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Impacts of Cattle Grazing as a Tool to Control *Phragmites australis* in Wetlands on Nitrogen, Phosphorus, and Carbon

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IMPACTS OF CATTLE GRAZING AS A TOOL TO CONTROL PHRAGMITES AUSTRALIS IN WETLANDS ON NITROGEN, PHOSPHORUS, AND CARBON

by

Brittany L. Duncan

A thesis submitted in partial fulfillment
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ABSTRACT

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by

Brittany L. Duncan, Master of Science

Utah State University, 2018

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Department: Watershed Sciences

While grazing in wetlands has been a great tool to decrease *Phragmites australis* biomass, many concerns remain about possible negative impacts to wetland nutrient cycling and ecosystem functioning. Despite the potential for large amounts of nutrient loading from fresh manure, we did not detect any significant increases of nutrients in the water, soil, or leaves in the grazed plots when compared to the control plots. Instead, we found that grazing had a temporal suppressive effect, where many nutrient levels did not increase as much over time in the grazed plots when compared to the control plots. We hypothesized this occurred for two primary reasons. First, when *Phragmites australis* was stressed, it was likely more rapidly assimilating nutrients to regrow. Secondly, the hoof action of the cattle mixing soil slurries was likely increasing oxygen within the soils, and thus, causing a shift from anaerobic to aerobic microbial processes within the soil. Another surprising finding was that soil carbon decreased by approximately half, which

is concerning since carbon storage is a primary wetland ecosystem function. Other research showed that carbon loss would have likely occurred with any restoration activities, and that the drying of the land alone likely contributed to this loss.

In the end, we concluded that, with careful water management, grazing over a short term will likely have little negative impacts on nutrient levels, except for carbon, within wetlands while also significantly decreasing *Phragmites australis* biomass.

(76 pages)

PUBLIC ABSTRACT

Impacts of Cattle Grazing as a Tool to Control *Phragmites australis* in Wetlands on Nitrogen, Phosphorus, and Carbon

Brittany L. Duncan

Phragmites australis is a plant that is causing problems in wetlands by outcompeting native plants that provide food and shelter for millions of migratory birds. Currently, managers try to control *Phragmites australis* by spraying herbicide, burning, and mowing, but these methods are costly, time consuming, and have low levels of success. Adding grazing as a tool to control *Phragmites australis* provides a cheap and low labor alternative. However, there are many concerns regarding if grazing will cause nutrient loading in our wetlands that will decrease water quality and alter beneficial functions of wetlands.

To better understand the effects of grazing in wetlands, we proposed a two-year study and received funding from many organizations including the Utah Department of Fire, Forestry, and State Lands, South Davis Sewer District, and the Utah Department of Environmental Quality and Water Quality. Also, the Utah Department of Natural Resources helped tremendously in allowing access to the sites, in the actual implementation of the project, coordinating with local ranchers who allowed for their cattle to be in the study, managed their cattle during the study, and assisted with fence installation, and many volunteers from Utah dedicated hunters helped with the fence installation.

We collected water, manure, soil, and leaf samples over time to analyze nutrient changes and measured changes in the plants, water levels, soil cover, and litter cover over time. We then compiled and analyzed this information to better understand how grazing impacts our wetlands. As a result, we were able to make some recommendations for future research and how best to graze in wetlands with minimal impacts according to the information we found.

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INTRODUCTION

The problem

Phragmites australis (Phragmites) was introduced from Eurasia to North America in the early 19th century (Lambertini, Sorrell, Riis, Olesen, & Brix, 2012; Saltonstall, 2002). Its invasion has resulted in negative impacts on North American wetlands including a decrease in native plant cover in many wetlands, corresponding loss of avian habitat, and other changes in ecosystem functioning (Chambers, Meyerson, & Saltonstall, 1999; Kettenring & Mock, 2012). To restore or conserve impacted wetlands, major efforts have been enacted to remove or control the spread of *Phragmites* (Hazelton, Mozdzer, Burdick, Kettenring, & Whigham, 2014; Martin & Blossey, 2013).

The most commonly used approach to controlling *Phragmites* is herbicide (Hazelton et al., 2014) because it can kill both above- and belowground parts of the plant if used correctly. Use of herbicide is problematic, however, because it requires multiple years of treatment (3+ years) and spot treatments are needed to prevent reinvasion (Hazelton et al., 2014). Herbicide is very costly and often not very effective at controlling *Phragmites* growth and spread in the long term (Martin & Blossey, 2013). In addition, native plant re-establishment post treatment is constrained by light limitation associated with dense standing dead stems and high litter levels produced by *Phragmites* (Minchinton, Simpson, & Bertness, 2006; Christine B. Rohal, 2018). As an alternative, herbicide use in combination with biomass removal approaches such as mowing or burning that remove or decrease litter can enhance recovery of native wetlands (Hazelton et al., 2014). However, mowing and burning for biomass removal in conjunction with

herbicide application has some drawbacks. Although highly effective for removing biomass, burning is typically not appropriate to utilize near urban centers. Most problematic are the air quality impacts, which are particularly notable in our study region with poor summertime air quality(C. B. Rohal, Kettenring, Sims, Hazelton, & Ma, 2018). Mowing does not have the same negative air quality impacts, but it is expensive, time consuming, and logistically difficult to get machinery into wetlands (Martin & Blossey, 2013; DWR Land Managers, *pers. comm.*; Rohal et al. 2018).

An alternative approach: Grazing

Cattle grazing has emerged as an alternative biomass removal tool that can potentially be used in conjunction with an herbicide application sequence or as a singular treatment approach (Hazelton et al., 2014). Grazing is attractive to managers seeking alternative, less expensive tools and addresses many of the challenges associated with mowing and burning. Grazing may decrease seed production and plant biomass, while stressing underground rhizomes and thereby minimizing further spread of *Phragmites.* Furthermore, cattle trampling may help break down the large amounts of dead litter accumulation in *Phragmites* stands and provide more light for plant growth (Ágoston-Szabó, Dinka, Némedi, & Horváth, 2006; Asaeda, Nam, Hietz, Tanaka, & Karunaratne, 2002; Costantini & Rossi, 2010; Schaller et al., 2013). Several studies show that grazing is effective at removing biomass, although this has not been well-studied as a broad-scale management treatment in North America (Hazelton et al. 2014).

However, the potential negative impacts of grazing *Phragmites* with cattle in wetlands have not been investigated. Cattle have the potential to impact wetlands through

changes to water quality, soil bioavailable nutrients, and nutrient pools in leaves, manure, and soil, namely through increased nutrient loading via excretion by cattle. The act of grazing could change soil nutrient inputs coming directly from the plants, transform nutrients from plants to more labile forms in manure, and remove nutrients from the ecosystem through assimilation, respiration, and other gaseous emissions. In the soil, grazing could alter many biogeochemical and physical processes that could affect nutrient cycling and nutrient levels within the soils. Furthermore, research is needed to determine if nutrients may be exported off site through natural drainage and further exacerbate nutrient issues in downstream waters such as Farmington Bay. Also, given that *Phragmites* is a high nutrient specialist (Mozdzer & Zieman, 2010) and numerous experiments have shown increased growth, reproduction, and occurrence of *Phragmites* under elevated nutrient conditions (Kettenring, McCormick, Baron, & Whigham, 2011; King, Deluca, Whigham, & Marra, 2007; Minchinton & Bertness, 2003; Saltonstall & Court Stevenson, 2007), cattle may have unintended consequences for future *Phragmites* invasions by concentrating nutrients in wetland soils. Nevertheless, the impacts of grazing on nutrient loading in Great Salt Lake wetlands have not been evaluated. It is important to assess changes in plant available soil nutrients pre- and post-grazing to determine if nutrients reach high levels that have been shown to promote invasion of non-native species, such as *Phragmites* (James, Drenovsky, Monaco, & Rinella, 2011; Kettenring & Adams, 2011; King et al., 2007; Rickey & Anderson, 2004; Saltonstall & Court Stevenson, 2007).

This research aims to answer the following questions and inform management decisions associated with using grazing as a tool to control *Phragmites*:

1. **Does grazing affect water quality?**

Expectation: Manure and urine nutrient inputs associated with grazing will increase the following nutrients, total dissolved nitrogen (hereafter TDN), dissolved organic nitrogen (hereafter DON), nitrate (hereafter NO_3-N), ammonium (hereafter NH_4-H) and soluble reactive phosphorus (hereafter SRP-P) in flood water and (soil) pore water that could potentially lead to downstream eutrophication.

2. **Does grazing alter soil bioavailable nutrients or biogeochemical drivers of nutrient cycling?**

Expectation: Grazing will cause a significant increase in bioavailable nutrients (NO_3-N and ortho-phosphate) in soils, and manure will contain a higher concentration of the nitrogen heavy isotope, $\delta^{15}N$, to trace the contribution of manure to bioavailable nutrients in soil, water, and plants.

3. **Will grazing alter carbon to nitrogen and carbon to phosphorus (hereafter C: N and C:P) nutrient ratios within *Phragmites* itself? Will nitrogen to phosphorus (hereafter N:P) ratios within *Phragmites* reveal changes in soils nutrient limitations?**

Expectation: Grazing will cause a significant drop in C: N and C:P ratios as *Phragmites* may compensate by re-sprouting after being grazed by increasing nitrogen (hereafter N) and phosphorus (hereafter P) assimilation for regrowth. N:P ratios will

show that the soil is N limited, but the grazed site will have higher N:P values than the control.

4. Does grazing alter overall nutrient pools in leaves, manure, and soils?

Expectation: Grazers will decrease nutrient pools of total nitrogen (hereafter TN), total phosphorus (hereafter TP), and total carbon (hereafter TC) by assimilating nutrients into grazer biomass, but overall grazing will largely transform nutrients into more bioavailable forms.

METHODS

Sites, grazing, and design

We established five paired 10.12 ha grazed and ungrazed plots in Utah's Great Salt Lake wetlands. One pair was located near the Crystal Unit at the Farmington Bay Waterfowl Management Area (FBWMA) (Table 1, Fig. 1). Two pairs of plots were located at Howard Slough WMA (HSWMA), and two pairs at Harold Crane WMA (HCWMA; Table 1, Fig. 1). All sites were near impoundments with water control structures, which allowed managers to control wetland water level inflow but not the outflow. Water flow was cut off to sites prior to grazing— to dry the land for the health and safety of the cattle—and flooded after the cattle were removed in the fall. At the start of the experiment, all plots were comprised of *Phragmites australis* with an overall minimum cover of approximately 80%.

Each of the five grazed plots was fenced with 3-strand barbed-wire fencing, while ungrazed plots were not fenced. Each year, grazing intensity on each grazed plot equaled

approximately 30 animal unit months (AUM) for 3-12 weeks depending on site conditions (i.e. water levels and availability of forage). The 30 AUMs consisted of 25 cow-calf pairs and occasionally 1 bull at Farmington Bay and the Howard Slough sites, and 30 yearling heifers and 1 bull grazed at Harold Crane. Grazing occurred during the summers of 2015 and 2016. At each site we collected samples and measurements during at least 1 of the following timeframes each year depending on relevance: pre-grazing, during grazing, and post-grazing (Table 2). For example, manure was only collected during grazing because that is when the cows were on the plots, and flood water was only taken post-grazing because that was when the sites were flooded.

All grazed and ungrazed plots, contained a mowed 100 m long corridor leading from plot edge to a 0.40ha circular area at the center of the plot (Fig. 2). In grazed plots, the mowed corridor allowed cattle access to the mowed circular area that served as an accessible area (in otherwise nearly impenetrable *Phragmites*) where cattle could ruminate.

Field sampling

We collected samples to assess water quality at one of our five sites, FBWMA. To assess nutrient pools across sites, we collected additional manure, soil, and leaf samples at all five sites, as well as soil bulk density samples, plant biomass samples, and other plant measurements. All but the water quality and manure samples were collected along 11m transects that were placed in permanent locations across the plot and revisited over time (Table 2, Fig. 3). The number of transects varied by grazing and mowing treatments to best capture the variability in plant responses.

Water quality

Field water sampling

At FBWMA, we collected water from 3 sources: the impounded pond (the source of water for the sites, which we assumed to be the same for both grazed and ungrazed sites), flood water, and well water. For the pond water, we collected 2 samples near the ungrazed and 3 samples near the grazed plot. We collected floodwater samples from 8 locations in the grazed plot and 8 locations in the ungrazed plot, 3 of which were in the mowed and 5 in unmowed areas.

For well water, we installed 16 (8 grazed, 8 ungrazed; including 3 mowed, 5 unmowed), 7.62cm PVC wells with slits, 0.92m deep. We collected pond and floodwater samples using the grab sample method (Danielson, 2014). We drained wells twice before grabbing a sampling from them. We filtered all water samples with 0.7µm ashed glass fiber filters within 24hrs of sampling. We froze the filtrate until sent to lab for nutrient analysis.

We collected well water and pond water pre- and post-grazing in both years (Table 2). We collected floodwater in the post-grazing timeframe in both years (Table 2).

Manure and soil leaching

From FBWMA, we used a portion of 15 manure samples and a portion of 15 (6 ungrazed—3 mowed and 3-unmowed; and 9 grazed—3 mow and 6 unmowed) soil samples that were also used for nutrient analysis discussed above. We mixed 25g of

sample (either soil or manure) in 500ml deionized water agitated on a table shaker for 5 minutes to mimic floodwater inundation and movement. We let the slurries settle for 10 minutes and then filtered the decanted liquid with a 0.7µm ashed glass fiber filter. We froze the filtered water until sent to the lab for nutrient analysis. We used samples from post-grazing 2016 soils and during-grazing 2016 manure (Table 2).

Manure

For manure nutrient sampling, from each grazed plot we collected 15 fresh (warm and moist) manure to minimize nutrient transformation. The samples were grouped, homogenized. A small portion was dried at 105°C overnight, weighed, ground with a Wiley Mill, and stored in a vial with a screw cap for analysis of TN, $\delta^{15}N$, TC, and $\delta^{13}C$ concentrations. The rest of the sample was frozen for TP concentration and leachate analysis.

