ABSTRACT

Title of Dissertation: WATER QUALITY IN MANAGEMENT INTENSIVE GRAZING AND CONFINED FEEDING DAIRY FARM WATERSHEDS

Rachel Esther Gilker, Ph.D., 2005

Dissertation Directed By: Professor Ray R. Weil, Natural Resource Sciences and Landscape Architecture

Dairy farm size has increased in the United States, while the profit margin has decreased. An alternative to confined feeding dairy farming is management intensive grazing (MIG), a grass-based system relying on rotational grazing for most of the herd's dietary requirements. Previous research has measured high levels of nitrate leaching under MIG, citing the liquid nature and high nitrogen (N) content of urine. However, this research included heavy N fertilizer applications or was conducted on monolith lysimeters with artificial leaching processes and did not accurately represent mid-Atlantic MIG dairy farms. Phosphorus (P) losses have typically been attributed to runoff and erosion but are now being ascribed to leaching as well. To measure the magnitude of N and P losses to groundwater, we sampled shallow groundwater and pore water on one confined feeding and two MIG-based Maryland dairy farms between 2001 and 2004. Transects of nested piezometers and ceramic-tipped suction lysimeters were installed in two watersheds on each farm. Two streams running through two of the grazed watersheds were also sampled to measure the effects of grazing on surface water. For three years, groundwater and surface water samples

were collected biweekly and pore water was collected when conditions made it possible. Samples were analyzed for inorganic N and dissolved reactive P and were digested for determination of dissolved organic N and P, pools previously not considered major sources of nutrient loss. Seasonal mean nitrate concentrations under the grazed watersheds remained below the EPA maximum contaminant load of 10 mg L^{-1} with only two exceptions on the grazed watersheds. Mean nitrate concentrations in the four grazed watersheds ranged from 3 to 7.44 mg L^{-1}. Nitrogen losses were closely correlated to farm N surpluses. Groundwater P concentrations exceeded the EPA surface water critical levels in all six watersheds. Geologic factors, rather than dairy farm management, played a large role in P losses In all watersheds, substantial pools of dissolved organic N and P were measured in groundwater. Low nitrate losses under MIG as well as the environmental advantages inherent in a grass-based system make grazing a viable Best Management Practice.

WATER QUALITY IN MANAGEMENT INTENSIVE GRAZING AND CONFINED FEEDING DAIRY FARM WATERSHEDS

By

Rachel Esther Gilker

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Advisory Committee:
Professor Ray R. Weil, Chair
Professor Robert Hill
Professor Rick Kohn
Professor Adel Shirmohammadi
Professor Patricia Steinhilber

Dedication

To Chris, my heart and steadfast best friend,
you give me strength and courage,
and to Ezra Louis, our sunshine

And to the memory of
Cousin Louis Dolin,
Bill Stout
and
Sue Dorcey, mwah.

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A quote has been sitting by my computer during the years of research and writing:

"'It's a long time yet,' said Maggie. 'You seem to have put an idea in his head.'
'Me? What?'
'He says he'd like to be a – dirt doctor.'
'Dirt doctor!'"
---*Who has seen the wind,* W.O. Mitchell

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Chapter 1: Dairy farm management and nutrient losses

Abstract

Dairy farm size has increased in the United States, while the profit margin has decreased. Farmers using confined feeding systems have had to increase herd size to maintain financial viability, requiring the purchase of feed and dietary supplements and leading to on-farm nutrient loading. An alternative to confined feeding dairy farm management is management intensive grazing (MIG), a grass-based system relying on rotational grazing for most of the herd's dietary requirements. The herd is rotated between small paddocks every 12-24 hours for the most efficient forage consumption. Farmers adopting MIG have found both economic and quality-of-life benefits, as MIG requires less capital outlay and allows the farm to be profitable with a smaller herd. Cycling of nitrogen and phosphorus is more direct when grazing animals return the majority of ingested nutrients directly to the pasture; however, research using monolith lysimeters and on fertilized grazed land has measured high nitrate concentrations in leachate. The research that suggests MIG leads to excessive nitrate leaching does not accurately represent mid-Atlantic grazing systems, and further investigation is required to determine the nature of losses under MIG in this region.

Dairy farm management: "Get big, or get out."

Dairy farming has changed in the United States since the end of World War II. Average farm and herd size has increased, while the number of dairy farms decreased (Etgen and Reaves, 1978). The total number of farms in the U.S. fell 69 percent between 1940 and 1997, and the number of farms with milk cows decreased by 97 percent

(Blayney, 2002). After the 1950s and 1960s, dairy farming was no longer a sideline of homestead farm operations, and only farms dedicated to dairying kept milk cows.

When farm and herd sizes were relatively small, cows and other livestock were raised on marginal, unproductive land dedicated as pastureland (Fick and Clark, 1998). Harvested hay was stored for feeding the livestock over winter months. As farm machinery became commonplace, dairy farming came to include crop production to provide high quality feed. Dairy farmers in the 1960s and 1970s began to use more efficient milking, feeding and waste-handling systems. Between 1975 and 2000, the average number of cows per dairy operation increased from 25 to 88.

Changes in milk parlor design, animal housing, and monitoring tools refined dairy farming, and farmers began to substitute mixed rations for pasture. With these technical innovations, milk production per cow increased. While the number of milk cows in the U.S. went from 22 million to 9.2 million in the last half of the 20th century, and milk production remained constant at 53 billion kg of milk (Blayney, 2002).

Under confined-feeding dairying, the herd was kept in enclosures, or barns, and their feed raised in the fields and/or purchased from off-farm and brought to them. Advances in animal nutrition, health, and breeding improved the production by each cow. Efforts were made to monitor and control the environment, creating conditions for optimal production. Manure from the herd was stored in pits or lagoons and may have been used to supplement field fertility. States established standards for milk, generally addressing sanitation and health issues, and handling of milk changed with the addition of new equipment set up to store and refrigerate thousands of gallons.

Although milk production steadily increased, profitability did not. Production costs continued to rise, while the demand for milk remained stable, and milk prices dropped. Because of narrowing profit margins per cow and per unit of milk sold, dairy farmers increased herd size, following the now-familiar maxim attributed to Secretaries of Agriculture Earl Butz and Ezra Taft Benson to “get big or get out.” Getting big can mean a deterioration in farmer quality of life (Berry, 1986), as one farmer put it in an interview with Studs Terkel, “the only way the farmers are making it today is the ones in business keep getting bigger, to kinda offset the acreage, the margin income. I don’t know what’s gonna happen in the future. I’m afraid it’s gonna get rough in time to come….I don’t believe farmers have as much ulcers as business people, ‘cause their life isn’t quite as fast. But I’ll say there will be more as time goes on. ‘Cause farming is changing more. It’s more a business now. It’s getting to be a big business.” (Terkel, 1972). In the 1990s, milk price support programs of the U.S. government were curtailed, removing the lower limit on milk prices and making profitability an increasingly pressing issue in the minds of dairy farmers.

Large-scale confined feeding dairy farming, with herd sizes in the thousands, has been described as conferring certain benefits (Cheeke, 1993). Large farms are run more like industry. There may be on-site services to oversee animal welfare, health, and feed nutrition. An increased level of control may help maintain product consistency. There is also the possibility of improvements in worker benefits. And, environmentally, a larger business is more likely to be able to afford the most advanced waste storage and disposal facilities to avoid nutrient pollution from the concentrated manure and nutrients.

However, there also may be an increased risk of environmental degradation, as larger farms mean greater concentrations of nutrients and manure.

"Getting big" is not the only option dairy farmers have for staying profitable. Another avenue toward profitability is to decrease the costs of production, an approach facilitated by use of grass-based systems. Dairy farmers in New Zealand, Europe, and other parts of the world have experienced benefits from low-input dairy farming, relying on rotationally-grazed pasture for the primary source of cow nutrition (Fick and Clark, 1998). A small percentage of North American dairy farmers has adopted this low-input farming system, but many consider it a choice appropriate only for farmers in financial hardship, rather than an attractive alternative to confined feeding (Muller and Holden, 1995).

Management intensive grazing

The first published English instructions for grazing management appeared in 1739, when Samuel Trowell recommended 21-30 day pasture rest periods (Fick and Clark, 1998). Since then, refinements have been added, but the basic principles of allowing 3-4 weeks to elapse between grazing remains a part of rotational grazing.

Grazing systems can be described by the period of rotation, intensity of management, or goals of the management system. Management intensive grazing (MIG) aims to maximize the quality of forage and livestock performance, or milk production (Murphy, 1998; USDA-SARE, 1995). Different terms are used to describe the MIG system, including phrases such as management intensive rotational grazing (Paine et al., 1999), rotational grazing (Mueller and Green, 1987), or intensive grazing (Volesky et al., 1990). In this work, MIG or grazing will be used to refer to low input, but management intensive, rotational grazing of dairy cows. In this system, dairy cows are typically rotated between paddocks once or twice each day, far more frequently than in typical beef cattle rotational grazing systems.

Land on a MIG-based farm is primarily in grassed pastureland, divided into small paddocks with lightweight, moveable electric fencing (Murphy, 1998). Paddock size is dependent upon herd size and management intensity, with paddocks large enough to supply enough dry matter for the grazing animals, and small enough to ensure that most of the forage plants present are consumed (Murphy, 1998). The herd is rotated between the paddocks every 12-24 hours.

Two main components of grazing management open to modification are the intensity and frequency of grazing. When the word ***intensive*** or ***intensity*** is used to describe grazing, it refers to the management being applied and refers to the amount of dry matter removed in each grazing session (Bryan et al., 2000). ***Intensity*** can be affected by the number of cows put in the paddock, and the length of time allowed for the grazing. If the grazing area allotted is greater than the herd requires, cows will graze selectively, choosing the most palatable plants. Under ***intensive*** management, the available land is typically divided into more but smaller paddocks, grazing periods are shorter, and generally the stocking rate is higher (Jackson-Smith et al., 1996).

Sward height and re-growth determine grazing frequency (Bryan et al., 2000). Plant growth rate and forage quality will affect the length of the recovery period needed (Murphy, 1998). By rotating the herd more frequently, pre-grazing plant height will be shorter, and plant quality may be greater. Plant growth rate, in turn, is affected by post-grazing plant height (Vough et al., 1994a). Re-growth post-grazing is slower initially because removal of leaves limits the photosynthetic capacity of the plant. New growth is fastest from newly initiated leaves along the tiller or shoot and stem. If grazing removes too much of the tiller, new growth will be slower and will come from basal or rhizome buds at the root of the plant, or from aerial tillers. When plant growth is more rapid, recovery periods may be shorter and paddocks smaller. In the spring, cool-season forages quickly reach the optimal pre-grazing heights, and recovery periods are shortest, at 12-24 days. Growth of cool-season forages is slower in midsummer because of drier conditions and hotter temperatures, and recovery periods may be 24-30 days. Warm season grasses may grow fastest during the summer months. As temperatures drop in the fall, and daylight shortens, recovery periods may lengthen to 24-42 days. Legumes may need longer recovery periods, of 28-35 days (Vough et al., 1994a).

Forage yields

Research has been undertaken on different forage species to find the highest-yielding, most palatable and digestible species for use in grazing. In Wisconsin, 91 cool-season grasses were planted in blocks on three dairy farms (Casler et al., 1998). Pre- and post-grazing estimates of initially available and ingested forage were 2.7-3.8 Mg ha^{-1} and 1-1.8 Mg ha^{-1} respectively, with intake positively related to available forage; reed canarygrass (*Phalaris arundinacea* L.) had the greatest amount of both available forage and intake. Similar intake rates were found in Illinois, where pastures were grazed by cow-calf herds for three days with 26-day rest periods, and 56% of forage produced by the sward was ingested (Kaiser et al., 1990).

An important aspect to managing a grazing system is estimating pasture forage yields. Some grazing guides provide estimates which are based on pasture height and species (Mueller and Green, 1987) and yields for hay production (Sullivan et al., 2000).

Tools have been developed to help graziers estimate the forage available for grazing. Graziers may also visually assess the forage available in the paddocks to schedule the grazing rotation. To extend the time livestock can obtain their sustenance from on-farm forage, farmers may stockpile grasses when paddock yields exceed the need of the herd.

Forage quality

Forage quality is a measurement of the forage's suitability to meet the cow's dietary requirements. Cows require carbohydrates and protein, and the concentration of these in the plant is affected by its stage of growth.

Carbohydrates provide energy for milk production, reproduction, and body maintenance. The main types of carbohydrates are fiber, sugars and starches. Sugars and starches, non-fiber carbohydrates found in growing plant cells and grains, are highly digestible and are the major source of energy. Fiber carbohydrates are lignin, cellulose, and hemicellulose. Cellulose and hemicellulose are found in the structural parts of plants, and in legumes, grasses, and corn silage. They are only partially digestible. Lignin is found in the cell walls of mature plants, alfalfa (*Medicago sativa* L.), and straw, and is practically indigestible.

Protein is also a necessary part of cow rations for milk production, growth and maintenance, reproduction, and to maintain the necessary rumen microbes (Vough et al., 1994a). Protein is found in higher concentrations in younger plants and is at its peak in the spring.

Forage quality is usually at its peak when plants reach a height of 15-20 cm and are grazed down to 2.5-5 cm (Murphy, 1998). Both pre- and post-grazing heights may be varied by sward species (USDA-NRCS, 1997). Forage quality is also dependent on the development of the sward (Vough et al., 1994b).

The morphological development of perennial forage grasses has been divided into five growth stages, described as: (1) germination, (2) vegetative, (3) elongation, (4) reproductive, and (5) seed ripening (Moore and Moser, 1995). Morphology can be affected by environment, plant species, and season. Development will be affected by sward management and grazing (Sheafferet al., 1998) . The proportion of leaves and stems affects forage quality because of differences between the two. Grass in a sward is made up of leaves, stems and inflorescence, in a proportion dependent upon the development stage. Generally, as the pasture sward matures, the proportion of leaves to stems drops, and protein and energy yielded decrease, while fiber and lignin content increase (Vough et al., 1994b). The greater the proportion of leaves, the higher the sward's forage quality, as leaves contain higher N content and approximately twice the crude protein of stems, and generally less of the lignified cell walls found in stems (Sheaffer et al., 1998).

Grazier quality of life

Grazing has environmental, social, and economic implications (Fick and Clark, 1998). Socially, rural communities may be strengthened, and farmer quality of life may improve after a switch to grazing. The work week for a farmer on a MIG farm can be between 20-50 hours shorter than that required of farmers on confined feeding farms, in part because grazing herds are often smaller than those on confined feeding farms (Jackson-Smith et al., 1996).

When considering labor on a per cow basis, the hours worked are similar, with 3.03 hours per cow per week needed on the grazing farm, compared to 2.87 on the confined feeding farm (Johnson, 2002). Jackson-Smith et al. (1996) found that grazing cows required 2.53 hours per cow, as compared to 2.19 hours for confined cows. However, in Michigan, when farms with similar herd sizes were compared, the effect on work week hours was the reverse, with annual labor hours of 83.9 for confined feeding cows and 75.8 for grazing cows (Nott, 2000). As one farmer said, "…what I like most about grazing dairy cows is that we're not so busy. My family is relaxed and my cows are relaxed. We have time to enjoy each other." (USDA-NRCS, 1997).

In a survey of more than 1,500 Wisconsin dairy farmers, the 150 respondents who had converted to grazing from confined feeding systems said their work hours and labor use had decreased after making the switch, while their family free time and net farm and household income had increased (Jackson-Smith et al., 1996). Of the dairy farmers surveyed, farmers on grazing-based farms were more likely than confined-feeding farmers to say their family's quality of life had improved over the past five years.

Economic effects of grazing

Studies indicate that it is possible to be financially successful producing milk from cows raised on grass, and that grazing is economically competitive with confined feeding. In two case studies, the potential profitability under grazing was greater than that under confined feeding (Elbrehi and Ford, 1995; Pillsbury and Burns, 1989). On dairy farms in Wisconsin, net farm income from operations per cow was higher on grazing farms than on confined feeding farms for all five years of the study (Kriegl, 2001). Another way of looking at profitability is through the rate of return on assets, considered to be the most telling piece of information on farm performance (Johnson, 2002). The rate of return on assets is a measure of the farm's profit generation based on its use of its assets of land, labor, management, and capital. In a study of 530 Wisconsin dairy farms, the ratio of return to assets was 17.8% for grazing compared to 9.6% for confined feeding (Jackson-Smith et al., 1996).

Ford and Musser (1998) reviewed literature comparing grazing and confined feeding profitability and found higher profitability per cow for grazing in all 22 studies examined, with the difference between the two systems ranging from \$47 to \$294 cow^{-1}. This comparison does not take into account the difference in cow breeds found on the different types of farms. Confined operators almost always use Holstein cows, while many graziers use smaller breeds.

The main reason for the economic advantage of grazing over confined feeding is the difference in production costs between the two operations. Several studies have recorded lower costs on grazing farms for chemicals, seeds, and veterinary bills, (Kriegl, 2001; Nott, 2000; Ford and Musser, 1998). The biggest single expense on confined feeding farms is feed, which may make up 45-69% of the milk production costs (Vough et al., 1994a).

In a comparison of 15 New York dairy farms, production costs dropped \$153 per cow after the adoption of grazing (Emmick and Toomer, 1991). The cost for production and harvest for an acre of conventionally-tilled corn (*Zea mays* L.) is approximately \$200 more than for an acre of pasture, and hay field production and harvest costs about $60 more per acre than does pasture (Ford and Musser, 1998). Another relevant comparison is the net farm income from operations (NFIFO). Per hundredweight of milk (cwt), graziers had a consistently higher NFIFO, earning $1.36-$1.94 more than conventional feeders over a five year period (Kriegl, 2001).

Ford and Musser (1998) also noted that expenses for repairs, utilities, fuel, machinery, and manure management all decrease under grazing management. Feed costs can be lower, although purchases for some feed may increase. Another savings is in veterinary expenses, which surveys have noted drop under grazing (Parker et al., 1992; Elbrehi and Ford, 1995; Ford and Musser, 1998). Improved herd health also translates to lower cull rates, and culls from grazing farms may be healthier and heavier than those from confined feeding farms, yielding a higher sale price (Ford and Musser, 1998). The healthier cows produce milk for more seasons and do not have to be replaced, or culled, as often. Not having to cull a cow and replace it with a heifer can mean a savings of $500-600. Other savings under grazing are in the decreased need for housing and bedding (Ford and Musser, 1998; Parker et al., 1992; Hanson et al., 1998) .

Ford and Musser (1998) also point out that there are costs associated with grazing that are not insignificant, such as fertilizer, repairs to equipment, general maintenance, and some machinery expenses. Additionally, there are costs related to the establishment of pasture systems, such as fencing, water supply systems, and forage stand establishment.

Milk production per cow may decrease after a switch to grazing from high-input confined feeding. The decrease reported from 60 dairy farms in the Pennsylvania-New

York region was in the range of 3-10% (Hanson et al., 1998). Estimates on the annual costs of lost milk are about $130 per cow (Ford and Musser, 1998). The decreased production is due to less control over nutritional intake and energy expenditure, and not all herds experience lowered production under grazing (Ford and Musser, 1998). Milk production may drop after making the switch to grazing, but as the grazier gains experience, improvements in management may lead to a rebound. In a Michigan study, milk production per cow averaged less on the grazing farms, but production on some of the grazing farms was on par with the confined feeding farms in the study (Johnson, 2002).

Environmental effects of grazing

Perennially grass-covered pastures are associated with a number of environmental benefits. Continuous grass cover leads to the accumulation of soil organic matter, sequestering carbon in the soil, and thereby reducing the potential for CO_2 accumulation in the atmosphere (Weil et al., 1993). The increase in soil organic matter is also related to soil quality, with improvements in soil structure, aeration and microbial activity (Brady and Weil, 2002; Skidmore et al., 1975; Banerjee et al., 2000). Grassed pasture can limit the amount of runoff (Alderfer and Robinson, 1947), which may reduce surface transport of nutrients and agricultural chemicals carried by sediments compared to losses from cropland (Fick and Clark, 1998). Soil erosion is 80-90% less from grassed pastures than from oats, corn silage, or corn grain production according to the USDA Universal Soil Loss Equation (USDA-NRCS, 1989).

Because cows do the harvesting and manure-spreading, a MIG farm is less reliant on large machinery and consumes much less fossil fuel than a confined-feeding system

(Fick and Clark, 1998). There are some requirements for machinery to establish grass swards, clip pastures, and move hay bales, but fuel and power costs for grazed pasture production and use are a small fraction compared to that of corn production (Fick and Clark, 1998). In a survey of four years of records for Maryland dairy farms, the average annual fuel cost per cow was $42 for ten grazing farms, and $53 for 27 confined feeding farms (Johnson et al., 2004). Additionally, the imported feed most confined feeding dairy farms rely upon is trucked far distances from the site of production to consumption (Lanyon, 1992). The imported feed also inevitably causes a concentration of nutrients at the site of consumption. Because grazing dairy farms can be profitable with smaller herd size, they can maintain the herd mainly with forage grown on the farm land, and thus can avoid importing high quantities of nutrients as feed (Lynch et al., 2003).

There are potential negative environmental impacts to grazing. Grazing on poor quality forage may cause the production of additional methane, and surface and ground water may be contaminated by nutrient leaching and runoff from pastures. Nutrient losses to groundwater from grazed pasture have been of particular concern in recent research.

Nutrient cycling

Because of direct deposition of excreta by animals on pastures, nutrient cycling under grazing is potentially more efficient than under confined feeding. In confined feeding systems, the manure is collected, and some of the nutrients are lost in the course of storage. In grazing systems, 75-80% of ingested nutrients typically are returned to pasture as excreted feces and urine (Jonker et al., 2002; Kemp et al., 1979). However, these excreta are deposited in feces and urine spots that may cover only 15-30% of the pasture area in a given year, and not only during the growing season (Petersen et al., 1956). Management is necessary to prevent cows from returning most of the excreta in a small area of the pasture where cows tend to congregate and rest (Peterson and Gerrish, 1992).

Nutrient cycling – Nitrogen

Nitrogen is typically brought onto the farm in both fertilizer and feed. About 10-25% of the ingested nitrogen (N) is exported from dairy farms in milk. Approximately three-quarters of the N in ingested feed is excreted, and either deposited directly onto the pasture by grazing cows, or applied to the fields by the farmer (Jonker et al., 2002; Kemp et al., 1979). If supplemental feed is imported, the return of N to the pasture may be equal to or greater than that removed in grazing.

Leguminous species of forage and crop plants can also import N to the farm through biological N-fixation. The addition of N through fixation is a function of the N in the soil as well as climatic conditions. When more plant-available N is present in the soil, N fixation is inhibited. When N is fixed, some portion of that fixed N may become available for plant uptake and may contribute to a net increase in pasture soil plant-available N (Bellows, 2001). Nitrogen is also imported onto the farm in both dry and wet atmospheric deposition, but this importation is not affected by management system.

Nitrogen may be lost from the dairy farm through volatilization of ammonia gas. Excreta-N volatilization occurs at a lower rate in grazing systems than in confined feeding systems (Meisinger and Jokela, 2000; Jarvis et al., 1989). When urine is excreted onto the pasture, only 5-25% of the urinary-N volatilizes, but of the manure excreted by confined herds, 40-99% of N volatilizes from land-applied manure, along with 30-35%

from excretions within the barns (Meisinger and Jokela, 2000; Sherwood, 1981). The lower volatilization rate of pasture-excreted urine and subsequent greater return to the nutrient cycle is due to rapid infiltration by the urine, and sorption of ammonium-N to soil colloids. The very high rates of volatilization occur during hot and dry periods, because as the urine is exposed to air and dries, water in the urine quickly evaporates and more of the ammonium present is converted to ammonia and volatilizes. Approximately 80% of the N excreted is in the urine, and most of that is in urea, from which N rapidly becomes available for plant uptake.

One pathway of N loss which is of great concern is leaching to the groundwater. As well as causing environmental damage, the N losses can also be a cause for economic concern, as loss of applied N can be considered an expense to the farmer (Groeneveld et al., 1998). For a more complete discussion, see “Nutrient losses to groundwater”.

Soil erosion and surface flow are generally not major pathways for N transport on well-managed grazed land, in part because the vegetative cover of the pastures limits runoff and soil erosion (Owens et al., 1983b; Van Doren et al., 1940). In beef cattle-grazed pastures in southern Ohio with more than 90% vegetative cover, very little runoff and sediment was collected from most precipitation events. Most of the runoff occurred during 5-10% of the events (Owens et al., 1983a). Average nitrate-N concentration in surface runoff from the beef cattle-grazed pastures was only 2.8 mg L^{-1} (Owens et al., 1983b). Surface flow accounted for 14-20% of the N transport measured in their research, with yearly total-N concentrations in runoff from the winter grazing area averaging 7.6 mg N L^{-1} over a five-year period. There was little N transported in sediment, as there was very little soil loss. Under regulated grazing, Van Doren et al.

(1940) measured less than 350 kg sediment loss per hectare from pasture with 8% slopes. Runoff was limited to less than 110 mm of the 1080 mm average precipitation from grazed pastures, even during a period of higher than normal rainfall (Owens et al., 1983a).

In South Carolina, dairy manure, poultry manure, sludge, and ammonium nitrate were applied to plots in a fescue pasture (McLeod and Hegg, 1984). Runoff was analyzed for nitrate-N, ammonium-N, and total Kjeldahl-N (TKN) concentrations. Less than 2% of the applied TKN in dairy manure was lost in runoff, and in all cases the number of rain events since application was the determining factor in runoff nutrient concentration. Poultry manure and chemical fertilizer caused higher initial TKN in runoff than did the sludge or dairy manure. Runoff nitrate concentrations from the chemical fertilizer exceeded acceptable limits, but overall losses made up a minimal proportion of each of the four applications. Manure applications made to simulate grazing on tall fescue (*Festuca arundinacea* Schreb.) plots in Kentucky produced low concentrations of N in runoff, totaling 7.4-21.1 g nitrate-N ha^{-1} and 49.9-182.8 g TKN ha^{-1} (Edwards et al., 2000). These results agree with those found in Arkansas, where the effect of poultry manure was compared to that of dairy feces and urine on surface runoff from tall fescue hayfields (Sauer et al., 1999). N losses from poultry litter treatments were more than six times higher, with only minimal N losses in runoff from the dairy excreta treatments.

Eghball and Gilley (1999) reported that surface application of manure or compost application to no-till fields can cause ammonium-N runoff losses at higher rates than when the applications are incorporated. Runoff did not appear to be a major source of nitrogen loss from no-till fields, however, when manure was applied at appropriate agronomic rates. Nutrient runoff can occur as a result of spray-irrigation of effluent, especially if high rates are applied in response to inadequate manure storage space (Karr et al., 2003). Manure applications are commonly applied at rates in excess of crop needs on confined feeding farms, especially where herd size results in manure accumulation exceeding storage capacity or crop needs (Bacon et al., 1990; Lanyon, 1992). An alternative to over-application is transportation of the manure to areas needing N application, but doing so may not be feasible because of financial or time constraints (Lanyon, 1992). Taken together, these results suggest that runoff is not a significant path for N loss from grazed pastures.

Nutrient cycling – Phosphorus

Phosphorus (P) cycles among soil, plants, and water, but without the atmospheric component found in the N cycle. There is usually not very much P in soil in non-agricultural or non-fertilized soil. Much of the P present is found in insoluble compounds with calcium, iron, or aluminum, or in organic compounds (Bellows, 2001). When soluble P is applied to fields, much of it may become fixed into these insoluble compounds that release P slowly.

Like N, P generally is introduced into the dairy farm nutrient cycle in imported feed. Phosphorus requirements had been reported as a function of body weight and fecal P excretion data, but more recent research demonstrated that a more appropriate approach is as a function of dry matter intake (DMI) (National Research Council, 2001). Current recommendations by the National Research Council (2001) state that dietary P should make up 2.2 - 4 g kg^{-1} of diet dry matter, but farmers on both confined feeding and grazing dairy farms typically provide rations of up to 4.8 g P kg^{-1} dry matter. Concern over reduced milk production and reproductive efficiency by the herd keep farmers from reducing the P content of rations (Ebeling et al., 2002). To avoid potential declines in milk production and fertility, dairy nutritionists may recommend higher than necessary levels, with high milk-production herds receiving 5 to 6 g P kg^{-1}. With these recommendations still being made, it is difficult to convince farmers higher levels of P are not necessary (Powell et al., 2001). Phosphorus intake can be reduced from these highs by 30-40% without decreasing milk production (Satter and Wu, 1999). Additional research shows that moderate decreases in dietary P do not affect fertility or milk production (Valk et al., 2000).

Almost 75% of the P consumed is excreted, and the more P ingested, the more present in excreted manure (Ebeling et al., 2002). Only 15-25% of the P ingested in herbage by grazing animals is used in weight gain or exported in milk sold. Of the P excreted, 95-98% is in feces, which averages 1.2% P by dry weight (Sharpley et al., 1985). The amount of P in manure often exceeds the amount needed for crop production (Lanyon and Beegle, 1989). This excess P may lead to more P in runoff when the manure is applied to the field, especially if manure application rates are calculated to fulfill N requirements, and the manure may contain more P than required. Also, manure may have high levels of P because dietary P values are higher than necessary; P may be purposefully over-applied to compensate for P fixation taking place in the field; or there may be more manure than storage space, so application becomes necessary (Ebeling et al., 2002; Lanyon and Beegle, 1989; Powell et al., 2001). Since much of the applications remain on the surface, P is more susceptible to loss in runoff, where it can end up in natural waters and contribute to eutrophication.

Phosphorus loss in surface runoff is either dissolved in water or sorbed to eroded soil particles. Much of the P may be in dissolved form, which is often more reactive and, therefore, more bio-available. More of this dissolved and reactive P will be removed in runoff in sites where soil erosion is unlikely and P losses are primarily due to surface flow (Nash et al., 2000). In such cases, where P transport is largely dissolved in runoff rather than sorbed to eroding soil particles, practices that abate erosion may not greatly reduce nutrient loss or transport.

The amount of P in runoff is inversely related to the time elapsed between the runoff event and the manure or fertilizer application or grazing period (Nash et al., 2000). The concentration of P in runoff is a function of soil conditions, amount and rate of runoff, and the P present near or at the soil surface. All else being equal, less P was lost in runoff from grazed land than from corn fields, at 5% and 13% of the applied P, respectively, from two fields of similar slope and soil conditions, (Magdoff et al., 1997). Total runoff volume and P losses were lower from grassland than from corn stubble/winter cereal over the same period of rainfall events (Withers et al., 1999).

If more P is added to a soil than is removed in crop harvest and runoff, soil build-up of P will occur (Powell et al., 2001; Wang et al., 1999). Phosphorus measured in soil tests may be at levels much higher than needed for optimal crop yields (Powell et al., 2001). In a Wisconsin field, repeated manure and fertilizer application increased soil test P (using Bray1 P extraction) over the years from 34 mg kg^{-1} in 1968-1973 to 50 mg kg^{-1} in 1990-1994, where 25-35 mg kg^{-1} is considered sufficient. In soils with excessive levels of P, especially where high levels of P have resulted from heavy, repeated manure or fertilizer applications, P movement through the soil profile is possible (Eghball et al., 1996). Phosphorus moved deeper through the profile under manure applications than did P from fertilizer when both were applied at similar loading, which may be due to the enhanced mobility or solubility of organic forms of P (Eghball et al., 1996).

Under monolith lysimeters installed in grassland receiving typical fertilizer applications of 40 kg P ha^{-1} for two years, Turner and Haygarth (2000) found leachate P concentrations high enough to be associated with eutrophication. Concentrations were regularly above 100 μg P L^{-1}, with most of the leachate P in the dissolved fraction (<0.45 μm) and 21-46% in the particulate fraction (Turner and Haygarth, 2000). These findings and those by Eghball et al. (1996) and Ulen et al. (1998), suggest subsurface flow may be a more significant pathway for P transport than previously believed.

Nutrient balances and management on dairy farms

Less than 30% of N and P imports on dairy farms are exported in animal products, mainly due to the high levels of nutrients imported in feed, and the low levels of nutrients expelled in milk (Anderson and Magdoff, 2000; Klausner, 1995; Klausner et al., 1998). By increasing herd size to make dairy farming profitable, it may become necessary to import more fertilizer and feed, causing nutrient surpluses on the farm (Knowlton et al., 2001; Powell et al., 2001). However, there is great potential for reducing the levels of P fed to dairy cows, as seen in the work done by Ebeling et al. (2002), discussed above.

Kuipers et al. (1999) developed a model to calculate ways to reduce nutrient losses from farms in the Netherlands. Nutrient losses could be reduced on grazing farms by using fertilizer more efficiently, feeding a more balanced diet, and restricting grazing.

Monitoring milk urea nitrogen (MUN) helped optimize protein intake and reduce N losses in the rumen. Using fertilizer to achieve target yields on grassland and to account for weather conditions was also found to reduce N inputs. The surplus of imported phosphorus can also be reduced by lowering P concentrations in rations.

On a case study farm in Nova Scotia, Canada, Lynch et al. (2003) found that by switching from confined feeding to grazing, N surpluses per acre were reduced to approximately half those found in a study of 17 NY confined feeding dairy farms (Klausner et al., 1998). Because most confined-feeding dairy farm nutrient imports are in feed, and grazing farms rely primarily on pasture, there is potentially a great reduction in imports and the resulting surpluses (Lynch et al., 2003). Some of the surplus may be stored within the pasture, immobilized in soil organic matter. Permanent grassed pasture may account for increased N storage of up to 70 kg N ha^{-1} y^{-1} over a 10-year period (Barry et al., 1993).

Nutrient losses to groundwater

Many studies in New Zealand, the United Kingdom, and the U.S. have shown high nitrate concentrations in groundwater under fertilized, grazed pastures (Barraclough et al., 1992; Cuttle et al., 1998; Hack-ten Broeke et al., 1996; Macduff et al., 1990; Owens et al., 1992; Ruz-Jerez et al., 1995; Ryden et al., 1984). In one of the seminal studies, Ryden et al. (1984) measured nitrogen losses under un-replicated grazed and cut ryegrass (*Lolium perenne* L.) swards in the U.K. from 1976-1982. Both plots received 420 kg N ha^{-1} per year for four years of the study, but one sward was cut while the other was grazed rotationally one week out of four each April to October. They found that nitrate loss under the grazed sward was 5.6 times those under the cut grass. The source of this nitrate was attributed to the return of nutrients by the grazing cattle, as up to 90% of the consumed nitrogen may be excreted.

Nitrate leaching was measured from 1989-1991 under three sheep-grazed pastures in New Zealand (Ruz-Jerez et al., 1995). Comparing ryegrass and white clover (*Trifolium repens* L.); an herbal ley of legumes, grasses, and deep-rooting herbs; and ryegrass receiving 400 kg N ha^{-1} y^{-1}, they measured nitrate in the soil solution extracted from soil cores from between 30 and 45 cm depth. The nitrate leaching from the fertilized grass pasture was six to seven times that lost from the two other pastures and was the only one of the three treatments to exceed acceptable water quality limits.

In the Netherlands, Hack-ten Broeke and de Groot (1998) measured nitrate concentrations under six fields of an experimental dairy farm for four years. Farm management was planned to reduce manure and fertilizer applications and losses to groundwater. Nine ha of the 55 ha farm were in permanent grassland for rotational grazing. The grassland received 150-275 kg N ha^{-1} as slurry and 85-200 kg N ha^{-1} in fertilizer per year in addition to the excreta left by the grazing animals. Similar to findings by other studies, high levels of nitrate leached from the heavily fertilized grazed land, and the authors postulated that reduced fertilization might decrease the losses.

Measuring leaching with ceramic cup lysimeters from grazed perennial ryegrass *(Lolium perenne* L.) and/or white clover (*Trifolium repens* L.) receiving different fertilizer N rates, the only swards with nitrate concentrations within EU limits were the unfertilized grass and grass/clover mix (Macduff et al., 1990). The other treatments had nitrate concentrations greater than 11.3 mg N L^{-1}, and under the pure clover sward, nitrate concentrations increased over the course of the study.

Stout et al. (1997 and 2000a) carried out studies in Pennsylvania designed to measure nutrient losses to the groundwater under conditions simulating grazing. Monolith lysimeters of 60 cm in diameter and 90 cm deep were installed and the leachate from the bottom of each soil column was collected and analyzed. In one study, measurements were made comparing six different pasture swards, with some receiving N fertilization (Stout et al., 2000a). Four swards were grass/legume mixes, and two were fertilized grass. Nitrate leaching losses increased under the grass/legume swards, something attributed to the death of nodules on alfalfa (*Medicago sativa* L.) and white clover during drought conditions. The balance between grass and legume species was related to the amount of nitrate leaching, and species management was suggested as a potential tool to minimize nitrate leaching under grazing.

Another study compared the effect of urine and feces applied in the spring, summer and fall (Stout et al., 1997). Applications of urine and feces were made to the tops of the monolith lysimeters. The research found high levels of nitrate leaching from pasture swards with leguminous mixes, and for the lysimeters receiving urine, especially in the fall. Concentrations of nitrate exceeded EPA maximum contaminant load of 10 mg NO_3-N L^{-1}.

Using the data from two leaching experiments, Stout et al. (2000b) estimated nitrate concentrations leaching from grazed land under different stocking rates. They found that even under relatively low stocking rates, the expected nitrate concentrations would exceed the EPA maximum contaminant load. This work may not have accurately represented field leaching processes because the lysimeter lip extended above the soil

Figure 1-1: Urine spots on grazed pasture with an 8% slope. The dark grass on this otherwise low fertility pasture shows the extent of the urine spread. The superimposed circle indicates the size of the monolith lysimeters used in the cited leaching research. Scale marked in decimeters. (Photo courtesy of Ray Weil)

surface. The lysimeter lip therefore confined the applied urine to a 0.2 m^2 area, compared to the 0.5 to 1.5 m^2 areas of unimpeded urine spots observed on pastures (the higher values being for more steeply sloped land; personal communication Ray Weil, June 2004). Confinement of the applied urine would cause momentary ponding which is known to result in rapid preferential flow throughsoil pores such as earthworm burrows, root channels, and shrinkage cracks. In addition, flow down the edges of the monolith lysimeters has also been observed to double apparent hydraulic conductivity rates compared to soil columns with edges sealed (Cameron et al., 1996).

Unlike most grazing systems in the research just reviewed, MIG systems in the mid-Atlantic typically use <100 kg ha^{-1} y^{-1} or no N fertilizer. The unfertilized treatments of Macduff et al. (1990) may more reasonably represent grazing systems in the mid-Atlantic, but most of thestudies that found high levels of N leaching do not replicate conditions similar to MIG systems in the mid-Atlantic.

Maryland recommendations call for little to no nitrogen fertilizer on grazed pastures with grass/legume mixes (Greene, 1998: Coale, 2002). Graziers may apply only enough to provide for the expected yields, at rates of between 50-100 kg N ha^{-1}, if any (Lynch et al., 2003; Rehme et al., 2001), far different from the levels of nitrogen fertilizer used by most researchers, who applied up to 400 kg ha^{-1}.

Conclusions

Management intensive grazing is an alternative to confined feeding and is appropriate for mid-Atlantic dairy farmers. The improved lifestyle and increased profits of farmers using MIG have been noted in comparisons to confined feeding dairy farmers around the U.S. Worldwide and in the U.S.; however, nutrient pollution issues have dogged the otherwise attractive management system. A number of studies have pointed to excessive nitrate leaching as the outcome of herds excreting urine directly on grazed pastureland. These studies do not accurately represent mid-Atlantic dairy farming, especially not under MIG. Many of the studies use N fertilizer applications at rates much higher than those used by MIG dairy farmers, who use little if any. Other studies have measured leachate under monolith lysimeters which have artificial hydrologic constructs and may inflate the rate of leaching. Research monitoring water quality under actively grazed mid-Atlantic dairy farms can determine whether MIG produces excessive levels of nutrient leaching.

Chapter 2: Inorganic nitrogen in groundwater and soil on confined feeding and grazing dairy farm watersheds

Abstract

Under confined feeding systems, dairy farms and herds have had to grow to remain financially viable. Larger herd sizes have required purchase of feed and other supplements, leading to nutrient loading on the farm. Management intensive grazing (MIG) is an alternative to confined feeding and uses grassed pasture to provide most of the herd's nutritional needs. The herd is rotated between small paddocks every 12-24 hours for the most efficient forage consumption. As a grass-based system, MIG provides a number of environmental advantages by improving soil quality, sequestering carbon and reducing runoff. Farmers adopting MIG have found both economic and quality-of-life benefits, as MIG requires less capital outlay and allows the farm to be profitable with a smaller herd. However, previous research on MIG measured high concentrations of nitrate under monolith lysimeters and pastures, suggesting risks of nitrate leaching under grazing. These studies do not accurately represent mid-Atlantic MIG dairy farms. Many of the field-based studies included excessive levels of nitrogen fertilization, and MIG farmers in this region use very little fertilizer, if any. Studies using monolith lysimeters to measure nitrate leaching under urine applied to simulate grazing also found high leaching losses. These studies were likely over-estimating losses because the applied urine ponded and caused preferential flow, increasing leaching rates. To determine the extent of nitrate losses to groundwater under MIG in the mid-Atlantic, we sampled shallow groundwater biweekly for three years on three Maryland dairy farms, one confined and two MIG-based farms. Transects of nested piezometers and ceramic-tipped

suction lysimeters were installed in two watersheds on each farm. Over the study, seasonal mean nitrate concentrations under the grazed watersheds remained below the EPA maximum contaminant load of 10 mg L^{-1} with only twograzed watersheds exceeding this level during two separate measurement periods. Average nitrate concentrations for all four grazed watersheds were between 4 and 7 mg L^{-1}, indicating MIG does not cause excessive nitrate leaching and should be considered as an environmental Best Management Practice.

Introduction

Nitrogen typically is imported onto dairy farms in both fertilizer and feed, and only 10-25% of dietary N is exported in milk. Leguminous forage and crop plants add to the stock of N on the farm through biological N-fixation. Nitrogen also is imported onto the farm in both dry and wet atmospheric deposition, but this is not affected by management system.

Nitrogen losses from the farm are related to management. Three-quarters of ingested N is excreted, deposited directly onto the pasture by grazing cows or collected and applied to fields by the farmer (Jonker et al., 2002; Kemp et al., 1979). Ammonia volatilization from excreta-N occurs at a lower rate in grazing than in confined feeding systems (Meisinger and Jokela, 2000; Jarvis et al., 1989). Urine contains almost 80% of excreted N, and pasture-excreted urine infiltrates rapidly, resulting in ammonium- N sorption to soil colloids and a greater return of urine-N to thesoil nutrient cycle (Meisinger and Jokela, 2000).

Nitrogen in surface runoff (dissolved or attached to sediment) is not usually a major pathway for N transport to streams from well-managed grazing land, in part because the vegetative cover of pasture vegetation limits runoff and soil erosion (Owens et al., 1983b; Van Doren et al., 1940). In simulated grazing, manure applications on fescue plots produced low concentrations of N in runoff, suggesting runoff is not a significant path for N loss from grazed pastures (Edwards et al., 2000). Surface application of manure or compost to no-till fields at rates which fulfill corn (*Zea mays* L.) N or P requirements can cause ammonium-N losses in runoff at higher rates than when the applications are incorporated (Eghball and Gilley, 1999). Runoff was not a major source of nitrogen loss from no-till fields, however, when manure or compost was applied at appropriate rates, and when rainfall did not immediately follow nutrient application (McLeod and Hegg, 1984). Nitrogen runoff from grazing animals was considerably lower than from applications of poultry litter, even when poultry litter carried several times the N (Sauer et al., 1999).

In contrast to surface runoff, leaching is a pathway for N loss that provokes great environmental concern. Studies in New Zealand, the United Kingdom, and the U.S. have suggested that deposition of soluble N in urine on fertilized, grazed pasture leads to very high levels of N leaching (Barraclough et al., 1992; Cuttle et al., 1998; Hack-ten Broeke et al., 1996; Macduff et al., 1990; Owens et al., 1992; Ruz-Jerez et al., 1995; Ryden et al., 1984). These findings may cause reluctance on the part of U.S. policymakers and regulators to promote grazing as an alternative to confined feeding systems.

Studies done in the U.K. and New Zealand used high levels of N fertilization, rates which are more appropriate for hay production where the nutrients are removed in harvest and not returned by grazing animals. Unlike most of the grazing systems in the research cited above, MIG systems in the mid-Atlantic typically use little (<100 kg ha^{-1} y^{-}

^{1}) or no N fertilizer. Therefore, most of the studies that found high levels of N leaching do not represent conditions similar to MIG systems in the mid-Atlantic. Maryland recommendations call for little to no nitrogen fertilizer on grazed pastures with grass/legume mixes (Coale, 2002; Greene, 1998). Graziers may apply only enough to provide for the expected yields, at rates of between 50-100 kg N ha^{-1}, if any (Lynch et al., 2003; Rehme et al., 2001), far different from the up to 400 kg N ha^{-1} used in studies that found high levels of groundwater nitrate under grazed dairy pastures.

