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Quantifying the Impact of Regular Cutting on Vegetative Buffer Efficacy for Nitrogen-15 Sequestration

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ABSTRACT

This study used the stable ^{15}N isotope to quantitatively examine the effects of cutting on vegetative buffer uptake of NO_3^-–N based on the theory that regular cutting would increase N demand and sequestration by encouraging new plant growth. During the summer of 2002, 10 buffer plots were established within a flood-irrigated pasture. In 2003, ^{15}N-labeled KNO_3 was applied to the pasture area at a rate of 5 kg N ha^{-1} and 99.7 atom % ^{15}N. One-half of the buffer plots were trimmed monthly. In the buffers, the cutting effect was not significant in the first few weeks following ^{15}N application, with both the cut and uncut buffers sequestering ^{15}N. Over the irrigation season, however, cut buffers sequestered 2.3 times the ^{15}N of uncut buffers, corresponding to an increase in aboveground biomass following cutting. Cutting and removing vegetation allowed the standing biomass to take advantage of soil ^{15}N as it was released by microbial mineralization. In contrast, the uncut buffers showed very little change in ^{15}N sequestration or biomass, suggesting senescence and a corresponding decrease in N demand. Overall, cutting significantly improved ^{15}N attenuation from both surface and subsurface water. However, the effect was temporally related, and only became significant 21 to 42 d after ^{15}N application. The dominant influence on runoff water quality from irrigated pasture remains irrigation rate, as reducing the rate by 75% relative to the typical rate resulted in a 50% decrease in total runoff losses and a sevenfold decrease in ^{15}N concentration.

In California, irrigated pasture provides a relatively low-cost source of green forage during the summer months when surrounding rangelands are dry and dormant. Irrigation rates vary by irrigation method, but for flood irrigation, rates are as high as 70 L s^{-1} at the top of the slope, applied continuously over an 8- to 14-h period (up to 12 cm). In the Sierra Nevada foothills, with slopes from 5 to 30%, this can generate runoff losses of up to 70% of the applied water (Tate et al., 2000b). Given that irrigated pasture is both fertilized and grazed, there is concern that runoff water contains toxic levels of pathogens and nutrients.

Nitrate N is a soluble nutrient commonly cited as a source of ground- and surface-water contamination. In the United States, the legal drinking water limit for NO_3^-–N is 10 mg L^{-1}, but concentrations as low as 1 mg L^{-1} can contribute to algal blooms (Mendez et al., 1999). Nitrate has been implicated in eutrophication in seawater and fresh water (Cole et al., 2004). Measured concentrations in runoff from irrigated pasture range from 0.2 to 5 mg L^{-1} (Bedard-Haughn, unpublished data, 2002).

Buffer strips are broadly defined as strips of vegetation that improve or maintain water quality downslope of an agriculture or forestry operation (Barling and Moore, 1994). Buffers function to remove pollutants by reducing or filtering surface runoff and/or by filtering ground water and stream water (Dosskey, 2001). Attenuation of NO_3^- by buffers is attributed to a combination of factors, including denitrification, infiltration, and plant uptake (Hill, 1996). The relative importance of each factor varies according to buffer characteristics such as hydrology, vegetation type (grass vs. forest), soil type (coarse vs. fine), buffer width, and pollutant type (Bharati et al., 2002; Schmitt et al., 1999). In irrigated pasture, infiltration and plant uptake appear to have a greater impact on NO_3^- attenuation than denitrification (Verchot et al., 1997). A recent field study in California using ^{15}N-enriched NO_3^- tracers in an irrigated pasture system found that up to 50% of applied ^{15}N was removed by plant uptake the first 10 d following application, making uptake the dominant mechanism for N attenuation (Bedard-Haughn et al., 2004). However, minimal uptake occurred over the remainder of the growing season, even with available N in the soil. Consequently, ^{15}N continued to be lost throughout the irrigation season via runoff, despite the presence of vegetative buffers. In examining grass buffer trapping efficiency for sediment and nutrients, Dillaha et al. (1989) reported higher levels of soluble nutrients leaving buffers than entering them, which they attributed to low trapping efficiency for soluble nutrients and to release of nutrients previously stored in the buffer. This contributes to concern that buffer efficiency may decrease over time, and that buffers will ultimately become a source of N (and other nutrients) rather than a sink (Mendez et al., 1999).

Plant N demand and uptake can be key factors in controlling N losses in many ecosystems (Mulholland et al., 2000). Demand and uptake vary with the N status of the vegetation, NO_3^- availability, plant growth rate, and plant age or phenology. All other factors remaining constant, plant N uptake during growth will be greater if the vegetation is N deficient or if there is an abundance of available N. Maximum N uptake occurs during the vegetative growth phase when roots are actively growing and soil moisture is high (Jackson et al., 1988); increasing plant age tends to decrease N uptake (Schenk, 1996). As Jackson et al. (1988) observed in the Sierra Nevada foothills, even well-watered grasses can senesce within weeks of anthesis, decreasing N demand. When new sources of N are introduced, microbial immobilization

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Abbreviations: DON, dissolved organic nitrogen; LME, linear mixed effects; SFREC, Sierra Foothill Research and Extension Center.

may compete effectively with plants for N (Jackson et al., 1989). Subsequent turnover and mineralization releases this previously immobilized N. In annual grasslands in the Sierra Nevada foothills, turnover of the microbial N pool can occur rapidly (less than one day) and continuously (Davidson et al., 1990; Jackson et al., 1989), necessitating continual plant demand for N to minimize nutrient losses.

It may be possible to increase plant N uptake via regular cutting, which would increase N demand by encouraging compensatory regrowth. Within two weeks after shoot harvest, uptake of N increases (Ourry et al., 1990). Matheson et al. (2002) found that although regular cutting of vegetation decreased new shoot production, it increased the NO_3^- assimilation capacity of shoots by a factor of 5 compared with shoots that were not cut, suggesting that even when total plant biomass is reduced by cutting, the positive effects on N sequestration might offset this.

The role of plant uptake in attenuating nutrients is diminished when nutrients are returned to the soil via decomposition, therefore periodic harvesting of buffer vegetation might improve the long-term effectiveness of buffers (Dosskey, 2001). Mowing alone will increase plant N uptake, but removal of the cut vegetation is required to prevent nutrient release via decomposition (Barling and Moore, 1994). Although grazing also removes vegetation, up to 60 to 90% of the ingested N can be returned to the pasture system, mostly as urine (Di and Cameron, 2002).

Applying ^{15}N-enriched techniques in the field provides a powerful insight into plant–soil N dynamics (Powlson and Barraclough, 1993), commonly within a single growing season (Bardgett et al., 2003; Di et al., 1999; Jackson et al., 1989; Mulholland et al., 2000). For this study, ^{15}N-enriched isotopes allowed new NO_3^- to be distinguished from NO_3^- already present in the system and to be quantitatively traced through the buffers (Bedard-Haughn et al., 2003).

Given the previously observed abatement in plant N uptake in mature buffers in irrigated pasture (Bedard-Haughn et al., 2004) and the potential for increasing plant N uptake via vegetation management, this study was designed to: (i) quantitatively determine whether regular cutting and removal of vegetation in buffer strips would increase plant ^{15}N uptake and retention, (ii) measure the impact of regular cutting on attenuation of runoff ^{15}N, and (iii) determine whether there was a corresponding impact on attenuation of ^{15}N in the soil solution and on ^{15}N sequestration in the soil. In addition, we considered whether decreasing irrigation rate affects buffer efficiency by comparison with results on an adjacent site. By examining the water quality measurements in conjunction with the soil and vegetation results, a complete ^{15}N recovery budget was developed, providing insight into the relative importance of the different N sinks and pools in the function of vegetative buffers in irrigated pasture.

MATERIALS AND METHODS

Site Description

The University of California Sierra Foothill Research and Extension Center (SFREC), located 100 km northeast of Sacramento, CA, has a xeric climate and hilly terrain. During the summer of 2002, 10 adjacent plots were established within an existing flood-irrigated pasture at SFREC. Each plot consisted of a 5-m-wide by 16-m-long (80 m^2) buffer area immediately downslope of a 25-m^2 pasture area (Fig. 1). The pasture-buffer areas were dominated by orchard grass (*Dactylis glomerata* L.), velvet grass (*Holcus lanatus* L.; also known as Yorkshire fog), and dallis grass (*Paspalum dilatatum* Poir.). Soils (Table 1) were fairly uniform throughout the site and were classified as fine-loamy, mixed, thermic, Mollic Haploxeralfs of the Auburn–Las Posas–Argonaut rocky loam association (Herbert and Begg, 1969) and site slope ranged from 15 to 18%. The entire study area was fenced to prevent disturbance by the cattle grazing the surrounding pasture.

