
Nitrogen loading in agricultural water drainage in Northfield, Minnesota

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Abstract

Fertilization of agricultural fields increases the amount of inorganic nitrogen in soils and local water sources. This anthropogenic change to the nitrogen cycle leads to the pollution of streams, lakes, rivers, and ocean water. Additionally, inorganic nitrogen in soil and water can cause an increase in nitrification and denitrification, two processes that produce the greenhouse gas nitrous oxide as a byproduct. Controlling the use of fertilizers and managing agricultural drainage are critical ways to reduce nitrogen pollution and nitrous oxide production in waterways. In this study, I examined the water chemistry from five difference sources of agricultural drainage water in Northfield, MN. Sampling sites induced fields with corn and soybean cultivation and no till, strip till, and conventional tilling practices. Drainage management varied across sample sites and included a field with surface drainage, fields with artificial subsurface drainage, and one example of artificial subsurface drainage connected to a saturated riparian buffer. During the sampling period during autumn 2016, I found no significant difference in nitrate, total nitrogen, or dissolved organic carbon concentrations between the five different sampling sites. Analysis of the saturated buffer strip demonstrated that the buffer removes nitrate and adds dissolved organic carbon to the drainage water. At one sampling site, artificial subsurface drainage was analyzed for dissolved nitrous oxide concentrations from October 2015 through February 2016. There was a significant positive relationship between nitrate and nitrous oxide concentrations in this site, with highest concentrations occurring after the field was harvested.

Introduction

Since the invention of the Haber-Bosch process that artificially fixes nitrogen gas, humans have dramatically changed the global nitrogen cycle. By increasing the amount of inorganic nitrogen through the use of artificial fertilizer, agricultural productivity has increased. However, excess nitrogen from fertilization is lost from fields in run-off that pollutes surface waters. Nitrifying and denitrifying bacteria metabolize some of this excess nitrogen, producing nitrous oxide (N_2O) as a byproduct. Agriculture is a large source of nitrous oxide, which can be emitted directly from soil and indirectly from surface and ground water (Outram and Hiscock 2012).

In Minnesota, an important anthropogenic change to the nitrogen cycle occurs when ammonia fertilizer is applied to agricultural fields. Ammonia is converted to ammonium, which is converted to nitrate (NO_3^-) by nitrifiers in an aerobic environment. In the process of nitrification, bacteria produce nitric oxide (NO) and nitrous oxide (N_2O) as byproducts. In the process of denitrification, microbes in an anoxic environment convert nitrate to nitrogen gas, also creating nitric oxide and nitrous oxide as byproducts. Increasing the amount of inorganic nitrogen in a system increases the potential for nitrification and denitrification and thus the potential production of nitrous oxide. For soils with up to 60% water filled pore space (WFPS), nitrifying bacteria metabolize ammonium and are responsible for the production of nitrous oxide (Linn and Doran 1984). High rates of nitrous oxide production are also observed when soils have greater than 60% WFPS, creating an anaerobic environment where denitrifying bacteria metabolize nitrate (Davidson and Verchot 2000) (Liu et al. 2007). Coupled nitrification-denitrification is likely an important source of nitrous oxide flux in fields fertilized with ammonium (Liu et al. 2007). Nitrification and denitrification remove inorganic nitrogen from soils, which can reduce pollution of waterways, but these processes produce nitrous oxide, a greenhouse gas with a strong warming potential.

In Minnesota, artificial subsurface "tile" drainages are commonly used to capture excess water from agricultural fields in perforated pipes. These pipes direct water into ditches, streams, and other outlets. Artificial subsurface drainage lowers the water table to prevent flooding and aerate soils, ultimately increasing productivity. However, subsurface flow that is directed through drainage pipes has few opportunities for nitrate removal, and these nitrates pollute natural streams and rivers (Jaynes and Isenhart 2014). High nitrate concentration in natural water sources can contaminate wells and drinking water, as nitrate is a health risk when above 10 mg/L (Keeler et al. 2016). Nitrates cause eutrophication of local streams and lakes, but can also be transported by large-order river to the ocean, where eutrophication and hypoxia harm marine habitats.