Research done in Pennsylvania used monolith lysimeters to estimate nitrate leaching from grazed lands under different stocking rates found high concentrations even at relatively low stocking rates (Stout et al., 2000a; Stout et al., 1997; Stout et al., 2000b). Because they prevent the applied urine from spreading over a larger area, lysimeters such as those used by Stout et al. (2000) may cause momentary ponding of applied urine that stimulates preferential flow and results in exaggerated estimates of N leaching potential. Monolith lysimeters are often prone to preferential flow and unnatural hydraulic conditions (Cameron et al., 1992).

This study was carried out to measure nitrogen concentrations in shallow groundwater and soil pore water under grazed pastures and manured crop fields to ascertain the nature of the environmental impact of MIG. By measuring the groundwater regularly on three well-managed dairy farms in Maryland, we hoped to determine if MIG does cause elevated levels of nitrate in the groundwater, or if those measured in previous research may be an artifice resulting from high levels of fertilization or unnatural hydrological processes associated with the use of monolith lysimeters (Cameron et al., 1992).

Materials and methods

Site selection

Three dairy farms in Maryland were selected for this study (Table 2-1). All three farms, herein designated Grazed 1, Grazed 2, and Confined, have been specialized dairy operations for at least 30 years and have included livestock for at least 100 years prior to the study. Grazed 1 and Grazed 2 began using a MIG system in 1995 and 1994, respectively. At the commencement of this study, Grazed 1 and Grazed 2 had been managed as MIG farms for six and seven years, respectively. The Confined farm uses conventional feeding system and has used some form of no-till management as a soil conservation practice on its cropland since 1962.

The grazing farms were selected from a very small pool of existing MIG farmers in Maryland. The reasons for choosing these two farms include the fact that both had two pasture watersheds suitable for groundwater monitoring and they were two of the earliest adopters of MIG in Maryland, increasing the likelihood thatsoils had approached steady state conditionsunder MIG and that the farmers had developed successful management systems. The confined farm was chosen because of its location adjacent to one of the MIG farms, with similar soils and topography, and because the confined farm had a long history of collaborating with Maryland Cooperative Extension and USDA Natural Resource Conservation Service personnel and was an early adopter of conservation tillage and nutrient management practices.

Table 2-1: Selected characteristics of the three study farms. Production attributes are based on averages from 2001-2003.

	Grazed 1	Grazed 2	Confined Feeding
Farm size, ha	83	71	245
Soils	Fauquier silt loam, fine, mixed, mesic, Ultic Hapludalf and Myersville silt loam, fine-loamy, mixed, active, mesic Ultic Hapludalf	Glenville loam, fine-loamy, mixed, active, mesic Aquic Fragiudult	Watershed A: Fauquier silt loam, fine, mixed, mesic, Ultic Hapludalf Watershed B: Highland silt loam, coarse-loamy, mixed, mesic Ultic Hapludalf
Farm location	Frederick County, Md.	Baltimore County, Md.	Frederick County, Md.
Switched to grazing	1995	1994	NA
Herd size, milking cows	105	150	400
Avg. cow weight[†], kg	363	499	590
AU[‡] ha^{-1}	0.95	2.2	2.1
AUD[§] ha^{-1}	348	810	NA
Vegetation	Pasture: 8% legume	Pasture: 24% legume	6 year rotation: corn/oats/alfalfa
Milk production per AU, L	5,990	4,240	8,556
Profit $/Mg milk	154	96	79

[†]Cow breed varied among farms.

[‡] AU = animal units of 454 kg (or 1000 lbs).

[§]AUD per ha = days of grazing by milk cow herd. Does not include heifers or calves. Assumes cows graze 365 days y^{-1}.

The farms were selected with the help of local agriculture extension agents who identified farmers who would be willing to participate in the study and had reputations as good managers and good land stewards. The three farmers joining the study allowed us to install nests of piezometers in their fields or pastures and shared their financial and nutrient management records for economic and nutrient balance analyses (Appendix A). For twelve months of the study, the three farmers collected bulk-tank milk samples which were analyzed for milk urea nitrogen (MUN), a predictor of cow N excretion (Jonker et al., 1998).

Grazed 1 and Confined occupy adjacent tracts of land in Frederick County, Maryland, in the lowland section of the Piedmont Plateau physiographic province, where the average precipitation is 1026 mm, and average annual temperature is 13°C. Soils on Grazed 1 are primarily Fauquier silt loams and Myersville silt loams. The soils on Confined are Fauquier silt loams and Highland silt loams. Grazed 2 occupies land in eastern Baltimore County, Maryland, in the upland section of the Piedmont Plateau physiographic province, where the average precipitation is 1039 mm, and average annual temperature is 13°C. Its soils are mainly Glenville loams, overlaying a Cockeysville marble (Cleaves et al., 1968).

Grazed 1 and Grazed 2 provide the dietary energy needs of the herd primarily through grazed forage. On these farms, hay is made to store excess forage for winter feed. Additional hay may be purchased when on-farm hay production is insufficient to support the herd through the winter. Relatively small quantities of purchased supplemental grain are imported to the farm. Grazed 1 cows have been supplemented at

3.6 kg cow^{-1} day^{-1} (roughly 1% of body mass) since 1999. Grazed 2 cows were fed supplemental grain at 3.6- 6.8 kg cow^{-1} day^{-1} during 2001-2002 and 3.6 kg cow^{-1} day^{-1} from 2003 onward.

Confined produces crops in a six-year rotation of corn-corn-oats-alfalfa-alfalfa-alfalfa (oats: *Avena sativa* L.; alfalfa: *Medicago sativa* L.). Additional feed and bedding is purchased to support the herd. Manure from the herd is applied in liquid form to cropland on the farm. Supplemental fertilizer, at 56 kg N ha^{-1}, is used when needed, mostly on the corn fields.

Within each farm, two watersheds, identified as A and B, were selected for groundwater monitoring using piezometers. The watersheds were chosen because the majority of the land within each watershed was under the management of the farmer, and topography suggested that the groundwater would be within the reach (8 m) of the drilling equipment available to the project. In each pair of watersheds from a given farm, one watershed was determined to be the 'homestead' watershed, which historically (100+ years) received a greater proportion of nutrients because of its close proximity or convenience to the barn and homestead. The three homestead watersheds (designated "home") were Confined A, Grazed 1A, and Grazed 2B. The other watershed on each farm was designated "away". One control piezometer was installed on each farm upslope of the farm management activities, to measure the baseline nitrate-N concentration levels in groundwater coming onto the farm.

A tipping-bucket rain gauge (Spectrum Technologies; Plainfield, IL) was installed within or near each watershed. The rain gauge closest to the farmer's house or barn was supplied with a digital display that could be viewed and recorded by the farmer, and the rain gauge in the watershed farther from the homestead was connected to a downloadable datalogger (HOBO® Shuttle and event logger; Onset Computer Corporation; Pocasset, MA). Recording by the farmers was not consistent and was influenced by their work schedules. If rain data were available from the neighboring farm, it was used if one farm did not have rain data for a period of time. In the rare cases that the recording monitor or rain gauge was not operative and the farmer's records were not complete, National Oceanic and Atmospheric Administration records from the nearest station were used.

Site preparation

In each watershed, a transect of nested piezometers was installed consisting of three nests spaced 18 m apart, starting at the watershed discharge point and following the flow line upslope. A hinged, slatted wood box 0.9 m x 0.9 m x 0.15 m high was installed over each nest to protect the tops of the piezometers from farm equipment and grazing cows.

Three piezometers initially were installed within each nest. The piezometers were made of 5-cm inner diameter polyvinylchloride pipe. The deepest 1 m of each piezometer was slotted. The piezometers were installed in the spring of 2001, when groundwater levels were beginning to drop. The shallowest of the three piezometers was installed to a depth where it could just reach the groundwater at the time of installation, with the next two piezometers installed approximately 1 and 2 m deeper. The most shallow piezometer depths within each nest ranged from 1-3 m and the deepest ranged from 5-6.6 m. A fabric filter sock (Drain-Sleeve ® Fabric Sock; Carriff Corporation, Inc.; Midland, NC) was fitted over the 1 m of slotting at the bottom of each piezometer and taped in place. Clean sand was poured into the installation hole around the piezometer. A plug of bentonite powder (Wyoming Bentonite for Water Well & Geotechnical Sealing; Drillers Service, Inc.; Hickory, NC) was used around the upper 30 cm of each piezometer to prevent flow around the wall of the piezometer or down the installation hole.

Extreme drought conditions in 2001 caused groundwater levels to drop below the reach of the deepest piezometers in two of the watersheds. Because this lack of recharge during the winter and spring of 2002 suggested that future sampling might also be limited, a fourth, 1 m deeper piezometer was added in each of the affected nests in October 2002 (Appendix B, Table B-1).

Within each nest box, ceramic-tipped suction lysimeters (Irrometer Company, Inc.; Riverside, CA) were installed at a 45° angle to the surface of the ground, using a drop-hammer device to make the pilot hole in soil. The two 2.5-cm diameter lysimeters, one 90-cm and the other 120-cm long, were installed with the tops of the lysimeters within the wooden nest box, and the ceramic tip end extending below ground into surrounding pasture or field, so as to place the ceramic tip 60 or 90 cm below the soil surface and at least 30 cm outside of the box.

Groundwater sampling

To determine how many times the piezometer should be bailed to collect a sample representative of the groundwater, samples were taken of the water in the piezometer prior to bailing and upon refilling after a first and second bailing. Because no changes were seen in the groundwater chemistry (pH, EC or nitrate-N) between the samples taken after the first and second bailing, we concluded that only one bailing was necessary to obtain a representative groundwater sample.

Groundwater samples were collected biweekly, beginning in May 2001. Prior to sampling, the depth to groundwater in each piezometer was measured using a water depth indicator. The shallowest piezometer in each nest containing at least one meter of groundwater was bailed. After approximately two hours, samples of 120-150 mL were taken from the bailed piezometers. The pH of each sample was measured, and the sample was acidified to pH<3 with 2-3 drops of 4 *M* H_2SO_4. The samples were returned to the lab on ice, where they were stored under refrigeration at 4°C until analysis.

Soil pore water sampling

Whenever soil moisture allowed, soil pore water samples were collected using ceramic-tipped suction lysimeters at the time of groundwater sampling. This occurred eleven times on each farm between June 2002 to June 2004, mainly in the fall and spring months for a total of 26 sampling dates. To collect the samples, a suction of 70 to 80 kPa was pulled on the lysimeters, using a hand vacuum pump (Irrometer, Inc.; Riverside, CA), and the lysimeter tubes clamped off to hold this vacuum. The sample was collected two to four hours later by drawing it into an Erlenmeyer flask and then pouring into a sealable plastic vial. The Erlenmeyer flask was rinsed with distilled water in between the collection of samples. The lysimeter water samples were acidified with 4*M* H_2SO_4, with one drop added for every 30-50 mL of sample. They were transported on ice to the lab, where they were refrigerated until analysis.

Soil sampling

Soil profiles from distinct topographic areas (landscape units) within each watershed were collected with a bucket auger in the spring of 2002 and 2003. Two to four profiles per watershed were placedn a 10 -cm diameter trough, divided into horizons, described with regard to color, texture, and other morphological features (Appendix B). Composite samples of the upper 15 cm of each area were collected with a hand-held corer. The composite samples and samples from each horizon were taken to the lab in sealed plastic bags on ice, where they were spread out and air-dried. Sub-samples were ground and analyzed for total C, H, and N by high temperature combustion using the CHN 2000 (LECO Corporation; St. Joseph, Michigan) (Campbell, 1992). Samples were also analyzed for pH (1:1 in water), percent organic matter (loss on ignition) and Melich 1 extractable, Mg, P, K, and Ca (Northeast coordinating committee on soil testing, 1995).

Soluble soil N in these profile samples were estimated by extraction with 0.5*M* K_2SO_4. Three grams of dried, ground, and sieved soil were shaken with 30 mL of extractant for 30 minutes at 100 rpm, then centrifuged for 10 minutes at 3000 rpm, and then filtered (No. 42 Whatman filter paper), and refrigerated for no more than 24 hours before being analyzed for nitrate as described below. The standards used for comparison were filtered nitrate standards in 0.5*M* K_2SO_4.

Lab analyses

Water samples were filtered under vacuum through 0.2 μm filters (polycarbonate membrane, Nuclepore ® Corporation Filtration Products; Pleasanton, CA). A fresh membrane was used for each sample, and the filter apparatus was rinsed with distilled water between samples. Once filtered, samples were transferred to fresh sample cups and stored at 4°C unless not analyzed within 14 days, in which case long-term storage was at

<-15°C. Frozen samples were brought to room temperature (approximately 22-23°C) before analysis.

Filtered samples were analyzed for NO_3-N using a Technicon Autoanalyzer II flow injection analyzer (Technicon Industrial Systems; Tarrytown, NY) with a cadmium reduction column and a 2:1 distilled water dilution loop at a rate of 30 samples h^{-1} (Technicon Industrial Method No. 487-77A, 1977). Standard NO_3-N solutions in the concentration range of 0-25 mg L^{-1} were prepared from KNO_3. To bring the samples to within the necessary pH range for use in the Autoanalyzer (5-9), 0.5 mL of 0.1*M* NaOH in 10% NaAc was added to each. This buffer had no colorimetric effect on the procedure, but its dilution effect was included in the calculation of NO_3-N concentration.

Ammonia concentration was determined using an Orion 9512 ammonia specific gas-sensitiveelectrode (Banwart et al., 1972). One mL of 5*M* NaOHionic strength adjusting (ISA) solution with pH color indicator was added to 10.0 mL of sample to bring the sample pH to >13. A Teflon-coated stir bar was added, the sample vial placed on a magnetic stirrer, and the electrode was then lowered into the sample for a reading. When the change in mV slowed to <1mV s^{-1}, the mV reading was recorded for samples and ammonium-N standards (0, 0.1, 1, 10 mg L^{-1}) and a logarithmic standard curve constructed.

Potentially mineralizable soil nitrogen

The samples were incubated at 60% water-filled pore space (Linn and Doran, 1984). The 16-day incubation was carried out according to the methods described by Sainju et al. (2002). Each 10.0 g soil sample was incubated in a gas-tight 1-L chamber. Based on a preliminary experiment, 3.0 mL of 0.50 *M* NaOH in a plastic vial was also placed within each 1-L chamber. Additionally, a vial of distilled water was put in each chamber to maintain soil humidity and soil water content over the period of incubation. The 16 chambers were placed in an incubator at 30 $\pm$ 1 °C for 16 days (Drinkwater et al., 1996). Six "blank" chambers containing vials of NaOH and distilled water but no soil were also incubated to determine the extent of background CO_2 absorption in the containers.

At the end of the incubation period, the NaOH was titrated with standardized HCl after adding $BaCl_2$, and each sample was transferred into a 50-mL polyurethane centrifuge tube with 20 mL of 0.1 *M* K_2SO_4 and shaken horizontally for 15 minutes at 100 rpm. After the sample settled for at least 20 minutes, the supernatant was filtered (VWR No. 494 filter paper, VWR International; Bridgeport, NJ) into 20-mL vials. A 0.2 mL aliquot of this filtrate was used to determine the extracted nitrate N with a salicylic colorimetric method modified from Cataldo (1975). Ammonia was measured using an ammonia-gas sensitive electrode and millivolt meter as described above for groundwater samples. Initial nitrate and ammonium-N in soil samples was measured by performing the same extraction with 20 mL of 0.1 *M* K_2SO_4 on additional subsamples that had been stored dry at room temperature. The difference in N extracted from incubated and non-incubated soil was considered to be mineralizable N.

Drainage and groundwater flow determination

Drainage was calculated using the WATBAL model (Appendix C; Vinten, 1999), a monthly water balance model which uses inputs of temperature, slope, vegetative cover, rainfall, and cloud cover to estimate evapotranspiration, changes in soil moisture, runoff, and finally, drainage. WATBAL determines the monthly balance based on the inflow of precipitation, and outflows and flux of evapotranspiration, drainage and soil water storage. The model is based on the equation:

$$P = ET + R \pm \Delta SM$$

where: P = precipitation, ET = evapotranspiration, R = runoff, and SM is soil moisture. Evapotranspiration is determined from insolation based on a global radiation submodel, rather than on temperature alone (Starr, 1999).

Temperature and cloud cover data were taken from Maryland Archives (Maryland State Archives, online). Rainfall data was collected from the rain gauges on the farms, with additional information from local climatological data stations. Large-scale (1:1200) topographic maps were made to determine the boundaries, slope, slope aspects, and area of each watershed.

To calculate the rate of groundwater flow through the six watersheds, hydraulic conductivity was calculated using slug tests performed on each watershed. Rising-head slug tests were carried out for 62 of the 65 piezometers, using standard methods (ASTM Standard Test Method D4044, American Society for Testing Materials, 1997). Hydraulic conductivity for each piezometer was calculated using the Bouwer and Rice (1964) methodology, based on the following equation:

$$K= [r_c^2 \ln (R_e/R)]/(2L_e) \bullet (1/t) \bullet [\ln (y_0/y_t)],$$

where

r_c = radius of casing,

y_0 = vertical difference between water level inside and outside well at t = 0, assumed = 0,

y_t = vertical difference between well water level and water table outside at time t,

R_e = effective radial distance over which head is dissipated, and varying with well geometry,

R = radial distance of undisturbed portion of aquifer from centerline,

L_e = length of screened portion of well,

t = time.

To determine the hydraulic conductivity of each watershed, checks were made for anomalies in both the hydraulic conductivity calculated for each piezometer in the watershed and in the particle size distribution in the soil profile at the depth of the piezometers. Hydraulic conductivity can be extremely variable within a field, both due to heterogeneous topography and layers of contrasting conductivity within the regolith (Schwartz and Zhang, 2003). In order to account for the vertical differences, piezometer depth and altitude was compared with soil texture data to determine if the calculated hydraulic conductivities coincided with certain horizons and regolith layers across the transect. The deepest piezometers from each nest within a given watershed tended to draw water from similar aquifer layers; this relationship was also seen for the middle and shallowest piezometers from each nest. The calculated hydraulic conductivity for each of these horizons was compared, and when values were similar across all three nests, the values were averaged. When the calculated hydraulic conductivity was an outlier by two or more orders of magnitude, it was not included because of its potential to greatly affect the estimated average hydraulic conductivity of the watershed while not accurately reflecting the groundwater flow. The presence of the outlier may have indicated a small scale heterogeneity that would not affect the watershed as a whole.

Statistical analyses

Statistical analyses for nitrate-N were conducted on data from samples collected from June 2001 through June 2004. Due to the dry conditions that occurred during the first leaching period, groundwater was not available in two of the grazed watersheds until October 2002 (Grazed 1B and Grazed 2B), and analyses including those watersheds cover the period from October 2002-June 2004 (Figure 2-1). Therefore, data were grouped for statistical analyses in two ways: 1) beginning in October 2002, when piezometers in all six watersheds provided groundwater; 2) beginning in July 2001, when piezometers in four of the watersheds provided groundwater. In addition, to test the effect of watershed proximity to the homestead and barns (see above), the data beginning in October 2002 was classified as either “home” or “away”.

The data on nitrate-N concentration in water samples were analyzed using repeated measures GLM (SYSTAT, 1998). The seasonal average for each period of sampling (e.g. Winter 2002, Spring 2002, Summer 2002) for each piezometer nest within each watershed was used to avoid pseudoreplication over time. The model included effects of farm, piezometer nest, and watershed nested within farm. Originally, the rainfall between sampling dates and sample pH were considered as covariates in the analysis, but were removed because they were not significant. An ANOVA was also run for nitrate in pore water taken via ceramic-tipped suction lysimeters, using the effects of watershed and lysimeter length.

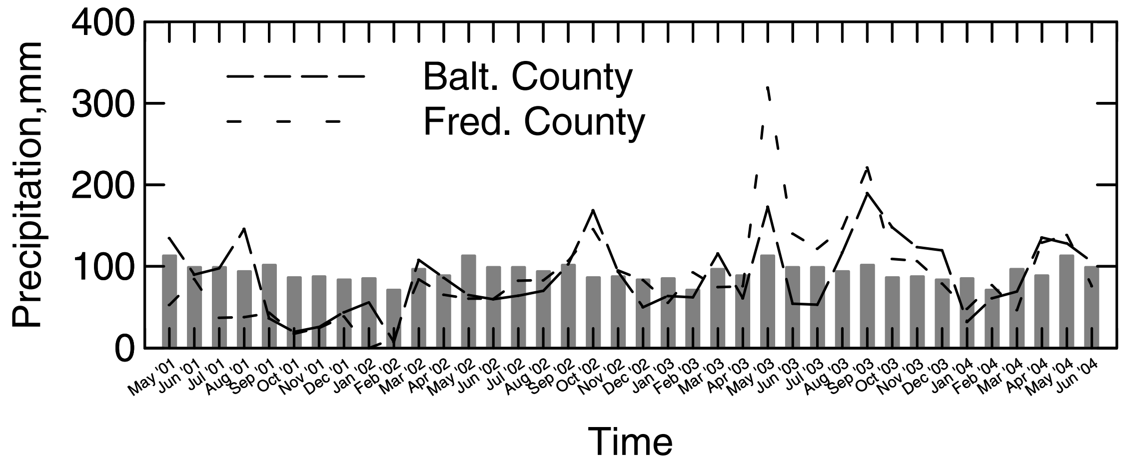

Figure 2-1: Monthly precipitation for the two farm areas sampled in 2001-2004. Grazed 1 and Confined are in Frederick County, and Grazed 2 is in Baltimore County. Bars represent 30-year average monthly precipitation for Maryland.

To test the hypothesis that groundwater nitrate-N concentrations from grazed watersheds would be below the 10 mg L^{-1} EPA standard, one-tailed t-tests were performed using watershed and season as grouping factors. Because of the drought and resulting interruption in sampling, separate t-tests were carried out for the watersheds sampled without interruption from July 2001-June 2004 and for all six watersheds from October 2002 – June 2004.

Results and discussion

Groundwater and pore water nitrate

From May 2001 to March 2002, Maryland experienced a drought (Figure 2-1), with little to no groundwater recharge. For almost a year following these extremely dry conditions, there was heavier than normal rainfall. During dry periods and through most of the growing season, plants take up both water and nitrogen, allowing little opportunity for nitrate leaching. It is during periods when precipitation exceeds evapotranspiration and groundwater recharges that nitrate leaching is most likely (Staver and Brinsfield, 1998).

Under the study watersheds, groundwater nitrate concentrations became especially high when periods of high leaching followed dry periods, flushing to the groundwater nitrate previously retained in the soil profile (Figure 2-2). This type of nitrate flush was also reported by Tyson et al. (1997) and Unwin (1986). The most notable nitrate concentration peaks occurred in the Fall 2002 and Winter 2003 (day 1000-1200 on Figures 2-2 and 2-3). Other fluctuations in nitrate concentrations in this study were similar to seasonal variations reported in other studies, with greater nitrate levels

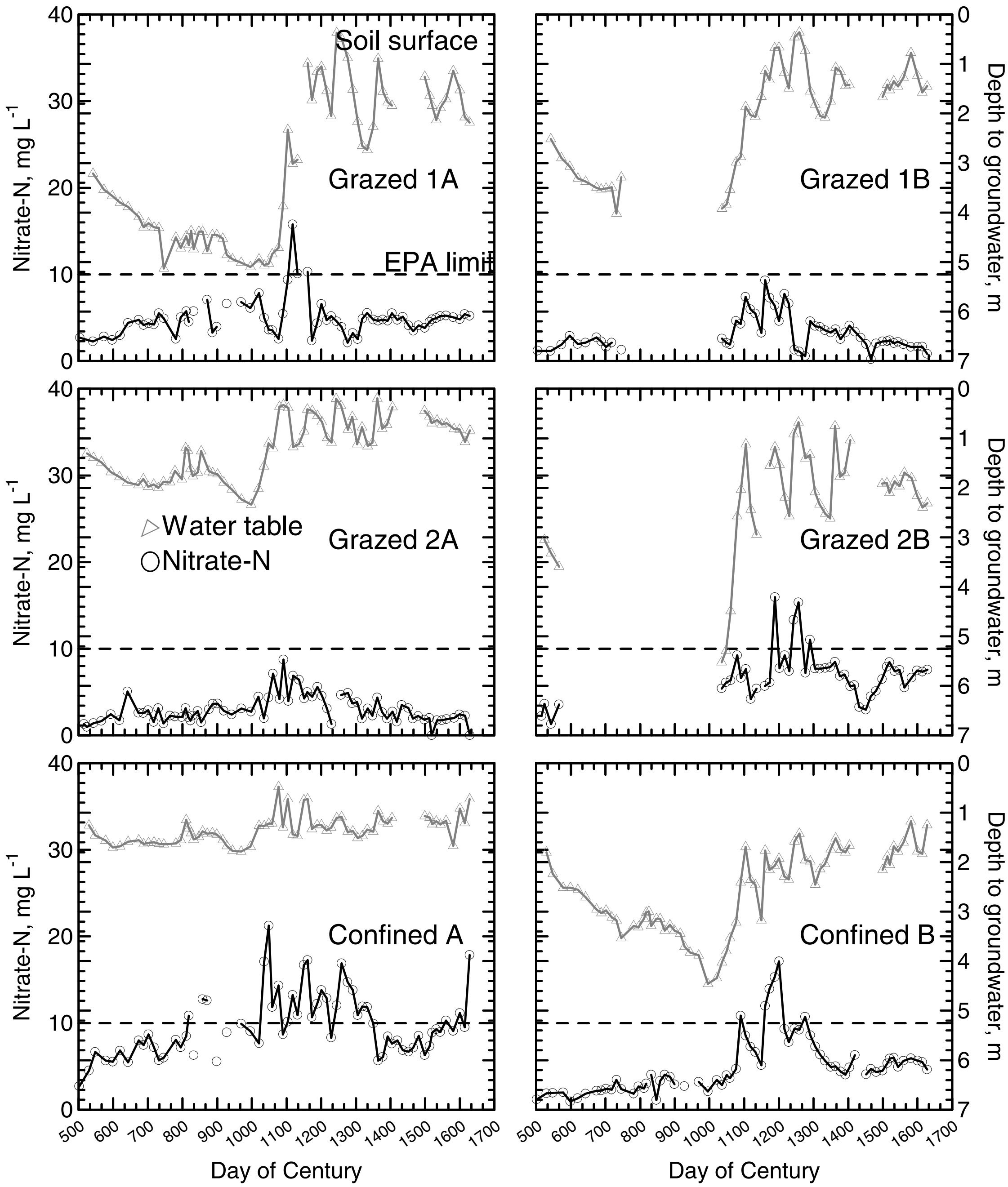

Figure 2-2: Nitrate-N in shallow groundwater and depth to groundwater in six watersheds on three dairy farms from July 2001- June 2004. Each measurement is an average of three samples collected from three piezometer nests within each watershed on one sampling date. The dotted line denotes the EPA maximum contaminant load of 10 mg NO_3-N L^{-1}.

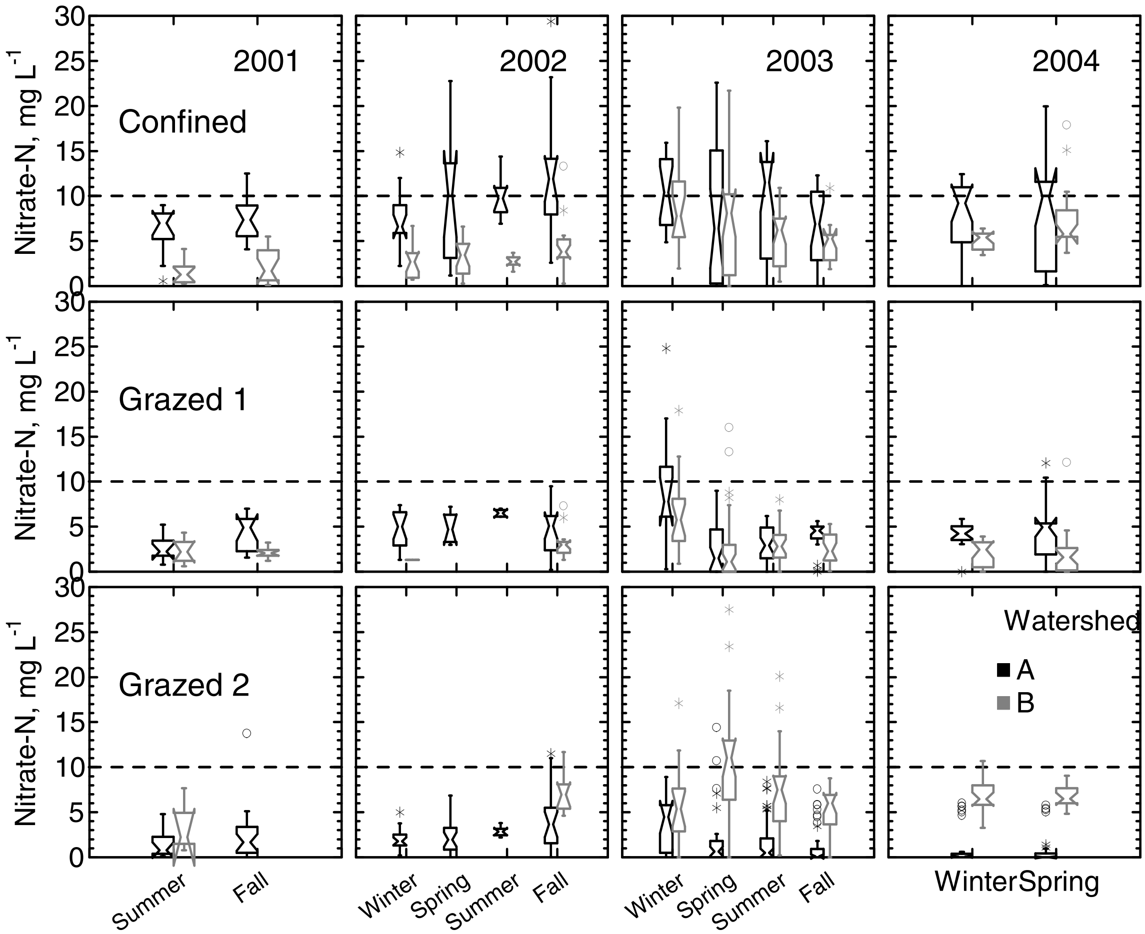

Figure 2-3: Shallow groundwater nitrate concentrations from three farms for each watershed. Samples were collected biweekly from June 2001 - June 2004. Boxes represent the central 50% of values, with the notches denoting 95% confidence intervals. The dotted line denotes the EPA maximum contaminant load of 10 mg NO_3-N L^{-1}.

measured in the wetter and colder months (Hack-ten Broeke et al., 1996; Kolenbrander, 1981; Owens et al., 1992; Saarijarvi et al., 2004; Stout et al., 1997).

Even with elevated nitrate concentrations following drought conditions, groundwater under grazed pastures did not reach the excessively high levels previous research had predicted. Predictive models developed by Stout et al (2000) suggested mean annual groundwater nitrate concentrations of 15 and 32 mg NO_3-N L^{-1} for the grazed watersheds with the stocking rates found on the two MIG farms in the present study. Instead, the mean annual nitrate-N concentrations actually observed on the four MIG watersheds were between 4 and 7 mg L^{-1}.

When all six watersheds included in the statistical model, the main effect of watershed nitrate concentration was significant, with the highest concentrations on the Confined farm (Table 2-2). Seasonal average groundwater nitrate concentrations in the two Confined watersheds ranged from 6.8 to 10.9 mg L^{-1} and exceeded the EPA maximum contaminant load ten times over the study period (Figure 2-2). The mean nitrate-N concentrations in Confined A were not significantly below 10 mg NO_3-N L^{-1} during eight of the 13 seasonal periods in the study and Confined B failed to stay below this concentration for two seasonal periods (Figure 2-2). The high nitrate-N levels in groundwater under manured cropland despite the implementation of an approved nutrient management plan is in keeping with the observation of Angle (1990) that nitrate-N concentrations are expected to exceed 10 mg L^{-1} under high yielding crops in Maryland conditions.

Table 2-2: Repeated measures GLM F-values for effects on nitrate concentrations in shallow groundwater on dairy farms. Three different models were used depending on the portion of the data set considered.

Effects in model	Six watersheds, 2002-2004		Four watersheds with uninterrupted sampling, 2001-2004[‡]		Proximity of watersheds[†], 2002-2004	
	F-value	df	F-value	df	F-value	df
Farm	13.745**	2	14.166**	2	13.745**	2
Watershed within Farm	7.671**	3	25.998**	1	NI	NI
Nest	4.138*	2	2.830*	2	4.138*	2
Proximity[§]	NI	NI	NI	NI	20.416**	1
Proximity x Farm	NI	NI	NI	NI	1.299	2
Error		10		5		10
Season	8.992***	6	4.391***	11	8.992***	6
Season x Farm	1.734	12	1.229	22	1.734	12
Season x Watershed within Farm	1.306	18	1.054	11		
Season x Nest	0.530	12	0.812	22	0.530	12
Season x Proximity	NI	NI	NI	NI	0.894	6
Season x Proximity x Farm	NI	NI	NI	NI	1.511	12
Error		60		55		60

[†] Proximity model compares watersheds closer to and farther from the homestead or barnyard. Homestead watersheds are Confined A, Grazed 1A, and Grazed 2B, those most likely to have received a greater proportion of nutrients historically.
[‡] The following watersheds were sampled from 2001-2004 without interruption: Confined A, Confined B, Grazed 1A, and Grazed 2A.
[§] Proximity refers to whether the watershed is considered to be a homestead watershed or a more distant watershed.
*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.
NI Not included in the model.

In contrast, during each seasonal period of sampling groundwater nitrate concentrations averaged 3.4 – 8.2 mg L^{-1} on the four grazed watersheds, with two exceptions. The 95% confidence interval of groundwater nitrate concentrations under grazed watersheds were significantly below the EPA maximum contaminant load except during two seasons of rapid groundwater recharge and heavier than normal precipitation levels: Winter 2003, when Grazed 1A averaged 9.6 mg NO_3-N L^{-1}, and Spring 2003, when Grazed 2B averaged 11.6 mg NO_3-N L^{-1}.

Proximity to the barnyard or homestead affected nitrate concentrations on all three study farms. Homestead watersheds on Confined and Grazed 2 had concentrations almost double those of the "away" watershed, while the homestead watershed on Grazed 1 had nitrate concentrations about 1.25 times those found under the "away" watershed (Figure 2-4).

Our study can not determine whether the differences in concentration due to proximity result from historical or current management. The homestead-related difference in nitrate-N concentrations in Grazed 2 may have related to the farmer's logistical constraints that lead him to graze the herd on the homestead watershed during the day and on the farther watershed through the night. It has been suggested that cows release more feces and urine during the day than during the night, possibly contributing to the higher concentrations found both in pore water and groundwater in Grazed 2B

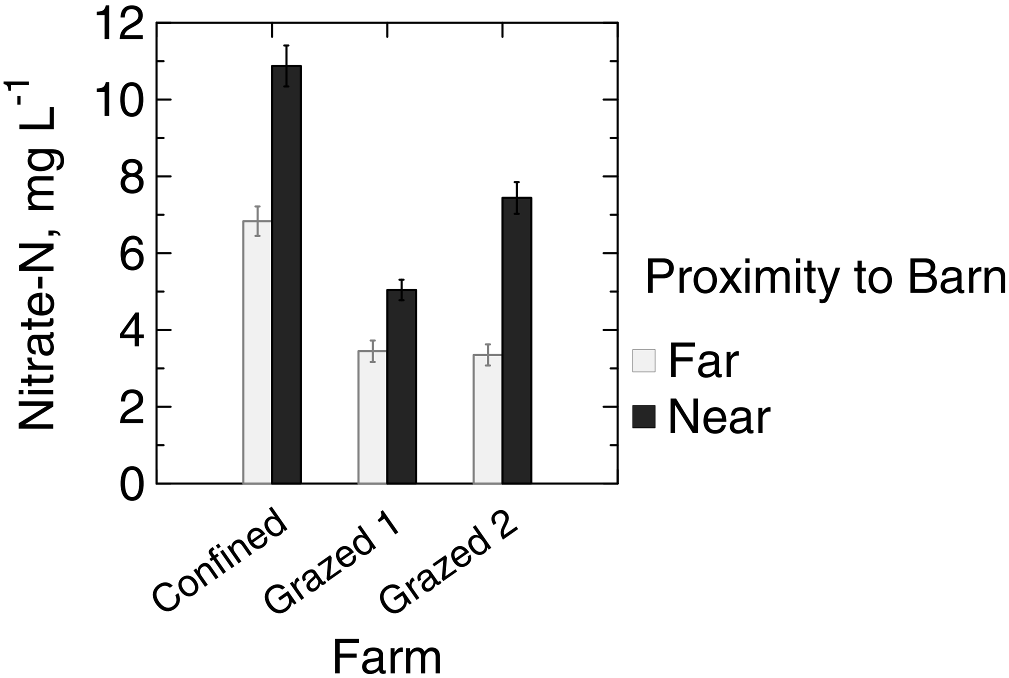

Figure 2-4: On all three farms, the watershed nearest the barns had higher mean nitrate-N concentrations in the groundwater. Bars show SE. Data for October 2002 – June 2004.

compared to Grazed 2A. On Confined, the lower part of the homestead watershed was planted in corn during the study, and the area's nearness to the barnyard combined with the N-requirements for corn may lead to continued heavy applications of manure to the area. The smaller difference between the two Grazed 1 watersheds may result from the fact that the low concentrations on the homestead watershed leave little room for an even lower concentration on a farther watershed.

Nitrate-N concentrations fluctuated widely in the three piezometer nests in each watershed, with concentration spikes occurring in different nests within a watershed on a given sampling date (Figure 2-5). However, repeated measures analyses showed that the main effect of nest was significant, with groundwater sampled from piezometer nest “a” (at the outlet) having the lowest mean nitrate-N concentration and that from nest “c” at the higher elevation having higher nitrate-N concentrations (Figure 2-6)

Nitrate-N concentration in the soil porewater from the six watersheds averaged from 0.2 to 8.0 mg L^{-1}, with the lowest average in Grazed 2A, and the highest in Grazed 2B (Figure 2-7). Analyses of variance on log-transformed data did not show an effect of watershed, depth of sample or the interaction of the two (Table 2-3).

Groundwater ammonia

Groundwater ammonia concentrations average 0.4 – 0.5 mg L^{-1} in each of the six watersheds, with very little variation. There was no effect of watershed or depth of groundwater on ammonia concentration.

Table 2-3: ANOVA F-values for effects on nitrate concentrations in pore water on dairy farms. Samples were collected on 26 dates from April 2003 – May 2004. Samples were obtained using ceramic-tipped suction lysimeters with tips at 60 and 90 cm below soil surface. ANOVA was performed on log-transformed data.

Effects	F-value†
Watershed	1.673
Nest	0.505
Depth of sample	2.332
Watershed*Depth of sample	1.656
Number of samples	132
Model R^2	0.272

† None of the effects were statistically significant at a probability level <0.05.

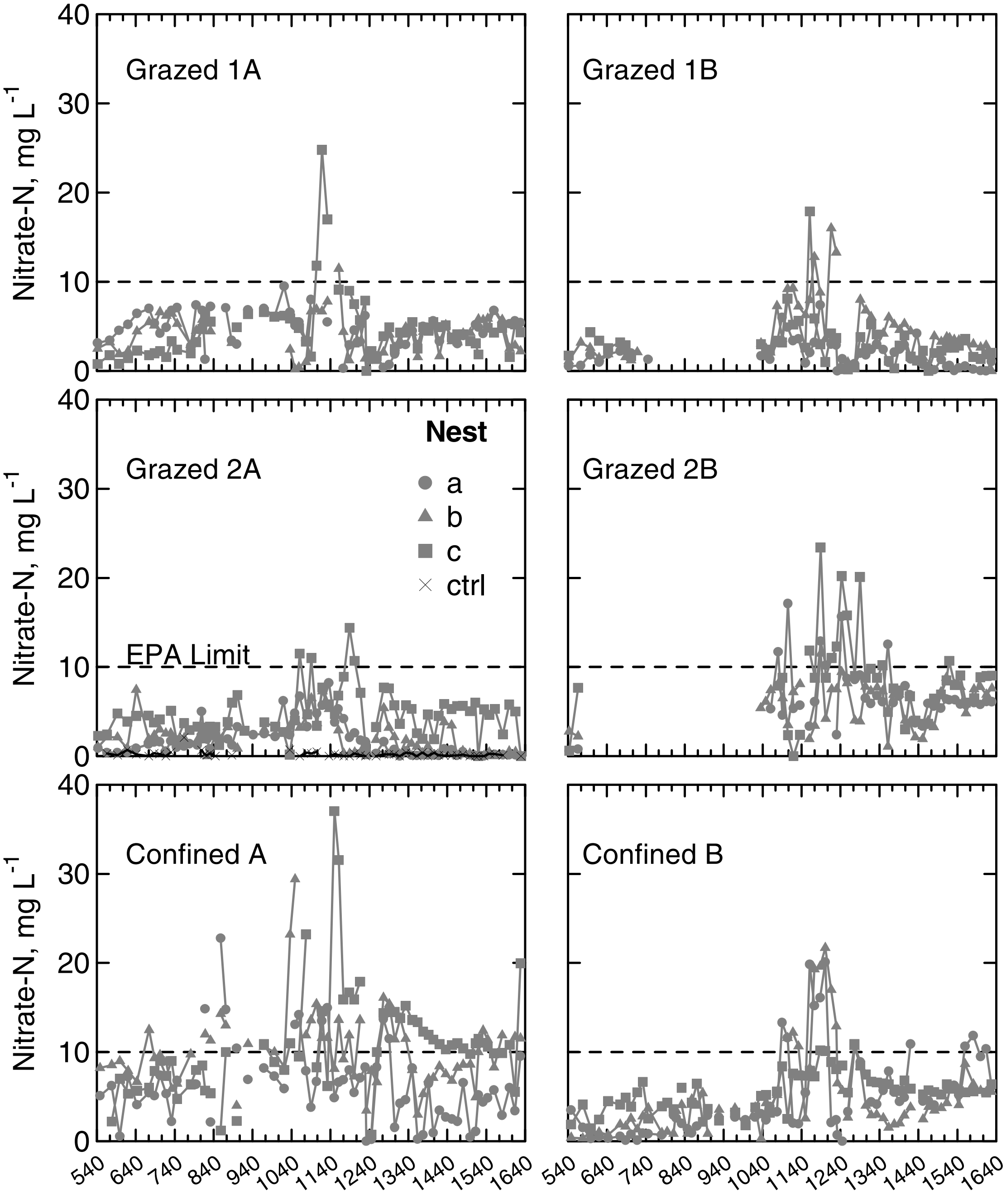

Figure 2-5: Nitrate - N in shallow groundwater samples from transects of nested peizometers in six dairy farm watersheds during the period from January 2002 - June 2004. The dotted line denotes the EPA maximum contaminant load of 10 mg NO_3-N L^{-1}.

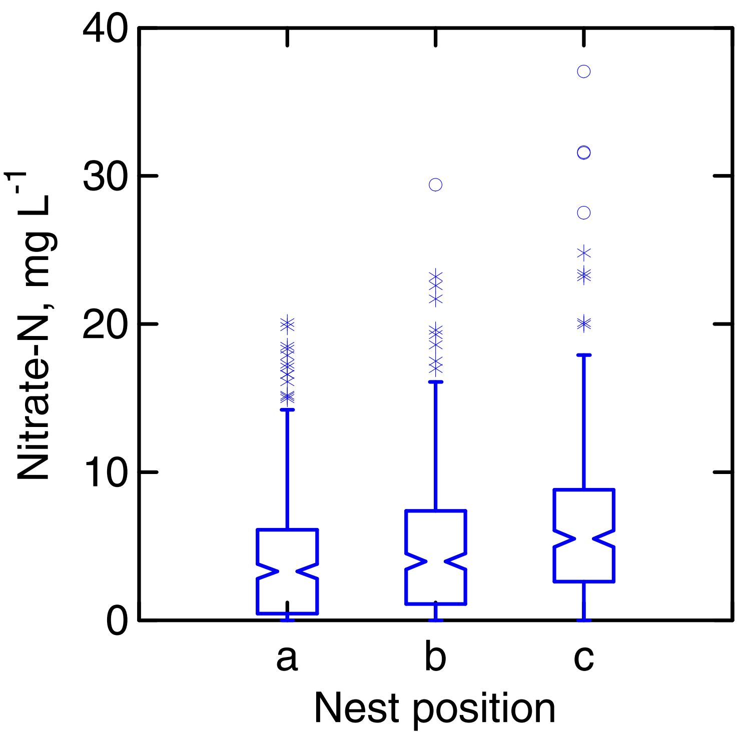

Figure 2-6: Nitrate-N concentrations were higher in piezometer nest c (father from watershed outlet) than in nest a (closest to watershed outlet). Boxes contain 50% of values and notches represent the 95% confidence interval.

