NUTRIENT REMOVAL AND GREENHOUSE GAS EMISSIONS FROM 20 YEAR OLD CONSTRUCTED WETLANDS RECEIVING TILE DRAINAGE WATER: A BIOGEOCHEMICAL ANALYSIS

BY

TYLER ANTHONY GROH

THESIS

Submitted in partial fulfillment of the requirements for the degree of Master of Science in Natural Resources and Environmental Sciences in the Graduate College of the University of Illinois at Urbana-Champaign, 2014

Urbana, Illinois

Master’s Committee:

Professor Mark B. David, Adviser
Emeritus Professor David A. Kovacic
Associate Dean George F. Czapar
ABSTRACT

Agricultural nutrient loss (N and P) from the Midwestern United States is an important issue, especially from tile drained land. There are a wide range of in-field and edge-of-field practices to reduce losses that have been studied and are currently recommended, such as cover crops, fertilizer management, drainage water management, wetlands, wood chip bioreactors, and saturated lateral buffers. In-field practices can lead to reduced crop yields and may cause problems with annual implementation, so edge-of-field methods are viewed as an alternative to avoid these problems. Constructed wetlands receiving tile drainage have the potential to remove and retain both N and P from agricultural runoff. This has been shown through many studies. However, little is known about how the N and P removal processes and efficiencies change as constructed wetlands age. In addition, there have been few measurements of greenhouse gas, GHG, emissions from agricultural constructed wetlands. This study evaluated N and P removal and greenhouse gas emissions in three 20 year old constructed wetlands that receive tile drainage from corn and soybean fields in southern Champaign County, Illinois.

All three wetlands were equipped with data loggers and pressure transducers at their inlets and outlets to measure flow, using Agri Drain control structures. Water samples were collected to determine N and P concentrations and loads. Wells were also installed in and outside of the berms of wetlands A and B in order to determine the volume of seepage water, and the nitrate load lost through this pathway. Using the inlet, outlet, and seepage nutrient load data, nutrient budgets were constructed, and N and P removal rates were calculated. Greenhouse gas fluxes, including carbon dioxide, methane, and nitrous oxide, were measured from the inundated and terrestrial portions of the wetlands via floating and static chambers, respectively. These fluxes were measured throughout the year, and were linearly interpolated in order to construct a cumulative flux.

The wetlands removed approximately 46% of the nitrate-N and 2% of the total P inlet load when they were first established. During two years of study (2012-2013) I determined that the wetlands
removed on average 59% of the nitrate-N and 32% of the total P inputs. Hydraulic loading had a strong, positive relationship with the mass of N removed per hectare with a $R^2$ value of 0.73. The predictability of N removal increased when the nitrate inlet flow weighted mean was also considered in a multiple linear regression. Together, the hydraulic loading and average nitrate concentration explained 85% of the variation in N removal. Total P wetland retention was more difficult to explain, and varied greatly between water years and wetlands, with removal ranging from -7 to 100%. Unlike N removal, P retention was not strongly related to hydraulic loading and inlet P flow weighted means.

The dominant GHG emitted from the wetlands was carbon dioxide, which made up between 75 and 96% of the total GHG emissions. The nitrous oxide flux, which was of special concern due to the denitrification process, only contributed between 3.7 and 13% of the total cumulative GHG flux. Methane made up even less, between 0.08 and 12%, of the total cumulative GHG flux. Further, the terrestrial portions of the wetlands emitted the majority, between 86 and 99%, of the total GHG emissions. Nitrous oxide emissions were 7.6 and 3.1% of the total nitrate loss from wetlands A and B in 2012 and 2013, respectively, with the larger percentage when the wetlands were mostly dry because most of the losses were from terrestrial portions of the wetlands. The GHG fluxes, from both the inundated and terrestrial portions of the wetland had a threshold with water and soil temperature respectively. GHG samples collected from water below 18°C or soil below 15°C typically had low concentrations, which ultimately translated to low fluxes. All of the large methane and nitrous oxide fluxes observed took place above these temperature thresholds. Soil moisture was also correlated to terrestrial GHG fluxes. Terrestrial GHG fluxes with a soil moisture level above 25% typically had larger fluxes than when they were drier.

Overall, these wetlands continue to function well in controlling nitrate and total P losses from tile drained agricultural fields in Illinois. Wetland age has not affected nitrate removal. Maximizing hydraulic loading leads to the greatest nitrate removal per ha of wetland. Nitrous oxide emissions were
a small percentage of nitrate removed, although this was larger during the drought year of 2012.
ACKNOWLEDGEMENTS

First, I would like to thank my advisor, Dr. Mark David, for the opportunity to conduct research in his laboratory and the chance to get my Master’s degree at the University of Illinois. I also want to thank him for the many semesters I was able to be a teaching assistant for him and other professors. Further, I would like to thank my committee members, Dr. David Kovacic and Dean George Czapar, for their insight and for taking their time to guide my thesis project, funding for which would not be possible without the USDA’s financial support.

Second, I really appreciated Drs. David, Darmody, and Arai for giving me the chance to be a teaching assistant for them. They provided great insight on how to teach at the university level, and I truly feel that I am a better lecturer and lab instructor because of my teaching experiences with them. Also, Dr. Darmody taught me how to take advantage of any situation, especially when it came to OPM.

Third, I would like to thank everyone who helped me conduct my research. This includes, but is not limited to, Corey Mitchell, Lowell Gentry, Dr. Candice Smith, Morgan Davis, Tito Lavaire, Mike Masters, and Dr. Carmen Ugarte. The aforementioned people were crucial in field sampling, running analyses, and helping me interpret my results. A project as big the Embarras Wetland Study could not have been done without this amazing group of scientists. I would also like to thank the many undergraduate workers for their hard work, early summer mornings, and dedication to this project.

Fourth, I want to take time to thank my family, both blood related and academically adopted. My lab mates Tito Lavaire and Morgan Davis, who I think of as adopted brothers, were continuously there for me when I needed them. A person could not ask for better friends and colleagues. In addition to their support, I would like to thank my family for all of their words of motivation, and for believing in me throughout my Master’s. This is especially true of my mom, who is my idol and motivation. Her hard work to provide my brothers and me with everything we ever wanted gave me the motivation to perform at my best every day.
Finally, I would like to dedicate this thesis to my grandpa, Vernon Schultz, who passed away during my time here at the University of Illinois. He was a huge supporter of my academic career path, and provided me with the confidence needed to make every moment of my Master’s degree count. My one hope is that I can one day live up to the legacy he left in my family.
# Table of Contents

**LIST OF FIGURES** ............................................................................................................................................ ix

**LIST OF TABLES** .............................................................................................................................................. xi

**INTRODUCTION** ............................................................................................................................................... 1

**OBJECTIVES** .................................................................................................................................................. 5

**LITERATURE REVIEW** .................................................................................................................................... 6
  - Wetlands ......................................................................................................................................................... 6
  - Nitrogen Removal .......................................................................................................................................... 7
  - Phosphorus Retention ................................................................................................................................. 9
  - Wetland Seepage Nitrogen and Phosphorus Removal .................................................................................. 12
  - Greenhouse Gas Emissions ......................................................................................................................... 14
  - Previous Embarras Wetland Studies ........................................................................................................... 19

**MATERIALS AND METHODS** ....................................................................................................................... 23
  - Site Description .......................................................................................................................................... 23
  - Wetland Water Samples ............................................................................................................................. 27
  - Seepage Well Water Samples ................................................................................................................... 28
  - Water and Nutrient Budgets ....................................................................................................................... 30
  - Greenhouse Gas Samples ........................................................................................................................... 32

**RESULTS** ....................................................................................................................................................... 36
  - Wetland Hydrology ..................................................................................................................................... 36
  - Wetland Nitrogen Budgets ......................................................................................................................... 38
  - Wetland Phosphorus Budgets ................................................................................................................... 41
  - Wetland Hydrology and Nutrient Load Relationships .............................................................................. 41
  - Seepage ....................................................................................................................................................... 48
  - Wetland Greenhouse Gases ....................................................................................................................... 52
  - Wetland Greenhouse Gas Relationships with Abiotic Controls ................................................................. 56
  - Seepage Berm Greenhouse Gases ............................................................................................................. 62

**DISCUSSION** .................................................................................................................................................. 65
  - Wetland Hydrology ..................................................................................................................................... 65
  - Wetland Nitrogen Removal ....................................................................................................................... 66
  - Wetland Phosphorus Retention ................................................................................................................ 70
  - Wetland Greenhouse Gas Fluxes ............................................................................................................. 73
<table>
<thead>
<tr>
<th>Section</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>Seepage Berm Greenhouse Gas Fluxes</td>
<td>75</td>
</tr>
<tr>
<td>Greenhouse Gas Flux Comparisons with Other Studies</td>
<td>75</td>
</tr>
<tr>
<td>CONCLUSIONS</td>
<td>77</td>
</tr>
<tr>
<td>REFERENCES</td>
<td>79</td>
</tr>
</tbody>
</table>
LIST OF FIGURES

Fig. 1. Aerial view of wetlands A, B, and D with pictures taken of each wetland.
The aerial view was done with Google Earth software. ..................................................25

Fig. 2. The 2012 and 2013 precipitation and nitrate loads for all three wetlands’ inlets and outlet. ........43

Fig. 3. Wetlands A, B, and D nitrate and total P removal rates, in both mass removal per ha yr and
percent removal, plotted against hydraulic loading for all years sampled. .............................46

Fig. 4. Nitrate (Top) and total P (Bottom) inlet flow weighted means for wetlands A, B, and D
for all years analyzed. .............................................................................................................47

Fig. 5. The average nitrate concentrations for both wetland A and B’s berm (the wells closest to the
wetlands) and riparian (the wells furthest from the wetlands and right next to the
Embarras River) wells during both water years. These box-and-whisker plots
contain the average nitrate concentrations obtained from the respective well locations.
Average values from each sampling date were placed into these figures. These graphs
therefore give the overall average, as well as extreme values, for each seepage well
location of the entire seepage nitrate dataset. ........................................................................49

Fig. 6. Wetland A berm and riparian wells’ nitrate concentration for both 2012 and 2013.
This box-and-whisker plot separated out all three of wetland A’s berm wells and
riparian wells in order to observe the nitrate reduction as water seeped out of the
wetland to the Embarras River, from the inlet to the outlet. Each box and whisker plot
contained all nitrate concentrations from all sampling dates. The results show that not
only does the water nitrate concentration decrease as water seeps from the wetland to
the river, but that both the berm and riparian wells have lower nitrate concentrations
going from wetland A’s inlet to its outlet. ................................................................................50

Fig. 7. Carbon dioxide, methane, and nitrous oxide cumulative fluxes for wetlands A and B,
displayed overtime while separating the terrestrial source of each gas from the total
cumulative flux. The orange lines outline the final dry down of each wetland for both
water years. The first line in each water year represents the last day of outlet flow,
and the second line represents when the wetlands are completely dry. ................................54

Fig. 8. Wetland A and B’s cumulative carbon dioxide, methane, nitrous oxide, and total fluxes (Top),
as well as the individual fluxes for each sampling day (Bottom), all expressed in carbon dioxide
equivalent units for the 2012 and 2013 sample years. The orange lines outline the final
dry down of each wetland for both water years. The first line in each water year represents
the last day of outlet flow, and the second line represents when the wetlands are completely dry. .................................................................55

Fig. 9. Surface water temperature correlations with carbon dioxide (Top), methane (Middle),
and nitrous oxide (Bottom) fluxes from inundated chambers for both wetlands A and B.
The carbon dioxide, methane, and nitrous oxide data were separated at the 18⁰ C inflection
point. The two graphs include all inundated fluxes from 2012 and 2013. ...............................57

Fig. 10. Terrestrial carbon dioxide (Top), methane (Middle), and nitrous oxide (Bottom) fluxes
plotted with soil temperature for wetlands A and B in 2012 and 2013. .....................................58

Fig. 11. Terrestrial carbon dioxide (Top), methane (Middle), and nitrous oxide (Bottom) fluxes
plotted with percent soil moisture for wetlands A and B in 2012 and 2013. .............................59

Fig. 12. Inundated methane (Top) and nitrous oxide (Bottom) fluxes compared to water nitrate
concentrations for both wetlands in 2012 and 2013. The data presented were from all
sample locations in wetlands A and B. .....................................................................................60

Fig. 13. Inundated methane (Top) and nitrous oxide (Bottom) fluxes relations with dissolved
methane and nitrous oxide concentrations respectively. The data presented were from
all sampling dates in 2012 and 2013 for each sample location in wetlands A and B. ..................61
Fig. 14. Cumulative carbon dioxide, methane, nitrous oxide, and total flux for wetland A’s riparian seepage berm during 2012 and 2013 (Top). Individual carbon dioxide, methane, nitrous oxide, and total flux during each 2012 and 2013 sampling date for wetland A’s berm (Bottom)....

Fig. 15. Nitrate removal in both mass per area (Top) and percent of inlet (Bottom) related to wetland hydraulic loading (Left) and the inlet flow weighted nitrate concentration (Right) for the current Embarras Wetland study and other agricultural constructed wetlands. ..................69

Fig. 16. Total P removal in both mass per area (Top) and percent of inlet (Bottom) related to wetland hydraulic loading (Left) and the inlet flow weighted total P concentration (Right) for the current Embarras Wetland study and other agricultural constructed wetlands. ..................72
LIST OF TABLES

Table 1. Dimensions and drainage area for wetlands A, B, and D................................................................. 26
Table 2. Water budget for wetlands A, B, and D for all study years................................................................. 37
Table 3. Hydrological data for wetlands A, B, and D for all years studied...................................................... 39
Table 4. Yearly inlet and outlet N loads along with mass removed per ha of wetland and percent removal for wetlands A, B, and D................................................................................................. 40
Table 5. Yearly inlet and outlet P loads along with percent P removed and mass P removed per ha of wetland for wetlands A, B, and D................................................................................................. 42
Table 6. Seepage nitrate removal rates for wetlands A, B, and D during all study years............................... 51
Table 7. Cumulative fluxes of carbon dioxide, methane, and nitrous oxide for wetlands A and B during the 2012 and 2013 water years................................................................. 53
Table 8. Carbon dioxide, methane, nitrous oxide, and total greenhouse gas fluxes for wetland A’s seepage berm in 2012 and 2013........................................................................................................ 63
INTRODUCTION

The hypoxic zone in the Gulf of Mexico, often referred to as the dead zone, has been identified as one of the major environmental concerns of the Mississippi River Basin, MRB, in the United States. Hypoxia is defined as any water that has a dissolved oxygen concentration less than 2 mg L\(^{-1}\) (Goolsby and Battaglin, 2000; Turner et al., 2012). The zone in the Gulf of Mexico forms when the oxygen in the water column decreases below this threshold concentration. This low oxygen level is usually isolated only to the benthic portion of the Gulf because of a strong halocline and thermal cline that forms in the summer, both of which effectively stratify the Gulf water (Rabalais et al., 1996; Goolsby and Battaglin, 2000). Ultimately, the low oxygen levels in the Gulf of Mexico can cause stress and potential death for any bottom dwelling organisms that cannot swim away. Hypoxia is caused by the stimulation of algal production by nitrogen (N) and phosphorus (P) additions, followed by decomposition by microbial organisms in the deeper depths of the Gulf of Mexico (Goolsby and Battaglin, 2000; Mitsch et al., 2001; Turner et al., 2012). Therefore, the hypoxic zone is typically larger if the amount of algal production increases.

Since the Gulf is a marine system, N is an especially limiting nutrient (Rabalais et al., 1996). The total amount of nitrate exported to the Gulf from the Mississippi River has nearly tripled since the 1950’s (Goolsby and Battaglin, 2000). However, with this overall increase in nitrate-N export to the Gulf, one of the smallest hypoxic zones, 4,400 km\(^2\), occurred in 2000, while the largest, 22,000 km\(^2\), was in 2002 (LUMCON, 2013). Therefore, the hypoxic zone is not increasing each year, but rather varies with nitrate load, determined by the conditions in the MRB and flow in the Mississippi River.

Tile drainage has clearly been shown to increase stream and river nitrate loads (Goolsby and Battaglin, 2000; Crumpton et al., 2006; McIsaac and Hu, 2004; David et al., 2010). Roughly 30 million ha of the MRB were drained in the 19\(^{th}\) and 20\(^{th}\) centuries (Mitsch and Day, 2006). The more tiles that are installed, the more effective farmers can be at lowering the water table and turning fertile, agriculturally
unusable soil into productive farmland. However, tiles not only increase the amount of productive farmland and agricultural productivity, but they also increase the amount of N exported to the Gulf of Mexico. In general, tiled land exports more N than areas without tiles (Goolsby and Battaglin, 2000; Crumpton et al., 2006; McIsaac and Hu, 2004; David et al., 2010). Tile water nitrate concentrations in the MRB may be 20 mg L−1 or higher (e.g., Goolsby and Battaglin, 2000) and many rivers have concentrations > 10 mg N L−1 (e.g., Royer et al., 2006). These high nitrate concentrations come from soil mineralization and applying large amounts of inorganic N fertilizer to land that has tile drainage (David et al., 2010).

