Title

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Evaluating the Effectiveness of Vegetated Buffers to Remove Nutrients, Pathogens, and Sediment Transported in Runoff from Grazed, Irrigated Pastures

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Abstract
This project examined the application of grass buffer strips to improve runoff water quality from irrigated pastures in the Sierra Nevada foothills. These flood irrigated pastures range up to 30% slope, and can generate significant runoff. Three experiments were conducted to determine: 1) the partitioning of nitrogen (N) between soil, plants and runoff within buffers; 2) whether buffer capacity for N decreases over time as buffer vegetation matures in the absence of grazing/cutting; and 3) the efficiency of buffers to attenuate *E. coli*, total phosphorus (P), dissolves organic carbon (DOC), and suspended solids in a rotationally grazed pasture scenario designed to offset the timing of grazing bouts from irrigation events. These experiments were conducted on irrigated pasture – buffer runoff plots at the UC Sierra Foothill Research and Extension Center near Brown's Valley, CA. Buffer size treatments were 0, 8, and 16 m and grazing – irrigation offset treatments were 2, 15, and 30 days. We used the nitrogen isotope (^{15}N) method in Experiments 1 and 2. Vegetative uptake was a major mechanism for attenuating new N in irrigated pasture systems, and nutrient cycling within vegetative buffers was serving as both a sink and a source for N in runoff. Buffers were effective for attenuating nitrate (NO_3-^{15}N), slightly more effective for ammonium (NH_4-^{15}N), and least effective for dissolved organic nitrogen (DON-^{15}N). For DON, the 16 m buffer was actually less effective than the 8 m buffer, indicating that the 16 m buffers themselves were serving as a source for this less plant-available form of N. Monthly cutting of buffer vegetation doubled ^{15}N uptake compared to uncut buffers, confirming that regular cutting and harvest of buffer vegetation increases vegetative buffer efficacy for N uptake. Under the irrigation application – runoff – transport capacity scenario examined in this study, we could attribute no significant reduction in dissolved organic carbon, total suspended sediment, *E. coli*, or total phosphorus load (kg/ha) in irrigation runoff to 3 year non-grazed/cut vegetative buffers either 8 or 16 m in width. DOC load was actually significantly ($P<0.05$) increased on plots with a 16 m buffer, and there were apparent increases in load for TSS, VTSS, and *E. coli* for both 8 and 16 m buffer widths compared to no buffer control plots. Pollutant load was positively related to runoff volume, indicating that reductions in runoff volume will result in reduced pollutant transport. Pollutant load was significantly reduced by increasing days rest from grazing prior to irrigation from 2 to 15 days. Extending this rest to 30 days gained only slight additional reduction in pollutant load. The general failure of buffers to reduce DOC, TSS, VTSS, *E. coli*, and P loads in Experiment 3 under the high irrigation application – runoff – transport capacity scenario examined in this study should not be extrapolated to conclude that vegetative buffers have no merit for water quality improvement in this system. Rather, it is clear that application of buffers to irrigated pastures without a simultaneous effort to balance irrigation rates with soil infiltration capacity and plant-soil water demand will certainly not achieve water quality protection. It is also clear that management of buffer vegetation will be required to maintain buffer capacity for nutrients, and to reduce the potential for buffers to become a source for DOC and DON, and habitat for rodents shedding *E. coli* in their feces.

Introduction and Problem Statement
A vegetative buffer strip is a management practice designed to attenuate pollutants in runoff from agricultural and other land uses (Dosskey, 2001). A significant body of literature concludes that vegetative buffers can be an effective water quality improvement practice, but that buffers are not uniformly effective in all circumstances (Chaubey et al.,

1995; Chaubey et al., 1994; Dillaha et al., 1989; Schwer and Clausen, 1989; Bingham et al., 1980; Doyle et al., 1977). The attenuation efficiency of a vegetative buffer depends upon factors such as runoff volume, pollutant load and type, and soil type (Bharati et al., 2002; Schmitt et al., 1999; Castelle et al., 1994). In addition to the application of appropriate buffer design specifications (e.g., size, vegetation type) to accommodate site characteristics (e.g., runoff volume, soil type), long-term management of buffer vegetation is required to sustain buffer efficiency for some pollutants.

In this project we were specifically interested in evaluating the efficiency of grass vegetated buffers to attenuate various pollutants in surface runoff from irrigated, grazed pastures in the western foothills of California's Sierra Nevada Mountains. Irrigated pasture provides relatively inexpensive green forage during dry summer months (May though mid October), and is critical to the viability of the State's livestock industry. A significant portion of California's surface drinking water supplies are derived from watersheds on the west-slope of the central and northern Sierra Nevada Mountain Range, and irrigated pastures occupy substantial acreage in the foothills of these watersheds (FRRAP 1988).

Various water pollutants of concern can be transported from grazed, irrigated pastures. These include microbial pollutants which can lead to human illness if drinking water concentrations reach infectious levels. The current microbial water quality standard for the region is that *Escherichia coli* (*E. coli*) cannot exceed 126 colony forming units (cfu) per 100 mL of surface water. Nitrogen (N) and phosphorus (P) are commonly implicated in eutrophication in fresh water reservoirs (Cole et al., 2004). For instance, nitrate (NO_3) concentrations as low as 1 mg/L can contribute to algal blooms (Mendez et al., 1999). Dissolved organic carbon (DOC) is a pollutant of particular concern in the region, due to the formation of carcinogenic derivatives at municipal drinking water treatment facilities when source water contains high DOC levels. Suspended sediments are of water quality concern because of potential degradation of spawning gravels and pool habitat critical to threatened and endangered fisheries by sedimentation.

In the Sierra Nevada's, irrigation water is typically applied to pastures using open ditch or gated pipe flood irrigation techniques. On foothill pastures, land slope can range up to 30%, with up to 70% runoff (Tate et al., 2000). These runoff rates are significantly (orders of magnitude) greater than reported in most studies of buffer efficiency reported in the scientific literature. Rotational grazing can be used to off-set the timing of active grazing bouts from irrigation events. Rotational grazing is the movement of livestock herds systematically through a set of pastures to achieve multiple benefits such as improved forage harvest efficiency and optimal forage growth rates. Irrigators in the region generally attempt to remove livestock from pastures prior to an irrigation event in order to avoid physical trampling damage to soil, vegetation, and irrigation conveyance ditches which occurs when pastures are irrigated while populated with livestock. The water quality benefits of this practice are not known, but rest from grazing prior to irrigation should allow for desiccation and decomposition of livestock fecal material and recovery of vegetative cover prior to irrigation – runoff – pollutant transport events.

The goal of this project was to examine the efficiency of grass buffers to attenuate various pollutants commonly found in runoff from grazed, irrigated foothill pastures.

Three experiments were conducted to examine buffer efficiency under several conditions including various buffer sizes, pasture grazing management, and buffer vegetation management.

Objectives

Objective A. Quantify the effectiveness of buffers to attenuate nutrients, *E. coli*, dissolved organic carbon, and suspended solids in surface water runoff from grazed, flood-irrigated pastures.

Objective B. Employ the N isotope method to quantify N partitioning within pasture, buffer, and runoff.

Objective C. Employ the N isotope method to determine whether buffer capacity for N decreases over time as buffer vegetation accumulates and matures in the absence of grazing.

Procedure

Project Location. All experiments were conducted on irrigated pasture at the UC Sierra Foothill Research and Extension Center near Browns Valley, California, approximately 100 km north of Sacramento. Two sets of runoff plots (Study Site A and B) were utilized for this project.

Study Site A. During the summers of 2000 and 2001, nine adjacent runoff plots were established within an existing flood-irrigated pasture at SFREC (Figure 1). A completely random study design was employed to allocate 3 buffer treatments in 3 replicates to 9 plots. Plots were established parallel to the slope and the direction of irrigation flow. Buffer treatments consisted of a 3:1 pasture to buffer area ratio; a 6:1 pasture to buffer area ratio; and a no buffer control. Buffers were fenced from livestock grazing in 2000 and remained un-grazed by livestock for the duration of the project. Each plot had a 240 m^2 (5 m wide by 48 m long) pasture area. The 3:1 pasture to buffer area treatment had a buffer area of 80 m^2, and the 6:1 pasture to buffer area treatment had a buffer area of 40 m^2. Buffer length for the 3:1 and 6:1 buffer treatment was 16 and 8 m, respectively. The plots were fenced from the surrounding pasture to prevent trespass grazing and to allow for application of grazing management treatments. 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 (Netafim, Model WT) that allowed measurement of both rate and quantity of water applied. Earthen berms separated adjacent plots to prevent surface water crossing from one plot to another. Poly vinyl chloride collection troughs, with a V-notch at one end for sample collection, were installed across the bottom of each treatment with the edge of the trough flush with the ground surface. Concrete aprons were used to prevent erosion along the upslope edge of the troughs, and to insure that surface water entered the collection troughs. The troughs allowed for the measurement of surface water runoff rates and collection of water samples for analysis. Collection troughs were fenced to exclude cattle. The pasture-buffer areas were dominated by *Dactylis glomerata* (orchard grass), *Holcus lanatus* (Yorkshire fog/velvet grass) and *Paspalum dilatatum* (dallis grass), with *Verbena bonariensis* (purpletop/tall verbena) also present in the buffer areas. Soils were classified as fine-loamy, mixed, thermic, Mollic Haploxeralfs of the Auburn-Las Posas-Argonaut Rocky

Loam association (Herbert and Begg, 1969). Slope ranged from 9.5 to 11.9%. The pasture area was fertilized with 170 kg/ha of 16-20-0 (N-P-K) in early May of each year of the study.

Study Site B. During the summer of 2002, ten adjacent plots were established within the same flood-irrigated pasture at SFREC which contained Study Site A as described above. 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 (Figure 2). The pasture-buffer areas were dominated by *Dactylis glomerata* (orchard grass), *Holcus lanatus* (Yorkshire fog/velvet grass), and *Paspalum dilatatum* (dallis grass). Soils at the site 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 15.0 to 18.3%. Beginning in June 2003, a cutting treatment was randomly allocated to five of the ten buffer areas. Adjacent pasture-buffer areas were separated by landscape edging to prevent runoff crossover between buffers. For the duration of the 2003 irrigation season, vegetation in the five cut buffers was trimmed monthly using nylon-line trimmers to levels corresponding to post-grazing height (5-10 cm) in the surrounding pasture. All 10 pasture areas were trimmed at the same intervals as the cut buffers to simulate grazing. 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 (Netafim, Model WT) 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. Collection troughs installed across the bottom of each buffer collected all surface water runoff, allowing for measurement of surface water runoff rates and collection of water samples for analysis. Collection troughs and entire plot area were fenced to exclude cattle for the duration of the experiment.

Experiment 1. Experiment 1 was conducted on Study Site A during the summer of 2002. The purpose of this experiment was to examine the efficiency of the buffers to attenuate N contributed from the pasture area as surface runoff (Objective A) and to examine the partitioning of this N between soil, plants and runoff (Objective B). This experiment was based upon the use of nitrogen isotopes, which are stable and non-radioactive, have been used extensively to follow the dynamics of N in soils and crops (Powlson and Barraclough, 1993). We utilized ^{15}N enriched material so that the added N could be detected and differentiated from inherent background variability in naturally occurring ^{15}N levels (Bedard-Haughn et al., 2003). Natural abundance background levels of ^{15}N in all N pools were measured prior to application of ^{15}N-labeled fertilizer to account for natural variability and dilution of the applied ^{15}N fertilizer by background ^{14}N.

In July 2002, ^{15}N-labeled KNO_3 was applied in solution at a rate of 5 kg N/ha and 99.7 atom % ^{15}N. The rate and atom % 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 all 9 plots along the entire width of the experiment. The area labeled was 1 m deep and 5 m wide and located 0.75 m above the buffer areas (Figure 1). Following application, the labeled fertilizer was watered in with 20 L of water per m^2. Watering-in was done by hand with watering cans for maximum precision; 20 L represented the

optimum amount to ensure that the applied ^{15}N-labeled KNO_3 was rinsed off of the foliar surfaces, but the volume was not so great as to cause deep leaching of the applied fertilizer. The ^{15}N application area was fenced to prevent redistribution of the ^{15}N-enriched material by the cattle.

Grazing in pasture areas was by mature beef cattle at a stocking density of 5 animal units (dry cow) on 0.216 ha for 2 days. Cattle were managed to replicate grazing and fecal loading rates typical of the region. Mean fecal loading rate per plot per grazing event was 336 kg/ha (±29.1). A 3-week rest period was maintained between grazing events to assure the sustained health and productivity of the pasture's vegetation. Buffer areas were neither fertilized nor grazed, but received the same irrigation treatment as the pasture areas. For this experiment the irrigation rate was calibrated to 4 L/sec per buffer for approximately 3 h every 11 days. This application rate approximates high application rates on pastures in the region, or a worst case transport scenario.

For a 14-week period following application, water samples were collected from the installed collection troughs during each irrigation trial (11 day schedule). Water samples (500 ml) were collected as "grab" samples from the V-notch at the end of each collection trough. Samples were taken at 0 (leading edge of runoff), 15, 30, 60, 90, and 120 min. following commencement of runoff from each treatment and were stored frozen until analysis. This sampling scheme represented a minimum sample number and is based upon previous experience with the timing of runoff and pollutant transport from these systems. At each sampling interval, runoff rate was determined by measuring the volume of runoff draining from the V-notch in the collection trough in a 5 s period. Subsurface water was collected using soil solution samplers (Soilmoisture Equipment Corp., Santa Barbara, CA), which were installed to a depth of 45 cm, the approximate depth of the heavy clay Bt horizon (Figure 1). Following each irrigation, vacuum was applied to the soil solution sampling tubes and allowed to draw moisture from the soil for 10 days (i.e., until the next irrigation). Although vacuum was not applied constantly over the 10 day period, suction was still present at sampling. Soil water samples were collected just prior to the subsequent irrigation and were stored frozen until analysis.

Runoff ^{15}N isotope analyses were performed on all three N pools: NO_3, NH_4, and total N for Days 1, 12, 31, 65, and 86 following application of the tracer. For Days 1 and 12, only the 0, 15, 60 and 120 minute time intervals were analyzed because preliminary experiments indicated that this was sufficient for characterization of maximum variation. For Days 31-86, even fewer intervals were needed to acquire sufficient information because there was no longer significant change between sampling days. Samples were filtered to remove sediment and vegetation residues from runoff. Ammonium-^{15}N and NO_3-^{15}N were determined by NH_3 diffusion onto polytetrafluoroethylene-encased acid traps (Stark and Hart, 1996). To measure NO_3-^{15}N, the Stark and Hart (1996) method was modified only slightly in that following diffusion of 100 ml samples for NH_4, 1 ml of 5M NaOH was added to each to bring the pH up to ≥12. Samples were heated uncovered at 95°C to remove any trace ammonium or labile organic N (DON) and to concentrate the volume down to 25 ml. In place of Devarda's alloy, $TiCl_3$ (Fisherbrand Titanous Chloride Solution, 20%) was then added (typically 1/20th sample volume) to reduce NO_3 to NH_3. Soil solution samples (25 ml aliquots) were analyzed for NO_3-^{15}N via $TiCl_3$ diffusion as above, except no concentrating step was required. Titanous chloride has been found

preferable to Devarda's alloy due to its low cost, low N contamination, and availability in solution form (Cho et al., 2002; Cresser, 1977; Crumpton et al., 1987). Samples were sealed and incubated at 50°C for 72 hours. Nitrate standards with field-level N concentrations had mean N recovery of 94% (SD±5%) using this modified method.

Total ^{15}N was determined on a separate 20 ml aliquot by performing a persulfate digestion (APHA, 1989) to convert the DON and NH_4 to NO_3, and samples were then diffused for NO_3 as above (without concentration step). DON-^{15}N for each sample was calculated using an isotope mixing model via difference from total ^{15}N (Shearer and Kohl, 1993). Following diffusion, acid disks were removed from polytetrafluoroethylene packets and analyzed via mass spectrometry (Europa Integra Integrated Stable Isotope Analyzer, UK) at the University of California, Davis Stable Isotope Facility. The current sensitivity of our stable isotope ratio mass spectrometers is 0.0002 atom % ^{15}N.

Representative plant samples from the pasture and buffer areas were taken prior to each irrigation trial. To determine how far the ^{15}N-fertilizer had moved into the buffer strip, plants were sampled across the width of the buffer at down slope intervals with a sample spacing of 1 m immediately above and below the zone of ^{15}N application, and spacing of 2 m further into the buffer. The buffer vegetation samples were separated between grasses and verbena, the native shrub in the buffers. Following each grazing (every second irrigation), the fenced ^{15}N application area was clipped and the vegetation removed to simulate grazing. All plant samples were oven-dried at 65°C and analyzed for ^{15}N isotopic composition via mass spectrometry (van Kessel et al., 1994).

Soil samples were taken monthly to 15cm depth in two increments (0-7, 7-15), corresponding to the depth of the A horizon. Samples were taken at 0, 1, and 5 m from the ^{15}N application at 12, 43, and 86 days following ^{15}N application. Samples were also taken at 8 and 16 m on day 86. Sample quantity, depth, and diameter were limited due to concurrent sampling at the site that analyzed total suspended sediment in runoff. Soil samples were oven-dried at 40°C and analyzed for total N and ^{15}N via mass spectrometry. Isotopic levels for the soils and plants 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 source. Background levels are based on pre-application samples. Where possible, atom % ^{15}N excess amounts were extrapolated to get the total amount of ^{15}N in a given pool by weight and thus to determine a ^{15}N budget. Note that it was not possible to perform budget calculations for the vegetation in the buffer areas as accurate biomass measurements over the course of the summer season would have required destructive sampling that would have confounded subsequent measurements.

The results were analyzed using linear mixed effects model analysis. 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 both fixed (buffer treatment) and random (irrigation date) effects on buffer ^{15}N uptake levels. Treating time as a random effect provided a direct test for whether buffered plots were significantly different from non-buffered plots when results were considered over the duration of the study. The magnitude and direction (+/-) of the coefficient for buffer effect was used to define the relationship between ^{15}N loading in runoff and buffer treatment. This approach allowed for robust evaluation of the data while accounting for the repeated

measures (group effect – plot identity) embedded in the data structure. 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., 2003).

