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By Margaret McCahon

Entitled Strategically Siting Constructed Wetlands to Target Nitrate Removal in Tile-Drained Agricultural Watersheds

For the degree of Master of Science in Engineering

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STRATEGICALLY SITING CONSTRUCTED WETLANDS TO TARGET NITRATE REMOVAL IN TILE-DRAINED AGRICULTURAL WATERSHEDS

A Thesis

Submitted to the Faculty

of

Purdue University

by

Margaret McCahon

In Partial Fulfillment of the

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of

Master of Science in Engineering

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In loving memory of Grandfather Mack Drake, who loved this land, and in whose footsteps I am now following.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

McCahon, Margaret. M.S.E., Purdue University, August, 2010. Strategically Siting Constructed Wetlands to Target Nitrate Removal in Tile-Drained Agricultural Watersheds. Major Professor: Indrajeet Chaubey.

Wetlands offer a variety of services, such as pollutant removal from point and nonpoint sources, flood attenuation, and habitat for biodiversity. Constructed wetlands can be used in agricultural watersheds to protect surface waters from pollution by agricultural activities. A particular concern for the agricultural Midwest is the high nitrate export from agricultural fields that affects water quality at local and regional scales, including hypoxia in the Gulf of Mexico. Nitrogen export primarily originates in the Upper Mississippi and Ohio River Basins. Tile-drained lands, characteristic of west central Indiana, have greater nitrate losses than un-tiled lands. Constructed wetlands have been proposed as a partial solution to intercept nitrate between agricultural lands and downstream waters.

In order to most efficiently use constructed wetlands to treat nitrate exported from tile-drained lands, these wetlands should be carefully placed in the landscape to intercept high nitrate loads and sized according to their contributing areas. In this thesis a methodology is presented for strategically placing constructed wetlands in the landscape, and this methodology is applied to an 8-digit hydrologic unit (HUC) in west central Indiana. Results showed 19 sites that are suitable for wetland placement, requiring conversion of 0.1% of the entire 8-digit watershed. These wetlands would intercept approximately 3% of nitrate-rich waters from tile-drained lands, removing approximately 1% of all nitrate exported.

To better estimate nitrate removal in these headwater wetlands, the Soil and Water Assessment Tool (SWAT) was applied to a watershed within the 8-digit HUC where three potential wetland sites were found. Simulated wetlands removed nitrate in every month having flow. These three modeled wetlands consistently removed 17-36% of annual incoming nitrate, culminating in a 5% decrease in average nitrate loads at the watershed outlet. If placed strategically, wetlands can efficiently remove nitrate from tile-drained flow, but the extent of their impact in the landscape is dependent on the suitability of local conditions for wetland placement, as well as the specific criteria used for siting wetlands.

CHAPTER 1. INTRODUCTION

Nitrate export from agricultural practices in the Midwest is a widespread problem affecting surface and ground water quality at scales ranging from first order streams to the Mississippi River Basin and the Gulf of Mexico. High nitrate export from these areas has resulted in hypoxia, a condition of low oxygen levels in the water (less than 2 mg/L) in the Gulf of Mexico (Goolsby et al., 1999). The extent of this hypoxic zone is about 15,000 km^2 (Gulf Hypoxia Action Plan 2008). Global climate change is expected to increase hypoxia around the world, because higher temperatures will cause greater stratification of coastal waters and higher net primary productivity combined with greater nutrient loadings (Rabalais et al., 2009). Hypoxia and anoxia cause death of many marine organisms and collapse of aquatic communities (Diaz, 2001).

Hypoxia in the Gulf of Mexico is a result of nutrient (primarily nitrate and phosphorus) loadings from the Mississippi River (Justic et al., 1993). The excess nutrients induce eutrophication resulting in a rapid abundance of aquatic plants. As aquatic plants decay the oxygen is depleted faster than it is replenished due to a stratified water column in the warm summer months. The primary contributors of nitrogen to the Gulf of Mexico are located in the Upper Mississippi River and Ohio River Basins, of which Indiana is a part, due to high nutrient inputs to row crops such as corn (Burkart and James, 1999) and extensively tile-drained land. This region was historically rich with wetlands, due to its relatively flat topography, moist climate, and poorly drained soils. Wetland soils are quite fertile and are ideal for cropping when drained, so most of these original wetlands have been drained by open ditches or underground tile drains (perforated pipes).

However, these drains allow waters concentrated with nitrate to flow rapidly beneath ground, where it is free from uptake by the biologically active upper soil layers. Consequently, flow from tile drains can carry large loads of nitrate to surface waters. Extensive tile drainage in the Upper Mississippi River Basin, combined with the loss of original wetlands, has resulted in significant leaching of nitrate to surface waters. For example, the Upper Mississippi and Ohio River Basins are estimated to contribute a total of 60-90 percent of the total nitrogen load reaching the Gulf (Alexander et al., 2008; Gulf Hypoxia Action Plan, 2008; Goolsby et al., 1999).

The 2008 EPA action plan aims to reduce the hypoxic zone to about one third of its size by 2015 (Gulf Hypoxia Action Plan, 2008). Such a reduction will require significant nutrient load reductions from large contributing areas throughout the Midwest. A recent estimate by the EPA Science Advisory Board (2007) finds that a reduction of 45% of both nitrogen and phosphorus may be necessary to meet the goal by 2015. Yet with climate change and other uncertainties, it is possible that reductions of greater than 45% will be required (Donner and Scavia, 2007). Such a significant reduction of nitrogen loading may be difficult to achieve while maintaining agricultural production and may require a combination of technical knowledge and policy instruments. To remove nitrate most efficiently, a technical solution should consider many possibilities of changing agricultural practices and land use. For instance, Alexander et al. (2000) found that nitrate removal is greater in smaller streams than larger ones; therefore nitrate control can most effectively occur in the headwaters of the landscape. The most efficient solutions are likely to occur at the field or small watershed scale. A number of technical solutions are needed to address site-specific conditions, and may even be used in combination (Robertson and Vitousek, 2009). In order to make a significant difference on the large scale of the Upper Mississippi River Basin, policy instruments are needed to regulate nonpoint source nutrients in the Upper Mississippi River Basin (Rabalais et al., 2002a).

The problem of capturing a high percentage of nitrogen loads originating from agricultural areas before entering receiving waterbodies is quite difficult to solve. Many best management practices (BMPs) have been established to reduce nitrogen loadings from farms. These BMPs include the timing and rate of nitrogen application to agricultural fields, crop rotations and tillage practices, controlled drainage structures, riparian buffers and constructed wetlands (Dinnes et al., 2002). Constructed wetlands show particular promise in acting as a buffer between tile drains and streams. Wetlands slow water flow and store water during high flow events (Bullock and Acreman 2003), process nutrients (Fisher and Acreman, 2004) and provide the substrate for denitrification (Hernandez and Mitsch, 2007) and nitrogen uptake by plants. Wetlands also provide other ecosystem services such as wildlife habitat and recreation. Nitrate removal in agricultural wetlands generally ranges from 23-53% of incoming nitrate (Jordan et al., 2003; Poe et al., 2003; Kovacic et al., 2000; Kovacic et al., 2006). Yet nitrate removal efficiencies are dependent on wetland design. Crumpton (2001) demonstrated that strategically placing wetlands in the landscape and designing them according to their contributing areas may achieve efficient nitrate removal.

1.1. Objectives

The overall goal of this study was to evaluate how constructed wetlands can be used to treat nitrate in tile-drained agricultural lands. The specific objectives were to: 1) Determine suitable wetland sites in an 8-digit watershed in Indiana using GIS methods and wetland siting criteria; 2) Create preliminary wetland designs at each site according to wetland design criteria; and 3) Estimate nitrate removal provided by each design, using a simple regression-based model and a mechanistic approach with the Soil and Water Assessment Tool (SWAT) model.

CHAPTER 2. LITERATURE REVIEW

Nitrogen loading from the agricultural Midwest USA has significant ecological consequences, including hypoxia in the Gulf of Mexico. Nitrate is a dominant form of nitrogen exported from tile-drained lands, which are characteristic of west central Indiana. The discussion in the subsequent sections of this chapter reviews the relationship between nitrate export and tile drainage, outlines a number of best management practices that have been implemented to reduce nitrate exports, and discusses the role of constructed wetlands in intercepting high nitrate flows. A strategy for siting efficient wetlands is presented, along with models for estimating nitrate removal from these wetlands, including the Soil and Water Assessment Tool (SWAT).

2.1. Nitrogen in the Agricultural Midwest

2.1.1. Nitrate and Tile Drainage

Nitrogen application is a critical agricultural input to intensive agricultural systems, and is probably necessary to provide food for the large human population (Robertson and Vitousek, 2009). While hypoxia in the Gulf of Mexico may be the primary environmental concern for the Midwest, there are many other consequences of nitrogen leaching involving the complex nitrogen cycle, such as returning to the atmosphere in the form of nitrous oxide, and increased nitrogen deposition (Robertson and Vitousek, 2009). It may not be possible to eliminate the effects of nitrogen application because this large, widespread nutrient input is not natural or even in equilibrium with the surrounding environment. Yet its

damages must be minimized to address problems like hypoxia in the Gulf of Mexico.

Many factors affect nitrate leaching from agricultural practices. There are generally greater nitrate losses with increasing nitrogen fertilizer application rates (Jaynes et al., 2001). Cropping patterns play a role as well, and nitrate losses are generally higher with row crops and annuals than with perennial crops (Randall et al., 2001). Hydrology also has an impact, and it has been found that precipitation, timing of fertilizer applications, soil moisture, and evapotranspiration rates affect nitrate loadings (Goswami et al., 2009).

Artificial drainage, common throughout the Upper Mississippi River Basin, appears to strongly influence nitrate losses. Historically, this region was rich with wetlands, but most have been drained for agriculture and urbanization. Approximately 37% (20.6 million hectares) of all cropland in the Great Lakes and Cornbelt (Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, Ohio, and Wisconsin) is tile-drained (Fausey et al., 1995). In 1850, wetlands covered approximately 24-31% of the land area in Indiana (McCorvie and Lant, 1993). By the 1980s, an estimated 87% of Indiana's original wetlands have been drained (Mitsch and Gosselink, 2007). About half of Indiana's cropland was drained by 1985 (Pavelis, 1987) through a combination of open ditches and underground tile drains.

Tile drains allow nitrate-rich water to flow rapidly beneath ground, where it is free from uptake by the biologically active upper soil layers. Consequently, flow from tile drains can carry large loads of nitrate into surface waters (Hickey and Doran, 2004). Because the Upper Mississippi River Basin is extensively tile-drained, and the majority of its original wetlands are lost, nitrate leaches readily into surface waters. In heavily tile-drained watersheds, tiles are often the primary path of nitrate loss (David et al., 1997; Drury et al., 1993; Gentry et al., 2009). In

locations with few tile drains, base flow may be the greatest source of nitrate to surface waters (Schilling and Lutz, 2004; Goswami et al., 2009). Kladivko et al. (1991 and 2004) found that drains spaced closer together led to higher flows and consequently greater nitrate losses. In regions with heavily tile-drained land, the resulting nitrate concentration in surface waters has often been found to be above the drinking water standard of 10 mg/L (Drury et al., 1993; Jaynes et al., 2001; Schilling and Lutz, 2004; Kladivko et al., 2004).

2.2. Constructed Wetlands

2.2.1. Best Management Practices (BMPs)

To reduce the negative environmental impacts associated with nitrate losses from agricultural areas, many best management practices (BMPs) have been established. These BMPs include nutrient management plans, such as changing fertilizer application rate and timing, cropping, and tillage practices, as well as off-farm structures, including riparian buffers and constructed wetlands designed to intercept high-nitrate flows before they reach surface waters (Dinnes et al., 2002). These BMPs and other technical solutions may be considered on a site-by-site basis to determine an effective way to minimize nitrogen losses.

Many studies have considered these BMPs and their relevance to the problem of nitrogen losses. Reducing nitrogen application rates have been proposed as one solution for minimizing nitrogen loss (Dinnes et al., 2002; Mitsch et al., 2001; Randall et al., 2001). A farmer may over-apply fertilizers if not all sources of nitrogen are considered. However, fairly high application rates are required to maintain high crop yields. Application timing has also been considered (Dinnes et al., 2002), and application in the spring is usually preferred over fall application (Mitsch et al., 2001) because it reduces fertilizer losses in the winter, when nutrients are not needed as no crops are grown.

Other solutions proposed to reduce nitrogen losses include cropping patterns and tillage practices. Corn-soybean crop rotations have reduced nitrate losses when compared to continuous corn (Dinnes et al., 2002). Similarly, cover crops in the cold season retain nutrients on the field, thus reducing nitrogen losses over the winter (Dinnes et al., 2002; Drury et al., 1993). Alternative crops, especially perennials, have lower nitrate losses (Randall et al., 2001; Misch et al., 2001), due in part to their lower fertilizer demands. Conservation tillage practices, such as ridge tillage and no tillage, have sometimes been found to reduce nitrate losses from corn (Dinnes et al., 2002; Drury et al., 1996) by a combination of less water leaching through tiles and a lower concentration of nitrate in tile drain water (Drury et al., 1993).

Off-farm practices and structures that provide a barrier or buffer through which water must pass before reaching surface waters can be utilized as a BMP to reduce off-site transport of nitrate. Controlled drainage structures allow the water table to rise and store nitrate-rich water high in the landscape, where it may denitrify, or may be used by plants and cannot readily enter the surface waters (Dinnes et al., 2002; Drury et al., 1996). Similarly, bioreactors are buffers through which the water passes that have high organic matter content to encourage denitrification (Dinnes et al., 2002). Riparian buffers adjacent to streams are common practices to protect the stream from pollution, and have been found to be net sinks of nitrogen (Mitsch et al., 2001). However, they are not very effective in tile-drained land because the tile drains bypass the riparian buffer (Dinnes et al., 2002). Constructed wetlands also have potential for reducing nitrate loads through denitrification and plant uptake, and moderating flows through water storage (Dinnes et al., 2002; Randall et al., 2001; Mitsch et al., 2001).

2.2.2. Ecosystem Services Provided by Wetlands

In recent years, there has been rising recognition of the importance of wetlands within the landscape for the ecosystem services they uniquely provide. Ecosystem services are “benefits people obtain from ecosystems” (Reid et al., 2005). Generally, these services include provisioning, regulating, cultural, and supporting services. There are many critical services provided by wetlands, such as flood and climate regulation, water purification, and sustaining biodiversity (Zedler, 2003). In the current era of global climate change, projections indicate increasing hydrologic variability, accompanied by more extreme events of flooding and drought (Milly et al., 2005). Such hydrologic shifts will make flood regulation more difficult, and the Midwest is more vulnerable without its original wetlands.

Wetlands generally attenuate peak flows in high frequency floods by providing storage that slows water flow. This flood regulation is important in the Midwest, especially where land is relatively flat and highly susceptible to damage by flooding. A number of factors are involved in flooding, such as peak flow, flood event volume, length of flooding, and time to peak. For instance, one wetland constructed to detain water on a farm in Louisiana was found to successfully control flooding on that farm during a major hurricane (Millhollon et al., 2009). After synthesizing data from wetlands worldwide, Bullock and Acreman (2003) determined that floodplain wetlands reduce or delay flooding, while headwater wetlands may or may not provide this service. Bullock and Acreman’s (2003) data for the Midwest also indicate that wetlands generally attenuated the smaller floods (2-year return period; 14 out of 15 studies), though the trend was not clear for lower frequency flooding (3 out of 6 studies attenuate peak flows). Wetlands also did not consistently alter flood volume (no trend, 5 studies), and they may have actually prolonged flooding events (4 out of 4 studies). For those hydrologic aspects with no clear trend, there was also no trend evident across wetland type. It appears from the combination of these studies that Midwest

wetlands have a role in reducing flood peak intensity in higher frequency events, most likely by delaying high flows, which in turn attenuates peak flows and increases flood duration.

Wetlands may act as a carbon sink in the landscape and may regulate climate. Carbon dioxide and methane are two naturally produced carbon-based greenhouse gases. Carbon dioxide is more common in the atmosphere, but methane is more potent in causing global warming. Wetlands often act as a carbon dioxide sink and a methane source due to their anaerobic respiration. Another gas that wetlands may introduce in the process of denitrification is nitrous oxide, a potent greenhouse gas. Therefore, a wetland can either contribute to or mitigate global warming, depending on its balance of carbon dioxide, methane and nitrous oxide production. Altor and Mitsch (2008) analyzed soils from a Midwest floodplain in mesocosms to study carbon dynamics under two different flooding regimes. They found that continuously inundated, hydric soils produced the most methane, while non-hydric soils under intermittent flooding had the lowest methane emissions. Net carbon dioxide uptake was similar across mesocosms and appeared to outweigh methane emissions. This demonstrates a common finding that natural wetlands generally act as a carbon sink, providing climate regulation, and that carbon dynamics will vary based on flooding regime and other wetland properties. While wetland restorations alone are unlikely to provide a sufficient reduction of atmospheric carbon dioxide levels, wetlands have potential for use in a suite of strategies to confront the problem of climate change.

As diverse transition zones between terrestrial and aquatic ecosystems, wetlands also play an important role in sustaining biodiversity. Biodiversity is the basis behind ecosystem services and life on earth, and it is under increasing pressure due to a warming climate and growing population (Reid et al., 2005). The diverse assortment of wetlands on earth promotes tremendous biodiversity

(Semlitsch et al., 1998). Wetlands are capable of uniquely providing many ecosystem services, and even small wetlands on marginal lands may be useful for these tasks. In fact, the most strategic placement of wetlands may be in various sized segments of marginal cropland that was historically wetland (Zedler, 2003).

2.2.3. Nitrate Removal

Wetlands are ecosystems containing biota such as plants and bacteria, water, and soils. As ecosystems, they process and extract value from available energy sources that pass through. This means they process nutrients, and are also capable of self-assembly of biota, so they may require less long-term maintenance than other technical (non-biological) solutions (Mitsch et al., 1998). Wetlands are known to play a part in nutrient cycling, and act as a filter to reduce nitrogen loads through processes such as denitrification (Hernandez and Mitsch, 2007). Natural wetlands are generally a sink for both nitrogen and phosphorus (Fisher and Acreman, 2004). Wetlands have been constructed in many different ways and for a variety of purposes. For instance, constructed wetlands for human wastewater treatment are reported to generally remove 40-50% of the influent nitrogen (Lee et al., 2009). Wetlands constructed for treatment of wastewater often prove cost-effective when compared with conventional treatment plants (Lee et al., 2009). Constructed wetlands have even been used to remove nitrate from groundwater (Lin et al., 2008).