The major inputs of N to the MRB are inorganic N fertilizers (52% of inputs), N fixation (40% of inputs, primarily by soybean), and atmospheric deposition (8% of inputs) (David et al., 2010). David et al. (2010) showed that the tile-drained agricultural fields of the upper MRB (Minnesota, Iowa, Illinois, Indiana, and Ohio) were the source of most of the nitrate load in the Mississippi River.

In-field and edge-of-field methods can be used in agricultural production systems to reduce N and P losses (USEPA, 2008). One in-field practice is to reduce the amount of N and P fertilizer used in the MRB (Day et al., 2003). Farmers in the MRB typically apply between 100 and 200 kg N ha−1 yr−1 (David et al., 1997a). Using the SWAT model, it was determined that lowering the N fertilizer rate by 10, 20, 30, and 50% may decrease riverine nitrate flux by 10, 18, 29, and 43% respectively (Hu et al., 2007). However, with the fertilizer decrease, there would also be yield decreases. The same model from Hu et al. (2007) predicted yield decreases of 6% when the fertilizer was reduced by 10%. This result may have been due to the assumptions used in the model, but also may be a warning sign for farmers that want to maximize their crop yields.

Other in-field methods for reducing the N and P loads include improving the timing of fertilizer application, using nitrification inhibitors, investing in slow release fertilizers, and using cover crops (USEPA, 2008; Gentry et al., 2009). This is especially true for reducing the amount of N fertilizer that leaches from the fields. The practices discussed in Gentry et al. (2009) are good in theory, but are
difficult to implement on a large scale with conventional agriculture (Stevenson and Cole, 1999).

Vitousek et al. (2009) discussed that food security should be considered first and foremost for policy makers, and that the farming agencies do not have the jurisdiction to control farming practices. Therefore, more options for reducing the amount of fertilizer runoff into adjacent streams need to be given to farmers.

There are a wide range of edge-of-field practices that have been studied to reduce N and P losses from agricultural fields, including drainage water management (DWM), saturated lateral buffers, woodchip bioreactors, and constructed wetlands. Skaggs et al. (2012) summarized DWM studies to date and found that nitrate losses from tile drained fields were reduced by 18 to 85%, with most of the studies showing >50% removal. However, a limitation of this work is that the fate of the held back water and nitrate has not been determined, which may reduce effectiveness at a watershed scale (Woli et al., 2010). Saturated lateral buffers can reduce nitrate losses by taking some of the tile flow that would normally directly enter a ditch or stream and routing it through a buffer strip, which allows the water to slowly seep to the ditch. This allows for denitrification and plant uptake of the nitrate along the flow path. At this time they are poorly studied, with only one published report. Jaynes and Isenhart (2014) found that 55% of the tile flow was able to be directed to pass through the forested buffer, with all of the nitrate removed from this flow (45% of the overall tile load was therefore removed by the buffer system). Woodchip bioreactors involve directing some of the tile flow through a trench filled with woodchips, a high C:N material. Bioreactors are not all constructed the same way, but all have shown that they can reduce agricultural nitrate loads. Percent reduction for these bioreactors can range from 12 to 99.5% depending on water source, retention time, woodchip temperature, wood chip age, and seasonality (Woli et al., 2010; Chun et al., 2010; Robertson, 2010). Drainage water management, saturated lateral buffers, and woodchip bioreactors have not been evaluated for P removal, as the focus of these techniques has been on nitrate removal.
Constructed wetlands have been used to remove nitrate and retain P from agricultural tile drains. These wetlands can be created by breaking and daylighting a tile line and excavating an area large enough to hold the tile water long enough for denitrification to occur (Kovacic et al., 2000). Like most other end-of-pipe nutrient removal techniques, wetlands vary in their ability to remove N and P. Nutrient removal depends on soil temperature, hydraulic retention time, total hydraulic load, wetland physical dimensions, and seasonality of inlet flow (Phipps and Crumpton, 1994; Mitsch et al., 1995; Raisin and Mitchell, 1995; Woltemade, 2000; Spieles and Mitsch, 2000; Braskerud, 2002; Reinhardt et al., 2005; Song et al., 2010; Hey et al., 2012). The wetlands discussed in these studies had varying levels of nutrient removal efficiencies for both N and P. Another, not well studied, characteristic of wetlands that can determine how well they remove nutrients is the wetland’s age. The lack of knowledge of how constructed wetlands’ age affects nutrient removal is the primary focus for this study.

Another primary focus for this study included measuring GHG emissions. Agricultural constructed wetlands have been shown to increase environmental quality by removing nutrients from agricultural runoff, but they may be negatively impacting the environment by emitting carbon dioxide, methane, and nitrous oxide (Altor and Mitsch, 2008a). These GHGs, especially methane and nitrous oxide, have a potent global warming effect. Therefore, it was imperative to measure carbon dioxide, methane, and nitrous oxide emissions alongside wetland nutrient removal in this study.
OBJECTIVES
The objectives of my study were to:

1) determine N and P budgets for established constructed wetlands receiving tile drainage in order to determine their N and P removal efficiencies and compare them with efficiencies following construction;

2) estimate seepage nitrate removal rates to determine overall removal rates of the wetland-buffer system; and

3) measure greenhouse gas (methane, carbon dioxide, and nitrous oxide) emissions from the wetlands.
LITERATURE REVIEW

Wetlands

Wetlands, by nature, are difficult to characterize and to delineate (Gray et al., 1999). They can take on many forms, and may include swamps, marshes, and bogs. However, there are three defining characteristics for a wetland: wetland hydrology, soils, and vegetation. Wetlands may receive their water from either above or below ground sources. Usually wetlands are in areas where the presence of water is noticed at or above the soil surface during all or part of the year.

Wetlands by definition are found with wet, or hydric, soils (Gray et al., 1999). These soils develop grey mottles from being saturated for an extended period of time, usually in terms of several weeks. Not all hydric soils are the same; they can have a range of moisture contents. Hydric soils that have a mixture of brown and grey mottles near the surface horizons are usually in wetlands with a fluctuating water table. On the other hand, soils that have dark grey mottles are in wetlands with near continuous inundation.

The third and final defining characteristic that wetlands need is hydrophilic plants (Gray et al., 1999). Hydrophilic plants are water loving plants that can survive soil that is completely saturated. Some of these wetland plants can be classified as obligate hydrophytes, while some are facultative, meaning they can survive wet or dry conditions. Having wetland plants is a clear byproduct of both the wetland hydrology and soils. However, an area cannot be classified as a wetland without hydrophilic plants.

Wetlands can provide many services, both environmental and socio-economical. These functions include shoreline stabilization and erosion control, flood control, sediment trapping and nutrient removal, wildlife habitat, recreation, and education (Gray et al., 1999). For the purpose of this study, the literature review will focus on nutrient removal, and how this service’s efficiency changes through time in constructed wetlands.
Nitrogen Removal

Many studies have shown that constructed wetlands can be effective at removing nutrients, especially N, from water (O’Geen et al., 2010). Specifically, constructed wetlands are efficient at removing nitrate. This is advantageous since the majority of the N in agricultural water, especially tile water, is in the form of nitrate (Baker, 1998). The two major mechanisms that play a role in N removal from wetlands include plant and periphyton uptake and microbial denitrification (Baker, 1998; Mitsch et al., 1999; Mitsch et al., 2000; Day et al., 2003; O’Geen et al., 2010). Denitrification is especially important since wetlands that receive agricultural runoff and tile water obtain most of their water in late winter to early spring before the wetland plants emerge (Hoagland et al., 2001). In addition, Xue et al. (1999) found that plants in mesocosms only removed up to 10 percent of the nitrate available. Therefore, the remaining 90 percent was either lost in the wetland’s water column or was denitrified.

Plants may be thought of as a temporary sink or a conversion mechanism (Hoagland et al., 2001). This is due to the fact that plants convert inorganic N to organic N, which may ultimately leave the wetland (Mitsch et al., 1999). Also, plants can be decomposed by microorganisms and the nutrients that were once bound up in organic matter can be mineralized back into the wetland water. However, plants can be a permanent sink if the layers of organic matter accumulate in anaerobic sediments over the age of a wetland (Kadlec, 1999). This removal mechanism usually occurs in wetlands that do not dry down, thus allowing the sediments to remain anaerobic. Algae, along with plants, may also be thought of as a somewhat temporary sink for N (Hoagland et al., 2001). Algae tends to accumulate in wetlands due to a high concentration of nutrients. This is especially true in warmer climates. Hoagland et al. (2001) found that 10 kg N ha\(^{-1}\) was removed via algal production. The authors also noted that this rate was during a year with minimal algal production due to lack of wetland water. They speculated that wetter years would have greater algal production and N removal.

Along with nitrate, ammonium may also be a substantial source of the total N in agricultural water draining into constructed wetlands. This is especially true when high precipitation or rapid
thawing events occur in early spring after fertilization (Kovacic et al., 2000). Ammonium, like nitrate, can be taken up by plants and algae to be processed through these temporary sinks. Some ammonium may be retained on the soil’s cation exchange sites if the soil stays anaerobic so that nitrification does not occur (Mitsch et al., 1999). This, however, is rare since wetland soils can have an aerobic zone on their surface that nitrifies ammonium to nitrate.

Regardless of the form of N that enters the wetland, the total amount of inorganic N removed via microbial processes depends on hydraulic retention time, organic C availability, redox potential in wetlands soil, N input concentration, and water column depth (Fink and Mitsch, 2004). This is especially true for nitrate. Hydraulic retention time is arguably the most important factor in N removal in wetlands. The longer agricultural water is retained in wetlands, the greater amount of time the denitrifying communities have to remove nitrates in the water. Organic C availability is also important since the denitrifiers have to oxidize an organic C source to carbon dioxide when they reduce nitrate to dinitrogen gas and nitrous oxide (Stevenson and Cole, 1999). In Hernandez and Mitsch (2007), the amount of organic matter doubled in the 0 to 9 cm depth in the wetland, which correlated to a 25-fold increase in the denitrification potential in the same soil layer. Redox potential is also critical for N removal since nitrate will only be reduced once all of the oxygen is removed from the wetland’s substrate. Much is known about how wetlands remove N from their influent, but little is known about how this removal rate changes over time.

Other, somewhat minor, factors that affect the amount of N removed from constructed wetlands through both microbial and plant processes include temperature, hydrology, and wetland vegetation types. Mitsch et al. (2005) looked at two wetlands in Ohio and a wetland complex in Louisiana. In this study, the authors found that the Louisiana wetland complex removed more N, 46 g-N m⁻² yr⁻¹, than the average of the Ohio wetlands, 39 g-N m⁻² yr⁻¹. The main conclusion from this study was that the warmer climate and longer growing season in Louisiana were the two main reasons why the
Louisiana wetland system’s N removal rates were greater than the Ohio wetlands’ average. Song et al. (2010) worked on the same Ohio wetlands that were in Mitsch et al. (2005). However, Song et al. (2010) were more focused on how hydrology affected denitrification rates and microbial communities. Denitrification rates decreased for all areas except for the portion near the wetland inlet during the inundated period. During the reflooding event, denitrification rates went from near zero during dryness to the same, if not higher, rates than the soils had during the inundated period. These rates decreased as the wetland dried down, but easily rebounded when the wetlands become inundated again. This may have been due to increased N mineralization, which increases the substrate for the denitrifying communities. Also pointed out was the fact that the denitrifying communities present were not the reason for this changing denitrification rate. Rather, the hydrology was the main reason for the change in rates. This study found that microbial communities were not affected by the flashy hydrology of the study.

Finally, wetland vegetation types and how they affected denitrification rates was studied by Clement et al. (2002). They examined how forested, understory vegetation, and a herbaceous grass sites varied from each other in terms of denitrification rates. The authors found that the denitrification rates between the sites were not statistically significant. This could have been due to the fact that each vegetation type supplied the denitrifiers with enough organic C substrate to support denitrification. Regardless, the denitrification rate did vary significantly with soil depth. Clement et al. (2002) found that there was a higher denitrification rate in the 0-25 cm soil layer, which corresponded to the more organically rich portion of the soil, than in deeper soil depths. This emphasizes the importance of organic substrate in the process of denitrification. Also, even though the highest rates of denitrification was seen in the upper 25 cm, observable denitrification rates were measured down to 75 cm.

*Phosphorus Retention*

In addition to N, wetlands also are able to remove P. Unlike N, P does not have a prominent
gaseous phase (Richardson and Craft, 1993). Therefore, there is no process like denitrification that can remove P from wetland influent. Rather, wetlands must rely on retention processes, like sedimentation, to remove P, especially dissolved reactive phosphorus, DRP. This makes the wetland’s parent material chemical makeup extremely important (Mitsch, 1995). The parent material ultimately determines the pH, iron, aluminum, calcium, and in situ P concentrations in the soil, which in turn, affects how much DRP is retained in the wetland soils (Richardson and Craft, 1993; Richardson et al., 1997). The pH and redox potential affect how mobile DRP is in the wetland. The more acidic a soil is, the more DRP will absorb on iron hydroxides and precipitate out of solution. However, the more basic the soil is, the more DRP will bind with calcium to form precipitates (Richardson et al., 1997). When iron is reduced from its +3 to its +2 redox state, DRP that was once precipitated may be redissolved (Richardson and Craft, 1993). On the other hand, reduction can cause some aluminum and iron minerals to become amorphous and better at precipitating DRP from wetland influent. Wetland soils’ parent material chemical makeup also determines the amount of sorption sites in the wetland soil. Dissolved reactive P can be retained on soil sorption sites, but this retention process is thought to be limited and may only last for a couple of months after a wetland is created (Kadlec, 1999).

Another short-term P retention process includes both plant and algal biomass growth. Both plants and algae can uptake DRP for use in their tissues. Therefore, DRP retention may increase when new biomass and plant species are added to a wetland system, but this increase would be temporary (Richardson et al., 1997). Even though biomass uptake does not improve long-term DRP retention, it does dominate short-term DRP uptake (Richardson et al., 1997; Hoagland et al., 2001). Further, phytoplankton have the ability to remove more DRP than some macrophytes (Richardson and Craft, 1993). Over the life of a wetland, it is expected that the biomass removal process will slow or stop when the total wetland biomass reaches a larger, more stable size (Kadlec, 1999). Also, the DRP that is taken up by plants or algae may be exported out of the wetland as organic P, or the biomass may decompose.
eventually and release between 35 and 75% of the DRP they took out of the water (Richardson and Craft, 1993; Hoagland et al., 2001). Both of these processes would negatively affect the DRP removal efficiency of the wetland system.

The third P retention mechanism, burial of organic materials, has a potential for being a long-term sink (Kadlec, 1999). The burial process may be both sediment and peat-like matter accumulation (Richardson et al., 1997). The majority, possibly up to 80%, of the P in this pool is usually retained in the top 20 cm of buried material. This emphasizes the amount of P that can be removed by recent vegetation if it is added to the somewhat stable organic material at the bottom of wetlands. Also, since this can be a long term P retention mechanism, wetlands should be designed to maximize peat and sediment accumulation (Richardson et al., 1997).