Experiment 2. Experiment 2 was conducted on Study Site B during the summer of 2003. The purpose of this experiment was to determine whether buffer capacity for N decreases over time as buffer vegetation matures in the absence of grazing (Objective C). The ^{15}N-enriched tracer method was used to quantify N uptake by pasture and buffer vegetation (Objective B), and loss to runoff (Objective 1).

For this experiment the irrigation rate was calibrated to 1 L/sec per buffer for approximately 3 h every 9 d. Total duration of each irrigation event varied according to the volume needed to balance soil water content, which was determined using evapotranspiration data from the California Irrigation Management Information System weather station located at SFREC. This rate is 75% lower than the rate applied in Experiment 1; a lower irrigation rate was used in an effort to reduce runoff losses and improve irrigation efficiency.

In June 2003, ^{15}N-labeled KNO_3 was applied in solution at a rate of 5 kg/N/ha 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 (Figure 1). Application rate and area were based on Experiment 1. 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 not so great as to cause deep percolation. The entire study area was fenced to prevent disturbance or redistribution of the ^{15}N-enriched material by the cattle grazing the surrounding pasture. Natural abundance background levels of ^{15}N in all N pools were measured prior to application of ^{15}N-labeled fertilizer to account for natural variability and dilution of the applied ^{15}N fertilizer by ^{14}N. Isotopic levels for the plants 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 source. Atom % ^{15}N excess amounts were extrapolated to obtain the total amount of ^{15}N in a given pool by weight and thus to determine a ^{15}N budget.

Vegetation samples were collected 3, 11, 21, 42, 60, 79, 98, and 114 days after ^{15}N application. There were more frequent sampling dates early in the project to capture maximum variation. To determine how far the ^{15}N-fertilizer had moved into the buffers, plants were sampled immediately 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 represent 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 (van Kessel et al., 1994).

Of the two plots that did not receive ^{15}N application, one received the same regular cutting as the cut buffers while the other was left to mature the same as the uncut buffers. The species composition, vegetation age, and irrigation rates of these two non-labeled 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 non-labeled buffers (Figure 2). 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.

Soil samples were taken at 0, 1, 4 and 12 m from the ^{15}N application at 3 and 114 days following ^{15}N application. On both dates, samples were taken to 15 cm depth in two increments (0-7, 7-15), corresponding to the depth of the A horizon. 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 (APHA, 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 Experiment 1. Microbial ^{15}N was determined by difference between fumigated- and non-fumigated samples for both dates (0-15 cm only) for the 0 and 1 m distances. Soil texture was determined on the 114 d samples using laser diffraction and reported in volume percent (Eshel et al., 2004).

Runoff samples were taken from the collection ditches 15 minutes following the leading edge of runoff and again just before the end of the irrigation event and were stored frozen until analysis. Our findings in Experiment 1 establish that the 15-minute 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 drainage pipe in the runoff collection trough. Runoff rates were determined at regular intervals by measuring the volume of runoff from the drainage pipe in a 5 s period. Runoff ^{15}N isotope analyses were performed on three N pools: NO_3, NH_4, and total N as described for Experiment 1.

The results were analyzed using linear mixed effects model analysis as described for Experiment 1 (Pinheiro and Bates, 2000) and was used here to account for the influence of fixed (cutting) effects on buffer ^{15}N uptake and loss levels and for the repeated measures (group effect – plot identity) embedded in the data structure.

Experiment 3. Experiment 3 was conducted on Study Site A during the summer of 2003. The purpose of this experiment was to examine the efficiency of buffers to attenuate *E. coli*, P, DOC, and suspended solids in a rotationally grazed pasture scenario (Objective

A) designed to offset the timing of grazing bouts from irrigation events by 2, 15, and 30 days (i.e., 2, 15, and 30 days rest from grazing prior to irrigation). Buffer treatments were the primary treatment examined in the experiment and are described above in the section "Study Site A". At the time of this experiment (May through September 2003) the buffers had been excluded from grazing by domestic livestock for 3 years (May 2000), and no livestock grazing occurred within the buffers during the course of this experiment.

For this experiment the irrigation rate was calibrated to 2 L/sec per buffer for approximately 3 h every 9 d. Total duration of each irrigation event varied according to the volume needed to balance soil water content, which was determined using evapotranspiration data from the California Irrigation Management Information System weather station located at SFREC. This rate is 50% lower than the rate applied in Experiment 1; a lower irrigation rate was used in an effort to reduce runoff losses and improve irrigation efficiency.

Duration of pasture rest from grazing prior to irrigation was the secondary treatment examined in this study. Rotational grazing was used to generate conditions of 2, 15, and 30 days rest from grazing before irrigation. Irrigation application scheduling, temporary electric fencing, and short duration grazing bouts (2 day) were used to apply each rest from grazing treatment to each plot a minimum of 2 times over the course of 8 irrigation events (Trial 1 through 8) conducted between May and September 2003. All plots were irrigated and sampled in each trial. Table 1 reports the grazing treatment to plots for each trial. Stocking rate (grazing pressure) was constant for each grazing bout over the course of the study. Grazing throughout the study was implemented with beef cattle at a stocking rate of 8 animal units (1 animal unit = 1 mature beef cow) on 0.216 ha for 2 days (grazing bout). Cattle were provided with water and a dry loafing area adjacent to the runoff plots to facilitate even grazing and fecal deposition across the actual study plots. Fecal loading rate per grazing event was measured for each plot and included in statistical analysis as a covariate.

Water samples (500 ml) were collected as "grab" samples from the V-notch at the end of each collection trough. Samples were taken at 0 (leading edge of runoff), 15, 30, 60, 90, and 120 min. following commencement of runoff from each treatment and were stored frozen until analysis. This sampling scheme represented a minimum sample number and is based upon previous experience with the timing of runoff and pollutant transport from these systems. At each sampling interval, runoff rate was determined by measuring the volume of runoff draining from the V-notch in the collection trough in a 5 s period.

All water samples were analyzed for total phosphorus (P), phosphate (PO_4), total suspended solids (TSS), volatile total suspended solids (VTSS), dissolved organic carbon (DOC), and *E. coli* concentration. Turbidity (ntu) was also determined on all water samples. Irrigation event load (flux) of each pollutant (kg/ha/event) from each plot was calculated by multiplying sample concentration (mg/L) and instantaneous runoff at the time of sample collection (L/s) for each sample collection event (0, 15, 30, 60, 90, 120 min). The area under this instantaneous load curve (event load) was then estimated by fitting a cubic spline to these 6 data points through time and solving the integral from time = 0 to 120 min (as explained in volume 2 of *Stata Statistical Software: Release 7.0*, *Reference H-P*, p. 530-534 [Stata Corporation, College Station, Tex.], 2001).

Total P concentration was determined on a non-filtered sample following persulfate oxidation. PO_4 concentration was determined by ascorbic acid reduction of phosphomolybdate complex and quantitative measurement by flow injection analysis (APHA 1989). Enumeration of *E. coli* (cfu/100 mL) was accomplished by serially diluting each sample (10^{-2}, 10^{-3}, 10^{-4}), followed by direct membrane filtration and culturing the membrane onto EEC CHROMagar at 44.5 C for 24hr (APHA 1989). Total suspended solid concentration was determined via vacuum filtration of a 500 to 1000 mL sample aliquot through a pre-weighed, desiccated 1.0 μm porosity filter, and desiccation to a constant weight on a scientific balance (APHA 1989). VTSS was then determined by combustion of TSS filters in a muffle furnace and resulting loss of mass (APHA 1989). DOC was determined on a 25 mL sample aliquot pre-filtered through a 0.45 μm porosity filter then processed with a UV – persulfate TOC analyzer.

Concentration and load data collected across 8 trials were analyzed using linear mixed effects model analysis (Pinheiro and Bates, 2000). Individual analyses were conducted for each pollutant (dependent variable). Plot identity was treated as a group effect to account for repeated measures on individual plots (experimental unit) over the course of 8 trials. Buffer treatment (control, 3:1, 6:1), runoff (L/ha/event), rest from grazing (2, 15, 30 days) and fecal load (kg/ha) were treated as fixed (independent variables) effects in the analysis. The magnitude and direction (+/-) of the coefficients for buffer treatment, runoff, rest from grazing, and fresh fecal load were used to define the relationships between these factors and pollutant concentration and load in pasture runoff.

Results

Experiment 1. With few exceptions, the non-buffered treatment had the highest runoff concentrations of ^{15}N, with the difference between the buffered and non-buffered treatments being greatest at the leading edge of runoff (t=0) and diminishing over the course of a given irrigation event (Figure 3). Following the leading edge, the concentration increased slightly for the NO_3- and DON-^{15}N pools, and then decreased corresponding to a rapid increase in runoff levels as the irrigation proceeded. Typically, initial (t=0) runoff levels were approximately 0.4 L/s/plot, increased rapidly to 2 L/s/ plot by 30 minutes, and then leveled at a steady rate of approximately 3 L/s/plot by 60 minutes. During the second post-application irrigation (Day 12), the NO_3-^{15}N concentration started similar to the concentration at the end of the previous irrigation, but for the other pools, there was a slight increase in concentration at the leading edge of runoff. By Day 31, the pattern was well-established, with a slight increase in concentration at the start of each irrigation event, followed by a rapid decrease to a steady level. The NO_3-^{15}N levels showed the greatest change over the course of the summer, from having the highest concentration at Day 1 to the lowest at Day 86. The NH_4-^{15}N levels tended to remain relatively constant. By Day 31, the DON-^{15}N pool established a new steady level and remained constant for the remainder of the summer. Differences between the 8 m- and 16 m-buffers could also be observed during some of the earlier irrigations, but did not display the same consistent pattern.

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 (Figure 4). Regardless of the buffer treatment,

maximum ^{15}N loads were observed in the first irrigation following application. Note, however, that for NH_4-^{15}N, the loads were relatively low and constant for the first two irrigations following application, and overall, remained quite steady over the course of the summer. Nitrate-^{15}N load started at a much higher level than the other pools, but decreased rapidly to a lower level and continued to be detectable throughout the summer. Although DON-^{15}N load decreased after the first irrigation, it established a higher steady-state level, similar to that of NH_4-^{15}N. Typically, the greatest differences between the buffered and non-buffered treatments were observed in the first month after ^{15}N application, but by later in the summer, there were minimal differences among treatments. Note, however, that by the end of the summer, the buffered treatments occasionally exhibited higher ^{15}N loads for NO_3^- and DON than the non-buffered treatment (Figure 4).

Linear mixed effects analysis of the ^{15}N runoff load over the course of the entire summer indicated that when compared to the non-buffered treatments, the buffered treatments had significantly less ^{15}N ($P = 0.05$) for all N pools except for the NO_3 pool in the 8 m buffer and the DON pool in the 16 m buffer (Table 2). For the NO_3 and NH_4 pools, the log mean load of ^{15}N in runoff decreased from the non-buffered to the 8 m- to the 16 m- buffer (from $e^{-0.19}$ to $e^{-0.42}$), illustrating that ^{15}N load decreased as buffer length increased (Table 2). In contrast, the log mean load of DON-^{15}N was greater for the 16 m buffer than for the 8 m buffer ($e^{0.06}$ versus $e^{0.01}$), suggesting that although buffered treatments had less ^{15}N load than non-buffered, the 8 m buffer had a more substantial effect on load than the 16 m buffer.

There were detectable levels of ^{15}N in the 45-cm soil solution samplers (Table 4), with a very slight decrease in atom % ^{15}N excess from the non-buffered to the 8 m buffer to the 16 m buffer, but this trend was not statistically significant (data not shown). The majority of the ^{15}N not lost via runoff was stored in vegetation and soils. Based on conservative estimates of pasture biomass, approximately 10.3 g (SD±1.4) were stored in the pasture grasses immediately underneath the zone of ^{15}N application within 11 days of application. This represents 46% of the 22.5 g of ^{15}N applied across all treatments (2.5 g per treatment). By the end of the summer, only 1 g of ^{15}N (4% of total applied) remained in the pasture biomass, but because the pasture biomass was regularly clipped and removed to simulate grazing, ^{15}N was actually removed from the system and was not recycled into the buffers. Within the buffers, most of the ^{15}N was stored in the first 4 m downslope of the zone of application, as indicated by the higher values of atom % ^{15}N excess (Figure 5). The amount of ^{15}N then decreased further downslope, but note that ^{15}N was observed in the vegetation at the end of the longest buffer even at the first sampling following application. For the grasses, the ^{15}N enrichment decreased over time, indicating dilution of the ^{15}N signature via uptake of non-enriched N. The only exceptions to this dilution occurred at 6 and 8 m downslope. For the verbena, the ^{15}N enrichment decreased over time for the first 8 m, but generally remained constant further downslope. Between Days 43 and 86, there was very little change in ^{15}N levels in the vegetation. Additional measurements were performed 3 and 6 months after the last irrigation (data not shown). Compared to Day 86, there was little change in vegetation ^{15}N levels at 3 months, but by 6 months after the last irrigation, ^{15}N levels had decreased by approximately 50%.

Of the ^{15}N applied, approximately 23% was immediately stored in the upper 15 cm of the soil immediately beneath the zone of application (Table 3); however this was subject to redistribution further downslope during subsequent irrigations (Figure 6). In the 0-7 cm layer, the ^{15}N levels immediately under the zone of application (0 m) decreased over the summer irrigation season. Further downslope, the ^{15}N levels started lower, and increased over the season, suggesting lateral movement within the 0-7 cm layer. A similar pattern was observed in the 7-15 cm layer except that by the end of the season, there was another slight decrease in ^{15}N levels at all distances. Unlike the vegetation measurements, soil measurements 6 months after the last irrigation indicated similar soil ^{15}N levels when averaged across all plots, but the spatial distribution changed.

The ^{15}N tracer was observed in all measured pools (Table 4). Levels were at a maximum for the first sampling date following ^{15}N application, but within a month of application, levels in all pools had dropped to a lower level of steady enrichment. The ^{15}N could still be measured within the system but was neither increasing nor decreasing further.

Experiment 2. At the first vegetation sampling following ^{15}N application (11 d), vegetation atom % ^{15}N excess (i.e., % ^{15}N present in excess of background ^{15}N levels) 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 (Figure 7). 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. Cut or uncut, there was a general decrease in atom % ^{15}N excess 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, Figure 2).

Comparing the atom % ^{15}N excess of the dominant species present in the uncut buffers showed few consistent patterns (Figure 8). On day 11, *Holcus lanatus* had slightly higher enrichment than the other species. Both *Holcus lanatus* and *Dactylis glomerata* showed decreasing atom % ^{15}N excess with distance from the ^{15}N application area, whereas *Paspalum dilatatum* did not. Later in the season (60 d, 114 d), atom % ^{15}N excess consistently decreased with distance, regardless of species, and there were no differences among species.

There were differences in biomass between the cut and uncut buffers (Figure 9). 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 3 species, *Paspalum dilatatum* had the highest biomass (per m^2) and *Holcus lanatus* had the lowest. The total biomass values for a given uncut buffer varied according to the cover distribution of the species within that area.

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 (Figure 10) was determined by multiplying the mg ^{15}N/g vegetation values times biomass values (g/m^2) and extrapolating to the whole area using cover data. When biomass (Figure 9) and percent cover distribution data were used to determine

mass of ^{15}N for each dominant species in a given uncut buffer, *Dactylis glomerata* tended to sequester the majority of the ^{15}N, whereas *Holcus lanatus* sequestered the least (Figure 10). Although *Paspalum dilatatum* had the highest biomass per m^2 (Figure 9), it was intermediate in its ^{15}N storage. The mass of ^{15}N sequestered by a given species remained relatively constant 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 (Figure 11). Values for uncut buffers reflect the ^{15}N in the standing biomass on a given date, whereas 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, with a slight increase between days 21 and 42. 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 to the uncut.

The linear mixed effects model confirms that cutting effect on ^{15}N uptake was time dependent (Table 5). 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 (Figure 11). 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 (Figure 12). 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 (Figure 11).

A similar decrease in ^{15}N mass within the zone of ^{15}N application was observed in the soil microbial biomass (Figure 13). In both the 0-7 and 7-15 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.

Of the total amount of ^{15}N applied, 14-16% was taken up by the pasture vegetation within the zone of ^{15}N application (Table 6). 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 to 26 mg (1%) in the uncut buffers.

The maximum differences in ^{15}N concentration in surface runoff between the cut and uncut buffers occurred in the NO_3 and total dissolved N pools, and were greatest beginning on Day 42 (Figure 14). The uncut buffers had higher ^{15}N concentrations than the cut buffers; this same trend could be observed throughout the experiment in the NH_4 and DON pools, although it was not as significant.

The concentration of NO_3-^{15}N in the surface runoff was at a maximum during the first irrigation event following 15N application (Day 3, Figure 14). By Day 21, NO_3-^{15}N concentrations in runoff from the uncut buffers had reached a plateau, whereas for the cut buffers, they decreased sharply at Day 21 before leveling off. For NH_4-^{15}N, the concentrations started lower on Day 3, and increased to a plateau by Day 42. Overall, the NH_4-^{15}N concentrations were approximately an order of magnitude lower than the NO_3-^{15}N concentrations (Figure 14, note y-axes). DON-^{15}N concentrations remained relatively constant over the course of the summer. Within a given irrigation event, the ^{15}N concentrations for any given pool were slightly higher for the 15 min. measurement than for the end of irrigation measurement, but these differences were typically not significant. When ^{15}N concentrations were multiplied by runoff rates to calculate the total load of ^{15}N in runoff over a given irrigation event, the patterns were identical to those of the ^{15}N concentrations: in all pools, the ^{15}N load in runoff was greater from the uncut buffer than from the cut buffer after Day 42 (Figure 15). NO_3-^{15}N load decreased to a plateau by Day 42, NH_4-^{15}N load increased to a plateau by Day 42, and DON-^{15}N load remained relatively level throughout the study. Maximum NO3-^{15}N was lost in the first 21 days 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 to the volume during a typical irrigation event. Over 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.

Cutting effects are demonstrated in the linear mixed effect model of the ^{15}N load data (Table 7). 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 ($P \leq 0.05$) on Day 42, whereas for total dissolved N, cutting had a significant effect by Day 21 ($P = 0.05$). For the DON pool, cutting reduced the ^{15}N load ($P = 0.08$) 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.

