A crucial element in understanding current environmental issues is knowledge about regional balances of nitrogen. Accounting for nitrogen on a regional scale most closely represents the impact of terrestrial processes on aquatic and marine systems and relates land surface processes to atmospheric dynamics. Constructing regional balances presents a set of challenges qualitatively different from site-level budgets (Robertson 1982, 1986, Robertson and Rosswall 1986). First, very few data are available at the regional scale or have an appropriate spatial distribution such that they can be generalized to the regional scale. Second, it is not always simple to scale such data to the regional level, because many processes are difficult to assess, such as lateral or vertical transport processes (e.g., erosion, human transport of materials) or various land uses.

One of the major impacts of humans on regional nitrogen (N) balance is by means of agricultural practices (NAS 1978, Howarth et al. 1996, Jordan and Weller 1996). There have been numerous studies in the United States of the impact of grazing and cultivation on N pools and processes at the site level (Haas et al. 1957, Floate 1981, Aguilar et al. 1988, Detling 1988, Paul et al. 1996, and many others). All of these studies have demonstrated that cropping practices, much more than grazing, result in substantial alterations of N budgets, through fertilization, enhanced N cycling, enhanced N trace gas losses, and increased nitrate leaching. Relatively few studies have evaluated regional-scale consequences of cropland management (Miller and Smith 1976, Keeney 1979, Robertson and Rosswall 1986), because of the challenges of assessing processes at large spatial scales.

In this article, we provide a regional assessment of the N balance for the central grassland region of the United States. We focus on this region because it is a key area of US crop production, and it has been the focal point for several long-term regional studies. The data that have been collected are available in a variety of forms, including simulation analyses (Parton et al. 1987, Burke et al. 1991, 1997). We combine individual site data on processes, regional analyses of land use and soils, and historical data to assess patterns of nitrogen fluxes through time and space. A key value of a regional analysis is that it can identify the spatial patterns within which the individual site data sets are consistent.
specific processes occur. Our goal is to determine which processes and which subregions are of most concern at this time, rather than to construct a detailed regional budget.

**Description of the region**

The central grassland region of the United States (figure 1) encompasses some or all of 14 states, covering approximately $2.21 \times 10^6$ square kilometers ($\text{km}^2$) east of the Rocky Mountains and west of the Mississippi River, between the Canadian border and central Texas (Lauenroth et al. 1999). The region is characterized by strong environmental gradients, with precipitation varying from 300 millimeters (mm) annually in the rain shadow of the Rocky Mountains to over 1000 mm in the eastern portion of the region (figure 1). Mean annual temperature varies with latitude from less than 4°C in the north to greater than 18°C at the southern boundary. Before the arrival of European settlers, the vegetation consisted of native perennial grasslands. Since the early 1800s, much of the region has experienced conversion from perennial grasslands to annual crops; currently 64% of the region is cultivated, and most of the balance is used as rangelands for livestock production (Lauenroth et al. 1999). Cropping is most dominant in the eastern portion, where enough precipitation reliably falls for sustainable crop production (Lauenroth and Burke 1995, Burke et al. 1998).

The central grassland region contains the major crop-producing areas of the United States (figure 2). Approximately 75% of US land devoted to growing wheat is located in this region, 65% of the land used to grow corn, and 60% of that in soybeans. Management practices vary across the region, in accordance with average precipitation, temperature, and soil variation (Burke et al. 1998, Lauenroth et al. 2000). The western portion is managed for livestock production and winter wheat, except where irrigation water is available to support corn and cotton; as average annual precipitation increases in an easterly direction, wheat becomes an important crop. At about 800 mm average annual precipitation, corn and soybeans begin to dominate. Below about 500 mm average annual precipitation, winter wheat is grown with a summer fallow rotation (Lauenroth et al. 2000). Wheat and corn are fertilized with N across the entire region. Cotton is grown over relatively small areas of Texas, Oklahoma, and New Mexico, which nonetheless represent 45% of all US land planted in cotton.

