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A study of nitrogen fate and transport in agricultural landscapes at the field, wetland, and watershed scales

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A STUDY OF NITROGEN FATE AND TRANSPORT IN AGRICULTURAL LANDSCAPES AT THE FIELD, WETLAND, AND WATERSHED SCALES

by

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To Mom and Dad

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ABSTRACT

Reducing agricultural nutrient loading in Iowa is critical to achieving Gulf of Mexico hypoxia water quality goals. Iowa comprises 4.4% of the Mississippi-Atchafalaya River Basin but contributes an average of 29% of the annual nitrate (NO_3-N) load to the Gulf of Mexico (Jones et al., 2018). The main goal of this research was to study nitrogen fate and transport in agricultural areas of Iowa at different spatial scales using a unique combination of water monitoring and numerical modeling. High-frequency, continuous water quality monitoring provided valuable insights into stream and wetland NO_3-N dynamics. A biogeochemical model was written and coupled to a spatially distributed, surface-subsurface hydrologic model to perform continuous (multi-year) nitrogen fate and transport simulations at the field, wetland, and watershed scales.

Field scale simulations of a tile-drained, corn-soybean rotation under conventional agricultural management over a 5-yr period illustrated strengths and weaknesses of the soil nitrogen model. Using a simplified approach to describe soil organic matter dynamics, the simulated annual nitrogen balance and NO_3-N loss in tile drainage were comparable to observations and literature estimates. However, the model was not able to predict the correct response of NO_3-N loss in tile drainage to fertilizer rate, which was attributed in part to limitations with the current plant uptake function which did not capture the nonlinear relationship expected between fertilizer rate and crop nitrogen uptake.

NO_3-N removal was quantified at one of Iowa's largest constructed wetlands using high-frequency (15-min), continuous water quality monitoring and hydrologic modeling. The wetland reduced incoming NO_3-N concentrations 49% and loads by an estimated 61 kg day^{-1} from May-Nov over a 3-yr period. Wetland removal was influenced by both hydrologic and biological

conditions; mass removal was greatest in Jun when discharge and NO_3-N loading were highest, while percent removal was greatest in Aug when discharge was low, water residence times in the wetland were high, and warm water temperatures enhanced processing. The high-frequency monitoring captured NO_3-N dynamics not possible with traditional lower frequency grab sampling, including concentration dynamics connected to storm events telling of sources and pathways of NO_3-N delivery, diurnal variations in concentration indicative of biological processes, and the marked variability in wetland removal performance during low and high flow conditions. Over 5600 wetlands of similar removal performance treating over 60% of Iowa's area and costing $1.5 billion would be required to reduce the state's baseline NO_3-N load by 45%.

The high-frequency monitoring guided and informed numerical simulations of nitrogen fate and transport at the wetland and watershed scales. Wetland simulations using imposed discharge and water quality conditions upstream of the wetland (inlet) and first order, temperature dependent kinetics produced satisfactory daily and monthly predictions of NO_3-N concentration and water temperature downstream of the wetland (outlet) from May-Nov in 3/4 and 4/4 study years, respectively. NO_3-N predictions were most sensitive to the denitrification first order rate constant and temperature during low discharge periods and least sensitive to both during storm events. Temperature dependent kinetics were necessary to accurately predict wetland NO_3-N removal in late summer.

The continuous watershed simulations produced satisfactory monthly predictions of inlet and outlet NO_3-N concentration and outlet water temperature. Consistent with findings from other modeling studies, annual nitrogen components and NO_3-N dynamics were simulated reasonably well under average hydrologic conditions, while simulated NO_3-N dynamics weakened under extreme (wet) hydrologic conditions. Temperature was important for predicting the seasonality of wetland NO_3-N removal during the growing season, while other factors such as organic carbon and dissolved oxygen may be more influential outside the growing season when removal can still occur despite cold conditions.

A preliminary evaluation of six recently constructed wetlands that detain and process agricultural runoff from 12% of a 45 km^2 watershed in north central Iowa estimated sizable flood and NO_3-N reductions locally which diminished moving downstream. Continuous watershed simulations over a 13 month period following wetland implementation estimated peak flow reductions of 3-43% at the wetlands that dissipated with drainage area; similarly, the wetlands reduced NO_3-N loads by an estimated 7-25% locally and 2% at the watershed outlet. Further refinements to the biogeochemical-hydrologic model are needed to improve simulated NO_3-N dynamics in order to more reliably assess downstream flow and NO_3-N reduction benefits.

This work identified limitations with the current modeling approach, areas of future work, and offers recommendations to guide future conservation design. Sensible hydrologic predictions are imperative to the success and dependability of the water quality simulations, which may seem obvious but can be difficult to ascertain in ungauged catchments. Future work aspires to couple a complete agricultural systems model with a physically-based hydrologic model to simulate the nitrogen cycle in a more comprehensive manner to assess which field scale nitrogen processes are most important to accurately predict stream nutrient loading at the watershed scale. Constructed wetlands could provide greater flood and nutrient reduction benefits if the normal pool hydraulics were designed with smaller hydraulic structures that more effectively throttle down incoming flows and provide the opportunity for active rather than passive pool management. As the ultimate goal of this research and other like work is to quantify progress of water quality goals set forth by the Gulf Hypoxia Task Force and help guide future conservation practice implementation, continued investment in science-based water research, water monitoring, and water modeling is necessary.

PUBLIC ABSTRACT

Reducing nutrient loss from agricultural areas in Iowa is critical to improving local and regional water quality and reducing the Dead Zone in the Gulf of Mexico. Iowa comprises 4.4% of the Mississippi-Atchafalaya River Basin but contributes an average of 29% of the annual nitrate (NO_3-N) load to the Gulf of Mexico (Jones et al., 2018). The goal of this study was to study nitrogen fate and transport in agricultural areas of Iowa at different spatial scales using a unique combination of water monitoring and numerical modeling. Doing so ultimately enabled the flood and nitrogen reduction benefits of wetlands to be evaluated under variable hydrologic conditions.