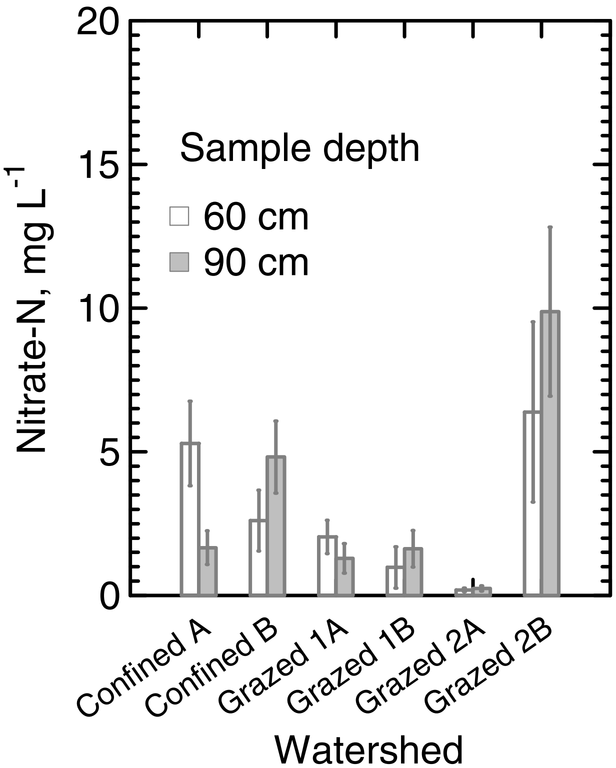

Figure 2-7: Nitrate in pore soil water in six dairy farm watersheds. Samples were collected via ceramic-tipped suction lysimeters installed under actively grazed or cropped watersheds. Collection at each farm took place on 11 dates from April 2003 through June 2004. ANOVA was performed on log-transformed data; data shown is untransformed.

Soil nitrate and mineralizable N

Based on a 16-day incubation, the upper 15 cm of soil from the six watersheds had 41.7 - 76.5 mg mineralizable N kg^{-1} (Table 2-4). The concentration of nitrate and mineralizable N in the soil was not affected by watershed or by farm. Most of the profiles had nitrate levels of 5-10 mg kg^{-1} in the surface horizons (A and E) or in the upper 30 cm (Figure 2-8).

Generally, concentrations decreased with depth. In Grazed 2, in both watersheds, profiles near the piezometer nests and watershed outlet showed a bulge in nitrate-N concentration at approximately 1 m, where nitrate concentrations increased from approximately 1 mg kg^{-1} to more than 15 (Grazer 2A) or 5 (Grazer 2B) mg kg^{-1}. The increase coincided with a horizon of coarser soil material (Appendix B). In the case of Grazer 1A profile 1, a buried A horizon was present just above the peak nitrate-N layer.

Total-N profile means from high temperature combustion analyses ranged from 1.19-1.30 mg kg^{-1} in the four grazed watersheds to 1.02 and 2.04 mg kg^{-1} in the two Confined watersheds (Appendix A). The average C present in the soil profiles ranged from 8.28 $\pm$ 1.38 mg kg^{-1} in Grazed 2B to 14.58 $\pm$ 3.67 mg kg^{-1} in Confined A, with each watershed having a C:N ratio of approximately 10:1. In the uppermost 15 cm, soil pH was 7.2 in Confined A and 6.7 in Confined B. Grazed 1 had pH levels of 6.6 and 6.0, while Grazed 2 ranged from 6.6-6.7. The percent organic matter in soil profiles was between 2 and 3.4% in the Grazed 2 watersheds, and 3.6 and 5% in Grazed 1A and 1B.

Table 2-4: Mineralizable N in soil samples from six watersheds in three dairy farms in Maryland. Each sample was collected as a composite of 10-20 cores from the upper 15 cm of a topographically distinct area within the watershed. Mineralizable N was determined by a 16-day incubation.

Watershed	Sample	Pre-incubation		Post-incubation		Mineralizable N
		NH_4-N	NO_3-N	NH_4-N	NO_3-N	(NO_3-N + NH_4-N)
		------------------mg kg^{-1}----------------				
Confined A	1	0.41	8.6	0.46	100.2	91.65
	2	0.30	12.2	0.36	68	55.86
Confined B	1	0.36	8	0.42	68.2	60.26
	2	0.21	7.6	0.24	61.6	54.03
Grazed 1A	1	0.33	7.4	0.37	99.8	92.44
	2	0.23	2.2	0.30	87.2	85.07
	3	0.40	0.6	0.46	33.6	33.06
	4	0.30	1	0.33	74.2	73.23
Grazed 1B	1	0.22	1.8	0.27	57.2	55.45
	2	0.33	2.3	0.39	81.6	79.06
	3	0.21	1	0.24	57.4	56.43
Grazed 2A	1	0.35	4.6	0.42	37.2	32.67
	2	0.34	1.8	0.37	21	19.23
	3	0.41	6	0.48	79.4	73.47
Grazed 2B	1	0.33	3	0.39	94	91.06
	2	0.49	2.8	0.56	64.8	62.08

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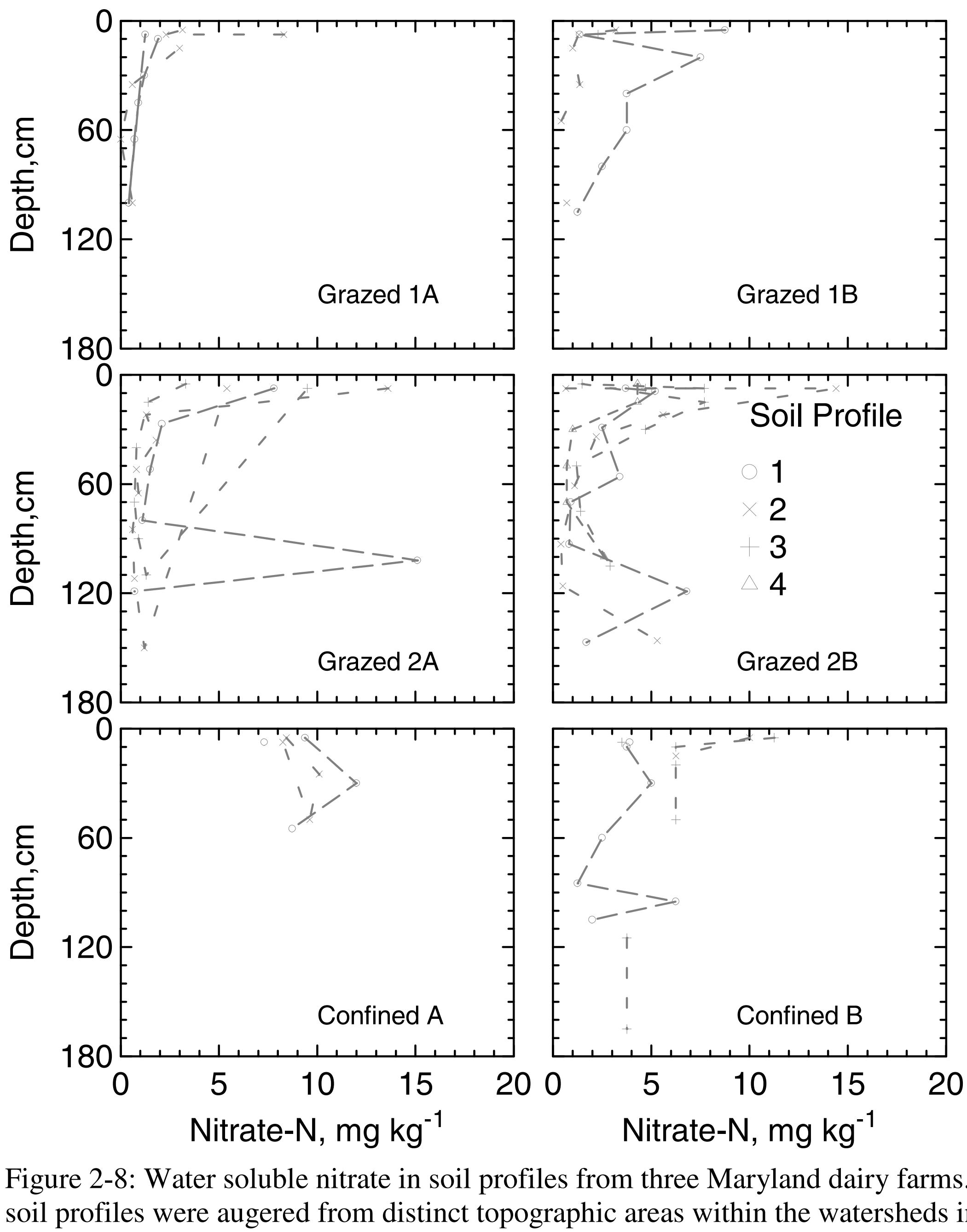

Figure 2-8: Water soluble nitrate in soil profiles from three Maryland dairy farms. The soil profiles were augered from distinct topographic areas within the watersheds in the spring of 2002 and 2003. Sample depth was dependent on the soil in the watershed; rocky conditions made deep samples unobtainable for some profiles.

Confined A and B had organic matter of 5% and 3.7% in the uppermost 15 cm. Soil fertility index values for the uppermost 15 cm in Confined ranged from 110-123 for Mg, from 29-253 for K, from 28-200 for P, and 220-455 for Ca (Appendix A). Grazed 1 had values of 108 for Mg, 61-105 for P, 126-238 for K, and 175-180 for Ca. Grazed 2 soil fertility index values were 62-107 for Mg, 61-90 for P, 99-100 for K, 158-179 for Ca.

Milk urea nitrogen

Levels on all three farms averaged 14.1 $\pm$ 0.5 mg MUN dl^{-1}, within range of 10-16 mg dl^{-1} considered optimal for efficient N utilization (Jonker et al., 1998) (Appendix A). The average on Confined was 13.64 mg MUN dl^{-1}, and all samples were within the desired range. Milk urea nitrogen on the grazed farms exceeded this range several times, with concentrations up to 18.78 mg MUN dl^{-1}. The higher MUN concentrations occurred when forage grew rapidly, in one case because of rainfall after a dry period, or in May 2002, during spring growth. The younger plant material had higher N concentrations, leading to higher MUN when the grazing animals ingested the new growth.

Groundwater N losses

Using rainfall and weather data and the WATBAL model, drainage was calculated as 268 mm y^{-1} in the Frederick County watersheds and 302 mm y^{-1} in the Baltimore County watersheds. Based on average nitrate concentrations in the groundwater and calculated drainage, the farms exported between 9.4 and 29.2 kg nitrate-N ha^{-1}, and these exports correlated to each farm's calculated N surplus (Table 2-5, Figure 2-7, Appendix A). Extrapolated to the whole farm, this would yield a total yearly

Table 2-5: Nitrate groundwater concentrations and exports. Concentrations based on groundwater samples collected biweekly from October 2002 – June 2004. Export rates based on drainage and flow calculations from WATBAL model and weather data. The estimated total farm nitrate export is calculated from the mean groundwater nitrate, drainage, and farm size. Farm surplus data is based on farm records and interviews with farmers (Appendix A). Surplus values are means for 2001-2003.

Watershed	Mean nitrate-N	Drainage	Nitrate-N export	Farm N surplus	Mean Nitrate-N export	Farm size
	mg L^{-1}	mm y^{-1}	---------- kg ha^{-1} y^{-1}----------			ha
Confined A	10.9	268	29.2	147	23.7	245
Confined B	6.8	268	18.2			
Grazed 1A	5.1	268	13.7	46	11.6	83
Grazed 1B	3.5	268	9.4			
Grazed 2A	3.3	302	10.0	64	16.2	71
Grazed 2B	7.4	302	22.3			

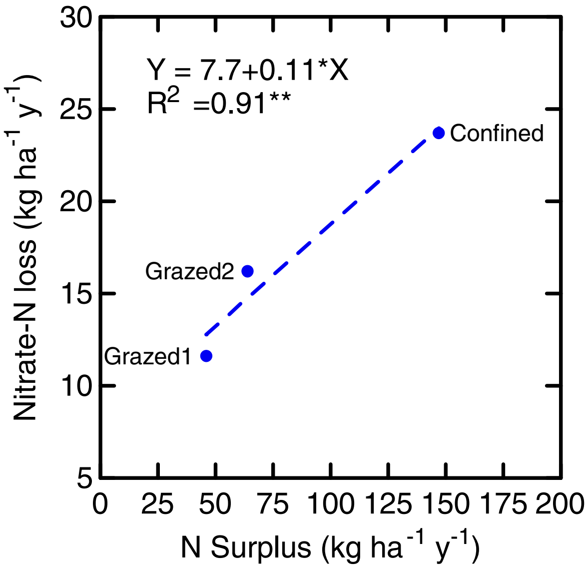

Figure 2-9: The relationship between calculated nitrogen (N) surplus and nitrate-N leaching losses to shallow groundwater on three Maryland dairy farms. Groundwater was sampled biweekly from shallow piezometers for 3 years (June 2001 – June 2004). Farm nutrient imports and exports were determined from 2001-2003 farm records. Surplus and losses shown are averages for measured periods.

export of 4459-7154 kgNO_3-N from the 245 ha Confined and yearly exports of 780-1137 kg from Grazed 1 (83 ha) and 710-1583 kg from Grazed 2 (71 ha). The much higher nitrate-N concentration found in groundwater under Confined A compared to those under Confined B may stem from the proximity of Confined A to the homestead and barns and a resulting long history prior to nutrient planning (ca. 1990) of greater manure applications. Organic N accumulated from many years of heavier manure application because of convenience to the barns may still be mineralizing and producing leachable nitrate-N. Almost twice as much mineralizable N was measured in the soil profiles in Confined A as in Confined B, supporting the hypothesis of higher levels of higher levels of nitrate-N leaching because of historical organic N accumulation. Also, Confined B was planted in alfalfa during most of the study, while Confined A was in corn. It has been suggested (Daliparthy et al., 1995; Owens et al., 1994) that alfalfa may reduce nitrate leaching compared to corn bytaking up nitrate deeper in the profile and limiting the water available for leaching.

Calculated N exports from the six watersheds were similar to the range of losses reported by others for grazed plots and monolith lysimeters. Cuttle et al (1998) estimated annual exports of 2-46 kg NO_3-N ha^{-1} from sheep-grazed plots, and Stout et al (1997) had predicteded losses under grazing of 16.8 – 31.5 kg NO_3-N ha^{-1} based in their monolith lysimeter studies. The estimated annual exports from the four grazed watersheds in this study were at the lower end of these measurements at 9.4 - 22.3 kg NO_3-N ha^{-1}, and were less than those from manured and fertilized corn (Jemison and Fox, 1994).

Conclusions

In three years of sampling, mean groundwater nitrate concentrations under grazed watersheds far lower than predicted by the model of Stout et al. (2000) and were unlikely to exceed the EPA maximum contaminant load of 10 mg NO_3-N L^{-1}. Generally, the highest nitrate-N concentrations occurred during periods of leaching and groundwater recharge, especially when following a relatively dry period. Even with the extreme rise in groundwater levels due to a very wet year following a very dry year, seasonal average nitrate concentrations under MIG were within the EPA maximum contaminant load. Thus, MIG does not appear to pose the environmental risk previous studies have suggested. Nitrate losses and exports from grazed land were lower than those from other agricultural land uses reported in the literature. These findings suggest that MIG has great potential as a Best Management Practice and as an alternative to confined feeding dairying, especially if grazing allows profitable dairy farming without herd sizes that require large imports of feed.

Changes in N held in the soil profile may provide clues as to future leaching or N storage under grazing. Our data suggests that historical N loading from uneven manure application on the farm may continue to influence N leaching more than a decade after adopting approved nutrient management plans and agronomic manure management. Grazing herds deposit N throughout the pasture with spatial variability. A long-term monitoring of soil N may be of value to further our understanding of the potential for nutrient losses under this management system. Future research should investigate long-term changes in soil properties under MIG which may be affected by the repeated deposition of urine-N or affect N losses. A relevant approach that answers the societal question of how to produce needed milk with the lowest water quality impact should correlate N losses with milk production to assess the level of environmental risk per unit of product. In the present study, not including N leached on other farms where imported feed was grown, nitrate-N leaching was 1.38 kg nitrate-N Mg^{-1}milk for Confined, compared with 2.28 and 1.69 kg nitrate-N Mg^{-1} milk for Grazed 1 and 2, respectively.

Chapter 3: Organic nitrogen in groundwater and soil on confined feeding and grazing dairy farm watersheds

Abstract

Research measuring groundwater quality and nitrogen (N) leaching has typically focused on nitrate concentrations. Recent studies have determined a significant amount of N is transferred to groundwater under forests as dissolved organic N (DON). Nitrogen losses under agriculture, especially animal-based systems, may also include substantial quantities of DON. Transects of nested piezometers and ceramic-tipped suction lysimeters were installed in six dairy farm watersheds. From January 2002 to June 2004, biweekly groundwater samples were collected. Pore water was collected when soil pore water made samples available. The samples were digested for the determination of total dissolved N (TDN). A rapid microwave digestion method was refined for use in analysis. Dissolved organic N was calculated as the difference between TDN and dissolved inorganic N. Dissolved organic N made up 12-27% of the TDN in shallow groundwater, approximately 0.9 – 21.9 mg DON L^{-1}. These losses are substantial and may affect water quality, suggesting DON should be included in future studies of N leaching, particularly under animal-based systems.

Introduction

Research in recent decades has found that much of the nitrogen (N) lost to groundwater from some temperate forest ecosystems is in the form of dissolved N-containing organic (DON) compounds (Perakis and Hedlin, 2002). Soluble organic N (SON) has been found to play a major role in soil-plant N cycling in forest systems

(Currie et al., 1996; Qualls and Haines, 1991; Yavitt and Fahey, 1986). Dissolved organic N is now considered to be an important N pool in the soil-plant-water system, and concentrations of DON may exceed those of inorganic N in forest floor leachates and subsurface runoff (Qualls and Haines, 1991; Yu et al., 1994).

More recently, research in agricultural systems has found that DON may account for a considerable proportion of the N lost to groundwater from grassland and cropland (Murphy et al., 2000; Streeter et al., 2003). Murphy et al. (2000) observed that several studies have shown SON can be extracted from agriculturalsoils in significant amounts, and that significant amounts of DON may be present in the soil solution and leachates from farmed soils. Streeter et al. (2003) reported that DON levels in isolated lakes within grassland watersheds were highly correlated wih DON in the grassland soils. They determined that over 60% of the N in both the lake and soil water was in organic form.

Dissolved organic N in soil may be divided into two pools, a pool rapidly cycled by microorganisms, and a second, more resistant pool, composed of humic substances. The latter soil pool was proposed as the principal source of DON found in freshwater lakes and streams (Jones et al., 2004). Murphy et al. (2000) suggested that DON losses may be especially large under animal-based systems because of repeated input of organic N in manures.

Typically, nitrate has been the form of N measured in leachate and groundwater, but advances in laboratory methods have made measurements of DON more feasible. In 1882, Johan Kjeldahl developed the Kjeldahl digest method to determine total N. The method required boiling a sample in sulfuric acid with a mercury or copper catalyst for several hours. The procedure allows the measurement of organic and ammonia N but not nitrate or nitrite. Precision using this method can be variable, and for the analysis of large numbers of samples this can prove costly and time consuming (Mason et al., 1999; Smart et al., 1981). An alternative digestion method uses an alkaline persulfate reagent (Smart et al., 1981). Potassium persulfate decomposes in water, releasing 2 mol H^+ per mole of potassium persulfate and causing the oxidizing solution to become acidic after oxidation:

$$K_2S_2O_8 + H_2O \rightarrow 2KHSO_4 + \frac{1}{2} O_2$$

The oxidation process converts all N and P into nitrates and orthophosphates and allows the simultaneous determination of total N and total P (Ebina et al., 1983). This method has been found to have greater efficiency and equal or greater precision than the Kjeldahl method of total N in natural waters, soil extracts, and microbial biomass extracts (Cabrera and Beare, 1993; Smart et al., 1981; Yu et al., 1994).

To accelerate the method, microwave digestions were used as an alternative heat source for both the Kjeldahl and persulfate oxidation methods. Pressurized chambers and microwave ovens allow the digestion times for the persulfate method to be reduced from 2-4 hours to 15-30 minutes (Mason et al., 1999; Johnes and Heathwaite, 1992). Autoclave and microwave digestion using the potassium persulfate method and the Kjeldahl digestion all have given similar recovery efficiency for total N and P (Maher et al., 2002).

Another method developed for total nitrogen determination is high temperature combustion (HTC). Using seawater samples, Sharp et al. (2004) compared persulfate oxidation to different HTC combustion instruments (Sharp et al., 2004). Total dissolved nitrogen measurements from the persulfate oxidation were slightly higher than those from automated HTC instruments (Shimadzu Scientific Instruments, Inc.; Columbia, MD). Although the different methods tested gave comparable results, there was variability between some of the HTC instruments used. Recovery of all known compounds was still within acceptable limits for all methods and instruments used.

There is increased interest in developing precise, economical, and efficient methodology and instrumentation for determination of TDN and DON, reflecting a growing awareness of the importance of the organic N in soil and water systems. In a three year study ofwater quality under manured cropland and management intensive grazing (MIG) pastures on Maryland dairy farms, we used a microwave digestion to measure DON in soil extracts, soil water, and groundwater. Our objective was to determine if DON was present in significant amounts in groundwater under animal-based agriculture.

Materials and methods

Sample collection

Shallow groundwater samples were collected from nested piezometers on six watersheds on three Maryland dairy farms. Two of the farms are under MIG, and are referred to as Grazed 1 and Grazed 2. The third farm is a confined feeding dairy farm, and is referred to as Confined. Grazed 1 and Confined are on adjacent tracts of land in Frederick County, and Grazed 2 is in Baltimore County Transects of nested piezometers ranging in depths from 1-3 m and 4-6 m were installed within each of the six watersheds. The nests were 18 m apart. A 90-cm and a 120-cm, 2.5-cm diameter, ceramic-tipped suction lysimeter were installed within each nest. Additional information on the three farms, sampling equipment installation and sampling methods is included in Chapter 2.

Groundwater was sampled biweekly, from May 2001 through June 2004. Samples were taken from the shallowest piezometer within each nest containing at least 1m of water. Prior to sampling, the piezometer from which the sample was to be taken was bailed and allowed 2-3 hours to recharge. Field pH was measured after the sample was taken, and beginning in January 2002, samples were acidified with 2-3 drops of 4 *M* H_2SO_4 for each 120-150 mL of sample after pH measurement. Samples were transported to the lab on ice where they were filtered under vacuum through 0.2 μm membranes before analysis. Samples were stored under refrigeration at 4°C, unless analytical procedures would not be carried out within 2 weeks, in which case they were stored at <-15°C until analysis.

Pore water samples were collected via the lysimeters when soil moisture made this possible. Samples were collected from each farm 11 times over the course of the study, for a total of 26 sampling dates. These samples were treated similarly to the groundwater samples collected via the piezometers. Complete information on pore water sample collection is found in Chapter 2.

Digestion methods

Total dissolved N (TDN) was determined by a modified alkaline persulfate microwave digestion (Cabrera and Beare, 1993; Hosomi and Sudo, 1986; Littau and Englehart, 1990; Johnes and Heathwaite, 1992). The digestion transformed TDN to nitrate which was measured using a cadmium reduction column and a Technicon autoanalyzer as described in Chapter 2. Organic N was calculated as the difference between TDN and dissolved inorganic N (DIN).

Samples were digested using an alkaline persulfate reagent (45g K_2SO_8 and 9.5 g NaOH per 1L). An aliquot of 10.0 mL of sample was put in each 120 mL high-pressure Teflon vessel (CEM Corporation; Matthews, NC), along with 10.0 mL of reagent. The vessels were closed using a mechanical capping station to achieve the torque necessary to maintain pressure. Sets of twelve samples were digested at a time. They were put on a rotating tray in the center of the microwave oven and digested at full power (actual output 675 W) for 750 seconds (12 ½ min). An aliquot of 1.0 mL buffer of 0.3 *M* NaOH in 10% NaAc was added to 10.0 mL of digestate prior to nitrate determination to raise the pH to within the range (5-9) required for the nitrate analysis. The buffer did not react colorimetrically with the cadmium column in nitrate determination. Recovery of total N by this method was tested on several known organic standards (nicotinamide adenine dinucleotide disodium salt, glutamine, urea, glycine) and averaged 70-95% (Appendix D).

Soil organic N

Soil N was determined for two or more soil profiles within each watershed. In the field, the augered soil profile was placed in a 10-cm diameter trough, divided into horizons, described with regard to color, texture, and other morphological features (Appendix B) and sampled by horizon. Samples from each horizon were taken to the lab in sealed plastic bags on ice, where they were spread out and air-dried. Sub-samples were ground and analyzed by the University of Maryland Soil Testing Lab for pH (1:1 in water), percent organic matter (loss on ignition), and Melich 1 extractable Mg, P, K, and Ca (Northeast coordinating committee on soil testing, 1995).

Using these samples, soluble soil N and P were estimated by extraction with 0.5*M* K_2SO_4 as described in Chapter 2. The extract was analyzed for DIN and also digested for total N using 10.0 mL of the filtered extract and 10.0 mL of the reagent described in the digestion methods above. The standards used for comparison were filtered potassium nitrate in 0.5*M* K_2SO_4. Soluble organic N (SON) was then determined as the difference between total N and DIN in the soil extracts.

Dissolved organic carbon and total dissolved N analyses of water samples

Selected groundwater samples to be analyzed for TOC were collected along with other samples and filtered, but were not acidified. Total organic carbon was determined using a Shimadzu TOC 5000 Total Organic Carbon Analyzer with an ASI 5000 Automatic Sample Injector (Shimadzu Scientific Instruments, Inc.; Columbia, MD). Standards were made using potassium hydrogen phthalate, and samples and standards were acidified to pH 2-3 with two drops of 2*M* HCl.

The dissolved organic N concentration was calculated as the difference between the TDN and the dissolved inorganic N (DIN). The ratios of DOC:TDN and DOC:DON were then compared to the calculated concentration of DON.

Statistical analyses

Statistical analyses for the variables dissolved organic N (DON), total dissolved N (TDN), and DON:TDN were done separately. Due to the dry conditions that occurred during the first leaching period, groundwater was not available in two of the grazed watersheds until October 2002 (Grazed 1B and Grazed 2B), and analyses including those watersheds cover the period from October 2002-June 2004 (Figure 2-1). Therefore, data were grouped for statistical analyses in two ways: 1) beginning in October 2002, when piezometers in all six watersheds provided groundwater; 2) beginning in January 2002, when piezometers in four of the watersheds provided groundwater. In addition, to test the effect of watershed proximity to the homestead and barns, the data beginning in October 2002 was classified as either "home" or "away" (see Chapter 2).

The data on DON, TDN and DON:TDN in water samples were analyzed using repeated measures GLM (SYSTAT, 1998). The seasonal average for each period of sampling (e.g. Winter 2002, Spring 2002, Summer 2002) for each piezometer nest within each watershed was used to avoid pseudoreplication over time. The model included effects of farm, piezometer nest, and watershed nested within farm. An ANOVA was also run for the N variables in pore water taken via ceramic-tipped suction lysimeters, using the effects of watershed and lysimeter length. A separate ANOVA was run for the DOC:DON ratio in groundwater samples, with the main effects of farm and watershed.

Results and discussion

Groundwater concentrations of DON in the six watersheds were substantial and were comparable to those measured in leachate from English agricultural soils (Streeter et al., 2003), where the DON:TDN ratio reached 0.60. The six watersheds averaged 1.0 – 2.0 mg DON L^{-1} in groundwater, making up an average of between 0.10 and 0.27 of the DON:TDN ratio (Table 3-1, Figure 3-1).

Table 3-1: Average concentrations of dissolved organic N (DON), total dissolved N (TDN), and the DON:TDN ratio in groundwater from six dairy farm watersheds. Samples were collected biweekly from October 2002 – June 2004 from transects of nested piezometers. Means are given with SE. Means within a column followed by the same letter do not differ significantly at $P < 0.05$.

Watershed	No. of Samples	DON	TDN	DON:TDN
		--------mg L^{-1}--------		
Grazed 1A	52	1.0 ± 0.5ab	6.2 ± 0.9b	0.12 ± 0.04
Grazed 1B	67	0.7 ± 0.3b	4.2 ± 0.8c	0.21 ± 0.04
Grazed 2A	62	1.0 ± 0.3ab	4.4 ± 0.9bc	0.27 ± 0.07
Grazed 2B	59	1.2 ± 0.4ab	8.5 ± 0.8b	0.10 ± 0.03
Confined A	85	1.9 ± 0.4a	13.1 ± 1.2a	0.13± 0.03
Confined B	92	1.6 ± 0.9ab	8.3 ± 1.4b	0.12 ± 0.03

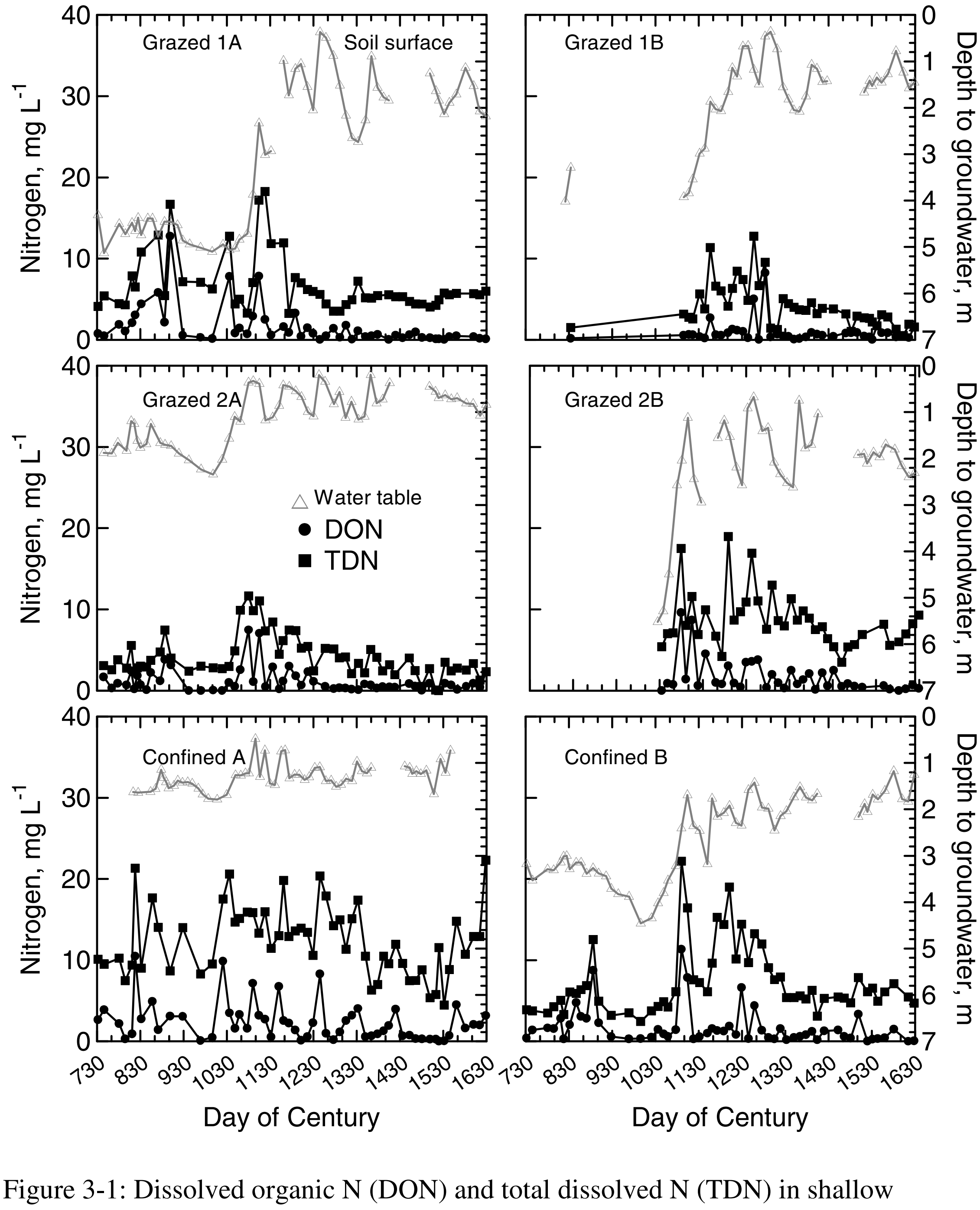

Figure 3-1: Dissolved organic N (DON) and total dissolved N (TDN) in shallow groundwater and depth to groundwater under six dairy farm watersheds. Samples were collected from January 2002-June 2004. Each measurement is an average of three samples collected from three piezometers nests within each watershed on one sampling date.

The subsequent increase in groundwater DON is similar to the flush of nitrate observed (Chapter 2) and may be attributed to the movement of N retained in the soil profile during drier conditions (Hack-ten Broeke and van der Putten, 1997; Stout et al., 2000a). The first year of monitoring took place during a period of drought with little, if any, groundwater recharge. During the second leaching season, precipitation exceeded normal conditions, with heavy rainfall through the Fall 2002 and Spring 2003. This may have caused a spike in groundwater nutrient concentrations, visible in Figure 3-1.

The only effect on groundwater DON was a difference among farms for the four watersheds sampled from January 2002 – June 2004 (Table 3-2). Concentrations of DON were higher in groundwater on Confined than in groundwater from the two grazed farms (Figures 3-2 and 3-3).

Total dissolved nitrogen also increased with groundwater recharge in Fall 2002 – Winter 2003. Total dissolved nitrogen concentration means ranged from 4.2 – 13.1 mg L^{-1}, with higher levels from Confined and Grazed 2B (Table 3-1). There was also an effect of watershed on TDN for the model comparing the four watersheds sampled throughout (Table 3-2). Again, Confined watersheds had the highest concentrations.

The critical observation is the presence of 1-2 mg DON L^{-1} under the six watersheds. Between 10-27% of the groundwater TDN is DON under these grazed and cropped watersheds. There was no difference in the DON:TDN ratio on the three farms, suggesting dairy farming, be it confined feeding or grazing, produces similar proportions of DON:TDN. The similarity of the DON:TDN ratio over time and in all six watersheds, even with higher concentrations of both DON and TDN in the Confined watersheds, points to a consistent presence of DON.

Table 3-2: Repeated measures GLM F-values for effects on dissolved organic N (DON), total dissolved N (TDN), and DON:TDN concentrations in shallow groundwater on dairy farms. Initially, three different models† were used depending on the portion of the data set considered. The only model with significant effects was that testing the four watersheds sampled from January 2002 – June 2004, shown below. ‡

Effects in model	DON		TDN		DON:TDN	
	F-value	df	F-value	df	F-value	df
Farm	8.359*	2	11.207*	2	0.342	2
Watershed within Farm	5.675	1	32.844**	1	0.237	1
Nest	2.144	2	0.549	2	1.500	2
Error		4		4		4
Season	0.855	9	2.499*	9	4.030**	9
Season by Farm	0.494	18	0.813	18	1.001	18
Season by Watershed within Farm	1.313	9	0.330	9	0.845	9
Season by Nest	0.756	38	0.813	18	1.506	18
Error		36		36		36

† A proximity model compared watersheds closer to and farther from the homestead or barnyard. Homestead watersheds are Confined A, Grazed 1A, and Grazed 2B, those most likely to have received a greater proportion of nutrients historically. A second model tested all six watersheds from October 2002 – June 2004.

‡ The following watersheds were sampled from 2001-2004 without interruption: Confined A, Confined B, Grazed 1A, and Grazed 2A.

*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.

NI Not included in the model.

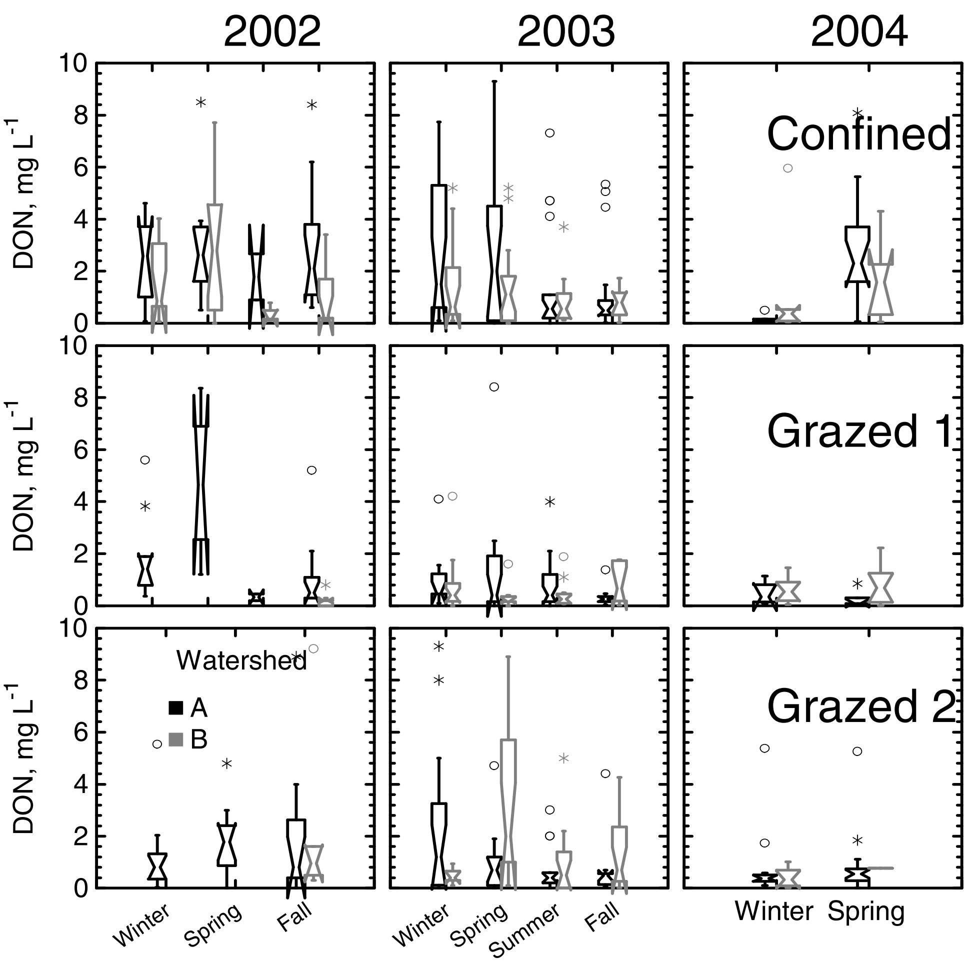

Figure 3-2: Dissolved organic N (DON) in shallow groundwater under six dairy farm watersheds. Samples were collected bi-weekly from January 2002 – June 2004. Sample availability was limited in Grazed 1B and Grazed 2B until Fall 2002. Boxes represent the central 50% of the values, with the notches showing the 95% confidence intervals.

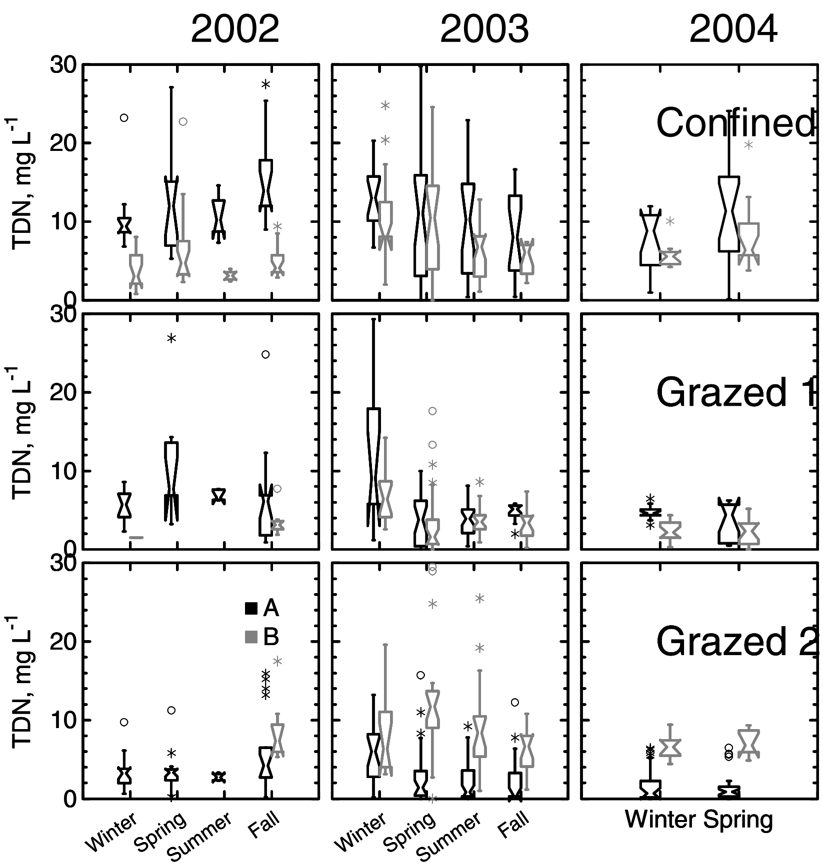

Figure 3-3: Total dissolved N (TDN) in shallow groundwater under six dairy farm watersheds. Samples were collected bi-weekly from January 2002 – June 2004. Sample availability was limited in Grazed 1B and Grazed 2B during the first three seasons of 2002. Boxes represent the central 50% of the values, with the notches showing the 95% confidence intervals.

Pore water dissolved organic nitrogen

Concentrations of DON in pore water were approximately 1 mg L^{-1}, with no significance of watershed or depth of sample collection (Table 3-3). There was also no significant effect of depth or watershed in concentrations of TDN. Sample variability was relatively low, and the ratio of DON:TDN was between 0.40-0.70 for the lysimeters within each watershed. The exceptions to this range of ratios were in shallower samples from Grazed 1B and samples from Grazed 2A, which had pore water with a proportion of DON:TDN closer to 1.00 (Figure 3-4). The proportions of DON in the TDN in soil pore water are 3 to 4 times greater than those found in groundwater, suggesting that most of the DON present at 60 to 90 cm depths may mineralize before it reaches the groundwater.

DOC:DON ratio in groundwater

There was a significant effect of watershed on the ratio of DOC:DON in groundwater on the three farms. This effect was due to the presence a very high ratio in samples from Grazed 2A; the proportion of DOC:DON in the other five watersheds was not significantly different from one another (Figure 3-5). In those five watersheds, the ratio ranged from 3.0 to 5.9, and in Grazed 2A, the ratio was 19.3. All six watersheds had soil C:N ratios of approximately 10 (see Chapter 2).

The high proportion of DOC:DON found in Grazed 2A could be related to the buried A horizon found in the watershed. The organic matter from the buried A horizon could have increased the magnitude of nitrate-N losses through denitrification as was found in a similar study by Weil et al. (1990), leaving DON as a larger proportion of the total N in solution. The buried A horizon occurred at or near the depth that much of the

Table 3-3: ANOVA F-values for dissolved organic N (DON) and total dissolved N (TDN) in pore water from six dairy farm watersheds. Samples were collected on 26 dates from April 2003 – May 2004. Samples were obtained using ceramic-tipped suction lysimeters with tips at 60 and 90 cm below soil surface.

