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Soil Nitrogen Budgets

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Nitrogen budgets, or N balances, are a valuable tool for expanding our understanding of the soil N cycle. Nitrogen budgets have been used to estimate the size of various N pools, N gains from the atmosphere, N losses to the environment, and to study the interactions among soil N cycle processes. Nitrogen budgets have also been used to compare the effects of management practices on the soil–crop N cycle. A major advantage of constructing an N budget is that it requires a "systems approach", i.e., the identification and estimation of the major N cycle processes in a system. Constructing N budgets requires a synthesis of the individual N processes, described throughout this monograph, into a coherent view of the entire cycle. High-quality N budgets should lead to more efficient use of N, and lower environmental losses.

Constructing a nutrient budget for N is particularly difficult because N exists in oxidation states from +5 (NO_3), to 0 (N_2), to −3 (NH_4). Nitrogen is also transformed by diverse agents, including microbes, chemical reactions, plants, and animals. Furthermore, it can be transported between compartments by air and water. Nitrogen exists in compounds with a wide range of stabilities, from long-lived soil organic matter to short-lived urea, as inorganic forms with slow reactivity (clay-fixed ammonium) to rapid reactivity (solution ammonium N), and as a nearly inert gas (N_2) to a highly reactive gas (NH_3). These facts explain why N budgets have challenged many generations of soil scientists, and why N budgets remain a challenge today.

The goals of this chapter are to review, analyze, and interpret selected N budgets, and the components making up budgets, to illustrate different approaches for constructing budgets and to examine the soil N cycle in various agricultural systems. This chapter is not a comprehensive literature review, but will emphasize N balances on the field-plot scale, and larger spatial scales.

Table 13–1. Nitrogen balance sheet for four 19th century rotations in Alsace, France, as reported by Boussingault (1841, 1843).

Crop rotation description†	Manure N input‡	Total crop N outputs	Difference, over entire rotation	Average difference, annually
	kg N ha^{-1}, entire rotation			kg N ha^{-1} yr^{-1}
Fallow–wheat–wheat	83	87	+ 4	nil
Pot.–Wht.–Clover–Wht./turnip–Oats	203	251	+ 48	+ 10
Pot.–Wht.–Cl.–Wht./turnip–Peas–Rye	244	354	+ 110	+ 18
Luc.–Luc.–Luc.–Luc.–Luc.–Wheat	224	1078	+ 854	+ 142

† Pot. = Potato (*Solanum tuberosum* L.), Wht. = wheat, Cl. = clover, turnip = turnip (*Brassica rapa* L.) cover crop, oat = oat (*Avena sativa* L.), rye = rye (*Secale* cereale L.) Luc. = lucerne (alfalfa).

‡ The manure had a C/N = 18.

Previous Soil Nitrogen Budget Studies and Summaries

Early Soil Nitrogen Budgets

Nitrogen budgets have been studied for over 170 yr. The early budgets were designed to estimate N transfers between the atmosphere and the soil–plant system. Although some readers may consider early N budgets to be only of historical interest, one only needs to recall that all the soil N transformations that are active today were also active during the early N investigations: what has changed are our research methods and the accumulated knowledge of the individual N cycle processes.

Boussingault

Boussingault (1841) was the first to employ an N balance sheet for field plots, when he evaluated crop rotations on his Bechelbronn farm in Alsace, France (Aulie, 1970; McCosh, 1984). His studies utilized rotations that had been practiced for many decades on the same fields and his balance sheets covered an entire rotation cycle, which is equivalent to assuming a steady-state soil N condition over the course of the rotation (see later discussion). His studies began in about 1836 (Aulie, 1970) when he evaluated multiyear rotations by analyzing manure N inputs, N outputs in crops, and assembled the data into a balance sheet to see if crop N requirements could be met by manure, or if other N inputs were involved. He analyzed the manures and crops for N, C, H, and O utilizing the newly developed combustion method of his colleague Dumas (1834). Boussingault's 1841 data are summarized in Table 13–1 and show that: (i) rotations without legumes depended on N inputs from manure to maintain soil fertility, (ii) including a crop of clover (*Trifolium pratense* L.) in the rotation resulted in an added N output of about 48 kg N ha^{-1} over the rotation, and (iii) utilizing both clover and peas (*Pisum sativum* L.) in a rotation added about 110 kg N ha^{-1}. However, his most striking result was the large N yield from 5 yr of lucerne, i.e., alfalfa (*Medicago sativa* L.), followed by wheat (*Triticum aestivum* L.), which removed over 850 kg N ha^{-1} over the rotation, translating into an average annual input of about 140 kg N ha^{-1}. It is noteworthy that these estimates of N_2 fixation are consistent with modern estimates derived from much more sophisticated techniques (Russelle, 2008, see Chapter 9). Boussingault (1841, 1843) hypothesized that the source of the additional legume

N was nonammoniacal atmospheric N, but the final identification of this N source had to wait until later in the 19th century with the development of microbiology. Boussingault's rotation data also led him to the conclusion that N was the primary nutrient needed by crops in a rotation, with secondary needs for the mineral salts of P and K (Boussingault, 1841).

Boussingault utilized N budgets throughout his career and applied them to pot studies and animal nutrition studies (Aulie, 1970). In later studies, Boussingault (1855) and Lawes et al. (1861) constructed N budgets for enclosed potted plants to see if grasses or legumes could directly utilize the N_2 gas in "the great sea of nitrogen—the atmosphere" (McCosh, 1984). Both investigators concluded that neither grasses nor legumes could directly utilize atmospheric N_2 when grown in sterile conditions. Yet, earlier field trials had clearly shown that legumes were high in N and had contributed to higher yields of succeeding cereal crops (Boussingault, 1841, 1843). The solution to this enigma had to wait until later in the 19th century when Hellriegel and Wilfarth (1888) and Schloesing and Laurent (1892) showed substantial N gains from nodulated legumes that came from the fixation of N_2 by bacteria in the root nodules. Thus, these early scientists were challenged by gaps in their N budgets that shaped ensuing studies and expanded our understanding of the agricultural N cycle.

Lawes and Gilbert

The second field-plot N budget arose from studies begun in 1843 at Rothamsted, United Kingdom (Lawes and Gilbert, 1864, 1884, 1885; Lawes et al., 1882) with the initial objective of testing Leibig's Mineral Theory of Plant Nutrition (Leibig, 1840). By 1852 sufficient data had been collected to disprove the Mineral Theory and the treatments were modified, transforming the study into the long-term Broadbalk Winter Wheat Experiment. Many of the treatments in this experiment have continued for over 160 yr (Johnston and Garner, 1969; Dyke et al., 1983; Rothamsted Research, 2006, p. 52). The Broadbalk Experiment is unreplicated, but has treatments applied in long strips (6 by 300 m) on a gentle slope with a central tile drain running the length of each strip. Wheat has been grown annually on at least part of the field (except for occasional fallow years), with both grain and straw removed annually. The soil is mapped as a Batcombe Series (U.S. classification, Aquic Paleudalf; FAO (1990) classification, Chromic or Vertic Luvisol) and has a surface texture of clay loam to silty clay loam (19–33% clay in plots considered herein), overlying a subsoil containing over 50% clay (Avery and Catt, 1995). The fine-textured soil is free draining due to a dual-pore water transport system, with about 20% of the average annual percolate of 245 mm moving downward through large-pore rapid-flow pathways, and about 80% via slower small-pore drainage paths (Goulding et al., 2000). The soil was heavily chalked in the 18th century and early 19th century (Powlson et al., 1986a; Jenkinson, 1991). Some plots still retain free calcium carbonate and the others have received lime to maintain a pH of 7.5 to 8.0. Broadbalk has been arable, i.e., tilled, since at least 1623 and was probably first cultivated in Roman times—the foundations of a Roman cemetery lie less than 100 m away.

The total N content of the Broadbalk soil in both the control plot and in the inorganically fertilized plot receiving 144 kg N ha^{-1} yr^{-1} has changed little since about 1865 (see later discussion in the section "Steady-State, or Equilibrium, Soil Nitrogen Concept"), the date of the first comprehensive soil sampling. The original Broadbalk N budget constructed by Lawes et al. (1882) for 1879 and 1880 documented N inputs and N removals in grain plus straw (Table 13–2), as well as estimated drainage losses from a comprehensive sampling of the tile drains. The most prominent feature of Table 13–2 is the unexplained input of 29 kg N ha^{-1} yr^{-1} to the plot receiving no fertilizer N. Lawes et al. (1882) originally attributed this to a decline in total soil N, an explanation based on soil bulk density measurements that were later found to be incorrect (Dyer, 1902). It is now accepted (see Jenkinson, 1977) that there has been little or no change in the total N contents of any of the plots in Table 13–2 over the period 1865 to 2000. We are still uncertain about the source of the additional N input on the control plot that has produced regular crop N outputs year after year—although various hypotheses will be considered in the section "Approaches for Extending Labeled Nitrogen Budgets to Total N Budgets".

The 1879–1880 Broadbalk N budget in Table 13–2 contains other noteworthy features. The overall mass of N lost from the system increased with additions of fertilizer N, especially at the highest N rate (Table 13–2). The losses needed to balance the budget can most likely be attributed to gaseous outputs, probably dominated by denitrification losses of N_2 and N_2O, with a possible additional loss of NH_3 from the ammonium sulfate that was then used as a N fertilizer on this calcareous soil (Francis et al., 2008, see Chapter 8). Quantitatively partitioning the gaseous losses into denitrification and ammonia emissions remains a challenge to this day, although denitrification is generally considered to be the largest loss mechanism in this fine-textured soil.

Another significant feature of the early Broadbalk N budget is the percentage recovery of fertilizer N, as estimated for the different N rates by subtracting the control plot N removals from the fertilized plot N removals, and dividing by the fertilizer N rate. This approach (the traditional difference method) produces similar crop fertilizer N recoveries of 24 to 27% for the three N rates (Table 13–2). The difference method tacitly assumes that the N inputs in the control plot are similar to those in the fertilized plot. However, it is difficult to determine if this assumption is correct. For instance, do the plots receiving fertilizer N gain as much N from the unidentified inputs as the controls, e.g., from a relatively uniform input such as atmospheric deposition? Or do the N-stressed plots contribute to more undocumented N inputs than the N-sufficient plots through an input such as algal N_2 fixation, as suggested by Witty et al. (1979)?

The difference-method approach can also be used to estimate the apparent drainage loss of fertilizer N, again assuming that the fertilized plots leach the same quantity of nonfertilizer N as the control. Thus calculated, the plot receiving 49 or 99 kg N ha^{-1} yr^{-1} lost 12 to 13% to leaching, while the plot receiving 148 kg N ha^{-1} lost 20% (Table 13–2). These findings by Lawes and Gilbert, have been confirmed much more recently by Goulding et al. (2000), who showed that leaching losses do not increase much until the capacity of the crop to take up N is exceeded.

Previous Nitrogen Budget Reviews

Two classic reviews of the soil N budgets were published by Allison in 1955 and 1966. The first focused on N processes and drew on lysimeter experiments, greenhouse data, and field-plot experiments. These N balance studies typically emphasized crop uptake, leaching, runoff, changes in soil organic N, and an "unaccounted for" term (by difference) that represented total gaseous losses. Allison

Table 13–2. Nitrogen budgets (kg N ha^{-1} yr^{-1}) from 1879 to 1880 for N rate treatments of the Broadbalk Continuous Winter Wheat Experiment (Lawes et al., 1882). Fate of N, as a percentage, calculated for fertilizer N using the control plot as a reference†.

N budget component	Without fertilizer N; with P, K, and Mg‡ (Plot 05)	With fertilizer N; and P, K, and Mg ‡					
		49 kg N ha^{-1} in spring (Plot 06)		99 kg N ha^{-1} in spring (Plot 07)		148 kg N ha^{-1} in spring (Plot 08)	
	kg N ha^{-1} yr^{-1}						
N inputs							
Soil N Change	0	0		0		0	
Seed	3	3		3		3	
Rain (wet dep.)	5	5		5		5	
Fertilizer N	0	49		99		148	
N input sum	8	57		107		156	
N outputs			FNF†		FNF†		FNF†
Grain and straw	18	30	24%	45	27%	55	25%
Drainage N lost §	19	25	12%	32	13%	48	20%
To balance¶	+ 29	–2		–30		–53	

† FNF is fertilizer N fate, which is the N output value substracting the Plot 05 output value, and dividing by fertilizer rate (difference method).

‡ Annual mineral inputs (kg ha^{-1}): P, 35; K, 90; Mg, 11; and Na, 16 (Johnson and Garner, 1969).

§ Mean annual drainage, spring 1879 to spring 1881.

¶ + = a N input, – = a N loss.

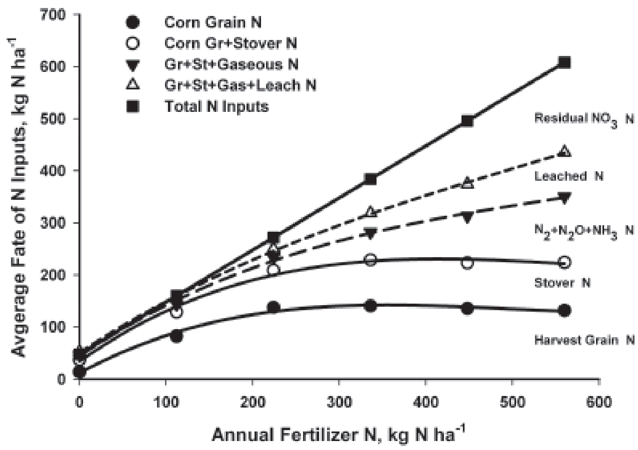

Fig. 13–1. Average yearly distribution of total N inputs for irrigated corn grown on a sand loam soil at Kearney, CA, in 1973–1975. The N balance was solved for excess inorganic N, which was derived from fertilizer N additions, plus natural N inputs from rainfall and irrigation water (8 kg N ha^{-1}, Tanji et al., 1977), plus N from mineralization of soil organic N and crop residues (Broadbent and Carlton, 1978) using an annual mineralization rate of 4%. Nitrogen outputs that were subtracted were N in harvested corn grain (Broadbent and Carlton, 1979), total gaseous N losses of 22% of the inorganic N (Fig. 5, Legg and Meisinger, 1982, p. 535), and leaching losses of 14% of the inorganic N estimated from the water-input leaching fraction (Tanji et al. (1977, 1979). The areas identified within the figure estimate the fate of the total N inputs to the designated N process. This is an update of the previous Fig. 7 of Legg and Meisinger (1982, p. 554), which contains a detailed description of the estimation techniques used in summarizing the original data of Broadbent and Carlton (1978, 1979) and Tanji et al. (1977, 1979).

(1955) concluded that most N balance studies failed to balance, a situation that popularized the phrase "N enigma", because 10 to 20% of the added N was commonly unaccounted for. However, direct estimates of denitrification and ammonia volatilization were lacking in these budgets. In the later review, Allison (1966) noted that marked progress had been made in ascertaining the fate of applied N, which he attributed to the use of ^{15}N, improved instruments, and new techniques for direct measurement of various N loss processes. Allison's 1966 review focused on N loss processes, especially chemical and biological gaseous losses. Allison summarized the 1966 review by noting that crop N uptake commonly amounted to about 50% of the added N, with gaseous losses and leaching losses (in humid regions) accounting for the remainder.

In 1982 Legg and Meisinger reviewed soil N budget research, focusing on N losses from experimental plots. They reported N balances from various cropping systems, such as corn (*Zea mays* L.), small grains, rice (*Oryza sativa* L.), grassland, and forest systems. Their summary noted a consistent, although highly variable, loss of N to the gaseous pathways of denitrification and/or ammonia volatilization. They also summarized a classic total N balance study for irrigated corn on a Hanford sandy loam (Typic Xerorthent) at Kearney, CA, from data reported by Broadbent and Carlton (1978, 1979) and by Tanji et al. (1977, 1979). An updated synopsis derived from the original figure of Legg and Meisinger (1982, p. 554) is given in Fig. 13–1, which summarizes the 3-yr total-N input budget on an annual basis. A nonsteady-state condition (see later discussion in "Steady-State, or Equilibrium, Soil Nitrogen Concept") was utilized to allow for the significant accumulations of inorganic N at the high N rates and an accumulation of soil organic N in the coarse-textured soil that contained only about 200 mg N kg^{-1} soil at the start of the study. The Kearney data in Fig. 13–1 illustrate an important N balance principle: in the N responsive part of the yield curve, below 224 kg N ha^{-1}, the crop N utilization is rather efficient (75–80% recovery of total N inputs in grain plus stover) if the crop is fertilized in phase with N demand with the fertilizer applied below the soil surface. This is shown by the close agreement between the total N input and crop N uptake curves below 224 kg N ha^{-1} (see Fig. 13–1). An important corollary to this principle is that N losses increase rapidly once N inputs exceed crop assimilation capacity, as shown by the increasing areas in Fig. 13–1 above 224 kg N ha^{-1} that are attributed to denitrification, leaching, or an accumulation of residual NO_3 in this subhumid climate. In the excess N range above 224 kg N ha^{-1}, the recovery of total N inputs in the grain plus stover declined steadily from 75% at 224 kg N ha^{-1} to 59, 45, and 37%, respectively; for fertilizer rates of 336, 448, and 560 kg N ha^{-1}. Legg and Meisinger (1982) also concluded that N budgets are highly variable from year-to-year, from soil-to-soil, and from crop-to-crop, because the final budget is the result of biological, chemical, and physical processes that continuously interact with each other over time.

A recent survey of N balances, compiled by Mosier et al. (2004), provides a good description of contemporary N cycling in agriculture using a global perspective. They examined agricultural systems in developed and developing countries, in high- and low-N input systems, and considered the social, economic, and environmental aspects of N.

Nitrogen Budget Principles

Nitrogen budgets seek to summarize the complex agricultural N cycle by documenting the major flow paths of N as it enters and emerges from various N pools. The large-scale N cycle also contains smaller N cycles, for example N is continuously cycled between mineral and organic forms as part of the mineralization–immobilization turnover (MIT) process that has been described by Jansson and Persson (1982). Likewise, N_2 gas can be cycled into plant or microbial protein through biological N_2 fixation, only to return quickly to N_2 when some of the legume residues undergo denitrification. Nitrogen budgets spanning long time intervals, e.g., many years, emphasize the large-scale N cycling while short time-interval budgets, e.g., several months or a growing season, emphasize the smaller-scale N cycles nested within the larger agricultural N cycle.

Conservation of Mass

Soil N budgets are based on the principle of conservation of mass, which simply states that N inputs minus the N outputs equals the change in N stored within the system. However, this deceptively simple statement requires thoughtful definition of the N budget goals, careful definition of the system boundaries in space and time, and appropriate estimates of the N flows that cross system boundaries. The general mass–balance equation for a soil–crop system defined in space and time is:

$$N_{inputs} - N_{outputs} = \text{Change in Soil N Storage } (\Delta N_{soil}) \quad [1]$$

The first step in constructing an N budget is a clear statement of goals. Nitrogen budgets have been used to estimate major N processes that are not easily measured (e.g., denitrification), to identify knowledge gaps (e.g., Boussingault's unexplained N inputs from N_2 fixation), or to study the affect of fertilization practices on soil N pools and N losses (e.g., the N budgets of Lawes and Gilbert).