We collected manure while the cattle were on the plots (during grazing) in 2015 and 2016 (Table 2). In 2015, we were unable to collect manure from FBWMA because the cattle were removed from the plot due to high water levels before the collection could occur, and in 2016, we were unable to collect from one of the Howard Slough WMA plots because high water levels resulted in the cattle being put on site late and removed early before collection could occur.

Soil

We collected 6 cores along the transects used for leaf/plant collections (Fig. 3). During coring, we removed the litter layer, cored, removed the O-horizon, and kept 30cm

of the remaining core. Then, we used a soil sieve to homogenize the soil and remove roots. We dried a portion of this soil between 105°C for approximately 24hrs to calculate soil moisture levels and ground it for TC, TN, $\delta^{15}N$, and $\delta^{13}C$ analysis. We mixed about 1 tablespoon of moist, sieved soil in a 0.5 M sodium bicarbonate solution to extract P. Finally, we mixed about 2 tablespoons of moist, sieved soil in a 2M KCL solution for extracting NO_3-N. The weight of the cup, the solution plus the cup, the solution plus the cup and the soil sample was recorded, Finally, the soil moisture levels for each soil sample was used with the weights to calculate the final nutrient concentrations.

In addition to collecting soil for nutrient analysis, we collected an additional core per transect for bulk density measurements (Fig. 3, Table 2). We dried the core at 105°C for approximately 48hrs before weighing.

We took soil samples for nutrients, soil moisture, and bulk density in pre- and post-grazing timeframes in both 2015 and 2016 (Table2).

Phragmites

We collected leaf samples, plant measurements (stem density, flowering density, percent cover) and plant biomass samples along our original sampling transects (Fig. 3). In addition, we added new transects throughout the growing season in the grazed plots as newly grazed areas emerged. Along each of the transects, three 1m x 1m quadrats were placed at 3m intervals and used as our focal sampling areas (Fig. 3).

To evaluate nutrients in leaves, we collected one of the top three leaves of a plant. We did this for 12 plants evenly spaced plants per transect and homogenized the leaves (Fig. 3). We dried the samples overnight at 50-60°C. We used a Wiley mill to grind the

dried leaves and stored the samples in vials with screw caps until sent for nutrient analysis.

To upscale leaf biomass to site-level assessments of nutrient pools and to evaluate changes in *Phragmites* primary production with grazing, we collected biomass samples by clipping all the stalks of *Phragmites* in multiple 1x1m^2 areas along each transect location (Fig. 3). These samples were dried at 50-60 °C for >48 hours and then weighed.

For leaf nutrient analysis, we collected in pre- and post-grazing timeframes in both 2015 and 2016 (Table 2). For plant biomass, we collected these samples only in post-grazing 2016 because of the destructive nature of the sampling (Table 2).

Chemical analysis for nutrients

We tested all water samples (collected in field and leachates) for TDN, NH_4-N, NO_3-N , and SRP-P through Dr. Michelle Baker's Aquatic Biogeochemistry Lab at Utah State University using an Astoria Pacific Autoanalyzer (Baker, 2011).

Manure, soil, and leaves were tested for TN, $\delta^{15}N$, TC, and $\delta^{13}C$ concentrations at the University of Utah Stable Isotopes for Environmental Research (SIRFER) Laboratory (Salt Lake City, UT). Soils collected in 2015 were digested with 1 N HCl until a pH of 4 was achieved (in 2016, we did not measure TC and $\delta^{13}C$ in the soils).This process converted soil carbonates to CO_2 gas, resulting in a sample with bioavailable C. Samples were rinsed multiple times with deionized water and dried overnight at 65 °C. Subsamples of leaves, soil and manure were analyzed for total C and N, as well as isotopic C and N ratios ($\delta\ ^{13}C$, $\delta\ ^{15}N$), with a Thermo Finnigan Delta Plus isotope ratio mass spectrometer connected to a Carlo Erba 1005 EA via Thermo Finnigan Conflo III.

Percent elemental content (%C, %N) was calculated based on area under curve (chromatograph) and calculated K factor derived from the known internal lab reference materials. Stable isotope values for the samples were determined by normalizing with respect to the lab internal standards. The SIRFER lab uses a two-point normalization model with a third point as quality control. The standard uncertainty for stable isotope analysis was ± 0.2 permil for both carbon and nitrogen isotopes and 0.3% for both %C and %N. The internal lab reference materials are calibrated against internationals reference materials USGS 40 and USGS 41. The following equation was used to determine the raw delta (δ) values for samples:

$$\partial X(‰) = \left(\frac{\partial X_{sample} - \partial X_{standard}}{\partial X_{standard}}\right) \times 1000,$$

where X is ^{13}C or ^{15}N (Peterson 1999).

Manure P was analyzed by the Utah State University Analytical Lab using a Thermo iCAP 6300 ICP-OES (Gavlak, Horneck, & Miller, 2005). Soil NO_3-N and TP were analyzed by the USDA Range Research lab using a spectrophotometer and a Lachate Flow injection analyzer (Knepel, 2012; Olen, n.d.). Leaf P was analyzed by University of California-Davis analytical lab using an automated Lachat Flow Injection Analyzer (Prokopy, 1985).

Statistical analysis and mathematical computations

Objective 1: Water nutrient changes and implications for water quality

Field water sampling

Flood/pond ratio analysis

We analyzed nutrient concentrations of TDN, DON, NO_3-N, NH_4-N, SRP-P in flood/pond water and wells. We divided flood nutrient values by an average of the pond nutrient values. If the resulting value is > 1, the flood water contains more nutrients than the pond water and vice versa. To normalize our data, we transformed it using the natural log (ln).

DON was calculated by the following formula:

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

If the DON values were negative, it was likely that there was little to no DON in the sample. To adjust for the negative values, we assigned half of the lower detection limit for TDN (0.005) to those values. Also, if any nutrient concentrations were below detection limits, we assigned that sample at half of the lower detection limit for that nutrient. We built a 3-way factorial ANOVA model with the following fixed effects: grazing treatment (2 levels), mowing treatment (2 levels), and season (2 levels: post_2015 and post_2016) and all interactions. Also, an outlier was dropped that incorrectly provided low TDN and DON values and high NO_3-N, NH4, and SRP values.

Pore water analysis

To analyze pore water, we conducted 2 different mixed model ANOVAs. The first model compared the grazed and ungrazed plots over time. The fixed effects were the grazing treatment (2 levels), the mowing treatment (2 levels), and season (3 levels: post-2015, pre-2016, and post-2016; we dropped pre-grazing 2015 because we did not have data for the ungrazed) and the interactions among these effects. The random effect was the number of wells nested within the grazing and mowing treatment categories. Data were natural log (ln) transformed except for SRP-P, which was square root transformed. For values that were below the detection limit, we assigned a value that was half of the lower level detection limit. Also 2 outliers were dropped that clearly were incorrect values.

The second model compared all four time periods within the grazing treatment. The fixed effects were the mowing treatment (2 levels), the year (2 levels; 2015 and 2016), and the season (2 levels; pre- and post-grazing). The random effect was the number of wells nested within the grazing and mowing treatment. TDN, DON, NH4N and NO3N were log transformed to meet model assumptions. SRP-P was not transformed, and one outlier was removed for the SRP-P analysis.

Manure and soil leachate study

We analyzed nutrient levels of soil leachate using a 2-factor ANOVA model. The fixed effects were grazing treatments (2 levels), mowing treatment (2 levels), and the interactions between the two. Soil TDN and NO_3 leachate data were square-rooted

transformed. Soil DON, NH_4-N, and SRP-P leachate data were transformed using natural log. We scaled up manure and soils nutrients in the leachate to site levels by converting the leachate nutrient values from mg/l in the solution to g of nutrient leached per g of sample. We scaled up manure nutrient leachate by multiplying the grams of nutrient leached per gram of sample by the estimated grams of manure produced on site per season. We scaled up soil nutrient leachates by multiplying grams of nutrient leached per gram of soil sample by the averaged bulk density per site. Calculations for site-level manure production and soil volume are discussed further in Objective 4: Manure and Soil sections below.

Objective 2: Soil nutrient bioavailability

Soil nutrient ratios

For C:N ratios, we only had a complete data set for post-grazing 2015. The fixed effects were grazing treatment (2 levels; grazed and ungrazed) and mowing treatment (2 levels; mowed and unmowed) and the interaction between grazing and mowing treatments. The model had plots (10 levels) and its interactions as the random effects.

Soil NO_3-N

The fixed effects were the grazing treatment (2 levels), mowing treatment (2 levels), year (2 levels, 2015 and 2016), and season (2 levels, pre and post) and the interactions between these factors. Plot (10 levels) and its interactions with sites were the random effects. Soil NO_3-N data was transformed using a natural log function to normalize the data.

Soil $\delta^{15}N$

We conducted a variety of statistical tests with our $\delta^{15}N$ data. First, we constructed a model with fixed effects of sample type (3 levels; leaves, manure, and soils), year (2 levels, 2015 and 2016), and the interactions between them to assess differences between sample types over time. The years included only post-grazing for the soils. The random effects included the sites (5 levels; Harold Crane WMA 1 and 2, Howard Slough WMA west and east, and FBWMA) and its interactions.

Next, we investigated whether there was a difference in the soils due to treatment over time; that model included fixed effects of grazing treatment (2 levels), mowing treatment (2 levels), season (3 levels, post-2015, pre-2016, and post-2016), and the all possible interactions among the fixed effects. The random effects were plot (10 levels) and its interactions.

Finally, we examined how $\delta^{15}N$ in soils changed from pre- to post-grazing in 2016 only. The model included the fixed effects of grazing treatment (2 levels), mowing treatment (2 levels), and season (2 levels). The random effects were plot (10 levels) and its interactions.

Soil P

For soil P, we used the same model as we used for soil NO_3-N.

Objective 3: *Phragmites* nutrient changes

Leaf nutrient ratios

We used the following fixed effects: grazing treatment (2 levels), mowing treatment (2 levels), year (2 levels), and season (2 levels). The random effect was the plots (10 levels) and the interactions with plots. We square root transformed C:P data.

Objective 4: Total nutrient pools

To address objective 1A, we calculated how mass nutrient pools for TN, TP, and TC in leaves, manure, and soils were altered in the grazed plot compared to the ungrazed plot. We used the data from post-grazing 2016 for all nutrients except for TC which was from post-grazing 2015.

Phragmites

For the plant nutrient pool, we estimated plot level leaf nutrient input by using the data from the plant biomass collections. This information allowed us to roughly estimate the average mass of an individual leaf in the ungrazed plots and the number of leaves per plant, based on stem height. These estimates were then used to calculate total leaf biomass per m^2 for all other treatment combinations. We assumed the following: 1. all leaves were intact when we collected biomass samples, 2. leaf size was similar for plants found in ungrazed and grazed plots, and 3. node frequency (i.e. leaf frequency) would remain the same even if the plant was grazed. This change in node length could lead to a possible underestimation of leaf frequency within the grazed plot. Nonetheless, we

calculated the input TN, TP, and TC from leaves by multiplying the total nutrient content (%) of leaves by the averaged leaf biomass (g/m^2).

Manure

For the manure nutrient pool, number of cows, type of cattle (heifer, bull, yearling, calf), and number of days the cattle were on the plot grazing was recorded. The ranchers provided average masses for these different types of cattle. We averaged data from two different sources to estimate manure production per pound of animal per day (0.06 g manure/g animal × day) (Barker, Hodges, Walls, & Services, 2002; USDA/NRCS, 1995). We used this value to estimate how much manure (g/m^2) was produced throughout the grazing season. Again, we made several assumptions: (1) the manure was distributed evenly, (2) cattle eating *Phragmites* produce the same amount of manure as the same breed eating range grasses, and (3) mass, but not age, cattle breed, or sex, affects the quantity of manure production. Like the leaf bulk nutrient input, input of total N, C, and P from manure was calculated by multiplying the total nutrient content (%) of manure by the averaged manure biomass (g/m^2).

Soil

Finally, for our soil pool, we estimated soil nutrient content by calculating average soil bulk density (g/cm^3) multiplied by the total nutrient content (%) of soils, resulting in average of nutrient content per volume (g/cm^3). We then assumed that the nutrients were equally distributed throughout the top 30cm of soil, and then converted the volume of the bulk density into an area (g/m^2) to match the other nutrient calculations.

Because the mowed areas only represent 4% of the total soil, the averages presented for this section is an average of the unmowed areas only.

Because our estimates of nutrient inputs to soils and soil nutrient pools are based on approximations and several assumptions, no statistical analyses were performed. Instead, we used these estimates to give a general idea of how these nutrient pools may be changing with grazing.

For all the nutrient pool analysis, nutrients from leaves were treated as the main input of nutrients into the soils to the soils for the ungrazed plots. Nutrients from leaves and manure were treated as the main source of nutrient inputs to the soils for the grazed plots. We assumed that air and water input was the same for both the grazed and ungrazed plots due to their proximity and generally same source of water in this analysis.

General statistical information

All statistical analyses were completed using JMP® 13.0.0. For all factors and interactions that showed significant differences, we used a Tukey HSD test if there was more than 1 comparison between factor levels or interactions between factors, or we used students t-test if there was only one comparison between factor levels to determine actual differences when a factor or interaction was significant.

RESULTS

Objective 1: Water nutrient changes and implications for water quality

Field water sampling

Flood/pond ratio

To determine actual changes in water quality due to grazing, we analyzed nutrient levels in the flood/pond ratios, which represents changes in the above ground water as it moves over the plot and picks up or loses nutrients. We found no differences in TDN, DON, and NO_3-N concentrations between post seasons and grazing treatments, and mowing treatments (Table 3, Fig. 4). However, for NH_4-N, we found significant seasonal changes occurring in the ungrazed plot, but grazing suppressed seasonal increases from pre- to post-grazing (Fig. 4). More specifically, NH_4-N was significantly higher in post-2016 than in post-2015, the unmowed treatment was significantly higher than the mow treatment, and post-2016 ungrazed was significantly higher than all other treatment season combinations (Table 3, Fig. 4). Also, SRP-P was significantly different in the season × treatment interaction (Table 3, Fig. 4) where post-2015 ungrazed appears to be less than all other treatments. Noise in the data may be driving some of these differences in TDN, DON, and SRP-P. These nutrients did have some outliers, but there was no justifiable reason to drop the outliers.

