

Chapter 7

Ecologically Based Nutrient Management

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NUTRIENT MANAGEMENT AS APPLIED ECOLOGY

Ecologically based nutrient management is an integrated approach that applies ecological knowledge to optimize soil fertility, crop production, ecosystem services, and long-term sustainability. In particular, ecological nutrient management applies concepts from community and ecosystem ecology. An *ecosystem* is a dynamic complex of organisms, and the physical environment with which they interact. An *agroecosystem* is simply an ecosystem which is managed to achieve agricultural outcomes, including the production of food, fodder, and fiber. Application of an ecosystem framework provides agriculturalists with a flexible systems approach that can be used to organize the complex, dynamic interactions between organisms and their environment that ultimately govern nutrient dynamics and crop yields. *Ecosystem functions* are processes such as nutrient cycling, water and energy flows, soil retention, and primary production or crop yield that result from complex interactions among living and nonliving ecosystem components (i.e., plants, decomposers, climate, soil environment, etc.). When we are considering ecosystem functions and their benefits to human wellbeing, we refer to them as *ecosystem services*, in order to emphasize their value to humans.

In applying principles and concepts from ecology, the scope of nutrient management is expanded to include a wide range of soil nutrient reservoirs, soil organisms, and biogeochemical processes. For example, rather than focusing solely on soluble, inorganic plant-available pools, ecologically based nutrient management seeks to optimize organic and mineral soil reservoirs that are more efficiently retained in the soil, such as organic matter (OM) and sparingly soluble forms of phosphorus. This framework also expands the scope of management beyond the normal focus which is limited to crop nutrient uptake. Instead, efforts are directed toward managing

organisms and nutrient cycling processes occurring at a variety of spatial and temporal scales, from the rhizosphere, to field and landscape scales. Integrated management of the full array of ecosystem processes that regulate the cycling of nutrients and carbon in soil can improve productivity, while also increasing nutrient use efficiency (NUE) and ecosystem services over the long-term.

In this chapter we will explain ecologically based nutrient management, and emphasize strategies that integrate management of nitrogen, phosphorus, and carbon. Nitrogen and phosphorus are the two nutrients that most commonly limit crop production in agroecosystems. Biological processes play major roles in regulating the cycling of N and P, and, for this reason, the fate of these nutrients is strongly linked to the flow of carbon. We will discuss strategies for managing N, P, and C cycling processes, giving particular attention to the role of C in influencing the fate of N and P. Although we emphasize these major nutrients, many of the concepts we discuss are widely applicable to the other macro and micronutrients important for plant growth. Because a basic understanding of these cycles is fundamental to ecological nutrient management, we will first briefly review the key features of these elemental cycles.

THE BASICS OF NUTRIENT CYCLING

To understand nutrient cycling we must consider distribution, fluxes, and the regulatory mechanisms that make up an elemental cycle. Nutrients move from one *compartment* or *pool* to another. *Reservoir* is another term commonly used to refer to stores of nutrients in the soil. A compartment is usually defined by physical boundaries, while distinct pools can exist within a single compartment. For example, the soil compartment has several distinct pools of N. Plant uptake of NO_3^- results in the movement of N from the soil compartment (or more specifically, from the inorganic soil pool) to plant biomass. The distribution of nutrients among compartments and pools in agroecosystems varies in terms of the absolute amounts, depending on soil, climate, biotic, and management factors.

We refer to the rate of transfer from one pool to another as a *flux*. The flux of nutrients is often framed in terms of *source/sink* transfers when we want to emphasize the role of a particular process in regulating nutrient flows. *Source* simply refers to the pool where the nutrient came from, whereas a *sink* is the pool actively taking up the nutrient. All fluxes are regulated by a process, which can be either *biotic* (controlled by living organisms, i.e., mineralization) or *abiotic* (controlled by chemical and/or physical mechanisms, i.e., precipitation). The flux from one pool to another often entails a chemical modification of the nutrient. The most common chemical modifications are organic/inorganic and oxidation/reduction reactions.

We use models to depict relationships between location, form, and transfer of nutrient cycles. These models can be adapted to represent nutrient

cycling at any scale, with varying degrees of detail. An ecosystem can be divided up into very few compartments, i.e., the simplest nutrient cycle might only distinguish between plant and soil compartments. As more compartments and pools are added, the cycling model becomes more complex. To address nutrient flows at the landscape level, individual fields or farms and adjacent waterways would be the designated compartments. A very simple depiction of N flows is shown in Fig. 7.1A.

Only three compartments are shown, with two biologically mediated processes that control the flux of N from soil organic matter (SOM) into the inorganic pool (Flux A, mainly controlled by microorganisms), and then from the inorganic N pool to plant biomass (Flux B, regulated by the plant). If mineralization and plant assimilation are equal (N moving in and out of the inorganic N pool is the same), and if these two processes are the dominant fluxes regulating this pool, then the size of the inorganic N pool will remain a constant, even though NO_3^- is actually moving in and out. This situation is called a *steady state*. You can see that if we collected monthly soil samples and extracted inorganic N under steady state conditions, the NO_3^- pool will appear static since the concentration remains constant through time. We would miss the dynamics that are actually taking place, i.e., N is moving in and out of this pool. This is one of the difficulties in using static measurements of pool size as indicators of nutrient availability. The limitation of static measurements is particularly prominent when standing pools of

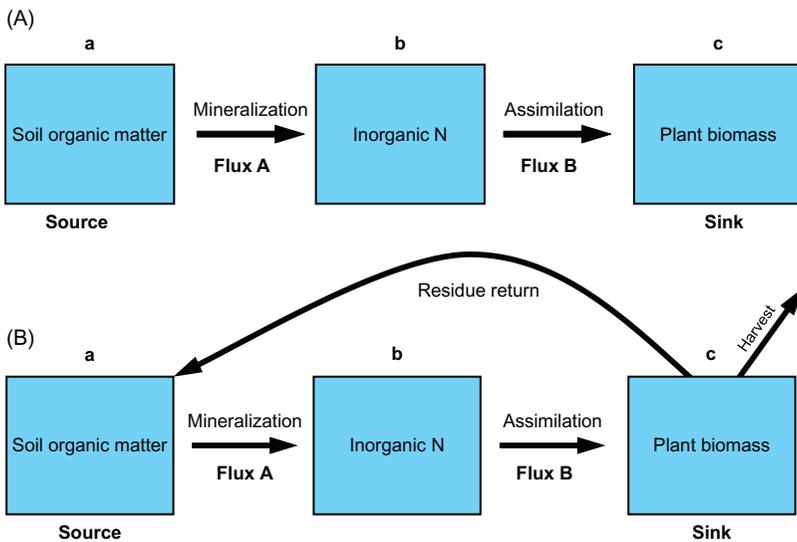


FIGURE 7.1 A simple model demonstrating the use of compartments and fluxes to depict nutrient flows. (A) Three compartments (a, b, and c) are shown with two biologically mediated fluxes. (B) The cycle is closed when plant residues are returned to the soil. Net export of N occurs through harvest.

inorganic N are very small. Small standing pools of NO_3^- are usually interpreted as an indicator of low N fertility. However, if plant assimilation is keeping up with mineralization, you can have a very large flux in a very small inorganic N pool. This is often the case in fields where organic residues have been used as nutrient sources for many years.

We can close the cycle in this simple model by adding two more fluxes (Fig. 7.1B). These two new processes are the result of human management. In this model, harvest removes the N from our agroecosystem, and we do not consider the fate of the harvested N which could be going to animals, and/or humans, and the N remaining in crop residues is left in the field to become part of the SOM. In this case, harvest is considered to be an *export*, since the harvested N leaves our system while the other three fluxes are part of the *internal* N cycle.

When developing nutrient management strategies, it is important to remember that the rates of different processes can vary by orders of magnitude. This impacts the distribution of nutrients among pools, and results in compartments with widely varying turnover times. Turnover time is defined as the time it will take to empty a reservoir if the source is cut off, and if sinks remain constant. In other words, fluxes out of the compartment continue but the influx is shut off. Understanding turnover time, and how to manipulate different pools of nutrients and OM over space and time, is at the foundation of ecologically based nutrient management, and will be the topic of this next section.

A useful way to compare the dynamics of different compartments is mean residence time, or the average amount of time a nutrient spends in the compartment before being transferred out. Mean residence time is calculated as the pool size/flux, assuming the pool is close to steady state conditions (i.e., in \approx out). For example, to estimate the mean residence time of nitrous oxide in the atmosphere on a global basis we calculate the total size of the pool and then divide by the estimated global rate of production:

$$\begin{aligned} [\text{N}_2\text{O}] &= 300 \text{ ppb, Total N}_2\text{O} = 2.3 \times 10^{15} \text{ g} \\ \text{Rate of production} &: 20 \times 10^{12} \text{ g/year} \\ \text{MRT} &= 2.3 \times 10^{15} \text{ g} / 20 \times 10^{12} \text{ g/year} = 110 \text{ years.} \end{aligned}$$

Both turnover time and mean residence time require detailed knowledge of fluxes and pool size that can be difficult to accurately measure in soils without the use of expensive tracer experiments. However, the mean residence time for important soil nutrient pools varies widely. For example, mean residence time for NO_3^- is about a day, while stabilized components of SOM have a mean residence time of hundreds to thousands of years (Tan, 2003). Given the huge difference in the temporal dynamics of these pools, it is not necessary to make measurements in your agroecosystems to apply these useful concepts. Estimates from the literature can be very helpful as a starting point.

Nitrogen Cycling

In unmanaged terrestrial ecosystems, the soil N cycle is driven by SOM, which contains approximately 50% C and 5% N, of which typically <5% is in labile forms. Available N normally enters ecosystems through biological N fixation, although during the past two centuries many unmanaged ecosystems also received anthropogenic N derived from fossil fuel burning and other human activities, through wet and dry deposition. In agroecosystems, large quantities of N are added as inorganic fertilizer or organic residues from biological N fixation and various soil amendments, such as compost or animal manure, also playing a major role in driving the N cycle. The breakdown, or depolymerization, of the large, complex molecules that make up organic residues is facilitated by extracellular enzymes secreted mainly by soil fungi and prokaryotes. With the exception of a few enzymes targeting phosphorus, such as phosphatase, plants do not release exoenzymes into the soil. As a result, plants are largely dependent on the primary decomposers to release nutrients from complex organic residues. Exoenzymes catalyze the release monomers, such as amino acids and sugars, which are small enough to be transported into cells by microbes and plants. These labile compounds are recycled and reused through microbial metabolism, faunal grazing of microbes, as well as the microbial death and damage that are caused by stress, such as wet–dry or freeze–thaw cycles (Schimel and Bennett, 2004; Fig. 7.2). Plants also contribute to internal cycling via root exudation of a diverse array of organic compounds which are decomposed, but can serve as signals to soil organisms (Bais et al., 2006).

Soil microorganisms play a dominant role in regulating soil N cycling. Mineralization occurs when heterotrophic microbes break down the nutrient rich organic monomers freed by exoenzymes and obtain energy, NH_4^+ , and other nutrients. Ammonia can be assimilated by heterotrophs, or used as an energy source by ammonia-oxidizing microbes to produce nitrite (NO_2^-) that is quickly converted to NO_3^- (nitrification). During nitrification, some nitric oxide (NO) and nitrous oxide (N_2O) are also produced and lost from the soil (Godde and Conrad, 2000; Fig. 7.2). Alternatively, NH_4^+ can be lost from the soil through the emission of ammonia (NH_3) gas if soil pH is greater than 8. Nitrate can be lost from the system through several processes. Denitrification is a metabolic process which takes place when heterotrophic bacteria under oxygen limitation use NO_3^- as an alternative electron acceptor to produce N_2O and N_2 . The leaching of NO_3^- , which contaminates groundwater, occurs when rainfall exceeds evapotranspiration, especially in coarse-textured soils. Runoff also carries N in various forms to surface waters. A second anaerobic pathway that helps to retain NO_3^- in the soil involves the conversion of NO_3^- to NH_4^+ (dissimilatory nitrate reduction to ammonium or DNRA). This pathway can compete with denitrification, and can be the dominant dissimilatory reduction pathway in some tropical soils (Silver et al., 2001).

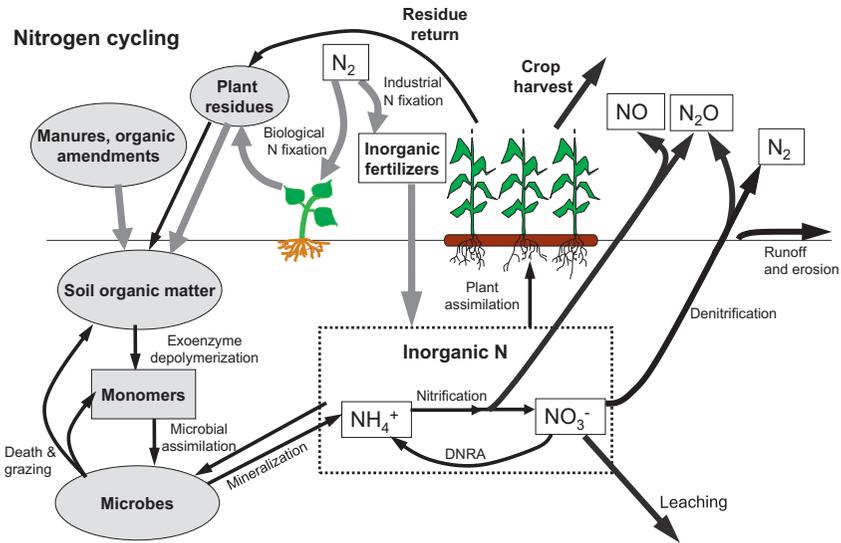


FIGURE 7.2 Nitrogen cycling in agroecosystems. See text for full discussion of cycling processes. New N is added through biological N fixation, synthetic fertilizers or organic amendments such as manure or compost (gray arrows). The main pathways of removal are through harvested exports, leaching, and denitrification (thick black arrows). Some gaseous losses also result from nitrification during the conversion of ammonium to nitrate (thick black arrows). Internal cycling processes occur through human management of residues, plant assimilation, and microbially-mediated transformations (thin black arrows).