The second step is a clear definition of the conceptual boundaries in space (for example a field plot in three dimensions, farm fields including/excluding adjacent ecosystems, or a watershed) and time (for example a single growing season, a calendar year, or several rotation cycles). Defining the boundaries is essential for developing an N budget because these elements define the "system." The system boundaries, in turn, determine the N flow paths that must be documented to construct the budget, so that flows crossing system boundaries are included in the budget. Meisinger and Randall (1991) have discussed system boundaries in detail for several agricultural systems. Meisinger (1984) noted that a major division between N recommendation systems based on N balances is the "whole crop" vs. the "aboveground crop" approach, with the whole-crop system containing the crop root system and a steady-state approximation while the aboveground system does not. Chapter 14 of this monograph (Meisinger et al., 2008) gives an in-depth discussion of whole crop and aboveground crop approaches as related to crop N fertilization.

The third step, of course, is the documentation of the major N_{inputs}, $N_{outputs}$, and ΔN_{soil} to derive an actual budget; one that hopefully narrows unaccounted-for N to one or two pathways. The remainder of this section will discuss estimation of the ΔN_{soil} component because it is an important element in determining if a steady-state approximation is appropriate. Approaches for estimating the N_{inputs} and $N_{outputs}$ components are the most commonly studied aspects of N budgets and are discussed throughout this monograph. They will also be discussed by examining several example N budgets in the section "Applications Of Nitrogen Budgets To Various Spatial And Temporal Scales".

Estimating the Change in Soil Inorganic Nitrogen

Estimating the ΔN_{soil} term involves both the soil inorganic N pool and the organic N pool. The change in soil inorganic N is dominated by the soil NO_3–N pool because soil NH_4–N levels are usually low and change little over time, except after recent additions of NH_4–N fertilizers or manures. Soil NO_3–N is a highly active N pool commonly containing between 30 and 300 kg N ha^{-1} in a 1-m-deep root zone, with the low levels being indicative of N deficiency and the high levels of excessive N (e.g., area in Fig. 13–1 for residual nitrate).

Many N studies (e.g., Broadbent and Carlton, 1978; Bigeniego et al., 1979; Jokela and Randall, 1997) have noted depletion of the soil nitrate N pool by 1st-year crops, especially in the nonfertilized control. This causes control-plot yields to be higher the 1st year than in succeeding years and can cause interpretation difficulties in ^{15}N studies. It is not uncommon for the soil root zone to contain 200 kg NO_3–N ha^{-1} at the beginning of an N budget, and 50 kg NO_3–N ha^{-1} after several years of low N inputs. Other causes for soil NO_3–N depletions are unusually large crop N removals due to favorable weather, or high N losses due to leaching or denitrification in wet years. The reverse case, of NO_3–N accumulation, is also common following drought years or if N inputs substantially exceed crop N removals, especially in subhumid climates as shown in Fig. 13–1 and as documented by Jokela and Randall (1997) and Stevens et al. (2005).

The best approach for estimating the change in soil inorganic N is through direct sampling of the root zone, followed by conventional mineral N analysis. Direct NO_3–N sampling has large coefficients of variation (CVs) commonly 30 to 60% (Meisinger, 1984), with much of the total field variability present within a few square meters (Beckett and Webster, 1971). Brye et al. (2003) reported that soil sampling presented a significant limitation for estimating the inorganic N component of N budgets in agricultural ecosystems, although the Brye et al. (2003) study utilized only two 2-cm diam. cores plot^{-1}. Based on the reported field variability of NO_3–N, Meisinger (1984) estimated that a composite of 10 to 20 cores plot^{-1} would estimate the NO_3–N mean to within about ±20% of the true value on three-fourths of the plots. Thus, accurate estimates of the change in soil NO_3–N requires intense sampling, Hauck et al. (1994, p.930) give detailed suggestions for soil sampling.

Estimating the Change in Soil Organic Nitrogen

The change in the soil organic N pool is more difficult to estimate because it is large, commonly 3000 to 5000 kg N ha^{-1} in the surface soil alone, and is only slowly reactive with 1 to 3% commonly mineralized annually (Bremner, 1965; Broadbent, 1984; Jenkinson, 1977). The slow dynamics of the soil organic N pool means that sampling must be at long intervals, normally 5 to 15 yr, if changes are to be measured accurately. The sampling frequency will depend on climate, cropping system, tillage practices, and soil properties. It is essential to follow consistent sampling and analytical protocols including measurement of soil bulk density, if changes in the soil organic N pool are to be measured accurately. Organic N inputs, such as manure, usually lead to higher soil organic N levels than corresponding nonmanured plots, as illustrated by the manure treatments in the three studies summarized in Fig. 13–2. The Broadbalk plot accumulated about 58 kg N ha^{-1} yr^{-1} over the first 50 yr of the experiment, or about 25% of the annual application of 225 kg N ha^{-1} in manure.

Effects of Tillage on Soil Organic Nitrogen

Tillage practices can affect soil organic N contents, with the modern high surface residue systems conserving more N than conventional clean-tillage systems. For example, Dolan et al. (2006) summarized a Minnesota study growing corn and soybeans [*Glycine max* (L.) Merr.] on a silt loam soil that was sampled to 45 cm after 23 yr of no-tillage or moldboard plow tillage. They found higher soil N in the 0- to 45-cm depth with no-tillage that translated into an average N accumulation of 17 kg N ha^{-1} yr^{-1}. They also noted the importance of sampling the majority of the root zone in tillage studies, because reduced tillage generally results in higher N contents in the surface layers and lower N contents in the subsurface compared with moldboard plow tillage. An Ohio study (Puget and Lal, 2005) compared organic N contents through the 0- to 80-cm depth in a silty clay loam soil after 8 yr of no-tillage or moldboard-plow tillage. They reported no significant effects of tillage on the standing stock of soil N, although no-tillage did exhibit the usual accumulation of N in the surface layers. Changes in soil organic N over time are closely linked to the changes in soil C, because the soil C/N usually remain within the range of 10 to 12. The large volume of literature on soil C sequestration can thus provide insight into the potential for long-term changes in soil organic N as

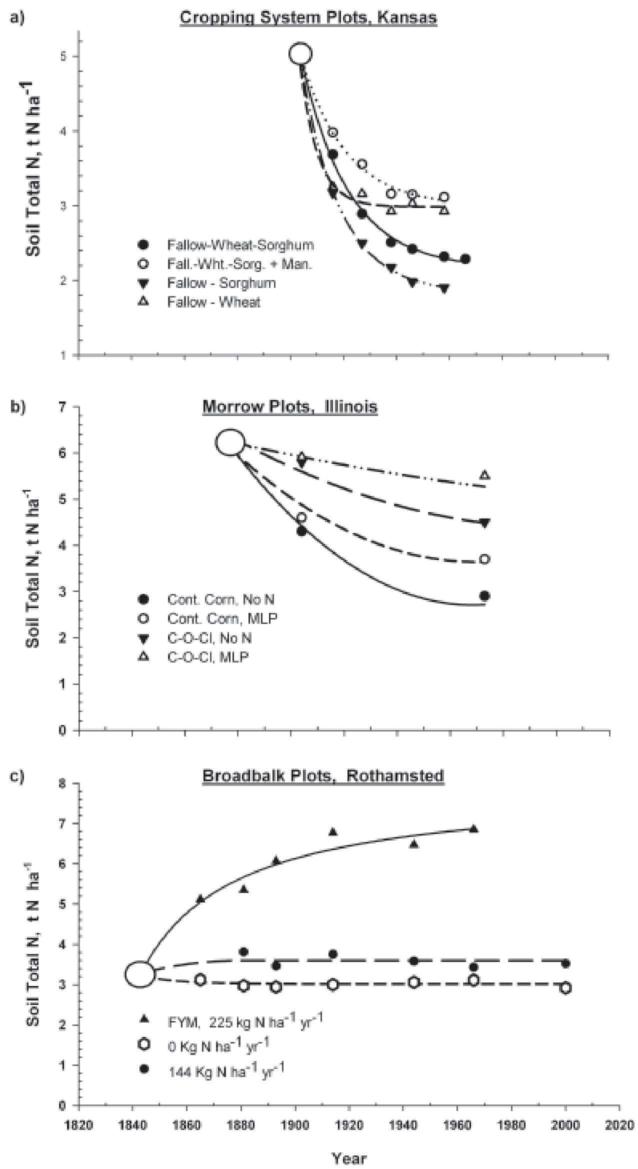

Fig. 13–2. Time sequence of topsoil N content in several long-term studies as affected by cropping system and amendments for the Kansas Cropping System plots (a) from Hobbs and Brown (1965) and Hobbs and Thompson (1971); the Illinois Morrow plots (b) from Illinois Agricultural Experimental Station. (1982); and the Rothamsted Broadbalk Continuous Winter Wheat Experiment (c) from Jenkinson (1977) with 2000 data from Paul Poulton (personal communication, 2005). All panels are plotted against the same horizontal time scale. The MLP amendment of the Morrow plots is manure plus lime plus P, and the FYM of the Broadbalk study is farmyard manure. The large open-circle data point in each panel is the projected soil N at the beginning of each study, as estimated by samples of border areas or informal soil data collected at the start of the study.

affected by management practices. West and Post (2002) summarized 67 long-term C-sequestration experiments across the world containing 267 paired-treatments of various cultural practices. This study estimated that for annually cropped systems (nonfallow systems), a change from conventional-tillage to no-tillage would result in the sequestration of 57 ± 14 g C m^{-2} yr^{-1}, which translates into an N sequestration of about 50 ± 13 kg N ha^{-1} yr^{-1} assuming a C/N of 11. They also estimated that a new plateau level of soil C would be reached in about 15 to 20 yr after converting to reduced tillage. A similar analysis by Puget and Lal (2005) used 56 paired no-till vs. conventional-till experiments and estimated an average C sequestration of 33 g C m^{-2} yr^{-1} with a 95% confidence interval of 5 to 62 g C m^{-2} yr^{-1}. The corresponding annual N sequestration would be about 30 kg N ha^{-1} with a 95% confidence interval of 4 to 56 kg N ha^{-1}. Thus, changes in tillage can result in significant, although quite variable, changes in soil organic N that will take several decades to reach completion. Such changes should be taken into account in N budget studies.

Effects of Cropping System on Soil Organic Nitrogen

Cropping practices can affect soil organic N levels, with rotations that include a forage legume conserving more organic N than continuous cereals. Accordingly, the Morrow Plot soil that supported a cropping system with clover (inverted triangles in Fig. 13–2b) retained an additional 17 kg N ha^{-1} yr^{-1} compared with the continuous corn treatments (circles in Fig. 13–2b). The conversion of cropland to forest also leads to an accumulation of organic N in the soil, as shown by Poulton et al. (2003) who found that old arable land reverting to woodland accumulated an average of 37 kg N ha^{-1} yr^{-1} over a period of 120 yr. Likewise, the conversion of tilled cropland to conservation–reserve grassland resulted in an accumulation of 7 kg N ha^{-1} yr^{-1} (Kucharik et al., 2003). The global data analysis of C sequestration by West and Post (2002) also examined the factors of increasing cropping system intensity, i.e., eliminating fallow periods, increasing the number of crop species in a rotation, or changing from monoculture to rotated cropping (but excluding corn–corn to corn–soybean). They estimated that increasing cropping intensity sequestered 20 ± 12 g C m^{-2} yr^{-1}, which translates into 18 ± 11 kg N ha^{-1} annually. The estimated time for this rotational N and C sequestration to reach completion was 40 to 60 yr (West and Post, 2002). Soil N increases can also occur from the return of greater crop residues, as shown in a 12-yr study by Halvorson et al. (2002) who compared a 2-yr rotation of wheat→fallow with a 2-yr rotation of spring wheat→winter wheat→sunflower (*Helianthus annuus* L.) and reported that the elimination of the fallow year resulted in an average N sequestration of 42 kg N ha^{-1} yr^{-1} in the 0- to 30-cm depth of soil. Small soil N increases can also arise from greater return of crop residues due to fertilization, as shown in Fig. 13–2c for the inorganically fertilized Broadbalk plots compared with the control plot.

The most striking organic N changes in soils often results from the conversion of grassland to cultivated cereals, as shown in Fig. 13–2a (solid circles) for the unfertilized fallow–wheat–sorghum [*Sorghum bicolor* (L.) Moench] rotation that lost an average of 89 kg N ha^{-1} yr^{-1} over the first 25 yr (Hobbs and Brown, 1965; Hobbs and Thompson, 1971). Similar declines are shown in Fig. 13–2b for unfertilized corn in Illinois, with an average decline of about 35 kg N ha^{-1} yr^{-1} over almost 100 yr (Illinois Agricultural Experiment Station, 1982). Conversion of grassland to reduced-tillage cereals gave an average loss of 28 kg N ha^{-1} yr^{-1} in two 9-yr Nebraska studies (Doran and Power, 1983). Diekow et al. (2005) also reported an average annual loss of over 30 kg N ha^{-1} from a sandy clay loam Paleudult soil in Brazil after native tropical grassland was tilled. Cessation of long-term N inputs can also cause soil N to fall, as shown when farmyard manure (FYM) applications ceased on one treatment of the continuous barley (*Hordeum vulgare* L.) experiment at Rothamsted, causing a decline of about 23 kg N ha^{-1} yr^{-1} over the next 41 yr (Jenkinson and Johnson, 1977).

The above summary shows that major changes in land management, such as the tillage of natural grassland, reversion of farmland to woodland, or initiation/cessation of manuring, produce significant changes in soil organic N. These changes should always be taken into account in drawing up N budgets, despite the difficulties of measuring the resulting changes in soil organic N and soil bulk density. However, in many multiyear N budgets the annual changes in organic N are relatively small (e.g., <15 kg N ha^{-1} yr^{-1}) compared with the uncertainties in other N budget components such as N_2 fixation, denitrification, or the change in soil inorganic N, so that approximate estimates can be used without great error.

Steady-State, or Equilibrium, Soil Nitrogen Concept

Ecosystems generally gain or lose N at a diminishing rate until an equilibrium, or steady-state N level is reached (Jenny, 1941), as illustrated in Fig. 13–2a (open triangles) and Fig. 13–2c (circles). Under steady-state conditions, the N mineralized from organic N is equal to organic N returned in aboveground residues, roots, root exudates, and new soil microbial biomass. The mathematical description of the steady-state is that the ΔN_{soil} term of Eq. [1] equals zero, resulting in N inputs equaling N outputs. Exact steady-state conditions rarely occur on an annual basis, because of year-to-year variability in soil N processes due to weather and the slow reaction rates of the soil organic N. Accordingly, the steady-state condition can be viewed as an underlying long-term theme, partially masked by overlying temporal and spatial variations in the soil N cycle. Although exact steady-state conditions are seldom realized in nature, it is often a useful approximation in N budgets, particularly if the ΔN_{soil} term of Eq. [1] is within some acceptably small value.

Examples of Steady-State Soil Nitrogen Contents

The soil total N vs. time relationships shown in Fig. 13–2a can be mathematically analyzed using the straightforward differential equation proposed by Jenny (1941, p.256), $dN/dt = -k_1N + k_2$. In this one-compartment first-order model, N is the soil N content (Mg N ha^{-1}), t is time (yr), k_1 is the first-order rate constant (yr^{-1}), and k_2 the quantity of N that is returned to the soil N annually (Mg N ha^{-1} yr^{-1}). The solution of this differential equation is $N = N_{ss} + (N_o - N_{ss}) \exp(-k_1t)$, for the initial condition that at time = 0, the beginning of the study, the original soil N content is N_o and at steady-state it is N_{ss}, with N_{ss} being mathematically equal to k_2/k_1. This equation was fitted to the Kansas Cropping System data (using the SAS nonlinear regression procedure; SAS Institute, 2001) and adequately summarized the data, with R^2 being >0.99 for each treatment. The fallow–wheat–sorghum system (Fig. 13–2a, solid circles) produced an N_{ss} estimate of 2.2 Mg ha^{-1}, while N_{ss} for manured fallow–wheat–sorghum was 3.0 Mg ha^{-1} (Fig. 13–2a open circles), which illustrates the N conservation effect of manure, also seen in the Morrow plot data (Fig. 13–2b). The N_{ss} for the fallow–wheat system was 2.9 Mg ha^{-1} (Fig. 13–2a, open triangles), while the fallow–sorghum system was only 1.8 Mg ha^{-1} (Fig. 13–2a, closed triangles), a difference attributed to the more frequent cultivations given to the sorghum row crop (Hobbs and Thompson, 1971). Similar soil N declines are evident in all the Morrow plot data (Fig. 13–2b) following breaking of the native prairie in 1876, with the rate of decline affected by rotation and by manure additions. In the Broadbalk Experiment (Fig. 13–2c) the plot receiving manure had more than doubled its N content in 100 yr. The capacity of manures to increase organic matter is illustrated by Jenkinson's (1990) organic matter model, which showed that a five-compartment model could successfully described all the Broadbalk plots in Fig. 13–2c, provided the farmyard manure was modeled as being further along the decomposition pathway than fresh plant residues due to passage through the animal's digestion system. The Broadbalk fertilized plot in Fig. 13–2c (solid circles) transitioned to a modestly higher steady-state N level compared with the unfertilized control (about 0.5 t N ha^{-1} higher), a change attributed to increased crop residues from fertilization (Jenkinson, 1990; Glendining et al., 1996).

Examples of Steady-State Time Prerequisites

The time required to approach steady-state is quite variable, being a function of climate, soil properties, tillage practices, rate and source of N, and cropping system. Many years are often required to approach a quasi steady-state condition, e.g., Fig. 13–2a solid circles, Fig. 13–2b open or closed circles, and Fig. 13–2c solid triangles. In other cases the time is shorter, e.g., Fig. 13–2a open triangles, or Fig. 13–2c solid circles that apparently transitioned to steady-state during the first few decades after 1852 (Jenkinson, 1977).

A convenient estimate of the response time to steady-state can be derived from the k_1 parameter (k_1 defined above) in Jenny's N-vs.-time equation. The time for the change in soil N, $|(N_o - N_{ss})|$, to increase/decrease to one-half of its initial value can be defined as $t_{1/2} = \ln(0.5)/k_1$. The estimated halving-times for the Kansas Cropping systems in Fig. 13–2a are: 12 yr for fallow–wheat–sorghum with or without manure, 10 yr for fallow–sorghum, and 5 yr for fallow–wheat. The differences in $t_{1/2}$'s are likely due to the different tillage practices (Hobbs and Thompson, 1971) with the intensely cultivated sorghum row–crop having longer $t_{1/2}$'s and the largest decline in soil N. In estimating the halving-time it is well to remember that both k_1 and k_2 of Jenny's equation are assumed to be constant throughout the period being considered. This description is a first approximation because the early years of a study emphasize the easily degraded portion of the soil N pool, while later years the more recalcitrant portions. Nevertheless, k_1 can provide initial estimates of the time required for any specified degree of approach to steady-state. For example, after four $t_{1/2}$'s the soil N would have traversed about 94% of the way to steady-state, although the strictly defined mathematical time for completely reaching steady-state would be infinity. To illustrate, the Kansas unmanured fallow–wheat–sorghum system (solid circles in Fig. 13–2a) has an N_o of 5.1 Mg ha^{-1}, an N_{ss} of 2.2 Mg ha^{-1}, a k_1 of 0.056 yr^{-1} and after four $t_{1/2}$'s (about 50 yr) the soil would have an annual decline of about 10 kg soil N ha^{-1} yr^{-1}; a value that would be extremely difficult to measure experimentally and would be within acceptable tolerances for advocating a steady-state approximation in many N budgets. The desired degree of approach to steady state will depend on the N budget's desired level of precision, the uncertainties in other components of the budget, and the size of the actual change in soil N, i.e., the $|(N_o - N_{ss})|$.

