, soils, and climates (Allaire et al. 2015). Improving the heterogeneity of agroecosystem configuration and composition can serve as insurance against natural and anthropogenic disturbances because they can preserve ecosystem functioning. Landscape configuration will also alter nutrient cycling because some areas will have an outsized effect on nutrient cycling due to their proximity to a water body or other natural system. Although agriculture as a whole is considered to be a non-point source of pollution to waterways, a farm close to a stream or creek may behave more like a point source of pollution. The ecological and economic benefits of complex landscape mosaics can be difficult to quantify, limiting the restoration of natural vegetation and wetland habitat within agroecosystems. In recent years, advances in economic evaluation tools and valuation of ecosystem services have helped to guide landscape conservation practices to remediate nutrient pollution (Roley et al. 2016).

Below, we review farm management strategies that promote tight nutrient cycling because (1) the effects of field-level changes have been well-described in scientific literature and through controlled, replicated experiments, (2) improved management practices can be implemented by farmers directly, and (3) policy and market incentives designed to mitigate nutrient pollution are typically constrained to a single land use or landowner.

Effects of agroecological practices on nutrient cycling

Farm management practices can help promote tighter nutrient cycling by improving the biological, physical, and chemical properties that regulate nutrient transformations. Greater nutrient retention can be achieved by reducing runoff, erosion, leaching, and soil greenhouse gas emissions and by enhancing soil organic matter content, microbial biomass, water holding capacity, and crop yields.

Erosion in agricultural systems is the result of soil loss via wind or water and is closely linked to soil tillage, which leaves soil bare prior to planting, between plants, and between seasons. Practices that reduce tillage and maintain crop residue on the field can reduce soil erosion and runoff, prevent evapotranspiration, and tighten nutrient cycling (Palm et al. 2014; Verhulst et al. 2010). Leaching is the downward movement of nutrients through the soil profile via infiltrating water. Thus, leaching rates are controlled mainly by the rate of fertilizer additions and soil infiltration rates. When soil nutrients are leached below the primary crop rooting zone, they are "lost" from the system. The presence of trees on farms may improve nutrient cycling in agroecosystems because the deep roots of trees are able to capture nutrients lost below the root zone of crop plants (e.g., safety-net hypothesis; Babbar and Zak 1995; Mekonnen, Buresh, and Jama 1997; Udawatta et al. 2002). Similar to leaching, agricultural emissions of nitrous oxide (N_2O) tend to be controlled by fertilizer application rate, soil water content, and soil C (Davidson 2009; Firestone and Davidson 1989). As N_2O is a significant contributor to global warming, even small reductions in emissions from agricultural soils will have a huge impact on both local and global nutrient cycles. Unfortunately, it is challenging to find agricultural practices that consistently reduce N_2O emissions, which are highly temporally and spatially variable (Basche et al. 2014; Venterea et al. 2012).

Maintaining or increasing soil organic matter (SOM) is a central principle of sustainable agroecosystem nutrient management because of its role in the biological, chemical, and physical functions of soil (Reeves 1997). Soil organic matter provides many ecosystem services from serving as binding sites for mobile NO_3 and PO_4, enhancing water holding capacity, and soil microbial biomass. Farm management practices that provide or maintain SOM can help mitigate nutrient losses by enhancing water and nutrient residence time and therefore optimize plant uptake (Franzluebbers 2002; Phillips et al. 1980; Tisdall and Oades 1982). Agroecological practices that enhance microbial biomass may improve internal nutrient cycling because of the important role microbes play in the breakdown of SOM and subsequent release of inorganic nutrients. Soil microbial biomass is often used as an indicator of changes in nutrient cycling because microbial biomass C and N pools are more responses to management practices than bulk soil C and N pools (Joergensen, Anderson, and Wolters 1995). Water holding capacity is the total amount of water that a soil can hold at field capacity, and it is controlled mainly by soil texture and soil organic matter content. Soils with higher water holding capacity allow more time for biotic nutrient uptake and abiotic transformations and thus may reduce N leaching losses. Finally, one of the best ways to balance both food and environmental objectives is to increase the amount of added or residual nutrients that are utilized by the crop (increase NUE). As most agroecosystems systems are only 30–50% NUE, there is a clear need to promote practices that promote both high yields and reduce environmental costs.

We focused our literature search on six management practices that, when employed across the greater agroecosystem, have been shown to vastly improve nutrient cycling compared to conventional practices: Intercropping; agroforestry; cover cropping; integrating crops and livestock; organic matter amendments; and conservation tillage. We looked for studies conducted in one of these six management systems and measured at least one of the seven indicators of nutrient cycling efficiency: (1) reduced runoff and erosion; (2) reduced leaching; (3) low greenhouse gas emissions; (4) improved C storage; (5) improved microbial biomass; (6) improved water holding capacity; and (7) improved yields. When reviewing the literature, we categorized our findings into different confidence levels of robustness of evidence and agreement across studies (Mastrandrea et al. 2010). There are three levels of agreement: low, medium, and high depending on the number of studies that had similar findings. There are also three levels of evidence (e.g., the type, quality, and consistency): limited, medium, and robust. We considered there to be high agreement and high evidence when at least three published field studies showed that a particular management strategy *enhanced* the same indicator of nutrient cycling (e.g., reduced N leaching losses). We considered there to be low agreement and high evidence if a particular management practice suppressed the same indicator of nutrient cycling (e.g., increased N_2O emissions). We considered there to be low evidence and low agreement when multiple studies showed conflicting results on the impact of a management strategy on a particular indicator of nutrient cycling. Findings with a high agreement and robust evidence indicate a high level of confidence. On the other hand, findings with low agreement and limited evidence indicate a low level of confidence.

Intercropping

Intercropping is the practice of planting two or more plants at the same time in the same field. In some cases, all are crop plants (e.g., maize, beans, and squash). In other cases, a non-crop plant is cultivated because of beneficial services it provides to the main crop. For example, *Desmodium* sp. grown alongside maize can reduce parasitic witchweed (*Striga hermonthica*), which can devastate low-input, smallholder farms (Khan et al. 2002). The term intercrop is used to describe many different systems, some of which are discussed in other sections of this review (e.g., agroforests, cover crops, and legumes). Here, we consider studies that focus only on the co-planting of two or more harvestable crop plants.

Much of the literature on intercrops focuses on yields and land use efficiency as measured by the land equivalent ratio (LER), which is an expression of the land required to produce the same yield in sole crops compared to intercrops (Equation (1)).

$$L_{crop1} = \frac{Ycrop1IC}{Ycrop1SC}; L_{crop2} = \frac{Ycrop2IC}{Ycrop2SC}; LER = L_{crop1}{}^{+}L_{crop2} \quad (1)$$

where Ycrop1IC and Ycrop2IC are the intercrop yields, and Ycrop1SC and Ycrop2SC are the sole crop yields of crop 1 and crop 2, respectively. L_{crop1} and L_{crop2} are called partial LERs, which are then summed for the total LER for the intercrop. When analyzing LERs, it is important to note that some farmers may care more about the yield of one crop compared to the other or that they may only be concerned about the overall field yield. However, since the focus of this review is on nutrient cycling rather than the production efficiency of intercropping systems, we do not provide a comprehensive review of LER. Overall, we found that intercropping led to higher yields than when crops were grown in monoculture (Fan et al. 2016; Ghaley et al. 2005; Hauggaard-Nielsen, Ambus, and Jensen 2003; Mapa 1995; Mariotti et al. 2016; Rusinamhodzi, Murwira, and Nyamangara 2006; Sharma and Banik 2014). In many cases, crops performed better individually (high partial LER) when intercropped than when grown in monoculture. This is similar to a recent review that found wheat intercrops consistently produced high LERs (Aziz et al. 2015). We note here that high LERs may likely be correlated with high nutrient use efficiency.

Many intercrops consist of a grain and legume mix, and we found that legumes tend to perform better in intercropped systems at low fertilizer rates because biological N fixation (BNF) is suppressed at high N availability. When BNF is suppressed, this gives the cereal a competitive advantage, which reduces the performance of the legume. Further, we found that legumes tended to derive more of their N from fixation when intercropped than when grown in monocultures. On average, the percent N derived from N fixation (Ndfa) was higher in intercropped legumes (90%) compared to sole cropped legumes (74%; Ghaley et al. 2005; Hauggaard-Nielsen, Ambus, and Jensen 2003; Mariotti et al. 2016; Pelzer et al. 2012; Rusinamhodzi, Murwira, and Nyamangara 2006). This suggests that there is an enhanced nutrient efficiency when grains and legumes are intercropped as the legume is "forced" to supply its own N, leaving more N available to the cereal and perhaps reducing the need for high fertilizer application rates.

While there was no significant effect of intercrops on soil C sequestration, microbial biomass tended to be enhanced in intercrops compared to monocultures (Rivest et al. 2010; Sharma and Banik 2014; Zhou, Yu, and Wu 2011), but the community structure differed among the different intercrops (Song et al. 2006; Tang et al. 2014). Intercrops tended to show a higher N recovery than sole crops, which might suggest that intercrops had smaller N losses through leaching and soil emissions. Research on leaching from intercrops tended to focus on the loss of N following the cropping cycle (sole crop vs. intercrop), and although there was some evidence that leaching losses were smaller following intercrops (Kanwar et al. 2005; Mariotti et al. 2016; Nie et al. 2012), other studies found no effect at all (Amossé et al. 2013; Hauggaard-Nielsen, Ambus, and Jensen 2003). Similarly, there was no strong effect of intercropping on erosion or runoff control and that rates were species-specific (Bertol et al. 2013; Fan et al. 2016; Lima et al. 2014; Oshunsanya 2013; Salah, Prasse, and Marschner 2016; Zougmore et al. 2000). There were very few studies that examined N_2O emissions in sole vs. intercrop systems, and emissions were crop-specific (Cruvinel et al. 2011; Dyer, Oelbermann, and Echarte 2012; Nie et al. 2012).

The literature provides high agreement and robust evidence that microbial biomass is higher in intercrops compared to monocultures, but there is little evidence that this leads to higher soil organic matter pools in intercropped systems. Further, there are low evidence and low agreement on the effect of intercrops on erosion, although some evidence suggests that compared to monoculture cereals, intercrops can reduce erosion. There were minimal evidence and low agreement that intercropping reduced N losses via leaching or soil emissions (Table 2).

Agroforestry/alley cropping/intercropping with trees

Agroforestry is the simultaneous cultivation of woody plants (trees or shrubs) and crops. The understory crop may consist of annual crops (e.g., maize, cassava) or perennial crops (e.g., coffee or cacao). Trees are planted on farms for many reasons: supplementary income (e.g., fruit or timber), nitrogen fixation (e.g., green manure), wind breaks, and nutrient and sediment traps on the edges of fields. The inclusion of N-fixing legume trees, such as *Erythrina poeppigiana*, will enhance N cycling (Harmand et al. 2007; Tully, Lawrence, and Wood 2013a) by providing organic material with a high N content. Deeply rooted trees are able to capture nutrients and reduce N leaching loss compared to tree-less monocultures (Babbar and Zak 1995; Mekonnen, Buresh, and Jama 1997; Udawatta et al. 2002). When planted on

Table 2. Summary table of evidence and agreement from the literature.

	Reduced runoff and erosion	Reduced leaching	Low GHG emission	Improved C storage	Improved microbial biomass	Improved WHC	Improved yield
Intercropping	H evidence H agreement	L evidence L agreementt	L evidence L agreement	L evidence M agreement	H evidence H agreement	L evidence L agreement	H evidence H agreement
Agroforestry	H evidence, H agreement	H evidence H agreementt	M evidence L agreement	H evidence, H agreement	H evidence H agreement	H evidence H agreement	M evidence L agreement
Cover crops	H evidence H agreement	H evidence M agreement	H evidence L agreement	H evidence H agreement	H evidence H agreement	L evidence H agreement	H evidence L agreement
Mixed cropping-livestock	L evidence L agreement	L evidence L agreement	L evidence L agreement	M evidence M agreement	L evidence L agreement	L evidence L agreement	L evidence M agreement
OM amendments	H evidence H agreement	L evidence M agreement	M evidence M agreementt	M evidence H agreementt	L evidence L agreement	H evidence H agreement	H evidence M agreement
Cons. tillage	H evidence H agreement	L evidence M agreement	L evidence L agreement	M evidence M agreement	M evidence M agreement	H evidence H agreement	H evidence M agreement

The table shows how agroecosystem management practices affect different indicators of efficient of nutrient cycling, where L = low, M = medium, and H = high/robust. The shade of gray indicates the confidence in that effect of a particular management practice on an indicator of efficient of nutrient cycling with dark gray representing the highest confidence and light gray representing the lowest confidence.

the edge of fields, trees can reduce runoff and erosion and N loading in streams (Lamichhane 2013; Udawatta et al. 2002). One way trees reduce runoff is through water uptake. Trees can compete with crops for water and studies show conflicting results—with some indicating that trees reducing soil moisture in the immediate root zone (Govindarajan et al. 1996), and others showing higher soil moisture under trees than annual crops (Chirwa, Nair, and Kamara 1994). Overall, the presence of fine roots may increase the overall water holding capacity of soils in farms with trees and other perennial plants (Lamichhane 2013; Rosecrance, Rogers, and Tofinga 1992; Velmourougane 2015). In the tropics, trees are often planted on farms to buffer financial and climate risk by providing a second crop or providing microclimatic conditions that support crop growth (Huxley 1999). In temperate climates, trees are often planted on farms to mitigate negative environmental impacts and improve real estate values (Williams et al. 1997).