Phosphorus Cycling

While N transformations are primarily controlled by microbially mediated processes, the soil P cycle is regulated by both biological and geochemical processes that compete with one another for the small amounts of soluble, inorganic P which are typically present in the soil solution (Cross and Schlesinger, 1995). A second major distinction shaping the P cycle is that it cannot be converted into gaseous forms that can be lost from the system. For convenience, the P cycle is portrayed as consisting of two subcycles reflecting the abiotic and biotic mechanisms (Fig. 7.3). The geochemical and biological subcycles are composed of processes that are distinct from one another, with the exception of the weathering of primary minerals, which is mediated by both biological and geochemical mechanisms (Schlesinger, 2005). Biological weathering occurs at rates many times faster compared to abiotic weathering processes. The biological transformations involving P are fewer compared to N, and begin with reactions mediated by exoenzymes that release P. Phosphorus mineralization is the microbial conversion of organic P to orthophosphates (H₂PO₄⁻ or H₂PO₄⁻², depending on soil pH) which can, in turn, be assimilated by either plants or microorganisms.

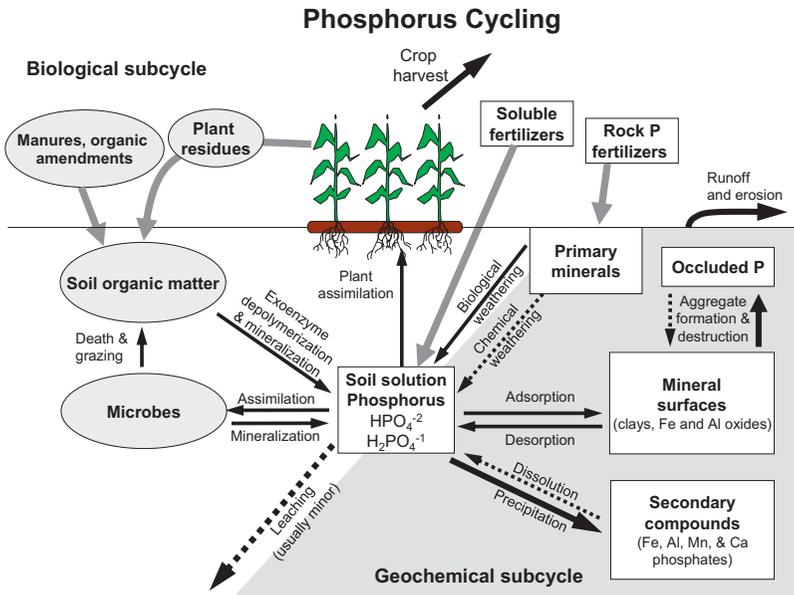


FIGURE 7.3 The phosphorus cycle consists of biological and geochemical subcycles. See text for full discussion of cycling processes. New P is added through soluble synthetic fertilizers, sparingly soluble amendments, such as rock P, or organic amendments such as manure or compost (gray arrows). The main pathways of removal are through harvested exports, erosion, occlusion, precipitation, with small losses occurring through leaching in some systems (thick black arrows). Internal cycling processes occur through human management of residues, plant assimilation, microbially-mediated transformations, and geochemical processes (thin black arrows). Dotted arrows indicate processes with smaller fluxes.

The microbial P will become available over time as the microbes die or are grazed. As with N, SOM and newly added organic residues can serve as an important source of P.

The geochemically mediated sinks for orthophosphates compete with biological assimilation and include two types of inorganic reactions; these are precipitation–dissolution and sorption–desorption processes. Precipitation–dissolution reactions involve the formation and dissolving of precipitates. Precipitation reactions occur with dissolved iron, aluminum, manganese (acid soils), or calcium (alkaline soils) to form phosphate minerals. The rate of dissolution is negligible for these precipitates, with the exception of the calcium phosphates. Calcium phosphates, such as apatite, account for 95% of P found in primary minerals of the earth’s crust, and are commonly referred to as *sparingly soluble P*, since these minerals can be dissolved by chemical and biological weathering. Apatite is the primary constituent of rock P, which can be added to infertile soils as a slow source of P. Sorption–desorption reactions involve sorption and desorption of ions and molecules at the surfaces of mineral particles. Adsorption is a reversible

chemical binding of P to soil particles. In some soils, adsorbed phosphate may become trapped on the surface of soil minerals when a Fe or Al oxide coating is formed on the mineral. The trapped phosphate is then described as being occluded. For all practical purposes, P that becomes occluded is no longer agronomically relevant.

Carbon Cycling

All biologically mediated cycling processes are dependent on C, either for energy or as the backbone of biomolecules that must be synthesized for life to exist. SOM is defined as all carbon-containing soil constituents, and is therefore the major biologically relevant soil reservoir for N and P in most arable soils. Because SOM is the result of all life, the biochemistry of SOM constituents is complex, reflecting the diverse array of compounds produced by plants, microbes, and larger soil organisms. The chemical composition and the accessibility of SOM to decomposing organisms (i.e., the actual size of the SOM and whether or not it is protected by soil minerals through occlusion or surface interactions) regulate the rate of decomposition with the former, being more important in the early stages of decomposition, and the latter, exerting more influence later in the process (von Lutzow et al., 2006). For practical purposes, SOM is conceptualized as a series of pools with varying flux rates that reflect differences in chemical composition and the degree of physical accessibility to microorganisms (Fig. 7.4). Decomposition of SOM is mediated primarily by bacteria and fungi, who release the majority of the C as CO₂ via respiration while incorporating a small portion of the C into cellular structures through biosynthesis (growth and reproduction). Growth, reproduction, and death, combined with interactions among soil organisms as part of the soil food web such as grazing, predation, and parasitism, regulate the flow of C and accompanying nutrients such as N, P, and other elements present in living organisms.

The bulk of SOM has rather long mean residence times, ranging from 250 to 1900 years; clearly beyond the time frame of planning for most agricultural settings. This SOM, referred to as the “stabilized fraction,” has long been the source of much controversy in terms of the actual chemical constituents and the mechanism of stabilization. Recently, extraction methods involving stepwise digestion which were used to study SOM composition for the past 100 years have been replaced by technologies which can discern the molecular structure of organic residues in situ. These approaches have revealed that humic substances (i.e., humic and fulvic acids), which were once thought to dominate the stable fraction, are actually present in soils in very limited quantities. This work, along with studies using ¹³C labeling to trace turnover times of various SOM constituents, has resulted in significant changes in our understanding of the mechanisms that enable organic residues to persist in soils, as well as the chemical composition of this stable fraction

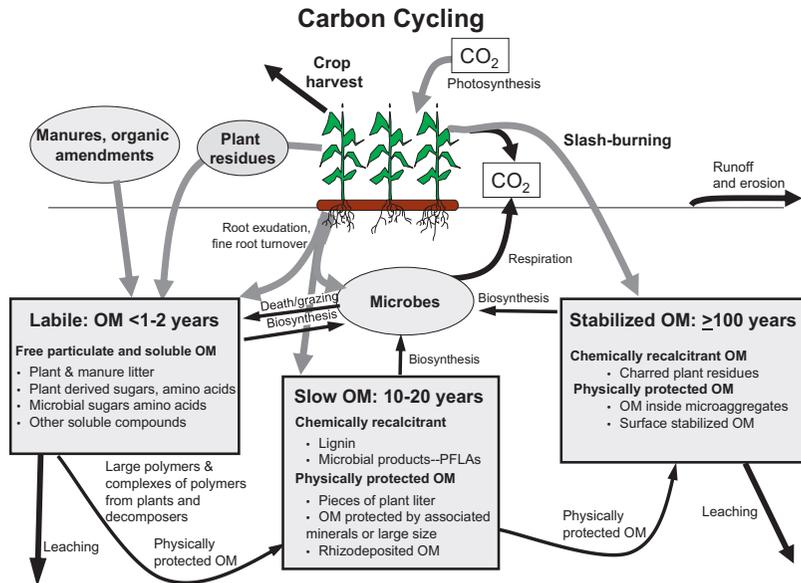
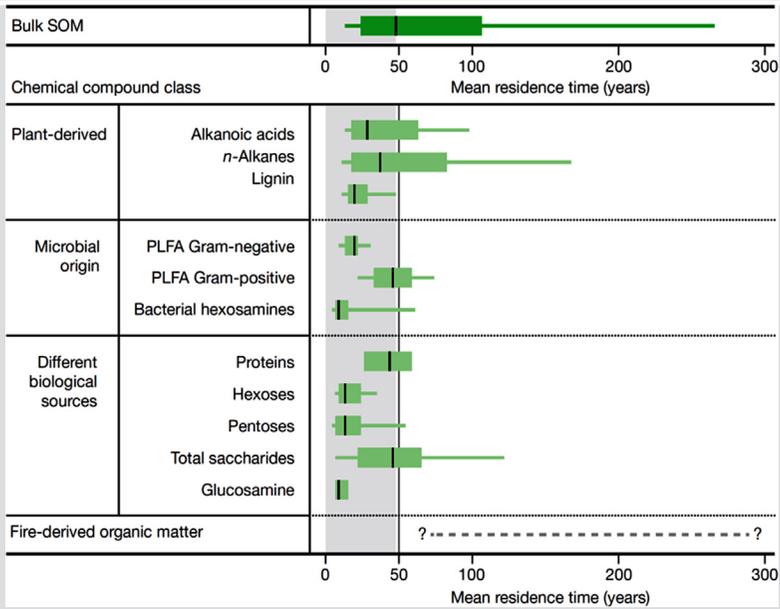


FIGURE 7.4 Carbon cycling in agroecosystems. See text for full discussion of cycling processes. The level of soil organic matter (OM) is determined by the balance between photosynthesis or new OM additions and decomposition. Decomposition encompasses two distinct processes that reflect the dual function of C: (1) respiration (energy), and (2) biosynthesis (growth and reproduction). Biosynthesis results in C from the various substrates actually being incorporated into microbial biomass, while respiration results in the release of CO_2 into the atmosphere. In this diagram we separate out OM pools based on their approximate rates of turnover. The stable OM pool is by far the largest, usually accounting for >80% of soil OM. The only route to stabilized OM that is directly under management control is through charcoal production. The vast majority of OM in the stabilized pool has undergone some form of microbial processing, and some of it has cycled through other trophic levels of the soil food web (i.e., grazers that feed on bacteria). In addition to the biological processes of respiration/biosynthesis, there are abiotic mechanisms which contribute to the stabilization of OM including adsorption, adventitious chemical reactions, and physiochemical interactions between clay particles and organic compounds that end up physically protecting these compounds making them inaccessible to decomposers or exoenzymes (von Lutzow et al., 2006; Schmidt et al., 2011). Thus, initially the rate of OM composition is controlled by the lability/recalcitrance of the compound. Some form of physical protection is required for OM to become stabilized in the soil for 100 years or more. Aggregate formation, which results in occluded OM, is mediated by both soil organisms and abiotic processes.

(Schmidt et al., 2011). During decomposition of plant biomass in incubation experiments, some plant-derived compounds (classically, long-chain alkanolic acids, *n*-alkanes, lignin, and other structural tissues) often persist longer than others. This led to the idea that chemically recalcitrant compounds such as these would persist in soils, while decomposers rapidly consumed labile compounds such as proteins and sugars. However, the newer methods show that in mineral soil the importance of chemical composition decreases over time, so that the initially fast-cycling compounds are just as likely to persist

as the slower cycling molecules (Table 7.1; Schmidt et al., 2011). Thus, we now understand that physical protection is the primary driving force in stabilizing organic compounds in soil over the long-term. This is contrary to the long held view that chemical recalcitrance is a key factor in determining how long organic substrates persist in soils. Instead, mechanisms such as aggregate formation and adsorption to mineral particles play a key role in OM stabilization. Furthermore, due to this shift in mechanisms leading to longevity, it is clear that the stable fraction is composed of smaller organic compounds that persist simply because they cannot be accessed by decomposers. Lignin is relatively short-lived in soil compared to polysaccharides, which are “sticky” and are intimately associated with mineral soil particles, and most soils do not have a significant “humus” fraction. This stabilized

TABLE 7.1 Molecular Structure Does Not Control Long-Term Decomposition of Soil Organic Matter (SOM)



This table compiles data from surface horizons of 20 long-term field experiments (up to 23 years) in a temperate climate, using ¹³C labeling to trace the residence time of bulk SOM and of individual molecular compounds. The variation in turnover time is also seen in the compounds of microbial origin, including phospholipid fatty acids (PLFA) produced by Gram-negative and Gram-positive bacteria and amino sugars (hexosamines).

Source: Schmidt et al. (2011). Used with permission

pool represents a sizable N reservoir, and the elemental composition of stable SOM is fairly consistent across soil types and climatic zones, with C and N contents of 50–60% and 2–4%, respectively (Tan, 2003).

A smaller, but more active SOM fraction that responds to agricultural management within shorter time durations (1–10 years) is particulate OM. Particulate OM refers to pieces of plant residues, including roots and shoots, that are the size of sand (53 μm to 2 mm) and have mean residence times ranging from a single growing season to 10–20 years. Particulate OM can serve as a significant source of nutrients, and also plays a key role in aggregation in some soils. Lastly, the soil microbial biomass is not only important for its decomposing function, but also serves as a labile pool of nutrients. The amount of N and P in soil prokaryotes is nearly equal to the amount in terrestrial plants (Whitman et al., 1998). For cultivated systems, the estimated N and P in soil bacteria amounts to an average of 630 and 60 kg/ha, respectively, in the first meter of soil (Whitman et al., 1998).

APPLYING AN ECOLOGICAL NUTRIENT MANAGEMENT STRATEGY

The plethora of processes controlling the cycling of nutrients in agroecosystems presents ample opportunities for enhancing the flows of N, P, and C. The over-arching strategy guiding ecosystem-based nutrient management is distinct from the conventional approach that has focused primarily on fertilizer management for the past 50 years. Table 7.2 compares the two management schemes. The underlying theory guiding conventional nutrient management emphasizes developing optimum delivery systems for soluble inorganic fertilizers, and managing the crop to create a strong sink for fertilizer by removing all other growth limiting factors. The primary difficulty with this strategy is that soluble inorganic forms of N and P are fast cycling and are subject to multiple pathways of loss. As you might predict, based on the nutrient cycling diagrams, when the pool of soluble inorganic N or P is greatly increased, undesirable fluxes also increase, and a proportion of these added nutrients is lost. So, while this approach has resulted in greater yields, it has also resulted in poor NUE, and major losses of fertilizers to the environment are widespread (Drinkwater and Snapp, 2007a). Soil degradation is also a secondary consequence of these intensive, fertilizer-driven cropping systems, mainly due to the use of intensive tillage combined with reduced inputs of organic residues and bare fallows.

