2017

Greenhouse gas emissions from Iowa Conservation Reserve Enhancement Program wetlands

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by

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A thesis submitted to the graduate faculty

in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Major: Environmental Science

Program of Study Committee:
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Iowa State University

Ames, Iowa

2017

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ACKNOWLEDGEMENTS

I would like to thank my committee chair, Dr. William Crumpton, and my committee members, Dr. Steven Hall and Dr. Mathew Helmers, for their guidance and support throughout the course of this research.

In addition, I would like to thank my friends and colleagues, Especially Jana Stenback for her guidance, and overall help throughout my undergraduate and graduate programs, David Green for his help with all things GIS, Greg Stenback for all his work with the GHG database, data analysis, and overall guidance, and lastly Tyler Groh, for providing comparison data. I also want to show my appreciation for my undergraduate students (Kristine Doetsch, Jerry Gentry, Colleen Windham, Daniel Frey, and Rhiannon Gors), without them my field work would not have been the same.

ABSTRACT

Nitrate loads from agricultural sources raise major water quality concerns for the US Corn Belt. Wetland restoration has been identified as a promising strategy to reduce nonpoint source nitrogen loads from agricultural watersheds. However, there is concern over increased nitrous oxide (N_2O) emissions from wetlands subject to elevated nitrogen loads. The major purpose of this research was to quantify N_2O and CH_4 emissions from wetlands targeted to intercept and reduce nitrate loads in agricultural watersheds. We measured nitrate removal and N_2O and CH_4 emission rates at three wetlands subject to different nitrate loads. Nitrate loads and losses were estimated based on close interval monitoring of inflows and outflows. N_2O and CH_4 emissions were estimated using floating chambers during synoptic studies conducted from late spring through early fall in 2015-2016. N_2O emission rates averaged 3.5 mg N_2O-N m^{-2} day^{-1}, similar to rates from cropland even though wetlands received more N per area than croplands. N_2O emission rates were correlated to nitrate concentrations, loading rates, and loss rates. CH_4 emission rates averaged 793 mg CH_4 m^{-2} day^{-1}, similar to rates for restored depressional wetlands in Iowa.

CHAPTER 1. GENERAL INTRODUCTION

Agricultural impacts on water quality present major environmental concerns in the US Corn Belt (Crumpton et al., 2001; Crumpton et al., 2008; Dale et al., 2010). Nitrate (NO_3) is of particular concern because of its effects on drinking water and its contribution to hypoxia in the Gulf of Mexico (Dale et al., 2010). The Hypoxia Action Plan calls for a 45% reduction in nitrogen loads in the Mississippi River Basin (Dale et al., 2010; The Hypoxia Action Plan, 2008). The states of Iowa, Illinois, and Minnesota have developed nutrient reduction strategies intended to achieve 45% reductions in nitrogen loads for each state (Iowa Nutrient Reduction Strategy, 2012; Illinois Nutrient Reduction Strategy, 2015; Minnesota Nutrient Reduction Strategy, 2013). Iowa has set an additional goal of reducing non-point source nitrogen loads by 41% (Iowa Nutrient Reduction Strategy, 2012). A combination of in-field and edge-of-field practices will be needed to achieve these goals (Iowa Nutrient Reduction Strategy, 2012). In-field practices such as fertilizer management and cover crops are needed to reduce the amount of nitrogen exported from fields. Edge-of-field practices such as saturated buffers, bioreactors, and wetlands are needed to intercept exported nitrogen and reduce the loads that reach downstream waters (Iowa Nutrient Reduction Strategy, 2012).

The US Corn Belt was once rich in wetlands, and in many areas, farming was made possible only by draining wetlands (Dahl et al., 1990; Pavelis et al., 1987). Wetlands losses exceed 85% in the Corn Belt states of Iowa, Illinois, Indiana and Ohio (Dahl et al., 1990). Mitsch et al. (2005) suggested that the hypoxic zone in the Gulf of Mexico is directly related to this large loss of wetlands. According to van der Valk and Jolly et al. (1992), wetland restoration is one of the most promising strategies for reducing surface water contamination and reducing nitrogen entering the Gulf of Mexico.