One new method for nitrate removal is rerouting part of the subsurface tile drainage into a saturated riparian buffer strip. In the buffer strip, nitrate is used taken up by plants, immobilized by microbes, and processed by bacterial denitrification (Ranalli and Macalady 2010). Saturated buffers are effective at removing agricultural nitrates, but these shallow saturated zones often have high greenhouse gas flux rates (Anderson et al. 2014). Further research is necessary to better understand the nutrient cycling and greenhouse gas production in agricultural drainage areas.

Till methods and crop types can change the need for artificial fertilizer, the amount of nitrogen lost from the field due to run-off, and the potential for greenhouse gas flux. Less intensive till methods, such as no-till and strip-till, conserve the amount of organic matter and nutrients in the soil, reducing the need for chemical fertilizers. Since reducing the use of nitrogen fertilizers reduces nitrate run-off, there is evidence that no-till and strip-till soils lose less nitrogen to nitrous oxide (Jacinthe and Dick 1997). Conversely, a different study found evidence that there were higher nitrous oxide fluxes from no-till soil, due to higher percent water filled pore space in no-till soils (Liu et al. 2007). Soybean fields require less fertilizer than corn fields since they fix nitrogen, so soybean crops tend to lose less nitrogen in the form of nitrous oxide (Jacinthe and Dick 1997). It is unclear whether the nitrogen is conserved due to the soybean plant's ability to fix nitrogen, or indirectly because less nitrogen fertilization is required.

Further studies of drainage systems, nitrogen fertilization, soil microbial communities, and agricultural farming methods are needed to better understand water pollution and greenhouse gas flux from agricultural catchments. To gain a better understanding of how agriculture affects the nitrogen loading of natural waterways, I have designed a study in Northfield, Minnesota. The objectives of this study were:

1. To investigate how crop type and tilling practices impact water composition of agricultural drainage.
2. To compare nitrate and DOC concentrations from drainage from fields with artificial subsurface drainage and no artificial drainage.
2. To determine the how effective a saturated riparian buffer strip is at processing nitrogen from agricultural run-off.
3. To examine any seasonal changes in nitrate and nitrous oxide concentrations from artificial subsurface drainage in a conventionally farmed agricultural field.

Methods

To gain an understanding of the water quality of agricultural drainage in Northfield, Minnesota, I sampled five different sites agricultural drainage discharge from corn and soybean fields with different till methods and various methods of managing drainage. The first location (DL) was a farm managed by Dave Legvold, located at 5103 315th Street W in Northfield, MN. Dave used strip-till on this cornfield and there was artificial subsurface tiling drainage directed into a saturated riparian buffer. Before the harvest at the DL location, water samples were taken from three subsurface locations: the perforated tile main before the saturated buffer, the perforated pipe halfway down the saturated buffer, and at the outlet of the saturated buffer. During harvest, the water level was too low to saturate the buffer and only the outlet was sampled. My second sampling location (RT) was a no-till soybean field with subsurface tile drainage owned by St. Olaf College and located near Cedar Avenue and North Avenue in Northfield, MN. I sampled a second St. Olaf-owned field (EF) located next to Eaves Avenue, which is strip-tilled, planted with corn, and artificially drained with subsurface perforated pipes. Additionally, I sampled surface drainage running from a cornfield that has no artificial subsurface drainage, located on 320^{th} Street W and Eveleth Avenue (KS). My final sample location was a conventionally farmed, artificially drained corn-soybean rotated field located on 100^{th} Street W and Decker Avenue (RP).

At the RP sample location, weekly surface water samples and dissolved gas samples were taken directly after discharge from the subsurface drainage mainline from October 8, 2015 to May 6, 2016. Surface water samples at DL, EF, RT, KS, and RP locations occurred throughout October and November 2016. Duplicate surface water samples were filtered in the field and analyzed for nitrates with a SmartChem 200 discrete analyzer, and for DOC and TN with a Shimadzu TOC-V analyzer. Dissolved gas samples were taken in triplicate by taken by shaking 30 mL atmospheric gas with 30mL of sample water for one minute and inserting 20 mL of gas into a septum cap gas vial. Headspace gas samples were analyzed for CH_4 and N_2O concentrations with a ThermoScientific gas chromatograph.