The ^{15}N concentration of the soil solution (Figure 16) was similar in range to the ^{15}N concentration of the NO_3-^{15}N in runoff (Figure 14), but the soil solution ^{15}N concentrations tended to be much more variable. This was particularly true in the first 42 days after ^{15}N application during which time the samples were collected 10 days after irrigation, versus after 3 days.

In the cut buffers, the solution samplers at 15 cm depth tended to have decreasing ^{15}N concentrations with increasing distance from the zone of ^{15}N application (Figure 17). In contrast, those samplers at the same depth in the uncut buffers developed a pattern of increasing ^{15}N concentration with increasing distance by Days 101 and 116. The samplers at 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 ^{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 ^{15}N concentrations in the subsurface water was not reflected in the 0-15 cm soil atom % ^{15}N excess (Figure 17). 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.

The ^{15}N lost via runoff was relatively small compared to 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 8). Maximum recovery occurred in the soil, where approximately 38-49% of the applied ^{15}N was measured as total soil ^{15}N within the zone of ^{15}N application. A further 21-22% was measured in the soil within the buffers. The vegetation within the zone of ^{15}N application recovered 14-16% 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 net recovery of the ^{15}N applied at the beginning of the study was on average 88% for the cut buffers and 76% for 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.

Experiment 3. Following 3 years of no grazing or cutting of buffer vegetation, we found the buffers at Study Site A resulted in elevation of DOC concentrations in runoff relative to no buffer control plots (Figure 18). Mean DOC concentrations of 11.93, 12.82, and 13.59 mg/L were measured for no buffer control, 6:1 and 3:1 pasture to buffer area treatments, respectively. DOC concentrations for both buffer sizes were significantly ($P<0.05$) higher than control plots (Table 9). There were no significant differences ($P>0.05$) in runoff concentrations between buffer plots and control plots for TSS, VTSS, E. coli, or turbidity (Table 9, Figure 18, concentration data for *E. coli* not displayed). Total phosphorous concentrations were apparently lower for the 6:1 and 3:1 pasture to buffer compared to the control ($P=0.094$; $P=0.169$; Table 9).

There was a significant ($P<0.05$) and negative association between runoff rate (L/s) and concentration for all 6 pollutants, reflecting the dilution effect of increased flow (Table 9). Figure 21 illustrates the relationship between runoff rate and DOC, TSS, VTSS, and turbidity reported in the linear mixed effect model in Table 9. Sample collection time (runoff time) was significantly, negatively associated with DOC, TSS, VTSS concentrations and turbidity indicating that concentrations decreased as the duration of runoff increased from 0 to 120 minutes. This in part reflects the dilution due to increasing runoff over the course of the trial, but also illustrates the flushing of pollutants realized

with initial runoff and implies that transport capacity exceeded pollutant supply over the course of a trial (Figure 18).

Rest from grazing was significantly ($P<0.05$) related with the concentration of all 6 pollutants. Basically, as time since grazing increased from 2 to 15 to 30 days there was a resulting decrease in mean concentration for DOC, TSS, VTSS, P, and turbidity (Table 9). *E. coli* concentrations were significantly ($P<0.001$) lower for 15 and 30 days rested pastures compared to 2 day rested pastures. However, *E. coli* concentrations were actually higher for 30 compared to 15 day rested pastures. An explanation for this result is not readily available, with the exception that increased rodent activity was observed in 30 day rested pastures compared to both 2 and 15 day rested pastures.

DOC load (kg/ha) was significantly higher ($P=0.049$) for the 3:1 pasture to buffer treatment relative to the no buffer control, and apparently higher ($P=0.201$) for the 6:1 pasture to buffer treatment compared to the control (Table 10; Figure 19; Figure 20). While mean TSS and VTSS were also greater for buffer plots compared to control plots, these increases in TSS and VTSS load were not statistically significant (Table 10; Figure 19; Figure 20). *E. coli* loads (log kg/ha) were 1.15, 1.21, and 1.31 for control, 6:1, and 3:1 pasture to buffer area plots respectively (Figure 23). These increases in load from buffer plots were also not significant (Table 10). There was no significant reduction in total phosphorous load from buffered plots compared to control plots (Table 10).

Runoff volume (m^3/ha/irrigation event) was significantly, positively associated with load for DOC, TSS, VTSS, and P (Table 10: Figure 22). Indicating that as runoff volume increased, so did pollutant load transported from the plot. Runoff volume was not significantly ($P<0.05$) related to *E. coli* load. Days rest from grazing was significantly, negatively related to load for all 6 pollutants. As duration of rest increased, pollutant loss was reduced (Table 10).

Conclusions

Experiment 1. Although net ^{15}N runoff losses were relatively low (3%), this study is of significance for a greater understanding of buffer function. By examining only new N inputs distinct from the much larger background N pool, this study clearly illustrates that (1) vegetative uptake is a major mechanism for attenuating new N in irrigated pasture systems and (2) nutrient cycling within vegetative buffers is indeed serving as both a sink and a source for N in runoff. Over the course of the study, buffers were effective for attenuating NO_3-^{15}N, slightly more effective for NH_4-^{15}N, and least effective for DON-^{15}N. For NO_3 and NH_4, the 16 m buffer was slightly more effective than the 8 m buffer, likely due to greater potential for plant N uptake. Nitrogen cycling within the soil was likely the major source of runoff mineral N later in the season. For DON, the 16 m buffer was actually less effective than the 8 m buffer, indicating that the 16 m buffers themselves were serving as a source for this less plant-available form of N. The majority of the applied ^{15}N was attenuated via plant uptake within the zone of ^{15}N application; a smaller percentage was stored in the first few meters of the buffer vegetation. However, without proper planning, the N sequestered in vegetation may be lost to decomposition, resulting in net N losses. To maximize long-term effectiveness and sustainability of buffer, the potential for increasing vegetation demand and uptake through buffer management must be explored.

Experiment 2. Monthly cutting of buffer vegetation doubled ^{15}N uptake compared to uncut buffers, confirming that regular cutting and harvest of buffer vegetation increases vegetative buffer efficacy for N uptake. 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 positive effects of cutting require careful management of cutting intensity to minimize belowground nutrient losses and the removal of cut residues to prevent nutrient losses via decomposition. Maximum ^{15}N sequestration occurred in the pasture area (~15% of applied), but over-cutting of the vegetation within the zone of ^{15}N application ultimately led to belowground N losses during the irrigation season. Regular cutting of vegetation in 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.

Experiment 3. Under the irrigation application – runoff – transport capacity scenario examined in this study, we could attribute no significant reduction in dissolved organic carbon, total suspended sediment, *E. coli*, or total phosphorus load (kg/ha) in irrigation runoff to 3 year non-grazed/cut vegetative buffers either 8 or 16 m in width. DOC load was actually significantly ($P<0.05$) increased on plots with a 16 m buffer, and there were apparent increases in load for TSS, VTSS, and *E. coli* for both 8 and 16 m buffer widths compared to no buffer control plots. Pollutant load was positively related to runoff volume, indicating that reductions in runoff volume will result in reduced pollutant transport. Pollutant load was significantly reduced by increasing days rest from grazing prior to irrigation from 2 to 15 days. Extending this rest to 30 days gained only slight additional reduction in pollutant load. The increase in DOC load due to buffer establishment has serious ramifications for the establishment and management of buffer near or above drinking water reservoirs or in watersheds providing drinking water. Volatile (organic) total suspended solids comprised 84 to 86% of total suspended solids regardless of buffer treatment, indicating the majority of TSS transported from the plots are organic, with limited mineral sediment transport. The vegetation in these plots must be managed to minimize accumulation of decaying vegetation.

Overall. The successful application of vegetative buffers to improve runoff water quality on grazed, irrigated pastures is not as simple as excluding livestock from the bottom portion of a pasture. Under irrigation application rates typical of the region we found that buffers failed to reduce the load of several important pollutants, and potentially serve as a source for pollutants such as DOC and DON derived from accumulation of heavy biomass within un-grazed/cut buffers. The potential for regular cutting of buffers to increase N uptake and sequestration is clearly evident, and would logically address the problem of elevated DOC from buffered plots. The dominant factor affecting ^{15}N concentration in surface runoff from irrigated pasture is the irrigation rate itself. Reducing the irrigation rate in Experiment 2 by 75% compared to Experiment 1 substantially decreased both the volume of runoff and the concentration of ^{15}N within the runoff. The positive relationship between runoff volume and all other pollutants in Experiment 3 also illustrates the dominant role runoff rate has upon pollutant transport in

these systems. The general failure of buffers to reduce DOC, TSS, VTSS, *E. coli*, and P loads in Experiment 3 under the high irrigation application – runoff – transport capacity scenario examined in this study should not be extrapolated to conclude that vegetative buffers have no merit for water quality improvement in this system. Rather, it is clear that application of buffers to irrigated pastures without a simultaneous effort to balance irrigation rates with soil infiltration capacity and plant-soil water demand will certainly not achieve water quality protection. It is also clear that management of buffer vegetation will be required to maintain buffer capacity for nutrients, and to reduce the potential for buffers to become a source for DOC and DON, and habitat for rodents shedding *E. coli* in their feces. Improvement of runoff water quality from these grazed, irrigated pastures very likely will depend upon the implementation of several management measures such as rotational grazing management to achieve pasture rest from grazing prior to irrigation, improved irrigation efficiency, fertility management, and implementation of managed buffers. Preliminary findings from this study have been used to secure funding for additional study of buffer efficiency in these systems within a holistic framework of management improvement.

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List of Publications

Bedard-Haughn, A., K.W. Tate, C. van Kessel. 2004. Using ^{15}N to quantify vegetative buffer effectiveness for sequestering N in runoff. J. Environmental Quality. In Press.

Bedard-Haughn, A., K.W. Tate, C. van Kessel. 2004. Quantifying the impact of regular cutting on vegetative buffer efficacy for ^{15}N sequestration. J. Environmental Quality. In Press.

Table 1: Grazing treatment application to runoff plots (numbers 1 through 9) over 8 irrigation events on Study Site A for Experiment 3 (2003).

		Grazing Treatment		
Trial	Day	2 day rest	15 day rest	30 day rest
1	2	1-9	none	none
2	17	4-9	1-3	none
3	32	7-9	4-6	1-3
4	47	1-3	7-9	4-6
5	62	4-6	1-3	7-9
6	77	7-9	4-6	1-3
7	92	1-3	7-9	4-6
8	107	4-6	1-3	7-9

Table 2: Linear mixed effects analysis of runoff data. ^{15}N load was transformed via natural log to account for greater variability immediately post-application. Negative log mean ^{15}N values reflect mean values of less than 1 mg (i.e., $e^{-0.12} = 0.89$, $e^{0.27} = 1.31$). Coefficients quantify the expected effect of buffer treatment on log mean ^{15}N load.

^{15}N pool	Factor	Log mean ^{15}N load		Regression coefficient (95% CI)	*P* value
		mg	±SD		
NO_3^-	No buffer	-0.12	2.65	0	
	8m buffer	-0.19	2.24	-0.33 (-0.86, 0.21)	0.1855
	16m buffer	-0.42	2.29	-0.56 (-1.09, -0.02)	0.0437
	Intercept			1.49 (1.18, 1.81)	<0.0001
NH_4^+	No buffer	-0.29	0.63	0	
	8m buffer	-0.77	0.66	-0.42 (-0.55, -0.29)	0.0002
	16m buffer	-0.96	0.56	-0.65 (-0.78, -0.52)	<0.0001
	Intercept			-0.31 (-0.39, -0.24)	<0.0001
DON	No buffer	0.27	1.04	0	
	8m buffer	<0.01	1.03	-0.23 (-0.36, -0.10)	0.0046
	16m buffer	0.06	0.80	-0.10 (-0.23, 0.02)	0.0946
	Intercept			-0.40 (-0.48, -0.33)	<0.0001
Total dissolved N	No buffer	1.43	1.70	0	
	8m buffer	1.12	1.55	-0.45 (-0.52, -0.37)	<0.0001
	16m buffer	1.01	1.48	-0.33 (-0.41, -0.25)	<0.0001
	Intercept			0.73 (0.68, 0.77)	<0.0001

Table 3: ^{15}N budget for soil and runoff as mean percentage of applied ^{15}N recovered by buffer treatment. Soil data differentiates between samples taken within the zone of ^{15}N application and samples taken in the buffer areas. Vegetation values not given due to lack of precise biomass measurements. Differences in soil ^{15}N between Day 12 and Day 86 represents losses via runoff, lateral and vertical leaching, denitrification or volatilization.

Soil (% ^{15}N recovery ± S.D.)[†]

	Depth	No buffer	8m buffer	16m buffer
		^{15}N zone		
Day 12	0-7cm	17.5 ± 4.3	19.1 ± 6.6	21.7 ± 10.6
	7-15cm	1.7 ± 0.3	2.5 ± 0.6	6.8 ± 7.7
		Buffer		
	0-7cm	n/a	0.2 ± 0.2	0.3 ± 0.1
	7-15cm	n/a	0.4 ± 0.4	0.6 ± 0.2
		^{15}N zone		
Day 86	0-7cm	3.4 ± 3.2	4.6 ± 5.6	2.2 ± 1.7
	7-15cm	0.6 ± 0.6	0.7 ± 0.6	0.3 ± 0.3
		Buffer		
	0-7cm	n/a	2.2 ± 2.0	2.7 ± 3.1
	7-15cm	n/a	1.2 ± 0.9	1.2 ± 1.2

Runoff (% ^{15}N recovery ± S.D.)

	Form	No buffer	8m buffer	16m buffer
Cumulative total (Day 1-86)	NH_4^+	0.3 ± 0.04	0.2 ± 0.02	0.1 ± 0.01
	NO_3^-	3.8 ± 1.2	2.1 ± 1.3	1.7 ± 0.7
	DON	0.6 ± 0.2	0.5 ± 0.4	0.4 ± 0.1
	Total dissolved N	4.6 ± 1.4	2.8 ± 1.6	2.2 ± 0.8

[†]2500 mg ^{15}N applied per buffer treatment

Table 4: Changes in atom % ^{15}N excess by N pool over the course of the study. Runoff values are for total dissolved N, vegetation values are for grasses only, soil solution values are for NO_3 only, and soil values are total N.

Days after ^{15}N	Runoff[a]		Days after ^{15}N	^{15}N zone vegetation[b]		Buffer vegetation[c]		Soil solution[d]		Soil[e]	
	Atom % ^{15}N excess	±SD		Atom % ^{15}N excess	±SD	Atom % ^{15}N excess	±SD	Atom % ^{15}N excess	±SD	Atom % ^{15}N excess	±SD
1	0.127	0.166	12	3.524	0.684	0.012	0.017	0.018	0.007	0.011	0.018
31	0.008	0.007	43	0.862	0.163	0.007	0.006	0.005	0.002	0.004	0.007
75	n/a[†]	n/a	86	0.278	0.076	0.007	0.005	0.005	0.001	0.002	0.004

[†]Runoff ^{15}N not analyzed for Day 75

Background atom % ^{15}N values: a 0.3666, b 0.3659, c 0.3667, d 0.3666, e 0.3676

Table 5: Linear mixed effects model predicting ^{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-value
Intercept	20.2	10.2, 30.2	0.0002
Treatment			
Uncut†	0.0	-	-
Cut	-13.2	-30.3, 3.9	0.1
Days after 15N application			
11d†	0.0	-	-
21d	-0.3	-10.7, 10.2	1.0
42d	9.1	-1.3, 19.6	0.1
60d	5.1	-5.4, 15.6	0.3
79d	9.5	-1.0, 19.9	0.1
98d	6.1	-4.3, 16.6	0.2
114d	5.6	-4.8, 16.1	0.3
Treatment x Days after ^{15}N			
Cut x 11d†	0.0	-	-
Cut x 21d	6.4	-8.4, 21.2	0.4
Cut x 42d	3.5	-11.3, 18.3	0.6
Cut x 60d	19.9	5.1, 34.7	0.01
Cut x 79d	26.7	11.9, 41.5	0.0008
Cut x 98d	38.8	24.0, 53.6	<0.0001
Cut x 114d	46.6	31.8, 61.4	<0.0001

† Reference category for variable

‡ 95% confidence interval for coefficient (lower, upper)

Table 6: The ^{15}N budget for vegetation after final irrigation (Day 114). % 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 mg ^{15}N recovered per pasture-buffer area (± S.D.)		
Vegetation type	Cut buffer	Uncut buffer
	^{15}N zone	
Grass (composite)	395 ± 52	338 ± 50
	Buffer	
Grass (composite)	59 ± 10	26 ± 5
- *Dactylis glomerata*	n/a	17 ± 6
- *Holcus lanatus*	n/a	1 ± 2
- *Paspalum dilatatum*	n/a	7 ± 5
	Total	
	454 ± 61	364 ± 47
% recovery[†]	18 ± 2	15 ± 2

[†]Each plot received 2500 mg ^{15}N

Table 7: Linear mixed effects analysis of runoff data. ^{15}N load for NO_3^- and Total dissolved N was transformed via natural log to account for greater variability immediately post-application. Coefficients quantify the expected effect of cutting and/or time on mg ^{15}N lost per buffer relative to the reference level. (Table continued next page)

^{15}N pool	Model term	Regression coefficient	95% CI‡	*P*-value
NO_3^-†	Intercept	6.9	6.4, 7.5	<0.0001
	Treatment			
	Uncut§	0.0	-	-
	Cut	0.3	-0.6, 1.3	0.5
	Days after ^{15}N application			
	3d§	0.0	-	-
	11d	-1.4	-2.2, -0.6	0.0008
	21d	-1.4	-2.2, -0.6	0.0012
	42d	-2.1	-2.9, -1.4	<0.0001
	79d	-2.1	-2.9 ,-1.4	<0.0001
	98d	-1.7	-2.5, -0.9	0.0001
	114d	-2.5	-3.3, -1.7	<0.0001
	Treatment x Days after ^{15}N			
	Cut x 3d§	0.0	-	-
	Cut x 11d	0.0	-1.1, 1.1	0.9907
	Cut x 21d	-0.4	-1.5, 0.7	0.4579
	Cut x 42d	-1.1	-2.3, 0.0	0.0451
	Cut x 79d	-2.3	-3.4, -1.2	0.0002
	Cut x 98d	-2.0	-3.2, -0.9	0.0007
	Cut x 114d	-2.6	-3.8, -1.5	<0.0001
NH_4^+	Intercept	13.2	-4.0, 30.3	0.1274
	Treatment			
	Uncut§	0.0	-	-
	Cut	-4.2	-33.5, 25.1	0.7372
	Days after ^{15}N application			
	3d§	0.0	-	-
	11d	6.3	-8.2, 20.9	0.3824
	21d	39.2	24.6, 53.7	<0.0001
	42d	55.5	41.0, 70.1	<0.0001
	79d	46.3	31.8, 60.9	<0.0001
	98d	58.8	44.3, 73.4	<0.0001
	114d	50.4	35.8, 64.9	<0.0001
	Treatment x Days after ^{15}N			
	Cut x 3d§	0.0	-	-
	Cut x 11d	7.5	-13.1, 28.1	0.4659
	Cut x 21d	2.9	-17.6, 23.5	0.7741
	Cut x 42d	-25.9	-46.5, -5.3	0.0150
	Cut x 79d	-18.0	-38.6, 2.6	0.0843
	Cut x 98d	-14.4	-35.0, 6.1	0.1643
	Cut x 114d	-22.1	-42.7, -1.5	0.0361
DON	Intercept	302.3	242.7, 361.9	<0.0001
	Treatment			
	Uncut§	0.0	-	-
	Cut	-88.4	-191.0, 14.1	0.0793

Total dissolved N[†]	Intercept	7.2	6.7, 7.6	<0.0001
	Treatment			
	Uncut[§]	0.0	-	-
	Cut	0.2	-0.5, 1.0	0.4672
	Days after ^{15}N application			
	3d[§]	0.0	-	-
	11d	-1.0	-1.6, -0.3	0.0035
	21d	-0.8	-1.4, -0.1	0.0180
	42d	-1.1	-1.8, -0.5	0.0007
	79d	-1.0	-1.7, -0.4	0.0021
	98d	-0.6	-1.3, 0.0	0.0437
	114d	-1.0	-1.6, 0.3	0.0033
	Treatment x Days after ^{15}N			
	Cut x 3d[§]	0.0	-	-
	Cut x 11d	-0.2	-1.1, 0.7	0.5930
	Cut x 21d	-0.9	-1.8, 0.0	0.0538
	Cut x 42d	-1.2	-2.1, -0.3	0.0087
	Cut x 79d	-1.0	-1.9, -0.1	0.0271
	Cut x 98d	-1.0	-1.9, -0.1	0.0288
	Cut x 114d	-0.8	-1.7, 0.0	0.0618

[†] Values transformed using natural log to account for greater variability at early dates

[‡] 95% confidence interval for coefficient (lower, upper)

[§] Reference category for variable

Table 8: Total budget for recovery of applied ^{15}N in runoff, soil and vegetation on Day 114 (after final irrigation).