The nitrogen cycle is complex, and nitrogen can go through many transformations within a wetland, including nitrogen fixation, plant uptake, microbial transformations, volatilization of ammonia, and denitrification of nitrate. But few of these transformations actually remove nitrogen from the system – instead, they might store the nitrogen for a time only to become a source of nitrogen in the future. Denitrification is a primary mechanism of nitrogen removal from the inflow, especially when nitrate concentrations are high, as they are in

agricultural drainage (Vymazal, 2007). Denitrification is an irreversible process in which nitrogen oxides (nitrate and nitrite) are the terminal electron acceptor for microbial respiration. As such, denitrification requires organic matter substrates, microbial communities (which are generally present under wetland conditions), and primarily anaerobic conditions (otherwise oxygen is used as the electron acceptor). Because it requires microbial activity, which is subdued in the cold months, denitrification is more effective in the warm summer months. In one small wetland (about 0.1% of its contributing area's size) in North Carolina, denitrification alone removed 10-16% of dissolved inorganic nitrogen (primarily nitrate and nitrite) entering the wetland in the summertime (Poe et al., 2003).

Nitrate removal efficiencies range depending on a wetland's size relative to its contributing area, the frequency and magnitude of precipitation, temperature/season, biological activity, and many other factors. Generally, wetlands are more efficient at removing nitrate from highly concentrated inflows, during the warm months, and under lower hydraulic loading rates. Residence time of water in a wetland, the inverse of hydraulic loading rate, is the primary determination of its overall efficiency because nitrate removal is a time-dependent process.

A two year study of a wetland treating agricultural runoff in Maryland under variable flow regime found nitrate removal efficiencies of 52% (Jordan et al., 2003). One relatively small wetland (0.1% of its contributing area) in North Carolina removed 53% of the incoming nitrate from row crops (Poe et al., 2003). In Illinois, Kovacic et al. (2000) studied small wetlands placed between tile drains and the river, with wetland areas ranging from 3-6% of their (tile-drained) contributing areas. In the 3-year study average inflow nitrate concentrations were approximately 10 mg/L, and average nitrate removal was 37%. In another study in Illinois, two wetlands with primarily tile-drained contributing areas, sized at 3-4% of their contributing areas, were found to retain 23-44% of entering

nitrate, with an average of 36% efficiency (Kovacic et al., 2006). Corresponding flow-weighted nitrate concentrations were reduced by 31-42%. From these studies, nitrate removal in agricultural wetlands are generally reported to range from 23-53% of incoming nitrate loads.

2.3. Strategically Siting Wetlands

2.3.1. Watershed Perspective

If wetlands were to be the sole solution for reducing nutrient loadings to meet US EPA goals, millions of hectares (21,000-52,000 km^2) of wetlands would be necessary to see any significant reduction in nitrogen loadings to the Gulf of Mexico (Mitsch et al., 2001). Mitsch and Day (2006) estimated that a 40% decrease in nitrogen loading could be attained by creation of 2.2 million hectares of wetlands, which is less than 1% of the area of the Mississippi River Basin. The authors also reported that the majority of these wetlands may need to be in the Upper Mississippi River Basin, where nitrogen loadings are relatively greater.

Crumpton (2001) used a modeling approach to show that wetlands can be much more efficiently placed when considering the size of their contributing areas. Wetlands were placed according to two different approaches over 1% of the entire watershed. A ‘conventional’ approach to siting wetlands without this consideration resulted in a 4:1 ratio of drainage area to wetland size. The wetlands intercepted 4% of the flow from the entire watershed. When wetlands were instead placed to maximize their efficiency by intercepting flow from large contributing areas – a ‘watershed’ approach – they intercepted 50% of the flow from the entire watershed. These simulated wetlands removed 35% of annual nitrate loss from the entire watershed, and were effective during both low and high flow events.

2.3.2. Wetland Siting and Design

Restoring wetlands in a strategic way can regain ecosystem services that were lost by draining wetlands. “Strategic restoration” involves targeting historic wetlands converted to marginal cropland, locating and sizing the wetlands according to their surroundings, and adaptive management (Zedler, 2003). A number of policy tools are available to provide incentives to farmers to restore or construct wetlands in these locations (e.g. Farm Bill; Conservation Reserve Enhancement Program (CREP)). Several criteria have been considered for the strategic siting and design of constructed wetlands to target nitrate reductions in a tile-drained watershed.

Tomer et al. (2003) performed an analysis using ArcGIS and publicly available data layers to strategically site riparian buffers and constructed wetlands in Iowa. Their criteria for wetland placement and design included the following: 1) the wetland should be located on cropland and intercept tile drainage or a small stream; 2) the wetland should intercept flow from at least 200 hectares of tile-drained cropland; 3) wetland size should be 0.5-2% of its drainage area; 4) no more than 25% of the wetland should be greater than 1 meter deep; 5) a buffer region must surround the wetland stretching from the wetland surface to 1.5 meters in elevation above it, and the area of this region should not exceed twice the size of the wetland area. Using these criteria, they found a number of successful wetland locations that could intercept 40% of the entire study watershed. However, only one quarter of these wetland sites were ideal for wetland creation.

2.4. Modeling Nitrate Removal in Wetlands

Wetland nitrate reductions are often modeled as a first-order kinetic process (Kadlec and Knight, 1996), where concentration reductions are a function of time and a nitrate removal rate constant. The flux of nitrate removal over the area of a wetland is:

$$J = \frac{dC}{dt} \times d = k \times C \qquad \text{Eq. 2.1}$$

where *J* is the area-based nitrate loss due to denitrification in units g/m^2/yr, *d* is the average depth of the wetland, *k* is the area-based, first order nitrate loss rate with units m/yr, and *C* is the concentration of nitrate in the wetland in mg/L (which is equivalent to g/m^3). The integrated rate law becomes:

$$C = Co \times e^{-kt/d} \qquad \text{Eq. 2.2}$$

where *t* is time (yr) and C_o is the initial concentration in the wetland (mg/L).

Nitrate removal rates also depend on temperature, as plant and microbial activity is limited in the cold months. Temperature dependence can be modeled directly as (Crumpton, 2001):

$$J = k_{20} \times C \times \theta^{(T-20)} \qquad \text{Eq. 2.3}$$

where k_{20} is the nitrate loss rate at 20 ℃, θ is the temperature coefficient of denitrification, and *T* is the temperature in ℃. Other models include te mperature indirectly through different rate constants in each season (Burchell et al., 2007).

Other researchers have created regression models to show how nitrate removal depends on various factors. Crumpton et al. (2006) calculated annual nitrate removal based on a regression of annual hydraulic loading rate from data taken on constructed wetlands in Iowa. In this regression, percent mass of nitrate removed (*% N removed*) is a function of annual hydraulic loading rate (*HLR*, in m/yr), which is the depth of flow a wetland receives within a year:

$$\%\, N\, removed = 103 \times HLR^{-0.33} \qquad \text{Eq. 2.4}$$

Using this model, Crumpton et al. (2006) estimated nitrate loads in the Upper Mississippi River Basin and found that 80% of the nitrate entering the Mississippi River originates in less than 30% of the basin. Then they extended their nitrate reduction model to estimate nitrate removal that can be expected by strategically placed wetlands in the Upper Mississippi River Basin. The authors determined

that 210,000 to 450,000 hectares of wetlands could achieve a 30% overall reduction of nitrate exported from the basin. Their modeled wetlands had wetland to contributing area ratios of 2%, and resulting nitrate removal efficiencies from 40-60%.

2.4.1. Soil and Water Assessment Tool (SWAT)

The SWAT model is a watershed-scale tool used to estimate long-term impacts of land management on nonpoint source pollution (Arnold et al., 1998). It is a physically based model that simulates upland and in-stream water and chemical transport. Its major upland components include climate, hydrology, nutrients, soil erosion/sedimentation, plant growth, and management practices. In-stream processes include routing of flow, sediment, and nutrients, as well as nutrient transformations.

Within the model, a watershed is divided into subwatersheds based on flow paths of water. These subwatersheds are further divided into hydrologic response units (HRUs), which are defined by similar land use and soils, and do not retain spatial reference beyond their placement in the subwatersheds. Inputs to the model are at the HRU level, and include subwatershed characteristics (elevation data, soil), climate data (precipitation, temperature, relative humidity, wind speed, and solar radiation), management practices (land use data), as well as plant growth data. Simulations provide daily, monthly and annual estimations of water balance, nutrient, sediment, pesticide, and bacteria losses, as well as plant biomass and yield at the subwatershed scale. The SWAT model has been shown to perform satisfactorily at monthly and annual time-scales (Borah and Bera, 2004; Gassman et al., 2007). SWAT has been extensively used to model the effect of changing management practices on hydrology and water quality (Gassman et al., 2007).

SWAT is capable of modeling water bodies in the form of reservoirs, ponds, and wetlands (Neitsch et al., 2005). All of the wetlands in a subwatershed are modeled as one lump sum of land area, and the user can specify the fraction of the subwatershed that drains to the wetlands ($frac = \frac{contributing\ area}{sub-watershed\ area}$). If the wetland is located at the outlet of the subwatershed, *frac = 1*. In SWAT, wetland hydrology is simulated using the following water balance for water volume *V* on the daily time scale:

$$V = Vstored + Vflowin - Vflowout + Vpcp - Vevap - Vseep \qquad \text{Eq. 2.5}$$

where *V* represents water volume (m^3), *Vstored* is the water stored in the wetland at the start of the day, *Vflowin* is the water that flows into the wetland during the day, *Vflowout* is the water that flows out of the wetland during the day, *Vpcp* is the precipitation falling directly onto the wetland during the day, *Vevap* is the water evaporating from the wetland surface that day, and *Vseep* is the water lost by seepage during the day.

These daily water balance measures are calculated from user inputs including wetland surface area and volume at normal pool and maximum water levels, precipitation data, soil data (saturated hydraulic conductivity of the wetland soil), and the fraction of the subwatershed that drains to the wetlands:

$$Vstored = V\ of\ the\ previous\ day \qquad \text{Eq. 2.6}$$

$$Vflowin = frac \times 10 \times (Qsurf + Qgw + Qlat) \times (Area - SA) \qquad \text{Eq. 2.7}$$

$$Vflowout = \begin{bmatrix} 0 & if\ V < Vnor \\ \frac{V - Vnor}{10} & if\ Vnor \leq V \leq Vmx \\ V - Vmx & if\ V > Vmx \end{bmatrix} \qquad \text{Eq. 2.8}$$

$$Vpcp = 10 \times Rday \times SA \qquad \text{Eq. 2.9}$$

$$Vevap = 10 \times \eta \times Eo \times SA \qquad \text{Eq. 2.10}$$

$$Vseep = 240 \times Ksat \times SA \qquad \text{Eq. 2.11}$$

where *frac* is the fraction of the subwatershed that drains to the wetland, *Qsurf* is the surface runoff in the subwatershed during that day (mm H_2O), *Qgw* is the groundwater flow in the subwatershed during that day (mm H_2O), *Qlat* is the lateral flow in the subwatershed that day (mm H_2O), *Area* is the subwatershed area (ha), *SA* is the wetland surface area (ha), *Vnor* is the normal pool volume in the wetland (m^3), *Vmx* is the maximum volume of the wetland (m^3), *Rday* is the amount of rainfall to fall directly on the wetland that day (mm H_2O), η is a coefficient of evaporation, *Eo* is the potential evapotranspiration that day, and *Ksat* is the saturated hydraulic conductivity of the wetland soil.

SWAT also calculates the nutrient dynamics within the wetland. The model assumes that the flow in the wetland is completely mixed, and the incoming nutrients are immediately distributed evenly throughout the wetland. Nutrient transformations are calculated at the daily timescale, assuming that nutrients are removed solely by settling (i.e. there is no exchange between nutrient pools, such as NO_3, NO_2, and NH_4). The settling rate of a nutrient is calculated from a user-input settling velocity and concentration of the nutrient in the wetland. The final nutrient balance is:

$$V \times \frac{dc}{dt} = W(t) - Q \times c - v \times c \times SA \qquad \text{Eq. 2.12}$$

where *V* is the volume of water in the wetland (m^3 H_2O), *c* is the concentration of a given nutrient (kg/m^3) in the wetland at time *t*, *W(t)* is the mass of nutrient entering the wetland that day (kg/day), *Q* is the flow rate of water leaving the wetland (m^3 H_2O/day), *v* is the apparent settling velocity of the nutrient (m/day), and SA is the surface area of the wetland (which is assumed to be equal to the sediment-water interface, as the wetland is assumed to have a uniform depth).

The SWAT model has been used extensively to model flow and nutrient transport from agricultural watersheds. Yet there are few studies in which wetlands have been specifically analyzed using the SWAT model. Arnold et al. (2001) used the SWAT model to estimate the water budget of a proposed constructed wetland

near Dallas, Texas. They calibrated, validated, and ran the SWAT model over 14 years. The wetland generally remained nearly full, and appeared to give a realistic description of wetland hydrology. Bosch (2008) used the SWAT model to analyze the impact of ponds and reservoirs on the export of nutrients from two watersheds in southeast Michigan. The SWAT model was calibrated, validated and run for ten years, and water bodies were designated as reservoirs if they were greater than 50 hectares in size and located near subwatershed outlets; otherwise they were modeled as ponds if they were greater than 10 hectares in size. They ran scenarios in which these water bodies were removed or additional reservoirs were added to examine the effect of water bodies on nutrient export from the subwatersheds. They found that these water bodies considerably lowered nutrient export, including nitrogen. In one subwatershed the removal of wetlands caused a nearly 100% increase in nitrogen loss from the subwatershed. They found that waterbody placement near nutrient sources (agricultural fields) resulted in greater nutrient reductions. They also found that a number of small ponds were more efficient in reducing nitrogen than one large reservoir.

2.5. Summary

Nitrogen loadings from the Upper Mississippi River Basin are a primary driver of hypoxia in the Gulf of Mexico. Nitrogen losses in the form of nitrate are significant in tile-drained lands, and contribute considerable nitrogen to surface waters. Because wetlands have been extensively drained in the Upper Mississippi River Basin, constructed wetlands are one potential solution to the problem. In tile-drained watersheds, wetlands are probably a more appropriate BMP than commonly-placed riparian buffers, because drains bypass these buffers. Wetlands enhance denitrification, which removes nitrogen from water and returns it to the atmosphere. Wetlands also have many other ecosystem services, such as flood regulation, wildlife habitat, and possible carbon sequestration. The most efficient placement of wetlands in tile-drained lands

involves taking a watershed perspective, where wetlands receive high flows, and wetlands are sized according to their contributing areas. Nitrate removal in wetlands has been modeled in a number of ways, including the Soil and Water Assessment Tool (SWAT), which is a physically-based watershed-scale model appropriate for estimating the outcome of management decisions on nonpoint source pollution. While SWAT has had limited application in wetlands in the Midwest, it is a powerful tool with all of the necessary components to model nitrate removal by wetlands in tile-drained lands.

CHAPTER 3. STRATEGICALLY SITING CONSTRUCTED WETLANDS TO TARGET NITRATE REMOVAL: A GIS METHOD APPLIED TO AN AGRICULTURAL WATERSHED IN WEST CENTRAL INDIANA

3.1. Introduction

Intensification of agricultural practices in the Midwest USA has had enormous economic benefits, such as increased yields in food, fuel, and feed. Yet this intensified production has come with unintended consequences, especially in the area of environmental degradation. One case pertaining especially to the agricultural Midwest is the hypoxia in the Gulf of Mexico. Nitrogen loads from agricultural production in the Mississippi and Atchafalaya River Basins have caused increased eutrophication and seasonal hypoxia in the Gulf of Mexico (Rabalais et al., 2002b). This is a zone with low oxygen levels (< 2mg/L) that forms each summer off the coast of Louisiana and threatens many ecological, recreational and commercial aspects of the Gulf. Its extent fluctuates, but on average this zone occupies 15,000 km^2 at its peak.

The primary driver of hypoxia in the Gulf of Mexico is nitrogen loads from the Mississippi River (Justic et al., 1993), and the majority of this nitrogen originates from nonpoint sources in the 'Upper Mississippi River Basin,' which is the combination of the Upper Mississippi, Ohio, and Tennessee River Basins, including west central Indiana. The Upper Mississippi River Basin contributes an estimated 60-90% of the total nitrogen transported to the Gulf (Alexander et al., 2008; Gulf Hypoxia Action Plan, 2008; Goolsby et al., 1999). Reasons for this region's high nitrogen contributions include high flows due to its moist climate, shorter flow paths to the large river stems that allow for less in-stream nutrient removal (Alexander et al., 2008), and corn dominated agriculture (Burkart and

James, 1999). Artificial drainage is also common in this region, especially in the relatively flat central Indiana, and this drainage appears to strongly influence nitrogen losses in the common form of nitrate.

The 2008 EPA Action Plan calls for a reduction of the hypoxic zone to about thirty percent of its five-year running average by 2015 (Gulf Hypoxia Action Plan, 2008). To meet the goals of the Action Plan, nitrogen loadings from the Upper Mississippi must be significantly reduced. A recent estimate by the EPA Science Advisory Board (2007) is that a reduction of 45% of both nitrogen and phosphorus is necessary to meet the goal by 2015. Previously, the target was a reduction of only 30% of nitrogen to the Gulf. With climate change and other uncertainties, it is possible that reductions of greater than 45% will be required (Donner and Scavia, 2007).

Much of the Upper Mississippi River Basin was historically rich with wetlands, but for the most part these have been drained for agriculture and urbanization. In 1850, wetlands covered approximately 24-31% of the land area in Indiana (McCorvie and Lant, 1993). By the 1980s, an estimated 87% of Indiana's original wetlands had been drained (Mitsch and Gosselink, 2007). Another estimate shows that half of Indiana's cropland was drained in 1885 (Pavelis, 1987) by a combination of open ditches and underground tile drains. Drains allow nitrate-rich water to flow rapidly beneath the ground, where it is free from uptake by the biologically active upper soil layers. Consequently, flow from tile drains can carry large loads of nitrate into surface waters (Hickey and Doran, 2004). In heavily tile-drained watersheds, tiles are often the primary path of nitrate loss (David et al., 1997; Drury et al., 1993; Gentry et al., 2009). Kladivko et al. (1991 and 2004) found that drains spaced closer together led to greater flows and consequent loads of nitrate loss. In regions with extensively tile-drained land, the resulting nitrate concentration in surface waters was often found above the drinking water standard of 10 mg/L (Drury et al., 1993; Jaynes et al., 2001; Schilling & Lutz,

2004; Kladivko et al., 2004). Because much of the Upper Mississippi River Basin is tile-drained, and the majority of its original wetlands are lost, nitrate leaches readily into surface waters.