Wetland P retention, regardless of the mechanism used, does not have much seasonality, even in colder climates (Kadlec, 1999). The quantity of P that each mechanism can retain varies depending on the wetland system, but they tend to follow the same order: adsorption/precipitation>plants>periphyton (Richardson et al., 1997). These mechanisms are efficient with the fast uptake of P, but are finite and will eventually run out. However, the accumulation of peat and soil sediment can be thought of as a somewhat permanent P retention process. The chemical and biological nature of a wetland system ultimately controls the P-retention mechanisms, but physical factors are also important. For instance, increased water input, greater wetland depth, and lower hydraulic retention time may limit a wetland’s ability to retain P (Richardson et al., 1997). On average, wetlands may be able to retain 0.5 g P m$^{-2}$ yr$^{-1}$ (Richardson et al., 1997). This average may change if the inlet P concentration goes beyond a certain threshold value. This value is believed to be different for each wetland. In Richardson et al. (1997), this threshold value was assumed to be 1 g P m$^{-2}$ yr$^{-1}$. Any inlet P concentration below this level allowed for constant, low output concentrations of P. Similar to wetland N removal, wetland P retention mechanisms are well known, but little is known about how this retention rate changes over time.
Due to the variation in wetland soils’ parent material and the concentration of inlet P, studies that analyze wetland P removal often have various retention rates. Mitsch et al. (1995) looked at four wetlands, two receiving high and two receiving low inlet flow. There was no significant difference between the high and low flow wetlands in the first two years, having retention rates between 63 and 92% and 62 and 90% respectively. However, during the third year, the wetlands receiving low flow had a greater P retention percentage than the high flow wetlands. All four wetlands increased their total P retention between years 2 and 3. The systems retained between 74 and 87% of the total P that came into them. Soluble reactive P, SRP, retention ranged from 80 to 90% in wetland 4 to between 74 and 92% in wetland 5. Roughly two thirds of the SRP was retained near the inlet. There was no relationship between macrophyte-established areas and those without for SRP sorption. For three out of the four wetlands, SRP retention decreased from year to year. This may be the ultimate result of SRP retention in ageing wetlands. This study also found that water temperature is not as vital of a control on P retention as other physical and chemical controls can be. The authors concluded that more research is needed to truly see how wetlands’ ageing affects their P retention.

Fink and Mitsch (2004) also studied P retention. This study looked at a 1.2 ha wetland in Ohio. The authors calculated a soluble reactive P reduction, SRP, of 54.4 and 59.4% for the first and second years of the study respectively. The average SRP loading rate for these years was 4.7 g P m\(^{-2}\) yr\(^{-1}\). There was no relationship between rainfall or volume of water and SRP reduction. The total P reduction was 74.4 and 40.6% in the first and second water years respectively. The average total P loading rate was 7.1 g P m\(^{-2}\) yr\(^{-1}\). The decrease in total P removal was somewhat to be expected when wetlands age, but it may also have been due to the relatively high loading rate of total P.

**Wetland Seepage Nitrogen and Phosphorus Removal**

Constructed wetlands, by default, are usually designed in a way that allows water to seep from the inside of the wetland to an area of groundwater or an adjacent surface water body down gradient of
the wetland. Also, constructed wetlands are usually surrounded by either natural grassland or forested area to act as a buffer strip between the agricultural field and the adjacent surface water body. Both the seepage and grass buffer strips may be efficient at removing nutrients seeping from the wetland and from agricultural surface runoff. Grass buffer strips adjacent to surface water bodies are usually effective at removing sediments from overland runoff (Osborne and Kovacic, 1993). These strips may also be effective at removing nutrients that are dissolved in overland runoff through microbial and vegetative uptake and absorption by both organic and inorganic portions of the soil.

Usually, nitrate removal in buffer strip zones varies and is not homogeneous throughout the zone (Addy et al., 1999). Riparian buffer strips are heterogeneous both horizontally and vertically with their hydrology, sediment properties, and biogeochemical pathways (Hill, 1996). Poorly drained soils may be a little better at nitrate removal than better drained soils due to their anaerobic characteristics (Addy et al., 1999). Also, along the lines of biogeochemical pathways, having a higher concentration of organic C in specific areas of the seepage berm may lead to a greater potential for denitrification. However, the authors in Addy et al. (1999) showed that there was no statistical difference in nitrate removal between a forested or a mowed buffer strip. However, they did find that the amount of denitrification gas, nitrous oxide, was related to the nitrate concentration in the groundwater. This relationship had a coefficient of determination equal to 0.87. Both the mowed and forested area had similar nitrous oxide production. The authors also found that the forested buffer strip had higher C concentrations in batches of the profile than the mowed strip. This adds to the idea of heterogeneity of buffer strips both temporally and spatially.

Seepage and buffer strips will not remove nitrate at all if the strips have tile lines running through them. Therefore, the only way of removing nutrients is to have the water slowly infiltrate from one end of the buffer strip to the other. If water is allowed to slowly seep through the buffer strip, nitrate can be removed from the water through microbial and vegetative uptake along with
denitrification (Groffman et al., 1992; Haycock and Pinay, 1993; Hill, 1996). Jordan et al. (1993) found that the majority of the nitrate in the shallow ground water gets removed in the first 35 m of a riparian forested buffer strip. The authors measured 8 ppm nitrate entering the strip and 0.4 ppm roughly 35 m into the forest using shallow groundwater wells. The removal efficiency may also be different for areas that have seen high nitrate levels for a long time and those that have not seen nitrate-rich water at all (Groffman, Gold, and Simmons, 1992). Dilution, caused by an external source of groundwater, may also affect the calculation of removal efficiencies for seepage and buffer strips. However, dilution may be estimated by looking at the chloride:nitrate ratios of the water samples taken along the flow path through the strip (Hill, 1996).

Riparian buffer strips may not do as well at retaining P compared to their N removal efficiencies (Osborne and Kovacic, 1993). The Osborne and Kovacic (1993) study focused on comparing nutrient removal in grass, forested, and crop buffer strips. The authors found that the forested buffer strip concentrated P in the upper soil horizons, and that the grass buffer strip was not statistically different from the crop buffer strip. Therefore, neither the forested or grass buffer strip did much for retaining P. However, it was speculated that if the trees were harvested from the forested buffer strip, the strip would have removed P from the shallow ground water. This was especially true since the forested buffer strip was considered a mature forest ecosystem. On the other hand, nitrate was found to be significantly lower in the grass and forested buffer strips when compared to the crop land buffer strip. This was most likely due to the process of denitrification. Therefore, similar to wetlands, buffer strips cannot retain P as well as they can remove N because there is no gaseous elimination step in the P cycle.

Greenhouse Gas Emissions

Wetlands can provide environmental benefits such as nutrient, N and P, removal, however, they also may emit greenhouse gases, GHGs. The three most important GHGs emitted from wetlands include methane, carbon dioxide, and nitrous oxide. These three gases can be emitted through natural
processes in constructed wetlands. Wetlands provide anaerobic, and therefore reducing, conditions that may increase the creation of methane in wetland sediments (Altor and Mitsch, 2008a). As a whole, wetlands may make up between 20 and 25% of the total world methane emissions, which comes out to be between 115 and 227 Tg CH$_4$ yr$^{-1}$ (Mitsch et al., 2012). Another study by Sovik et al. (2006) states that most of the methane production from terrestrial systems comes from wetlands, rice paddies, and landfills. Schlesinger (1997) also states that methanogenesis from wetlands is the dominant form of natural methane emissions. Furthermore, with atmospheric methane concentration increasing at a rate of 1% yr$^{-1}$, it is extremely important to look at wetland methane fluxes (Schlesinger, 1997). However, wetlands may offset this methane production by sequestering C, mainly carbon dioxide, from the atmosphere (Altor and Mitsch, 2008a). Also, wetland plants may determine the amount of methane emitted since plants excrete root exudates at various rates, thus providing a varying amount of carbonaceous material for methane emissions.

Wetland hydrology controls the redox potential of wetland sediments and, by extension, how much methane is produced. Altor and Mitsch (2008a) looked at two wetlands in Ohio for two years under varying hydrology. In the first year, the wetlands were subjected to flashy hydrology, and in the second year, the wetlands were under more steady-state hydrology regulation. The authors found that fully inundated areas were the only zones of the wetlands that differed statically from each other between the two different hydrology years. The fully inundated areas under pulsed hydrology had a lower methane flux, 5.65 ± 1.01 mg CH$_4$-C m$^{-2}$ hr$^{-1}$, while the steady state year produced a higher flux, 11.06 ± 2.16 mg CH$_4$-C m$^{-2}$ hr$^{-1}$. Therefore, the fluctuation between anaerobic and aerobic conditions resulted in a lower methane emission than the prolonged anaerobic conditions. This may have implications for wetlands that are not fully inundated for the entire year.

Carbon dioxide may also be emitted from wetlands. The amount of carbon dioxide emitted depends on climate, hydrology, and soil temperatures, but the decomposition of wetland organic matter
also plays a significant role in carbon dioxide emissions (Altor and Mitsch, 2008a). Decomposition can take place in both aerobic and anaerobic conditions (Bernal and Mitsch, 2008). The speed of the wetland organic matter decomposition depends on climate, both temperature and moisture, and the soil’s organic matter quality (Schlesinger, 1997). For some soils, an increase of 10° C results in a doubling of soil organic matter decomposition (Schlesinger, 1997). This would therefore increase the amount of carbon dioxide emitted from a wetland. Another source of carbon dioxide may come from methanotrophic bacteria. These microbial communities can convert methane to carbon dioxide in the presence of oxygen. Anaerobic wetland sediments may have oxygen pockets where this transformation can take place, thus making GHG emissions hard to predict (Altor and Mitsch, 2008a). As mentioned earlier, wetlands may also be sinks for C. Wetland plants can photosynthesize and turn atmospheric carbon dioxide into organic C. This organic C can accumulate in wetlands over time, especially if the wetland is highly anaerobic.

Determining the rate of organic C accumulation along with methane and carbon dioxide emissions is important when trying to make an accurate C budget for a wetland. Wetlands can both accumulate organic C via new plant and periphyton growth, but they can lose C via both aerobic and anaerobic decomposition (Badiou et al., 2011). The byproducts of these forms of decomposition are carbon dioxide and methane respectively. In Mitsch et al. (2012), the authors looked at soil C accumulation and methane emission rates in 7 temperate and tropical wetlands. The temperate wetlands were in Ohio, while the tropical wetlands were located in Costa Rica and in Botswana, Africa. The natural temperate wetland had a C accumulation rate of 143 g C m$^{-2}$ yr$^{-1}$, and the two created temperate wetlands had a C accumulation rate of 219 and 267 g C m$^{-2}$ yr$^{-1}$. The natural wetland had the highest methane emissions, 57 g C m$^{-2}$ yr$^{-1}$, for the temperate wetlands. The tropical wetlands accumulated between 42 and 306 g C m$^{-2}$ yr$^{-1}$. For the tropical wetlands, the floodplain wetland had the highest methane emissions, 220-263 g C m$^{-2}$ yr$^{-1}$. The two constructed temperate wetlands and the flow
through tropical wetland had lower methane emissions, an average of 30 and 33 g C m$^{-2}$ yr$^{-1}$ respectively. The study concluded that constructed wetlands have lower methane emissions than natural ones over the first 13-15 years of the wetlands. Also, this study suggests that wetlands in temperate climates seem to do better at sequestering C and thus having lower methane emissions and higher soil C accumulation than wetlands in tropical climates.

Sha et al. (2011) looked at methane emissions from the same two wetlands in the Altor and Mitsch (2008a) study in addition to an oxbow lake and a bottomland hardwood forest. All of these research sites were in the same area of the research park. For the two wetlands with controlled flow, wetland 1 had a mean methane flux of 13.5 mg CH$_4$-C m$^{-2}$ hr$^{-1}$, and wetland 2 had a mean methane flux of 21.5 mg CH$_4$-C m$^{-1}$ hr$^{-1}$ during the growing season. Both of these wetlands had significantly lower methane fluxes in the non-growing season. The open water sites in both wetlands had higher methane emissions than the transitional zones. Wetland 1, which was planted with wetland plants when established, had a methane flux of 68 g CH$_4$-C m$^{-2}$ yr$^{-1}$. Wetland 2, which was left to naturally colonize with plants, had a methane flux of 114 g CH$_4$-C m$^{-2}$ yr$^{-1}$. The higher methane flux in wetland 2 was thought to be because wetland 2 was more productive, as far as primary production, than wetland 1. Therefore, a greater C sequestration in wetland 2 may be the reason why it produced 68% more methane than wetland 1. The oxbow site was only inundated during the wet season and had a mean methane flux of 0.01 mg CH$_4$-C m$^{-2}$ hr$^{-1}$. This rate did not differ much between the growing and non-growing season, and there was no significant difference for methane emission rates between the inundated area and the transitional zones. This low average methane emission may be due to the fact that it is only inundated for part of the year, and the water it did receive in spring was cool water, thus slowing microbial methanogenesis. Overall, the oxbow lake had a methane flux of 0.3 g CH$_4$-C m$^{-2}$ yr$^{-1}$, and the forested riverside had a methane flux of 379 g CH$_4$-C m$^{-2}$ yr$^{-1}$. Soil temperatures correlated well with methane emissions in each of the four areas studied.
The third GHG that wetlands can emit is nitrous oxide. Nitrous oxide, being 310 times more potent than carbon dioxide, is an important component in the GHG effect (Solomon et al., 2007). Therefore, even though nitrous oxide has a lower atmospheric concentration than carbon dioxide, it can still be a significant contributor to global climate change (Omonode et al., 2011). In addition to warming the atmosphere, nitrous oxide can also cause ozone depletion (Ravishankara et al., 2009). Currently, nitrous oxide is increasing by 0.26% each year. About 70% of the anthropogenic and natural nitrous oxide sources come from the soil and soil related emissions (Sovik et al., 2006). Most of the anthropogenic nitrous oxide comes from the increased microbial release of nitrous oxide in both terrestrial and aquatic systems compared to what the natural background levels were before the systems were altered (Beaulieu et al., 2010). These microbial releases include both nitrification and denitrification. Like methane and carbon dioxide, nitrous oxide emissions for wetlands are mostly influenced by the system’s hydrology (Altor and Mitsch, 2008a; Altor and Mitsch, 2008b). Greater periods of standing water will lead to greater reducing conditions in the wetland compared to flashy hydrology with periods of dry down (Altor and Mitsch, 2008a). Flashy hydrology may vary biogeochemical processes depending on the extent of the periods of dry down. This kind of wet/dry hydrology may also increase primary productivity by allowing various plant species to establish, which may increase the amount of C stored thus offsetting wetland GHG emissions (Altor and Mitsch, 2008b). However, flashy hydrology leads to a variety of results, and needs to be looked into further in order to make better predictions.

Beaulieu et al. (2010) studied nitrous oxide emissions from a 153 km long pool of the Ohio River near Cincinnati, Ohio. The Ohio River watershed was 508,202 km², and 4% of this area was in agriculture and urban development. Both ammonium and nitrate concentrations did not vary significantly throughout the year. The ammonium concentration was about 50 µg L⁻¹ and the nitrate concentration was around 0.82 mg L⁻¹. The highest dissolved nitrous oxide concentrations were during
the summer months, and the lowest were in the winter. Temperature accounted for 70% of this seasonal change. Nitrous oxide emissions emitted from this pool were positively correlated with the dissolved nitrous oxide concentration. Like the nitrous oxide concentrations, nitrous oxide emissions also varied by season, with temperature explaining 36% of the variation. The saturation ratio of nitrous oxide increased from 1.6 to 7.4 when going from the upstream to the pool’s dissolved nitrous concentration. Due to this saturation increase, the nitrous oxide emissions also increased from upstream to the pool with emission rates of 16.3 and 623 µg-N₂O-N m⁻² hr⁻¹ respectively. Both the nitrous oxide concentrations and emissions were claimed to be controlled by nitrification rather than denitrification. However, further research is needed to be conclusive.

Sovik et al. (2006) examined GHG emissions for an open water wetland in Finland. This wetland had both an open water portion and a shallow, vegetated portion. The deepest part of the wetland was 1 to 2 meters deep. The nitrous oxide fluxes were higher in the areas that had plants vs the open water sections of the wetlands. Summer nitrous oxide fluxes were around 0.40±0.25 mg N₂O-N d⁻¹, while winter had lower fluxes around 0.09±0.017 mg N₂O-N d⁻¹. The lower winter nitrous oxide flux could be due to cooler temperatures and thus a slowdown of microbial processes like denitrification and nitrification. The total nitrous oxide flux was 1.6% of the total N load. The summer methane flux was lower than the winter flux: 29±6.4 and 46±12 mg CH₄-C d⁻¹. However, the summer carbon dioxide flux was greater than the winter flux: 1200±420 and 210±75 mg CO₂-C d⁻¹.

Previous Embarras Wetland Studies

Kovacic et al. (2000) measured how well three wetlands receiving tile drainage in the Upper Embarras Watershed removed N and P. When wetlands A and D were built, they were excavated with cuts varying in dimensions, from 3.0 to 3.7 m wide and 0.4 to 0.9 m deep. The soil from the excavation of these cuts were used to make the wetlands’ berms. Each wetland had an inlet and an outlet with weir structures, pressure transducers, and data loggers for measuring flow. Water entering the wetlands was
from the adjacent agricultural field's tile drainage system. Wetlands A, B, and D had drainage areas of 15, 5, and 25 ha respectively. The total amount of inlet flow for wetlands A, B, and D from 1995 to 1997 was 144,000, 47,700, and 245,600 m$^3$ respectively. Roughly 38% of the nitrate was removed from the three wetlands during this three year study, or 333 kg NO$_3$-N ha$^{-1}$ yr$^{-1}$. For nitrate, eight out of the total nine wetland water years had lower nitrate concentrations going out than coming in. During these eight years, the removal efficiencies for the wetlands ranged from 11 to 37%. For total N, wetlands A, B, and D had removal efficiencies of 40, 44, and 31% respectively. Together, all three wetlands removed 1697 kg N, roughly 37%, of the total N. For total P, the removal efficiencies were between 64 and 80%. One wetland water year had a greater amount of P leave the wetland than what entered. This could have been due to not taking into consideration the potential for surface runoff under high precipitation events. Also, the majority of the P came into the wetlands in the winter and early spring when the vegetation was still dormant. Therefore, little of this P could have been taken up by wetland vegetation.