Average ^{15}N recovery per pasture-buffer area (mg ± S.D.)		
Runoff		
Depth	Cut buffer	Uncut buffer
NH_4^+	1 ± 0.2	1 ± 0.2
NO_3^-	3 ± 0.6	4 ± 1.3
DON	3 ± 0.3	4 ± 0.7
Total dissolved N	7 ± 0.6	9 ± 2.5
Soil		
Depth	Cut buffer	Uncut buffer
	^{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	Cut buffer	Uncut buffer
	^{15}N zone	
Grass	395 ± 52	338 ± 50
	Buffer	
Grass (composite)	59 ± 10	26 ± 5
Total recovery		
	Cut buffer	Uncut buffer
^{15}N recovered (mg)	2207 ± 522	1891 ± 444
Total recovery (%)†	88 ± 20	76 ± 18

† 2500 mg ^{15}N applied per area

Table 9. Results of linear mixed effects analysis to determine the effect of vegetative buffer treatment, runoff, and grazing management factors on dissolved organic carbon (DOC) (mg/L), total suspended (solids) TSS (mg/L), volatile TSS (VTSS) (mg/L), total phosphorus (P) (mg/L), *E. coli* (cfu/100mL) concentrations and turbidity (ntu) from flood irrigated foothill pastures grazed by beef cattle. Analysis is based upon 432 observations of concentration (8 irrigation events by 9 plots by 6 runoff samples collected per each 2 hour irrigation event) collected across 9, 240 m^2 irrigated pasture – buffer runoff plots located at the UC Sierra Foothill Research and Extension Center in Yuba County, CA. Table continues on next page.

Constituent	Model Factor[†]	Mean (S.E.)[‡]	LME Coefficient (S.E.)[§]	P[¶]
DOC (mg/L)	Buffer Treatment			
	No Buffer Control[#]	11.93 (0.27)	0.00	--
	6:1 Pasture to Buffer	12.82 (0.26)	0.98 (0.78)	0.256
	3:1 Pasture to Buffer	13.59 (0.28)	1.95 (0.78)	0.047
	Runoff Time (min)	--	-0.01 (0.004)	0.004
	Runoff (L/s)	0.84 (0.02)	-3.03 (0.49)	<0.001
	Rest from Grazing			
	2 Days[#]	14.28 (0.21)	0.00	--
	15 Days	11.84 (0.28)	-2.52 (0.25)	<0.001
	30 Days	11.13 (0.27)	-3.04 (0.26)	<0.001
	Fecal Load (kg/ha)	1,831.7 (37.4)	0.0003 (0.001)	0.057
	Intercept	--	15.94 (0.66)	<0.001
lnTSS (mg/L)	Buffer Treatment			
	No Buffer Control[#]	1.34 (0.041)	0.00	--
	6:1 Pasture to Buffer	1.38 (0.072)	0.06 (0.16)	0.696
	3:1 Pasture to Buffer	1.37 (0.071)	0.16 (0.16)	0.354
	Runoff (L/s)	0.84 (0.02)	-1.02 (0.08)	<0.001
	Rest from Grazing			
	2 Days[#]	1.53 (0.063)	0.00	--
	15 Days	1.30 (0.077)	-0.20 (0.08)	0.015
	30 Days	1.14 (0.062)	-0.34 (0.09)	<0.001
	Intercept	--	2.29 (0.13)	<0.001
lnVTSS (mg/L)	Buffer Treatment			
	No Buffer Control[#]	1.04 (0.062)	0.00	--
	6:1 Pasture to Buffer	1.02 (0.060)	0.005 (0.11)	0.964
	3:1 Pasture to Buffer	1.03 (0.058)	0.089 (0.11)	0.464
	Runoff (L/s)	0.84 (0.02)	-0.84 (0.07)	<0.001
	Rest from Grazing			
	2 Days[#]	1.23 (0.051)	0.00	--
	15 Days	0.93 (0.066)	-0.27 (0.07)	<0.001
	30 Days	0.79 (0.054)	-0.40 (0.07)	<0.001
	Intercept	--	1.89 (0.10)	<0.001
Turbidity (ntu)	Buffer Treatment			
	No Buffer Control[#]	0.66 (0.058)	0.00	--
	6:1 Pasture to Buffer	0.89 (0.054)	0.18 (0.40)	0.671
	3:1 Pasture to Buffer	0.91 (0.052)	0.36 (0.39)	0.400
	Runoff Time (min)	--	-0.002 (0.001)	0.099
	Runoff (L/s)	0.84 (0.02)	-0.62 (0.32)	0.056
	Rest from Grazing			
	2 Days[#]	0.94 (0.040)	0.00	--
	15 Days	0.67 (0.063)	-0.22 (0.09)	0.009
	30 Days	0.78 (0.072)	-0.25 (0.1)	0.010
	Fecal Load (kg/ha)	1,831.7 (37.4)	0.0005 (0.0003)	0.081
	Intercept	--	1.16 (0.42)	0.006

ln*E.coli*(cfu/100ml)	Buffer Treatment			
	No Buffer Control[#]	4.27 (0.06)	0.00	--
	6:1 Pasture to Buffer	4.29 (0.06)	0.05 (0.32)	0.874
	3:1 Pasture to Buffer	4.32(0.05)	-0.03 (0.32	0.907
	Runoff (L/s)	0.84 (0.02)	-0.36 (0.06)	<0.001
	Rest from Grazing			
	2 Days[#]	4.59 (0.04)	0.00	--
	15 Days	3.86 (0.06)	-0.85 (0.05)	<0.001
	30 Days	4.26 (0.07)	-0.30 (0.06)	<0.001
	Fecal Load (kg/ha)	1,831.7 (37.4)	0.0007 (<0.0001)	<0.001
	Intercept	--	3.71 (0.24)	<0.001)
P (mg/L)	Buffer Treatment			
	No Buffer Control[#]	0.58 (0.03)	0.00	--
	6:1 Pasture to Buffer	0.37 (0.02)	-0.18 (0.09)	0.094
	3:1 Pasture to Buffer	0.40 (0.02)	-0.14 (0.09)	0.169
	Runoff (L/s)	0.84 (0.02)	-0.10 (0.02)	<0.001
	Rest from Grazing			
	2 Days[#]	0.61 (0.03)	0.00	--
	15 Days	0.39 (0.02)	-0.12 (0.02)	<0.001
	30 Days	0.36 (0.02)	-0.19 (0.02)	<0.001
	Intercept	--	0.74 (0.07)	<0.001

[†] Fixed effect in linear regression model.

[‡] Mean (1 standard error) DOC concentration (mg/L) for each buffer treatment and rest from grazing (day), and mean (1 standard error) runoff rate (L/s) and cumulative fecal load (kg/ha) per irrigation event realized across all 8 irrigation events and 9 plots.

[§] Coefficient (1 standard error) for each significant factor (*e.g.*, buffer treatment, runoff) in linear mixed effects model evaluating relationships between factors and DOC concentration (mg/L). Coefficient value indicates the effect (+ or -) and the magnitude of the relationship between the factor and DOC. For continuous variables (runoff time, runoff rate, and fecal load) the coefficient indicates the change in DOC associated with each incremental unit change in the factor.

[¶] Significance (regression coefficient $\neq 0$) of each factor, estimated using restricted maximum likelihood.

[#] For categorical factors (buffer treatment, rest from grazing), one level of the factor is set as the referent condition (buffer treatment = control, rest from grazing = 2 days) to which all other levels (buffer treatment = 1:6 and 1:3 pasture to buffer area treatments, rest from grazing = 15 and 30 days) of the categorical factor are compared.

Table 10: Results of linear mixed effects analysis to determine the effect of vegetative buffer treatment, runoff, and grazing management factors on dissolved organic carbon (DOC) (kg/ha), total suspended solids TSS (kg/ha), volatile TSS (VTSS) (kg/ha), total phosphorus (P) (kg/ha), *E. coli* (cfu/ha) load from flood irrigated foothill pastures grazed by beef cattle. Analysis is based upon 72 observations of DOC flux (8 irrigation events over 9 plots) collected across 9, 240 m^2 irrigated pasture – buffer runoff plots located at the UC Sierra Foothill Research and Extension Center in Yuba County, CA. Table continues on next page.

Constituent	Model Factor[†]	Mean (S.E.) [‡]	LME Coefficient (S.E.) [§]	P[¶]
DOC (kg/ha)	Buffer Treatment			
	No Buffer Control[#]	358.0 (23.2)	0.00	--
	6:1 Pasture to Buffer	438.6 (32.1)	43.4 (30.2)	0.201
	3:1 Pasture to Buffer	518.2 (33.6)	75.7 (30.7)	0.0488
	Runoff (m^3/ha)	351.3 (12.7)	1.2 (0.09)	<0.001
	Rest from Grazing			
	2 Days[#]	470.4 (28.2)	0.00	--
	15 Days	404.4 (34.0)	-73.6 (16.3)	<0.001
	30 Days	418.9 (36.6)	-87.1 (17.1)	<0.001
	Intercept	--	19.4 (34.9)	0.579
lnTSS (kg/ha)	Buffer Treatment			
	No Buffer Control[#]	2.53 (0.13)	0.00	--
	6:1 Pasture to Buffer	2.88 (0.12)	0.27 (0.15)	0.120
	3:1 Pasture to Buffer	2.98 (0.12)	0.25 (0.15)	0.156
	Runoff (m^3/ha)	351.3 (12.7)	0.003 (0.001)	<0.001
	Rest from Grazing			
	2 Days[#]	2.98 (0.11)	0.00	--
	15 Days	2.71 (0.12)	-0.28 (0.14)	0.048
	30 Days	2.56 (0.15)	-0.50 (0.14)	0.001
	Intercept	--	1.78 (0.21)	<0.001
lnVTSS (kg/ha)	Buffer Treatment			
	No Buffer Control[#]	2.17 (0.11)	0.00	--
	6:1 Pasture to Buffer	2.42 (0.11)	0.15 (0.12)	0.265
	3:1 Pasture to Buffer	2.56 (0.12)	0.17 (0.12)	0.223
	Runoff (m^3/ha)	351.3 (12.7)	0.003 (0.001)	<0.001
	Rest from Grazing			
	2 Days[#]	2.57 (0.10)	0.00	--
	15 Days	2.27 (0.10)	-0.31 (0.11)	0.007
	30 Days	2.16 (0.13)	-0.50 (0.12)	<0.001
	Intercept	--	1.38 (0.17)	<0.001
ln*E.coli* (cfu/ha)	Buffer Treatment			
	No Buffer Control[#]	1.15 (0.15)	0.00	--
	6:1 Pasture to Buffer	1.21 (0.15)	0.09 (0.31)	0.766
	3:1 Pasture to Buffer	1.31 (0.14)	0.07 (0.31)	0.842
	Rest from Grazing			
	2 Days[#]	1.51 (0.10)	0.00	--
	15 Days	0.76 (0.14)	-0.83 (0.13)	<0.001
	30 Days	1.21 (0.18)	-0.27 (0.13)	0.058
	Fecal Load (kg/ha)	1,831.7 (37.4)	0.0006 (<0.0001)	<0.001
	Intercept	--	0.42 (0.33)	0.205

lnP (kg/ha)	Buffer Treatment			
	No Buffer Control[#]	1.19 (0.06)	0.00	--
	6:1 Pasture to Buffer	1.08 (0.05)	-0.14 (0.10)	0.237
	3:1 Pasture to Buffer	1.13 (0.06)	-0.13 (0.10)	0.275
	Runoff (m^3/ha)	351.3 (12.7)	0.0015 (0.0002)	0.0002
	Rest from Grazing			
	2 Days[#]	1.25 (0.07)	0.00	--
	15 Days	1.09 (0.04)	-0.16 (0.05)	0.005
	30 Days	1.07 (0.05)	-0.21 (0.5)	0.0005
	Intercept	--	0.93 (0.12)	<0.001

† Fixed effect in linear regression model.

‡ Mean (1 standard error) DOC flux (kg/ha) for each buffer treatment and rest from grazing (day), and mean (1 standard error) runoff rate (m^3/ha) per irrigation event realized across all 8 irrigation events and 9 plots.

§ Coefficient (1 standard error) for each significant factor (*e.g.*, buffer treatment, runoff) in linear mixed effects model evaluating relationships between factors and DOC flux (kg/ha). Coefficient value indicates the effect (+ or -) and the magnitude of the relationship between the factors and DOC. For continuous variables (runoff time, and runoff rate) the coefficient indicates the change in DOC associated with each incremental unit change in the factor.

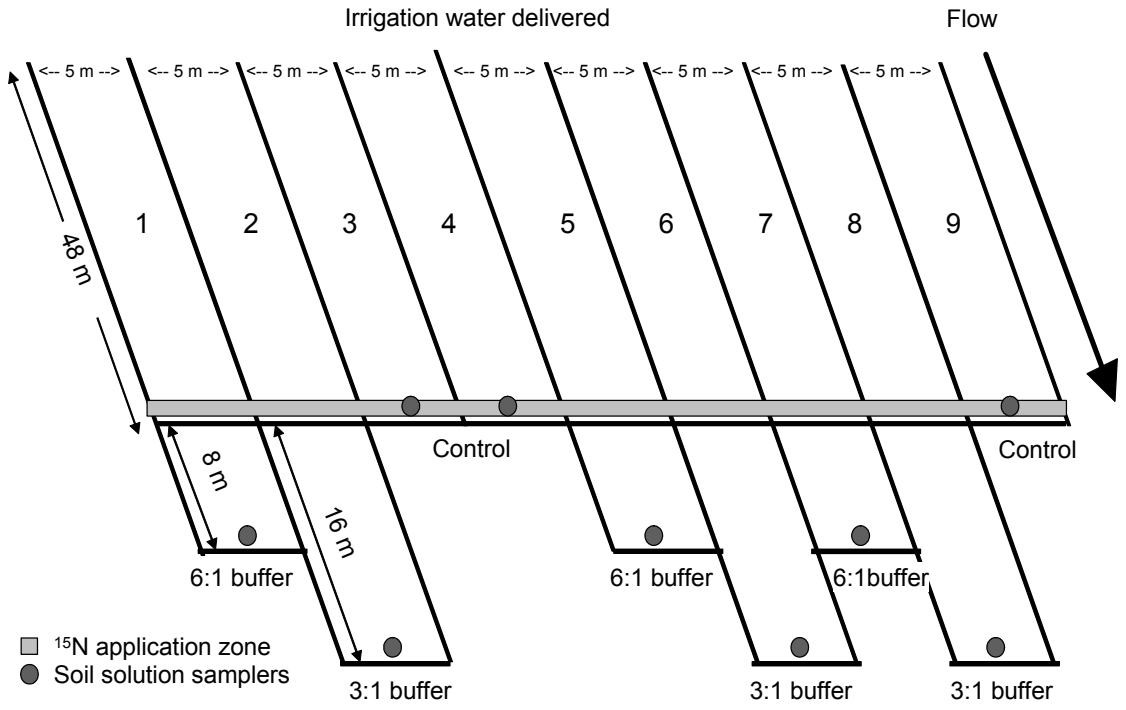

Figure 1: Schematic of Study Site A plot design (not to scale). Collection troughs installed at the bottom of each treatment (downslope of solution samplers).