Porewater

To determine if grazing can influence nutrient levels in the water below ground, we quantified TDN, DON, NO_3-N, and SRP-P in the water in the wells. Our first model

compared the grazing treatment, mowing treatment, and 3 seasons (post-2015, pre-2016, and post-2016). For TDN, DON, and SRP-P, we found a seasonal difference but no significant effect of grazing (Table 4, Fig. 5). TDN and DON increased seasonally over time in the ungrazed plots (Fig. 5), but this seasonal increase was suppressed in the grazed plots and was compounded even further in the grazed-mowed areas (Fig. 5). SRP-P seemed to decrease over time, but again we found seasonal suppression of this change in the grazed plots, especially in the mowed areas (Fig. 5). With NO_3-N and NH_4-H, we found a significant difference in the grazing treatment and season interactions (Table 4, Fig. 5), but this was likely due to noise in the data (NO_3-N) or lack of sample replication (NH_4-H).

For our second model comparing all four seasons for the grazing plot, we found no significant difference in TDN and DON concentrations between year and season and mow treatment. The NH_4-N results showed a significant, temporal differences but no grazing effect (Table 5, Fig. 6). SRP-P had a significant season × year interaction but, again, no grazing effect (Table 5, Fig. 6).

Manure and soil leachate

The manure leachate had nutrient levels that were several magnitudes higher than the soil where TDN was 69x, DON was 98x, and NH_4-N was 77x more concentrated than soil (Table 6). NO_3-N in the manure leachate was slightly less concentrated than the soil leachate (~16%) (Table 6). Even though the manure leachate TDN, DON, and NH_4-N values were much higher than the soil, soil leachate was not affected by the manure

addition as there was no difference between TDN, DON, NH_4-N, and NO_3-N between the different treatments and treatment interactions in the soil (Table 7, Fig. 7).

We also tested SRP-P during the leachate study and found that SRP-P was 43x more concentrated in manure leachate than in the soil leachate (Table 6). In soil leachate, there was no difference in SRP-P levels across the grazing and mowing treatments and the interaction between the treatments (Table 7, Fig.7).

Objective 2: Soil nutrient bioavailability

Soil NO_3-N

There were no significant differences in NO_3-N by grazing treatment, mowing treatment, and time (Table 8, Fig. 8).

Soil $\delta^{15}N$

We found no difference between the leaf and manure pool sources of nitrogen. When we compared grazing treatments and mowing treatments over 3 season-year combinations, we only saw seasonal differences. Despite this lack of manure/leaf differences, we detected changes of $\delta^{15}N$ in soils when we compared seasons in 2016 only, where we saw a significant season and season × grazing treatment effect on soils (Table 10; Fig. 10). More specifically, in the ungrazed plots, in 2016 soils became more enriched in $\delta^{15}N$ from pre- to post-grazing, but in contrast, this temporal increase was suppressed in grazed plots (Table 10; Fig. 10). However, these changes were contrary to our prediction that grazed sites would be more enriched in $\delta^{15}N$ than ungrazed sites,

although the model assumptions of normality may not have been fully met due to highly skewed data even after transformation.

Soil ortho-P

There was no effect of grazing and mowing on phosphate within the soils (Table 11, Fig. 11), but we did see substantial temporal variation (Table 11, Fig. 11).

Objective 3: Phragmites nutrient changes

Leaf nutrient ratios

In the leaf nutrient ratio model, we again found that grazing suppresses temporal increases for both C: N and C:P. For C:N, there was a significant treatment × season interaction where the post-grazing timeframe in the ungrazed plots were significantly higher than all other combinations (Table 12, Fig.12).For the C:P data, we found a significant grazing treatment × mow treatment × season interaction where ungrazed mowed post-grazing was higher than all other samples and ungrazed unmowed post was significantly higher than grazed × unmowed × pre (Table 12, Fig. 12).

For N:P, we only found 1 significant difference in the grazing treatment × year interaction (Table 12, Fig. 12), but the Tukey test found no significant differences at $\alpha=0.05$. Therefore, it was likely noise in the data.

Objective 4: Total nutrient pools in leaves, manure and soils

Total nitrogen (TN)

We quantified site level TN nutrient pools and how grazing changes the nutrient pools. Leaves in the ungrazed plot had about 40% more TN than the leaves in the grazed

plot (6.2 ± 0.4g/m^2 and 3.7 ±0.6g/m^2 respectively) (Fig. 13). Manure pools had 4.2 ± 0.9 g/m^2 of TN (Fig. 13). To compare total inputs of the grazed and ungrazed plots, we added the leaves and the manure of the grazed plot together and compared leaf input of the ungrazed plots to the leaf + manure input of the grazed plots (Fig. 13). The total input of the grazed plot would then be approximately 7.94g/m^2(Fig. 13). While this value is slightly higher than the ungrazed plot (about 12%), when standard errors are considered, it is likely overlapping. If all other inputs held constant (atmospheric and water inputs), the total potential inputs to the ungrazed soils (leaves) and to the grazed soils (leaves + manure) are relatively similar. Although the ungrazed soil pool had about 14% more TN, the values are likely overlapping between the ungrazed and the grazed plots when data distribution is considered (26.5 ± 2.6 g/m^2 and 24.0 ± 2.3 g/m^2 respectively) (Fig. 13).

Total phosphate (TP)

There was approximately 50% more TP in the leaf pool of the ungrazed plot compared to the grazed plot (Fig. 14). The manure pool had 0.19 ± 0.06 g/m^2 (Fig. 14). When the manure and grazed plot leaves are added together, the total potential input to the grazed soils was 0.54 g/m^2 (Fig. 14). This difference is approximately 24% less than the potential input from the ungrazed leaf pool, but the values are likely overlapping considering the data distribution. For the grazed plots, the soil TP pool was approximately 16% higher than the ungrazed plots (Fig. 14). Even though the potential inputs are slightly less in the grazed plots, the soil TP slightly increases.

Total carbon (TC)

For the TC in the leaf pools, the ungrazed plots had on average 53% more TC than the grazed plots (Fig. 15). The manure pool of the grazed plot only contains 12.70 ± 3.38 g/m^2, so even when leaves and manure carbon are added (total grazed input = 71.35 g/m^2), the grazed plot still had about 42% less TC to potentially be added to soils (Fig. 15). The soils pools reflect this difference with the ungrazed plots having approximately 34% more TC than the grazed plots (Fig. 15).

DISCUSSION

Objective 1: Water nutrient changes ad implications for water quality

Changes of nutrient levels on site within floodwater and pore water

If nutrients from manure enter the water column as water moves across a site or through the soil, downstream nutrient loading and eutrophication can occur (Hubbard, Newton, & Hill, 2004). We hypothesized that the addition of manure during grazing would increase nutrient loading within flood water and in porewater samples in areas that were grazed. Surprisingly, grazing did not significantly increase TDN, DON, and NO_3-N within the floodwater or porewater samples. Even after a second year of grazing (2016), variation in these samples seemed to be due to seasonal factors unrelated to grazing.

The elevated NH_4-N in floodwater and NO_3-N in porewater in *Phragmites* stands that were not grazed indicate that grazing may be altering nitrogen pathway rates and rates of nitrogen uptake by *Phragmites*. The anaerobic conditions within wetland areas that are not grazed could be inhibiting nitrification of NH_4-N to NO_3-N, thus causing the

elevated NH_4-N levels (Mitsch & Gosselink, 2015). In contrast, when grazed soils were aerated through trampling, nitrification (NH_4-N to NO_3-N) rates were likely increasing. Within areas that were grazed, any additional NH_4-N added by manure or NO_3-N in the porewater was also likely rapidly assimilated by plants or transformed to NH_3-N or NO_3-N. Although conducted on different plants, many studies have shown that grazing often increases rates of nitrogen uptake by plants (Anderson, Dong, & Mcnaughton, 2006; McInenly, Merrill, Cahill, & Juma, 2010; Sun, Schleuss, Pausch, Xu, & Kuzyakov, 2018). This phenomenon is likely the case with grazing *Phragmites* as well, but further research would be needed to confirm this.

In the flood water, we were also surprised to see no grazing effect on SRP-P. The large year-to-year variation in SRP-P indicates that grazing is not the primary driver of SRP-P in the flood water but SRP-P may instead be influenced by soil or water pH (Mitsch & Gosselink, 2015), a relationship that warrants further study. For SRP-P in pore water, the trends seem to be related to grazing but are completely opposite in the two seasons of grazing, where SRP-P increased from pre- to post-grazing in 2015 and decreased from pre- to post-grazing in 2016. Therefore, it is likely that SRP-P levels in the pore water are driven more by unquantified biogeochemical factors than by grazing.

Potential nutrient contribution from manure to floodwater (leaching study)

The leaching study quantified nutrients that could be leached from the manure and soils once the site was flooded. Manure leachate results indicated the potential for grazing to add large concentrations of all nutrients except NO_3-N to soils and flood water, yet soil leachates did not differ significantly in grazed vs. ungrazed areas. Also, the flood water

discussed above showed no significant increase in NO3-N in grazed plots relative to ungrazed. Even though the manure has the potential to add large amounts of labile nutrients, we did not see an increase in bioavailable nutrients within the soil leachate or within the water samples mentioned above. Although our leachate study displayed this huge potential for nutrient loading, the leachate study was conducted on fresh samples of manure. In contrast, a large portion of the manure in our sites was old, desiccated, and sometimes even broken down into small pieces when the sites were flooded. This additional time allowed the manure to undergo biogeochemical and physical processes before flooding and may have decreased the levels of nutrients that can be leached from the manure. This loss of N occurs as NH_4-N is converted to NH_3-N and then volatilized (Mitsch & Gosselink, 2015), and elsewhere up to 75% of nitrogen can be lost after 1 month of manure remaining on the soil surface due to this process (Barker et al., 2002). This incongruity between the manure and soils/water could also indicate a rapid uptake by the plants (Anderson et al., 2006; McInenly et al., 2010; Sun et al., 2018).

Interestingly, manure did not have higher levels of the most plant available form of nitrogen, NO_3-N, suggesting that nitrification had not yet occurred (Mitsch & Gosselink, 2015; NRCS, 2007). Also, since NO_3-N is the most mobile form of N, having NH_4-N as the main source of N in manure could mean a lower risk of transportation of N downstream than we had originally anticipated (Mitsch & Gosselink, 2015).

Objective 2: Soil nutrient bioavailability

Because our sites were mostly dried out during the growing season, changes in nutrient forms and bioavailability can potentially impact *Phragmites* and native plant

recovery and water quality when the sites are flooded. Our plant nutrient ratio results, discussed below, indicate that our sites are N-limited, and therefore microorganisms and plants are likely to rapidly assimilate any additional N that is added to the system.

NO_3-N is the most labile and bioavailable form of nitrogen, yet contrary to our hypothesis, there was no significant difference between treatments or seasons. This result does not preclude the possibility that grazing may alter the underlying biogeochemical processes. Grazing may have influenced denitrification. In wetlands, stagnant water causes anaerobic conditions and, consequently denitrification, which reduces nitrate to N_2 gas (Mitsch & Gosselink, 2015). In stands of *Phragmites* that were not grazed, the increase in $\delta^{15}N$ isotopes over time are likely indicative of this denitrification process. In contrast, the lack of change in $\delta^{15}N$ levels over time in grazed plots may be indicative of suppressed denitrification due to cow hooves aerating soils by turning over soils and mixing soil slurries. Although we did not directly test soil and water oxygen, several of our results are consistent with this interpretation. First, our flood/pond results showed an increase in NH_4-N in ungrazed plots likely were due to anaerobic conditions. Second, we found changes in soil $\delta^{15}N$ but not NO_3-N associated with grazing, suggesting that the mechanism by which N leaves the soils and water column is likely different in areas that were grazed vs. ungrazed. In ungrazed plots, NO_3-N soils and flood water could be lost due to denitrification from anaerobic conditions, but in the grazed plots, NO_3-N could be lost from soils and the water column due to rapid assimilation by microorganisms and the stressed *Phragmites* plants.

Phosphorus (P) is a key nutrient for plant growth and water eutrophication at high levels of P, and although manure has the potential to add large quantities of P to soils, we did not find significant grazing effects on soil P. Unlike N and C, P cannot be lost to the atmosphere through volatilization or respiration and can only be removed via plant assimilation, precipitation to soils, and leaching with flood water. It is likely all three of these biogeochemical processes are counteracting the addition of P from manure quickly enough that we are not seeing increases of ortho-phosphate within the soils.

Objective 3: Phragmites nutrient changes

Leaf nutrient ratios

Changes in C: N and C:P ratios can give us a more complete understanding of how grazing is affecting not only the plants but also nutrient cycling on site. For both C: N and C:P, we saw general trends of increasing over time in the ungrazed plots. In contrast, the grazed plots did not show a difference over time. In the grazed plots however, more labile forms of manure were added, as described by the above leaching study, and although we did not see this increase in the flood water or the soils, we saw effects on grazed plants. The grazing pressure and access to more N likely increased the rate of N assimilation within plants. A study on steppe habitats also found that grazing causes an increase in the C:N ratios (Bai et al., 2012). It should be noted that the leaves collected in post-grazed sites were fully matured, but at the grazed sites, the mature leaves had been eaten, and new young leaves were re-sprouting. The different stages of leaf growth when collected may be influencing the pattern we are seeing (Fig. 12). Nonetheless, these results indicate that *Phragmites* is allocating resources to regrowth

instead of undergoing its natural senescence cycle and storing nutrients within rhizomes, but also the process of grazing is providing the extra nutrients that are needed for *Phragmites* to resprout.

N:P ratios within the plant can be an indicator of nutrient limitation within the soil (Koerselman & Meuleman, 1996). According to Koerselman and Meuleman (1996), an N:P ratio <14 within a plant indicates N limitation. *Phragmites* leaves within our study had very low N:P ratios, indicating that our sites in general are very N limited. Because our sites are so N limited, plants are likely rapidly assimilating any additional nitrogen, and this assimilation may be why we are not seeing major increases in N within the soils or the flood water.