Ecological nutrient management seeks to harness soil processes that foster internal cycling and retention of nutrients, while reducing those that contribute to nutrient loss and soil degradation. Ecological nutrient management is a multifaceted approach that aims to achieve optimal yields, balance nutrient exports with additions, maintain soil nutrient reservoirs, and minimize losses of nutrients and soil to the environment. In agroecosystems with

TABLE 7.2 Characteristics of the Conventional Agronomic Approach (Balasubramanian et al., 2004; Cassman et al., 2002) Compared to an Ecosystem-Based Approach to Nutrient Management

	Agronomic Framework	Ecological Framework
Goals	Maximize crop uptake of applied N, P to achieve yield goal and reduce environmental losses	Achieve optimal yields and maintain soil reservoirs while balancing nutrient additions and exports as much as possible
Nutrient management strategy	Manage crop to create a strong sink for fertilizer by removing all growth limiting factors and by providing an optimum delivery system (Balasubramanian et al., 2004)	Manage agroecosystem to increase internal cycling capacity to: (1) maintain nutrient pools that can be accessed through plant- and microbially mediated processes; and (2) conserve N and P by creating multiple sinks in time and space
Nutrient pools actively managed	Inorganic N and P	All N and P pools, organic, and inorganic
Processes targeted by nutrient management	Crop uptake of N and P	Plant and microbial assimilation of N and P, C cycling, N, P, and C storage, other desirable processes that conserve nutrients
Strategy toward microbially-mediated N transformations	Eliminate or inhibit as much as possible	Promote processes that conserve N, reduce processes that lead to losses (i.e., denitrification) by maintaining small inorganic N and P pools
Strategies for reducing NO ₃ leaching, P occlusion/precipitation	Increase crop uptake of added N, use chemicals that inhibit nitrification	Minimize inorganic pool sizes through management of multiple processes: cover cropping, additions of N and P with organic residues
Assessment of NUE	Based on fertilizer uptake of the crop in one growing season	Based in budgeting framework, reflect agroecosystem level retention, multiyear
Typical experimental approaches	Short-term, small-plot, empirical, factorial experiments dominate	Participatory, systems approaches, on-farm research is important, spatial and temporal scales of experiments are determined by the processes of interest

Modified from Drinkwater and Snapp, 2007b.

poor or degraded soils, an additional goal is to restore soil fertility and agroecosystem functions.

To implement this comprehensive set of goals, ecological nutrient management must target a variety of nutrient reservoirs and cycling processes. The basic strategy is to conserve or even enhance nutrient pools that can be accessed through plant- and microbially mediated processes by creating sinks for inorganic N and P that will promote nutrient retention and internal cycling of nutrients. The nutrient reservoirs that are targeted include labile and stabilized SOM, microbial biomass, and sparingly soluble P. Management aims to promote processes that conserve these pools, while minimizing those that lead to nutrient losses. For example, practices that encourage plant and microbial assimilation of N and P, and other processes leading to N and P storage such as aggregate formation, are favored. While flux through the inorganic N and P pools may be very large in these systems, a central objective of this strategy is to reduce the size of these pools that are the most susceptible to loss. Examples of management practices that contribute to these outcomes include diversifying crop rotations and nutrient sources, cover cropping and intercropping, and legume intensification. The particular suite of cropping practices used are site specific, and reflect the environmental characteristics of the agroecosystem (climate and soils), the crops that are being grown, resources available to the farmer, and the livelihood goals of the household.

THE ROLE OF PLANTS AND MICROORGANISMS IN CYCLING NUTRIENTS

Cycling Processes Influenced by Plants

Effective use of plant diversity in agroecosystems requires some understanding of the roles played by different crop species in nutrient cycling. Plants and their associated microbes regulate ecosystem processes which ultimately control C, N, and P cycling (Hooper and Vitousek, 1997). Intentional use of plant diversity based on the capacity of a species to enhance particular ecosystem processes is an important strategy in ecological nutrient management. Agroecosystem plant species diversity can be increased, either by introducing additional cash crops or noncash crops (i.e., cover crops, intercrops) selected to serve specific ecosystem functions.

The most easily defined plant functional roles are those relating to phenology, productive potential, and above- and below-ground architecture. *Phenology* refers to plant life-cycle characteristics such as germination, growth, flowering, and reproduction, that are controlled by climatic conditions and seasons. For example, the functional role of legumes varies with phenology (Fig. 7.5). Many legume species used for grain production are short duration annuals, with determinant flowering, and a high harvest index

Which legume growth type?

Plant phenology varies from short-duration, determinant to long-duration, indeterminant (flowers repeatedly)

Short-duration
annuals:
**bean, peanut,
soybean**

Short-lived perennials:
**pigeonpea, tephrosia,
mucuna, crotalaria**

Perennials:
gliricidia, sesbania



FIGURE 7.5 Examples of legumes with differing phenology. Legumes can be integrated into cropping systems using a number of different strategies, depending on their life-cycle.

(proportion of above-ground biomass that is harvestable product). Green manure legume species are at the other end of the spectrum. They provide large amounts of nutrient-rich residues, and are generally short or long-lived perennials, with indeterminant flowering and low-to-zero harvest index. Differences in the seasonal niche of plants can be used to expand the amount of time a field is covered with actively growing plants, increasing nutrient uptake in space and time, and reducing nutrient losses (McCracken et al., 1994; Snapp and Silim, 2002). Increased plant growth has cascading effects on internal cycling processes in agroecosystems. For example, if bare fallow periods are replaced with a cover crop, rhizodeposition provides C to the soil microbial community for a greater part of the year, increasing the potential for assimilation of nutrients into the microbial biomass (Drinkwater et al., 1998). The tremendous variation among plant genotypes in root/shoot partitioning and root architecture can be exploited to complement cash crop characteristics and optimize plant-mediated processes below-ground. For example, root biomass makes greater contributions to SOM than shoots, which tend to decompose more rapidly (Puget et al., 2000).

Plant species characteristics, such as biochemical composition of litter and root exudates, fine root turnover, and the characteristics of the rhizosphere environment, influence ecosystem function through their impact on processes related to decomposition, such as net mineralization of nutrients, aggregate formation, and stabilization of OM. Striking plant species effects have been documented for decomposition dynamics and net mineralization of N and P (Wedin and Tilman, 1990; Fierer et al., 2001), aggregate

formation (Tisdall and Oades, 1979; Angers and Mehuys, 1989; Haynes and Beare, 1997), availability of nutrients such as Ca, Mg, and P from mineral sources (Marschner and Dell, 1994; Johnson et al., 1997; Neumann and Roemheld, 1999; Kamh et al., 1999), and microbial community composition (Kennedy, 1999). Many of these observed plant species impacts on nutrient cycling processes are actually mediated by microorganisms associated with the roots.

Beyond the particular impacts associated with particular plant functional groups or species, there is evidence that increasing plant diversity, intercropping, or diverse rotations (those adding one or more crops in rotation to a monoculture) increases total soil C and N (Gardner and Drinkwater, 2009; McDaniel et al., 2014; Cong et al., 2015). When rotations were diversified by adding cover crops, total C increased by 8.5%, and total N 12.8% (McDaniel et al., 2014; 122 publications). In addition to these impacts on nutrient cycling, plant biodiversity can also enhance disease suppression (Abawi and Widmer, 2000), reduce weed competition and herbicide requirements (Gallandt et al., 1999), and foster beneficial arthropod communities (Lewis et al., 1997). Inclusion of all of these functions is integral to agroecological management of crop production. One useful approach is to compile information on the functional traits of potential cover crops (Table 7.3). Decisions about rotation and intercropping that impact plant species composition can contribute to reducing the need for agrochemical inputs (Drinkwater and Snapp, 2007a,b).

Plant–microbial Interactions

While plants themselves can directly impact biogeochemical processes through nutrient assimilation and the quantity and quality of litter and root exudates, many influences on nutrient cycling are the result of plant–microbial interactions. The *rhizosphere* is the region of soil that is immediately adjacent to the plant root, and is the site of plant–microbial interactions (Fig. 7.6). The importance and extent of plant–microbial interactions that take place in this microenvironment has not been fully appreciated in the past (Drinkwater and Snapp, 2007b).

Mycorrhizal Fungi

Arbuscular mycorrhizae (AM) fungi, which are endosymbionts, are the most important fungal symbiont in agroecosystems. Plant–mycorrhizal associations are the major mechanism for phosphorus uptake in over 80% of plant species. Colonization of roots by mycorrhizal fungi provides the plant with a well-distributed and extensive absorbing system in soil, and a greater chance of encountering fertile microsites not available to roots alone. The ability of mycorrhizal fungi to access small soil pores (Drew et al., 2003), and their

TABLE 7.3 Example of Cover Crop Functions That Can Be Evaluated.

Forbs		Legumes					Grasses			
	Brassicac	Bell Beans	Medics	Rose Clover	Strawberry Clover	Vetches	Barley	Oats	Orchard Grass	Tall Fescue
Function										
Adds N to soil		X	X	X	X	X				
N retention	X						X	X	X	X
Erosion control	X			X			X	X	X	X
Weed suppression			X						X	X
Improves soil structure and water infiltration							X	X	X	X
Inhibits nematodes	X			X						
Attracts beneficial insects	X					X	X	X	X	X
Opens up heavy soils	X	X								

Note that the legumes supply new soil N while Brassicas and grasses excel at N retention. This is a simple yes (X) or no (blank) assessment, however, a more detailed evaluation could provide a ranking or some other more quantitative information.

Source: Modified from Eviner, V.T., Chapin, F.S., 2001. Plant species provide vital ecosystem functions for sustainable agriculture, rangeland management and restoration. California Agr. 55, 54–59 (Eviner and Chapin, 2001).

ability to quickly respond to localized nutrient patches (Tibbett, 2000; Cavagnaro et al., 2005), increases the plant's access to these nutrients. This is of particular significance in soils of low nutrient status, and for immobile nutrients, such as NH_4^+ and PO_4 (Ames et al., 1983; Menge, 1983; Hetrick, 1991). Also, under drought stress, the role of mycorrhizal uptake of NO_3^- becomes more important since the NO_3^- supply to the roots via mass flow is reduced (Nichols et al., 1985). The N uptake mechanisms are largely unknown, but NH_4^+ is preferentially used. For example, corn plants colonized by *Glomus aggregatum* took up to 10 times more N from a $^{15}\text{NH}_4^+$ patch than from a $^{15}\text{NO}_3^-$ patch (Tanaka and Yano, 2005). While AM fungi increase the recovery of ^{15}N from decomposing plant residues in soil, it is unclear how much they rely on organic N, or if they accelerate OM decomposition (Hodge, 2004).

Background soil fertility and species diversity can influence the role of mycorrhizal contribution to nutrient cycling in agroecosystems. The species type and extent of mycorrhizal diversity can greatly influence nutrient uptake efficiency, ecosystem function, and NPP (van der Heijden et al., 1998). Under the nutrient-rich conditions that occur in industrialized agricultural systems, formation of mycorrhizal associations may become a cost to the plant, as the plant is able to satisfy its own nutrient requirements (Johnson et al., 1997). Agricultural production practices appear to have inadvertently reduced diversity, function, and efficiency in plant – mycorrhizal symbiosis (Daniell et al., 2001). In a meta-analysis of AM and ectomycorrhizal studies, colonization generally declined in response to N and P fertilization, although N effects on AM abundance were less strong than those for P (Treseder, 2004). One explanation is that mycorrhizae may be less important in

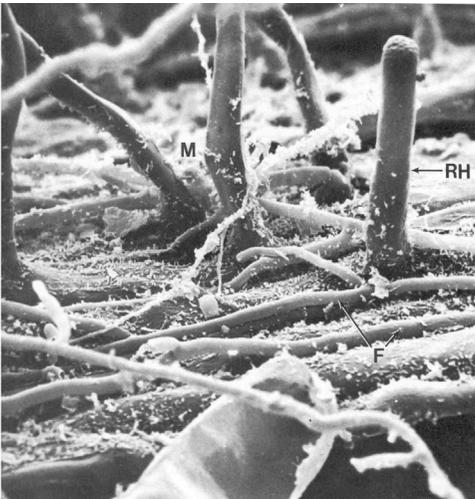


FIGURE 7.6 Electron micrograph of the root surface. Dense bacterial colonization can be seen on the root surface as well as fungal hyphae (F), root hairs (RH), and mucigel (M) from root exudates. From Foster, R.C., Rovira, A. D., Cook, T.W., 1983. *Ultrastructure of the Root-Soil Interface*. American Phytopathological Society St. Paul, MN, 157 p Foster et al. (1983).

facilitating plant uptake of NO_3^- , due its availability via mass flow, except in very N-limited ecosystems. In an organic farming system, a mycorrhiza-defective tomato mutant had 12% lower N content than the mycorrhizal wild-type, and there was more soil NO_3^- as well (Cavagnaro et al., 2006), indicating that AM are important in farming systems where fungicides and P fertilizers are not used. Manipulation of mycorrhizal populations to develop more efficient plant–symbiont combinations is in its infancy, but strategies that can be pursued include use of sparingly soluble rock P, reduced tillage, and integration of auxiliary plants that are highly mycorrhizal.

Biological N Fixation

Plants lack enzymes that can convert N_2 gas into a usable form and, as a result, most plants in natural ecosystems rely on the N released via microbial decomposition of SOM. Some bacteria, known as *diazotrophs*, do produce the enzyme nitrogenase that catalyzes the reduction of N_2 into NH_3 , a plant available form of N. This microbially mediated process is referred to as biological nitrogen fixation (BNF). Because N_2 is a stable molecule with a strong triple bond, BNF is an energy intensive process. Carbon is the primary energy source for the range of different types of bacteria, called diazotrophs, which carry out BNF. The high energy demand of BNF may explain why it is not more universally found in plant–microbe associations, and why N cycling in natural ecosystems is driven primarily by the recycling of previously fixed N through mineralization and immobilization of N from OM pools. Globally, BNF in unmanaged ecosystems is estimated to convert about 150 Tg of N_2 gas into plant-available N every year (Vitousek et al., 1997). In managed ecosystems, only an estimated 33 Tg N/year of nitrogen is fixed by cultivated legumes, while more than 100 Tg N/year is fixed nonbiologically through the fossil-fuel based production of synthetic nitrogen fertilizers (Galloway et al., 2003).