A number of best management practices (BMPs) have been developed in order to reduce environmental threats imposed by intensive agriculture. These BMPs include on-farm practices pertaining to nitrogen application, cropping, and tillage practices, as well as off-farm structures such as riparian buffers and constructed wetlands designed to intercept high-nitrate flows before they reach surface waters. Wetlands are known to play a part in nutrient cycling, acting like a filter to reduce nitrogen loads through processes such as denitrification (Hernandez and Mitsch, 2007). In tile-drained watersheds, wetlands are probably a more appropriate BMP than riparian buffers, because sub-surface flow from tile drains bypasses the riparian buffers (Dinnes et al., 2002). Because wetlands have been extensively drained in the Upper Mississippi River Basin, and the tile drains have altered considerably the hydrologic characteristics of the land, constructed wetlands may also be a partial restoration of a more sustainable landscape. Both natural and constructed wetlands are generally a sink for nitrogen and phosphorus (Fisher & Acreman, 2004) through denitrification and plant uptake, and moderating flows through water storage (Dinnes et al., 2002; Randall et al., 2001; Mitsch et al., 2001).

Nitrate removal efficiencies are affected by many factors, including wetland size relative to its subwatershed, frequency and magnitude of precipitation, wetland hydrology, temperature/season, and biological activity. Generally, wetlands are more efficient at removing nitrate from highly concentrated inflows, under lower hydraulic loading rates, and during the warm months. A two year study of a wetland treating agricultural runoff in Maryland under a variable flow regime found nitrate removal efficiencies of 52% (Jordan et al., 2003). One relatively small wetland (0.5% of its contributing area) in North Carolina removed 53% of

the incoming nitrate from row crops (Poe et al., 2003). In Illinois, Kovacic et al. (2000) studied small wetlands placed between tile drains and the river, with wetland areas ranging from 3-6% of their (tile-drained) contributing areas. In the 3-year study average inflow nitrate concentrations were around 10 mg/L, and average nitrate removal was 37%. In another study in Illinois, two wetlands sized at 3-4% of their primarily tile-drained contributing areas were found to retain 23-44% of entering nitrate, with an average of 36% efficiency (Kovacic et al., 2006). Corresponding flow-weighted nitrate concentrations were reduced by 31-42%. Overall, these wetlands removed 23-53% of incoming nitrate and were generally sized within the range of 0.5-4% of their contributing areas. While these nitrate removal efficiencies may appear great, at the watershed scale the impacts of these wetlands would appear small unless they are placed all throughout the landscape.

If wetlands were to be the sole solution to reduce nutrient loadings to meet US EPA goals, millions of hectares (21,000-52,000 km^2) of wetlands would be necessary to see any significant reduction in nitrogen loadings to the Gulf of Mexico (Mitsch et al., 2001). Mitsch and Day (2006) estimated that a 40% decrease in nitrogen loading could be attained by creation of 2.2 million hectares of wetlands, which is less than 1% of the total Mississippi River Basin. The majority of these wetlands may need to be placed in the Upper Mississippi River Basin, where nitrogen loadings are greater. In order to place wetlands efficiently in the landscape, a watershed-scale perspective is needed (Crumpton, 2001).

Determining suitable wetland sites for effective nitrate removal may be performed using spatial data and a Geographic Information System (GIS). Tomer et al. (2003) performed an analysis using ArcGIS and publicly available data layers to strategically site riparian buffers and constructed wetlands in Iowa. They found a number of successful wetland locations that could intercept 40% of the entire study watershed. However, only one quarter of these wetland sites were ideal

for wetland creation according to their criteria and other factors (such as inability of a wetland to cross a road). The critical basis of this GIS analysis was calculating the contributing area from tile-drained land that drains to each location. This ensures that wetlands intercept high nitrate loads and maximize nitrate reduction in the landscape. This concept forms the basis of an effort in Iowa to place large constructed wetlands for efficient nitrate reductions (Tomer et al., 2003), funded by the Conservation Reserve Enhancement Program (CREP).

Iowa's CREP formed the initial basis of this work. CREP is managed by the USDA's Farm Service Agency to fund conservation of sensitive agricultural lands, with a primary focus on high-priority problems such as nitrate losses from agricultural land in the Upper Mississippi and Ohio River Basins. Strategic placing of wetlands in the watershed achieves nitrate reductions at the lowest cost, increasing the opportunities to fund such a project. This work was solicited by the Indiana State Department of Agriculture (ISDA) to provide a methodology for prioritizing the most efficient wetland designs to fund through CREP in Indiana. Therefore, criteria were formulated to provide cost-effective nitrate removal in the study region.

The primary costs in wetland creation are the area of land retired and the construction costs. Retired land is not only the wetland itself, which should be sized according to its contributing area, but it also includes a buffer surrounding the wetland that rises 1.2 meters in elevation above the wetland surface. Increasing the size of this buffer greatly increases the cost while providing no additional nitrate removal. Therefore, the buffer size should be minimized. One variable in construction costs is the length of dam necessary to create the ponded area of the wetland. All other factors held constant, a larger dam will increase the cost of the project without improving nitrate reductions. Therefore, the length of the dam should also be minimized.

In order to test the efficacy of a methodology for strategically siting wetlands, it is important to estimate the nitrate removal that can be expected from them. Wetland nitrate reductions are often modeled as a first-order kinetic process (Kadlec and Knight, 1996), where concentration reductions are a function of time and a nitrate removal rate constant. Nitrate removal rates also depend on temperature, as plant and microbial activity is limited in the cold months. Some models include temperature directly (Crumpton, 2001), or indirectly through different rate constants in each season (Burchell et al., 2007).

Other researchers have created regression models to show how nitrate removal depends on various factors. Crumpton et al. (2006) calculated annual nitrate removal based on a regression of annual hydraulic loading rate from data taken on constructed farm wetlands in Iowa. In this study, percent mass of nitrate removed was determined as a function of annual hydraulic loading rate. They estimated nitrate removal that could be expected by strategically placing wetlands in the Upper Mississippi River Basin, and determined that 210,000 to 450,000 hectares of wetlands could achieve a 30% overall reduction of nitrate exported from the basin. Their modeled wetlands had wetland to contributing areas ratios of 2%, and resulting nitrate reduction efficiencies ranging from 40-60%.

3.1.1. Objectives

The main goal of this work was to develop a methodology for targeted wetland placement to remove nitrate from tile-drained agricultural lands. The specific objectives were to: 1) Determine suitable wetland sites in an 8-digit HUC in Indiana using GIS methods and wetland siting criteria; 2) Create preliminary wetland designs at each site according to wetland design criteria; and 3) Estimate the nitrate removal provided for each wetland design, using a simple regression-based model. A key aspect of this GIS analysis was calculating the tile-drained contributing area draining to each location, which ensured that these

wetlands intercepted large flows and maximized nitrate reduction in the landscape. This watershed-scale approach is critical to efficiently implement best management practices, such as constructed wetlands to reduce nitrate losses from agricultural watersheds.

3.2. Description of Study Site

The watershed used for this analysis was the Middle Wabash–Little Vermillion 8-digit Hydrologic Unit Code (HUC) 05120108. While a small part of the HUC reaches into Illinois, all the analyses were performed only in the Indiana portion of the watershed because the work was intended for use in Indiana (Figure 3.1). A number of data were obtained to describe the dataset (Table 3.1). The Indiana portion of this watershed is approximately 5,346 square kilometers in size and composed of 67% agricultural land use, of which 62% is estimated to be tile-drained. The watershed was characterized by poor soil drainage and historically contained numerous wetlands due to a relatively flat topography. Hydric soils cover 17% of the watershed. Most of these wetlands were artificially drained for agricultural purposes, with tile drains that lie about 1 meter (3 feet) below the ground surface.

3.3. Materials and Methods

3.3.1. Data Layers

A number of publicly available data layers were used in the GIS analysis. Table 3.1 gives an explanation of their sources and how they were used in the analysis. ArcGIS 9.3 (Johnston et al., 2001) was the GIS software used for the entire analysis.

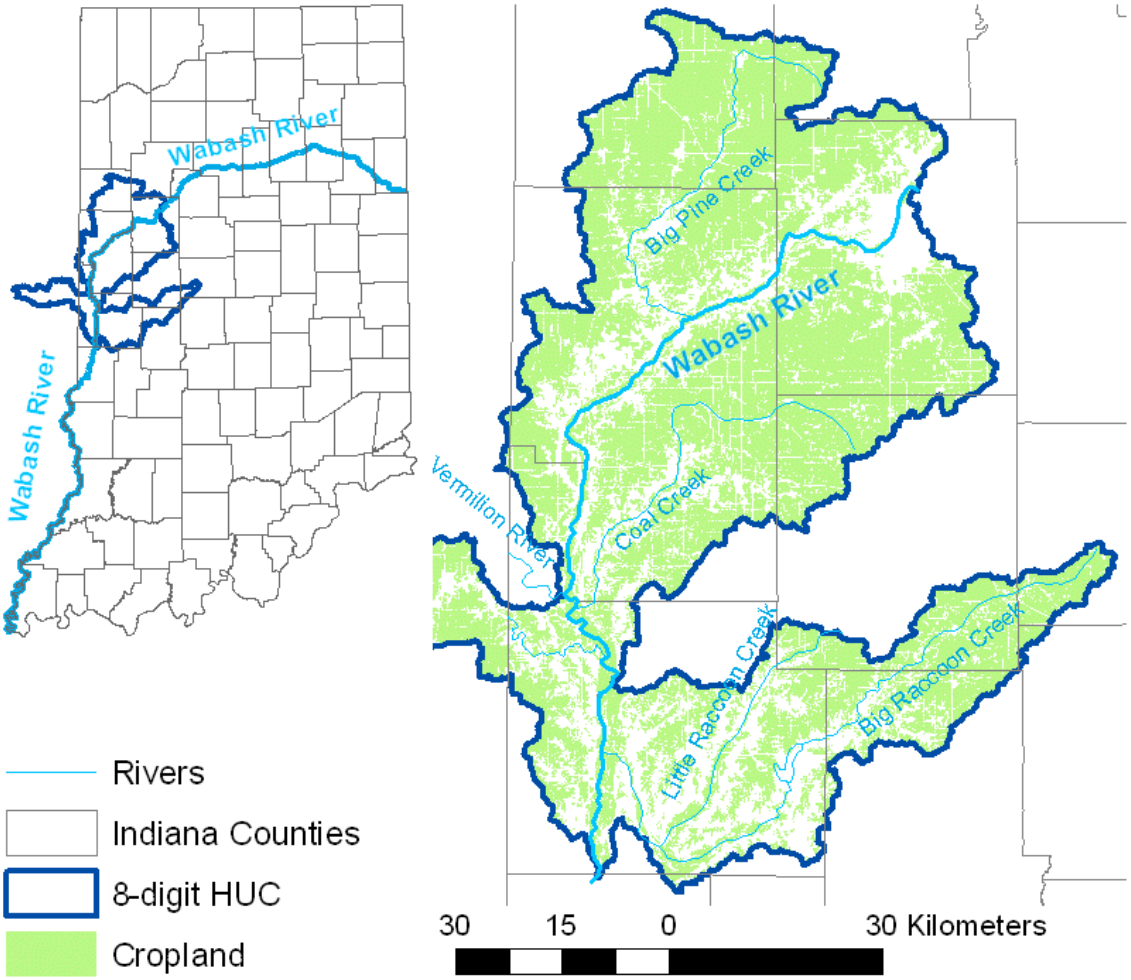

Figure 3.1 The Middle Wabash–Little Vermillion watershed (8-digit HUC 05120108).

3.3.2. Criteria for Wetland Placement and Design

Several criteria were used to place wetlands in the watershed, and also to determine feasibility and efficiency of each wetland placed (Table 3.2). The criteria were developed based upon those used in the Iowa CREP program and the work of Tomer et al. (2003), knowledge of Indiana agricultural practices, and discussions with members of the Indiana State Department of Agriculture (ISDA). This work was performed for the ISDA, so that they might replicate this methodology in other 8-digit HUCs in Indiana. The precise methodology developed to satisfy each of the siting criteria is detailed in the subsequent sections.

3.3.2.1. Sufficient contributing area from tile-drained land

To make the greatest impact in the landscape, each wetland was placed strategically to intercept a large flow of water from tile-drained land, which is the source of much of the nitrate in Indiana landscapes. These locations were targeted in two steps: (1) finding locations that have contributing areas between 2 and 8 km^2 (500 and 2000 acres) in size, and (2) narrowing these locations to only those that drain at least 2 km^2 (500 acres) of tiled land. The lower limit in Step (1) is intended to eliminate low-flow sites where appropriately sized wetlands would be quite small and yet have large buffers, decreasing cost-effectiveness. The upper limit in Step (1) is set sufficiently high to eliminate only sites that would be found within streams (see Criterion 2). Step (2) ensures that wetlands will intercept high flows from tile-drained lands where nitrate concentrations are highest.

Step (1) was completed by processing 10-meter grid National Elevation Data (NED) in Arc Hydro to determine flow accumulation – the paths along which water should flow in a landscape. Locations with very large contributing areas appeared as single paths because water flow is channelized. Step (2) was similar to Step (1), except a weighted flow accumulation was used that only takes

Table 3.1 GIS data layers

Data layer	Source dataset	How the layer was used in the analysis
Streams	National Hydrography Dataset (NHD), high resolution streams, downloaded June 2009	Used to eliminate streams as suitable locations for placing wetlands, and to approximate the location where closed drains empty into open drains
Elevation	National Elevation Dataset (NED), with one-third arc second resolution	Used to delineate watersheds, describe how water flows through the landscape, and create contours for wetland design
Roads	TIGER 2008 Census data, “edges”	Used to break up separate crop fields and also for locating different wetland sites
Cropland	National Land Cover Dataset (NLCD), selecting the field “cultivated crops”	Used for locating suitable sites for wetland placement and also to approximate tile-drained land in the landscape
Tile-drained land (estimate)	From SSURGO soils and Cropland (poorly, very poorly, and somewhat poorly drained cropland)	Used to estimate sites that drain a large contributing area of tile-drained land
Hydric soils	Soil Survey Geographic (SSURGO) Database	Used to give a general idea of areas that were formerly wetland
Orthophotos	Indiana Spatial Data Framework, seamless statewide basemap	Used for visualization of sites and determination of locations where closed drains empty into open ones

Table 3.2 Criteria for wetland siting and preliminary designs

Wetland Siting Criteria
1. Wetland has sufficient contributing area from tile-drained land (2-8 km^2 (500-2000 acres) of tile-drained land)
2. Wetland must not intercept a stream or open waterway
3. Wetland is on cropland
4. Topography lends itself to wetland placement
Wetland Design Criteria
5. Desired wetland size is 0.5-2% of its contributing area
6. No more than 25% of the wetland is more than 1 meter (3 feet) deep (“deep wetland”)
7. The surrounding buffer must extend 1.2 meters (four feet) above wetland surface, and should not exceed 4 times the size of the wetland

into account flow originating from tile-drained land. Results of each step are shown in Figure 3.2.

3.3.2.2. Wetland must not intercept a stream or open waterway

Streams were not considered suitable wetland sites because placing a wetland in an open waterway would greatly increase regulatory barriers. The NHD Streams dataset was used to eliminate any sites found within a stream. However, the flow paths determined above did not always line up directly beneath the NHD Streams data. To account for discrepancies between stream and flow accumulation datasets, the NHD Streams dataset was buffered 100 meters and all locations within this region were excluded from the analysis (Figure 3.3).

Initially, placing wetlands at the interface between closed and open drains was a separate criterion. Interrupting a closed drain could be problematic because water leaving the wetland is a combination of flows above and below ground and potentially difficult or even impossible to route back through a closed tile. Locating the wetland downstream in an open drain (ditch or stream) may require permitting from various regulatory agencies. However, this criterion became unduly limiting and it was removed from the analysis.

3.3.2.3. Wetland is on cropland

Wetlands constructed under funding from the Conservation Reserve Enhancement Program (CREP) or similar programs must be placed on cropland. To determine cropped fields for wetland placement, those suitable sites located on cropland were selected. First, the most downstream outlet point for each flow path was selected and its contributing area (watershed) delineated. These contributing areas represented all locations where a wetland could possibly be placed. Then, the contributing areas were intersected with the “cultivated crops” data. Fields that contained a location with sufficient contributing area were

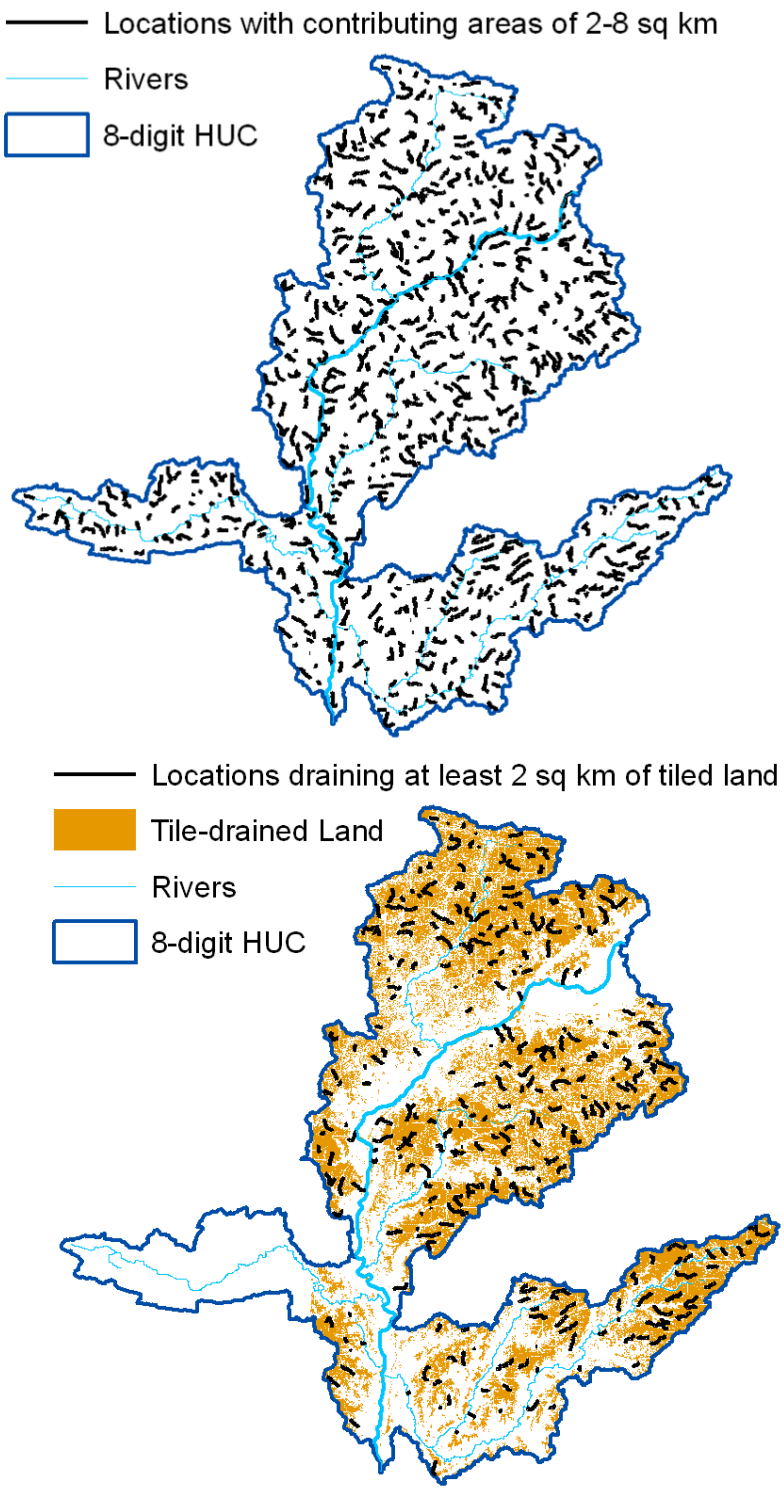

Figure 3.2 Determining locations that have sufficient contributing area from tile-drained land. Top: Step (1) locations with large (2-8 km^2) contributing areas. Bottom: Step (2) using estimate of tile-drained land to target high nitrate flows.