Objective 4: Total nutrient pools in leaves, manure, and soil

Grazing effects on total nitrogen (TN) pools

Alteration of TN pools through grazing could alter plant production and composition, and increase eutrophication of adjacent bodies of water (Carpenter et al., 1998; Engloner, 2009; Kettenring et al., 2011; Uddin & Robinson, 2018)). However, in our study, cattle grazing did not affect overall pools of TN in terms of inputs into the soil and the soil pool itself. This result was surprising and could indicate that these wetlands are well-buffered against changes in TN. No change in TN could also mean that plants were quickly assimilating any additional N added by grazing since these sites are likely N limited.

In general, grazing is associated with the addition of TN into soils and water (Carpenter et al., 1998; Hubbard et al., 2004). One study found that goat grazing

increased TN in *Phragmites* leaves (Brundage, 2010), but like our study, also found no effect of grazing on soil TN. This study also found that overall, the wetland system lost N through goat assimilation and loss to the atmosphere. In contrast, we did not see an overall loss of TN.

Grazing effects on total phosphorus (TP) in plants, manure, and total ortho-P in soils

While phosphate is a key nutrient contributing to eutrophication, wetlands are generally able to filter out natural levels of phosphate and store them in soils and plant biomass (Peruzzi et al., 2009). Our study indicates that, although the total inputs into the soil do not seem to differ, ortho-P within the grazed soils was slightly higher than the ungrazed.

Since TP input of the grazed (leaf + manure) and ungrazed (leaf only) plots is very similar, cattle grazing does not seem to affect the TP inputs. Although the total P input is similar, manure is a much more labile source of P that is immediately available compared to the slow release P from leaves through decomposition. For example, Asaeda et al. (2002) found that leaf litter decays only 33-48%, and then after the first year the litter is stored in the anaerobic layers of soil. Also, trampling of leaf litter in grazed areas potentially increasing the rate of leaf decomposition in the grazed plot relative to the ungrazed. Additionally, hoof action churns the soils, unearthing leaves that have been stored in the anaerobic layer, thereby the release phosphorus into the soil. Grazing has a great potential to increase the release P that is typically stored within plant litter.

Ortho-P is the most bioavailable form of phosphorus, and increased total levels of ortho-P in wetland soils could result in many problems such as decreased plant

biodiversity, increased invasive species, and possible leaching of ortho-P into adjacent bodies of water (Croel & Kneitel, 2011; Khan & Ansari, 2005). Our study found that the soil in the grazed plots contained slightly more SRP in the soil nutrient pool than the ungrazed plots, indicating that manure may be adding a more bioavailable form of P or that plant decomposition rates are increasing. This difference was not seen in the soil on a µg/g level. Also, other factors such as pH of the soil that may also affect phosphate levels because phosphate is the most plant available for of P at slightly acidic to neutral pH conditions (Mitsch & Gosselink, 2015). Further studies into how abiotic factors, such as pH and dissolved oxygen, change with grazing *Phragmites* will help us have a clearer understanding of the drivers of nutrient changes.

Grazing effects on total carbon (TC) pools

In wetlands, C storage is a highly valued ecosystem service that helps regulate climate change by storing C in plant biomass and the soil (Davidson et al., 2017; Duarte, Losada, Hendriks, Mazarrasa, & Marbà, 2013). Our study indicates grazing may be decreasing the ability of Great Salt Lake wetlands to maintain current levels of TC and may be impeding the ability of wetlands to sequester more TC over time by decreasing *Phragmites* biomass and by altering TC in soils.

Phragmites, manure, and TC

Plants in wetlands can be a sink for carbon by absorbing and storing TC in their biomass (González-Alcaraz et al., 2012). Like our study other studies also have seen the loss of TC in plant material due to plant removal and/or grazing. One study suggested

that the removal of above ground *Phragmites* biomass will likely result in a decrease in carbon storage and CO_2 emissions when it is demineralized (González-Alcaraz et al., 2012). Though this study was not specifically discussing grazing as a form of removal, it is consistent with our results. Although carbon changes in *Phragmites* due to grazing has not been well-studied in wetlands, rangeland studies have shown that grazing decreases photosynthesis, and thus root size (Klumpp et al., 2009), and this relationship indicates a decrease in TC storage within the roots as well as the above ground biomass. Other rangeland studies have shown that grazing in general slows plant growth and rhizome elongation (Schuster, 1964). This slowing effect would indicate that there is less carbon storage in plant biomass because the biomass itself is decreasing due to physiological changes in the plant rather than just the physical removal of above ground plant matter.

Soils and TC

Soils in wetlands are also effective at sequestering carbon (Davidson et al., 2017; Villa & Bernal, 2018). Sequestration is primarily driven by anaerobic qualities of the waterlogged soil that slow plant decomposition and therefore increase carbon storage (Davidson et al., 2017; Fisher et al., 2011; Sigua, Kang, & Coleman, 2006). Our study showed that TC in soils in the grazed plot was drastically less than the ungrazed. This reduction could be occurring for many reasons. First, the input of TC to the grazed soil was much less than the ungrazed, so the stores of TC were not replenished. Also, the plots were mostly dried out before grazing for the safety and health of the cattle. Because of the change from anaerobic to aerobic state of the soil, this likely increased the decomposition of any stored leaf material (Davidson et al., 2017; Fisher et al., 2011;

Sigua et al., 2006). Asaeda et al., 2002, projected that litter decomposition could take 6 months to 2 years with litter decomposing more quickly in an aerobic environment. Also, the trampling action of the hooves could have aerated and churned the soil and organic matter within also causing an increase in decomposition rates. This mixing seemed to occur in soils that were not completely dry—forming more of a soil slurry—that the cattle would sink into and "stir" as they walked through. Other studies have shown mixed results with the relationship between grazing and soil TC. A meta-analysis by Davidson et al. (2017) showed a consistent decrease in soil carbon in wetland grazing studies conducted in the Americas, but not in European studies. They hypothesized many reasons for this pattern including the fact that the U.S. tends to have more organogenic soils whereas Europe has more mineralogenic soils (Davidson et al., 2017). This decrease in soil carbon in consistent with our findings. A smaller scale grazing study using goats showed no changes in carbon levels in the soil (Brundage, 2010), but this could be due to a variety of factors including smaller size of the animals. In contrast, another study found that a wetland that had been converted to a grazing pasture for 63 yrs. had 96% less total organic carbon (TOC) in the soils when compared to a natural wetland (Sigua et al., 2006)

Trade-offs between carbon storage and wetland restoration

While we are seeing carbon losses with grazing to remove *Phragmites*, *Phragmites*-dominated wetlands are not ideal for many other wetland ecosystem services such as plant diversity and wildlife habitat (Bertness, Ewanchuk, & Silliman, 2002; Chambers et al., 1999; Fell et al., 2003; Gratton & Denno, 2006). *Phragmites* is a

massive plant that can store a lot of carbon within its structure and litter (González-Alcaraz et al., 2012). Also, the litter of *Phragmites* takes a long time to decompose. One study suggested it would take 6 months to 2 years depending on time in aerobic layer of water (Asaeda et al., 2002). Because of the plants size, accumulation of litter, and slow rates of decomposition other wetland functions such as biodiversity and wildlife habitat may be impeded (Bertness et al., 2002; Chambers et al., 1999; Fell et al., 2003; Gratton & Denno, 2006). Another study showed that deep *Phragmites* roots may be depleting deep soil stores of carbon compared to a native wetland plants (Bernal, Megonigal, & Mozdzer, 2017). An additional study showed mixed results where, in some cases, *Phragmites* increases gaseous CH_4 release, but this result was highly variable and site-specific (Mueller et al., 2016). In contrast, another study showed that *Phragmites* is likely to increase production with increased CO_2, and thus increase TC storage over time (Caplan, Hager, Megonigal, & Mozdzer, 2015). Although *Phragmites* has a complex involvement with the carbon cycle and may be an overall sink for carbon, this benefit likely does not compensate for other ecosystem functions.

Management implications

Grazing has the potential to be an effective *Phragmites* management tool especially if integrated with other options such as herbicide, and with careful management, it could have minimal unwanted nutrient impacts. Drying and flooding at specific intervals seems to play a key role in managing for optimal nutrient levels and ecosystem functioning. Our water nutrient analysis indicated that short-term grazing at moderate stocking rates in wetlands does not negatively impact water quality. However,

fresh manure has the potential to add large amounts of bioavailable nutrients, so drying the wetland during grazing and waiting to flood for a time after grazing could minimize water quality issues (Barker et al., 2002). On the other hand, flooding after grazing may be critical to help minimize the loss of carbon (Davidson et al., 2017; Fisher et al., 2011), and it may also have co-benefits of decreasing *Phragmites* seedling reproduction (Hayball & Pearce, 2004; Mauchamp & Méthy, 2004) and creating a healthy regrowth of *Phragmites* shoots susceptible to herbicide treatment. Monitoring downstream water quality would be a good practice to ensure the natural filtering systems of wetlands do not become overwhelmed with nutrient loading especially with longer periods of grazing or higher intensity grazing practices. Also, since *Phragmites* is a high nutrient specialist that thrives in enriched and disturbed areas, it could lead to a greater return of *Phragmites* when grazing has stopped (Chambers et al., 1999; Kettenring et al., 2011; Saltonstall & Court Stevenson, 2007; Uddin & Robinson, 2017). Our study also has shown evidence of *Phragmites* utilizing the nutrient additions to recover from the impacts of grazing. Therefore, we would recommend that *Phragmites* should be treated with herbicide or some other form of treatment when grazing has ceased. According to our data and research, we would recommend that the site be dried during grazing, remain dry for 2-4 weeks after grazing, and then flooded again, and integrating grazing with other *Phragmites* control methods could increase the effectiveness of both management tools.

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TABLES & FIGURES

Table 1. Latitudes and Longitudes of plot locations.

Plot	Latitude	Longitude
Farmington Bay Grazed	40°52'56.73"N	112° 1'38.18"W
Farmington Bay Control	40°53'4.20"N	112° 1'37.90"W
Howard Slough East Grazed	41° 6'52.89"N	112° 8'21.86"W
Howard Slough East Control	41° 6'53.28"N	112° 8'12.46"W
Howard Slough West Grazed	41° 6'54.42"N	112° 9'2.17"W
Howard Slough West Control	41° 6'57.97"N	112° 9'12.66"W
Harold Crane Grazed 1	41°21'58.86"N	112° 8'49.16"W
Harold Crane Control 1	41°21'58.51"N	112° 9'0.04"W
Harold Crane Grazed 2	41°21'50.23"N	112° 9'25.25"W
Harold Crane Control 2	41°21'45.38"N	112° 9'33.17"W

Table 2. Sampling schedule pre, during, and post grazing at five study sites.

Sample	Pre 2015	During 2015	Post 2015	Pre 2016	During 2016	Post 2016
Pore/well water nutrient sampling*	X		X	X		X
Pond water nutrient sampling*	X		X	X		X
Flood water nutrient sampling*			X			X
Leaf and soil nutrient sampling	X		X	X		X
Manure nutrient sampling		X			X	
Soil bulk density	X		X	X		X
Live and litter biomass sampling						X

* Samples collected from only one of the five sites, Farmington Bay Waterfowl Management Area

Table 3. Flood/pond water ratio model results addressing question 1 (does grazing affect water quality?) for ln-transformed TDN, DON, NH_4-N, NO_3-N, and SRP-P values.

TDN model results			
Source	DF	F-ratio	P-value
Season-year	1	0.55	0.47
Grazing treatment	1	0.09	0.77
Mow treatment	1	1.00	0.33
Season-year*grazing treatment	1	0.80	0.38
Season-year*mow treatment	1	0.37	0.55
Grazing treatment*mow treatment	1	1.04	0.32
Season-year*grazing treatment*mow treatment	1	2.47	0.13
DON model results			
Source	DF	F-ratio	P-value
Season-year	1	2.07	0.17
Grazing treatment	1	0.19	0.67
Mow treatment	1	0.71	0.41
Season-year*grazing treatment	1	0.11	0.74
Season-year*mow treatment	1	0.50	0.49
Grazing treatment*mow treatment	1	1.37	0.26
Season-year*grazing treatment*mow treatment	1	2.60	0.12
NH_4-N model results			
Source	DF	F-ratio	P-value
Season-year	1	14.28	0.00
Grazing treatment	1	2.13	0.16
Mow treatment	1	4.55	0.05
Season-year*grazing treatment	1	25.48	0.00
Season-year*mow treatment	1	0.45	0.51
Grazing treatment*mow treatment	1	0.27	0.61
Season-year*grazing treatment*mow treatment	1	0.04	0.85
NO_3-N model results			
Source	DF	F-ratio	P-value
Season-year	1	0.0362	0.8509
Grazing treatment	1	1.0883	0.3093
Mow treatment	1	0.3541	0.5585
Season-year*grazing treatment	1	0.0021	0.964
Season-year*mow treatment	1	4.3386	0.0503
Grazing treatment*mow treatment	1	0.0704	0.7934
Season-year*grazing treatment*mow treatment	1	0.0001	0.9911
SRP-P model results			
Source	DF	F-ratio	P-value
Season-year	1	0.2541	0.6197
Grazing treatment	1	0.8035	0.3807

Table 3. (cont.)

Mow treatment	1	0.5846	0.4534
Season-year*grazing treatment	1	6.0876	0.0228
Season-year*mow treatment	1	0.1502	0.7024
Grazing treatment*mow treatment	1	1.5508	0.2274
Season-year*grazing treatment*mow treatment	1	1.2582	0.2753

Table 4. Pore water model results addressing question 1 (does grazing affect water quality?) between grazing treatments, mowing treatments, and 3 season-year combinations for ln transformed TDN, DON, NH_4-N, NO_3-N, and square root transformed SRP-P.