**Sources of data for a regional analysis**

Several different types of data are available for assessing the regional N balance of the central grassland region. First, there are several sources of generalized, complete-coverage spatial data, such as state- or county-level information on crop fertilization and harvest yields (USDA 1995), remote-sensing imagery of land use (Loveland et al. 1991), and soil association data (USDA 2002a). Second, research data are available for various networks of sites within the region (summarized recently by Paul et al. 1996). These are data for individual sites that may be generalized to the region for dense networks, such as the National Atmospheric Deposition Program (http://nadp.sws.uiuc.edu/) or the National Weather Service (www.nws.gov).
For the less dense networks, such as the USDA-and state-sponsored agricultural experimental stations, the data may be used to generalize about the controls over ecosystem processes, to generate statistical models (Sala et al. 1988, Burke et al. 1989), or to develop or test ecosystem simulation models that may then be applied across the entire region. It is not possible to obtain continuous records for all the relevant data across large regions. Because of this, both statistical modeling and simulation modeling have played important roles in regional assessments to date (e.g., Running et al. 1989, 1994, Burke et al. 1990, 1991, 1997, Aber et al. 1997). For such analyses, adequate data must be available to test the models and to provide input for simulations of the region. In this article, which focuses primarily on assessing important N-balance processes, we utilize existing data of all types and provide a statistical assessment using large-scale data. Similar assessments using simulation analyses are available (Burke et al. 1990, 1997).

Nitrogen budget

Inputs. Atmospheric inputs of nitrogen vary with precipitation and range from 1.5 to 8 kilograms of N per hectare per year (kg N·ha⁻¹·y⁻¹) wet deposition (NADP 1998). Rates and amounts of dry deposition are not known but are probably similar (Holland et al. 1999). Deposition is probably much higher in the local areas surrounding livestock feedlots, where ammonia volatilization is very high.

A much more significant source of nitrogen for the central grasslands region is the direct addition of nitrogen to agricultural systems through fertilization. The majority of the land in wheat, corn, and cotton is currently fertilized with nitrogen (USDA 2002b), but a substantially lower proportion of land in soybeans (17%) receives fertilizer, because soybeans are an N-fixing crop. The levels of N fertilization have varied through time and across crops, with gradually increasing levels of fertilizer applied since about 1950, when industrially produced fertilizer was introduced. Between 1964 and 1976, the average rate of N fertilization for corn, the predominant crop in the eastern half of the region, doubled from about 75 kg N·ha⁻¹·y⁻¹ to nearly 150 kg N·ha⁻¹·y⁻¹ (figure 3). Corn receives the highest levels of fertilizer, followed by cotton, because these crops are grown under conditions of high water availability and therefore nitrogen has a high likelihood of limiting productivity. Wheat, which is most frequently limited by water availability (Lauenroth et al. 2000), receives fertilization at rates currently averaging about 60 kg N·ha⁻¹·y⁻¹. The states that have the longest growing season and the highest moisture availability have the highest rates of N fertilization for wheat.

Internal recycling. Both cultivation and grazing substantially increase the turnover rate of nitrogen within ecosystems (e.g., Floate 1981, Burke et al. 1996, Paustian et al. 1996, Knapp et al. 1999). Cultivation initially increases the amount of nitrogen mineralized from organic matter (Schimel 1986, Ihori et al. 1995), as a result of breaking down aggregates and exposing them to higher rates of mineralization and of mixing residues into soils; subsequently, N turnover is reduced as the pool of organic matter declines. Grazing directly affects the N cycle by removing plant biomass and by returning a significant proportion (50% to 75%) as urine and feces (Dean et al. 1975, Lauenroth and Milchunas 1992).

Increases in the rate at which N is cycled may result in increased N losses from ecosystems. Nitrogen losses most often occur from inorganic forms, and higher turnover rates result in more flow through the inorganic compartments.