NO_3-N removal was quantified at one of Iowa's largest constructed wetlands using high-frequency (15-min), continuous water quality monitoring and hydrologic modeling. The wetland reduced incoming NO_3-N concentrations 49% and loads by an estimated 61 kg day^{-1} from May-Nov over a 3-yr period. Over 5600 wetlands of similar removal performance treating over 60% of Iowa's area and costing $1.5 billion would be required to reduce the state's baseline NO_3-N load by 45%.

A coupled biogeochemical-hydrologic model was developed to perform continuous nitrogen fate and transport simulations. Wetland simulations of water temperature and NO_3-N concentration from May-Nov were in good agreement with observations during most of the four study years and the influence of temperature and the stream denitrification removal parameter on the NO_3-N predictions was greatest during low flow periods. The watershed simulations predicted the annual nitrogen balance in a reasonable manner and NO_3-N dynamics reasonably well under average hydrologic conditions, while simulated NO_3-N dynamics weakened under

extreme (wet) hydrologic conditions. Similar performance trends were noted in other modeling studies.

A preliminary evaluation of six recently constructed wetlands in a 45 km^2 watershed estimated sizable flow and NO_3-N reduction benefits locally near the wetlands that dissipated moving downstream. Continuous watershed simulations over a 13 month period following wetland implementation estimated peak flow reductions of 3-43% at the wetlands that dissipated with drainage area; similarly, the wetlands reduced NO_3-N loads by an estimated 7-25% locally and 2% at the watershed outlet. Revisions to the biogeochemical-hydrologic model are necessary to improve the reliability of these initial estimates.

This work identified limitations with the current approach, areas of future work, and offers recommendations to guide future conservation design. Sensible hydrologic predictions are imperative to the success and dependability of the water quality simulations, which may seem obvious but can be difficult to ascertain in ungauged catchments. Constructed wetlands could provide greater flood and nutrient reduction benefits if the wetland hydraulics were designed to more effectively throttle down incoming flows and provide the opportunity for active rather than passive pool management. As the ultimate goal of this research and other like work is to quantify progress of water quality goals set forth by the Gulf Hypoxia Task Force and help guide future conservation practice implementation, continued investment in science-based water research, water monitoring, and water modeling is necessary.

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1. INTRODUCTION

Motivation: Gulf of Mexico Hypoxia

The "Dead Zone" in the northern Gulf of Mexico is the second largest coastal hypoxic zone in the world (Rabalais et al., 2002; Scavia et al., 2018). Hypoxia refers to when a waterbody's dissolved oxygen (DO) is too low (< 2 mg/l) to support aquatic life. Coastal eutrophication caused by nutrient enrichment from the Mississippi and Atchafalaya Rivers provides an abundant food source for algae and phytoplankton production, but overpopulation, death, and biodegradation by bacteria that ensues consumes DO. Gulf Hypoxia is of most concern during summer when nutrient loads from the Mississippi-Atchafalaya River Basin (MARB; Figure 1) are greatest and the secondary effect of thermal stratification between the warmer, less dense freshwater and cooler, denser saltwater further inhibits vertical mixing of DO.

In 2001, the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force – a group of tribal, state, and federal agencies – released an Action Plan describing a strategy for hypoxic area reduction. The primary goal of the 2001 Action Plan was to reduce the 5-yr average size of the summer hypoxic zone to 5000 km^2 by 2015 (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2001). Summer monitoring (typically mid to late July) of hypoxic area began in 1985 (LUMCON, 2018) and MARB nutrient load estimates were first documented in 1979 (Aulenbach et al., 2007). The 2001 Action Plan estimated a 30% reduction in annual nutrient (nitrogen and phosphorus) loading (relative to the 1980-1996 average annual nutrient loads) from the MARB would be required to achieve the 5000 km^2 hypoxic area goal (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2001), but subsequent research indicated a 45% reduction was more likely (EPA Science Advisory Board, 2007). A revised Action Plan in 2008 maintained the same water quality goals and called for 12 states in the MARB to develop and implement nutrient reduction strategies using a combination of point and non-point source practices (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2008). In 2015, the 5-yr average size of the summer hypoxic zone was 14024 km^2, so the Task Force extended the target date for the water quality goals to 2035 with an intermediate goal of a 20% reduction in annual nutrient loading by 2025 (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2015). In 2017, the largest hypoxic zone on record was measured (22720 km^2; Figure 1). Most recently, the 5-yr average hypoxic zone area (15032 km^2; 2012-17), MARB total nitrogen load (1.3 billion kg N; 2012-16), and MARB total phosphorus load (163 million kg P; 2012-16) were 3.0, 1.5, and 2.2 times greater than the 2035 goals, respectively.

Although other factors contribute, Gulf Hypoxia is largely driven by nitrogen loading from the MARB, mainly in the form of nitrate-nitrogen (NO_3-N) which accounts for approximately two-thirds of the total nitrogen flux in the MARB (Turner et al., 2006). The May NO_3-N load is a particularly good predictor of hypoxic zone size, as it explained 50% of the variation in hypoxic area from 1978-2004 while both May NO_3-N load and year explained 82% of its variability (Turner et al., 2006). The primary source of NO_3-N in the MARB is from row crop agriculture in the Corn Belt region of the U.S. Midwest (David et al., 2010). The Upper Mississippi River Basin (UMRB), which accounts for 15% of the MARB by area and comprises large portions of several Corn Belt states, delivered 45% of the annual NO_3-N load in the MARB from 2000-2015 (Aulenbach et al., 2007).

The Corn Belt state of Iowa is one of the leading producers of corn, soybeans, and livestock in the U.S. Approximately 65% of Iowa is devoted to corn and soybean production

(Figure 1). At the time of the 2012 U.S. Census, Iowa ranked first among U.S. states in harvested area of corn (5.4 million ha) and soybeans (4.0 million ha), first in number of hogs (20.5 million), first in number of poultry (12.6 million), and sixth in number of cattle/calves (3.9 million) (USDA, 2017). There are 6.5, 4.0, and 1.2 times as many hogs, poultry, and cattle, respectively, in Iowa as there are people. An estimated 92% of Iowa's NO_3-N load is derived from non-point sources (Libra et al., 2004), primarily in the form of row crop agriculture covering two-thirds of the state. Since agricultural intensity has previously been correlated with NO_3-N loading (Schilling and Libra, 2000; David et al., 2010; Jones et al., in review), Iowa's contribution to the MARB NO_3-N load is expected to be significant.