Larson et al. (2000) studied the same three wetlands that were in Kovacic et al. (2000), but focused on the seepage of water from each wetland to the Embarras River. For his study, 10 wells were installed for wetland A and 5 wells for wetland D. For wetland A, there were three N to S transects of wells: wetland, berm, and riparian. The wetland transect consisted of three wells just inside wetland A on the side closest to the Embarras River. These wells were spaced out so that one well was closest to the inlet, one was in the middle, and one was closest to the outlet. The berm wells were installed on the outside of wetland A, across the berm from each wetland well. The riparian wells were installed roughly 3 m from the Embarras River, in line with each wetland and berm well. There was also a field duplicate well installed in the riparian well transect. Together, the wells made three transects perpendicular to wetland A’s berm, thus providing sampling locations for seepage water. Since wetland B was thought to mimic wetland A in seepage, no wells were installed for estimating seepage in wetland B. Rather, the seepage estimate from A was scaled down to estimate wetland B’s seepage. Wetland D only had 4
wetland wells and one riparian well to estimate seepage water and seepage N removal. When calculating the N removal via seepage, the authors took the average nitrate concentration in the berm wells for the nitrate load into the riparian buffer strip, and the highest nitrate concentration in the riparian wells. This was done to provide a conservative estimate for seepage N removal. The wetland A berm wells had a nitrate concentration range of <0.1 to 8.5 mg N L\(^{-1}\), while the riparian wells ranged from <0.1 to 8.0 mg N L\(^{-1}\). For each sampling day, the average nitrate concentration for the riparian wells was never higher than the average for the wetland wells. Roughly 61.3 kg of NO\(_3\)-N seeped out of wetland A in 1997. The wetland A complex, with the seepage berm included, removed between 47 and 60% of the nitrate-N in 1997. The amount of nitrate removed ultimately depended on which well’s nitrate concentration was used for the estimation. The 47% removal rate was considered the conservative estimate. The wetland wells for wetland D had nitrate concentrations ranging from 0.3 to 3.8 mg L\(^{-1}\), while the riparian well for wetland D had a nitrate-N concentration range from 0.2 to 4.6 mg N L\(^{-1}\). Wetland D’s total nitrate-N removal with its seepage strip varied between 28 and 34%, with 28% being the conservative estimate.

Xue et al. (1999) studied the same Embarras River wetlands, using the acetylene blockage and N-15 tracer methods to quantify denitrification rates. Both methods required PVC rings to be installed in the wetland soil and incubated over an amount of time. The results indicated that the in situ denitrification rates were between 2.0 and 11.8 mg N m\(^{-2}\) hr\(^{-1}\). The higher rate was observed in June with a higher sediment temperature, 25\(^{\circ}\)C, and background nitrate concentration of 10.5 mg N L\(^{-1}\). The lower rate was observed in February. The authors also found that the other nitrate removal mechanisms were plant and SOM, about 6 to 10%, and seepage, 30%.

Hoagland et al. (2002) studied plant and algae composition, biomass, and nutrient removal in wetland B of the Embarras River wetlands described previously. The authors collected above and below ground biomass five times throughout the sampling year. The total biomass ranged from 12,000 to
30,000 kg ha\(^{-1}\). The greatest amount of biomass was collected in September, and the majority of the biomass collected, between 54 and 77%, was from below ground. The total biomass N values reached a maximum, 370 kg N ha\(^{-1}\), in August, whereas the total biomass P reached a maximum, 57 kg P ha\(^{-1}\), in July. Both the N and P maximum were followed by a decline in biomass nutrient composition. Below ground N and P contents reached their maximum, 250 kg N ha\(^{-1}\) and 36 kg P ha\(^{-1}\), in August. A total of 18 plant species were collected from all five sampling dates. However, only four of these species were found at each sampling period. The authors did point to plants contributing to N and P removal, however, the majority of the inlet flow, and thus N and P load, occurred in late winter and early spring when there was little to no plant growth. Further, some of the N and P taken up by the plants may have come from soil organic matter mineralization. Therefore, it was difficult to estimate how much of the N and P load was taken up by the wetland plants. Algae on the other hand must take up all of their nutrients from the water column. The total amount of N uptake by algae was estimated to be roughly 10 kg ha\(^{-1}\). This low N removal rate was due to only two sampling dates that had algae present. The authors predicted that wetlands with longer periods of standing water would have a larger N removal rate via algal production. The total amount of P in algae was equal to the inlet minus the outlet total P load. This, however, could have been due to algae leaving the wetland outlet as organic P, and some of the algae dying and being incorporated into the wetland soil when the wetlands dried up. Finally, the authors concluded that wetland plants and algae may be both a sink and a source of N and P for the system as a whole, especially if the plants and algae are not harvested.
MATERIALS AND METHODS

Site Description

The wetlands were located in the Embarras River Watershed in east-central Illinois, where about 60% of the annual precipitation occurs in early spring to late summer, between March and August (Hu et al., 2007). This watershed is dominated by row crop agriculture. David et al. (1997a) estimated that 91% of this watershed was planted with corn and soybean. According to Hu et al. (2007), the majority of land in the Embarras River Watershed is flat with <1% slope. Therefore, most fields have subterranean tiles to drain the shallow groundwater from their fields in order to maximize their yield potential. In fact, between 75 and 80% of the land in agriculture in the Embarras Watershed was estimated to be drained with tiles (David et al., 1997a). This percentage was expected to have increased since the 1990s since farmers are going from random to pattern drainage. Pattern drainage consisted of parallel pipes, laterals, which varied between 7 to 20 m apart, and then hooked up to a larger main pipe that eventually branched to drain into the nearest river or drainage ditch. These pipes are perforated and are usually between 1 to 1.5 m deep. The Embarras wetlands in this study obtained their water from the tiles of both random and patterned drained land.

The six Embarras wetlands (A-F) are located 32 km south of Champaign, IL and about 5 km north of the Champaign-Douglas County line (Kovacic et al., 2000). These wetlands were built in 1994 on Colo series soil. Colo is a fine-silty, mixed, superactive, mesic Cumulic Endoaquoll. Before the wetlands were constructed, the land was a pasture that supported native wet prairie plants (David et al., 1997b).

When the wetlands were constructed, wetlands A, D, and E had excavated trenches dug into them. The soil from this excavation was used to build a berm around all six wetlands in order to retain tile water (Kovacic et al., 2000; Larson et al., 2000). Wetlands B, C, and F were constructed without disturbing the original soil profile. According to Kovacic et al. (2000), these berms were built 15.3 m from the Embarras River in order to keep with USDA protocol. Each berm was built 15 cm at a time and
were compacted with a sheep’s foot roller. Since the six wetlands were built in pairs, wetlands A and B, C and D, and E and F, there was also a berm installed between the pairs to help prevent seepage and over flow between wetlands. When the berms were finished, they were 1.4 m high. Each wetland had an emergency spillway installed in the berm, consisting of an opening that measured 2.4 m wide and 0.3 m deep (Kovacic et al., 2000). These spillways allowed for water to exit the wetland in large flood events in order to prevent damage to the berm or risk backing water up in the field.

The wetlands were not planted, but rather were allowed to vegetate naturally using their soil's seed bank (Kovacic, 2000). The plants that where originally established in the wetlands included ironweed (*Vernonia gigantean*), pigweed (*Amaranthus viridis* L.), yellow nutsedge (*Cyperus esculentus* L.), swamp smartweed (*Polygonum amphibium* L.), lady’s thumb (*Polygonum persicaria* L.), prairie cordgrass (*Spartina pectinata* Link), barnyard grass (*Echinochloa crus-galli* (L.)P. Beauv.), hop sedge (*Carex lupulina* Muhl. Ex Willd.), and reed canary grass (*Phalaris arundinacea* L.). However, when the plants started growing in 2012, the first year of this study, the wetlands were dominated by reed canary grass (*Phalaris arundinacea* L.) that made up nearly all of the biomass.

For this study, only three wetlands, A, B, and D, were sampled (Fig. 1). Wetlands C, E, and F were not measured due to their high susceptibility to overland runoff and complications with being undersized. Overland runoff is difficult to measure, and made constructing tight water and nutrient, budgets difficult. This is especially true for P budgets since particulate P is often elevated in surface runoff events. Also, wetlands A, B, and D were the most intensively studied wetlands upon establishment, and would therefore be the best wetlands to compare through time (Xue et al., 1999; Kovacic et al., 2000; Larson et al., 2000; Hoagland et al., 2001). Dimensions and drainage area for these three wetlands are summarized in Table 1. Wetlands A, B, and D had drainage areas of 15, 5, and 26 ha respectively. This drainage area has increased since the 1990s due to the addition of tile drainage in the surrounding area. When the current drainage areas were calculated, it appeared that wetland A did not
Fig. 1. Aerial view of wetlands A, B, and D with pictures taken of each wetland. The aerial view was done with Google Earth software.
Table 1. Dimensions and drainage area for wetlands A, B, and D.

<table>
<thead>
<tr>
<th>Wetland</th>
<th>Surface area (ha)</th>
<th>Volume (m³)</th>
<th>Tile drainage area 1995-1998 (ha)</th>
<th>Tile drainage area 2012-2013 (ha)</th>
<th>Average Depth (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>0.6</td>
<td>5400</td>
<td>15</td>
<td>15</td>
<td>0.9</td>
</tr>
<tr>
<td>B</td>
<td>0.3</td>
<td>1200</td>
<td>5</td>
<td>9</td>
<td>0.4</td>
</tr>
<tr>
<td>D</td>
<td>0.8</td>
<td>5200</td>
<td>25</td>
<td>70</td>
<td>0.7</td>
</tr>
</tbody>
</table>
have any new tile added to its drainage network. Wetland B’s drainage area increased from 5 to 9 ha since the previous study. Between 1998 and 2012, an additional 45 ha of tile drained land was added to the wetland D drainage area. Also, an additional 30 ha of tile was added to wetland D’s system in August 2013. These last 30 ha of drainage area for wetland D were not reported in Table 1 because they did not affect the majority of the 2013 water year’s budget. All water and nutrient budgets were converted into standardized units in order to compare wetlands of various sizes and drainage areas.

**Wetland Water Samples**

Each of the three wetland inlets had a water table control structure (an inline structure from Agri Drain) equipped with a pressure transducer and a data logger. This system allowed for continuous measurement (30 minute intervals) of wetland inlet flow from the tiles. Similar to the inlets, each wetland had an outlet (Agri Drain structure) with a pressure transducer and data logger for wetland outlet flow measurements.

The inlet and outlet structures also provided a location to take water samples, which were collected during or right after major rain events. For the larger rain events when the temperature was above freezing, an automatic water sampler (ISCO) was used to take water samples at regular time intervals in order to capture the fluctuating nutrient concentrations at the rise and fall of the hydrograph. For the extended periods of this study without a major rain event, inlet samples were taken once a week until the water stopped flowing over the v-notch weir in the control structures. With each grab sample taken, the height over the v-notch weir was taken with a meter stick in order to help calibrate and check the accuracy of the pressure transducer. Grab samples were taken from the outlet structures around large precipitation events and whenever the inlet samples were taken until the wetland water level fell below the outlet v-notch weir. Water samples collected from both the inlet and outlet structures were taken in 500 mL white Nalgene bottles that were not acid washed.
All samples were transported back to the lab, immediately filtered through a 0.45 µm filter, and separated into the appropriate aliquots. For samples that may have had a high amount of suspended particles, a GF/C glass fiber filter was used as a preliminary filtration before using a 0.45 µm filter. Each water sample was separated into three aliquots. The first aliquot was analyzed for nitrate, and ammonium, the second for total N and total P, and the third for DRP. The aliquot for nitrate and ammonium was placed in the freezer until analysis. The sample was then run on a Dionex DX 120 Ion Chromatograph for nitrate and a Quik Chem FIA+8000 Series Lachat for ammonium following standard methods (APHA, 1998). Each run for nitrate and ammonium was done with standards, duplicates, spikes, and external quality controls. The second aliquot measured was for total N and total P. This sample was not filtered and was preserved with sulfuric acid until the solution reached a pH of below 2. Once acidified, the sample was refrigerated until it was processed on the Lachat. The Lachat was run similarly to the ammonium samples with the exception of the standards, external quality standards, and manifold used. This aliquot was only run for a particular wetland when the wetland had both inlet and outlet flow. The third aliquot was prepared and analyzed for DRP. This aliquot was filtered through a 0.45 µm filter and refrigerated in a Nalgene until it was processed. Dissolved reactive P was determined colorimetrically using the Lachat with appropriate standards, duplicates, spikes, and external quality control standards.

Seepage Well Water Samples

Seepage well design, spatial arrangement, and installment followed the guidelines set out in Larson et al. (2000). For the seepage zone between wetland A and the Embarras River, three transects were set up that consisted of at least three wells each. The first transect, consisting of wells 1 through 3, were the closest to the inlet. The third transect, made up of wells 7 through 9, were closest to the outlet. Therefore, wells 4 through 6 were in the middle transect, transect 2. Each transect had a well just inside the wetland, wells 1, 4, and 7, a well just outside of the berm, wells 2, 5, and 8, and a well 3.0 m
from the Embarras River, wells 3, 6, and 9. These wells were called wetland, berm, and riparian wells in this study. The average well screen depths for the wetland, berm, and riparian wells were 1.0, 1.6, and 2.3 m respectively. Most of the wells, all except 1, 2, and 6, were from the original seepage study laid out in Larson et al. (2000). Well 6 was installed as a pair of wells, 6N and 6S, to serve as a field duplicate. Wells 3 and 5 were located near a tree. Therefore, new wells, wells 3 and 5N, were installed to see how the rhizosphere influences nitrate removal in seepage water from wetland A to the Embarras River.

The wells in wetland B followed the design, spatial arrangement, and installment guidelines from wetland D in Larson et al. (2000). Wetland B had three wells just inside the wetland, wetland wells, and one well about halfway between the berm and the Embarras River, the riparian well. The wetland wells were 1 meter deep and the buffer strip well was 2 meters deep. These wells were not monitored in the 2012 water year.

All of the wells that were installed were made out of 5.08 cm diameter PVC pipe. A 15 cm long screen with 0.1 cm slits was cut into these PVC pipes 5.08 cm from the bottom. An 8 cm diameter soil auger was used to dig a hole large enough for these wells. The holes were backfilled with pea-sized gravel, bentonite, and the soil that was removed by the auger respectively. The pea-sized gravel kept the screen open and prevented soil from clogging the screen and the wells’ interior. The bentonite prevented water from moving vertically from the soil surface to the well and thus diluting the water samples. Finally, the hole was backfilled with soil to insure that the well was stable and the screen was closed off from the surrounding environment.

For each sampling date, the water height in the wells was measured using a water depth measurer (Solinst). Then, the wells were pumped out with an electrical peristaltic pump (Solinst). In order to collect water samples that were not affected by the stagnant microcosm of the well, the wells were pumped to dryness or for three times its volume, whichever was first. An hour was used as an appropriate amount of time to pass for water to reenter the wells in order to collect a sample. The
water sample was collected by a hand pump and transferred to a 125 mL white Nalgene bottle for transportation to the lab. From these samples, one aliquot was taken to measure nitrate, chloride, sulfate, and ammonium. This aliquot was treated the same way that the wetland water samples for the same analytes were handled.