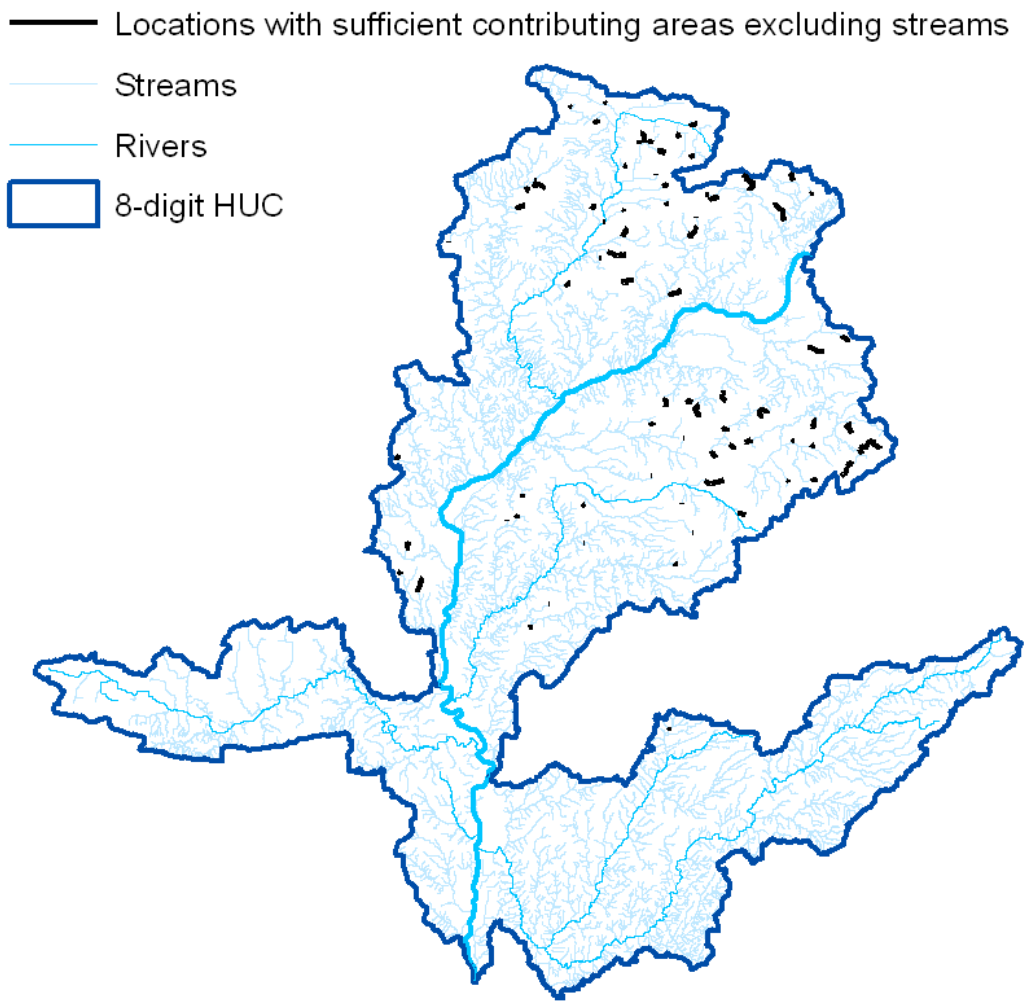

Figure 3.3 Using NHD streams data to eliminate locations found in open waterways.

selected. The final output looks like the example shown in Figure 3.4, where the cross-hatched field is the only site where a wetland could be placed. A total of 113 crop fields were found, though not all were truly suitable and were narrowed further in the next steps.

3.3.2.4. Topography lends itself to wetland placement

Topography is an important element of placing wetlands. Very flat topography, which is characteristic of parts of the study site, lacks the elevation drop necessary to hold water when a dam is placed. The ideal topography for a constructed wetland would be somewhat “bowl-shaped” – a large basin to create the wetland, surrounded by steep walls that allow for a small amount buffer land between wetland and cropland, and a narrow exit that allows for a small dam to be placed to minimize cost. Testing for appropriate topography involved qualitative analysis using contours. All suitable crop fields were examined qualitatively to determine whether topography was appropriate for wetland creation at each site. Wetland designs were visualized using 0.3-meter (1 ft) elevation contours. Orthophotos and the streams data layer were used to accurately determine the interface between closed and open drains. Good “bowl-shaped” topography was considered to be a small dam, indicated by dense contours, and a large and steep topographic rise of over 1.2 meters for buffer placement. A good candidate site is shown in Figure 3.5. Finally, preliminary designs were created at each site with suitable topography.

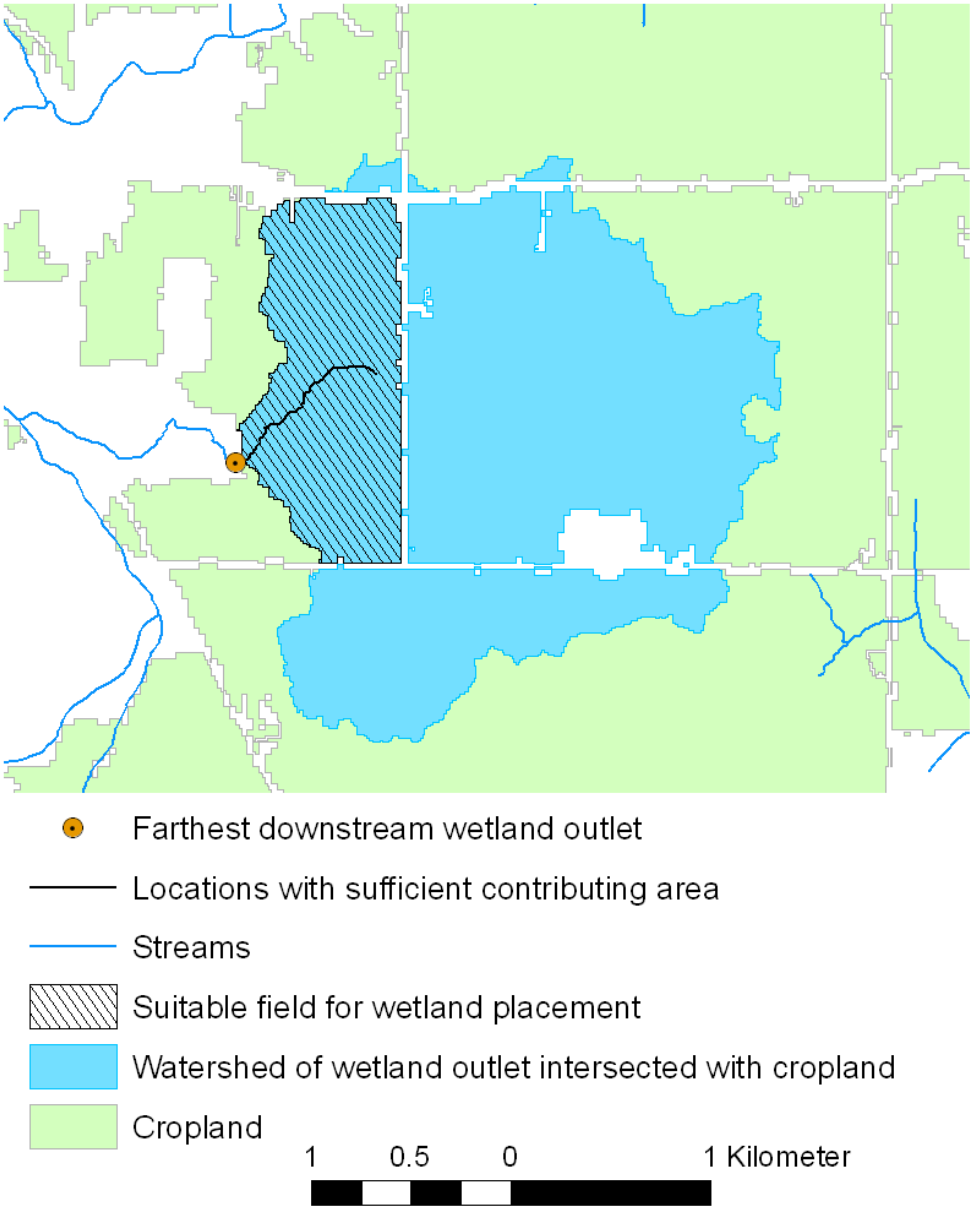

Figure 3.4 An output of the methods for siting wetlands. First an outlet point is found where the locations with sufficient contributing area (black line) empty into an open stream (light blue line). Next the contributing area for this outlet point is delineated. Then the contributing area and cropland layers are combined (blue areas), and any crop field that contains a location with sufficient contributing area is selected as potentially suitable for wetland placement.

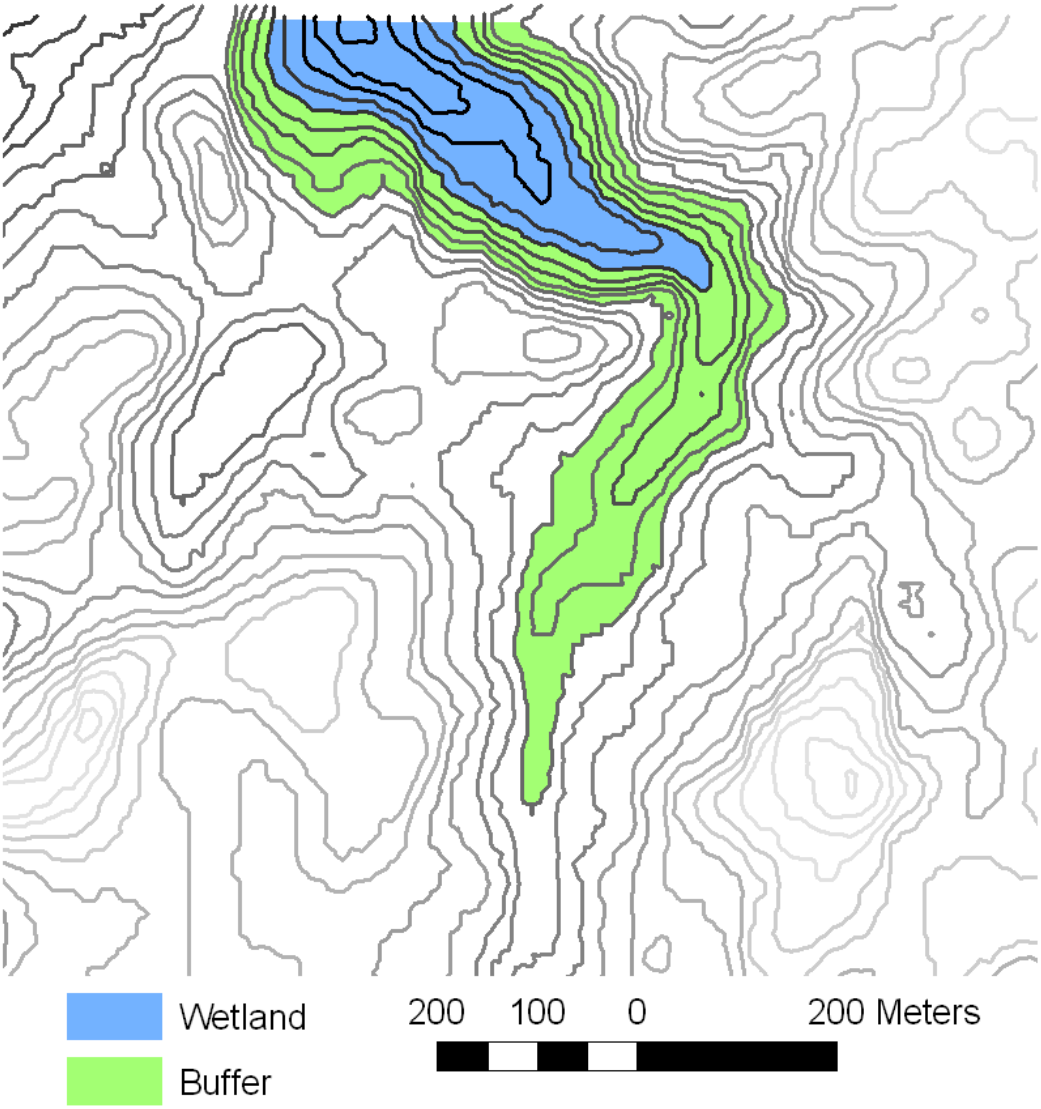

Figure 3.5 An example of topography that lends itself to wetland placement. Topographic contours of 1.2 m (1 ft) are shown from low (dark black) to high (light grey).

3.3.3. Wetland Preliminary Design

3.3.3.1. Desired wetland size is 0.5-2% of its contributing area

Wetlands must be large enough to treat the volume of water they receive from its contributing area. However, cost of wetland construction increases for larger wetlands. Wetland size depends on the size (and corresponding depth of flows) of its contributing area and the degree of treatment desired. In Iowa, the desired size for these wetlands is 0.5-2% of its contributing area. Yet Indiana generally has higher rainfall, which could produce higher flows in Indiana, and wetlands should be designed accordingly. Since Indiana conditions are wetter than those in Iowa, wetlands may need to be relatively larger than the Iowa CREP wetlands in order to get sufficient retention time. This analysis aimed for the desired wetland range of 0.5-2% of its contributing area, but designs were allowed to deviate above the desired range to 4% of the contributing area.

3.3.3.2. No more than 25% of the wetland is greater than 1 meter deep

The primary process of nitrate reduction in a wetland is usually denitrification by bacteria, which takes place most efficiently under oxygen depleted conditions (such as wetlands provide). Denitrifying bacteria need a food source such as wetland plants. Wetland plants can only establish in relatively shallow conditions (Mitsch and Gosselink, 2007). Therefore, most of a wetland should be less than one meter deep to achieve nitrate treatment efficiency.

3.3.3.3. The surrounding buffer must extend 1.2 meters above the wetland, and should not exceed four times the size of the wetland

When a wetland is installed, the local water table level will be at the wetland surface elevation. If the surface of a wetland is above a nearby tile, the tile will be submerged, hindering proper drainage from the cropland drained by that tile.

Therefore the land surrounding the wetland surface should not be farmed, but rather converted to a buffer of vegetation that is tolerant to wet conditions. In Indiana, tiles are usually located approximately one meter below the soil surface. Therefore, wetlands were designed at least 1.2 meters (four feet) below the surrounding land. This means that the buffer strip extends from the edge of the wetland to the farmable land, which is a 1.2 meter rise.

A 1.2 meter rise in elevation around a wetland basin is a rather large topographic feature in the local Indiana landscape. Many locations do not have such a rise, and others may rise so gradually that the buffer strip is excessively large relative to the size of the wetland. A 4:1 ratio of buffer to wetland area limited the buffer size (and therefore cost) while allowing for successful wetland design at numerous sites.

3.3.4. Nitrate Removal Estimation

Nitrate removal takes place by plant uptake and microbial processes in oxygen-poor environments such as wetlands. A number of factors affect the rate of nitrate removal, including hydraulic loading rate/hydraulic retention time, the concentration of nitrate in the inflow water, the temperature of the water, soil conditions, vegetation processes, and flow characteristics in the wetland. The nitrate removal model used in this study was developed from data on Iowa CREP wetlands (Crumpton et al., 2006), which should resemble the behavior of similar wetlands in northeast Indiana. In this model nitrate removal from these wetlands depends only on annual hydraulic loading rate, such that

$$N\ Removed = 103 * HLR^{-0.33} \qquad \text{Eq. 3.1}$$

where *N Removed* is the annual percent of nitrate removed from the wetland and HLR is the annual hydraulic loading rate to the wetland (in m/yr). This simple equation was shown to describe nitrate removal in these Iowa wetlands fairly well.

Hydraulic loading rate for each wetland was determined based on the size of the wetland (m^2), the depth of water contributing to the wetland (m), and size of the wetland's contributing area (m^2). The depth of water contributing to the wetland was calculated to be 0.45 m/yr over the contributing area. This value came from an average of daily flow measurements over two years in four ditches in northeast Indiana, where tile mains empty (Rice, 2003).

3.4. Results and Discussion

3.4.1. Suitable Sites

A total of 19 sites were found to meet all the criteria outlined in Table 3.2. In addition, multiple wetland designs of different wetland dimensions were possible at many sites. For each site with multiple designs, the wetland best addressing the design criteria was chosen. Figure 3.6 shows the spatial locations of these sites in the entire watershed.

3.4.2. Nitrate Removal Estimates

The estimated nitrate removal of wetlands at 19 sites ranged from 24.5 to 45.8 percent, with an average of 32.5 percent (see Figure 3.7). However, the land that is converted to create the wetlands and buffers is taken out of cropland. The conversion of existing cropland into wetland area will further reduce the nitrate input from these areas. If land conversion is taken into account, the average nitrate reduction increased to approximately 39%.