Cutting and Irrigation

Beginning in June 2003, a cutting treatment was randomly allocated to 5 of the 10 buffer areas. Adjacent pasture-buffer areas were separated by landscape edging, which effectively prevented runoff crossover between buffers. Preferential flow along the edging was minimized by typical irrigation management techniques. For the duration of the 2003 irrigation season (June–October), vegetation in the five cut buffers was trimmed monthly using nylon-line trimmers to levels corresponding

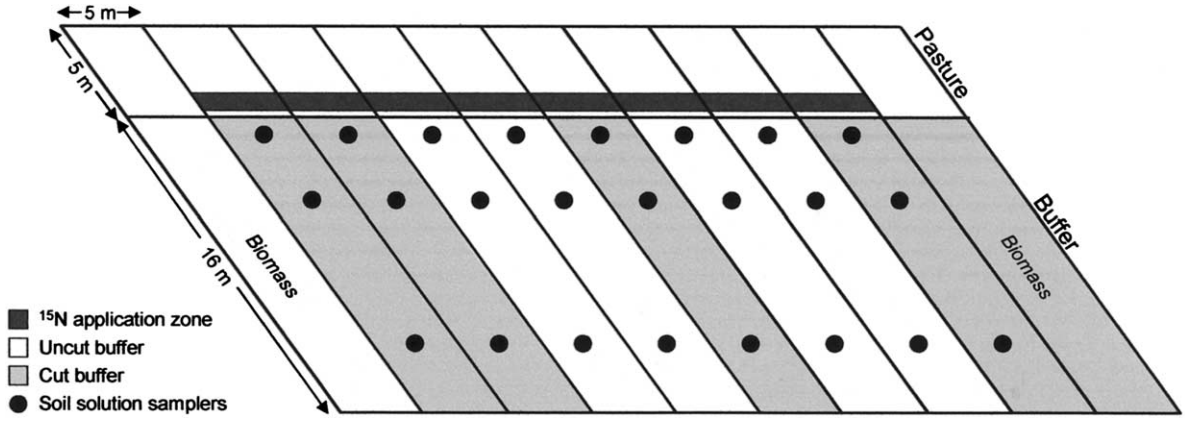

Fig. 1. Schematic of pasture-buffer plot layout. Not to scale. "Biomass" buffers received no ^{15}N and were used to get quantitative estimates of aboveground biomass. Soil samples were taken at the same downslope distances as the soil solution samples.

to post-grazing height (5–10 cm) in the surrounding pasture. Cutting frequency was based on observed recovery time required between grazing events in the surrounding pasture. All 10 pasture areas were trimmed at the same intervals as the cut buffers. Cut residues were collected and removed from the site. Uncut buffers were not trimmed.

Plot irrigation was by gated pipe, which delivered water separately to each pasture-buffer area. Irrigation rate was controlled by a valve and monitored by flow meters (Model WT; Netafim, Tel Aviv, Israel) that allowed measurement of both rate and quantity of water applied. Water was applied 5 m upslope from the buffer–pasture interface to maximize control of water distribution within the study area. During this project, the irrigation rate was calibrated to 1 L s^{-1} per buffer for approximately 3 h every 9 d. This irrigation rate is 75% lower than the rate applied in Bedard-Haughn et al. (2004) study on an adjacent set of plots (Table 1). A lower irrigation rate was used in an effort to reduce runoff losses and improve irrigation efficiency. On average, 29% of the applied irrigation water was lost as runoff. Total duration of each irrigation event varied according to the volume needed to restore soil water content, which was determined using evapotranspiration data from the California Irrigation Management Information System weather station located at SFREC. Climate and plant growth conditions during the 2003 growing season were normal for the region.

Collection troughs installed across the bottom of each buffer collected surface water runoff. Three pairs of ceramic soil solution samplers (Soilmoisture Equipment, Santa Barbara, CA) were installed in each buffer area at 1, 4, and 12 m downslope of the ^{15}N application (Fig. 1). Samplers were installed to depths of 15 and 45 cm, the average depths to the bottom of the A horizon and the top of the heavy clay Bt horizon, respectively.

Nitrogen-15 Application

In June 2003, four days after the first cutting and three days before the first irrigation, ^{15}N-labeled KNO_3 was applied in solution at a rate of 5 kg N ha^{-1} and 99.7 atom % ^{15}N. The rate and atom % ^{15}N concentration were selected to provide an approximation of post-irrigation fertilizer N levels while allowing the tracer to be detectable in all N pools throughout the duration of the experiment. The ^{15}N solution was applied across 8 of the 10 plots (4 cut, 4 uncut). The area labeled was 1 m wide across the width of each plot and located 0.75 m above the buffer areas (Fig. 1). Application rate and area were based on Bedard-Haughn et al. (2004). Following application, the ^{15}N fertilizer was watered in with 18 L of water per m^2; under field conditions, this volume was sufficient to rinse the ^{15}N solution off of the foliar surfaces but allowed only minimal percolation. To ensure uniform distribution of both the ^{15}N fertilizer solution and the additional water, the application area was subdivided into m^2 plots. Natural abundance background levels of ^{15}N in all N pools were measured before application of ^{15}N-labeled fertilizer to account for natural variability and dilution of the applied ^{15}N fertilizer by ^{14}N. Within a given plant species, the standard deviation of natural abundance atom % was within ±0.0002 atom %.

Isotopic levels are reported as atom % ^{15}N excess, which refers to the amount of ^{15}N present relative to the average naturally occurring background ^{15}N levels for that particular N pool. Atom % ^{15}N excess amounts were extrapolated to obtain the total amount of ^{15}N in a given pool by weight and/or volume and thus to determine a ^{15}N budget.

Table 1. Values (mean ± SD) for field site properties averaged across all buffers. Soil properties are average values for the 0- to 15-cm layer across all buffers.

Property	Current study	Bedard-Haughn et al. (2004)
C, %	3.3 ± 0.6	3.0 ± 0.4
N, %	0.3 ± 0.1	0.3 ± 0.04
C to N ratio	10.4 ± 0.6	10.4 ± 0.4
Sand, %†	36.8 ± 4.9	31.7 ± 3.8
Silt, %†	52.5 ± 4.9	50.6 ± 1.6
Clay, %†	10.6 ± 1.0	11.0 ± 0.9
Slope, %	16.9 ± 1.1	10.9 ± 0.8
Runoff losses, %‡	29.2 ± 12.7	56.8 ± 16.4

† Particle size expressed on measured volume % basis for 2003, calculated volume % basis for 2002 (Eshel et al., 2004).
‡ Runoff volume/irrigation volume.

Vegetation Sampling and Analysis

Grab samples of vegetation were collected 3, 11, 21, 42, 60, 79, 98, and 114 d after ^{15}N application. To determine how far the ^{15}N fertilizer had moved into the buffers, vegetation samples were collected along a cross-slope transect within the zone of ^{15}N application and at downslope distances of 1, 4, 8, 12, and 16 m from the application area. The uncut buffer vegetation samples were separated by the three dominant grass species, whereas cut buffer vegetation samples represented composites of all species present due to identification obstacles associated with newly clipped vegetation. All plant samples were oven-dried at 65°C and analyzed for ^{15}N isotopic composition via mass spectrometry (Integra Integrated Stable Isotope Analyzer; Europa Scientific, Crewe, UK) at the University of California, Davis, Stable Isotope Facility (van Kessel et al., 1994). The current sensitivity of our stable isotope ratio mass spectrometers is 0.0002 atom % ^{15}N.

Of the two plots that did not receive ^{15}N, one received the same regular cutting as the cut buffers whereas the other was left to mature the same as the uncut buffers. The species composition, vegetation age, and irrigation rates of these two nonlabeled buffers were equivalent to the labeled buffers. Accurate biomass measurements could not be taken from the labeled buffers without compromising results, so on each sampling day, representative biomass measurements were taken from the two nonlabeled buffers (Fig. 1). All living biomass within a randomly placed 0.1-m^2 quadrat was collected, dried, and weighed. For the cut buffer, three composite quadrat measurements were collected on each day. For the uncut buffer, one representative measurement was taken for each of the three dominant species. Although this lower number contributed to greater variability for uncut biomass values, it allowed for regular sampling over the season without eradicating the less prevalent species. Cover measurements for the uncut buffers were taken on Days 11, 42, and 114 using the line intercept method (Canfield, 1941) to determine the relative dominance of each of the three dominant grass species.

Vegetation N content was multiplied by atom % ^{15}N excess values to get the mass (mg) of ^{15}N in each g of vegetation. The total mass (mg) of ^{15}N sequestered in vegetation in a given buffer area was determined by multiplying the mg ^{15}N g^{-1} vegetation values times biomass values (g m^{-2}) and extrapolating to the whole area using cover data.

Runoff Sampling and Analysis

Runoff samples were collected on the same dates as vegetation samples. Samples were taken from the collection troughs 15 min following the leading edge of runoff and again just before the end of the irrigation event and were stored frozen until analysis. Based on results from an adjacent irrigated site,

these two measurements captured the maximum variability during the irrigation period (Bedard-Haughn et al., 2004). The 15-min interval provided a measurement of maximum ^{15}N concentration, whereas the event-end sample reflects the minimum ^{15}N concentration, but the maximum ^{15}N load. Sample collection (500 mL) was as a "grab" sample from the runoff collection trough. Runoff rates were determined at regular intervals by measuring the volume of runoff in a 5-s period. Runoff rate data were used to determine runoff losses (Table 1).

Runoff ^{15}N isotope analyses were performed on three N pools: NO_3^-, NH_4^+, and total N. Samples were filtered to remove sediment and vegetation residues from runoff. The NH_4^+–^{15}N and NO_3^-–^{15}N were determined by NH_3 diffusion of a 100-mL aliquot onto polytetrafluoroethylene-encased acid traps (Stark and Hart, 1996). To measure NO_3^-–^{15}N, the Stark and Hart (1996) method was modified using $TiCl_3$ (Titanous Chloride Solution, 20%; Fisher, Hampton, NH) to reduce NO_3^- to NH_3 as outlined in Bedard-Haughn et al. (2004). Total ^{15}N was determined on a separate 20-mL aliquot by performing a persulfate digestion (American Public Health Association, 1989) to convert the dissolved organic nitrogen (DON) and NH_4^+ to NO_3^-, and samples were then diffused for NO_3^- as above. Following diffusion, acid disks were removed from polytetrafluoroethylene packets and analyzed via mass spectrometry. The DON–^{15}N for each sample was calculated using an isotope mixing model via difference from total ^{15}N (Shearer and Kohl, 1993):

$$^{15}N_{DON} = \frac{^{15}N_{NT}m_{NT} - {}^{15}N_{NH_4}m_{NH_4} - {}^{15}N_{NO_3}m_{NO_3}}{m_{NT}} \quad [1]$$

where $^{15}N_x$ refers to the atom % ^{15}N value for a given N form (NT = total dissolved ^{15}N) and m_x refers to the quantity of N in μg. For irrigations where samples were not collected (Days 30, 51, 70, 80, and 106), runoff, concentration, and ^{15}N values were estimated by calculating the linear relationship between adjacent sampling dates.