**Water and Nutrient Budgets**

The major source and loss of water for the wetlands was through the wetlands’ inlets and outlets respectively. The pressure transducer and data logger recorded the height of water in the control structure. This height, in millivolts, was converted to centimeters and plugged into the equations for the manual control structures (Agri Drain) determined in Chun and Cooke (2008). These equations determined the flow in each inlet and outlet structure in liters per second. During periods of time when the Embarras River was out of its banks and flooded the wetlands, accurate outlet flow could not be determined. Therefore, outlet flow was assumed to equal inlet flow. Another important source of water was precipitation. This value was estimated using daily observational data from the National Oceanic and Atmospheric Administration’s, NOAA’s, climate station in Philo, IL. This station was approximately 6 miles from the wetland site, and was considered to be affected by the same precipitation events. Evapotranspiration was estimated similar to Kovacic et al. (2000), using the Illinois State Water Survey daily evapotranspiration potential and scaling these values up to each wetland. Both evapotranspiration and precipitation were measured only from when the inlet started flowing until when it stopped. Groundwater was considered minimal to nonexistent in terms of a source of water for each wetland due to a confining layer under the wetlands (Kovacic et al., 2000).

Seepage was estimated from wetland A for both water years of the study and from wetland B for the 2013 water year following Larson et al. (2000). This included using Darcy’s Law in an abbreviated form: \( V = K \times A \times i \). \( V \) was the total amount of seepage water, \( K \) was the apparent hydraulic conductivity, \( A \) was the total effective seepage area, and \( i \) was the hydraulic gradient. The apparent hydraulic
conductivity was determined separately for wetland A and B by analyzing periods of time with little to no precipitation, subtracting outlet from inlet flow adjusted for evapotranspiration for these time periods, and solving the equation above by adjusting the K value until it equaled the amount of water missing. The effective seepage area for wetland A for this study was the same as reported in Larson et al. (2000), 445 m². Wetland B’s effective seepage area was determined by measuring the length of the berm, 140 m, and multiplying it by the depth of soil to the impermeable layer after this depth was adjusted for the additional permeability of the soil due to standing water, 2.41 m. The hydraulic gradient for both wetlands A and B was determined using the difference of water elevation between the wetlands and the Embarras River, and dividing this difference by the average distance between the wetland berm and the river, 18.3 m. The river had a pressure transducer that continuously monitored the water’s height. Both wetlands’ water elevations were measured using the outlet pressure transducers during periods of outlet flow, and by the wetland wells during periods of inundation without outlet flow. The river’s pressure transducer, both wetlands’ outlet pressure transducers, and the wells were all surveyed with laser leveling equipment to get accurate elevation differences. Since wetland B did not have wells installed in the 2012 water year, the seepage data from wetland A was proportionally scaled down to wetland B in order to estimate its seepage rate. Wetland D’s seepage rate was estimated by determining the difference between the inlet and outlet flow, adjusted for evapotranspiration, for time periods without precipitation. This was done for 7 time periods and an average seepage rate was determined by assuming that the water missing equaled the amount of seepage water. This seepage rate was used to estimate wetland D’s seepage throughout the year. All three wetlands were flooded by the Embarras River two times in the 2013 water year. Seepage was assumed to be zero for the time period leading up to, during, and after the flooding.

Once the water budget was constructed, N and P budgets were made. Each N and P concentration determined was imported, along with the flow data, to SAS 9.2 (SAS Institute, 2008). SAS
9.2 was used to linear interpolate between sampling dates to find missing concentrations in order to get a yearly budget. Both dry and wet atmospheric N depositions, determined by using the data from the Bondville IL Environmental and Atmospheric Research Site, were added to the inlet load to accurately account for N from the atmosphere. Like evapotranspiration and precipitation, wet and dry deposition were measured from when the inlet started flowing until it stopped.

An N seepage budget was determined. This was done following Larson et al. (2000). The amount of seepage water determined using Darcy’s Law was the amount of water entering and leaving the seepage berm. This volume of water was multiplied by the average berm well N concentration to get the N load coming into the seepage berm. The volume of seepage water was then be multiplied again by the average N concentration of the riparian wells to determine the N load leaving the seepage berm. The difference in these two loads was considered the total amount of N removed through seepage. In addition, the N load that seeped out of the wetlands was taken into consideration when determining the amount of N removed by the wetland.

**Greenhouse Gas Samples**

Both wetlands A and B were monitored for GHGs during this experiment. The basic procedure followed the GRACEnet protocol for chamber sampling (Blowes et al., 2003). The analytes included methane, carbon dioxide, and nitrous oxide. Each wetland was equipped with terrestrial static rings, and was also sampled by floating chambers when there were inundated areas. Wetland A had four transects of static rings for the 2012 water year and two transects for the 2013 water year. These consisted of transects coming from both of the largest ponds in the wetland that were consistently inundated, even under low flows, until wetland A dried up completely. Wetland B had two transects of rings, one near the inlet and one near the outlet. Each ring in the transect was located approximately 4 ft from the preceding one. There were 5 rings, rings A through E, in each wetland transect. The rings that were sampled at each sampling date depended on the water level and which rings were inundated. This
method provided a gradient of moist to dry soil, thus capturing the wetlands’ dynamic nature. The seepage berm for wetland A also had two transects consisting of two rings each to determine the GHG flux from the seepage berm. All static rings were made of PVC pipe with an inner diameter of roughly 8 in. PVC pipe end caps outfitted with weather stripping, septa, and ventilation tubes were made to fit on the green PVC rings. Each ring was incubated for 30 minutes, and 15 mL samples were taken at times 0, 10, 20, and 30 with a syringe. The samples were placed into an evacuated glass vial with a grey butyl septum. Along with these samples, a LI-COR model LI-8100 was used to obtain a carbon dioxide flux from each terrestrial ring.

Any area of the wetland that was inundated by water with a depth of 10 cm or more was sampled for GHGs via floating chambers. These chambers were made from plastic tubs and were outfitted with septa, a handle, and a piece of foam for buoyancy. The floating chambers were also spray painted a light silver color to help reflect the sunlight and to avoid absorbing heat. Like the terrestrial chambers, the floating chambers were incubated for 30 minutes and 15 mL samples were taken at times 0, 10, 20, and 30 with a syringe and placed into a glass vial with a grey butyl septum. For each inundated area, three floating chambers were used to obtain an average flux of GHGs. Also, when the wetland was entirely inundated and had outlet flow, the floating chambers were solely used to obtain gas fluxes. Floating gas samples were not collected from wetland B during the 2012 water year due to the severe drought and little to no standing water inside of the wetland. Also, carbon dioxide was not measured in the floating chamber samples for the 2012 water year in wetland A. This was due to technical problems with the gas chromatograph.

Once the gas samples were collected, they were transported back to the lab and run on a GC-2014 gas chromatograph by Shimadzu. The gas chromatograph was used to determine both nitrous oxide and methane. The GC was also used to check the accuracy of the LI-COR’s carbon dioxide flux reading and to obtain a floating CO₂ flux in 2013. Along with running the samples, the GC also ran a set
of calibration standards. The standard curve created from the standards was used to determine the concentration of methane, nitrous oxide, and potentially carbon dioxide in each sample. These concentrations were scaled up from individual rings and floating chambers to the full area of the wetland by using the GPS location of each ring to estimate the portion of the wetland that was inundated. The average floating chamber flux for each gas measured was scaled up to the inundated area calculated, and the average terrestrial gas flux was used to scale up to the portion of the wetland that was not covered with water. Once a mass per day was calculated for each portion of the wetland, the inundated and terrestrial masses were summed and divided by the area of the wetland to give the final flux for the wetland that day. Linear interpolation was used to determine the fluxes for the days between field measurements. Floating chambers measurements were made relatively soon after a precipitation event to determine how these events affected GHG fluxes. All GHG fluxes were converted to CO$_2$-equivalent units in order to standardize each gas’s influence on the atmosphere.

Dissolved gas samples for methane and nitrous oxide were taken in 300 mL Wheaton BOD bottles. These samples were taken from each area where the floating chambers were used. Each sample was collected in a standing pool of water with a beaker attached to a telescoping pole. In order to take the sample, the beaker was slowly submersed so as to not agitate the sample and cause a false flux of gases into or out of the water sample. The water collected was then slowly poured into the Wheaton BOD bottle and appropriately capped off without any air pockets. Once collected, these samples were stored in the refrigerator until further processing.

The procedure for analyzing the dissolved gases was done with great care in order to avoid either gases degassing from or dissolving into the sample. Sample vials (20mL) were evacuated first and labeled with the appropriate sample identification. Each sample was done in triplicate to check the accuracy and precision of the method. Once the vials were prepared, samples were processed in groups of six in a small cooler in order to avoid warming the sample and affecting its gas solubility. Each vial
received 10 mL of sample and 15 mL of ultrapure helium. Once each vial had a sample, they were placed on a shaker table for 30 minutes on high. This was considered enough time to allow the gases in the sample to come into equilibrium between the liquid and gas phases in the vial. Once done shaking, the samples were allowed to equilibrate for an hour before being analyzed. Standards were made, and the samples and standards were run on a GC. The dissolved gas concentrations were plotted against the flux obtained in order to determine if there was a relationship between these values. Carbon dioxide could not be measured on the GC that was used with the dissolved samples, and therefore the carbon dioxide fluxes from the floating chambers were not correlated with a dissolved carbon dioxide concentration. Surface water and soil temperature, soil moisture, and water nitrate concentrations were also collected to determine if they affect greenhouse gas emissions.
RESULTS

Wetland Hydrology

The study area had a drought during the majority of the 2012 water year. The total amount of precipitation that occurred in 2012 was 644 mm. From this total, 97 mm of precipitation fell before February 1\textsuperscript{st}, and 288 mm fell after August 1\textsuperscript{st} when the topsoil and subsoil were under drought conditions. In fact, the tropical storm that resulted from Hurricane Isaac brought 77 mm of precipitation in two days on September 1\textsuperscript{st} and 2\textsuperscript{nd}, and helped bring central Illinois out of the drought. The 2013 water year was a more typical year with 981 mm of precipitation. There were two major storm events that occurred during this year. The first one occurred from April 16\textsuperscript{th} to April 19\textsuperscript{th}, with 119 mm of rain in four days. The second event was from June 23\textsuperscript{rd} to June 26\textsuperscript{th}, and was smaller than the April event with only 66 mm of precipitation.

The wetland inlets and outlets had a minimal amount of flow in the 2012 water year (Table 2). Due to the lack of outlet flow, most of the water that entered the wetlands seeped out. Wetlands A and B had 13,087 and 3,926 m\textsuperscript{3} of seepage water, respectively, in 2012. The differences in the water budgets for wetland A and B after the outputs (outlet, seepage, and evapotranspiration) were subtracted from the inputs (inlet and precipitation), were -200 and -966 m\textsuperscript{3} respectively. These negative values came from the large precipitation event on February 1\textsuperscript{st} that caused a large, unmeasured surface runoff event.

The 2013 water year had a greater amount of inlet flow for both wetlands A and B (Table 2). Wetland D was also monitored during this time, and had the largest recorded inlet flow volume for the Embarras wetlands, 248,450 m\textsuperscript{3}. Outlet flow and seepage estimates also increased with increasing inlet flow. The differences in the water budgets for wetlands A, B, and D were calculated to be 3,273, 1,124, and 46,266 m\textsuperscript{3} respectively. These differences indicated that some of the water that entered the wetlands exited through their unmeasured emergency spillways. This spillway flow occurred during the large precipitation events in April and June.
Table 2. Water budget for wetlands A, B, and D for all study years.

<table>
<thead>
<tr>
<th></th>
<th>Inlet</th>
<th>Precipitation</th>
<th>Outlet</th>
<th>Seepage</th>
<th>ET</th>
<th>Difference†</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>m³</td>
<td>m³</td>
<td>m³</td>
<td>m³</td>
<td>m³</td>
<td>m³</td>
</tr>
<tr>
<td>1995</td>
<td>30,500</td>
<td>3,500</td>
<td>14,100</td>
<td>19,700</td>
<td>5,000</td>
<td>-4,700</td>
</tr>
<tr>
<td>1996</td>
<td>59,200</td>
<td>4,200</td>
<td>37,800</td>
<td>20,300</td>
<td>4,400</td>
<td>800</td>
</tr>
<tr>
<td>Wetland A</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1997</td>
<td>54,300</td>
<td>4,900</td>
<td>27,700</td>
<td>23,600</td>
<td>5,500</td>
<td>2,500</td>
</tr>
<tr>
<td>1998</td>
<td>N/C‡</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2012</td>
<td>14,025</td>
<td>1,999</td>
<td>11</td>
<td>13,087</td>
<td>3,126</td>
<td>-200</td>
</tr>
<tr>
<td>2013</td>
<td>55,404</td>
<td>3,696</td>
<td>33,108</td>
<td>20,118</td>
<td>2,601</td>
<td>3,273</td>
</tr>
<tr>
<td>1995</td>
<td>13,000</td>
<td>1,600</td>
<td>7,200</td>
<td>9,600</td>
<td>2,100</td>
<td>-4,300</td>
</tr>
<tr>
<td>1996</td>
<td>19,300</td>
<td>2,400</td>
<td>10,000</td>
<td>8,600</td>
<td>2,800</td>
<td>200</td>
</tr>
<tr>
<td>Wetland B</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1997</td>
<td>15,400</td>
<td>2,800</td>
<td>7,200</td>
<td>7,800</td>
<td>3,100</td>
<td>200</td>
</tr>
<tr>
<td>1998</td>
<td>26,282</td>
<td>3,030</td>
<td>17,070</td>
<td>12,300</td>
<td>3,090</td>
<td>-3148</td>
</tr>
<tr>
<td>2012</td>
<td>3,400</td>
<td>954</td>
<td>0</td>
<td>3,926</td>
<td>1,394</td>
<td>-966</td>
</tr>
<tr>
<td>2013</td>
<td>35,200</td>
<td>2,032</td>
<td>13,892</td>
<td>20,658</td>
<td>1,558</td>
<td>1,124</td>
</tr>
<tr>
<td>1995</td>
<td>58,400</td>
<td>3,700</td>
<td>39,900</td>
<td>15,300</td>
<td>5,500</td>
<td>1,300</td>
</tr>
<tr>
<td>1996</td>
<td>106,000</td>
<td>4,800</td>
<td>99,000</td>
<td>16,000</td>
<td>5,200</td>
<td>-9,000</td>
</tr>
<tr>
<td>Wetland D</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1997</td>
<td>81,200</td>
<td>6,100</td>
<td>76,000</td>
<td>20,700</td>
<td>7,000</td>
<td>-16,300</td>
</tr>
<tr>
<td>1998</td>
<td>N/C‡</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2012</td>
<td>N/C‡</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2013</td>
<td>248,450</td>
<td>6,309</td>
<td>130,215</td>
<td>71,410</td>
<td>6,868</td>
<td>46,266</td>
</tr>
</tbody>
</table>

† Negative differences come from surface runoff that was not quantified in the water budget. Positive values indicated a portion of the inlet flow exited through the wetland’s emergency spillway.
‡ Data not collected.
The tiles that flowed into wetland A and B ran between 149 and 176 days in the 2012 and 2013 water years (Table 3). Wetland D’s inlet flowed for a longer period of time, 275 days, in 2013. In fact, wetland D’s inlet did not stop flowing once it started on December 20th 2012. Wetland A and B’s inlet also took less time in 2012 to reach the 75% total hydraulic loading volume than the same inlets in 2013. Finally, both wetlands A and B had longer retention times in 2012 than in 2013. Wetlands B and D had the same retention time for 2013, 6 days, which also happened to be the shortest retention time for the Embarras wetlands.

Wetland Nitrogen Budgets

The Embarras wetlands were effective at removing N with inlet loads being larger than the corresponding outlet loads for nitrate and total N (Table 4). Nitrate made up the majority of the total N load for all wetlands in all study years. Ammonium and organic N were measured during this study, and had removal rates 18 and 100% and between -107 and 98% respectively. However, since these N compounds made up a small percentage of the total N inlet load, between 0.1 and 0.9% for ammonium and between 0 and 2.2% for organic N, these removal rates did not significantly influence total N removal rates. Rather, the factors that affected the amount of nitrate removal from these wetlands also influenced total N removal. When comparing the combined nitrate removal rates for wetlands A and B between 2012 and 2013, it was obvious that the wetlands had an overall higher percent nitrate removal rate (86% in 2012) than 2013, when the removal % was 45. However, when the combined wetland nitrate removal rate for the 2012 and 2013 water years were compared on a mass removal basis, wetlands A and B removed 326 kg N ha\(^{-1}\) in 2012, compared to 1571 kg N ha\(^{-1}\) in 2013. Therefore, the larger percent nitrate removal in 2012 corresponded to a lower mass removal when compared with the 2013 water year. Wetland D had the largest nitrate removal rate for this study in the 2013 water year. The roughly 1000 kg of nitrate N removed was a little over half of the total inlet nitrate load.
Table 3. Hydrological data for wetlands A, B, and D for all years studied.

