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Final report to Utah Division of Forestry, Fire & State Lands

Impacts of grazing for *Phragmites* control on nutrient inputs, pools, and limitations in Great Salt Lake wetlands

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Introduction

One of the biggest threats to Utah wetlands is the invasion of *Phragmites australis* (common reed or *Phragmites*), which currently covers an estimated 9,308 hectares on the eastern third of the Great Salt Lake alone (Kettenring, et al. 2012; Long et al. 2012). This perennial grass, which was introduced from Eurasia, outcompetes native wetland vegetation and can rapidly transform healthy wetland habitat to a dense monoculture that is unusable by most wildlife, particularly waterfowl. Herbicide and mowing are commonly used to control this species, but over the past few years, grazing has garnered great interest along the Great Salt Lake as a *Phragmites* management tool that may be even more effective when used in conjunction with herbicide. Vast areas of protected lands on the Great Salt Lake (e.g., Ogden Bay, Farmington Bay, Howard Slough, and Harold Crane Waterfowl Management Areas and Bear River Migratory Bird Refuge) are currently exposed to cattle grazing for *Phragmites* management. Experimental research studies are needed to determine the costs and benefits of this approach.

Cattle may promote wetland restoration in two main ways. First, cattle that are accustomed to graze in wetlands can open up wetland habitat to promote native species. Second, grazing can potentially be timed to prevent *Phragmites* from producing seed, the main form of *Phragmites* spread (Kettenring and Mock 2012). Nonetheless, grazing may have unwanted impacts on wetlands, namely through increased nutrient loading via excretion by cattle. Nutrient cycling is very complex. 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 and Zieman 2010) and numerous experiments have shown increased growth, reproduction, and occurrence of *Phragmites* under elevated nutrient conditions (Kettenring et al. 2011; King et al. 2007; Minchinton and Bertness 2003; Saltonstall andStevenson 2007), cattle may have unintended consequences for future *Phragmites* invasions by concentrating nutrients in wetland soils, but 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* (Rickey and Anderson 2004; King et al. 2007; Saltonstall and Stevenson 2007; James et al. 2011; Kettenring et al. 2011). If that were the case, then grazing as a *Phragmites* management tool would need to be used more strategically, based on manager concerns about nutrient conditions in managed lands and the potential for continued *Phragmites* invasions. Alternatively, the contribution of cattle to soil nutrient pools may be relatively minor compared to baseline levels. In that instance, grazing may be used as a management tool with little concern for negative impacts on soil nutrients.

Our broad goals for this work was to determine the effects of cattle grazing on nutrient inputs, pools, and limitations via soils and plant tissue in Great Salt Lake wetlands.

To meet these goals, we addressed the following more specific questions:

Q1) *How do nutrient inputs to soils change due to grazing in five Great Salt Lake wetland sites?*

Q2) *How does grazing affect soil nutrient pools and nutrient limitation in five Great Salt Lake wetland sites?*
Q3) *What quantity of nutrients is easily mobilized from cow manure and soils?*

General Methods

Sites and Plot Design

We established five paired 10.12 ha grazed and ungrazed ("control") plots in Great Salt Lake wetlands. One pair was located near the Crystal Unit at the Farmington Bay Waterfowl Management Area (WMA). Two pairs of plots were located at Howard Slough near the Ogden Bay WMA, and two pairs at the Harold Crane WMA unit near Willard Spur (Fig. 1). All sites were near impoundments such that managers could control wetland water level. The plots were invaded by *Phragmites australis* with an overall minimum cover of approximately 80%.

In each 10.12 ha plot, we mowed a 100 m long corridor and 0.40 ha circular area at the end of the corridor (Fig. 2). In grazed plots, this corridor allowed the cattle to access the interior of the plot, and the 10.12 ha circular area served as a ruminating area where cattle were expected to return repeatedly throughout the grazing season due to ease of movement and accessibility to water from impoundments. We expected cattle grazing activity in the rest of the plot to attenuate with increasing distance from this center area, thereby creating a gradient of grazing intensity (Thrash and Derry 1999; Todd 2006).

Measurements and sampling

We collected soil, manure, and leaf nutrient samples in all plots. At Farmington Bay, we also collected water samples. We collected soil, leaf, and water samples pre- and post-grazing in both 2015 and 2016. Manure was collected during grazing in 2015 and 2016.

For questions 1 and 2, we were interested in N, C, and P in soil, manure, and leaf nutrient pools, and nutrient limitations as shown with nutrient ratios. Total N and total C were analyzed by mass spectroscopy at the University of Utah SIRFER lab. Soil P and nitrate were analyzed by the USDA Range Research lab using a spectrophotometer and a Lachate Flow injection analyzer (Knepel 2012; Olen n.d.). Manure P was analyzed by the Utah State University Analytical Lab. Leaf P was analyzed by University of California-Davis analytical lab using an automated Lachate Flow Injection Analyzer (Prokopy 1985).

For question 3, our primary interest was in water nitrogen and phosphorus levels and if water becomes more enriched with these nutrients as it moves through the grazed plots compared to the control plots. We collected water samples at Farmington Bay from the plots' water source (the impounded ponds), pore water from installed wells, and on-site flood water. Also, we collected soil and water samples at Farmington Bay. We leached distilled water through the soil and manure samples, filtered the leached water, and analyzed it to see what nutrients could be leached from the soil and manure once the sites had been flooded. We tested all water samples (collected in field and leached water) for total dissolved nitrogen (TDN), nitrate (NO3), ammonium (NH4), and soluble reactive phosphorus (SRP) through Dr. Michelle Baker's Aquatic Biogeochemistry Lab at Utah State University using an Astoria Pacific Autoanalyzer (Baker 2011).

Q1) How do nutrient inputs to soils change due to grazing in five Great Salt Lake wetland sites?

Methods

We addressed this question by comparing the quantity of nitrogen (N), carbon (C), and phosphorus (P) entering the system via direct inputs of *Phragmites* leaf tissue vs. inputs to the system via leaf tissue converted to cow manure, which represents an indirect, more labile form of nutrient transfer. We assumed that water and atmospheric inputs were the same for the grazed verses the control plots. Nitrogen-fixing bacteria could also be another source of nitrogen. This source could be altered by grazing, but we did not address this source in this study. We used these data to estimate site-level inputs of N, C, and P to soils from less labile *Phragmites* vs. more labile cow manure (see also *Q4*). All data are from the post-grazing 2016 sampling timeframe except soil carbon, which is from post-grazing 2015 sampling due to the expense and time associated with digesting soils to remove carbonates. Soil digestion with weak acid is required to quantify the C content that is available to soil biota.

We estimated leaf nutrient input by collecting a 1 m^2 area of plant biomass, which was then dried and weighed. We used 10 biomass samples from the control plots to calculate an average mass per m^2 for stems, inflorescences per m^2 and an average number of nodes per plant (indicating number of leaves per plant). This information allowed us to roughly estimate the average mass of an individual leaf in the control 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 control and grazed plots, and 3. node frequency (i.e. leaf frequency) would remain the same even if the plant was grazed. Our assumptions may have introduced error into our estimates for several reasons. Leaf loss over the season and during collection likely occurred, leading to an underestimate of leaf mass. Grazed plant leaves tended to be smaller re-sprouts based on general field observations. Hence, our estimates may be slightly inflated for the grazed plots. It is possible that the distance between nodes is shortened as plants are grazed. This change in node length could lead to a possible underestimation of leaf frequency within the grazed plot. Nonetheless, we calculated the input of total N (TN), total C (TC), and total P (TP) from leaves by multiplying the total nutrient content (%) of leaves by the averaged leaf biomass (g/m^2).

Nutrient input from manure was estimated through a different method. The 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 et al. 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).

We estimated soil nutrient pools 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). 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.

Results

Overall, there was more leaf biomass in the control plots. Thus, the contribution of nutrients from leaves to soils was larger in the control plots as compared to the grazed plots. In general, the leaves in control plots were a source of approximately 40-53% more nutrients (TC, TN, and TP) to the soil than the leaves in the grazed plot (Table 1; Fig. 3 and 5). Assuming cattle assimilate none of this leaf tissue into their own biomass, nutrient inputs from manure should account for this difference (Fig. 6). However, when the total grazed plot inputs (manure plus leaf nutrients) are compared to the leaf resource inputs in the control plot, the inputs are very similar for TN and TP but not carbon. Even when leaves and manure carbon are added, the grazed plot has about 42% less carbon added to soils per m^2.

The quantity of nutrients in soils is very small for both the control and the grazed plots. However, the control plot had about 14% more TN, 40% more TC, but 16% less TP when compared to the grazed plots (Table 1; Fig. 4 and 5). When standard errors are considered, these differences are likely not significant.