Effect	DON	TDN	DON:TDN
Watershed	0.538	1.672	2.525*
Nest	4.238	0.419	2.237
Depth of sample	1.288	1.496	4.349*
Watershed*Depth of sample	0.562	2.122	3.167*
Number of samples	150	169	137
Model R^2	0.159	0.338	0.356

*,**,*** for $P < 0.05$, 0.01, and 0.001, respectively

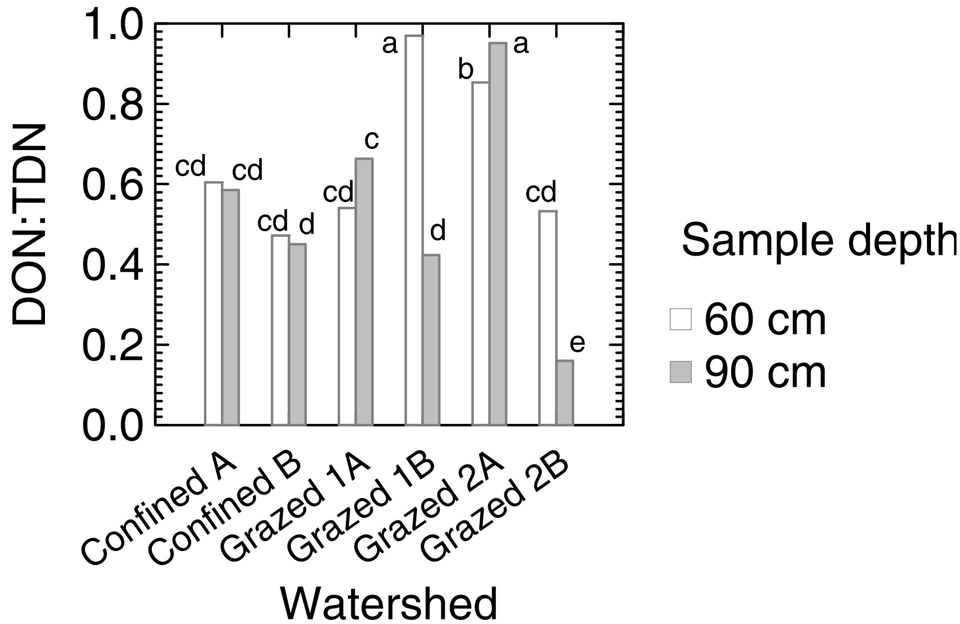

Figure 3-4: Ratio of pore water dissolved organic nitrogen (DON) to total dissolved nitrogen (TDN) from six dairy farm watersheds. Samples were collected via ceramic-tipped suction lysimeters installed under actively grazed or cropped watersheds. Collection took place on 11 dates from April 2003 – June 2004. Data was log-transformed for ANOVA analysis; un-transformed data is shown. Bars with the same letter do not differ significantly at $P < 0.05$.

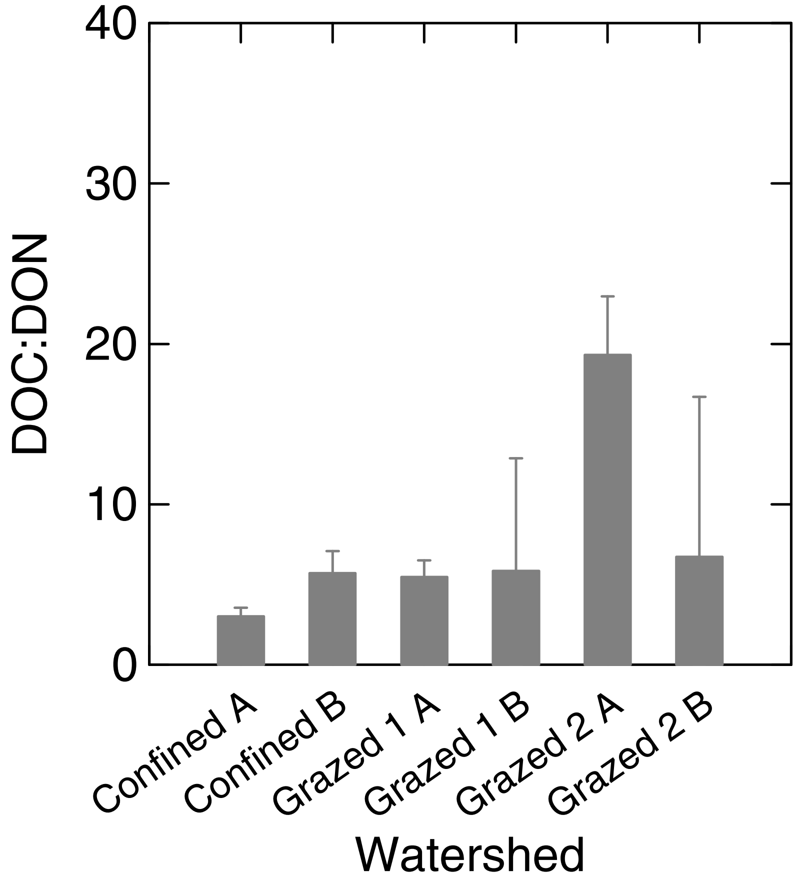

Figure 3-5: Dissolved organic carbon to dissolved organic nitrogen (DOC:DON) in shallow groundwater samples collected from six watersheds on three Maryland dairy farms. Samples were collected from shallow piezometers on two dates in October and November 2003. Bars indicate SE.

groundwater and pore water collected, and organic matter from this horizon would be likely to affect the ratio of DOC and DON found in leachate and soil pore water.

Soil nitrogen

Soluble organic nitrogen and total soluble nitrogen extractable from the soil were at higher concentrations closer to the surface, and levels decreased deeper in the profile (Figures 3-6 and 3-7). Surface concentrations of SON were approximately 50 mg kg^{-1} and decreased to 10 mg kg^{-1}. Similarly, TSN concentrations were at 60 mg kg^{-1} in the upper 15-20 cm of the soil profile and dropped to 10-15 mg kg^{-1} with depth. There were no significant differences by watershed or proximity.

Groundwater DON and TDN losses

Overall, substantial concentrations of DON were found in shallow groundwater under three Maryland dairy farms. Concentrations ranged from 0.9 to 1.9 mg DON L^{-1}, levels which could result in annual exports of 2.7 – 6.7 kg DON ha^{-1}, based on calculated drainage and measured groundwater concentrations (Table 3-4). These losses are not inconsequential. The watersheds Grazed 1A and B and Grazed 2A had the smallest losses; Confined A and B and Grazed 2B had the largest (Figure 3-8). Estimated DON farm losses were between 303 and 373 kg y^{-1} from the two grazed farms and 1703 kg y^{-1} from the Confined farm. As 10-27% of the TDN was in DON, future studies of water quality under animal-based agriculture should include measurements of organic N.

Table 3-4: Groundwater DON and TDN concentrations and exports. Concentrations based on groundwater samples collected biweekly from October 2002 – June 2004. Export rates based on drainage and flow calculations from WATBAL model and weather data. The farm export estimates are the averaged TDN exports per ha; farm surplus data is based on farm records and interviews with farmers (Appendix A). Surplus values are means for 2001-2003.

	Mean DON	Mean TDN	Drainage	DON export	TDN export	Farm N Surplus	Mean TDN export	Farm size
Watershed	-----mg L^{-1}-----		mm y^{-1}	-------------kg ha^{-1} y^{-1}-------------				ha
Confined A	2.5	13.7	268	6.7	36.7	147	32.4	245
Confined B	2.7	10.5	268	7.2	28.1			
Grazed 1A	1.7	7.0	268	4.6	18.8	46	15.9	83
Grazed 1B	1.0	4.8	268	2.7	12.9			
Grazed 2A	1.6	5.5	302	4.8	16.6	64	22.5	71
Grazed 2B	1.9	9.4	302	5.7	28.4			

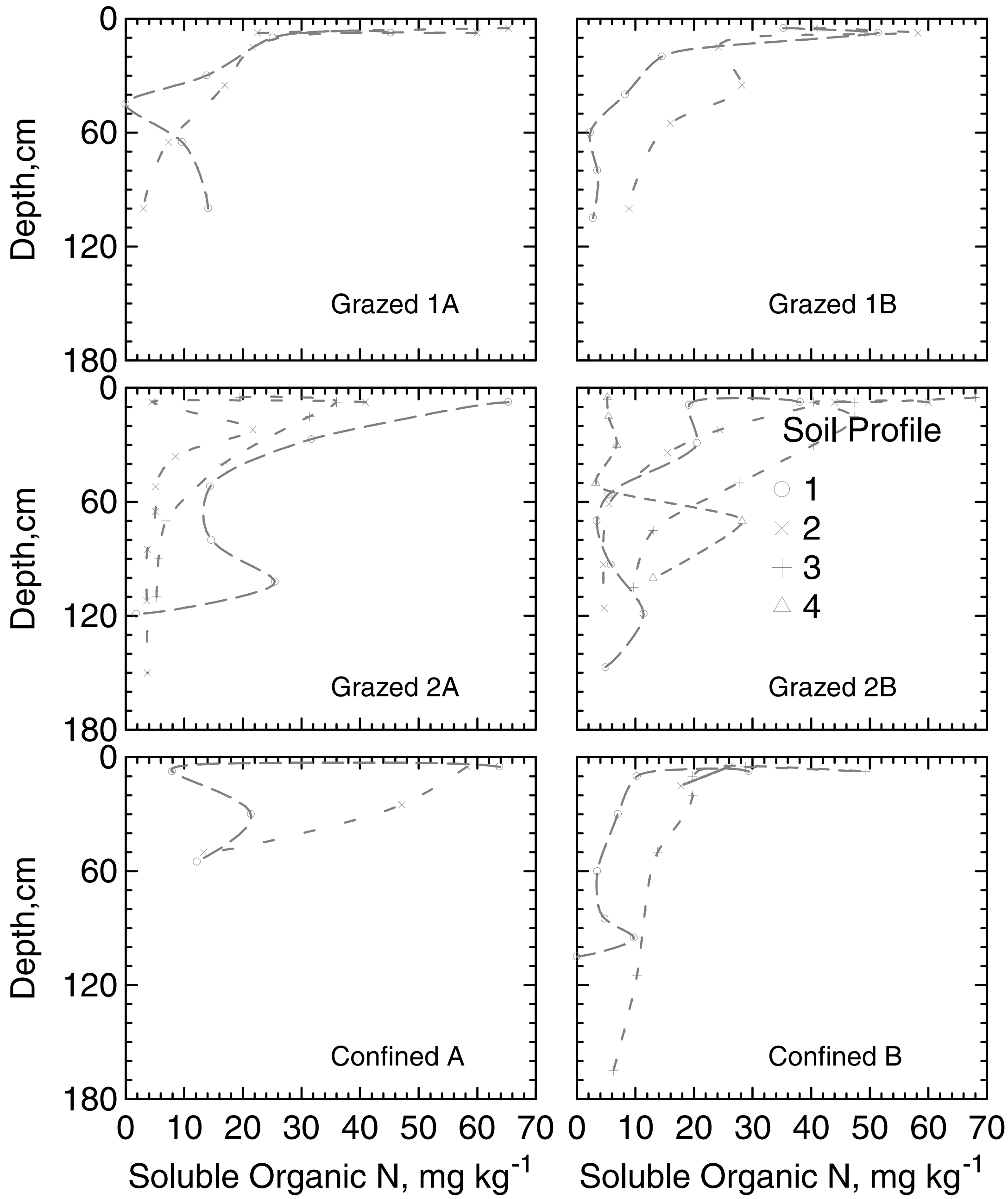

Figure 3-6: Soluble organic nitrogen in soil profiles on six dairy farm watersheds. The soil profiles were augered from distinct topographic areas within the watersheds in the spring of 2002 and 2003. Sample depth was dependent on the soil in the watershed; rocky conditions made deep samples unobtainable for some profiles.

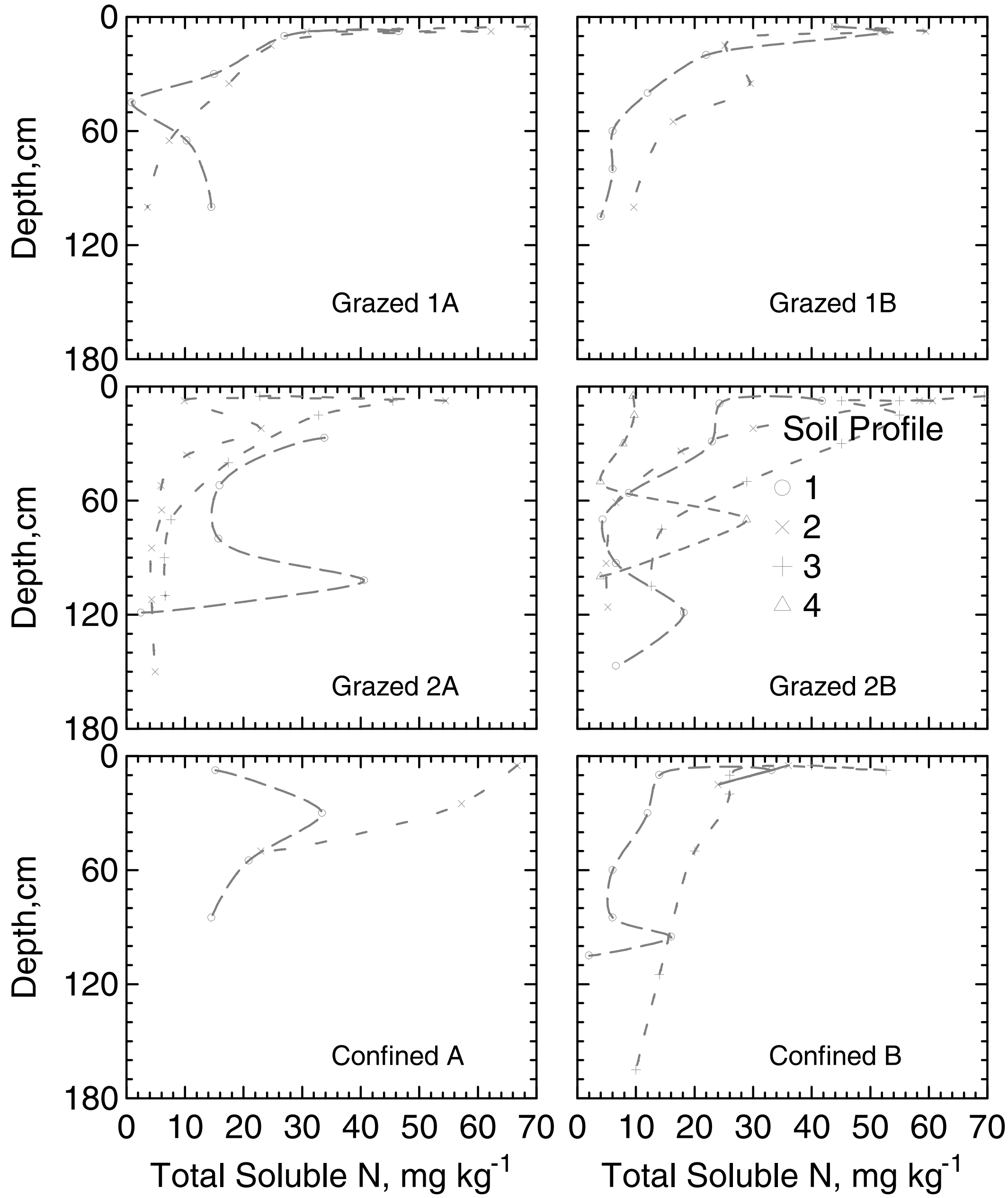

Figure 3-7: Total soluble nitrogen in soil profiles from six dairy farm watersheds. The soil profiles were augered from distinct topographic areas within the watersheds in the spring of 2002 and 2003. Sample depth was dependent on the soil in the watershed; rocky conditions made deep samples unobtainable for some profiles.

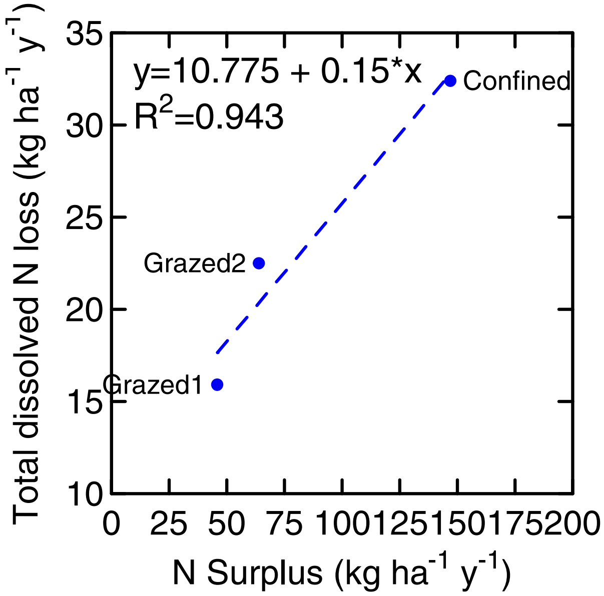

Figure 3-8: The relationship between calculated nitrogen (N) surplus and total dissolved N leaching losses to shallow groundwater on three Maryland dairy farms. Groundwater was sampled biweekly from shallow piezometers for 2 ½ years (January 2002 – June 2004). Farm nutrient imports and exports were determined from 2001-2003 farm records. Surplus and losses shown are averages for measured periods.

Annually, TDN losses in groundwater were from 12.8 – 28.4 kg ha^{-1} from grazed watersheds and 28-36 kg ha^{-1} from the cropped watersheds. Estimated yearly TDN losses were from 1320 to 1598 kg y-1 from the two grazed farms and 7938 kg y-1 from the Confined farm. Losses were strongly correlated with N surpluses for each farm, suggesting nutrient balance and management is critical to avoiding nutrient pollution (Figure 3-9, Appendix A).

Conclusions

Management intensive grazing did not appear to increase the magnitude of organic N losses to the groundwater. The proportion of DON:TDN was fairly constant, at approximately 16-17% in four of the six watersheds, enough to warrant consideration in future water quality studies. It remains to be determined whether the DON present is likely to mineralize rapidly and provide nutrients to microorganisms and plants, making it an immediate risk for eutrophication.

Chapter 4: Inorganic phosphorus in groundwater and soil on confined feeding and grazing dairy farm watersheds

Abstract

Phosphorus (P) losses have been linked to runoff and erosion from agricultural land use, causing nutrient pollution and subsequent eutrophication. Recent research has found that phosphorus leaching is also a pathway for significant P transfer. To determine the nature of P leaching losses under dairy farming, shallow groundwater was monitored for two and a half years on three Maryland dairy farms. Two of the dairy farms are under management intensive grazing (MIG), and the third farm is managed as a confined feeding system. Biweekly samples were collected from transects of nested piezometers in two watersheds on each farm. Samples were also collected via ceramic tipped suction lysimeters to measure P content in pore water. Groundwater dissolved reactive P (DRP) concentrations ranged from 0.034 – 0.233 mg L^{-1}, and differences in concentration were not related to management. The lowest concentrations, averaging 0.03-0.04 mg DRP L^{-1}, were found under one of the MIG farms. The soils there are mapped as having underlying calcareous geologic material which we hypothesized to sorb P and reduce groundwater concentrations. Groundwater on all three farms exceeded the EPA critical limit for surface water dissolved reactive P content.

Introduction

Pollution from cropland, pasture, and rangeland is the principal cause of degradation in 60% of the damaged river miles and 50% of the affected lake acres in the U.S. Nutrients from all sources constitute the leading pollutants in lakes and the third greatest in rivers (Parry, 1998). Phosphorus (P) loading of aquatic systems is of great environmental concern because P is the limiting nutrient in most freshwater bodies and leads to eutrophication at concentrations as low as 0.01 mg L^{-1} for dissolved reactive P (DRP) and 0.1 mg L^{-1} for total P (TP) (McDowell and Sharpley, 2004; USEPA, 2001).

Dairy farming is a major source of nutrient pollution in certain regions. Often dairy herds are larger than the farm can support, requiring feed, minerals and fertilizer imports. The imports introduce P into the dairy farm nutrient cycle when excreted in manure; approximately 75% of the P consumed is excreted, almost entirely in the feces (Ebeling et al., 2002; Powell et al., 2001). The resulting manure may be more than the farm requires for crop production or can assimilate, leading to over-application of manure and increasing the risk of P transfer in surface or subsurface runoff (Butler and Coale, 2005; Daniel et al., 1998; Ebeling et al., 2002; Knowlton et al., 2001; Lanyon and Beegle, 1989; Powell et al., 2001).

In a review of P imports and exports on 46 confined and MIG dairy farms in the northeast, two-thirds of the imported P was in feed and minerals, and 57% of imported P was not exported in milk or other products (Anderson and Magdoff, 2000). Phosphorus loading also may occur because farmers typically exceed current recommendations for dietary P, providing rations of up to 4.8 g P kg^{-1} dry matter (National Research Council, 2001). Concerns about negative effects on milk production and herd reproduction appear to keep farmers from reducing the P content of rations (Ebeling et al., 2002). However, research has shown that P intake can be reduced from current levels by 30-40% without decreasing milk production or fertility (Satter and Wu, 1999; Valk et al., 2000). In addition to decreasing the amount of P imported, reducing supplemental P in the diet will also reduce P in excreta, potentially reducing P losses in runoff (Ebeling et al., 2002).

Phosphorus transfer

Historically, P was thought to be lost exclusively through surface runoff, and, indeed, most nonpoint source P pollution is due to surface runoff and sediment transport, (Schlegel et al., 1996; Anderson and Magdoff, 2000). Much of the P transfer via runoff occurs when manure application is followed closely by a runoff event (Sherwood and Fanning, 1981; Nash et al., 2000). Sherwood and Fanning (1981) found that when high rates of slurry P were applied, dissolved reactive P (DRP) in runoff was exceedingly high for at least six weeks after the application, decreasing exponentially after application. The greatest P losses tend to occur during the largest storm events (Hillbricht-Ilkowska et al., 1995).

Recent research suggests that leaching is also a potential route for P transfer, albeit at a lower level than in runoff (Hansen et al., 2002; Kleinman et al., 2003; McDowell et al., 2001). Exports of P in sediments eroded from arable land are often greater than 3 kg ha^{-1} y^{-1}, while P exports in leaching may range from 0.1-0.9 kg ha^{-1} y^{-1}(Hillbricht-Ilkowska et al., 1995). Leachate P concentrations from monolith lysimeters were frequently high enough to be associated with eutrophication, even under lysimeters receiving generally typical P applications (Turner and Haygarth, 2000; Leinweber et al., 1999).

In a comparison of P leaching from mineral fertilizer, dairy manure and compost, and poultry manure and compost applied to monolith lysimeters, the highest concentration of P in leachate was found under dairy manure, at 88 mg DRP and 148 mg total P L^{-1} (McDowell and Sharpley, 2004). Leaching losses decreased over time after P application, but there appeared to be a threshold soil P level (100 mg DRP and 130 mg total P kg^{-1} in Mehlich-3 soil extracts) beyond which P concentrations in drainage waters were estimated to take 3-6 years to decrease to acceptable limits. In contrast, a study following a seven-year crop rotation measured P leaching from four soilsin monolith lysimeters receiving inorganic fertilizers, manure, or grass compost and found no difference in leachate P concentrations between the applications, suggesting manure applied in appropriate amounts should not be a risk for increased leaching (Ulen, 1999).

Phosphorus losses in leachate from agricultural, sandy soils ranged from 1-10 kg ha^{-1} y^{-1}, depending on the level of water-soluble P present (Schoumans et al., 1997). Annual losses in drainage water from grass ley growing on heavy clay soils were less than 1 kg P ha^{-1} (Turtola and Jaakkola, 1995). These findings and those by Eghball et al. (1996) and Ulen et al. (1998) suggest subsurface flow and leaching may be more significant pathways for P transport than previously believed.

On 11 sites with different chemical properties and management histories, Hooda et al. (2000) determined the amount of P desorbed and susceptible to leaching was most closely related to the level of P saturation for that soil. If more P is added to a soil than is removed in crop harvest and runoff, soil build-up of P will occur (Powell et al., 2001; Wang et al., 1999). This accumulation, as well as the nature of dairy manure, may lead to P leaching (Breeuwsma et al., 1995; Hooda et al., 2000; Kleinman et al., 2003). Butler and Coale (2005) found P levels under fields receiving dairy manure were higher than under those with broiler litter application and postulated that P sorption sites may be blocked by organic matter, leading to a greater likelihood of P leaching under dairy manure applications.

While nitrate leaches at a rate relative to a soil's hydraulic conductivity, phosphorus leaching is influenced by several additional factors. The two main mechanisms identified for P transport during leaching are 1) attachment to mobile colloids flowing through pores or 2) movement in solution through the soil matrix (Butler and Coale, 2005; McGechan, 2003). Inorganic forms of P are more likely to sorb to colloids within the soil matrix. Pore flow, or preferential flow, occurs when pore spaces are water-filled, and P is more likely to leach via these pathways during wet conditions or in soluble forms (McGechan, 2003). Preferential flow may occur when solutes move through only some of the total available pore space, such as worm or root channels, or cracks (Ulen et al., 1998). These pathways are more important for P loss during storm events that follow periods of drought (Simard et al., 2000). Depending on the type of flow through pores, sorption rate may have more of an effect on P leaching than sorption capacity (Gjettermann et al., 2004).

Until very recently, models of nutrient pollution assume that P losses from agricultural land are due to runoff and erosion (Hansen et al., 2002; Toor et al., 2005). However, with recent evidence suggesting the potential importance of P leaching, this study aimed to characterize P in groundwater under six dairy farm watersheds as relating to dairy management and geologic factors.

Materials and methods

Site selection

Three Maryland dairy farms, two based on intensively managed grazing and one based on conventional confined feeding, were selected for this study, and are hereafter referred to as Grazed 1, Grazed 2, and Confined. All three farms have been primarily dairy operations for at least 30 years and have been in farming that included livestock for at least 100 years prior to the study. Further information on the three farms is included in Chapter 2.

Within each farm, we identified two watersheds for groundwater monitoring. In each watershed, we installed piezometers in a transectconsisting of three nests spaced 18 m apart, starting at the watershed discharge point and following the flow line upslope. Each nest was covered by a 20-cm high wooden box to protect the instruments from cattle and machinery. Initially, we installed three piezometers within each nest, extending to 1, 2, and 3 m below the seasonal high water table in 2001. We installed a fourth piezometer in several nests after drought conditions caused groundwater levels to drop below the reach of the deepest piezometers. Details on piezometer depths are given in Appendix B. One control piezometer was installed on each farm upslope of the farm management activities, for the purpose of measuring baseline concentration levels coming onto the farm. Within each nest box, two 2.5-cm diameter, ceramic-tipped suction lysimeters (90 and 120 cm in length) were installed at a 45° angle to the ground surface so that the tips extended horizontally approximately 30 or 45 cm beyond the box.

Groundwater sampling

Groundwater samples were collected biweekly, beginning in May 2001, as described in Chapter 2. Beginning in January 2002, each sample was acidified to pH<3 with 2-3 drops of 4*M* H_2SO_4 after measuring pH. The samples were returned to the lab on ice, where they were stored under refrigeration at 4°C until analysis.

Soil pore water samples were collected using the lysimeters as described in Chapter 2. Pore water samples were obtained 11 times on each farm between June 2002 to June 2004, mainly in the fall and spring months.

Statistical analyses on dissolved reactive P (DRP) were conducted on samples collected from January 2002 through June 2004. Samples collected prior to 2002 were not acidified in the field and may have had bacterial growth which would affect analyses.

Lab analyses

Water samples were filtered under vacuum and stored as described in Chapter 2. Filtered samples were analyzed for DRP using the ascorbic acid method (American Public Health Association et al., 1992). A 10.0 mL aliquot of sample was used, with 1.6 mL of the reagent containing 5 *N* sulfuric acid, ammonia molybdate solution, 0.1 *M* ascorbic acid, and potassium antimonyl tartrate solution. Absorbance at 880 nm was measured and compared to a standard curve of six standards ranging from 0 to 1 mg L^{-1}.

Soil P

Two or more soil profiles within each watershed were sampled using a bucket auger and described. The augered profile soil was laid out in a 10 cm diameter trough, divided into horizons, described with regard to color, texture and other morphological features (Appendix B) and sampled by horizon. Samples from each horizon were taken to the lab in sealed plastic bags on ice, where they were spread out and air-dried. Sub-samples were ground and analyzed by the University of Maryland Soil Testing Lab for pH (1:1 in water), percent organic matter (loss on ignition), and Melich 1 extractable, Mg, P, K, and Ca (Northeast coordinating committee on soil testing, 1995).

Soluble soil P in profile samples was estimated by extraction with 0.5*M* K_2SO_4. Three grams of dried, ground and sieved soil was shaken with 30 mL of extractant for 30 minutes at 100 rpm, centrifuged for 10 minutes at 3000 rpm and then filtered (No. 42 Whatman filter papers). The processed samples were stored under refrigeration for no more than 24 hours before being analyzed. The extract was analyzed for DRP using the ascorbic acid method (American Public Health Association et al., 1992). The standards used for comparison were filtered orthophosphate standards in 0.5*M* K_2SO_4.

Drainage and transfer via groundwater

Because P leaching is considered to occur when P sorbed to mobile colloids moves through the profile under preferential flow or through the soil matrix in soluble forms, calculated groundwater flow using hydraulic conductivity is not as relevant as it is with nitrate movement. Estimations of P transfer to the groundwater and from the watershed was based on the drainage determined using the WATBAL model (see Chapter 2 and Appendix C), combined with groundwater concentration measurements and watershed size. Due to the highly variable nature of P flow, these calculations are not meant to represent the actual P losses from the watersheds, but to estimate relative losses in the various watersheds.

Statistical analyses

Statistical analyses for dissolved reactive P were conducted on data from samples collected from January 2002 through June 2004. Due to the dry conditions that occurred during the first leaching period, groundwater was not available in two of the grazed watersheds until October 2002 (Grazed 1B and Grazed 2B), and analyses including those watersheds cover the period from October 2002-June 2004 (Figure 2-1). Therefore, data were grouped for statistical analyses in two ways: 1) beginning in October 2002, when piezometers in all six watersheds provided groundwater; 2) beginning in July 2001, when piezometers in four of the watersheds provided groundwater. In addition, to test the effect of watershed proximity to the homestead and barns, the data beginning in October 2002 was classified as either "home" or "away". One of the two watersheds on each farm was designated "home", as the watershed historically (based on interviews with the farmers) receiving a greater proportion of nutrients because of its proximity or access to the barn and homestead for the past 100 years or more. The three watersheds considered to be "home" watersheds were: Confined A, Grazed 1A and Grazed 2B. Both Confined A and Grazed 1A are next to the barns and homestead. Grazed 2A is separated by a stream from the barns and homestead and historically was used as an exercise yard. Grazed 2B is similarly distanced from the barns but is not separated from them by a stream and therefore was treated as the homestead watershed. The other three watersheds were designated "away".

The DRP concentration data were analyzed using repeated measures GLM (SYSTAT, 1998). The seasonal average for each period of sampling (e.g. Winter 2002, Spring 2002, Summer 2002) for each piezometer nest within each watershed was used to avoid pseudoreplication over time. The model included effects of farm, piezometer nest, and watershed nested within farm. Originally, the rainfall between sampling dates and sample pH were used as covariates in the analysis, but were removed because they were not significant. An ANOVA was also run for DRP in pore water taken via ceramic-tipped suction lysimeters, using the effects of watershed and lysimeter length.

To test the hypothesis that groundwater from grazed watersheds was within acceptable limits, one-tailed t-tests (H_0: $\mu_A = 0.01$ mg PO_4-P L^{-1}) were performed using watershed and season as grouping factors. Because of the drought and resulting interruption in sampling, separate t-tests were carried out for the watersheds sampled without interruption from January 2002 - June 2004 and for all watersheds from October 2002 - June 2004.

Results and discussion

Groundwater phosphorus

Phosphorus concentrations under all six watersheds exceeded the EPA-recommended surface water quality level of 0.01 mg DRP L^{-1} (Figure 4-1) (USEPA, 1994; USEPA. 2001). Average concentrations of groundwater DRP differed by farm and ranged from 0.034 to 0.233 mg L^{-1}, with significantly lower concentrations in Grazed 2 watersheds (Tables 4-1 and 4-2). These high concentrations should not be surprising, as other studies have found leachate concentrations under grassland regularly surpass that critical level (Turner and Haygarth, 2000).

We speculate that the effect of farm may explained by the presence of calcareous material underlying Grazed 2. Calcareous material has a high capacity to sorb P and thus could remove P from recharge and groundwater. This explanation is supported by

Table 4-1: Average dissolved reactive P (DRP) concentration in shallow groundwater under six dairy farm watersheds. Means plus/minus SE are for samples collected biweekly from transects of nested piezometers from October 2002 to June 2004. Samples with the same letter are not significantly different.

Watershed	N	DRP mg L^{-1}
Confined A	119	0.180 $\pm$ 0.026a
Confined B	115	0.191 $\pm$ 0.035a
Grazed 1A	114	0.233 $\pm$ 0.036a
Grazed 1B	122	0.179 $\pm$ 0.023a
Grazed 2A	121	0.042 $\pm$ 0.005b
Grazed 2B	114	0.034 $\pm$ 0.010b

Table 4-2: Repeated measures GLM F-values for effects on dissolved reactive P concentrations in shallow groundwater on dairy farms. Three different models were used depending on the portion of the data set considered.

Effects in model	Six watersheds, 2002-2004		Four watersheds with uninterrupted sampling, 2001-2004‡		Proximity of watersheds†, 2002-2004	
	F-value	df	F-value	df	F-value	df
Farm	4.500*	2	7.413*	2	4.500*	2
Watershed within Farm	0.166*	3	0.039*	1	NI	NI
Nest	1.255*	2	3.941	2	1.255	2
Proximity§	NI	NI	NI	NI	0.147	1
Proximity by Farm	NI	NI	NI	NI	0.175	2
Error		10		5		10
Season	11.227***	6	11.598***	9	11.227***	6
Season by Farm	2.188*	12	4.0168***	18	2.188*	12
Season by Watershed within Farm	1.004	18	1.772	9	NI	NI
Season by Nest	0.594	12	1.702	18	0.594	12
Season by Proximity	NI	NI	NI	NI	0.916	6
Season by Proximity by Farm	NI	NI	NI	NI	1.340	12
Error		60		45		60

† Proximity model compares watersheds closer to and farther from the homestead or barnyard. Homestead watersheds are Confined A, Grazed 1A, and Grazed 2B, those most likely to have received a greater proportion of nutrients historically.
‡ The following watersheds were sampled from 2001-2004 without interruption: Confined A, Confined B, Grazed 1A, and Grazed 2A.
§ Proximity refers to whether the watershed is considered to be a homestead watershed or a more distant watershed.
*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.
NI Not included in the model.

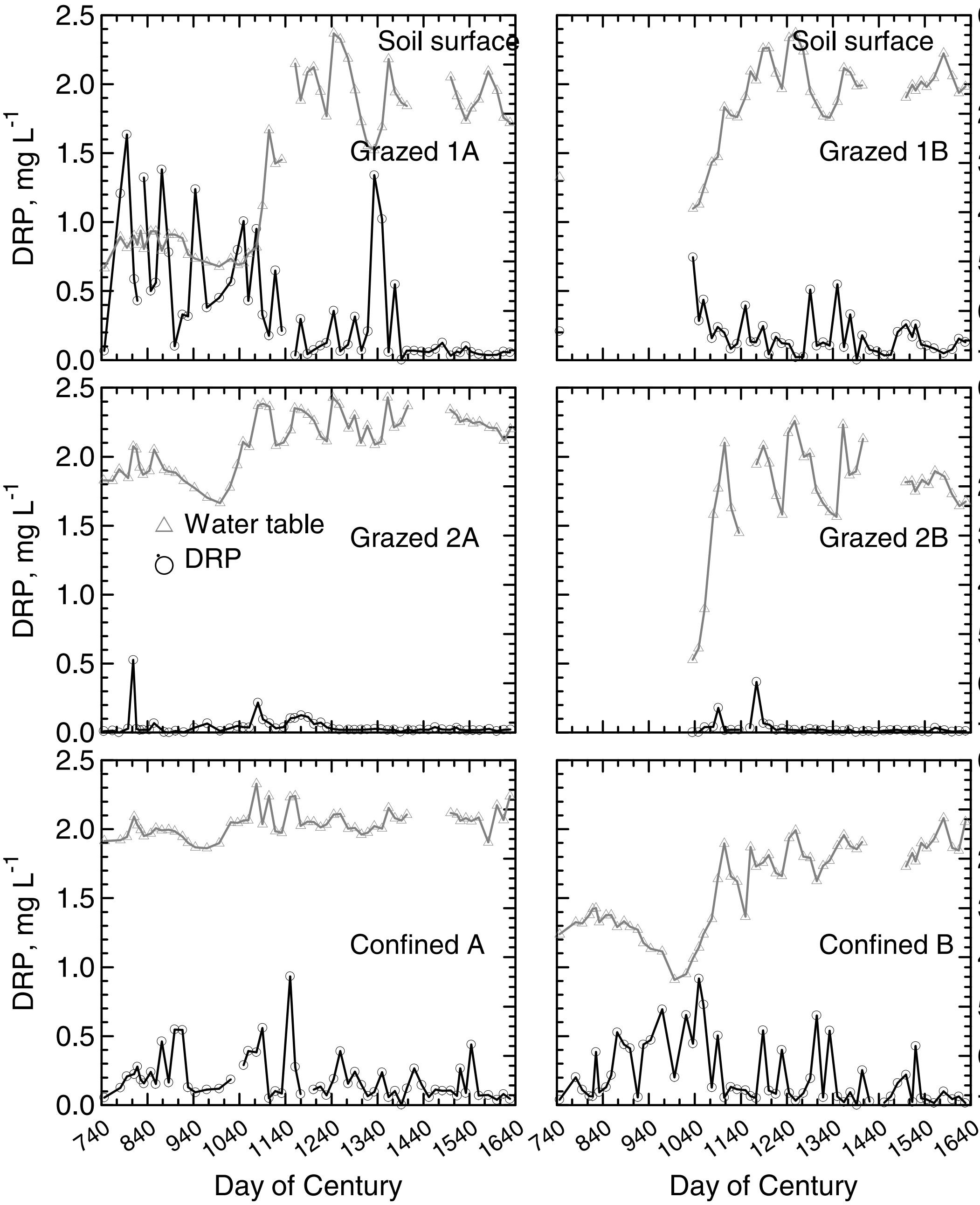

Figure 4-1: Dissolved reactive phosphorus (DRP) concentrations and depth to groundwater for six watersheds on three Maryland dairy farms. Each measurement represents an average of three samples collected from three piezometer nests within each watershed on one sampling date. Samples were collected from transects of nested piezometers biweekly from January 2002- June 2004.

geological maps of the area and by soil pore water data (Cleaves et al., 1968). However, we did not determine total calcium concentrations or P sorption capacity in subsoil materials in the Grazed 2 watersheds, and therefore can not confirm this explanation. Groundwater DRP concentrations differed between the three farms, but not between grazing and confined management and not by proximity to the homestead. Therefore our data does not support the hypotheses that greater historical manure applications at the homestead watersheds results in greater groundwater P loading. There is some evidence of a proximity effect in soil test results, to be discussed below.

There was a seasonal effect present in all three models, as well as a farm by season interaction. The farm by season interaction is likely linked to the difference between concentrations from the Frederick County farms and Grazed 2 in Baltimore County (Figure 4-2). Generally, as groundwater recharged, DRP levels decreased, possibly because of dilution of groundwater DRP by recharge, implying that current leachate is not carrying much DRP. The dilution trends are also consistent with the hypothesis that the level of DRP in groundwater was related to historical rather than current land management, and that present land management is not the source of much of the DRP present in groundwater.

Long term no-till cropping and permanent pastures on these farms can be expected to encourage high levels of earthworm activity, as well as the development of deep root channels from grass and alfalfa plants. Much of the P movement found in the six watersheds could occur by preferential flow such as wormholes and root channels (McDowell et al., 2001). None of the farmers till their land and have not in years (more than 4 decades on Confined), leading to increased earthworm populations with developed

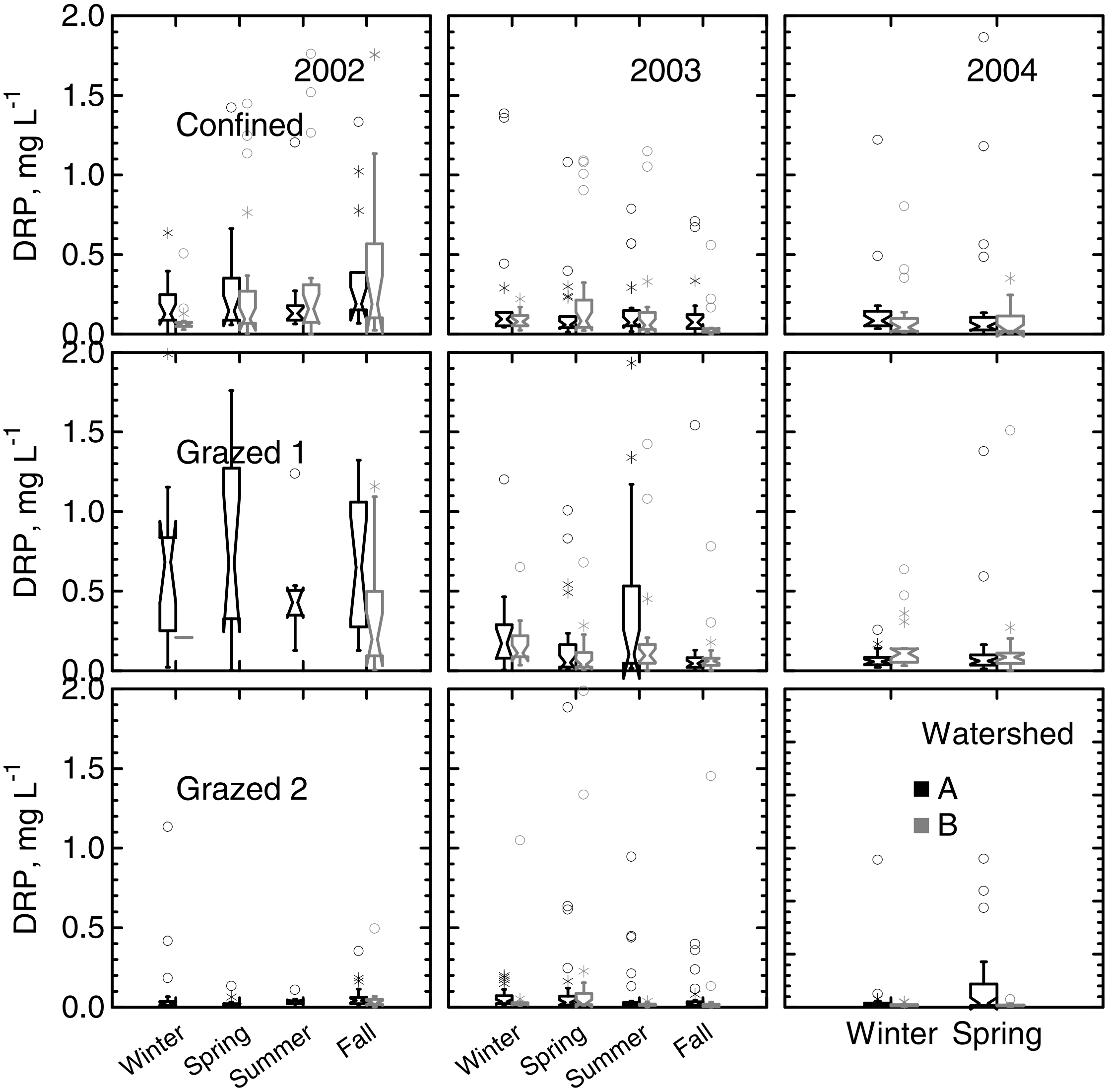

Figure 4-2: Seasonal averages of dissolved reactive phosphorus (DRP) in shallow groundwater on three Maryland dairy farms. Samples were collected biweekly from nested piezometers from January 2002 – June 2004. Boxes represent the central 50% of values, with notches indicating the 95% confidence intervals.

channels, as well as deep root channels from years of grass and alfalfa growth. These biopores are often the route for preferential flow and may be a source for P transfer to the groundwater. Even with the occurrence of pores, it seems unlikely that any of the three farms are experiencing much movement of P through leaching, as P concentrations decreased as groundwater recharged. This suggests that the water moving through the profile was the P concentrations, rather than transporting more P to the shallow aquifer. However, the quantity of P present indicates dairy farming contributes or has contributed substantial amounts of P to groundwater.

Pore phosphorous

Pore water DRP concentration showed no significant effects of field or nest, but there was an effect of depth of sample collection with a mean of 0.52 mg DRP L^{-1} at 60 cm and 0.22 mg DRP L^{-1} at 90 cm (Table 4-3, Figure 4-3). Means and variance for DRP concentration were both greater in samples taken via the shallower lysimeters (Figure 4-3). Variability was high in part because of limited sample availability; only 114 samples were collected from all six watersheds, 49 from the 90-cm lysimeters and 65 from the 120-cm lysimeters.

