Assessing the Effect of Management Intensive Grazing on Water Quality in the Northeast U.S.

W.L. Stout, S.L. Fales, L.D. Muller, R.R. Schnabel, G.F. Elwinger, and S.R. Weaver

ABSTRACT: Stocking rate is the key determining effect of management intensive grazing on dairy farm profitability. However, increased stocking rate can increase NO\textsubscript{3}\textsuperscript{-} leaching from pastures. Increasing stocking rate increases NO\textsubscript{3}\textsuperscript{-} loss through leaching because the bulk of the N consumed by the animal is excreted in concentrated areas of the pasture mainly in urine. We used experimental data from the northeast U.S. and the literature to assess the relationships between stocking rate and NO\textsubscript{3}\textsuperscript{-} leaching losses beneath an intensively grazed pasture. A relatively low cumulative seasonal stocking rate of about 200 mature Holstein ha\textsuperscript{-1} could result in a 10 mg l\textsuperscript{-1} NO\textsubscript{3}-N concentration in the leachate beneath a fertilized, intensively grazed pasture. This means that while management intensive grazing can improve farm profitability and help control erosion, it can have a significant negative effect on water quality beneath pastures. The extent to which this effect occurs within specific watersheds needs to be evaluated in context of the other cropping systems and lands uses within the watershed.

Keywords: Nitrate, water quality

To maintain dairy farm profitability in the face of rising fuel and machinery costs and decreasing federal subsidies, some dairy farmers in the northeast U.S. are including management intensive grazing (MIG) as a component of the production system on their farms (Fales et al. 1993). Management intensive grazing is a production system in which animals are rotated rapidly through a series of paddocks in order to maximize livestock production on either a per hectare or per animal basis. Economic surveys in Pennsylvania (Parker et al. 1992) and New York (Emmick and Toomer 1991) have shown that inclusion of MIG in a dairy farm can increase net profitability by $121 to $150 per cow\textsuperscript{-1} per year. The key management variable in determining effect of grazing on profitability is stocking rate (Leaver 1985). In a Pennsylvania study, increasing stocking rate from 2.5 to 4.0 cows ha\textsuperscript{-1} (ac) for a 180 day grazing season increased returns over cost by $1188 ha\textsuperscript{-1} (ac) (Fales et al. 1995). However, the 2.5 cows ha\textsuperscript{-1} (ac) stocking rate was most profitable on a per animal basis, increasing returns over costs by $36 per animal\textsuperscript{-1}.

In such temperate maritime regions of the world as the UK, the Netherlands, and New Zealand, increased stocking rate has been shown to be the main factor increasing N leaching losses from pasture (Ball and Ryden 1984; Cuttle and Schofield 1994; Steenvoordeen et al. 1986). Increasing stocking rate affects NO\textsubscript{3}\textsuperscript{-} loss through leaching because the bulk of the N consumed by the animal is excreted unevenly over the pasture in urine (Ball and Ryden 1984; Jarvis et al. 1989). The uneven deposition of urine N in pastures has caused NO\textsubscript{3}\textsuperscript{-} leaching from pastures to be higher than that from similarly fertilized cut grasslands; and it has been shown to pose a threat to water quality in many parts of the world (Ball et al. 1979; Ryden et al. 1984; Steenvoordeen et al. 1986; Stout et al. 1997).

Nitrate leaching from pastures is characterized by high losses from urine patches superimposed over lower losses from the pasture as a whole (Cuttle and Schofield 1994). Losses from urine patches are affected by animal size, animal type, and forage quality (Whitehead 1995), while N loss from the pasture as a whole can be affected by soil hydrologic properties, fertilization rate, and pasture species composition (Cuttle and Schofield 1994). In the northeast U.S., about 25% of the N excreted in cattle urine can leach below the root zone (Stout et al. 1997). The purpose of this paper is to assess the potential effect of cattle urine on water quality as affected by intensive pasture management in the northeast U.S.