In the final assessment, we conclude that attaining a quasi steady-state condition requires consistent application of the same management practices over many years. The rate and magnitude of change to steady-state will depend on climate, soil properties, tillage, N additions, and cropping system; these are all factors that should be carefully considered before invoking the steady-state assumption. In general, the approach to steady-state will be faster in warm than cold soils, in coarse-textured soils rather than fine-textured soils, and in well-drained soils rather than poorly drained soils (Jenny, 1941; Meisinger and Randall, 1991). The transition to a quasi steady-state that allows a tolerable change in soil N for N budgeting can vary from a few decades in rapid N turnover systems or for small changes in soil N (small $|(N_o - N_{ss})|$), to more than a century in slowly reactive systems with large soil N changes. The appropriateness of the steady-state approximation in an N budget will depend on the desired precision of the N budget, the size of the anticipated change in soil N, and the uncertainties in other N budget processes. Fried et al. (1976) and Tanji et al. (1977) discuss the validity of steady-state assumptions when estimating long-term N management effects on leaching.

Unlabeled vs. Labeled Nitrogen Budgets

Nitrogen budgets can be constructed based on a total N basis or on a labeled N basis. These two approaches are fundamentally different, but once these differences are understood each approach can provide useful information for understanding the soil N cycle. An illustration will clarify this point. Consider the addition of a singly labeled ammonium nitrate fertilizer to two identical uncropped plots; one receives $^{15}NH_4NO_3$ and the other $NH_4{}^{15}NO_3$. Now consider the effects of a rainfall event, say 40 mm, which occurs the day after application and is recorded by three independent scientists. One scientist follows the unlabeled total N budget and would observe a very small N input from rain, moderate N losses to surface runoff, and modest N losses due to leaching and denitrification. The second scientist follows the $^{15}NO_3$–N budget and records significant losses to surface runoff, leaching, and denitrification because the labeled NO_3 was subject to all these processes. The third scientist follows the $^{15}NH_4$–N budget and records only small N losses from surface runoff since most of the N was adsorbed on cation exchange sites, and virtually no losses to leaching or denitrification. The question then arises: Which budget is correct? The answer is: All the budgets are correct—because each scientist constructs the correct N budget for their *individual frame of reference*, but their frames of reference differ, i.e., their "N budget systems" are different. The total N budget focuses on the total N inputs and losses of the entire system, while the labeled budgets focus on the *fate of the labeled N including the labeled N's interaction within the soil N cycle*. It is important to understand the fundamental differences between a conventional total N budget and a labeled N budget when formulating research objectives and when interpreting the research results. In the last two subdivisions of this section we will examine how labeled N budgets can expand our understanding of traditional total N budgets, and in the process point out the benefits and drawbacks of each approach.

Problems and Opportunities with Labeled Nitrogen Budgets

Tracer techniques provide a tool for following the fate of the added ^{15}N in soil: they can extend but do not supplant nonisotopic methods. Complex problems often arise during the interpretation of isotopic experiments on the soil–plant system (Jenkinson et al., 1985). Labeled N studies often show that ^{15}N fertilized plants take up more unlabeled soil N than plants not given N—an effect often described as an "Added N Interaction" (ANI). This phenomenon has sometimes been termed a "priming effect", a term first introduced by Bingeman et al. (1953). An ANI can be a real effect if it increases the soil N uptake, e.g., if the fertilizer N increases the crop root zone. Or it can be an apparent effect if the labeled N merely stands proxy for unlabeled N that would otherwise be removed from the soil ammonium N or nitrate N pool, for example by microbial assimilation of N.

A specific example will illustrate this state of affairs. Consider a soil containing x kg inorganic N, of which y kg is immobilized during the following day: at the end of that day $(x - y)$ remains as soil mineral N. If $^*x$ kg labeled fertilizer N had been added to the soil at the beginning of the same day, then $(x + {}^*x) - y$ of labeled plus unlabeled N would remain, assuming that the inorganic fertilizer had no effect on the quantity of N immobilized. But of the added labeled N, a portion $y{}^*x/(x + {}^*x)$ would have been immobilized, standing proxy for unlabeled N that would have otherwise have been immobilized. The portion of *un*labeled N that would be immobilized at the end of the day is $yx/(x + {}^*x)$, leaving $x - [yx/(x + {}^*x)]$ remaining in the soil. The ANI is defined as the *un*labeled inorganic N remaining in the fertilized treatment, minus the inorganic N remaining in the control, i.e., the difference between soil N in the plot receiving labeled fertilizer and that in the control plot. The mathematical expression for this is: ANI = $[x - yx/(x + {}^*x)] - (x - y)$; which simplifies to:

$$\mathrm{ANI} = y{}^*x/(x + {}^*x) \qquad [2]$$

In this example the *apparent* ANI is positive; that is, if the fertilized soil had been analyzed at the end of the day, it would have contained more unlabeled N than the unfertilized soil. An equivalent crop-based mathematical expression for the relative ANI is given by Harmsen (2003) as: ANI/NF = ANR – ^{15}NR, where NF is the rate of N fertilizer, and ANR and ^{15}NR are the fractional N recoveries estimated by the traditional difference method or the ^{15}N method, respectively.

A more sophisticated treatment in Jenkinson et al. (1985) describes how ANIs can arise when immobilization and mineralization occur simultaneously, as usually happens (Jansson and Persson, 1982). Immobilization is not the only process than can generate ANIs by pool substitution in experiments using labeled N additions: so can denitrification. Under certain conditions, both positive and negative apparent ANIs can be observed when plants growing in unlabeled soil are treated with labeled fertilizer– this point is discussed further by Jenkinson et al. (1985) and by Hart et al. (1986).

A possible crop-based cause of ANIs is the continual exchange of ammonia between plant intercellular spaces and atmospheric ammonia during reproductive growth and senescence, especially when plants receiving labeled N senesce under water- or heat-stressed conditions. This exchange can lead to replacement of labeled ammonium N within ^{15}N-fertilized plants by unlabeled atmospheric ammonia (e.g., Francis et al., 1997), producing an ANI during senescence. Whether this plant–atmosphere ammonia exchange results in a net loss of labeled N, a net loss of unlabeled N, or simply an exchange producing negligible losses, depends on the behavior of NH_3 within the crop canopy. The effects of plant–atmosphere ammonia exchange on unlabeled N balances and on ^{15}N balances must await further research.

Another crop-based contribution to ANIs is the differing uptake patterns of labeled vs. unlabeled soil N with depth. This phenomenon arises because virtually all label N studies add ^{15}N to the surface soil, leaving the subsoil N with much lower ^{15}N concentrations. As a crop grows it first utilizes topsoil N that is more highly labeled, and then utilizes subsoil N as the root system extracts water from deeper in the soil. The result is that early in the season the ANI is negative (^{15}NR > ANR), due to a high reliance on topsoil ^{15}N by the young root system, but as the season progresses the ANI becomes positive (ANR > ^{15}NR) as the lower enriched subsoil N resources are utilized. Garabet et al. (1998a, 1998b) grew ^{15}N-fertilized winter wheat in Syria and sequentially sampled the total aboveground crop from fertilized and unfertilized plots during the entire growing season over 2 yr. They observed negative ANIs for the first 15 wk of the season that averaged −5 kg N ha^{-1}, which transitioned into positive ANIs of +13 kg N ha^{-1} during Weeks 15 to 23, and finally ended at maturity with an ANI of +10 kg N ha^{-1} after 25 wk. This study offers a clear example of the sequential change in ANI during crop development due to the nonuniform labeling of soil N with depth and the progressive uptake of N from lower soil depths as a crop matures under semiarid conditions.

In general, apparent ANIs can arise whenever both unlabeled N and labeled N are present in the same N pool, and in the same chemical form, at the same time. If an experiment with labeled fertilizer generates an ANI, positive or negative, the first task is to determine if the ANI is apparent, i.e., arising because of pool substitution. Only then should the possibility be considered that the added fertilizer causes real changes to N transformations already occurring in the unfertilized soil, such as immobilization, mineralization, or plant N uptake. Readers are referred to Jenkinson et al. (1985) for theoretical examination of ANIs and how they can affect the interpretation of experiments with isotopes, as well as a discussion of the relationship between ANIs and fertilizer N uptake efficiencies.

Extending Labeled Nitrogen Budgets to Total Nitrogen Budgets

Powlson et al. (1986a) developed an approach for using labeled-N data to construct a more complete picture of the total N budget processes. We will first examine this approach, and then the assumptions that are necessary for applying this theory.

Formulas for Extending Nitrogen-15 Budget Data to Total Nitrogen Budgets

We begin by considering the soil–crop system in a specified area, to a specified depth, over a defined time, but excluding the soil organic N. Under steady-state conditions the mathematics of Eq. [1] becomes

$$\mathrm{N_{inputs}} = \mathrm{N_{outputs}}$$

$$F + I + S_{\mathrm{m}} = H + L + S_i \qquad [3]$$

where F is fertilizer N input over the budget time interval, Δt; I is input of nonfertilizer N over Δt; H is output of N in harvested crop over Δt; L is N losses from the soil–crop system (i.e., leaching, denitrification, volatilization) in Δt; S_i is N returned to the soil by immobilization (i.e., immobilized by soil organisms, plus N in crop residues and all root residues) over Δt; and S_{m} is N released from the soil by mineralization of organic N over Δt.

The fractional recoveries of the total N input ($F + I + S_{\mathrm{m}}$) is given by R_t, so that $(1 - R_t)$ is the fractional loss of total N from the soil–crop system and

$$L = (1 - R_t)(F + I + S_m) \quad [4]$$

The fractional recovery of the total N input *retained in the soil* is R_s, e.g., N returned in crop aboveground residues, roots, root exudates, etc., so that $S_i = R_s(F + I + S_m)$. Under steady-state conditions $S_i = S_m = S$, and the preceding equation simplifies to

$$S = [R_s/(1 - R_s)][F + I\ ] \quad [5]$$

In most agronomic studies the fertilizer N input (F) and harvested crop N (H) will be directly measured in Eq. [3], leaving unknowns I, L, S_i, and S_m. Under steady-state conditions S_i equals S_m, and the two remaining unknowns can be solved by use of R_t and R_s. The N losses from the system can also be expressed in terms of I and F, from Eq. [4] and Eq. [5]:

$$L = [(1 - R_t\ )/(1 - R_s)][F + I] \quad [6]$$

And an equation for nonfertilizer N inputs can be derived from Eq. [3] and Eq. [6] as follows:

$$I = [(1 - R_s)/(R_t - \mathrm{R}_s)][H] - F \quad [7]$$

Jenkinson et al. (2004) have presented a formally similar treatment for cut grassland, in which there is an additional term for return of part of the harvested grass to the soil. If soil organic N is not at steady-state, then S_i and S_m are not equal. However the equations can still be solved, provided the difference between S_i and S_m can be estimated with the desired precision to satisfy the goals of the N budget.

Assumptions for Extending Nitrogen-15 Budget Data to Total Nitrogen Budgets

The development of the above equations has involved several important assumptions. The key assumption is that the fractional recovery of N in crop plus soil (R_t) is similar for all N inputs to the soil–crop system; i.e., the behavior of the labeled fertilizer N input is similar to that of all the other inputs (rainfall N, irrigation N, etc.). A quantity of fertilizer N applied to a growing crop at the optimal time and in an optimal position is clearly likely to be taken up more efficiently than (say) N arriving in rain during the winter season, when growth is slow. The validity of this assumption should be assessed for each individual input by considering the losses, and recoveries, from that input over the whole period of the N budget. An experimental approach for this assessment is considered below, but a sensitivity analysis can also be used, as shown by Jenkinson et al. (2004). The sensitivity analysis varied R_t for each N input to put limits on the calculated values of L, S, and I for fertilized grassland. This analysis showed that in their grassland study I and S were relatively insensitive to changes in R_t and to changes in R_s.

The second assumption is that R_s (the fractional recovery of N in the soil) is the same for *all* of the incoming N that is retained by the soil, labeled and unlabeled. This assumption is probably valid for labeled and unlabeled inorganic N reaching the roots at the same time, but more questionable if they arrive at different times or by different pathways. The partition of N between roots and shoots may well be different for unlabeled N taken up just after germination and labeled N applied later at midvegetative stages. Again, the partition between roots and shoots will almost certainly not be the same for N arriving from the soil as for N reaching the leaves by dry deposition, say as NH_3. Similarly, the R_s of fertilizer N would likely differ from that of manure N, due to the slow release of manure N over time, the greater gaseous losses from manure, and the greater potential for N immobilization with N sources containing C.

One experimental approach to evaluate these assumptions is to add the labeled N to microplots at different times of the year within the same treatment of an experiment, as done by Powlson et al. (1986a, 1986b), who compared the fate of labeled N added in the fall and in the spring. The times of addition should preferably be in different seasons of the hydrologic year and/or the crop growth cycle, to provide insight into the fate of N over the entire year. In addition, the parameters R_t and R_s should be estimated from averages over several years, because both are readily influence by weather conditions, as shown by their CV's of 10 to 20% (Powlson et al., 1986a).

It bears repeating that the validity of the above two assumptions needs to be examined on a case-by-case basis for each individual N budget. Great caution is needed before applying the above equations to N budgets in complex soil–crop systems; for example in long-term rotations with legumes, in systems utilizing organic and inorganic N sources, and in systems experiencing wide ranges in aerobic vs. anaerobic conditions.

Integration of Labeled Nitrogen Data into a Total Nitrogen Budget

Data obtained by Powlson et al. (1986a, 1986b) from field experiments with ^{15}N will now be used to illustrate how Eq. [5] through [7] can be utilized to estimate nonfertilizer inputs (I), total N losses (L), and soil N cycling (S). Powlson et al. (1986a, 1986b) began by superimposing 2-m square microplots receiving ^{15}N-labeled fertilizers within the large permanent plots of the Broadbalk Experiment. The large plots were those receiving the traditional dressings of P, K, and Mg plus either 0, 48, 96, 144 or 192 kg fertilizer N ha^{-1} annually, with the labeled fertilizer added at virtually the same N rate and at the same time as the large plot. Six microplots were established within each permanent plot, three received $^{15}NH_4{}^{15}NO_3$ in mid-April 1980 and the other three in mid-April of the following year. The wheat was a modern high-yielding, short-stemmed variety. Powlson et al. (1986a) measured total N and labeled N in wheat grain, straw, chaff, stubble, and in the soil to a 23- or 50-cm depth, separating the soil N into inorganic and organic forms.

In an important supplementary experiment, Powlson et al. (1986b) also examined the fate of fall-applied ^{15}N applied at 45 kg N ha^{-1} in the same two cropping years as the spring-applied ^{15}N. An abridged summary of their data (Table 13–3) shows a marked difference in R_t, the average total recovery of labeled N in crop plus soil, for the spring vs. fall applications with the former amounting to about 83% and the latter to only 47%. Fall-applied N is subject to higher leaching losses during the winter, when evapotranspiration is low (Goulding et al., 2000; Lawes et al., 1882; Powlson et al., 1986b) and crop N demand is also low (Powlson et al., 1986b; Widdowson et al., 1984). Higher losses of fall-applied N were first reported by Lawes et al. (1882) who monitored tile drainage from Broadbalk plots given 96 kg N ha^{-1} either in spring or fall, and reported annual drainage losses of 32 and 83 kg N ha^{-1}, respectively. It is also interesting to note that the additional 51 kg N ha^{-1} that Lawes et al. (1882) reported as drainage losses from fall N represents about 53% of the added fertilizer, leaving about 47% for crop recovery plus gaseous losses; these values are strikingly similar to the modern-day recoveries of fall-applied N.

Table 13–3. Winter wheat removals of labeled and unlabeled N from fertilizer applied in the spring or fall in 1980–1981 and 1981–1982 cropping seasons on the Broadbalk Winter Wheat Experiment. The plots received nominal rates of 145 kg N ha^{-1} in the spring (Powlson et al., 1986a) or nominal rates of 45 kg N ha^{-1} in the fall (Powlson et al., 1986b). The data illustrate the calculation of total ^{15}N recovery (R_t) and ^{15}N recovery in the soil (R_s) for use in estimating nonfertilizer N inputs, N losses, and soil N cycling, see text for equations and discussion.

Year, time of application of ^{15}N fertilizer	Rate of ^{15}N labeled fertilizer	Nitrogen harvested in grain plus straw			^{15}N in 50 cm soil plus crop residues	Fraction of ^{15}N in 50 cm soil and total in crop (R_t)†	Fraction of ^{15}N returned to 50 cm soil (R_s)‡
		Labeled	Unlabeled	Total			
	kg N ha^{-1}						
Spring ^{15}N							
1980–1981, April	147	69	63	132	44	0.77	0.30
1981–1982, April	143	77	41	118	51	0.90	0.35
Spring Avg.	145	73	52	125	48	0.83	0.33
Fall ^{15}N							
1980–1981, October	45§	12	151§	163	14	0.57	0.30
1981–1982, October	45§	6	127§	133	10	0.37	0.23
Fall Avg.	45§	9	139§	148	12	0.47	0.27
Growing season recoveries (same as spring averages):						0.83	0.33
Establishment season recoveries (avg. of spring and fall recoveries)¶:						0.65	0.30
Estimated weighted average annual recoveries#:						0.80	0.32

† For example, $R_t = (69 + 44)/147 = 0.77$.

‡ For example, $R_s = (44/147) = 0.30$.

§ Received nominal rate of 144 kg N ha^{-1} of unlabeled N in the spring.

¶ For example, R_t = mean of 0.83 and 0.47, R_s = mean of 0.33 and 0.27.

Weighting factors of 0.82 for growing season and 0.18 for establishment season, see text for discussion; e.g., calculation of $R_t = 0.82*0.83 + 0.18*0.65. = 0.80$.

Table 13–4. Fate of labeled N in the Broadbalk Winter Wheat Experiment, averaged over the 1979–1980 and 1980–1981 cropping seasons. Fertilizer N recovery calculated on a control-plot basis (difference method), and labeled N basis (Powlson et al., 1986a).