Samples collected from 60-cm deep lysimeters generally were more than twice as high in DRP than samples from the 90-cm deep lysimeters. Additionally, concentrations in 60-cm deep pore water were approximately five times higher than those measured in groundwater, suggesting P sorption occurred as water moved through the soil profile. One striking difference between pore water and groundwater was in samples from Grazed 2B. Pore water in Grazed 2B had the highest concentrations of DRP, but groundwater had the lowest DRP concentrations of the six watersheds. The much lower groundwater

Table 4-3: ANOVA F-values for effects on dissolved reactive phosphorus (DRP) concentrations in pore water on three Maryland dairy farms. Samples were collected on 11 dates over a 2 year period via ceramic-tipped, suction lysimeters with tips at 60 and 90 cm below soil surface when soil water conditions permitted. ANOVA was performed on log-transformed data.

Effects	F-value
Watershed	1.691
Nest	1.038
Depth of sample	8.776**
Watershed*Depth of sample	1.640
Number of samples	114
Model R^2	0.225

*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.

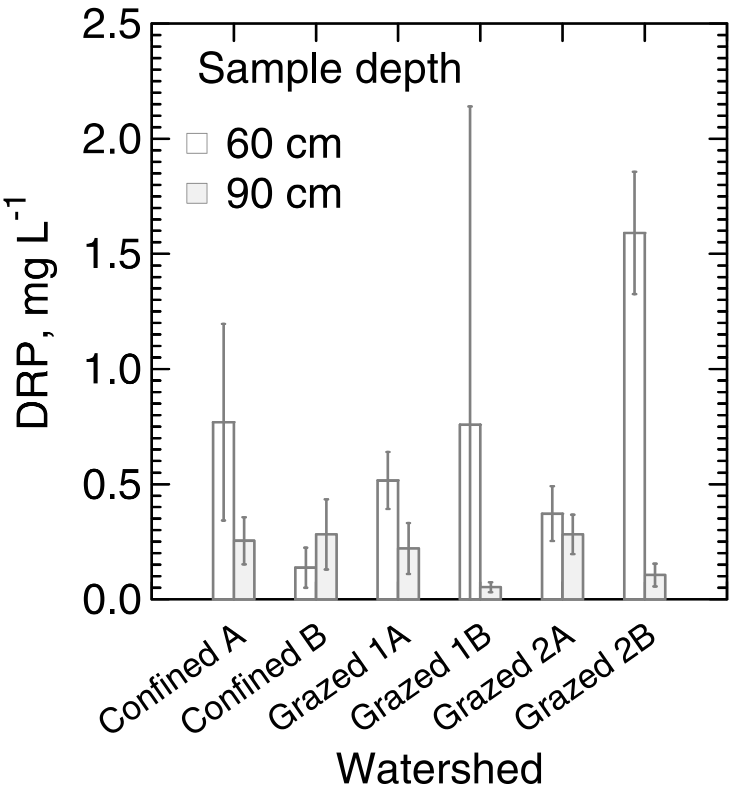

Figure 4-3: Concentrations of dissolved reactive phosphorus (DRP) in pore water sampled via ceramic-tipped suction lysimeters. Samples were collected on six watersheds in three dairy farms from October 2002 to June 2004. Tips of 90-cm lysimeter were approximately 60 cm below soil surface; tips of 120-cm lysimeters were approximately 90 cm below soil surface. Bars indicate SE.

concentrations suggest high levels of P sorption within the soil profile at depths between 90 cm and the groundwater table.

Soil phosphorus

Soluble (K_2SO_4-extractable) P concentrations in the soil ranged from 10-25 mg kg^{-1} in samples from the uppermost 10-20 cm. The highest concentrations were in samples collected in Grazed 1A (Figure 4-4). Below the A horizon, the concentration decreased greatly, to $\leq$ 1 mg L^{-1} at depths below 30 cm, with the exception of one sample in Grazed 2B. These concentrations and this pattern of decrease with depth are similar to that measured under other agricultural fields (Butler and Coale, 2005).

Melich 1 extractable soil P in the uppermost 10-20 cm ranged from soil fertility index values of 31 to 179 (Table 4-4). These index values are used to determine the plant availability of P, and to relate whether P is needed to improve yields. A low index value means more P is needed; a medium value indicates P may be needed, and an optimum value means enough P is present for the growth of most crops. Excessive P values mean there is more than enough P present for crop growth. The index values on the six watersheds correspond to between 75 and 406 kg P_2O5 ha^{-1}.

Soil P in both Frederick County homestead watersheds was at excessive soil fertility levels (Coale, 2002), while the farther watersheds had P index values that were low or medium. This supports the hypothesis of greater historical applications of manure on the watershed closest to the barnyard. The high P values on the two homestead watersheds may indicate sorption of DRP, also preventing a proximity effect for DRP groundwater concentrations.

Table 4-4: Melich 1 extractable P and P_2O_5 in soil samples from uppermost 10-20 cm in six Maryland dairy farm watersheds.

	N	Melich 1 extractable P Soil Fertility Index Value[†]	P_2O_5 kg ha^{-1}
Grazed 1A	7	151.3 ± 27.4	345.1
Grazed 1B	7	41.3 ± 8.5	98
Grazed 2A	9	83.0 ± 16.8	191.7
Grazed 2B	10	74.6 ± 4.3	173
Confined A	4	178.5 ± 25.3	406
Confined B	8	31.2 ± 10.0	75

[†] Soil Fertility Index Values: Low = 0-25, Medium = 26-50, Optimum = 51-100, Excessive >100.

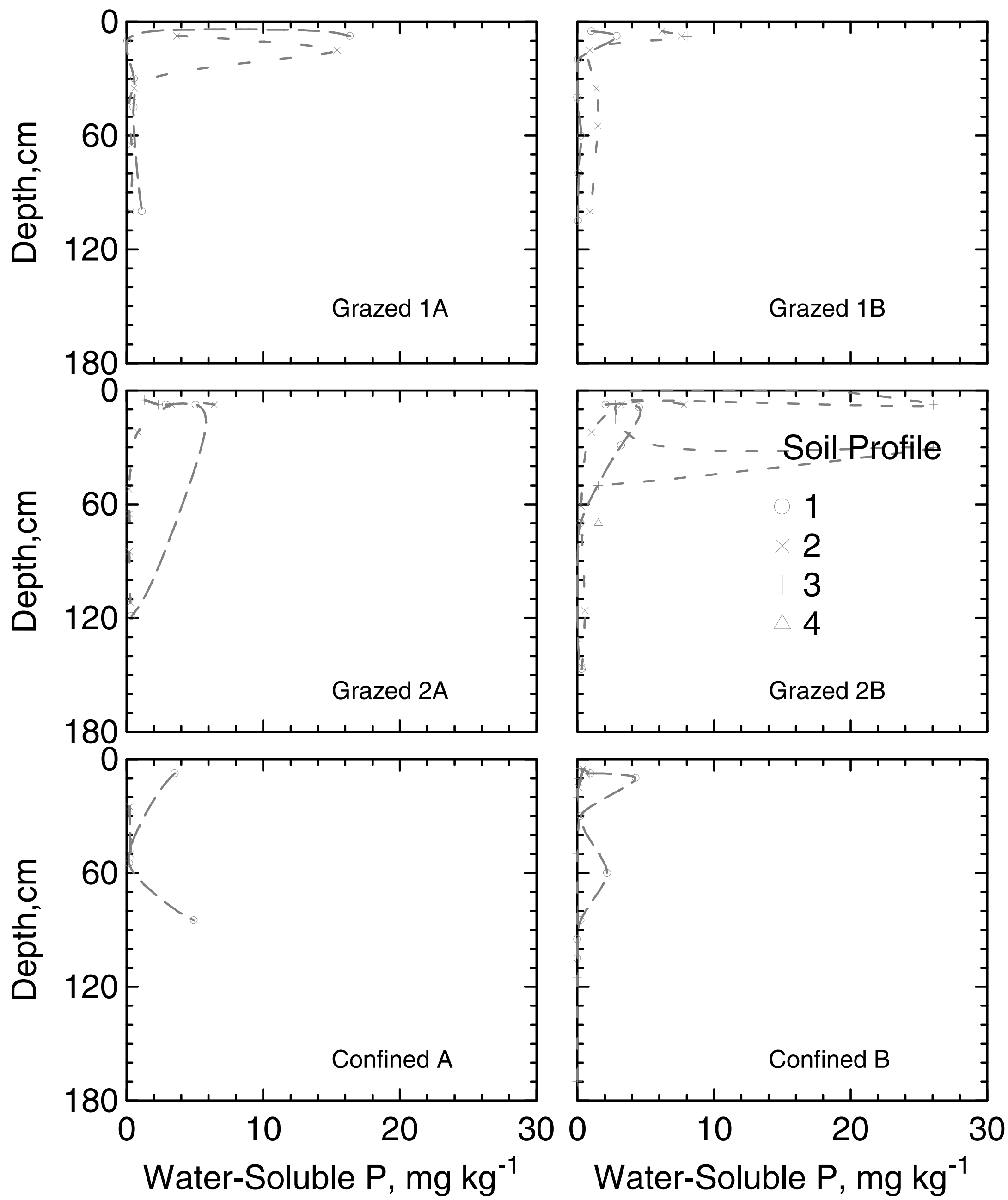

Figure 4-5: Water soluble phosphorus (P) in soil profiles from three Maryland dairy farms. The soil profiles were augered from distinct topographic areas within the watersheds in the spring of 2002 and 2003. Sample depth was dependent on the soil in the watershed; presence of very rocky conditions made deep samples unobtainable for some profiles.

On Grazer 2 in Baltimore County, there was no distinction in soil P between the two watersheds, and both watersheds had P index values in the optimum range. The similarity in soil P values may be a result of land management or soil type.

Groundwater phosphorus losses

Based on calculated drainage and measured groundwater concentrations, estimated annual DRP losses from the four Frederick County watersheds were 482-616 g ha^{-1} and 91-120 g ha^{-1} from Grazed 2A and 2B in Baltimore County (Table 4-5). These levels are similar to the 200-594 g DRP ha^{-1} y^{-1} reported by Turner and Haygarth (2000) based on their monolith lysimeter studies of leachate from four grassland soil types. Other studies report losses of up to 2.15 kg TP $ha^{-1}y^{-1}$ based on tile drainage, which tends to cause higher P concentrations (Kleinman et al., 2003; Simard et al., 2000; Sims et al., 1998). Losses from the two Frederick County farms in this study were therefore quite a bit lower than the 1 kg P ha^{-1} y^{-1} found in tile drainage under a New Zealand dairy catchment (McDowell and Wilcock, 2004) and close to the 0.33-0.42 kg^{-1} ha^{-1} y^{-1} from under barley and grass ley (Sims et al., 1998).

Estimated yearly DRP losses via groundwater from each farm were 118-125 kg from Confined, 40-51 kg from Grazed 1, and 6-8 kg from Grazed 2. Groundwater losses from the three farms were not correlated to the farm P surpluses (Appendix A). Soil P sorption processes, most notably in Grazed 2, probably affected this relationship with leaching P.

Conclusions

The risk of P leaching may be greater under dairy manure applications than would be expected because organic acids found in dairy manure can prevent P sorption either by blocking sorption sites with organic acids, or by inhibiting crystallization of Ca-P, maintaining P solubility and movement through the profile (Eghball et al., 1996; Harris et al., 1994). Additionally, the largest proportion of P in dairy manure is water soluble, making its movement through the soil profile more likely (He et al., 2004).

Because of the long-term agricultural land use and the nature of dairy manure, leaching losses are a potential pathway for P transfer from animal-based agricultural systems. The decrease in concentration between shallow pore water and groundwater indicates sorption does halt some of the P en route to groundwater. In this study, the type of dairy farm management did not affect the level of P losses. Instead, soil chemistry and geologic material probably played key roles in P leaching. The farm that had the lowest levels of DRP in groundwater was believed to be situated on a calcareous geologic material. The other two farms had similar levels of losses to each other, although one was a confined system and the other a grass-based, grazing system. Much of the DRP returning to the fields in applied manure or excreta may be sorbed to the soil, but levels found in groundwater under these three Maryland dairy farms demonstrate that P leaching and groundwater P should be considered in water quality models.

Table 4-5: Groundwater dissolved reactive phosphorus (DRP) concentrations and exports. Concentrations are based on overall means from groundwater samples collected biweekly from October 2002 – June 2004. Export rates are based on drainage and flow calculations from WATBAL model and weather data. The farm export estimates are based on per hectare exports and farm size. Farm surplus data are means for 2001-2003 and are based on farm records and interviews with farmers (Appendix A).

Watershed	Mean DRP	Drainage	DRP export	Farm P surplus	Mean DRP export	Farm size
	$mg\ L^{-1}$	$mm\ y^{-1}$	----------$kg\ ha^{-1}\ y^{-1}$----------			ha
Confined A	0.18	268	0.48	9.9	0.50	245
Confined B	0.19	268	0.51			
Grazed 1A	0.23	268	0.62	1.5	0.55	83
Grazed 1B	0.18	268	0.48			
Grazed 2A	0.04	302	0.12	7.8	0.11	71
Grazed 2B	0.03	302	0.09			

Chapter 5: Organic phosphorus in groundwater and soil on confined feeding and grazing dairy farm watersheds

Abstract

Recent research has found that phosphorus (P) leaching is a potentialpathway for significant P transfer. Previously, P leaching had been discounted as a source of P loss because of the likelihood of sorption and retention within the soil profile. Dissolved organic phosphorus (DOP) is less likely to be sorbed and, therefore, more likely to move through the soil profile and reach the groundwater. To determine the magnitude of DOP and total dissolved P (TDP) present in groundwater under dairy farming, shallow groundwater was monitored for two and a half years on three Maryland dairy farms. Two of the dairy farms are run under management intensive grazing (MIG), and the third farm is managed as a confined feeding system. Biweekly samples were collected from transects of nested piezometers in six actively grazed and cropped watersheds. Pore water samples were also collected via ceramic-tipped suction lysimeters. Groundwater TDP was measured using a microwave digestion method. Dissolved organic P was calculated as the difference between TDP and pre-digestion dissolved reactive P. Concentrations of TDP were between 25 and 45% DOP. Dissolved organic P in pore water did not decrease by depth, suggesting it moves more freely through the soil profile than does inorganic P. One of the MIG farms is over a calcareous geologic material which we hypothesized sorbed inorganic P but not organic P, resulting in the highest proportion of DOP in the TDP. Groundwater on all three farms exceeded the EPA critical limit for surface water TDP concentration.

Introduction

Phosphorus (P) in runoff from agricultural land is often in particulate or organic form (Daniel et al., 1998), and P can contribute to eutrophication in freshwater bodies as these forms convert to orthophosphate (Correll, 1998). Leaching has also been recognized as a pathway for P loss, and, although relatively unnoticed until recently, a substantial proportion of the P in soil solution and leachate may be in dissolved organic P (DOP) (Chardon et al., 1997). Phosphorus movement through the profile is either through preferential flow through pores, or carried in soluble form through the soil matrix (McGechan, 2003). Dissolved organic P may be more likely to leach than inorganic P, as DOP may not readily sorb to soil in the profile.

In animal-based agriculture, manure and slurry can contribute substantially to the DOP present in soil solution and leachate. Under plots which had received applications of either manure or fertilizer P since 1953, P from manure moved deeper in the soil profile than did chemical fertilizer even under equivalent application rates (Eghball et al., 1996). Phosphorus under manured plots was found below a P-sorbing layer of calcium carbonate, suggesting that P movement was in organic forms.

In a study using pig slurry applied to columns and lysimeters, between 70-90% of the total P leaching was in DOP (Chardon et al., 1997). In contrast, composted manure from beef feedlots was applied to plots of corn (*Zea mays* L.) in a study done in Nebraska (Eghball, 2003). The relative amount of DOP present was much lower in the cattle manure, at less than 25% of the total P. After 4 months of composting, less than 16% of the total P was in organic form, suggesting mineralization during that time.

Similar levels of DOP were found in leachate under monolith lysimeters receiving poultry manure (Kleinman et al., 2003) and fertilizer (Turner and Haygarth, 2000). Almost three-quarters of the TP concentration in leachate under the poultry manure was DRP, as compared to only 5% prior to manure application (Kleinman et al., 2003). The leachate from grassed monolith lysimeters receiving fertilizer had between 29 and 38% DOP (Turner and Haygarth, 2000). The presence of at least one-quarter to one-third of TP in leachate as DOP suggests that organic P may be a substantial source of P loading to groundwater.

The objective of this study was to determine if DOP was a substantial proportion of P lost to groundwater from dairy farms under contrasting manure and crop management. We measured DOP and TDP in shallow groundwater for several years under three Maryland dairy farms in order to determine the relative size of P transfers as well as the influence of grazing versus confined feeding management on this form of P loss.

Materials and methods

Shallow groundwater samples were collected from nested piezometers on six watersheds on three dairy farms. Two of the farms are MIG-based systems, and are referred to as Grazed 1 and Grazed 2. The third farm is a confined feeding dairy farm, and is referred to as Confined. Grazed 1 and Confined are on adjacent tracts of land in Frederick County, Maryland, and Grazed 2 is in Baltimore County, Maryland. Transects of nested piezometers were installed within each of the six watersheds. Ceramic-tipped suction lysimeters were installed to 45 and 60 cm depths at each nest. Additional information on the farms, instrumentation and sampling methods is included in Chapter 2 and Appendix A.

We sampled groundwater biweekly, from July 2001 through June 2004. For each nest, samples were taken from the shallowest piezometer containing at least 1m of water. Prior to sampling, the piezometer from which the sample was to be taken was bailed and allowed 2-3 hours to recharge. The pH of each sample was measured in the field. Beginning January 2002, 2-3 drops of 4 M H_2SO_4 were added for each 120-150 mL of sample to prevent microbial growth. Samples were transported to the lab on ice where they were filtered under vacuum through 0.2 μm membranes before analysis. Samples were stored under refrigeration at 4°C until analysis, unless analytical procedures would not be carried out within 2 weeks, in which case they were stored at <-15°C until analysis.

When soil water content was sufficiently high, pore water samples were collected via the lysimeters. Soil pore water samples were collected from each farm 11 times during the study. These samples were treated similarly to the groundwater samples collected via the piezometers. Complete information on sample collection is found in Chapter 2.

Total dissolved P (TDP) was determined by a modified alkaline persulfate microwave digestion described in Chapter 3 and dissolved organic P was calculated as TDP – DRP (Cabrera and Beare, 1993; Hosomi and Sudo, 1986; Littau and Englehart, 1990; Johnes and Heathwaite, 1992). The digestion transformed TDP to orthophosphate which was measured using the ascorbic acid method, as described in Chapter 4 (American Public Health Association et al., 1992).

Two or more topographically distinct areas within each watershed were identified. For each area, an augered soil profile was placedn a 10 -cm diameter trough, divided into horizons, described with regard to color, texture, and other morphological features (Appendix B). Samples from each horizon were taken to the lab in sealed plastic bags on ice, where they were spread out and air-dried. Sub-samples were ground and analyzed by the University of Maryland Soil Testing Lab for pH (1:1 in water), percent organic matter (loss on ignition), and Melich 1 extractable Mg, P, K, and Ca (Northeast coordinating committee on soil testing, 1995).

Soil sub-samples were also used to estimate water-soluble soil P by extraction with 0.5*M* K_2SO_4 as described in Chapter 2. The extract was analyzed for DRP and digested for total P determination using 10 ml of the filtered extract and 10 ml of the digestion reagent described above. The standards used for comparison were filtered orthophosphate standards in 0.5*M* K_2SO_4. Soluble organic P (SOP) was then determined as the difference between total P and DRP in the soil extracts.

Drainage and total losses to groundwater

Because P leaching can occur when P sorbed to mobile colloids moves through the profile under preferential flow or through the soil matrix in soluble forms, groundwater flow using measurements of hydraulic conductivity is not as relevant as it is with nitrate movement. Total losses to groundwater and from the watershed are based on the drainage determined using the WATBAL model (see Appendix C), combined with groundwater concentration measurements and watershed size. Due to the highly variable nature of P flow (Chardon et al., 1997), these calculations are not meant to represent the actual P losses from the watersheds, but rather to give estimations and the relative size of the losses from each.

Statistical analyses

Statistical analyses for the variables dissolved organic P (DOP), total dissolved P (TDP), and DOP:TDP were done separately. Due to the dry conditions that occurred during the first leaching period, groundwater was not available in two of the grazed watersheds until October 2002 (Grazed 1B and Grazed 2B), and analyses including those watersheds cover the period from October 2002-June 2004 (Figure 2-1). Therefore, data were grouped for statistical analyses in two ways: 1) beginning in October 2002, when piezometers in all six watersheds provided groundwater; 2) beginning in July 2001, when piezometers in four of the watersheds provided groundwater. In addition, to test the effect of watershed proximity to the homestead and barns, and a historical source of manure (see Chapter 2), one of the two watersheds on each farm was classified as either "home" or "away", and the data since October 2002 was analyzed. Confined A, Grazed 1A, and Grazed 2B were determined to be "home" watersheds, and the other three were determined to be "away" watersheds.

The data on DOP, TDP and DOP:TDP were analyzed using repeated measures GLM (SYSTAT, 1998). The seasonal average for each period of sampling (e.g. Winter 2002, Spring 2002, Summer 2002) for each piezometer nest within each watershed was used to avoid pseudoreplication over time. The model included effects of farm, piezometer nest, and watershed nested within farm. An ANOVA was also run for the P variables in pore water taken via ceramic-tipped suction lysimeters, using the effects of watershed and lysimeter length.

To test the hypothesis that groundwater from grazed watersheds would be within critical levels for TDP, one-tailed t-tests were performed using watershed and season as grouping factors. Because of the drought and resulting interruption in sampling, separate t-tests were carried out for the watersheds sampled without interruption from January 2002 – June 2004 and for all watersheds from October 2002 – June 2004.

Results and discussion

All three farms have been managed in a way that promotes the establishment of continuous pores, or channels, either from deep perennial grass and legume roots, or from thriving earthworm populations. While these pores may reduce runoff, they may also increase infiltration, and may allow P movement by preferential flow, something especially likely to occur when heavy rainfall follows dry periods (Chen et al., 1996; Simard et al., 2000).

Because organic P is relatively unaffected by sorption on soil surfaces, DOP movement can be an important source of P transfer to the groundwater from animal-based systems (Eghball et al., 1996; Ulen, 1999). Concentrations of DOP ranged from 0.02 - 0.06 mg DOP L^{-1}, making up 25 - 45% of the TDP critical level for acceptable surface water set by the EPA (Table 5-1). Organic P can account for between 2-23% of the water-soluble P in dairy manure (He et al., 2004), which is likely the source for DOP under these watersheds.

DOP and TDP concentrations, as well as the DOP:TDP ratio, differed between the three farms (Table 5-2). The lowest concentrations of DOP and TDP were found on Grazed 2, as well as the highest proportion of DOP:TDP (Table 5-1, Figures 5-1 and 5-2). Concentrations of DOP in all six watersheds were observed to increase during periods

Table 5-1: Average dissolved organic phosphorus (DOP) and total dissolved phosphorus (TDP) concentrations, ratio of DOP:TDP in shallow groundwater under six dairy farm watersheds. Samples collected biweekly from transects of nested piezometers from October 2002 to June 2004. Samples with the same letter are not significantly different.

	N	DOP	TDP	DOP:TDP
		mg L^{-1}		
Confined A	128	0.063 $\pm$ 0.013a	0.266 $\pm$ 0.041a	0.25 $\pm$ 0.02b
Confined B	115	0.037 $\pm$ 0.009a	0.207 $\pm$ 0.034a	0.25 $\pm$ 0.02b
Grazed 1A	121	0.045 $\pm$ 0.007a	0.266 $\pm$ 0.039a	0.25 $\pm$ 0.02b
Grazed 1B	118	0.041 $\pm$ 0.007a	0.192 $\pm$ 0.024a	0.25 $\pm$ 0.02b
Grazed 2A	147	0.016 $\pm$ 0.001b	0.059 $\pm$ 0.006b	0.35 $\pm$ 0.02a
Grazed 2B	110	0.023 $\pm$ 0.006b	0.061 $\pm$ 0.018b	0.45 $\pm$ 0.02a

Table 5-2: Repeated measures GLM F-values for effects on dissolved organic P (DOP), total dissolved P (TDP), and DOP:TDP concentrations in shallow groundwater on dairy farms.

Effects in model		DOP	TDP	DOP:TDP
Model: Six watersheds†	df		F-value	
Farm	2	5.484*	5.552*	19.276***
Watershed within Farm	1	0.350	0.332	1.769
Nest	2	1.070	1.319	0.850
Error	4			
Season	6	3.279**	9.700***	32.563***
Season by Farm	12	1.327	2.079*	1.608
Season by Watershed within Farm	18	1.313	1.180	0.650
Season by Nest	12	0.863	0.863	1.160
Error	60			
Model: Four watersheds†‡	df		F-value	
Farm	2	11.551*	NI	12.423*
Watershed within Farm	1	0.246	NI	2.248
Nest	2	5.034	NI	3.873
Error	5			
Season	9	4.500**		8.922***
Season by Farm	18	2.697**		5.355***
Season by Watershed within Farm	9	1.343		0.926
Season by Nest	18	1.764		1.138
Error	36			
Model: Proximity†	df		F-value	
Farm	2	5.484*	5.552*	19.276***
Proximity	1	0.629	0.393	0.592
Farm by Proximity	2	0.211	0.302	2.358
Nest	2	1.070	1.319	0.850
Error	10			
Season	6	3.279**	9.700***	32.563***
Season by Farm	12	1.327	2.079*	0.114
Season by Proximity	6	0.551	0.483	0.837
Season by Farm by Proximity	12	1.694	1.529	0.746
Season by Nest	12	0.863	0.863	1.160
Error	60			

† One model tested all six watersheds from October 2002 – June 2004. A second model tested four watersheds sampled from 2001-2004 without interruption: Confined A, Confined B, Grazed 1A, and Grazed 2A. The proximity model used data from October 2002 – June 2004, and compared watersheds closer to and farther from the homestead or barnyard. Homestead watersheds are Confined A, Grazed 1A, and Grazed 2B.
‡ Total dissolved P was not analyzed for this model, because data was incomplete.
*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.
NI Not included in the analyses.

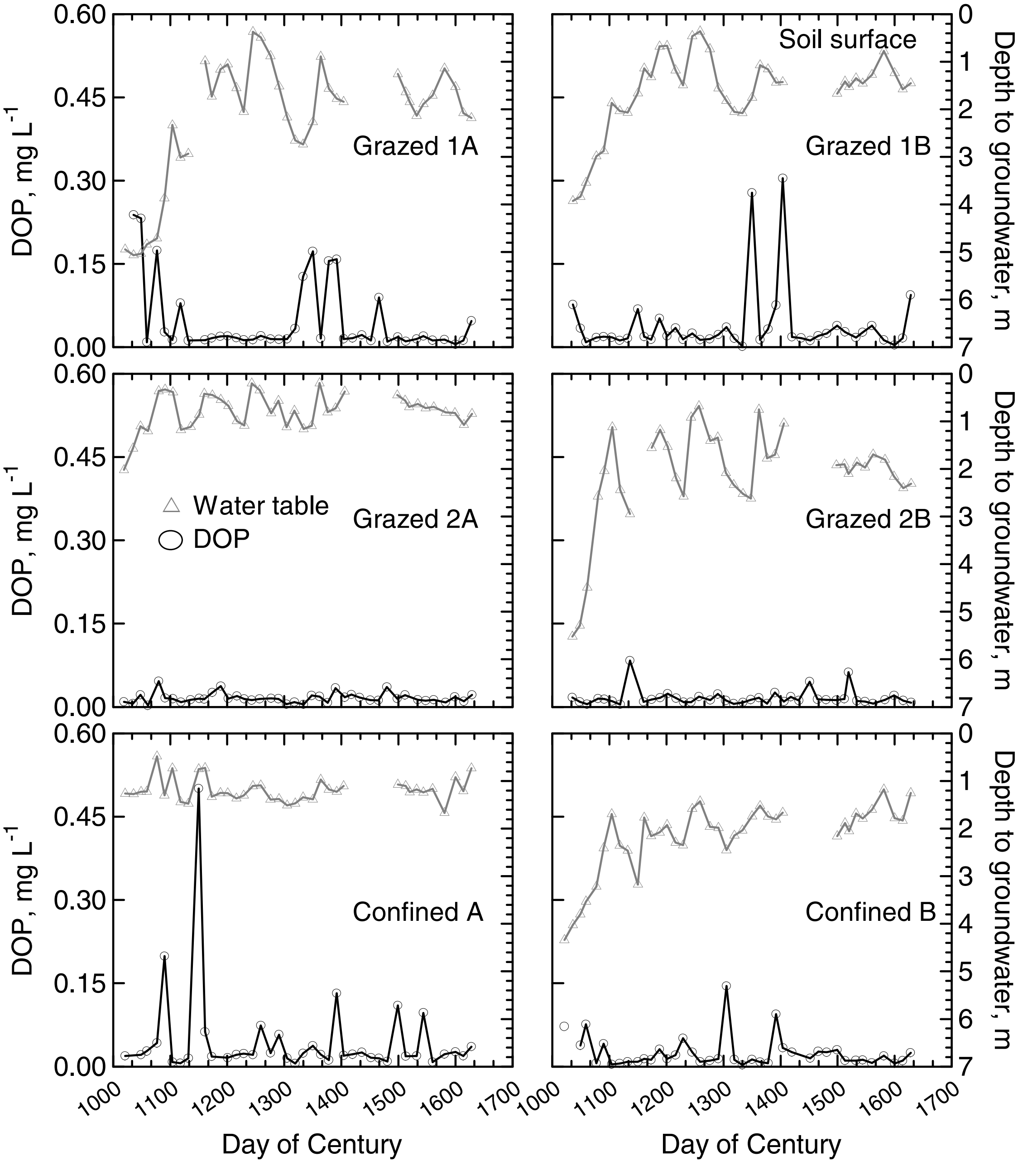

Figure 5-1: Dissolved organic P (DOP) and depth to groundwater from January 2002 – June 2004 on three Maryland dairy farms. Each measurement is an average of three samples collected from three piezometer nests within each watershed on the same sampling date. Samples were collected biweekly from transects of nested piezometers installed into actively grazed and cropped watersheds.

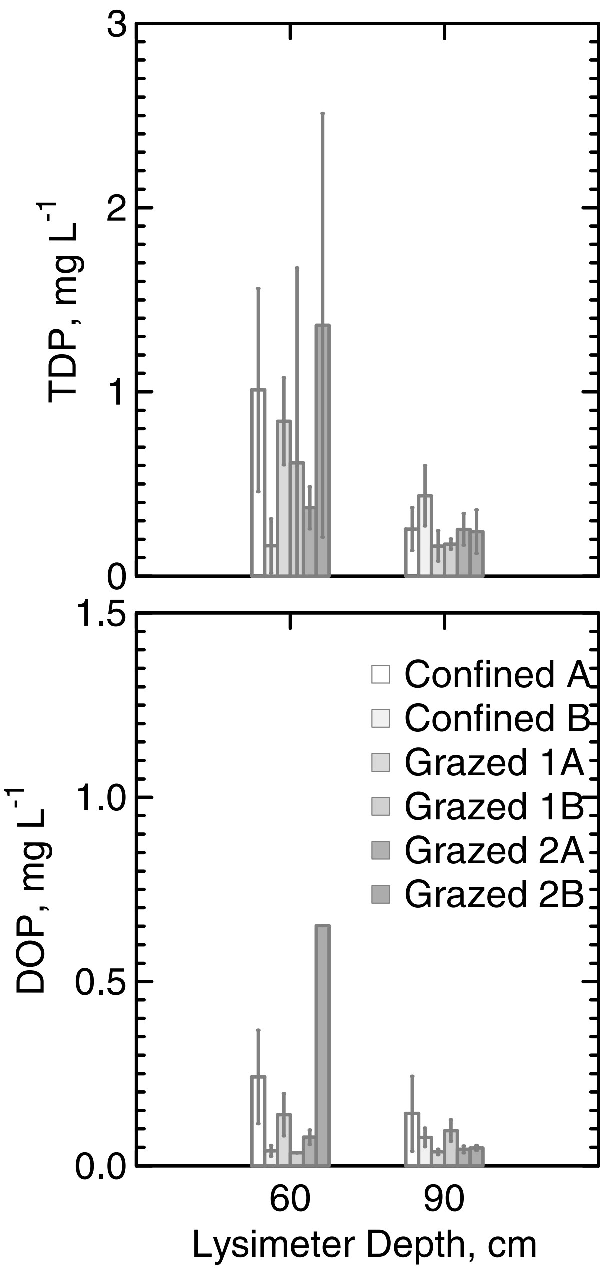

Figure 5-2: Total dissolved phosphorus (TDP) and dissolved organic phosphorus (DOP) in pore water on three Maryland dairy farms. Samples were collected via ceramic-tipped suction lysimeters when soil moisture made it possible. Collection took place 11 times on each farm between January 2002 and June 2004. Bars indicate SE.

groundwater recharge (Table 5-2, Figure 5-3). The increases in groundwater DOP concentration during recharge is in contrast to DRP concentrations, which decreased in groundwater during recharge (see Chapter 4). Dissolved organic P can move through the soil differently than inorganic P, because sorption by soil colloids may have only little effect on DOP movement through the profile, especially in preferential flow (Chardon et al., 1997). The increase in groundwater DOP during recharge indicates that P is leaching in organic forms, while DRP may be fixed within the soil profile. The lack of sorption of DOP was also seen in pore water, where there was no difference in DOP concentration in pore water samples collected at 60-cm and 90-cm depths (Figure 5-2).

Historical land management could have accounted for higher groundwater P concentrations in the watersheds more convenient to the barnyard, but there was no difference in concentration due to proximity to the barnyard. Because DOP can move through the soil profile without being sorbed, the changes in DOP concentrations were due to season and groundwater recharge, indicating DOP concentrations are a result of current land use and manure applications. The absence of a proximity effect points to the effectiveness of current nutrient management planning, where manure is no longer applied to the field based simply on convenience.

Total dissolved P in groundwater was lowest on Grazed 2, in the Baltimore County watersheds (Table 5-1). The lower concentration of TDP is likely related to the lower DRP concentrations found under Grazed 2. We hypothesized the lower concentrations on Grazed 2 may have been due to the underlying calcareous geologic material, something which was mapped by Cleaves et al. (1968), however we did not measure total Ca in the soil profile samples. Melich extractable Ca was not higher in

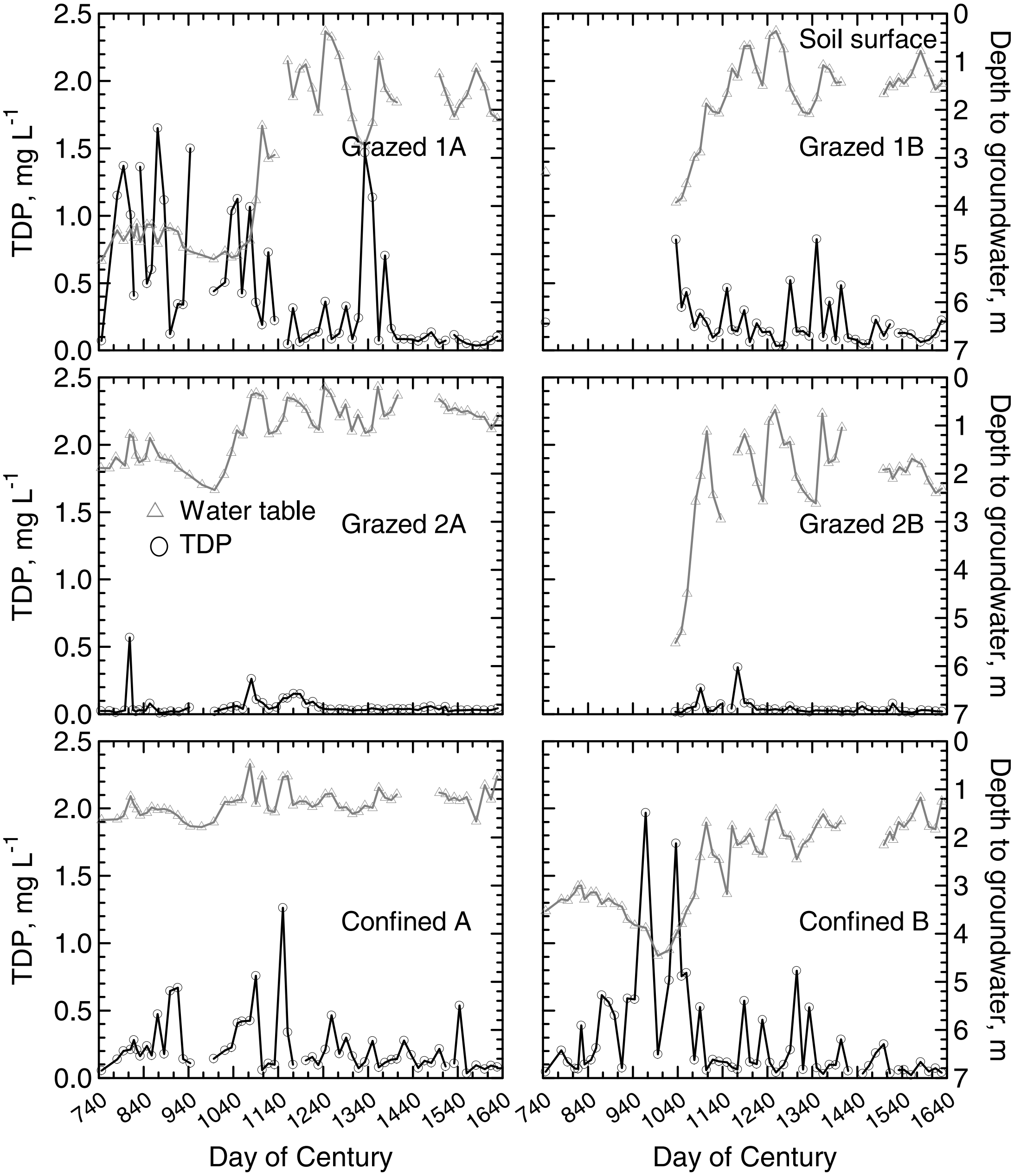

Figure 5-3: Total dissolved phosphorus (TDP) in shallow groundwater and depth to groundwater on three Maryland dairy farms. Samples were collected biweekly via nested piezometers from January 2002 – June 2004. Each point represents an average of three samples collected from three piezometers nests within each watershed on one sampling date. Grazed 1B and Grazed 2B were not regularly sampled until October 2002 because drought conditions made groundwater unavailable.

Baltimore County watersheds than in the Frederick County watersheds. In the other four watersheds (Frederick County), TDP levels exceeded the 0.1 mg L^{-1} EPA critical level for acceptable surface water quality.

In pore water, TDP, primarily DRP, was affected by depth, with higher concentrations taken from the shallower lysimeters, before sorption was able to take place within the soil profile (Figure 5-2). In four of the watersheds, mean pore water TDP concentrations were between 0.6 – 1 mg L^{-1} at a depth of 60 cm, but only 0.2 – 0.4 mg L^{-1} at 90 cm.

On Grazed 2 watersheds, groundwater TDP held 0.35-0.45 DOP, but in the four Frederick watersheds, TDP was close to 0.25 DOP (Table 5-1). The proportion of DOP:TDP differed by season on the six watersheds in all three statistical models (Table 5-2, Figure 5-4). The proportion was consistently higher in Grazed 2 because of the much lower DRP concentrations found there. Other seasonal effects on DOP:TDP are likely due to increased DOP concentrations and decreased TDP during recharge.

Soil phosphorus

Concentrations were highest at the surface, at between 5 – 10 mg soluble organic P kg^{-1} and 10 – 20 mg total soluble P kg^{-1} (Figures 5-5 and 5-6). Levels dropped with depth. A slight bulge is present in soil profiles from Grazed 2B.

Groundwater P losses

Based on calculated drainage and measured groundwater concentrations, estimated annual DOP losses were between 61-161 g ha-1 and TDP losses were from 181-724 g ha-1 ~~y~~ 1 (Table 5-3). These exports do not include surface runoff, considered

Table 5-3: Groundwater dissolved organic phosphorus (DOP) and total dissolved phosphorus (TDP) concentrations and exports. Concentrations based on overall means from groundwater samples collected biweekly from October 2002 – June 2004. Export rates based on drainage and flow calculations from WATBAL model and local weather data. Total farm exports are based on farm size and per hectare exports for that farm. Farm surplus data is based on farm records and interviews with farmers (Appendix A). Surplus values are means for 2001-2003.

Watershed	Mean DOP	Mean TDP	Drainage	DOP export	TDP export	Farm P surplus	Mean TDP export	Farm size
	-----mg L^{-1}-----		mm y^{-1}	-------------kg ha^{-1} y^{-1}-------------				ha
Confined A	0.06	0.27	268	0.16	0.72	9.9	0.64	245
Confined B	0.04	0.21	268	0.11	0.56			
Grazed 1A	0.05	0.27	268	0.13	0.72	1.5	0.62	83
Grazed 1B	0.04	0.19	268	0.11	0.51			
Grazed 2A	0.03	0.11	302	0.09	0.33	7.8	0.26	71
Grazed 2B	0.02	0.06	302	0.06	0.18			

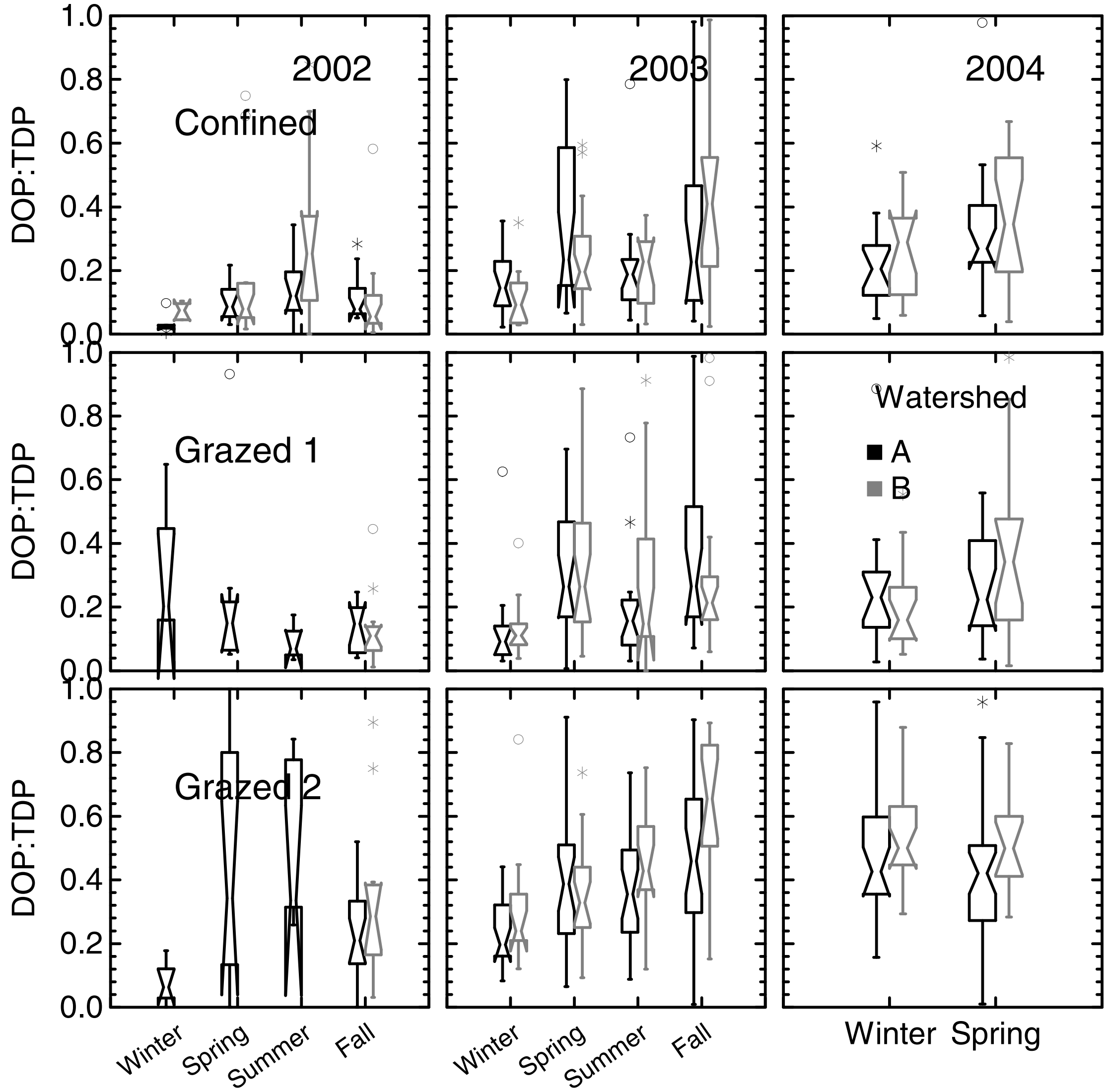

Figure 5-4: Ratio of dissolved organic phosphorus (DOP) to total dissolved phosphorus (TDP) in shallow groundwater under three Maryland dairy farms. Samples were collected from actively grazed or cropped watersheds via transects of nested piezometers. Boxes represent the central 50% of the values, with notches indicating the 95% CI.