Discussion

For leaf TC, the control plot is adding about 53% more C from the leaves. This contribution is likely due to grazing decreasing the plant biomass, not only through ingesting the leaves, but possibly also by stressing the plant and stunting its growth. Even when the manure and the leaf input are added in the grazed plot, the control plot still has a total input of 42% more C (55.06 g/m^2) into the soils than the grazed plot. Based on the generally accepted trophic level assimilation rule (Lindeman 1942), cattle should only be assimilating a small fraction of what they are eating. Therefore, assimilation alone does not account for this large difference. Other pathways for C loss from the system could be through cattle respiration of CO_2 or excretion of methane (Dong et al. 2006). Even despite these potential losses, this number is surprisingly large and may be due to some of the original assumptions made in calculating the nutrient pools. The difference in TC input between grazing and control plots reflects the difference in soil TC levels (approximately 40% less in the grazed plot). This pattern indicates that the TC input is what drives the TC soil levels rather than changes to the biogeochemical and physical processes that are occurring in the soil. Because the cattle appear to be removing carbon from the wetland system, the cattle are likely a sink for carbon. 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.

In looking at nutrients and grazing in wetlands, we were primarily interested in understanding how these nutrient changes could ultimately affect water quality and problems with eutrophication in the Great Salt Lake, but when discussing C, we need to also address how changes in soil C levels are affecting blue carbon. Blue carbon refers to the ability of oceans and costal ecosystems such as wetlands to store carbon (Davidson et al. 2017). In grazing *Phragmites* in wetlands, this function is being inhibited as seen by the huge loss of carbon. Although this is true, it may be a trade-off that land managers are willing to accept if *Phragmites* can be removed and wetlands dominated by native species can be restored (Rohal et al. 2018).

For TP, the leaf P input in the grazed plot is approximately half of the control leaf P input. As with C, this pattern is probably due to decreasing plant biomass through ingestion. The total P input of the grazed (leaf + manure) and control (leaf only) plots is very similar, so cattle assimilation or stunted plant growth does not seem to affect the inputs. Even though the inputs are roughly the same, the soil in the grazed plots has approximately 16% more TP. Since the total inputs are roughly the same, the change in soil TP is likely due to biogeochemical or physical processes that differ between treatments. Manure has a much more labile source of P that is immediately available compared to the slow release P from leaves through decomposition. For examples, in a review, Asaeda et al. (2002) found that leaf litter decays only 33-48 % in the first year depending on length of time the litter is in the aerobic layer of water, and then afterward, the litter is stored in the anaerobic layers of soil. This study also predicts that a higher ratio of P will be stored in the leaf material in the anaerobic layer than N (Asaeda et al. 2002). Also, the leaves and leaf litter that have not been eaten are fragmented by cattle hooves, potentially increasing the rate of leaf decomposition in the grazed plot compared to the control. Another possible explanation for these patterns is that hoof action churns the soils, unearthing leaves that have been stored in the anaerobic layer, thereby increasing rates of decomposition and releasing of phosphorus into the soil. This explanation is plausible based on our bulk density data that showed a decrease in soil density after grazing in 2015 (Table 2). The pH of the soil also plays a large part in the nutrient cycling of phosphorus, where phosphorus is most plant available at slightly acidic to neutral pH conditions (Mitsch and Gosselink 2015). According to research conducted by Christine Rohal and Chad Cranney (*unpubl. data*), average soil pH in Great Salt Lake wetlands is 7.9, which is neutral to slightly basic. The meta-analysis by Davidson et al. (2017) indicated that soil pH was not affected by grazing. Although pH is the primary driver of P availability, it does not appear to be the cause of the increase in P availability in our grazed plots.

While the leaf inputs of TN in the grazed plots were about half that of the control plots, the total inputs of TN and soil TN pools were very similar in the grazed and control plots. Surprisingly, cattle do not seem to affect the overall pools of TN. However, grazing may induce changes in the form of available nitrogen, and we will explore this further in questions 3 and 4.

<u>Conclusion</u>

Nutrient cycling in wetlands is a complex process, especially where grazing occurs. Overall, grazing reduced the amount of carbon being added to the wetland soils from reduced biomass production on site and losses of carbon to cows likely due to assimilation and gaseous emissions. This finding indicates that cattle are a sink for carbon from wetlands, but this carbon may be adding to atmospheric pollution. For P, cattle do not appear to be a sink for the nutrient, but the action of grazing seems to be altering soil P pools through decreased P inputs via leaf litter, and possibly stimulated rates of decomposition and mineralization of P previously bound within buried leaf material or other compounds within soil. Finally, the TN pools do not appear to be affected by grazing.

Q2) How does grazing affect soil nutrient pools and nutrient limitation in five Great Salt Lake wetland sites?

<u>Methods</u>

We addressed this question by comparing soil inorganic C:N:P ratios in ungrazed vs. grazed sites. We used the same TC, TN, and TP sources that we used to address question 2. We converted the elemental content of samples (i.e., g C or nutrient/g sample) to molar units (i.e., mol C or nutrient/g sample) and then calculated molar ratios of C:N, C:P, and N:P. We did not have values for TC in 2016, and we had limited samples from pre-grazing 2015. Therefore, we only display results containing carbon for 2015 post-grazing.

Results

For all C:N:P ratios, the grazing treatment does not appear to differ from the control (Tables 3 and 4; Fig. 6). For C:N ratio, the 2015 post-grazed mean in the control plot is slightly higher than the control, but considering the data distribution, it is likely not significant (Table 3; Fig. 6A). For C:P ratios, the data distributions largely overlap (Table 3; Fig. 6B). Although the grazed plot C:P is slightly elevated, it is also likely not significant (Table 3; Fig. 6B).

For N:P, we are again seeing very little difference even between seasons and years. The mean ratio values are also all very small (less than 4) (Fig. 6C; Table 3). We do see a large jump in the mean N:P value in the control in post-2016, but the error is very large and likely not significant (Fig. 6C; Table 3).

The C:N:P are approximately 15:3:1 for the control and 23:2:1 for the grazed averaged across both years (Table 4). When error is considered, these values do not appear to be different. On average, all sites and treatments combined have an C:N:P of 19:2:1.

Discussion

Lowered soil C:N and C:P ratios at grazed relative to control sites or post-grazing relative to pre-grazing would be another indication that the addition of cow manure input or a decrease in leaf litter input is altering nutrient pools. However, we are not seeing this lowering of ratios. One study showed that differences in C:P and N:P were only found in the top 10 cm of soil (Wang et al. 2014). For this study, we collected and homogenized approximately the top 30 cm of soil. Sampling to this depth of soil could have diluted or hidden any real differences if they existed. Also, the top organic litter layer of the soil sample (the O horizon) was removed during sampling which could also explain why we are not seeing any differences.

A literature review by Cleveland and Liptzin (2007) showed that global C:N values ranged from 2-30. Our values for C:N ranged from approximately 11 to 37, with an average around 21 (Tables 5 and 6). This finding indicates that our soils are slightly to very limited in N. This finding is not surprising because flooded soil is often limited in N due to denitrification that occurs under anaerobic conditions (Mitsch and Gosselink 2015).

One study found that N:P values range from 1 to 77 on a global scale (Cleveland and Liptzin 2007), and our values range from a high of approximately 8 to less than 1. Our low N:P values indicate that our soils are limited by N and not by P.

Global C:N:P values in soils from forests and grasslands are relatively restricted to 186:13:1 and soil microbial biomass at 60:7:1 (Cleveland and Liptzin 2007). A more recent global synthesis for forest, grassland, shrub, tundra, wetland, desert, cropland, and pasture found mean C:N:P ratios of 287:17:1 for soil and 42:6:1 for soil microbial biomass (Xu, et al. 2013). This same study showed that ratios in wetlands alone averaged 1347:72:1 (Xu et al. 2013). Our study values were low around 19:2:1, which is lower than but more closely reflects the global ratios for soil organisms (60:7:1 to 42:6:1 depending on source) rather than soil itself. This discrepancy could be due to a variety of factors including that the P was analyzed using different preparation methods and a different analytical machine.

Conclusion

C:N:P ratios in soils of the Great Salt Lake are low relative to reports from global syntheses. Wetland sites appear to be N limited, given high C:N and low N:P ratios in soils. Grazing did not appear to significantly affect the C:N:P ratios of soils.

Q3) What quantity of nutrients is easily mobilized from cow manure and soils at Farmington Bay?