To compare differences in water drainage from the five agricultural fields, ANOVA tests were used to determine any significant differences between mean nitrate, DOC, and TN concentrations. For analysis comparing the five fields, I only included DL samples that were taken after the saturated buffer as the drainage entered the stream. ANOVA tests were used to determine any significant changes in mean nitrate, DOC, and TN concentrations as drainage flowed through subsurface tile drainage, through a saturated buffer strip, and into a stream at the DL field. A linear regression was used to examine the relationship between nitrate concentrations and dissolved nitrous oxide concentrations in the RP drainage. ANOVA tests were used to compare seasonal (autumn, winter, spring) differences in nitrate and dissolved nitrous oxide concentrations, as well as pre- and post-harvest differences in nitrate and dissolved nitrous oxide concentrations. For each ANOVA test, a Bartlett test of equal variances and a

Shapiro-Wilks test for normality were used to confirm that the conditions for ANOVA were satisfied. Tukey's multiple comparison of means was completed as a post-hoc analysis of ANOVA results to determine which groups were significantly different. For datasets that were not normally distributed, I used a Mann-Whitney U test or Kruskal-Wallis test to compare the means of groups. All statistical analyses were completed in R using RStudio version 0.99.484.

Results

When comparing the data I collected in autumn 2016, I found no significant differences in nitrate concentrations ($F=1.1229$, $p= 0.52$), DOC concentrations ($F = 0.4729$, $p = 0.7642$), or total nitrogen concentrations ($F = 2.2221$, $p = 0.3336$) among the five fields. Mean nitrate concentrations during the season ranged from 1.37 mg/L at KS to 6.99 mg/L at RP, displayed in Figure 1. Total nitrate concentrations followed similar trends, also shown in Figure 1. Mean DOC concentrations ranged from 1.79 mg/L at RT to 6.05 mg/L at DL, shown in Figure 2.

I did find significant differences in nitrate concentrations when comparing means from samples that were taken on the same day, November 11, 2016. All the fields were harvested by this date, and environmental factors such as precipitation were similar across sites. Between the five fields on November 11, there was a significant difference in mean nitrate concentrations ($F = 2582.8$, $p < 0.001$), and displayed in Figure 3. There was a significant difference in DOC in different drainage sources on November 11, 2016 ($F = 26.965$, $p = 0.001$), with DOC concentrations were significantly higher in EF and KS than in RP and RT. Mean DOC concentrations on this date are displayed in Figure 4.

From the samples collected before, during, and after flow through the saturated buffer at the DL field, I saw significant differences in mean nitrate ($F = 1790.9$, $p < 0.001$), DOC ($F = 165.66$, $p < 0.001$), and TN ($F = 122.88$, $p <0.001$) concentrations. All these differences were significantly different between perforated pipe to mid-buffer samples, and between mid-buffer to stream samples. I found no significant differences in water chemistry in before-harvest drainage on October 1st when the buffer was saturated, and post-harvest drainage on November 11th when the water level was lowered in the buffer. Mean nitrate, DOC, and TN concentrations at the saturated buffer are shown in Figures 5 and 6.

Mean nitrate concentrations from RP from October 8, 2015 to May 5, 2016 are displayed in Figure 7. Mean dissolved nitrous oxide concentrations from Oct 8, 2015 to February 12, 2016 are displayed in Figure 8. There was no statistically significant seasonal variation in nitrate or dissolved nitrous oxide concentrations at this site. There was also no significant difference in mean nitrate concentration in the drainage before and after harvest, which occurred between October 29th and November 5th. However, there was a significant difference in nitrous oxide concentrations before and after harvest ($W = 21$, $p = 0.017$). Additionally, there was a positive linear relationship between nitrate concentration and dissolved nitrous oxide concentration in this subsurface drainage outlet, displayed in Figure 9 (adjusted $R^2 = 0.6612$, $p = 0.002$).