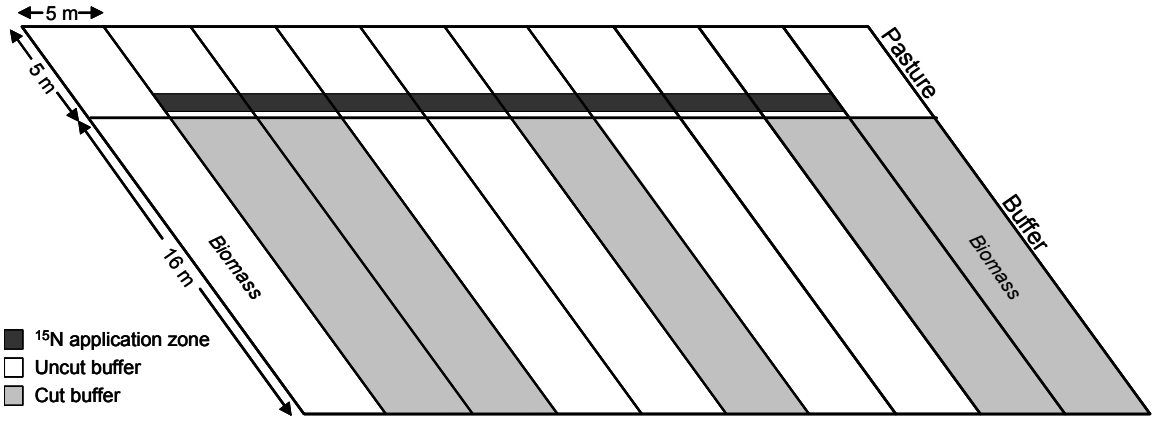

Figure 2: Schematic of Study Site B plot layout. Not to scale. "Biomass" buffers received no ^{15}N and were used to get quantitative estimates of aboveground biomass.

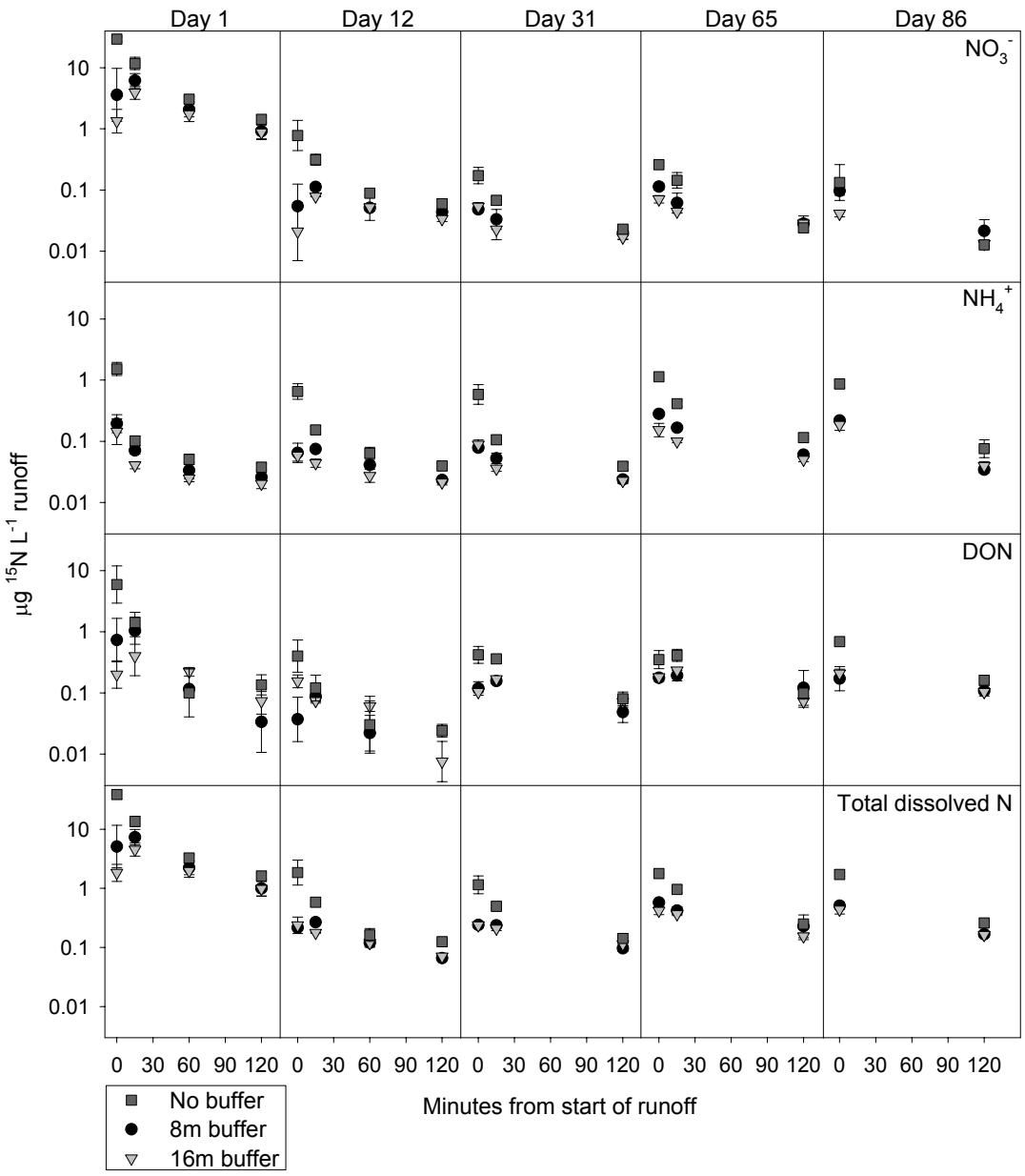

Figure 3: The ^{15}N concentrations within and between irrigations. Values are averaged by buffer treatment and time; error bars represent standard error. Note log y-axis.

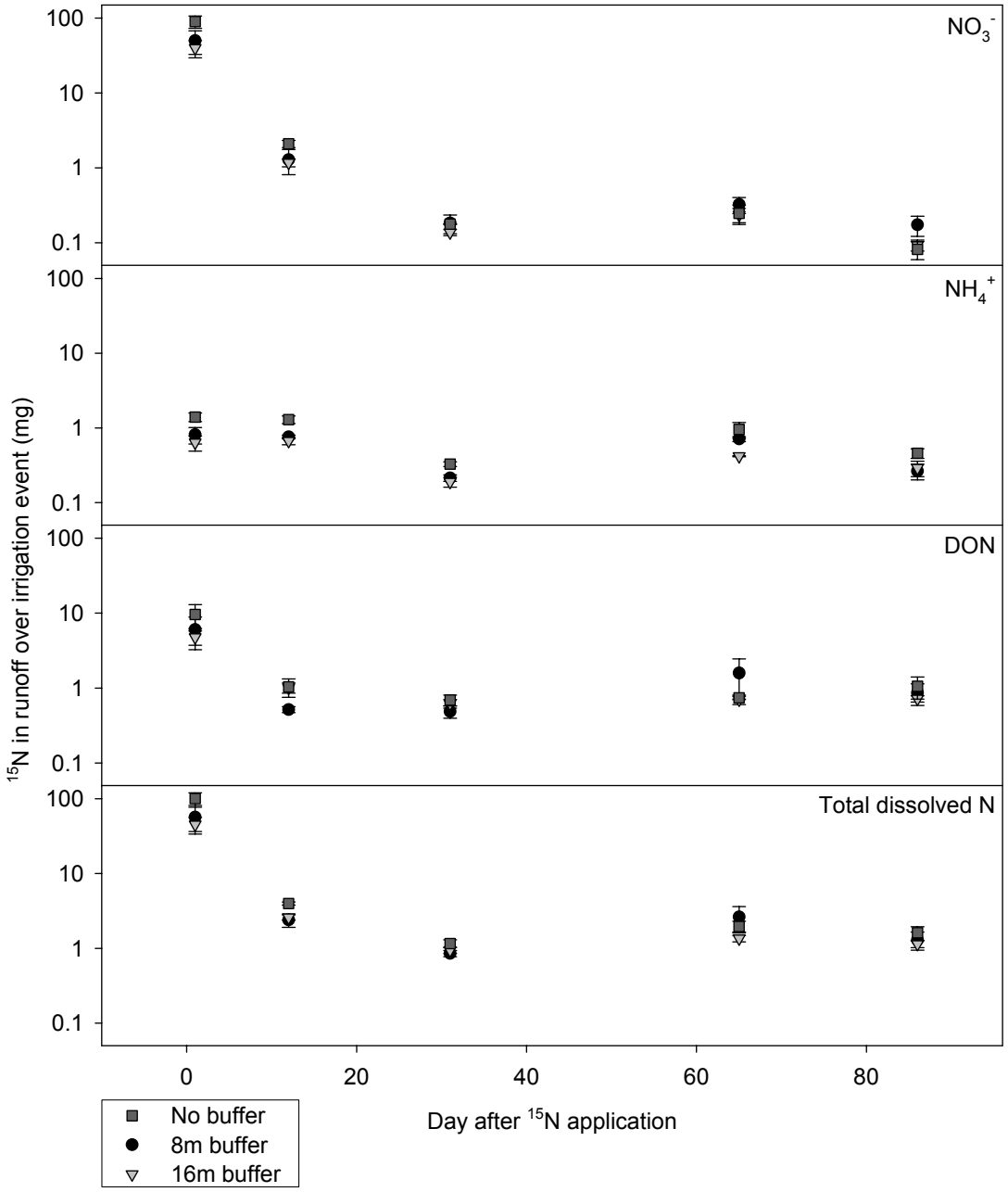

Figure 4: The ^{15}N load over the course of the summer. Values are averaged by buffer treatment and time; error bars represent standard error. Note log y-axis.

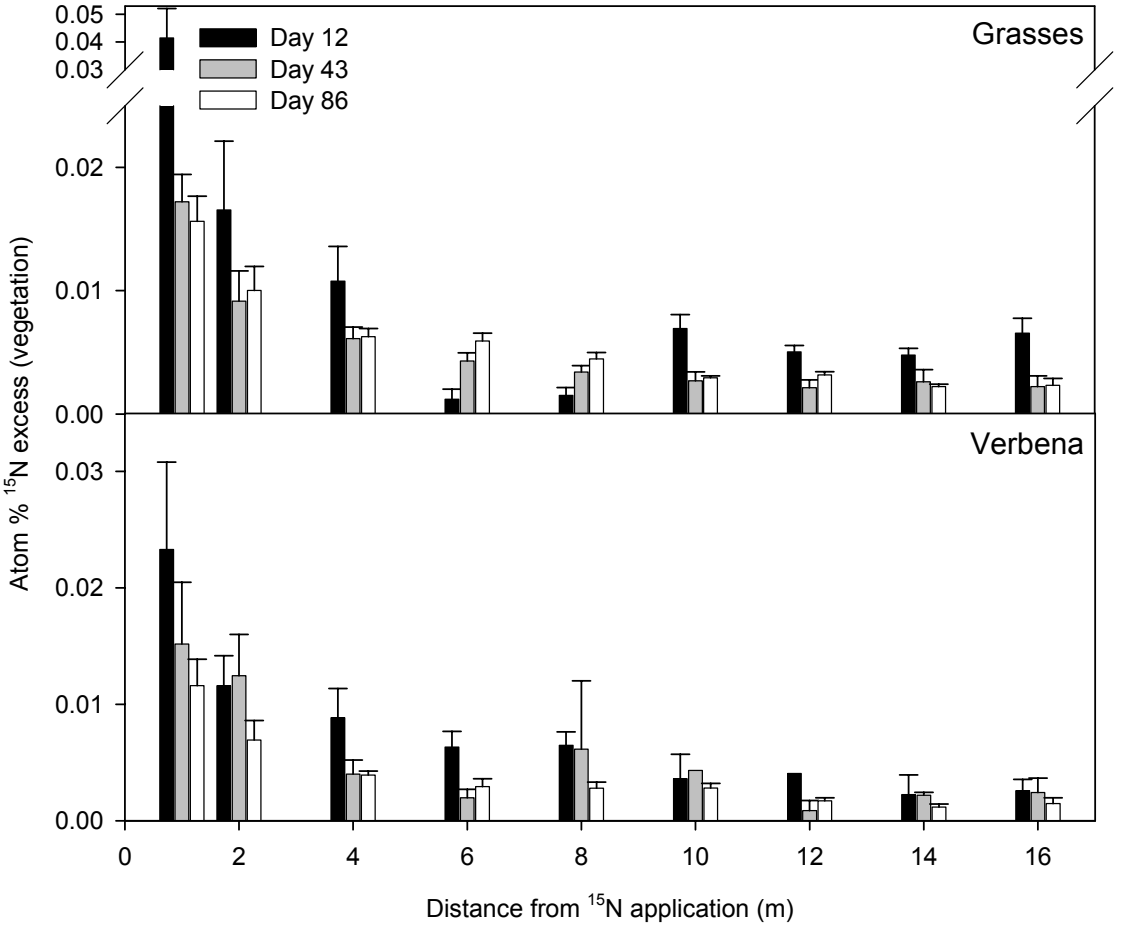

Figure 5: Atom % ^{15}N excess in vegetation by distance. Values are averaged by time and distance across all treatments; error bars represent standard error. Data from the ^{15}N application zone not shown here due to graphical limitations.

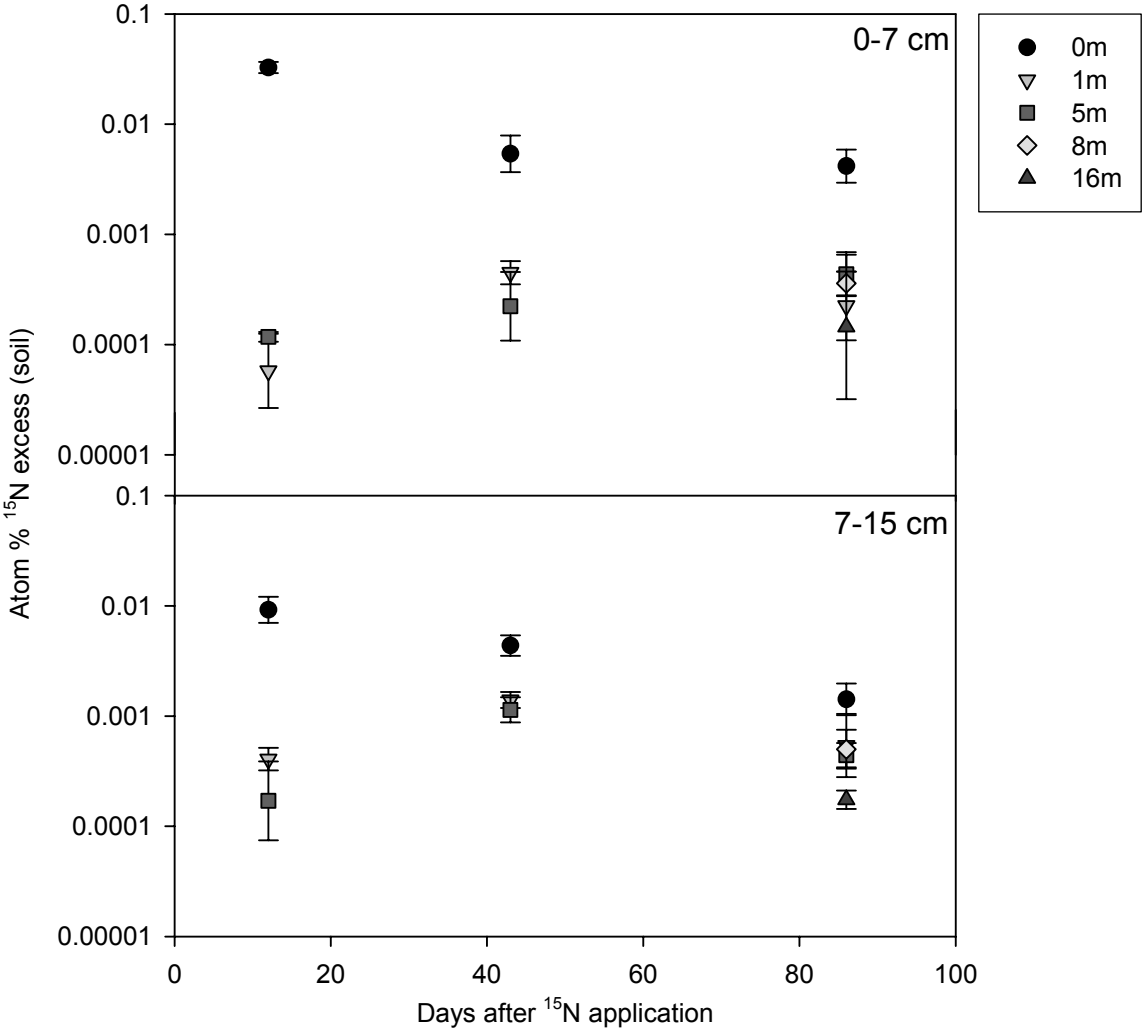

Figure 6: Atom % ^{15}N excess in soils by time. Values are averaged by time and distance across all treatments; error bars represent standard error. 8 m and 16 m data only available for Day 86.