Iowa's NO_3-N load has been estimated previously and accounts for a significant fraction of the total NO_3-N load entering the Gulf of Mexico. Table 1 summarizes several estimates of Iowa's average annual NO_3-N load and contribution to the MARB. Despite comprising only 4.4% of the area in the MARB, Iowa accounts for 15-30% of the MARB NO_3-N load on average and substantially more in flood years. Iowa also contributes a disproportionate amount of NO_3-N to the Missouri River Basin (MRB) and UMRB (Figure 1). Approximately 31% of Iowa drains to the MRB (3.3% of the MRB area), while 69% of the state drains to the UMRB (20.5% of the UMRB area). In 2016, the portions of Iowa draining to the Missouri and Mississippi Rivers contributed 72% and 47% of the NO_3-N loads to the MRB and UMRB, respectively (Jones et al., 2018). Iowa's long term (1999-2016) average NO_3-N load was recently estimated to represent 45% and 55% of the UMRB and MRB average NO_3-N loads, respectively (Jones et al., 2018). Clearly, focusing nitrogen reduction efforts in Iowa is critical to achieving Gulf Hypoxia water quality goals.

Wetlands: an Important Agricultural Conservation Practice

Wetlands are one agricultural conservation practice proven to be effective at reducing NO_3-N loads from cropped areas. Monitoring studies indicate natural or constructed wetlands can reduce agricultural NO_3-N loads by 20-70% and retain several hundred kg NO_3-N ha^{-1} wetland yr^{-1} (Drake et al., 2018). The Iowa Nutrient Reduction Strategy (INRS) estimated that wetlands could reduce NO_3-N concentrations by 11-92% with an overall average of 52% used for statewide nutrient reduction scenario calculations (ISU, 2014). Clearly, wetland NO_3-N removal performance is variable and dependent on a number of factors including hydrologic conditions, NO_3-N supply constraints, and biological conditions that are difficult to predict for planning and implementation purposes.

Numerical Modeling

Numerical modeling provides a viable, cost effective approach for evaluating the water quantity and quality benefits of conservation practices at meaningful spatial and temporal scales. Paired with sustained water monitoring, watershed modeling can quantify nutrient reduction progress and guide conservation practice implementation and management. While dozens of watershed models now exist, relatively few perform nutrient fate and transport simulations (Imhoff et al., 1983; Hansen et al., 2009; Li et al., 2010; Ikenberry et al., 2017). Therefore, the major goal of this thesis was to develop a water quality model encompassing the major processes of the nitrogen cycle and couple to a physically-based hydrologic model to perform watershed nitrogen fate and transport simulations to evaluate the effectiveness of wetlands for reducing agricultural NO_3-N loads in Iowa under variable hydrologic conditions to help achieve Gulf Hypoxia water quality goals.

Water quality modeling for agricultural applications has traditionally been performed with two different modeling approaches depending on the spatial scale of interest. Agricultural systems models are one-dimensional, process-based models that describe hydrologic and nutrient cycling and crop growth dynamics at the scale of an agricultural field. These models describe in detail the biogeochemical processes affecting plant growth, soil organic matter, and nutrient fate in a vertical soil column but do not consider the lateral transport of water or nutrients. The model domain (e.g. agricultural field) is typically represented as a single unit area and water and nutrient fluxes and transformations are resolved in the vertical direction only. Examples of agricultural systems models include the RZWQM (Root Zone Water Quality Model; Ma et al., 2000), APSIM (Agricultural Production Systems Simulator; Holzworth et al., 2014), and DAISY (Hansen et al., 1991). Agricultural systems models have been used to assess the impacts of various agricultural management, including cover crops (Hansen et al., 1991; Qi et al., 2011; Malone et al., 2014; Martinez-Feria et al., 2016), fertilizer rate (Thorp et al., 2007; Qi et al., 2012; Puntel et al., 2016), and crop rotations and tillage (Al-Kaisi et al., 2016; Puntel et al., 2016), on hydrologic and nitrogen cycling, crop yield, and NO_3-N loss in subsurface drainage.

In contrast, larger scale, downstream impacts of agriculture on hydrology and water quality have typically been evaluated with conceptual, lumped parameter watershed models. With this modeling approach, watershed processes are described in a more conceptual (rather than explicit) manner often using empirically-based relationships. The model domain (e.g. watershed) is decomposed into a number of subbasins and river reaches, and the primary goal is to describe how water and nutrient fluxes calculated at the subbasin scale (tens to hundreds of hectares in size) are transported through the river network. These models do not consider nutrient availability in simulating plant growth and biogeochemical processes influencing soil organic matter and nutrient transformations are represented in a simpler manner compared to agricultural systems models. Two conceptual, lumped parameter watershed models commonly used for nitrogen fate and transport simulations are HSPF (Hydrological Simulation Program-Fortran; Bicknell et al., 1993) and SWAT (Soil and Water Assessment Tool; Neitsch et al., 2011). Both have been used to evaluate the water quality impacts of agriculture in Iowa at the watershed scale. HSPF was used to evaluate the impact of conservation tillage and contouring on annual water, sediment, and nutrient balances in small (52 km^2) and large (7240 km^2) Iowa watersheds during a 5-yr study (Donigian et al., 1983). More recently, Bradley et al. (2015) performed a 65-yr continuous nitrogen fate and transport simulation with HSPF for a 1696 km^2 agricultural eastern Iowa watershed, but no conservation practices were simulated. SWAT was used to assess changes in land use and agricultural management on stream NO_3-N loading in two large Iowa watersheds (9400-16175 km^2; Jha et al., 2007; Schilling and Wolter, 2009). More recently, Ikenberry et al. (2017) evaluated the flow and nutrient removal algorithms for wetlands in SWAT using two small case study wetlands in Iowa (drainage areas less than 3.1 km^2). The revised equations provided similar or improved performance of simulated daily wetland outflows and NO_3-N removal and required less calibration compared to the original wetland equations.