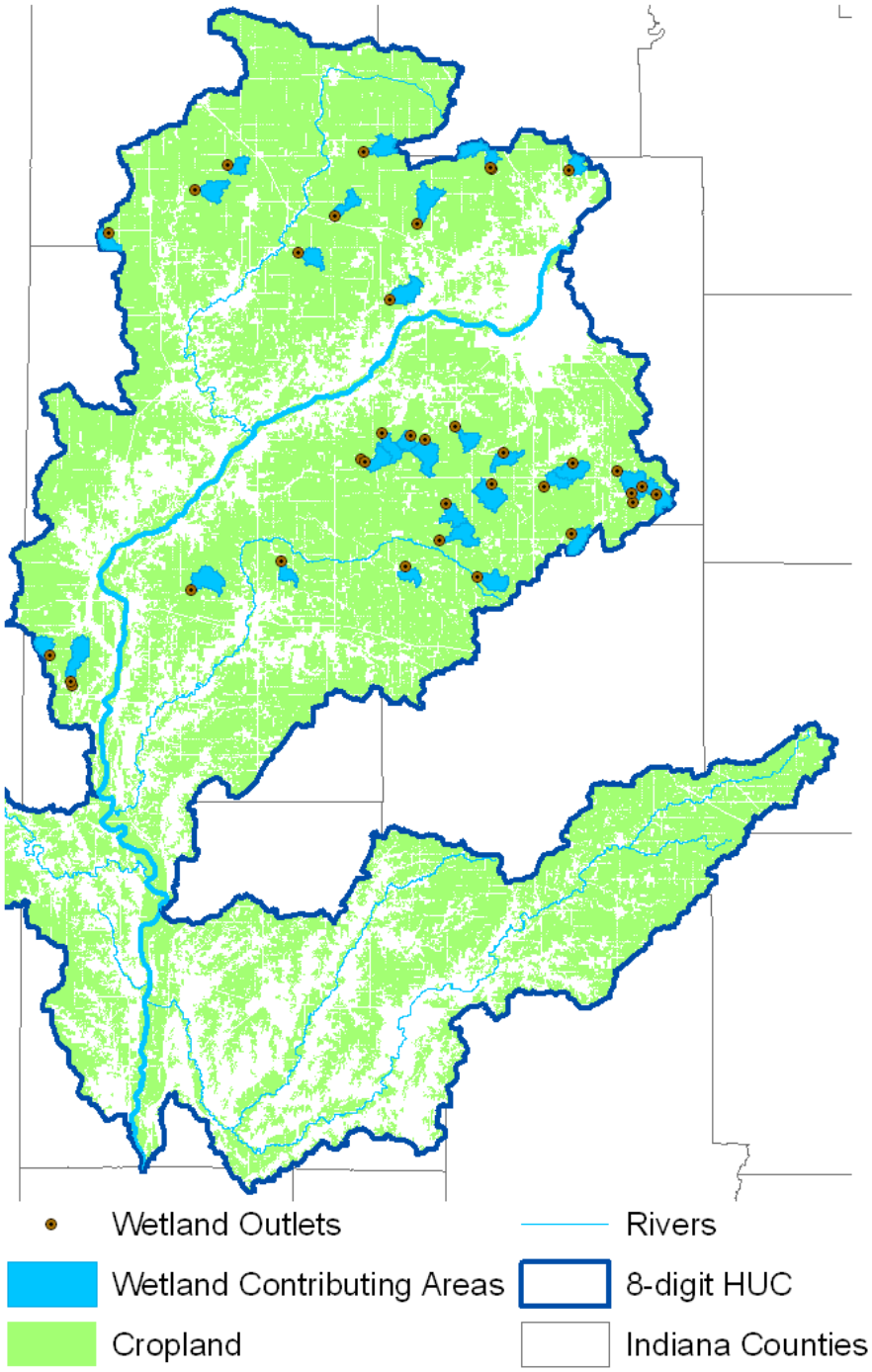

Figure 3.6 Left: Suitable wetland designs were created at 19 sites. The contributing areas of all potential wetlands show the drainage that would be treated by the wetlands, accounting for three percent of the entire tile-drained portion of the 8-digit watershed.

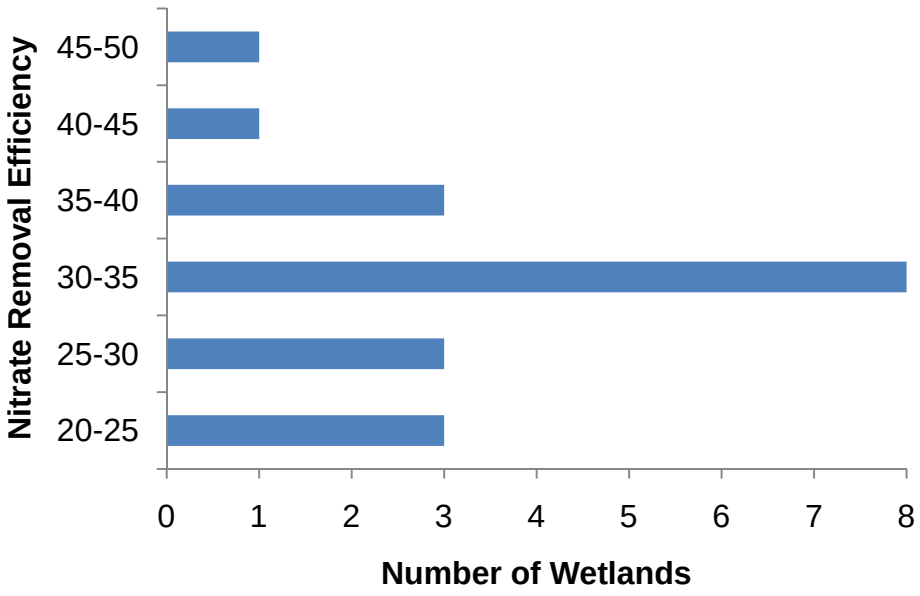

Figure 3.7 Histogram of nitrate removal efficiency for wetlands at all 19 sites.

3.4.3. Impact of Wetlands in the Landscape

If all 19 wetlands were to be placed in this 8-digit HUC, their contributing areas would intercept three percent of all tile-drained land. With approximately 33% efficiency, the wetlands would remove one percent of all nitrate from tile-drained land in the 8-digit watershed. This does not take into account land conversion – the wetland and buffer are taken out of cropland and no longer receive high nutrient application rates – so the estimated nitrate reduction efficiencies would rise to 39%. The land for wetland and buffer placement amounts to 0.14% of all cropland in the watershed, or 0.10% of the entire 8-digit watershed. Therefore, in an agricultural watershed with 67% cropland, 62% of which is tile-drained, these 19 wetlands will require conversion of 0.10% of the watershed, but will treat three percent of nitrate-rich waters from tile-drained lands and remove one percent of all nitrate exported from tile-drained lands.

Such strategic siting of wetlands may result in efficient results, but the impact in the landscape – 1% reduction of nitrate from tile-drained lands – is relatively small compared to the required nitrate reductions of 30-50%. If these wetlands were to make a bigger impact, more of them should be placed in the watershed. In order to place more wetlands, some criteria must be revised. The resulting set of wetlands may have lower nitrate removal efficiencies or higher costs of implementation (for instance if the wetlands were not sized efficiently according to their contributing areas). The sensitivity of each criterion is discussed in the following section, and may be useful in informing a decision of how to best revise the criteria for selection of additional wetlands.

Alternatively, if only the most strategically sited wetlands are to be implemented, then additional best management practices (BMPs) should be implemented in the watershed as well. Some of these BMPs may even work synergistically with wetlands to provide greater reductions in combination. For instance, it is possible that two-stage (“natural”) ditches would perform at greater nitrate

removal efficiency if placed downstream of a constructed wetland, because the wetland would attenuate high flows and allow water to move more slowly through the two-stage ditch system. Or, implementation of controlled drainage, where tile drain flow is reduced for much of the year, would allow for greater storage upstream and attenuation of peak flows entering a constructed wetland, increasing its performance. Most likely, one or two BMPs cannot be the whole solution to the nitrate reductions required in the Midwest USA. Instead, a wide range of BMPs should be studied and considered for placement strategically within the landscape.

3.4.4. Sensitivity of Each Criterion

In locating sites for wetland placement, each step in the analysis narrowed the set of potentially suitable sites. A sensitivity analysis was used to analyze each criterion's influence on the number of suitable sites identified for the watershed. This is not the traditional use of "sensitivity," where one parameter is varied while others are held constant. Instead, it is the contribution of each criterion to narrowing potentially suitable sites at the time it is used in the method. Note that the order of the analysis is important in determining the sensitivity of each parameter. If some parameters have a relatively small effect on narrowing the set of suitable sites, they may be considered unnecessary and the methodology can be further simplified. Or, the small impact may be a function of the Indiana landscape, and the factor with little influence may be highly influential in another setting with different landscape characteristics. Alternatively, if a particular criterion has disproportionately high impact on the set of suitable sites, then we might consider expanding the criterion so that it will result in more sites as suitable. Below we discuss the effect that each criterion makes in the methodology.

3.4.4.1. Sufficient contributing area from tile-drained land

The two steps involved in satisfying this criterion – Step (1): contributing area is 2-8 km^2 and Step (2): contributing area drains at least 2 km^2 tile-drained land – both narrowed the result. Step (1) resulted in 947 flow paths (a total of 1611 km) that had watersheds of 2-8 km^2. Because this is the first step in the entire methodology, it is not considered to have limited the sites (0% of sites eliminated in Table 3.3). Step (2) narrowed the number and total distance of flow paths 54% and 53%, respectively. In total, using this criterion resulted in a 54% (53%) elimination of the number (total distance) of suitable flow paths. Clearly this criterion is responsible for considerable narrowing of suitable sites.

The location of suitable flow paths is also interesting. After Step (1), flow paths occur throughout the entire watershed, as would be expected (Fig 3.2). Step (2) appears to preferentially eliminate sites near the main stem of the Wabash River and Little and Big Raccoon Creeks. This spatial selection is probably due to a combination of less agricultural land use in these river valleys, as well as fewer poorly drained soils. The result is a notable selection for the headwaters of the landscape. Altering the criteria could provide a greater number of resulting sites. Specifically, a much larger set of solutions could be found if smaller contributing areas are accepted (e.g. 250+ acres), and if land that is not tile-drained is considered. However, these would likely decrease the efficiency of nitrate removal by wetlands; the former, by treating comparably smaller contributing areas, and the latter through dilution of nitrate-rich tile drainage with less concentrated surface runoff that leads to higher hydraulic loading rates.

Table 3.3 Sensitivity of results to wetland siting and design criteria. Note that the order of the analysis is important in determining the sensitivity of each parameter.

	% Sites Eliminated	**# Sites Remaining**	**% Distance Eliminated**	**Distance Remaining**
Wetland Siting Criteria				
1. Sufficient contributing area from tile-drained land	54%	431 paths	53%	754 km
Step 1: Contributing area 500-2000 acres	*0%*	*947 paths*	*0%*	*1611 km*
Step 2: Drains >500 acres tile-drained land	*54%*	*431 paths*	*53%*	*754 km*
2. Excluding streams	76%	105 paths	88%	86 km
3. On cropland	12%	92 paths, 113 sites		
4. Wetland topography	73%	31 sites		
Wetland Design Criteria				
5. Wetland size (0.5-2 (4) % of contributing area)	23%	24 sites		
6. Deep wetland size (<25% of wetland size)	0%	24 sites		
7. Buffer size (1.2 m elevation rise, <4 times wetland size)	21%	19 sites		

3.4.4.3. Wetland must not intercept a stream or open waterway

This criterion has the most clear geographic bias of all the criteria, as it eliminates all sites in the southern portion of the watershed. Perhaps this is due to the hillier land in the south, where large contributing areas are more likely to be located in a stream. It appears that in the south (Parke and adjacent counties) the stream network is denser, perhaps because of hillier terrain or decreased tile drainage of this land. This criterion reduced the number and total distance of flow paths more severely, by 76% and 88%, respectively. Removing or altering this criterion and allowing for placement of wetlands within open streams would result in a vast increase in potentially suitable sites.

3.4.4.4. Wetland is on cropland

When these flow paths were intersected with cropland, 113 fields were selected. These fields contained 92 flow paths, reducing the number of suitable locations by only 12%. In this HUC this criterion has very little impact on the overall selection of suitable sites. The likely reason for this is found in redundancy in the model. In Criterion 1, the contributing area is narrowed to those with considerable tile drainage. This tile drainage occurs only on agricultural fields. Therefore, suitable flow paths are already located in the vicinity of a high concentration of cropland. Due to this redundancy, this criterion may be considered for removal from the methodology. However, in the case of CREP wetlands, funding is available only for placement of wetlands directly on cropland. For these particular wetlands, this criterion may have small impact but it is still critical for selecting acceptable sites.

3.4.4.5. Topography lends itself to wetland placement

Suitable topography narrowed the search considerably, a total of 73% reduction in suitable sites. The greatest topographical limitation was finding sites with more than a 1.2 meter elevation rise surrounding the wetland, which may be

more common in hillier terrain. The entire effect of topography (and other features in the landscape that cannot be crossed, such as roads) was quite large in narrowing the sites. Clearly this criterion is important in the methodology.

3.4.4.6. Wetland design criteria (5-7)

The three wetland design criteria were applied to 31 designed sites, and only 19 sites were found to meet (or nearly meet) the criteria. Sizing the wetland in proportion to its contributing area (Criterion 5) caused elimination of 23% of the sites. All of these sites were eliminated because wetlands were too small for their contributing areas, not too large. This is partially due to the expansion of the range from wetland size of 0.5-2% to 0.5-4% of its contributing area, which allowed for selection of larger wetlands. No sites were eliminated for amount of deep wetland (Criterion 6), for in the relatively flat landscape, the shallow bottom to the “bowl-shaped” topography was not hard to find. A further 21% of sites were eliminated due to buffers larger than the 4:1 ratio. As expected, the flat topography made the relatively steep rise necessary for the buffer region difficult to locate. Overall, these design criteria eliminated 39% of the wetland designs. While by this measure they do not appear to have a large narrowing effect, in fact the narrowing by these parameters was primarily executed in Criterion 4, looking for suitable wetland topography. In other words, in most locations where a wetland could possibly be designed with a surrounding buffer, at least one design was found to be suitable.

3.4.5. Situational Considerations

The results of this analysis are sensitive to the criteria used to determine wetland sites. In this watershed, the selection was most sensitive to the contributing area, exclusion of streams, and topography criteria. The contributing area criterion was only satisfied in heavily agricultural lands, where large contributing areas were extensively tile-drained. Exclusion of streams resulted in suitable

sites only found in relatively flat regions, where large contributing areas could be found within tile drains before emptying into open waterways. Criteria did not have the same limiting effect across this 8-digit watershed, as in the southern part of the watershed large contributing areas were only found within streams, likely due to hillier terrain. Results of this work could have been quite different given another set of criteria. If this methodology were applied in another location, with different terrain or climate (e.g. hilly, arid), or different goals (e.g. phosphorus, water quantity), care would have to be taken to design the correct criteria to determine the number of suitable sites (e.g. the most efficient, targeted approach, or inclusion of as many sites as possible).

3.5. Summary and Conclusion

This analysis has shown that, in a heavily tile-drained agricultural 8-digit watershed, placement of 19 wetlands will require conversion of 0.10% of the watershed but will treat three percent of nitrate-rich waters from tile-drained lands, removing one percent of all nitrate exported from tile-drained lands. This impact is not negligible, and the targeted watershed-scale approach leads to more efficient solutions. However, it is clear that this approach alone will not achieve the magnitude of reductions called for in the 2008 EPA Action Plan. Rather, a suite of best management practices will be required to meet water quality goals in the agricultural Midwest.

CHAPTER 4. ESTIMATING NITRATE REMOVAL FROM HEADWATER CONSTRUCTED WETLANDS USING THE SOIL AND WATER ASSESSMENT TOOL (SWAT) MODEL

4.1. Introduction

A particular concern for the agricultural Midwest is the hypoxia in the Gulf of Mexico, which is primarily caused by high nitrogen exports from agricultural lands in the Upper Mississippi and Ohio River Basins (Alexander et al., 2008; Goolsby et al., 1999). Nitrate losses in tile-drained lands, characteristic of west central Indiana, are generally greater than in un-tiled lands (Hickey and Doran, 2004; David et al., 1997; Drury et al., 1993; Gentry et al., 2009). Constructed wetlands have been proposed as a partial solution to intercept nitrate losses from agricultural fields (Mitsch et al., 2001; Mitsch and Day, 2006). However, constructed wetlands should be strategically placed in the landscape to maximize water quality benefits (Crumpton, 2001).

The Soil and Water Assessment Tool (SWAT) can be used to estimate nitrate removal that might be provided by constructed wetlands placed in a watershed. SWAT is a physically based watershed-scale model that simulates upland and in-stream water and chemical transport. Its major upland components include climate, hydrology, nutrients, soil erosion/sedimentation, plant growth, and management practices. In-stream processes include routing of flow, sediment, and nutrients, as well as nutrient transformations. Wetland nitrate removal efficiency depends upon a variety of design and environmental factors. In general wetlands are more efficient at removing nitrate from highly concentrated flows, during warm months, and under lower hydraulic loading rates. SWAT has been used extensively to model flow and nutrient transport in agricultural

watersheds, yet few studies have specifically analyzed wetland nutrient dynamics using SWAT (see full discussion of SWAT in Chapter 2.4.1).

4.1.1. Objectives

The goals of this work were to estimate the nitrate reduction impact of strategically placed constructed wetlands in a tile-drained agricultural watershed. The specific objectives were to: 1) Design wetlands in the headwaters of the watershed, 2) Setup and calibrate a SWAT model using measured flow and water quality data available for one year at the outlet of the watershed, and 3) Simulate wetlands in SWAT and estimate nitrate removal provided by each wetland.

4.2. Description of Study Site

The study was conducted in the Little Pine Creek watershed, located northwest of Lafayette, IN (Figure 4.1). The watershed is 5820 ha in size, and 86% of its land area is cropland, primarily corn and soybeans. Soils are generally poorly drained, with 42% characterized as hydric soils. A total of 81% of cropland in the watershed is estimated to be tile-drained (see Chapter 3 for explanation of estimation). Topography is generally flat with an average slope of 0.9 percent.

The outlet point for the Little Pine watershed was a recently installed US Geological Survey gaging station (033356786 at Montmorenci, IN). Daily stream flow data have been collected at this station since 3/21/2009 (available at http://waterdata.usgs.gov/nwis/uv?033356786). Weekly water samples were collected and analyzed for nitrite plus nitrate, ammonia, total phosphorus, dissolved organic carbon, total suspended solids, *E. coli*, and total coliforms as a part of another on-going project since 5/1/2009 (unpublished data).