Outputs. There are several important losses to consider when constructing a regional N budget. First, crop and livestock removal results in N removal from the region, or its redistribution. The ultimate fate of such organic matter removed from croplands or rangelands is varied; grain may be fed to humans within or outside of the region, and a high proportion is fed to livestock in feedlots within the region. Any type of feeding results in large N losses as urine and feces, which may be routed to septic systems or sewage treatment plants (in the case of humans) or feedlot manure storage and reapplication systems (in the case of livestock), lost to the atmosphere through ammonia volatilization, or leached into surface or ground water. In our analysis, we provide a broad picture of the effects of crop and livestock removal and do not
attempt to trace the N further. Because direct removal of livestock biomass has been estimated to represent a very small N export (approximately 0.6 kg N·ha⁻¹·y⁻¹ in tallgrass prairie) (Parton and Risser 1980), we focus our analysis on crop removal.

Important losses of N from croplands and rangelands occur through several other mechanisms, including volatilization upon burning, nitrate leaching, trace gas losses such as nitric and nitrous oxide production, ammonia volatilization, and erosion. Interestingly, regional data on such processes do not exist, though individual studies have been conducted to estimate site-level losses of N through such vectors. We do not attempt to provide a detailed review of the site-level studies that have been conducted, but we do present boundary values for the possible rates across the region. We do not consider erosion specifically, because this may represent more of a redistribution than a loss at the regional scale.

**Crop removal.** Removal of N in crops represents a major output and is very likely the largest loss from cultivated lands on an annual basis. Losses of N in crops may be calculated from yield data and average N concentrations for various grains or forage. For the major crops within the central grasslands region, N removal is highest for corn and soybeans (grown under highest water availability), crops that have high grain yield and, in the case of soybeans, very high N concentration. Our calculated average N losses for grain or silage removal vary between 25 and 150 kg N·ha⁻¹·y⁻¹, with values exceeding 200 kg N·ha⁻¹·y⁻¹ for some states for corn silage and cotton (figure 4). Important sources of uncertainty in these numbers are their high variability and the lack of data regarding return of residue to fields.

**Nitrate leaching.** There are relatively few data on regional losses of N by means of leaching in the central grassland region, other than through interpretation of large-scale watershed data (Howarth et al. 1996) or through extrapolation of individual site data on nitrate leaching. Even at the site level, investigators often identify rates of nitrate leaching as one of the unknowns (including volatile losses) needed to balance N budgets (Puckett et al. 1999). Site data indicate that summer fallow–winter wheat systems may contain between 70 and 215 kg N·ha⁻¹ as nitrate in the soil profile during the summer fallow phase (Wood 1990, Campbell and Zentner 1993, Evans et al. 1994, Jones et al. 1995); this nitrate is vulnerable to leaching, but in the low-precipitation zones of the summer fallow region, it is unclear what the average or maximum leaching rates are. Up to an order of magnitude more N is lost through leaching under wet conditions and from feedlots compared with the dry conditions that prevail in the western portion of the region (Stewart et al. 1968). Leaching is much more likely to be important in the wetter, eastern portion of the region. David and colleagues (1997) es-
timated that as much as 25 to 44 kg N·ha⁻¹·y⁻¹ may be leached annually from corn and soybean watersheds in Illinois; losses were highly dependent upon conditions associated with individual years, with highest losses associated with high precipitation and fertilization.

**Volatile losses of nitrogen.** Nitrous oxide (N₂O) and nitric oxide (NO) are both volatile gases that, when lost to the atmosphere, remove N from the soil. Regional data are not available for assessing volatile N losses from the centralgrassland region; however, site-level data indicate that many land-use management practices, including grazing by livestock, burning of tallgrass prairie, and cropping, increase N volatilization.