Since the mid-1980s, a variety of state and federal programs have been used to promote wetland restoration in the US Corn Belt region, however some these wetlands were not specifically restored for water quality purposes. The Iowa Conservation Reserve Enhancement Program (CREP) wetlands are strategically placed in positions on the landscape to intercept high nitrate loads in tile drainage waters and promote denitrification (Iowa Conservation Reserve Enhancement Program, 2016). As of 2016, 83 total wetlands, totaling pool areas of 759 acres have been restored through the Iowa Conservation Reserve Enhancement Program with a primary goal of water quality improvement (Iowa Conservation Reserve Enhancement Program, 2016).

Studies throughout the US Corn Belt have demonstrated that wetlands can be effective at reducing non-point source nitrate loads (Crumpton et al., 2008; Freeman et al., 1997; Mitsch et al., 1999; Mitsch et al., 2005). In addition to removing nitrate, wetlands can be important carbon sinks and are typically much more effective than agricultural fields at sequestering carbon (Hey et al., 2012). However, wetlands can also be significant sources of nitrous oxide (N_2O) and methane (CH_4). Nitrous oxide and CH_4 are both potent greenhouse gases with 310 and 21 times the global warming potential of CO_2, respectively (over a 100-yr time horizon) (Solomon et al., 2013). There is some concern that widespread restoration of wetlands to intercept and reduce non-point source nitrate loads could substantially increase emissions of greenhouse gases and in particular N_2O emissions (Verhoven et al. 2006).

Understanding wetlands and their role on the GHG cycle is one of many reasons for conducting this study. Relatively few studies have quantified N_2O and CH_4 emissions from wetlands receiving nonpoint source nitrate loads and these represent a relatively small range of loads (Altor and Mitsch 2006; Groh et al. 2015; Hernandez and Mitsch 2006; Verhoven et al., 2006). This research is to quantify N_2O and CH_4 emissions and get a better understanding for the importance for wetlands in agricultural watersheds with elevated nitrate loads.

The primary objectives for this study were to quantify N_2O and CH_4 emissions from wetlands subject to a wide range of nonpoint source nitrate loads and evaluate the effect of nitrate loading rate on N_2O and CH_4 emission rates.

Thesis Organization

With growing concern over harmful emission from wetlands and agricultural fields, this thesis explores N_2O and CH_4 emissions from Iowa CREP wetlands to determine daily emissions. Chapter II explores the N_2O and CH_4 emissions from three CREP wetlands and the relationship to nitrate concentrations, loading, and temperature. Chapter III draws general conclusions about N_2O and CH_4 emissions and evaluates effectiveness of targeted wetland restoration on water quality improvements and greenhouse gas emissions.

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CHAPTER 2. NITROUS OXIDE AND METHANE EMISSIONS FROM WETLANDS RECEIVING NON-POINT SOURCE NITRATE LOADS

A paper to be submitted to the *Journal of Environmental Quality*

Hannah L. Hoglund, William G. Crumpton, and Greg A. Stenback

Abstract

Nitrate loads from agricultural sources raise major water quality concerns for the US Corn Belt. Wetland restoration has been identified as a promising strategy to reduce nonpoint source nitrogen loads from agricultural watersheds. However, there is concern over increased nitrous oxide (N_2O) emissions from wetlands subject to elevated nitrogen loads. The major purpose of this research was to quantify N_2O and CH_4 emissions from wetlands targeted to intercept and reduce nitrate loads in agricultural watersheds. We measured nitrate removal and N_2O and CH_4 emission rates at three wetlands subject to different nitrate loads. Nitrate loads and losses were estimated based on close interval monitoring of inflows and outflows. N_2O and CH_4 emissions were estimated using floating chambers during synoptic studies conducted from late spring through early fall in 2015-2016. N_2O emission rates averaged 3.5 mg N_2O-N m^{-2} day^{-1}, similar to rates from cropland even though wetlands received more N per area than croplands. N_2O emission rates were correlated to nitrate concentrations, loading rates, and loss rates. CH_4 emission rates averaged 793 mg CH_4 m^{-2} day^{-1}, similar to rates for restored depressional wetlands in Iowa.