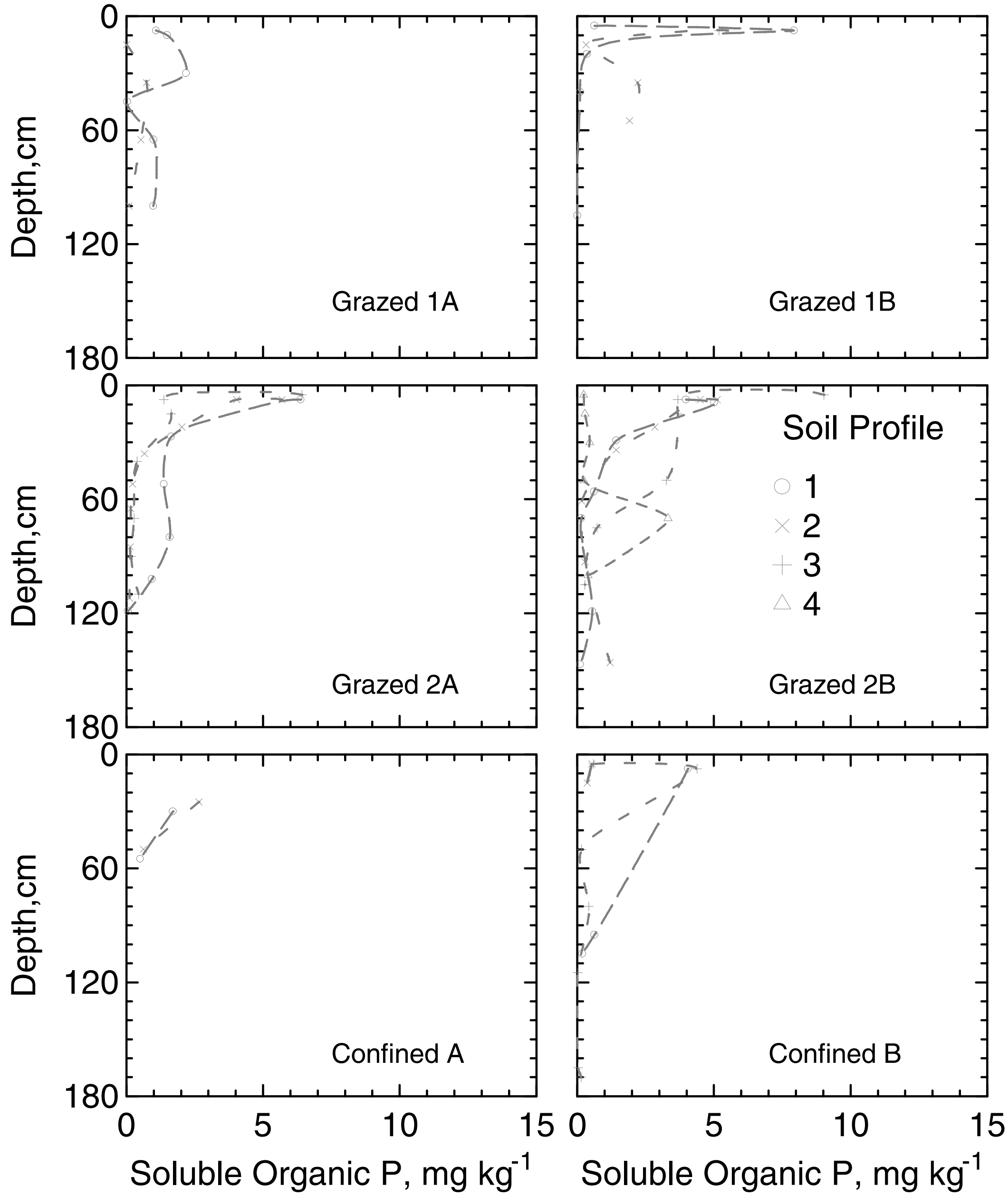

Figure 5-5: Soluble organic phosphorus (P) in soil profiles from three Maryland dairy farms. The soil profiles were augered from distinct topographic areas within the watersheds in the spring of 2002 and 2003. Sample depth was dependent on the soil in the watershed; rocky conditions made deep samples unobtainable for some profiles.

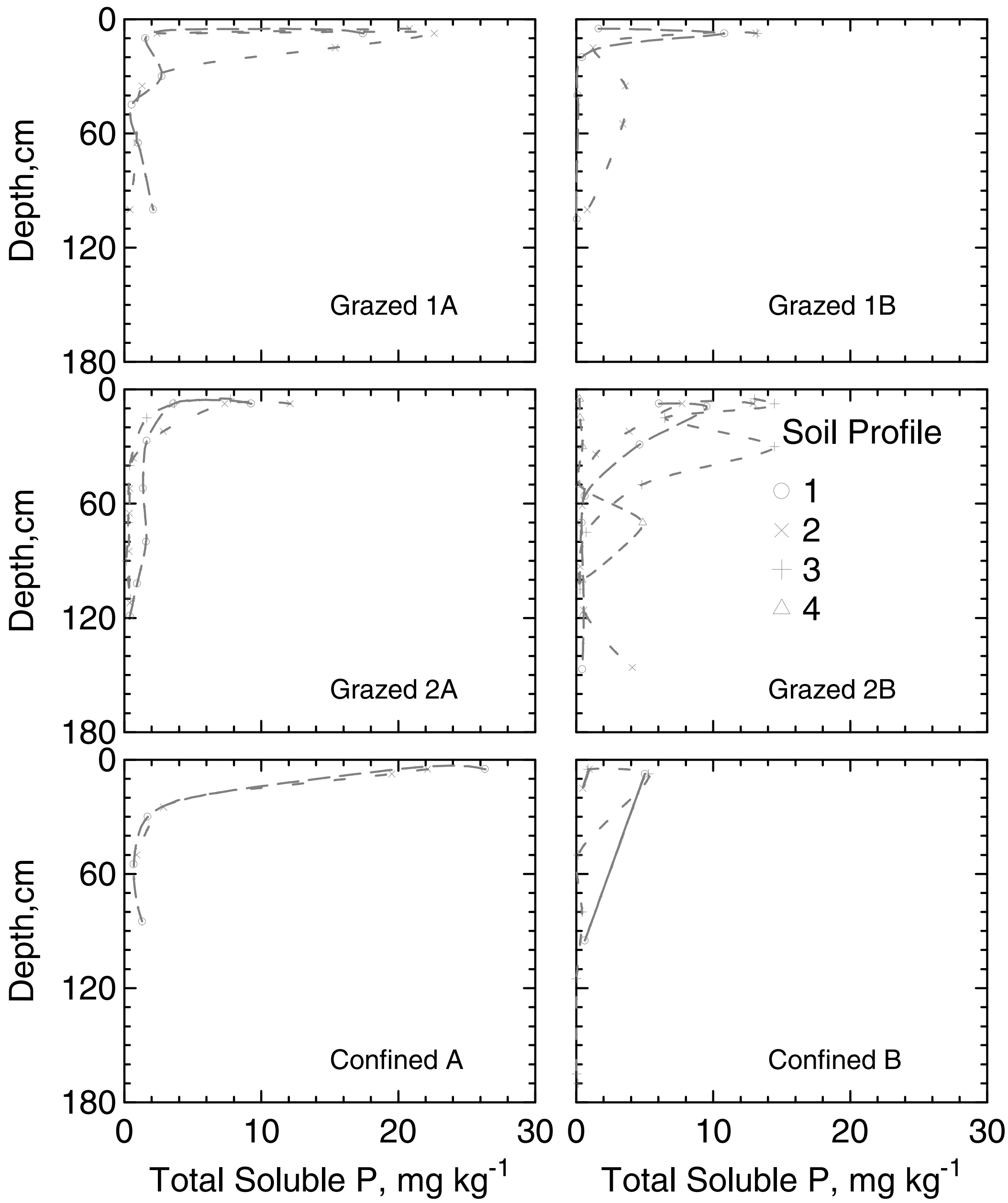

Figure 5-6: Total soluble phosphorus (P) in soil profiles from three Maryland dairy farms. The soil profiles were augered from distinct topographic areas within the watersheds in the spring of 2002 and 2003. Sample depth was dependent on the soil in the watershed; rocky conditions made deep samples unobtainable for some profiles.

to be the pathway for the greatest quantity of P transfer. Calculated P surpluses were not closely related to P losses to groundwater.

The interaction between soil and P accounts for the absence of a relationship between P losses to groundwater and the calculated P surpluses on each farm. Grazed 1 has the lowest P surpluses per hectare, yet per hectare, P losses to groundwater were the highest. Grazed 2 has much higher P surpluses, but because of the low TDP groundwater concentrations, possibly due to P sorption in the soil profile, P losses were lower than those on Grazed 1.

Conclusion

Dissolved organic P and TDP were measured under three Maryland dairy farms at levels which warrant the inclusion of groundwater losses in models describing P transfers. There was no effect of farm management, but there was a difference between the two farms in Frederick County and the third farm in Baltimore County. This may be due to an unconfirmed geologic factor, where a supposed underlying calcareous material sorps leaching P. Because of the high proportion of organic P in dairy manure, its movement through the soil profile, and the presence of biopores under many farm fields, P may be less affected by soil sorption than previously estimated, making leaching a potential pathway for DOP and TDP loss. The concentrations and estimated losses seen here point to the potential for significant P losses to groundwater under animal-based agricultural land.

Chapter 6: Nitrogen and phosphorus in surface water

Abstract

Nutrient losses via runoff are a major source of nutrient pollution to surface water. These losses are often the cause of eutrophication and resulting environmental degradation. To determine the nature of losses to surface water from a dairy farm under management intensive grazing (MIG) two streams were monitored from 2001 to 2004. Samples were collected at five sampling sites along each stream as it flowed through a grazed watershed. Collection took place biweekly, under both base and storm flow. Samples were analyzed for nitrogen (N) and phosphorus (P). During base flow, N and P concentration did not increase in either stream. During some storm events, one stream had an higher P concentration attributed to runoff from an area where the cows congregated, situated alongside the stream. The other stream showed no effect of runoff or increase in P concentration during storm events. These results suggest that under proper management, grazing does not lead to high levels of nutrient runoff and resulting nutrient pollution.

Introduction

Nutrient losses to surface water are a major cause of algal growth, eutrophication, and environmental degradation in the Chesapeake Bay watershed. Nitrogen (N) and phosphorus (P) may reach streams and waterways through both surface runoff and subsurface flow (Sharpley and Syers, 1979). Generally, P has received more attention than has N as a pollutant of surface waters, partly because N loading is largely by subsurface leaching, atmospheric deposition, and biological N fixation, while P loading is more associated with surface runoff and sedimentation, two processes subject to control by management (Powell et al., 2001). Additionally, P is considered the limiting nutrient for eutrophication of freshwater bodies (Parry, 1998), while N is more often limiting in brackish waters.

Runoff of both N and P can be abated by management that reduces erosion, such as the maintenance of adequate vegetative cover, or proper timing of manure application (Hansen et al., 2002). The use of buffer strips to filter runoff before it reaches streams and waterways can also reduce the impact of nutrient pollution, especially N, and if the buffer strip is wide enough, P as well, through sedimentation or sorption, denitrification and plant uptake within the buffer strip (Heathwaite et al., 1998; Hillbricht-Ilkowska et al., 1995). The extent of N and P losses to surface water from intensively grazed pastureland has not been well documented.

Phosphorus transfer in runoff

Eutrophication is accelerated by the presence of excess P (Correll, 1998); the EPA has set critical values for dissolved reactive P (DRP) and total dissolved P (TDP) in surface water at 0.01 and 0.1 mg L^{-1}, respectively (USEPA, 1994; USEPA. 2001). Phosphorus runoff from agricultural land is a major source of pollution of lakes and streams (Parry, 1998). Phosphorus pollution increases with discharge; during wet periods, rain storms cause greater P transport by rivers (Sharpley et al., 1995).

Phosphorus lost from the soil surface is transported in dissolved form in runoff water or sorbed to eroding soil particles. Between 75-90% of P in surface runoff from cultivated land is sorbed to eroding soil particles or organic matter, with most of the remainder transported as DRP (McDowell et al., 2001). This relationship is inverted in runoff from grassed land, which carries little eroded sediment, and most of the P is dissolved in the runoff. Dissolved reactive P is considered readily algal available, and therefore a greater risk for immediate eutrophication, but sorbed P may also become bioavailable quickly (Hansen et al., 2002). Both dissolved and sorbed P in runoff water are typically due to losses from the uppermost 5 cm of soil and is related to the quantity and reactivity of P present, as well as the individual soil type and its management.

If even small proportions of applied P are lost, the amount of P transferred to waterways can lead to eutrophication (Hansen et al., 2002). Surface runoff of about 1-5% of the applied P from agricultural lands usually results in P losses of approximately 0.5-2 kg ha^{-1} y^{-1}, (Breeuwsma et al., 1995). In the Midwest, where recommended P application rates are from 25 to 45 kg $ha^{-1}y^{-1}$, losses of 1-2 kg P ha^{-1} y^{-1} can cause environmental harm to surface water (Hansen et al., 2002).

Research modeling P runoff to waterways found that agricultural land in the Netherlands contributed 0.82 kg total P $ha^{-1}y^{-1}$ (Pieterse et al., 2003). Runoff from watersheds in the U.S. Atlantic Coastal Plain yielded 0.31 kg total P $ha^{-1}y^{-1}$ from forested land and 2.41 kg total P $ha^{-1}y^{-1}$ from agricultural land (Sharpley et al., 1995).

Surface runoff losses of P from agricultural land are typically increased by repeated application of manure from animal-based operations. Increased losses often occurs when the farm holds a herd larger than its feed production can support, requiring more P to be imported than is exported (Daniel et al., 1998; Powell et al., 2001). Another cause for high levels of P in runoff from dairy operations is that generally dietary P in rations exceeds the rates required to support milk production, growth, and reproduction, resulting in more P excreted in manure (National Research Council, 2001; Ebeling et al., 2002). It has been suggested that dietary P could be reduced by 30-40% without any adverse affects to production (Satter and Wu, 1999; Valk et al., 2000).

Runoff losses also occur when the amount of P in the top 0 – 5 cm of soil P exceeds a threshold soil P level, (Weld et al., 2001). In order to more accurately predict P losses, threshold soil P level has to be used in conjunction with the potential for surface runoff and erosion (Sharpley et al., 2001; Weld et al., 2001).

The length of time between a runoff event and fertilizer application or grazing has an effect on the amount of P lost in surface runoff (Nash et al., 2000; Sherwood and Fanning, 1981). More than four days after fertilizer application, the total P (TP) lost in surface runoff decreases by half compared to losses from runoff less than two days after application (Sherwood and Fanning, 1981).

Total runoff volume and P losses may be lower from grassland than from corn (*Zea mays* L.) stubble/winter cereal during the same series of rainfall events (Withers et al., 1999). There is less of an effect of time between grazing and runoff on TP, possibly because the concentration of P in feces excreted by grazing dairy cowsis relatively low when compared to that of fertilizer, averaging a total of 26.2 g per day in 12 fecal patches, with individual fecal patch volume estimated at 1L (McGechan, 2003). Also, because feces as applied by grazing cows have a smaller surface area than does the fertilizer, less of the fecal deposit is exposed to runoff, which may make it less affected by runoff events. In Ohio, P transport was less than 1.0 kg ha^{-1} y^{-1} in surface runoff of P from beef cattle-grazed pastures receiving P fertilizer only when indicated by soil tests (Owens et al., 2003). In a comparison of 928 U.S. watersheds, those dominated by forest or grazed grassland had the lowest levels of P losses, at no more than 0.1 kg P ha^{-1} y^{-1}, whereas watersheds under other agricultural land uses had between 0.2 – 0.3 kg P ha^{-1} y^{-1} (Hillbricht-Ilkowska et al., 1995).

Conversely, infiltration may be reduced in New Zealand cattle-grazed pastures because of hoof impact and compaction of the soil (Sharpley and Syers, 1976). This may increase surface runoff and erosion, as seen by Sharpley and Syers (1976) when they measured surface runoff from plots with grazing cattle. However, the runoff and sediment carried may be intercepted by vegetation within the pasture or buffers along waterways, moderating and limiting the potential pollution.

Nitrogen transfer in runoff

Nitrogen losses in runoff can be due to N deposition in precipitation that causes the runoff. Nitrogen deposition via precipitation ranges from 5.6 to 19.8 kg ha^{-1} y^{-1} in the continental U. S. (Carlisle et al., 1966; Olson et al., 1973; Taylor et al., 1971). Nitrogen in runoff from an Iowan watershed contour-planted with corn and fertilized at 168 kg N ha^{-1} held only two-thirds of the N measured in precipitation over the site, while the soil acted as a sink for the remainder of the N received (Schuman and Burwell, 1974).

The rate of N loss in surface runoff from grazed swards is relatively low, and may be even less than that received via deposition (Jarvis et al., 1987). Runoff losses from summer-grazed pastures in Ohio were less than 1.0 kg ha^{-1} y^{-1} (Chichester et al., 1979). In Ohio, N runoff from an unfertilized, grazed pasture was low and was similar to losses from adjacent woodland (Owens et al., 1983c). Runoff from pasture receiving 224 kg N $ha^{-1}y^{-1}$ with rotationally grazed beef cattle also showed low N losses, with nitrate concentrations well below 10 mg L^{-1} (Owens et al., 1983b). From a beef cattle summer-grazing area, annual surface runoff over five years averaged 4.2 mg nitrate-N L^{-1} and 8.2

mg total-N L^{-1} for annual transport of 2.1 and 4.0 kg ha^{-1} for the nitrate and total-N, respectively. Runoff and transport from the winter-grazed area was similar. As with P, most N runoff from animal manure occurs when a runoff event follows shortly after slurry applications (Jarvis et al., 1987).

Although research has been done on runoff related to fertilizer and manure applications, and simulated grazing, little information is available on the effect of intensive rotational grazing on nutrient concentrations in runoff. Most of the research agrees that N loss in surface runoff is relatively low, and that the greatest influences on N losses in runoff are the length of time between application and runoff event and the presence of adequate vegetative cover.

Subsurface flow and nutrient transfer

During base flow conditions, streams and rivers are typically fed by groundwater and subsurface flow (Schwartz and Zhang, 2003). Subsurface flow is a term usually used to describe soil water that moves laterally to surface water bodies as shallow perched, groundwater, found above the water table. Some of the nutrients in subsurface flow may move to the deeper groundwater and leave the watershed of origin; however, nutrients transported in subsurface flow may be carried to nearby streams and waterways, directly impacting local watersheds.

Both N and P can reach surface waters by subsurface transport. Owens et al. (1983b) found low levels of N loss in surface runoff from fertilized grazed pastures, but measured nitrate concentrations as high as 18 mg L^{-1} in subsurface flow beneath these pastures. These concentrations were measured under pastures receiving 224 kg N ha^{-1} y^{-1} and were higher under winter-grazed areas than under summer-grazed areas. Most research on N losses in subsurface transport is in reference to N leaching to deeper groundwater, but it should be recognized that subsurface N transport can also reach local streams via lateral subsurface flow.

Phosphorus can be transported in water-soluble forms or sorbed to soil particles (Hesketh and Brookes, 2000; McDowell et al., 2001; McGechan, 2003). Subsurface movement of P is regulated in part by the P sorption capacity of soil and may be substantial in profiles with low P sorption capacity (Hansen et al., 2002; Hooda et al., 2000). Subsurface transport of P is not necessarily tied to mineral fertilizer applications, as increased levels of fertilization did not lead to greater P leachate losses in a German study comparing soils and leachate in monolith lysimeters receiving different crop rotations and P amendments (Leinweber et al., 1999). Phosphorus transport via subsurface flow is often measured in artificial drainage systems which may directly contribute P to local waterways (Sims et al., 1998). The presence of a drainage system often increases the leaching of P (Simard et al., 2000).

Organic P applied in animal manure may contribute to P mobility through the soil profile, because it is less likely to be sorbed and therefore more available for subsurface transport (Alloush et al., 2003; Chardon et al., 1997; Eghball et al., 1996; Kleinman et al., 2003). Kleinman et al. (2003) examined soil profiles for evidence of P movement and found that subsurface transport may be due to preferential flow, as the relationship between soil P and leachate P was poor, suggesting that P did not move within the soil matrix, but through pores. Subsurface P transport may be especially critical after storm events that may cause large amounts of preferential flow (Simard et al., 2000).

Phosphorus contributions to a stream in New Zealand draining a grazed watershed were greater from surface transport than from subsurface transport over three years, in part because stream bank erosion and release of P from suspended particles in runoff made up large proportions of the P transported from the watershed (Sharpley and Syers, 1979).

As part of a study investigating the effects of management intensive grazing (MIG) on water quality, we monitored N and P concentrations in two streams flowing through grazed watersheds on a Maryland dairy farm. For each stream, we sampled biweekly for three years at the point where the stream entered the grazed pasture part of the watershed and at four other sampling stations, each 100 m farther downstream in the grazed pasture land. We expected that stream nutrient concentrations would rise along this sampling transect in proportion to the nutrient loading by surface and subsurface flow from the grazed pasture land. Because P losses are primarily due to surface runoff during storm events, we hypothesized the effects of the grazed watershed on P concentrations would be due to surface flow during storm events. Since N losses are generally due to leaching and subsurface flow, we hypothesized that stream N would be likely to be affected most during base flow, when the streams are fed by groundwater.

Materials and methods

Site selection and sample collection

Grazed 2 is a dairy farm in Baltimore County, Maryland and was a part of a study measuring the effects of management intensive grazing (MIG) on water quality. Grazed 2 has been farmed by the same family for more than 100 years, has been in dairy operations for more than 30 years, and has been under MIG since 1994, seven years before stream monitoring began. Grazed 2 is located in the upland section of the Piedmont Plateau physiographic province, where the average precipitation is 1039 mm and average annual temperature is 13°C. The soils are mainly Glenville loams, overlaying a Cockeysville marble (Cleaves et al., 1968; personal communication, Robert Shedlock, US Geological Survey, June 3, 2003).

In each of the two watersheds on the farm, a stream bisects actively grazed pastureland that accounts $\geq$90% of the land within the watershed on both sides of the stream. Five sites along each stream were identified for regular sampling, with the first site located at the point of the stream's entrance onto the farm, and each consecutive site approximately 100 m farther downstream.

Before Stream A enters the watershed, it flows through a wooded area for approximately 200 m. Stream A meanders through 6.1 ha Watershed A and has grassy banks and two stone-stabilized stream crossings for the herd. A paved farm driveway passes over the stream via a small bridge after sampling site 3, and a man-made pond is located adjacent, but separated from the stream, downstream of the driveway. Cattle were excluded from the stream most of the time, but the herd had some access to the stream, mainly between sampling sites 2 and 4. A paddock located on the east bank of Stream A between sampling sites 3 and 4 was used prior to 2004 as a "sacrifice pasture" where the herd was assembled and fed hay during the winter and where the herd was held when paddocks were too wet tobe grazed without damage. This paddock developed into what is herein termed a "camping area", a place where the herd tends to congregate repeatedly and trample the pasture. This paddock, therefore, suffered visible pasture degradation, denudation, compaction, and erosion.

Stream B, flowing through 5.4 ha Watershed B, was channelized circa 1950 and has perennial woody vegetation along its banks in a riparian zone about 3 m wide along each bank. The herd has no access to Stream B, with the exception of once or twice a year when they are allowed to graze the banks near sampling sites 4 and 5. The stream passes through a 1.5 m-diameter culvert under an unpaved farm lane prior to sampling site 3. Upstream from sampling site 1, before Stream B enters the study farm, it flows through a neighboring dairy farm, which uses both confined feeding and conventional tillage in its management.

Grab samples (130 mL) were collected biweekly from May 2001 through June 2004. Sixty-one sampling dates occurred during base flow conditions and 11 occurred during storm flow conditions. During storm flow, stream levels were substantially higher and much more sediment laden and turbid than during base flow conditions.

Sample pH was measured in the field immediately upon collection, and, beginning in January 2002, each sample was preserved by acidifying to pH<3 with 2-3 drops of 4 *M* H_2SO_4. The samples were transported on ice to the lab where they were stored under refrigeration at 4°C until analysis, unless analysis would not take place within 14 days, in which case they were stored at <-15°C..

A tipping-bucket rain gauge (Spectrum Technologies; Plainfield, IL) was installed on the roof of the milking parlor, approximately 350 m from both streams. On a few occasions when this instrument was not functioning properly, National Oceanic and Atmospheric Administration records were used from a location approximately 10 km away. No attempt was made to monitor stream flow volumes.

Stream sample lab analyses

Stream samples were filtered under vacuum through a 0.2 µm filter (polycarbonate membrane, Nuclepore ® Corporation Filtration Products; Pleasanton, CA). The filtrates were analyzed for nitrate-N, ammonium-N, total dissolved nitrogen (TDN), dissolved reactive phosphorus (DRP), and total dissolved phosphorus (TDP). A fresh membrane was used for each sample, and the filter apparatus was rinsed with distilled water between samples. The membranes from 26 base flow and 8 storm event sampling dates were dried and weighed for sediment measurement and analyses. Once filtered, samples were transferred to fresh sample cups and stored at 4°C unless they were not to be analyzed within 14 days, in which case, long-term storage was at <-15°C. Frozen samples were brought to room temperature (approximately 22-23°C) before analysis.

Filtered stream samples were analyzed for NO_3-N using a Technicon Autoanalyzer II flow injection analyzer (Technicon Industrial Systems; Tarrytown, NY) with a cadmium reduction column and a 2:1 distilled water dilution loop at a rate of 30 samples per hour (Technicon AutoAnalyzer II, 1977). Standard NO_3-N solutions in the concentration range of 0-25 mg L^{-1} were prepared from KNO_3. To bring the samples within the necessary pH range for use in the Autoanalyzer (5-9), 0.5 mL of 0.1 *M* NaOH in 10% NaAc was added to each. This buffer had no colorimetric effect on the procedure, but its dilution effect was included in the calculation of NO_3-N concentration.

Ammonia concentration was determined using an Orion 9512 ammonia specific gas-sensitive electrode and E640 Orion mV specific ion meter (Banwart et al., 1972). One mL of 5*M* NaOH ionic strength adjusting (ISA) solution with thymophalein pH color indicator was added to 10.0 mL of sample in a 20 mL vial to bring the sample pH to >13. Readings were taken while stirring and a logarithmic standard curve was constructed.

Filtered samples were analyzed for DRP using the ascorbic acid method (American Public Health Association et al., 1992). Ten mL of sample was used, with 1.6 mL of the reagent containing 5 *N* sulfuric acid, ammonia molybdate solution, 0.1 *M* ascorbic acid and potassium antimonyl tartrate solution.

Total dissolved nitrogen (TDN) and TDP were determined by a modified alkaline persulfate microwave digestion (Cabrera and Beare, 1993; Hosomi and Sudo, 1986; Johnes and Heathwaite, 1992; Littau and Englehart, 1990). Samples were digested using an alkaline persulfate reagent (45g K_2SO_8 and 9.5g NaOH per 1.0L) heated in a microwave in 120 mL high-pressure Teflon vessels (CEM Corporation; Matthews, NC). Ten mL of sample was put in each vessel with 10 mL of reagent. The vessels were closed using a mechanical capping station to achieve the amount of torque necessary to maintain pressure. Sets of 12 samples were digested on a rotating tray in the center of the microwave oven at full power (actual output 675 W) for 750 seconds (12.5 min). Nitrate and orthophosphate were analyzed in the digested subsamples (now representing TDN and TDP) using the methods described above.

Dissolved organic N (DON) was calculated as:

$$DON = TDN - (NH_4\text{-}N + NO_3\text{-}N)$$

Dissolved organic P (DOP) was calculated as:

$$DOP = TDP - DRP$$

Sediment sample analyses

The amount of total suspended solids (TSS) was determined by drying and weighing each sediment- laden filter after filtration, subtracting the original weight of the filter to give the sediment weight, and dividing this by the volume of the stream sample. Each filter was then microwave digested as described above with 10.0 mL distilled water and 10.0 mL digestion reagent.

The resulting digestate was filtered using a syringe with 25 mm qualitative filters (P5; Fisher Company; Pittsburg, PA) or 25 mm membranes (0.2 µm polycarbonate Acrodisc LC; Pall Corporation; East Hills, NY) if sediment visible to the naked eye remained in the digestant after filtering with qualitative filters. The filtered digestate was then analyzed for TDN and TDP as described above. Particulate N and P were calculated as:

Particulate nutrient concentration = mg particulate-associated nutrient/mL sample collected.

The sediment load was calculated as:

Sediment load = mg sediment/L sample collected.

Statistical analyses

Sediment concentration data were analyzed using GLM (SYSTAT, 1998). Sample pH and season were used as covariates in the model. Stream water data were analyzed using repeated measures GLM (SYSTAT, 1998) with post-hoc hypothesis testing. Seasonal averages were used for the repeated measures (e.g. Fall 2002 = October – December 2002, Winter 2003 = January – March 2003). Analyses were done for both streams together considering the main effects of sampling site and flow, and the interaction of site, flow and stream. A third set of analyses were on base and storm flow data separately to take advantage of the larger data set available for base flow data. For both sediment and stream samples, effects of watershed, flow type (base or storm) and sampling site, and the interaction between site and the other two effects were included.

Because the primary pathway for N transport is through leaching to groundwater, N concentrations at sampling site 5 would be expected to be higher than at sampling site 1 during base flow (Schwartz and Zhang, 2003), and N concentrations at the sites were compared at base flow. Because P is typically transported during surface runoff in storm events, to determine if MIG pasture increased P in surface water, the model compared P concentrations in samples from the five sites in each stream during storm flows.

Results and discussion

Groundwater dropped below the depth of the stream beds because of drought conditions during September 2001 – March 2002. In the spring and summer of 2002, both streams were dry for a period until the second leaching season recharged the groundwater. Fall 2003 and Winter 2004 had higher than normal precipitation (Figure 2-1). In periods of high rainfall, both watersheds experienced temporary surface ponding, which could contribute to both preferential flow of P and runoff.

The average sediment loads during base flow were 0.019 $\pm$ 0.003 and 0.015 $\pm$ 0.002 mg L^{-1} in Streams A and B, respectively (Figure 6-1). During storm flow, the average sediment load increased to 0.033 $\pm$ 0.006 in Stream A and 0.025 $\pm$ 0.005 mg L^{-1} for Stream B.

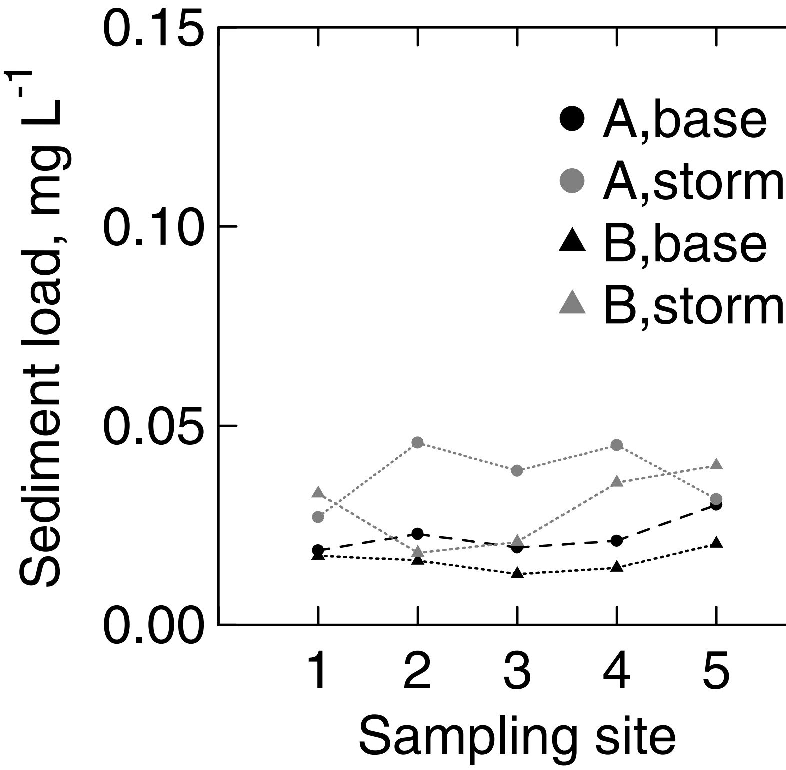

Figure 6-1: Sediment load for Streams A and B during base flow and storm events. Each stream runs through a grazed watershed on a Maryland dairy farm under management intensive grazing. Samples were collected at five sampling sites along each stream. Sampling site 1 for each stream was at the point where the stream entered the grazed watershed, with each site approximately 100 m farther downstream. Sampling site 5 was near the point where the stream exited the dairy farm. Samples were collected biweekly during 47 base flow dates and 11 storm flow events.

Surface water phosphorus

Base flow concentrations of DRP exceeded the EPA critical level of 0.01 mg DRP L^{-1} for surface water, with averages of 0.025 and 0.029 mg L^{-1} for Streams A and B, respectively (Figure 6-2). Base flow TDP concentrations were 0.042 and 0.048 mg L^{-1} in Streams A and B. Average base flow DRP and TDP concentrations remained constant among all five sampling sites within both streams as each flowed through the grazed watersheds (Table 6-1, Figure 6-2). With the addition of an average of 0.009 mg particulate-P L^{-1}, base flowtotal -P concentrations were well within EPA critical levels of 0.1 mg total-P L^{-1}.

In both streams, season affected TDP, DRP, and DOP concentrations during both base flow and storm events (Table 6-2, Figure 6-3). The stream by season effect for base flow TDP suggests that the two streams have different responses to fluctuations in the groundwater, which feeds surface water during base flow.

Increased runoff during winter seasons caused a significant season by flow effect for DRP and TDP through the study period (Table 6-3, Figure 6-3). This was especially pronounced during Winter 2004, when P concentrations increased dramatically during storm events. Winter 2004 followed a period of above-average rainfall which had saturated soil conditions, reducing the already limited infiltration typical of winter months and increasing runoff.

Storm flow DRP and TDP increased to >0.3 mg L^{-1} from approximately 0.1 mg L^{-1} during storm events in the other sampling seasons. The increase was greater in Stream A than in Stream B (Figures 6-2 and 6-3), probably due to the lack of a woody riparian

Table 6-1: Repeated measures GLM F-values for effects on dissolved reactive phosphorus (DRP), total dissolved phosphorus (TDP), dissolved organic phosphorus (DOP) concentrations, and DOP:TDP ratio in two streams in two grazed watersheds. Repeated measures use seasonal means[†]. Samples were collected biweekly at five sites along each stream from January 2002 to June 2004. Seasons included in analyses are: Winter 2002 and Fall 2002 – Spring 2004.

Effect	df	DRP	TDP	DOP	DOP:TDP
Site	4	0.795	0.334	0.312	0.361
Flow	1	0.062	2.822	0.553	2.546
Site x Flow x Stream	4	0.710	0.579	0.366	0.202
Error	10				
Season	7	3.230**	14.487***	8.430***	30.039***
Season x Site	28	0.854	0.321	0.683	0.820
Season x Flow	7	4.540***	20.169***	1.287	7.187***
Season x Site x Flow x Stream	28	0.762	0.567	1.514	1.384
Error	70				

[†] Each season of sampling is represented separately in the model, e.g. Spring 2002, Summer 2002, Fall 2002.

*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.

Table 6-2: Repeated measures GLM F-values for effects on dissolved reactive phosphorus (DRP), total dissolved phosphorus (TDP), dissolved organic phosphorus (DOP) concentrations, and DOP:TDP ratio in two streams in two grazed watersheds. Repeated measures use seasonal means†. Samples were collected biweekly at five sites along each stream from October 2002 to June 2004. Nine measurement periods had a complete set of base flow data, only eight of which had a complete set of storm flow data.

	df	DRP	TDP	DOP	DOP:TDP
Base Flow					
Stream	1	0.997	1.880	0.103	6.207
Site	4	1.556	0.226	0.015	1.108
Error	1				
Season	8	3.976**	48.848***	2.335	14.070***
Season x Stream	32	1.978	7.475***	0.658	1.125
Season x Site	8	0.973	0.885	0.259	0.715
Error	32				
Storm Events					
Stream	4	17.298*	10.183*	4.617	0.001
Site	1	0.413	0.511	0.396	0.347
Error	4				
Season	7	19.197***	19.854*	7.185***	24.661***
Season x Stream	28	4.455**	3.313*	1.289	3.643**
Season x Site	7	0.765	0.573	1.201	0.841
Error	28				

† Each season of sampling is represented separately in the model, e.g. Spring 2002, Summer 2002, Fall 2002.

*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.

Table 6-3: Post-hoc hypothesis testing F-values following the repeated measures ANOVA shown in Table 6-1. The effect of flow (base or storm) during different seasons of sampling was tested. Concentrations of dissolved reactive P (DRP), total dissolved P (TDP), dissolved organic P (DOP), and DOP:TDP were measured in surface water samples collected biweekly from 2 streams running through grazed watersheds on a Maryland dairy farm.

Season	DRP	TDP	DOP	DOP:TDP
Winter 2002	22.755**	17.531*	4.499	20.595*
Fall 2002	4.087	57.128**	5.813	41.074**
Winter 2003	29.708**	128.818***	14.547*	9.861*
Spring 2003	4.092	10.446*	1.137	61.640**
Summer 2003	2.907	2.741	0.813	38.081**
Fall 2003	0.382	0.042	0.018	0.000
Winter 2004	186.347***	48.083**	0.629	1.479
Spring 2004	5.124	12.664**	92.982***	2.659

*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.

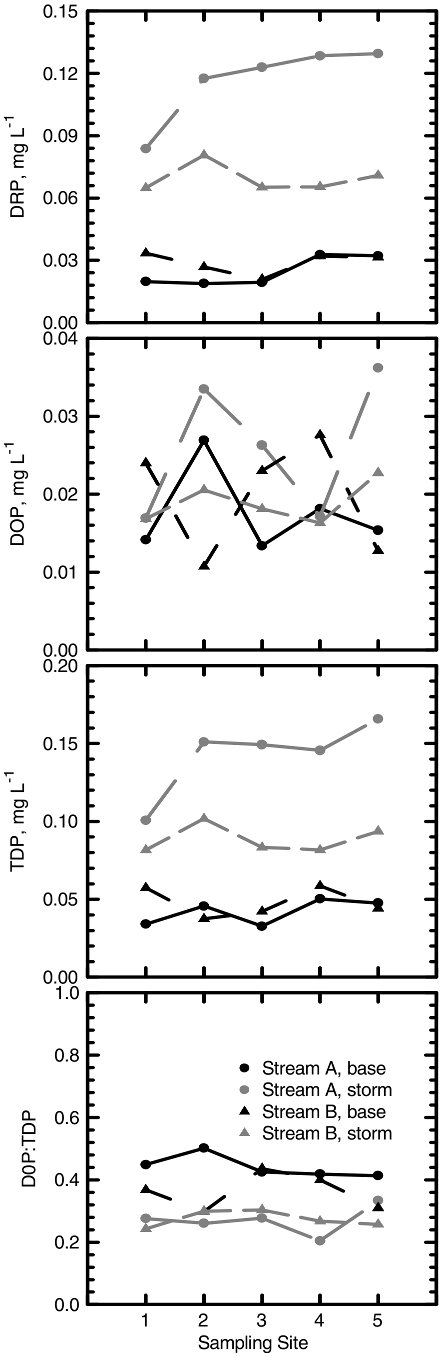

Figure 6-2: Average dissolved reactive phosphorus (DRP), dissolved organic phosphorus (DOP), total dissolved phosphorus (TDP), and DOP:TDP for stream water in two grazed watersheds on a Maryland dairy farm under management intensive grazing. One stream ran through Watershed A, and one stream through Watershed B. Sampling site 1 for each stream was at the point where the stream entered the grazed watershed, with each site approximately 100 m farther downstream. Sampling site 5 was near the point where the stream exited the dairy farm. Samples were collected biweekly during 47 base flow dates and 11 storm flow events.

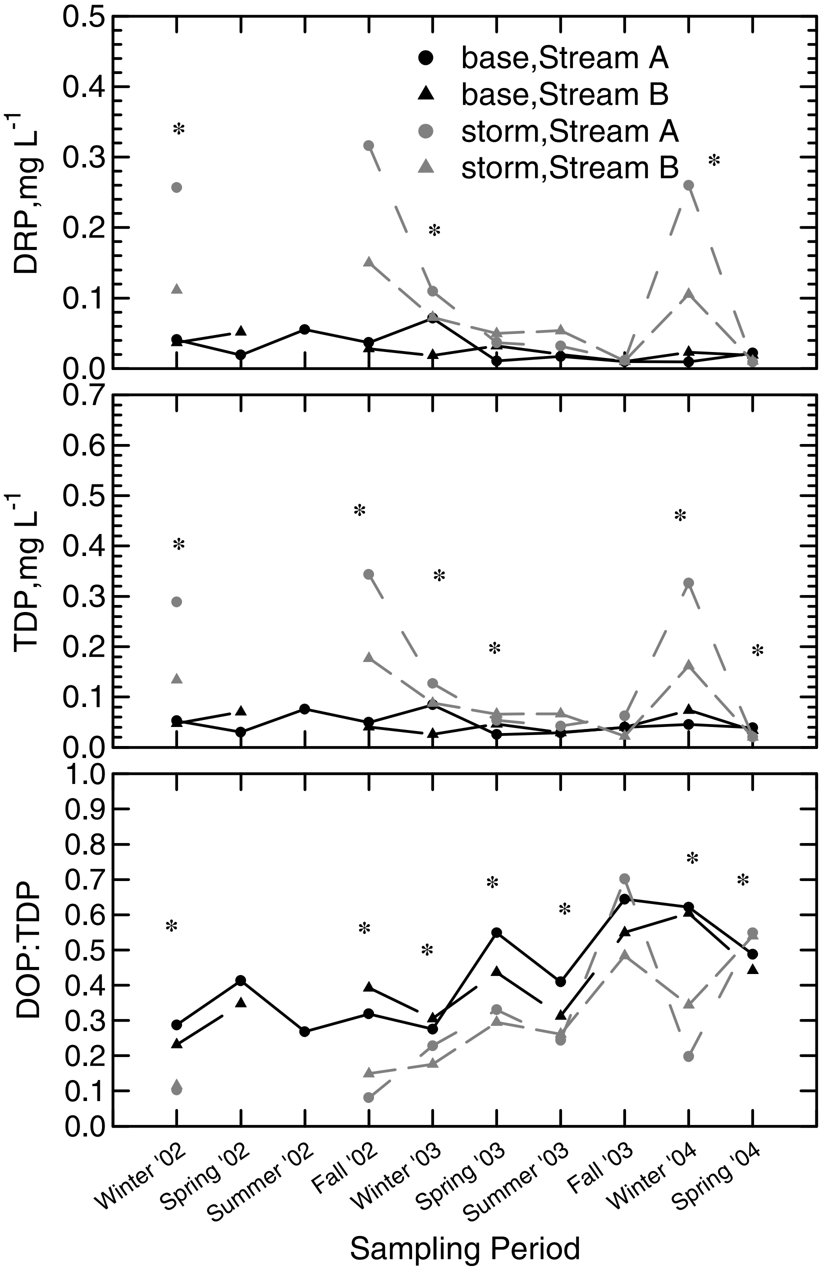

Figure 6-3: Dissolved reactive phosphorus (DRP), total dissolved phosphorus (TDP), and dissolved organic phosphorus (DOP) in two streams flowing through grazed watersheds on a Maryland dairy farm. Samples were collected biweekly at five sites along each stream during 47 base flow and 11 storm flow events. Asterisks indicate significant season by flow interactions.

zone and runoff from the camping area adjacent to sampling sites 3-4 of Stream A (Figures 6-2 and 6-3). Only in Winter 2002 and Winter 2004 did DRP appear to increase across the watershed in Stream A during storm events. The absence of a significant effect of sampling site indicates the grazed pastures were not a significant source of P in runoff, and that P concentration remained unchanged as the streams flowed through the watersheds.