Diazotrophs and plants have evolved different degrees of association that facilitate the transfer of photosynthetically derived carbon from plant to bacteria to support BNF. Most diazotrophs are heterotrophic, and rely on plant-derived carbon to support BNF. Symbiotic diazotrophs that fix nitrogen within nodules of leguminous plants are the most familiar example of BNF, and are typically referred to collectively as rhizobia (Fig. 7.7). In most cases, the legume–rhizobia symbiosis is highly specific. Complex chemical signaling has evolved between legume species and specific rhizobial strains to initiate nodule formation. Symbiotic rhizobia in the nodule receive a direct supply of C in exchange for N-fixation for plant growth. The nodule provides physical protection for the rhizobia, while increasing the capture of the fixed N by the plant.

More recent work has identified numerous other diazotrophs that are associated with plant roots, either externally or internally within intracellular root



FIGURE 7.7 Leguminous plants are important in cropping systems worldwide. (A) In the Potosi region of Bolivia, Tarwi (*Lupinus mutabilis* Sweet) is a multipurpose legume that serves as a fertility source and grain crop. (B) The root nodules on Tarwi are large and numerous.

spaces. These associative diazotrophs utilize labile carbon root exudates as an energy source to support BNF. The plant has less control over the fate of the fixed N in this situation, compared with the N fixed within a root nodule. The N fixed by associative diazotrophs is incorporated into the bacterial biomass. This N becomes available for plant uptake when grazers feed on these bacteria, and trophic interactions in the rhizosphere food web result in a net N release (Clarholm, 1985). Due to the rapid turnover of microbial biomass in comparison with the much longer life-cycle of the plant, significant quantities of associatively fixed N end up in the associated plant (Ladha and Reddy, 2003). Some free-living diazotrophs, such as cyanobacteria, are autotrophs, and they are capable of both fixing carbon via photosynthesis and fixing N via BNF. Many cyanobacteria, in spite of their relative self-sufficiency, form symbiotic partnerships with plants, and the plants again are eventually the beneficiaries of the fixed N (Yoneyama et al., 1987).

The ecology of BNF is complex, and many things must be considered to optimize this valuable process. For example, at the field scale BNF is regulated by interactions between plant species, climate, and soil type. Similarly, at microscales, BNF is regulated by plant–microbe–microsite environmental interactions. The complexity of the ecology of BNF is reflected in the high variability found in BNF rates in natural and agroecosystems (Ojiem et al., 2007; Walley et al., 2001). The soil environment exerts influence on BNF through direct and indirect effects on the plants and microorganisms involved in N fixation. High soil temperatures ($>27\text{--}40^\circ\text{C}$), water stress, soil acidity ($\text{pH} < 5$), low soil P availability, and Al toxicity—all common conditions in certain tropical systems with highly weathered soils—can limit rhizobial growth and nitrogenase activity (Hungria and Vargas, 2000; Graham and Vance, 2003). The availability of molybdenum (Mo), a key component of the nitrogenase enzyme, can also be an important limiting factor in BNF in some soils (O’Hara, 2001). Because BNF only supplies new N to

agroecosystems while recycling other nutrients, integration with other soil amendments is critical to both the ability of the system to support the nutritional demands of BNF, and to maintain longer-term nutrient balances.

Nonsymbiotic N Fixation

Associative and free-living diazotrophs are commonly found in the rhizosphere of graminaceous species, such as rice, maize, sugar cane, and tropical pasture grasses. Field measurements of the contributions from associative and free-living diazotrophs reveal extreme variability, with contributions of 0–80 kg N/ha to crop growth (Bremer et al., 1995; Peoples et al., 2001; Table 7.3). With the rapid development of improved molecular methods, we are only beginning to scratch the surface of identifying the variety of diazotrophs responsible for fixing nitrogen in soils, and we still do not have methods that provide accurate measures of how much N free-living and associative diazotrophs are fixing in different agroecosystems (Buckley et al., 2007).

A promising area of biological N fixation research is in understanding the ecology and importance of associative N fixation with cereal species such as rice, maize, and sorghum. Watanabe et al. (1979) found that 80% of bacteria in rice roots are capable of fixing N. *Acetobacter*, a nonobligate diazotroph commonly found in sugar cane roots, can fix up to 150 kg N/ha per year (Boddey and Dobereiner, 1995). *Azospirillum* and *Herbaspirillum* are examples of diazotrophs commonly found associated with rice, sugar cane, maize, and sorghum roots. The amount of N fixed by these associative N-fixers has been highly debated, from a maximum of 5 kg N/ha (Giller, 2001) to a range of 1–25 kg N/ha for semiarid Australian cereals (Gupta et al., 2006). A recent study found that even in the presence of mineral N fertilizer, multiple tropical maize lines derived 12–33% of their N from associative N fixers (Montañez et al., 2009); however, evidence that associative BNF contributes significant N to maize in temperate grain production systems is lacking. It is important to note that much research remains to be done, and a further challenge is the nonspecificity of diazotroph–plant interactions that makes it more difficult to select and inoculate highly effective N-fixing strains compared to the more specific legume–rhizobial symbioses.

Partnerships With the Rhizosphere Community

Because the rhizosphere is the site of increased C availability, there are numerous other kinds of interactions that involve free-living or rhizosphere microorganisms that are not obligate symbionts. Rhizosphere microbial communities, referred to as rhizobiomes, vary between plant species, and even between different crop cultivars (Grayston et al., 1998; Peiffer et al., 2013; Turner et al., 2013). For example, the abundance of disease suppressing bacteria in the rhizosphere of wheat varies among wheat cultivars, resulting in differences in resistance to this pathogen across cultivars. Differences in the rhizosphere community are most commonly detected when there is either significant genetic

variation between plants (Bouffaud et al., 2014) or when closely related plants have distinct phenotypical differences relating to the rhizosphere (Briones et al., 2002, 2003; da Mota et al., 2002). Some evidence suggests that crop breeding under nutrient rich conditions has disrupted these plant–microbial interactions. In soybean, breeding history has altered the ability of modern cultivars to suppress rhizobia that are not fixing N for the plant, and as a result, these cultivars are vulnerable to parasitic Rhizobia strains (Kiers et al., 2007). Altered composition of nitrifying bacteria in the rice rhizosphere associated with differences in plant anatomy and physiology between cultivars resulted in different nitrification rates and N use efficiency (Briones et al., 2002). Another study of invasive grass species found that community compositions of nitrifying bacteria varied across grass species, giving rise to variations in the nitrification potential. These linkages between plant genotype–phenotype, rhizobiome composition, and nutrient assimilation suggest that cultivars which can access soil nutrient reservoirs and improve nutrient retention can be developed by targeting these plant–microbial interactions in the rhizosphere.

Since crop plants mainly take up NH_4^+ and NO_3^- rather than large polymer organic forms of N, mineralization is important for the N supply to plants in the absence of inorganic N fertilizer additions. The role of these microbial–plant interactions in stimulating N mineralization has been studied intensively. Plants can stimulate mineralization of organic substrates by supplying labile C to decomposers in the rhizosphere (Clarholm, 1985; Cheng et al., 2003; Hamilton and Frank, 2001; Kuzyakov and Xu, 2013). The rate of decomposition and N-mineralization varies with plant species (Cheng et al., 2003), rhizosphere community composition (Clarholm, 1985; Ferris et al., 1998; Chen and Ferris, 1999), and nutrient availability (Tate et al., 1991; Liljeroth et al., 1994). The release of nutrients for plant uptake appears to be enhanced by the involvement of secondary consumers feeding on the primary decomposers due to differences in the stoichiometry (the ratio of elements to one another e.g., N:P or C:N) between the two trophic levels (Clarholm, 1985; Ferris et al., 1998; Chen and Ferris, 1999; Fig. 7.8). This trophic cascade provides a mechanism for the primary producers to influence nutrient mineralization, similar to the so-called microbial loop in aquatic ecosystems where primary producers have been shown to increase excretion of soluble C under nutrient limiting conditions (Elser and Urabe, 1999). There is some evidence suggesting that terrestrial plants can influence the rate of net N mineralization through this mechanism, based on their need for nutrients by modifying the amount of soluble C excreted into the rhizosphere (Hamilton and Frank, 2001; Fig. 7.9).

Greater reliance on SOM as a nutrient source increases the importance of microbially mediated processes such as decomposition and mineralization. The tight coupling that occurs in the rhizosphere between net mineralization of N and P and plant assimilation reduces the potential for nutrient losses. Inorganic nutrient pools can be extremely small in ecosystems, while high rates of plant growth are maintained if N mineralization and plant assimilation are spatially and temporally connected in this manner (cf. Jackson et al., 1988). The

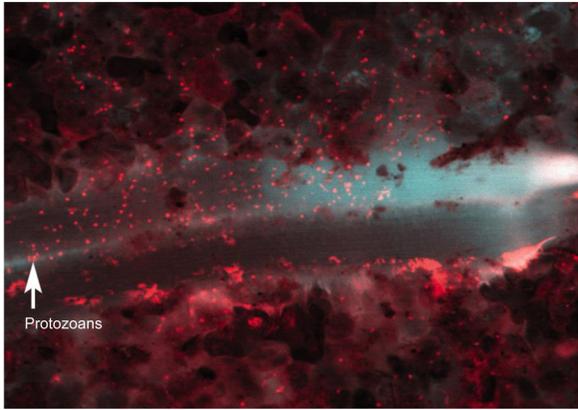


FIGURE 7.8 A swarm of protozoa grazing on red fluorescent bacteria in the rhizosphere. The rhizosphere is the home of numerous organisms that influence nutrient cycling and plant access to nutrient through food web interactions. From Bringhurst, R.M., Cardon, Z.G., Gage, D.J., 2001. Galactosides in the rhizosphere: utilization by *Sinorhizobium meliloti* and development of a biosensor. *Proc. Natl. Acad. Sci. U.S.A.* 98, 4540–4545. With permission copyright (2001) National Academy of Sciences, U.S.A.

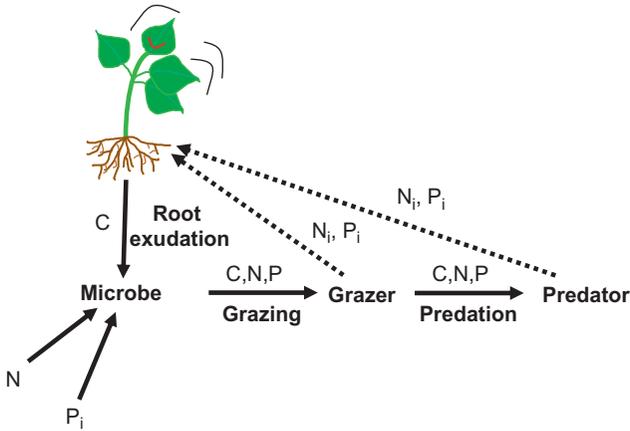


FIGURE 7.9 Feeding interactions across trophic levels increase net mineralization of N and P in the rhizosphere. Plants supply C to microbes who take up N and P from sources that are not available to plants, grazing of these microbes increases the rate of mineralization.

identity of the soil OM pools that are accessed through this mechanism remains unknown; however, phytoremediation studies show that decomposition of chemically recalcitrant substrates is accelerated in the rhizosphere compared to bulk soil (Reilley et al., 1996; Siciliano et al., 2003). This evokes the intriguing possibility that plants may be able to promote access to stabilized SOM pools through partnerships with rhizosphere microorganisms.

Microbial Mediation of Nutrient Cycling

Microorganisms represent a substantial portion of the standing biomass in agricultural ecosystems and contribute to the regulation of C sequestration, N availability and losses, and P dynamics. The size and physiological state of the standing microbial biomass is influenced by management practices, including rotational diversity (Anderson and Domsch, 1990), tillage (Holland and Coleman, 1987), and the quality and quantity of C inputs to the soil (Kirchner et al., 1993; Wander and Traina, 1996; Lundquist et al., 1999; Fliessbach and Mader, 2000). The mechanisms that control community structure and functional characteristics of below-ground ecosystem processes remain largely unknown, however, this is an area of active research and much progress has been made in the last decade.

While some plants are able to produce and secrete enzymes required for P mineralization (Vance et al., 2003) release of nutrients from organic compounds is largely carried out by heterotrophic microorganisms (Paul and Clark, 1996). Microbial production of extracellular enzymes that can attack polymers and release small, soluble molecules is an important mechanism contributing to the internal cycling of N, P, and S (McGill and Cole, 1981; Paul and Clark, 1996). Microbial community composition and metabolic status determine the balance between C released through respiration, and C assimilation into biomass during decomposition, as well as the biochemical composition of that biomass and the net release of plant available nutrients. Decomposers in soils with greater plant species diversity or greater abundance of C relative to N have reduced energy requirements for maintenance, and therefore convert a greater proportion of metabolized C to biomass (Anderson and Domsch, 1990; Fliessbach et al., 2000; Aoyama et al., 2000). Changes in microbial community structure can lead to increased C retention if the management practices result in fungal-dominated decomposer communities (Holland and Coleman, 1987).

Microbial Control of N Cycling

The relative abundance of C and N strongly influences the rates of competing microbial processes, and offers opportunities for farmers to optimize N cycling through manipulating microbial metabolism. For example, plant litter with a high C:N ratio initially increases microbial N immobilization, and decreases NH_4^+ and NO_3^- availability to plants. As microbial decomposition of these residues continues, and cascading effects on grazers and other trophic levels increase, net N release increases (Booth et al., 2005). When large amounts of inorganic N are added, as in industrialized cropping systems, inorganic N pools expand beyond the capacity of crop and microbial uptake, so that pathways of loss such as denitrification and leaching are increased. Soluble

fertilizer additions appear to stimulate preferential decomposition of some soil OM pools, including particulate OM (Neff et al., 2002).