Soil Sampling and Analysis

Soil samples were taken at 0, 1, 4, and 12 m from the ^{15}N application at 3 and 114 d following ^{15}N application. On both dates, samples were taken to a 15-cm depth in two increments (0–7 and 7–15 cm) using a slack hammer (Ben Meadows Company, Janesville, WI), corresponding to the depth of the A horizon. Soil texture was determined on the 114-d samples using laser diffraction and reported in volume percent (Eshel et al., 2004). On Day 114, soil samples were also taken to a 1-m depth in two increments (0–40 and 40–100 cm) to allow for a more complete estimate of the final ^{15}N budget. Soil samples were oven-dried at 40°C and analyzed for total N and ^{15}N via mass spectrometry. Soil C was analyzed by mass spectrometry in conjunction with soil N. Bulk density measurements for all depth increments were done on oven-dried intact cores and were within the range of values measured by Dahlgren et al. (1997) in SFREC grazed pasture.

Soil microbial ^{15}N was measured using fumigation–extraction method (Brookes et al., 1985), with fumigation for 48 h with chloroform vapor and extraction with 0.5 *M* K_2SO_4. Extract ^{15}N was determined by persulfate digestion (American Public Health Association, 1989) to convert the DON and NH_4^+ to NO_3^-, and diffusion using a modification of the Stark and Hart (1996) method, as outlined in Bedard-Haughn et al. (2004). Microbial ^{15}N was determined by difference between fumigated and nonfumigated samples for both dates (0–15 cm only) for the 0- and 1-m distances.

Soil Solution Sampling and Analysis

Immediately before each irrigation event, vacuum was applied to the soil solution sampling tubes and allowed to draw moisture from the soil for 10 d before sample collection (Bedard-Haughn et al., 2004). Although vacuum was not applied continuously over the 10-d period, suction was still present at sampling in most sampling tubes. After Day 42, the time between irrigation and sample collection was shortened from 9 to 3 d, which substantially improved the reliability of the suction in the tubes and the volume of sample collected. Soil solution samples were stored frozen until analysis for NO_3^-–^{15}N via the $TiCl_3$ diffusion (25-mL aliquots) as outlined in Bedard-Haughn et al. (2004).

Nitrogen-15 Recovery Budget

The ^{15}N recovery budget illustrates the mass of ^{15}N sequestered and/or measured in runoff relative to the mass of ^{15}N applied at the beginning of the study. For soil and vegetation samples, atom % ^{15}N excess amounts were extrapolated using total N content, soil bulk density (mass/volume), and vegetation biomass (mass/area) to obtain the total amount of ^{15}N in a given sink by mass and thus to determine the total amount of ^{15}N stored in the pasture-buffer areas. The total amount of ^{15}N lost via runoff (^{15}N load) during a given irrigation event was determined by multiplying runoff volume by ^{15}N concentrations for each measured interval and integrating over time. Summing these values for the measured irrigation events, together with quantitative estimates for intervening irrigation events where runoff was not measured, provided a value for the total amount of ^{15}N lost as runoff from the pasture-buffer areas. There were no total flux measurements for soil solution, so total subsurface ^{15}N load could not be calculated.

Statistical Analysis

The results were analyzed using linear mixed effects model analysis (S-PLUS; Insightful Corporation, 2001). Linear mixed effects analysis can be applied to both structured and observational studies (Pinheiro and Bates, 2000) and was used here to account for the influence of fixed (cutting) effects on buffer ^{15}N uptake levels and for the repeated measures (group effect–plot identity) embedded in the data structure. Treating time as a fixed effect provided a test of how response varied over the duration of the study. The magnitude and direction (±) of the coefficient for treatment and time effects was used to define the relationship between ^{15}N uptake and runoff ^{15}N load and cutting effects. This flexible model also allowed within-group variance and correlation structures for handling within-group (plot) heteroscedasticity and temporally correlated errors (irrigation series within year) (Pinheiro and Bates, 2000). This approach has been used in modeling other complex longitudinal datasets (Atwill et al., 2002; Tate et al., 2000a, 2003). The soil and soil solution data were analyzed using the nonparametric Wilcoxon rank sum test (S-PLUS; Insightful Corporation, 2001), which is not restricted by assumptions of normality.

RESULTS

Aboveground Vegetation

Cut or uncut, there was a general decrease in atom % ^{15}N excess (i.e., % ^{15}N present in excess of background ^{15}N levels) with increasing distance from the ^{15}N application zone. However, there was ^{15}N present in vegetation at the 16-m distance even after a single irrigation event (11 d; Fig. 2). At the first vegetation sampling following

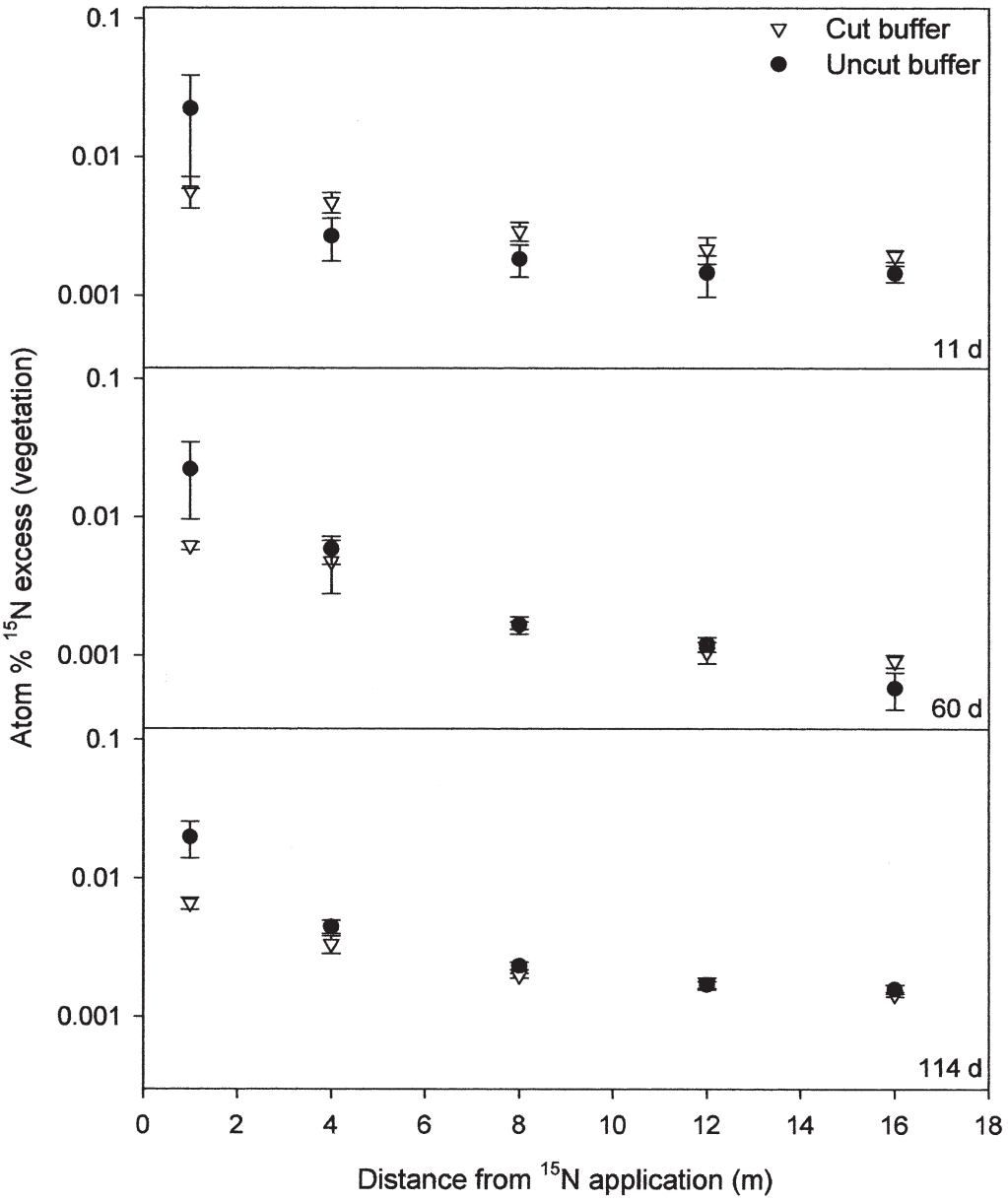

Fig. 2. Atom % ^{15}N excess in buffer vegetation by distance from ^{15}N application (averages, standard error bars). From top to bottom, days after ^{15}N application = 11 d, 60 d, 114 d. Note log *y* axis.

^{15}N application (11 d), vegetation atom % ^{15}N excess was higher for uncut buffers within 1 m of the ^{15}N application zone, whereas further downslope, vegetation atom % ^{15}N excess was higher for cut buffers (Fig. 2). As the irrigation season progressed (60 d, 114 d), atom % ^{15}N excess values remained higher in uncut buffers for the 1-m sampling distance, but there were no downslope differences between cut and uncut buffers.