Introduction

Agricultural impacts on water quality present major environmental concerns in the US Corn Belt (Crumpton et al., 2001; Crumpton et al., 2008; Dale et al., 2010). Nitrate is of particular concern because of its effects on drinking water and its contribution to hypoxia in the Gulf of Mexico (Dale et al., 2010). The Hypoxia Action Plan calls for a 45% reduction in nitrogen loads in the Mississippi River Basin (Dale et al., 2010; The Hypoxia Action Plan, 2008). The states of Iowa, Illinois, and Minnesota have developed nutrient reduction strategies intended to achieve 45% reductions in nitrogen loads for each state (Iowa Nutrient Reduction Strategy, 2012; Illinois Nutrient Reduction Strategy, 2015; Minnesota Nutrient Reduction Strategy, 2013). Iowa has set a goal of reducing non-point source nitrogen loads by 41% (Iowa Nutrient Reduction Strategy, 2012). A combination of in-field and edge-of-field practices will be needed to achieve these goals (Iowa Nutrient Reduction Strategy, 2012). In-field practices such as fertilizer management and cover crops are needed to reduce the amount of nitrogen exported from fields. Edge-of-field practices such as saturated buffers, bioreactors, and wetlands are needed to intercept exported nitrogen and reduce the loads that reach downstream waters (Iowa Nutrient Reduction Strategy, 2012).

Wetland restoration is one of the most promising strategies for reducing surface water contamination and reducing nitrogen entering the Gulf of Mexico (Hey et al. 2012; Mitsch et al. 2005). Studies throughout the US Corn Belt have demonstrated that wetlands can be effective at reducing non-point source nitrate loads (Crumpton et al., 2008; Freeman et al., 1997; Mitsch et al., 1999; Mitsch et al., 2005). In addition to removing nitrate, wetlands can be important carbon sinks and are typically much more effective than agricultural fields at sequestering carbon (Hey et al., 2012). . However, wetlands can also be significant sources of nitrous oxide (N_2O) and methane (CH_4). Nitrous oxide and CH_4 are both potent greenhouse gases with 310 and 21 times the global warming potential of CO_2, respectively (over a 100-yr time horizon) (Solomon et al., 2013). There is some concern that widespread restoration of wetlands to intercept and reduce non-point source nitrate loads could substantially increase emissions of greenhouse gases and in particular N_2O emissions (Verhoven et al., 2006). Relatively few studies have quantified N_2O and CH_4 emissions from wetlands receiving nonpoint source nitrate loads and these represent a relatively small range of nitrate loads (Altor and Mitsch 2006; Groh et al. 2015; Hernandez and Mitsch 2006; Verhoven et al., 2006).

The primary objectives for this study were to quantify N_2O and CH_4 emissions from wetlands subject to a wide range of nonpoint source nitrate loads and evaluate the effect of nitrate loading rate on N_2O and CH_4 emission rates.

Materials and Methods

Study sites

Study sites were selected in part to ensure a range in nitrate loading and loss rates. The wetlands were located to intercept tile drainage and surface runoff from three agricultural catchments ranging from 307 to 576 ha in size and with 81-92% of the catchment areas in row crops (Table 1). The wetland pools range in size from 1.45 ha to 3.11 ha and occupy 0.47%-0.57% of their catchment areas (Table 1). Prior to restoration, all three sites had been cropped or pastured and were restored by creating low earthen dikes with integrated outflow structures. This resulted in shallow wetland pools on predominately hydric soils and with average full pool depths ranging from 0.60 to 0.73 m. At the time of restoration, vegetated buffers ranging from 6.2 to 11.3 ha were established around each wetland by seeding with a mix of native grasses and forbs (Table 1). Wetland pools were allowed to vegetate naturally and at the time of this study had relatively sparse emergent zones (Halpin, 2016) and extensive submersed aquatics dominated by *Potamogeton* and *Ceratophyllum* (Eeling, 2017).

N_2O and CH_4 emissions

Nitrous oxide and CH_4 gas emissions were sampled at three wetlands from May through August for a total of 36 synoptic events in 2015 and 2016 including 15 events at wetland KS, 10 at wetland RS, and 11 at wetland LICA. Using ArcMap (10.3), evenly spaced transects were located perpendicular to the centerline of each wetland and chamber measurement locations were evenly spaced along these transects (on average 9 chambers for each synoptic event). Locations were transferred to a Trimble GPS unit, which was used to mark measurement locations in the field.