Discussion

While there was variation in mean nitrogen concentrations in the five sample fields, I did not find any significant differences among fields, possibly due to the small sample size. While not statistically significant, the fields with subsurface tiling drainage (DL, EF, RT, and RP) had higher nitrate concentrations than KS, which did not have any artificial drainage systems. These higher nitrate concentrations are likely due to the perforated pipes causing faster drainage, decreasing the ability for microbes to immobilize or process the nitrogen leached from the soil (Ranalli and Macalady 2010). Also statistically insignificant but important to note was the higher DOC from saturated buffer strip (DL) and natural surface drainage (KS), indicating greater transport of carbon from system with these drainage methods. In all the drainage sites sampled in October and November 2016, the nitrate concentrations were below the EPA health standard of 10 mg/L. However, the nitrate concentrations at RP were consistently above 10 mg/L from October 2015 through April 2015. The lowering of nitrate concentrations at RP could be due to a reduction in fertilization, different patterns of precipitation, or the rotation of crops from corn in 2015 to soybean in 2016.

Analysis of the saturated riparian buffer strip indicated that the buffer strip removed nitrogen from the drainage water, and added dissolved organic carbon. These changes in water chemistry were noticeable as water flowed from the perforated pipes, slowly moved through the buffer, and drained from the buffer into a natural stream. The reduction of nitrate concentrations at the DL site is consistent with the research on saturated buffer strips completed by Jaynes and Isenhart (2014). However, the buffer strip only functions when the water level is raised within the buffer, slowing the water's movement. During harvest, water level was lowered to dry the fields, allowing water to move quickly past the buffer and directly into the stream. While the difference in means is statistically insignificant, there was an increase in nitrate concentrations and a decrease in DOC concentrations after the water level in the buffer was lowered.

While the study of nitrous oxide flux was limited to one sampling location, analysis of the nitrate and dissolved nitrous oxide concentrations at RP showed a moderately strong, positive, linear correlation between nitrate and nitrous oxide flux. Since nitrification produces nitrate as a product and nitrous oxide as a byproduct, and denitrification uses nitrate to produce nitrogen gas a product with nitrous oxide as a byproduct, this positive relationship is not surprising (Groffman et al. 1998, Davidson and Verchot 2000, Anderson et al. 2014). Interestingly, the highest nitrous oxide flux occurred in the winter, after harvest occurred. While colder temperatures decrease microbial respiration, this increased winter flux could be caused by a few factors. Crops are harvested in the autumn, reducing the nitrogen uptake by plants and potentially increasing nitrogen availability to nitrifying and denitrifying bacteria. Snow and ice cover may create an anoxic environment, such that the primary form of respiration switches from nitrification to denitrification. Snow may also trap nitrous oxide in the soil, increasing the dissolved gas concentrations in groundwater. Lastly, we must consider how cold temperatures decrease the solubility of gas, such that dissolved gas concentrations may increase even if the net nitrous oxide flux to the atmosphere remains the same.

Since the size of the data set was limited and there were many important factors unaccounted for, it was difficult to determine any difference in agricultural discharge from fields using different drainage methods or with different crop types. For an accurate comparison of these variables, I would need to design an experimental field to control the amount of fertilizer and timing of fertilizer application, the drainage type and groundwater flow, the amount and timing of precipitation, and the date of harvest. In an ideal study, water filled pore space and soil organic matter would be analyzed for different tilling methods.

For a more complete analysis of nitrous oxide flux, frequent dissolved gas samples would need to be taken from many sources of agricultural drainage. It would be useful to compare ^{14}N isotope fractioning in these samples to determine whether the processes of nitrification or denitrification are producing the most nitrous oxide in soils. It is also important to note that we are unable to estimate annual flux of nitrous oxide from an agricultural field with just dissolved water samples. Gas flux from soil directly to the atmosphere would need to be measured, and the flux from surface waters would need to be estimated using the dissolved gas samples. Another relevant experimental study would be to compare the amount of nitrate run-off with varying levels of fertilizer application and different timing of fertilizer application. Applying fertilizer right before a major precipitation event can increase the loss of inorganic nitrogen, and fertilizing during different times of year could influence plant productivity.