Methods

We addressed this question by conducting leaching studies of cow manure and soils collected from Farmington Bay Waterfowl Management Area (FBWMA). First, we collected 15 fresh, cow manure samples and 15 soil samples (6 control and 9 grazed). In the lab, 25g of sample (wet weight of soils or cow manure) was mixed in 500ml deionized water agitated on a table shaker for 5 minutes to mimic floodwater inundation and movement. Slurries were settled for 10 minutes and decanted liquid was filtered with a 0.7µm ashed glass fiber filter. The leachate solutions were analyzed of soluble reactive phosphate (SRP-P), total dissolved nitrogen (TDN), total dissolved organic nitrogen (TON), ammonium (NH_4-N), and nitrate (NO_3-N) concentration as describe in the general methods section. We did not obtain NO_3-N concentrations for manure because the leachate was too cloudy to produce accurate results. We estimated site-level inputs of nutrients to soils or flood water from cow manure by combining these chemical analyses with estimates soil bulk density and cow manure production discussed previously in question 2. Soil leaching studies were carried out post-grazing in 2016. We estimated site-level flux of nutrients from soils to surface waters by comparing leaching study data to chemistry data for pond water, floodwater, and well water at FBWMA. Pond and well water samples were collected pre- and post-grazing in both years, and flood water was collected only post grazing in both years when the site was flooded. Flood and pond water was collected using a grab sample method. Acid-washed Nalgene bottles were rinsed 3 times in the water type of interest. The bottle was then filled as full as possible to minimize air being trapped in the bottle. The samples were kept cold until processed within 24hrs of collection. In the lab, the samples were filtered with a 0.7µm ashed glass fiber filter, and frozen until analyzed by the lab. The well water was collected from the wells after the wells had been drained and refilled twice. The acid washed bottles were rinsed 1-3 times (depending on water availability) with the same water we were collecting from. Like the other water samples, they were kept cold (in a cooler or refrigerator) until processed (within 24hrs). We filtered the samples and froze them until ready for analysis. These water samples were also tested for SRP-P, TDN, TON, NH_4-N, and NO_3-N concentrations. We subtracted the pond water results from the flood water results to determine if the floodwater was being enriched with nutrients as it traveled over the site as compared with the original pond water concentrations. We also examined water from the wells in all pre- and post-grazing timeframes to determine if the soil pore water is becoming enriched in the grazed vs. the control plots.

Results

The soil leachate nutrient levels did not seem to differ for SRP-P, TDN, TON, or NH_4-N between the grazed and the control sites (Table 5; Fig. 7). However, NO_3-N levels in the soil leachate appear to be elevated in the grazed sites compared to the control sites (0.09 ±0.01 mg/L and 0.06 ±0.01 mg/L respectively) (Table 5; Fig. 7). The manure leachate had nutrient levels that were several magnitudes higher than the soil with SRP-P increased by 43x, TDN increased by 62x, and NH_4-N increased by 75x (Table 5; Fig. 7-8). We do not have data for NO_3-N in manure leachate.

To determine if these nutrients were indeed being mixed into the flood water, we examined the difference in the nutrient concentrations between flood water and pond water. The results for each nutrient form are as follows:

- Floodwater was consistently elevated in SRP-P relative to pond water and differences between the two water sources were greater in 2015 relative to 2016 (Table 7; Fig. 9A). In addition, the grazed site contributed approximately 2x and 3x more P to floodwater than control site in 2015 and 2016, respectively (Table 7; Fig. 9A).
- Trends in TDN differences between the two water sources were less clear than SRP-P patterns. The difference in TDN concentrations between pond water and floodwater increased in the control plot between 2015 and 2016 but decreased in the grazed plots (Table 7; Fig. 9B). The grazed site contributed nearly 2x more TDN to floodwater than the control site in 2015, whereas the grazed site contributed 1.5x less TDN to floodwater relative to the control site in 2016 (Table 7; Fig. 9B).
- The grazed site was a source of DON to floodwater in 2015, but a sink for DON in 2016 when 2.36 mg DON/L water was sequestered (Table 7; Fig. 9C).
- Floodwater and pond water had similar concentrations of NH_4-N, except the grazed site was a source of NH_4-N to floodwater in post-2016 when floodwater concentration was 2.46 mg/L relative to <0.01mg/L in pre-grazing in 2016 (Table 7; Fig. 9C). This pattern opposes the pattern we saw with DON (Fig. 9B and C), suggesting flooding may spur mineralization of organic forms of N resulting in elevated NH_4-N availability. There was also one high data point that could have driven this result in grazed site in 2016 (mean without outlier for grazed site post-2016=0.07 mg/L).
- Both control and grazed sites were sinks for NO_3-N, given that we observed negative values when we compared NO_3-N concentrations in flood water and pond water (Table 7; Fig. 9D). More NO3-N was sequestered by wetland sites in 2016 relative to 2015. In both years, more NO_3-N was sequestered by the control plot relative to the grazed plot (Table 7; Fig. 9D).

Finally, we assessed nutrient concentrations of groundwater at both control and grazed sites. In theory, these wells collected water as it was moving through the soil. These values should reflect what is being leached from the soil itself, not just what is moving over top of the soil.

- Patterns in well water SRP-P showed interannual variation. In 2015, well water SRP-P increased from pre- to post-grazing sampling dates (Table 8; Fig. 10A). The opposite occurred in 2016, when SRP-P was higher in wells prior to grazing for both the control and the grazed plots (Table 8; Fig. 10A). However, SRP-P concentrations were similar amongst the control and grazed plots (Table 8; Fig. 10A).
- TDN concentrations in well water were relatively stable in 2015 (Table 8; Fig. 10B). In 2016, TDN concentrations increased from pre- to post-grazing sampling dates in both the control sites and the grazed sites (Table 8; Fig. 10B
- Concentrations of DON were relatively constant for all sampling times and treatments (Table 8; Fig. 10C). The variation for post-grazing at the grazed plot in 2016 is very high, but the mean is like other treatments and sampling dates (Table 8; Fig. 10C).

- NH_4-N was also relatively similar for all treatments and sampling dates (Table 8; Fig. 10D). However, grazed sites in post-2016 had much variation.
- For NO_3-N, all values were very small (<0.05mg/L) (Table 8; Fig. 10E) and similar to one another, with the exception of post-2016 control sites at which values were approximately 2x larger than other treatments and sampling dates and had a very large standard error range.

Discussion

The results from our in-lab leachate experiment and our in-situ flood minus pond water experiment differed. While the leachate study showed little difference between the soil SRP-P, TDN, DON, or NH_4-N nutrient levels, the flood minus pond water analysis did display differences. The leachate study showed an increase in NO_3-N in the grazed plots, but there was no difference between the pond and flood water NO_3-N levels. This finding is a little surprising because as we discussed, as part of our mass balance comparisons, only total levels of C and P changed in the soils but not N. There was also a lot of variation within the well water, but none of it could be clearly linked to grazing.

In the flood minus pond water experiment, SRP-P increased in both the control and grazed sites over time, but the numbers were overall higher in the final grazed site sampling, having 1.17 mg/l more SRP-P on average than the final control numbers (Table 7). This finding is likely because manure transforms leaf P into the more labile form of P, SRP-P. P cannot be lost to the atmosphere, and its bioavailability is highest around neutral to somewhat acidic conditions (Mitsch and Gosselink 2015). Since our sites have a pH close to neutral, it is likely that a large percentage of P at our sites were in this bioavailable form.

The flood minus pond water experiment showed that grazed sites were sinks for TDN and DON but sources of NH_4-N. These N transformations are not surprising, given we found our sites to be limited with respect to N. Nitrogen mineralization is the process that changes DON to NH_4-N. A study conducted in estuaries in China found that temperature can be a strong driver of nitrogen mineralization (Lin et al. 2016). With the opening of the *Phragmites* canopy through grazing, it is likely that our water column and soils were much warmer in the grazed vs. the control plot. This increase in temperature could have caused the drop in TDN and DON and the spike in NH_4-N. Further study would be needed to verify that temperature is indeed the driver. Also, we had a value that was uncharacteristically high that could have made these values seem more extreme than reality.

NO_3-N was lower in the flood water than in the pond water which contrasts with the leachate study. These findings indicate that NO_3-N is being lost through denitrification or assimilation, or that production of NO_3-N through nitrification is being slowed. One of the strongest drivers of denitrification is temperature (Mitsch and Gosselink 2015), but since both the grazed plot and the control plot are undergoing this loss, increased temperatures through opening of the canopy may not be the driver. Also, dissolved oxygen (DO) levels could have been too low to support conversion of NH4-N to NO3-N (Mitsch and Gosselink 2015). This explanation is contrary to some of the other hypotheses, however, because we posited that the hoof action of the cattle is increasing the DO levels within the water and upper soil surfaces. Lastly, we hypothesize that biotic uptake of NO3-N by *Phragmites*, other plants, or microbes also may have been responsible for the observed losses of NO3-N from floodwater.

The levels of nutrients that can be leached from manure are large but not reflected in floodwater nutrient concentrations. Our leaching study showed that 43-75x more nutrients can be leached from the manure than from the soil. During the study, we also observed that manure is easily broken down into soluble organic matter when water is added. This slurry contains a large amount of nutrients that could potentially flow off site to Farmington Bay. Our field values did not show such dramatic increases likely due to dilution in a large volume of floodwater. For the leaching study, we added 25g of sample to 500ml of water. When roughly calculating manure (g) to water (L) ratios, the leachate study had 50 g of manure to 1L of water. In 2015, the grazed plot had approximately 10g of manure to 1L of water, and in 2016, 1g of manure to 1L of water. Therefore, the leachate study was 5-50x more concentrated than what we were seeing in the field. Also, dramatic increases in floodwater nutrients may not have been seen because manure was added slowly over time rather than all at once. This allowed time for nutrients in the manure to be diluted through rain and fixed by plants and bacteria or through biogeochemical transformations such as volatilization and lost to the atmosphere. One study tested the decomposition of manure and found that uncovered and watered manure lost nutrients over time (DICKINSON and CRAIG 1990). In fact, TP nutrient content of uncovered and watered manure decreased by 54 and 72% respectively over 85 days (Dickinson and Craig 1990). For TN, there was a decrease of 64% and 77% in the uncovered and watered manure respectively (Dickinson and Craig 1990). Nonetheless, this study also found that these values were not reflected in the soil (Dickinson and Craig 1990). They hypothesized that root growth was stimulated and was utilizing nutrients as soon as they were entering the soils. It is likely that we are seeing a similar loss, and the combination of dilution, biogeochemical changes over time, and plant and microbial assimilation account for the differences between the leachate study (which was conducted with fresh manure) and the in-situ water sampling.