TDN model results			
Source	DF	F-ratio	P-value
Season-year	2	4.95	0.02
Grazing treatment	1	1.00	0.34
mow treatment	1	0.25	0.63
Season-year*grazing treatment	2	1.50	0.25
Season-year*mow treatment	2	0.22	0.81
Grazing treatment*mow treatment	1	1.10	0.32
Season-year*grazing treatment*mow treatment	2	1.66	0.22
DON model results			
Source	DF	F-ratio	P-value
Season-year	2	8.66	0
Grazing treatment	1	1.34	0.27
mow treatment	1	0	0.99
Season-year*grazing treatment	2	1.59	0.23
Season-year*mow treatment	2	1.14	0.34
Grazing treatment*mow treatment	1	0	0.99
Season-year*grazing treatment*mow treatment	2	2.16	0.14
NH_4-N model results			
Source	DF	F-ratio	P-value
Season-year	2	3.58	0.05
Grazing treatment	1	0.04	0.84
mow treatment	1	0.61	0.45
Season-year*grazing treatment	2	5.64	0.02
Season-year*mow treatment	2	0.53	0.6
Grazing treatment*mow treatment	1	0.4	0.54
Season-year*grazing treatment*mow treatment	2	0.63	0.55

Table 4. (cont.)

NO_3-N model results			
Source	DF	F-ratio	P-value
Season-year	2	2.07	0.15
Grazing treatment	1	12.42	0.00
mow treatment	1	3.61	0.08
Season-year*grazing treatment	2	8.00	0.00
Season-year*mow treatment	2	3.31	0.06
Grazing treatment*mow treatment	1	4.71	0.05
Season-year*grazing treatment*mow treatment	2	3.43	0.05
SRP-P model results			
Source	DF	F-ratio	P-value
Season-year	2	7.41	0.00
Grazing treatment	1	0.09	0.77
mow treatment	1	0.04	0.84
Season-year*grazing treatment	2	0.49	0.62
Season-year*mow treatment	2	0.82	0.46
Grazing treatment*mow treatment	1	0.01	0.93
Season-year*grazing treatment*mow treatment	2	0.84	0.45

Table 5. Pore water model results addressing question 1 (does grazing affect water quality?) in grazed plot only between mowing treatments, seasons, and year combinations for ln transformed TDN, DON, NH_4-N, NO_3-N, and non-transformed SRP-P data.

TDN model results			
Source	DF	F-ratio	P-value
Year	1	1.14	0.31
Season	1	1.18	0.30
Mow treatment	1	0.29	0.61
Year*season	1	0.15	0.70
Year*mow treatment	1	1.17	0.30
Season*mow treatment	1	0.04	0.84
Year*season*mow treatment	1	1.83	0.20

Table 5. (cont.)

DON model results			
Source	DF	F-ratio	P-value
Year	1	4.17	0.07
Season	1	0.87	0.37
Mow treatment	1	0.13	0.73
Year*season	1	4.34	0.06
Year*mow treatment	1	2.8	0.13
Season*mow treatment	1	0.29	0.6
Year*season*mow treatment	1	4.03	0.07
NH_4-N model results			
Source	DF	F-ratio	P-value
Year	1	0.172	0.69
Season	1	11.52	0.01
Mow treatment	1	0.81	0.4
Year*season	1	5.98	0.03
Year*mow treatment	1	1.15	0.31
Season*mow treatment	1	2.03	0.18
Year*season*mow treatment	1	1.03	0.33
NO_3-N model results			
Source	DF	F-ratio	P-value
Year	1	1.52	0.24
Season	1	13.91	0.00
Mow treatment	1	0.00	0.96
Year*season	1	1.94	0.19
Year*mow treatment	1	0.32	0.58
Season*mow treatment	1	1.43	0.25
Year*season*mow treatment	1	0.06	0.81
SRP-P model results			
Source	DF	F-ratio	P-value
Year	1	2.21	0.17
Season	1	1.01	0.34
Mow treatment	1	0.19	0.68
Year*season	1	19.79	0.00
Year*mow treatment	1	0.81	0.39
Season*mow treatment	1	0.15	0.70
Year*season*mow treatment	1	0.00	1.00

Table 6. Manure and soil leachate means ± 1 standard error addressing question 1 (Does grazing affect water quality?). The soil included both the grazed and the ungrazed soils combined. Magnitude difference is calculated using the following formula: (manure-soil)/soil.

	Manure (mg/L)	Soil (mg/L)	Magnitude difference
TDN	7.40 ± 0.31	0.11 ± 0.02	69x
DON	4.27 ± 0.24	0.04 ± 0.01	98x
NO3	0.04 ± 0.01	0.05 ± 0.01	-0.16x
NH4	3.08 ± 0.18	0.04 ± 0.01	76.71x
SRP	2.44 ± 0.15	0.05 ±0.00	43.38x

Table 7. Soil leachate model results addressing question 1 (does grazing affect water quality?) between grazing and mowing treatments for square root transformed TDN and NO_3-N data and ln transformed DON, NH_4-N, and SRP-P data.

TDN model results			
Source	DF	F-ratio	P-value
Grazing treatment	1	0.04	0.84
Mowing treatment	1	1.92	0.19
Grazing treatment*mowing treatment	1	2.26	0.16
DON model results			
Source	DF	F-ratio	P-value
Grazing treatment	1	0.01	0.93
Mowing treatment	1	4.15	0.07
Grazing treatment*mowing treatment	1	2.11	0.17
NH_4-N model results			
Source	DF	F-ratio	P-value
Grazing treatment	1	1.27	0.28
Mowing treatment	1	2.40	0.15
Grazing treatment*mowing treatment	1	0.40	0.54
NO_3-N model results			
Source	DF	F-ratio	P-value
Grazing treatment	1	0.04	0.84
Mowing treatment	1	1.84	0.20
Grazing treatment*mowing treatment	1	0.20	0.66
SRP-P model results			
Source	DF	F-ratio	P-value
Grazing treatment	1	0.05	0.82
Mowing treatment	1	0.82	0.38
Grazing treatment*mowing treatment	1	0.12	0.74

Table 8. Soil NO_3-N model results addressing question 2 (Does grazing alter soil bioavailable nutrients or biogeochemical drivers of nutrient cycling?) between grazing treatments, mowing treatments, year, and season effects where the NO_3-N data was ln transformed.

Soil NO_3-N model results			
Source	DF	F Ratio	Prob > F
Grazing treatment	1	1.88	.
Mowing treatment	1	2.73	1.00
Grazing treatment*mowing treatment	1	1.31	1.00
Year	1	92.82	1.00
Year*grazing treatment	1	0.00	1.00
Year*mowing treatment	1	0.03	1.00
Grazing treatment*mowing treatment*year	1	0.39	1.00
Season	1	0.01	1.00
Season*grazing treatment	1	4.96	1.00
Season*mowing treatment	1	1.18	1.00
Season*year	1	26.07	1.00
Grazing treatment*mowing treatment*season	1	0.57	1.00
Year*grazing treatment*season	1	0.70	1.00
Year*mowing treatment*season	1	1.03	1.00
Grazing treatment*mowing treatment*year*season	1	0.19	1.00

Table 9. Soil $\delta^{15}N$ model results addressing question 2 (Does grazing alter soil bioavailable nutrients or biogeochemical drivers of nutrient cycling?) between grazing treatments, mowing treatments, and season-year effects where the $\delta^{15}N$ data was ln transformed.

Soil $\delta^{15}N$ model results			
Source	DF	F Ratio	P-value
Grazing treatment	1	0.11	0.75
Mow treatment	1	0.75	0.41
Grazing treatment*mow treatment	1	0.34	0.58
Season-year	2	6.71	0.00
Grazing treatment*season-year	2	1.94	0.16
Mow treatment*season-year	2	0.06	0.95
Grazing treatment*mow treatment*season-year	2	0.05	0.95

Table 10. Soil $\delta^{15}N$ model results addressing question 2 (Does grazing alter soil bioavailable nutrients or biogeochemical drivers of nutrient cycling?) between grazing treatments, mowing treatments, and season effects in year 2016 where the $\delta^{15}N$ data was ln transformed.

Soil $\delta^{15}N$ 2016 model results			
Source	DF	F Ratio	P-value
Grazing treatment	1	0.026	0.88
Mow treatment	1	2.06	0.19
Grazing treatment*mow treatment	1	0.33	0.58
Season	1	12.20	0.00
Grazing treatment*season	1	7.94	0.01
Mow treatment*season	1	0.12	0.73
Grazing treatment*mow treatment*season	1	0.25	0.62

Table 11. Soil ortho-P model results addressing question 2 (Does grazing alter soil bioavailable nutrients or biogeochemical drivers of nutrient cycling?) between grazing treatments, mowing treatments, year, and season effects in year 2016. Data was not transformed.

Soil ortho-P model results			
Source	DF	F Ratio	P-value
Grazing treatment	1	0.77	0.41
Mow treatment	1	0.81	0.40
Mow treatment*grazing treatment	1	0.56	0.48
Year	1	12.80	0.00
Year*grazing treatment	1	1.86	0.19
Year*mow treatment	1	0.52	0.48
Mow treatment*grazing treatment*year	1	0.72	0.41
Season	1	22.08	0.00
Season*grazing treatment	1	2.91	0.10
Season*mow treatment	1	0.06	0.80
Season*year	1	14.44	0.00
Mow treatment*grazing treatment*season	1	0.41	0.53
Mow treatment*grazing treatment*year*season	1	0.77	0.39
Year*grazing treatment*season	1	1.15	0.29
Year*mow treatment*season	1	0.44	0.51

Table 12. Leaf nutrient ratios (moles) model results addressing question 3 (Will grazing alter nutrient ratios within *Phragmites* itself (C: N and C:P)? Will N:P ratios within *Phragmites* reveal changes in soils nutrient limitations?) between grazing treatments, mowing treatments, and season-year effects in year 2016. C: N and N:P data was not transformed; C:P data was transformed using square root.

Leaf C: N model results			
Source	DF	F Ratio	P-value
Grazing treatment	1	11.37	0.01
Mow treatment	1	0.00	0.95
Grazing treatment*mow treatment	1	2.47	0.16
Year	1	11.80	0.00
Grazing treatment*year	1	0.06	0.81
Mow treatment*year	1	0.23	0.64
Grazing treatment*mow treatment*year	1	0.65	0.43
Season	1	44.09	0.00
Year*season	1	3.42	0.07
Grazing treatment*year*season	1	0.00	0.97
Mow treatment*year*season	1	0.11	0.74
Grazing treatment*mow treatment*year*season	1	1.02	0.32
Grazing treatment*season	1	28.19	0.00
Mow treatment*season	1	0.13	0.73
Grazing treatment*mow treatment*season	1	2.08	0.16
Leaf C:P model results			
Source	DF	F Ratio	P-value
Grazing treatment	1	44.98	0.00
Mow treatment	1	4.41	0.07
Grazing treatment*mow treatment	1	4.52	0.07
Year	1	4.22	0.06
Grazing treatment*year	1	5.24	0.04
Mow treatment*year	1	0.37	0.55
Grazing treatment*mow treatment*year	1	0.03	0.87
Season	1	22.83	0.00
Year*season	1	3.22	0.08
Grazing treatment*year*season	1	0.51	0.48
Mow treatment*year*season	1	0.09	0.77
Grazing treatment*mow treatment*year*season	1	0.16	0.69
Grazing treatment*season	1	10.55	0.00
Mow treatment*season	1	0.02	0.89
Grazing treatment*mow treatment*season	1	5.51	0.03

Table 12. (cont.)

Leaf N:P model results			
Source	DF	F Ratio	P-value
Grazing treatment	1	0.07	0.79
Mow treatment	1	2.44	0.16
Grazing treatment*mow treatment	1	0.62	0.45
Year	1	0.14	0.72
Grazing treatment*year	1	13.64	0.00
Mow treatment*year	1	0.71	0.41
Grazing treatment*mow treatment*year	1	0.01	0.94
Season	1	0.00	0.96
Year*season	1	0.23	0.63
Grazing treatment*year*season	1	0.70	0.41
Mow treatment*year*season	1	1.01	0.32
Grazing treatment*mow treatment*year*season	1	0.25	0.62
Grazing treatment*season	1	2.73	0.11
Mow treatment*season	1	0.16	0.70
Grazing treatment*mow treatment*season	1	1.49	0.23

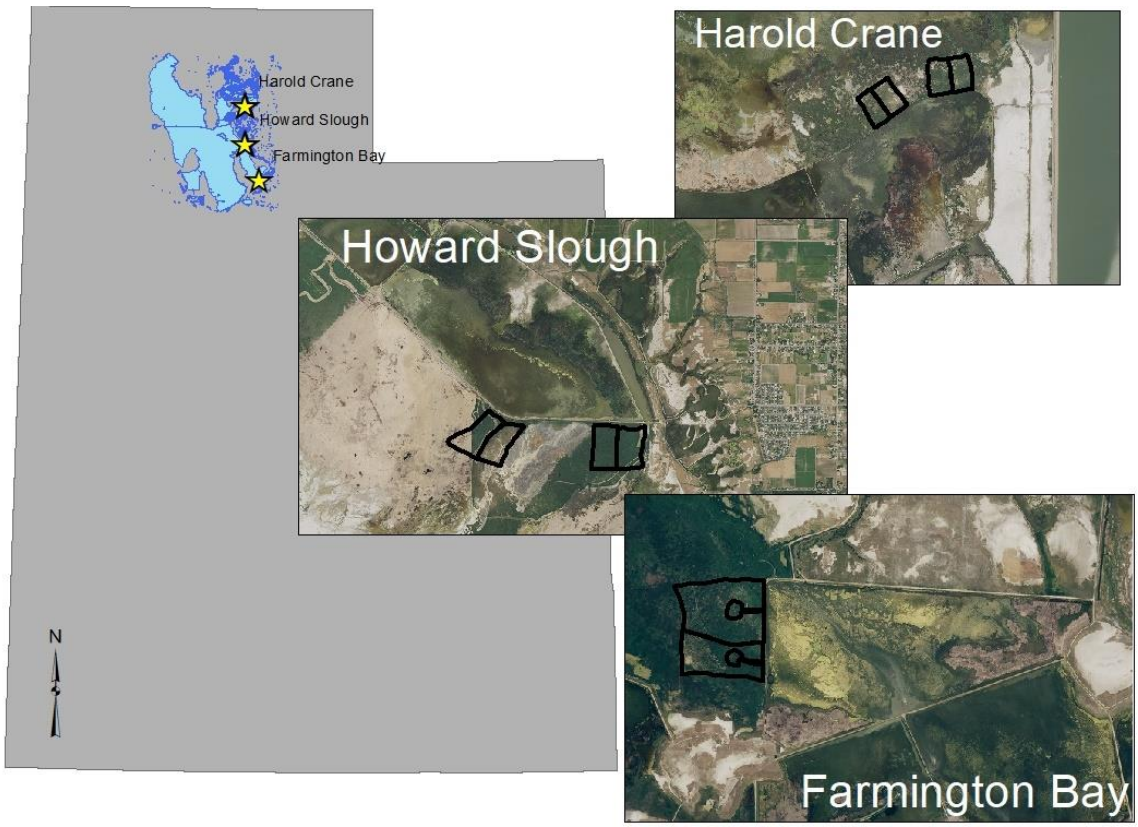

Fig. 1. Map of site locations around Great Salt Lake with yellow stars marking study locations and aerial imagery showing the individual plots.