Floating chambers were made from a 4.5L opaque plastic bucket, equipped with a sampling tube (6-8 inches long), pressure equilibrium tube (2 feet long), handle, and tube of floating foam. While sampling, the chambers were attached to pre-marked flags at each wetland, and secured with a string connected to a brick at the sediment. The GRACEnet protocol was followed to sample N_2O and CH_4 gas emissions (Parkin and Venterea, 2010). Gas samples were taken at intervals ranging from 20-45 minutes for a total of 1-1.5 hours. Gas samples were collected with syringes, transferred to evacuated 21 ml glass vials, and stored room temperature (22 °C) until analyzed. Gas samples were analyzed for N_2O and CH_4 concentrations using a SRI 8610C gas chromatographer, equipped with a flame ionization detector and electron capture detector, and using ultra-high purity hydrogen and nitrogen gases. Emission rates were calculated based on the rate of change in concentrations within the chamber headspace. The slope was determined by the linear portion of the response curve.

At each chamber, a water grab sample was taken for nitrate analysis and temperature was taken with a Hach HQ30d dissolved oxygen meter. Water samples were analyzed for nitrate by second-derivative spectroscopy (Crumpton et al., 1992).

Nitrogen mass balance

Wetland pool elevations, inflows and outflows were monitored at five-minute intervals using a combination of stage recorders (Solinst Leveloggers, Model 3001) and submerged area velocity (SAV) meters (Mace FloPro XCi). Point discharge measurements and SAV based discharge measurements were used to develop stage-discharge equations for stream channels and to calibrate discharge equations for wetland outflow structures. Precipitation (P) and evapotranspiration (ET) were estimated based on Iowa Environmental Mesonet for nearby stations.

Daily composite water samples were collected at wetland inflows and outflows using programmable, automated samplers (QCEC QLS Portable Wastewater Sampler). Sample bottles were pre-acidified using sulfuric acid to preserve samples at pH<2.

Water samples were analyzed for nitrate by second-derivative spectroscopy (Crumpton et al., 1992) and for total nitrogen using the persulfate digestion method (Standard Methods 1998). Spectroscopic analyses were performed using an Agilent UV/VIS spectrophotometer running Hewlett Packard ChemStation software.

The mass of nitrate transported with inflow (N_{in}), outflow (N_{out}), seepage (SN_{out}), and precipitation (PN_{in}) was calculated as the sum of the daily product of nitrate concentrations and volumes associated with each input or output. Concentrations in seepage were assumed to equal those in wetland outflows. The concentration in precipitation was taken as 0.32 mg/L, the average reported in the National Atmospheric Deposition Program (NADP, 2014) database from stations surrounding the study sites (NADP Stations AI23, AI08 and MN27). To calculate the mass of nitrate removed in the wetland (N_r), ΔN, N_{in}, PN_{in}, N_{out}, and SN_{out} were input as values and N_r was calculated based on solving the system mass balance (Equation 1).

$$\Delta N = N_{in} + PN_{in} - N_{out} - SN_{out} - N_r \qquad [1]$$

Results and Discussion

Nitrate loading and loss rates

Seasonal flow weighted average (FWA) nitrate concentrations varied about five fold across sites and years, from 4.5-26.3 mg N/L (Table 2). Average seasonal nitrate loading rates varied by an order of magnitude across sites and years, ranging from 385-6927 mg N m^{-2} day^{-1} (Table 2). On average, nitrate comprised 87.6% of the total nitrogen (TN) load, ranging from 75-97%. All three wetlands were sinks for nitrate and TN during the study seasons. On average, the wetlands removed 731 mg nitrate-N m^{-2} day^{-1} with an average percent reduction of 41%.

Nitrous oxide emissions

Average N_2O emission rates for individual synoptic events ranged from 0.6-20.1 mg N_2O-N m^{-2} day^{-1}. Seasonal N_2O emission rates averaged across events by site and year ranged from 1.4-7.1 mg N_2O-N m^{-2} day^{-1} (Table 3, Figure 1) and averaged 3.5 mg N_2O-N m^{-2} day^{-1} across sites and years. Seasonal average N_2O emission rates were correlated with FWA nitrate concentrations (R^2 0.98, Figure 2), nitrate loading rates (R^2 0.71, Figure 3) and nitrate loss rates (R^2 0.41, Figure 4). These results are consistent with the general expectation of increasing N_2O emissions as nitrogen inputs increase (Verhoven et al., 2006). After accounting for nitrate concentrations, event and seasonal average N_2O emission rates had no significant correlation with water temperature.