Conclusion

Our leaching study indicated the potential for nutrient transformations in wetland sites due to grazing and flooding. We found increases in SRP-P and NH_4-N in floodwaters relative to pond water that inundated sites. Floodwater with these nutrients drain to Farmington Bay, which may exacerbate existing eutrophication of the bay. Surprisingly, NO_3-N levels, the most mobile form of N (Mitsch and Gosselink 2015), were near zero. Also, the grazed wetland plots still served as an excellent filter of TDN and DON, actually reducing values dramatically when compared with the other data points. Because more NH_4-N is being formed than DON being lost, this grazed site is a “hot site” for N transformation.

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Tables

Table 1. Bulk nutrient means ± 1 standard error (SE)

Sample type	Treatment	Mean g N m^{-2} post-2016	SE g N m^{-2} post-2016	Mean g C m^{-2} post-2016	SE g C m^{-2} post-2016	Mean g P m^{-2} post-2016	SE g P m^{-2} post-2016
Leaves	Control	6.19	0.43	123.86	7.70	0.58	0.044
Leaves	Grazed	3.74	0.59	58.65	7.37	0.29	0.057
Manure	Grazed	4.19	0.85	12.70	3.38	0.19	0.055
Total input	Grazed	7.94	1.44	71.35	10.75	0.48	0.11
		Mean g N cm^{-3} post-2016	**SE g N cm^{-3} post-2016**	**Mean g C cm^{-3} post-2015**	**SE g C cm^{-3} post-2015**	**Mean g P cm^{-3} post-2016**	**SE g P cm^{-3} post-2016**
Soil	Control	0.00014	1.89E-05	0.0082	0.0014	3.96E-07	4.83E-08
Soil	Grazed	0.00012	1.68E-05	0.0049	0.0035	4.74E-07	2.83E-08

Table 2. Summary of soil bulk density values.

Treatment	Season	# of samples	Mean bulk density (g cm^{-3})	SE bulk density
Control	Pre_2015	5	1.04	0.07
Control	Post_2015	5	1.08	0.02
Control	Pre_2016	5	0.23	0.01
Control	Post_2016	5	0.22	0.01
Grazed	Pre_2015	5	1.17	0.04
Grazed	Post_2015	4	0.87	0.22
Grazed	Pre_2016	5	0.24	0.01
Grazed	Post_2016	5	0.24	0.01

Table 3. Soil nutrient ratio values by season and treatment. Missing values reflect when carbon data was not able to be obtained.

Treatment	Season	# of samples	Mean C:N (moles)	SE C:N	Mean C:P (moles)	SE C:P	Mean N:P (moles)	SE N:P
Control	Pre_2015	1	38.07		12.73		0.78	
Control	Post_2015	5	13.49	2.34	18.71	8.06	1.51	0.58
Control	Pre_2016	5					2.65	1.34
Control	Post_2016	5					4.50	3.49
Grazed	Pre_2015	3	15.38	1.77	21.12	14.45	1.35	0.87
Grazed	Post_2015	5	11.80	0.77	22.94	6.82	1.87	0.43
Grazed	Pre_2016	5					2.27	0.58
Grazed	Post_2016	5					1.39	0.62

Table 4. Soil nutrient ratios by treatment.

Treatment	# of samples	Mean C:N (moles)	SE C:N	Mean C:P (moles)	SE C:P	Mean N:P (moles)	SE N:P
Control	5	15.03	3.86	15.25	5.23	2.73	1.05
Grazed	5	22.71	4.13	41.79	11.83	1.78	0.34

Table 5. Mean nutrient content of the leachates of soil and manure ± 1 standard error (SE). There are no data for manure NO_3-N.

Leachate sample type	Treatment	Mean NH_4-N (mg/L) ± 1 SE	Mean SRP_P (mg/L) ± 1 SE	Mean TDN (mg/L) ± 1 SE	Mean NO_3-N (mg/L) ± 1 SE
Soil	Control	0.05 ± 0.00	0.05 ± 0.00	0.13 ± 0.02	0.06 ± 0.01
Soil	Grazed	0.04 ± 0.01	0.06 ± 0.01	0.10 ± 0.02	0.09 ± 0.01
Manure	Grazed	3.08 ± 0.18	2.44 ± 0.15	7.40 ± 0.31	

Table 6. Mean ± 1 standard error (SE) and median manure leachate nutrients upscaled to total input at the site level in 2016.

Manure leachate	Mean ± 1 SE	Median
TDN g/site	8304.90 ± 344.76	8249.46
NH4-N g/site	3460.51 ± 206.91	3341.11
SRP-P g/site	2738.90 ± 173.70	2936.83

Table 7. Nutrient content of the water collected on site (flood water) minus the source of water to the site (pond) water.

Treatment	Season	SRP-P (mg/L)	TDN (mg/L)	TON (mg/L)	NH_4-N (mg/L)	NO_3-N (mg/L)
Control	Post_2015	0.09	0.30	0.25	-0.05	-0.01
Control	Post_2016	0.36	0.72	0.61	0.13	-0.02
Grazed	Post_2015	0.57	0.99	1.00	0.00	0.00
Grazed	Post_2016	1.53	0.28	-2.36	2.46	-0.01

Table 8. Nutrient content of water collected from wells on site by treatment and season ± 1 standard error (SE).

Water source	Treatment	Season	Mean SRP-P (mg/L) ± 1 SE	Mean TDN (mg/L) ± 1 SE	Mean TON (mg/L) ± 1 SE	Mean NH_4-N (mg/L) ± 1 SE	Mean NO_3-N (mg/L) ± 1 SE
Well	Grazed	Pre_2015	1.54 ± 0.21	1.09 ± 0.22	0.81 ± 0.24	0.27 ± 0.12	0.02 ± 0.01
Well	Control	Post_2015	2.58 ± 0.56	0.86 ± 0.05	0.55 ± 0.02	0.30 ± 0.05	0.01 ± 0.00
Well	Grazed	Post_2015	2.50 ± 0.08	1.01 ± 0.16	0.43± 0.17	0.57 ± 0.01	0.01 ± 0.00
Well	Control	Pre_2016	2.53 ± 0.31	1.17 ± 0.12	0.76 ± 0.10	0.37 ± 0.06	0.04 ± 0.01
Well	Grazed	Pre_2016	2.03 ± 0.21	1.34 ± 0.31	0.92 ± 0.16	0.41 ± 0.16	0.02 ± 0.00
Well	Control	Post_2016	1.54 ± 0.36	2.23 ± 0.29	1.70 ± 0.25	0.30 ± 0.05	0.23 ± 0.14
Well	Grazed	Post_2016	1.43 ± 0.22	4.08 ± 2.59	1.03 ± 0.43	3.05 ± 2.77	0.00 ± 0.00

Table 9. Nutrient data for the flood and pond water samples ± 1 standard error (SE).

Treat-ment	Sample type	Season	Mean SRP-P (mg/L)	SE SRP-P	Mean TDN (mg/L)	SE TDN	Mean DON (mg/L)	SE DON	Mean NH_4-H (mg/L)	SE NH4-H	Mean NO_3-N (mg/L)	SE NO3-N
Control	Flood	Post_2015	0.62	0.30	0.81	0.03	0.65	0.12	0.04	0.01	0.01	0.00
Control	Flood	Post_2016	0.63	0.16	1.85	0.18	1.66	0.15	0.18	0.05	0.01	0.01
Grazed	Flood	Post_2015	0.93	0.31	1.74	0.75	1.64	0.74	0.09	0.01	0.00	0.00
Grazed	Flood	Post_2016	1.84	1.42	1.53	0.30	-1.21	2.65	2.55	2.48	0.00	0.00
Control	Pond	Post_2015	0.54	0.06	0.51	0.07	0.40	0.08	0.10	0.02	0.02	0.01
Control	Pond	Post_2016	0.27	0.01	1.13	0.01	1.05	0.03	0.05	0.01	0.03	0.03
Grazed	Pond	Post_2015	0.36	0.18	0.75	0.05	0.65	0.08	0.09	0.03	0.01	0.00
Grazed	Pond	Post_2016	0.31	0.02	1.25	0.01	1.14	0.01	0.09	0.01	0.02	0.02