Storm flow concentrations of particulate-P were higher than base flow particulate-P in both streams, but there was no difference between the two streams, or among sampling sites within, and averaged 0.039 mg L^{-1}. Storm flow total P was therefore 0.18 mg L^{-1} in Stream A (0.142 mg TDP L^{-1}) and 0.13 mg L^{-1} in Stream B (0.088 mg TDP L^{-1}), exceeding the EPA guideline of 0.1 mg total-P L^{-1}. Because there was no difference in particulate P from site to site within each stream, the increase in storm flow could be due to re-suspension of the stream bed sediments or from materials carried from upstream. The levels found in both streams were lower than those measured in runoff from calcareous soils on English dairy farms, which held 2.1-9.5 mg TP L^{-1} (Withers et al., 1999).

Concentrations of DOP averaged 0.019 mg L^{-1} and did not differ between the two streams during base flow (Table 6-2). Both streams had an effect of season on DOP, with concentrations close to 0.06 mg DOP L^{-1} during Winter 2004 (Figure 6-3).

The ratio of DOP:TDP was constant among sampling sites in both streams during base flow. However, the ratio differed significantly between the two streams over the course of the study, with an interaction of season and stream during storm events (Table 6-1, Figure 6-2). There was no effect of sampling site, suggesting the differences between the streams may have originated upstream of the study farm, but were not due to effects of the grazed watersheds.

Concentrations of DRP and TDP in Stream A were significantly higher during storm events than those in Stream B. There were no changes in P as Stream B flowed through the grazed watershed, suggesting the stream's increase in P concentration during storm flow was likely due to activity upstream of Grazed 2, with insignificant amounts of P added from within the MIG farm. However, under storm conditions, Stream A had higher P concentrations than did Stream B, with the increases likely attributable to the presence of the camping area. During three storms, Stream A did show an effect of sampling site. However, the absence of a site effect during most storm events makes it unlikely that the grazed land supplied substantial amounts of P in runoff, but the camping area probably contributes some P in runoff during heavy storm events (Figure 6-2). Because Stream A had low levels of suspended sediment but contained higher levels of DRP during storm flow, P transfer in runoff was likely in soluble forms, considered to be a major pathway for P export from grazed catchments (Nelson et al., 1996). Most high concentrations in runoff can be attributed to a localized area within the watershed, such as the camping area in this case, with severe storm events more likely to lead to greater P runoff (Edwards et al., 2000a; Withers et al., 1999).

The highest concentrations of both DRP and TDP measured during storm flow occurred during events with more than 3 cm of rainfall. Stream A concentrations during high rainfall storm events were far greater than those in Stream B, most likely due to greater volumes of runoff carrying more DRP and TDP from the camping area (Figure 6-4). Changes in management of the camping area may improve water quality by reducing

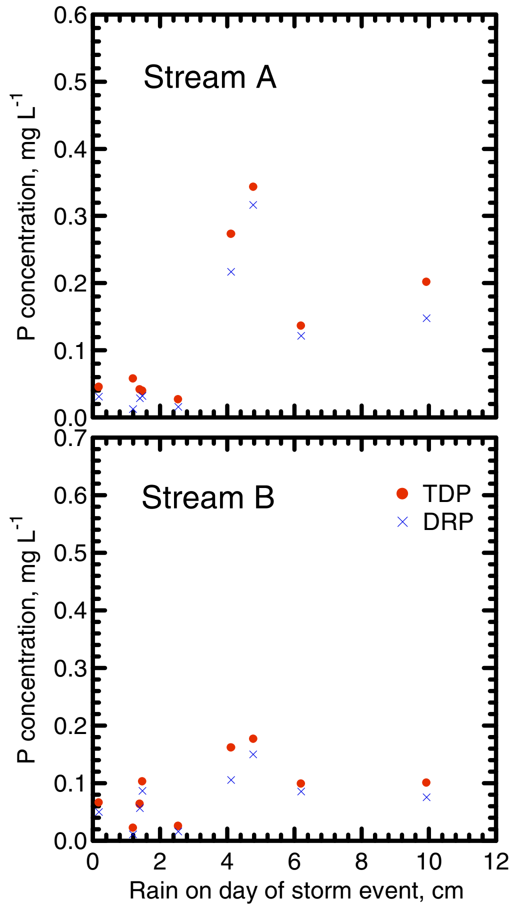

Figure 6-4: Stream DRP and TDP during storm events for two streams running through grazed pastures of Maryland dairy farm. Samples were collected during 11 events over two and a half years. Concentrations for each event are an average of five samples taken along the length of each stream passing through the grazed watershed.

runoff and erosion. During the winter of 2004, the farmer carried out some rehabilitation of the paddock near Stream A after sampling site 2, but the effect was not yet discernible in the data by the end of this study.

Because sampling site was not an effect in either storm or base flow in both streams, we conclude that the grazed pastureland did not contribute significant P to surface water. The camping are may be a source of some P contributions, but these were not evident in the analyses as changes over the course of Stream A's flow through the watershed. The localized pasture degradation associated with a winter camping area along Stream A has been remedied since the study began, and overall, grazing has demonstrated limited transfer of P to surface waters. These results are in agreement with other researchers who have determined that grass-based MIG can limit runoff, and the dispersal of fecal matter in MIG pastures makes runoff less likely to cause P loading to surface water (McGechan, 2003; Owens et al., 2003).

Surface water nitrogen

Nitrogen concentrations in each stream stayed constant through the grazed watersheds and levels were comparable those found in a stream adjacent to a North Carolina dairy farm (Karr et al., 2003). In neither stream was there any difference in nitrate-N, TDN, or DON concentration among the five sampling sites, indicating there were no contributions of N from the surrounding grazed watersheds (Table 6-4).

Nitrate concentrations in Streams A and B were relatively low during base flow ($<$2-3 mg L^{-1}), when surface water is fed by groundwater. Nitrate and TDN concentrations were higher in Stream B than in Stream A, under both base and storm flow conditions (Tables 6-4 and 6-5, Figure 6-5).

Table 6-4: Repeated measures GLM F-values for effects on nitrate-N, total dissolved nitrogen (TDN), dissolved organic nitrogen (DON) and the DON:TDN ratio in two streams in two grazed watersheds. Repeated measures use seasonal means†. Samples were collected biweekly at five sites along each stream from October 2002 to June 2004. Seasons included in analyses are Winter 2002 and Fall 2002 – Winter 2004.

Effects	df	Nitrate-N	TDN	DON	DON:TDN
Site	4	0.012	0.012	0.108	0.168
Flow	1	0.578	0.578	0.062	3.240
Site x Flow x Stream	4	0.118	0.118	2.076	0.384
Error	10				
Season	7	5.723***	14.487***	7.053	7.197***
Season x Site	28	0.965	0.321	0.445	0.905
Season x Flow	7	6.560***	20.169***	2.250	3.380***
Season x Site x Flow x Stream	28	0.345	0.567	1.030	0.837
Error	70				

† Each season of sampling is represented separately in the model, e.g. Spring 2002, Summer 2002, Fall 2002.

*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.

Table 6-5: Repeated measures GLM F-values for effects on nitrate-N, total dissolved nitrogen (TDN), dissolved organic nitrogen (DON) and the DON:TDN ratio in two streams in two grazed watersheds. Repeated measures use seasonal means†. Samples were collected biweekly at five sites along each stream from October 2002 to June 2004. Nine measurement periods had a complete set of base flow data, only eight of which had a complete set of storm flow data.

	df	Nitrate-N	TDN	DON	DON:TDN
Base Flow					
Stream	1	1074.452***	138.979***	0.540	10.317*
Site	4	9.659*	0.785	0.045	2.735
Error	1				
Season	8	19.311***	20.089***	5.028**	8.837***
Season x Stream	8	5.990***	8.198***	0.955	2.223
Season x Site	32	0.731	0.886	0.685	0.287
Error	32				
Storm Events					
Stream	4	912.629***	53.657**	4.488	8.205*
Site	1	5.074	0.215	0.105	0.328
Error	4				
Season	7	24.109***	9.324***	4.791**	6.238***
Season x Stream	28	7.973***	5.087**	1.619	1.844
Season x Site	7	0.895	0.915	0.775	1.079
Error	28				

† Each season of sampling is represented separately in the model, e.g. Spring 2002, Summer 2002, Fall 2002.

*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.

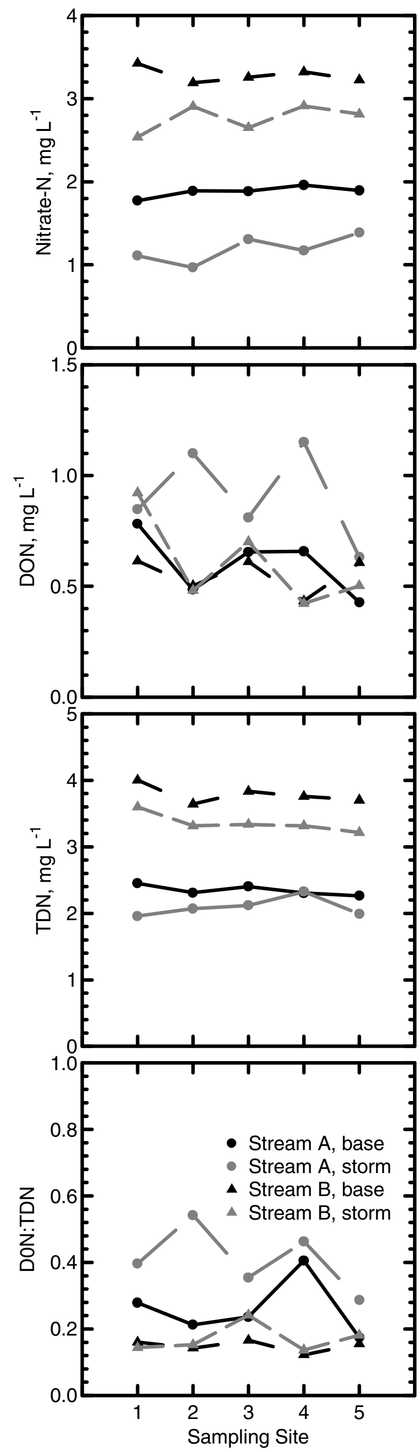

Figure 6-5: Average Nitrate-N, dissolved organic nitrogen (DON), total dissolved nitrogen (TDN), and DON:TDN for streams in two grazed watersheds on a Maryland dairy farm under management intensive grazing. Sampling site 1 for each stream was at the stream's entrance into the grazed watershed, with each site approximately 100 m farther downstream. Samples were collected biweekly from July 2002 through June 2004 during 47 base flow dates and 11 storm flow events.

The levels of nitrate-N found in base flow of the two study streams corresponded closely with the average nitrate-N levels found in groundwater under the two grazed watersheds, which were 2-4 mg L^{-1} in Watershed A, and 5-7 mg L^{-1} in Watershed B (Chapter 2). These levels are slightly higher than those found in the streams at base flow, but the differences in groundwater nitrate-N correspond to the differences measured in the two streams. Since nitrate transport is typically through leaching and subsurface flow, the lack of change in concentration over the course of the streams' base flow demonstrated that shallow groundwater under the grazed watersheds does not contribute significant levels of nitrate to surface waters.

Nitrate-N concentrations in both streams decreased during storm flow (1.2 and 2.8 mg L^{-1} in Streams A and B) than during base flow (1.9 and 3.3 mg L^{-1} in Streams A and B). There was a significant interaction between season and flow (Table 6-6 and Figure 6-6).

Lower nitrate concentrations occur during storm flow because runoff, which typically does not carry much N (Edwards et al., 2000b; McLeod and Hegg, 1984), diluted the stream water. When the soil was saturated from previous rainfall, infiltration was reduced, and there was more runoff. This occurred most significantly in Winter 2004 (Table 6-6).

Particulate-N additions were somewhat higher during storm flow, but still added less than 0.5 mg N L^{-1}. There was no difference in particulate-N between the streams or sampling sites, suggesting very little N transfer from overland flow.

Dissolved organic N did not differ significantly between the two streams during base flow, averaging 0.6 mg DON L^{-1} (Table 6-5, Figure 6-5). There was a difference

Table 6-6: Post-hoc hypothesis testing F-values following the repeated measures ANOVA shown in Table 6-4. The effect of flow (base or storm) during different seasons of sampling was tested. Concentrations of nitrate-N, total dissolved N (TDN), dissolved organic N (DON), and DON:TDN were measured in surface water samples collected biweekly from 2 streams running through grazed watersheds on a Maryland dairy farm.

Season	Nitrate-N	TDN	DON	DON:TDN
Winter 2002	24.175**	88.918**	NI	14.742*
Fall 2002	210.178***	27.997**	0.157	0.556
Winter 2003	24.593**	6.032	0.764	38.077**
Spring 2003	17.597*	2.769*	0.977	3.270
Summer 2003	6.471	4.371	1.037	0.154
Fall 2003	27.912**	28.924**	1.633	0.010
Winter 2004	772.086***	14.172*	6.034	230.783***
Spring 2004	6.188	55.585**	0.049	2.365

NI Not included.

*, **, *** Significant at <0.05, 0.01, and 0.001 probability levels, respectively.

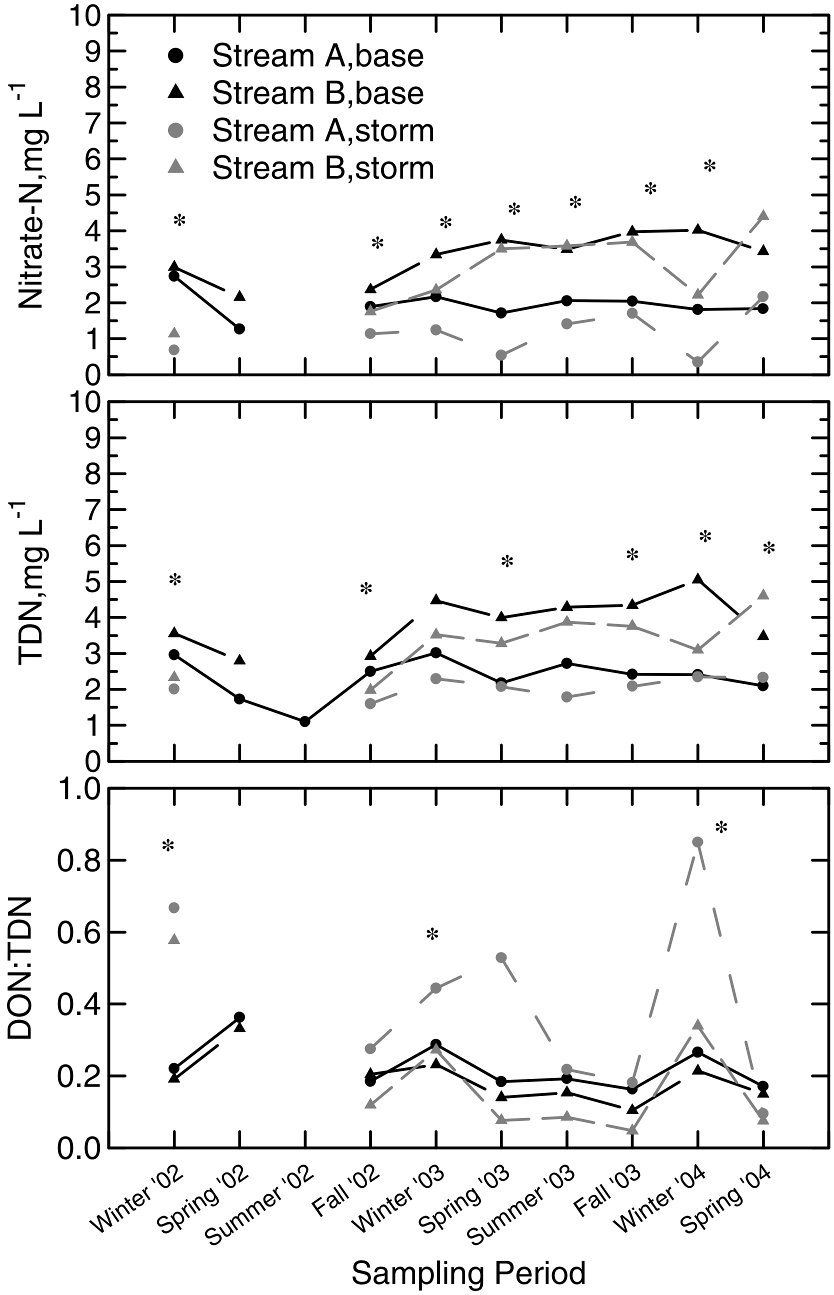

Figure 6-6: Nitrate, total dissolved nitrogen (TDN) and the ratio of dissolved inorganic nitrogen (DON) to TDN in two streams flowing through grazed watersheds on a Maryland dairy farm. Samples were collected biweekly at five sites along each stream during 47 base flow and 11 storm flow events. Asterisks indicate significant flow by season interaction.

between the two streams in DON:TDN, likely because of the difference in nitrate concentrations. The ratio of DON:TDN in Stream B did not differ much between base and storm flow (0.15-0.17 DON), but Stream A DON:TDN ranged from 0.26 (base flow) to 0.41 (storm events).

The higher proportion of DON in sections of Stream A during fall and winter storm flow may be due to a higher volume of runoff from denuded camping area along that stream (Figure 6-5). Additionally, Watershed A, through which Stream A flows, had a DON:TDN of 0.46 in shallow groundwater (Chapter 2). A buried A horizon was found in Watershed A adjacent to sampling site 2 at a depth of 65-79 cm (Figure 6-6, Appendix B). Typically, the A horizon has a higher concentration of organic matter which may have contributed to the elevated DON:TDN in groundwater and stream water. The higher ratio at sampling site 2, therefore, could have been a result of subsurface flow from the buried A horizon found 20 m away.

Increased DON:TDN was also evident during storm events with greater rainfall (Figure 6-7). Runoff during heavier rainfall probably carried DON associated with manure and surface soil at the camping area situated between sampling sites 3-4. Watershed B had a lower proportion of DON:TDN in groundwater, at approximately 0.19, and had lower DON in surface water as well. No buried A horizon or signs of heavy runoff were noted in that watershed.

Concentrations of TDN (Tables 6-3 and 6-4, Figure 6-5) were lower during storm flow (2.1 and 3.4 mg L^{-1} for Streams A and B) than during base flow (2.4 to 3.8 mg L^{-1} for Streams A and B). Both streams had an interaction of flow and season, although the differences were not as dramatic as those of nitrate and P concentrations. Mean

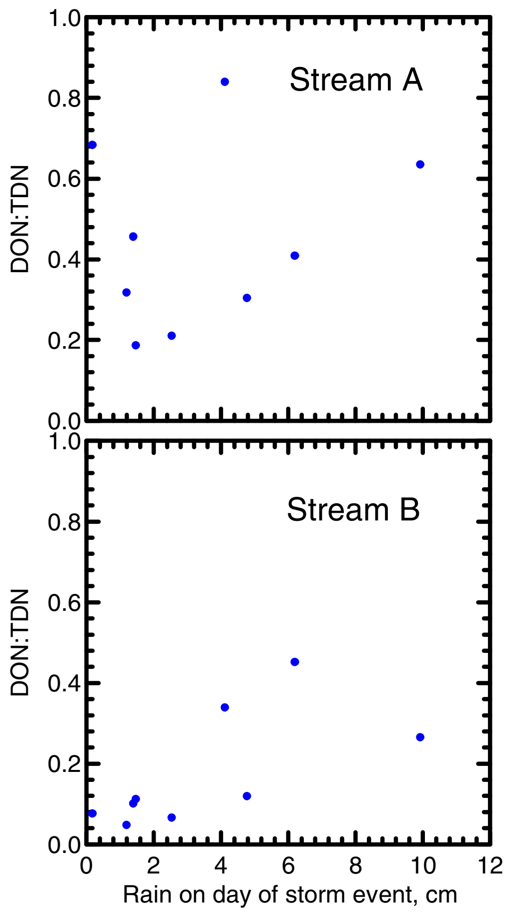

Figure 6-7: Stream dissolved organic nitrogen to total dissolved nitrogen ratio (DON:TDN) during storm events for two streams running through grazed pastures of Maryland dairy farm. Samples were collected during 11 events over two and a half years. Concentrations for each event are an average of five samples taken along the length of each stream passing through the grazed watershed.

particulate-N concentration was higher in storm flow (0.319 mg L^{-1}) compared to base flow samples (0.136 mg L^{-1}). There was no difference in the particulate-N concentration between the streams or among sampling sites within each stream.

Ammonia-N concentrations averaged 0.8 mg L^{-1} during base flow (Figure 6-8). Ammonia concentrations were measured from only one storm, averaging 0.5 mg L^{-1}. While this is a measurable and substantial amount, it did not appear to be affected by grazing practices. Concentrations did not differ between the two streams or among sites within either stream.

Conclusions

Proper management of grazed watersheds, including maintaining vegetation to maximize pasture cover, can reduce nutrient pollution and runoff (Nelson et al., 1996). Nitrate and TDN did not increase in the two streams as they flowed through the two grazed watersheds, under either base or storm flow. In storm flow when P runoff is most likely to cause nutrient pollution, a winter holding area was the likely source of P transfer to one stream, but there were no significant changes in P concentration as the stream flowed through that watershed. P levels in the second stream did not change over the course of the stream's path through the watershed. The lack of increased nutrient concentrations as these streams traversed the study pastures suggests that MIG can be practiced without posing a significant environmental risk for nutrient pollution in surface waters.

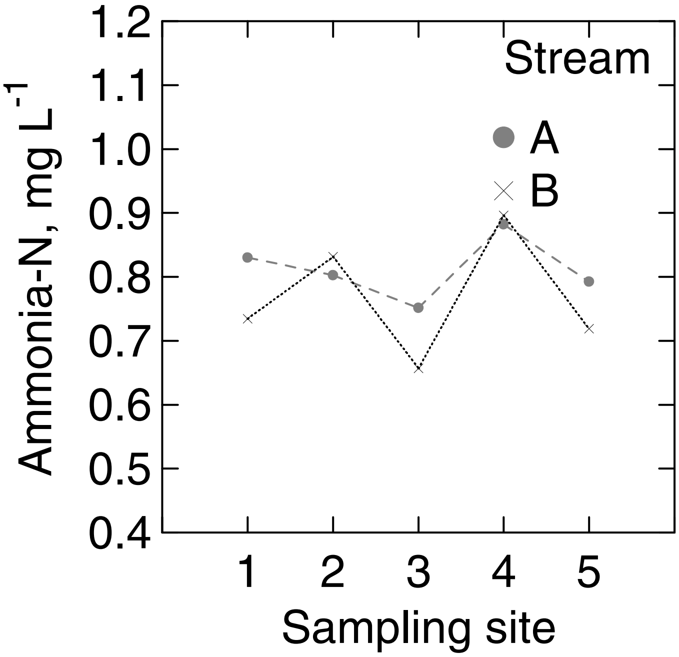

Figure 6-8: Base flow ammonia-N concentrations from two streams running through two grazed watersheds on a Maryland dairy farm operated under management intensive grazing. Sampling site 1 for each stream was at the stream's entrance into the grazed watershed, with each site approximately 100 m farther downstream.

Chapter 7: Conclusions

Groundwater nitrogen and phosphorus

Three years of monitoring surface, pore and ground water in four grazed and two cropped watersheds on three dairy farms did not show excessive leaching under grazing. Unacceptably high levels of nutrient pollution, especially nitrate leaching, had been predicted by most previous research. Those studies indicated that, among other things, the high nitrogen content and liquid form of urine made it an easily leachable source of N.

The present study took place during a period that included an extremely dry year followed by an unusually wet year, which, taken together, provided a very high potential for nutrient leaching. Extremely dry conditions in the first leaching season of the study limited groundwater recharge and allowed nutrients to be retained within the soil profile. This drought period was followed by heavier than normal precipitation, which then flushed the accumulated nutrients through the soil profiles and recharged the groundwater. Even under these circumstances, nitrate leaching from grazing was not in excess of the EPA 10 mg L^{-1} maximum contaminant load. Overall, nitrogen losses correlated to calculated farm surpluses, suggesting whole-farm nutrient balances play an important role in nutrient management. Nitrogen loading, rather than grazing, appears to be a greater risk for nutrient pollution.

Phosphorus concentrations under all six watersheds were greater than projected in literature under agricultural land. The rather high P concentrations found in groundwater in this study suggests that substantial P leaching is taking place, or has taken place, under these animal-based agricultural systems, probably as a result of P surpluses built up over decades. In ecologically sensitive areas, such as the Chesapeake Bay watershed, groundwater containing the levels of P found in our study could represent a significant source of P loading and should be considered in nutrient management planning and water quality policy Traditionally, P loading to streams has been considered to be almost entirely an issue of surface runoff, mainly as P attached to eroded sediment. Losses through surface runoff and erosion would call for very different management approaches than through leaching.

All three farmers involved in the study have incorporated nutrient management into their planning, and P concentrations in groundwater were still high on two of the three farms. The third farm had lower concentrations, and we hypothesize that this was because geologic material underlying this farm was calcareous in nature, causing P to be sorbed on carbonates and form highly insoluble calcium phosphate compounds within the profile, thus greatly decreasing losses of P to groundwater.

The high level of P found in groundwater could be due, in part, to a long history of manure application that caused P to build up. However, we did not find a higher level of P in watersheds identified as nearest the homestead versus concentrations in groundwater under the farther watershed. Higher P in proximity to the homestead was expected due to the practices in previous decades of applying much more manure on the more convenient, closer fields. Instead, P levels in groundwater were similar in watersheds both closer and farther from the homestead, suggesting P leaching under animal-based agriculture is a potential source of nutrient pollution because of current activity, not just previous manure applications. Because of the long history of agriculture on all three farms, soils may have reached a threshold sorption capacity, allowing P leaching in years before the farmers took up nutrient management methods.

In most of the hundreds of water samples we analyzed, there were substantial amounts of dissolved organic N and P. Both DON and DOP may be transformed into algal or plant-available forms, adding to the measured environmental pollutants of inorganic N and P. Advances made in this study and in other research have developed more efficient and safer methods for DON and DOP determination. With these improvements, a greater understanding of the potential for DON and DOP leaching should be established easily. Currently, most researchers studying N and P leaching to groundwater have measured only inorganic N and P on the assumption that these were the only important forms leaching. Because of the measurable presence of DON and DOP in ground and pore water, these pools should be considered in future research.

Surface water nitrogen and phosphorus

Our research found no indication that the pastures were a source of nutrient pollution to the two streams running through grazed watersheds. Each stream was sampled at five points along its course through the farm, and samples were collected biweekly during both base and storm flow events. During base flow, when streams are fed by groundwater, neither stream showed any increase in N or P concentration with distance flowed through the grazed paddocks. Even during storm flow, the two streams did not pick up additional N as they flowed through the grazed watersheds. Similarly, even though most P loading is thought to occur during storm flow, the only increase in P concentration observed was in one watershed where the stream flowed past a paddock in which the dairy herd had congregated for convenient winter feeding. This "camping area," trampled by high numbers of cattle and enriched in nutrients by importing large quantities of hay, may have been the source of the measured increase in streamwater P. The otherwise low impact of grazing found in these two streams suggests that properly managed grazed pastures do not pose a major environmental risk for pollution in surface waters.

Because MIG protected the watershed from erosion and most nutrient runoff and produced relatively low levels of nitrate in groundwater, this dairy management system can be considered as a potential environmental Best Management Practice. Grazing has provided dairy farmers with an alternative to confined feeding. Many farmers have found MIG to be an improvement, both economically and socially. This study shows that MIG is an appropriate choice environmentally as well.

Appendix A: Additional farm data and nutrient balances on three Maryland dairy farms

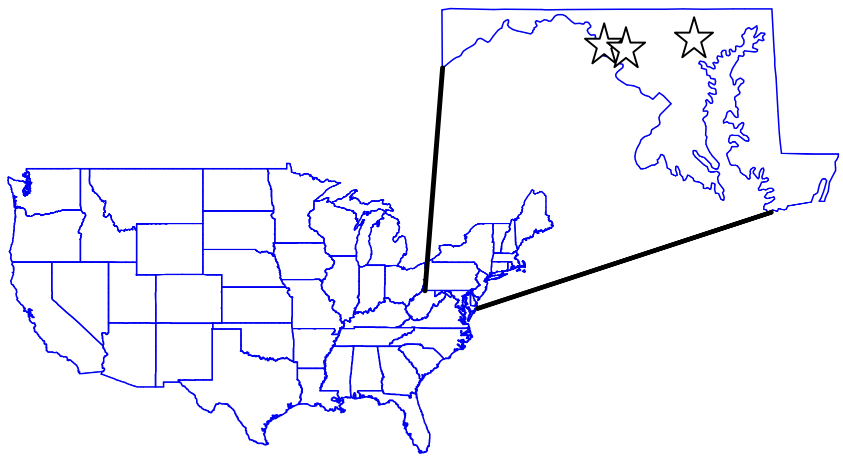

Figure A-1: Study dairy farm locations in Maryland. Farms are marked with stars; Confined and Grazed 1 are located on adjacent tracts of land in Frederick County, and Grazed 2 is in Baltimore County.

Nutrient balance information

Each farmer completed several questionnaires detailing nutrient imports and exports. Questions included: the amount and type of fertilizer purchased and applied, manure application and test results, hay and bedding material purchased or sold, the amount and type of feed purchased, daily ration information (including net energy intake, % crude protein, % phosphorus), herd size and breeds, and milk and animals sold. One of the questionnaires was from the Maryland Nutrient Balance system (Kohn, 2000). The questionnaires were followed with interviews with the individual farmers for clarification and further information on farm history. Each interview was at least one hour in length. Information on crops grown and acreage (including leguminous crops) was shared by the Confined farmer, and forage quality information (discussed below) was used to determine the proportion of nitrogen fixing species in grazed pastures.

The nitrogen and phosphorus content of imports and exports were calculated. The fixation rates were estimated as 125 kg ha^{-1} for clover and 200 kg ha^{-1} for alfalfa. Milk nitrogen was calculated using milk urea nitrogen data (discussed below). Nitrogen in rations was calculated from crude protein content (Protein = N*6.25). Exports in animals, hay, and silage sold were calculated based on weight and estimated nutrient contents.

Exports were subtracted from imports. The total was divided by the area of the farm for a per hectare value (Table A-1).

Table A-1: Nutrient balance information from three Maryland dairy farms. Data was collected from farmers using questionnaires and interviews. Data shown is an average of 2001-2003.

		Grazer 1	Grazer 2	Confined
Farm, ha		83	71	245
AU† ha^{-1}		0.95	2.2	2.1
Vegetation		Pasture: 8% legume	Pasture: 24% legume	6 y rotation: corn-oats-alfalfa
AUD‡ or Manure		348	810	23 t ha^{-1} y^{-1}
Surplus, kg ha^{-1} y^{-1}	N	54	71	173
	P	1.5	7.8	9.9

† AU = one animal unit of 454 kg.

‡ AUD = annual AU days per ha.

Milk urea nitrogen

Milk urea nitrogen (MUN) can be used as a predictor of cow N excretion (Jonker et al., 1998). Dairy cows consuming excess N will have higher MUN levels and will excrete more N in urine and feces. Milk samples were collected monthly on each farm for a year, or during lactation on the grazing farms. The farmers took samples from the bulk tank as close to the first of each month that the tank was filled and sent the samples in a preservative to the Lancaster D.H.I.A. lab. The lab analyzed the samples for MUN, fat and protein content, soluble oxygen, and somatic cell count using a Bentley wet-chemistry MUN analyzer.

Data was reported back to the farmers, as it could be applied to immediately improve herd diet. Levels were generally within the acceptable range, showing appropriate nutrient management and ration supplies on each farm.

Table A-2: Milk urea nitrogen in bulk tank milk samples on three Maryland dairy farms. Samples were collected by the farmers and analyzed at the Lancaster D.H.I.A. lab.

Month	Confined	Grazed 1	Grazed 2
	---------------mg MUN dl^{-1}---------------		
December 2001	15.31	14.45	6.98
January 2002	13.81	-	-
February 2002	14.16	-	-
March 2002	14.90	-	-
April 2002	12.44	12.51	-
May 2002	15.04	-	11.26
June 2002	14.40	15.31	-
July 2002	14.66	13.90	13.65
August 2002	13.34	12.47	-
September 2002	10.27	18.78	15.56
October 2002	11.92	-	18.45
November 2002	13.46	-	-
March 2003	-	14.04	-
April 2003	-	16.93	16.75
May 2003	-	17.36	-
May 2003, 2nd sample	-	13.69	-

Pasture forage quality

Forage samples were taken from the grazed watersheds five times during one growing season. Samples were always collected in paddocks that were ready to be grazed within 24 to 48 hours. Forage was collected from within five 0.25m^2 quadrats along a diagonal transect of the paddock. The forage was cut to a height of 2.5 cm from the soil surface. Forage from within each of the five quadrats was put in five perforated brown paper bags and brought back to the lab where each bagged sample was sorted into grass, legume, and weed samples. The forage was dried, along with a composite of five 15-cm by 50-cm swatches for wet chemistry analysis. The wet chemistry composite sample was bagged in a plastic sample bag and transported to the lab on ice.

Each of the sorted subsamples were dried in a plant-drying oven for 72 hours and weighed. The dry samples were ground, put in capsules and analyzed for CHN by high temperature combustion at the University of Maryland Soil Testing Lab using the CHN 2000 (LECO Corporation; St. Joseph, Michigan) (Campbell, 1992).

The wet chemistry samples were stored at <-15° C until sent for analysis at the Cumberland Valley Analytical Services Lab. Prior to shipment, the samples were freeze-dried to avoid affecting the protein and fiber analyses (Kohn and Allen, 1992). Samples underwent a standard group of analyses, including determination of moisture, dry matter, crude protein, soluble protein, degradable protein, and total digestible nutrients.

The data were made available to the farmers and showed expectable variation due to seasonal differences. Plant composition and nitrogen content may be used in construction of nutrient balances for the grazed farms.

Table A-3: Soil test results for profile and composite core samples collected from three Maryland dairy farms. Sites were distinctly different topographic areas within each watershed. Soil samples were collected from two watersheds on each farm, corresponding groundwater and pore water monitoring. Soil samples were analyzed at University of Maryland Soil Testing lab. Extractable Mg, P, K, and Ca were measured using Melich 1. C, H, and N were determined by high temperature combustion. Percent organic matter (OM) was measured by loss on ignition.

Watershed	Site	Depth	Texture	pH	Mg	P	K	Ca	OM	C	H	N
		cm			----Soil Test Index†----				-----%-----			
Confined A	1	0-10	sil	7.3	123	234	250	422	5.6	2.890	0.701	0.281
Confined A	1	0-15	sil	7.1	117	202	202	318	4.5	2.310	0.654	0.218
Confined A	1	20-40	sil	6.8	107	26	31	191	3.1	0.910	0.598	0.113
Confined A	1	40-70	sil	7.0	92	5	20	184	2.3	0.369	0.781	0.045
Confined A	1	70-100	sil	7.0	95	7	21	292	2.5	0.184	0.781	0.559
Confined A	2	0-10	sil	7.2	123	162	257	496	4.5	2.500	0.666	0.228
Confined A	2	0-15	sil	7.2	121	117	169	558	4.3	2.590	0.662	0.231
Confined A	2	20-30	sil	7.1	105	34	56	228	2.6	1.040	0.638	0.103
Confined A	2	30-70	sil	7.1	102	10	23	142	1.8	0.360	0.659	0.055
Confined B	1	0-20		6.92	102	6.7	22	223	2.9	0.800	0.619	0.092
Confined B	1	20-40		6.81	99	5.2	18	208	3.4	0.509	0.689	0.062
Confined B	1	0-15	sil	6.7	103	28	61	229	4.0	1.740	0.652	0.180
Confined B	1	40-80		6.87	121	3.2	13	207	3.4	0.313	1.030	0.058
Confined B	1	80-90		6.36	82	6.5	13	184	2.2	0.162	0.665	0.043
Confined B	1	90-100		6.72	109	12.7	18	116	3.0	0.858	0.766	0.090
Confined B	1	100-110		6.86	98	9	15	221	2.5	0.120	0.671	0.032
Confined B	1	110-120		6.58	94	11	16	199	2.7	0.174	0.421	0.038
Confined B	2	0-10		6.73	117	36.6	40	219	4.8	1.870	0.698	0.199
Confined B	2	10-20		6.84	111	22.2	28	207	3.5	1.150	0.642	0.119
Confined B	2	160-180		6.51	95	6.2	22	127	2.3	0.357	0.612	0.062
Confined B	3	0-10		6.41	116	26.6	40	181	3.4	1.740	0.639	0.176
Confined B	3	0-15	sil	6.3	103	20	61	179	4.3	1.990	0.723	0.205
Confined B	3	0-20		6.89	124	97.5	42	238	4.3	1.550	0.601	0.179
Confined B	3	10-30		6.20	116	12.1	23	158	3.7	1.010	0.396	0.122
Confined B	3	30-70		6.56	128	7.6	14	127	2.9	0.843	0.545	0.098
Confined B	3	70-90		6.57	114	10.2	16	109	2.9	0.812	0.481	0.086
Confined B	3	110-120		6.66	107	15.2	21	94	2.8	0.751	0.490	0.094
Confined B	3	160		6.51	100	6.3	24	109	2.6	0.375	0.698	0.056
Confined B	3	160-180		6.31	94	6.1	21	133	2.5	0.303	0.662	0.048

Watershed	Site	Depth	Texture	pH	Mg	P	K	Ca	OM	C	H	N
		cm			-----Soil Test Index-----				-----%-----			
Grazed 1A	1	0-20	sil	6.60	110	100	106	175	3.7	1.98	0.595	0.177
Grazed 1A	1	0-15	sil	6.60	108	95	165	184	3.6	1.75	0.625	0.188
Grazed 1A	1	20-40	sil	7.10	113	88	47	171	2.1	0.645	0.482	0.085
Grazed 1A	1	40-50	sil	7.20	111	70	31	143	1.7	0.346	0.488	0.032
Grazed 1A	1	50-80	sil	6.80	114	95	17	192	2.2	0.511	0.511	0.077
Grazed 1A	1	80-120	sil	7.00	113	103	14	195	1.6	0.477	0.511	0.049
Grazed 1A	2	0-15	sil	6.70	109	110	146	179	3.5	1.430	0.567	0.146
Grazed 1A	2	0-15	sil	6.90	114	216	258	212	4.6	2.400	0.733	0.2328
Grazed 1A	2	0-10	sil	6.80	122	262	376	204	5.1	2.540	0.706	0.254
Grazed 1A	2	10-20	sil	7.10	113	198	229	213	2.8	1.14	0.878	0.134
Grazed 1A	2	20-50	sil	7.20	97	9	90	151	1.9	0.273	0.724	0.038
Grazed 1A	2	50-80	sil	7.00	93	8	45	175	2.0	0.196	0.782	0.025
Grazed 1A	2	80-120	sil	6.20	80	7	34	179	2.2	0.137	0.826	0.043
Grazed 1B	1	0-10		6.71	102	42.6	194	190	6.6	3.270	0.779	0.330
Grazed 1B	1	0-15		6.50	102	20	109	186	4.4	2.180	0.686	0.211
Grazed 1B	1	10-30		6.82	107	16.4	55	152	3.5	1.050	0.570	0.126
Grazed 1B	1	30-50		6.79	100	6.4	22	149	2.7	0.455	0.652	0.075
Grazed 1B	1	50-70		6.91	77	5.3	19	211	2.4	0.271	0.687	0.051
Grazed 1B	1	70-90		6.74	81	6.7	17	197	2.5	0.328	0.657	0.046
Grazed 1B	1	90-120		6.90	109	5.0	14	241	2.9	0.139	0.833	0.043
Grazed 1B	2	0-10	sil	7.00	113	80	282	170	4.2	1.960	0.647	0.19
Grazed 1B	2	0-15	sil	6.60	101	4.6	193	160	3.9	1.840	0.628	0.169
Grazed 1B	2	10-20	sil	7.00	105	27	117	135	2.9	1.000	0.579	0.116
Grazed 1B	2	20-50	sil	7.20	107	22	66	119	2.5	0.862	0.548	0.093
Grazed 1B	2	50-60	sil	6.90	99	17	32	116	2.0	0.601	0.554	0.066
Grazed 1B	2	60-80	sil	6.30	99	22	23	102	2.2	0.603	0.490	0.078
Grazed 1B9	2	80-120	sil	6.90	102	18	242	110	1.7	0.313	0.611	0.057
Grazed 1B	3	0-15		6.60	99	57	147	191	4.2	1.940	0.668	0.192
Grazed 2A	1	0-15	sil	6.60	116	50	118	201	6.0	3.180	0.780	0.297
Grazed 2A	1	0-15	sil	6.20	117	92	80	168	4.7	2.480	0.618	0.250
Grazed 2A	1	15-37	sil	6.50	89	28	36	89	2.2	0.939	0.447	0.096
Grazed 2A	1	37-65	sil	6.60	93	11	26	112	2.4	0.993	0.780	0.297
Grazed 2A	1	65-93	sil	6.60	107	16	25	148	3.5	1.410	0.447	0.096
Grazed 2A	1	93-108	sil	6.70	120	122	114	147	1.9	0.784	0.459	0.092
Grazed 2A	1	108-128	sil	7.00	110	77	24	99	0.7	0.299	0.725	0.121
Grazed 2A	2	0-15	sil	6.50	104	10	24	64	4.3	2.310	0.573	0.072

Watershed	Site	Depth	Texture	pH	Mg	P	K	Ca	OM	C	H	N
		cm			-----Soil Test Index-----				-----%-----			
Grazed 2A	2	0-15	sil	6.80	108	105	92	157	3.7	2.080	0.601	0.192
Grazed 2A	2	15-27	sil	6.90	104	7	22	70	2.0	1.000	0.293	0.025
Grazed 2A	2	27-45	l	7.00	100	5	19	65	1.5	0.367	0.583	0.214
Grazed 2A	2	45-58	l	6.90	95	5	16	67	1.3	0.285	0.438	0.101
Grazed 2A	2	56-70	l	6.90	92	4	17	56	1.3	0.185	0.458	0.055
Grazed 2A	2	70-100	l	6.70	107	5	22	50	1.2	0.130	0.483	0.041
Grazed 2A	2	100-122	sicl	5.20	111	60	59	96	1.6	0.150	0.463	0.035
Grazed 2A	2	122-172	sicl	5.60	103	30	21	59	1.5	0.147	0.506	0.022
Grazed 2A	3	0-10	sil	6.60	98	6	21	60	4.5	2.410	0.759	0.030
Grazed 2A	3	0-15	sil	6.60	127	190	62	158	3.8	1.740	0.567	0.173
Grazed 2A	3	10-20	sil	7.00	100	5	23	42	2.6	0.755	0.615	0.029
Grazed 2A	3	30-50	sil	6.70	101	6	22	28	1.6	0.333	0.691	0.242
Grazed 2A	3	50-80	l	5.90	102	11	23	157	1.5	0.180	0.496	0.095
Grazed 2A	3	80-100	sil	5.60	108	105	92	168	1.3	0.163	0.566	0.070
Grazed 2A	3	100-120	l	5.30	117	92	80	158	1.3	0.123	0.466	0.034
Grazed 2B	1	0-15	SIL	6.5	112	95	111	161	4.1	1.940	0.557	0.185
Grazed 2B	1	0-17	SIL	6.3	108	69	44	123	3.6	1.770	0.546	0.167
Grazed 2B	1	17-33	SIL	6.5	105	20	21	60	2.0	1.110	0.527	0.110
Grazed 2B	1	33-70	SIL	6.6	107	7	23	69	1.2	0.289	0.418	0.047
Grazed 2B	1	70-83	L	6.7	109	7	28	98	1.4	0.202	0.565	0.041
Grazed 2B	1	83-103	L	6.7	108	8	25	96	1.2	0.137	0.482	0.034
Grazed 2B	1	103-133	SIL	6.5	111	16	31	105	1.9	0.602	0.532	0.077
Grazed 2B	1	133-157	L	6.4	116	11	16	40	1.2	0.124	0.457	0.033
Grazed 2B	2	0-15	SIL	6.7	104	78	90	158	3.8	1.970	0.632	0.179
Grazed 2B	2	0-15	SIL	6.6	113	89	127	159	4.0	1.980	0.558	0.193
Grazed 2B	2	15-27	SIL	6.4	106	57	24	93	2.7	1.180	0.530	0.121
Grazed 2B	2	27-40	SIL	6.4	107	16	22	74	2.2	0.663	0.495	0.079
Grazed 2B	2	40-80	L	6.6	114	6	24	99	1.7	0.263	0.600	0.051
Grazed 2B	2	80-100	L	6.3	112	10	18	45	1.7	0.132	0.484	0.029
Grazed 2B	2	100-125	L	5.6	92	10	23	27	1.2	0.112	0.466	0.031
Grazed 2B	2	125-173	L	5.7	116	9	17	30	1.1	0.072	0.441	0.024
Grazed 2B	3	0-15	SIL	6.6	105	77	97	156	4.4	2.130	0.638	0.201
Grazed 2B	3	0-15	SIL	6.5	109	87	86	138	3.9	1.930	0.586	0.182
Grazed 2B	3	0-10	SIL	6.3	106	56	85	126	4.1	2.030	0.639	0.203
Grazed 2B	3	10-20	SIL	6.5	106	52	34	113	3.0	1.240	0.583	0.129
Grazed 2B	3	20-40	SIL	6.4	109	5	25	71	2.3	0.451	0.745	0.059
Grazed 2B	3	40-60	SIL	6.3	112	5	22	69	2.1	0.252	0.762	0.062
Grazed 2B	3	60-90	L	6.5	110	6	19	73	2.5	0.256	0.934	0.067
Grazed 2B	3	90-120	L	6.5	115	5	19	67	2.7	0.273	1.110	1.110
Grazed 2B	4	0-10	L	6.2	107	71	28	111	3.3	1.620	0.553	0.167

Watershed	Site	Depth	Texture	pH	Mg	P	K	Ca	OM	C	H	N
		cm			-----Soil Test Index-----				-----%-----			
Grazed 2B	4	10-20	SIL	6.3	111	72	22	88	2.9	1.270	0.547	0.130
Grazed 2B	4	20-40	L	6.2	106	6	22	63	1.8	0.307	0.566	0.053
Grazed 2B	4	40-60	L	6.5	107	6	23	98	1.8	0.269	0.681	0.049
Grazed 2B	4	60-80	L	6.3	108	7	24	90	1.6	0.165	0.641	0.044
Grazed 2B	4	80-120	SIL	6.2	110	9	28	83	1.4	0.110	0.571	0.047

† Results are reported as Soil Fertility Index.