The orientation and management of trees on farms play a major role in how nutrients are cycled throughout the system. For example, in commercial coffee agroforests (e.g., "shade-grown coffee"), trees are planted in the same row as coffee plants every 6–10 m (Tully, Wood, and Lawrence 2013b). In contrast, in alley cropping systems, trees are managed as hedgerows and annual crops (like maize) are planted in the "alleys" between the tree rows. In both cases, trees are periodically pruned (1–3 times per year; Tully, Wood, and Lawrence 2013b), and prunings are used as green manure (if incorporated into the soil) or mulch (if left to decompose on the soil surface). "Cut-and-carry" systems, such as those in East Africa, incorporate N-fixing legume trees along farm boundaries, but crops are not grown in the shade of trees. In these systems, nutrient-rich, fresh leaves are *cut* from the border trees, *carried* and to the field to be used as green manure (Rufino et al. 2006). Similar to crops, trees are managed on an intensification spectrum. For example, coffee agroforests range from the inclusion of coffee bushes planted under partially cleared forests, to traditional polycultures, to commercial polycultures, to shaded monocultures (Perfecto et al. 2003, 2005). Not surprisingly, the quantity of C stored in aboveground biomass tends to decrease along this spectrum. Microbial biomass also tends to increase in agroforests compared to monocultures (Chander et al. 1998; Hergoualc'h et al. 2008; Mazzarino, Szott, and Jimenez 1993; Paudel et al. 2011; Ramesh et al. 2013; Tian, Cao, and Wang 2013; Unger et al. 2012; Velmourougane 2015). There are few studies that estimate soil C storage in temperate agroforestry systems (Howlett et al. 2011; Mosquera-Losada, Freese, and Rigueiro-Rodríguez 2011; Upson and Burgess 2013), and in some cases, these studies lack control plots without trees, but show elevated SOC in temperate agroforestry systems (Cardinael et al. 2017). High leaf litter and root-turnover (even when grown in monoculture) can enhance soil C pools and internal nutrient cycling. Agroforestry systems have thus been lauded for their ability to sequester C in both plants and soil (Lenka et al. 2012; Nair 2011; Roshetko, Delaney, and Hairiah 2002; Sharrow and Ismail 2004).

In contrast to N leaching losses, there is evidence that N_2O emissions are enhanced in tropical agroforestry systems compared to monoculture systems (Babbar and Zak 1995; Dick et al. 2006; Guo et al. 2008; Hergoualc'h et al. 2008) likely due to the provisioning of N and C through litterfall and prunings. However, these gaseous fluxes represent a small part of the overall N balance (~6 kg N/ha/yr) compared to N exported in the harvest (79 kg N/ha/yr) and N lost through leaching (119 kg N/ha/yr; Rosenstock et al. 2014). Further, some studies did not measure N_2O at a frequency adequate to capture episodic pulses of gaseous N loss due to mineralization or rainfall events, which means that interpolation could lead to erroneous estimates of cumulative N_2O emissions on an annual basis (Rosenstock and Tully unpublished data). In a temperate riparian buffer, N_2O emissions were lower than in adjacent crop fields because buffers were unfertilized and due to tree N uptake (Dong-Gill 2008).

Yields tended to be lower in agroforestry systems (Odhiambo and Bomke 2001; Soto-Pinto, Perfecto, and Castillo-Hernandez 2000; Tully, Lawrence, and Scanlon 2012). For organic coffee, this yield loss is offset by price premiums for organic/shade-grown coffee. In subsistence systems, this is a harder sell. Farmers have to sacrifice yield when they take land out of production for an improved fallow, for example. However, agroforestry can improve yields on "vulnerable" farms with even small inputs of green manure from legume trees (Tully et al. 2015). This suggests that at high-levels of productivity, farmers may see a yield reduction, but at low-levels of productivity, agroforestry can help improve yields.

The scientific literature provides robust evidence and high agreement that agroforestry practices can sequester C, enhance microbial biomass, improve water holding capacity, and that they attenuate soil erosion and runoff and nutrient leaching compared to traditional monoculture systems (Table 2).

Cover crops

Cover cropping is an ancient farming technique, dating back 3,000 years to the Chou dynasty in China (Pieters 1927) where it was used to improve soil fertility. Cover cropping is the practice of planting a "service" crop at a time of the year when a cash crop is not grown. In temperate systems, cover crops tend to be planted in the fall to capture nutrients during the winter months. In tropical systems, where temperature doesn't inhibit crop growth, "improved fallows" may be planted during the dry season or for several years to replenish soil fertility. Improved fallows are the deliberate planting of fast-growing species (often legumes), which can be either herbaceous (e.g., *Mucuna pruriens*) or woody species (e.g., *Gliricidia* *sepium*; Sanchez 1999). Improved fallows that include woody species are sometimes considered agroforests.

Unlike agroforests, temperate cover crops include annual species typically planted in the fall after the cash crop harvest and removed in the spring before the next cash crop planting. Cover crops are lauded for their ability to reduce soil erosion, runoff, and weed pressure, and enhance biodiversity. However, the types and extent of services provided by cover crops vary by species and management such as planting and kill dates (Clark et al. 1997; Lawson et al. 2015), and method of cover crop incorporation (Wortman et al. 2013). Cover crops accumulate larger quantities of above- and belowground biomass enhancing soil organic matter provisioning. Studies show that cover crops can sequester soil C (Bayer et al. 2009; Reicosky and Forcella 1998; Sainju and Singh 1997; Sainju, Whitehead, and Singh 2005) and enhance microbial biomass (Chirinda et al. 2010; Sanz-Cobena et al. 2016; Steenwerth and Belina 2008).

In temperate systems, rainfall in the fall and spring leads to high N leaching. Fall and winter annuals are often planted to capture residual soil N following cash crops and to reduce drainage by increasing evapotranspiration (Meisinger et al. 1991). Some crops will winter-kill (e.g., rape and radishes), and others will overwinter and grow rapidly in the early spring (e.g., ryegrass and oats). Cover crops are killed in the spring (either mechanically or chemically), and organic N stored in the plant tissues is mineralized and can be made available to the next cash crop. Similar to inorganic fertilizers, if cover crop-derived nutrients are mobilized when there is no crop to use them, they are prone to leaching and gaseous losses. Thus, there is a lot of interest among agroecologists to better synchronize nutrient release from cover crop residue to cash crops (Wagger 1998).

Legume cover crops can be incorporated into rotations as an additional N source or green manure. Winter annual legumes can provide 50–150 kg of plant available N per hectare (Cook et al. 2010; Ebelhar, Frye, and Blevins 1984; Ranells and Wagger 1996). Addition of a legume cover can be used to offset mineral fertilizer application, although cash crop yields tend to be lower (~10%) when grown under 100% green manure treatments compared to crops receiving 100% mineral fertilizer (Tonitto, David, and Drinkwater 2006). Legume tissues are of higher quality (lower C:N) than grasses, leading to higher N availability following legume covers (Clark et al. 1997; Ranells and Wagger 1996). However, higher N availability can also lead to higher N losses. For example, non-legume covers reduced N leaching by 70% while legume covers only reduce N leaching by 23% (Meisinger et al. 1991). The research on leaching indicated that N is generally reduced compared to monoculture systems (Feaga et al. 2010; Restovich, Andriulo, and Portela 2012; Snapp et al. 2005), but that the magnitude of reduction is highly dependent on species and termination date (Heinrich, Smith, and Cahn 2014; Herrera et al. 2010; Sainju and Singh 1997). Inconsistencies in the effect of cover crops (and other practices) on N leaching may also be due to the fact that there is no standard method for measuring soil solution N in the field (Fares, Deb, and Fares 2009; Lamba et al. 2013; Siemens and Kaupenjohann 2004; Tully, Wood, and Lawrence 2013b; Zhu, Fox, and Toth 2003) or modeling water flux (Ajdary et al. 2007; Groenendijk et al. 2014; Moreels et al. 2003; Perego et al. 2012; Van Der Laan et al. 2014).

Overall, studies show that cover crops reduce erosion and runoff (Battany and Grismer 2000; Malik et al. 2000; Snapp et al. 2005). N_2O emissions also tend to be higher in legume vs. non-legume cover crops (Basche et al. 2014; Gomes et al. 2009; Pappa et al. 2011; Sanz-Cobena et al. 2016) although others showed no effect (Chirinda et al. 2010; Parkin and Kaspar 2006). Cover crops can alter denitrification potential by altering (1) soil mineral N content, (2) soil moisture via evapotranspiration losses, (3) soil moisture during the mulch phase, and (4) soil organic C sources. As cover crop species and management will have a strong impact on these drivers, the effect of cover cropping on N_2O emissions differs greatly among studies. About 40% of studies found that cover crops suppressed N_2O emissions and 60% increased N_2O emissions compared to a no-cover control (Basche et al. 2014).

The literature provides robust evidence and high agreement that cover cropping can reduce erosion and runoff, sequester soil C, and enhance soil microbial biomass compared to control plots. In general, there is robust evidence that cover crops tend to reduce N leaching; however, the overall reduction in leaching varies with residual N levels and species (medium agreement on their efficacy). There is robust evidence, but low agreement on whether cover crops enhance or reduce N_2O emissions (Table 2).

Integrating livestock and cropping systems

Consumption of meat, dairy, and other animal products is increasing rapidly throughout the world. This trend is largely attributed to a growing global population and shifts in dietary preferences. The livestock sector has large global nutrient, water, and energy footprints (Steinfeld and Wassenaar 2007), and the spatial decoupling of crops and livestock exacerbate regional nutrient imbalances (Hilimire 2011). In decoupled production systems, animals are raised in concentrated conditions and rely on imported feed grains and mineral supplements that are typically produced with synthetic fertilizers. Concentrated livestock operations produce a disproportionate amount of manure leading to over-application and/or inefficient reuse of manure nutrients (Kleinman et al. 2012). Incorporating crops and livestock production is one strategy to minimize environmental harm and maximize environmental benefits of livestock production through the tighter coupling of biogeochemical cycles. Integrating crop and livestock production at the farm level can also reduce the need for external inputs of fertilizer and feed (Regan et al. 2017).

A large portion of the world's agricultural production relies on some degree of integration of livestock and cropping components, known as "mixed agroecosystems" (Gerber et al. 2013). Smallholder mixed agroecosystems systems are still very common in Africa, Asia, and Latin America and often called agropastoral, silvopastoral, or agro-silvopastoral systems which combine crops and livestock, trees and livestock, and crops, trees, and livestock, respectively. As countries and economies develop, livestock farming systems shift from pastoral to mixed agroecosystems and then to industrialized livestock systems (Herrero et al. 2016). Sustainable mixed agroecosystems are characterized by a balance in animal numbers with the quantity of feed crops grown in the system (Malcolm et al. 2015). Integration of animal and crop enterprises occurs at different levels from a single farm to regional cooperation of farmers across a region (Bell and Moore 2012; Regan et al. 2017). Mixed agroecosystems can tighten nutrient cycles across large geographic spaces by reducing dependency on synthetic fertilizer inputs. At the farm-scale, nutrient flows are altered by high diversity of grain and forage crops and by the reuse of organic nutrients in manure. Indirect impacts of manure on soil cation exchange capacity, water infiltration, and soil structure can also increase the ability of a soil to retain nutrients.