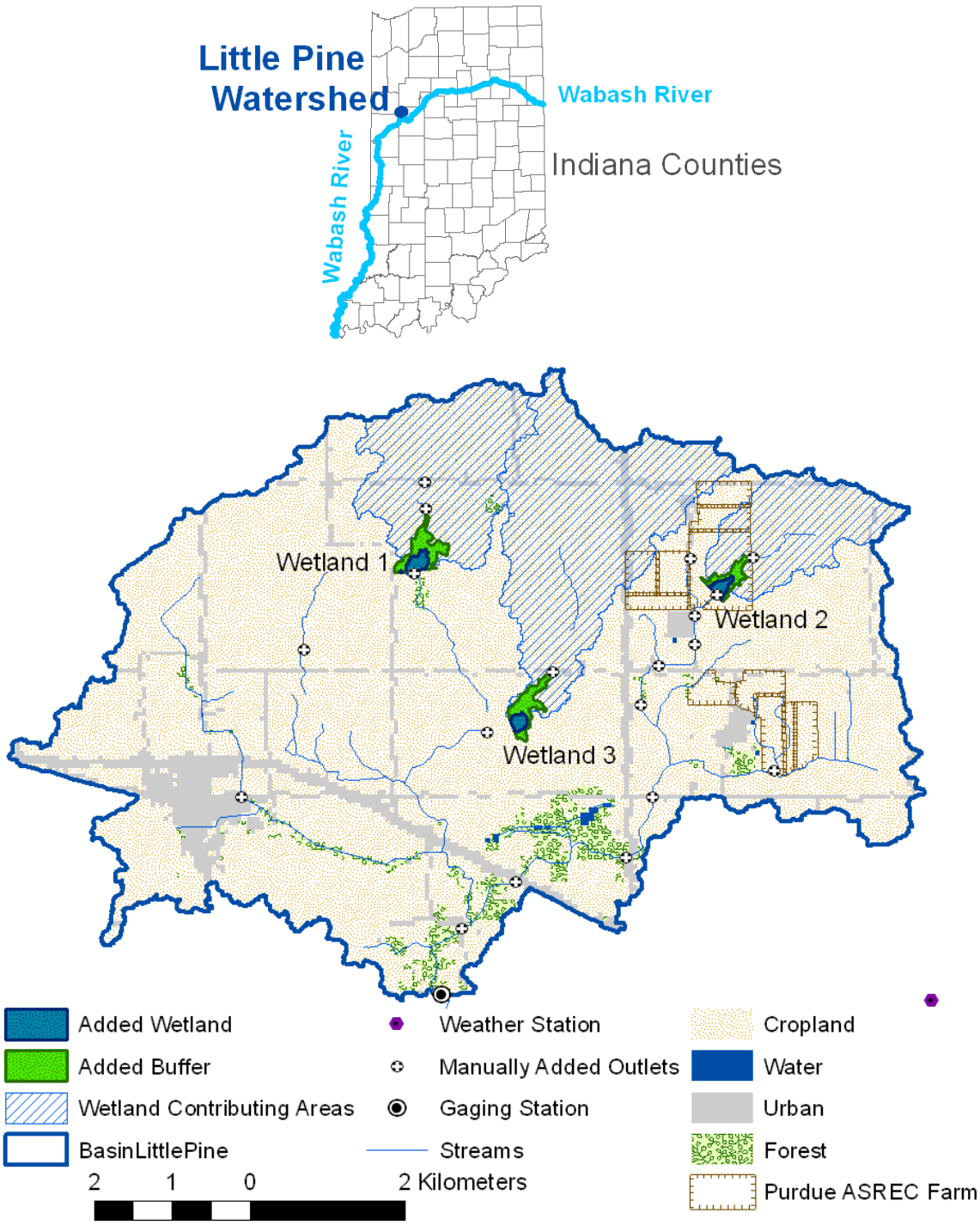

Figure 4.1 Little Pine watershed with added wetlands and their contributing areas. Most common land use distribution in the watershed is shown as well.

4.3. Materials and Methods

4.3.1. Data Layers

A number of datasets (Table 4.1) were used to set up the SWAT model and alter input files to accurately represent the study region. Many of these data layers are publicly available and can be acquired for locations throughout the United States. Because the study region is relatively small for implementation of the SWAT model, high resolution datasets were used whenever possible.

4.3.2. Wetlands

Three wetland sites were designed (Figure 4.1) according to methodology of strategic siting as described in Chapter 3. Two of these wetland sites – Wetlands 1 and 2 – did not match the criteria entirely, as they were located within an open ditch/stream. Wetlands were sized according to the existing topography and the size of their contributing areas. Wetland 1 was 1.4% of its contributing area, wetland 2 was 1.7% of its contributing area, and wetland 3 was 0.6% of its contributing area. The three wetland contributing areas were a total of 1246 hectares, 21.4% of the entire 5820 ha watershed.

4.3.3. SWAT Model Scenarios

The SWAT model was set up for the Little Pine watershed, which is within the 8-digit HUC described in Chapter 3 (Figure 4.1). To determine the difference these wetlands could make in the landscape, the SWAT model was set up with two scenarios: first a “control” scenario with current land use, and second a “wetland” scenario, which had land use altered to incorporate wetlands and buffer regions.

Table 4.1 Data layers used in SWAT modeling

Data layer	Source dataset	Use in SWAT model
Elevation	National Elevation Dataset (NED), with one-third arc second resolution	Used for streams definition and watershed delineation
Streams	National Hydrography Dataset (NHD), high resolution streams, downloaded June, 2009	Burned in to elevation data to force flow paths to follow streams
Land use	National Agricultural Statistics Service (NASS) 2009 and NLCD 2001	Combined datasets used for land use approximation and creating HRUs
Soils	Soil Survey Geographic (SSURGO) Database	To model soil conditions and create HRUs
Climate	National Climatic Data Center (NCDC), for period of 01/01/2009 – 11/30/2009, and from Indiana State Climate Office for 11/30/2009-03/01/2010	Precipitation, and minimum and maximum temperature
Stream flow	U.S. Geological Survey (USGS) Real Time Water Data at gaging station 033356786	For calibration of flow
Nitrate concentration	Weekly concentrations (mg/L) of nitrate plus nitrite at gaging station 033356786 (unpublished data)	For calibration of nitrate
Manure Applications	Detailed data on loads and timing of manure applications to Purdue's Animal Sciences Research and Education Center (ASREC)	Used to estimate application rates and manure nutrient composition on the farm
CAFO animal numbers	Animal numbers and sizes for one CAFO in the watershed, 1999-2000	Used, along with ASABE Standard, to estimate manure applications near CAFO

National Hydrography Dataset (NHD) high definition streams were burned into the elevation data, which had raster resolution of 10 m by 10 m. The subwatersheds were delineated with a stream threshold of 50 hectares, meaning flow paths with at least 50 hectares of contributing land are modeled to be streams. Subwatershed outlet points were also manually added for each wetland inlet and outlet, and at locations where flow and/or water quality data were collected in previous studies in this watershed. A total of 86 subwatersheds were delineated in this watershed. For the wetland scenario, NASS land use was altered to include all three wetlands as wetland land use (WETL), and their corresponding buffer regions as pasture (PAST). Hydrologic response units (HRUs) were defined by a single slope, with thresholds of 2 ha land use, 10 ha soils, and 100 ha slope, resulting in a total of 354 HRUs. These HRUs are regions with similar land use and soils, which are combined together within a subwatershed.

Climate data of daily precipitation and max/min temperature were obtained from a nearby weather station, located approximately 2 km south-east of the watershed boundary (Co-operative Daily Station 129430, located in West Lafayette 6NW). Other climate inputs were simulated by the SWAT model. The SWAT model was run from 01/01/1997-12/31/2010. The first three years were a warm-up period, and data was analyzed for the following 11 years.

4.3.4. Crop Management Inputs

Crop management input files were altered according to a number of data sources, and these practices were assumed to occur throughout the study region at the following dates. Corn and soybean crops were generally assumed to have a two-year corn-soybean rotation, with corn and soybean management practices based on knowledge of Indiana conditions and the Tri-State Recommendations for fertilizer applications (Vitosh et al., 1995). For corn, planting was assumed to occur on May 6, the same day as disk plowing. The previous fall, a chisel

plowing occurred on October 15. Two weeks prior to the planting of corn, 173 kg/ha anhydrous ammonia (NH_3-N) and 64 kg/ha P_2O_5 and a pesticide application of 1.41 kg/ha atrazine were input on April 22. Harvest was assumed to occur on October 14. Soybeans were planted with no-till on May 24, and were fertilized two weeks earlier with 51 kg/ha P_2O_5. All soybean fields were assumed to be harvested on October 7.

A number of fields in the watershed are located in the Purdue Animal Sciences Research and Education Center (ASREC), where manures are known to be the sole fertilizer application. The primary hard pack (dewatered) manures applied were dairy and beef, and lagoon effluents were derived from dairy, poultry, swine, and beef waste. Crops within the ASREC farm were given manure applications in the simulations based on detailed application rates and timing available for the period of 2008-2009. These application rates were aggregated by month and averaged over the two years to derive an average monthly application rate of a particular manure to a given field. These manures were applied on the 15th of each month in all the HRUs in a subwatershed intersecting that field. Inorganic fertilizers were applied in other cropland areas which did not receive animal manure. A number of manure analysis reports on the ASREC manures performed in 2008-2009 were used to estimate water content and nutrient composition of the manures. This dataset included detailed nutrient composition information for beef hard pack and the dairy lagoon, which were added to the SWAT fertilizer file (fert.dat). Moisture content of the hard packs was estimated using water content of beef and sheep hard pack. Lagoon moisture content was estimated from data on the dairy lagoons. Moisture contents were used to convert between wet weights of manures, in the dataset, to dry weights, which is required by SWAT. Consult the Appendix for further detail on these calculations.

Another special case was a swine confined animal feeding operation (CAFO) located in the north-west part of the watershed. To estimate manure applications on the surrounding fields, the maximum numbers of particular sizes of swine in the farm were found from a permit for the years of 1999-2000. Using the ASABE Standard for Manure Production and Characteristics, the dry weight of manure produced was calculated (ASABE, 2005). This procedure is detailed in the Appendix. Specific dates for the manure application were not available. It was assumed that the manure was applied on the 15th of each month to the three subwatersheds closest to the farm. When the swine manure was not adequate to meet the crop nutrient requirements, supplemental inorganic fertilizers described in the preceding sections were used.

4.3.5. Calibration

The SWAT model was calibrated at the Little Pine watershed outlet for both flow and nitrate from 03/21/2009 to 05/31/2010, the time period for which flow and water quality data were available at the outlet. Simulated flow was calibrated at the daily time scale using the measured flow data. Nitrate was calibrated as an average monthly load (kg), derived from weekly water quality measurements at the gauging station and the LOADEST model (Runkel et al., 2004) using the following regression equation:

$$Ln(load) = a_0 + a_1 \times lnQ + a_2 \times (lnQ)^2 \qquad \text{Eq. 4.1}$$

where *load* is the daily load of nitrate (kg), *Q* is the daily flow rate (cms), *lnQ* is centered by its mean value, and a_0, a_1, and a_2 are regression coefficients. The resulting coefficients based on the Adjusted Maximum Likelihood Estimation (AMLE) were a_0 = 5.0779, a_1 = 1.3949, and a_2 = -0.0687. The AMLE fit the measured data well, with an overall R^2 = 0.86.

The objective functions used in evaluating the adequacy of the model calibration included the coefficient of determination (R^2), the Nash-Sutcliffe coefficient (E_{NS}), and the relative mean error (*RME*) defined as:

$$R^2 = \left[\frac{\sum_{i=1}^{n}(O_i - \bar{O})(P_i - \bar{P})}{\sqrt{\sum_{i=1}^{n}(O_i - \bar{O})^2 \sum_{i=1}^{n}(P_i - \bar{P})^2}}\right]^2 \qquad \text{Eq. 4.2}$$

$$E_{NS} = 1 - \frac{\sum_{i=1}^{n}(O_i - P_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2} \qquad \text{Eq. 4.3}$$

$$RME = \frac{\sum_{i=1}^{n}(P_i - O_i)}{\sum_{i=1}^{n}(O_i)} \qquad \text{Eq. 4.4}$$

where O_i is the observed value on a given day (month for nitrate), P_i is the value predicted by SWAT on a given day (month for nitrate), $\bar{O}$ is the average observed value, $\bar{P}$ is the average predicted value, and *n* is the number of days (months) over which these values are compared. R^2 varies from 0 to 1, with 1 representing a perfect regression fit of the data. E_{NS} varies from negative infinity to 1, with 1 representing a perfect fit. *RME* varies from negative infinity to positive infinity, and indicates whether there is a negative bias (underprediction) or a positive bias (overprediction), along with the magnitude of the bias.

To calibrate the SWAT model, flow was first calibrated to fit the observed data, and then nitrate was calibrated second because nitrate loads are dependent on flow. Parameters in the model that are likely to have an impact on the flow and nitrate are considered for adjustment within physically reasonable limits. In this study, default values for a number of potentially sensitive parameters were dictated by past work in modeling nitrate flows from tile-drained lands in Indiana (Sui, 2007). Sui (2007) calibrated a number of parameters for flow and nitrate (Table 4.2). These calibrated values were used as defaults in the control SWAT model. Then parameters that were most sensitive in her study (CN2, NPERCO, SOL_ORGN, CMN, SOL_NO3, DEP_IMP) were adjusted in model calibration.

4.3.6. Nitrate Removal Estimation

After calibrating the control model, the calibrated parameters (Table 4.2) were applied to the wetlands model. Additionally, a number of parameters specific to wetlands were estimated for each of the three wetlands (Table 4.3). In SWAT, default parameters for wetlands in the wetland files (.pnd) are initially all 0, presuming that wetlands do not impact flow or water quality. To accurately simulate the hydraulic characteristics of each wetland, the wetland surface area and volume were estimated from the original shapefiles and elevation data. To simulate nitrogen removal in each wetland, SWAT uses a parameter called the "apparent nitrogen settling rate," (NSETL) which combines all forms of nitrogen removal in a wetland. NSETL was estimated from data taken on a similar, nearby wetland (Poole, 2006). This wetland pumps water from a nearby ditch that drains a large area of cropland. Notable differences between this wetland and the SWAT simulated wetlands include the lack of treatment in winter when pumps are turned off, and the steady inflow to the wetlands by pumps rather than the variable flow provided by direct access to the tile network/stream. To estimate NSETL for one year (*dt*), flow-weighted average nitrate concentration (*C*), the mass of nitrogen removed in a year (*MSETL*), and the wetland surface area (A_s) were obtained from over a year of data (Poole, 2006). These were input into SWAT's equation to solve for NSETL: $MSETL = NSETL \times C \times A_s \times dt$. The resulting NSETL was 39 m/yr.

At each wetland outlet, daily flow and nitrate loads were compared between the control and the wetland model. Daily flow and nitrate loads were aggregated to the annual time-scale, and reductions determined by SWAT were compared to reduction estimates from the simple regression model (Crumpton et al., 2006) as discussed in Chapter 3. Daily data were also averaged by month over the 11 years of simulation to determine the average monthly nitrate reduction expected from each wetland.

4.4. Results and Discussion

4.4.1. Calibration

The only parameter found to significantly impact flow was the surface runoff curve number (CN2), which was calibrated to -25 percent of its original value in order to reduce surface runoff on tile-drained fields (Table 4.2). CN2 should be lowered in tile-drained fields, because drained fields have lower surface runoff and greater infiltration. The only factor adjusted for nitrate calibration was SDNCO, which was returned to its default state, meaning no denitrification occurs in the soil (Table 4.2).

Calibration of flow resulted in $R^2 = 0.70$, $E_{NS} = 0.59$, and $RME = 0.12$. Calibration of nitrate resulted in $R^2 = 0.48$, $E_{NS} = 0.66$, and $RME = 0.23$. These statistics are generally within the recommended ranges: E_{NS} greater than 0.50 (Santhi et al., 2001; Moriasi et al., 2007; Engel et al., 2007), R^2 greater than 0.60 (Santhi et al., 2001; Engel et al., 2007), and *RME* fairly close to 0. The positive *RME* for flow and nitrate means that both flow and nitrate are generally over predicting slightly. Figure 4.2 shows time-series plots of the flow and nitrate at the watershed outlet during the calibration period. The predicted streamflow closely followed the trend shown by the measured flow for 2009, but in the winter the model generally underpredicted peak and base flow. Nitrate was difficult to capture in the cold months and, like the flow, shows significant underprediction during this time. No further improvements could be made by altering the other relevant parameters. The remaining error may be due to the sampling period of just over one year for nitrate and flow data, and the model performance could likely be improved if additional data were available for model calibration.

Table 4.2 Calibrated parameters[1] for flow and nitrate were used as default values in this study.

Parameter	Description	Default Value	Calibrated Value (Sui)	Calibrated Value Final	Input File
CN2	SCS runoff curve number for moisture condition II	varies	Reduced by 9%	Reduced by 25%	.mgt
SDNCO	Threshold value of water content for denitrification	1.1	0.9	1.1	.bsn
DDRAIN	Depth of tile drains (mm)		1000	1000	.mgt
TDRAIN	Time to soil field capacity (hr)		24	24	.mgt
GDRAIN	Lag time of flow in tiles (hr)		48	48	.mgt
SOL_K	Soil saturated hydraulic conductivity (mm/hr)	varies	Increased by 15%	Increased by 15%	.sol
DEP_imp	Depth to restricted layer (mm)	6000	3000	3000	.hru
NPERCO	Nitrate percolation coeff.	0.2	0.97	0.97	.bsn
CMN	Rate of humus mineralization of active organic nutrients	0.0003	0.019	0.019	.bsn
RS4	Rate for organic N settling in the reach at 20 deg C (1/day)	0.05	0.001	0.001	.swq
BC1	Rate for biological oxidation of NH_4^+ to NO_2^- in reach (1/day)	0.55	0.95	0.95	.swq
BC2	Rate for biological oxidation of NO_2^- to NO_3^- in reach (1/day)	1.1	2.0	2.0	.swq
BC3	Rate for hydrolysis of organic N to NH_4^+ in reach (1/day)	0.21	0.4	0.4	.swq

[1] Calibrated values used by Sui (2007).