<table>
<thead>
<tr>
<th></th>
<th>Number of flow days</th>
<th>Days to 75% of hydrologic loading†</th>
<th>Retention time‡</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetland A</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1995</td>
<td>254</td>
<td>50</td>
<td>35</td>
</tr>
<tr>
<td>1996</td>
<td>175</td>
<td>51</td>
<td>8</td>
</tr>
<tr>
<td>1997</td>
<td>207</td>
<td>54</td>
<td>19</td>
</tr>
<tr>
<td>1998 N/C§</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2012</td>
<td>175</td>
<td>29</td>
<td>58</td>
</tr>
<tr>
<td>2013</td>
<td>154</td>
<td>108</td>
<td>13</td>
</tr>
<tr>
<td>Wetland B</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1995</td>
<td>272</td>
<td>58</td>
<td>17</td>
</tr>
<tr>
<td>1996</td>
<td>252</td>
<td>79</td>
<td>14</td>
</tr>
<tr>
<td>1997</td>
<td>172</td>
<td>66</td>
<td>11</td>
</tr>
<tr>
<td>1998</td>
<td>320</td>
<td>231</td>
<td>12</td>
</tr>
<tr>
<td>2012</td>
<td>149</td>
<td>87</td>
<td>34</td>
</tr>
<tr>
<td>2013</td>
<td>176</td>
<td>121</td>
<td>6</td>
</tr>
<tr>
<td>Wetland D</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1995</td>
<td>314</td>
<td>35</td>
<td>26</td>
</tr>
<tr>
<td>1996</td>
<td>299</td>
<td>45</td>
<td>13</td>
</tr>
<tr>
<td>1997</td>
<td>228</td>
<td>44</td>
<td>11</td>
</tr>
<tr>
<td>1998 N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2012 N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2013</td>
<td>275</td>
<td>164</td>
<td>6</td>
</tr>
</tbody>
</table>

† The number of days it took to get to 75% of the total inlet flow.
‡ Retention time = (wetland volume/(sum of inlet flow, precipitation, and surface runoff)) multiplied by number of days of tile flow.
§ Data not collected.
Table 4. Yearly inlet and outlet N loads along with mass removed per ha of wetland and percent removal for wetlands A, B, and D.

<table>
<thead>
<tr>
<th></th>
<th>NO$_3^-$N</th>
<th></th>
<th></th>
<th></th>
<th>NO$_3^-$N</th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>In (kg N)</td>
<td>Out (kg N)</td>
<td>Seepage Out (kg N)</td>
<td>Mass Removed (kg N)</td>
<td>Removed (kg N ha$^{-1}$)</td>
<td>In (kg N)</td>
<td>Out (kg N)</td>
<td>Seepage Out (kg N)</td>
</tr>
<tr>
<td>Wetland A</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1995</td>
<td>374</td>
<td>203</td>
<td>50</td>
<td>121</td>
<td>202 (32)$\ddagger$</td>
<td>384</td>
<td>209</td>
<td>50</td>
</tr>
<tr>
<td>1996</td>
<td>861</td>
<td>463</td>
<td>107</td>
<td>291</td>
<td>485 (34)</td>
<td>868</td>
<td>467</td>
<td>107</td>
</tr>
<tr>
<td>1997</td>
<td>635</td>
<td>225</td>
<td>61</td>
<td>349</td>
<td>582 (55)</td>
<td>783</td>
<td>309</td>
<td>61</td>
</tr>
<tr>
<td>1998</td>
<td>N/C§</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2012</td>
<td>147</td>
<td>0.12</td>
<td>5</td>
<td>142</td>
<td>236 (96)</td>
<td>152</td>
<td>0.13</td>
<td>5</td>
</tr>
<tr>
<td>2013</td>
<td>619</td>
<td>246</td>
<td>38</td>
<td>335</td>
<td>558 (54)</td>
<td>642</td>
<td>278</td>
<td>38</td>
</tr>
<tr>
<td>Wetland B</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1995</td>
<td>105</td>
<td>49</td>
<td>17</td>
<td>39</td>
<td>130 (37)</td>
<td>109</td>
<td>51</td>
<td>17</td>
</tr>
<tr>
<td>1996</td>
<td>238</td>
<td>85</td>
<td>43</td>
<td>110</td>
<td>367 (46)</td>
<td>241</td>
<td>88</td>
<td>43</td>
</tr>
<tr>
<td>1997</td>
<td>149</td>
<td>47</td>
<td>30</td>
<td>72</td>
<td>240 (48)</td>
<td>156</td>
<td>50</td>
<td>30</td>
</tr>
<tr>
<td>1998</td>
<td>214</td>
<td>72</td>
<td>N/C</td>
<td>142</td>
<td>473 (66)</td>
<td>220</td>
<td>77</td>
<td>N/C</td>
</tr>
<tr>
<td>2012</td>
<td>50</td>
<td>0</td>
<td>23</td>
<td>27</td>
<td>90 (54)</td>
<td>52</td>
<td>0</td>
<td>23</td>
</tr>
<tr>
<td>2013</td>
<td>544</td>
<td>114</td>
<td>126</td>
<td>304</td>
<td>1013 (56)</td>
<td>555</td>
<td>127</td>
<td>126</td>
</tr>
<tr>
<td>Wetland D</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1995</td>
<td>506</td>
<td>303</td>
<td>24</td>
<td>179</td>
<td>224 (35)</td>
<td>520</td>
<td>313</td>
<td>24</td>
</tr>
<tr>
<td>1996</td>
<td>999</td>
<td>610</td>
<td>43</td>
<td>346</td>
<td>433 (35)</td>
<td>1011</td>
<td>634</td>
<td>43</td>
</tr>
<tr>
<td>1997</td>
<td>578</td>
<td>363</td>
<td>25</td>
<td>190</td>
<td>238 (33)</td>
<td>618</td>
<td>424</td>
<td>25</td>
</tr>
<tr>
<td>1998</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2012</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2013</td>
<td>1975</td>
<td>640</td>
<td>334</td>
<td>1001</td>
<td>1251 (51)</td>
<td>2038</td>
<td>712</td>
<td>334</td>
</tr>
</tbody>
</table>

$\dagger$ Includes N from dry and wet atmospheric deposition, ammonium, and organic N.

$\ddagger$ Values in parentheses indicate percent removal.

§ Data not collected.
**Wetland Phosphorus Budgets**

Wetland P retention rates from the current study varied greatly by water year and wetland (Table 5). The 2012 water year had nearly 100% P retention due to the lack of outlet flow. The 2013 water year P retention was not as large. Wetland B had the lowest total P retention rate of -7%, or -1 kg P ha\(^{-1}\), whereas wetland A had the greatest total P retention rate of 44%, or 8.7 kg P ha\(^{-1}\). This total P retention rate for wetland A was also the largest seen in the Embarras wetland study.

**Wetland Hydrology and Nutrient Load Relationships**

Daily inlet nitrate loads corresponded closely with daily inlet flow (Kovacic et al., 2000). This meant that large nitrate loads occurred when there was a large inlet flow event. The same was true for this study. Therefore, only the precipitation and nitrate loads for both the wetlands’ inlet and outlet were plotted over time to represent the wetlands’ hydrographs (Fig. 2). The total P loads also followed a similar trend. The main difference between the nitrate and total P peaks was that the total P peaks were shorter lived, with the total P loads returning to baseline values quicker than the nitrate loads. The inlets had seasonal flow patterns, with most of the inlet flow occurring in winter and spring (Fig. 2). The average percent of total inlet flow that occurred in winter and spring for wetlands A, B, and D was 90, 92, and 93% respectively. The smallest amount of flow occurred in fall for each wetland, whereas the summer usually had the second smallest volume of inlet flow. Tiles usually stopped in summer due to the increased evapotranspiration in the corn and soybean fields. The inlet for Wetland D did not stop during the 2013 water year due to the increased drainage area added to the system.

Water year 2012 started on December 30\(^{th}\), 2011 due to the installation date for the Agri Drain structures (Fig. 2). There were two 21 mm precipitation events in January 2012 that caused the wetlands to fill. There was only 11 m\(^3\) of outlet flow for wetland A during this early precipitation event. Wetland B never had outlet flow during the 2012 water year due to the lack of precipitation. The January precipitation event caused wetland B to fill, but water never reached the bottom of the v-notch weir of
Table 5. Yearly inlet and outlet P loads along with percent P removed and mass P removed per ha of wetland for wetlands A, B, and D.

<table>
<thead>
<tr>
<th>Wetland A</th>
<th>DRP</th>
<th>Organic-P</th>
<th>Total P</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>In</td>
<td>Out</td>
<td>Mass Removed</td>
</tr>
<tr>
<td>1995</td>
<td>5.4</td>
<td>4.3</td>
<td>1.1</td>
</tr>
<tr>
<td>1996</td>
<td>9.9</td>
<td>8</td>
<td>1.9</td>
</tr>
<tr>
<td>1997</td>
<td>15</td>
<td>8</td>
<td>7.2</td>
</tr>
<tr>
<td>1998</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2012</td>
<td>0.3</td>
<td>0.002</td>
<td>0.3</td>
</tr>
<tr>
<td>2013</td>
<td>5.7</td>
<td>4.2</td>
<td>1.5</td>
</tr>
<tr>
<td>Wetland B</td>
<td>DRP</td>
<td>Organic-P</td>
<td>Total P</td>
</tr>
<tr>
<td>1995</td>
<td>1.5</td>
<td>0.2</td>
<td>1.3</td>
</tr>
<tr>
<td>1996</td>
<td>2.4</td>
<td>2.3</td>
<td>0.1</td>
</tr>
<tr>
<td>1997</td>
<td>2.1</td>
<td>1.3</td>
<td>0.8</td>
</tr>
<tr>
<td>1998</td>
<td>3.06</td>
<td>1.9</td>
<td>1.6</td>
</tr>
<tr>
<td>2012</td>
<td>0.05</td>
<td>0</td>
<td>0.1</td>
</tr>
<tr>
<td>2013</td>
<td>1.6</td>
<td>2.3</td>
<td>-0.7</td>
</tr>
<tr>
<td>Wetland D</td>
<td>DRP</td>
<td>Organic-P</td>
<td>Total P</td>
</tr>
<tr>
<td>1995</td>
<td>5</td>
<td>3</td>
<td>2</td>
</tr>
<tr>
<td>1996</td>
<td>11</td>
<td>8</td>
<td>3</td>
</tr>
<tr>
<td>1997</td>
<td>11</td>
<td>14</td>
<td>-3</td>
</tr>
<tr>
<td>1998</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2012</td>
<td>N/C</td>
<td>N/C</td>
<td>N/C</td>
</tr>
<tr>
<td>2013</td>
<td>5.7</td>
<td>4.9</td>
<td>0.8</td>
</tr>
</tbody>
</table>

† Values in parentheses indicate percent removal.
‡ Negative values indicate that more P was exported from the wetlands than came into them.
§ These values could not be determined due to the divisibility of numbers by zero.
¶ Data not collected.
Fig. 2. The 2012 and 2013 precipitation and nitrate loads for all three wetlands' inlets and outlet.
the outlet structure. There was a large precipitation event on September 1st and 2nd due to Hurricane Isaac that reached central Illinois as a tropical storm. This event caused 77 mm of precipitation, but was not enough to start the tiles that fed the wetlands due to the low antecedent moisture conditions from the extremely dry summer. However, this event was enough to bring the soil moisture levels to near normal conditions for the 2013 water year.

Wetland D, due to its larger drainage area, was the first wetland to have inlet flow in the 2013 water year starting on January 9th 2013. Wetland A and B followed, and had a short lived flow event starting on January 13th after a 23.1 mm precipitation event. The start to the inlet flow for both wetlands A and B began on January 30th. This was when the tiles had constant flow, supplying the wetlands with inlet water. The first real precipitation and inlet event occurred from February 26th through the 28th. This event brought 40.4 mm of precipitation, and was enough to cause both a spike in inlet and outlet nitrate load hydrographs for all three wetlands. The other main precipitation events occurred on and around April 18th, June 1st, and June 26th. These events had precipitation totals of 119, 42.7, and 66 mm respectively.

The main difference between the nitrate load hydrographs for wetland D and wetlands A and B was that wetland D had an extended flow period after each precipitation event. This meant that the hydrograph was less pulsed, and had more of a gradual decline to its baseflow. Wetland D also had a much greater volume for its baseflow than wetlands A and B. Both of these contrasting characteristics can be seen when comparing the nitrate load hydrographs. Wetlands A and B had narrow peaks of inlet nitrate load after each event (Fig. 2). Wetland D, on the other hand, had a broader inlet nitrate load shown in Fig. 2 as a thicker and elevated base nitrate load. Outlet nitrate load peaks corresponded well with inlet flow pulses. For the precipitation events that caused the Embarras River to flood the wetlands, April 18th and June 26th, outlet loads were equal to inlet loads because the river and the wetlands were thought to be one system. This allowed for conservative estimates for nutrient removal during these
days as well. Finally, these two flood events also were enough to slow the tile flow entering the wetlands due to the river applying back pressure and preventing the tile from reaching its maximum flow potential. This was especially true for wetland D. Back pressure was the reason why the nitrate load peaks in wetland D had a flat top during high precipitation events instead of a sharp maximum peak.

Total wetland hydraulic loading was thought to be a good predictor for N and P removal/retention (Fig. 3). This figure has all points for each year sampled. Both the wetlands’ nitrate and total P removal rates, when expressed as mass per area, had a positive relationship with the wetlands’ hydraulic loading. However, only the nitrate removal relationship with hydraulic loading had a strong relationship explaining 73% of the variation. On the other hand, there was an overall weak negative relationship between nitrate and total P removal, expressed as a percentage, and the wetlands’ hydraulic loading.

Along with total wetland hydraulic loading, nitrate and total P flow weighted means were determined (Fig. 4). The nitrate inlet flow weighted mean for wetland A was greater than wetland B’s nitrate flow weighted mean in 1995, 1996, and 1997. However, in 2012 and 2013, wetland B’s nitrate inlet flow weighted means were greater than the means for wetland A. The 2012 and 2013 wetland B flow nitrate weighted means, 14.8 and 15.5 mg N L\(^{-1}\), were also the largest for this study. Wetland D had a smaller nitrate inlet flow weighted mean than both wetland A and B for all years analyzed, except in 1995 when wetland D had a slightly larger nitrate flow weighted mean than wetland B. The total P flow weighted means for the wetlands varied per water year, but wetland A consistently had the largest total P flow weighted mean while wetland D consistently had the smallest.

Since both hydraulic loading and nutrient flow weighted means affect the total nutrient load into a wetland, and ultimately the amount of nutrient removal, multiple linear regression was used to with hydraulic loading rate and nutrient flow weighted means to predict the amount of that nutrient removed. After the multiple regression was performed, all of the relationships in Fig. 3 were improved,
Fig. 3. Wetlands A, B, and D nitrate and total P removal rates, in both mass removal per ha yr and percent removal, plotted against hydraulic loading for all years sampled.
Fig. 4. Nitrate (Top) and total P (Bottom) inlet flow weighted means for wetlands A, B, and D for all years analyzed.
but none of them, besides the nitrate mass per area removal correlation with hydraulic loading had a strong relationship. The $R^2$ was 0.73 when just considering hydraulic loading, and increased to 0.85 when considering both hydraulic loading and the nitrate concentrations. This multiple linear regression had the formula: \( \text{Nitrate Removal (kg N ha}^{-1}\text{ yr}^{-1}) = -428.61 + 44.323X_{\text{Hydro}} + 43.176X_{\text{FWM}} \) with both variables being significant ($p_{\text{Hydro}} < 0.0001$, $p_{\text{FWM}} = 0.01$).

**Seepage**

Not all of the nitrate that exited the wetlands directly entered the adjacent Embarras River. The seepage buffer strip allowed for nitrate-rich water to slowly seep from the wetlands to the Embarras River, providing a chance for denitrification and plant uptake to occur. Nitrate concentrations in wells decreased as water seeped from the wetlands to the river (Fig. 5). Since the average nitrate concentration in the berm wells was greater than the average for the riparian wells, it can be assumed that nitrate removal did occur in both wetland A and B’s buffer strips. Further, wetland A’s berm wells had greater nitrate concentrations closer to the wetland’s inlet than the outlet (Fig. 6). The same pattern was observed for wetland A’s riparian wells.