While conceptual models predict more efficient nutrient cycling in mixed agroecosystems, the use of livestock into an agroecosystem may still contribute to nutrient losses. Manure is a significant source of CH_4 and N_2O emissions resulting from its handling, storage, and land application. These nutrient losses occurring at the farm-scale are difficult to quantify and have the potential to offset some or all of the benefits from reducing fertilizer inputs. As with cropping systems, there are several farm management practices that mitigate nutrient losses from manure management and reuse. In dairies and other systems where manure can be collected, storage practices can be optimized to reduce N losses from volatilization and leaching (Owen and Silver 2014). Where grazing results in urine patches and hotspots of N activity, nitrification inhibitors can be used to prevent the formation of nitrate and subsequent leaching losses (Cai and Akiyama 2016).

Over the last several decades, the abundance of mixed agroecosystems has declined in response to trade and technological opportunities and replaced with production systems in which animal and crops are decoupled in space and management (Wilkins 2008). In areas of rapid expanse of decoupled cropping systems, fertilizer use tends to increase disproportionately, following the removal of livestock (Gavier-Pizarro et al. 2012), in addition to reduced area and vegetative diversity of surrounding pasture areas

(Carvalho and Batello 2009). However, revitalization or modernization of mixed agroecosystems is thought to be an important pathway to achieve the dual goals of food security and environmental sustainability (Herrero et al. 2016; Lemaire et al. 2014).

There is high agreement but a lack of robust evidence in the literature that the integration of livestock and cropping systems tightens nutrient cycles and reduces losses to the environment. The body of scientific research focusing on the environmental impacts and life cycle assessments of livestock production systems has grown rapidly over the last decade. Much of this research has focused on feed use efficiency and other technologies used to improve the resource use efficiency as a function of product output (e.g., (Aguerre et al. 2011). However, there are critical gaps in our understanding of the impacts and potential benefits of integrated crop-livestock systems on soil C sequestration, water infiltration, or the provisioning of other ecosystem services.

Organic matter amendments

Amending agricultural soils with organic material can improve nutrient management directly by supplying a complex source of nutrients in soil and indirectly by improving soil properties. Organic amendments that are diverted from waste streams and recycled can also improve lifecycle greenhouse gas and nutrient footprints (DeLonge, Ryals, and Silver 2013). Organic amendments can be sourced from diverse waste streams, including animal and human excrement, crop residues, yard debris, and waste from food processing industries. Organic amendments can be applied to soil in their raw forms or after treatment, such as composting, anaerobic digestion, pyrolysis, gasification, or solid/liquid separation. In tropical countries with rapidly growing population and consumption patterns, recycling of organic wastes via composting, biochar, and mulching have been proposed as a strategies to achieve multiple goals of sustainable waste management, reduced dependency on fertilizer inputs, and improving fertility of highly weathered, nutrient-poor soils. Farmers in these regions who adopt organic matter amendments do so because of improvements to soil; however, adoption rates are limited by the labor requirements of manual application, lack of specialized machinery, cost, and lack of information about quality (Paul et al. 2017). Differences in the source and treatment of organic matter amendments results in a wide range of chemical, biological, and physical characteristics that, in turn, can have differing impacts on nutrient cycling and loss pathways.

Organic forms of nutrients provide a slow-release of nutrients compared to highly soluble and reactive, synthetic fertilizers. Low mineralization rates supply a consistent source of nutrients to the soil solution for plant to utilize over time. Initial C:N and decomposition rate of the amendment regulate immobilization and mineralization rates. Organic amendments with initial C:N ratios exceeding 20:1 tend to immobilize N, P, and S during the initial phase of decomposition, making these nutrients less immediately available for plant growth or gaseous and aqueous loss pathways (Hadas et al. 2004).

Organic matter is a large source of cation exchange capacity (Schulten and Schnitzer 1997). Therefore, organic matter amendments or other practices that increase soil organic matter greatly improve the soil's ability to adsorb and resupply nutrients. Organic matter amendments improve soil structure by lowering bulk density, improving aggregation, and increasing water holding capacity in most soils (Jeffery et al. 2011; Zebarth, Neilsen, and Hogue 1999). Improvements in the physical properties of soil reduce risks of soil erosion, promote water infiltration, and reduce leaching losses. However, over- or mis-application of organic material can result in nutrient N and P runoff and leaching, posing risks to water quality (Carpenter et al. 1998). Management practices that incorporate manure reduce risks of nutrient runoff but increase vulnerability of erosional losses of N and P.

Organic matter amendments are rich in C—an essential energy source for soil microbes that cycle nutrients. Compared to inorganic fertilization, long-term fertilization with organic soil amendments has been shown to boost microbial biomass, rates of extracellular enzyme activities, and favor more diverse soil microbial communities (Francioli et al. 2016). In semi-arid climates, soil amended with organic matter foster microbial communities that are more resistant and resilience to drought stress (Hueso, Hernández, and García 2011; Sun et al. 2017), suggesting the important of this practice a climate change adaptation strategies. Organic amendments can also mitigate climate change via C sequestration (Ryals and Silver 2013). Carbon sequestration can be achieved through organic matter amendments when ecosystem C uptake via photosynthesis exceed losses of C via soil heterotrophic respiration and dissolved organic C losses, or when mean residence time of soil C increases (Paustian et al. 1997). The potential for C sequestration or loss with organic amendments varies among field studies and is sensitive to the quality of the organic inputs (Berti et al. 2016), soil characteristics, and other management practices. Compost additions to grazing lands were shown to increase ecosystem C storage (Ryals and Silver 2013) and increase C stored in physically protected soil fractions (Ryals et al. 2014). In contrast, fresh plant residues decompose more quickly and may not lead to net increases in soil C.

There is high agreement in the literature the organic matter amendments increase soil water holding capacity (Gomiero, Pimentel, and Paoletti 2011), improve aggregation and soil structure (Udom, Nuga, and Adesodun 2016), decrease bulk density (Reeves 1997), and promote diverse, resilient soil microbial communities (Francioli et al. 2016; Sun et al. 2017). Plant uptake and nutrient loss pathways from organic soil amendments are primarily a function of soil texture, the characteristics of the amendment, and the timing and method of application. Organic matter amendments have been shown to increase, decrease, or result in no change in greenhouse gas emissions compared to inorganic fertilizers (Paustian et al. 2016). Compost and biochar amendments tend to have lower soil greenhouse gas emissions relative to fresh or slurried manures. Manure amendments tend to have large, labile pools of ammonium and are a large, global source of soil N_2O, NH_3 volatilization, and N leaching (Castán et al. 2016; Davidson 2009). Over short timescales, crop yields can either increase if the amendment relieves nutrient or water limitations (Ryals et al. 2016) or decrease if the amendment immobilizes nutrients (Hadas et al. 2004). Long-term studies with single or repeated organic amendments suggest that elevated crop yields can be sustained over time due to the indirect improvements in soil biological, chemical, and physical properties (Diacono and Montemurro 2010).

Conservation tillage

Conservation tillage refers to a range of soil tillage practices that reduce or eliminate physical turnover of the soil and leave crop residues on the soil surfaces. The primary objective of conservation tillage is to conserve soil resources. Examples of conservation tillage methods include no-till, minimum till, ridge and furrow till, chisel plow, and mulch till. In the US, conservation tillage is operationally defined as a tillage system that maintains at least 30% of ground covered by crop residues (Unger, Stewart, and Singh 1991). However, broader definitions that emphasize the end goal of soil conservation, rather than strict adherence to residue requirements, have been adopted in semi-arid regions where residues are limited and are used to feed livestock (Nyakudya and Stroosnijder 2015). Conservation tillage was first implemented in response to large soil erosion events (e.g., The Dust Bowl). Conservation tillage has widespread implementation, particularly in high-input, large-scale, mechanized agricultural systems in temperate climates and high-yielding smallholders rice-wheat region of South Asia (Palm et al. 2014). Promotion and adoption of conservation tillage by smallholder farmers in low-input, low-yielding systems have increased in recent years, but are limited by socioeconomic constraints and uncertainties in crop yield responses (Kassam et al. 2009). Low amount of plant residuals, reliance on manual or animal labor, and limited access to markets further constrain widespread adoption of conservation tillage in these regions (Thierfelder, Cheesman, and Rusinamhodzi 2012).

There is an appreciable amount of scientific literature documenting the beneficial impact of conservation tillage on soil structure and concomitant reductions in runoff and erosion rates and increases in water infiltration (as reviewed by Palm et al. 2014). A multitude of paired comparisons shows significant decreases in erosional losses at the plot or field scale (Lal, Reicosky, and Hanson 2007). At the catchment scale, soil erosional losses from conservational tillage were lowered by an order of magnitude compared to conventional tillage (Prasuhn 2012).

Crops grown with conservation tillage practices have variable yield responses compared to conventional tillage systems (Pittelkow et al. 2015b). Yield responses between tillage systems are a function of crop type, climate, soil type, other management practices, and duration of tillage management. A recent global meta-analysis of 678 studies found variable crop yields responses in no-tillage compared to conventional tillage systems, with no yield difference for oilseed, cotton, and legume crops, small yield declines for wheat, and a larger decline for rice and maize (Pittelkow et al. 2015a). Greatest declines in crop yield under no-tillage occur in tropical and subtropical climates, and the yield gap can be remedied with low rates of N fertilization (Lundy et al. 2015). Yield advantages with conservation tillage tend to be greater in low rainfall conditions (Nyakudya and Stroosnijder 2015). In semi-arid climates, higher soil water retention can enable crops to survive under low rainfall or drought conditions (Biazin et al. 2012), suggesting that conservation tillage may be an important climate change adaptation strategy in more vulnerable, drier regions of the world (Pittelkow et al. 2015a).

There is high confidence that conservation tillage increases topsoil organic matter compared to conventional tillage, primarily due to improved aggregate stability and residue retention and decomposition. Soil microbial composition and activities are highly regulated by the physical impacts of tillage. Conservation tillage tends to increase soil microbial biomass and the activities of extracellular enzymes involved in nutrient acquisition, and lower rates of soil organic matter mineralization (Raiesi and Kabiri 2016).

Conservation tillage systems are widely touted as C sequestering, yet field research studies are not in complete agreement. There are many studies documenting increases, decreases, and no effect on soil C with conservation tillage compared to conventional tillage. These differences are attributed to variations among conservation tillage systems, amount of residue cover, climate, and whether cover cropping or other conservation practices were used in concert with conservation tillage.

Leveraging agroecology to improve nutrient cycling

The management strategies described above (intercrops, agroforestry, cover crops, organic amendments, integrated crop-livestock, and conservation tillage) all have the potential to tighten nutrient cycling in agroecosystems. However, in order to balance food and environmental objectives, several of them should be employed simultaneously on farms. Research designs tend to isolate one or two practices and test the effect of those practices on one or two metrics for improved nutrient cycling. Studies that examine multiple metrics and compare across many strategies are rare (DeLonge, Miles, and Carlisle 2015). Nevertheless, some combinations of management practices have been described in the literature and show great promise for improving the efficiency of nutrient cycling without reducing crop yields. We describe three potential combinations: reduced-tillage organic agriculture combined with perennial crops; integrated crop-livestock systems with cover crops; and organic matter amendments to agroforest soils.

Reduced-tillage organic agriculture combined with perennial crops

Organic farming prohibits the use of synthetic herbicides; thus, organic farmers tend to rely on intensive tillage operations to control weeds (Armengot et al. 2012). However, farmers and researchers are eager to develop reduced-tillage strategies for organic farming systems (OFRF 2004; Sooby, Landeck, and Lipson 2007). Current strategies include (1) cover crop-based organic rotational no-tillage, which adopts planting and weed control practices that minimize tillage operations over the course of a crop rotation and (2) incorporating a perennial crops into the rotation, thereby avoiding tillage for the duration of the perennial crop. Organic grain growers often depend on legume cover crop rotations as a source of organic N to the subsequent cash crop; thus, cover cropping with legumes is common in organic grain systems. Research has also shown that integrating herbaceous perennials (e.g., alfalfa) into annual organic grain rotations can increase corn grain yield and economic stability while reducing weed pressure, predicted soil erosion, animal manure inputs, and soil P loading (Cavigelli et al. 2013).

Integrated crop-livestock systems with cover crops

Mixed agroecosystems have the potential to combine many of the practices we describe above. For example, a variety of cover crops can be used as forage for dairy cows such as, triticale, a rye and wheat hybrid (Houser, Harkcom, and Hall 2012 & 2013; Long, Ketterings, and Czymmek 2013). The addition of trees on pasture can provide shade to cattle. Leguminous trees (which are common in the tropics) can improve feed quality and digestibility (Thornton and Herrero 2014) and increase species richness and abundance at the farm-level (Kremen and Miles 2012) while also providing climate change mitigation benefits (Bryan et al. 2013; FAO 2010).