From this study, it is clear that conventional agriculture in the Northfield, MN increases nitrate loading to natural water sources. The concentration of nitrates in drainage waters varies field to field, but variables causing these differences could not be determined in this study. Nitrate concentrations were highest in fields with artificial subsurface drainage. Preliminary study of a saturated riparian buffer strip demonstrated that it is a useful way of removing nitrates from agricultural subsurface drainage. Finally, this study found a positive relationship between nitrate and dissolved nitrous oxide concentrations in an artificial subsurface drainage outlet. From this study, it is apparent that agricultural fields in Northfield, MN are a source of nitrogen to natural waterways and the atmosphere. It is important that further studies are completed to better understand the mechanisms of microbial nitrogen processing and to determine the fate of inorganic nitrogen that is applied to agricultural fields. Agriculture in the Midwestern United States has a known impact on water quality and is a known source of nitrous oxide emissions to the atmosphere. Further studies of agricultural drainage waters aid our understanding of the global nitrogen cycle and will provide insight on how we can change agricultural practices to reduce water pollution and greenhouse gas emissions.

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Tables and Figures

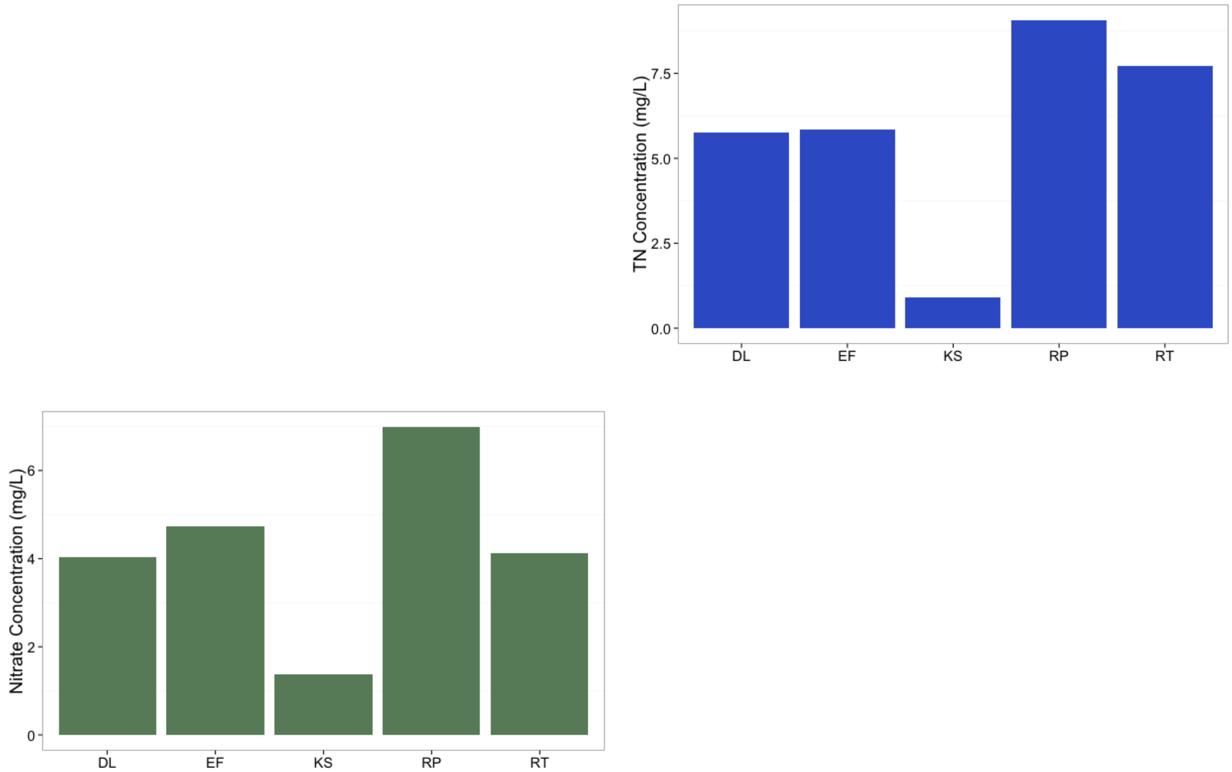


Figure 1. Mean nitrate and total nitrogen concentrations in drainage sampled from 5 fields throughout October and November 2016.