This dissertation proposes a new approach for simulating nitrogen fate and transport in agricultural landscapes that allows both field scale processes and downstream impacts to be assessed within a single modeling framework. The new approach involves coupling a physically-based spatially distributed hydrologic model to a biogeochemical model to perform nitrogen fate and transport simulations. The spatial resolution of the hydrologic-biogeochemical model is fine enough to evaluate field scale nitrogen processes while the model domain (watershed) enables downstream impacts to be assessed. Few studies have attempted to perform nitrogen fate and transport simulations with a physically-based hydrologic model (Styczen and Storm, 1993; Hansen et al. 2007, 2009; Vervloet et al. 2018), in large part because of the added computational expense and large number of input parameters required (Refsgaard, 1997). The few studies that have did so with a two-step sequential modeling approach. First, an agricultural systems model (DAISY) simulated crop growth dynamics and soil nitrogen cycling to derive estimates of NO_3-N leaching below the root zone for all combinations of agricultural management, soils, and crop sequences. Following, the NO_3-N leaching time series were imposed in the spatially distributed hydrologic model for the watershed nitrogen simulations. The watershed simulations sometimes simulated denitrification in the river network and/or groundwater. In general, these studies predicted annual nitrogen components at the catchment scale in a reasonable manner, while seasonal nitrogen dynamics were simulated unsatisfactorily, particularly in wet or dry years (Hansen et al., 2009). Poorly simulated NO_3-N dynamics were attributed to errors in simulated hydrology, spatial averaging of hydrologic and water quality inputs, and the sequential rather than dynamic coupling between the agricultural systems model and the hydrologic model (Hansen et al., 2007, 2009).

Dissertation Funding

The primary funding source for this dissertation was from a USDA Conservation Innovation Grant awarded to the Iowa League of Cities in September 2015. The grant objective was to develop a water quality trading framework in Iowa to help achieve INRS water quality goals. Water quality trading is a voluntary, conceptual framework to improve water quality in which point sources could invest in upstream conservation practices (primarily on agricultural land in Iowa) to reduce nutrient loading at a potentially lower cost than with traditional point source removal technologies. While this approach is appealing to point sources who are federally

regulated to meet water quality standards because of the potential cost savings, implementation in Iowa and across the nation has been limited thus far in part because of the high uncertainty associated with nutrient reduction estimates for agricultural conservation practices (Selman et al., 2009; Hoag et al., 2017). IIHR-Hydroscience & Engineering was contracted by the Iowa League of Cities to develop the scientific framework for the water quality trading program and serves on the technical advisory committee. This 3-year, $700,000 grant concludes in the fall quarter of 2018.

Funding for this dissertation was also provided by the Iowa Watershed Approach (IWA) project. In January 2016, Iowa was awarded a 5-year, $97 million grant from the U.S. Department of Housing and Urban Development aimed at flood mitigation, nutrient reduction, and increasing social resiliency to flooding. Nine Iowa watersheds impacted by flooding from 2011-13 were selected to participate. Hydrologic assessments are currently being completed for these watersheds that summarize watershed conditions and practice scenarios that quantify the effectiveness of different hypothetical flood mitigation strategies and practices. The second part of the project will focus on construction of conservation practices for flow and nutrient reduction in targeted areas of each watershed. Approximately 88% of IWA dollars will be used for building projects with landowners responsible for a 10% cost share. The long term goal of this project is to develop a program for flood reduction and water quality improvement that is scalable and replicable across Iowa, the Midwest, and the U.S.

Goals and Objectives

The goal of this research was to study nitrogen fate and transport in agricultural areas of Iowa at different spatial scales. Specific objectives included:

1. Develop spatially distributed, surface-subsurface continuous hydrologic models of the study areas.
2. Develop a coupled biogeochemical-hydrologic model to perform nitrogen fate and transport simulations.
3. Complete a detailed parameter assessment for nitrogen cycling.
4. Use high-frequency, continuous water quality monitoring to assess wetland nitrogen removal.
5. Use continuous numerical simulations to evaluate nitrogen fate and transport at the field, wetland, and watershed scales.

Thesis organization:

- Chapter 2: Development of a simple soil nitrogen model to predict soil nitrogen cycling and NO_3-N leaching. Field scale simulations of a tile-drained corn-soybean research plot in north central Iowa were compared to observational data.
- Chapter 3: Assessment of NO_3-N dynamics and removal at Iowa's second largest CREP wetland (Slough Creek) using high-frequency, continuous water quality monitoring and hydrologic modeling.
- Chapter 4: Development of a simple stream nitrogen model to predict wetland nitrate removal. Wetland and watershed simulations performed at the Slough Creek wetland.
- Chapter 5: Watershed nitrogen fate and transport simulations to evaluate six recently constructed wetlands for flow and nitrate reduction in a 45 km^2 agricultural HUC-12 watershed.
- Chapter 6: Summary and conclusions, study limitations, potential areas of future work.

Tables and Figures

Table 1. Estimates of Iowa's NO_3-N load contribution to the MARB.

Study	Time Period	Description/Method	NO_3-N Load (10^6 kg yr^{-1})	NO_3-N Yield (kg ha^{-1})	Fraction of MARB NO_3-N Load
Goolsby et al., 1999	1980-1996	Multivariable regression	251-298	17-20	16-19%
Goolsby et al., 1999	1993 (flood year)	Multivariable regression			35%
Libra 1998	1987-1996	Monthly to bimonthly water monitoring data	204-222	14-15	25%
Libra et al., 2004	2000-02	Daily discharge and monthly monitoring from 68 watersheds covering 80% of the state	180	12	20%
Iowa Nutrient Reduction Strategy (ISU, 2017)		Literature review differentiated by geographic landform region	279	19	32%[1]
Jones et al., 2018a	2016	High frequency (15-min) monitoring from 13 sensors capturing water from 82.5% of the state's area	477	33	41%
Jones et al., 2018b	1999-2016	Daily discharge and monthly grab sampling from 23 sites capturing water from 80% of the state's area	255	18	29%