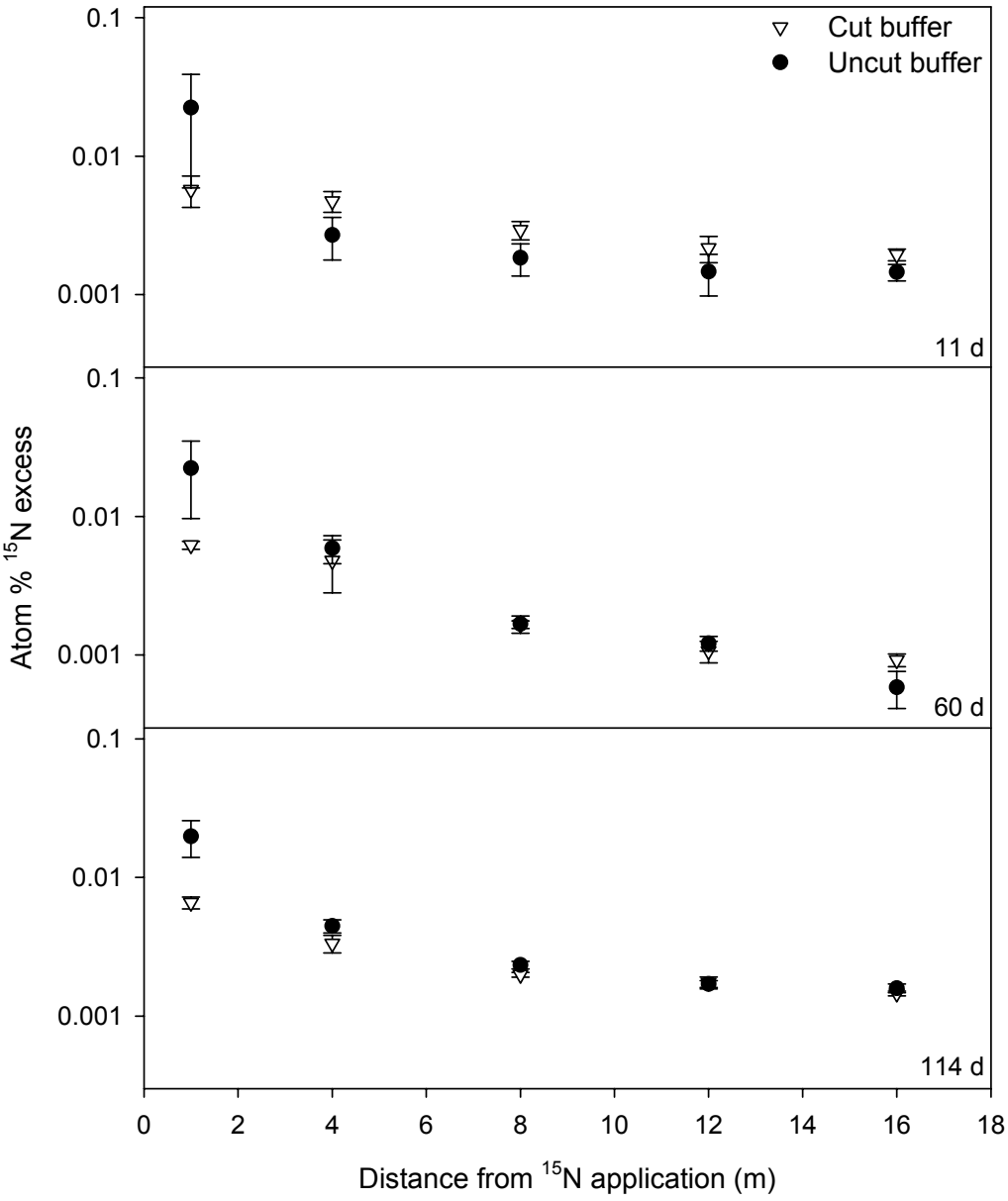

Figure 7: Atom % ^{15}N excess in buffer vegetation by distance from ^{15}N application (averages, standard error bars). From top to bottom, days after ^{15}N application = 11 d, 60 d, 114 d. Note log y-axis.

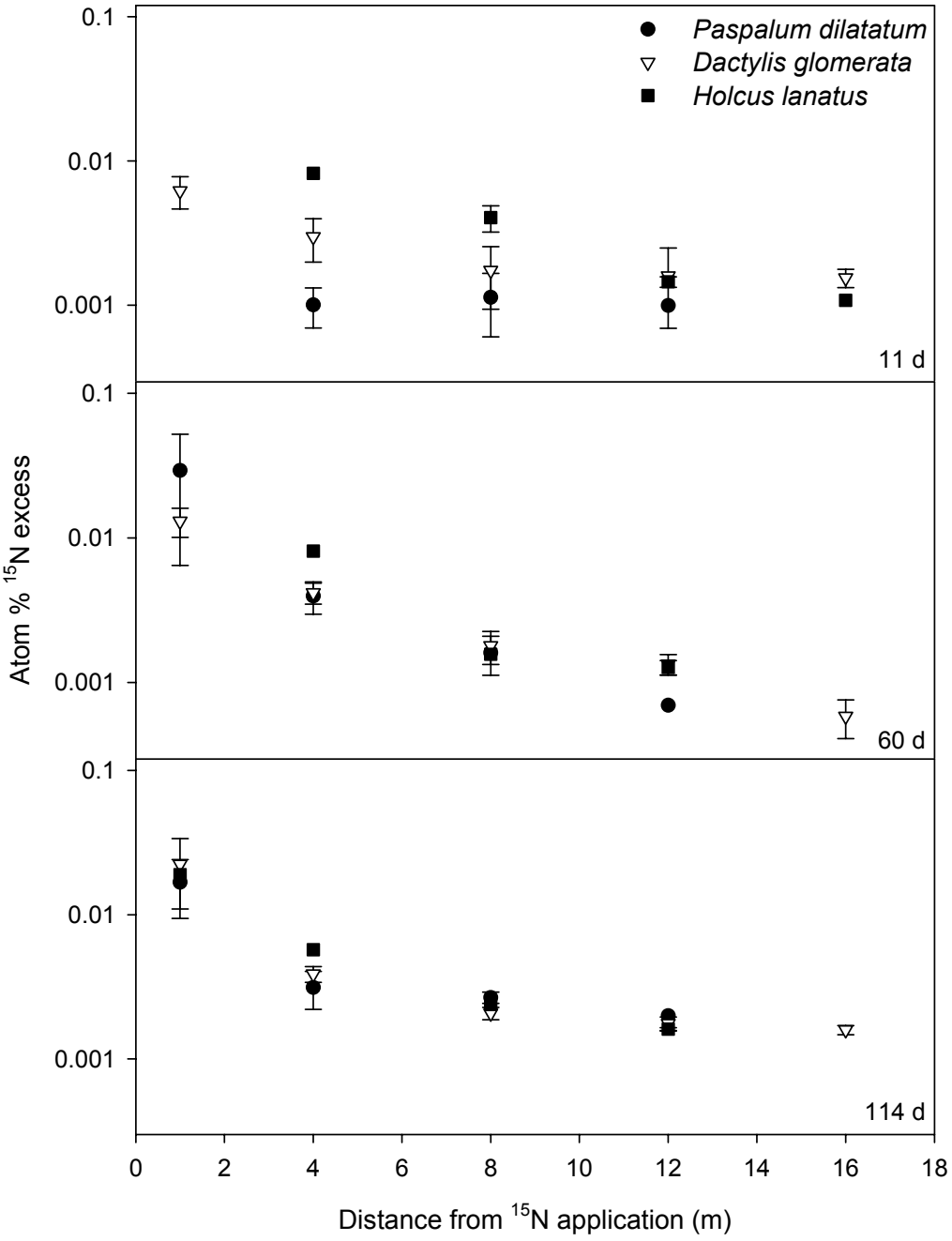

Figure 8: Atom % ^{15}N excess for each of the three dominant species in uncut buffers by distance from ^{15}N application (averages, standard error bars). From top to bottom, days after ^{15}N application = 11 d, 60 d, 114 d. Note log y-axis. Not all species were present at all distances in measurable amounts for each sampling date.

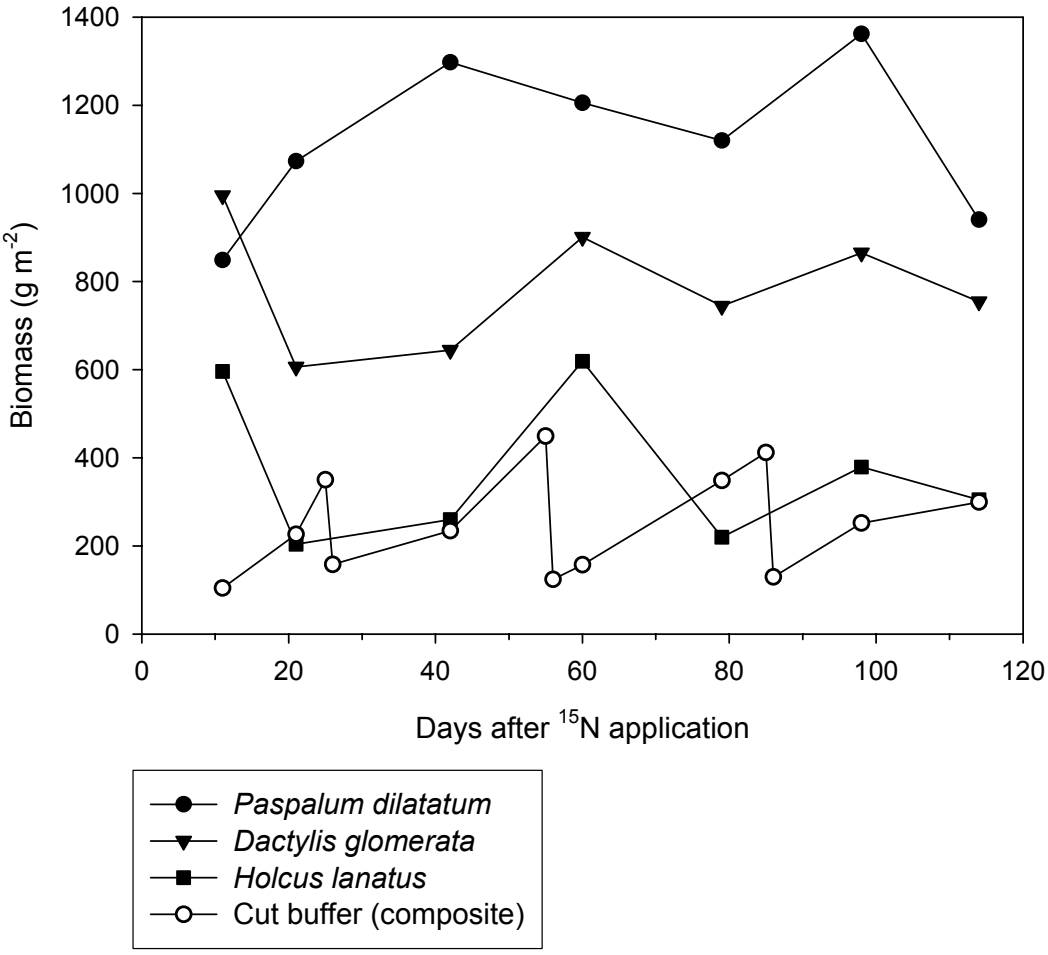

Figure 9: Vegetation biomass (g/m^2) by time since ^{15}N application for each of the three dominant species within the uncut buffers and for a composite of all species present per m^2 within the cut buffers.

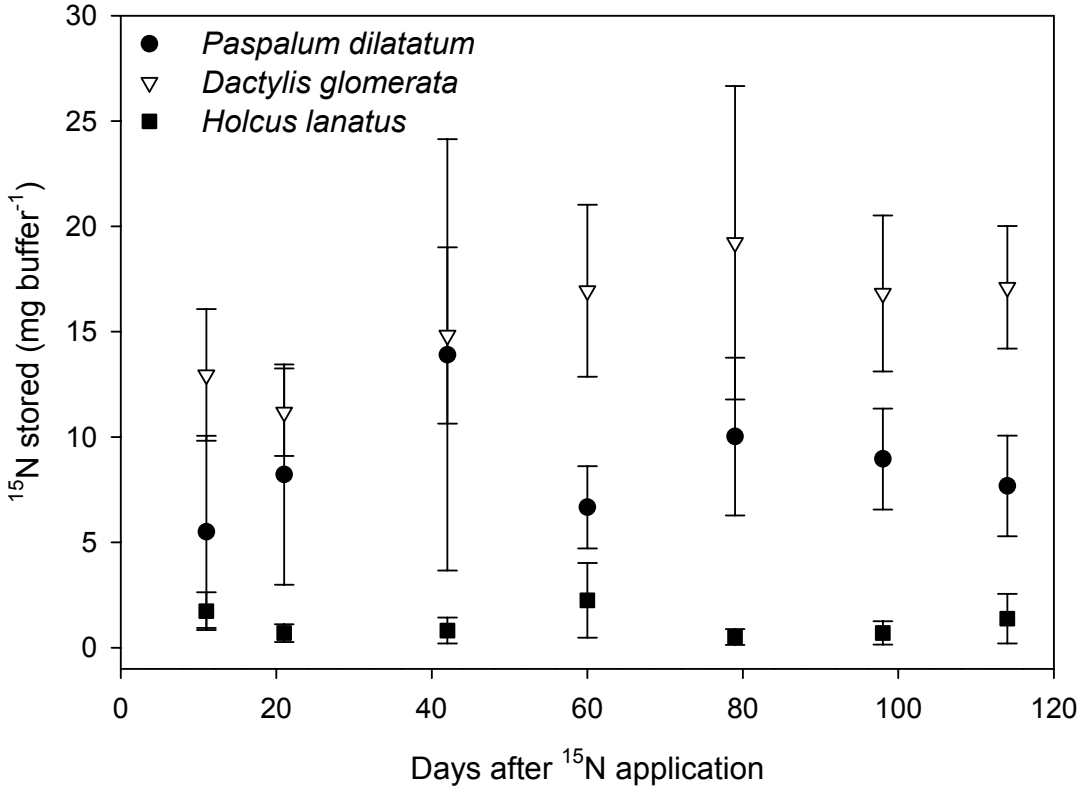

Figure 10: Total mass (mg) ^{15}N sequestered by each of the three dominant species in uncut buffer areas by time since ^{15}N application.

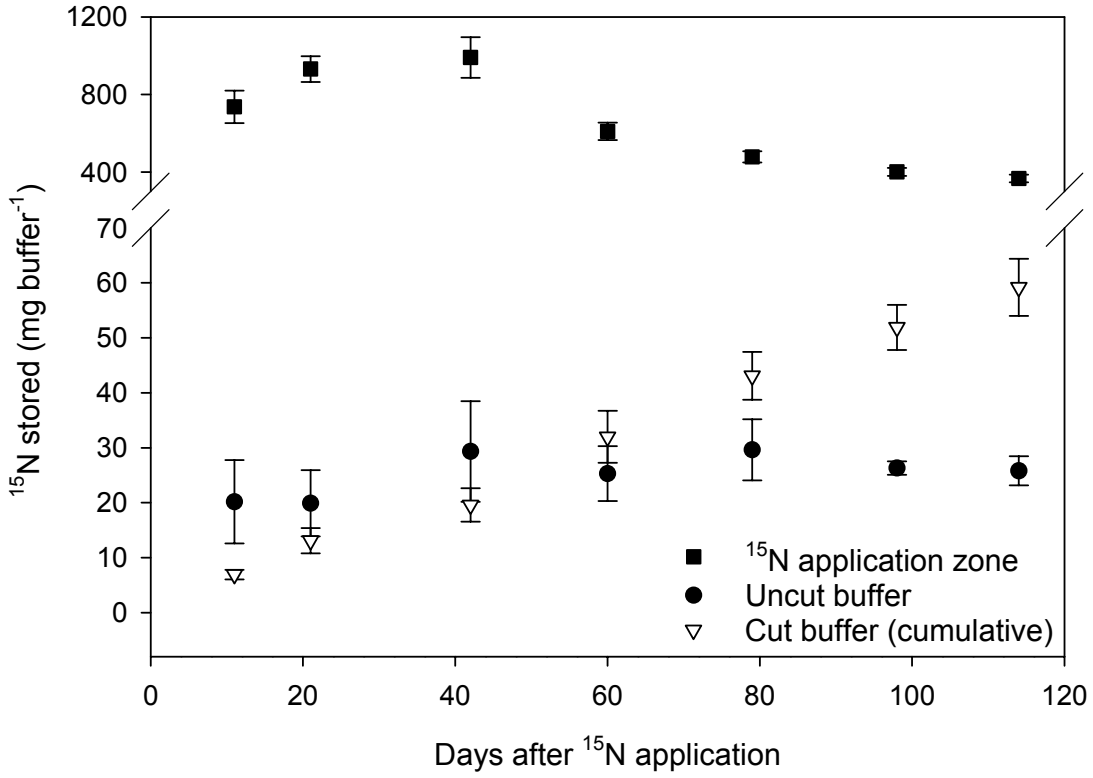

Figure 11: Total mass (mg) ^{15}N sequestered in a given buffer area by time since ^{15}N application for pasture areas, uncut buffer areas, and cut buffer areas, where pasture and cut buffer areas are cumulative, including ^{15}N removed by clipping during the irrigation season.

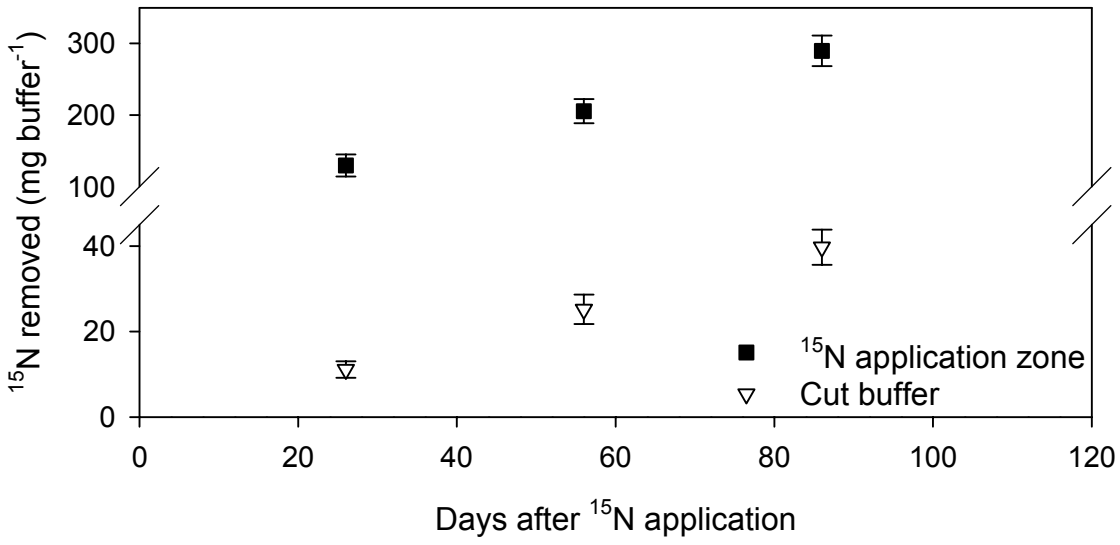

Figure 12: Total cumulative mass (mg) ^{15}N removed from pasture and cut buffer areas by clipping during the irrigation season.