In agroecosystems with low N fertility, plants and soil microbes compete for NH_4^+ and NO_3^- . In short-term studies, i.e., one to several days, microbes take up more inorganic ^{15}N than plants, presumably because they have higher substrate affinities, larger surface area to volume ratios, and faster growth rates than plants (Hodge, 2004; Schimel and Bennett, 2004). But after a month or so, plants contain an increasing proportion of the added ^{15}N , because the gradual release of microbial ^{15}N into the soil becomes available for root uptake, and plants hold on to N longer than microbes (Harrison et al., 2007).

Increased soil stocks of labile C substrates contribute to N conservation through both aerobic and anaerobic pathways (Silver et al., 2001; Burger and Jackson, 2003). Studies in agricultural soils simulating conditions in bulk soil indicate that the major fate of NH_4^+ is nitrification (Shi and Norton, 2000; Burger and Jackson, 2003). Competition between heterotrophs and nitrifiers for NH_4^+ is strong, resulting in very small NH_4^+ pools, and nitrification rates can be two- to threefold greater than NH_4^+ immobilization (Burger and Jackson, 2003). Nevertheless, soils receiving greater C additions supported a larger, more active microbial biomass, resulting in a greater proportion of NO_3^- assimilation and reduction of standing NO_3^- pools (Burger and Jackson, 2003).

Carbon abundance also influences dissimilatory NO_3^- reduction pathways in ways that support N conservation and reduce environmental impacts. In one study, denitrification in soils receiving organic N amendments reduced the proportion of N lost as N_2O (Kramer et al., 2006). Carbon abundance can also favor a second anaerobic pathway, DNRA (Silver et al., 2001). This process was thought to be limited to extremely anaerobic, C-rich environments such as sewage sludge and estuarine or lake sediments (Giles et al., 2012), but has recently been detected in a broad range of unmanaged terrestrial ecosystems (Giles et al., 2012), and in agricultural soils (Yin et al., 2002). Silver et al. (2001) reported average rates of DNRA were threefold greater than denitrification in humid tropical forest soils. The resulting reduction in NO_3^- availability to denitrifiers and leaching may contribute to N conservation in these ecosystems (Silver et al., 2001). In rice paddies, soils with greater levels of SOC due to additions of straw mulch had threefold greater DNRA compared to soils where straw was removed and endogenous SOC was reduced (Yin et al., 2002).

As with N, soil OM levels and C abundance influence microbially-mediated processes that control the uptake of P by microbes, as well as mineralization and biological weathering. For example, microorganisms solubilize sparingly soluble inorganic P through several mechanisms if they have adequate C substrates for growth and reproduction but are lacking P (Illmer et al., 1995; Oberson et al., 2001). Direct excretion of phosphatase enzymes is one mechanism of phosphate-solubilization. Another is local acidification

through organic acid excretion, such as occurs in the soil fungus *Penicillium radicum* isolated from a low-P rhizosphere of unfertilized wheat (Whitelaw et al., 1999). In this system, phosphate-solubilization from insoluble or sparingly soluble complexes with calcium, colloidal aluminum, and iron was related to titratable acidity and gluconic acid concentration. Organic acid excretion not only alters pH, but also may chelate Al_3^+ or other cations, directly further enhancing the solubilization of phosphate (Erich et al., 2002).

The assimilation of inorganic phosphorus by microbes may protect phosphorus from geochemical adsorption reactions with soil particles, through microbial turnover and OM mineralization processes which are synchronized with plant and further microbial uptake. Indirect evidence for this is the enhanced levels of microbial P and cycling of P from inorganic to organic and plant forms associated with managed systems that had enhanced soil biological activity and legumes present (Oberson et al., 2001). Labeled glucose and residue studies have recently shown that biomass P turnover is rapid, approximately twice as fast as C (Kouno et al., 2002). This indicates that the potential for microbial P pools to support plant P requirements may have been markedly underestimated.

CONCEPTS AND STRATEGIES TO OPTIMIZE ECOLOGICAL NUTRIENT MANAGEMENT

Using Spatial and Temporal Scales to Organize Management Decisions

The use of both temporal and spatial scales to organize nutrient flows into a logical structure is fundamental to developing a coherent set of management strategies that act together to achieve the goals of ecological nutrient management. The spatial scales we must consider range from microns to the plant, field, and farming community, or regional scales. We can think of these spatial scales as nested within one another, so that at any level, we are able to identify the location of the processes we are aiming to manage. For example, the use of rock phosphate as a source of P involves processes occurring at the micron, plant, and field scales (Fig. 7.10).

In using a sparingly soluble P source, the farmer is aiming to modify the solubilization of P, a microscale process which is mediated by microorganisms and the rhizosphere of some plant species. The background soil environment and climate affect processes occurring at every level, including the farmer's decisions. Assuming that P is a limiting factor in this field, plant productivity will be impacted by field-scale management decisions and the resulting rate of P-solubilization. The field-scale decisions that will directly influence this process are: (1) choice of amendments at the field scale, (2) selection of plant species, and (3) inoculation of P-solubilizing microbes (may be an option in the future!). Interactions across these scales impact one

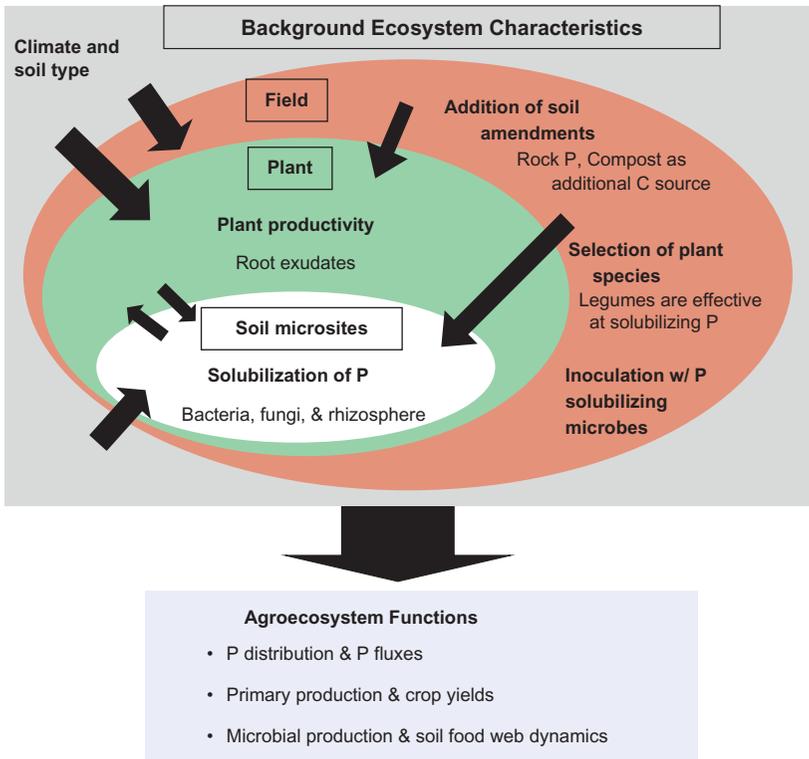


FIGURE 7.10 Processes occurring at multiple scales must be considered in nutrient management decision-making. Here the processes that impact the decision to use rock P and the ultimate outcomes are illustrated. See text for full discussion.

another, i.e., field management impacts the plant and soil microbes, P solubilization influences plant productivity and creates feedback, because increased productive capacity increases the ability of the plant to stimulate P solubilization through direct and indirect means. Adding rock P in conjunction with planting a legume can be particularly effective, because legumes are able to access sparingly soluble P. If rock P was added without consideration of plant species, or use of an additional C source (such as compost or manure) to support microbial P-solubilization, then it is possible that there may not be a detectable improvement in crop yields in the first growing season because microbial activity is limited by access to C (energy), not P. The inclusion of a supplemental C-source is particularly important if a nonmycorrhizal crop is to be planted. This example illustrates how systematic analysis of processes occurring at different scales can help in planning management interventions.

Just as interactions across spatial scales were important in conceptualizing the key processes in the rock P example, interactions among processes

occurring at different rates is also a useful organizing principle. This is particularly important when a major change in an agroecosystem's management regime is implemented, such as increased inputs of organic residues or a change in tillage regime. There are major differences in the process rates and the flux through various pools. As a result, the MRTs of relevant nutrient pools range from minutes for transient inorganic N forms, such as nitrite, to centuries for stabilized pools of SOM. Generally speaking, the spatial and temporal scales of ecological processes are commonly linked. Small-scale or local processes are often ephemeral and rapid. Examples are nutrient transformations controlled by microorganisms, such as nitrification or mineralization, and nutrient uptake by a single fine root. These rapid, small-scale processes and interactions can be highly variable in space and time, but in aggregate, they determine agroecosystem functions at the field scale. For example, two competing biological processes that occur very rapidly yet play a significant role in regulating the amount of N lost from a field on an annual basis are the flux of NH_4^+ into fine roots versus the conversion of NH_4^+ to NO_3^- by nitrifying microorganisms. If the former predominates, then the NH_4^+ available to nitrifiers is reduced and NO_3^- pools remain small. On the other hand, when nitrifiers have access to ample NH_4^+ , then NO_3^- production is elevated resulting in larger NO_3^- pools and increased losses of N.

Managing agroecosystems to modify reservoirs in support of longer mean residence times requires planning for management that occurs over longer time frames compared to small-scale processes and pools that are fast cycling. For example, soil degradation and restoration results from slow changes that accrue over decades rather than years, and that represent the sum of many shorter-term processes and events. Yet it is these longer-term processes that are critical to the long-term sustainability of agroecosystem production.

One approach that has been used in the case of soil OM is to focus efforts on OM pools that can be influenced by management in shorter time frames. Because of the different mean residence times of the soil organic pools that are impacted by management changes, the shift to new steady state conditions will occur over multiyear, decades, and even longer time-scales. Agroecosystems that are undergoing changes in ecological processes are considered to be "in transition" (Liebhardt et al., 1989). During this transition period, there are clear signs of directional change. For instance, when soluble fertilizers are replaced with organic nutrient sources, subsequent shifts in C and N cycling impact crop yields and the distribution of SOM pools (Wander et al., 1994; Liebhardt et al., 1989). In the short-term, organic inputs will have a greater impact on faster cycling processes. To impact slower cycling SOM reservoirs requires that nutrient management strategies consider time frames of 5–10 years. Fig. 7.11 illustrates how a green manure incorporation impacts SOM pools with differing mean residence times and their contribution to crop N supply.

While many ecological processes that govern nutrient availability fall somewhere along the continuum from small-scale and fast to large-scale and

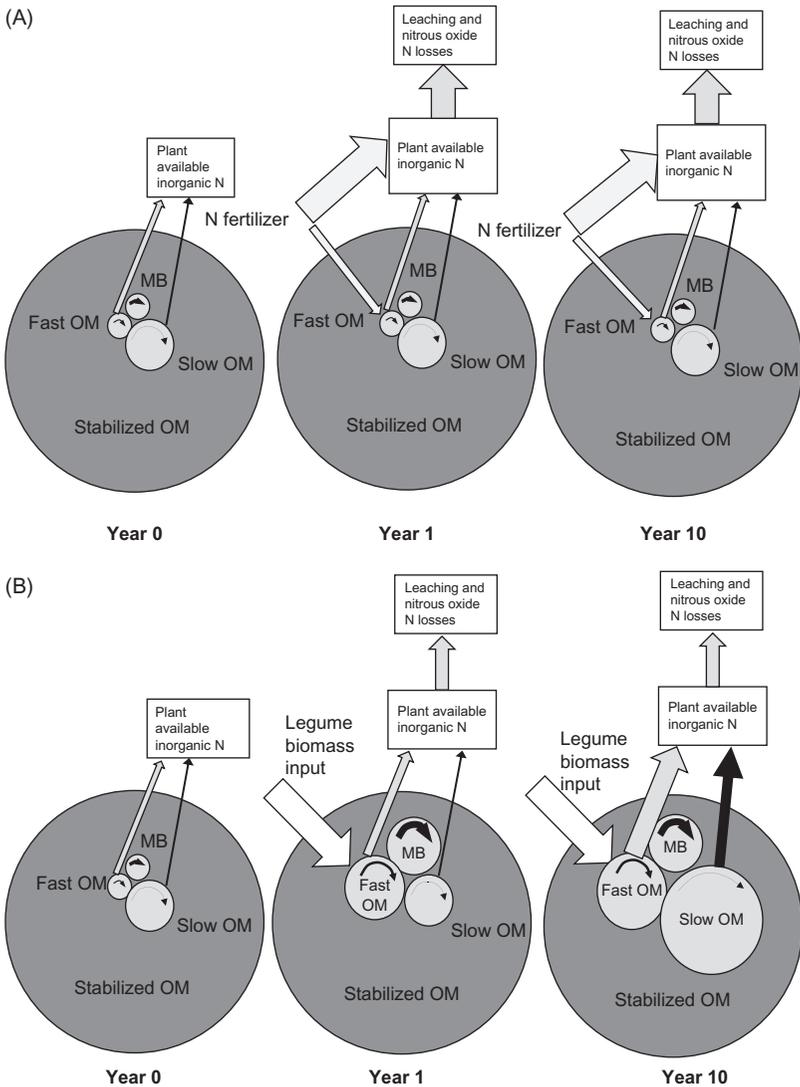


FIGURE 7.11 The effect of sole use of N fertilizer compared to legume additions on soil organic matter pools and N availability over time. Four soil organic matter pools are represented in each diagram: microbial biomass (MB), fast-cycling organic matter pool primarily composed of recent litter additions (Fast OM), slow-cycling organic matter pool primarily composed of partially decomposed litter (Slow OM), and the much larger stabilized SOM pool which is unavailable for microbial decomposition and plant N uptake. In both A and B, year 0 represents a relatively degraded soil with low soil organic matter levels and low N supplying capacity from decomposition of fast OM with small contributions from slow OM. (A) Uncoupled nutrient management using N fertilizer without any added C sources. Fertilizer applied alone without carbon sources (residues, compost, manure, and legume vegetation and roots) is taken up by crop plants, but very little is retained as soil OM. About 40% or more of N fertilizer applied is lost through leaching and gaseous loss pathways. Even after 10 years of management with N fertilizer, soil organic N reserves have not increased. (B) Integrated management using legumes as an N source resulting in carbon-mediated retention of

(Continued)

slow, there are exceptions. In agricultural systems, it is not uncommon to have large-scale processes that occur very rapidly. Management interventions such as crop harvest, burning, and tillage are examples. These events cause rapid, large-scale changes, and result in dramatic shifts in virtually all the smaller-scale ecosystem processes from one moment to the next. A well-known example is the pulse in soil respiration that occurs after tillage.