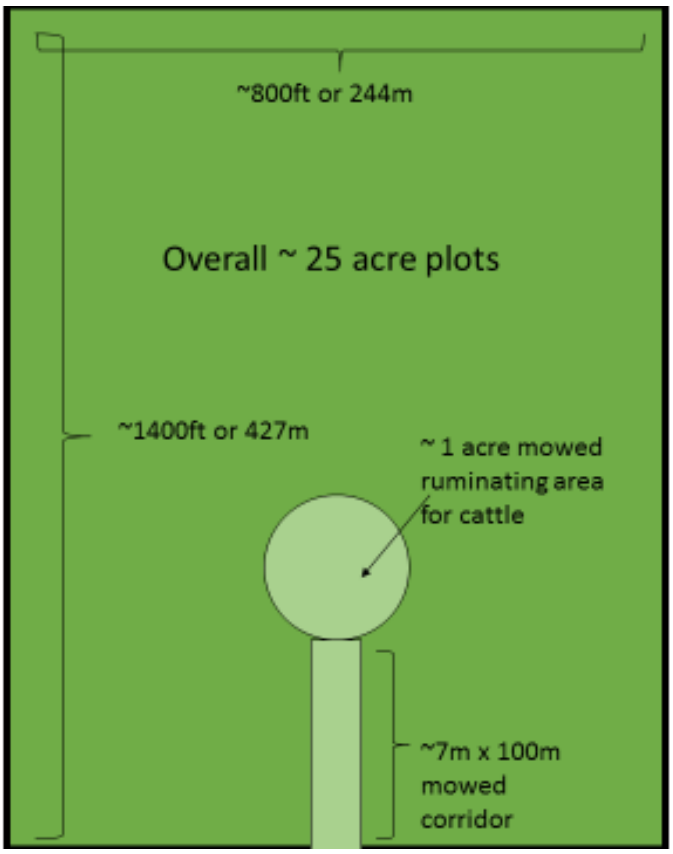

Fig. 2. Plot size and layout.

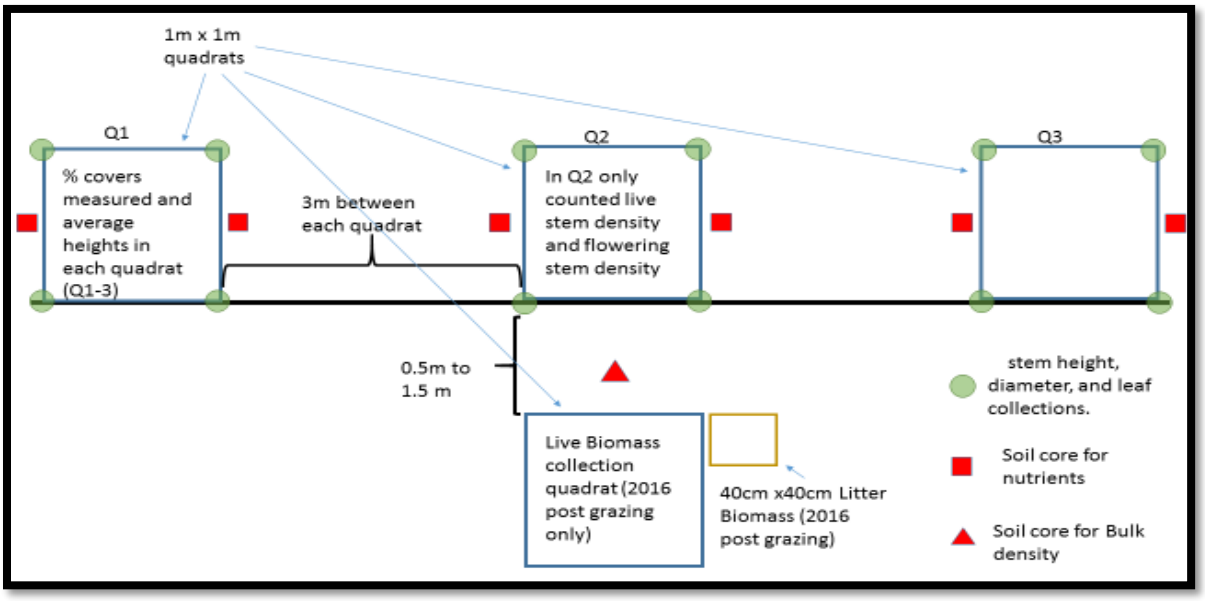

Fig. 3. Transect layout with 3 quadrats per transect. Measurements taken at each quadrat are shown.

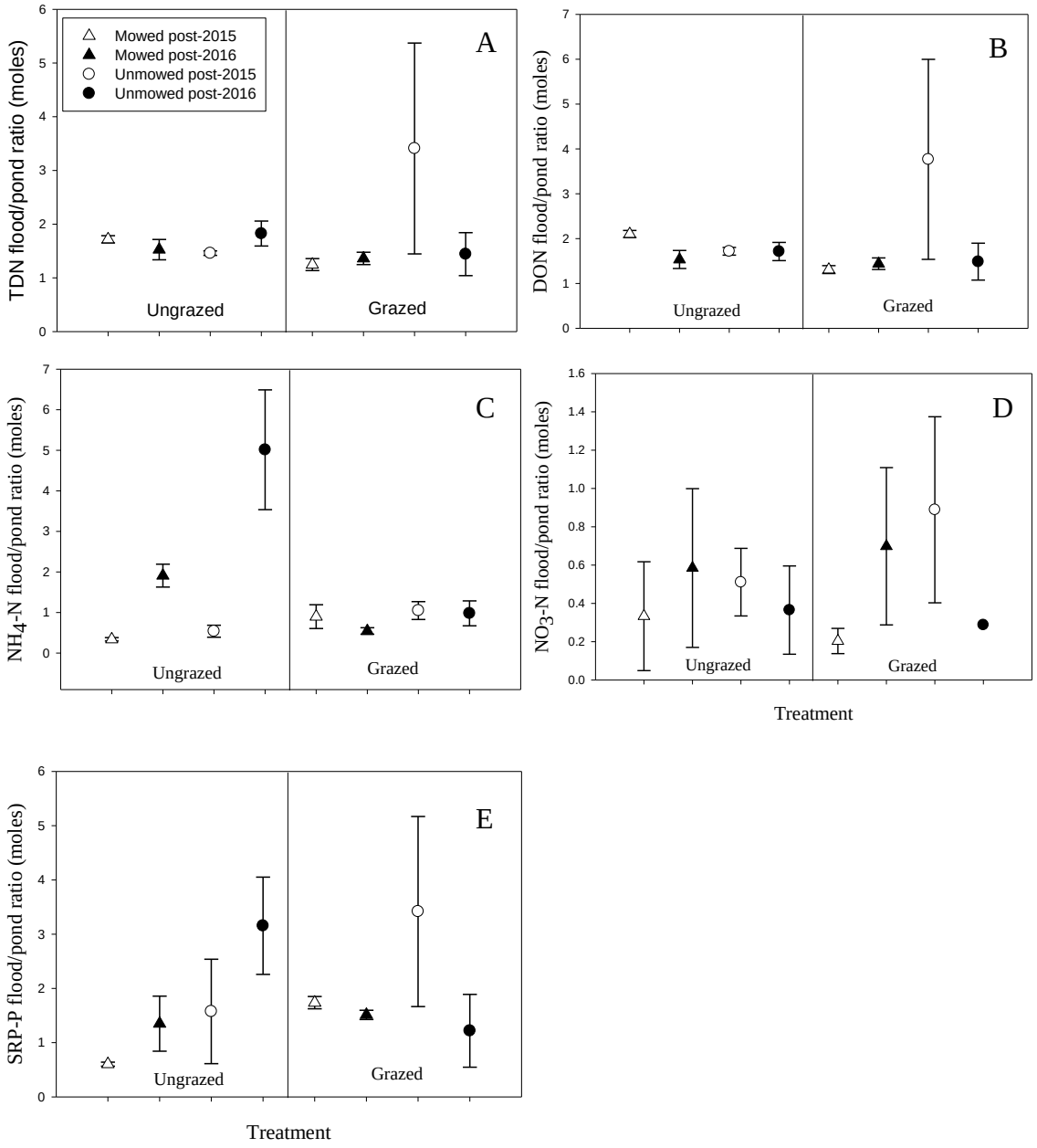

Fig. 4. Flood/pond ratio means ± 1 standard error (SE) by grazing and mowing treatments over 2 seasons. Each graph represents a different nutrient: A. TDN, B. DON, C. NH_4-N, D. NO_3-N, and E. SRP-P.

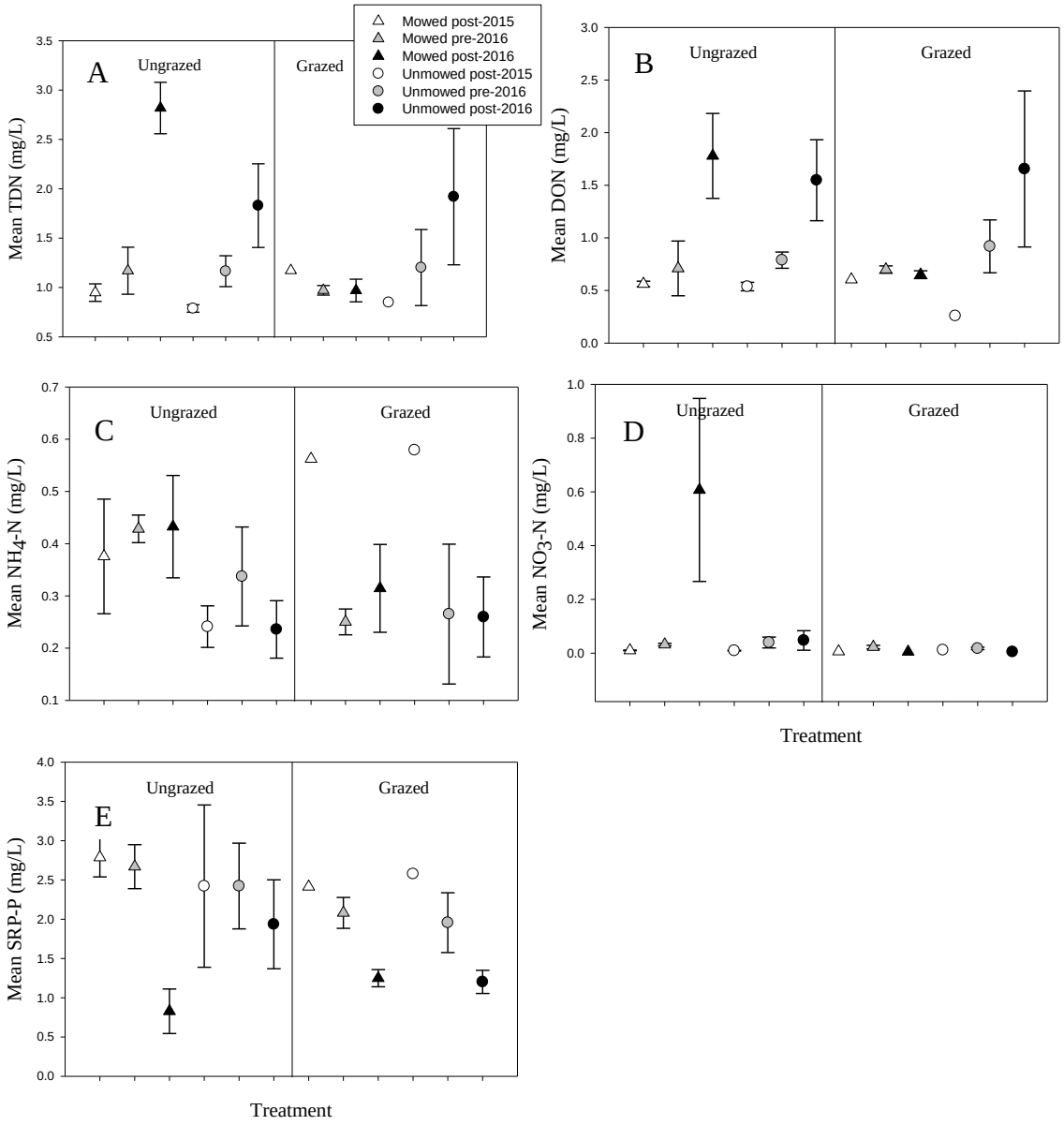

Fig. 5. A comparison in pore water nutrients by grazing and mowing treatments over 3 seasons (post-2015, pre-2016, and post-2016). Each graph represents a different nutrient: A. TDN, B. DON, C. NH_4-N, D. NO_3-N, and E. SRP-P. For the grazed plots in pre-2015, we only had 1 sample. Therefore, there are no error bars on those points.