There were differences in seasonal biomass trends between the cut and uncut buffers (Fig. 3). The cut buffer biomass values reflect the effects of regular cutting, with increasing biomass values between cuttings and sharp drops in biomass on the actual cutting dates. For the uncut buffers, biomass values varied nonlinearly throughout the season, but generally, of the three species, dallis grass had the highest biomass (per m^2) and velvet grass had the lowest. The total biomass values for a given uncut buffer varied according to the cover distribution of the species within that area.

When biomass (Fig. 3) and percent cover distribution data were used to determine mass of ^{15}N for each dominant species in a given uncut buffer, orchard grass tended to sequester the majority of the ^{15}N, whereas velvet grass sequestered the least (Table 2). Although dallis grass had the highest biomass per m^2 (Fig. 3), it was intermediate in its ^{15}N storage. The mass of ^{15}N sequestered by a given species did not change significantly over the course of the season.

The amount of ^{15}N sequestered by each species was summed to get the total mass (mg) of ^{15}N sequestered per uncut buffer (Fig. 4). Values for uncut buffers reflect the ^{15}N in the standing biomass on a given date, whereas

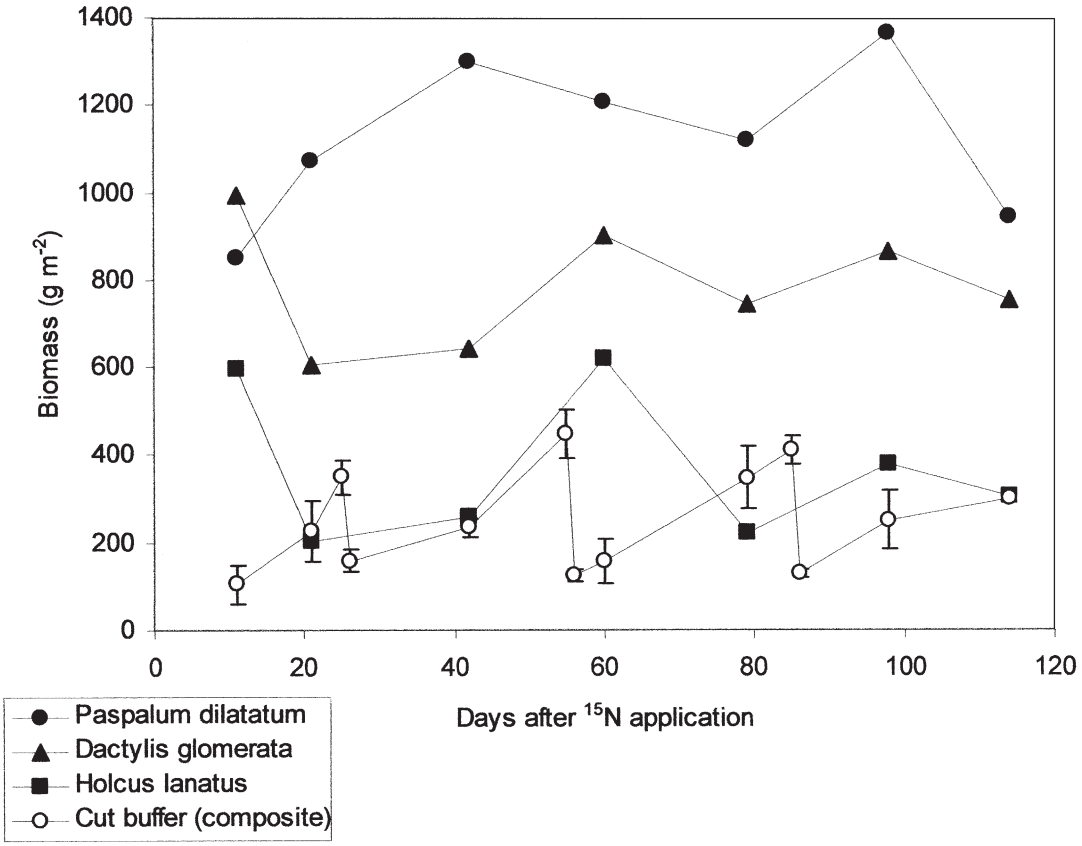

Fig. 3. Vegetation biomass (g m^{-2}) by time after ^{15}N application for each of the three dominant species within the uncut buffers and for a composite of all species present per m^2 within the cut buffers.

values for cut buffers are cumulative, reflecting the ^{15}N in the standing biomass as well as the ^{15}N removed from the plots by cutting. Overall, the uncut buffers had a constant mass of ^{15}N sequestered over the course of the season, regardless of biomass fluctuations. In contrast, the cut buffers had a lower mass of ^{15}N sequestered immediately following ^{15}N application, indicative of the lower biomass in these buffers on Day 11. Over the course of the season, however, there was a linear increase in the mass of ^{15}N in the cut buffers such that by the end of the season there was nearly double the amount of ^{15}N sequestered in the cut buffers compared with the uncut.

The linear mixed effects (LME) model confirms that cutting effect on ^{15}N uptake was time dependent (Table 3). Cutting alone resulted in a decrease in ^{15}N uptake by buffer vegetation (coefficient = −13.2, P = 0.1); however, if the interaction with time is taken into consideration, cutting substantially increased the amount of ^{15}N sequestered, with the most significant differences between cut and uncut buffers occurring at the end of the season (coefficient = +46.6, P = <0.0001).

The majority of ^{15}N sequestration by vegetation occurred within the ^{15}N application zone (Fig. 4). As for the cut buffers, the ^{15}N application zone was cut regularly and the ^{15}N contained in the vegetation was removed, so there was a steady increase over the season of ^{15}N removed. The difference in mass of ^{15}N removed from the application zone versus the cut buffers was nearly an order of magnitude, despite a much smaller reference area. Unlike the cut buffers, the total cumulative mass of ^{15}N sequestered in the standing vegetation of the application zone did not increase over the season; it increased from Day 11 to Day 42 and then decreased to a new lower level, suggesting ^{15}N losses from the standing vegetation (Fig. 4).

A similar decrease in ^{15}N mass within the zone of ^{15}N application was observed in the soil microbial biomass (Fig. 5). In both the 0- to 7- and 7- to 15-cm depth increments, the amount of microbial ^{15}N decreased between Days 3 and 114. In contrast, just 1 m downslope, the amount of microbial ^{15}N increased between Days 3 and 114 in both depth increments. There were no significant differences in microbial ^{15}N content between the cut and uncut buffers, regardless of date.

Within 114 d of ^{15}N application, 14 to 16% of the total amount of ^{15}N applied was taken up by the pasture vegetation within the zone of ^{15}N application (Table 2). However, the observed differences in recovery between the cut and uncut buffers were most notable in the buffer vegetation, where the cut buffers recovered an average of 59 mg (2.4%) of the applied tracer compared with 26 mg (1%) in the uncut buffers.

Surface Runoff

Runoff rates averaged 0.4 L s^{-1} $plot^{-1}$ (SD ± 0.1) within 15 min of the start of runoff and leveled off at

Table 2. The ^{15}N budget for runoff, soil, and vegetation after final irrigation (Day 114). Percent recovery refers to the mass of ^{15}N recovered in a given pasture-buffer area relative to the total mass applied (2500 mg) in the zone of ^{15}N application.

	Average ^{15}N recovery per pasture-buffer area (±SD)	
	Cut buffer	Uncut buffer
	mg	
Runoff		
Depth		
NH_4^+	1 ± 0.2	1 ± 0.2
NO_3^-	3 ± 0.6	4 ± 1.3
Dissolved organic nitrogen (DON)	3 ± 0.3	4 ± 0.7
Total dissolved N	7 ± 0.6	9 ± 2.5
Soil		
Depth	^{15}N zone	
0–7 cm	571 ± 223	419 ± 102
7–15 cm	79 ± 31	77 ± 37
15–40 cm	513 ± 460	384 ± 349
40–100 cm	52 ± 35	81 ± 35
Total (0–100 cm)	1215 ± 429	961 ± 293
	Buffer	
0–7 cm	91 ± 18	95 ± 25
7–15 cm	23 ± 14	18 ± 2
15–40 cm	157 ± 61	224 ± 33
40–100 cm	260 ± 115	220 ± 94
Total (0–100 cm)	531 ± 144	557 ± 135
Vegetation		
Vegetation type	^{15}N zone	
Grass	395 ± 52	338 ± 50
	Buffer	
Grass (composite)	59 ± 10	26 ± 5
Orchard grass	NA†	17 ± 6
Velvet grass	NA	1 ± 2
Dallis grass	NA	7 ± 5
Total recovery		
^{15}N recovered	2207 ± 522	1891 ± 444
Total recovery, %†	88 ± 20	76 ± 18

† Cut buffers have composite values only.

approximately 0.7 L s^{-1} $plot^{-1}$ (SD ± 0.2) by the end of the 3-h irrigation event. When ^{15}N concentrations were multiplied by runoff volume to calculate the total load of ^{15}N in runoff over a given irrigation event, the ^{15}N load in runoff in all N pools was greater from the uncut buffer than from the cut buffer after Day 42 (Fig. 6). The NO_3^-–^{15}N load decreased to a steady state by Day 42, NH_4^+–^{15}N load increased to a steady state by Day 42, and DON–^{15}N load remained relatively level throughout the study. Maximum NO_3^-–^{15}N was lost in the first 21 d after ^{15}N application, and maximum differences in NO_3^-–^{15}N load between the cut and uncut buffers appeared after Day 60. For the NH_4^+ and DON pools, significant differences between the cut and uncut buffers started to appear as early as Day 42. The data gap on Day 60 is due to the occurrence of an isolated precipitation event on that sampling day; the total volume of precipitation was comparable with the volume during a typical irrigation event. Over one-half of the precipitation fell within 1 h; the total duration of the event was 8 h. For Day 60, vegetation and soil solution samples could be collected, but there was no measurable runoff.