The production of N₂O and NO is increased dramatically by fertilization, irrigation, and cultivation (Hutchinson and Mosier 1979, Mosier and Hutchinson 1981, Mosier et al. 1986, 1991, 1997, Bronson and Mosier 1993, Thornton and Valente 1996, Veldkamp and Keller 1997). Rates of N₂O production vary from 0.1 to 1.0 kg N·ha⁻¹·y⁻¹ in rangelands to 0.4 to 2.0 kg N·ha⁻¹·y⁻¹ in dryland wheat to 3 to 6 kg N·ha⁻¹·y⁻¹ in irrigated systems. The increased N losses in cropped lands are the result of enhanced nitrification and denitrification. Fermentation increases ammonium content, tillage increases microbial activity, and irrigation enhances denitrification (Davidson and Kingerlee 1997). Few data are available on the production of NO (lost through the process of nitrification) in agricultural systems (Thornton and Valente 1996, Davidson and Kingerlee 1997, Jambert et al. 1997, Skiba et al. 1997); however, the importance of NO relative to N₂O decreases with increasing anaerobic conditions (high moisture or fine textured soils). Data on the shortgrass steppe from Mosier and colleagues (1997) suggest that, in nonirrigated systems (either rangeland or dryland wheat), NO losses may be 5 to 10 times that of N₂O, or 1 to 5 kg N·ha⁻¹·y⁻¹. Emission of dinitrogen (N₂) is small in dry areas, because denitrification events are generally of limited importance in N budgets where water is limiting (Parton et al. 1988). NO flux is likely to be similar to N₂O flux in irrigated systems (3 to 6 kg N·ha⁻¹·y⁻¹) (Hutchinson and Brams 1992). Total N losses through nitrification and denitrification are small, less than 10 kg N·ha⁻¹·y⁻¹, in well-drained irrigated soils (Mosier et al. 1986), but they are most likely higher in irrigated soils that have lower infiltration rates (Ryden and Lund 1980).

Ammonia volatilization from fertilizers and ammonification may also be significant, accounting for as much as 1 to 5 kg N·ha⁻¹·y⁻¹ (Bowman et al. 1997). In grazed systems, ammonia volatilization from urine may be as high as 50% of total N in urine and very variable across landscapes. Such losses have been estimated to be about 0.7 kg N·ha⁻¹·y⁻¹ in the shortgrass steppe (Schimel et al. 1986).

Direct N losses from burning of tallgrass prairie may result in losses of 12 to 30 kg N·ha⁻¹, representing 63% to 89% of the N in aboveground biomass, per burn (Ojima et al. 1990). The importance of fire to regional N budgets is determined by the intensity and frequency of burning and the areal extent of tallgrass prairie. Currently, little tallgrass prairie remains, with most having been converted to cropland (Lauenroth et al. 1999).

Our estimates of total volatile losses of N are 2 to 7 kg N·ha⁻¹·y⁻¹ in western rangelands, 2.5 to 9 kg N·ha⁻¹·y⁻¹ in dryland wheat systems, and 7 to 17 kg N·ha⁻¹·y⁻¹ in irrigated crops or crops in the wet, eastern part of the region. Losses of N in tallgrass prairie depend upon the fire management strategy.

**Controls over nitrogen retention in the central grassland region.** One of the most important predictions that can be made at the regional scale is that any process that causes carbon losses, by means of reduced production or increased decomposition of organic carbon (C), will result in N losses (Asner et al. 1997). The ratio of N to C in organic matter is relatively fixed in ecosystems (Redfield 1958, Reiners 1986). If insufficient organic matter is available to retain N, inorganic N accumulates and is subjected to numerous vectors of loss, including denitrification, ammonia volatilization, and nitrate leaching. Any management practice that results in losses of organic C from soil causes net ecosystem N losses as well. The ability of an ecosystem to retain N from anthropogenic inputs, such as atmospheric deposition or fertilization, depends on the system’s C balance. In addition, it has frequently been shown that increases in N result in higher capability of agricultural systems to maintain soil C (Paul et al. 1996); management of C and N are interdependent.

Two questions remain for an assessment of the regional N budget of the central grassland region. First, of the original, or native, N present in these systems, how much remains? If we can answer this question, then we can estimate the total amount of N that has been released from the region (native plus anthropogenic); such a historical analysis can also provide a baseline for us to assess the influence of current management practices. Second, what are the direct anthropogenic effects on N balance in the region on an annual basis, and how much N is exported either to the atmosphere or to aquatic systems through runoff and leaching? To address these questions, we have conducted two straightforward analyses.