Organic matter amendments to agroforest soils

Trees provide multiple ecosystem services, including creating microclimates, providing food for pollinators, and reducing nutrient loss via leaching and erosion. The inclusion of woody perennials within any of the farming systems we describe will have positive environmental outcomes (Zomer et al. 2009). Shade trees in agroforestry systems can be pruned to moderate light availability to the understory crop. Tree pruning residues can be further treated and stabilized as mulch or compost and returned to the system as a soil amendment. There are few studies comparing organic and conventional agroforestry systems. However, some research has shown similar soil C in organic vs. conventional agroforests as both systems receive high leaf litter inputs (Tully, Lawrence, and Wood 2013a). Novel organic matter additions, like biochar, to agroforestry systems have the potential to alter soil biophysical and chemical properties, improve nutrient cycling, and increase LER (Deng et al. 2016).

Research and management opportunities and challenges

Significant advances in understanding of ecological principles underpinning nutrient cycling in agroecosystems have greatly improved our ability to balance food and environmental objectives. Despite these advances, there remain important uncertainties the hinder the evaluation and recommendation of sustainable nutrient management practices. Based on this literature review, we identify four broad priority areas to focus future research efforts.

(1) *Standardized field studies replicated across different soil and climate types* are needed, with a focus on aspects of nutrient cycling and management with low evidence and/or low agreement. For instance, the effect of cover crops, intercrops and mixed-crop livestock systems on water holding capacity is not well-defined. As water holding capacity is a strong driver of nutrient and water movement in the plant-soil system, research should focus on enhancing our understanding of how these practices can be optimized to improve water retention. Research opportunities also exist for better understanding the how mixed-crop livestock systems and intercrops can reduce nutrient leaching. Finally, the greatest uncertainties and inconsistencies in the literature surround N leaching and N_2O emissions from different cropping systems. Measuring leaching can be a great challenge in the field, and there is no standard method for measuring concentrations or water flux. Inconsistent trends in N_2O emissions among studies were often due to infrequent measurements (i.e., poor quality data), extrapolation beyond a reasonable period, or a lack of agreement on the effect of these practices. Many studies examine the effect of management on N_2O emissions, but there is a clear lack of agreement on how best to manage systems to suppress emissions.

(2) *Consistent metrics for evaluating nutrient cycling efficiency* are essential for comparing agroecological approaches. This challenge goes deeper than simply the multitude of ways studies can report NUE (Ladha et al. 2005). Methods for measuring leaching and N_2O may make the data difficult to compare (Russo and Tully *in review*; Rosenstock and

Tully *unpublished data*), and the evaluation of carbon C changes is confounded by flaws in C accounting methodology (Palm et al. 2014). Still, comprehensive reviews such as this one go a long way towards evaluating where opportunities exist to optimize these management practices for multiple ecosystem services.

(3) *Nutrient cycles should be considered at multiple temporal and scales.* Research that links farm-scale management to landscape-scale environmental outcomes will more effectively identify those practices that balance both food and environmental objectives (Wezel et al. 2016). Systems-based and whole-farm studies are challenging to conduct, replicate, and fund (DeLonge, Miles, and Carlisle 2015). Thus, the appropriate scale and characteristics of a mixed agroecosystems remain highly uncertain and may vary considerably across climate, soil type, and land use history. Recent advances in whole-farm models of nutrient budgets, including the Farm Energy Analysis Tool (FEAT) and Manure-DNDC, provide a helpful method for evaluating nutrient flows and losses within a diversified crop-livestock farm (Li et al. 2012; Malcolm et al. 2015). Likewise, life cycle assessments are continually refined to compare different agricultural production systems and to estimate ecosystem services (Rowntree et al. 2016). These tools can be further strengthened with more empirical research studies, systems-based studies, and evaluations of social dynamics among cooperating farmers, and by including other nutrients alongside C and N.

(4) *Coupling of ecological information with economic, social, and cultural information* is needed. Integration of information and perspective will help land managers and decision-makers implement practices that go beyond the farm-scale and consider how the configuration and composition of the landscape can be utilized to improve the efficiency of nutrient cycling. We cannot expect farmers to change their practices without public and private support. Farmers, citizens, and policy-makers must collaboratively use science, technology, and policy instruments to support local adoption of sustainable management practices in order to affect a global change.

Summary and conclusions

In the face of a growing human population and shifting diets, we must manage our agricultural systems to promote both food and environmental objectives. Nutrient cycles are the currency by which inputs can be tracked through agroecosystems and are a useful tool for evaluating the efficiency of these systems. We focused on several ecosystem services that help promote efficient nutrient cycling, yet there are many others (e.g., promotion of biodiversity, pollination, disease control), which were beyond the scope of this study.

Overall, we found that organic amendments and conservation tillage practices promoted water holding capacity. Runoff and erosion control are best promoted by agroforestry and cover crop systems, and agroforestry was the only practice that consistently reduced nutrient leaching over conventional practices. Agroforestry and cover crops improved carbon sequestration, and agroforestry, cover crops and intercrops all enhanced microbial biomass compared to conventional monoculture systems. On the other hand, agroforestry tends to suppress harvestable yields, mainly due to shading of the main crop (e.g., coffee). Most other practices we evaluated had either a neutral or positive effect on yields.

We also identified research priority areas and literature inconsistencies and uncertainties. Future research opportunities should use standardized, replicated field experiments and prioritize topics where there is low agreement among existing observations, across studies, and when scaled from field to landscapes. There is a strong need for consistent metrics to evaluate nutrient cycling efficiency and for tools for evaluating nutrient dynamics across multiple scales. Finally, to move from theory to practice, ecological information should be coupled with economic, social, and cultural aspects of agroecology.

ORCID

Kate Tully http://orcid.org/0000-0002-6190-2679

References

Aguerre, M. J., M. A. Wattiaux, J. M. Powell, G. A. Broderick, and C. Arndt. 2011. Effect of forage-to-concentrate ratio in dairy cow diets on emission of methane, carbon dioxide, and ammonia, lactation performance, and manure excretion. *Journal of Dairy Science* 94:3081–93. doi:10.3168/jds.2010-4011.

Ajdary, K., D. K. Singh, A. K. Singh, and M. Khanna. 2007. Modelling of nitrogen leaching from experimental onion field under drip fertigation. *Agricultural Water Management* 89:15–28. doi:10.1016/j.agwat.2006.12.014.

Allaire, S. E., C. Sylvain, S. F. Lange, G. Thériault, and P. Lafrance. 2015. Potential efficiency of riparian vegetated buffer strips in intercepting soluble compounds in the presence of subsurface preferential flows. *PLoS One* 10:e0131840–21. doi:10.1371/journal.pone.0131840.

Amossé, C., M.-H. Jeuffroy, B. Mary, and C. David. 2013. Contribution of relay intercropping with legume cover crops on nitrogen dynamics in organic grain systems. *Nutrient Cycling in Agroecosystems* 98:1–14. doi:10.1007/s10705-013-9591-8.

Armengot, L., L. José-María, L. Chamorro, and F. X. Sans. 2012. Weed harrowing in organically grown cereal crops avoids yield losses without reducing weed diversity. *Agronomy for Sustainable Development* 33:405–11. doi:10.1007/s13593-012-0107-8.

Aziz, M., A. Mahmood, M. Asif, and A. Ali. 2015. Wheat-based intercropping: A review. *Journal of Animal & Plant Sciences* 25:1–18.

Babbar, L. I., and D. R. Zak. 1995. Nitrogen loss from coffee agroecosystems in Costa Rica: Leaching and denitrification in the presence and absence of shade trees. *Journal of Environment Quality* 24:227–33. doi:10.2134/jeq1995.00472425002400020003x.

Baligar, V. C., N. K. Fageria, and Z. L. He. 2001. Nutrient use efficiency in plants. *Communications in Soil Science and Plant Analysis* 32:921–50. doi:10.1081/CSS-100104098.

Basche, A. D., F. E. Miguez, T. C. Kaspar, and M. J. Castellano. 2014. Do cover crops increase or decrease nitrous oxide emissions? A meta-analysis. *Journal of Soil and Water Conservation* 69:471–82. doi:10.2489/jswc.69.6.471.

Battany, M. C., and M. E. Grismer. 2000. Rainfall runoff and erosion in Napa Valley vineyards: Effects of slope, cover and surface roughness. *Hydrological Processes* 14:1289–304. doi:10.1002/(ISSN)1099-1085.

Bayer, C., J. Dieckow, T. J. C. Amado, F. L. F. Eltz, and F. C. B. Vieira. 2009. Cover crop effects increasing carbon storage in a subtropical no-till sandy acrisol. *Communications in Soil Science and Plant Analysis* 40:1499–511. doi:10.1080/00103620902820365.

Bell, L. W., and A. D. Moore. 2012. Integrated crop–livestock systems in Australian agriculture: Trends, drivers and implications. *Agricultural Systems* 111:1–12. doi:10.1016/j.agsy.2012.04.003.

Berti, A., F. Morari, N. D. Ferro, G. Simonetti, and R. Polese. 2016. Organic input quality is more important than its quantity: C turnover coefficients in different cropping systems. *European Journal of Agronomy* 77:138–45. doi:10.1016/j.eja.2016.03.005.

Bertol, I., R. R. Ramos, F. T. Barbosa, A. P. González, J. C. Ramos, and D. H. Bandeira. 2013. Water erosion in no-tillage monoculture and intercropped systems along contour lines. *Revista Brasileira De Ciência Do Solo* 37:521–28. doi:10.1590/S0100-06832013000200023.

Biazin, B., G. Sterk, M. Temesgen, A. Abdulkedir, and L. Stroosnijder. 2012. Rainwater harvesting and management in rainfed agricultural systems in sub-Saharan Africa: A review. *Physics and Chemistry of the Earth* 47–48:139–51. doi:10.1016/j.pce.2011.08.015.

Bryan, E., C. Ringler, B. Okoba, J. Koo, M. Herrero, and S. Silvestri. 2013. Can agriculture support climate change adaptation, greenhouse gas mitigation and rural livelihoods? Insights from Kenya. *Climatic Change* 118:151–65. doi:10.1007/s10584-012-0640-0.

Cai, Y., and H. Akiyama. 2016. Nitrogen loss factors of nitrogen trace gas emissions and leaching from excreta patches in grassland ecosystems: A summary of available data. *Science of the Total Environment* 572:185–95. doi:10.1016/j.scitotenv.2016.07.222.

Cardinael, R., T. Chevallier, A. Cambou, C. Béral, B. G. Barthès, C. Dupraz, C. Durand, E. Kouakoua, and C. Chenu. 2017. Increased soil organic carbon stocks under agroforestry: A survey of six different sites in France. *Agriculture, Ecosystems and Environment* 236:243–55. doi:10.1016/j.agee.2016.12.011.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8:559–68. doi:10.1890/1051-0761(1998)008[0559:NPOSWW]2.0.CO;2.

Carvalho, P. C., and C. Batello. 2009. Access to land, livestock production and ecosystem conservation in the Brazilian Campos biome: The natural grasslands dilemma. *Livestock Science* 120:158–62. doi:10.1016/j.livsci.2008.04.012.

Castán, E., P. Satti, M. González-Polo, M. C. Iglesias, and M. J. Mazzarino. 2016. Managing the value of composts as organic amendments and fertilizers in sandy soils. *Agriculture, Ecosystems and Environment* 224:29–38. doi:10.1016/j.agee.2016.03.016.

Cavigelli, M. A., S. B. Mirsky, J. R. Teasdale, J. T. Spargo, and J. Doran. 2013. Organic grain cropping systems to enhance ecosystem services. *Renewable Agriculture and Food Systems* 28:145–59. doi:10.1017/S1742170512000439.

Chander, K., S. Goyal, D. P. Nandal, and K. K. Kapoor. 1998. Soil organic matter, microbial biomass and enzyme activities in a tropical agroforestry system. *Biology and Fertility of Soils* 27:168–72. doi:10.1007/s003740050416.

Chirinda, N., J. E. Olesen, J. R. Porter, and P. Schjønning. 2010. Soil properties, crop production and greenhouse gas emissions from organic and inorganic fertilizer-based arable cropping systems. *Agriculture, Ecosystems and Environment* 139:584–94. doi:10.1016/j.agee.2010.10.001.

Chirwa, P. W., P. Nair, and C. S. Kamara. 1994. Soil moisture changed and maize productivity under alley cropping with Leucaena and Flemingia hedgerows at Chalimbana near Lusaka, Zambia. *Forest Ecology and Management* 64:231–43. doi:10.1016/0378-1127(94)90297-6.