Figures

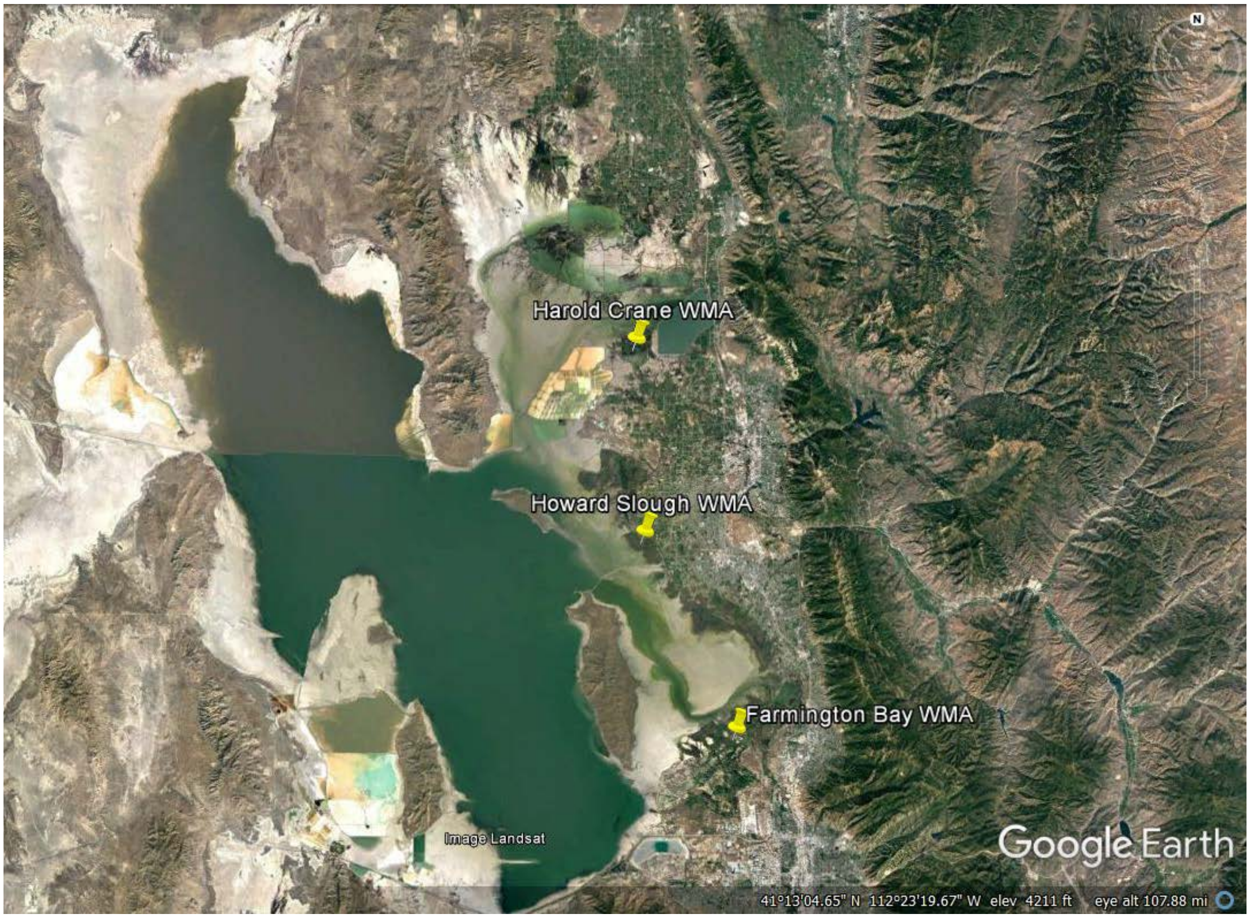

Fig. 1. Map of Great Salt Lake with yellow pins marking study sites. Basemap courtesy of Google Earth. The coordinates of the sites were:

- FB Grazed plot Lat: 40°52'56.73"N; Long: 112° 1'38.18"W
- FB Control plot Lat: 40°53'4.20"N; Long: 112° 1'37.90"W
- HS East Grazed plot Lat: 41° 6'52.89"N; Long: 112° 8'21.86"W
- HS East Control plot Lat: 41° 6'53.28"N; Long: 112° 8'12.46"W
- HS West Grazed plot Lat: 41° 6'54.42"N; Long: 112° 9'2.17"W
- HS West Control plot Lat: 41° 6'57.97"N; Long: 112° 9'12.66"W
- HC Grazed 1plot Lat: 41°21'58.86"N; Long: 112° 8'49.16"W
- HC Control 1 plot Lat: 41°21'58.51"N; Long: 112° 9'0.04"W
- HC Grazed 2 plot Lat: 41°21'50.23"N; Long: 112° 9'25.25"W
- HC Control 2 plot Lat: 41°21'45.38"N; Long: 112° 9'33.17"W

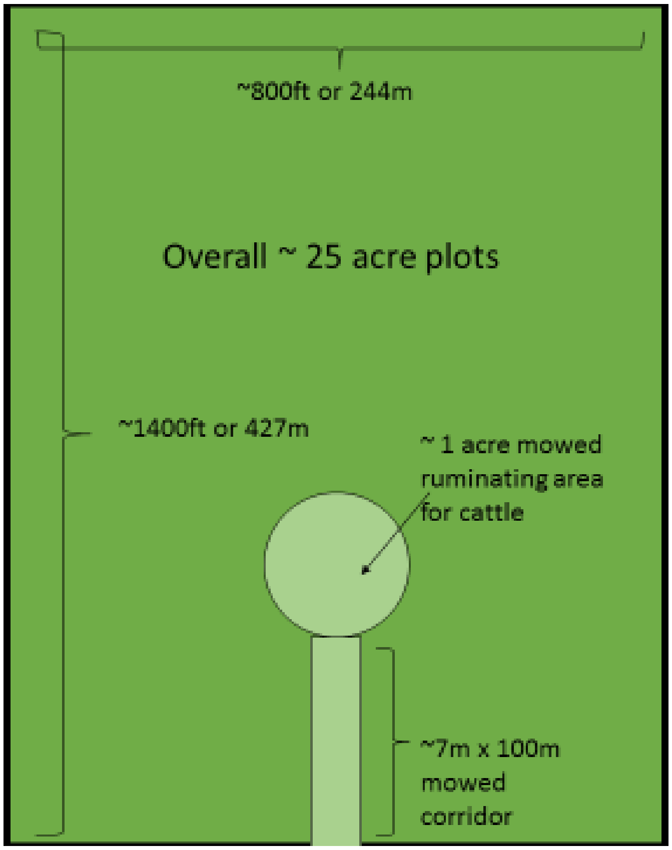

Fig. 2. Plot size and layout. Each site contained two plots, one grazed and one control.

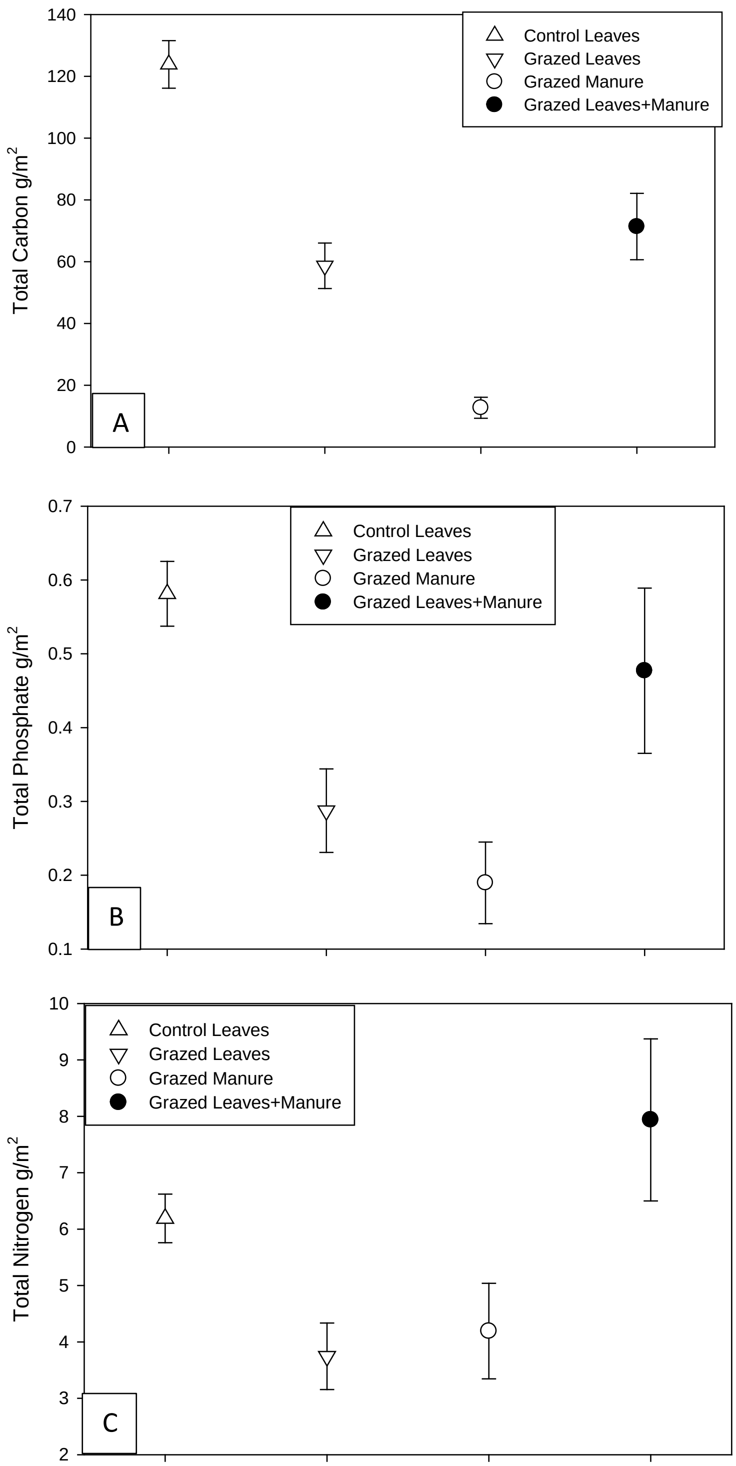

Fig. 3. Leaf, manure, and leaf plus manure bulk nutrient values for A. carbon, B. phosphate, and C. nitrogen.