**Historical changes in native nitrogen.** Over the long term, historical cultivation has resulted in large losses of native soil C and N across the grassland region (Haas et al. 1957, Aguilar et al. 1988, Burke et al. 1989). Soils represent the largest storage pool for biologically important elements in native grasslands worldwide (Ajtay et al 1979); thus, the data suggest a significant long-term loss of N. Burke and colleagues (1989) utilized soil C and N data from a USDA Natural Resource Conservation Service database for over 800 rangeland and cultivated soils to develop a statistical model of C and N losses across the central grassland region (figure 5). Similar analyses have been conducted with a simulation model, with comparable results (Burke et al. 1997). The data and statistical model suggest that highest absolute N losses have been experienced in the areas with highest initial soil organic matter, which are those areas with high precipitation and...
thus high net primary production, and thus low decomposition. Across the region, this translates into highest losses in the northeastern section. Losses of N from cultivated fields since settlement average 1080 kg N·ha−1, or 30.8%. Evaluated over the entire region, including native rangeland areas, which we assume have had a balanced N budget (Milchunas and Lauenroth 1993), this represents regional losses of approximately 20% of the original N capital, or $1.69 \times 10^{11}$ kg N over 75 to 100 years of cultivation. This amount of N is roughly equivalent to annual levels of terrestrial biological N fixation globally (Schlesinger 1997).

**Current nitrogen balance of the region.** Are losses of native nitrogen still occurring across the region? It is difficult to know whether the region still has a net negative N balance. Many data suggest that the major losses of organic matter due to cultivation occur during the first 10 to 20 years (Haas et al. 1957, Jenkinson and Rayner 1977), and as most of the area has been cultivated for much longer, it is likely that losses of native N have decreased substantially.

How are direct anthropogenic effects such as fertilization and crop export influencing the annual N balance of the region? Data for 1996 (for Kansas only; data are missing for either crop yields or fertilization usage for other states) suggest that approximately 25 kg N·ha−1·y−1 more N is added as industrial fertilizer than is removed as crops, for wheat, corn grain, and corn silage (figure 6). For soybeans, N removal exceeds N additions by more than 100 kg N·ha−1·y−1; this is accounted for by biological N fixation by the crop, so that the soy system is not in a negative N balance. Including N inputs from precipitation, a minimum estimate of the average amount of N added to cropped systems in the central grassland region is $35$ kg N·ha−1·y−1.

What is the ultimate fate of this added N? There are relatively few data to help us answer this question, particularly at the regional scale. The land uses at particular locations may be increasing N capital due to additions, keeping N losses balanced with equal inputs, or decreasing N capital as a result of losses exceeding inputs. Isotope labeling experiments suggest that half of the N added as $^{15}$N-labeled inorganic fertilizer remains in the ecosystem; however, these results are insufficient for estimating the total N losses from the ecosystem, because different N isotopes may be lost at different rates. Experimental data have indicated that minimum tillage or no-tillage systems and a high level of rotation with N-fixing species can result in accumulation of soil organic matter and, hence, N under continuous cropping (Haas et al. 1957, Campbell et al. 1996); it is unclear how common this management strategy is across the region. The summer fallow wheat systems of the drier portion of the region are unlikely to be accumulating N for two reasons. First, even data from no-tillage management suggest that summer fallow systems do not experience a significant increase in total N (Wood 1990). Second, more than 95% of farmers using summer fallow rotations utilize tillage, rather than herbicides, as a weed control practice (USDA 1995).