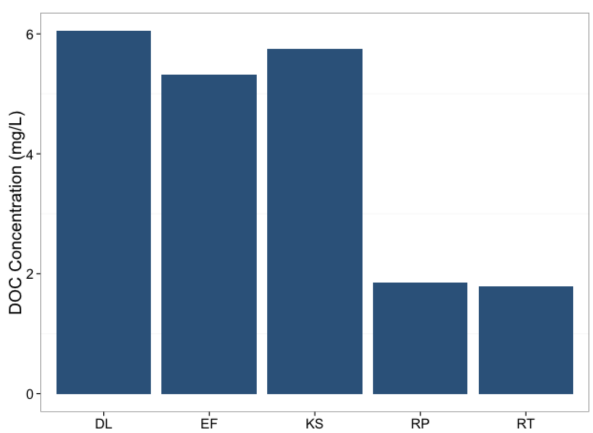
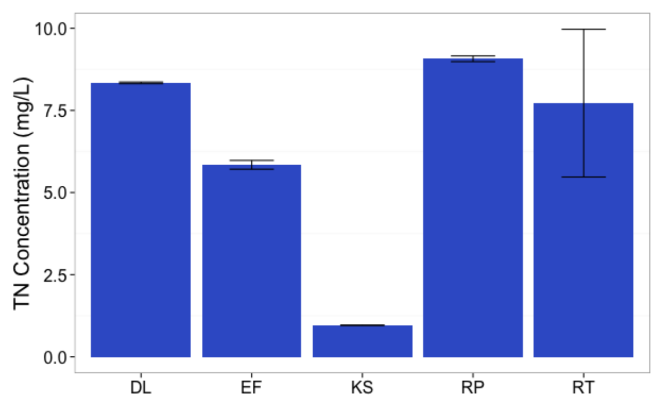


Figure 2. Mean DOC concentrations in drainage sampled from 5 fields throughout October and November 2016.



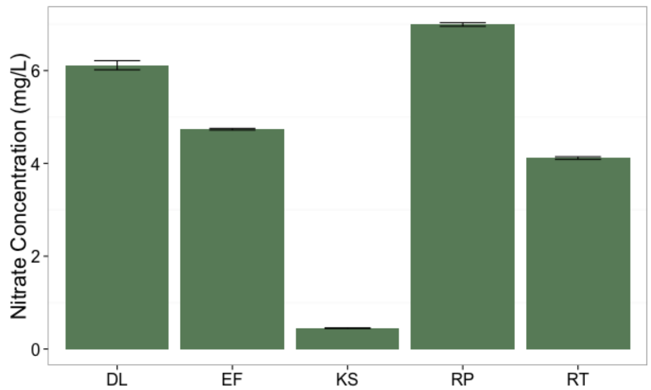
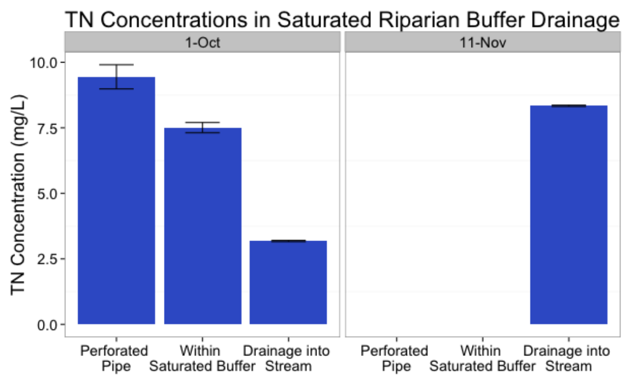


Figure 3. Mean nitrate and total nitrogen concentrations in drainage sampled from five fields on November 11, 2016 (n=2). Error bars display standard error.