Crop or soil N component with descriptions	Without fertilizer N; control plot with P, K, Mg (Plot 05)	With fertilizer N in spring and P, K, and Mg†							
		48 kg N ha^{-1} (Plot 06)		96 kg N ha^{-1} (Plot 07)		144 kg N ha^{-1} (Plot 08)		192 kg N ha^{-1} (Plot 09)	
Grain yield, Mg ha^{-1} yr^{-1}	1.45	3.72		6.07		6.45		6.89	
N component:	Total N	Total N	^{15}N	Total N	^{15}N	Total N	^{15}N	Total N	^{15}N
Crop grain, straw, chaff, & stubble, kg N ha^{-1} yr^{-1}	31	70	25	127	60	161	87	190	117
Soil N, 0–23 cm, kg N ha^{-1} yr^{-1}	2900	3090	16	3370	20	3470	27	3260	28
Crop N recovery:									
Labeled N ‡, % of added fertilizer N	–	52		63		61		62	
Control (Plot 05)§, % of added fertilizer N	–	81		100		91		83	
Labeled N recovery in soil plus crop, % of added fertilizer N	–	84		83		79		77	
Labeled N lost, % of added fertilizer N	–	16		17		21		23	

† Annual mineral inputs (kg ha^{-1}): P, 35; K, 90; Mg, 11; and Na, 16 (Johnson and Garner, 1969).

‡ Labeled N recovery is the crop labeled N content divided by the labeled fertilizer N rate.

§ Control plot N recovery is the fertilized plot crop total N content minus Plot 05 total N content, divided by fertilizer N rate.

A more detailed summary for the 1979–1980 and 1980–1981 wheat on Broadbalk (Table 13–4) show that the recoveries of labeled fertilizer by the aboveground crop (grain, straw, chaff, and stubble) ranged from 52 to 63%, substantially lower than fertilizer recoveries estimated by reference to the traditional unfertilized control plot, which ranged from 81 to 100%. This difference probably arises because the long-term unfertilized plot is not an appropriate control for fertilizer N recovery estimates, due to a build up of mineralizable N in the plots receiving annual fertilizer N (Glendining et al., 1996). Another possibility is that the denser crop canopy on the fertilized plot absorbs more combined atmospheric N than the sparse crop on the control plot.

Estimating Nitrogen Recovery in Crop plus Soil (R_t) and Nitrogen Recovery in Soil (R_s)

The ^{15}N data in Table 13–3 for Broadbalk Plot 08 (receiving 144 kg N ha^{-1} yr^{-1}) will be used to illustrate how nonlabeled N inputs (I of Eq. [7]), N losses (L of Eq. [6]), and soil N cycling (S of Eq. [5]) can be calculated. An annual time step will be used in the calculation with the soil assumed to be at steady-state (see Fig. 13–2c). A soil depth of 0 to 50 cm, rather than the traditional Rothamsted depth of 23 cm, was used to include as much of the root zone as possible. This was done by allotting 3% of the added ^{15}N to the 23- to 50-cm layer, in accord with other measurements made by Powlson et al. (1986a).

The labeled N recoveries, R_t and R_s, were estimated on an annual basis from a weighted combination of R_t and R_s for the two major crop development seasons of winter wheat. The annual growth cycle of winter wheat can be divided into two major seasons: (i) the main growing season (April through August) when crop N uptake is rapid and leaching small, due to a combination of low rainfall and high crop water use, although significant rainfalls can induce denitrification events; and (ii) the establishment season (September through March) when crop uptake is relatively slow and leaching is substantial (Powlson et al., 1986a, 1986b; Jenkinson, 1977; Goulding et al., 2000). The estimates of R_t and R_s for the spring-applied ^{15}N averaged over the two growing seasons were 0.83 and 0.33, respectively (Table 13–3). The values of R_t and R_s for fall-applied ^{15}N were 0.47 and 0.27, respectively (Table 13–3). Now, some of the N in the wheat at the end of the establishment season (March) would have just been taken up (with recoveries similar to the summer growing season) and some would have been taken up in the fall (with recoveries similar to the fall-applied ^{15}N). We therefore used mean values of the spring and fall N recoveries to represent the establishment season (see details in fourth footnote of Table 13–3). Finally, the estimates of R_t and R_s on an annual basis were derived from a weighted combination of the establishment-season and growing-season recoveries. The weighting was done from the ratio of aboveground uptake of N just before the labeled fertilizer was applied (27 kg N ha^{-1}, mean of four seasons, see Powlson et al., 1986a), to the total uptake of N at harvest (157 kg N ha^{-1}, again the mean of four seasons from Powlson et al., 1986a) which produced weighting factors of 0.18 (= 27/157) for the fall establishment season and 0.82 for the summer growing season. This weighting strategy gave greater emphasis to the N recoveries during times of high N cycle activity (high nitrate N concentrations and high biological activity of summer) and a lower emphasis to N recoveries during low N cycle activity. The final annual estimates of R_t and R_s are 0.80 and 0.32, respectively (see fifth footnote of Table 13–3 for details).

Estimating non-Labeled N Inputs, "I"; N Losses, "L"; and Soil N Cycling, "S"

Updated estimates of the nonlabeled N inputs (I of Eq. [7]), N losses (L of Eq. [6]), and soil N cycling can now be calculated for Broadbalk Plot 08 using the annual total N input recovery (R_t) of 0.80 and the total N input recovery in the soil (R_s) of 0.32 (see Table 13–3), the steady-state assumption, and the average yearly N harvests of grain plus straw for 1990–1997 (Table 13–5). This time period was chosen to correspond to the most recent estimates of drainage losses on the Broadbalk plots (Goulding et al., 2000). The estimated input of nonfertilizer N from Eq. [7] is approximately 40 kg N ha^{-1} yr^{-1} {I = [(1 − 0.32)/(0.80 − 0.32)*129] − 144}. Part of this 40 kg N ha^{-1} is immediately attributable to the 4 kg N ha^{-1} in seed, and part to wet deposition of 7 kg N ha^{-1} yr^{-1} for 1990–1997 (T. Scott, personal communication, 2006). However, the source of the remaining 29 kg is less well established. Witty et al. (1979) found that there were surface crusts of blue–green algae on some Broadbalk plots and proposed that algae fix significant quantities of atmospheric N_2. Using the acetylene reduction technique, they found that fixation was highly variable, depending on surface moisture, previous desiccation, soil mineral N, and sunlight intensity at the soil surface. Witty et al. (1979) estimated that algal fixation on the N-deficient plot was about 25 kg N ha^{-1} yr^{-1}, but fixation was minimal on high N treatments such as Plot 08, where the crop cover was more complete. However, further attempts to quantify the algal fixation produced highly variable and uncertain results (P. C. Brookes, personal communication, 2005), so the algal N_2 fixation hypothesis for the additional N remains an open question. Azotobacter are present in Broadbalk (Ziemiecka, 1932) but the present view is that they make a negligible contribution to N_2 fixation. Another potential input is atmospheric dry deposition, which is currently considered to be the main source of the N that reaches the Broadbalk plots every year. Goulding et al. (1998) used deposition velocity calculations to estimate dry deposition at Rothamsted in 1996 at 34 kg N ha^{-1}. Over 85% of this deposition was attributed to oxides of N (NO_2 and HNO_3) originating from off-site sources, probably associated with urbanization. The remaining 15% was attributed to reduced N, with NH_3 accounting for less than 5% of the total dry deposition and particulates the remainder. Some of the NH_3 may have been of local origin, arising from plot-to-plot transfers from the nearby manured plot or fertilized plots, but quantitatively assessing these potential local sources will have to await future research.

The annual N losses from Plot 08 as estimated from Eq. [6] are 55 kg N ha^{-1} {L = [(1 − 0.80)/(1 − 0.32)]*(144 + 40)}. The average leaching loss measured over 1990–1998 on Plot 08 was 22 ± 6 kg N ha^{-1} (Goulding et al., 2000) and varied greatly from year-to-year (CV of 75%), with higher losses occurring after water-stressed years. Deducting 22 kg N ha^{-1} for leaching, leaves a total gaseous losses of 33 kg N ha^{-1}, most likely due to denitrification, which is also highly variable because losses depend on the transient concurrence between NO_3, high soil water content, O_2 demand, and temperature. Losses that are greater from denitrification than leaching are consistent with the conclusions of Addiscott and Powlson (1992), who examined 13 winter wheat experiments and used models to partition unrecovered ^{15}N between leaching and denitrification. Their general conclusion was that denitrification losses were probably twice as large as leaching losses, although the partition between leaching

Table 13–5. Nitrogen budgets (kg N ha^{-1} yr^{-1}), averaged over the 1990 to 1997 cropping seasons, for different N applications on the Broadbalk Continuous Winter Wheat Experiment. The fate of N, as a percentage, calculated for fertilizer N using the control plot as a reference, and from total N inputs (see footnotes for details). Crop data supplied by Dr. Paul Poulton (personal communication, 2005).

Nitrogen budget component	Without Fertilizer N; with P, K, Mg† (Plot 05)	With spring fertilizer N and P, K, Mg†											
		48 kg N ha^{-1} (Plot 06)			96 kg N ha^{-1} (Plot 07)			144 kg N ha^{-1} (Plot 08)			192 kg N ha^{-1} (Plot 09)		
	kg N ha^{-1} yr^{-1}												
N inputs:													
Soil N change	0	0			0			0			0		
Seed	4	4			4			4			4		
rain (wet dep.)	7	7			7			7			7		
N Dep. (dry dep.)‡	30	30			30			30			30		
Fertilizer N	0	48			96			144			192		
N input total:	41	89			137			185			233		
N outputs			FNF§	TNF¶		FNF§	TNF¶		FNF§	TNF¶		FNF§	TNF¶
Grain and straw	19	50	65%	56%	93	77%	68%	129	76%	70%	147	67%	63%
Drainage N#	12	12	0%	13%	15	3%	11%	22	7%	12%	30	9%	13%
To balance††	–10	–27			–29			–34			–56		

† Annual mineral inputs (kg ha^{-1}): P, 35; K, 90; Mg, 11; and Na, 16 (Johnson and Garner, 1969).

‡ N dry deposition estimated from calculated nonfertilizer input (I, see text) and from deposition velocity calculations of Goulding et al. (1998) with final average rounded to nearest 5 kg N ha^{-1}.

§ FNF is fertilizer-N fate, which is the N output value, subtracting the corresponding Plot 05 output value, and dividing by fertilizer N rate.

¶ TNF is the total-N fate, which is the N output value divided by the sum of the N inputs.

Mean annual drainage losses from Goulding et al. (2000) for 1990–1997.

†† + = a N input, – = a N loss.

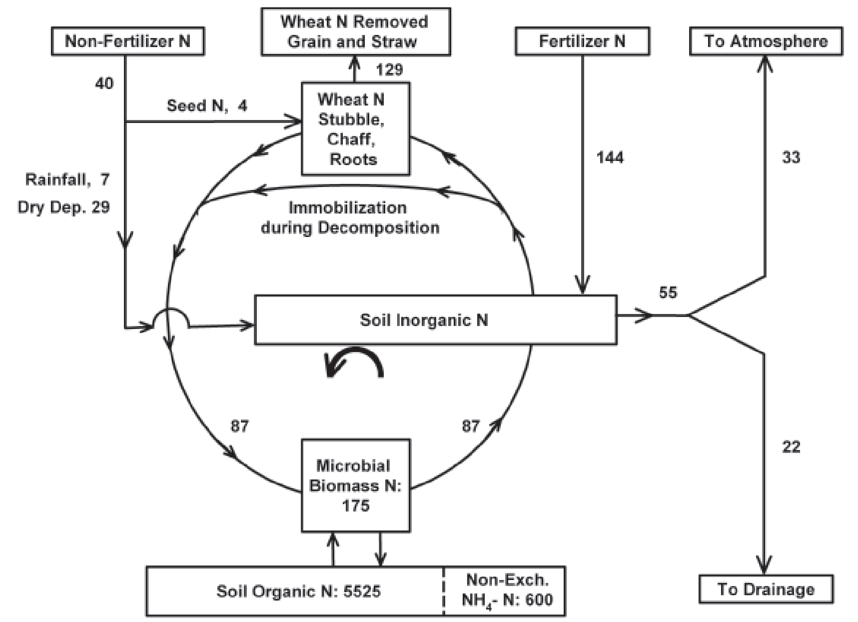

Fig. 13–3. Annual N cycle for 0 to 50 cm of soil and continuous winter wheat for 1990–1997 on the Broadbalk Continuous Winter Wheat Experiment Plot 08 receiving 144 kg N ha^{-1} annually. Values include estimates of nonfertilizer N inputs (I), total N losses (L), and soil N cycling (S) derived from labeled N studies as described in text. Drainage losses are from Goulding et al. (2000) and crop N data are from Paul Poulton (personal communication, 2005). The units for values within boxes are kilograms N per hectare and units for all other values are kilograms N per hectare per year, uncertainties are approximately ± 5 to 10% of the values shown.

and denitrification varied considerably between years and between experiments. Other potential gaseous losses are probably small, since micrometeorological studies at Rothamsted with wheat growing on the same soil type estimated ammonia emissions to be only 1 to 2 kg N ha^{-1} yr^{-1} (Yamulki et al., 1996) and NO and N_2O emissions to total only about 2 kg N ha^{-1} yr^{-1} (Yamulki et al., 1995).

The quantity of N entering (S_i) and leaving (S_m) the soil organic N pool annually is estimated to be about 87 kg ha^{-1}, as calculated from Eq. [5] {S = [(0.32)/(1 – 0.32)]*(144 + 40)}, assuming that S_i and S_m are similar, i.e., that steady-state conditions prevail. There is about 5500 kg organic N ha^{-1} in the top 50 cm of Plot 08 on Broadbalk, so the gross turnover rate is 63 yr. It is also interesting to note that the quantity of soil N cycling annually is about one-half of the N in the soil microbial biomass pool (Fig. 13–3), which gives a biomass turnover rate of 2 yr, very similar to that in old grassland at Rothamsted (Jenkinson et al., 2004).

Total-Nitrogen Budget for the Broadbalk Plot 08, Fertilized with 144 Kilograms Nitrogen per Hectare per Year (144 kg N ha^{-1} yr^{-1})

The above estimates of nonfertilizer inputs, N losses, and soil N cycling, have been integrated into Fig. 13–3 to produce the 1990–1997 total N budget for the Broadbalk plot that has been fertilized with 144 kg N ha^{-1} annually since 1852. The total annual N input of 184 kg N ha^{-1} into the system can be partitioned into crop removals (grain plus straw) of 70% (129 kg N ha^{-1}), leaching losses of 12% (22 kg N ha^{-1}), and gaseous losses of 18% (33 kg N ha^{-1}). The total N input recovery in the harvested crop on this plot over 1990–1997 was 70%, about 6% lower than the 76% crop recovery calculated from the traditional control-plot difference method (see Table 13–5), probably because the control plot mineralized less N as concluded by Glendining et al. (1996). Crop N recoveries computed from labeled N data in 1980–1981 were also lower than those calculated by the control plot method, which was 76% (Table 13–4). This overestimation of crop N recovery by the difference method using long-term controls can also lead to a corresponding underestimation of N losses to drainage, as shown by the drainage loss on this plot of 12% based on total N input, but 7% based on the control plot (Table 13–5). Long-term control plots should therefore be used with caution when estimating crop N recoveries and corresponding N losses.

The soil N diagram in Fig. 13–3 shows the complexity of the soil–crop N cycle, even in a relatively simple system of continuous winter wheat receiving inorganic N. Soil–crop systems that include legumes, crop rotations, perennial crops, or manure inputs, like those studied long ago by Boussingault (1841), are much more complex. However, important knowledge gaps still remain in Fig. 13–3. For example, how and when does N in dry deposition enter the soil–crop system, and when are the products of denitrification, N_2 and N_2O, released? These and related questions will test the ingenuity of future scientists, but their solution should lead to the more efficient use of N.

Applications of Nitrogen Budgets to Various Spatial and Temporal Scales

Nitrogen budgets are also a basic tool for summarizing and analyzing data on a wide range of spatial and temporal scales. The spatial scales covering a few square meters, tens of hectares, tens of square kilometers, and regional budgets covering thousands of square kilometers. The accuracy of an N budget will usually be greatest for small-scale budgets and will necessarily decrease with increasing size due to the inclusion of more heterogeneous ecosystems, the necessity of simplifying assumptions, and paucity of data for key processes, such as denitrification in riparian zones or drainage systems. Nevertheless, N budgets on larger areas can still identify major N sources and sinks and qualitatively evaluate management scenarios.

Field Plot Studies

Field plot studies are the most commonly used scale because they can be precisely managed allowing accurate treatment comparisons and their smaller size reduces variability and allows replication. However, small plots have limitations by minimizing opportunities for ecosystem interactions and are limited for studying larger-scale N transformations, such as surface runoff and volatilization.

Highly Instrumented Confined Microplots

Rolston and colleagues (Rolston and Broadbent, 1977; Rolston et al., 1978, 1979) reported one of the most complete field ^{15}N budgets that illustrate the dynamic nature of the soil–crop N cycle and the impact of climate, cropping, and manures on the fate of labeled nitrate. The study followed the fate of N added as $Ca(^{15}NO_3)_2$ at 300 kg N ha^{-1} to 1-m^2 plots of well-drained Yolo loam (Typic Xerochrepts). It used two soil water levels corresponding to soil–water pressure heads of about 1 kPa (about 90% saturation) and about 6 kPa (about 80% saturation) that were maintained with an automatic traveling spray boom. The study was conducted in both the summer and winter seasons and included a noncropped control, a ryegrass (*Lolium perenne* L.) cropped treatment, and a noncropped manured treatment. The manured plots received the equivalent of 34 t ha^{-1} of beef feedlot manure (about 40% C) that was incorporated into the surface 10 cm of soil 2 wk before the addition of labeled nitrate. The plots were heavily instrumented throughout the 1.2-m undisturbed soil profile with soil solution samplers, tensiometers, soil atmosphere samplers, thermocouples, and neutron probe access tubes to monitor soil moisture and estimate ^{15}N leaching. Temporary covers were also placed over each plot periodically to collect labeled N_2O and N_2 to directly estimate denitrification of labeled N. Following the 115-d study eight soil cores (2.5-cm diam.) were taken to a depth of 1.2 m and labeled N was determined in the organic and inorganic fractions of soil. Ammonia loss was not a factor in this study since the labeled N was in the NO_3 form.

Fate of Labeled Nitrate in Summer

The fate of the $^{15}NO_3$–N is summarized in groups of bar graphs in Fig. 13–4 with treatments listed in rows (cropped, or soil alone, or manured) and environmental conditions of water content (90 or 80% saturation) and season (summer or winter) in columns. The data from the summer high-water environment (first column Fig. 13–4) show that the uncropped control plot (middle bar graph group of first column) lost most of the labeled N through leaching (87%), and a small portion was transformed into soil organic N compounds (9%). The ryegrass crop (upper bar graph group of first column) utilized only a small fraction of the labeled N (11%) with the major loss occurring through leaching (66%). The uncropped manure treatment (lower bar graph group) markedly increased ^{15}N losses to denitrification (79% lost), particularly compared with the uncropped no-manure treatment. The high denitrification losses in the warm summer months (23°C) were encouraged by the wet soil, the available C from the manure, and the high NO_3 concentrations. The high denitrification with manure also decreased leaching losses from 87 to 12% compared with the uncropped no-manure treatment.