Our average emission rate is higher than that reported by Groh et al. (2015) and by Hernandez and Mitsch (2006) for Corn Belt wetlands receiving non-point source nitrate loads, but our systems had much higher average nitrate loading rates. Hernandez and Mitsch (2006) measured N_2O emissions from two permanently flooded wetlands receiving river water at an average loading rate of 175 mg N m^{-2} day^{-1}. They reported average emission rates of 0.24 mg N_2O-N m^{-2} day^{-1} for permanently inundated sites (Hernandez and Mitsch 2006). Groh et al. (2015) measured N_2O gas emissions from two small, intermittently flooded wetlands receiving tile drainage. These wetlands were flooded less than 45% of the sampling season and loading rates averaged only 595 mg N m^{-2} day^{-1} over the period the wetlands were inundated (Groh et al., 2015; Groh personal communication). Groh et al. (2015) reported average emission rates of 1.6 mg N_2O-N m^{-2} day^{-1} for inundated areas, which is similar to our result at comparable nitrate loading rates (Figure 3). Groh's emission rates for individual wetlands and years ranged from 0.4-1.9 mg N_2O-N m^{-2} day^{-1} and were very similar to our emission rates from our wetlands at similar nitrate loading rates (Groh personal communication, Figure 3).

The wetlands had very high nitrate conversion efficiencies, measured as N_2O-N emission as a fraction of nitrate-N removed (0.49%, Figure 4). Hernandez and Mitsch (2006) reported similar efficiencies with nitrous oxide emissions averaging 0.26-0.32% of nitrogen removal. Groh et al. (2015) reported fractional emissions of 1.3-3.2% of nitrate removal, but that included emissions when sites were not inundated. During periods of inundation, their fractional emission rate (average 0.48 %) is very similar to that of our wetlands (Figure 4; Groh personal communication).

In combination, results from the current study, Groh et al. (2015), and Hernandez and Mitsch (2006) support the expectation that N_2O emissions will increase with increases in nitrate loading. However, results also demonstrate very nigh nitrate conversions efficiencies, with N_2O-N emissions averaging <0.5% of nitrate removal when wetlands are inundated. The fraction of nitrogen loading that would be transformed to N_2O is much higher in cropland or downstream riverine or marine systems than in wetlands (Hey et al., 2012). Wetlands received much more nitrogen per area than cropland, but still had a comparable amount of emissions to cropland. Parkin and Kaspar (2006) and Smith et al. (2013) reported average N_2O emissions ranging from 0.66 to 2.35 mg N_2O-N m^{-2} day^{-1} for soybean and corn respectively.

Methane emissions

Average CH_4 emission rates for individual synoptic events ranged from 313-2418 mg CH_4 m^{-2} day^{-1}. Seasonal CH_4 emission rates averaged across events by site and year ranged from 530-1508 mg CH_4 m^{-2} day^{-1} (Table 3, Figure 1) and averaged 793 mg CH_4 m^{-2} day^{-1} across all sites and dates. Seasonal average CH_4 emission rates were not correlated with FWA nitrate concentrations, nitrate loading rates, or temperature. However, we did not measure emissions at low temperatures (<15 ^{o}C) which have been shown to substantially reduce CH_4 emissions in similar systems (Altor and Mitsch, 2006; Groh et al., 2015).

Our CH_4 emission rates are lower than those reported by Tangen et al. (2015) for hydrologically restored, semi-permanent wetlands in Northern Iowa (939 mg CH_4 m^{-2} day^{-1}) but higher than rates reported by Groh et al. (2015) and Altor and Mitsch (2006) for wetlands receiving non-point source nitrate loads, and especially for intermittently flooded sites. Altor and Mitsch (2006) reported average emission rates of 256 mg CH_4 m^{-2} day^{-1} for permanently flooded sites, but only 112 mg CH_4 m^{-2} day^{-}1 for intermittently flooded sites when inundated (The wetlands Altor and Mitsch (2006) studied were the same as described in Hernandez and Mitsch (2006)). Groh et al. (2015) reported even lower CH_4 emissions from intermittently flooded wetlands during periods of inundation (52 mg CH_4 m^{-2} day^{-1}). These results are consistent with the general expectation of higher CH_4 emissions from permanently flooded sites (Altor and Mitsch, 2006).