Clark, A. J., A. M. Decker, J. J. Meisinger, and M. S. McIntosh. 1997. Kill date of vetch, rye, and a vetch-rye mixture: II. Soil moisture and corn yield. *Agronomy Journal* 89:434–41. doi:10.2134/agronj1997.00021962008900030011x.

Clark, R. B., and R. R. Duncan. 1991. Improvement of plant mineral nutrition through breeding. *Field Crops Research* 27:219–40. doi:10.1016/0378-4290(91)90063-2.

Cook, J. C., R. S. Gallagher, J. P. Kaye, J. Lynch, and B. Bradley. 2010. Optimizing vetch nitrogen production and corn nitrogen accumulation under no-till management. *Agronomy Journal* 102:1491–99. doi:10.2134/agronj2010.0165.

Cordell, D., J.-O. Drangert, and S. White. 2009. The story of phosphorus: Global food security and food for thought. *Global Environmental Change* 19:292–305. doi:10.1016/j.gloenvcha.2008.10.009.

Cordell, D., and S. White. 2014. Life's bottleneck: Sustaining the world's phosphorus for a food secure future. *Annual Review of Environment and Resources* 39:161–88. doi:10.1146/annurev-environ-010213-113300.

Cruvinel, Ã. B. F., M. M. D. C. Bustamante, A. R. Kozovits, and R. G. Zepp. 2011. Soil emissions of NO, N_2O and CO_2 from croplands in the savanna region of central Brazil. *Agriculture, Ecosystems and Environment* 144:29–40. doi:10.1016/j.agee.2011.07.016.

Daoji, L., and D. Daler. 2004. Ocean pollution from land-based sources: East China Sea, China. *AMBIO: A Journal of the Human Environment* 33:107–13. doi:10.1579/0044-7447-33.1.107.

Davidson, E. A. 2009. The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860. *Nature Geoscience* 2:659–62. doi:10.1038/ngeo608.

DeLonge, M. S., A. Miles, and L. Carlisle. 2015. Investing in the transition to sustainable agriculture. *Environmental Science & Policy* 55:266–73. doi:10.1016/j.envsci.2015.09.013.

DeLonge, M. S., R. Ryals, and W. L. Silver. 2013. A lifecycle model to evaluate carbon sequestration potential and greenhouse gas dynamics of managed grasslands. *Ecosystems* 16:962–79. doi:10.1007/s10021-013-9660-5.

Deng, B., P. Tammeorg, O. Luukkanen, J. Helenius, and M. Starr. 2016. Effects of Acacia seyal and biochar on soil properties and sorghum yield in agroforestry systems in South Sudan. *Agroforestry Systems* 91:137–48. doi:10.1007/s10457-016-9914-2.

Diacono, M., and F. Montemurro. 2010. Long-term effects of organic amendments on soil fertility. A review. *Agronomy for Sustainable Development* 30:401–22. doi:10.1051/agro/2009040.

Dick, J., U. Skiba, R. Munro, and D. Deans. 2006. Effect of N-fixing and non N-fixing trees and crops on NO and N_2O emissions from Senegalese soils. *Journal of Biogeography* 33:416–23. doi:10.1111/jbi.2006.33.issue-3.

Dobermann, A., and K. G. Cassman. 2004. Environmental dimensions of fertilizer nitrogen: What can be done to increase nitrogen use efficiency and ensure global food security. In *Agriculture and the Nitrogen cycle assessing the impacts of fertilizer use on food production* *and the environment*, eds. A. R. Mosier, J. K. Syers, and J. R. Syers, 261–278. Washington, DC: Island Press.

Dong-Gill, K. 2008. Nitrous oxide and methane fluxes in riparian buffers and adjacent crop fields. Dissertation thesis, Iowa State University, Ames, IA.

Dyer, L., M. Oelbermann, and L. Echarte. 2012. Soil carbon dioxide and nitrous oxide emissions during the growing season from temperate maize-soybean intercrops. *Journal of Plant Nutrition and Soil Science* 175:394–400. doi:10.1002/jpln.v175.3.

Ebelhar, S. A., W. W. Frye, and R. L. Blevins. 1984. Nitrogen from legume cover crops for no-tillage corn. *Agronomy Journal* 76:51.

Epstein, E., and A. J. Bloom. 2005. *Mineral nutrition of plants*. Sunderland, MA: Sinauer Associates Incorporated.

Fan, Z., T. An, K. Wu, F. Zhou, S. Zi, Y. Yang, G. Xue, and B. Wu. 2016. Effects of intercropping of maize and potato on sloping land on the water balance and surface runoff. *Agricultural Water Management* 166:9–16. doi:10.1016/j.agwat.2015.12.006.

Fares, A., S. K. Deb, and S. Fares. 2009. Review of vadose zone soil solution sampling techniques. *Environmental Reviews* 17:215–34. doi:10.1139/A09-010.

Feaga, J. B., J. S. Selker, R. P. Dick, and D. D. Hemphill. 2010. Long-term nitrate leaching under vegetable production with cover crops in the Pacific Northwest. *Soil Science Society of America Journal* 74:186–10. doi:10.2136/sssaj2008.0178.

Firestone, M. K., and E. A. Davidson. 1989. Microbiological basis of NO and N_2O production and consumption in soil. In *Exchange of trace gases between terrestrial ecosystems and the atmosphere*, eds. M. O. Andreae and D. S. Schimel, 7–21. New York, NY: Wiley

Food and Agriculture Organization of the United Nations (FAO). 2002. *Crops and drops. Making the best use of water for agriculture*. Rome, Italy: FAO.

Food and Agriculture Organization of the United Nations (FAO). 2010. *ClimateSmart agriculture: Policies, practices and financing for food security, adaptation and mitigation*. Rome, Italy: FAO.

Francioli, D., E. Schulz, G. Lentendu, T. Wubet, F. Buscot, and T. Reitz. 2016. Mineral vs. organic amendments: Microbial community structure, activity and abundance of agriculturally relevant microbes are driven by long-term fertilization strategies. *Frontiers in Microbiology* 7:215–16. doi:10.3389/fmicb.2016.01446.

Franzluebbers, A. J. 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. *Soil and Tillage Research* 66:197–205. doi:10.1016/S0167-1987(02)00027-2.

Galloway, J. N., J. D. Aber, J. W. Erisman, S. P. Seitzinger, R. W. Howarth, E. B. Cowling, and B. J. Cosby. 2003. The Nitrogen Cascade. *BioScience* 53:341–56. doi:10.1641/0006-3568(2003)053[0341:TNC]2.0.CO;2.

Gavier-Pizarro, G. I., N. C. Calamari, J. J. Thompson, S. B. Canavelli, L. M. Solari, J. Decarre, A. P. Goijman, R. P. Suarez, J. N. Bernardos, and M. E. Zaccagnini. 2012. Expansion and intensification of row crop agriculture in the Pampas and Espinal of Argentina can reduce ecosystem service provision by changing avian density. *Agriculture, Ecosystems and Environment* 154:44–55. doi:10.1016/j.agee.2011.08.013.

Gerber, P. J., H. Steinfeld, B. Henderson, A. Mottet, and C. Opio. 2013. *Tackling climate change through livestock: A global assessment of emissions and mitigation opportunities*. Rome, Italy: Food and Agriculture Organization of the United Nations (FAO).

Gerloff, G. C., and W. H. Gabelman. 1983. Genetic basis of inorganic plant nutrition. In *Encyclopedia of plant physiology*, eds. A. Lauchli and R. L. Bieleski, 453–80. Berlin: Springer-Verlag.

Ghaley, B. B., H. Hauggaard-Nielsen, H. Høgh-Jensen, and E. S. Jensen. 2005. Intercropping of wheat and pea as influenced by nitrogen fertilization. *Nutrient Cycling in Agroecosystems* 73:201–12. doi:10.1007/s10705-005-2475-9.

Gomes, J., C. Bayer, F. De Souza Costa, M. De Cássia Piccolo, J. A. Zanatta, F. C. B. Vieira, and J. Six. 2009. Soil nitrous oxide emissions in long-term cover crops-based rotations under subtropical climate. *Soil and Tillage Research* 106:36–44. doi:10.1016/j.still.2009.10.001.

Gomiero, T., D. Pimentel, and M. G. Paoletti. 2011. Environmental impact of different agricultural management practices: Conventional vs. organic agriculture. *Critical Reviews in Plant Sciences* 30:95–124. doi:10.1080/07352689.2011.554355.

Govindarajan, M., M. R. Rao, M. N. Mathuva, and P. K. R. Nair. 1996. Soil-water and root dynamics under hedgerow intercropping in semiarid Kenya. *Agronomy Journal* 88:513–20. doi:10.2134/agronj1996.00021962008800040003x.

Groenendijk, P., M. Heinen, G. Klammler, J. Fank, H. Kupfersberger, V. Pisinaras, A. Gemitzi, S. Peña-Haro, A. García-Prats, M. Pulido-Velazquez, A. Perego, M. Acutis, and M. Trevisan. 2014. Performance assessment of nitrate leaching models for highly vulnerable soils used in low-input farming based on lysimeter data. *Science of the Total Environment* 499:463–80. doi:10.1016/j.scitotenv.2014.07.002.

Guo, Z. L., C. F. Cai, Z. X. Li, T. W. Wang, and M. J. Zheng. 2008. Crop residue effect on crop performance, soil N_2O and CO_2 emissions in alley cropping systems in subtropical China. *Agroforestry Systems* 76:67–80. doi:10.1007/s10457-008-9170-1.

Hadas, A., L. Kautsky, M. Goek, and E. Erman Kara. 2004. Rates of decomposition of plant residues and available nitrogen in soil, related to residue composition through simulation of carbon and nitrogen turnover. *Soil Biology and Biochemistry* 36:255–66. doi:10.1016/j.soilbio.2003.09.012.

Harmand, J.-M., H. Ávila, E. Dambrine, U. Skiba, S. De Miguel, R. V. Renderos, R. Oliver, F. Jiménez, and J. Beer. 2007. Nitrogen dynamics and soil nitrate retention in a *Coffea arabica—Eucalyptus deglupta* agroforestry system in Southern Costa Rica. *Biogeochemistry* 85:125–39. doi:10.1007/s10533-007-9120-4.

Hauggaard-Nielsen, H., P. Ambus, and E. S. Jensen. 2003. The comparison of nitrogen use and leaching in sole cropped versus intercropped pea and barley. *Nutrient Cycling in Agroecosystems* 65:289–300. doi:10.1023/A:1022612528161.

Heinrich, A., R. Smith, and M. Cahn. 2014. Winter-killed cereal rye cover crop influence on nitrate leaching in intensive vegetable production systems. *HortTechnology* 24:502–11.

Hergoualc'h, K., U. Skiba, J.-M. Harmand, and C. Hénault. 2008. Fluxes of greenhouse gases from andosols under coffee in monoculture or shaded by *Inga densiflora* in Costa Rica. *Biogeochemistry* 89:329–45. doi:10.1007/s10533-008-9222-7.

Herrera, J. M., B. Feil, P. Stamp, and M. Liedgens. 2010. Root growth and nitrate-nitrogen leaching of catch crops following spring wheat. *Journal of Environment Quality* 39:845–10. doi:10.2134/jeq2009.0306.

Herrero, M., B. Henderson, P. Havlík, P. K. Thornton, R. T. Conant, P. Smith, S. Wirsenius, A. N. Hristov, P. Gerber, M. Gill, K. Butterbach-Bahl, H. Valin, T. Garnett, and E. Stehfest. 2016. Greenhouse gas mitigation potentials in the livestock sector. *Nature Climate Change* 6:452–61. doi:10.1038/nclimate2925.

Herridge, D. F., M. B. Peoples, and R. M. Boddey. 2008. Global inputs of biological nitrogen fixation in agricultural systems. *Plant and Soil* 311:1–18. doi:10.1007/s11104-008-9668-3.

Hilimire, K. 2011. Integrated crop/livestock agriculture in the United States: A review. *Journal of Sustainable Agriculture* 35:376–93. Doi:10.1080/10440046.2011.562042.

Houser, C., W. S. Harkcom, and M. H. Hall. 2012 & 2013. *Forage trials report: Short-lived cool-season grass trial*. University Park, PA: Cooperative Extension Service Pennsylvania State University.

Howlett, D. S., G. Moreno, M. R. Mosquera Losada, P. K. R. Nair, and V. D. Nair. 2011. Soil carbon storage as influenced by tree cover in the Dehesa cork oak silvopasture of central-western Spain. *Journal of Environmental Monitoring* 13:1897–98. doi:10.1039/c1em10059a.