On the summer plots at 80% saturation (second column of bar graphs) the untreated soil accounted for most of the labeled N as inorganic soil N (86%) with no leaching losses and only small losses to denitrification (6%). The ^{15}N budgets for the untreated plots at the two water levels contrast sharply, 87% of the labeled N was leached in the high moisture treatment while 86% remained as soil nitrate N on the lower moisture treatment. The cropped plots of the low moisture summer treatment accumulated most of the labeled N in soil organic forms (45%) with plant uptake increasing to 21% and denitrification amounting to 13%. The cropped plots at 80% saturation had approximately double the plant N uptake and soil organic N compared with the 90% saturation treatment. Leaching losses were also reduced from 66% with 90% saturation to undetectable with lower soil moisture. However, the cool-season ryegrass was not a highly effective N sink, taking up 11 to 21% of the ^{15}N, during the warm summer months of the study. The manured plots at the lower moisture also accumulated most of the labeled N as soil nitrate with denitrification amounting to 26%, which was about four times greater than the unmanured soil. Denitrification in the manured high-moisture plots was reduced about one-third by lowering soil moisture from 90 to 80% saturation, indicating the major impact that soil aeration has on this process.

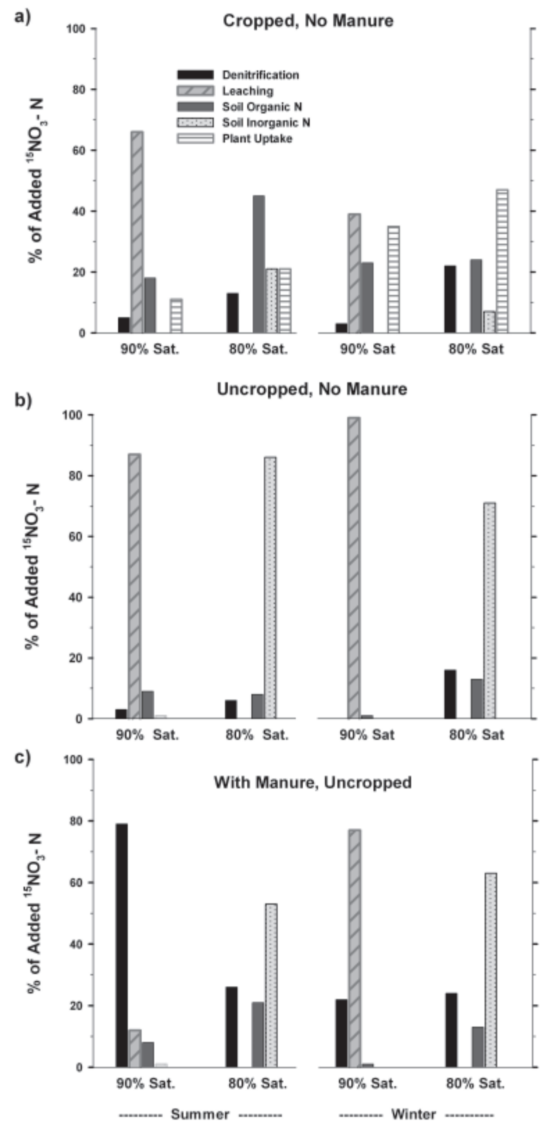

Fig. 13–4. Nitrogen budgets summarizing the data of Rolston and Broadbent (1977) and Rolston et al. (1978, 1979) showing the fate of labeled nitrate N added to a loam soil in California as affected by two water regimes (80 or 90% saturation) and (a) cropped to annual ryegrass without manure, (b) uncropped without manure, or (c) uncropped with manure. Study was conducted in the summer (two sets of figures on left) and winter (two sets of figures on right) seasons. See text for discussion of measurement techniques and interpretation of results.

Several N budget principles are illustrated in these summer data, namely: the rapid loss of nitrate to leaching under wet soil moisture conditions and the absence of leaching under drier conditions, the increased loss of N to denitrification with available C (manure addition) plus high water (90% saturation), and the marked decrease in both leaching and denitrification with the modestly lower soil moisture levels.

Fate of Labeled Nitrate in Winter

Data from the untreated soil during the winter (third and fourth columns of bar graphs in Fig. 13–4) show that virtually all (99%) of the ^{15}N was lost via leaching under high moisture conditions while most accumulated as soil inorganic N (71%) under lower soil moisture levels. Adding a winter ryegrass crop (upper row of bar graphs) reduced leaching from 99 to 39% at the high soil moisture treatment and resulted in 35% of the ^{15}N accumulating in the aboveground grass and another 23% immobilized in the roots and soil organic N. Adding manure and maintaining high soil moisture without a winter crop resulted in major losses to leaching (77%) and only secondary losses to denitrification (22%) during the cool (8°C) winter months.

The lower soil moisture environment during the winter season virtually eliminated leaching for all treatments, which is similar to the summer study. The lack of leaching for the lower soil moisture treatment resulted in a substantial accumulation of ^{15}N as soil inorganic N in the uncropped plots, while the cropped plots accounted for most of the ^{15}N through plant uptake (47%) and immobilization into soil organic N (24%). Denitrification accounted for about 20% (range 16–24%) of the ^{15}N across the treatments of the 80% saturation plots.

Seasonal Comparison and Summary of Fate of Labeled Nitrate

The fate of the labeled N under wet soil moisture regimes after manure additions was markedly different in the cool (8°C) winter season where leaching dominated (third column of Fig. 13–4), compared with the warm (23°C) summer season (first column of Fig. 13–4) where denitrification dominated. This was likely due to reduced microbial activity in the cool winter season, but denitrification did not totally cease at lower temperatures. The winter data clearly show the benefit of growing a winter crop for reducing leaching losses and the level of residual nitrate N, a recurring conclusion in many cover crop studies (Dabney et al., 2001; Meisinger et al., 1991; Shipley et al., 1992).

Several precepts can be gleaned from the above study. One is that the fate of labeled nitrate is strongly affected by factors such as: available C (affecting microbial activity and oxygen demand), water regime (affecting leaching and oxygen status), temperature (affecting microbial activity, crop growth, and water use), cropping practices (creating sinks for N and water), and by soil properties that interact with all of the above factors. If the goal is to channel N into denitrification, then conditions should be managed to juxtapose wet soil conditions (high soil moisture regimes, drainage management), high available C (recent manure additions), and high microbial activity (warm temperatures). If the goal is to minimize leaching, then management should focus on controlling soil moisture (irrigation management, drainage management) and maintaining an actively growing crop on the soil.

An important corollary to the above statements is that the fate of soil nitrate N will depend on the environment encountered within several weeks after application, which is likely to differ for seasons within a year, for water regimes, and for crop management systems. As previously noted, the N balance for labeled N represents the fate of the labeled N atoms plus its interaction with the soil N cycle. The fate of

^{15}N may or may not represent the fate of the total N flowing through the soil N cycle during the course of an entire year. Therefore, important knowledge of the soil–crop N cycle can be obtained by tracing the fate of ^{15}N applied in various seasons, and in various agronomic management systems. The above seasonal differences support the approach of developing annual N budgets by integration of ^{15}N budgets from different seasons of the year into a comprehensive annual N budget, as described in "Integration of Labeled Nitrogen Data into a Total Nitrogen budget".

Paddy Rice with Two Gaseous Loss Processes

A labeled N balance was conducted with flooded rice in the Philippines on a Maligaya silty clay (Isohyperthermic Vertic Tropaquept) to determine the effect of three fertilizer placement and water management strategies on the fate of the ^{15}N urea fertilizer (DeDatta et al., 1989). Nitrogen budgets with urea in flooded rice are difficult because there are two significant avenues for gaseous N loss, ammonia volatilization and denitrification. The study determined the ^{15}N uptake in the aboveground rice, the roots, and that remaining in the soil to 50 cm that was sampled in depth increments of 0 to 5, 5 to 15, 15 to 30 and 30 to 50 cm. Leaching losses were minimal in the fine-textured soil, as shown by the absence of labeled N below 15 cm. Runoff losses were also minimal because of berms that isolated each plot, preventing runoff. Ammonia volatilization was estimated with the bulk aerodynamic method, which measured NH_3 loss from a circular plot with a 25-m radius using a simplified mass balance approach and simultaneously monitoring the primary variables driving ammonia volatilization, namely: pH, total ammoniacal N, temperature, and wind speed. These variables were also measured in the small plots, which allowed estimates of ammonia volatilization in the large circle to be translated into the conditions of the individual plots receiving various fertilizer management practices. Documenting the above major N pathways allowed denitrification to be estimated by difference.

Management Practices Comparison

The management practices studied centered on the first urea application that occurred near transplanting. The treatments were broadcast application of labeled urea into 5 cm of floodwater without incorporation, broadcast application into 5 cm of floodwater plus incorporation into the soil, and broadcast onto wet soil and incorporated before flooding. The last treatment allowed the urea to react with the soil for 2 d before returning 5 cm of floodwater to the plots. The fate of the first application of 80 kg of labeled urea N ha^{-1} was documented by estimating NH_3 losses and collecting soil and plant samples 10 d after fertilization. Soil and plant samples were also taken at crop maturity to document the fate of the total application of 120 kg ^{15}N ha^{-1}.

Fate of Labeled Urea for Placement and Water Management Practices

Incorporating the first application of urea with no floodwater present resulted in greater recoveries of ^{15}N in the aboveground crop (Table 13–6), in greater quantities of exchangeable NH_4–N, and also resulted in substantially lower ammonia volatilization losses (7% vs. ~50%). However, this treatment also had the largest denitrification losses, 25% vs. 15% or 3%, but there was significantly more ^{15}N re-

Table 13–6. Nitrogen budget for ^{15}N urea applied to flooded rice by traditional broadcasting into 5 cm of floodwater, by broadcast and soil incorporated with 5 cm flood water present, or by broadcast and incorporated with no floodwater present with 5 cm flood water returned 2 d later, data summarized from DeDatta et al. (1989).

Fate of ^{15}N Urea	Fertilizer and flood water management treatments			LSD %
N placement:	Broadcast, no incorporation	Broadcast, w/ incorporation	Broadcast w/ incorporation	
Floodwater:	5 cm	5 cm	0 cm	
	—— % of ^{15}N urea applied at transplanting† ——			
Aboveground plant N uptake	15	14	26	8
Soil exchangeable NH_4–N	11	7	18	7
Soil organic N and roots	15	21	24	9
Total recovery within system	41	42	68	13
Ammonia volatilization	56	43	7	8
Denitrification (by difference)	3	15	25	13§
	—— % of total ^{15}N Urea applied to crop‡ ——			
Rice grain N	24	24	34	8
Rice straw N	10	12	15	2
Total aboveground crop recovery	34	36	49	8§
Soil organic N and roots	21	26	28	6
Total recovery within system	55	62	77	8
Ammonia volatilization	37	29	7	5
Denitrification (by difference)	8	9	16	8§

† Percentage of 80 kg ^{15}N urea ha^{-1} applied at transplanting, sampled 10 d after application.

‡ Percentage of total 120 kg ^{15}N urea ha^{-1} applied to crop as 80 kg N ha^{-1} at transplanting plus 40 kg N ha^{-1} at panicle initiation or boot stage, sampled at end of season.

§ LSD not reported; assumed to be equal to most variable component in calculation.

tained in the soil–plant system than in the other treatments (68 vs. 42%). The NH_3 volatilization losses occurred rapidly, within 7 d after application, and were driven by high floodwater NH_4–N concentrations (15 mg N L^{-1}), pH's above 8, and windy conditions (3–5 m s^{-1}). Thus, incorporating the urea into the wet soil without floodwater and letting it react with the soil produced high exchangeable NH_4–N that effectively lowered ammonia losses, although some of the exchangeable N was likely oxidized to NO_3 and subsequently lost to denitrification.

The final N budget for the total application of 120 kg N ha^{-1} (lower panel of Table 13–6) shows higher ^{15}N sequestered in the rice grain and in the straw for the incorporated urea without floodwater, compared with the other treatments. Incorporation without floodwater also resulted in higher total ^{15}N recoveries (77%) than the other treatments (62% or 55%). The substantial reductions in ammonia loss listed for the final N budget were a direct result of the lower losses from the first urea application, because the second application gave only negligible gaseous losses due to rapid crop uptake and shading of the floodwater.

This study illustrates the importance of N management in a high N loss environment like flooded rice. Protecting urea N from ammonia volatilization can be achieved by keeping it out of the high loss environment of the floodwater. The soil N cycle par-

Table 13–7. Fate of ^{15}N labeled cover crop residues (*Phacelia tanacetifolia* Benth.) with a C/N = 19 applied to a lettuce production system at three dates after residue incorporation. All soil data for 0- to 30-cm depth, potential nitrate leaching estimated by summing nitrate N captured by ion-exchange resins at 60 cm plus nitrate N in the 30- to 60-cm depth (Jackson 2000).

Soil or crop N pool	Days after incorporation of labeled cover crop (C/N = 19)		
	14 (30 d preplant)	72 (midcrop)	116 (crop harvest)
	% of cover crop N		
Soil mineral N, ($NO_3 + NH_4$)–N	11	3	1
Soil microbial biomass N	4	2	1
Soil organic N residues, etc.	87	75	61
Potentially leachable N below 30 cm	0	2	5
Crop N uptake, roots and shoots	NA	1	21
Gaseous losses, (by difference)	nil	17	11
Approx. std. error of soil organic N Mean	± 7	± 2	± 4

tially counterbalanced the lower ammonia losses with higher denitrification losses, which again illustrates the interconnections between soil N cycle processes. This interconnectedness results from the principle of conservation of mass, i.e., a change in one soil N cycle process will likely result in changes in other soil N cycle processes.

Fate of Cover Crop Nitrogen in Vegetable Production

Jackson (2000) reported labeled N data that followed the fate of ^{15}N labeled cover crop residues (*Phacelia tanacetifolia* Benth.) applied to a Chualar loamy sand (Typic Argixeroll) growing irrigated lettuce (*Lactuca sativa* L.) in California. The study used 25-cm-diam. polyvinyl chloride (PVC) cylinders that were sampled throughout the lettuce production cycle by soil sampling for ^{15}N as nitrate, ammonium, microbial biomass, and organic N. The N pathways of crop uptake and potential leaching (from buried ion-exchange resin bags) were also documented. Denitrification and other gaseous losses were estimated by difference.

Jackson's (2000) data allowed construction of labeled N budgets a few weeks after incorporation (14 d), at midcrop (72 d), and at final harvest (116 d) as shown in Table 13–7. The ^{15}N budget at 14 d not only provides a measure of short-term N transformations, but is also a straightforward approach to validate sampling and analytical protocols, as shown by the complete recovery of labeled N (see Table 13–7). At 14 d about 85% of the ^{15}N was in organic forms, about 10% had been mineralized to NO_3 (C/N of residues was 19), and about 4% of the ^{15}N was in the recent ephemeral increase in microbial biomass. Between 14 and 72 d the organic N and mineral N declined with most of the decrease attributed to gaseous losses of about 17% (see Table 13–7), which was probably encouraged by the 217 mm of rainfall plus irrigation during the 8-wk interval. Between 72 and 116 d, a period of major crop growth, labeled organic N declined from 75 to 60% with crop uptake increasing to about 20%. The fate of the cover crop ^{15}N at the end of the first lettuce crop (Table 13–7) was: crop uptake 20%, soil organic N 60%, potentially leachable

N 5%, gaseous losses 11%, with small quantities of ^{15}N in the soil mineral N and microbial biomass N pools.

These data show that of the 40% of cover crop N that mineralized, about one-half was taken up by the crop, about one-fourth was lost in gaseous forms, and the remainder attributed to potential leaching plus inorganic N and microbial biomass. An interesting feature of these data is the small ^{15}N contribution to microbial biomass, which Jackson (2000) attributed to a greater immobilization of unlabeled N by the biomass as cover crop C was being decomposed. However, the biomass ^{15}N could also be underestimated because pieces of crop residue >2 mm were excluded from the biomass assay, thus excluding biomass directly associated with decomposing residues.

While the primary fate of the first-year cover crop N was soil organic N, the study also showed that residual cover crop N becomes slowly available to subsequent crops, as revealed by the succeeding crop of lettuce recovering only about 5% of the original cover crop ^{15}N. The low availability of residual organic ^{15}N has been a common observation in labeled N studies (e.g., Jansson, 1958; Legg and Meisinger, 1982) for both labeled organic sources (e.g., Seo et al., 2006; Varco et al., 1989; Jackson, 2000) and from immobilized inorganic ^{15}N (e.g., Jansson, 1963; Broadbent, 1980; Ladd and Amato, 1986; Janzen et al., 1990).

Fate of Manure Nitrogen in a Soil–Crop System

In "Highly Instrumented Confined Microplots" we illustrated the significant effects that manure can have on the fate of N in the soil nitrate pool. However, the fate of manure N itself, i.e., N in feces and urine, is also important because it is one of the most difficult N sources to manage. Chapter 21 of this monograph (Beegle et al., 2008) has discussed many of these challenges and management approaches using nonlabeled manure, however many recent studies have also developed methods to label manure with ^{15}N.

Labeling Manure for Nitrogen Budget Studies

In principle, the best method to label manure for N budgets is to grow an animal exclusively fed on rations from uniformly labeled feed stocks, i.e., label the entire animal plus all manure produced. However this approach would be prohibitively expensive, so researchers have used alternative pulse-labeling techniques. The short-term pulses usually feed ^{15}N labeled ration components to an animal for several days or weeks, with successive collection of the manure.

The inorganic N fraction in manure has been frequently labeled by spiking excreted urine with ^{15}N ammonium salts (e.g., Trehan and Wild, 1993) or labeled urea (e.g., Bronson et al., 1999). Labeling manure organic N presents more difficulties because it is a complex mixture of partially digested feed, digestive tract cells or excretions, living and dead microbial cells from the intestine and hind gut, and rumen or intestinal microbes. Nonetheless, ^{15}N labeling of manure has been reported for ruminants by feeding labeled urea (e.g., Rauhe and Bornhak, 1970; Powell et al., 2004) or labeled forages (e.g., Rauhe et al., 1973; Sorensen et al., 1994; Sorensen and Jensen, 1998; Powell et al., 2004). Chicken and swine manures have also been labeled by feeding labeled cereal- and legume-grains (Thomsen, 2004; Sorensen and Thomsen, 2005). All of these studies have clearly shown that ^{15}N enriched manure can be produced—but an important question remains regarding the uniformity of the labeled organic N and the consistency of the ^{15}N pool produced when the manure is mineralized. It is important to have the manure ^{15}N sufficiently uniform in N cycling studies, if the resulting ^{15}N data are expected to be representative of the major N fractions in the manure.

One common approach to evaluate the uniformity of ^{15}N labeling is to sequentially monitor the ^{15}N concentration of the excreted manure, or in the urine and feces. Thomsen (2004) fed six chickens a diet containing labeled barley (7.78 atom% ^{15}N) and labeled field pea (4.94 atom% ^{15}N) for 20 d with manure collected twice daily. The manure ^{15}N concentration increased during the first 7 d on the enriched diet that contained 6.43 atom% ^{15}N, but after 7 d the manure stabilized at about 4.18 atom% ^{15}N in the total N and 4.02 atom% ^{15}N in the NH_4–N fraction. Apparently, the ^{15}N in the ration was diluted with unlabeled N from slowly reacting sources within the animal, but the labeled manure was deemed satisfactory for tracing the fate of manure N in field studies (see next section, "Using Labeled Poultry Manure to Evaluate Manure Timing"). Sorensen and Thomsen (2005) also fed 45 to 50 kg swine a diet containing 2.37 atom% ^{15}N excess made from barley and field pea containing 4.47 and 1.72 atom% ^{15}N excess, respectively. They monitored the ^{15}N concentrations in the urine and feces for 11 d and noted a rapid increase in ^{15}N concentration for the first 3 d, then a very slow increase from 5 to 13 d that produced feces and urine averaging about 2.08 and 1.76 atom% ^{15}N excess, respectively. Again, both feces and urine contained lower ^{15}N enrichments than the diet, due to the dilution with slower reacting unlabeled N sources within the animal. It is noteworthy that the feces and urine enrichments differed, leading the investigators to conclude that an evaluation of the manure (the mixture of urine and feces) should be done as separate treatments, i.e., the labeled urine should be evaluated with unlabeled feces, and vice versa for the feces [as done by Thomsen et al. (1997) and by Jensen et al. (1999)].