Evaluating Management Impacts on Agroecosystem-Scale Nutrient Flows

To efficiently gauge the impact of farm management on longer-term soil fertility and sustainability, it is crucial to consider the flow of nutrients across boundaries of management units, as well as the larger landscape in which they are embedded. To analyze the movement of nutrients across field and farm boundaries, net flows of nutrients can be estimated by constructing nutrient budgets. This approach is used in ecosystem ecology to compare fluxes into and out of a defined compartment, which can be as small as a patch of organic residue in the soil (Hodge, 2004), or as large as the entire atmosphere (Schlesinger, 2005), in order to find out whether the balance of these fluxes are positive or negative. Over the last 10 years, the value of nutrient budgeting as a tool for analyzing nutrient flows in agroecosystems and agricultural landscapes has become apparent, and the approach has been widely applied at a variety of scales. Depending on the questions that are being addressed, the scale of land unit used can be individual fields, farms, watersheds, or even whole regions and countries.

To conceptualize how management is affecting the nutrient status of a field or other management unit, we treat the field as a compartment, and focus on inputs and exports across the field boundary. The simplest nutrient budgets emphasize the flow of nutrients that are controlled by the manager, such as fertility inputs and harvested exports (Fig. 7.12). These fluxes are usually the dominant flows that regulate the transfer of nutrients into and out of a field or farm. These simple mass balances provide a starting point for managing smaller-scale processes that are regulating the fate of nutrients in agroecosystems. While this balance does not quantify internal nutrient

◀ N though increased coupling of N and C cycles. In year 1, legume biomass is incorporated into the cropping system. Biomass enters the fast OM pool and drives a rapid increase in the size and cycling rate of MB. Labile carbon and nitrogen compounds are decomposed rapidly by microbes, resulting in a quick burst of N availability. The slow OM pool is not immediately impacted by the legume addition. Carbon and N from the legume biomass flows through the food web to eventually become part of the slow OM pool on a decadal time scale. By year 10, the size of MB and fast OM pools are maintained, and a larger slow OM pool also contributes to the N supplying capacity of the soil. The net effect is greater N availability from these soil reserves for crop uptake, along with small gains in total soil OM to support longer-term nutrient supplying capacity.



FIGURE 7.12 Major nutrient flows in a smallholder cropping system in the Potosi region of Bolivia where animals are an integral part of fertility management. Farmers harvest nutrients from rangeland through grazing their animals on marginal lands and manure is deposited in corrals (*dotted arrows*). Internal transfers of P also occur when harvested crop residues serve as forages for animals (*thin solid arrow*). Manure is used primarily on fields that are closest to the homestead, although some is transported to farther fields (*thick black arrows*). The manure contains nutrients that have been captured from communal rangelands as well as recycled from cropping fields. Nutrients leave the agroecosystem as harvested crops and through losses to the environment (*gray arrows*). Erosion is the major environmental nutrient loss pathway in these systems.

cycling processes or environmental losses resulting from these internal processes, it provides useful information for assessing whether surplus or inadequate nutrients are being added, and thus is useful in developing nutrient management strategies. All things being equal, environmental losses are directly related to the level of N and P availability. Soils with excess applications will lose more through microbially mediated processes compared to soils that do not have surplus nutrients (Aber et al., 1989).

Construction of a field-scale mass balance entails calculating the difference between inputs and harvested exports over the course of a rotation cycle (Drinkwater et al., 1998). All fertilizers, soil amendments, and N-fixing crops must be accounted for as inputs. One of the most challenging aspects of using this budgeting approach is the determination of N inputs from leguminous cover crops. A common practice for legumes is to measure N in standing biomass for green manures as an estimate of N from BNF, and to consider no net gain or loss of N for leguminous grain crops (Drinkwater et al., 1998). The exports are all harvested crops or animals, including grains,

BOX 7.1 How Do Different Nutrient Management Strategies Affect N Mass Balance in Grain Systems?

Fifteen-year N balances for three distinct grain production systems: (1) MNR—integrated system with grains, forages, and legumes, with animal manure returned to the field. (2) Cash grain systems with leguminous green manures as the only N source. (3) Cash grain system based on soluble N fertilizer inputs (modified from Drinkwater et al., 1998). All systems include maize and soybean, while only the MNR and LEG also grow wheat and leguminous green manures.

Nitrogen Balance	MNR (kg/ha)	LEG (kg/ha)	CNV (kg/ha)
Nitrogen inputs	1365	740	1310
Nitrogen in grain exports	-825	-745	-790
Surplus or deficit: (Total inputs—Exports)	540	-5	520
Net change in soil nitrogen	415	110	-495
N not accounted for (Total inputs—Exports)—Soil N change	125	-115	1015

These simple input – output balances show that the LEG system is running close to steady state, i.e., inputs are roughly equal to harvested exports, while the MRN and CNV have accrued comparable surpluses of N over this 15-year period. These differences in N balance are driven mainly by the inputs rather than yields, since the harvested N in these three cropping systems is similar. If we include data on soil N using samples conducted at the beginning of the experiment and then 15 years' later, we can detect a significant increase in soil N for the MNR, while the CNV system shows a significant decline in soil N for the same time period. The small increase shown for the LEG system is not statistically significant.

forages or crop residues removed, manure or animal biomass removed. This simple budgeting method can be very useful as an indicator of directional change and the relative efficiency of divergent nutrient management strategies (Box 7.1). Negative balances indicate that deficits are accruing, and that nutrients are being extracted from the soil (Box 7.2). In this case, if nutrient management practices are not modified, soil fertility will continue to be depleted and production will decline. Chronic surpluses may indicate that over-application is a problem, however, to fully determine whether or not these surpluses are being retained in the field, additional analysis of soil stocks (i.e., Box 7.1) and potential loss pathways such as erosion (Box 7.2) will need to be evaluated.

BOX 7.2 Intercropping of Pigeon Pea Reduces Erosion and Increases Grain Yields and P Recycled Through Active Soil OM Pool.

In the table below, net exports of P in yields and through erosion at two sites with differing erosion rates in Songani, Southern Malawi. Erosion P losses were estimated after [Stoorvogel et al., 1993](#) where erosion rates were estimated as follows: (1) site 1 (2% slope) erosion was estimated at 5 ton ha year; and (2) site 2 (30% slope) at 20 t/ha per year ([Snapp et al., 1998](#)). Based on percentage ground cover, we estimated that erosion, and the resulting P loss, was reduced by 25% when maize was intercropped with pigeon pea intercrop, compared to monoculture maize. Long-duration pigeon pea extends the period of soil cover over a 4-month period of intermittent rains.

	Yield Maize	Yield P'pea	P Harvested in Grain	Erosion P Loss	Net P Balance	P Recycled in Crop Residues (Internal P Cycle)
	t ha ⁻¹	t ha ⁻¹	kg/ha ⁻¹	kg/ha ⁻¹ per year ⁻¹	kg/ha ⁻¹ per year ⁻¹	kg/ha ⁻¹ per year ⁻¹
Maize, low erosion site 1	1	0	2	2.3	-4.3	7.7
Maize, high erosion site 2	0.5	0	1	9	-10.0	3.8
Maize + P'pea,site 1	1.1	0.4	3.4	1.7	-5.1	17.4
Maize + P'pea,site 2	0.5	0.3	1.9	7.8	-11.7	9.8

Because there are no inputs of P for this maize crop, all P balances at the end of the season are negative, indicating that a net export of P has occurred. Phosphorus lost through erosion is threefold greater in the steeper field. The addition of pigeon pea into this system increases the export of harvested P while also reducing P lost through erosion. However, because of the increase in harvested yields, overall P removal is accelerated with pigeon pea + maize. As a result, although erosion losses are reduced at each site, the need to add P through soil amendments is increased by intercropping. The last column reports the P content of crop residues from maize or maize + pigeon pea, and shows how the inclusion of the legume more than doubles the amount of P that will be recycled back into labile OM pools.

Integrating Background Soil Fertility Into Nutrient Management Planning

Nutrient cycling in agroecosystems reflects interactions between the environment, management, and the organisms present in the system. While management practices can exert a strong influence on shorter-term outcomes such as crop nutrient uptake and yields, the particular effect of identical management strategies will vary across farms, depending on climate, soil type, and the legacy of past management decisions. These inherent characteristics of the agroecosystem need to be considered in developing overall nutrient management strategies. Fig. 7.13 illustrates how management practices can have different results, depending on the initial fertility status of a site. In this diagram, we have laid out three different management scenarios for two fields that differ in terms of the initial fertility status. The cause this difference in soil fertility is inconsequential, it could be due to either soil type difference, or past management history.

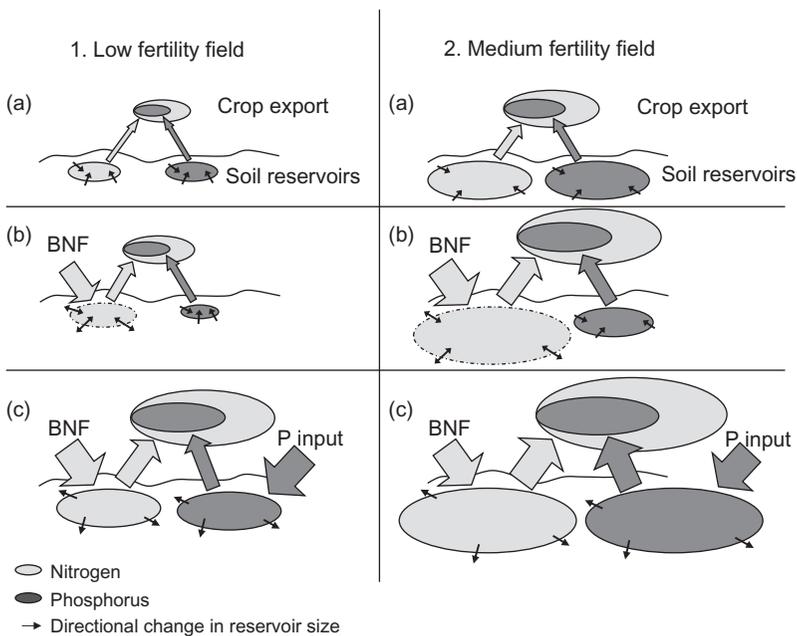


FIGURE 7.13 Management practices will have differing consequences depending on background soil fertility. A low fertility (1) and medium fertility (2) background are compared for three management options: (a) absence of any fertility additions, (b) inclusion of N-fixing crop in the system where at least part of the N-fixed is retained in the system, and (c) N-fixing crop combined with modest addition of rock P. The effect of these three management regimes is illustrated in terms of relative flows of N and P into soil pools and crop harvest, changes in soil pools, and crop N and P content. See text for full discussion.

For a low fertility field (Fig. 7.13 (1a)), small soil N, P, and OM reservoirs support low productivity. With crop exports, small soil reservoirs continue to shrink, leading to a downward spiral of soil degradation. If legumes alone are incorporated into a cropping system in this field (Fig. 7.13 (1b)), BNF will only provide a small benefit as legume growth will be limited by a small and increasingly shrinking soil P reservoir. The presence of legumes in the system can increase the availability of soil P (Bah et al., 2006), but this will only increase the rate of P depletion over the long-term. In this scenario, legumes are not likely to increase productivity, but they may help maintain soil N reservoirs and SOM status. If incorporation of legumes is paired with modest P fertilizer additions (Fig. 7.13 (1c)), BNF can make much larger contributions to overall productivity. If the P fertilizer is in organic form, such as manure, SOM reservoirs can also be increased. It is at higher levels of productivity that retaining crop residues becomes more economically feasible for farmers, reinforcing the maintenance of the SOM reservoir. As the OM reservoir increases, the capacity of the soil to retain N and P in relatively available forms increases.

For a medium fertility field (Fig. 7.13 (2a)), modest crop production can be sustained over the short-term by the mining of existing soil N and P reservoirs. With the incorporation of legumes into this field (Fig. 7.13 (2b)), BNF can provide substantial benefits because biomass production is not limited by P and other nutrient availability. Legumes can improve the P status of the soil by moving P from less to more available soil pools (Bah et al., 2006). Long-term dependence on just legume BNF input will eventually lead to P depletion, and this medium fertility soil could shift to a low fertility status as in Fig. 7.1B. Again, if incorporation of legumes is paired with modest additions of P fertilizer (Fig. 7.13 (2c)), BNF can make much larger contributions to productivity over the long-term. As in Fig. 7.13 (1c), the form of P fertilizer and the quantity of crop residues retained affects the longer-term SOM reservoir.

Synthetic N fertilizers, as an alternative to BNF, could help boost productivity in any of the non-P limiting scenarios. However, inorganic N fertilizers can exacerbate soil P depletion (Lupwayi et al., 1999), and do not contribute to longer-term nutrient cycling capacity of the system if they are not coupled with C inputs.

The size of nutrient exports relative to crop residues retained in the field determines the directional change in soil reservoir size. Legume grain crops, e.g., tend to export almost as much N as they fix (Alves et al., 2003). As yields increase with increasing soil fertility, grain crops can export more N than they fix (Ojiem et al., 2007). Incorporation of green manures as intercrops or relay crops can help balance N exported in a cash crop with N fixed by the legume (Lupwayi et al., 1999).

Strategic Use of Soil Amendments and Cover Cropping to Enhance Linkages Between Cycles

We have discussed the role of plants and microbes in connecting N and P cycling with C flows, and emphasized how these linkages support internal

cycling capacity and promote nutrient retention. Here we provide specific examples of how use of various nutrient sources can either promote or impair these linkages. For example, while the initial crop uptake of inorganic, soluble fertilizers is greater than crop uptake from other forms of amendments such as organic residues or rock phosphate, retention of these soluble forms in the ecosystem through conversion to soil OM is reduced, resulting in greater environmental losses (Bundy et al., 2001; Ladd and Amato, 1986; Drinkwater et al., 1998). This greater loss of soluble fertilizers occurs because the processes that sequester soluble nutrients are saturated (Azam et al., 1985; Ladd and Amato, 1986; Hodge et al., 2000) leaving NO_3^- vulnerable to leaching/denitrification. In contrast, microbial assimilation of N from organic sources is two- to four-fold greater than for N from inorganic fertilizer, leading to increased storage of legume-derived N in SOM pools. Likewise, soluble, surplus P sources push P cycling into absorbed, precipitation, and occluded pools.