According to the runoff LME model, for the NO_3^-–, NH_4^+–, and total dissolved ^{15}N pools, cutting alone did not have a significant effect on the ^{15}N load. There was, however, a significant effect when the interaction with time was taken into consideration, with the cut buffers having less ^{15}N load in runoff as shown by the negative regression coefficients. For the NO_3^- and NH_4^+ pools, the effect of cutting only became statistically significant (LME $P \leq 0.05$) on Day 42, whereas for total dissolved N, cutting had a significant effect by Day 21 (LME $P = 0.05$). For the DON pool, cutting reduced the ^{15}N load (LME $P = 0.08$) regardless of time since ^{15}N application; adding time as a fixed effect improved the significance slightly, but not enough to warrant its inclusion in the model.

Subsurface: Soil Solution and Soil

The NO_3^-–^{15}N concentration of the soil solution (Fig. 7) was similar in range to the ^{15}N concentration of the NO_3^-–^{15}N in runoff (Fig. 6), but the soil solution NO_3^-–^{15}N concentrations tended to be much more variable. This was particularly true in the first 42 d after ^{15}N application during which time the samples were collected 10 d after irrigation, versus after 3 d.

In the cut buffers, the solution samplers at the 15-cm depth tended to have decreasing NO_3^-–^{15}N concentrations with increasing distance from the zone of ^{15}N application (Fig. 7). In contrast, those samplers at the same depth in the uncut buffers developed a pattern of increasing NO_3^-–^{15}N concentration with increasing distance by Days 101 and 116. The samplers at the 45-cm depth did not demonstrate any clear patterns associated with distance from the zone of ^{15}N application. Regardless of sampler depth and distance from the zone of ^{15}N application, the NO_3^-–^{15}N concentrations were significantly higher for soil solution in the uncut buffer than in the cut buffer ($P = 0.002$, Wilcoxon rank sum test).

This difference between the cut and uncut buffers for NO_3^-–^{15}N concentrations in the subsurface water was not reflected in the 0- to 15-cm soil atom % ^{15}N excess (Fig. 8). There was no significant difference in soil atom % ^{15}N excess between the cut and uncut buffers on either sampling date ($P = 0.7$, Wilcoxon rank sum test). There was also no difference between sampling dates. The only general pattern was a decrease in atom % ^{15}N excess with increasing distance from the zone of ^{15}N application.

Nitrogen-15 Recovery Budget

The ^{15}N lost via runoff was relatively small compared with the amount applied: 0.3% of the applied ^{15}N was lost in runoff from the cut buffers and 0.4% of the applied ^{15}N was lost in runoff from the uncut buffers (Table 2). Maximum recovery occurred in the soil, where approximately 38 to 49% of the applied ^{15}N was measured as total soil ^{15}N within the zone of ^{15}N application. A further 21 to 22% was measured in the soil within the buffers. The vegetation within the zone of ^{15}N application recovered 14 to 16% of the applied ^{15}N over the course of the study. Only a small amount was recovered by the buffer vegetation itself: 2% in the cut buffers and 1% in the uncut buffers. The difference in ^{15}N recovery between the cut and uncut buffers was not significant for any pool except for the within-buffer vegetation (Table 2). Total ^{15}N recovery from the cut buffers was

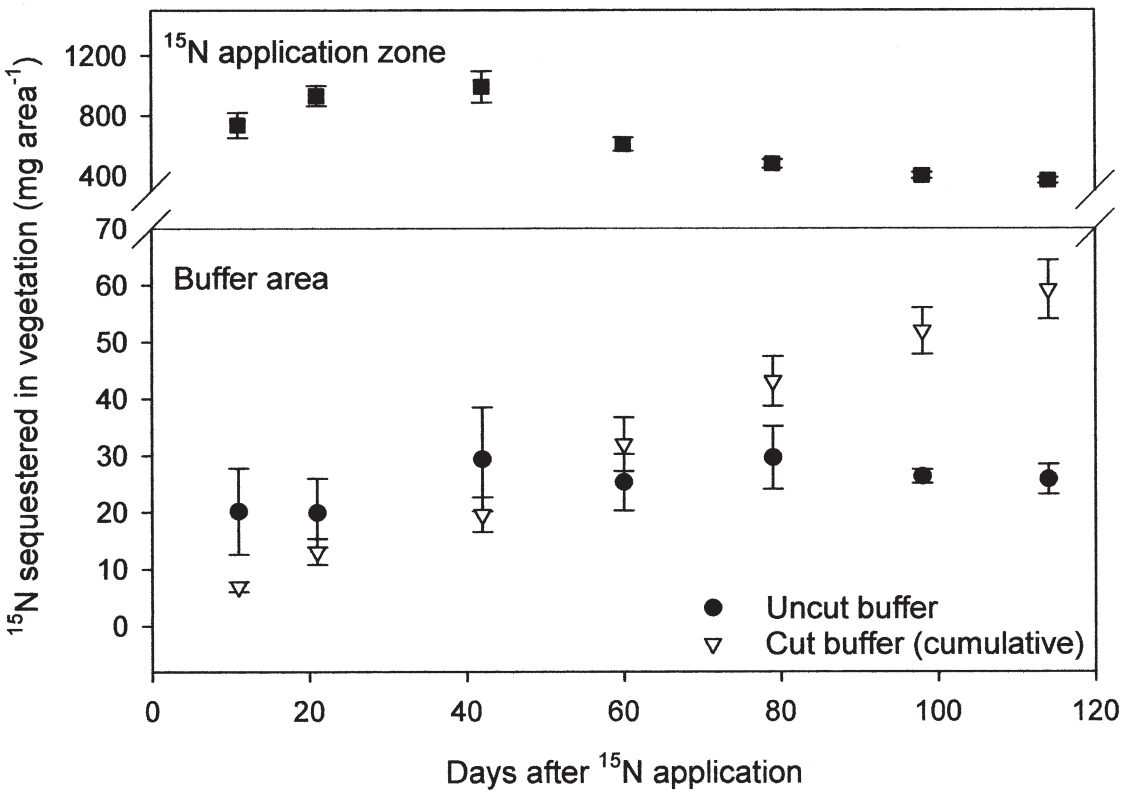

Fig. 4. Total mass (mg) of ^{15}N sequestered in aboveground vegetation within a given area by time after ^{15}N application for ^{15}N application zone, uncut buffer areas, and cut buffer areas, where ^{15}N application zone and cut buffer areas are cumulative, including ^{15}N removed by clipping during the irrigation season.

88% (SD ± 20) and from the uncut buffers was 76% (SD ± 18).

DISCUSSION

Effects of Cutting on Plant Nitrogen-15 Uptake

The effect of cutting on plant ^{15}N uptake was not significant in the first few weeks following ^{15}N application (Table 3); higher ^{15}N sequestration in the uncut vegetation reflects the greater initial biomass available for N uptake. Soon thereafter, however, ^{15}N sequestration increased in the cut buffers as regular cutting maintained plant N demand.

Table 3. Linear mixed effects model estimating ^{15}N uptake by buffer vegetation over time by treatment (uncut versus cut). Coefficients quantify the expected effect of cutting and time on mg ^{15}N sequestered per buffer area relative to the reference level.

Model term	Coefficient	95% CI†	*P*
Intercept	20.2	10.2, 30.2	0.0002
Treatment			
Uncut‡	0.0	–	–
Cut	−13.2	−30.3, 3.9	0.1
Days since ^{15}N application			
11 d‡	0.0	–	–
21 d	−0.3	−10.7, 10.2	1.0
42 d	9.1	−1.3, 19.6	0.1
60 d	5.1	−5.4, 15.6	0.3
79 d	9.5	−1.0, 19.9	0.1
98 d	6.1	−4.3, 16.6	0.2
114 d	5.6	−4.8, 16.1	0.3
Treatment × days after ^{15}N			
Cut × 11 d‡	0.0	–	–
Cut × 21 d	6.4	−8.4, 21.2	0.4
Cut × 42 d	3.5	−11.3, 18.3	0.6
Cut × 60 d	19.9	5.1, 34.7	0.01
Cut × 79 d	26.7	11.9, 41.5	0.0008
Cut × 98 d	38.8	24.0, 53.6	<0.0001
Cut × 114 d	46.6	31.8, 61.4	<0.0001

† 95% confidence interval for coefficient (lower, upper).
‡ Reference category for variable.

In comparing the N capacity of cut and uncut buffers over the irrigation season, the cut buffers sequestered twice the ^{15}N of the uncut buffers (Fig. 4). Given that the cut and uncut buffers had very similar atom % ^{15}N excess values for much of the irrigation season, the difference in sequestration can be attributed primarily to increases in biomass in the cut buffers. The increase in biomass following each cutting (Fig. 3) was a typical compensatory response to defoliation (Ferraro and Oesterheld, 2002). This period of growth should be a period of high N demand (Jackson et al., 1988). Cutting of aboveground vegetation can increase shoot N assimilation by more than 5 times (Matheson et al., 2002). Cutting and removing vegetation from the buffers allowed the standing biomass to take advantage of immobilized soil ^{15}N as it was released by microbial mineralization (Bardgett et al., 2003). Removal of the cut vegetation is essential, otherwise decomposition will simply return nutrients to the system, increasing the potential for losses via runoff or leaching (Dosskey, 2001). In contrast, the uncut buffers showed very little change in ^{15}N sequestration throughout the irrigation season (Fig. 4), suggesting the occurrence of senescence and a corresponding decrease in N demand (Jackson et al., 1988) or the absence of net growth during the study.