The berm and riparian well concentrations were multiplied by the volume of seepage water to determine the nitrate load into and out of the seepage buffer strip, which was ultimately used to calculate the removal rate for each seepage buffer strip (Table 6). The seepage nitrate removal rates for all three wetlands during the 1995 and 1996 water years were estimated assuming that the entire nitrate load that seeped out of the wetland was removed through the buffer strip. This was probably an over estimation, and the 1995 and 1996 seepage nitrate removal rates were probably artificially high. However, it should be noted that wetland B’s buffer strip had a greater nitrate removal rate than wetland A’s buffer strip for both the 2012 and 2013 water years. Wetland D’s seepage removal was not estimated because a riparian well was never installed in this buffer strip. Therefore, a seepage nitrate removal rate could not be calculated.
Fig. 5. The average nitrate concentrations for both wetland A and B’s berm (the wells closest to the wetlands) and riparian (the wells furthest from the wetlands and right next to the Embarras River) wells during both water years. These box-and-whisker plots contain the average nitrate concentrations obtained from the respective well locations. Average values from each sampling date were placed into these figures. These graphs therefore give the overall average, as well as extreme values, for each seepage well location of the entire seepage nitrate dataset.
Fig. 6. Wetland A berm and riparian wells’ nitrate concentration for both 2012 and 2013. This box-and-whisker plot separated out all three of wetland A’s berm wells and riparian wells in order to observe the nitrate reduction as water seeped out of the wetland to the Embarras River, from the inlet to the outlet. Each box and whisker plot contained all nitrate concentrations from all sampling dates. The results show that not only does the water nitrate concentration decrease as water seeps from the wetland to the river, but that both the berm and riparian wells have lower nitrate concentrations going from wetland A’s inlet to its outlet.
Table 6. Seepage nitrate removal rates for wetlands A, B, and D during all study years.

<table>
<thead>
<tr>
<th></th>
<th>Wetland A</th>
<th>Wetland B</th>
<th>Wetland D</th>
</tr>
</thead>
<tbody>
<tr>
<td>1995</td>
<td>132</td>
<td>78</td>
<td>81</td>
</tr>
<tr>
<td>1996</td>
<td>283</td>
<td>198</td>
<td>145</td>
</tr>
<tr>
<td>1997</td>
<td>74</td>
<td>87</td>
<td>17</td>
</tr>
<tr>
<td>2012</td>
<td>11</td>
<td>46</td>
<td>N/D†</td>
</tr>
<tr>
<td>2013</td>
<td>61</td>
<td>276</td>
<td>N/D</td>
</tr>
</tbody>
</table>

† Rate was not determined.
Both inundated and terrestrial fluxes of carbon dioxide, methane, and nitrous oxide were sampled from wetlands A and B during this study. The years described in the GHG portion of this study do not represent water years, but rather the sampling season for each wetland. Both wetlands A and B were sampled from March 22nd to October 29th in 2012 and from March 3rd through November 20th in 2013. Carbon dioxide was the dominant GHG for both wetlands in both study years (Table 7). The majority of the total carbon dioxide flux came from the terrestrial portions of the wetlands. The same was true for the methane and nitrous oxide fluxes. The inundated portions of the wetland only made up 14% of wetland A’s total cumulative flux and 7.7% of wetland B’s total cumulative flux during the 2013 sample year. Wetland B’s total GHG flux, 27,085 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$, was roughly half of wetland B’s total flux in 2013 and of wetland A’s total flux in both 2012 and 2013.

The large percentage of the total GHG flux that the terrestrial portions produced can be seen graphically by looking at the cumulative GHG flux over time (Fig. 7). For each GHG, the total cumulative flux curve over time took on the same shape as the cumulative terrestrial flux. There was also little space between the two lines, indicating that the GHG flux from the inundated portions of the wetland made up a small portion of each GHG’s total cumulative flux. The methane and nitrous oxide total and terrestrial cumulative fluxes increased at a greater rate during the final dry down events for the wetlands. This final dry down period was marked off by the orange lines in Fig. 7. The cumulative carbon dioxide flux started to increase at a greater rate after the final wetland dry down.

By combining the cumulative terrestrial and inundated fluxes for each GHG, it became even more apparent that carbon dioxide did make up the majority of the total GHG flux from wetlands A and B for both sampling years (Fig. 8). This trend was observed throughout both sample years. Also, the increased terrestrial methane and nitrous oxide emission rates during final wetland dry down was observed when analyzing the fluxes on the actual sampling dates. The daily methane flux even
Table 7. Cumulative fluxes of carbon dioxide, methane, and nitrous oxide for wetlands A and B during the 2012 and 2013 water years.

<table>
<thead>
<tr>
<th>Wetland</th>
<th>Year</th>
<th>Inundated</th>
<th>Terrestrial</th>
<th>Inundated</th>
<th>Terrestrial</th>
<th>Inundated</th>
<th>Terrestrial</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetland A</td>
<td>2012</td>
<td>N/C†</td>
<td>41,505 (90)‡</td>
<td>147 (0.3)</td>
<td>189 (0.4)</td>
<td>16 (0.03)</td>
<td>4,030 (8.8)</td>
<td>45,887</td>
</tr>
<tr>
<td></td>
<td>2013</td>
<td>3,720 (9)</td>
<td>26,865 (66)</td>
<td>1,050 (3)</td>
<td>3,696 (9)</td>
<td>930 (2)</td>
<td>4,340 (11)</td>
<td>40,601</td>
</tr>
<tr>
<td>Wetland B</td>
<td>2012</td>
<td>N/C</td>
<td>26,072 (96)</td>
<td>N/C</td>
<td>21 (0.08)</td>
<td>N/C</td>
<td>992 (3.7)</td>
<td>27,085</td>
</tr>
<tr>
<td></td>
<td>2013</td>
<td>2,430 (5.3)</td>
<td>37,607 (82)</td>
<td>273 (0.6)</td>
<td>777 (1.7)</td>
<td>802 (1.8)</td>
<td>3,751 (8)</td>
<td>45,640</td>
</tr>
</tbody>
</table>

† Data was not collected.
‡ Values in parentheses represent the percent of the total cumulative flux in carbon dioxide equivalents.
Fig. 7. Carbon dioxide, methane, and nitrous oxide cumulative fluxes for wetlands A and B, displayed over time while separating the terrestrial source of each gas from the total cumulative flux. The orange lines outline the final dry down of each wetland for both water years. The first line in each water year represents the last day of outlet flow, and the second line represents when the wetlands are completely dry.
Fig. 8. Wetland A and B’s cumulative carbon dioxide, methane, nitrous oxide, and total fluxes (Top), as well as the individual fluxes for each sampling day (Bottom), all expressed in carbon dioxide equivalent units for the 2012 and 2013 sample years. The orange lines outline the final dry down of each wetland for both water years. The first line in each water year represents the last day of outlet flow, and the second line represents when the wetlands are completely dry.
surpassed the carbon dioxide flux during one wetland A sampling event in July of 2013. This date was during the final dry down event.

*Wetland Greenhouse Gas Relationships with Abiotic Controls*

Since GHG emissions are driven by microbial processes, temperature was thought to be an important controlling variable. For methane and nitrous oxide fluxes, both water and soil temperature were important in inundated and terrestrial fluxes respectively (Fig. 9 and Fig.10). Inundated methane and nitrous oxide fluxes had a threshold at 18°C. Any flux that occurred in water temperature below 18°C was limited when compared to fluxes that were emitted in water above 18°C. This same trend was observed for terrestrial methane and nitrous oxide fluxes. However, the temperature threshold for terrestrial fluxes was centered around 15°C. Carbon dioxide did not have the same threshold characteristics.

Similar to the temperature threshold, the terrestrial methane and nitrous oxide fluxes had a soil moisture threshold at ~25% (Fig. 11). Both terrestrial methane and nitrous oxide fluxes stayed at a baseline level until the soil moisture was at 25% or higher. Also similar to the temperature threshold, carbon dioxide did not respond to this soil moisture threshold.

In addition to temperature, the inundated nitrous oxide and methane fluxes were correlated with the water’s nitrate concentration (Fig. 12). The inundated methane flux was negatively related to the amount of nitrate in the water, while the nitrous oxide flux was positively related to the nitrate concentration. Both relationships were weak, but there were clear trends for both.

The final correlation for inundated methane and nitrous oxide fluxes was with the dissolved concentrations of the respective gas (Fig. 13). The inundated methane fluxes were not related to the dissolved methane concentration. Regardless, it was noted that methane fluxes were relatively low when the dissolved methane concentrations were elevated. On the other hand, the larger inundated nitrous oxide fluxes occurred with greater dissolved nitrous oxide concentrations. This relationship was
Fig. 9. Surface water temperature correlations with carbon dioxide (Top), methane (Middle), and nitrous oxide (Bottom) fluxes from inundated chambers for both wetlands A and B. The carbon dioxide, methane, and nitrous oxide data were separated at the 18°C threshold. The two graphs include all inundated fluxes from 2012 and 2013.
Fig. 10. Terrestrial carbon dioxide (Top), methane (Middle), and nitrous oxide (Bottom) fluxes plotted with soil temperature for wetlands A and B in 2012 and 2013.
Fig. 11. Terrestrial carbon dioxide (Top), methane (Middle), and nitrous oxide (Bottom) fluxes plotted with percent soil moisture for wetlands A and B in 2012 and 2013.
Fig. 12. Inundated methane (Top) and nitrous oxide (Bottom) fluxes compared to water nitrate concentrations for both wetlands in 2012 and 2013. The data presented were from all sample locations in wetlands A and B.
Fig. 13. Inundated methane (Top) and nitrous oxide (Bottom) fluxes relations with dissolved methane and nitrous oxide concentrations respectively. The data presented were from all sampling dates in 2012 and 2013 for each sample location in wetlands A and B.
weak, but still observable.

**Seepage Berm Greenhouse Gases**

Similarly to the wetlands’ fluxes, wetland A’s riparian seepage berm total greenhouse gas fluxes were made up primarily of carbon dioxide, 99.4 and 96.7% for 2012 and 2013 respectively (Table 8). The next most important gas was nitrous oxide. Nitrous oxide made up 0.5 and 3 percent of the total greenhouse gas emission in 2012 and 2013 respectively. This meant that 2012 and 2013 methane flux from wetland A’s riparian seepage berm was only 0.03 and 0.2 percent of the total for these years respectively.

The total cumulative flux for wetland A’s seepage berm closely followed its carbon dioxide flux (Fig. 14). The only clear separation between the total and carbon dioxide cumulative flux happened between late June and early July in 2013. There was a relatively large release of nitrous oxide during this time as shown by looking at the average daily terrestrial flux measurements for that time period. This increased nitrous oxide flux was short lived, and soon returned back to baseline levels.
Table 8. Carbon dioxide, methane, nitrous oxide, and total greenhouse gas fluxes for wetland A’s seepage berm in 2012 and 2013.

<table>
<thead>
<tr>
<th></th>
<th>CO₂</th>
<th>CH₄</th>
<th>N₂O</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>2012</td>
<td>50086 (99.4)†</td>
<td>15 (0.03)</td>
<td>265 (0.5)</td>
<td>50366</td>
</tr>
<tr>
<td>2013</td>
<td>45447 (96.7)</td>
<td>85 (0.2)</td>
<td>1478 (3)</td>
<td>47010</td>
</tr>
</tbody>
</table>

† Values in parentheses represent the percent of the total cumulative flux in carbon dioxide equivalents.
Fig. 14. Cumulative carbon dioxide, methane, nitrous oxide, and total flux for wetland A’s riparian seepage berm during 2012 and 2013 (Top). Individual carbon dioxide, methane, nitrous oxide, and total flux during each 2012 and 2013 sampling date for wetland A’s berm (Bottom).
DISCUSSION

Wetland Hydrology

The drought during the 2012 water year greatly limited the amount of water available to both wetlands A and B. In fact, wetland A’s total inlet flow for 2012 was only 30% of the average inlet flow from the 1995-1997 study (Table 2). Wetland B also had limited inlet flow with the total only being 18% of its average flow from the 1995-1998. This was especially significant for wetland B because it was estimated that this wetland’s drainage area increased from 5 to 9 ha since Kovacic et al. (2000) (Table 1). The lower inlet flow also increased the retention time for each wetland. Wetland A and B had retention times of 58 and 34 days respectively (Table 3). These retention times were nearly double that of Kovacic et al. (2000). The 2012 drought also severely reduced the amount of outlet flow for both wetlands. Therefore, during the 2012 water year, the water exited the wetland mostly through seepage and evapotranspiration with seepage making up 93 and 115% of the total inlet flow for wetlands A and B respectively (Table 2). The negative difference values in Table 2 represent the water that entered the wetland as unmeasured surface flow. This water most likely entered the wetland during the large precipitation event at the end of January 2012. This event occurred during the winter when there was little vegetative growth and the soil had elevated antecedent moisture. Therefore, it was likely that some of the precipitation did not infiltrate into the soil, leading to surface runoff.

The 2013 water year precipitation total was closer to the average from Kovacic et al. (2000). The average inlet flow for wetland A for the 1995 to 1997 water year was 48,000 m³, whereas wetland A had an inlet flow of 55,404 m³ (Table 2). Both wetland B and D had a much larger inlet hydraulic load for the 2013 water year when compared to their average hydraulic load from 1995 through 1998. This was likely due to the increase in drainage area for each wetland (Table 1). The retention times for all three wetlands were reduced in the 2013 water year due to the higher inlet flow. Wetlands A, B and D had retention times of 13, 6, and 6 days respectively (Table 3). The 2013 retention times for Wetlands B and
D were the lowest for all years studied due to the increased drainage area and corresponding inlet flow. All three wetland water budget differences in Table 2 were positive, indicating unmeasured water leaving through the wetlands’ emergency spillways, although most of differences are small in magnitude.

*Wetland Nitrogen Removal*

Due to the low hydraulic loading of 2012, the wetlands also had a limited amount of N loading, which limited the mass N per area removed that water year. Conversely, the reduced outlet flow in 2012 allowed for elevated percent removal of nitrate for that water year. Since the 2013 water year was more typical, the wetlands did not experience the same inlet N limitation. Rather, wetlands B and D saw the greatest N removal rate during the 2013 water year, and had more typical percent removal rates. Overall, the wetlands performed as well as they did when they were first created (Fig. 3). The current study’s removal rates fell on the same regression line as removal data collected in Kovacic et al. (2000). The only exception was wetland B.

Wetland B removed a greater amount of N than what its hydraulic load predicted using linear regression (Fig. 3). A large portion of the total hydraulic loading for this wetland most likely came during the large precipitation events. However, the inlet for wetland B quickly tapered off after the events. Wetland B’s outlet was usually the first outlet to stop flowing after large events. This hydrological pattern, combined with the largest inlet N flow weighted mean, helped wetland B attain the large N removal rate with a lower hydraulic loading rate (Fig. 4).

On the other hand, wetland D had one of the smallest average inlet N concentrations in 2013 (Fig. 4). However, since this wetland had such a large inlet hydraulic load, it was never N limited, and ultimately had the largest N removal rate (Fig. 3). Both wetlands B and D had their largest N removal rates despite a short hydraulic retention time of 6 days (Table 4). This was counterintuitive because many studies have found that increasing a wetland’s retention time also increased its N removal
efficiency. However, both wetland B and D’s high N removal rates can be explained by analyzing the number of flow days for the 2013 water year. Wetland B’s inlet delivered water for 176 days, which was the third lowest in the study for this wetland including the 2012 drought year. Wetland D’s inlet flowed for 275 days, which was the second lowest in this wetland’s studied history. The large hydraulic loading for both wetlands, combined with the low number of flow days, indicate that a few large flow events may have caused the hydraulic residency time to decrease (Fig. 2). The largest of these flow events occurred on April 18th 2013. This event produced an inlet flow of over 3500 and 3300 m$^3$ d$^{-1}$ for wetlands B and D respectively. Wetland A had an even larger inlet flow of roughly 6700 m$^3$ d$^{-1}$. Even though wetland D had a greater drainage area, it did not have a greater inlet flow per day for the April 18th event. This was due to the low landscape position of wetland D’s inlet, and the back pressure applied to the inlet flow from the water inside of the wetland. However, wetland D’s high flow extended for 8 days due to this back up. With this prolonged elevated inlet flow, wetland D did receive a greater amount of water than wetlands A and B. Also, since the nitrate load closely followed the inlet hydraulic loading, the 8 days after April 18th also produced large nitrate loads for wetland D (Fig. 2). The hydraulic retention time for both wetlands B and D would be greater if large inlet events, like the one that occurred on April 18th, were taken out of the retention time equation. This increased hydraulic residency time for base flow conditions during the majority of the water year probably helped wetlands B and D remove a large amount of N.