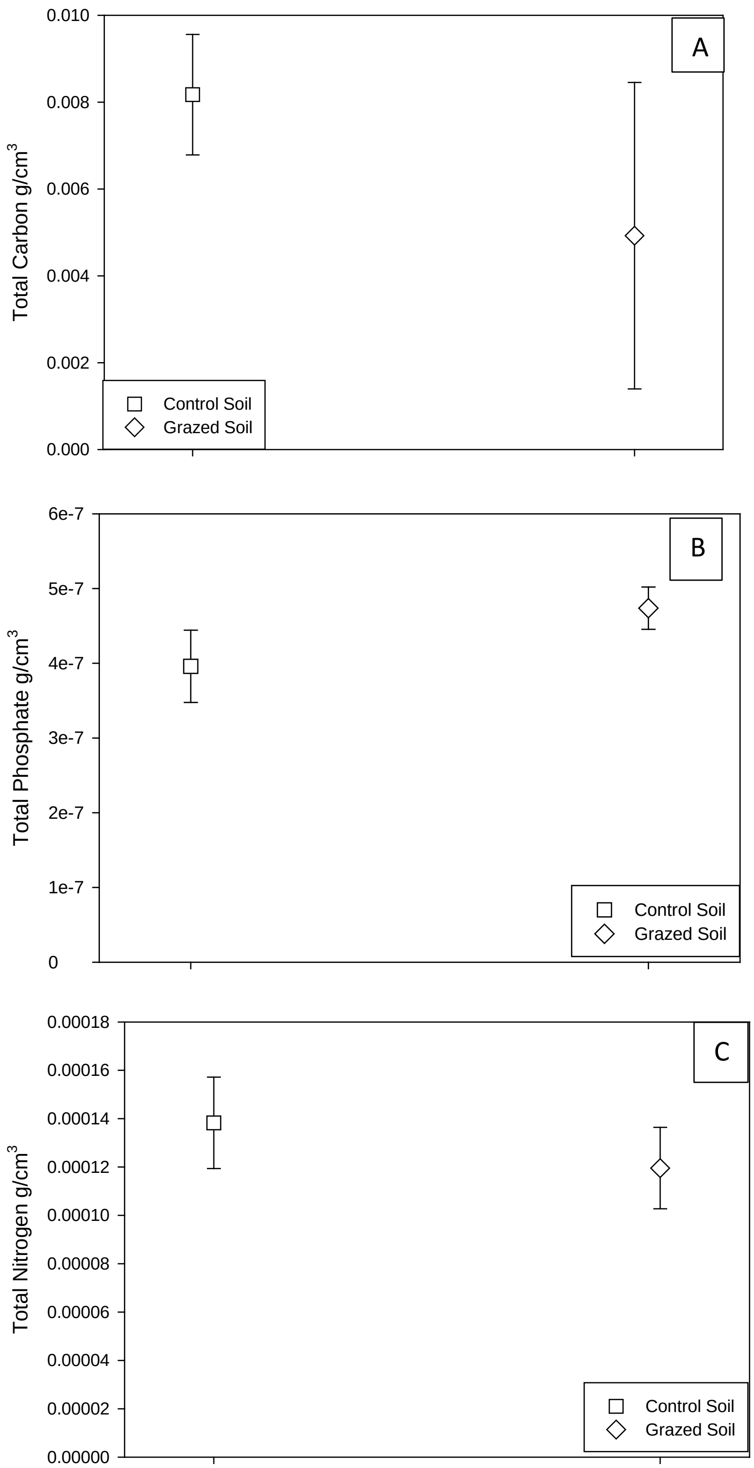

Fig. 4. Soil bulk nutrient values for A. carbon, B. phosphate, and C. nitrogen.

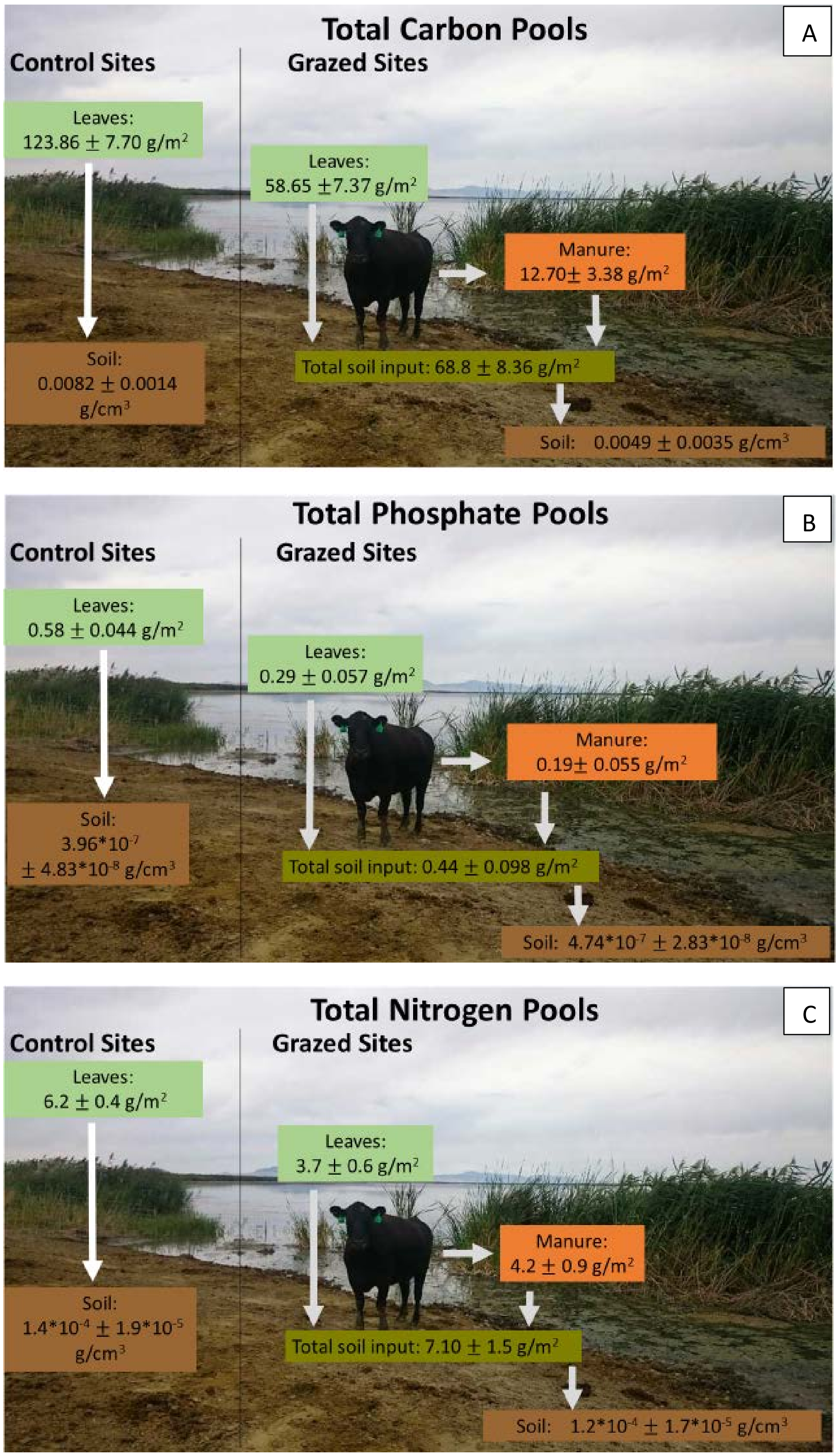

Fig. 5. Nutrient pool flow charts for A. carbon, B. phosphate, and C. nitrogen.