Examining interspecific differences in ^{15}N was ex-

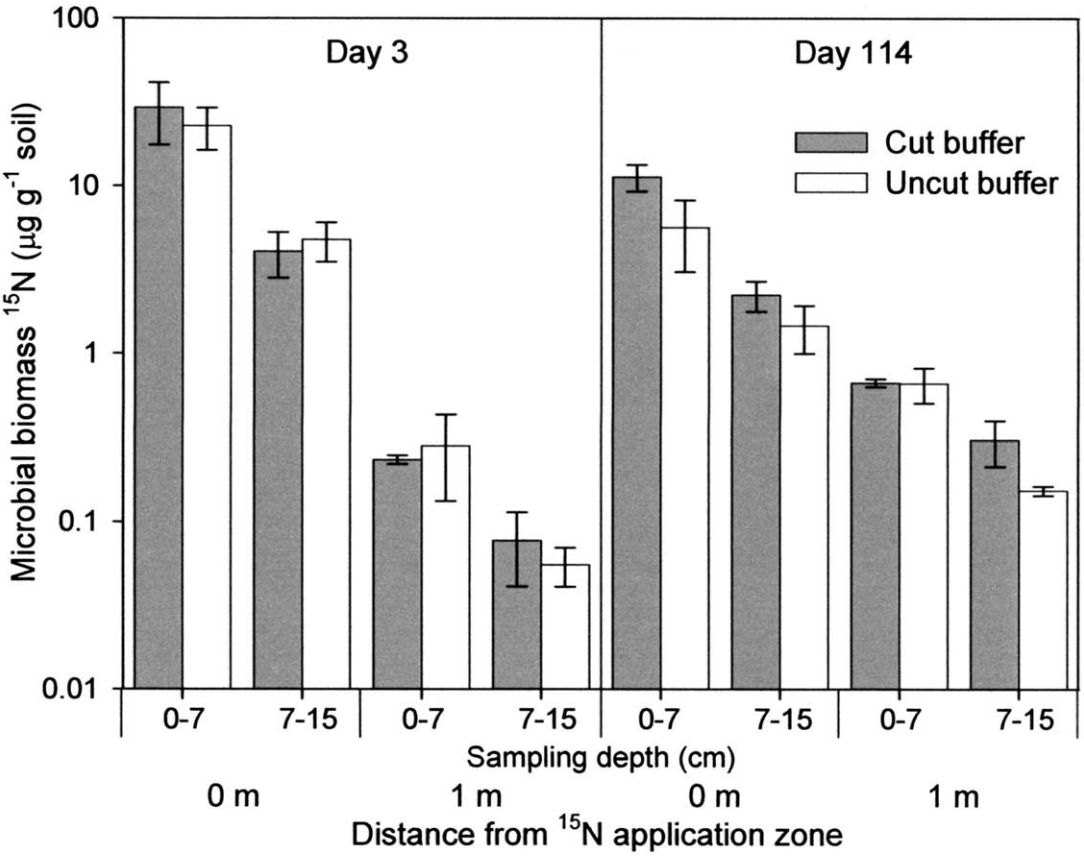

Fig. 5. Soil microbial biomass ^{15}N (μg ^{15}N g^{-1} soil) by depth, distance from ^{15}N application, and time after ^{15}N application. Note log *y* axis.

pected to provide insight into the functioning of the uncut buffers, and to help determine whether one species might be better suited for buffers in irrigated pasture systems; however, the primary determinant of ^{15}N sequestration was total aboveground biomass within the buffer. Dallis grass had greater biomass per m^2 than orchard grass (Fig. 3), but orchard grass was by far more prevalent with the buffers (averaging 60% of buffer area, compared with 20% of buffer area for dallis grass), and so served to sequester the most ^{15}N of the three species. Although the cut buffers were not examined by species, orchard grass appeared to be the dominant species within the cut buffers (and in the surrounding pasture), attributable in large part to its rapid regrowth following grazing or cutting, compared with moderate and slow regrowth for dallis grass and velvet grass, respectively (USDA, 2005). Rapid regrowth following cutting and extent of ground cover are likely the best predictors of plant uptake ability in managed buffers.

Cut or uncut, maximum plant uptake of ^{15}N occurred within the first 4 m of the buffer (Fig. 2). The higher vegetation atom % ^{15}N excess observed in the first meter downslope of the application area for the uncut buffers compared with the cut buffers may be attributable in part to differences in root biomass. Although soil moisture is a major factor controlling fine root production in annual grasslands (Cheng and Bledsoe, 2002), cutting may have reduced the root biomass (Williams et al., 2003), increasing root turnover, or inhibited the production of new roots (Matheson et al., 2002). If the uncut buffers had greater root biomass, the vegetation immediately downslope of the zone of application may have been better able to take advantage of ^{15}N moving downslope via lateral movement, or, particularly if there was significant lateral root development, drawn on the much higher concentrations of ^{15}N available within the zone of application itself.

Effects of Cutting on Surface Runoff Nitrogen-15

For the first 21 d following application of the ^{15}N tracer, there were no differences in surface runoff NO_3^-–^{15}N between the cut and uncut buffers (Fig. 6). Regardless of cutting treatment, there was excess ^{15}N measured in the surface runoff during the first irrigation event after the ^{15}N was applied. Similarly, there was ^{15}N measured in the soil solution (Fig. 7), soil (Fig. 8), and vegetation (Fig. 2) at the furthest distance from the zone of ^{15}N application following the first irrigation event. This suggests that both the cut and uncut buffers were attenuating some ^{15}N, but during this period, the NO_3^-–^{15}N tracer was extremely mobile and its redistribution via surface runoff was identical for both the cut and uncut buffers. As Di and Cameron (2002) observed, maximum NO_3^- leaching tends to occur whenever NO_3^- is present in the soil profile during periods of significant drainage, as would be associated with irrigation events. It is interesting to note, however, that the sharp decrease in surface runoff NO_3^-–^{15}N between Days 21 and 42 (Fig. 6) corresponds to the first post–^{15}N application cutting of the

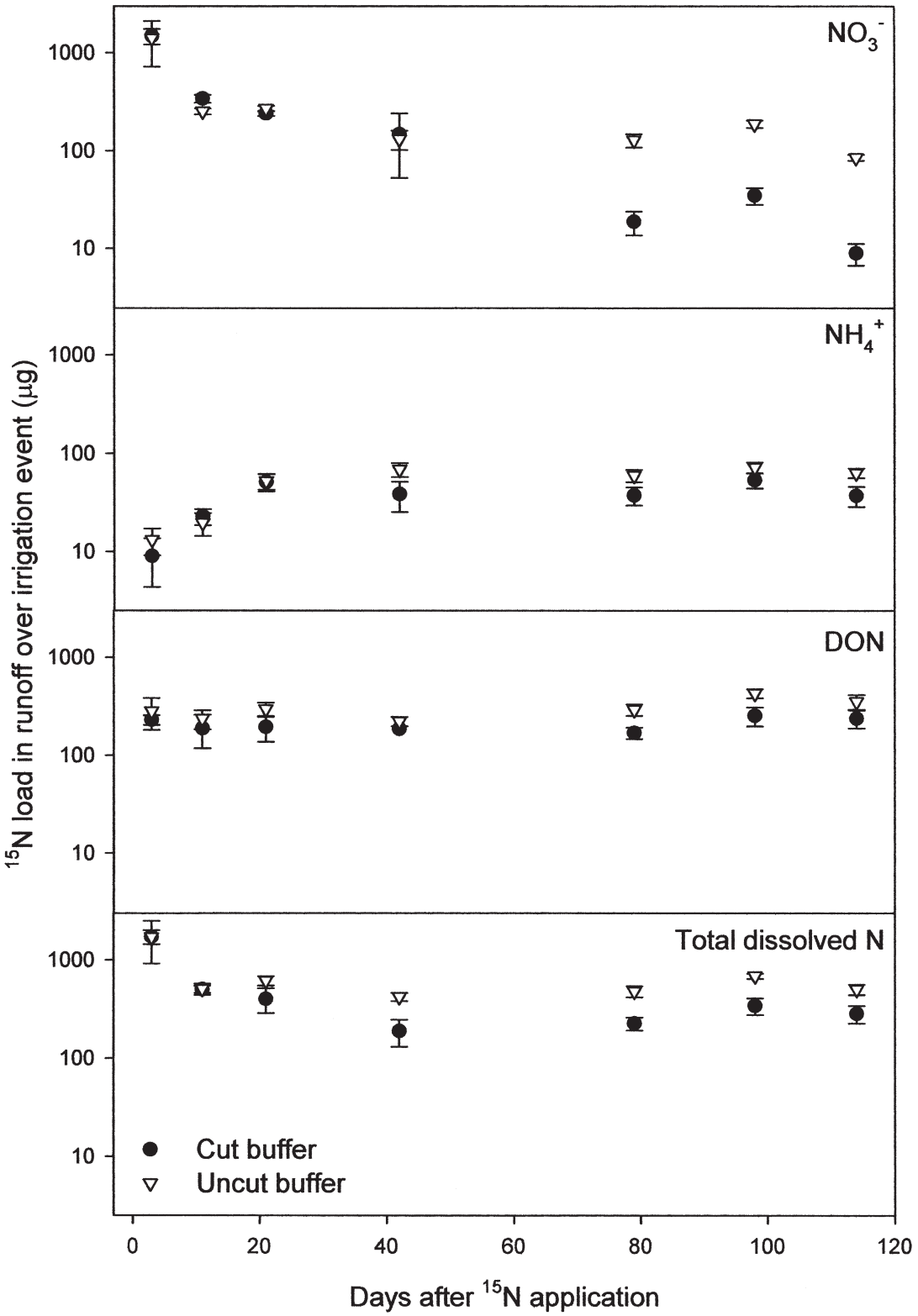

Fig. 6. Runoff ^{15}N load over the course of the irrigation season. Values are averaged by buffer treatment and time; error bars represent standard error. Note log *y* axis.

buffer vegetation, suggesting a very strong initial cutting effect on runoff water quality.