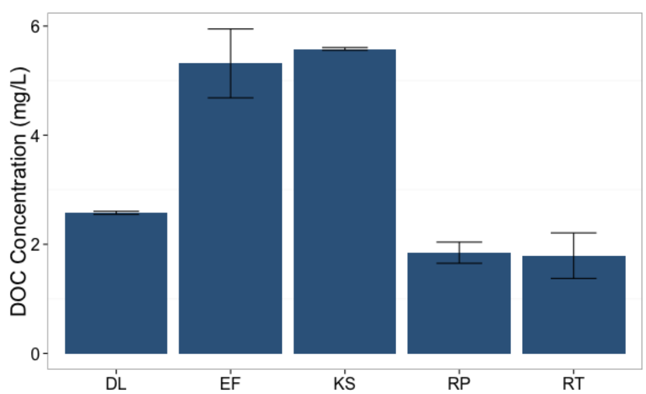


Figure 4. Mean DOC concentrations in drainage sampled from 5 fields on November 11, 2016 (n=2). Error bars display standard error.

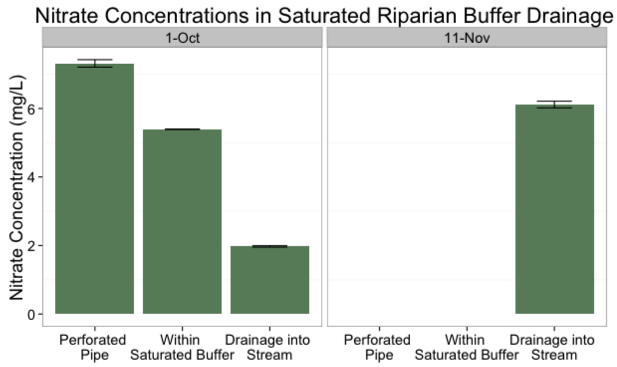


Figure 5. Mean nitrate concetrations from water samples taken at DL field before harvest on October 1, 2016 and after harvest on November 11, 2016 (n=2). Error bars display standard error.

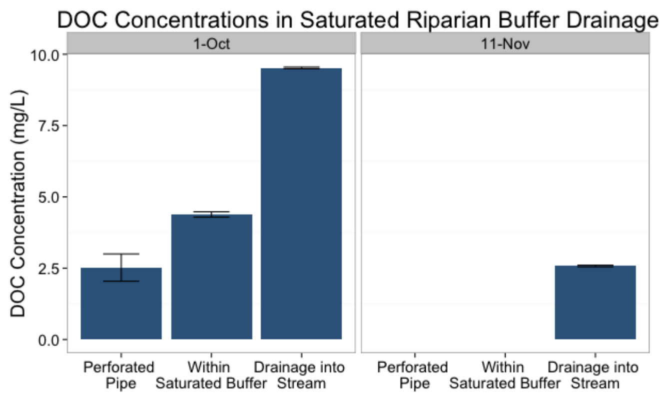


Figure 6. Mean DOC and total nitrogen concentrations from water samples taken at DL field before harvest on October 1, 2016 and after harvest on November 11, 2016 (n=2). Error bars display standard error.

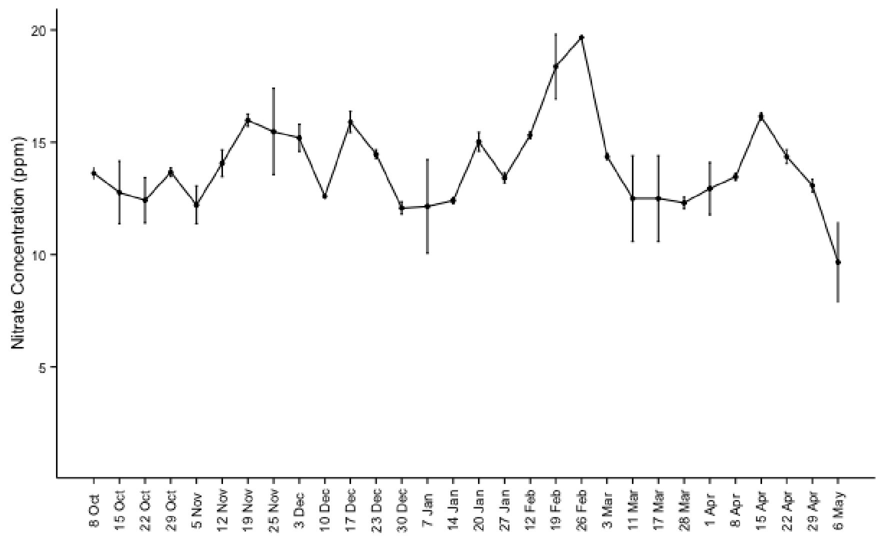


Figure 7. Mean nitrate concentrations from RP artificial subsurface drainage from 8 October 2015 to 6 May 2016 (n=3). Error bars display standard error.