Hueso, S., T. Hernández, and C. García. 2011. Resistance and resilience of the soil microbial biomass to severe drought in semiarid soils: The importance of organic amendments. *Applied Soil Ecology* 50:27–36.

Huxley, P. 1999. *Tropical agroforestry*. Oxford: Blackwell Science.

Jansa, J., A. Weimken, and E. Frossard. 2006. The effects of agricultural practices on arbuscular mycorrhizal fungi. In *Function of soils for human societies and the environment*, eds. E. Frossard, W. E. H. Blum, and B. Warkentin, 89–115. London, UK: Geological Society.

Jeffery, S., F. G. A. Verheijen, M. Van Der Velde, and A. C. Bastos. 2011. A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis. *Agriculture, Ecosystems and Environment* 144:175–87. doi:10.1016/j.agee.2011.08.015.

Joergensen, R. G., T. H. Anderson, and V. Wolters. 1995. Carbon and nitrogen relationships in the microbial biomass of soils in beech (*Fagus sylvatica L.*) forests. *Biology and Fertility of Soils* 19:141–47. doi:10.1007/BF00336150.

Kanwar, R. S., R. M. Cruse, M. Ghaffarzadeh, A. Bakhsh, D. L. Karlen, and T. B. Bailey. 2005. Corn-soybean and alterative cropping systems effect on NO_3-N leaching losses in subsurface drainage water. *Applied Engineering in Agriculture* 21:181–88. doi:10.13031/2013.18151.

Kassam, A., T. Friedrich, F. Shaxson, and J. Pretty. 2009. The spread of conservation agriculture: Justification, sustainability and uptake. *International Journal of Agricultural Sustainability* 7:292–320. doi:10.3763/ijas.2009.0477.

Khan, Z. R., A. Hassanali, W. Overholt, T. M. Khamis, A. M. Hooper, J. A. Pickett, L. J. Wadhams, and C. M. Woodcock. 2002. Control of witchweed *Striga hermonthica* by intercropping with *Desmodium* spp., and the mechanism defined as allelopathic. *Journal of Chemical Ecology* 28:1871–85. doi:10.1023/A:1020525521180.

Kleinman, P., K. S. Blunk, R. Bryant, L. Saporito, D. Beegle, K. Czymmek, Q. Ketterings, T. Sims, J. Shortle, J. McGrath, F. Coale, M. Dubin, D. Dostie, R. Maguire, R. Meinen, A. Allen, K. O'Neill, L. Garber, M. Davis, B. Clark, K. Sellner, and M. Smith. 2012. Managing manure for sustainable livestock production in the Chesapeake Bay Watershed. *Journal of Soil and Water Conservation* 67:54A–61A. doi:10.2489/jswc.67.2.54A.

Kremen, C., and A. Miles. 2012. Ecosystem services in biologically diversified versus conventional farming systems: Benefits, externalities, and trade-offs. *Ecology and Society* 17:40–25. doi:10.5751/ES-05035-170440.

Ladha, J. K., H. Pathak, T. J. Krupnik, and J. Six. 2005. Efficiency of fertilizer nitrogen in cereal production: Retrospects and prospects. *Advances in Agronomy* 87:85–156.

Lal, R., D. C. Reicosky, and J. D. Hanson. 2007. Evolution of the plow over 10,000 years and the rationale for no-till farming. *Soil and Tillage Research* 93:1–12. doi:10.1016/j.still.2006.11.004.

Lamba, J., P. Srivastava, T. R. Way, S. Sen, C. W. Wood, and K. H. Yoo. 2013. Nutrient loss in leachate and surface runoff from surface-broadcast and subsurface-banded broiler litter. *Journal of Environmental Quality* 42:1574–79. doi:10.2134/jeq2013.02.0064.

Lamichhane, K. 2013. Effectiveness of sloping agricultural land technology on soil fertility status of mid-hills in Nepal. *Journal of Forestry Research* 24:767–75. doi:10.1007/s11676-013-0415-0.

Lamy, T., K. N. Liss, A. Gonzalez, and E. M. Bennett. 2016. Landscape structure affects the provision of multiple ecosystem services. *Environmental Research Letters* 11:124017–10. doi:10.1088/1748-9326/11/12/124017.

Lawson, A., C. Cogger, A. Bary, and A.-M. Fortuna. 2015. Influence of seeding ratio, planting date, and termination date on rye-hairy vetch cover crop mixture performance under organic management. *PLoS One* 10:e0129597–19. doi:10.1371/journal.pone.0129597.

Lemaire, G., A. Franzluebbers, P. C. Carvalho, and B. Dedieu. 2014. Integrated crop–livestock systems: Strategies to achieve synergy between agricultural production and environmental quality. *Agriculture, Ecosystems and Environment* 190:4–8. doi:10.1016/j.agee.2013.08.009.

Lenka, N. K., P. R. Choudhury, S. Sudhishri, A. Dass, and U. S. Patnaik. 2012. Soil aggregation, carbon build up and root zone soil moisture in degraded sloping lands under selected agroforestry based rehabilitation systems in eastern India. *Agriculture, Ecosystems and Environment* 150:54–62. doi:10.1016/j.agee.2012.01.003.

Li, C., W. Salas, R. Zhang, C. Krauter, A. Rotz, and F. Mitloehner. 2012. Manure-DNDC: A biogeochemical process model for quantifying greenhouse gas and ammonia emissions from livestock manure systems. *Nutrient Cycling in Agroecosystems* 93:163–200. doi:10.1007/s10705-012-9507-z.

Lima, P., M. Silva, N. Curi, and J. Quinton. 2014. Soil loss by water erosion in areas under maize and jack beans intercropped and monocultures. *Ciência E Agrotecnologia* 38:129–39. doi:10.1590/S1413-70542014000200004.

Long, E., Q. Ketterings, and K. Czymmek. 2013. Survey of cover crop use on New York dairy farms. *Crop Management* 12:1–5. doi:10.1094/CM-2013-0019-RS.

Lundy, M. E., C. M. Pittelkow, B. A. Linquist, X. Liang, K. J. Van Groenigen, J. Lee, J. Six, R. T. Venterea, and C. Van Kessel. 2015. Nitrogen fertilization reduces yield declines following no-till adoption. *Field Crops Research* 183:204–10. doi:10.1016/j.fcr.2015.07.023.

Malcolm, G. M., G. G. T. Camargo, V. A. Ishler, T. L. Richard, and H. D. Karsten. 2015. Energy and greenhouse gas analysis of northeast U.S. dairy cropping systems. *Agriculture, Ecosystems and Environment* 199:407–17. doi:10.1016/j.agee.2014.10.007.

Malik, R. K., T. H. Green, G. F. Brown, and D. Mays. 2000. Use of cover crops in short rotation hardwood plantations to control erosion. *Biomass and Bioenergy* 18:479–87. doi:10.1016/S0961-9534(00)00016-7.

Mapa, R. B. 1995. Effect of intercropping coconut lands on soil water retention. *Biological Agriculture & Horticulture* 12:173–83. doi:10.1080/01448765.1995.9754735.

Mariotti, M., A. Masoni, L. Ercoli, and I. Arduini. 2016. Nitrogen leaching and residual effect of barley/field bean intercropping. *Plant, Soil and Environment* 61:60–65. doi:10.17221/PSE.

Marschner, H. 1995. Functions of mineral nutrients. In *Mineral nutrition of higher plants*, ed. H. Marschner, 229–312. 2nd ed. London, UK: Academic Press.

Mastrandrea, M. D., C. B. Field, T. F. Stocker, O. Edenhofer, K. L. Ebi, D. J. Frame, H. Held, E. Kriegler, K. J. Mach, P. R. Matschoss, G.-K. Plattner, G. W. Yohe, and F. W. Zwiers. 2010. *Guidance note for lead authors of the IPCC fifth assessment report on consistent treatment of uncertainties. Intergovernmental Panel on Climate Change (IPCC)*. Jasper Ridge, CA: Jasper Ridge.

Mazzarino, M. J., L. Szott, and M. Jimenez. 1993. Dynamics of soil total C and N, microbial biomass, and water-soluble C in tropical agroecosystems. *Soil Biology and Biochemistry* 25:205–14. doi:10.1016/0038-0717(93)90028-A.

Meisinger, J. J., W. L. Hargrove, R. L. Mikelsen, J. R. Williams, and V. W. Benson. 1991. Effects of cover crops on groundwater quality. In *Cover crops for clean water*, ed. W. L. Hargrove, 57–68. Ankeny, IA: Soil and Water Conservation Society.

Mekonnen, K., R. J. Buresh, and B. Jama. 1997. Root and inorganic nitrogen distributions in sesbania fallow, natural fallow and maize fields. *Plant and Soil* 188:319–27. doi:10.1023/A:1004264608576.

Mikkelsen, R. L. 2007. Managing potassium for organic crop production. *HortTechnology* 17:455–60.

Mikkelsen, R. L., and T. W. Bruulsema. 2005. Fertilizer use for horticultural crops in the U.S. during the 20th century. *HortTechnology* 15:24–30.

Molden, D., M. Vithanage, C. De Fraiture, J. M. Faures, L. Gordon, F. Molle, and D. Peden. 2011. Water availability and its use in agriculture. In *Treatise on water science*, ed. P. Wilderer, Vol. 4, 707–32. Oxford: Academic Press.

Moreels, E., S. De Neve, G. Hofman, and M. Van Meirvenne. 2003. Simulating nitrate leaching in bare fallow soils: A model comparison. *Nutrient Cycling in Agroecosystems* 67:137–44. doi:10.1023/A:1025526802717.

Mosquera-Losada, M. R., D. Freese, and A. Rigueiro-Rodríguez. 2011. Carbon sequestration in European agroforestry systems. In *Carbon sequestration potential of agroforestry systems*, eds. B. M. Kumar and P. K. R. Nair, 43–59. Dordrecht, The Netherlands: Springer.

Nair, P. K. R. 2011. Carbon sequestration studies in agroforestry systems: A reality-check. *Agroforestry Systems* 86:243–53. doi:10.1007/s10457-011-9434-z.

Nie, S.-W., A. E. Eneji, Y.-Q. Chen, P. Sui, J.-X. Huang, and H. Shao-Min. 2012. Nitrate leaching from maize intercropping systems with N fertilizer over-dose. *Journal of Integrative Agriculture* 11:1555–65. doi:10.1016/S2095-3119(12)60156-7.

Nyakudya, I. W., and L. Stroosnijder. 2015. Conservation tillage of rainfed maize in semi-arid Zimbabwe: A review. *Soil and Tillage Research* 145:184–97. doi:10.1016/j.still.2014.09.003.

Odhiambo, J. J. O., and A. A. Bomke. 2001. Grass and legume cover crop effects on dry matter and nitrogen accumulation. *Agronomy Journal* 93:299–9. doi:10.2134/agronj2001.932299x.

Organic Farming Research Foundation (OFRF). 2004. *Final results of the Fourth National Organic Farmers' Survey: Sustaining organic farms in a changing organic marketplace*. Santa Cruz, CA: Organic Farming Research Foundation.

Oshunsanya, S. O. 2013. Spacing effects of vetiver grass (*Vetiveria nigritana* Stapf) hedgerows on soil accumulation and yields of maize–cassava intercropping system in Southwest Nigeria. *Catena* 104:120–26. doi:10.1016/j.catena.2012.10.019.

Owen, J. J., and W. L. Silver. 2014. Greenhouse gas emissions from dairy manure management: A review of field-based studies. *Global Change Biology* 21:550–65. doi:10.1111/gcb.12687.

Palm, C., H. Blanco-Canqui, F. DeClerck, L. Gatere, and P. Grace. 2014. Conservation agriculture and ecosystem services: An overview. *Agriculture, Ecosystems and Environment* 187:87–105. doi:10.1016/j.agee.2013.10.010.

Pappa, V. A., R. M. Rees, R. L. Walker, J. A. Baddeley, and C. A. Watson. 2011. Nitrous oxide emissions and nitrate leaching in an arable rotation resulting from the presence of an intercrop. *Agriculture, Ecosystems and Environment* 141:153–61. doi:10.1016/j.agee.2011.02.025.

Parkin, T. B., and T. C. Kaspar. 2006. Nitrous oxide emissions from corn–soybean systems in the Midwest. *Journal of Environment Quality* 35:1496–11. doi:10.2134/jeq2005.0183.