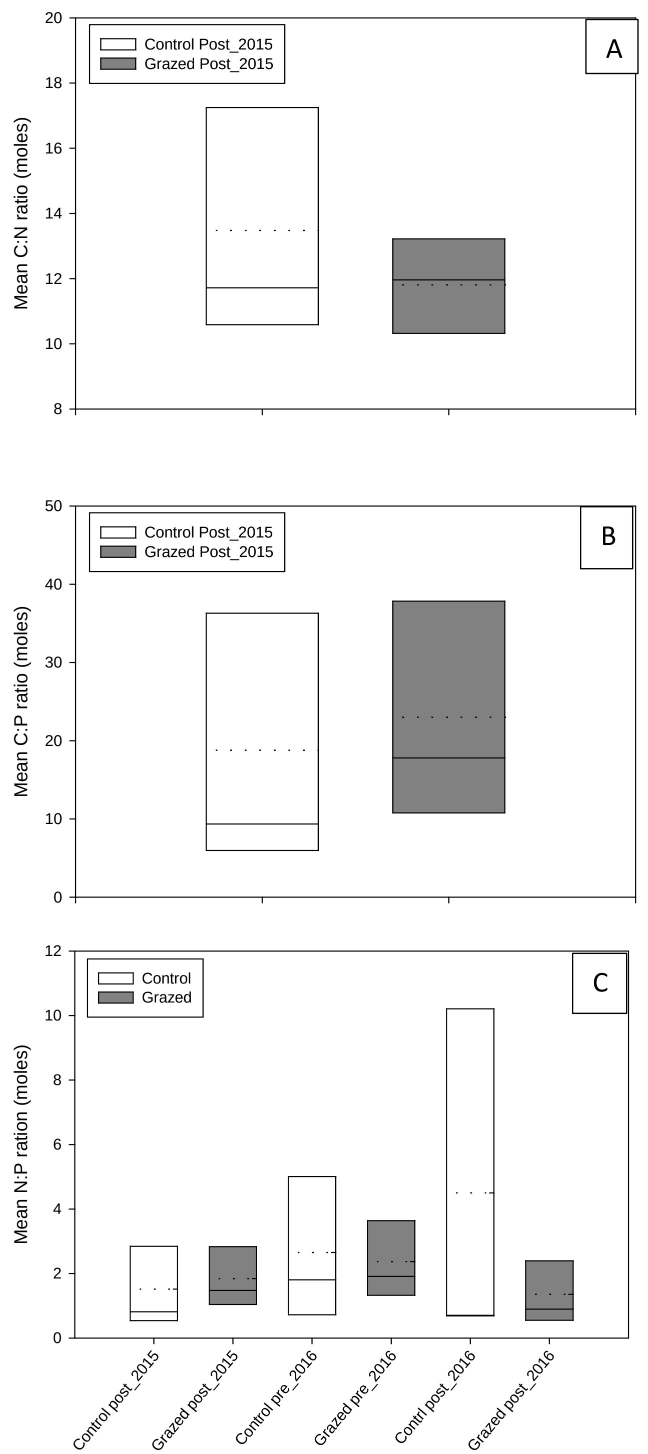

Fig. 6. Soil nutrient ratios comparing grazed and control plots in a box plot where the solid line is the median and the dotted line is the mean. The box contains 100% of the data because we have no outliers for A. mean C:N, B. mean C:P, and C. mean N:P.

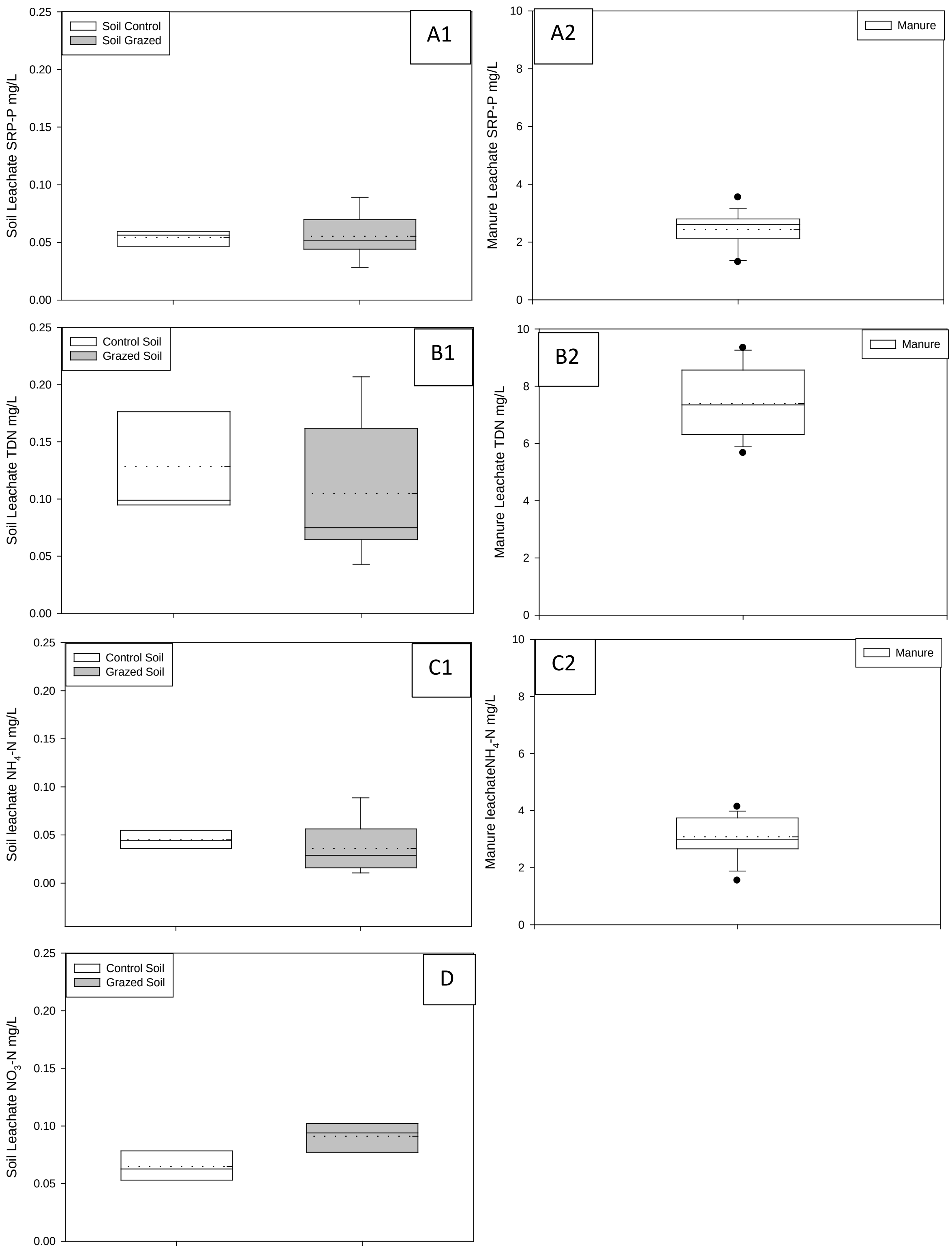

Fig. 7. Box and whisker graphs showing median (solid line) and mean (dotted line) of nutrients from leachate solutions. The box contains 25-75% of the data, and whiskers contain 10-90% of the data. Dots represent outliers. This figure includes A1) SRP-P (mg/L) of soil leachate for the control and grazed plots,

A2) SRP-P (mg/L) of manure leachate, B1) TDN (mg/L) of soil leachate for the control and grazed plots, B2) TDN (mg/L) of manure leachate, C1) NH_4-N (mg/L) of soil leachates of the control and grazed plots, C2) NH_4-N (mg/L) of the manure leachates, and D) NO_3-N (mg/L) of soil leachates of the control and grazed plots. The manure samples were too cloudy to get nitrate readings.

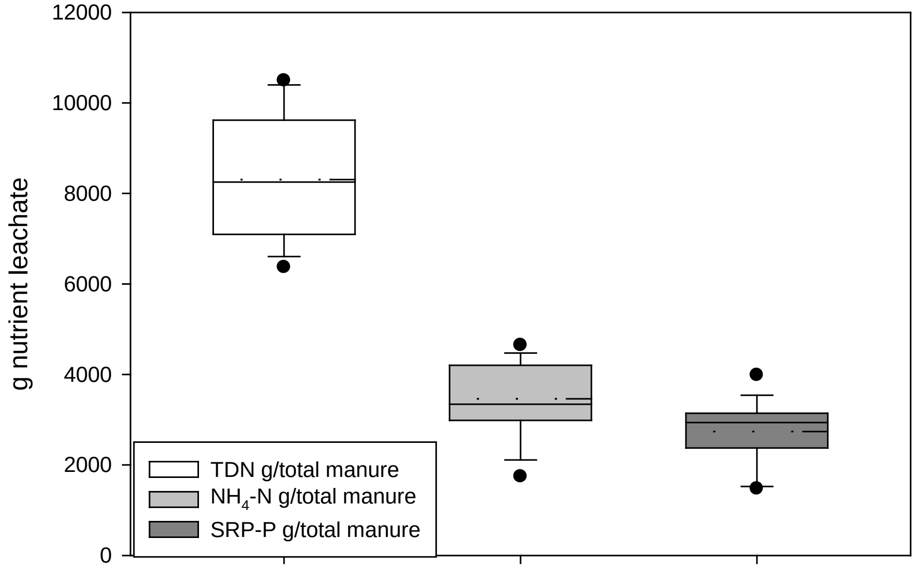

Fig. 8. An estimate of total nutrients (TDN, NH_4-N, and SRP-P) leached from all manure produced on site in 2016. This box and whisker graph shows the mean as a dotted line and the median as a solid line. The box contains 25-75% of the data, and whiskers contain 10-90% of the data. Dots display outliers.