In cropping systems where soluble fertilizers are part of the overall nutrient management strategy, fertilizer additions should be managed to enhance assimilation in biologically regulated sinks. In rotations, inorganic sources can be preferentially applied to those crops with higher NUE. Improving the spatial/temporal connections between fertilizers and senescent crop residues appears to increase the retention of older SOM fractions (Clapp et al., 2000), suggesting it may be advantageous to add small portions of fertilizer when high C crop residues are being incorporated. A review of three long-term trials from temperate countries suggests that the fate of soluble P from fertilizers depends on whether P was added primarily as an organic or inorganic source, as well as soil characteristics (Blake et al., 2000). In these studies, P-use by plants was much more efficient if applied in balance with C.

Targeted use of animal manures facilitates plant and microbial uptake of P, through a range of mechanisms. These include direct competition for adsorption sites by manure-compounds, enhanced release of P from sparingly soluble pools through altered pH and soluble C addition, and enhanced microbial activity (Erich et al., 2002; Laboski and Lamb, 2003). Where manure is utilized at sustainable, moderate levels, and livestock are distributed extensively across the landscape, organic-P sources appear to be inherently less vulnerable than inorganic fertilizer sources to loss from occlusion, erosion, or leaching (Powell et al., 1999). While manure additions also contribute to N fertility and SOM pools, in the long-term, soil nitrogen status will depend in large part of the proportion of land devoted to symbiotically-fixing plant species. Use of animal manures serves as a mechanism to recycle N and P back to cropping fields where forages were produced.

Likewise, use of sparingly soluble inorganic P inputs should be combined with strategies to link P solubilization with C flows. Application of sparingly-soluble sources of P to crops (e.g., most legumes) that can assimilate P into biological pools is an efficient strategy to bypass desorption, precipitation, and occlusion of P (Oberson et al., 1999). In degraded soils, additions of rock P may be need to be combined with the use of shrubby, short-lived, mycorrhizal

plants that have been shown to reduce erosion, build OM, and assimilate N and P into plant accessible N and P pools. Two legumes species, pigeon pea and lupin, are notable for providing these multiple ecosystem services, and have also been shown to access sparingly soluble phosphorus pools. Interestingly, these crops are commonly integrated with nutrient-demanding crops in indigenous cropping systems. For example, pigeon pea is grown as an intercrop with maize in India, and lupin as a rotational crop just before potato in the Andes. The use of legumes to transfer P from mineral forms to labile OM pools increases the amount of P that is actively cycling via biological processes and can contribute to increased P uptake by subsequent crops that may not have the ability to access sparingly soluble P. An example is provided in [Box 7.2](#), where the amount of P recycled in crop residues increased nearly threefold when pigeon pea was intercropped with maize.

The consistent theme uniting all of these strategies is to evaluate the possible fates of various nutrient sources, and to also consider how to link use of these sources with enhanced C cycling. Furthermore, the greatest potential for effective nutrient cycling is realized when soil amendments are combined with use of biological N fixation.

Biological N-Fixation: A Key Source of Nitrogen

Effective management of biological N fixation is central to ecologically based nutrient management. The most familiar example of symbiotic nitrogen fixation is the close association between legumes and rhizobial bacteria (*Rhizobium*, *Mesorhizobium*, *Sinorhizobium*, and *Bradyrhizobium*) although associative and free-living diazotrophs are potentially important in several monocot crops.

Legumes can be incorporated into crop rotations either intercropped with nonlegumes or in sequential (relay) rotations. A disadvantage of relay cropping is that mineralization of N may not coincide with the subsequent crop N demand. Beneficial effects of relay cropping systems include the addition of OM and mineralization of N from residual legume biomass that can support the growth of subsequent, nonlegume crops. Grain legumes, such as soybeans, are typically grown as monocultures in rotation with nonlegume grain crops, such as maize. Grain legumes are the most common type of legume in cropping systems, because they provide essential human and livestock protein sources in a form that is easily stored and transported. Grain legumes, such as soybeans, can fix up to 200 kg N/ha per year ([Table 7.4](#)). However, most of this N is exported off the farm in the protein-rich seeds, resulting in low or negative net soil N balance. Most estimates of N fixation, however, do not include root biomass, which can be 16–77% of total plant N ([Table 7.5](#)). Root biomass is difficult to measure; however, from the limited data available, it is clear that legume species can vary greatly in root-to-shoot ratios. Perennial species tend to have a higher root:shoot ratios than annual species ([Antos and Halpern, 1997](#)). This is generally supported by recent below-ground N results ([Table 7.5](#)), where perennial legumes tend to

TABLE 7.4 Average and Upper Range of Biological N-Fixation Contributions to Tropical Cropping Systems

Associated Crop	Average N Fixed (kg N/ha per year)	Upper Range of N Fixed (kg N/ha per year)
Rice: <i>Cyanobacteria</i> ^b	30	Up to 80
<i>Azolla</i> : <i>Anabaena</i> in rice ^a	32	–
Sugarcane: <i>Acetobacter</i> ^c	–	Up to 150
Grain legumes ^a	77	Up to 200
Green manure legumes ^a	85	Up to 300
Pasture legumes ^a	78	Up to 250
Leguminous trees and shrubs ^a	150	Up to 275

^aFrom Giller (2001). Legume nitrogen fixation values do not include below-ground biomass and are, therefore, underestimates.

^bFrom Roger and Ladha (1992) As cited in Reis (2000).

^cFrom Boddey et al. (1995).

TABLE 7.5 Measured Legume Below-Ground N Biomass as a Percentage of Total Plant N

Legume	Primary Use	BGN as % of Total Plant N	Sources
Chick pea (<i>Cicer arietinum</i>)	Grain	29	Turpin et al. (2002)
Fava bean (<i>Vicia faba</i>)	Grain	25	Khan et al. (2003)
Fava bean (<i>Vicia faba</i>)	Grain	17	Mayer et al. (2003)
Field pea (<i>Pisum sativum</i>)	Grain	17	Mayer et al. (2003)
Jack bean (<i>Canavalia ensiformis</i>)	Green manure/ forage	39	Ramos et al. (2001)
Mucuna (<i>Mucuna aterrima</i>)	Green manure	49	Ramos et al. (2001)

have a higher below-ground N as a percentage of total plant N (average of 43%) than the annual grain legumes (average of 32%). Environmental conditions can also influence root biomass and root architecture. Generally, plant allocation to roots increases under drought conditions. If estimates of root biomass are included, grain legumes can provide modest positive N balances, even with high grain N exports.

Intercropping systems incorporate legumes into agroecosystems by planting legumes and nonlegumes together in close proximity in the same field. Examples of an annual intercropping system include maize–pigeon pea mixtures (Snapp et al., 2003). Legume intercrops can supply a slow, but steady supply of N for the nonlegume crop. Furthermore, intercropping can also reduce soil erosion and nutrient leaching, contribute to suppression of weeds and pathogens, and provide food and shelter for beneficial insects. To provide these benefits while increasing yields, intercrops must combine crop species that maximize complementarity and minimize competition for light, nutrients, and water. One of the major constraints to the adoption of legumes in cropping systems is the opportunity cost of taking land out of production in either space, as part of an intercrop, or in time as part of a legume relay cropping rotation. For this reason, successful adoptions are more likely when legumes serve multiple purposes of producing a net positive N balance, while still producing consumable products or livestock forage. Pigeon pea is one such example of a green manure crop that produces a high-protein vegetable product while maintaining a positive N balance (Ghosh et al., 2007).

In contrast with grain legumes, green manures are grown for the primary purpose of improving soil N fertility, and are typically incorporated into the soil at a maximal stage of biomass production. Tropical green manures, such as *Canavalia*, *Crotalaria*, and *Mucuna*, commonly fix over 100 kg N/ha per year, all of which is retained in the system, resulting in more positive N balances than grain legumes. Green manures as relay crops are more commonly used in temperate systems, because of lower land pressures and because they can be grown during the colder winter months when crop production is not possible. In tropical systems, relay green manures are less common due to high land pressures, limited labor supply, the ability to produce crops year-round in some regions, or the lack of water to support green manure growth during the dry seasons between cropping seasons. Intercropping of green manure crops to supply N to a simultaneously growing cash crop have been adopted in some systems. The aquatic fern, *Azolla*, and its symbiotic association with the cyanobacteria *Anabaena* provides an example of a green manure that is used exclusively as an N source when intercropped in lowland rice systems. With 80–95% of *Azolla* N derived from fixation, rice–*Azolla* intercrops can fix approximately 30 kg N/ha (Yoneyama et al., 1987; Choudhury and Kennedy, 2004). Some constraints to more widespread adoption of *Azolla* are pest pressures, P limitation, and limited irrigation availability in some regions (Giller, 2001).

Farmers that have limited land, labor, and other resources are interested in “dual purpose” legumes, which have an intermediate phenology. That is,



FIGURE 7.14 A Bolivian farmer shows off his fava bean crop. The previous potato crop failed due to unfavorable climatic conditions, leaving behind P from the manure application that is normally applied to potatoes but not to bean crops. As a result, the fava beans produced a very large biomass.

they provide a product, such as leaf, vegetable, or grain, while at the same time providing long-term benefits through residues that suppress weeds and build soil fertility (Fig. 7.14). There is a trade-off, as carbohydrate and nutrient invested in residues provides less resources for yield potential, thus residue biomass is inversely related to harvest index across legume species (see Fig. 3.12). Examples of dual purpose, low harvest index legumes include long-duration pigeon pea, forage soybean, and mucuna. Such species provide returns to farmers in the short-term—and thus the economic feasibility of adoption—while simultaneously contributing to ecosystem services.

Over the long-term, dual purpose plant types contribute to resilient cropping systems. This is both through the soil building properties of high quality residues, and the inherent ability of indeterminate growth types to recover from pest epidemics. Plant breeding efforts have historically focused on producing high yield potential phenotypes. Examples include the development of new varieties of pigeon pea and cowpea that are extra-early, and extra short duration. These crops often incorporate high harvest index traits, which has had the unintended consequence of reducing biomass available for fodder, weed suppression, and soil fertility enrichment. Producing a wider range of dual purpose genotypes with intermediate phenology, and experimenting with intercrops of short and long-duration crops are approaches that require careful consideration in the future.

Alley cropping involves the use of woody or shrub perennial legumes between “alleys” of nonlegume crops. Prunings from the legumes are used as livestock forage, and/or added to the soil as a N source for the nonlegume.

Inclusion of perennials in cropping systems provides important ecological benefits due to their extensive rooting systems that persist across multiple cropping seasons. Perennials can reduce soil erosion, access deeper soil pools of nutrients and water, provide critical microbial habitat between annual cropping seasons, and increase SOM. *Leucaena* and *Gliricidia* are two common leguminous alley crop species. *Leucaena* intercropped with sorghum increased sorghum yields by 73%, as compared to sorghum grown without N fertilizer, and yields were 43% greater than with a low rate N fertilizer application (Ghosh et al., 2007). Alley cropped legumes can fix between 200 and 300 kg N/ha per year (Giller, 2001). Some of the challenges in the adoption of alley cropping systems include the competition of the legume with the cash crop for moisture in dry years, the pruning labor required, and the use of land for a noncash crop. Selection of species that have complementary rooting systems with cash crops (i.e., a deep-rooted perennial legume cropped with a shallow-rooted annual), and species that grow at a manageable pace to supply N while not requiring excessive pruning inputs, are important considerations in the selection of legume species for alley cropping.

Lastly, while reliable data on the contributions of nonsymbiotic diazotrophs (free-living and those found in the rhizosphere) is limited, there are circumstances where it may be possible to increase N fixed by these microbes. Management practices that affect the availability of soil carbon should significantly impact the potential for BNF. For example, the retention of the carbon in straw from a wheat crop with a yield of 2 t/ha could theoretically fuel the production of 50–150 kg N/ha if utilized by diazotrophs to drive N fixation (Kennedy and Islam, 2001). In addition, crop selection and breeding can affect BNF potential, because plant species differ greatly in the quantity and quality of root exudates produced.

Agroecosystem-Scale Nutrient Use Efficiency

A central theme of any fertility management regime is the idea of evaluating the efficiency of nutrient inputs. In our experience, understanding and promoting nutrient efficiency is the key concern of resource-constrained farmers. It is much more important than determining the rate of nutrient application that will maximize agronomic return. This is because smallholder farmers with very limited assets need to optimize returns to their modest investments, rather than optimizing profitability per se. An efficiency approach is a different way to think about nutrient management compared to the majority of soil fertility management research and fertilizer recommendations developed around the globe, which focus on optimizing plant yields. When NUE is considered within industrial agriculture management regimes, it is usually measured as yield per nutrient input from fertilizer, i.e., kg maize/kg fertilizer N. In other words, the efficiency of a nutrient source is evaluated based on the estimated contribution to yield for a single growing season.

There are several drawbacks to this approach. First and foremost, the focus on the single process of plant assimilation of the nutrient input leaves out many processes that retain nutrients for crop use in subsequent years and are beneficial for long-term improvement of soil fertility. Furthermore, this metric is limited to a single growing season, so the fate of these fertilizer inputs over longer time frames is not factored into the assessments of NUE. You can see that reliance on this metric as an indicator of NUE leads to management decisions that are driven solely by consideration of immediate yield outcomes while more complex outcomes such as longer-term benefits to soil fertility or retention of nutrients in SOM do not factor into nutrient management strategies. An additional consequence is that organic amendments such as green manures or composts that contribute to building SOM are judged to be inefficient nutrient sources, and therefore inferior to inorganic, soluble fertilizers. One consequence of the wide application of this single NUE metric to drive nutrient management decisions is that farmers find themselves on a “fertilizer treadmill,” where their farming systems have become dependent on high inputs of soluble fertilizers simply to maintain acceptable yields (Drinkwater and Snapp, 2007a,b).