Paudel, B. R., R. P. Udawatta, R. J. Kremer, and S. H. Anderson. 2011. Soil quality indicator responses to row crop, grazed pasture, and agroforestry buffer management. *Agroforestry Systems* 84:311–23. doi:10.1007/s10457-011-9454-8.

Paul, J., J. Sierra, F. Causeret, L. Guindé, and J. M. Blazy. 2017. Factors affecting the adoption of compost use by farmers in small tropical Caribbean islands. *Journal of Cleaner Production* 142:1387–96. doi:10.1016/j.jclepro.2016.11.168.

Paustian, K., O. Andrén, H. H. Janzen, R. Lal, P. Smith, G. Tian, H. Tiessen, M. Noordwijk, and P. L. Woomer. 1997. Agricultural soils as a sink to mitigate CO_2 emissions. *Soil Use and Management* 13:230–44. doi:10.1111/sum.1997.13.issue-s4.

Paustian, K., J. Lehmann, S. Ogle, D. Reay, G. P. Robertson, and P. Smith. 2016. Climate-smart soils. *Nature* 532:49–57. doi:10.1038/nature17174.

Pelzer, E., M. Bazot, D. Makowski, G. Corre-Hellou, C. Naudin, M. Al Rifaï, E. Baranger, L. Bedoussac, V. Biarnès, P. Boucheny, B. Carrouée, D. Dorvillez, D. Foissy, B. Gaillard, L. Guichard, M.-C. Mansard, B. Omon, L. Prieur, M. Yvergniaux, E. Justes, and M. H. Jeuffroy. 2012. Pea–wheat intercrops in low-input conditions combine high economic performances and low environmental impacts. *European Journal of Agronomy* 40:39–53. doi:10.1016/j.eja.2012.01.010.

Perego, A., A. Basile, A. Bonfante, R. De Mascellis, F. Terribile, S. Brenna, and M. Acutis. 2012. Nitrate leaching under maize cropping systems in Po Valley (Italy). *Agriculture, Ecosystems and Environment* 147:57–65. doi:10.1016/j.agee.2011.06.014.

Perfecto, I., A. Mas, T. Dietsch, and J. Vandermeer. 2003. Conservation of biodiversity in coffee agroecosystems: A tri-taxa comparison in southern Mexico. *Biodiversity and Conservation* 12:1239–52. doi:10.1023/A:1023039921916.

Perfecto, I., J. Vandermeer, A. Mas, and L. S. Pinto. 2005. Biodiversity, yield, and shade coffee certification. *Ecological Economics* 54:435–46. doi:10.1016/j.ecolecon.2004.10.009.

Phillips, R. E., G. W. Thomas, R. L. Blevins, W. W. Frye, and S. H. Phillips. 1980. No-tillage agriculture. *Science* 208:1108–13. doi:10.1126/science.208.4448.1108.

Pieters, A. J. 1927. *Green manuring principles and practices*. New York, NY: John Wiley & Sons, Inc.

Pittelkow, C. M., X. Liang, B. A. Linquist, K. J. Van Groenigen, J. Lee, M. E. Lundy, N. Van Gestel, J. Six, R. T. Venterea, and C. Van Kessel. 2015b. Productivity limits and potentials of the principles of conservation agriculture. *Nature* 517:365–68. doi:10.1038/nature13809.

Pittelkow, C. M., B. A. Linquist, M. E. Lundy, X. Liang, K. J. Van Groenigen, J. Lee, N. Van Gestel, J. Six, R. T. Venterea, and C. Van Kessel. 2015a. When does no-till yield more? A global meta-analysis. *Field Crops Research* 183:1–13. doi:10.1016/j.fcr.2015.07.020.

Power, A. G. 2010. Ecosystem services and agriculture: Tradeoffs and synergies. *Philosophical Transactions of the Royal Society B: Biological Sciences* 365:2959–71. doi:10.1098/rstb.2010.0143.

Prasuhn, V. 2012. On-farm effects of tillage and crops on soil erosion measured over 10 years in Switzerland. *Soil & Tillage Research* 120:137–46. doi:10.1016/j.still.2012.01.002.

Raiesi, F., and V. Kabiri. 2016. Identification of soil quality indicators for assessing the effect of different tillage practices through a soil quality index in a semi-arid environment. *Ecological Indicators* 71:198–207. doi:10.1016/j.ecolind.2016.06.061.

Ramesh, T., K. M. Manjaiah, J. M. S. Tomar, and S. V. Ngachan. 2013. Effect of multipurpose tree species on soil fertility and CO_2 efflux under hilly ecosystems of Northeast India. *Agroforestry Systems* 87:1377–88. doi:10.1007/s10457-013-9645-6.

Ranells, N. N., and M. G. Wagger. 1996. Nitrogen release from grass and legume cover crop monocultures and bicultures. *Agronomy Journal* 88:777–882. doi:10.2134/agronj1996.00021962008800050015x.

Reeves, D. W. 1997. The role of soil organic matter in maintaining soil quality in continuous cropping systems. *Soil and Tillage Research* 43:131–67. doi:10.1016/S0167-1987(97)00038-X.

Regan, J. T., S. Marton, O. Barrantes, E. Ruane, M. Hanegraaf, J. Berland, H. Korevaar, S. Pellerin, and T. Nesme. 2017. Does the recoupling of dairy and crop production via cooperation between farms generate environmental benefits? A case-study approach in Europe. *European Journal of Agronomy* 82:342–56. doi:10.1016/j.eja.2016.08.005.

Reicosky, D. C., and F. Forcella. 1998. Cover crop and soil quality interactions in agroecosystems. *Journal of Soil and Water Conservation* 53:224–29.

Restovich, S. B., A. E. Andriulo, and S. I. Portela. 2012. Introduction of cover crops in a maize–soybean rotation of the Humid Pampas: Effect on nitrogen and water dynamics. *Field Crops Research* 128:62–70. doi:10.1016/j.fcr.2011.12.012.

Rivest, D., A. Cogliastro, R. L. Bradley, and A. Olivier. 2010. Intercropping hybrid poplar with soybean increases soil microbial biomass, mineral N supply and tree growth. *Agroforestry Systems* 80:33–40. doi:10.1007/s10457-010-9342-7.

Rockström, J., J. Barron, and P. Fox. 2002. Rainwater management for increased productivity among small-holder farmers in drought prone environments. *Physics and Chemistry of the Earth* 27:949–59. doi:10.1016/S1474-7065(02)00098-0.

Roley, S. S., J. L. Tank, J. C. Tyndall, and J. D. Witter. 2016. How cost-effective are cover crops, wetlands, and two-stage ditches for nitrogen removal in the Mississippi River Basin? *Water Resources and Economics* 15:43–56. doi:10.1016/j.wre.2016.06.003.

Rosecrance, R. C., S. Rogers, and M. Tofinga. 1992. Effects of alley cropped *Calliandra calothyrsus* and *Gliricidia sipium* hedges on weed growth, soil properties, and taro yields in Western Samoa. *Agroforestry Systems* 19:57–66. doi:10.1007/BF00130094.

Rosenstock, T. S., K. L. Tully, C. Arias-Navarro, H. Neufeldt, K. Butterbach-Bahl, and L. V. Verchot. 2014. Agroforestry with N2-fixing trees: Sustainable development's friend or foe? *Current Opinion in Environmental Sustainability* 6:15–21. doi:10.1016/j.cosust.2013.09.001.

Roshetko, J. M., M. Delaney, and K. Hairiah. 2002. Carbon stocks in Indonesian homegarden systems: Can smallholder systems be targeted for increased carbon storage? *American Journal of Alternative Agriculture* 17:138–48. doi:10.1079/AJAA200116.

Rowntree, J. C., R. Ryals, M. S. DeLonge, W. R. Teague, M. B. Chiavegato, P. Byck, T. Wang, and S. Xu. 2016. Potential mitigation of midwest grass-finished beef production emissions with soil carbon sequestration in the United States of America. *Future of Food: Journal on Food, Agriculture and Society* 4:31–38.

Rufino, M. C., E. C. Rowe, R. J. Delve, and K. E. Giller. 2006. Nitrogen cycling efficiencies through resource-poor African crop–livestock systems. *Agriculture, Ecosystems and Environment* 112:261–82. doi:10.1016/j.agee.2005.08.028.

Rusinamhodzi, L., H. K. Murwira, and J. Nyamangara. 2006. Cotton–cowpea intercropping and its N2 fixation capacity improves yield of a subsequent maize crop under Zimbabwean rain-fed conditions. *Plant and Soil* 287:327–36. doi:10.1007/s11104-006-9080-9.

Russelle, M. P., E. J. Diebert, R. D. Hauck, M. Stevanovic, and R. A. Olson. 1981. Effects of water and nitrogen management on yield and 15N-depleted fertilizer use efficiency of irrigated corn. *Soil Science Society of America Journal* 45:553–58. doi:10.2136/sssaj1981.03615995004500030024x.

Ryals, R., V. T. Eviner, C. Stein, K. N. Suding, and W. L. Silver. 2016. Grassland compost amendments increase plant production without changing plant communities. *Ecosphere* 7: e01270–15. doi:10.1002/ecs2.1270.

Ryals, R., M. Kaiser, M. S. Torn, A. A. Berhe, and W. L. Silver. 2014. Impacts of organic matter amendments on carbon and nitrogen dynamics in grassland soils. *Soil Biology and Biochemistry* 68:52–61. doi:10.1016/j.soilbio.2013.09.011.

Ryals, R., and W. L. Silver. 2013. Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grasslands. *Ecological Applications* 23:46–59. doi:10.1890/12-0620.1.

Sainju, U. M., and B. P. Singh. 1997. Winter cover crops for sustainable agricultural systems: Influence on soil properties, water quality, and crop yields. *HortScience* 32:21–28.

Sainju, U. M., W. F. Whitehead, and B. P. Singh. 2005. Carbon accumulation in cotton, sorghum, and underlying soil as influenced by tillage, cover crops, and nitrogen fertilization. *Plant and Soil* 273:219–34. doi:10.1007/s11104-004-7611-9.

Salah, A. M. A., R. Prasse, and B. Marschner. 2016. Intercropping with native perennial plants protects soil of arable fields in semi-arid lands. *Journal of Arid Environments* 130:1–13. doi:10.1016/j.jaridenv.2016.02.015.

Sanchez, P. A. 1999. Improved fallows come of age in the tropics. *Agroforestry Systems* 47:3–12. doi:10.1023/A:1006287702265.

Sanz-Cobena, A., L. Lassaletta, E. Aguilera, A. Del Prado, J. Garniere, G. Billen, A. Iglesias, B. Sánchez, G. Guardia, D. Abalos, D. Plaza-Bonilla, I. Puigdueta-Bartolomé, R. Moral, E. Galán, H. Arriaga, P. Merino, J. Infante-Amate, A. Meijide, G. Pardo, J. Álvaro-Fuentes, C. Gilsanz, D. Báez, J. Doltra, S. González-Ubierna, M. L. Cayuela, S. Menéndez, E. Díaz-Pinés, J. Le-Noë, M. Quemada, F. Estellés, S. Calvet, H. J. M. Van Grinsven, H. Westhoek, M. J. Sanz, B. S. Gimeno, A. Vallejo, and P. Smith. 2016. Strategies for greenhouse gas emissions mitigation in Mediterranean agriculture: A review. *Agriculture, Ecosystems and Environment* 238:5–24. doi:10.1016/j.agee.2016.09.038.

Schulten, H. R., and M. Schnitzer. 1997. Chemical model structures for soil organic matter and soils. *Soil Science Society of America Journal* 162:115–30. doi:10.1097/00010694-199702000-00005.

Sharma, R. C., and P. Banik. 2014. Baby corn-legumes intercropping systems: I. Yields, resource utilization efficiency, and soil health. *Agroecology and Sustainable Food Systems* 39:41–61. doi:10.1080/21683565.2014.942764.

Sharrow, S. H., and S. Ismail. 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. *Agroforestry Systems* 60:123–30. doi:10.1023/B:AGFO.0000013267.87896.41.

Siemens, J., and M. Kaupenjohann. 2004. Comparison of three methods for field measurement of solute leaching in a sandy soil. *Soil Science Society of America Journal* 68:1191–96. doi:10.2136/sssaj2004.1191.

Smil, V. 2000. Phosphorus in the environment: Natural flows and human interferences. *Annual Review of Energy and the Environment* 25:53–88. doi:10.1146/annurev.energy.25.1.53.

Smith, S. E., and D. J. Read. 2008. *Mycorrhizal symbiosis*, 3rd ed. Cambridge: Academic Press.