![Figure 5. Estimates of precultivation soil nitrogen (N) and N loss due to cultivation for the Great Plains. Based on Burke et al. (1989).](https://academic.oup.com/bioscience/article-abstract/52/9/813/248758)
Tillage increases turnover of organic matter, decreases organic storage of N, and increases N losses (Mosier et al. 1991). If we assume that, on average, ecosystems of the central grassland region are currently balanced with respect to total organic matter and thus N content, this suggests that a minimum of 25 to 35 kg N·ha⁻¹·y⁻¹ is lost from the wheat and corn systems each year (fertilizer addition plus atmospheric deposition minus crop removal). This N may be lost by means of ammonia volatilization, NO or N₂O losses, or in surface runoff and nitrate leaching. We anticipate the highest losses in sites where atmospheric deposition of N is high (eastern section); fertilization rates are high (corn crops, mostly in the eastern section); storage potential of soil organic matter is low (highly tilled soils, sandy textures); and precipitation or irrigation results in wet soil conditions, which increases rates of volatile N loss, surface runoff, and leaching. The partitioning among these losses is very important for understanding the N balance of the central grassland region and its influence on greenhouse gas concentrations, photochemistry of the atmosphere, and N enrichment of surface waters, but data are insufficient at this time for either constructing a regional budget or partitioning losses.

We can estimate the N allocation to livestock and humans if we assume that all N removed in crops is consumed by humans or livestock, and that biomass of humans and livestock remain relatively constant. Under these conditions, all N removed in crop yield is converted to metabolic wastes. The average amount of N removed is approximately 75 kg N·ha⁻¹·y⁻¹ for all of the cropped land within the region (64% of 2.210 × 10⁶ km²), or about 1.1 × 10¹⁰ kg N per year. This N may be exported from the region for consumption elsewhere, or it may be stored within the region as biosolids, lost through volatile fluxes from feedlots or sewage treatments plants, or lost to aquatic systems through overland flow or septic or sewage treatment loss.

**Case study from Rooks County, Kansas**

One tool that may help ecologists construct regional N budgets for the central grassland region is the use of agricultural census data to construct county-level nutrient analyses that can then be applied to entire regions. It is possible to evaluate nutrient flows at the county level and then to apply the same methodology to any county in the United States to assess regional trends in nutrient management. Good historical data are available at the county level for the entire country, beginning as early as 1850, in the US Census of Agriculture. These data are now readily accessible for all counties in 12 Great Plains states (Gutmann et al. 1998). Here we present an example using data for Rooks County, Kansas, from 1910 and 1978, to assess human-mediated impacts on the county N balance.

Homesteaders began to settle Rooks County, in west central Kansas, in the 1870s. They planted a mix of winter wheat and corn. Since the 1930s, drought-hardy sorghum has replaced corn, and the cash grain crop has been grown on a summer fallow rotation. By 1920 all of the arable land in the county—about 50% of its total area—had been plowed and cropped. The remaining 50% of the county is in native grass used to graze cattle. The even division between cropped land and grazed land has remained quite steady for the past 75 years.

**Assumptions.** The purpose of this analysis is to investigate county-level estimates of human-mediated changes to the N budget, not to be an all-inclusive model of N flux for the system. The area is divided into two separate N systems, one for cropland (tables 1 and 2) and one for rangeland (table 3). Because the census did not report total area of land in crops prior to 1925, the crop estimates are the sum of the hectares of the 14 most important crops (including hay) in the county. We assume all farmland not in crops to be rangeland. The unit of analysis is the county’s layer of topsoil. We assume that all crops harvested were exports from the soil, whether shipped out of the county or fed to livestock locally. Crop N fed locally to livestock resulted in the generation of manure N that we treat as an input into the soil. It is possible to consider both crop exports and manure N to be local recycling within a region, but to be consistent, we elect to treat those as fluxes out and in. Crop residues represent an internal flux, and these are counted as neither exports from nor inputs into the soil.
Seed planted each season represented an N input into the soil. We include only input and export parameters that can be estimated with a degree of certainty; we therefore exclude important inputs such as atmospheric N deposition and important exports such as leaching and denitrification, because there are no empirical data from this county for those fluxes.