A second approach to evaluate manure ^{15}N uniformity is to mineralize the feces in sand or soil, with regular monitoring of the ^{15}N composition of the resulting mineral N, as described in Sorensen et al. (1994) and in Sorensen and Thomsen (2005). The swine fecal N described in the preceding paragraph was mineralized for 12 wk (Sorensen and Thomsen, 2005) and produced somewhat lower mineral ^{15}N compositions than the fecal source during the first few weeks, but no significant differences during the remaining 10 wk. These results led Sorensen and Thomsen (2005) to conclude that the fecal labeling was sufficiently uniform to permit use in N cycling studies without corrections for nonuniform labeling.

The above studies show that production and use of labeled manure in soil–crop N budgets should include an evaluation of the uniformity of the labeled manure and the separation of the labeled urine and feces if their ^{15}N concentrations differ.

Using Labeled Poultry Manure to Evaluate Manure Timing

Thomsen (2004) used the ^{15}N labeled poultry manure described above to study the effect of time of manure application and bedding material on crop and soil ^{15}N recoveries. After collection the manure was mixed with modest amounts of bedding material, either wood chips or straw or no bedding, and then stored for 10 d to mimic a short-term storage before application. It is noteworthy that all manures lost nearly 20% of the C and 7 to 10% of the N during the 10-d storage, which highlights the fragile nature of manures and the need to standardize pretreatment procedures in manure research. The stored manures were then applied

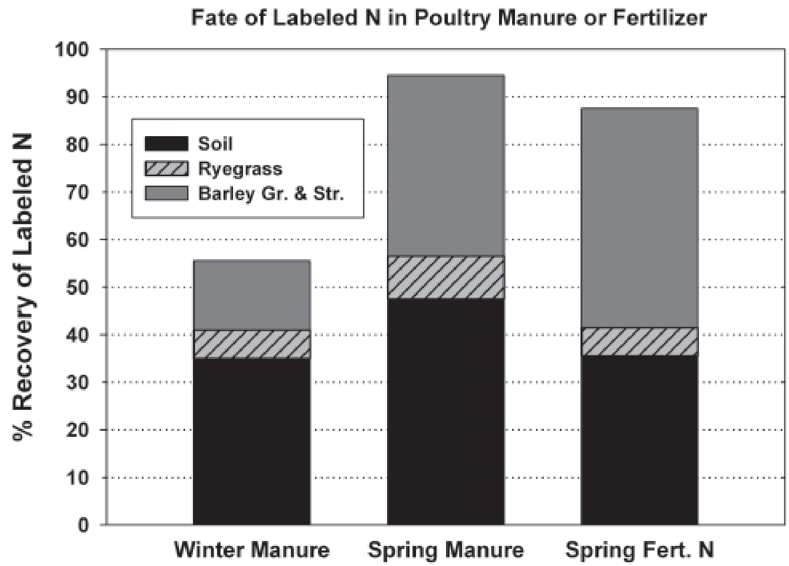

Fig. 13–5. Three-year fate of ^{15}N labeled poultry manure or fertilizer N applied in the fall or spring to barley (Thomsen, 2004). Residual ^{15}N determined by two years of ^{15}N uptake by subsequent crops of annual ryegrass and by ^{15}N recovery in the loamy sand soil (0–20 cm) at the end of the study.

at about 19 g N m^{-2} for spring barley in the preceding fall, or in the spring about 2 wk before planting the barley that was undersown with ryegrass. The ryegrass was grown for two full seasons after the barley establishment year to measure residual ^{15}N availability, with the 2nd- and 3rd-year ryegrass fertilized with modest rates of unlabeled N (3.8 g N m^{-2} in spring before the first cutting and 2.0 g N m^{-2} after the first and second cuttings) to assure a healthy stand. The plots consisted of 30-cm diam. PVC cylinders that were pressed 28 cm into the loamy sand soil (Typic Hapludult) leaving 5 cm aboveground to eliminate runoff. The manure was applied by removing 15 cm of surface soil, mixing the manure into the soil, and then replacing the manured soil into the cylinder. The treatment design was a complete factorial of the two application times and three bedding treatments. The bedding treatments all produced insignificant main effects and insignificant interactions with the timing variable, so results have been averaged across all bedding materials. Three types of control treatments were also included: a high (10 g N m^{-2}) and low (5 g N m^{-2}) rate of $^{15}NH_4{}^{15}NO_3$, and a nonfertilized control. All treatments were repeated in triplicate, and all cylinders received supplemental P and K fertilizer. The entire 3-yr serial experiment was repeated at a neighboring site the next year, which provided data representing weather conditions in two consecutive years for all phases of the cropping sequence.

The barley ^{15}N uptake in Thomsen's (2004) study showed that applying poultry manure in the winter produced substantially lower recoveries of 15% compared with the 38% recovery for the spring application (Fig. 13–5). Both manure treatments, however, were lower than the average 46% recovery from the two fertilized treatments. The apparent N recoveries for the barley, calculated by the difference method from data in Thomsen (2004), produced somewhat higher values than the ^{15}N recoveries with apparent recoveries averaging 18% for the fall manure, 43% for the spring manure, and 56% for the fertilizers. The higher recoveries estimated with the difference method is a common occurrence, reflecting isotope equilibration in soil–crop systems and/or crop N uptake patterns in relation to the distribution of labeled N in the soil (see "Problems and Opportunities with Labeled N Budgets"). The difference between the apparent N recoveries and labeled N recoveries translates into a positive ANI that was virtually the same for the manure and fertilizer treatments, amounting to an average of about 7 kg N ha^{-1}.

The labeled N budget from Thomsen (2004) shows that soil N was the other major sink for ^{15}N (see Fig. 13–5). The labeled N retained in the top 20 cm of soil was highest for the spring-applied manure, 48%, with the other two treatments retaining about 36% in the soil. The residual availability of the ^{15}N harvested in the ryegrass was similar for all treatments, amounting to a 2-yr sum of 6 to 9%, which is consistent with many ^{15}N studies that show annual residual ^{15}N availabilities of 2 to 5% from organic N sources (Seo et al., 2006; Ladd and Amato, 1986; Janzen et al., 1990). The total recovery of ^{15}N in all crops plus the soil was 95% for the spring-applied manure, 88% for the fertilized treatments, and 56% for the fall-applied manure (Fig. 13–5). Thomsen (2004) attributed the lower recoveries of fall-applied manure to greater leaching losses due to the 150 to 180 mm of winter precipitation that could leach mineralized N out of the soil before crop uptake begins in the spring. Ammonia volatilization was considered to be small due to soil incorporation of the manure and denitrification was also likely to be small due to the soil's coarse texture (82% sand).

The results of this study show that manures can be an excellent source of N, producing crop and total N recoveries comparable with fertilizers, but they need to be applied in phase with crop N demand to avoid N losses to the environment.

Large-Scale Nitrogen Budgets

What do we gain from estimating N budgets for large spatial scales? We already know the primary components of the within-field soil–plant N cycle, but the interplay of these within-field components with the N cycles of on-farm or off-farm systems are equally important in determining the final fate of N (Kowalenko, 2000). Large-scale budgets can identify the spatial distribution of the major N sources and sinks, can map the N flow paths that can identify areas for more detailed N evaluation, and can identify situations where moderate N surpluses on individual farms could accumulate across many individual farms to produce a large regional surplus.

Large-area budgets also provide a background to evaluate potential N management strategies (Brisbin, 1995). However, this evaluation will depend on the soundness of the assumptions, the quality of the input data, and the conceptual model of the large-scale budgets, which all affect the estimates of environmental losses and the potentials for improved N management scenarios. Thus, large-scale N budgets, although inherently less precise than traditional field-scale budgets, provide a useful broad-spectrum tool for evaluating options for improving the N balances within large areas. Zebarth et al. (1998, 1999) has provided an excellent example of using large-scale N budgets to assess N inputs and evaluate management strategies.

Background and Description

The Abbotsford-Sumas Aquifer underlies southwest British Columbia, Canada, and northwestern Washington on the U.S. side of the border. The aquifer has extensive areas of high nitrate groundwater that have been attributed to widespread nonpoint sources of contamination. Liebscher et al. (1992), Carmichael et al. (1995), Zebarth and Paul (1995), Wassenaar (1995), Zebarth et al. (1998), and Hii et al. (1999) all reported consistent high nitrate concentrations in the aquifer's shallow and deep wells with 30 to 50% of the wells being above the 10 mg NO_3–N L^{-1} health advisory limit, and up to 80% having concentrations above 8 mg NO_3–N L^{-1} (Wassenaar, 1995).

Evaluating Nitrate Sources for Water Quality

Mitchell et al. (2003) reported >10 mg NO_3–N L^{-1} throughout the aquifer in northwestern Washington, with shallow groundwater commonly having twice this value, but the groundwater delta ^{15}N ($\delta^{15}N$) data of Mitchell et al. (2003) failed to clearly identify the N sources. Wassenaar (1995) used $\delta^{15}N$ and delta ^{18}O ($\delta^{18}O$) data from the aquifer to identify likely sources of the NO_3 and concluded that poultry manure, and to a lesser extent fertilizer N, were the primary sources. However, the $\delta^{15}N$ approach has relatively low discriminatory power for identifying N sources, even when coupled with $\delta^{18}O$ data, because of: (i) the small $\delta^{15}N$ signature of agricultural N sources, e.g. $\delta^{15}N$ of fertilizer and soil N is –5 to +5‰ (parts per thousand), while septic N and all manure N sources are commonly +10 to +20‰; (ii) the difficulties of collecting representative samples and of highly precise isotope analysis; and (iii) the groundwater samples may represent a mixture of several $\delta^{15}N$ sources or may have been enriched in $\delta^{15}N$ by denitrification (Kendall, 1998; Herbel and Spalding, 1993; Fogg et al., 1998; Hauck et al., 1972). Another limitation of the $\delta^{15}N$ approach is that if it identifies a general source of N, e.g., manure N, it cannot identify the animal species contributing to the loss if several species are in the groundwater recharge area, nor can it suggest management practices that could mitigate the N losses.

On the other hand, an aggregated N budget approach based on farm enterprise sectors can identify likely sources of excess N, can define locations for monitoring N losses, and can suggest opportunities for improved N management. Zebarth et al. (1998) reported that the main activity that appeared to correlate with the rising nitrate concentrations was the change in agricultural activities from 1971 to 1991. As a result, Zebarth et al. (1998, 1999) employed an aggregated N budget approach for typical farm activities to estimate N losses and to suggest options to improve water quality.

Historical Changes in Agricultural Practices in Large-Scale Nitrogen Budget Area

The main study area of Zebarth et al. (1998, 1999) was the Matsqui South district of the Lower Fraser Valley that is directly over the Abbotsford aquifer, in southwestern British Columbia, just north of the U.S. border. It contains about 6,600 ha with about half in agriculture and the other half in forestry, rural homes and vacant land, and riparian trees along streams. The area's proximity to the Pacific Ocean results in a humid and moderate climate that receives about 1,500 mm of precipitation annually, with about 450 mm between April and October, and the remaining 1,050 mm in the nongrowing season. The nearby ocean moderates temperatures, with average monthly highs varying from 15 to 24°C during the growing season and 5 to 15°C in winter. The soils are seldom frozen. The dominant soils are well-drained silt loams derived from loess that was deposited over sand/gravel glacial outwash that would be classified as Haplorthods in the U.S. system. The soils commonly have high organic matter contents with values often near 8% (Zebarth et al., 1998). The medium-textured soils over coarse-textured material in a high-rainfall climate sets the stage for high percolation during the winter and high potential nitrate leaching to groundwater. Any nitrate remaining in the soil after the growing season, or mineralized during the fall–winter, is highly vulnerable to winter leaching (Kowalenko, 1987, 1989).

The agriculture in the Matsqui South district in the 1990s contained a high concentration of poultry production and specialty horticultural crops, especially raspberries (*Rubus idaeus* L.), with these enterprises developing over the past 35 yr (Zebarth et al., 1998). Earlier, in 1971, the district's main livestock species were laying hens and dairy with about two-thirds of the land producing hay, pasture, or silage corn. In 1981 the number of dairy animals had declined about 40% and the accompanying area of hay, pasture, and silage corn also declined 40%. The animal enterprises replacing dairies were meat-producing poultry units for chickens and turkeys, which increased about 270% above 1971 levels. In addition, a shift of land use into raspberry production occurred, that was a 240% increase compared with 1971 levels (Zebarth et al., 1998). The shift away from dairy and into poultry meat production continued between 1981 and 1991 as dairy animal numbers declined another one-third while numbers of poultry animals increased another 230% compared with 1981 levels. Accompanying the continued shift from dairy to poultry between 1981 and 1991 was a continued 45% decline in forage production compared with 1981 levels (Zebarth et al., 1998).

The effects of the above changes in soil–crop–livestock practices on the area's N cycle can be evaluated by estimating the N budgets for agricultural land in 1971, 1981, and 1991. These N budgets were estimated from Census of Agriculture data, producer surveys, direct field estimates of N pools, and the assumption of steady-state soil N levels (see Zebarth et al., 1998, for details). The N budgets for 1971 to 1991 (see Table 13–8) show only a small increase in total N additions, but a marked change in N sources with manure inputs increasing and fertilizer N inputs decreasing due to the expansion of concentrated animal production facilities and the change from the high fertilizer rates used on forages to lower rates on horticulture crops. Estimated N outputs in crops declined from about 175 kg N ha^{-1} in 1971 to about 90 kg N ha^{-1} in 1991. This decline was attributable to the replacement of grass–hay N outputs, which are commonly 300 kg N ha^{-1}, with raspberry N outputs that are usually less than 30 kg N ha^{-1}. The N needed to balance the N inputs vs. N outputs over the 20 yr increased from about 135 kg N ha^{-1} in 1971 to 245 kg N ha^{-1} in 1991 (Table 13–8), a quantity that estimates potentially leachable N, plus possible increases in soil organic N, plus N lost to other pathways not specifically listed in the table. Although the large-area N budgets in Table 13–8 should not be used to estimate NO_3 leaching losses, they do indicate that the potential for leaching substantially increased over the 20 yr and show that the lower crop N removals with the fruit crops compared with forage production were a large factor in this

Table 13–8. Nitrogen balances (kg N ha^{-1}) for agricultural land for the 1971, 1981, and 1991 in the Matsqui South study area (Zebarth et al., 1998).

N budget component	1971	1981	1991
	kg N ha^{-1}		
N inputs			
Inorganic fertilizer	138	113	89
Manure†	152	180	231
Atmospheric	40	40	40
total additions	329	333	359
N outputs, except leaching			
Crop removal	175	127	92
Denitrification	20	21	22
Total, except leaching	195	148	114
N needed to balance‡	134	185	245

† Manure input adjusted for estimated ammonia loss.

‡ N attributable to potential leaching, runoff losses, change in soil N, etc.

increase. The second contributing factor for this increase is the rise in concentrated animal production units that require importing protein and carbohydrates for the avian diets, as opposed to locally grown protein and carbohydrates that were the basis of the previous dairy rations, these imports show up as increases in manure N. The above discussion illustrates that large-scale N budgets can provide insight into the effects of changing agricultural practices on the levels of surplus N.

Nitrogen Budgets for Individual Sectors of Farm Enterprises

Estimating N budgets for individual sectors of a farm can provide the "building blocks" for understanding N flows of the whole-farm system, especially N flows involving livestock. These small-sector budgets can also identify opportunities for improved N management.

Field Crop Sector Nitrogen Budgets

The N budgets for two major agricultural enterprises in the Matsqui South district, raspberry and forage-crop production, are given in Table 13–9 and are derived from components estimated from producer surveys, from direct field measurements on replicated plots, or from the literature (Zebarth et al., 1996; Zebarth et al., 1998; Paul and Zebarth, 1997a). The N inputs consisted of fertilizer and/or manure plus atmospheric deposition of 40 kg N ha^{-1}, which is about twice the common estimate to allow for deposition of locally volatilized NH_3 within the district. Nitrogen outputs consisted of harvested crops, with raspberry values estimated from producer surveys and N concentration in the fruit (Kowalenko, 1994). The silage corn outputs were derived from direct measurement of corn N removals on replicated plots. Denitrification was estimated at 8% for the well-drained soils common to raspberry production, and at 18% for the silage–corn soils from acetylene block measurements using intact soil cores (Paul and Zebarth, 1997a). Ammonia volatilization was assumed to be 20% for poultry litter N (Zebarth et al., 1998) and 17% for dairy slurry N that had been incorporated within 24 h after application as suggested by the British Columbia Ministry of Agriculture, Fish-

Table 13–9. Nitrogen budgets (kg N ha^{-1}) for fertilized or manured raspberry production and silage–corn production, data from Zebarth et al. (1998, 1996) and Paul and Zebarth (1997a), all values rounded to nearest 5 kg N ha^{-1}.

N component	Raspberry production		Silage–corn production
	Fertilizer	Poultry litter	Dairy slurry and fertilizer
N inputs		kg N ha^{-1}	
Inorganic fertilizer	70	50	100
Manure	0	400	300
Atmospheric†	40	40	40
Total additions	110	490	440
N outputs (except leaching, etc.)			
Crop removal	20	20	245
Denitrification‡	5	35	70
Ammonia volatilization§	nil	80	50
Change in soil total N¶	nil	80	30
Total, (except leaching, etc.)	25	215	395
N to balance, or N surplus	85	275	45

† Atmospheric input from Zebarth et al. (1998).

‡ Assumes denitrification of 8% of fertilizer and manure N in raspberries, and 18% of fertilizer and dairy slurry N in silage corn (Paul and Zebarth, 1997a,b; Zebarth et al.,1998).

§ Assumes 20% ammonia loss for poultry litter (Zebarth et al., 1998) and 17% loss during first 24 h before incorporation for dairy slurry (British Columbia Ministry of Agriculture, Fisheries & Food, 1993).

¶ Assumes 20% of poultry litter and 10% of dairy slurry N converted to long-term soil organic N (see discussion in "Estimating the Change in Soil Organic N").

eries and Food (1993). Soil total N contents were originally assumed to be at a steady-state by Zebarth et al. (1998). However, Table 13–9 assumes quasi steady-state only for the fertilized raspberry system, while the manured systems were assumed to sequester about 20% of the poultry litter N and about 10% of the dairy slurry N, which are first approximation estimates from the examples discussed in "Estimating the Change in Soil Organic N" and "Examples of Steady-State Soil N Contents". The above approach provides a basic N budget for the main soil–crop systems that can identify areas at risk for N loss. The resulting estimates of surplus N in Table 13–9 are 85 and 275 kg N ha^{-1} for the fertilized and manured raspberry systems, respectively, and 55 kg N ha^{-1} for silage corn. The N budgets also show that these surpluses arose from the low crop N removals of raspberries compared with corn and the high N inputs in the manured raspberry system.