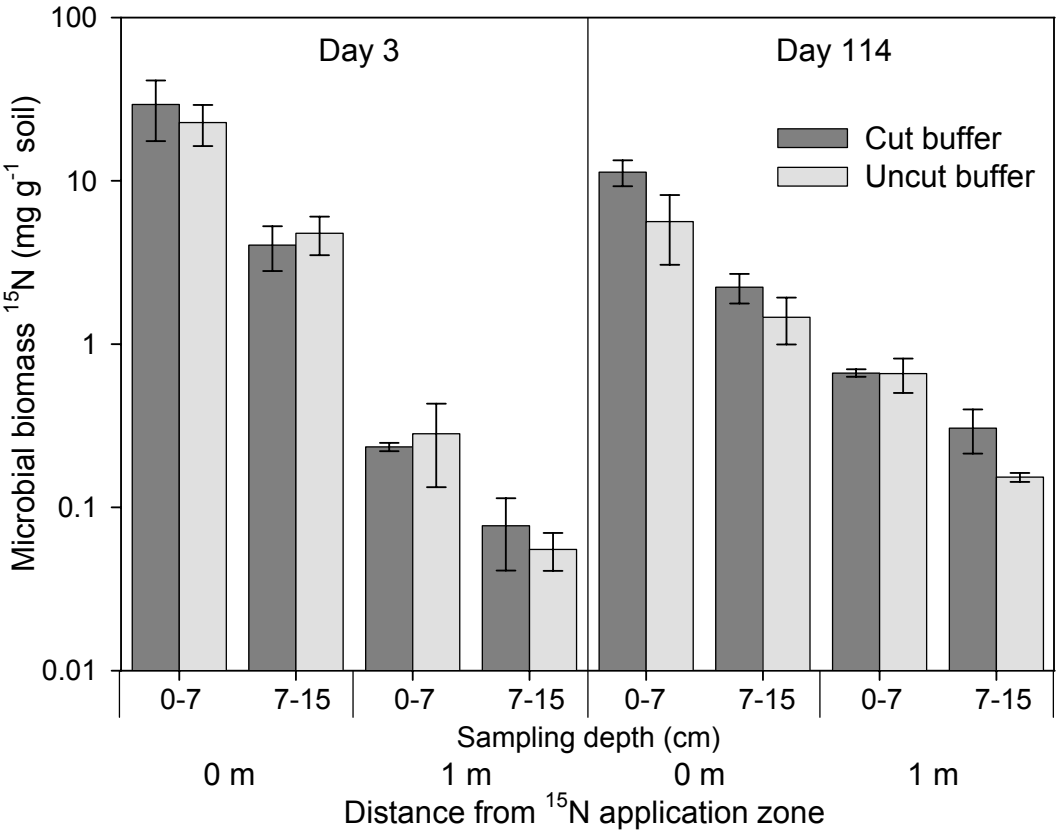

Figure 13: Soil microbial biomass ^{15}N (mg ^{15}N/g soil) by depth, distance from ^{15}N application, and time since ^{15}N application. Note log y-axis.

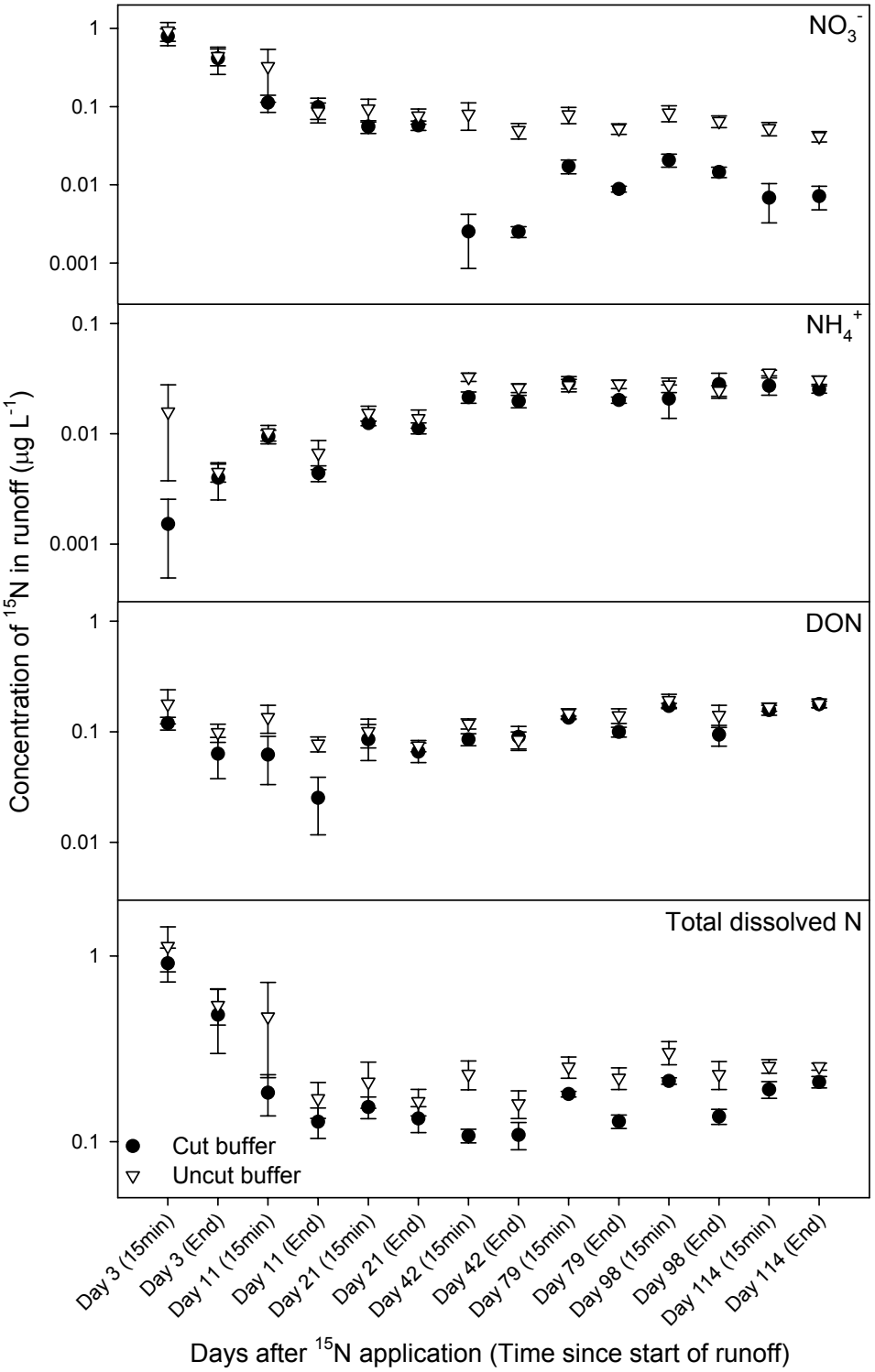

Figure 14: Runoff ^{15}N concentrations within and between irrigations. Values are averaged by buffer treatment and time; error bars represent standard error. Note log y-axis and differences in y-scale between N pools.

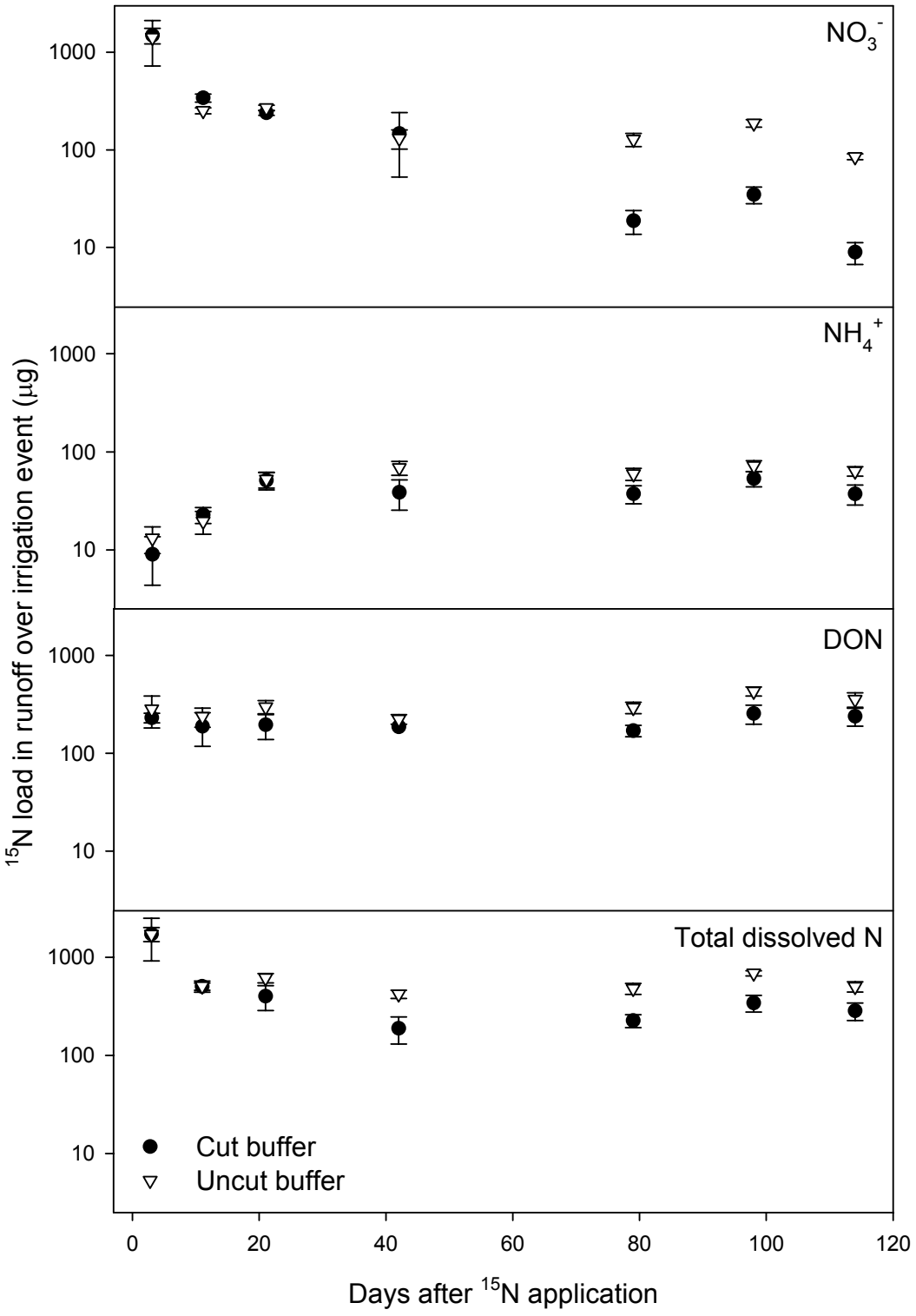

Figure 15: Runoff ^{15}N load over the course of the irrigation season. Values are averaged by buffer treatment and time; error bars represent standard error. Note log y-axis.

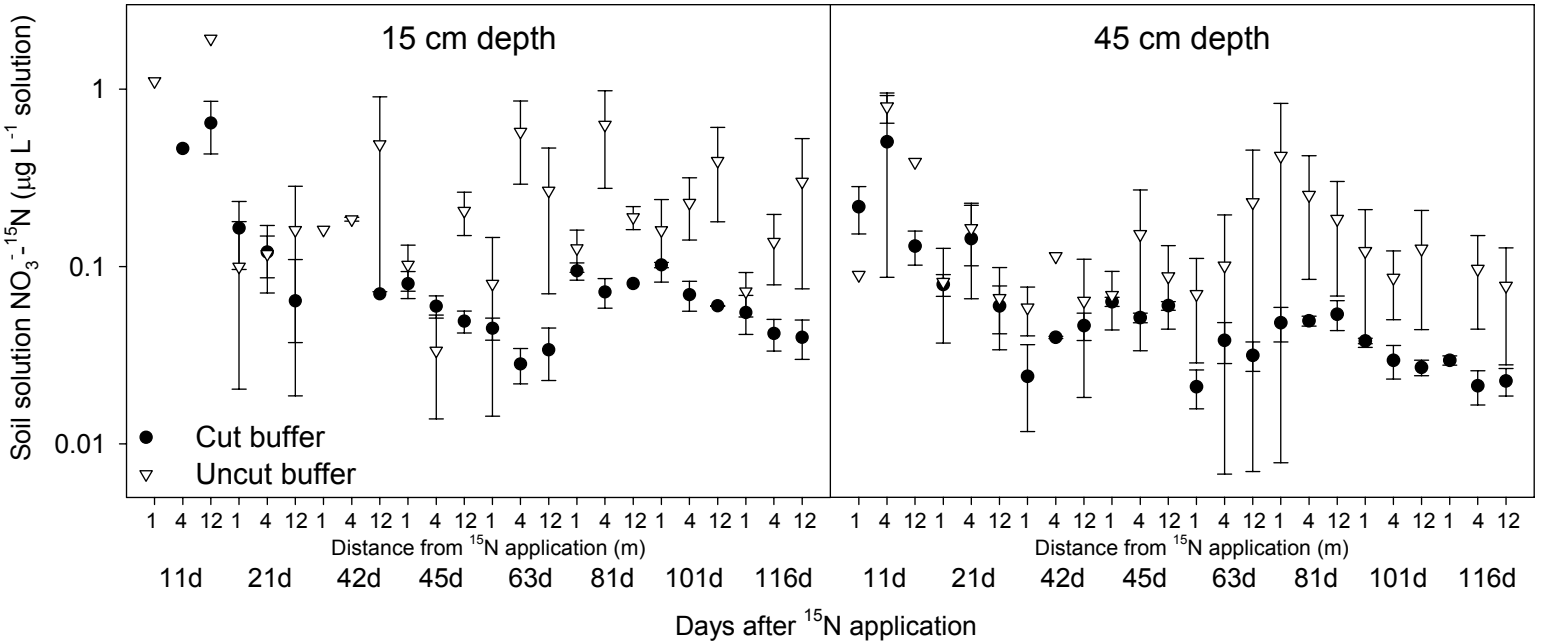

Figure 16: Soil solution NO_3-^{15}N concentrations by time and distance from ^{15}N application. Values are averaged by buffer treatment, time, and distance; error bars represent standard error. Note log y-axis.

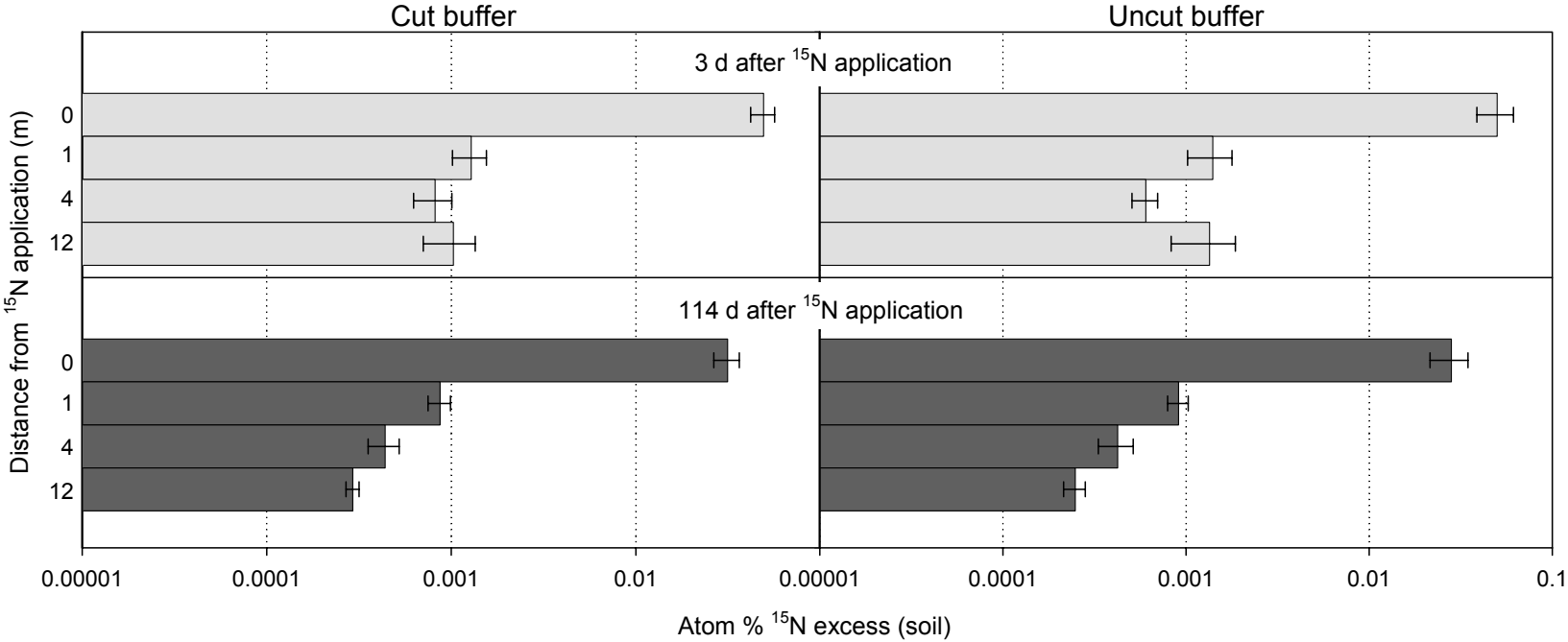

Figure 17: Atom % ^{15}N excess in soils by distance from ^{15}N application at 3d (top) and 114 d (bottom) after ^{15}N application. Values are averaged by treatment and distance; error bars represent standard error