Conclusions

Nitrous oxide and CH_4 emissions from the study wetlands averaged 3.5 mg N_2O-N m^{-2} day^{-1} and 793 mg CH_4 m^{-2} day^{-1} across all wetland sites and dates. The wetlands were highly efficient at denitrifying nitrate to N_2, with N_2O-N emissions averaging only 0.49% of nitrate removal. In combination, results from the current study and others support the expectation that N_2O emissions will increase with increases in nitrate loading (R^2 0.71, Figure 3). However, results also demonstrate that N_2O emissions from wetlands constructed or restored on former agricultural land are similar to emissions from cropland that the wetlands replace. The water quality benefits, habitat benefits, and favorable greenhouse balances of wetlands support a strategy of widespread restoration and construction of wetlands to intercept and reduce surface water nitrate loads in the US Corn Belt.

Acknowledgements

This work was supported in part by funding from the USDA National Institute of Food and Agriculture and the Iowa Department of Agriculture and Land Stewardship. We thank Jana Stenback, and David Green for technical assistance and support.

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Tables

Table 1.General information about the three study sites

Wetland:	% Row Crop	Catchment Area (ha)	Buffer Area (ha)	Pool Area (ha)	Average Depth (m)
KS	89.6	307	6.2	1.45	0.60
RS	91.7	576	11.3	3.11	0.73
LICA	81.2	336	6.3	1.93	0.67

Table 2. Nitrogen budget for each wetland for 2015 and 2016.

Wetland	Year	seasonal FWA ([NO3-N] mg L^{-1})	Seasonal nitrate Load (mg N m^{-2} day^{-1})	Seasonal loss mass balance (mg N m^{-2} day^{-1})	% nitrate loss	Nitrate as percent of TN
KS	2015	26.34	6927	1128	16	93.5
KS	2016	21.65	2480	672	27	96.6
RS	2015	12.32	3317	1063	32	83.7
RS	2016	12.32	978	549	56	94.4
LICA	2015	7.66	1867	663	35	82.3
LICA	2016	4.52	385	312	81	75.3

Table 3. Average emission rates for nitrous oxide and methane by wetland and year. Standard error in parentheses.

	mg N_2O- N m^{-2} day^{-1}			mg CH_4 m^{-2} day^{-1}		
Wetland	2015	2016	2 YR AVG	2015	2016	2 YR AVG
KS	7.1 (1.1)	5.6 (0.6)	6.2 (0.6)	572 (87.6)	793 (78.6)	705 (58.8)
RS	2.8 (0.5)	2.5 (0.4)	2.7 (0.3)	586 (80.6)	1508 (311)	1047 (107)
LICA	1.8 (0.2)	1.4 (0.2)	1.6 (0.1)	530 (115)	711 (131)	629 (88.4)
AVG	3.9	3.2	3.5	563	1004	793