**Inputs.** Manure application and crop rotation with cultivated legumes were the main methods of N manipulation available to farmers before the advent of industrial fertilizer. We utilize the census data of number of livestock and acres of cropland, along with the N content of different manures (MWPS 1985), to estimate the maximum amount of N that farmers in Rooks County could add to their cropland. We assume the manure accumulated in open lots and, when applied to crops, was broadcast without immediate incorporation into the soil, practices that would result in an estimated 47.5% reduction in manure N due to volatile losses of ammonia (MWPS 1985, Tisdale et al. 1993). Using legume fixation rates (Tisdale et al. 1993), the area of land planted in legumes, and the N content of legume crops to estimate N losses from harvest, we calculate the net N input from symbiotic N fixation. It is likely there are differences between modern legume fixation rates and the rates that occurred during 1910 because of differences in both the legume varieties used and soil N availability; however, we can devise no other method of estimating this input. Using seeding rate estimates from Hutcheson et al. (1936) for 1910 and Miller (1971) for 1978, and estimates of N in crops (NRC 1982), we estimate the amount of N returned to the soil through seeding. Industrial fertilizer inputs for 1978 are calculated from fertilizer expenditure and application information for Rooks County in the 1978 Census of Agriculture and from proportional fertilizer applications rates outlined in Miller (1971).

**Exports.** We treat harvested crops as exports from the soil. Part of the crop was shipped out of the county, while the remainder was fed locally to livestock, in which case N reentered the soil as a separate input. On rangeland, the main N losses were from livestock sold and manure volatilization. The N content of livestock is calculated from Frissel (1978), and we assume N losses due to manure volatilization to be 22.5% (MWPS 1985).

**County budget.** A comparison of the N budgets from 1910 and 1978 for Rooks County, Kansas, shows significant human-mediated changes in N distribution. In 1910, there was a deficit in the anthropogenically controlled N balance on the...
cropland, resulting in a county-level loss of 3.8 kg N per ha for croplands. The analysis suggests a net anthropogenic addition of 32.6 kg N per ha in 1978 due to the application of industrial fertilizers and symbiotic N-fixing crops. The N balance for both years was slightly negative on rangeland, although 1978 was more negative than 1910 because significantly more livestock were raised. This method of calculating human-mediated changes to county N budgets—using the agricultural census—can be scaled to entire regions, providing important insight into large-scale N fluxes over time.

**Benefits of regional analysis.** What do we learn from the regional analysis that was not previously known from site-scale studies? The simplest answer is that regional analysis places individual site studies into a context that is related to their regional importance. For example, knowing the magnitude of a specific N loss from summer fallow systems for raising wheat has little regional significance in the absence of knowledge about the spatial importance of wheat cultivation in the region. In the most straightforward case, knowing a spatial pattern such as land use tells us how to weight the processes that occur in each land use type to calculate the regional importance of the processes. For example, loss of N to fire in tallgrass prairie very likely has little regional consequence because of the limited extent of remaining tallgrass. Understanding the regional importance of processes helps scientists understand how to allocate their efforts to improve knowledge about, or implement changes in, regional N dynamics.

Although our initial analysis of N in the central grassland region of the United States is far from demonstrating the full benefits to be gained from regional analysis, we have identified the key spatial patterns in climatic and surface features that affect N balance, the important sources of data that are crucial for a full regional N budget, and perhaps most important, the gaps in our knowledge that preclude a complete analysis.

**Regional data gaps**

There are several key data gaps that must be filled before we can complete a regional N balance for the central grassland region. The two largest unknowns are trace gas losses and nitrate leaching. In a review of the uncertainties in nutrient budgets, Oenema and Heinen (1999) identified spatially and temporally variable fluxes as the important unknowns limiting assessments of nutrient balance. A network of sites with process-level data representing the current management practices across the region would contribute substantially to improving the regional budgets. Because land-use management practices (fertilization, tillage, crop, irrigation practice and amount) are the keys to control of N balance, high-resolution spatial data on land-use management are crucial. Such information may guide decisions to decrease excess N applications to ecosystems and increase nutrient use efficiency.

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