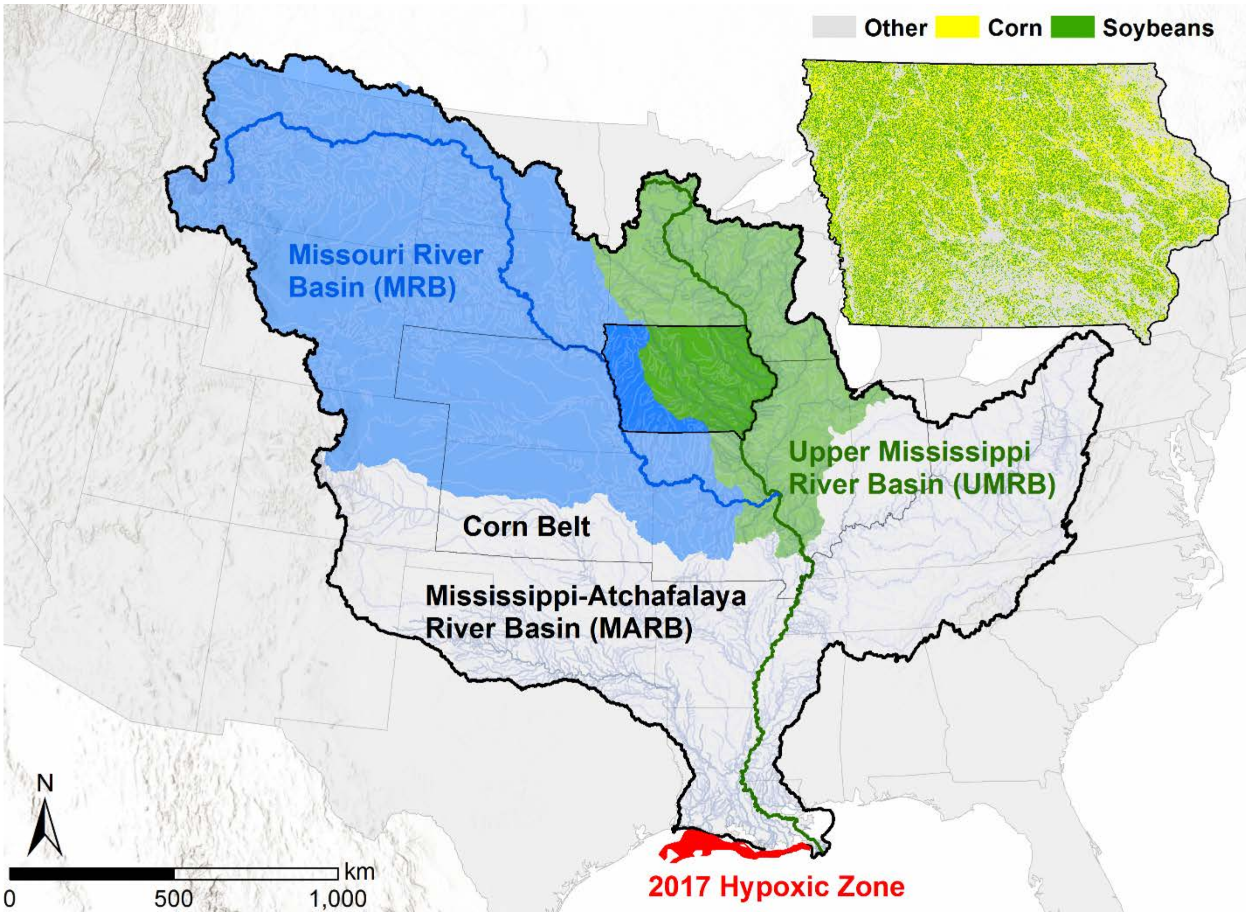

Figure 1. Iowa, the Mississippi-Atchafalaya River Basin (MARB), and the hypoxic zone in the Gulf of Mexico.

2. PREDICTING ANNUAL NITROGEN COMPONENTS AND NITRATE LOSS IN SUBSURFACE DRAINAGE FROM A CONVENTIONAL AGRICULTURAL FIELD

Abstract

Evaluating baseline conditions and the potential of agricultural conservation practices to mitigate nitrogen runoff using computer models requires simulating soil nitrogen processes in a reasonable manner. Simulating nitrogen fate and transport with a physically-based watershed model has been relatively unexplored. The objectives of this study were to develop a soil nitrogen model using MIKE SHE ECO Lab and evaluate its performance in simulating annual nitrogen components and nitrate-nitrogen (NO_3-N) loss from a tile-drained, corn-soybean plot in Iowa under conventional agricultural management. A simple soil nitrogen model was developed based on a review of several field and watershed nitrogen models and describes the fate of organic nitrogen, soluble ammonium (NH_4-N), and NO_3-N in the root zone and shallow groundwater. The soil nitrogen model contains 11 parameters and simulates net mineralization, nitrification, denitrification, crop uptake, and sources of inorganic nitrogen from precipitation and fertilizer inputs. Using a 5-yr field and modeling study for reference, MIKE SHE ECO Lab simulated several nitrogen components in a reasonable manner annually. Simulated annual NO_3-N loss in subsurface drainage was considered unsatisfactory by several statistical measures, but results were comparable with literature estimates and consistent with the simulated hydrology. The MIKE SHE ECO Lab model was not able to predict the response of NO_3-N concentration to fertilizer rate observed and modeled previously at the site, which is partially attributed to limitations with the current plant uptake function. While limitations exist, the current MIKE SHE ECO Lab model still shows promise for simulating the annual soil nitrogen mass balance in a reasonable manner.