Figures

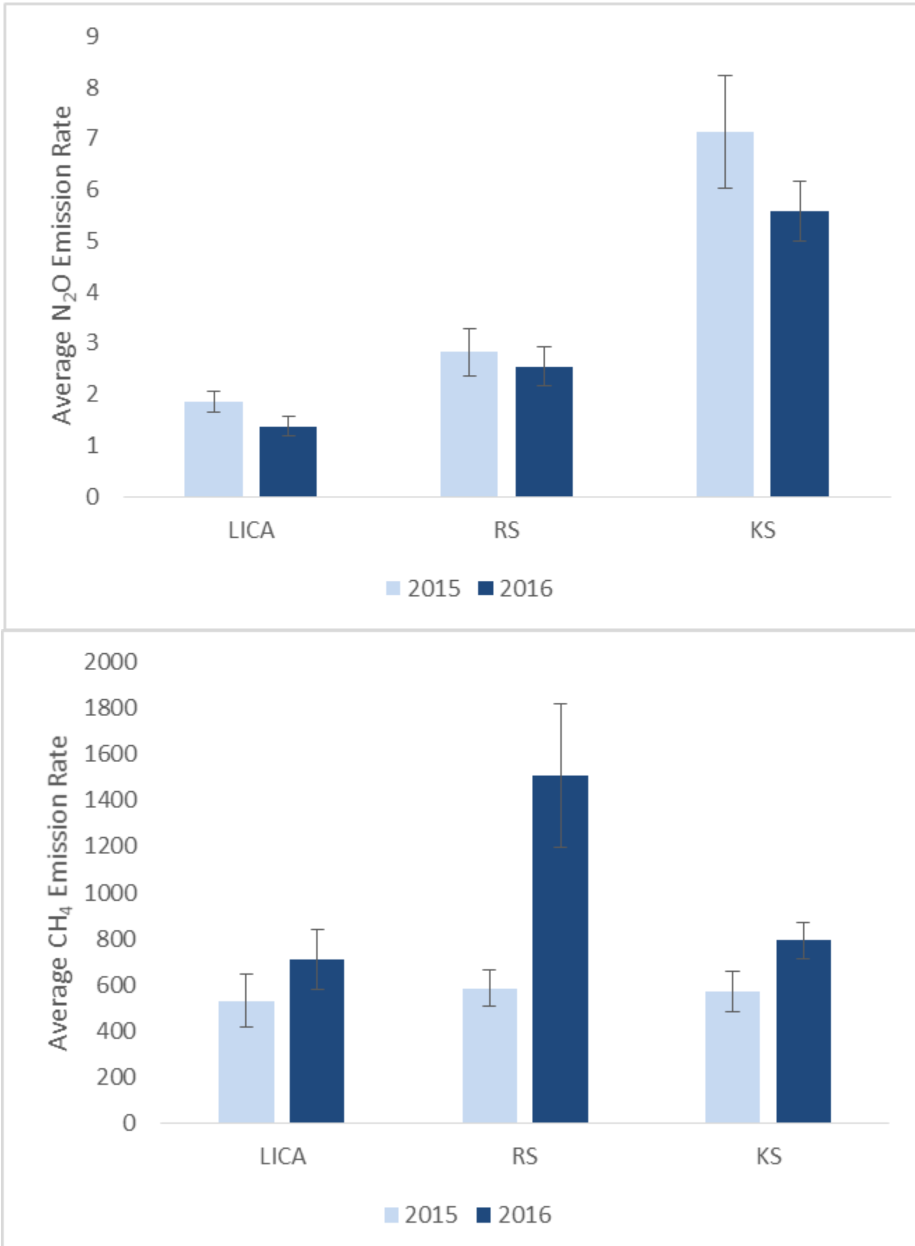

Figure 2
Nitrous oxide (top) and methane (bottom) emissions with standard error by wetland and year. Emission rates are in mg N_2O- N m^{-2} day^{-1} and mg CH_4 m^{-2} day^{-1}.

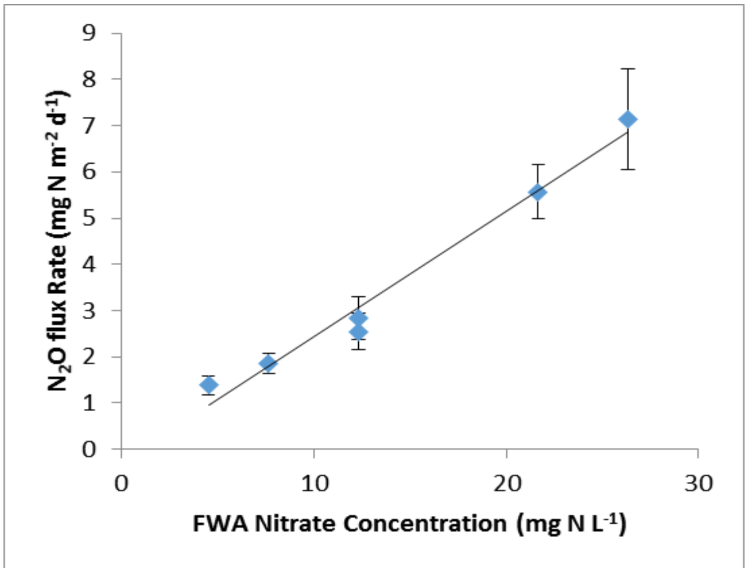

Figure 2
Nitrous oxide emissions (average and standard error) vs flow weighted average (FWA) nitrate concentration by wetland and year (R^2 0.98).