During Days 3 to 42, some of the NO_3^-–^{15}N appears to have been immobilized by microbial biomass and cycled into other N pools, as shown by the parallel decrease in runoff NO_3^-–^{15}N load and increases in runoff DON– and NH_4^+–^{15}N (Fig. 6). One possible pathway for the movement between the NO_3^-– and NH_4^+–^{15}N pools

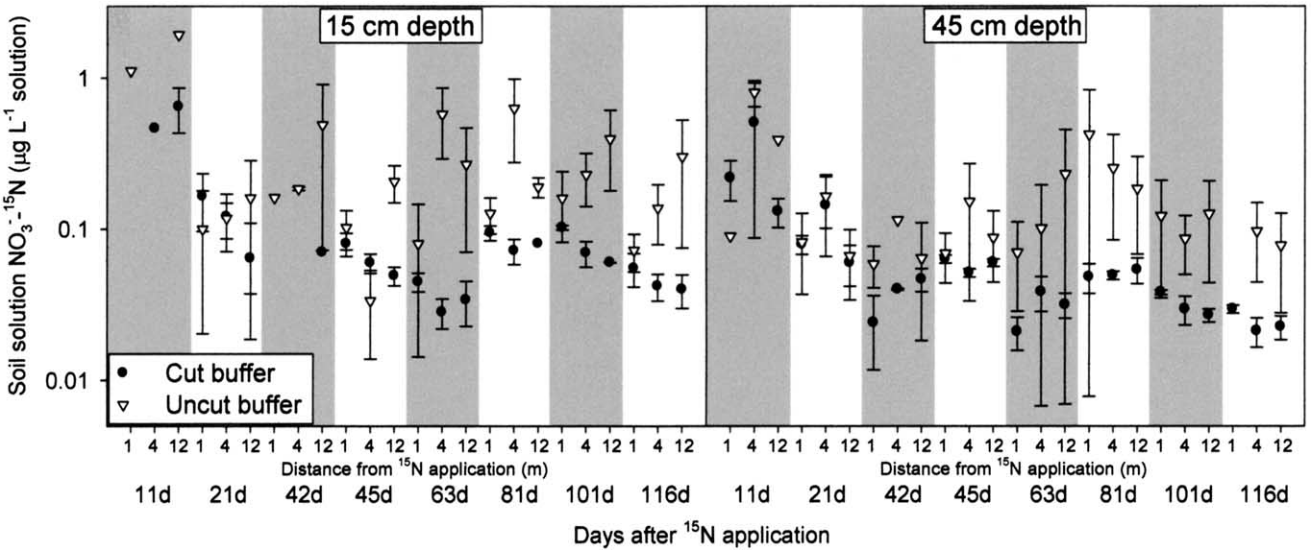

Fig. 7. Soil solution NO_3^-–^{15}N concentrations by time and distance from ^{15}N application. Values are averaged by buffer treatment, time, and distance; error bars represent standard error. Note log *y* axis.

is dissimilatory nitrate reduction to ammonium. Although it is unlikely here given the inherently low soil NH_4^+ levels, this could not be confirmed with our field tracer study design. In a microcosm study, Matheson et al. (2002) found that within 32 d, up to 49% of applied NO_3^-–^{15}N was reduced to NH_4^+–^{15}N and up to 25% was immobilized in the microbial biomass. After Day 42, there were significantly lower ^{15}N loads in runoff from the cut buffers compared with the uncut buffers for both the NO_3^-– and NH_4^+–^{15}N pools (Fig. 6). This corresponds with the observed increase in aboveground plant biomass and plant ^{15}N storage in the cut buffers (Fig. 4). However, the difference between cut and uncut buffers did not increase substantially after 42 d despite continued growth in the cut buffers, indicating that even though there was a continual demand for ^{15}N, mineral N was available for ^{15}N runoff losses, albeit at extremely low concentrations (<1 μg total dissolved N L^{-1} runoff).

Unlike the runoff NH_4^+–^{15}N pool, there was no lag time between the application of ^{15}N and the leveling off in runoff DON–^{15}N load. This may reflect the observation of Davidson et al. (1990) that these grassland soils have a significant heterotrophic microbial sink for NO_3^-, particularly when NH_4^+ availability is low. There was also minimal temporal effect on the efficacy of cutting for reducing runoff DON–^{15}N load: cut buffers had consistently lower runoff DON–^{15}N load than uncut buffers throughout the experiment ($P = 0.08$; Fig. 6). The higher DON–^{15}N in runoff from the uncut buffers may reflect slightly greater partitioning of mineral ^{15}N to the microbial pool in the absence of significant plant demand. For example, Jackson et al. (1989) observed that microbial immobilization of NO_3^- and NH_4^+ was greater than plant uptake regardless of plant growth stage, and for NH_4^+, the relative dominance of immobilization was even more pronounced after plant senescence. In this study, however, the partitioning was not significant enough to be reflected in microbial biomass ^{15}N (Fig. 5). The constancy of the DON–^{15}N load throughout the experiment is indicative of a rapid N turnover; Davidson et al. (1990) observed a turnover time of 0.3 to 1.6 d in grassland soils at SFREC.

Effects of Severe Cutting on Nitrogen-15 Retention

The buffer areas that were cut regularly exhibited good ^{15}N retention due to the continual plant demand for N as it was released by the microbial biomass (Fig. 4). The uncut buffers also had good N retention within the time frame of the irrigation season, but previous research in irrigated pasture (Bedard-Haughn et al., 2004) suggests that plant decomposition during the winter months would ultimately contribute to N losses from the uncut buffers. The rate and amount of new growth and hence new N demand within uncut buffers will determine how much of the recycled ^{15}N will be retained over the long term.

Even during the course of the irrigation season, there were ^{15}N losses observed within the application zone vegetation (Fig. 4), despite regular cutting and removal of vegetation ^{15}N. This may be due to unintended severe cutting (i.e., too short) of the vegetation in the application zone compared with the buffer areas; vegetative growth and vigor in this zone after 42 d was limited. Severe cutting has been found to contribute to elevated rates of root death (Jarvis and Macduff, 1989). Cutting can also give rise to increased partitioning of N to the belowground biomass and increased rhizodeposition (Paterson and Sim, 2000). In a study using ^{15}N-enriched synthetic sheep urine, Williams et al. (2003) observed more ^{15}N in the soil when vegetation was subject to regular cutting. This belowground partitioning associated with aboveground cutting and the increased potential for root death, coupled with the high C levels already present in the rhizosphere, provide optimum conditions for microbial ^{15}N uptake (Jackson et al., 1989). As observed, soil microbes can compete effectively for both

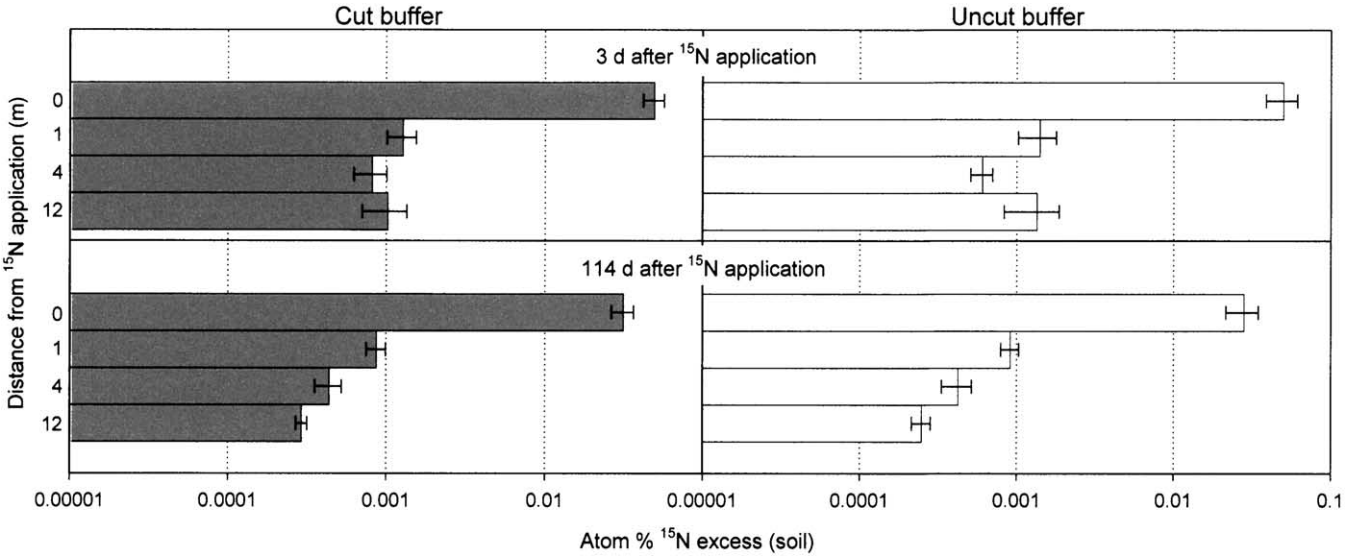

Fig. 8. Atom % ^{15}N excess in soils by distance from ^{15}N application at 3 d (top) and 114 d (bottom) after ^{15}N application. Values are averaged by treatment and distance; error bars represent standard error.