Introduction

Reducing nitrate-nitrogen (NO_3-N) loss from agricultural areas – the primary source of NO_3-N loading in the Mississippi-Atchafalaya River Basin (MARB) (David et al., 2010) – is critical to improving local water quality and reducing Gulf of Mexico Hypoxia. Agricultural intensity in the MARB is greatest in the Upper Mississippi River Basin (UMRB), which comprises several Corn Belt states including Iowa. Long term (>15 years) monitoring indicates the UMRB (15% of the MARB by area) and Iowa (4.4% of the MARB by area) contribute an average of 45% and 29% of the NO_3-N load to the MARB, respectively (Aulenbach et al., 2007; Jones et al., 2018). Within these agriculturally intensive regions, subsurface (tile) drainage – perforated pipes installed approximately one meter below the ground to remove excess soil water to enhance agricultural productivity – is the primary mechanism for NO_3-N loss from farm fields (Baker et al., 1975; Ikenberry et al., 2014). Historic estimates suggest 37% of the Corn Belt (Fausey et al., 1995) and 25% of Iowa (Baker et al., 2004) have subsurface drainage, and NO_3-N concentrations and yields from tile-drained, corn-soybean systems often exceed 20 mg l^{-1} and 40 kg ha^{-1} (Jaynes et al, 2001; Ikenberry et al., 2014).

Computer models can be cost effective tools for evaluating the hydrologic and water quality impacts of different agricultural scenarios. Depending on the spatial scale of interest, however, models of varying conceptual frameworks, process formulations, and complexity are used. Modeling of water, carbon, nitrogen, and crop dynamics at the scale of an agricultural field has traditionally been performed with one-dimensional, process-based models that describe the fate of nutrients and other constituents in the soil column in detail but do not consider lateral transport. The model domain (e.g. agricultural field) is typically represented as a single unit area and water and nutrient fluxes and transformations are resolved in the vertical direction only. The RZWQM (Root Zone Water Quality Model; Ma et al., 2001) and APSIM (Agricultural

Production Systems Simulator; Holzworth et al., 2014) are two examples of such agricultural systems models that have been used to evaluate agricultural management at the field scale in Iowa (Qi et al., 2011; Qi et al., 2012; Malone et al., 2014; Martinez-Feria et al., 2016, 2018). In contrast, watershed models describe the transport of water, nutrients, and other constituents at larger spatial scales but generally do not consider or lack the spatial resolution to resolve nutrient transformations that are important at the field scale. SWAT (Soil and Water Assessment Tool; Neitsch et al., 2011) and HSPF (Hydrological Simulation Program-Fortran; Bicknell et al., 1993) are examples of a limited number of watershed models that do consider nutrient fate and transport. Both have been applied to Iowa watersheds to evaluate water quality conditions and the effect of conservation measures (Donigian et al., 1983; Imhoff et al., 1983; Schilling et al., 2009; Bradley et al., 2015). HSPF and SWAT are classified as conceptual, lumped parameter watershed models in which water and nutrient fluxes are calculated at the subbasin scale typically ranging from tens to hundreds of hectares in size.

We propose the use of a different water quality modeling methodology that has the potential to resolve water and nitrogen dynamics at the field scale yet allow integrated watershed simulations. This approach involves the use of a physically based watershed model coupled with a process-based ecological model to perform nitrogen fate and transport simulations. The modeling platform is MIKE SHE ECO Lab. MIKE SHE is a physically based watershed model that describes the movement of water across the landscape based on the governing equations of fluid flow (Graham and Butts, 2005). ECO Lab is a generic equation editor containing user-defined state variables and processes for the environmental problem of interest. Equations describing soil nitrogen transformations were defined in ECO Lab and coupled to the MIKE SHE solute transport module to perform nitrogen fate and transport simulations. While MIKE

SHE has been used previously to evaluate water and nutrient transport in agricultural watersheds (Styczen and Storm, 1993; Hansen et al., 2007, 2009; Vervloet et al., 2018), few studies have been conducted in Iowa (Frana 2012; Zhou et al., 2013) and none to our knowledge have used MIKE SHE ECO Lab.

The purpose of this study was to develop a soil nitrogen model using MIKE SHE ECO Lab and evaluate the model's ability to simulate soil nitrogen components and NO_3-N loss for a tile-drained corn-soybean research plot under conventional agricultural management. The modeling was conducted at a research plot in north central Iowa where subsurface drainage and NO_3-N loss were monitored during a 5-yr period 2005-09 and also modeled by RZWQM (Qi et al., 2011). The ability of the MIKE SHE ECO Lab model to simulate NO_3-N loss in subsurface drainage in response to variable nitrogen fertilizer rates was also evaluated and compared to previous monitoring (Lawlor et al., 2008) and modeling (Qi et al., 2012) studies at the same research site.

Materials and Methods

MIKE SHE ECO Lab

MIKE SHE is a physically based, spatially distributed hydrologic and solute transport model. The commercial code was first developed in 1977 by three European organizations and since the mid-1980s has been maintained by DHI Water & Environment in Copenhagen, Denmark (Graham and Butts, 2006). MIKE SHE simulates the various components of the terrestrial water cycle describing how precipitation is partitioned into evapotranspiration, surface runoff, infiltration, and groundwater flow. Based on the blueprint proposed by Freeze and Harlan (1969), surface and subsurface water fluxes are calculated at discrete locations by numerical solutions to partial differential equations describing conservation of mass (continuity) and momentum. Surface runoff (or overland flow) is calculated two-dimensionally using the diffusive wave approximation of the Saint Venant equations; infiltration and soil water retention in the unsaturated zone is calculated one-dimensionally in the vertical direction by Richards equation; saturated zone flow (groundwater) is calculated three-dimensionally according to Darcy's law (DHI 2017). Other processes such as evapotranspiration and snowmelt are calculated empirically. Evapotranspiration – the sum of canopy interception, evaporation from ponded water, soil water evaporation, and plant transpiration – is calculated by the Kristensen-Jensen method based on vegetative inputs and a potential evapotranspiration rate (Kristensen and Jensen, 1975). Snowmelt is calculated by a modified degree-day method. Solute transport can also be simulated for water quality applications using the advection-dispersion equation. MIKE SHE uses a modular structure to simulate the various hydrologic processes and calculations are performed on a square grid of a user-defined size for the surface and subsurface layers.