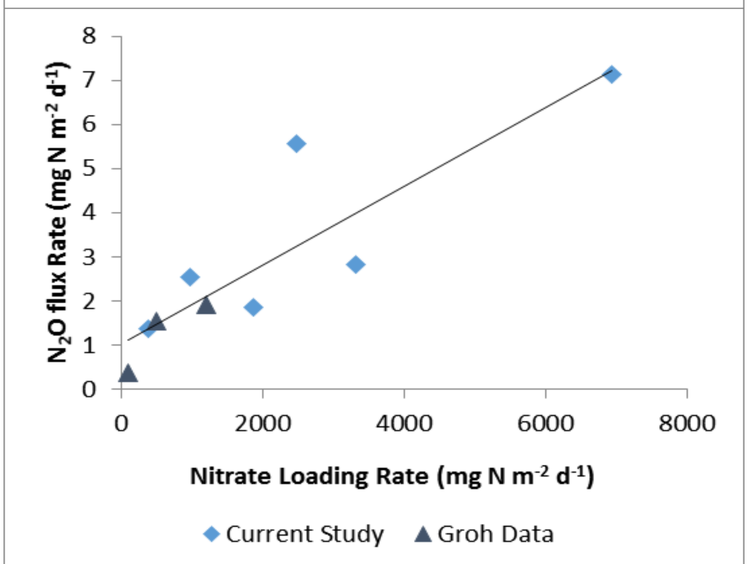

Figure 3
Nitrous oxide emissions vs nitrate loading into the system (R^2 0.71). R^2 was 0.78 when the Groh data was included. (Results from current study and Groh, personal communication).

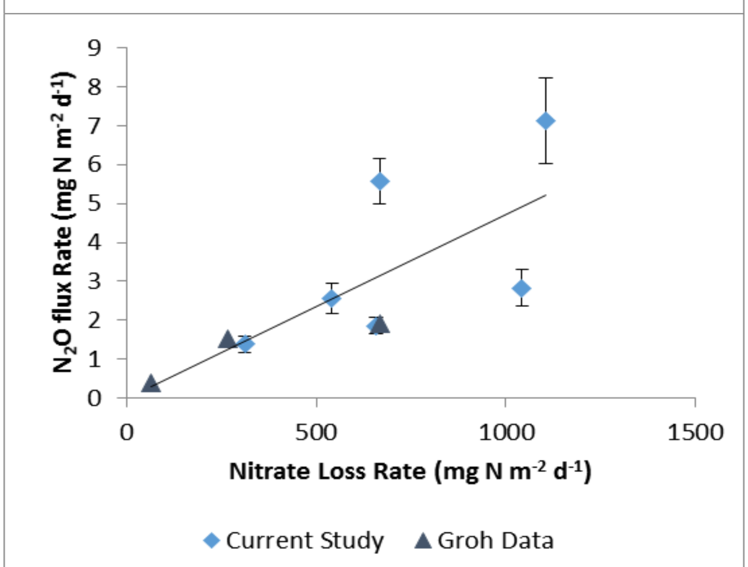

Figure 4
Average seasonal nitrous oxide flux by wetland and year vs nitrate loss (R^2 0.41). R^2 was 0.55 when the Groh data was included. (Results from current study and Groh, personal communication).

CHAPTER 3. GENERAL CONCLUSIONS

Summary & Conclusions

Nitrous oxide and CH_4 emissions from the study wetlands averaged 3.5 mg N_2O-N m^{-2} day^{-1} and 793 mg CH_4 m^{-2} day^{-1} across all wetland sites and dates. The wetlands were highly efficient at denitrifying nitrate to N_2, with N_2O-N emissions averaging only 0.47% of nitrate removal. In combination, results from the current study and others support the expectation that N_2O emissions will increase with increases in nitrate loading (R^2 0.71, Figure 3). However, results also demonstrate that N_2O emissions from wetlands constructed or restored on former agricultural land are similar to emissions from cropland that the wetlands replace.

Wetland greenhouse gas balances are likely better than moderately productive agricultural lands which restored wetlands would typically replace (Hey et al., 2012). Croplands usually have lower carbon sequestration and higher N_2O emissions when compared to Midwest US wetlands (Hey et al., 2012). The water quality benefits, habitat benefits, and favorable greenhouse balances support a strategy of widespread restoration and construction of wetlands to intercept and reduce surface water nitrate loads in the US Corn Belt.

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