The Embarras wetland systems had additional nitrate removal through their seepage buffer strips. Wetland B did have a larger seepage N load, and removal rate, than wetland A despite of its shorter berm and lower effective seepage area (Table 6). This was probably due to the larger average N concentration in the wetland B’s berm wells, 5 mg N L$^{-1}$, compared to wetland A’s berm wells, 1.5 mg N L$^{-1}$ (Fig. 5). This larger nitrate concentration came from two things: wetland B’s larger average inlet N concentration and wetland B being half the size of wetland A, thus limiting the amount of N removal
before the nitrate seeped out of the wetland. When wetland A and B’s N budgets were combined, their seepage buffer strips removed 7% of the total inlet N load for the 2012 and 2013 water years. This percent N removal is additive to the percent wetland N removal. Therefore, the total wetland N removal of 52% for all three wetlands in 2012 and 2013 can be increased to 59% after considering seepage N removal. This value relies on the assumption that wetland D’s seepage buffer strip also removed about 7% of its nitrate load. Overall, these wetlands had an average 44% N removal rate on a mass basis when combining the data from Kovacic et al. (2000) with the current data.

Other studies had similar nitrate removal rates to this study when plotted against hydraulic loading and average inlet nitrate concentrations (Fig. 15). The R² values for these relationships ranged from 0.27 to 0.62. The largest explanation of nitrate removal once again came from the mass of nitrate removed per area relationship with hydraulic loading. It should be noted that the Mitsch et al. (2005) data was removed from this regression due to the nitrate removal rates being artificially low for their corresponding large hydraulic loads. This was probably due to this study’s low average inlet nitrate concentration that limited denitrification. The wetlands in Mitsch et al. (2005) received water that was pumped from an adjacent river. Rivers, due to dilution, have a lower nitrate concentration than the tile drainage water that empties into them. Despite this potential N limitation, there was a clear positive trend between nitrate removal on a mass per area basis with hydraulic loading and a clear negative trend between nitrate percent removal with hydraulic loading.

These results have implications for future wetland managers. If the goal of constructed wetland installation is to reduce nitrate loss by a certain percentage, then the managers should be careful to not overload the wetlands with too much water. This involves planning for potential future drainage area additions similar to what happened to wetlands B and D in this study. On the other hand, the increased N removal in mass per area with increasing hydraulic loading may affect the way wetlands are thought of economically. The price per kilogram of N removed could potentially be reduced for a certain wetland
Fig. 15. Nitrate removal in both mass per area (Top) and percent of inlet (Bottom) related to wetland hydraulic loading (Left) and the inlet flow weighted nitrate concentration (Right) for the current Embarras Wetland study and other agricultural constructed wetlands.
with a larger mass per area removal rate, which means keeping the hydraulic load as large as possible.

**Wetland Phosphorus Retention**

Wetland P retention is often much more variable than N removal. This is largely because the P cycle does not have a gaseous phase. Rather, wetlands have to rely on sedimentation and sorption processes to remove P from inlet waters. Therefore, during any given year, the wetlands run a risk of having once deposited P flushed out due to large inlet events. Despite this risk, wetland A has always had positive DRP retention values. However, the amount of DRP retained varied greatly (Table 5). Kovacic et al. (2000) had DRP retention rates of between 1.8 and 12 kg P ha\(^{-1}\) for wetland A. The DRP retention values for wetland A during the 2012 and 2013 water year were 0.5 and 2.5 kg P ha\(^{-1}\), values much lower than the maximum from the Kovacic et al. (2000) study. Wetland B did not have the same DRP removal range as wetland A, and the 2013 DRP retention rate was negative. The average wetland B DRP retention rate for the current study, -1.1 kg P ha\(^{-1}\), was much lower than this wetland’s average from 1995 through 1998, 2.8 kg P ha\(^{-1}\). Wetland D was able to retain 1 kg P ha\(^{-1}\) in 2013, which was roughly the same as the average from 1995 through 1997, 0.8 kg P ha\(^{-1}\).

Similar to DRP, organic P retention can be variable. The inlet organic P loads from Kovacic et al. (2000) were all zero since it was believed that little organic P came from tile drainage water (Table 5). Due to this assumption, all organic P retention rates in kg P per ha were negative. Organic P was measured from the inlet water samples in the current study. The inlet organic P loads were all greater than the outlet loads, which led to positive organic P retention rates in both 2012 and 2013 for all three wetlands.

The total P retention rates in Table 5 combine both the DRP and organic P retention abilities for the wetlands. The average total P retention from Kovacic et al. (2000) was 1.1 kg P ha\(^{-1}\)yr\(^{-1}\), whereas the average total P retention for the current study was 3.4 kg P ha\(^{-1}\)yr\(^{-1}\). By comparing the two averages, the wetlands were able to retain more total P as they age. However, as stated previously, P retention varies
greatly and trends cannot be accurately observed with a few years of wetland P retention data. This is especially true since the current study’s wetland years had such variable hydrology. The overall, long term total P retention average for these wetlands was 13%. This included the data values from Kovacic et al. (2000) and the current study. Since the long term average percent P retention is positive, it can be assumed that the Embarras Wetlands are P sinks.

Another factor that complicated measuring wetland P retention was the amount of surface runoff and river water that entered the wetlands unmeasured in any given water year. Both surface runoff from the adjacent fields and the river water had the potential to bring a fair amount of particulate P into each wetland. When this particulate P settled to the anaerobic sediments of the inundated portions of the wetlands, it had the potential to become desorbed, and therefore leave the wetland. If this happened, the wetland P retention rate would be artificially reduced due to the inability of measuring surface runoff and river water P loads. The river could have also exported some of the P that was retained in the wetlands’ sediment. This export of P was not, and could not, be accurately measured, but does affect the P retention dynamics of the wetland system. Therefore, due to the complex nature of these wetlands, it was difficult to construct an accurate P budget with the input and output source taken into consideration.

There were no significant trends when the Embarras wetland P retention rate relationships with hydraulic loading and average P concentration were overlain with other agricultural wetlands’ P retention relationships (Fig. 16). The largest $R^2$, 0.29, was the relationship between total P removal on a mass per area basis and hydraulic loading. This was a positive linear regression indicating that larger hydraulic loads provide better conditions for increased P retention. This may be from a larger amount of particulate-bound P entering the wetlands in large precipitation events. If this particulate P entered the wetlands during the large events, it may settle out of solution before exiting the wetlands. This is especially true for P bound to large particulates that settle faster. Conversely, the percent total P
Fig. 16. Total P removal in both mass per area (Top) and percent of inlet (Bottom) related to wetland hydraulic loading (Left) and the inlet flow weighted total P concentration (Right) for the current Embarras Wetland study and other agricultural constructed wetlands.
removal had a weak, negative relationship with hydraulic loading. This could have been due to the re-suspension of once retained P due to large flushing events.

*Wetland Greenhouse Gas Fluxes*

The majority of the total cumulative fluxes for both wetlands A and B in both sample years came from the terrestrial portions of the wetlands. The largest terrestrial methane and nitrous oxide fluxes came during the final dry-down of the wetlands (Fig. 7). During this time, the soils were still saturated with water, and were thought to be anaerobic. In addition, once the water receded, the soil was exposed to a greater amount of solar radiation, thus warming the soil. These greater soil temperatures could have increased microbial activity, thus increasing the amount of methane and nitrous oxide emitted. Water has a high specific heat, and the wetlands needed a great amount of time to warm when they were inundated. This water was especially difficult to heat during periods of elevated inlet flow since colder tile water would pulse through the wetland without being warmed. Therefore, the soils must have been colder when they were inundated, lowering the amount of methane and nitrous oxide emitted when floating chambers were used. The soil temperature was not measured during floating chamber incubations. However, it can be assumed that the soil temperature was colder than the surface water temperature measured and also the terrestrial portion of the wetland at any given time. Both trends of warmer soil temperature and greater soil moisture leading to greater terrestrial nitrous oxide and methane fluxes were observed for this study (Figs. 10 and 11). In fact, soil temperatures of less than 15°C can severely limit the rate of denitrification (Stanford et al., 1975). Another potential reason why the terrestrial nitrous oxide fluxes were greater during the final dry-down could have been the startup of the aerobic process of nitrification. When ammonium gets nitrified to nitrate, some nitrous oxide is given off. This may be the nitrous oxide gas that was measured in this experiment when the wetlands dried down to the point where the soils were no longer fully saturated.

The Inundated fluxes for each greenhouse gas measured made up a small portion of the total
cumulative flux. The largest combined greenhouse gas Inundated flux, which made up 14% of the total year flux, came from wetland A in 2013 (Table 7). Assuming that the 2013 water year was a more typical water year than 2012, wetland A had a greater Inundated flux of methane, 1,050 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$, than nitrous oxide, 930 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$. Conversely, wetland B had a greater Inundated nitrous oxide flux, 802 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$, than Inundated methane flux, 273 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$. This may have been due to the nitrate concentration in the wetlands. Wetland B was smaller, with a larger inlet nitrate flow weighted mean, thus giving plenty of nitrate to denitrifying communities. Wetland A was larger with a smaller inlet nitrate flow weighted mean, thus giving the denitrifiers more of a chance to remove all or most of the nitrate in the water. This was especially true for the inundated location sampled near wetland A’s outlet. If all of the nitrate was removed before the water exited the wetland, the anaerobic microbes had to move on to another terminal electron acceptor. Eventually, the wetlands became reduced enough so that methane producing microbes took. These microbes can reduce carbon dioxide and other carbonaceous material to methane. This shift from inundated nitrous oxide to methane production was further confirmed by the relationship between inundated nitrous oxide and methane fluxes and nitrate concentration in Fig. 12. These relationships suggested that an area with a large methane flux could not also have a large nitrate concentration. Once again, this was because methanogens cannot survive in conditions that provide more energetically favorable electron acceptors.

A concern with installing constructed wetlands to remove nitrate from tile lines is the release of nitrous oxide during denitrification. Of the total nitrate removed by the wetlands, 6.7 and 3.1% was released as nitrous oxide in 2012 and 2013, respectively. Most of the nitrous oxide loss was from the terrestrial portions of the wetlands, which were much larger during the drought year of 2012. Little nitrous oxide was emitted from inundated areas. Therefore, the moist but not fully anaerobic terrestrial portions of the wetlands were where denitrification to N$_2$ was not as complete. Keeping the wetlands as flooded as possible would reduce nitrous oxide losses, by maintaining anaerobic conditions.
Wetland A for both sample years and wetland B for the 2013 sample year had roughly the same total cumulative flux values (Table 12). Wetland B had a lower cumulative flux in 2012. This could have been due to the extremely dry conditions that year. Wetland A had cuts excavated into its bottom when it was constructed. Being lower in the landscape, these cuts usually had a greater moisture content. Therefore, the soil in the excavated areas were not as dry as long in 2012, providing a greater potential for higher greenhouse gas fluxes. Dry soils, like those in wetland B, would have a lower microbial activity, and therefore a lower greenhouse gas flux, than moist soils due to the microbes’ reduced activity under drought conditions.

Seepage Berm Greenhouse Gas Fluxes

Similar to the wetlands’ cumulative GHG flux, wetland A’s berm cumulative GHG flux was made up mostly of carbon dioxide, and little nitrous oxide or methane was produced (Table 14). The majority of the soil profile in the buffer strip for wetland A was saturated for most of the year due to the hydraulic head between wetland A and the Embarras River. However, the surface of the soil was usually not completely saturated. The top few centimeters of soil were probably dominated by aerobic processes, thus limiting the amount of methane and nitrous oxide produced.

Greenhouse Gas Flux Comparisons with Other Studies

Knowing the cumulative GHG flux from the Embarras wetlands is important, but it is also important to know how these fluxes compare to other studies and other environmental conditions. Smith et al. (2013) looked at cumulative nitrous oxide fluxes from corn, Miscanthus x giganteus, switchgrass, and mixed prairie. The cumulative flux from these systems had ranges of 1,656 to 3,751, 292 to 682, 390 to 689, and 195 to 341 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$ respectively. Corn had the largest nitrous oxide fluxes due to the application of N fertilizer. The other plots did not have N fertilizer applied. Regardless, the range of corn nitrous oxide fluxes was comparable to the range of nitrous oxide fluxes found in this
study, 992 to 5,270 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$.

Adviento-Borbe et al. (2007) measured carbon dioxide, methane, and nitrous oxide fluxes from corn fields under various management practices. These greenhouse gases had ranges from 16,390 to 32,340, -161 to -9, and 672 to 4501 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$ respectively. Similar to Smith et al. (2013), these values were all taken from soils under aerobic conditions. This was why methane had a negative flux, and was actually consumed in Adviento-Borbe et al. (2007). On the other hand, the carbon dioxide and nitrous oxide flux ranges were similar to the ones found in this study. The carbon dioxide flux should, theoretically, be larger in aerobic systems since soil respiration rates are greater in aerobic conditions when compared to anaerobic environments. However, the largest carbon dioxide flux from Adviento-Borbe et al. (2007), 32,340 kg CO$_2$ ha$^{-1}$ yr$^{-1}$, was smaller than the largest carbon dioxide flux for the Embarras wetlands, 41,505 kg CO$_2$ ha$^{-1}$ yr$^{-1}$.

Finally, Altor and Mitsch (2006) measured methane fluxes from two constructed wetlands in Ohio under flashy hydrology with dry portions of the water year. The permanently inundated portions of the wetlands in Altor and Mitsch, (2006) had a cumulative flux of 11,760 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$, while the portions that had dry periods had a cumulative flux of 3,528 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$. The permanently inundated portions’ methane flux was well above the maximum methane flux measured in this study, 4,746 kg CO$_2$-e ha$^{-1}$ yr$^{-1}$, which came from wetland A during 2013. This also happened to be the only methane flux from this study that was greater than the methane flux from the portion of the wetlands in Altor and Mitsch (2006) that had flashy hydrology. By comparing these two studies, a major controlling factor on methane production from these wetlands is the amount of time the wetland is inundated.
CONCLUSIONS

Hypoxia in the Gulf of Mexico has environmental scientists evaluating practices to reduce N and P loss from agricultural fields, especially those that occur on tile drained land. There are a variety of in-field and end-of-pipe nutrient remediation practices that are currently recommended to reduce N and P losses, including constructed wetlands.

The wetlands studied here had similar rates of N removal to when they were first established in 1994 (Kovacic et al., 2000). Wetland age did not affect N removal. Rather, increased hydraulic loading in 2013 for wetlands B and D resulted in a large mass of nitrate removed per hectare of wetland. This was most likely due to the continuous supply of inlet N to denitrifying microbial communities. Hydraulic loading explained the majority of the variation involved in predicting the mass of N removed per area of wetland. This relationship was even more robust when the average inlet nitrate concentration was added in a multiple linear regression. The wetland seepage buffer strips removed an additional 7% of wetland inlet N, and increased the total inlet N removal to 59%.

Total P retention was more variable compared to N removal. There was a considerable amount of variability in total P retention among wetlands and water years. The wetlands, especially wetland B, were a source of P during some water years. Hydraulic conditions such as the amount of surface runoff and flooding events most likely complicated P retention estimations. Total P retention, on a mass per wetland area basis, had a weak positive relationship with hydraulic loading. This was possibly from the increased particulate P loading during large hydraulic loading events.

The majority of the GHG flux came from the terrestrial portions of the wetland, not the flooded areas. This was especially true during the final dry down periods of wetlands A and B in both sampling years. Terrestrial GHG fluxes had thresholds at 15°C and at 25% soil moisture content. The drought in 2012 severely limited the GHG budget for wetland B due to the low soil moisture levels. When the total cumulative GHG budget was separated out based on the gas emitted, carbon dioxide contributed the
majority of the total cumulative flux. The inundated nitrous oxide and methane fluxes were also found to be closely related to the water’s nitrate concentration. Nitrous oxide emissions were 6.7 and 3.1% of overall nitrate removal in 2012 and 2013, respectively. Moist soils in the terrestrial portions of the wetlands during 2012 where there was little inundation likely led to the larger percentage in 2012. This would limit how anaerobic the soils were and lead to greater nitrous oxide release.

Finally, there is still a need for more years of data collection from these wetlands, and a need for additional studies that look at constructed wetlands that receive the majority of their water from tile drainage. This is especially true for the northern portion of the Corn Belt. Hopefully continuing this study and potentially monitoring more wetlands can fill in the N removal-hydraulic loading linear regression, especially larger hydraulic loads. Future studies should look at wetlands that have various drainage-to-wetland area ratios. There is also a need to come back to the Embarras wetlands in the future to further assess their lifecycle to aid in budget analyses of nutrient reduction strategies. Finally, there were fairly consistent GHG trends during this study, but more years and constructed wetland GHG studies may be needed to confirm these trends.
REFERENCES


USEPA. 2008. Hypoxia in the northern Gulf of Mexico, an update by the EPA Science Advisory Board. EPA-SAB-08-003. USEPA, Washington, DC.