Methods

The data for this assessment were developed from two NO\textsubscript{3} leaching experiments, a grazing trial in central Pennsylvania, and the literature. The leaching experiments and the grazing study were conducted at the Pennsylvania State University Dairy Research Center (40°48'N, 77°53'W, 350 m elevation) on a Hagerstown silt loam (fine, mixed, mesic, Typic Hapludalf). The Hagerstown is a deep well-drained soil derived from limestone residuum. The soil is moderately permeable due to a high degree of blocky structure in the subsoil (USDA 1981). A detailed description of the hydrologic properties of the soil is presented in a previous paper (Stout et al. 1998). The weather conditions for the experimental period (April 1993 to March 1996) are summarized in Table 1 and also presented in detail in Stout et al. (1998).

The leaching studies involved measuring NO\textsubscript{3}-N losses directly beneath urine and fecal patches using drainage lysimeters (Stout et al. 1997; Stout et al. 1998) and beneath different pasture swards (Stout et al. 1996). Urine was applied to the lysimeters in the spring, summer, and fall. The feces were applied only in the summer.

<table>
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<th>Table 1. Seasonal weather summary for April 1993 to March 1996 for the Study Site.</th>
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<td>Average Air Temperature, °C</td>
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<td>Spring (Apr. - Jun.)</td>
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<td>Summer (Jul. - Sept.)</td>
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The grazing trial was a replicated farmlet experiment conducted for two years at The Pennsylvania State University (Fales et al. 1995). In this study, mature Holsteins weighing about 590 kg and producing about 25 kg milk day⁻¹ were pastured for a 25 week grazing season. This study provided data on pasture utilization and grain supplementation for the different levels of pasture utilization that are presented in the assessment.

For the purpose of this paper, we developed high and low input grazing scenarios with three stocking rates in each scenario. The high input scenario is orchardgrass fertilized with 246 kg ha⁻¹ N in split applications, and having a total herbage production of about 11000 kg (lb) dry matter ha⁻¹. Within each scenario, the three stocking rates were those that utilized 60% of the available forage, 80% of the available forage, and 80% of the available forage with grain supplementation equal to 33% of the dry matter intake of a lactating 512 kg (lb) dairy cow. For the purpose of the assessment, we assumed this dry matter intake to be 3% of the body weight.

A schematic of the method used in developing relationships between stocking rate and NO₃⁻ N leaching loss and NO₃⁻ concentration in the leachate for these scenarios is shown in Figure 1.

Component A (Figure 1) represents the background NO₃⁻ N leaching loss from pasture not impacted by excreta. Both this total loss and concentration data for this component were measured in the field using large drainage lysimeters (Stout et al. 1997). Lysimeters containing only orchardgrass were fertilized with N so as to simulate the pasture fertilization of an ongoing grazing study adjacent to the lysimeter site. Depending on the growing conditions within each year, (i.e., the number of grazing cycles) the amount of N applied ranged from 196 to 280 kg ha⁻¹ (lb ac) applied in four to six applications per year (Stout et al. 1997). Lysimeters containing both white clover and orchardgrass received no fertilizer N. Herbage was harvested from all of the lysimeters on a schedule that matched the grazing trial. Harvested samples were dried to determine sward yield, weighed, and analyzed for N. Plant N uptake was calculated from these data.

Component B represents the NO₃⁻ N leached from the daily urine excretions of a single animal. The data used in this component was collected from the previously described lysimeters treated with urine (Stout et al. 1997). The amount of N leached from these daily excretions is a function of the number of excretion events per animal per day and the amount and composition of the individual excretions. A dairy cow will urinate between 8 to 12 times per day (Whitehead 1995). For this study, we chose 12 urinations per day since the amount of urine produced by cattle from fresh forage, such as pasture, is about twice the volume of that produced from dry hay (Shechtner et al. 1980). The number of urinations per day was then adjusted for the amount of time that the animals would not be on pasture. About 10 to 15% of the excreta is deposited in gateways, alleyways, and milking sheds (Steele and Vallis 1987). Thus, the base level of 12 urinations a day⁻¹ was decreased to the equivalent of 10.5 urinations a day⁻¹.