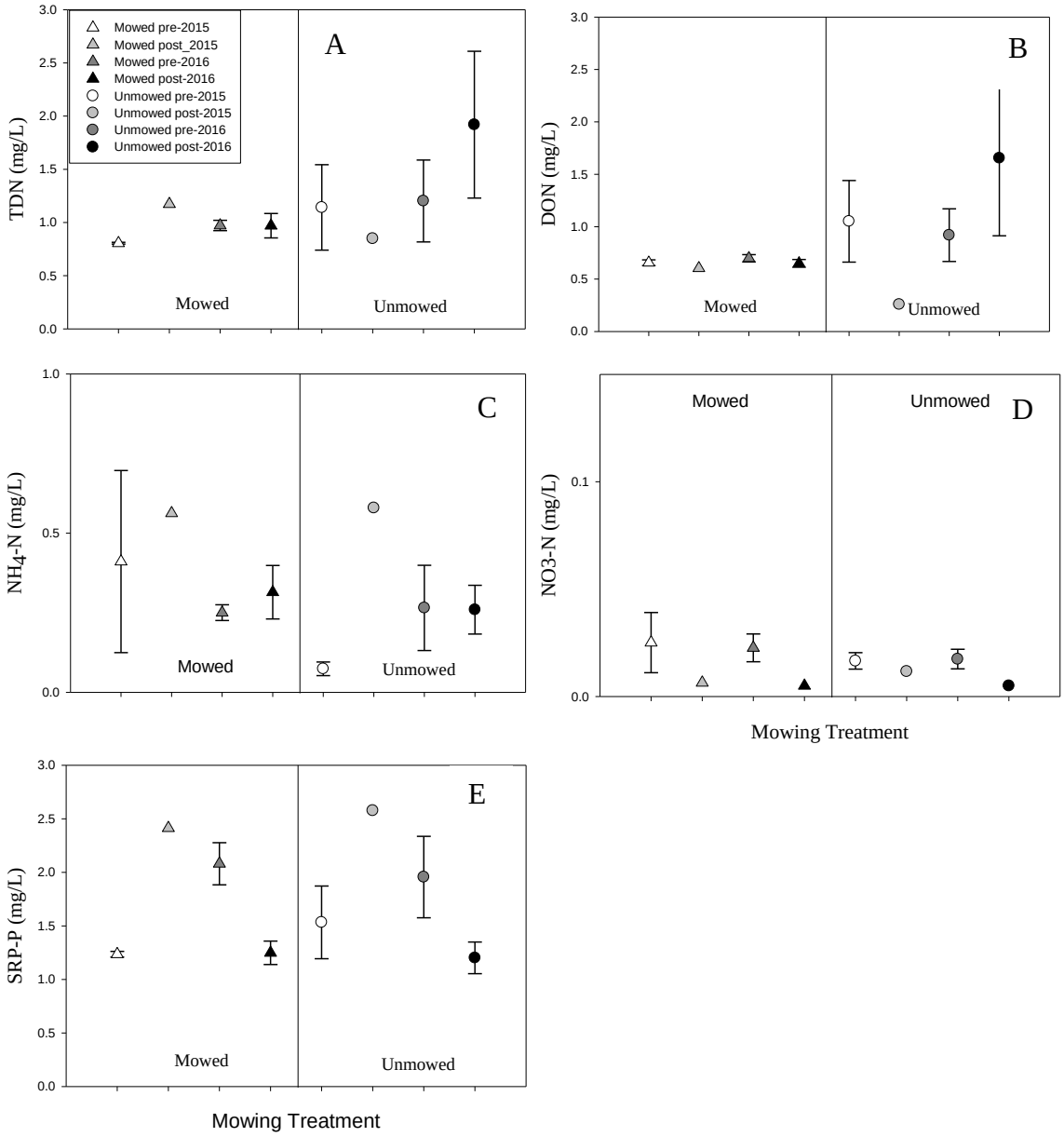

Fig. 6. Differences in pore water nutrients in grazing plot by mowing treatment and season-year combinations. Each graph represents a different nutrient: A. TDN, B. DON, C. NH_4-N, D. NO_3-N, and E. SRP-P. For pre-2015, we only had 1 sample. Therefore, there are no error bars on those points.

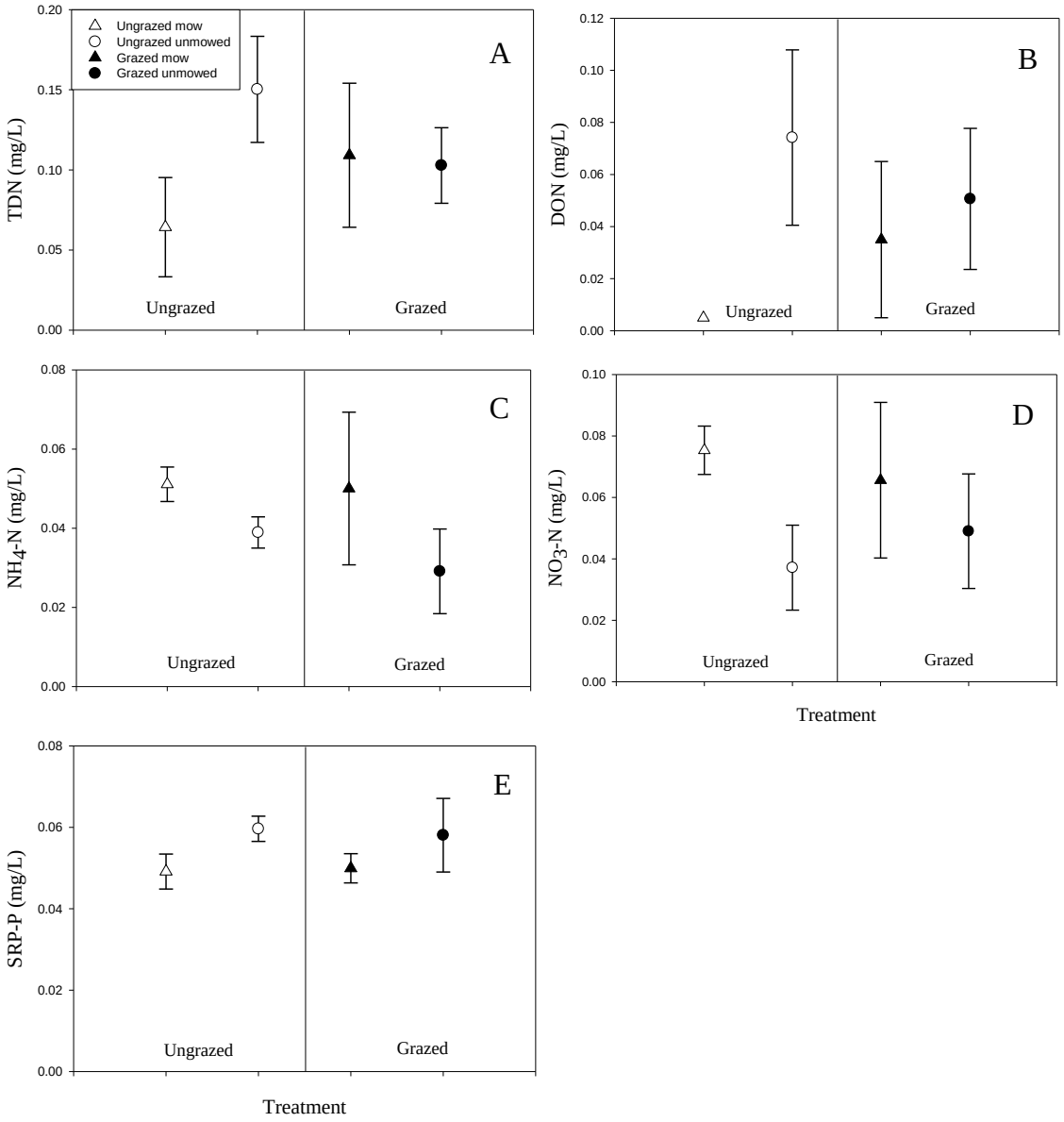

Fig. 7. Differences in soil leachate nutrient between the grazing and mowing treatments. Each graph represents a different nutrient: A. TDN, B. DON, C. NH_4-N, D. NO_3-N, and E. SRP-P.

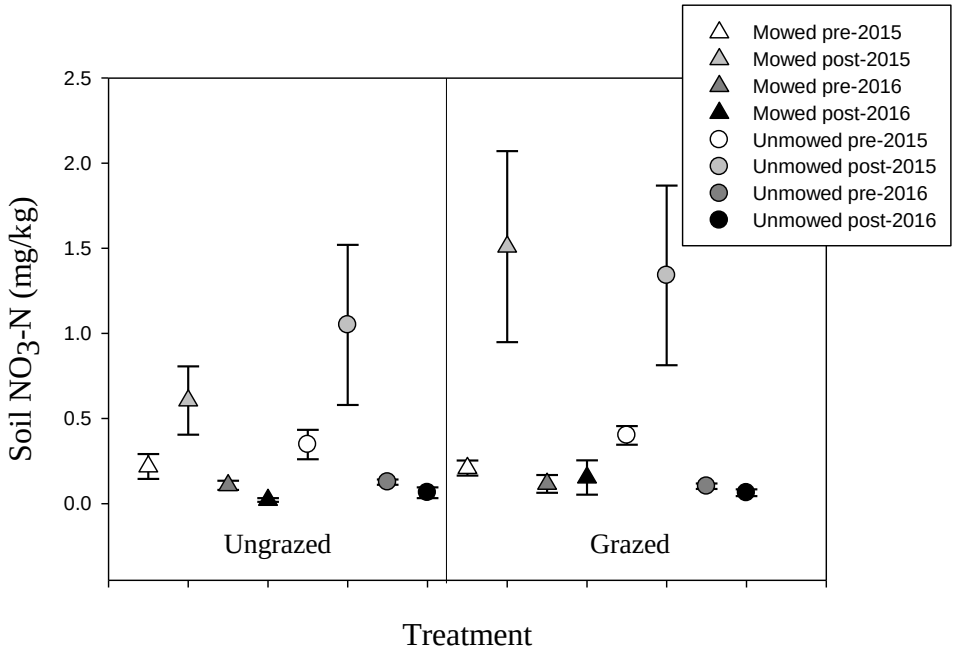

Fig. 8. Soil NO_3-N concentrations (mean ± 1 standard error) displayed by grazing and mowing treatments across all season-year combinations.

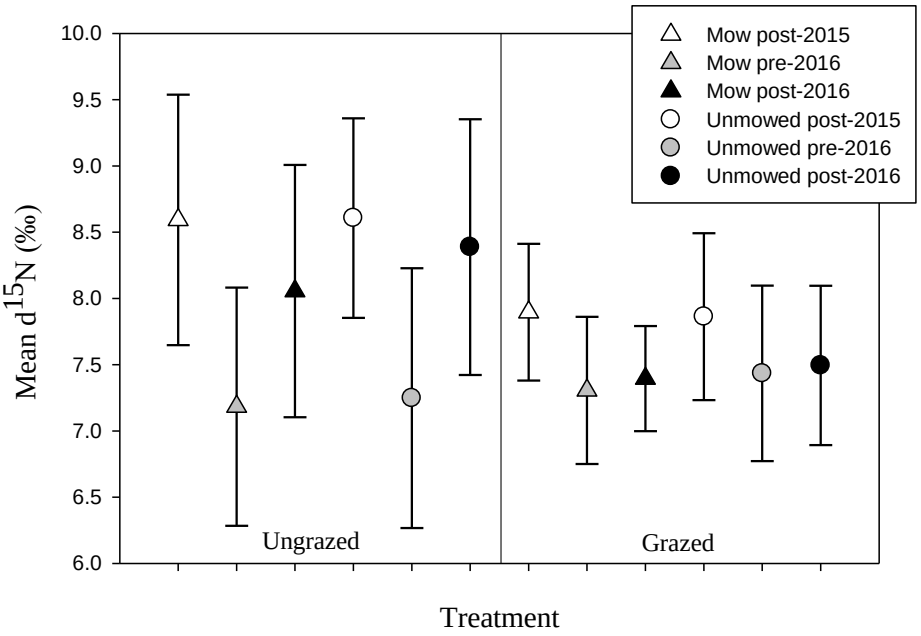

Fig. 9. Means ± 1 standard error of $\delta^{15}N$ concentrations in soils across grazing and mowing treatments across 3 season-year combinations.

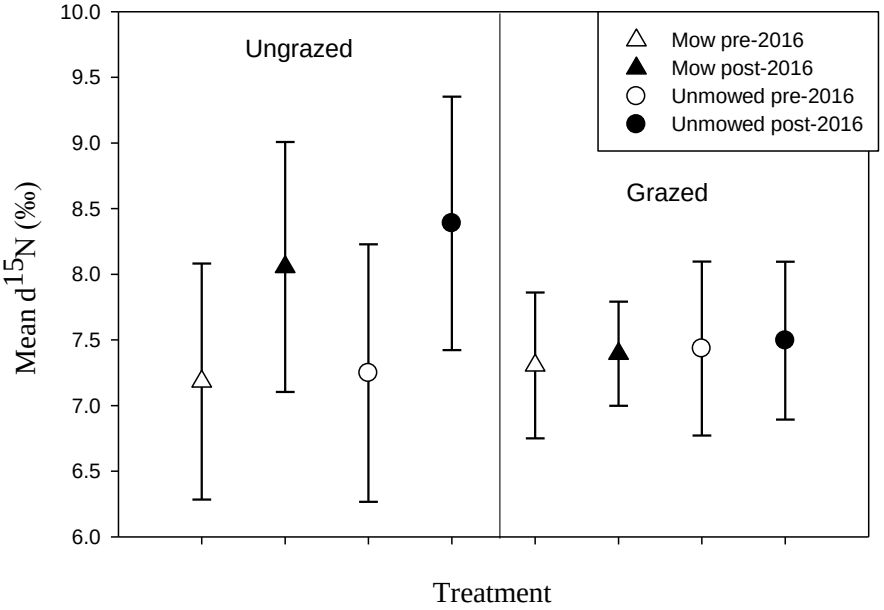

Fig. 10. Means ± 1 standard error of $\delta^{15}N$ concentrations in soils across grazing and mowing treatments across both season in 2016.