The N surplus estimates of Table 13–9 should also be compared with field data whenever possible. Such comparative data can come from end-of-season soil nitrate sampling and/or groundwater monitoring. Direct soil sampling (0–90 cm) of nitrate after raspberries was conducted in the fall of 1991 from 7 fertilized commercial fields and 14 manured fields (Zebarth et al., 1998). This fall soil sampling found an average of 165 ± 39 kg NO_3–N ha^{-1} and 355 ± 156 kg NO_3–N ha^{-1} on fertilized and manured fields, respectively. Corresponding NO_3–N contents for soil samples on the dairy–slurry plots were 114 ± 54 kg N ha^{-1} (Zebarth et al., 1996). These fall soil nitrate-N data support the *relative* difference between the N budgets of Table 13–9, with manured raspberry fields containing about 200 kg N ha^{-1} more surplus N than fertilized raspberries and silage corn having the lowest residual nitrate.

A close agreement between the field samples and the N budget estimates should not be expected, due to the high inherent variability of soil nitrate (see "Estimating the Change in Soil Inorganic Nitrogen") and because sampling occurred in only 1 or 2 yr. Groundwater monitoring data of Zebarth et al. (1998) also support the general differences between N surpluses in Table 13–9, with higher nitrate-N concentrations being common in areas with high density animal systems and low concentrations in areas with nonagricultural land. However, groundwater monitoring is an inherently less sensitive approach for evaluating N surpluses because of the uncertainties in knowing what source areas are represented in a water sample, uncertainties in determining the age of water in the sample due to uncertain hydrologic gradients and possible mixing, and uncertainties about transformations of nitrate within the aquifer (see previous discussion in "Evaluating Nitrate Sources for Water Quality"). Nevertheless, the independent field data did validate the relative differences between N surpluses derived from the N budgeting processes of Table 13–9.

The N surplus of Table 13–9 represents N susceptible to leaching and other gaseous losses, although leaching would be the most likely pathway in this humid region as shown by Paul and Zebarth (1997b) who estimated that over 80% of the loss of fall nitrate was attributable to leaching. The surplus N of Table 13–9 is also the same variable as "potentially leachable N" of Meisinger and Randall (1991), which was proposed to identify areas with a high risk for N leaching. An N surplus may or may not result in an environmental problem, depending on the sensitivity of the area's surface- and groundwater to N loading and to possible N transformations in ecosystems beyond the agricultural field (Groffman, 2008, see Chapter 19). In British Columbia researchers have reached a consensus that an N surplus of about 50 to 100 kg N ha^{-1} would maintain crop productivity while protecting the environment, based on the data of Zebarth et al. (1995) and Brisbin (1995). These values recognize that agricultural soils will lose some quantity of N to the environment, and that defining more specific targets would be highly subjective. An alternative justification for the 50 to 100 kg N ha^{-1} value is that if this quantity of NO_3–N was dissolved in the approximate 1000 mm of recharge water, it would produce a NO_3–N concentration of about 5 to 10 mg NO_3–N L^{-1}, a value below the health advisory level for drinking water.

Livestock Sector Nitrogen Budgets

Nitrogen budgets on the most common livestock sectors in the study area can also be constructed, but are more difficult than soil–crop budgets because of ammonia volatilization and the complexities of N losses from various manure management systems. Data for the main livestock systems of the study area, poultry meat or egg production, were derived from the Census of Agriculture data and summarized by Zebarth et al. (1998).

The N budget for a typical broiler house showed that N removed in broiler carcasses accounted for about 45% of the feed N (Zebarth et al., 1998), with the remaining N likely split into 20 to 25% as NH_3 volatilization and 30 to 35% as manure N. These values are supported by the reports of Patterson et al. (1998) and Coufal et al. (2006) who estimated that broiler carcasses account for 50 to 57% of the feed N, NH_3 losses 18 to 21%, and manure 22 to 31%. The layer house N budgets of Zebarth et al. (1998) estimated that 50% of the feed N could be attributed to egg production; the remainder was likely split into somewhat higher losses to NH_3 volatilization than broilers, say 25 to 30% (Liang et al., 2005; Yang et al., 2000), leaving 20 to 25% for manure. It is noteworthy that the 45 to 50% N recoveries by poultry are comparable with N recoveries by field crops.

Dairy N recoveries are lower than poultry with lactating cows commonly yielding about 20 to 30% of their feed N as milk during lactation (Bulley and Holbek, 1982; Wilkerson et al., 1997). The N losses after excretion are usually quite variable, being dependent on specific manure management systems and the crude protein content of the ration.

The above recoveries of feed N in livestock products generate N excretions of about 50 to 70% of the N entering as feed, which will ultimately appear as ammonia losses or as manure N that will usually be applied to cropland. Thus, livestock N excretions need to be considered in large-scale N budgets.

Combining Crop and Livestock into a Whole-Farm Nitrogen Budget

The next step is to integrate the soil–crop and livestock budgets and estimate the whole-farm N budget. The whole-farm approach is most useful for identifying major N flows, N sources/sinks, and estimates of N utilization efficiencies that can suggest areas for more detailed N evaluation (Lanyon and Beegle, 1989; Dou et al., 1998; Klausner, 1993).

Paul and Beauchamp (1995) have provided a good example of a whole-farm budget for the University of Guelph's dairy operation at the Elora Farm, 20 km north of Guelph. The dairy had a 145-cow milking herd plus 145 head of replacement calves and heifers. The representative soil for the farm is the well-drained Conestoga silt loam (Typic Hapludalf). The average annual whole-farm budget was derived from measurements over three consecutive years. Dairy N outputs were estimated from milk sales and protein concentrations (converted to N%) and animal N exports were estimated from animal sale weights assuming a 2.08% N in the whole animal (Maynard et al., 1979). The dairy's external N inputs were determined from records of purchased feed and bedding and the protein concentration of each feedstock. The feed N entering the dairy from within the farm was determined for individual field records of measured crop yields and periodic samples of crop N concentration. Records were also maintained on individual fields documenting N inputs from fertilizer, plus manure applications and manure analyses (see Paul and Beauchamp, 1995 for details). Atmospheric N input by wet deposition was estimated from local rainfall monitoring stations (Vet et al., 1988) and dry deposition of NO_3 and NO_2 from Barrie and Sirois (1986). The remaining N input was from N_2 fixation for alfalfa, which was grown on about one-third of the acreage, and for small areas of periodic crops of soybeans.

Estimating N_2 fixation has been a challenge since the mid-1850s when Boussingault estimated annual alfalfa N inputs of about 140 kg N ha^{-1} (Table 13–1 and "Boussingault"). Nitrogen fixation can be satisfactorily estimated on research plots by isotope dilution or growing nodulating and non-nodulating strains of a legume (Russelle, 2008, see Chapter 9 for further discussion). But field-scale estimates generally assume that the legume derives a constant percentage of its N from fixation. For example, Klausner (1993) in New York and Dou et al. (1998) in Pennsylvania both assumed a 60% fixation value for alfalfa, while Meisinger and Randall (1991) suggested values that varied from 30 to 85% for perennial forages depending on soil N availability. Paul and Beauchamp (1995) assumed that 100% of the legume N was fixed, but our summary has assumed that two-thirds of the legume N was

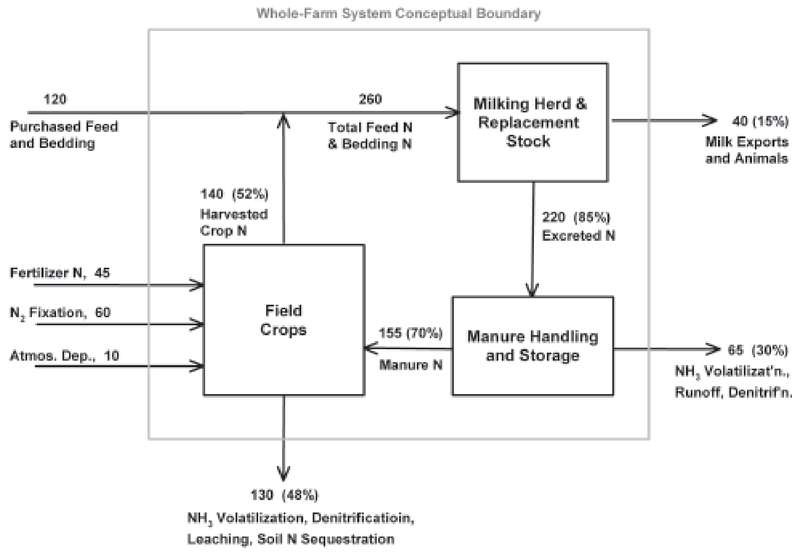

Fig. 13–6. Annual N flows on an Ontario dairy farm for an average hectare of cropland and associated dairy and manure storage facility (Paul and Beauchamp, 1995). Values are kilograms N per hectare per year for the cropland and approximate kilograms N per year per lactating cow (including her replacement stock). The percentages refer to the distribution of N entering a given N pool (ha of cropland, dairy, or manure storage) into the various N loss pathways or within-farm recycled N pathways. See text for discussion of estimation techniques and interpretation of results.

fixed as suggested by Meisinger and Randall (1991, p. 100). The legume N credit for alfalfa or soybeans that is commonly used for N recommendations for a succeeding cereal crop were also included as fixed N, with two-thirds of the legume credit attributed to N_2 fixation.

Estimating a whole-farm N budget also requires several assumptions. Paul and Beauchamp (1995) assumed that dairy animal numbers remained constant from year-to-year, that no significant feed surplus occurred in any year (a plausible assumption because three consecutive years were in the budget), and that the soil organic N was at a quasi steady-state condition.

The whole-farm N budget of Paul and Beauchamp (1995) is summarized in Fig. 13–6, which has adopted the point-of-view from the average soil–crop N budget (kg N ha^{-1} yr^{-1}) of the average field on the Elora Farm. Accordingly, the original flow diagrams of Paul and Beauchamp (1995) for the dairy component and the manure-handling component were rescaled relative to the average soil–crop N budget. This resulted in an annual dairy N input of about 260 kg N, which is about equal to the annual feed-N input for one cow plus her replacement stock. Thus, the annual N values in Fig. 13–6 can be viewed as kilograms N per hectare, or kilograms N per lactating cow including her replacement stock.

Figure 13–6 shows that the dairy operation exported only about 40 kg N, or about 15%, of the feed input. This value is consistent with reports from farms in The Netherlands of 17% (Aarts et al., 1992), and Pennsylvania of 15 to 19% (Lanyon and Beegle, 1989; Bacon et al., 1990), but are lower than other reports for lactating cows that range from 20 to 30% output efficiency (e.g., Bulley and Holbek, 1982; Van Horn et al., 1996). The difference in these N efficiencies is that the higher values refer to only the lactating herd, while the lower values include the N invested in growing replacement stock.

Manure excreted by the herd begins losing N immediately after excretion, mostly through ammonia volatilization, which occurs in the housing facility, during manure handling, and in the manure storage. The combined total losses from excretion to land application were estimated at about 65 kg N, or 29% of the excreted N (Paul and Beauchamp, 1995).

The largest N input to the soil–crop system was from manure (Fig. 13–6), which supplied about 57% of the total N added to the average field, followed by N inputs from N_2 fixation and fertilizer. The N removed in harvested crops accounted for about 140 kg N ha^{-1}, which is a little over 50% of the N entering the soil–crop system. The nonharvested N, 130 kg N ha^{-1} yr^{-1} represents N lost to the combined pathways of ammonia volatilization, denitrification, leaching, and any N that might be sequestered in soil organic N.

It is not possible to accurately determine the main pathways for N loss at this spatial scale. But the whole-farm budget does show the main areas for improving N utilization of the farm, one being the dairy herd itself to improve the 15% efficiency, and another being the soil–crop system to improve the 50% N use efficiency. The appropriate N management strategies to pursue for these areas will depend on a detailed analysis of the farm and the management expertise of the operator, but some potential options would be reducing excess N in the ration, better timing of manure applications, or substituting high-protein grass forage for alfalfa. The whole-farm budget identified the areas where such management improvements can have the largest impact. The primary area identified by Paul and Beauchamp (1995) for improving whole-farm N recovery was improved diets, which is the same conclusion reached by Dou et al. (1998).

Aggregating Sector Nitrogen Budgets for Evaluating Nitrogen Management Scenarios

The N budgets from representative farm systems can also be aggregated to estimate the N budgets for larger areas. This aggregation is usually more informative than taking a large-scale view and working down to the smaller areas because the smaller-scale budgets can be estimated with more accuracy and with more clearly understood assumptions than a large-scale budget. The approach of Zebarth et al. (1999), illustrates how small-scale budgets from the Matsqui South district contributed to the development of a large-scale budget of the entire district that was used to evaluate various N management approaches to reduce N losses.

The N budget for the Matsqui South district was estimated from a multilayered N budget model that included N flows for animal production units considering housing types, manure storage systems, and land application practices. The N flows from the livestock were merged with fertilizer inputs for the most common soil–crop production systems of the area to estimate field N recoveries and losses. These N recoveries and losses were estimated for a wide range of management scenarios to assess the effects of various combinations of manure, fertilizer, and livestock management strategies on the N surplus. Finally, the total N recoveries and losses for a given management scenario were calculated by an algorithm and summed across the district to estimate the N surplus for the specific set of management conditions (Brisbin, 1995; Zebarth et al., 1997).

Large-scale N budgets invariably include larger uncertainties than well-defined systems due to the varying degrees of data quality entering the budget and the validity of assumptions. Therefore, the numeric values from large-scale budgets are usually considered as first approximations. However, the *relative comparison* of the large-scale budgets over time or over N management scenarios, as compared to a reference scenario, can be very instructive.

Aggregating the Livestock Sectors into a Large-Scale Nitrogen Budget

The N budget algorithm for livestock in the Matsqui South district utilized a monthly time step and was based on data from the Census of Agriculture inventories of livestock numbers and land use, from surveys of producers, and from local agricultural experts. Nitrogen losses from manure management included losses to surface water, groundwater, and the atmosphere that were estimated from literature values for various housing systems, manure storage structures, and land application practices (Zebarth et al., 1997). Manure production was estimated for different livestock species (poultry, dairy, swine, etc.) and different production categories (layers, broilers, cows, heifers, etc.) with different excretion rates allowed for each species and production group as estimated from several reports in the literature (Zebarth et al., 1997, 1999). Livestock housing systems were partitioned into the commonly used production units such as free-stall barns, tie stalls, or pasture systems and N loss factors assigned to each type of unit. Manure storage systems were also varied according to common practices for each species and production group, such as earthen lagoons, concrete tanks, or solid storage under shelters. Factors for N losses to surface water, groundwater, and the atmosphere were then selected for each storage system and livestock production category.

The manure remaining after the estimated N losses from housing and manure storage system was assumed to be applied to the agricultural land within the district. Land application practices considered losses to surface water and the atmosphere for various methods of application (surface applied, injected, or incorporated) and the month of application. A noteworthy feature common to livestock systems are the large losses from ammonia volatilization as shown by Zebarth et al. (1999) who estimated the partitioning of the total ammonia emissions from livestock as 44% from housing, 26% from storage, and 30% from land application. A detailed description of the N budget algorithm, the estimation procedures, loss estimates for various management practices, and assumptions are given in Brisbin (1995) and Zebarth et al. (1997).

Aggregating the Soil–Crop Sectors into a Large-Scale Nitrogen Budget

The N algorithm for the soil–crop system utilized Census of Agriculture land use inventories and a root-zone soil depth with N inputs estimated from local manure production, fertilizer, and atmospheric sources. Nitrogen outputs included the harvested crop and denitrification. The typical land uses, as a percentage of the Matsqui South's area, were grass forages and improved pastures 20 to 25%, vegetable and horticulture crops about 60%, and unimproved pasture 10–15%. Manure N inputs were estimated as manure N remaining after housing, storage, and land application losses as described above. Fertilizer N inputs were estimated from the area of the various crop types and the recommended N fertilization rates, or the N fertilization rates used in local practice. Atmospheric inputs were set as proportional to the ammonia-N losses from manure applications and housing and it was assumed that about 65% of the emitted ammonia was redeposited on local land surfaces (Welte and Timmermann, 1987) with deposits to agriculture land being directly proportional to the percentage of the district's land in agriculture. This approach allowed both ammnonia volatilization and redeposition within the district, which produced a net effect that about 20% of the district's ammonia emissions were redeposited on the district's agricultural land (Belzer et al., 1997; Zebarth et al., 1999).

Nitrogen removals were estimated for the harvested portion of the crop based on land areas of each crop, typical yields for the area, and N contents of the harvested portions as determined from direct measurement or literature values (Kowalenko, 1994, 2000). Denitrification losses are the most problematic outputs to estimate. Paul and Zebarth (1997a) reported denitrification from inorganic fertilizer and liquid dairy slurry on corn–silage fields as varying between 9 and 18%, but losses of over 70 kg N ha^{-1} were observed on a poorly drained soil. Differences in soils and hydrology were taken into account by Zebarth et al. (1999) by varying the denitrification rates between 5 and 15% according to soil drainage and hydrologic setting, similar to the approach of Meisinger and Randall (1991). Soil N mineralization was assumed to be balanced by immobilization, i.e., the soil organic N in the root zone was in a steady-state condition.

Calculating the Nitrogen Surplus and Nitrogen Reference Scenario

The final step in the large-scale N budget algorithm combines the livestock and soil–crop components across the district and estimates the N surplus, which is the N remaining after subtracting N outputs from N inputs. It is important to note that the calculated N surplus reflects several key assumptions relating N management in the livestock sector and in the soil–crop sector. Examples of these assumptions include: the N excretions by different species of animals based on conventional diets, the N losses from classes of manure storage (runoff and gaseous losses), the N losses from land application practices (ammonia and runoff), and the N losses from various N rate and timing practices in specific cropping systems. On the positive side, these assumptions also provide a mechanism for evaluating how changes in various management practices will likely affect N surpluses across the district. However, before doing such an evaluation it is necessary to establish a point of reference for these relative comparisons.

The reference scenario for the Matsqui South district adopted the animal populations and species distribution from the 1991 Census of Agriculture, including the estimates of the 1991 animal housing units, the manure storage facilities, and the manure land application practices. The N budget for this reference scenario (Table 13–10) shows an average N input of about 360 kg N ha^{-1} for the agricultural land that was derived mostly from manure additions. Nitrogen outputs were estimated at about 110 kg N ha^{-1}, leaving an N surplus of about 250 kg N ha^{-1} (Zebarth et al., 1998, 1999). The estimated N surplus is well above the desired 50 to100 kg N ha^{-1} that was judged to be the goal for agricultural lands in the district (see previous discussion) and indicates a need to explore various scenarios for improved N management.