The range of volume per urination is 1.5 to 3.5 L per urination, with the small volumes typical of smaller breeds such as Jerseys and the large volumes typical of larger breeds such as Holsteins (Whitehead 1995). In this study, we used a urine volume of 3 L since the majority of the dairy cattle in the U.S. northeast are Holsteins and 3 L was the amount used in the previously reported leaching studies (Stout et al. 1997).

The composition of urine can vary widely due to forage quality and season (Whitehead 1995). In this assessment, we used N concentrations from urine collected during the spring, summer, and fall from grazing dairy cattle (Stout et al. 1997). Since urine composition was measured in the spring, summer, and fall, the composition of the urine used in the final calculations was a weighted mean based on the number of spring (55), summer (92), and fall (42) days in the 189 day grazing season.

Finally, only the NO₃⁻ from urine is used for the projections in this study. This simplification is justified by the fact that the contribution of feces to leachate N over the three years of the study was less than 0.2% of the amount contributed by urine (Stout et al. 1997). Since annual NO₃⁻ N deposition and leaching loss is much less in feces there would be a tendency for N in feces to be more efficiently recycled by plant uptake and redistributed throughout the pasture than N in urine. Over time this would tend to increase the leaching contribution of component B.

Component C represents the cumulative seasonal stocking rate. The points for each utilization level within each scenario (Figures 2 and 3) were determined by dividing the total herbage on offer in the pasture by the daily dry matter intake of that forage by the grazing animal. The amount of herbage on offer assumed in this study was that previously reported for N fertilized orchardgrass and an orchardgrass-white clover mixture (Stout et al. 1997). Daily dry matter intake was assumed to be 18 kg (lb) per day⁻¹ per animal⁻¹, about 3% of the body weight of a 590 kg Holstein cow. This intake estimation is based on data from a previous grazing study (Fales et al. 1995).

Component D represents a correction for multiple urine depositions with increased stocking rates (Peterson et al. 1956; Richards and Woolton 1976) on the same site in the pasture. This component adds the multiple deposition of urine N not used by the plant to the N amount leached. The number of these multiple depositions was shown to follow a negative binomial distribution (Peterson et al. 1956; Richards and Woolton 1976). Using this distribution, we increased NO₃⁻ N leaching under these multiple deposition areas as follows.

First, we assumed that biomass yield and N uptake were maximized by the N applied through fertilization or by biological fixation and the initial urine event. Also, the amount lost per event due to denitrification, immobilization, and volatilization remain constant for all urine depositions on a particular site. Next, plant N uptake due to urine deposition was calculated as the difference between plant N uptake of the control and urine treatment lysimeters (Stout et al. 1997). Thus, the N leached from multiple depositions is equal to the amount of N leached from the first urine deposition plus the sum of N from each subsequent urine deposition and the N uptake from the initial urine deposition. This accounts for the non linearity of the curves in Figures 2 and 3.

In making the assessment, it was necessary to make two assumptions. First, the NO₃⁻ N leached due to application of fertilizer application or biological fixation and excreta is additive. This means that for any given area the total amount of NO₃⁻ N leached is equal to the sum of the blanket fertilizer N application rate plus biological N fixation plus whatever excreta N was applied to the area. Also, we discounted the potential reduction in biologically fixed N beneath urine and...
fecal spots, since this would happen in only a small area of the pasture for only a portion of the growing season (i.e., excreta deposited at the end of the grazing season would have no effect on biological fixation for most of the growing season). Second, for the area covered by urine spots, the total amount of N removed by plants in a grazing season is a weighted average of the yearly amounts removed by plants from urine and treatments. The weights are the products of the number of grazing days per meteorological season and the dry matter weights of harvested biomass for the corresponding urine treatment. This implies the assumption of uniform stocking density grazing season being achieved by feeding excess forage harvested from paddocks during the flush of growth in the spring. In projecting the leaching impact of an
individual urine event to a hectare basis, we used the values of 0.28 m² for urine impact area (Peterson et al. 1956; Garwood and Ryden 1986). Thus, the amount of N [kg ha⁻¹ (lb ac)] in the leachate collected beneath urine (Stout et al. 1997) is attributed to a 0.28 m² area of pasture. The amount of N leached in a grazing season from the area covered by single urine spots is a weighted average of the amounts leached from the three urine treatments, where the weights are the numbers of grazing days per meteorological season.