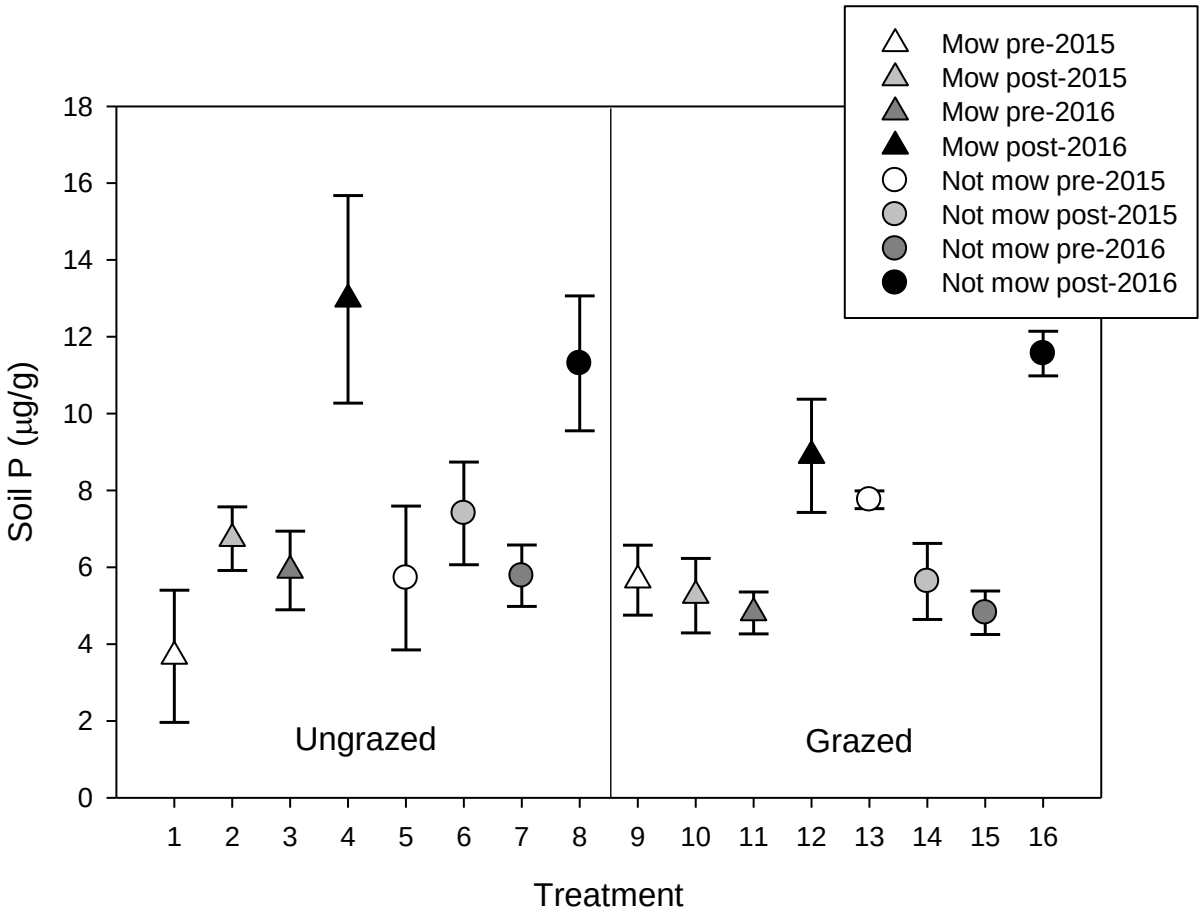

Fig. 11. Means ± 1 standard error of ortho-P concentrations in soils across grazing and mowing treatments across all season-year combinations.

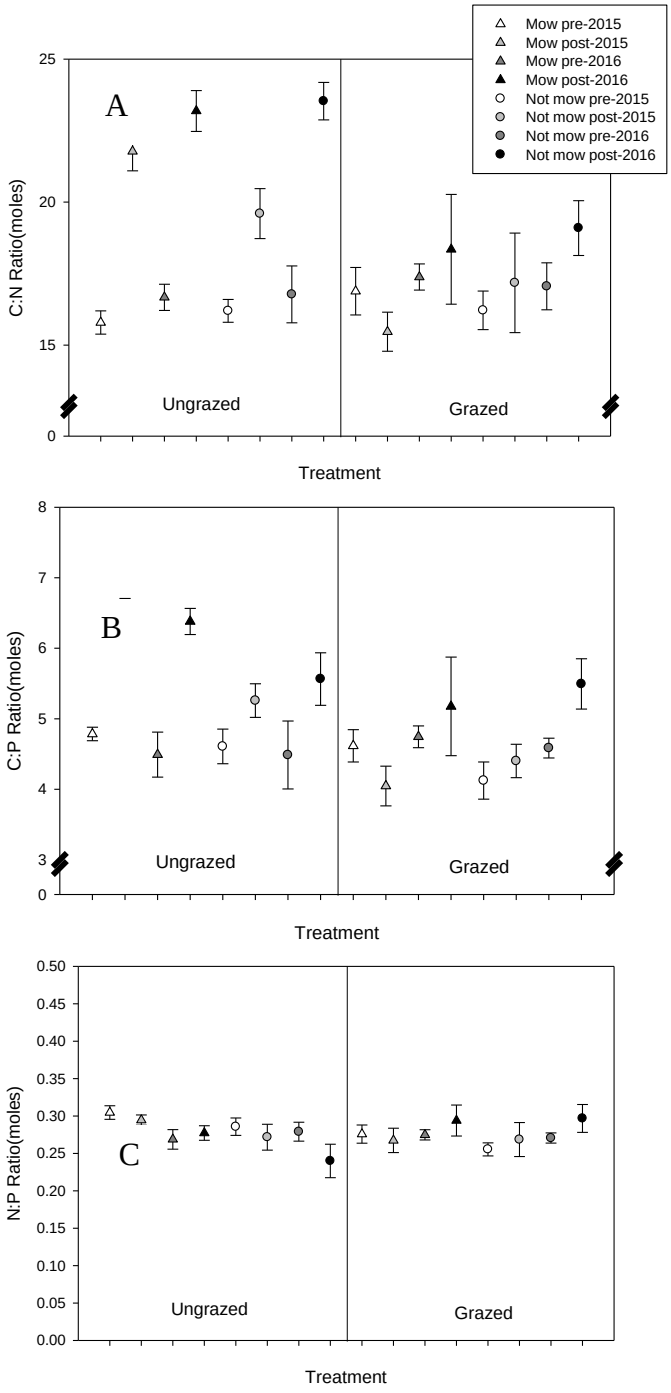

Fig. 12. Nutrient ratios (moles) in leaves with means ± 1 standard error. The hatch marks represent extra space where no data was present, so it was hidden to make the graphs easier to read. The following ratios are displayed on each graph: **A.** C: N, **B.** C:P, and **C.** N:P

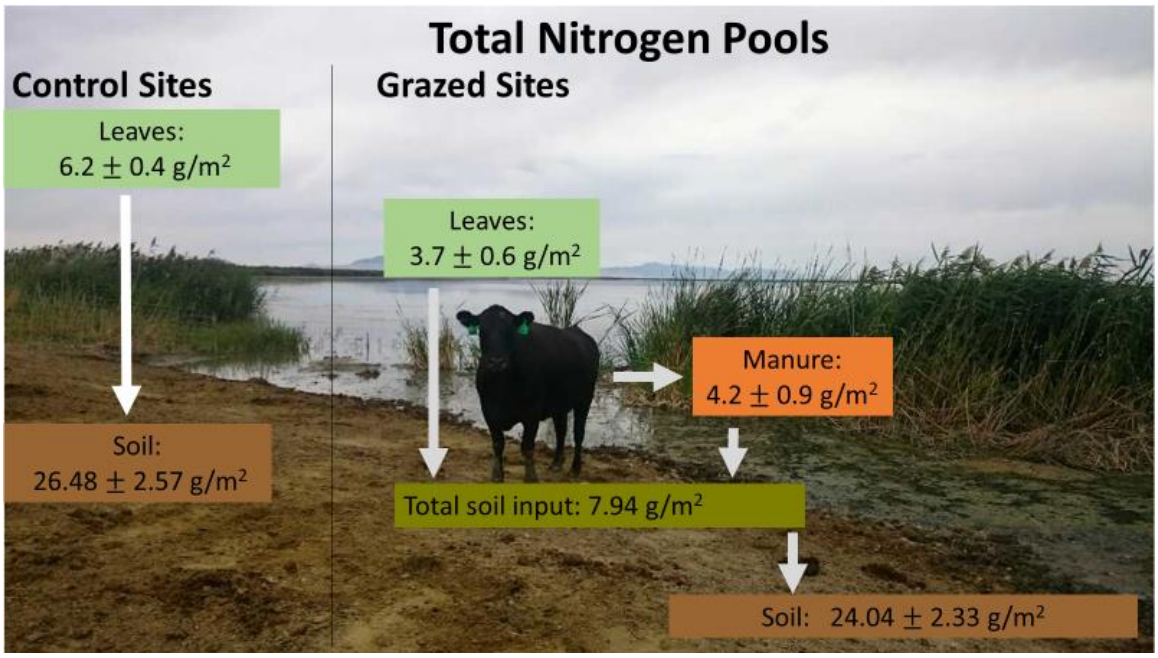

Fig. 13. Means ± 1 standard error of TN in different nutrient pools including leaves, manure, and soils. Total soil input is the mean leaf TN concentration plus the mean manure concentrations.

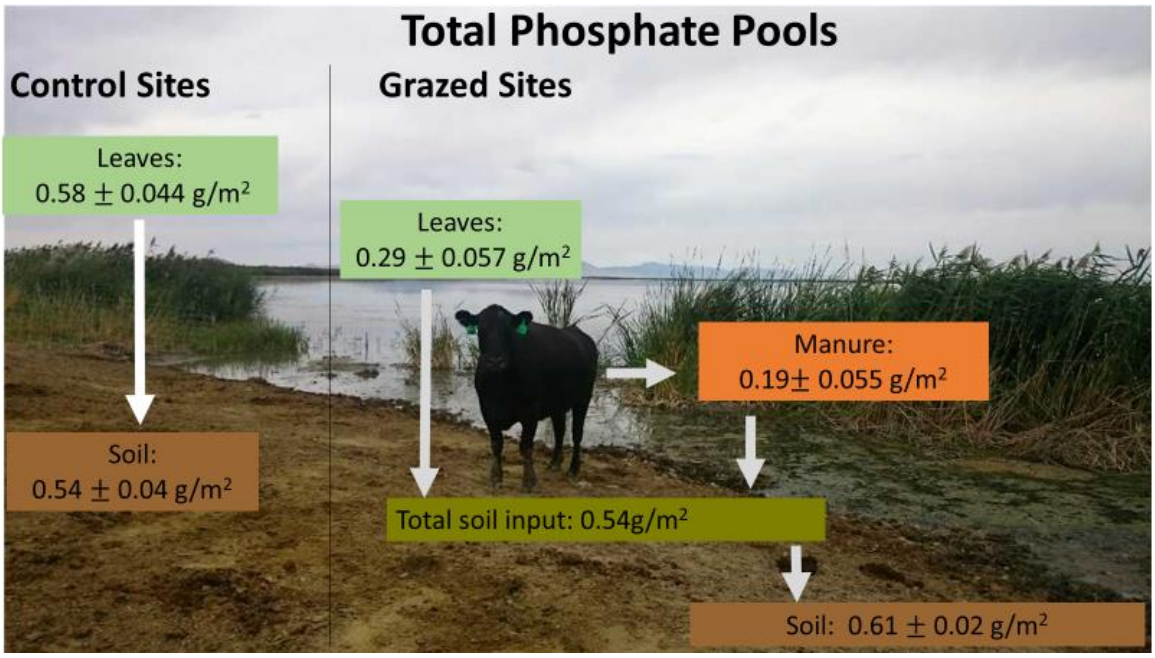

Fig. 14. Means ± 1 standard error of TP in different nutrient pools including leaves, manure, and soils. Total soil input is the mean leaf TP concentration plus the mean manure concentrations.

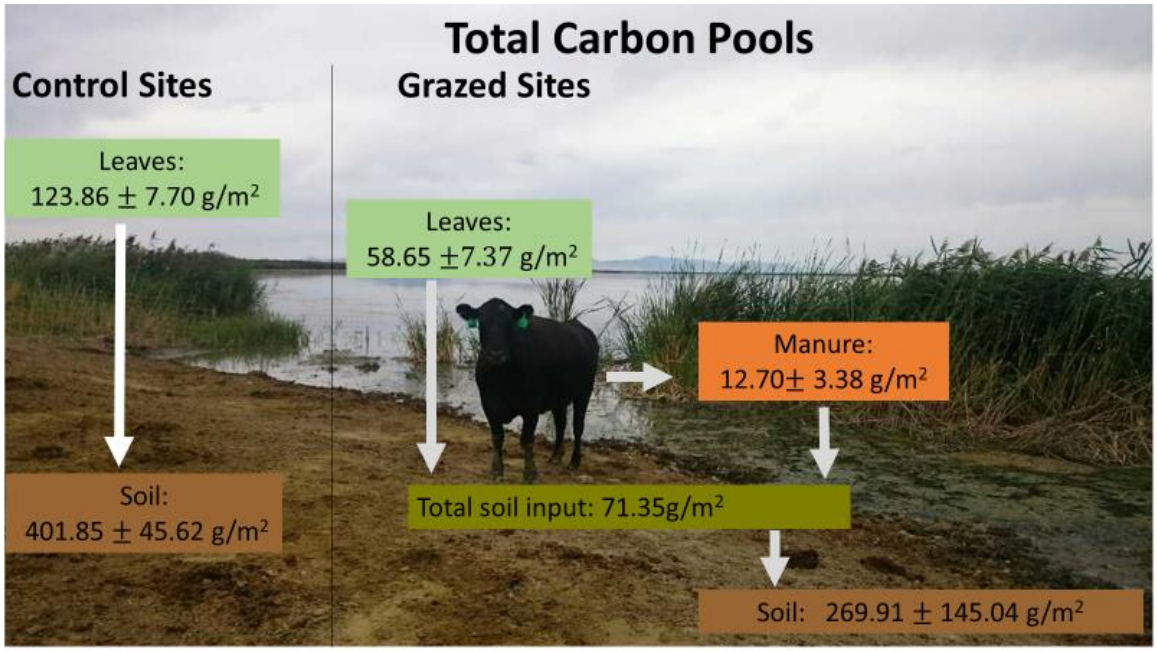

Fig. 15. Means ± 1 standard error of TC in different nutrient pools including leaves, manure, and soils. Total soil input is the mean leaf TC concentration plus the mean manure concentrations.