NH_4^+ and NO_3^- (Bardgett et al., 2003; Davidson et al., 1990). This immobilization of inorganic N can be an effective mechanism for minimizing leaching losses in agroecosystems (Di et al., 1999), but given the rapid turnover of the microbial N pool, the positive effects may be temporary (Jackson et al., 1989). This may be of particular importance where frequent wet–dry cycles occur, as is the case in irrigated pasture, because wetting cycles can cause significant pulses of N mineralization (Fierer and Schimel, 2002).

Analysis of the microbial biomass ^{15}N (Fig. 5) does show high microbial immobilization of the applied ^{15}N in the first few days following application. There is then a decrease in microbial ^{15}N in the soil within the area of application over the course of the irrigation season, but just 1 m downslope, the microbial ^{15}N increases. If this increase were due to decomposition of ^{15}N-enriched vegetation within the buffers themselves, there would likely be a difference between the cut and uncut buffers because the cut buffers did not show any evidence of senescence during the irrigation season. Instead, the increase in microbial ^{15}N at the 1-m distance may be attributable to losses via root exudation and/or decomposition in the rhizosphere of the zone of ^{15}N application, uptake by microbial biomass, and subsequent mineralization and lateral movement of inorganic N.

The differences in ^{15}N retention between the regularly cut buffers and the severely cut application zone (Fig. 4) highlight the importance of responsible buffer management; cutting must be managed to allow for maximum compensatory regrowth, otherwise any benefits associated with cutting may be lost.

Effects of Cutting on Subsurface Nitrogen-15

The primary effect of cutting in the subsurface environment is lower NO_3^-–^{15}N concentrations in the soil solution within the cut buffers. Within 45 d of ^{15}N application, the 15-cm soil solution samples from the cut buffers established the spatial pattern of decreasing NO_3^-–^{15}N with increasing distance from the zone of ^{15}N application; at the same time, the difference in soil solution NO_3^-–^{15}N concentration between the cut and uncut buffers became much more pronounced (Fig. 7). These patterns complement the decrease in ^{15}N load in the surface runoff from the cut buffers after 42 d (Fig. 6). Given that maximum root concentrations in California grasslands tend to occur in the top 10 to 20 cm of the soil profile (Cheng and Bledsoe, 2002; Jackson et al., 1988), these soil solution patterns likely reflect the increased root uptake associated with increased vegetation growth in the cut buffers (Ourry et al., 1990).

In the cut buffers, the 15-cm soil solution NO_3^-–^{15}N concentrations remained at a maximum closest to the zone of ^{15}N application, but decrease with distance due to vegetative buffer uptake (Fig. 7). In the uncut buffers, 15-cm soil solution NO_3^-–^{15}N concentrations were variable or increased with distance due to increased downslope movement via surface runoff and subsurface lateral flow (Bedard-Haughn et al., 2004) and due to lower plant N demand associated with senescence of the mature vegetation (Jackson et al., 1988). In the 45-cm soil solution samples (Fig. 7), the differences between the cut and uncut buffers likely reflect leaching from the root zone because the lack of significant root density at this depth (Cheng and Bledsoe, 2002) makes it unlikely that differences in plant uptake are the cause of differences in concentration.

A similar pattern of higher ^{15}N levels in the uncut buffers was expected for the 0- to 15-cm soil atom % ^{15}N excess (Fig. 8), but there were no significant differences between the cut and uncut buffers on either sampling date. However, the soil atom % ^{15}N excess values reflected a combination of the soil and the root biomass; roots were not analyzed separately. The cut buffers are likely to have greater belowground partitioning of ^{15}N into the root biomass due to stress effects of cutting

(Paterson and Sim, 2000; Williams et al., 2003). Given that the uncut buffers have higher NO_3^-–^{15}N concentrations in the 15-cm soil solution samples (Fig. 7), uncut solution ^{15}N and cut root ^{15}N may balance each other out, resulting in similar total soil ^{15}N values.

Nitrogen-15 Recovery Budget

The majority of the applied ^{15}N (59–71%) was recovered in the soil beneath the pasture and buffer areas, indicating that for this study, infiltration was the dominant mechanism for minimizing ^{15}N losses in surface runoff. A further 17 to 19% of the applied ^{15}N was recovered by vegetation uptake. Although the majority of the infiltration and uptake occurred within the zone of ^{15}N application itself, the buffers attenuated approximately 25% of the applied ^{15}N, mostly within the soil, indicating that the buffers themselves were effective, regardless of cutting treatment. Runoff losses represented less than 1% of the applied ^{15}N (Table 2). Note that the only permanent sink for the applied ^{15}N was that removed in the cut vegetation; all other ^{15}N could potentially be re-released at a later point and become available for leaching and runoff.

Although the amount of ^{15}N recovered within the buffer vegetation was low compared with the overall N pool, there was a significant difference in vegetation recovery between the cut and uncut buffers, with the cut buffers recovering approximately twice as much ^{15}N as the uncut buffers (Table 2). The absence of a significant difference in total ^{15}N recovery between the cut and uncut buffers does not reflect the temporal improvement in runoff water quality or vegetative uptake (Table 3). It does, however, reflect the absence of significant differences in vegetation, runoff, soil solution, and soil ^{15}N concentrations between the cut and uncut buffers in the first 21 d of the experiment, when ^{15}N concentrations in all N pools were at their highest.

The applied ^{15}N that was not recovered in the runoff, soil, or vegetation likely reflects losses due to denitrification, volatilization, or leaching within the soil profile to depths greater than 1 m. Note that runoff losses may be higher under the more typical granular fertilizer application.

Runoff and Nitrogen Losses

Reducing the irrigation rate from 4 to 1 L s^{-1} $plot^{-1}$ decreased the runoff losses by approximately 50% (Table 1) compared with Bedard-Haughn et al. (2004), a buffer study on an adjacent set of plots that used the more typical irrigation rates for the region. The initial runoff rate of 0.4 L s^{-1} $plot^{-1}$ was identical to that observed in Bedard-Haughn et al. (2004), but the maximum level of 0.7 L s^{-1} $plot^{-1}$ was considerably lower than the previously measured 3 L s^{-1} $plot^{-1}$. This smaller range of runoff rates was reflected in a smaller range of ^{15}N loads between the beginning and the end of a given irrigation event.

By reducing the irrigation rate by 75%, the total amount of dissolved N lost from a given buffer decreased by six- to eightfold, from 55 mg N $buffer^{-1}$ at the 4 L s^{-1} irrigation rate (Bedard-Haughn et al., 2004) to 7 to 9 mg N $buffer^{-1}$ in this study (Table 2). Vegetative growth and vigor were comparable with surrounding pasture irrigated at the typical higher rate. This emphasizes the critical importance of managing irrigation rates to minimize runoff as a primary method for reducing nutrient loading in surface water (Tate et al., 2000b). Vegetative buffers still have a significant impact on nutrient loading, but must remain secondary measures (Barling and Moore, 1994).

Reducing the irrigation rates by 75% also appears to have substantially increased the relative importance of infiltration, particularly within the zone of ^{15}N application. Verchot et al. (1997) also found infiltration to be a major mechanism for minimizing N losses in surface runoff under unsaturated conditions. At the end of this study, the amount of soil ^{15}N stored in the A horizon (0–15 cm) within the zone of ^{15}N application (Table 2) was approximately 10 times that stored when the higher irrigation rate was applied (Bedard-Haughn et al., 2004). However, some of this greater soil storage is related to belowground ^{15}N losses from the vegetation within the zone of application (Fig. 4).

CONCLUSIONS

Although both the cut and uncut buffers served to attenuate ^{15}N in surface runoff, regular cutting of vegetation in upland buffer areas contributed to a significant increase in plant ^{15}N uptake and a corresponding decrease in ^{15}N concentration of both the surface runoff and the subsurface water, indicating that cutting is a viable management technique for improving both the capacity and effectiveness of vegetative buffers in irrigated pasture. Monthly cutting of buffer vegetation doubled ^{15}N uptake compared with uncut buffers. Although mineralization of microbially immobilized ^{15}N provided an ongoing source of ^{15}N over the course of the irrigation season, vegetation in the cut buffers had greater N demand due to increased growth and potential for shoot assimilation.

The dominant factor affecting ^{15}N concentration in surface runoff from irrigated pasture is the irrigation rate itself. Reducing the irrigation rate by 75% substantially decreased both the volume of runoff and the concentration of ^{15}N within the runoff. This appears to be primarily due to greater infiltration within the zone of ^{15}N application. Given this increase in infiltration with the lower irrigation rate, consideration must be given to the long-term effectiveness of infiltration as a mechanism for attenuating ^{15}N. Nitrogen storage within the soil may be ephemeral and could eventually be leached to ground water unless removed by plant uptake and cutting or denitrification.

However, total buffer vegetation uptake was relatively small in this irrigated pasture, so the importance of the cutting effect needs to be considered under a broader range of N inputs and in other agroecosystems. Within-pasture fertilizer timing and irrigation management must still be considered the primary techniques for minimizing NO_3^- losses in irrigated pasture.

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