Index values: Low=0-25, Medium=26-50, Optimum=51-100, Excessive=>100

To convert to lbs A^{-1}: [P_2O_5= (P index + 2.327)/0.499] ; [K_2O= (K index + 0.439)/0.314]; [Mg= (Mg index- 0.271)/0.382]; [Ca=(Ca*-0.403)/0.058].

Appendix B: Watershed characteristics: piezometer depths and soil profile descriptions

Table B-1: Piezometer depths (m) by nest and watershed for three Maryland dairy farms. Piezometer nests were 18 m apart in a transect, with nest a located at or near the drainage outlet for each watershed.

	Grazed 1		Grazed 2		Confined	
	Watershed A	Watershed B	Watershed A	Watershed B	Watershed A	Watershed B
Nest A	5.12	1.44	1.11	1.96	1.07	2.40
	6.01	2.42	1.99	2.85	1.99	3.30
	6.57	3.33	2.86	3.75	2.89	4.17
		6.12		5.04		
Nest B	3.28	1.97	1.12	1.98	2.13	2.56
	4.04	2.96	1.99	2.81	2.84	3.42
	4.83	3.79	2.52	3.85	3.88	4.47
	5.34	6.36		6.04		
Nest C	2.64	2.08	1.53	2.29	2.03	5.04
	2.45	3.13	2.60	3.21	2.85	6.00
	4.53	4.00	3.51	4.34	3.82	6.62
		6.69		5.44		
Control	4.53			7.00	7.11	
	6.48					
Watershed, ha	2.63	18.8	6.12	5.4	1.71	11.58

Soil profile descriptions

The following soil profile descriptions were done in the field, on the dates given. Each profile described was intended to represent a distinct topographic area of the watershed.

Date	6/06/2003
Watershed	Confined A
Location	Between nests a and b in piezometer transect.
Site	1
Depth (cm)	Description
0-10	Silt loam, heavy. Very friable.10YR 4/6. Decaying crop residues in upper 3 cm.
10-30	Silt loam. Fine angular blocky throughout.7.5YR 4/6.
30-40	Silty clay loam, light. Strong friable. 7.5YR 4/6. Greenstone fragments.
40-50	Silty clay loam. 7.5YR 4/6. Mn stains. Clay skins. Wet. Water entered hole-perched water table? Weathered Mn and Fe.
50-60	Silty clay loam. 7.5YR 5/8. More Mn stains. Soil saturated.
60-70	Clayey-clay. 7.5YR 4/6 Coarse fragments.
70-80	Clay loam-clay. Water draining in, soil within peds is not saturated. Some sand.
80-90	7.5YR 4/6. Weathered greenstone. Rubbed color 10YR 4/6. Black stains: Mn, mild reduction (7.5YR 2/0).
90-100	7.5YR 5/6. 30% Mn. Very common Mn.

Date	6/06/2003
Watershed	Confined A
Location	Twenty m above nest c.
Site	2
Depth (cm)	Description
0-2	Silt loam. 7.5YR 3/4. Friable. Strong fine, angular blocky.
2-20	Silt loam-fine silty/clay loam. Fine angular blocky.7.5YR 4/4. Coarse gravel, coarse schist fragments.
20-30	Silty clay loam. Very friable. Fine angular blocky. 5YR 4/6. Earthworm channel covers (5YR 3/4).
30-40	Clay loam, percent of clay increasing (35-40% clay 5YR 4/6.). Common clay skins.
40-50	Clay (45-50%). Coarse fragments. 5YR 4/6
50-60	Coarse fragments common-quartz, greenstone- weathered parent material. Hematite nodule (10R 3/6).
60-70	Pseudolith of weathered parent greenstone – 5YR 6/4. Mg 1-2 mm concentration, hematite nodules.
70+	5YR 4/6. Small fragments of greenstone.

Date	9/10/2003
Watershed	Confined B
Location	Top of ridge, 75 m west of transect.
Site	1
Depth (cm)	Description
0-20	Silt clay loam. 5YR 4/6
20-40	Clay loam. 2.5YR 4/6. Plow layer ends at 25 cm.
40-80	Clay/clay loam. Common Mn concentration.
80-90	Less clay with more weathered material. Weathered rock, wet soil.
90-100	Greenstone parent material. 10YR 5/6.
100-120	10YR 5/6

Date	9/10/2003
Watershed	Confined B
Location	Between nests a and b, 7m east of the piezometer transect on the swale.
Site	3
Depth (cm)	Description
0-70	Silt loam. Very friable. 7/5YR 4/6 eroded sediment.
At 70	Barely noticeably yellower 7.5YR 4/6
70-90	Silt loam. Fine Mn stains/concentrations. Friable.
90-100	Large charcoal chunk of burnt wood (2 cm diameter) as a result of fire at the original land surface. 7.5YR 3/4.
100-110	7.5YR 3/4. Mottling. Buried A horizon from original land use.
110-120	7.5YR 4/6. Alfalfa (*Medicago sativa* L.) tap root. Clay increases
120-140	Finer texture. More clay. 7.5YR 5/8. Mn concretions 1-2 mm diameter.
140-150	Clay loam (buried B horizon?). 5G 5/2 mottles (former root channel?) in 7.5YR 5/6-5/8 matrix.
150-160	Clay loam. Thick red clay skins 5YR 4/6. Mn concretions. Wet-saturated pore.
160-170	Clay loam. Gray/green mottles prevalent 5G 6/2.
170-180	Clay loam. Roots present. Colors lose blue tinge when exposed to air. Gray/green mottling along roots. Water at 175cm, probably rising.
180-190	Clay loam. Root exudates, gleying, anaerobic conditions. Enter quartz gravel. Parent material. Heavy gleying, perched water table, 50% of soil surfaces at 5G 6/1, the rest at 7.5YR 5/6.

Date	9/10/2003
Watershed	Confined B
Location	35m northwest of nest c, ridge shoulder.
Site	2
Depth (cm)	Description
0-10	5YR 4/6 mixed with 7.5YR 5/8 topsoil. Very friable, all sizes, with 2+cm chunks of quartz, angular rock.
10-30	5YR 4/6 greenstone parent material, concretions. 2.5YR 5/6, some 5YR 4/6 in the mix.
30-40	2.5Y 5/6 mottles in 5YR 4/6 weathered greenstone
40+	Hit rocks, no sample- 10 cm chunk of greenstone removed.

Date	6/06/2003
Watershed	Grazed 1A
Location	25m up from slope break, at shoulder by the barn.
Site	2
Depth (cm)	Description
0-10	Silt loam. Granular. 7/5YR 4/6. Roots prevalent.
10-20	Silt loam. 7.5YR 3/4. Small quartz fragments. The top 20cm are Ap horizon.
20-40	Silty clay loam. 7/5YR 4/6.
40-50	Silty clay loam. Clay skins common, almost continuous. 7.5YR 4/6.
50-80	Silty clay loam. Clay skins common, almost continuous. 7.5 YR 4/6.
80-100	Silty clay loam. Coarse angular blocky breaking into fine organic blocky. 5YR 4/6. Common organic coatings in channels. Earthworms, roots, still common.
100-120	Silty clay loam, more clay. Coarse angular blocky breaking into fine organic blocky. Yellower than 2.5YR 4/8, redder than 5YR 5/8. Black root tracings common.

Date	6/06/2003
Watershed	Grazed 1A
Location	25 m towards barn from nest b.
Site	1
Depth (cm)	Description
0-10	Silt loam. Granular, fine, strong. 7.5YR 4/4 throughout profile. Many fine roots.
10-30	Silt loam. Granular. Common fine roots. Quartz fragments.
30-40	Silt loam. Weak, angular blocky. Alluvial sand from high flow episode.
40-50	Loamy sand. Saturated sandy soils, coarse-medium sand particles.
50-60	Fine sandy loam.
60-80	Silt loam.
80-90	Heavy silt loam. Silt particles vary. Sediment- no coarse fragments evident.
90-100	Silt loam. Start some lighter areas. Some Mn stains.
100-110	Some Mn stains. Saturated, a bit redder.
110-120	Earthworm channels.

Date	6/06/2003
Watershed	Grazed 1B
Location	30cm toward cemetery from nest c.
Site	2
Depth (cm)	Description
0-20	Granular, friable. 7.5YR 4/4. Roots very common.
20-30	Friable. 7.5YR 4/4. Quartz fragments.
30-40	7.5YR 4/6
40-50	Medium angular blocky. Structure inconclusive- breaks to small angular blocky to 90 cm depth.
50-60	Light areas of 7.5YR 5/8 in 7.5YR 4/6 matrix.
60-90	7.5YR 3/4
90-100	Silty clay loam (40% clay). Blocky structure. 7.5YR 4/6.
100-120	7.5YR 4/6

Date	4/19/2002
Watershed	Grazed 1B
Location	In waterway, between nests b and c.
Site	1
Depth (cm)	Description
0-30	Silt loam. 7.5YR 4/6. Ap horizon.
30-50	Silty clay loam. Increased clay. 7.5YR 5.8
50-70	Mottles (10YR 6/2) and Mn concentrations in 10YR 5/6. Some red 7.5YR 5/8.
70-90	Silt loam. 7.5YR 5/8. Mottles common.
90-120	Silt loam. Less clay. 7.5YR 5/8. Mottles.

Date	4/19/2002
Watershed	Grazed 2A
Location	Between nests a and b of the piezometer transect.
Site	1
Depth (cm)	Description
0-15	Fine sandy loam. Strong medium granular. Micaceous.10YR 3/3
15-65	Fine sandy loam to silt loam. Midrange medium granular.10 YR 4/4
65-79	Buried A horizon, oxidized root channel.
79-94	Clay loam to clay (40% clay).10YR 2/1 matrix with 10YR 4/6 Fe concentrations. Fine, few roots.
94-110	Sandy clay loam (40% c), 40% sand (mica).10YR 3/1 matrix with common fine 10YR 5/8 Fe concentrations.
110-130	10YR 4/1 (rubbed) sandy loam (20% c). 10YR 5/1 matrix with 10YR 5/6 Fe concentrations, fine, few.

Date	4/19/2002
Watershed	Grazed 2A
Location	Above nest c in piezometer transect, at the terrace/old fence line.
Site	2
Depth (cm)	Description
0-15	Silt loam (15% clay).
15-28	Silt loam (15% clay).
28-46	Silt loam (15% clay).
46-58	Silt loam (18-20% clay).
58-71	Silty clay loam (26% clay).
71-99	Silty clay loam (35% clay). Red Fe concentrations – mottles.
99-123	Clay (40%).
123-165	Clay (48%).

Date	6/09/2003
Watershed	Grazed 2A
Location	20 m uphill from cedar tree which is above nest c in piezometer transect.
Site	3
Depth (cm)	Description
0-10	Silt loam with micaceous sand. Fine strong granular structure. 10YR 4/4. Roots common.
10-20	Silt loam with micaceous sand. Fine strong granular structure. 10YR 4/6 Roots common.
20-30	Silt loam, granular, some platiness. Structure difficult to determine in augered soil.10YR 5/6
30-40	Silt loam. Sub angular blocky with platy tendency.10YR 5/6
40-50	Platy tendency. 7.5YR 5/8 Small redox mottles.
50-60	Silt loam. Platy tendency. 7.5YR 5/8 Mottles (10YR 7/1) present. Strong gray mottles on ped faces. Water enters auger hole, perched free water table.
60-80	Sub-angular blocky with platy tendency.5YR 5/8 Red and gray mottles present in 7.5YR 5/8.
80-90	Fine sandy loam. Less clay, weathered mica present. Prevalent red and gray mottles. Slightly platy.
90-100	More gray, less clay, more mica.
100-110	Drier, crumbled rock.
110-120	Hematite, almost sapprolite.

Date	4/19/2002
Watershed	Grazed 2B
Location	Near nest a.
Site	1
Depth (cm)	Description
0-18	A1. Silt loam, granular. 10YR 3/6. High bioactivity.
18-30	A2. Silt loam, increased clay content. 10YR 3/4.
30-40	Silt loam/silty clay loam, increased clay content.10YR 5/6.
40-59	Silt loam/silty clay loam, increased clay content.10YR 5/6.
59-71	Silty clay loam.10YR 6/8.
71-81	10YR 5/8 base, some black (Mn) flecking.7.5YR 5/8. Red mottles appear. Concentrations of redox. Concentrations are medium-sized (0.5 x0.5 cm).
81-94	Silt loam (25% clay).Matrix 7.5YR 5/8. Continued mottling 5YR 5/8. Depletion evident 10YR 6/8.
94-104	Silt loam (less clay). Same color.
104-118	Silt loam (less clay).Same color. 10YR 5/8 with red mottles (10YR 6/2).
118-123	Silt loam (less clay).7.5YR 5/8 Increasing red concentrations.
123-135	Fine sandy loam. Low clay, high micaceous sand.
135-147	Loamy sand. 7.5YR 5/6.
147-160	Slight mottling, highly micaceous. Small quartz pieces.

Date	4/19/2002
Watershed	Grazed 2A
Location	Near nest b.
Site	2
Depth (cm)	Description
0-15	Silt loam (15% clay). Strong granular. 10YR 4/4.
15-28	Silt loam (15% clay). Medium granular. 10YR 4/6.
28-41	Silt loam (18% clay). Medium granular. 10YR 4/6.
41-53	Silty clay loam (25% clay).10YR 5/6 with small mottles (7.5YR 5/8) begin.
53-66	Loam (25% clay).10YR 5/6 with small mottles (7.5YR 5/8).
66-81	Loam (32% clay- clay loam?).7.5YR 5/8 with small mottles.
81-91	Loam (20% clay).10YR 5/6 rubbed color. Coarse black stains on ped faces and sand weathered schist.
91-104	Sandy clay loam. 10YR 5/6 Weathered mica schist, gritty, sticky.
104-127	Loam to fine sandy loam. Angular block structure with gray (10YR 6/2) ped faces.10YR 5/6 matrix with medium Fe concentrations (7.5YR 5/8).
127-150	Fine sandy loam (10% clay, 10% fine mica sand), weathered schist with black stains, gray 10YR 6/2 depleted zones. Hit rock or hard fragments.

Date	6/09/2003
Watershed	Grazed 2B
Location	Twenty m toward nest C from western edge of paddock, on raised area which seemed rockier than surrounding paddock.
Site	3
Depth (cm)	Description
0-10	Silt loam. Strong medium granular. 10YR 4/4 Roots common.
10-20	Silt loam. Strong medium granular. 10YR 4/4. Roots common. Quartz gravel.
20-30	Clay loam-clay. Angular blocky from here down. 7.5YR 5/8. Few roots. Organic-filled worm channels (10YR 4/4). Some quartz fragments.
30-40	Silty clay loam, heavy clay. 7.5YR 5/8. Common fine roots. Mn flecks, some quartz fragments.
40-60	Silty clay loam, roots present.
60-80	Silty clay loam. 7.5YR 5/8. Gray mottles (10YR 7/1). Mn concentrations. Red mottles.
80-90	Silty clay loam. More gray and red mottles.
90-100	Silty clay loam. Quartz? Fragment.
100-110	Silty clay loam. Large quartz? May be limestone or marble.

Date	6/09/2003
Watershed	Grazed 2B
Location	Five m east of C in lower spot.
Site	4
Depth (cm)	Description
0-10	Silt loam. Strong medium granular.10YR 4/4. Common fine roots.
10-20	Silt loam. Strong medium granular. 10YR 4/4. Common fine roots. Some quartz fragments up to 3 cm.
20-30	Silty clay loam. 7.5YR 5/6. Weathered parent material: quartz and schist. Small flecks of quartz.
30-40	Silty clay loam, micaceous. 7.5YR 5/6. Redder color.
40-50	Heavy clay loam. 7.5YR 5/6. Mottles (5YR 4/6). Small rocks weathering. Mn stains common, and concentration. Water flow through this depth.
50-60	Sandy clay loam (coarse sand). Small rocks. Weathered schist and Mn concentrations.
60-70	Clay loam with a lot of micaceous sand.
70-80	Increasing red mottling and common black stains. Fe and Mn concentrations. Not saturated, but hole is filling with water.
80-100	Clay. Heavy black stains.
100-120	Clay. Yellow stains 10YR 6/8.

Appendix C: WATBAL Model

Topographic models were made for five of the six watersheds. For the sixth watershed, Grazed 2B, a hard-copy model was used, and area and slope were estimated from the map itself.

WATBAL is a monthly water balance calculation based on:

$$P = ET + R \pm \Delta SM$$

Calculations used to determine drainage were based on the following equations:

Drainage = P – (Snow cover – Initial Snow cover) – Runoff –

AET – abs(Δ Soil Moisture) – (Matrix losses for Soil Moisture)

Changes in soil moisture were determined based on the following conditions and corresponding calculations:

1) Temperatures less than 0°C, no evapotranspiration:

Initial Soil Moisture*(1 – matrix loss)

2) Precipitation plus snow melt exceeds PET, leading to AWC, with the potential for flux:

Soil Moisture at field capacity

3) The same conditions existing as (2), but AWC does not lead to flux:

(Initial Soil Moisture + Infiltration)*(1- Matrix loss)

4) ET exceeds P + Snowmelt, causing a draw on Soil Moisture. But Soil Moisture is above the critical content at which ET can no longer be met by PET:

(Initial Soil Moisture + (P - PET + Snowmelt*root uptake))*(1 – Matrix loss)

5) Soil Moisture is below the critical content described in (4):

(Initial Soil Moisture – (Soil Moisture * ((Initial Soil Moisture – Soil Moisture at wilting point)/AWC)(abs(P – PET) + Snowmelt)*root uptake)))*(1- Matrix loss)

6) ET causes a demand on the soil. Initial soil moisture is greater than the critical moisture content and withdrawal to critical content takes place at PET. After reaching critical content, ET is satisfied by withdrawal at AET.

Initial Soil Moisture – ((Initial Soil Moisture – Soil Moisture at wilting point – (Critical Soil Moisture * AWC)) + (((abs(P– PET + Snowmelt)*root) – (Initial Soil Moisture – Soil Moisture at wilting point – (Critical Soil Moisture * AWC))) * Soil Moisture rate * Critical Soil Moisture * (1 – Matrix loss)

7) Anything else = 0, and Soil Moisture is at plant wilting point

For these calculations:

P = Precipitation

ET = Evapotranspiration

AET = Actual evapotranspiration

PET = Predicted evapotranspiration

AWC = Available Water Content

Critical Soil Moisture is the soil moisture / AWC which controls AET

Matrix losses for soil moisture, equal to [(Soil Moisture at start + Infiltration)*Matrix loss] was used when the change in soil moisture was not equal to 0. This accounts for change in soil water storage.

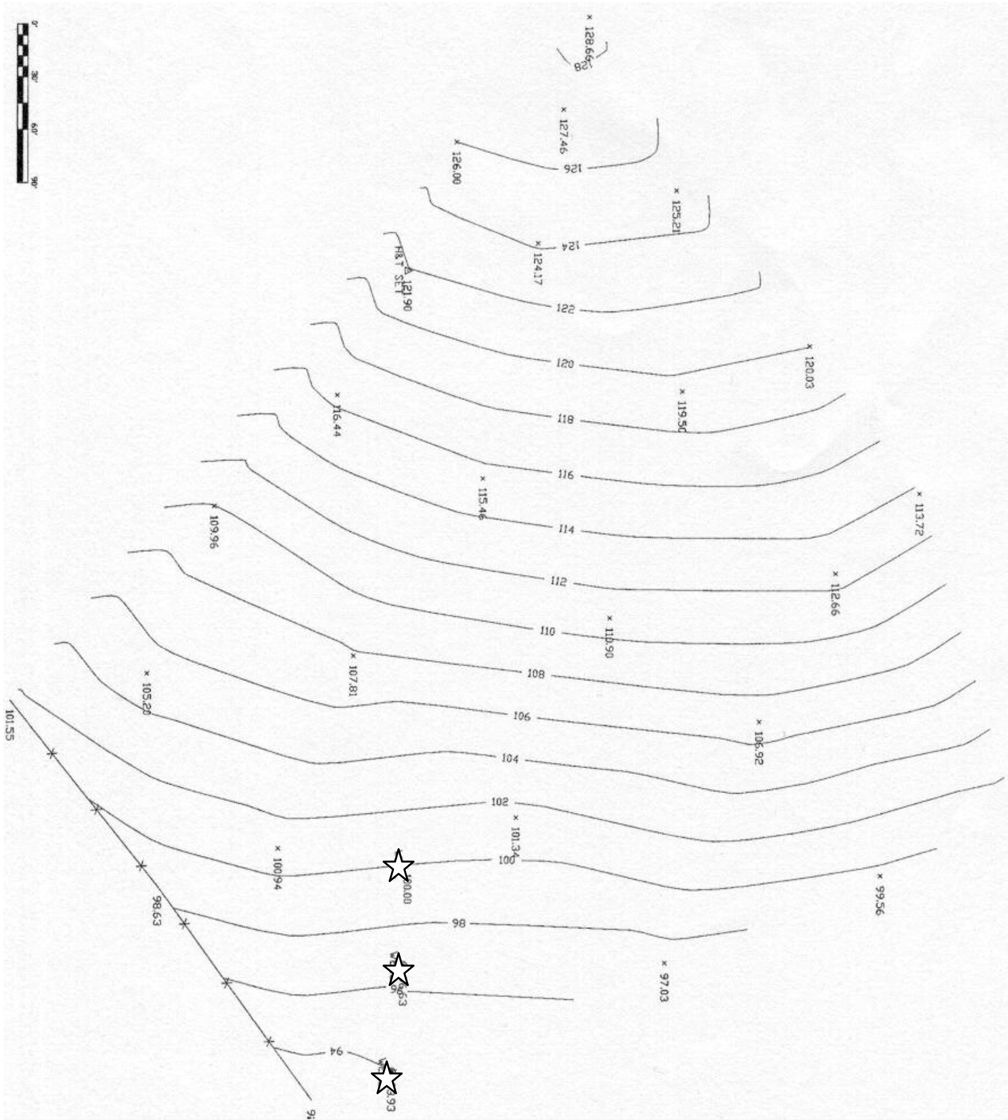

Figure C-1: Topographic map of Confined A watershed on Maryland dairy. Stars point out piezometer nests a, b, and c from top to bottom. Nests are 18 m apart, along the flow line. Bar is 90' long. Elevation changes 2' with each contour line.

Figure C-2: Topographic map of Confined B watershed on Maryland dairy farm. Stars mark piezometer nests a, b, and c, moving from left to right on the flow line. Nests are 18 m apart. Bar is 300’ long. Elevation changes 2’ with each contour line.

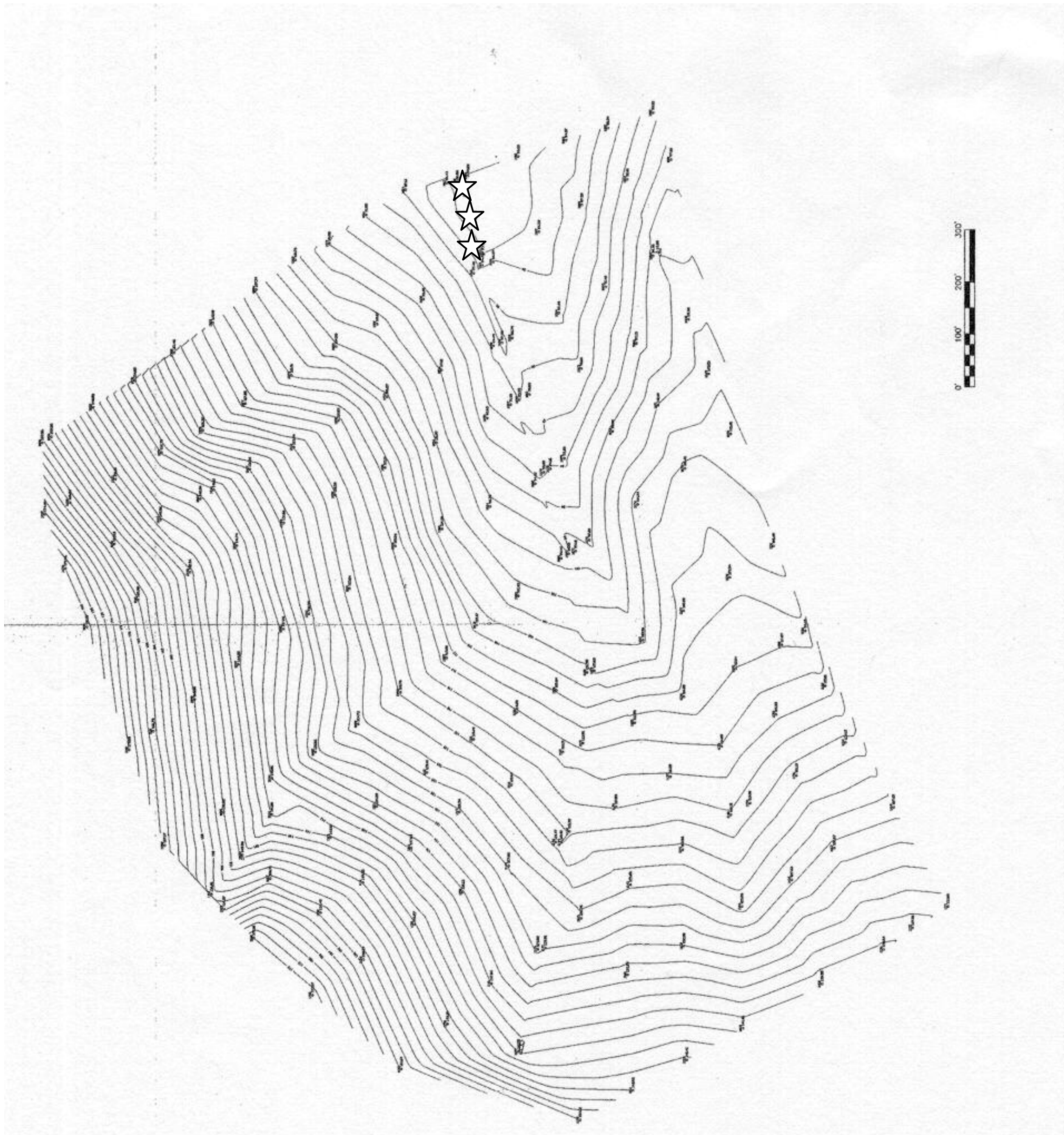

Figure C-3: Topographic map of Grazed 1A watershed on a Maryland dairy farm. Piezometer nests are marked with stars, with nests a, b, and c running from top to bottom, 18 m apart along the flow line. Elevation changes 2' with each contour line.

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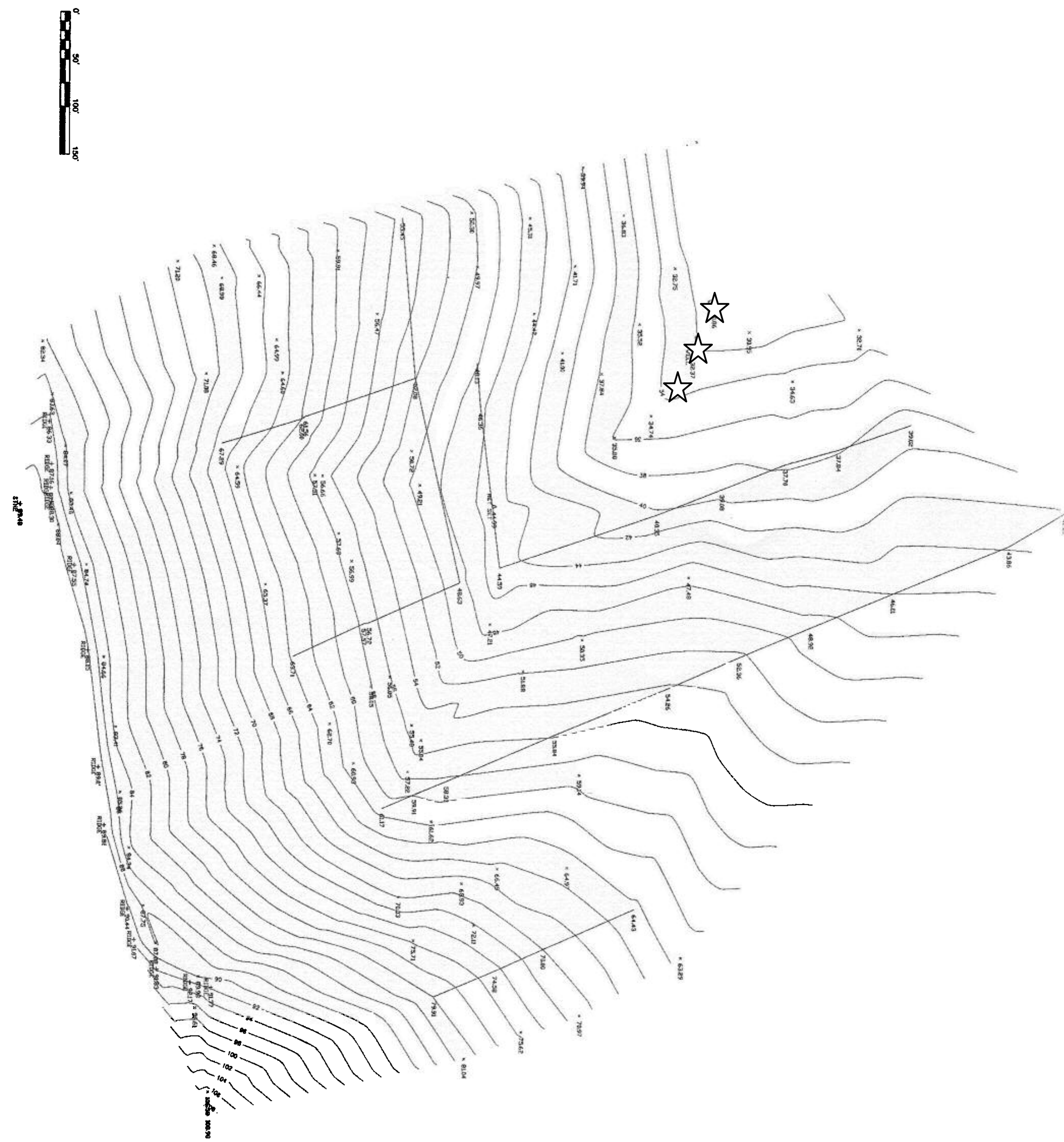

Figure C-4: Topographic map of Grazed 1B watersed on Maryland dairy farm. Bar in upper left corner is 150’; stars denote piezometer nests a, b, and c, from upper right to lower left. Elevation changes 2’ with each contour line.

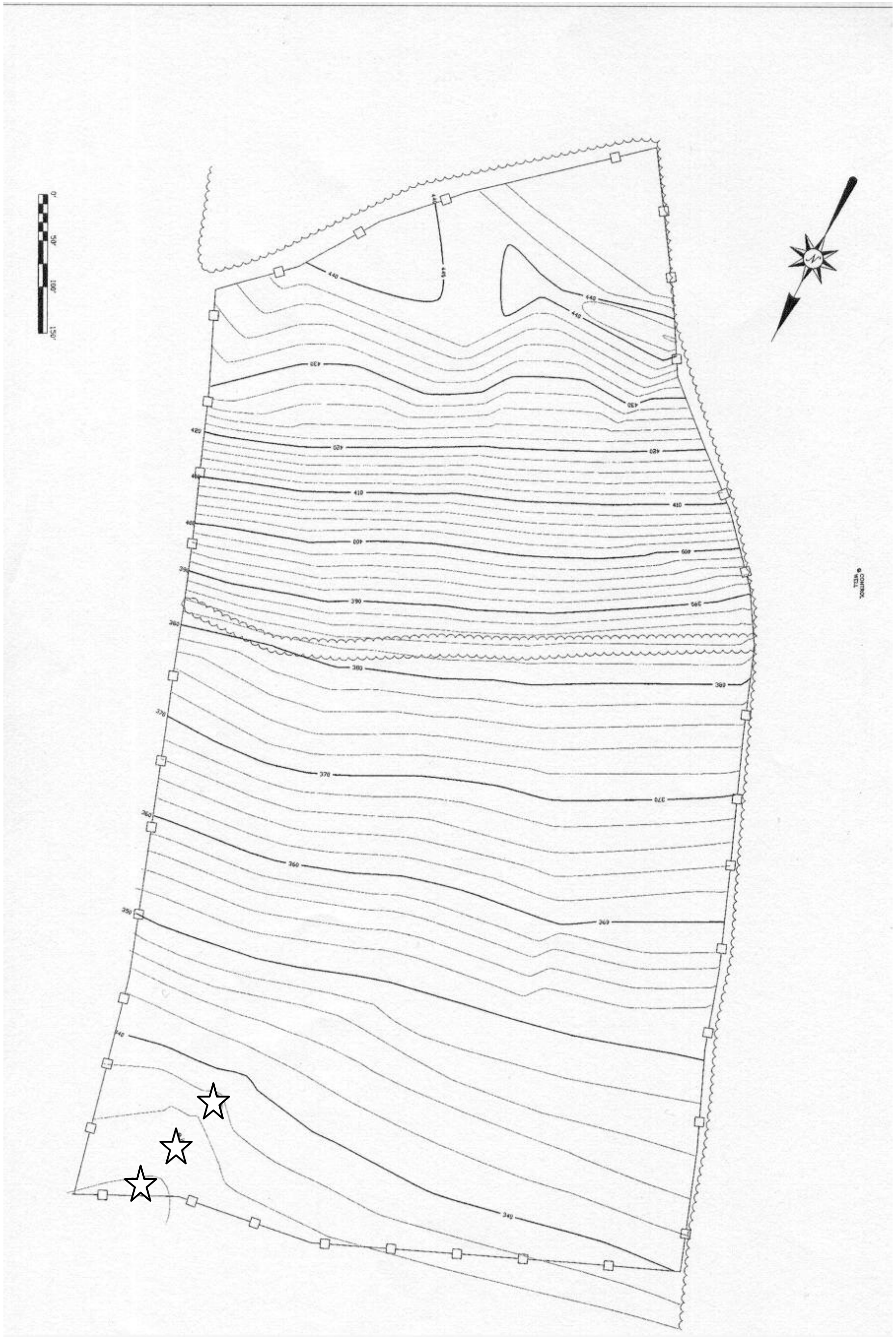

Figure C-5: Topographic map of Grazed 2A, a watershed on a Maryland dairy farm. Transect of nested piezometers is marked with stars, with nests a, b, and c situated 18 m apart from bottom to top along the flow line from the discharge point. Bar is 150’ long. Scalloped lines indicate tree line and hedgerow. Elevation changes 2’ with each contour line.

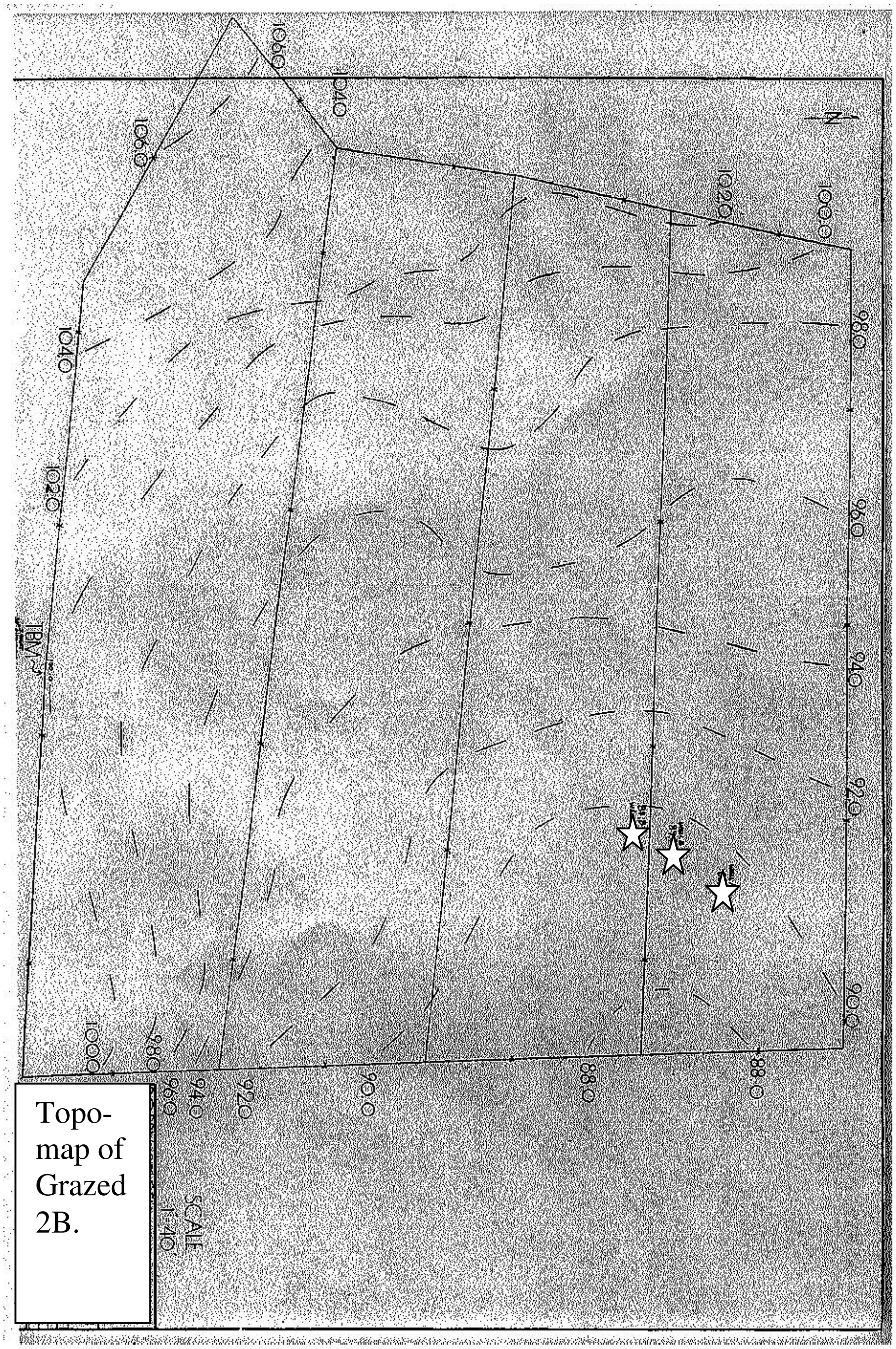

Figure C-6: Topomap of Grazed 2B. Stars mark nests a, b, and c, from upper left to lower right. Original scale is 1":40'. Elevation changes 2' with each contour line.

Appendix D: Microwave digestion methods

Total dissolved N (TDN) was determined by a modified alkaline persulfate microwave digestion which oxidized all nitrogen in the sample to nitrate (Cabrera and Beare, 1993; Hosomi and Sudo, 1986; Littau and Englehart, 1990; Johnes and Heathwaite, 1992). This procedure simultaneously oxidized total dissolved phosphorus to orthophosphate for the determination of dissolved organic phosphorus.

Samples were digested using an alkaline persulfate reagent (45g K_2SO_8 and 9.5 g NaOH per 1L) heated in a microwave in 120 mL high-pressure Teflon vessels (CEM Corporation; Matthews, NC). Ten mL of sample was put in each vessel, along with 10 mL of reagent. The vessels were closed using a mechanical capping station to achieve the torque necessary to maintain pressure. Sets of twelve samples were digested at a time. They were put on a rotating tray in the center of the microwave oven and digested at full power (actual output 675 W) for 750 seconds (12 ½ min).

The appropriate length of time and amount of reagent to use for digestion was determined using several known organic standards (nicotinamide adenine dinucleotide disodium salt, glutamine, urea, glycine) as well as several stream and groundwater samples. The point at which maximum N recovery was achieved without compromising the pressure of the vessels was determined to be 12 minutes and 30 seconds. Recovery of total N averaged 70-95%. Recovery was slightly higher with 13 min digestion time, but that length of time occasionally resulted in the loss of pressure and the release of an unknown amount of sample and reagent. Reducing the digestion time to 750 sec eliminated this sample loss.

Because the digestion process acidified the digestate, 1.0 mL of 0.3 *M* NaOH in 10% NaAc was added to each 10.0 mL of digestate immediately prior to nitrate measurement to bring the digestate to the pH range (5-9) optimal for measurement with the Technicon Autoanalyzer II flow injection analyzer (Technicon Industrial Systems; Tarrytown, NY). The buffer did not react colorimetrically with the cadmium column in nitrate determination.

An additional set of samples from the same sampling dates were analyzed by the Wye Research and Education Center Water Quality Lab. Those samples were collected and acidified in a manner identical to that used for the digested samples. They were then analyzed for total dissolved nitrogen (TDN) using an autoclave digestion with an acidic potassium persulfate reagent (Valderama, 1981). The results from the off-site lab analyses were used to compare with and validate the alkaline potassium persulfate microwave digestion results done in the University of Maryland Soil Quality Lab. In case of discrepancy, the TDN concentration as measured by the microwave digestion at the Soil Quality Lab was considered to be the amount present. The organic N concentration was calculated as the difference between the TDN and the dissolved inorganic N (DIN). The ratios of dissolved organic carbon (DOC) to TDN and DOC to DON were then compared to the calculated concentrations of organic N.

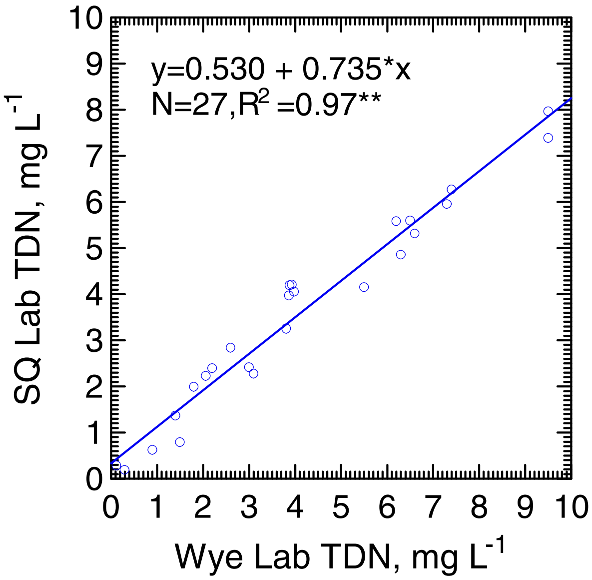

Figure D-1: Comparison of digestion results for total nitrogen analyses completed at University of Maryland Soil Quality Lab (SQ) and Wye Research and Education Center Water Quality Lab (Wye). Comparison was run on samples collected on the same dates. Samples included shallow groundwater samples from three Maryland dairy farms and surface water running through two grazed watersheds on one of the farms.

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