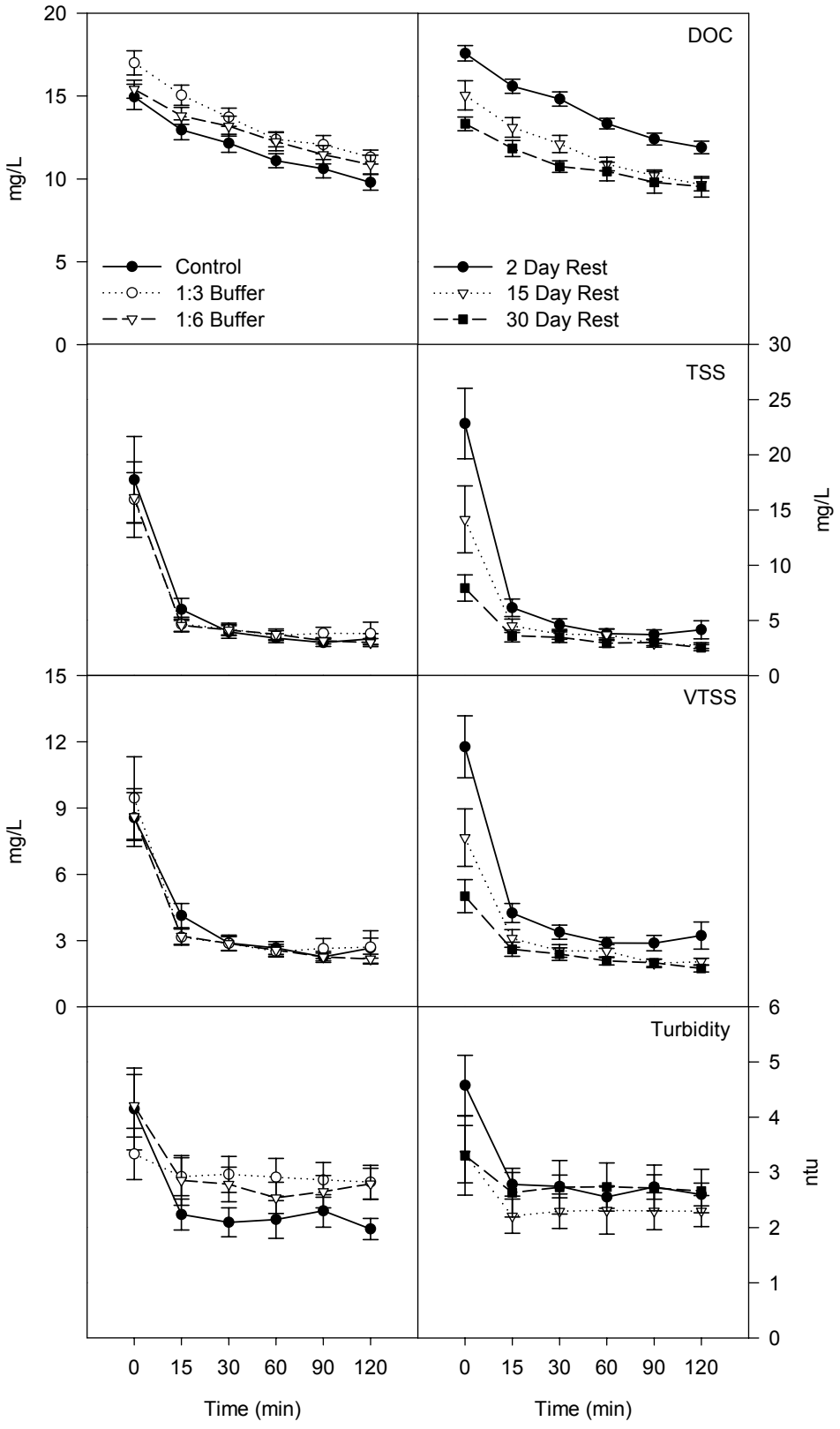

Figure 18. Mean (+/- SE) DOC, TSS concentration and turbidity for buffer and rest from grazing treatments across all plots and sample collection times from 8 trials.

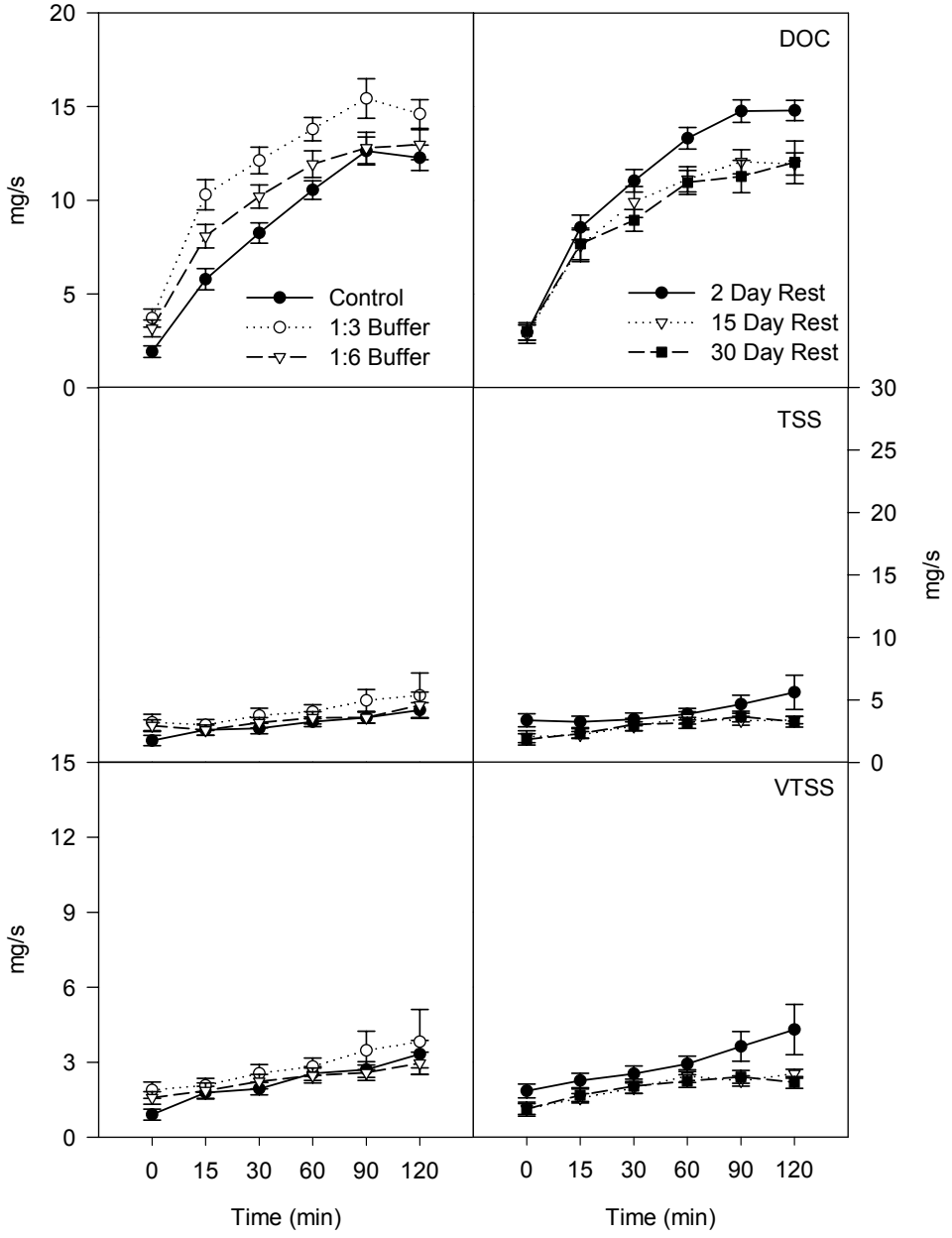

Figure 19. Mean (+/- SE) instantaneous load for DOC and TSS for buffer and rest from grazing treatments across all plots and sample collection times from 8 trials.

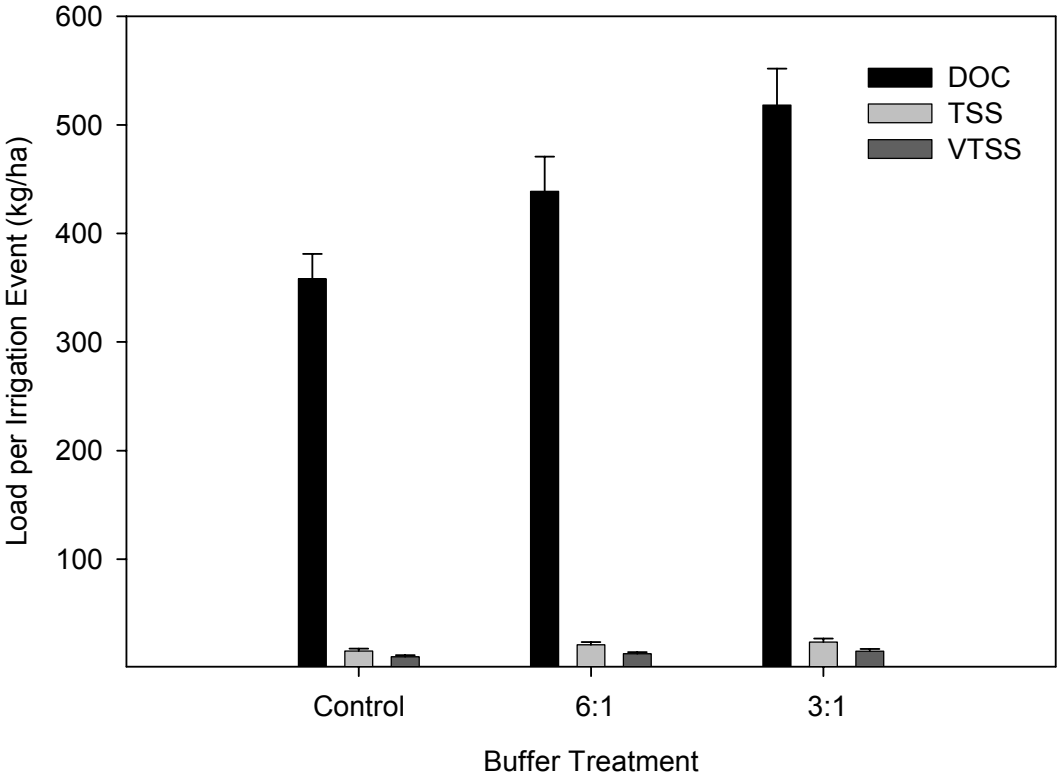

Figure 20: Mean (+/- SE) DOC and TSS flux for buffer treatments across all plots and sample collection times from 8 trials.

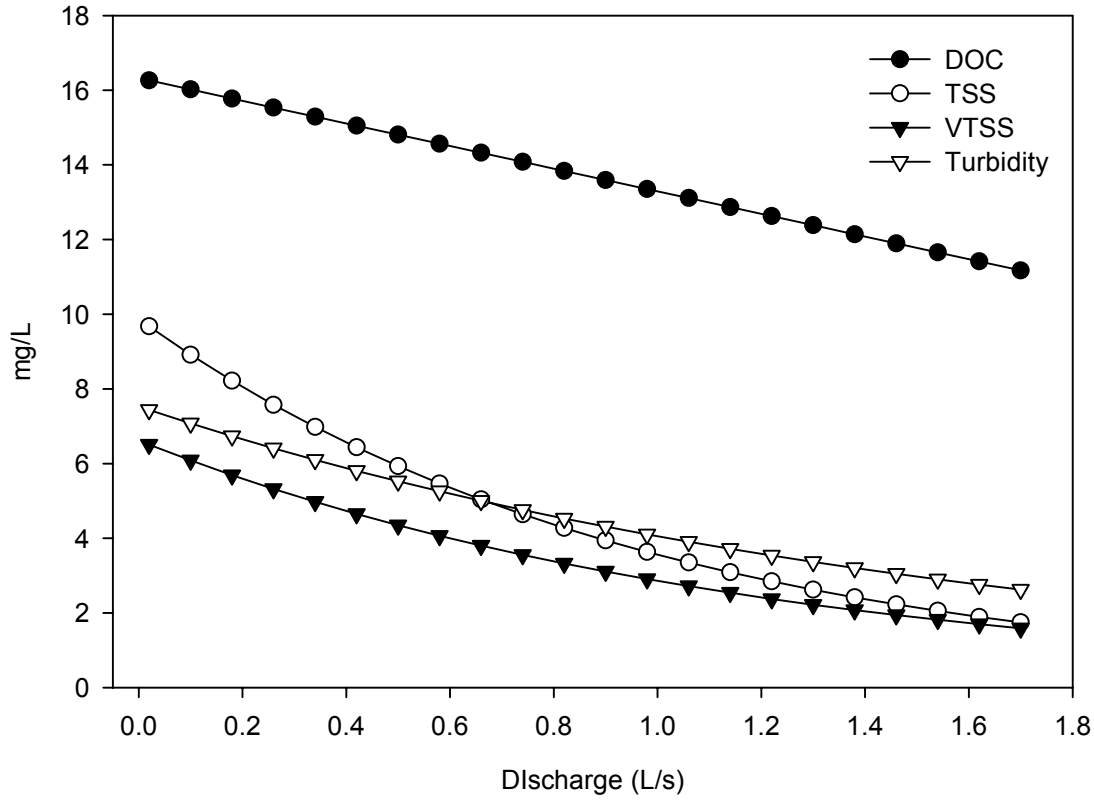

Figure 21: Illustration of DOC, TSS, and turbidity concentration LME model from Table 9.

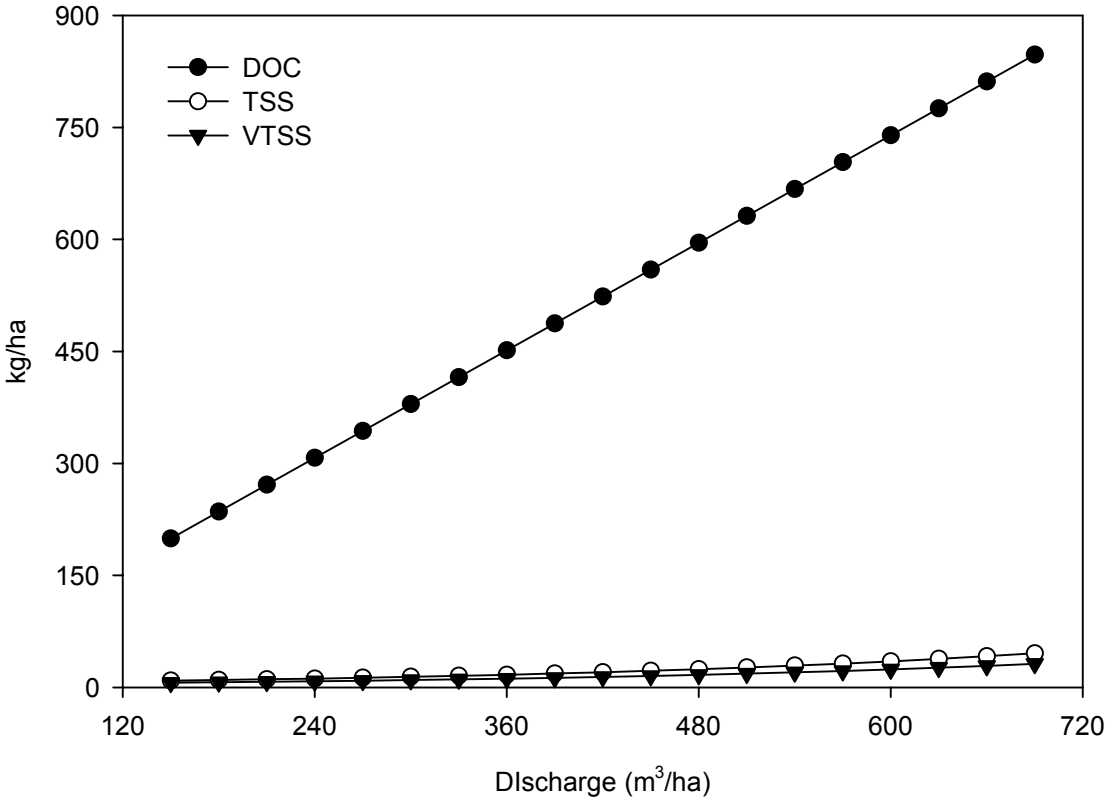

Figure 22: Illustration of DOC and TSS load LME model from Table 10.

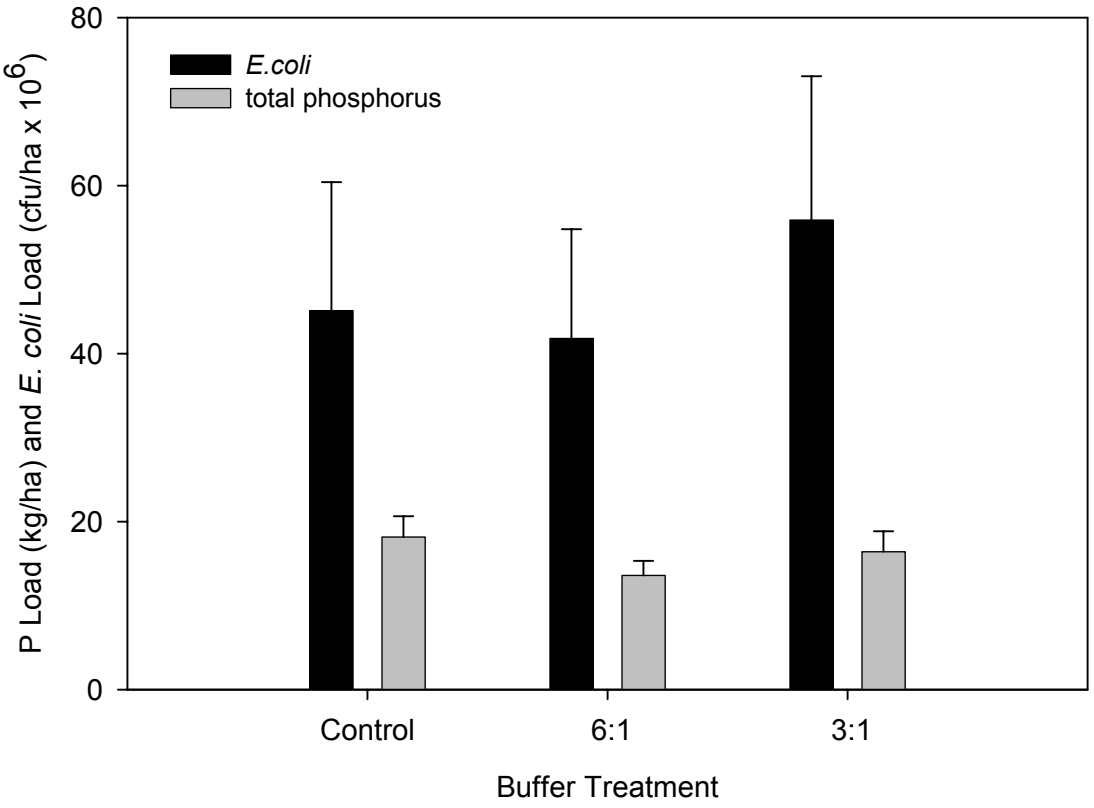

Figure 23: Mean (+/- SE) total phosphorus and *E. coli* load for buffer treatments across all plots and sample collection times from 8 trials.