ECO Lab is another MIKE module consisting of an equation editor intended for ecological modeling applications. State variables and processes representing different environmental phenomena are defined by the user. For each state variable, a mass balance equation is written as an ordinary differential equation that describes how that state variable changes with time due to different processes that can also influence other state variables. ECO Lab is coupled to the solute transport module of a particular MIKE SHE hydrologic module and acts as an additional source/sink term in the advection-dispersion equation. ECO Lab has been used previously to model stream temperature (Loinaz et al., 2013), fish growth (Loinaz et al., 2014), and NO_3-N retention processes in a Danish wetland (Christierson et al., 2015).

Soil Nitrogen Model

A soil nitrogen model was developed using MIKE SHE ECO Lab that describes nitrogen transformations taking place in the root zone (unsaturated zone in MIKE SHE) and shallow groundwater (saturated zone in MIKE SHE) (Figure 2). The model was developed based on a review of the soil nitrogen models implemented in RZWQM, APSIM, HSPF, and SWAT. The model considers the fate of three nitrogen species – particulate organic nitrogen, soluble ammonium (NH_4-N), and nitrate (NO_3-N) – and includes the processes of net mineralization, nitrification, denitrification, plant uptake, and sources of inorganic nitrogen (NH_4-N or NO_3-N) from precipitation (atmospheric wet deposition) and fertilizer inputs. The processes of net mineralization, nitrification, and denitrification are defined in ECO Lab while plant uptake and sources of inorganic nitrogen from precipitation and fertilizer are handled by MIKE SHE directly.

The root zone nitrogen model considers the fate of all three nitrogen species in the top meter of soil (Figure 2a). The processes of net mineralization, nitrification, and denitrification defined in ECO Lab are described by first order kinetics that include the influence of soil temperature and soil moisture on reaction rates. Soil temperature is modeled empirically by MIKE SHE at different soil depths based on air temperature (Klein, 1995) and the nitrogen transformations only take place when the soil temperature is above 5 ^{o}C (Neitsch et al., 2011; Bicknell et al., 1993). The soil water factors vary from 0-1 and were adapted from the European root zone model DAISY (Hansen et al., 1991) and APSIM (Holzworth et al., 2014). Net mineralization and nitrification rates are greatest at soil moisture saturations of 50-75% and 25-75%, respectively, while denitrification only occurs when the soil saturation is above 80% (Hansen et al., 1991). While agricultural systems models like RZWQM, APSIM, and DAISY consider soil organic matter dynamics which by extension requires carbon modeling, a simpler approach was adopted in this study. Following the approach implemented in an HSPF study of the Iowa River Basin (Imhoff et al., 1983), contributions to the organic nitrogen pool from plant roots, stubble, and crop residue were approximated by resetting the organic nitrogen concentrations in each soil layer to their initial values at the end of each growing season (a date of 1 Nov was assumed). One goal of this research was to assess if this simplified approach to modeling organic nitrogen would be adequate for the purposes of predicting NO_3-N leaching or if perhaps a more robust framework considering the complex interactions associated with soil organic matter would be required.

The shallow groundwater nitrogen model only includes NH_4-N, NO_3-N, and the processes of nitrification, denitrification, and plant uptake (Figure 2b). Nitrification and denitrification are described by first order, temperature dependent kinetics based on user-specified groundwater temperature.

Plant uptake and sources of inorganic nitrogen (soluble NH_4-N and NO_3-N) are handled directly by MIKE SHE. Plant uptake is treated as sink of nitrogen by MIKE SHE and is calculated empirically in proportion to the plant root water uptake and soluble inorganic nitrogen concentration in the soil. Sources of inorganic nitrogen from atmospheric wet deposition require user-defined concentrations in precipitation while fertilizer inputs require application rate and depth.

The MIKE SHE ECO Lab soil nitrogen model is simplified compared to field and watershed nitrogen models described earlier. As described previously, the major simplification of the MIKE SHE ECO Lab soil nitrogen model is in how organic nitrogen associated with soil organic matter is represented. The soil nitrogen model considers only one pool of organic nitrogen and a single net mineralization rate which lumps together the complex feedback processes associated with soil organic matter turnover and decomposition, mineralization, and immobilization. While this simplified representation of soil organic matter achieved reasonable annual results in a prior HSPF study (Imhoff et al., 1983), the other field and watershed models perform carbon modeling in which soil organic matter is subdivided into several carbon pools associated with soil humus, crop residue, and microbial biomass. Each carbon pool is characterized by a different reactivity rate (rate of decomposition/turnover) and carbon:nitrogen ratio which relates soil organic carbon content to organic nitrogen. Because MIKE SHE is primarily used for hydrologic applications at the watershed scale, the model structure does not support the simulation of plant growth processes considering nutrient availability and demand like in agricultural systems models. Finally, the soil nitrogen model does not consider legume fixation of nitrogen, NH_4-N adsorption and desorption processes (only soluble NH_4-N is considered), and volatilization losses.

MIKE SHE ECO Lab Simulation

A MIKE SHE ECO Lab simulation of a tile-drained Iowa field under a corn-soybean rotation from 2005-09 was conducted and simulated hydrology, crop uptake, and NO_3-N loss in subsurface drainage were compared to field measurements and model results from a prior RZWQM study (Qi et al., 2011). Details of the study area, agronomic activities, and RZWQM simulation are described in Qi et al. (2011), but a brief overview is provided here.

The field study was conducted at the Iowa State University (ISU) Agricultural Drainage Water Quality – Research and Demonstration Site (ADWQ-RDS) near Gilmore City in north central Iowa. The research site is located in the recently glaciated Des Moines Lobe, a landform region of low relief and poor surface drainage (Prior, 1991). The site was established in the late 1980s for long-term evaluation of subsurface drainage and nutrient management of different

Iowa cropping systems. The site contains 78 research plots, and the average slope of each plot is about 1% in the longitudinal direction and nearly flat in the transverse direction. Each 0.06-ha plot is 38 m in length and 15.2 m in width and contains a corrugated plastic drain tile installed at a depth of 1.06 m. During this five year (2005-09) study, corn was planted in odd years and soybeans in even years in mid-May. Aqueous ammonia-nitrogen fertilizer was applied to corn in the spring shortly following emergence (mid-May) at 140 kg N ha^{-1}.