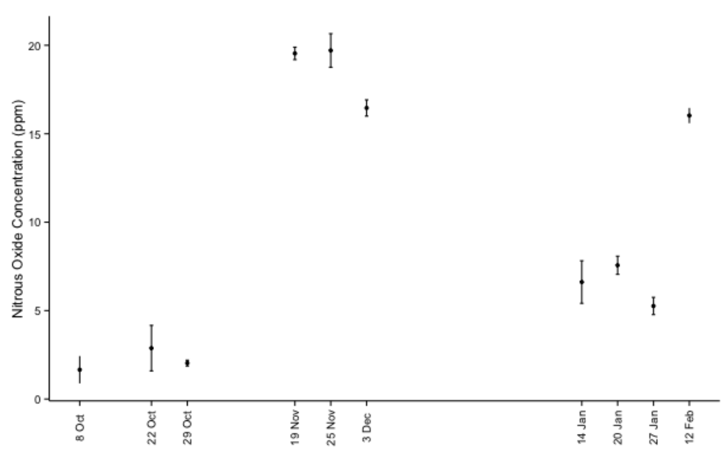


Figure 8. Mean dissolved nitrous oxide concentrations from RP artificial subsurface drainage from October 8, 2015 to February 12, 2016 (n=3). Error bars display standard error.

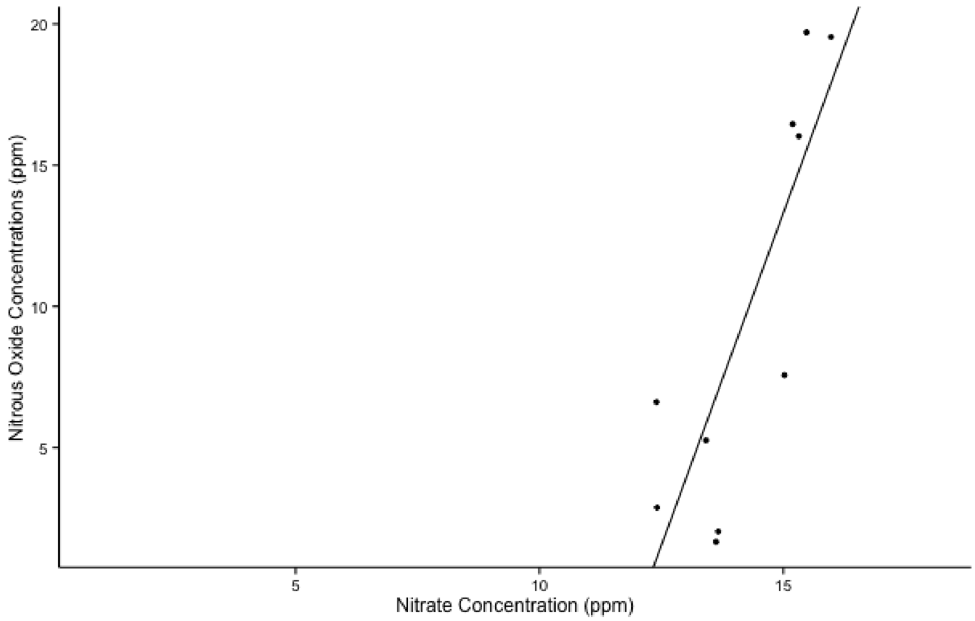


Figure 9. Linear relationship between nitrate concentration and dissolved nitrous oxide concentration in RP subsurface drainage from October 6, 2015 to February 12, 2016. Linear model is $y = 4.706x - 57.279$ (adjusted $R^2 = 0.6612$, $p = 0.002$).

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