Evaluating Nitrogen Management Scenarios

The improved management scenarios that Zebarth et al. (1999) evaluated included management practices for manure, fertilizers, or animal diet that were considered individually and in combination. In the improved manure management (IM) scenario

Table 13–10. Nitrogen budget for the reference scenario of the Matsqui South district estimated by Zebarth et al. (1998, 1999), and the differences between the reference scenario and various improved N management scenarios. All reference scenario values rounded to nearest 10 kg N ha^{-1}. See text for description of management scenarios.

	Matsqui South District Reference scenario (1991 base conditions)	Improved management scenario for components			
		Animal manure (IM)†	Manure and fertilizer (IMF)‡	Animal diet (ID)§	Manure and fertilizer and diet (IMFD)¶
N inputs	kg N ha^{-1}	% increase or decrease relative to reference scenario			
Inorganic	≈90	0	–63%	0	–63%
Manure	≈230	+33%	+33%	–23%	+2%
Atmospheric	≈40	–10%	–3%	–13%	–18%
Total inputs	≈360	+73%	+4%	–16%	–17%
N outputs					
Crop	≈90	0	0	0	0
Denitrification	≈20	+15%	+15%	–15%	0
Total outputs	≈110	+3%	+3%	–3%	0
N Surplus	≈250	+28%	+5%	–22%	–24%

† IM = improved manure management.
‡ IMF = improved manure management plus fertilizer N management scenario.
§ ID = improved diet.
¶ IMFD = improved manure plus fertilizer plus diet scenario.

the manure was kept in an appropriate storage facility, storage capacities were increased to 24 wk, and manure was incorporated into the soil at the optimal time for crop uptake (Zebarth et al., 1997; Brisbin, 1995). Manure N conservation in the IM scenario predictably reduced manure N losses to surface water because all the manure was contained in storage structures. The IM scenario also estimated reduced NH_3 losses due to soil incorporation of manure. However, the reduction in manure N losses resulted in a one-third increase in manure N added to soil (Table 13–10) that contributed to an approximate 30% increase in the N surplus because N removals were virtually unchanged. This illustrates the interaction of N budget components and the need to adjust other N inputs to accommodate increased manure N.

The improved manure plus fertilizer N management scenario (IMF) retained all the practices in the IM scenario, but fertilizer rates were reduced to accommodate the additional manure N. In the IMF scenario, the total N inputs were about the same as the reference scenario (Table 13–10) because the additional manure N was counter balanced by lower fertilizer N inputs. The N outputs for the IMF scenario also remained about the same as the reference scenario, resulting in an N surplus that was similar to the reference scenario. In the IMF scenario for the Matsqui South district, the N surplus remained substantially above the target of 50 to 100 kg N ha^{-1}. The estimated impact of implementing the IMF practices in the Matsqui South district was limited by an overabundance of local manure and the restraints of maintaining a low N-output cropping system.

For the improved diet (ID) scenario, the poultry, dairy, and swine diets were altered to lower animal N excretion rates (Zebarth et al., 1997; Brisbin, 1995). The assumed diet modifications were based on a summary of many literature reports that N excretions can be reduced by removing surplus dietary crude protein, by balancing protein and carbohydrate in the diet, and by balancing amino acids. These dietary N management practices were assumed to be capable of reducing poultry and swine N excretions 25%, and dairy N excretions 20%. The ID scenario reduced manure N production and consequently the manure N input about 25% compared with the reference scenario (Table 13–10). The atmospheric N input was also reduced somewhat due to reduced ammonia emissions from manures with lower N contents. These reductions in manure N and atmospheric N reduced the total N inputs about 16%, or about 60 kg N ha^{-1}, compared with the reference conditions (Table 13–10). The ID scenario reduced the N surplus about 22% compared with the reference scenario, to about 190 kg N ha^{-1}, due to reduced N inputs, however, the N surplus was still several fold higher than the target level of 50 to 100 kg N ha^{-1}.

The improved manure plus fertilizer plus diet scenario (IMFD) assumed that all of improved management assumptions described above were implemented. This scenario resulted in the greatest reduction in surplus N (24% lower) compared with the 1991 reference conditions (Table 13–10). But the N surplus was once again substantially higher than the goal. This indicates that in the Matsqui South district, improving agricultural N management alone should not be expected to bring N losses into an environmentally acceptable level. Consequently, additional measures such as manure export, developing nonagricultural uses for manure, installing manure treatment systems, limiting animal densities, or developing high N removal cropping systems need to be considered in addition to traditional agricultural N management practices.

Extending the District-Scale Nitrogen Budgets to Regional Estimates

The district-scale N budget described above for Matsqui South was also extended to the regional scale by applying the N budget algorithm of Brisbin (1995) and Zebarth et al. (1997, 1999) to the 20 districts that comprise the entire Lower Fraser Valley of southwestern British Columbia. This region contains over 70,000 ha of land, covering a range of livestock types (dairy, swine, and beef), and a wide range of cropping systems (cereal grains, forage crops, improved pastures, and vegetable crops). Extending the N budget approach to larger areas necessarily involves working with data that have highly varying degrees of accuracy, estimating N flows in soil–crop–livestock systems that are not frequently studied, and making a much greater number of assumptions. This inevitably produces less precise estimates, but the approach can provide a broad-spectrum evaluation that is useful for identifying opportunities for N management and can estimate the spatial distribution of surplus N across the region.

Estimating the Regional Nitrogen Budget for the Lower Fraser Valley

The N budget algorithm described above was also applied to each district within the entire Lower Frazer Valley. A full description of the regional-scale N budgeting model is given in Brisbin (1995) and Zebarth et al. (1997, 1999). The N budget model produced estimates of the N surplus for each district in an approach analogous to the one used in the Mataqui South district in Table 13–10.

The results of the N budget model for the Lower Fraser Valley for the 1991 reference scenario showed a wide range of N surpluses, which resulted from the wide range of soil–crop–livestock systems in the region (Table 13–11). The average N surplus across all districts was about 68 kg N ha^{-1}, which is about 25% of the

Table 13–11. Nitrogen budgets (kg N ha^{-1}) for the 1991 reference scenario (see text for details) in the Lower Fraser Valley, British Columbia, using districts categorized by major agricultural activities (Zebarth et al., 1999). Each category contains four districts, values for N Surplus rounded to nearest 5 kg N ha^{-1}.

	Description of district's major agriculture activities				
	>35% horticulture, crops, and low manure	<50% manure from dairy and >15% from nonconfined beef	>50% manure from dairy and <20% from poultry or swine	>50% manure from dairy and >20% from poultry or swine	>60% manure from confined poultry, swine, or beef feedlots
	≈kg N ha^{-1}				
N inputs					
Inorganic	105	117	143	155	115
Manure	35	91	84	144	195
Atmosphere	20	20	20	20	20
Total inputs	160	228	247	319	330
N outputs					
Crop	149	180	200	202	165
Denitrification	6	8	12	22	10
Total outputs	155	188	212	224	175
N surplus	5	40	35	95	155

average total N inputs across the region. However, the standard deviation of the N surpluses across districts was ±63 kg N ha^{-1} (CV of about 95%) indicating a very heterogeneous spatial pattern of N surpluses. Crop N removals were not greatly different across districts averaging 180 kg N ha^{-1} (CV of about 20%), but N inputs from manure were highly variable averaging 110 kg N ha^{-1} and having a CV of 60% (Zebarth et al., 1999). In fact, manure inputs explained 93% of the variation in N surpluses across the 20 districts in the Lower Fraser Valley. Zebarth et al. (1997) concluded that the level of N surpluses in the Valley indicated a substantial potential for root-zone N loss to groundwater and surface water in 1991.

The 20 districts were also grouped into five categories (4 districts in each category) that had similar agricultural activities. This classification of districts (Table 13–11) clearly illustrates that N surpluses, and likely environmental losses, increase primarily in response to increases in manure inputs and animal densities, i.e., an increase as animal numbers increase relative to the area of agricultural land. It is noteworthy (Table 13–11) that districts that had >50% of the land receiving manure from local land-based livestock systems (mainly dairies) but <20% from confined livestock (poultry and swine) had projected N surpluses below 50 kg N ha^{-1}, due to greater crop N removals with forage systems and the lower animal densities with dairies. The districts that had >50% of the land receiving manure from dairies and >20% from poultry or swine had higher N surpluses (about 95 kg N ha^{-1}), despite high crop N removals. Districts that had >60% of the land receiving manure from confined livestock enterprises had the highest N surpluses that averaged about 155 kg N ha^{-1}. The previously discussed Matsqui South district is in this last category that includes livestock systems that rely on imported feed and do not have a local land base for feed production or manure utilization. The N budgeting approach was therefore able to provide a linkage between the N surpluses and animal densities, with intensive animal operations located in areas with limited agricultural land being most problematic. The N budgets also provided semiquantitative estimates of the magnitude of the N surpluses and provided a view of the spatial distribution of the N surpluses across the region.

Evaluating Nitrogen Management Effects for the Lower Fraser Valley

The effect of various manure, fertilizer, and dietary N management scenarios were also evaluated with the regional-scale N budget for the Lower Fraser Valley. Details of these N management scenarios have already been given (see previous section, "Estimating the Regional Nitrogen Budget for the Lower Fraser Valley"). The following discussion will examine the effectiveness of these management strategies by examining the change in N surplus's resulting for their modeled implementation.

The IM scenario resulted in an estimated 50% reduction in N losses associated with manure storage areas, but if implemented alone would increase N surpluses for the root zone by 20% (Zebarth et al., 1999) as previously noted in Table 13–10. This reinforces the need to adjust fertilizer N rates according to N availability from manure. The IM scenario would decrease the estimated emission of ≈7000 t NH_3–N yr^{-1} by about 13%, due to improved storage facilities and soil incorporation of manure. The 50% reduction in N losses from improved manure storage was due to minimizing N losses via surface runoff and leaking storage facilities.

The N budget evaluation of the IMF scenario estimated a substantial reduction in the N surplus for the Lower Fraser Valley over the long-term, from 68 to 5 kg N ha^{-1} of agricultural land, although the three districts with the highest N surpluses still had values well above the 50 to 100 kg N ha^{-1}goal. Zebarth et al. (1997, 1999) also noted that achieving the benefits of the IMF scenario would require transporting manure throughout the region, equipping farms for manure use, and developing better diagnostic tools to manage manure N in place of fertilizer N. Zebarth et al. (1997) estimated that only about one-half of the potential benefit from the IMF scenario could be realized with acceptable costs to the producer, the remaining benefits would require larger investments and a longer time-frame.

The improved diet (ID) scenario reduced manure N production, which in turn decreased the N surplus in the Valley from 68 to 45 kg N ha^{-1}, with much of this reduction occurring through an estimated 23% reduction in ammonia emissions. The effects of the ID scenario were largest in districts with concentrated animal operations. A 25% reduction in N excretion was considered achievable without a loss in animal productivity and with little increase in producer costs (Zebarth et al., 1997). Thus, although the ID approach did not decrease N surpluses as much as the IMF, the ID scenario involves little direct investment in equipment and could be easier to fully implement than the IMF that requires larger long-term investments. Furthermore, the ID approaches would be directed toward those districts with the largest N surpluses and therefore would produce the greatest benefit in reducing potential N leaching over the whole Valley.

Combining all the improved N management scenarios (IMFD) resulted in a negligible N surplus relative to 1991 practices when averaged over the entire Lower Fraser Valley. However, the three districts with the largest 1991 reference scenario N surpluses still had substantial N surpluses, Matsqui South's being about 180 kg N ha^{-1}, but these surpluses were diluted when considering the average over the entire Valley.

Overview on Estimating Regional Nitrogen Budgets

Estimating large-scale N budgets presents both benefits and difficulties for scientists, farm managers, and policymakers. We will briefly discuss some of the supporting and dissenting views for these types of studies.

The very nature of estimating N budgets for large-scale systems is filled with difficulties because of the highly variable flows of N within soil–crop–livestock systems. These flows dictate the fate of N and are affected by many interacting factors such as weather, cropping system, N management practices, soil properties, and livestock husbandry practices—in fact, when one stands back and reviews the list of difficulties it is easy to understand why many have concluded that it is too difficult and fraught with uncertainty to pursue. In addition to the uncertainties about basic N flows, there are also difficulties of incomplete data describing large-scale agriculture activities, e.g., accurate farm fertilizer N rates, accurate manure rates, and accurate feed composition data. The collection of actual field-level data on N inputs with a consistent protocol over an extended period would greatly improve the estimation of large-scale budgets and would allow more accurate tracking of changes in agricultural practices.

On the other side of the issue, agricultural scientists are increasingly being asked to extend their basic process-level knowledge to real-world problems. Most real-life decisions are made with imperfect knowledge, and rely on moving in the right general direction rather than knowing the precise effect of a specific policy or the most efficient path toward a goal.

Therefore, large-scale N budgets are most useful for identifying the major N flows within an area, for locating areas of potential high N loss, for defining the spatial pattern of areas at risk for N loss, and for estimating if moderate N surpluses on individual farms could accumulate to produce a large regional surplus. This knowledge can define options for improving N management. The nature of the agricultural N cycle and the heterogeneity of factors affecting N transformations dictate that all large-scale N budgets will contain inaccuracies, varying degrees of questionable assumptions, and will be based on incomplete data. The results of large-scale budgets should therefore be considered as semiquantitative, at best. But, large-scale budgets derived from well documented budgets on smaller sectors are a good tool for identifying the major N processes, for focusing evaluations into a whole-system mode, and for identifying the general paths for improving N use efficiency in agriculture.

Summary

Nitrogen budgets have been used for over 170 yr to estimate the size of various N pools, N gains from the atmosphere, N losses to the environment, and to study the interactions among soil N cycle processes. A major advantage of an N budget is that it calls for a "systems approach" that requires the identification and estimation of the major N cycle processes, and their interactions, for a defined system.

The early N budget of Boussingault (1841) identified the importance of legume N additions to cropping systems, with alfalfa estimated to add about 140 kg N ha^{-1} yr^{-1}. Research by Lawes and Gilbert at Rothamsted provided the earliest long-term data on the N response of winter wheat (Lawes et al., 1882) and several treatments within the Broadbalk Winter Wheat Experiment have now continued for more than 140 yr. The most up-to-date total N budget using modern (1990–1997) production practices for the Broadbalk plot receiving 184 kg N ha^{-1} of total-N annually (144 kg fertilizer-N ha^{-1} yr^{-1}) partitions the total-N inputs into crop removals in grain plus straw of 70%, leaching losses of 12%, and gaseous losses of 18%.

Nitrogen budgets are based on the conservation of mass, with the deceptively simple statement that N inputs minus N outputs, equal the change of N within the system. The main N inputs and N outputs that are needed for constructing N budgets are described throughout this monograph, but the change in the soil organic N component is particularly difficult to estimate.

The consistent application of the same management practices over many years will cause an ecosystem to gain or lose N at a diminishing rate, until a quasi steady-state N level is reached (Jenny, 1941). Under steady-state conditions, the average N mineralized from organic N is equal to organic N returned in aboveground residues, roots, root exudates, and new soil microbial biomass. Significant changes (e.g., >15 kg N ha^{-1} yr^{-1}) in soil organic N are common with major changes in land management, such as tillage of grassland, reversion of farmland to woodland, or initiation/cessation of manuring. These changes should always be taken into account in drawing up N budgets, despite the difficulties of measuring the changes in soil organic N and soil bulk density. However, in many long-term budgets the annual changes in organic N are relatively small (e.g., <15 kg N ha^{-1} yr^{-1}) compared with the uncertainties in other N budget components, such as N_2 fixation or denitrification, so that approximate estimates can be used without great error. The appropriateness of a steady-state approximation in an N budget will depend on the desired precision of the budget, the size of the anticipated change in soil N, and the uncertainties in other N budget processes.

Nitrogen budgets have traditionally been based on the total N entering and leaving a system, but the past 20 yr has seen a proliferation of ^{15}N budgets; these two N budgeting approaches are not equivalent. The total N budget focuses on the total N inputs and losses of the entire system, while the labeled budgets focus on the fate of the ^{15}N including the ^{15}N's interaction within the soil N cycle. The interaction of ^{15}N with the soil N cycle can produce an ANI that can arise whenever both unlabeled N and labeled N are present in the same N pool in the same chemical form, at the same time. An ANI can be positive or negative, and can be real (e.g., expanded root depth due to fertilization) or apparent (e.g., arising due to N pool substitution). The choice of a total N budget or a ^{15}N budget will depend on the studies objectives and available resources, but careful consideration should be given to the fundamental strengths and weaknesses of each budgeting approach.

The results from a ^{15}N budget contributes to a greater understanding of N cycling from the labeled source, but can also contribute to the understanding of N flows for a conventional total N budget. However, several important assumptions need to be met to integrate ^{15}N budget data into a conventional N budget. These assumptions are: that the fractional recovery of N in crop plus soil is similar for all N inputs into the soil–crop system, and that the fractional recovery of N in the soil is the same for all incoming N retained by the soil. These major assumptions should be evaluated with great care before applying them, and their resulting equations, to a soil–crop N budget. However, with carefully organized studies using ^{15}N additions at key points in the soil–crop–hydrologic cycle, it is possible to integrate the approaches into an expanded N budget. This has been shown by the development of Fig. 13–3 for the 144 kg fertilizer-N ha^{-1} yr^{-1} treatment of the Broadbalk Winter Wheat Experiment.

Nitrogen budgets from several studies have been described in this chapter and illustrate that the final budget represents the product of numerous transformations performed by physical, chemical, and biological agents interacting with each other and the environment over time. The major N budget processes are usually crop N uptake, leaching or the accumulation of residual N, and gaseous losses through denitrification and ammonia volatilization. An important N budget principle is that crop N use efficiency can be rather high (70–80%) if the crop growth is increased by the N inputs, and the N is applied below the soil surface and in-phase with crop demand. An important corollary to this principle is that N losses increase rapidly once N inputs exceed crop assimilation capacity with lost N usually accounted for as increased leaching, denitrification, or an accumulation of residual nitrate. Nitrogen leaching losses are commonly 10 to 30% of total N inputs, but depend on soil nitrate content, quantity of surplus water (water inputs vs. evapotranspiration), soil texture and rooting depth, and pattern of water movement (preferential flow vs. complete displacement). Gaseous N losses to denitrification are highly variable but are commonly 5 to 25% of total N inputs, with losses depending on nitrate concentration, oxygen demand, available C, and temperature. Gaseous losses to ammonia volatilization are also highly variable with high losses of 10 to 25% being common for surface-applied manures or urea containing fertilizers, and small losses of less than 10% common for immediately incorporated N sources. Several studies have also shown the rapid stabilization of labeled N after it is converted to organic forms.

Large-scale N budgets, derived from documented smaller-scale budgets, have proven valuable for identifying the major N pathways and the spatial pattern of N surpluses. Large-scale budgets, particularly whole-farm budgets, have also proven valuable for evaluating scenarios for improving N recoveries within the soil–crop–animal system.

Soil N budgets have challenged generations of soil scientists, and will continue to challenge future generations of scientists by slowly revealing fundamental principles that are woven within a matrix of contrasting results and the inevitable variability of biological systems. By understanding these principles and the factors influencing them, scientists with have a stronger foundation for improving N use efficiency and concurrently reducing N losses to the environment.

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