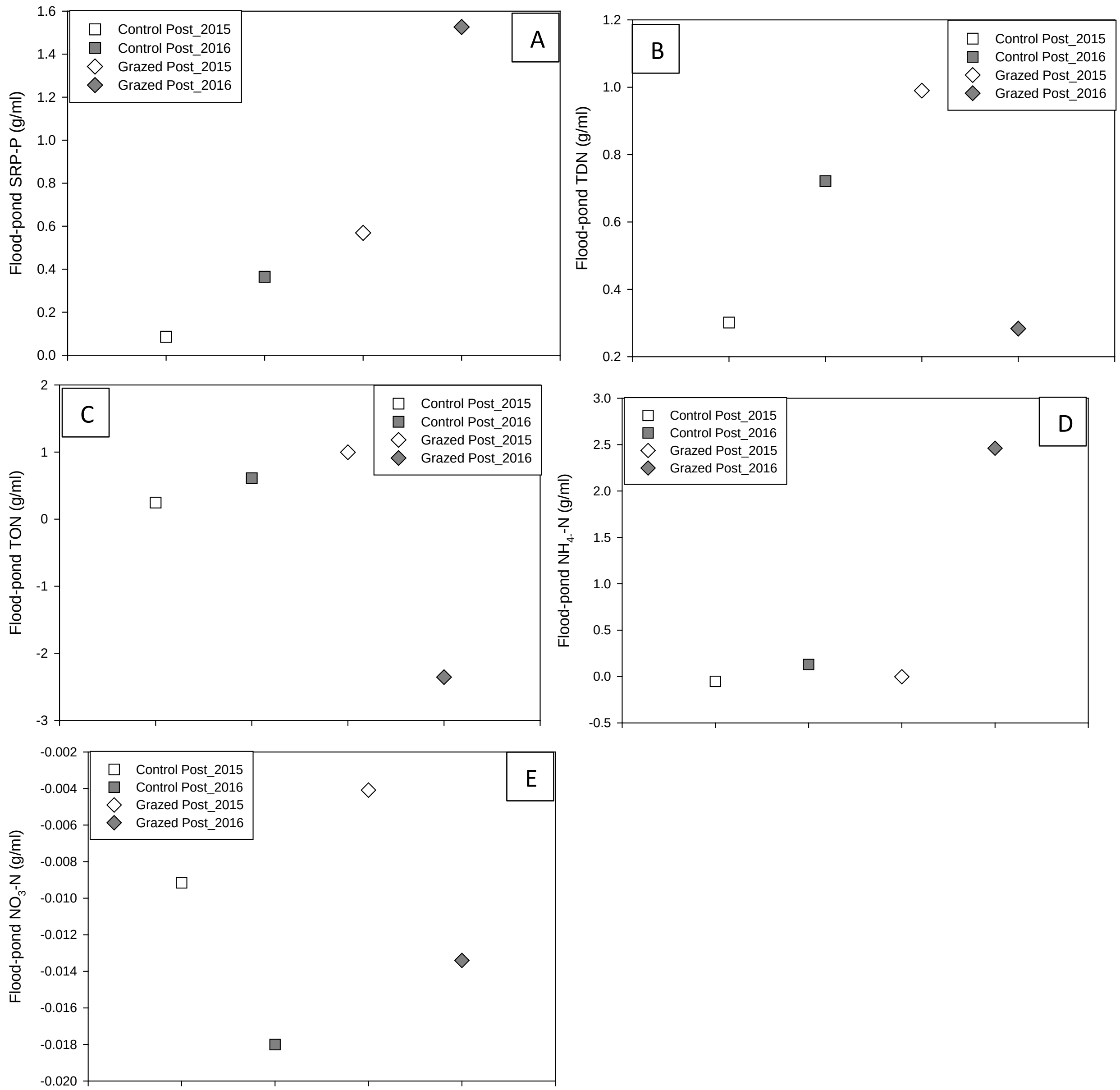

Fig. 9. Nutrients in flood water-pond water: A. soluble reactive phosphorus (SRP-P), B. total dissolved nitrogen (TDN), C. total organic nitrogen (TON), D. ammonium (NH_4-N), and E. nitrate (NO_3-N).

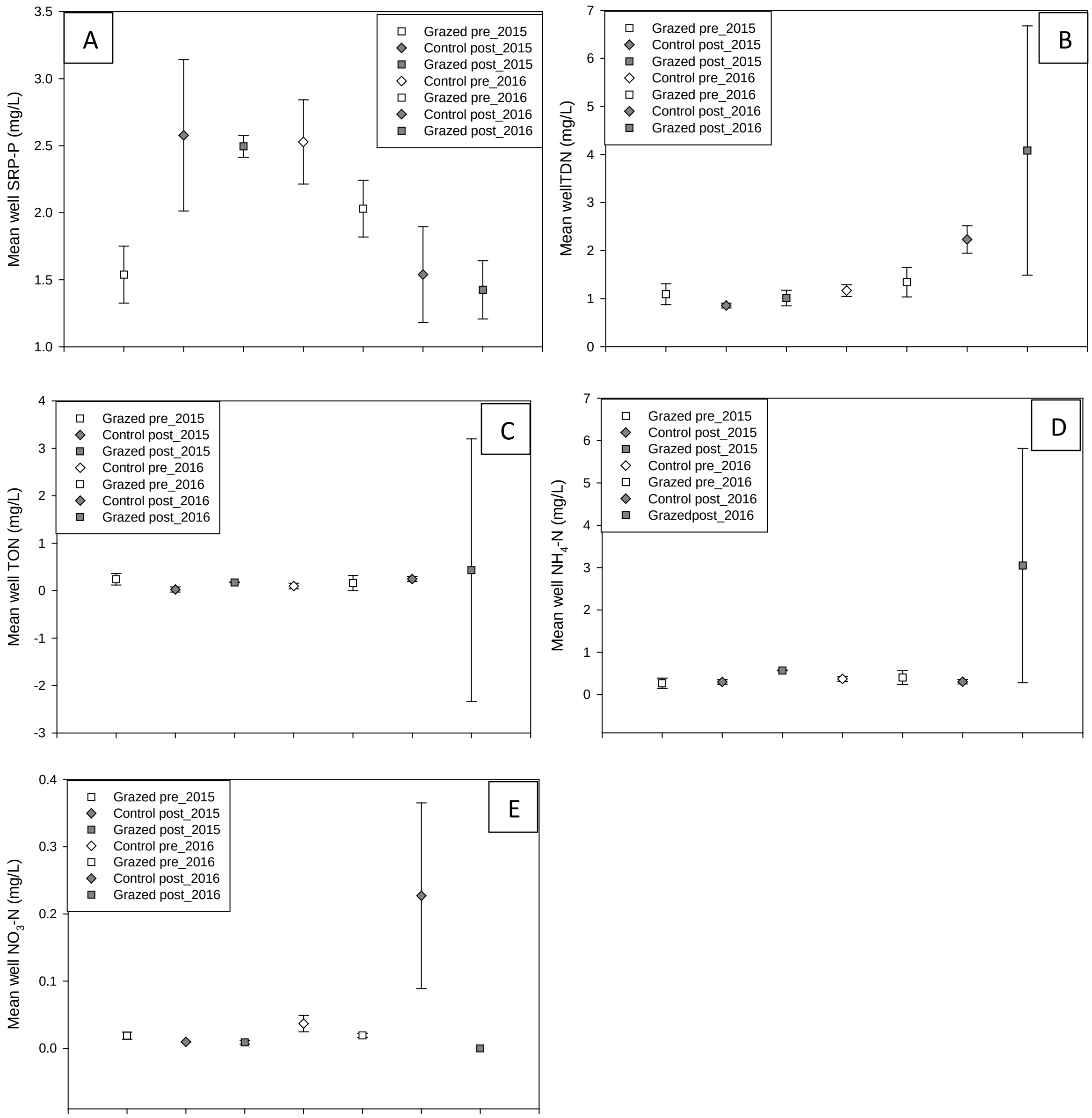

Fig. 10. Well water nutrient levels: A. soluble reactive phosphorus (SRP-P), B. total dissolved nitrogen (TDN), C. total organic nitrogen (TON), D. ammonium (NH_4-N), and E. nitrate (NO_3-N).