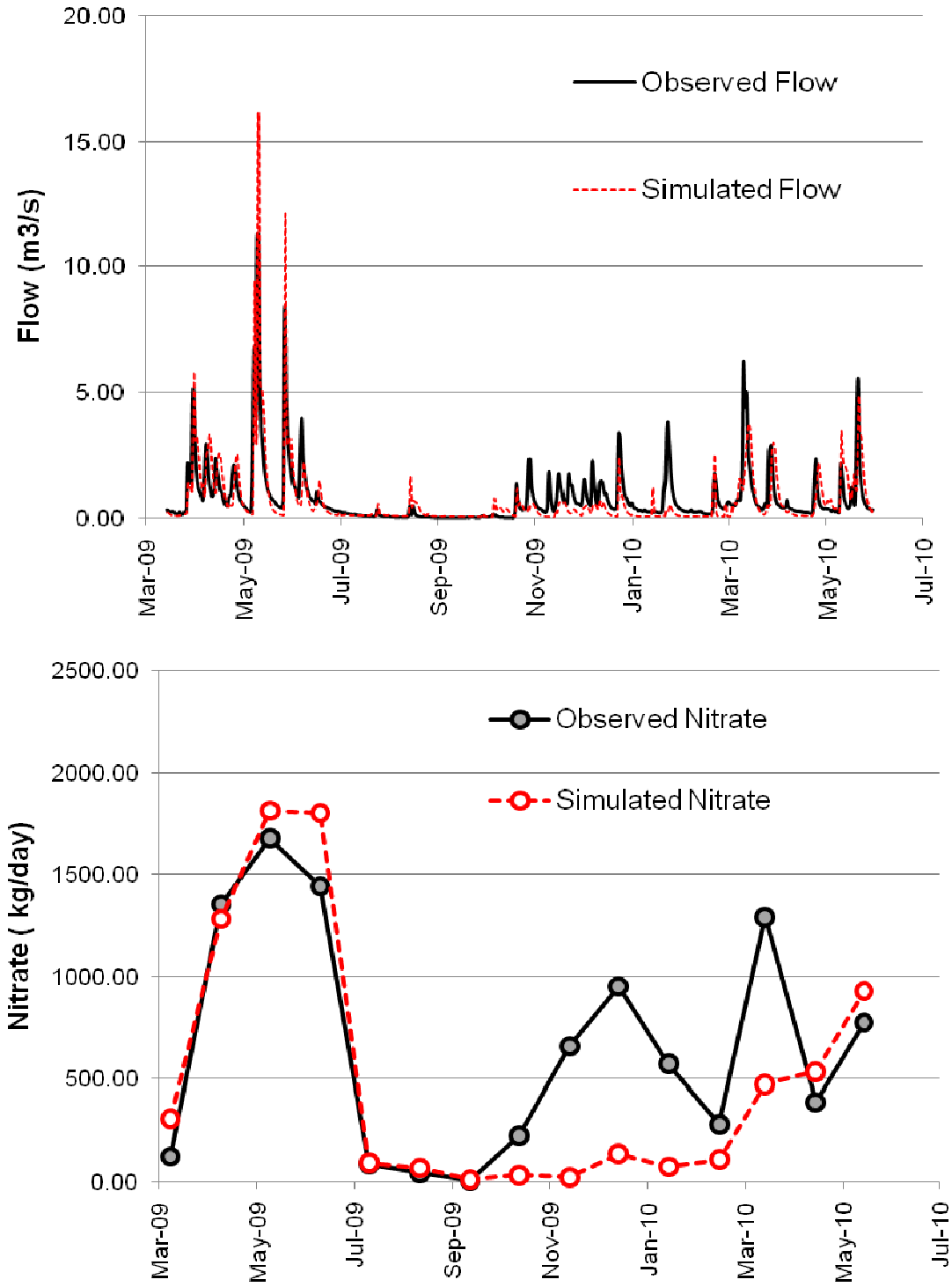

Figure 4.2 Flow and nitrate time-series at watershed outlet during the calibration period.

4.4.2. Nitrate Removal Estimation

Nitrate removal efficiencies were estimated by both the SWAT model and the model developed by Crumpton et al. (2006) (Table 4.4). All three wetlands provided considerable nitrate removal at the annual and monthly timescales (Table 4.4, Figures 4.3-4.4). Figure 4.3 shows the annual average nitrate reductions expected by the two different models. Average nitrate concentrations entering the wetlands estimated by SWAT were 4.95-7.05 mg/L, and average nitrate concentrations in the wetland effluent ranged from 2.97-5.45 mg/L. The two nitrate removal estimation models agree fairly well in their average nitrate removal efficiencies for each wetland over the 11-year study. This demonstrates that the Crumpton et al. (2006) model provides a good first approximation of nitrate removal. However, the two models do not agree well for wetland 3. The primary reason for this discrepancy is that the Crumpton model takes into account wetland surface area but not wetland volume in its calculation of nitrate reductions. Wetland 3 has a surface area comparable to the other wetlands but it is much shallower and has a smaller volume (Table 4.3), leading to over-estimation of nitrate reductions by the Crumpton model. Comparison of the two models therefore suggests that the Crumpton model predicts nitrate removal quite well for wetlands with a bowl-shaped structure, and not as well for shallow wetlands.

When considering smaller time-scales, the two models show greater differences. The SWAT model produces somewhat larger annual fluctuations than the Crumpton model, resulting in higher standard deviations averaged for the 11 years (Figure 4.3, Table 4.4). Factors affecting these annual fluctuations include temperature, influent nitrate concentration, and the timing of flows. The SWAT model is better able to provide variation in nitrate removal that can be expected at annual and smaller time scales.

In the control scenario with no wetlands, average annual flows at the watershed outlet and the wetland outlets were 21,626,516 m^3 (watershed outlet), 1,239,736 m^3 (Wetland 1 outlet), 777,793 m^3 (Wetland 2 outlet), and 2,264,860 m^3 (Wetland 3 outlet). Average annual nitrate loads were 159,108 kg (watershed outlet), 6,972 kg (Wetland 1 outlet), 3,741 kg (Wetland 2 outlet), and 15,261 kg (Wetland 3 outlet). The addition of wetlands consistently increased annual flows, with averages 21,808,177 m^3 (watershed outlet), 1,311,724 m^3 (Wetland 1 outlet), 820,560 m^3 (Wetland 2 outlet), and 2,331,421 m^3 (Wetland 3 outlet). Corresponding annual nitrate loads were decreased, with averages 150,985 kg (watershed outlet), 4,602 kg (Wetland 1 outlet), 2,434 kg (Wetland 2 outlet), and 12,710 kg (Wetland 3 outlet). These summary statistics show that Wetland 3 is clearly receiving much larger loads and flows than the other two wetlands, and yet Wetland 3 is of smaller surface area and much smaller volume than the others, so it is not surprising that it has significantly lower nitrate removal efficiency. In fact when nitrate load reduction is considered rather than percent, Wetland 3 removes the greatest load of nitrate (2,551 kg compared to 2,370 kg for Wetland 1 and 1,307 kg for Wetland 2). Therefore, despite its apparent low nitrate removal efficiency, Wetland 3 may be the most cost effective of the three wetlands if few wetlands and other best management practices are to be placed in the watershed.

Wetlands may not always be a sink of nitrogen, but at times may become a source, and this may depend on season. Figure 4.4 shows the SWAT simulated average monthly nitrate loads leaving wetlands compared with the control scenario where no wetlands were present in the watershed. On average, the wetlands reduced nitrate loads in every month with flow (there are periods in the winter when nitrate loads are already low due to low flows). However, comparing control and wetland nitrate flows on the daily time-scale for 11 years reveals that on average Wetland 1 acts as a nitrate source 10% of the time (37 days/year), Wetland 2 is a source 4% of the time (16 days/year), and Wetland 3 is a source

Table 4.3 Wetland parameters for three wetlands (input to the .pnd file).

Parameter	Description	Wetland 1	Wetland 2	Wetland 3
WET_FR	Fraction of subwatershed area that drains to wetland	1	1	1
WET_NSA	Surface area of wetland at normal water level (ha)	5.391	4.143	3.672
WET_NVOL	Volume of wetland at normal water level (10^4 m^3)	2.636	2.010	0.705
NSETLW	Nitrogen “apparent settling rate” (m/yr)	39	39	39

Table 4.4 Average annual nitrate removal performance of three wetlands at wetland outlets, averaged over 11 years. The control scenario has no wetlands implemented in the watershed.

	Percent Nitrate Reduction (%)				Nitrate Concentration (mg/L)	
	Crumpton Model		SWAT Model		Contol	Wetlands
	Average	*St. Dev.*	*Average*	*St. Dev.*	*Average*	*Average*
Wetland 1	37%	3%	34%	4%	5.82	3.51
Wetland 2	40%	4%	36%	8%	4.95	2.97
Wetland 3	27%	2%	17%	4%	7.05	5.45

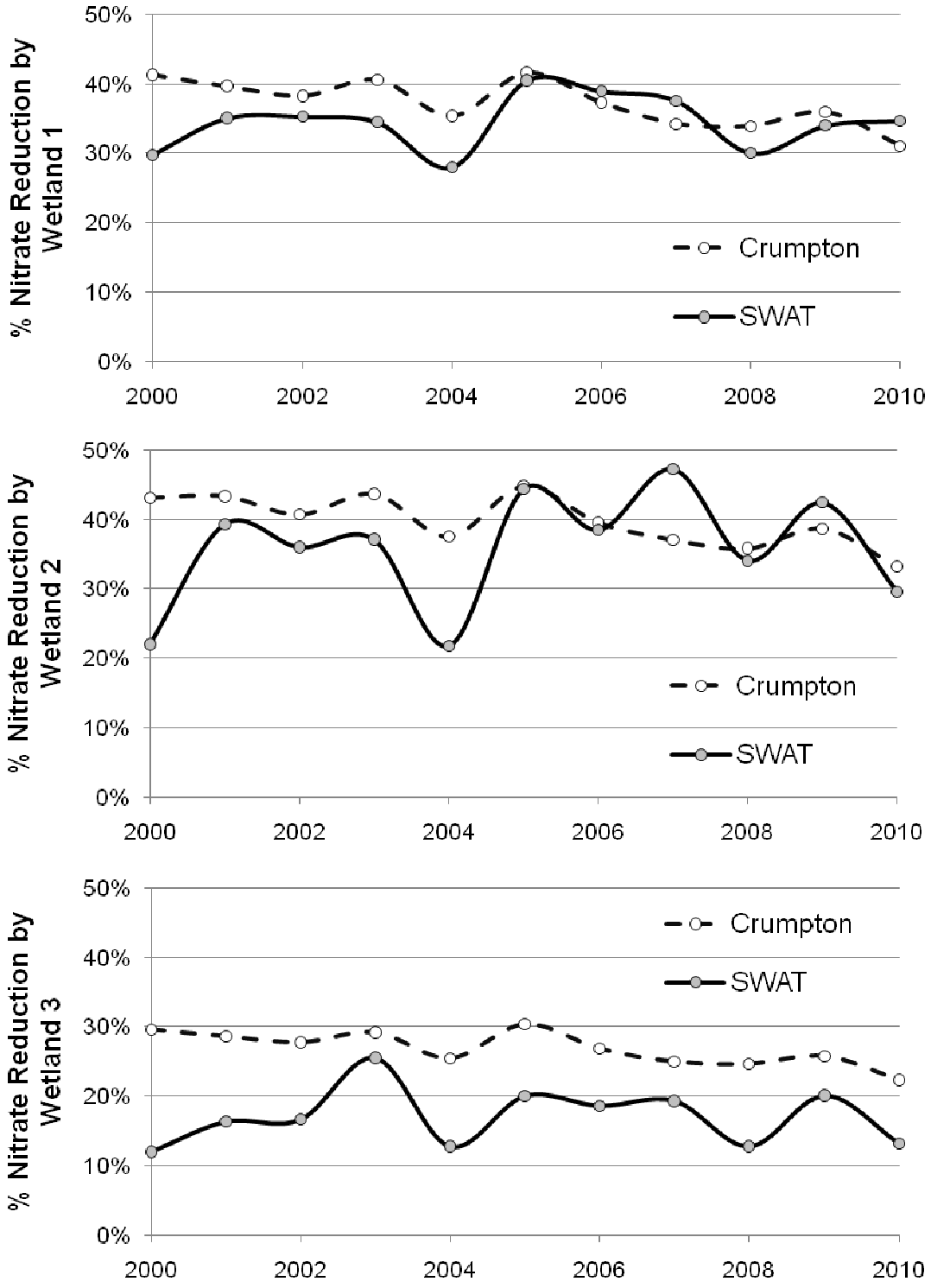

Figure 4.3 Annual nitrate reductions predicted by both the SWAT and Crumpton models at the outlet of each wetland.

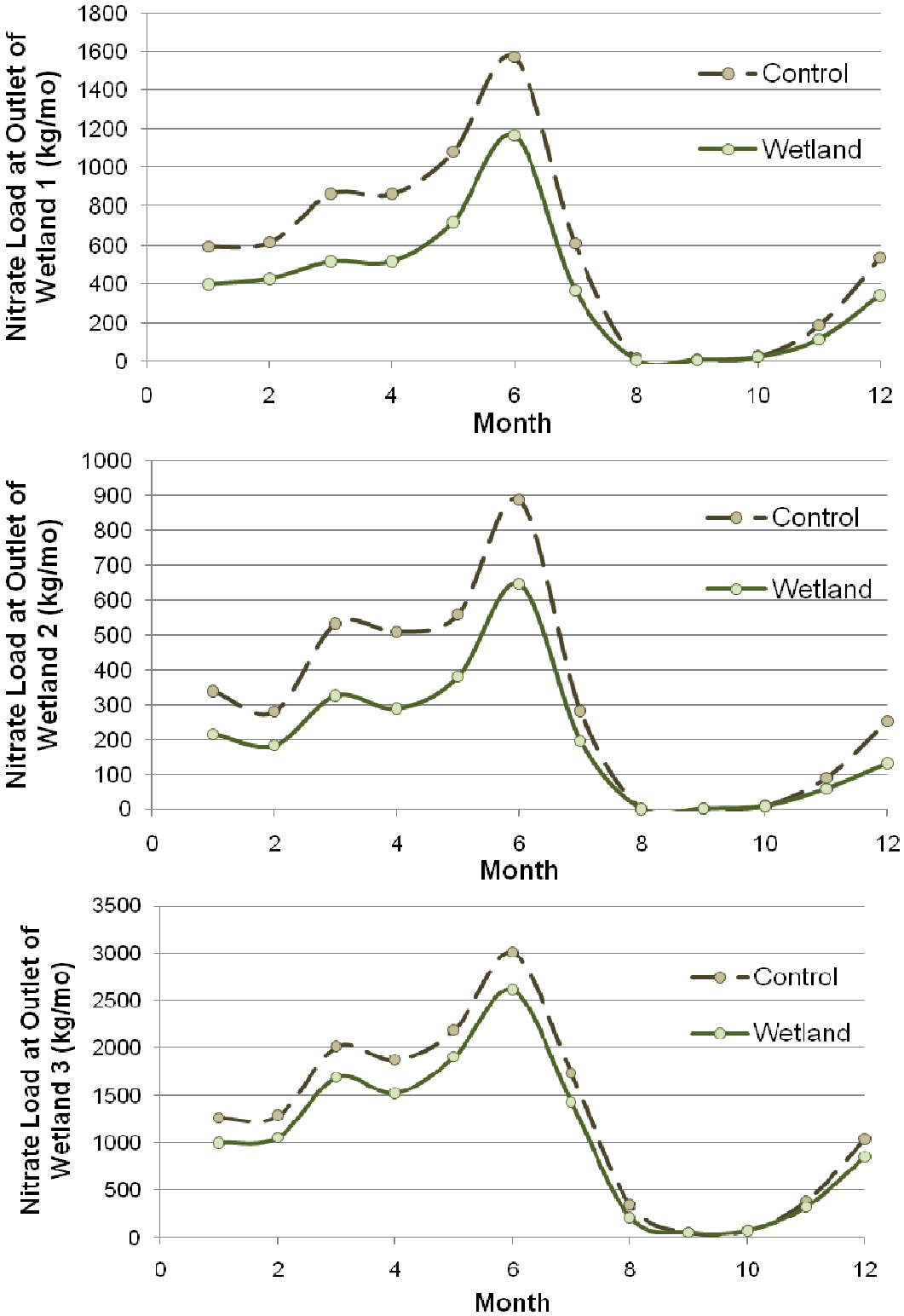

Figure 4.4 Average SWAT simulated monthly nitrate loads at wetland outlets compared to control scenario with no wetlands.

7% of the time (26 days/year). Therefore, while these wetlands are overall a sink of nitrate, they may for a time act as a source.

The main driver of nitrate reduction in the wetland was the “apparent nitrogen settling rate” (NSETL), which combined all forms of nitrogen removal from the system. This parameter should be considered carefully when implementing wetlands in the SWAT model, and if at all possible the values should be measured in the field. Here, NSETL was derived from fairly detailed data in a similar wetland nearby, yet this wetland did not have variable hydraulic loading rate, and data was taken over the course of only one year.

4.5. Summary and Conclusion

The three wetlands placed strategically in the region remove 17-36% of influent nitrate on an annual basis. The Crumpton et al. (2006) regression model estimated similar nitrate removal efficiencies, although the two models did not agree for wetland 3, which was unusually shallow. There was also a disparity in variability seen between the two models, which is due to the increased precision of the SWAT model. Wetlands were shown to be a nitrate sink during every month of the year. Average influent nitrate concentrations estimated by SWAT were 5-7 mg/L, while average effluent concentrations ranged from 3-5.5 mg/L.

One limitation of this work is that SWAT has a fairly simplistic way of modeling nutrient dynamics in wetlands. All mechanisms of nitrogen removal from wetlands are lumped into one parameter, called the “apparent nitrogen settling rate.” It would be of great interest to know the amount of nitrate removed by the irreversible process of denitrification as opposed to plant uptake or sediment-nitrogen interactions, which could be only temporary nitrogen sinks. The nitrogen settling rate is calculated for an annual time interval, so it should somewhat accurately represent the overall sink and source dynamics of a wetland. Yet the settling rate may have considerable variation throughout the year, and this is

currently not accounted for in this study. In SWAT, it is possible to input two different nitrogen settling rates for the cold and warm seasons. Future work in modeling wetlands in SWAT should include a better estimation of the “apparent nitrogen settling rate,” ideally from existing wetlands of the same type in the same geographic region, and separate estimations for the warm and cold seasons to account for increased nitrate removal in warm months.

The findings here support the nitrate reductions estimated in Chapter 3, although those results would be improved with a better estimation of hydraulic loading rate to the wetlands, which could be achieved by gathering more local data or further SWAT modeling. Constructing these wetlands would require conversion of 51 ha to wetlands and buffers, which is 0.9% of the watershed, or 1% of its cropland. Overall, if these three wetlands were placed in the Little Pine watershed, they would intercept 20% of its flows and remove 5% of the nitrate exported from the watershed.