A more comprehensive, ecologically based model for NUE assessment takes into account diverse nutrient fates over a longer time scale than a single growing season. From this holistic perspective, NUE is defined in terms of the retention of nutrients within the agroecosystem, usually at either the field or farm-scale, in conjunction with plant production related outcomes. Therefore, we distinguish between crop-scale NUE and agroecosystem-scale NUE. Crop-scale NUE, or yield/fertilizer input, is certainly one useful measure to consider in the context of nutrient management decisions, however, use of this metric cannot support integrated management. Agroecosystem-scale NUE can be estimated using the simple input–output mass balance approach we discussed earlier. This requires information on rotation, fertility inputs, and crop yields for at least one rotation cycle. Clearly, there are many sources of error in these simple budgets, however, we have found them to be a useful starting point for developing strategies to improve nutrient management in a variety of agroecosystems. In the future, it may be possible to use natural isotopic ratios of $^{15}\text{N}/^{14}\text{N}$ as an indicator of agroecosystem-scale NUE. While NUE is a useful concept, it should only be used as one of the many factors that contribute to the development of field-specific nutrient management planning.

Integrating Nutrient Management With Other Farming System Decisions

In addition to the processes which are directly linked to nutrient cycling, nutrient management practices have cascading effects on other agroecosystem processes, making it advantageous to integrate nutrient management

BOX 7.3 The Goat Dilemma: How Should Revenues From the Sale of a Goat Be Used?

A farmer sells her goat at the start of the planting season. Should she: (1) use the proceeds to buy fertilizer to apply at the recommended rate of 45 or more kg N/ha, which has been shown to be profitably applied to a maize crop? Or (2) should she use the proceeds to apply a moderate dose such as 17 kg N fertilizer per ha, and apply this over a larger area? She also needs to consider if she can afford to apply fertilizer and hire extra labor to weed the crop intensively. Her decisions need to take into consideration the value of concentrating the fertilizer in fields where she usually obtains high yields, versus a strategy that includes application of the fertilizer to low yield potential fields that might help enhance the yield output from the entire farm.

planning with tillage, pest management, marketing, and livelihood goals. A farmer perspective on the decision of how to best manage a fertilizer source use is illustrated by the “what to do with a goat’s worth of proceeds” dilemma described in text [Box 7.3](#). The question facing many smallholder farmers is how to optimize returns from the modest proceeds raised by selling one goat. Should this be invested in fertilizer, in improved seed, in hiring labor to carry out extra weeding, or in some combination of these strategies? Trade-offs need to be considered. Is it worthwhile to invest in fertilizer for parts of the farm where an extra weeding operation cannot be undertaken, due to labor or financial constraints? Integrated nutrient management occurs within the context of investment decisions such as these, which are made on a whole farm basis. This further complicates farmer decision-making, as an investment in fertilizer or compost at high rates in one field may preclude nutrient investment in other fields. An on-going question is the extent to which returns can be enhanced through targeting fertilizer to the highest performing fields, or through spreading fertilizer throughout a farm to obtain the high efficiency possible at low rates of fertilizer.

The interaction among these allocation decisions was studied using simulation modeling and on-farm research in southern Africa to evaluate combinations of weeding intensity and N fertilizer rates ([Dimes et al., 2001](#)). In these systems, N was the limiting nutrient, and therefore N fertilizer additions should have increased maize yields. However, yield increases were not achieved unless an extra weeding was carried out in fields receiving N fertilizer. For these site-specific management decisions, the most promising strategy is expected to vary, depending on the heterogeneity of resources across a farm, and the background rate of fertility, e.g., what production is obtainable without fertilizer, based on a minimal investment in planting, weeding, and harvest. To illustrate how allocation of resources to fertilizer applications

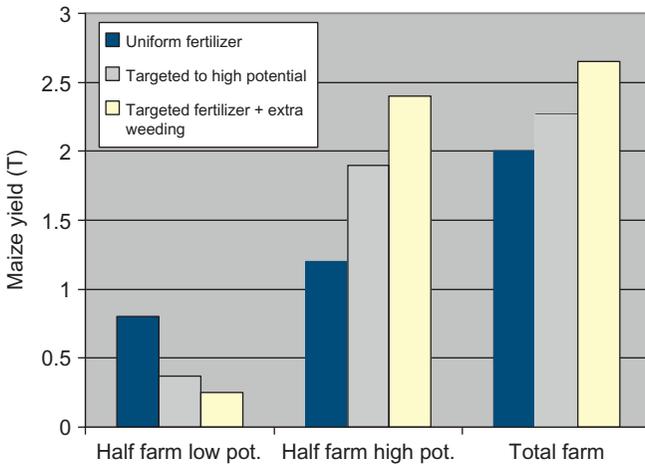


FIGURE 7.15 Effect of fertilizer and weed management decisions on total farm maize yield. Three possible scenarios are presented for investment in inputs by a smallholder farmer across a hypothetical farm, where half of the maize production area has low potential productivity (0.5 ha of 0.5 T/ha potential maize grain yield without inputs), and the other half has high potential productivity (fourfold higher yield potential without inputs: 0.5 ha with 2.0 T/ha yields). Maize production outcomes are presented for the two halves of the farm and on a total farm basis, for scenario (1) N fertilizer applied uniformly (solid blue bars), (2) N fertilizer targeted to the field with greater yield potential, and (3) a reduced amount of fertilizer combined with extra weeding, both targeted to the field with greater yield potential. The overall financial investment remains the same for all three scenarios.

and labor for weeding interacts with the inherent productivity at the farm-scale, we have compared the impact of three different management scenarios on maize yields (Fig. 7.15). Scenarios of targeted and homogenous application are explored for a farm with two maize production fields that vary in yield potential, one being low (0.5 t/ha without fertilizer), and the other high (2.0 t/ha without fertilizer). Uniform application of a 25 kg of fertilizer per ha rate across the farm lead to the lowest yield potential overall, although the poor yield potential site had a higher yield than in other scenarios. Scenario two targeted a higher dose of fertilizer to the high yield potential site, combined with a lower rate at the low yield potential site, and had a significant positive effect on the overall production of maize grain from the farm. Trading-off some fertilizer for an extra weeding, which is again targeted to the higher potential site, produced the largest maize yield overall for the same level of investment across the farm. Notice that, in this third scenario where fertilizer resources and weeding efforts are directed toward the more productive half of the farm, yields in the other half with poorer soils are exceedingly low.

The take home message from this example is that trade-offs occur across a farm, and the outcomes of management decisions will vary, depending on

the particular situation on that farm. Yield from the low potential fields on a smallholder farm may be at such a low level that the grain produced and response to input is minimal, and is not able to significantly influence overall productivity of the farm. Abandoning part of the farm as a minimal investment site, and intensifying production on higher potential sites, may be a useful strategy in some circumstances. If input resources are limited, e.g., farmers may not be in a position to apply all of the inputs that economic returns would justify. It is important to take into consideration the background level of fertility, the interactions of fertility and other inputs at different sites across the farm, and overall, the response of staple grains to complementary investments over the short and long-term, including weeding and SOM building practices.

DEVELOPING SITE-SPECIFIC ECOLOGICAL NUTRIENT MANAGEMENT SYSTEMS

Clarify Goals of Nutrient Management

A first step in managing the nutrient cycling to support agricultural goals is to identify nutrient management goals for your agroecosystem. What are the yield and fertility objectives? What is the relative balance between fertility and food or nutritional goals? Is there a perceived problem that needs to be addressed? Initially, the goals do not need to be prioritized or evaluated for whether or not they can be reasonably achieved. Refinement of goals will be easier after a concept map is developed.

Concept Map of Nutrient Flows

Drawing a conceptual diagram of nutrient flows, compartments, and processes regulating those flows, similar to some of the diagrams we have used in this chapter, can be a useful exercise. The use of conceptual models as communication and planning tools has proven to be a useful tool for facilitating communication and planning in groups with diverse perspectives (Heemskerk et al., 2003). A conceptual model is a visual representation of the system to be studied. Conceptual models are particularly useful in planning interdisciplinary agricultural systems research, because they require the team to graphically represent the problem to be addressed within a larger, systems context. Ideally, to be useful as a planning tool, a conceptual model should:

- Describe a system that encompasses the research questions/management issues, but has clear boundaries;
- Explicitly define the components of the system and how they interact with one another. For example, it should identify the factors that directly or indirectly contribute to production, environmental outcomes, or nutrient flows;

- Provide a logical framework for the problems or questions to be addressed;
- Be simple enough to be understood by scientists from a variety of disciplines and stakeholders;
- Be developed and agreed upon by all stakeholders and researchers.

Diagrams of agroecosystem nutrient flows can serve as an vehicle for achieving several outcomes which are prerequisites for successful implementation of ecologically based nutrient management. This process is important for:

1. *Facilitating information exchange*: Ensures that farmers and researchers have an agreed-upon understanding of nutrient management practices, while also helping scientists to share information about important soil processes that control nutrient availability with farmers.
2. *Organizing a complex system*: By laying out the relationships among the interacting processes that are occurring at different spatial and temporal scales, trade-offs and linkages between management strategies become apparent.
3. *Moving the local nutrient cycling knowledge system forward*: The process of agreeing upon a diagram that represents diverse perspectives helps to identify knowledge gaps, while also promoting the incorporation of innovations and new knowledge into the shared knowledge structure.

Resource Inventory

As part of the information gathering stage, it is important to define the agroecosystem characteristics that provide the backdrop for nutrient management decisions. These include:

1. *Background environment*: Soil types, soil fertility status, climate;
2. *Cropping system characteristics*: Crops that are grown, rotation, and proportion of land that is usually in each crop, relationship between crops, forage, and animal production, field sizes, locations, management intensity;
3. *Nutrient input sources*: Identify the sources of nutrients that are locally available, and constraints which impact their use by farmers;
4. *Relationship to other management practices*: How do other management issues such as weed control, and tillage systems impact nutrient management?
5. *Fate of crops*: Important to distinguish between crops grown for family consumption and those aimed at markets, identity of markets, relative value of cash crops.

There are numerous resources available outlining methods that can be used in characterizing agroecosystems and in problem diagnosis (i.e., [Gonsalves et al., 2005](#)).

Revisit and Refine Goals

With the above information in hand, it will be possible to prioritize, evaluate trade-offs, and identify which goals are easily achievable. At this point, a useful step might be to distinguish between long-term and short-term goals. If a farmer-identified problem is the catalyst for this evaluation, then the range of possible solutions should be evaluated using the conceptual diagram and information that has been gathered.

Quick Assessment of Consequences of Current Nutrient Management Practices

Before moving forward to develop nutrient management strategies to achieve the goals (or solve the problems) which have been identified, prioritized, and analyzed, construction of simple input–output balances is a further step that can be used to analyze the current management. This approach has proven useful in pin-pointing the most important weaknesses in nutrient management systems which are currently being used by farmers. In the United States, application of this tool indicated that organic vegetable growers were over-applying compost, leading to environmentally unsound levels of soil P. In Andean systems, this approach demonstrated that P management practices in fields closer to the community provided sufficient P, and were compatible with increased use of legumes for N fixation, while fields that were farther from communities did not receive adequate P to benefit from legume intensification (Vanek and Drinkwater, 2013). Further study of these systems revealed that potassium was being extracted at rates that far exceeded inputs, indicating that over the long-term, potassium limitations may reduce yields. An additional example is the resource allocation maps (RAMS) which are specifically designed to track nutrient flows at the farm or community scale, where transfers across fields, rangeland, and corrals are important (Box 7.4, Defoer, 2002). Readers should visit the website for this textbook for updates on tools which are being developed to facilitate the use of nutrient budgeting in developing management strategies.

Selecting and Testing Promising Nutrient Management Practices

Using this iterative process, a collaborative team can colearn with farmers regarding which management strategies are worthy of further testing and research. There is no single process that should be used in making these decisions, however, if a number of competing practices are identified, a simple method for comparing and contrasting these practices is to list the strengths and weaknesses of each. Also, the relationships between practices should be considered. Once you have agreement from farmers and other

BOX 7.4 Mapping Farm and Community Scale Nutrient Flows

Participatory research approaches have illustrated that farmer resource management can be improved through maps of agroecosystem nutrient resource flow, also called RAMS (Defoer, 2002). Farmers and researchers together develop the maps and use them to record, monitor, and analyze data and decision-making, which enhances understanding of soil fertility status, nutrient transfers, and degree of recycling associated with management options. Information gathered in this way is of value at different levels. This includes local participants who may be able to better assess where losses are potentially high on their farm, and thus where opportunities to recycle should be concentrated to improve overall nutrient efficiency. The RAMS approach illustrates the exciting potential of approaches that act as an interface between a “hard system” of knowledge (resource flow budgeting which can be used for modeling and comparisons with other systems), and a “soft system,” integrating knowledge gained from collaborating with farmers and improving understanding of farmer perception of losses, gains, and transformations within and across a farm. Participatory research that integrates qualitative and quantitative approaches may provide new insights into designing sustainable agricultural systems that are not only efficient from a bio-engineering perspective, but also are relevant to real world farmers.

At a community or small watershed scale, resource mapping is also being pursued as a means to enhance understanding and recycling of resources on a larger scale than the farm. In Nicaragua, e.g., participatory microwatershed studies were initiated through community meetings of stakeholders, where resource mapping, transect analysis, and indicator-based assessment was used to evaluate current status and opportunities for improvement.

Livestock-crop integrated systems are ideal ways to concentrate and transfer nutrients, as animal manure is collected by corraling animals at night, and during the day pasturing them over a wide area. A cow pastured on four hectares can provide sufficient nutrients to support half a hectare of nutrient-demanding crops such as maize. Thus, livestock transforms a widely spread, relatively unavailable nutrient source from wild or semiimproved pastures, or even urban streets, and concentrates these nutrients as manure, which can be targeted to specific crops. Transhumance, nomadic livestock systems that move through field crop areas and trade residue grazing for transient manure deposition, were once one of the most common land use systems in the world.

stakeholders about which practices are of the most interest, you can design research trials to evaluate and optimize these practices. This research should be conducted in farmer’s fields as much as possible, using participatory experimental designs such as the mother–baby scheme (Snapp et al., 2002). To succeed, research aimed at supporting ecological nutrient management must be conducted within a systems-context, and must apply participatory methodologies.

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