Values for NO₃-N concentrations in groundwater (mg L⁻¹) are derived by back calculating from the aerial quantities of leachate NO₃-N (i.e., kg ha⁻¹ (lb ac)) and measured leachate volume (Stout et al. 1997). The leachate volume used for this conversion is equal to: 1.) the volume of leachate collected from the fertilizer-only treatment (for the area not impacted by urine or feces), 2.) a weighted average of the volumes of leachate collected from the three urine treatments (for the area covered by urine), where the weights are the numbers of grazing days per meteorological season.

Results

The relationships between stocking rate and the yearly average NO₃ leachate loss under N fertilized orchardgrass and an orchardgrass-white clover mixture are shown in Figure 2. The Y intercept of the curve (no animals present) represents the NO₃-N leaching loss attributable to the application of commercial fertilizer in the case of the N fertilized orchardgrass or biological N fixation in the case of the orchardgrass-white clover mixture. The NO₃-N leachate loss from the N fertilized orchardgrass was about twice that of the orchardgrass-white clover mixture, but the yield of the N fertilized orchardgrass [11,000 kg ha⁻¹ (lb ac)] was only 47% greater than the yield of the orchardgrass-white clover mixture [7500 kg ha⁻¹ (lb ac)]. This indicates that even with no animals present the NO₃-N leaching will be higher under the N fertilized grass pastures orchardgrass than under pastures where N is provided by biological fixation a legume such as white clover. Also, the higher yield of the N fertilized orchardgrass would provide for a higher stocking seasonal stocking rate, further increasing NO₃-N leaching in relation to an orchardgrass-white clover pasture.

European studies indicate that N loss from grass pastures fertilized with 250 to 400 kg N ha⁻¹ (lb N ac) were 30 to 60 kg ha⁻¹ higher that those from ungrazed grasslands (Whitehead 1995). At 80% utilization of the on-offer herbage, our estimations for NO₃-N leaching losses were about 55 kg ha⁻¹ for orchardgrass fertilized with about 246 kg N ha⁻¹ (Figure 2). The resulting difference between the NO₃-N leaching loss at 80% utilization and the Y intercept (i.e., no grazing) was about 40 kg ha⁻¹. Since the three year mean N application in this study was 246 kg ha⁻¹, the increase in NO₃-N leaching loss that we attributed to grazing is very similar to those reported from the European studies.

The highest projected NO₃-N leaching loss from N fertilized orchardgrass was about 80 kg ha⁻¹ at 80% herbage utilization plus grain supplementation (Figure 2). This equates to a cumulative stocking rate of about 775 animal days ha⁻¹. For a 180 day grazing season, this daily stocking rate would be 4.3 animals ha⁻¹. This is roughly equivalent to the stocking rate of 4.0 animals ha⁻¹, the stocking rate reported to be the most profitable on a per hectare basis (Fales et al. 1995).

Nitrate-N leaching loss from cut and grazed ryegrass-white clover swards in the UK was 2.5 and 23 kg ha⁻¹, respectively (Garwood and Ryden 1986). In our study, estimated NO₃-N leaching losses from orchardgrass-white clover was 8 kg ha⁻¹ at the Y intercept (no grazing) and about 34 kg ha⁻¹ at 80% utilization, respectively (Figure 2). While our leaching losses were somewhat higher than those in the UK, the difference in leaching losses between cut and grazed swards in the UK (20 kg ha⁻¹) and our study (26 kg ha⁻¹) were similar.