CHAPTER 5. SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS FOR FURTHER RESEARCH

The goal of this study was to evaluate how constructed wetlands can be used to treat nitrate in tile-drained agricultural lands. The specific objectives included 1) Assembling criteria for wetland placement and implementing the methodology, 2) Creating preliminary wetland designs at these sites according to wetland design criteria, and 3) Estimating resulting nitrate removal provided by these wetlands.

To complete objective 1, criteria for wetland placement were developed based on the literature and knowledge of Indiana conditions. These criteria were then implemented in an 8-digit HUC in Indiana using publicly available data layers in ArcGIS 9.3 (Johnston et al., 2001) to select appropriate sites for wetland placement. For objective 2, criteria for wetland design were developed from preexisting wetland restoration criteria and knowledge of local soil, elevation, and tile drain characteristics. Preliminary wetlands were designed based on topography at every possible site. Then design criteria were used to eliminate designs that were not suitable. Objective 3 was addressed at every suitable site using local data for hydraulic loading rate and an annual regression equation developed on similar wetlands in Iowa. Then three wetlands were chosen to further analyze using the Soil and Water Assessment Tool (SWAT) model, which provided a more realistic and precise estimate of flow and nitrate dynamics in the study region.

The primary finding from objectives 1 and 2 (Chapter 3) was that constructed wetlands may be efficient solutions to nitrate removal, but the methodology for siting wetlands is too limiting for wetlands to provide considerable nitrate

reduction in the landscape. Suitable sites were found to intercept 3% of the flow from all tile-drained lands in the 8-digit HUC. Modeling shows that an average of 33% nitrate removal efficiency can be expected from these wetlands. Therefore, these wetlands would remove approximately 1% of all nitrate exported from tile-drained agricultural lands. However, revising the criteria could allow acceptance of more sites, especially those within small streams, so that wetlands could intercept a larger percentage of the high-nitrate flows.

Two models were used to satisfy objective 3 (Chapter 3 and Chapter 4). In Chapter 4, the implementation of the SWAT model showed that three wetlands – one designed in Chapter 3 and two additional wetlands that intercept small streams – could provide nitrate removal efficiencies within the range of 17-36% on an annual basis. These three wetlands would intercept 20% of the flows from tile-drained lands in the watershed, resulting in a 5% decrease in average nitrate loads at the watershed outlet. Average nitrate removal efficiencies provided by the SWAT model were in the same range as those estimated in Chapter 3, and for the most part the simple regression model based solely on annual hydraulic loading rate was found to predict quite well. Total nitrate removal expected at the watershed-scale from the 8-digit HUC in Chapter 3 (1%) differs from the SWAT modeled watershed in Chapter 4 (5%) by a factor a 5. One of the primary reasons for the greater impact in the SWAT modeled watershed is the revision of criteria to include small streams as suitable wetland sites, allowing two additional wetlands to be placed. This further supports the finding that wetlands are capable of large nitrate reductions in the landscape if criteria are altered to include more wetland sites.

5.1. Recommendations for Further Research

- The method may be validated by collecting water quality and flow data in strategically sited wetlands in Indiana, and comparing nitrate removal efficiencies to wetlands sited by other methods.
- A better estimate of the “apparent nitrogen settling rate” (NSETL) used to calculate nitrate removal in SWAT may be obtained by measuring water quality and flow in wetlands for the warm and cold seasons.
- The SWAT model may be extended to provide more realistic nitrogen removal mechanisms for wetlands. In particular, nitrogen removal may be split into temporary storage (plant uptake, sediment-nitrogen interactions) and complete removal (denitrification).
- The methodology may be implemented different geographic and climatic regions to better understand its limitations in application and the sensitivity of the method’s criteria.
- Synergies between different BMPs may be studied at the watershed scale. A suite of BMPs is likely required to obtain the necessary nitrate reductions, and yet little is known about exactly how BMPs will interact with one another.

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APPENDIX

APPENDIX

Manure inputs to agricultural lands were estimated separately for Purdue's ASREC Farm and a CAFO located in the SWAT-modeled watershed.

ASREC Farm Manure Estimation

1. Obtaining data
    a. Manure Application Data: Recorded manure applications for 2008-2009 were obtained from ASREC. The data included date of application, farm field to which manure was applied, application rate (ton/acre for hard packs/solids and gal/acre for slurries/liquids), and type of manure (e.g. dairy hard pack).
    b. Manure Composition Data: Moisture and nutrient content in the manures were obtained. These values were available for only a few samples. All available data was used in the analysis.
    c. Field Data: Size, shape and location of each farm field were determined from hand-drawn maps of the region in combination with orthophotos. Each farm field was then digitized in ArcGIS to the appropriate size, shape, and location.

2. Estimating monthly application rates to farm fields (kg/month dry manure)
    a. For each month in Manure Application Data, application rates were summed for identical farm fields and manure types. Because the actual date of application varies each year, application of all

manures was considered to take place on the 15th of the month during which it was applied.

b. manures was considered to take place on the 15th of the month during which it was applied.

c. For each manure type, the fraction of dry weight manure in the application was estimated using Manure Composition Data.
Dry weight fraction = 1 – (% moisture content/100)

d. Application rates were converted to total load (kg/month) of a manure to a given field.
For solid application rates:
Load applied (kg) = Rate (ton/acre) * Dry weight fraction * (907.18 kg/ton) * Field size (acre)
For liquid applications:
Load applied (kg) = Rate (gal/acre) * (0.00379 m^3/gal) * Density of water (1000 kg/m^3) * Dry weight fraction * Field size (acre)

e. For two years (2008 and 2009), the same months were averaged to determine the final output: average application rate of a certain manure in a given month to a particular field.

3. Estimating monthly application rates to SWAT subwatersheds

a. Using Field Data, digitized ASREC fields were intersected with SWAT subwatersheds.

b. For each entry (load applied to a field in a month), load to a field was split into load to the subwatersheds that intersect that field. The process is shown below:

$A_{sub,crop}$ = area of cropland in the subwatershed (ha)
$A_{sub,field}$ = intersection area of subwatershed and field (acre)
$F_{sub,out}$ = fraction of subwatershed that is not in ASREC farm

For a given manure application on a month to a field, a subwatershed intersecting this field receives the following manure application throughout the subwatershed (kg/ha):

$Manure_{sub,field}$ = [Load applied (kg) / Field size (acre)] * [$A_{sub,field}$/ $A_{sub,crop}$].

This calculation is repeated for every manure applied in each month to each field that intersects the subwatershed of interest.

Calculating fertilizer applications to a given subwatershed involves both manure applications from ASREC fields as well as traditional fertilizer applications to fields outside the ASREC farm:

$N_{traditional}$ = traditional anhydrous ammonia application rate (kg/ha) used to model fertilizers outside of ASREC farm

$P_{traditional}$ = traditional P_2O_5 application rate (kg/ha) used to model fertilizers outside of ASREC farm

$N_{sub,out} = F_{sub,out} * N_{traditional}$ = traditional anhydrous ammonia application rate to the subwatershed (kg/ha)

$P_{sub,out} = F_{sub,out} * P_{traditional}$ = traditional P_2O_5 application rate to the subwatershed (kg/ha)

The result of the above calculations for each subwatershed was a number of tables of month, manure type, and manure application rate spread across the subwatershed, along with traditional fertilizer amounts in the subwatershed.

4. Changing inputs to SWAT management files
    a. Because we had nutrient content data for a number of manures, we were able to add these manures to the SWAT manure database, the “fert.dat” file. The “fert” file tells SWAT the proportion of a given fertilizer that is mineral N, mineral P, organic N, organic P, and the ratio of NH_3-N to mineral N. New entries were made in the “fert” file

for each manure, with ID numbers 60-69. P2O5 was also added as number 99.

b. Crop management files had to be altered to reflect the manure applications (and decreased traditional fertilizer application) to the subwatersheds intersecting the ASREC farm. For each subwatershed all fertilizer applications were combined along with other management operations into corn/soy .mgt files. A matlab code was used to speed the process, but the same could be achieved through copy/paste directly into .mgt files. An example for subwatershed 18 HRU 1 is shown in Figure A.1.

000180001.mgt - WordPad

File Edit View Insert Format Help

```
 .mgt file Watershed HRU:66 Subbasin:18 HRU:1 Luse:CORN Soil: IN157Du-1 Slope: 0-9
               0    | NMGT:Management code
Initial Plant Growth Parameters
               0    | IGRO: Land cover status: 0-none growing; 1-growing
               0    | PLANT_ID: Land cover ID number (IGRO = 1)
            0.00    | LAI_INIT: Initial leaf are index (IGRO = 1)
            0.00    | BIO_INIT: Initial biomass (kg/ha) (IGRO = 1)
            0.00    | PHU_PLT: Number of heat units to bring plant to maturity (IG
General Management Parameters
            0.20    | BIOMIX: Biological mixing efficiency
           77.00    | CN2: Initial SCS CN II value
            1.00    | USLE_P: USLE support practice factor
            0.00    | BIO_MIN: Minimum biomass for grazing (kg/ha)
           0.000    | FILTERW: width of edge of field filter strip (m)
Urban Management Parameters
               0    | IURBAN: urban simulation code, 0-none, 1-USGS, 2-buildup/was
               0    | URBLU: urban land type
Irrigation Management Parameters
               0    | IRRSC: irrigation code
               0    | IRRNO: irrigation source location
           0.000    | FLOWMIN: min in-stream flow for irr diversions (m^3/s)
           0.000    | DIVMAX: max irrigation diversion from reach (+mm/-10^4m^3)
           0.000    | FLOWFR: : fraction of flow allowed to be pulled for irr
Tile Drain Management Parameters
           0.000    | DDRAIN: depth to subsurface tile drain (mm)
           0.000    | TDRAIN: time to drain soil to field capacity (hr)
           0.000    | GDRAIN: drain tile lag time (hr)
Management Operations:
               2    | NROT: number of years of rotation
Operation Schedule:
  5  6          6   61            0.00000
  5  6          1   19            0.00000   0.00     0.00000 0.00   0.00  0.00
  4 22          3    3           80.59859   0.00
  4 22          3   99           24.46726   0.00
  4 22          4    3            1.41000
 10 14          5                 0.00000
  4 15          3   66            1.02776   0.00
  9 15          3   61         2220.36235   0.00
 10 15          3   61         2220.36235   0.00
                0
  5 24          6    4            0.00000
  5 24          1   56            0.00000   0.00     0.00000 0.00   0.00  0.00
  5 10          3   99           19.49734   0.00
 10  7          5                 0.00000
 10 15          6   58            0.00000
  4 15          3   66            1.02776   0.00
  9 15          3   61         2220.36235   0.00
 10 15          3   61         2220.36235   0.00
```

For Help, press F1 NUM

Figure A.1 An example of one ".mgt" file showing manure applications for an HRU in a sample subwatershed intersecting the ASREC farm.

CAFO Manure Estimation

5. Obtaining data
    a. CAFO Data: Numbers of maximum numbers of swine of various sizes was obtained from the 1999/2000 IDEM Virtual File Cabinet online. The swine types were 400 nursery pigs (<18 kg), 400 grower/finishing hogs (>18 kg), 200 gestating sows, 48 sows with litters, and 12 boars.
    b. ASABE Standard: Tables 1 and 2 from the ASABE Standard for Manure Production and Characteristics (ASABE, 2005) were used to estimate the dry weight of manure produced in the CAFO.

6. Estimating annual dry weight manure production
    a. The estimation is shown step-by-step in Tables A1-A4, resulting in FRT_KG, an annual dry weight manure application to each of three subwatersheds located near the CAFO.
    b. In the absence of any data on what fields received the CAFO's manure applications, or at what time of year these applications occur, the applications were assumed to occur at the same rate in the three subwatersheds nearest the CAFO, divided equally and applied on the 15th of each month.
    c. Traditional fertilizers were assumed to be applied in these subwatersheds such that the nitrogen and phosphorus requirements were satisfied according to the assumptions for corn/soybean outside the CAFO and ASREC farm.

7. Changing management files in SWAT
    a. The management files (.mgt) were changed in the same way as for the ASREC farm (Appendix, part 4). SWAT's "swine fresh manure" in "fert.dat" was used to estimate nutrient composition of manures.

Table A.1 Types, sizes and maximum numbers of swine in CAFO (CAFO Data).

	A	B	C
Type	**Type**	**Size**	**Max capacity**
1	Nusery pigs	< 18 kg	400
2	Grower/finishing hogs	> 18 kg	500
3	Gestating sows		200
4	Sows with litters (total 1 unit)		48
5	Boars		12

Table A.2 Manure production standard for swine (ASABE, 2005).

From ASABE Standard, Tables 1 and 2

	D	E	F	G	H	I	J
Type	**Weight**	**Total solids**	**N**	**P**	**Total Manure**	**Moisture**	**Finishing period**
	kg	*kg/finished animal*				*% w.b.*	*days*
1	12.5	4.80	0.41	0.07	48.00	90	36
2	70	56.00	4.70	0.76	560.00	90	120
	kg	*kg/day-animal*				*% w.b.*	---
3	200	0.50	0.03	0.00	5.00	90	
4	192	1.20	0.09	0.03	12.00	90	
5	200	0.38	0.03	0.01	3.80	90	

Table A.3 Estimating total annual manure production by summing for each swine type.

Calculations

	K	L	M
Type	**Manure Rate**	**Manure Rate**	**Yearly Manure**
	(=H/J)	(=K*(100-I)/100)	(=L*C*365)
	kg/day/animal	*kg/d/a, dry weight*	*kg, dry weight*
1	1.33	0.1333333	19467
2	4.67	0.4666667	85167
3	5.00	0.5	36500
4	12.00	1.2	21024
5	3.80	0.38	1664

		N
Total Manure (kg/year)	(= SUM(M))	**163822**

Table A.4 Estimating manure application rates in three subwatersheds

O	P	R	S	T
Subwatershed	**Crop area**	**Watershed fraction**	**Yearly Manure**	**FRT_KG**
	(=corn+soy)	(=P/Q)	(=N*R)	**(=S/P)**
Subwatershed ID	*ha*	*fraction*	*kg/yr*	***kg/ha/yr***
2	85.01	0.29	46884	**552**
4	47.58	0.16	26243	**552**
15	164.45	0.55	90694	**552**

	Q
Total (=Sum(P))**:**	**297.04**

VITA

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ACADEMIC BACKGROUND

M.S.E., Ecological Sciences and Engineering, Purdue University, West Lafayette, IN, 2010. *Thesis Title:* Strategically Siting Constructed Wetlands to Target Nitrate Removal in Tile-Drained Agricultural Watersheds.

B.S., General Engineering with a concentration in Bioengineering, F. W. Olin College of Engineering, Needham, MA, 2008

EXPERIENCE

Entering Graduate Student Mentor, 2009-2010, Agricultural and Biological Engineering, Purdue University
I met weekly with a first-year graduate student in Agricultural and Biological Engineering through a mentoring program meant to facilitate a supportive environment in the department.

Ambassador to Agricultural and Biological Engineering, 2009-2010, Graduate Mentoring Program, Purdue University
I served as an ambassador to the department from the Graduate Mentoring Program. Responsibilities included facilitating events for the women of the department to assemble in a supportive and friendly environment.

Peer-to-Peer Mentor, August 2009 – May 2010, Ecological Sciences and Engineering Seminar, Purdue University
Acted as a mentor to new students in the Ecological Sciences and Engineering program and provided feedback on weekly written assignments.

Undergraduate Mentor, Summer 2009, Summer Undergraduate Research Fellowship (SURF) program, Purdue University
I co-mentored an undergraduate student in Agricultural and Biological Engineering at Purdue as he formulated and completed a summer research project

Biomaterials Researcher, January 2007 – May 2007, September 2007 – May 2008, Materials Science, Olin College of Engineering
Analyzed material properties of a transparent, waterproof nest cell lining of a *Colletes* bee. Work culminated in the presentation of a poster at the 2008 Materials Research Society Spring Meeting.

Safety and Ethics Coordinator & Project Manager, August 2007—May 2008, Senior Consulting Program for Engineering (SCOPE), Olin College of Engineering
Consulted for Boston Scientific to redesign an endoscopic band ligator, which is used to treat esophageal varices and hemorrhoids. Resulted in a provisional patent.

Environmental Engineering Researcher, Summer 2007, Research Experience for Undergraduates in CEES, University of Oklahoma, Norman, OK
Conducted an independent, self-designed research study relating riparian vegetation to stream morphology.

Course Assistant for Introductory Chemistry, Spring 2006 and 2007, Olin College of Engineering
Assisted in laboratory and held problem sessions for weekly problem sets.

Biology Researcher, January 2006—December 2006, Biology, Olin College of Engineering
Analyzed a tumor suppressor gene, RASSF1A, by DNA extraction, gel electrophoresis, protein purification, lipid blot analysis, western blot analysis, and growing and passing tumor cells.

Summer Intern, Summer 2005, Engineering Department, Department of Public Works, Amherst, MA
Surveyed with a total station and examined old plans, searching for lost structures, to improve the ArcMap town database.

HONORS/AWARDS

Honorable Mention, National Science Foundation Graduate Research Fellowship Program, 2009

Third Prize, Purdue's Ecological Sciences and Engineering Symposium Poster Competition, 2009

Honorable Mention, National Science Foundation Graduate Research Fellowship Program, 2008

Purdue Doctoral Fellowship, Purdue University, 2008-2010
U.S. Provisional Patent: McCahon, M., Austin, L., Cornelius, L. G., Doremus, N., Kneen, E. (2009). U.S. Provisional Patent: Ligating Band Dispenser Device. Filed September 30, 2009.

Full Tuition Scholarship, Olin College, 2004-2008

PROFESSIONAL AFFILIATIONS

Member, American Society of Agricultural and Biological Engineers (ASABE)
Member, American Water Resources Association (AWRA)

PRESENTATIONS

Margaret McCahon, Jane Frankenberger, Eileen Kladivko, Indrajeet Chaubey. Targeting Suitable Sites for Wetland Creation to Remove Nitrate in an Agricultural Watershed. *Poster presentation at the Interdisciplinary Graduate Programs Reception and Poster Session. Purdue University*. April 5, 2010.

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Margaret McCahon, Jane Frankenberger, Eileen Kladivko, Indrajeet Chaubey. Targeting Sites for Constructed Wetlands to Remove Nitrate in an Agricultural Watershed. *Poster presentation at Ecological Sciences and Engineering Symposium, West Lafayette, IN*. Sept. 25, 2009. Third prize.

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