Snapp, S., S. M. Swinton, R. Labarta, D. Mutch, J. R. Black, R. Leep, J. Nyiraneza, and K. ONeil. 2005. Evaluating cover crops for benefits, costs and performance within cropping system niches. *Agronomy Journal* 97:322–32.

Sollins, P., P. Robertson, and G. Uehara. 1988. Nutrient mobility in variable- and permanent-charge soils. *Biogeochemistry* 6:181–99. doi:10.1007/BF02182995.

Song, Y. N., F. S. Zhang, P. Marschner, F. L. Fan, H. M. Gao, X. G. Bao, J. H. Sun, and L. Li. 2006. Effect of intercropping on crop yield and chemical and microbiological properties in rhizosphere of wheat (*Triticum aestivum L.*), maize (*Zea mays L.*), and faba bean (*Vicia faba L.*). *Biology and Fertility of Soils* 43:565–74. doi:10.1007/s00374-006-0139-9.

Sooby, J., J. Landeck, and M. Lipson. 2007. *National Organic Research Agenda: Outcomes from the Scientific Congress on Organic Agricultural Research (SCOAR)*. Santa Cruz, CA, USA: Organic Farming Research Foundation.

Soto-Pinto, L., I. Perfecto, and J. Castillo-Hernandez. 2000. Shade effect on coffee production at the northern Tzeltal zone of the state of Chiapas, Mexico. *Agriculture, Ecosystems and Environment* 80:61–69. doi:10.1016/S0167-8809(00)00134-1.

Steenwerth, K., and K. M. Belina. 2008. Cover crops enhance soil organic matter, carbon dynamics and microbiological function in a vineyard agroecosystem. *Applied Soil Ecology* 40:359–69. doi:10.1016/j.apsoil.2008.06.006.

Steffen, W., K. Richardson, J. Rockström, S. E. Cornell, I. Fetzer, E. M. Bennett, R. Biggs, S. R. Carpenter, W. De Vries, C. A. De Wit, C. Folke, D. Gerten, J. Heinke, G. M. Mace, L. M. Persson, V. Ramanathan, B. Reyers, and S. Sorlin. 2015. Planetary boundaries: Guiding human development on a changing planet. *Science* 347:1259855–1259855. doi:10.1126/science.1259855.

Steinfeld, H., and T. Wassenaar. 2007. The role of livestock production in carbon and nitrogen cycles. *Annual Review of Environment and Resources* 32:271–94. doi:10.1146/annurev.energy.32.041806.143508.

Sun, D., K. Li, Q. Bi, J. Zhu, Q. Zhang, C. Jin, L. Lu, and X. Lin. 2017. Effects of organic amendment on soil aggregation and microbial community composition during drying-rewetting alternation. *Science of the Total Environment* 574:735–43. doi:10.1016/j.scitotenv.2016.09.112.

Sutton, M. A., and H. Van Grinsven. 2011. *European nitrogen assessment: Summary for policymakers*. Cambridge: Cambridge University Press.

Tang, X., L. Bernard, A. Brauman, T. Daufresne, P. Deleporte, D. Desclaux, G. Souche, S. A. Placella, and P. Hinsinger. 2014. Increase in microbial biomass and phosphorus availability in the rhizosphere of intercropped cereal and legumes under field conditions. *Soil Biology and Biochemistry* 75:86–93. doi:10.1016/j.soilbio.2014.04.001.

Thierfelder, C., S. Cheesman, and L. Rusinamhodzi. 2012. A comparative analysis of conservation agriculture systems: Benefits and challenges of rotations and intercropping in Zimbabwe. *Field Crops Research* 137:237–50. doi:10.1016/j.fcr.2012.08.017.

Thompson, J. R., K. F. Lambert, D. R. Foster, E. N. Broadbent, M. Blumstein, A. M. Almeyda Zambrano, and Y. Fan. 2016. The consequences of four land-use scenarios for forest ecosystems and the services they provide. *Ecosphere* 7:e01469–22. doi:10.1002/ecs2.1469.

Thornton, PK and M Herrero. 2014. Climate change adaptation in mixed crop-livestock systems in developing countries. *Global Food Security* 3:99–107.

Tian, Y., F. Cao, and G. Wang. 2013. Soil microbiological properties and enzyme activity in Ginkgo–tea agroforestry compared with monoculture. *Agroforestry Systems* 87:1201–10. doi:10.1007/s10457-013-9630-0.

Tilman, D. 2002. Agricultural sustainability and intensive production practices. *Nature*. doi:10.1038/nature01014.

Tisdall, J. M., and J. M. Oades. 1982. Organic matter and water-stable aggregates in soils. *European Journal of Soil Science* 33:141–63. doi:10.1111/ejs.1982.33.issue-2.

Tomich, T. P., S. Brodt, H. Ferris, R. Galt, W. R. Horwath, E. Kebreab, J. H. J. Leveau, D. Liptzin, M. Lubell, P. Merel, R. Michelmore, T. Rosenstock, K. Scow, J. Six, N. Williams, and L. Yang. 2011. Agroecology: A review from a global-change perspective. *Annual Review of Environment and Resources* 36:193–222. doi:10.1146/annurev-environ-012110-121302.

Tomich, T. P., S. B. Brodt, R. A. Dahlgren, and K. M. Scow. 2016. *The California nitrogen assessment: Challenges and solutions for people, agriculture, and the environment*. Oakland, CA: University of California Press.

Tonitto, C., M. B. David, and L. E. Drinkwater. 2006. Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: A meta-analysis of crop yield and N dynamics. *Agriculture, Ecosystems and Environment* 112:58–72. doi:10.1016/j.agee.2005.07.003.

Tully, K. L., D. Lawrence, and T. M. Scanlon. 2012. More trees less loss: Nitrogen leaching losses decrease with increasing biomass in coffee agroforests. *Agriculture, Ecosystems and Environment* 161:137–44. doi:10.1016/j.agee.2012.08.002.

Tully, K. L., D. Lawrence, and S. A. Wood. 2013a. Organically managed coffee agroforests have larger soil phosphorus but smaller soil nitrogen pools than conventionally managed agroforests. *Biogeochemistry* 115:385–97. doi:10.1007/s10533-013-9842-4.

Tully, K. L., S. A. Wood, M. Almaraz, C. Neill, and C. Palm. 2015. The effect of mineral and organic nutrient input on yields and nitrogen balances in western Kenya. *Agriculture, Ecosystems and Environment* 214:10–20. doi:10.1016/j.agee.2015.08.006.

Tully, K. L., S. A. Wood, and D. Lawrence. 2013b. Fertilizer type and species composition affect leachate nutrient concentrations in coffee agroecosystems. *Agroforestry Systems* 87:1083–100. doi:10.1007/s10457-013-9622-0.

Udawatta, R. P., J. J. Krstansky, G. S. Henderson, and H. E. Garrett. 2002. Agroforestry practices, runoff, and nutrient loss. *Journal of Environment Quality* 31:1214. doi:10.2134/jeq2002.1214.

Udom, B. E., B. O. Nuga, and J. K. Adesodun. 2016. Water-stable aggregates and aggregate-associated organic carbon and nitrogen after three annual applications of poultry manure and spent mushroom wastes. *Applied Soil Ecology* 101:5–10. doi:10.1016/j.apsoil.2016.01.007.

Unger, I. M., K. W. Goyne, R. J. Kremer, and A. C. Kennedy. 2012. Microbial community diversity in agroforestry and grass vegetative filter strips. *Agroforestry Systems* 87:395–402. doi:10.1007/s10457-012-9559-8.

Unger, P., B. A. Stewart, and R. P. Singh. 1991. Crop residue management and tillage methods for conserving soil and water in semi-arid regions. *Soil and Tillage Research* 20:219–40. doi:10.1016/0167-1987(91)90041-U.

Upson, M. A., and P. J. Burgess. 2013. Soil organic carbon and root distribution in a temperate arable agroforestry system. *Plant and Soil* 373:43–58. doi:10.1007/s11104-013-1733-x.

Van Der Laan, M., J. G. Annandale, K. L. Bristow, R. J. Stirzaker, C. C. Du Preez, and P. J. Thorburn. 2014. Modelling nitrogen leaching: Are we getting the right answer for the right reason? *Agricultural Water Management* 133:74–80. doi:10.1016/j.agwat.2013.10.017.

Velmourougane, K. 2015. Shade trees improve soil biological and microbial diversity in coffee based system in Western Ghats of India. *Proceedings of the National Academy of Sciences, India Section B: Biological Sciences* 87:489–97. doi:10.1007/s40011-015-0598-6.

Venterea, R. T., A. D. Halvorson, N. Kitchen, M. A. Liebig, M. A. Cavigelli, S. J. D. Grosso, P. P. Motavalli, K. A. Nelson, K. A. Spokas, B. P. Singh, C. E. Stewart, A. Ranaivoson, J. Strock, and H. Collins. 2012. Challenges and opportunities for mitigating nitrous oxide emissions from fertilized cropping systems. *Frontiers in Ecology and the Environment* 10:562–70. doi:10.1890/120062.

Verhulst, N., B. Govaerts, E. Verachtert, A. Castellanos-Navarrete, M. Mezzalama, P. Wall, J. Deckers, and K. Sayre. 2010. Conservation agriculture, improving soil quality for sustainable production systems? In *Advances in soil science: Food security and soil quality*, eds. R. Lal and B. A. Stewart, 137–208. Boca Raton, FL: CRC Press.

Wagger, M. G. 1998. Cover crop management and nitrogen rate in relation to growth and yield of no-till corn. *Agronomy Journal* 81:533–38. doi:10.2134/agronj1989.00021962008100030028x.

Weil, R. R., and N. Brady. 2015. *Nature and properties of soils*. 15th ed. Essex, UK: Pearson Education Limited.

Welch, L. F., D. L. Mulvaney, M. G. Oldham, L. V. Boone, and J. W. Pendleton. 1971. Corn yields with fall, spring, and sidedress nitrogen. *Agronomy Journal* 63:119–23. doi:10.2134/agronj1971.00021962006300010037x.

Wilkins, J. R. 2008. Eco-efficient approaches to land management: a case for increased integration of crop and animal production systems. *Phil Trans R Soc B* 363:517–525.

Wezel, A., H. Brives, M. Casagrande, C. Clément, A. Dufour and P. Vandenbroucke. 2016. Agroecology territories: places for sustainable agricultural and food systems and biodiversity conservation, *Agroecology and Sustainable Food Systems*, 40:132–144.

Williams, P., A. M. Gordon, H. E. Garrett, and L. Buck. 1997. Agroforestry in North America and its role in farming systems. In *Temperate agroforestry systems*, eds. A. M. Gordon and S. M. Newman, 9–48. Wallingford, UK: CAB International.

Wong, M., R. Hughes, and D. L. Rowell. 1990. Retarded leaching of nitrate in acid soils from the tropics: Measurement of the effective anion exchange capacity. *Journal of Soil Science* 41:655–63. doi:10.1111/ejs.1990.41.issue-4.

Wortman, S. E., C. A. Francis, M. A. Bernards, E. E. Blankenship, and J. L. Lindquist. 2013. Mechanical termination of diverse cover crop mixtures for improved weed suppression in organic cropping systems. *Weed Science* 61:162–70. doi:10.1614/WS-D-12-00066.1.

Zebarth, B. J., G. H. Neilsen, and E. Hogue. 1999. Influence of organic waste amendments on selected soil physical and chemical properties. *Canadian Journal of Soil Science* 79:501–04. doi:10.4141/S98-074.

Zhou, X., G. Yu, and F. Wu. 2011. Effects of intercropping cucumber with onion or garlic on soil enzyme activities, microbial communities and cucumber yield. *European Journal of Soil Biology* 47:279–87. doi:10.1016/j.ejsobi.2011.07.001.

Zhu, Y., R. H. Fox, and J. D. Toth. 2003. Tillage effects on nitrate leaching measured by pan and wick lysimeters. *Soil Science Society of America Journal* 67:1517–23. doi:10.2136/sssaj2003.1517.

Zomer, R. J., A. Trabucco, R. Coe, and F. Place. 2009. Trees on farms. ICRAF Working Paper No. 89, 1–72. Nairobi, Kenya: World Agroforestry Center.

Zougmore, R., F. N. Kambou, K. Ouattara, and S. Guillobez. 2000. Sorghum-cowpea intercropping: An effective technique against runoff and soil erosion in the Sahel (Saria, Burkina Faso). *Arid Soil Research and Rehabilitation* 14:329–42. doi:10.1080/08903060050136441.