The effect of stocking rate on NO₃-N concentration in leachate is shown in Figure 3. Regardless of the source of N (fertilizer or biological fixation), the weighted average NO₃-N concentration in leachate would be estimated to exceed the 10 mg L⁻¹ U.S. EPA drinking water standard (U.S. EPA 1987) at a relatively low cumulative seasonal stocking rate. This stocking rate was about 200 ad ha⁻¹ for N fertilized orchardgrass and about 280 ad ha⁻¹ for the orchardgrass-white clover mixture. The cause of this difference was less background NO₃ leaching (Component A, Figure 1) from the orchardgrass-white clover pasture.
The effect of stocking rates (i.e., the slope of the line on Figure 2) on NO$_3$-N concentration in the leachate was similar for both sward types. The factor controlling the maximum NO$_3$-N concentration in the leachate was pasture productivity and the subsequent utilization by the grazing animal. At comparable levels of utilization, the NO$_3$-N leachate concentration was higher under the N fertilized orchardgrass sward than under the orchardgrass-white clover sward. However, when the orchardgrass-white clover scenario was pushed to its maximum utilization (80% herbage utilization + grain supplementation), the NO$_3$-N concentration in the leachate would approach those under the N fertilized orchardgrass scenario at 80% herbage utilization (Figure 3).

For N fertilized grass swards, the most profitable cumulative stocking rate on the per hectare basis, according to the grazing study (Fales et al. 1995), was 775 ad ha$^{-1}$. At this stocking rate, the estimated NO$_3$-N concentration in the leachate would be about 30 mg l$^{-1}$ (Figure 2). However, if one considers the most profitable cumulative stocking rate on the per animal basis, 385 ad ha$^{-1}$ (Fales et al. 1995), the estimated NO$_3$-N concentration in the leachate would be about 15 mg l$^{-1}$. Averaged over both sward types and all utilization levels, the estimated NO$_3$-N concentration in leachate beneath intensively grazed pasture using our data and procedures for calculation would be about 19 mg l$^{-1}$. In a study with corn (Zea Mays L) in the same valley on the same soil type, Jemison and Fox (1994) reported yearly NO$_3$-N concentrations of 11.9 and 19.8 mg l$^{-1}$ in the leachate beneath corn fertilized 100 and 200 kg N ha$^{-1}$, respectively.

**Discussion**

The assessment of the effect of intensive grazing on water quality, as presented in this paper, closely relates to specific climatic and soil conditions and type of animal, and certainly does not constitute a predictive model. It does however indicate that NO$_3$-N amounts and concentrations in leachate beneath MIG systems in the northeast U.S. can be expected to be similar to those found under MIG systems in other parts of the world and to those found under cropping systems in the northeast U.S. Also, NO$_3$-N concentrations in the leachate can be expected to exceed U.S. EPA drinking water standards. This indicates that leachate from MIG systems should not be relied upon to mitigate high NO$_3$-N concentrations in leachate from other areas of a farm or watershed. All of this means that MIG systems need to be evaluated before the means of our calculations. There is, of course, a certain amount of variability associated with these lines. However, the only source of variability that we could include in the response lines would be that associated with the leaching data collected on our project. There are many other sources of variability such as: soil drainage, grass species, legume species, milk production levels in the animals, lactation stage of the animals, and size of the animal. If this type of assessment were to be included in water quality or nutrient management models, the sensitivity of such models to the aforementioned sources of variation, as well as others, would have to be evaluated before the models would be valid.

**Conclusion**

A relatively low cumulative seasonal stocking rate of as little as 200 ad [mature Holstein dairy cows (590 kg ha$^{-1}$)] would result in a 10 mg l$^{-1}$ NO$_3$-N concentration in the leachate beneath an intensively grazed pasture. This means that while management intensive grazing can improve the profitability of small and medium sized dairy farms and can provide erosion control, it can have a significant negative impact on water quality beneath pastures. The extent to which this impact can affect water quality within a specific watershed needs to be evaluated in context of the other cropping systems and lands uses within the watershed, and management techniques need to be developed to address the specific nutrient problems associated with MIG.

**REFERENCES CITED**


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