The MIKE SHE plot model was adapted from a prior study that evaluated the ability of MIKE SHE to simulate subsurface drainage from a different plot at the same site under a corn-soybean-rye rotation from 2006-09 (Zhou et al., 2013). Terrain, soil, and weather conditions were assumed to be representative for all plots. The MIKE SHE plot model consisted of 206 2-m square grid cells, 27 unsaturated zone layers ranging from 5-30 cm in thickness, and one saturated zone layer extending to a depth of 3.9 m. Several changes were made to the baseline hydrologic model to match the agricultural management described in Qi et al. (2011) and to improve the simulation of subsurface drainage for a corn-soybean rotation.

The MIKE SHE ECO Lab model was also evaluated by simulating NO_3-N loss in subsurface drainage in response to variable nitrogen fertilizer rates. The 2005-09 water quality simulation was repeated at nine different fertilizer rates (0-252 kg ha^{-1}) and simulated annual flow-weighted average NO_3-N concentration (FWANC) under each fertilizer scenario was compared to observations (Lawlor et al., 2008) and RZWQM results (Qi et al., 2012) conducted at the research site from 1989-2004. While the time period of this study (2005-09) is not the same as the field experiment (1989-2004), making direct comparisons difficult, the general relationship between fertilizer rate and FWANC observed by Lawlor et al. (2008) was assumed to still be valid.

Results and Discussion

Model Calibration

The MIKE SHE hydrologic model from Zhou et al. (2013) was altered to match study conditions described in Qi et al. (2011) and to improve the simulation of water balance components. Hourly precipitation and daily potential evapotranspiration datasets used by Zhou et al. (2013) were replaced with daily datasets used by Qi et al. (2011). Because the daily precipitation time series from Qi et al. (2011) already included snow water equivalent, snowmelt modeling was excluded from the simulation. In the evapotranspiration model, the dates and length of the growing season in the evapotranspiration model were adjusted to match plant/harvest dates reported in Qi et al. (2011), and the crop coefficients (K_c) for corn and soybeans were adjusted to reflect reference evapotranspiration calculated for a grass reference crop (Allen et al., 1998). Finally, a model coefficient controlling canopy interception was reduced to increase plant transpiration and soil evaporation while maintaining approximately the same total evapotranspiration to improve agreement with RZWQM-simulated evapotranspiration components (Qi et al., 2011). The hydrologic modeling was performed in two steps to minimize the effect of initial conditions on simulated water balance components and subsurface drainage (Ajami et al., 2014). First, a recursive simulation was ran using weather data from 2004 until a pseudo state-state equilibrium was achieved, defined as when the plot-integrated surface and subsurface storage changes between consecutive years approached zero. Using the end result from the recursive simulation, a continuous simulation was ran from 2004-09 with model results from 2004 excluded from the analysis.

The soil nitrogen model required 11 parameters to be specified that were assigned through a literature review and/or calibration (Table 2). The water quality simulation was ran from 2005-09, and the initial concentrations (in water, not soil) of NO_3-N and NH_4-N in the unsaturated and saturated zones were set to 10 mg l^{-1} and 0 mg l^{-1}, respectively.

Simulated Hydrology

Annual water balance components were simulated reasonably well by the revised MIKE SHE plot model (Table 3). The 5-yr average precipitation at the site was 827 mm, with annual totals ranging from 627 mm in 2006 to 1050 mm in 2007. The simulated 5-yr average subsurface drainage was 313 mm, close to the observed 5-yr average of 311 mm. Observed annual subsurface drainage varied considerably among individual corn-soybean plots, but the simulated annual drainage totals fell within the observed minimum and maximum range. Following recommendations from Moriasi et al. (2007), Qi et al. (2011) considered the simulation of annual subsurface drainage "satisfactory" when several statistical skill scores – Nash-Sutcliffe Efficiency (NSE), percent bias (PBIAS), and root mean square error to the standard deviation of measured data (RSR) – met certain quantitative criteria defined as NSE > 0.5, PBIAS within ±25%, and RSR $\leq$ 0.7. Based on these criteria, simulated annual subsurface drainage by MIKE SHE was satisfactory.

The simulated evapotranspiration was also comparable to RZWQM simulation results and other comparable studies. The simulated 5-yr average evapotranspiration of 497 mm was 10% greater than the 453 mm 5-yr average simulated by Qi et al. (2011) and 6% greater than the 468 mm simulated by Thorp et al. (2007) in a previous RZWQM study at the research site. Simulated evapotranspiration from May-Sep in the five years ranged from 379-466 mm, comparable to that simulated by Qi et al. (2011) (344-468 mm) and observed by Bakhsh et al. (2004) from 1992-94 in central Iowa (334-493 mm). Finally, the simulated average transpiration in 2007-09 was 248 mm, close to the 256 mm simulated by Qi et al. (2011), while the simulated average soil evaporation in 2007-09 of 165 mm was 23% less than the 214 mm simulated by Qi et al. (2011).

MIKE SHE also performed satisfactorily in predicting monthly drainage from April 2006 to Dec 2009 with values of -13%, 0.73, and 0.52 for PBIAS, NSE, and RSR, respectively (Figure 3). While simulated and observed monthly drainage totals were comparable during the growing season (May-Sep), the model struggled to replicate observations in the spring and fall as March drainage was always overestimated when no drainage was observed and Oct drainage was always underestimated. Qi et al. (2011) compared simulated and observed daily drainage during 2007 and 2008, noting that little drain flow was measured after mid-April 2006 and that the flow meter in the fall of 2009 did not work well, so it is possible that the high observed monthly drainage in Oct and Nov of 2009 may be questionable. Overestimation of simulated drainage in early spring and March is likely a result of snowmelt and freeze-thaw dynamics near the tile depth that were not accurately represented by MIKE SHE. However, it was not entirely clear if the field study monitored subsurface drainage throughout the year or only during the warmer months.

Simulated Nitrogen Cycling

In general, the simulated annual nitrogen balance was comparable to observations or literature estimates but was simulated in a poorer manner than the annual water balance (Table 4). The simulated 5-yr average net mineralization rate was 169 kg ha^{-1}, comparable to the 5-yr average RZWQM-simulated rate of 140 kg ha^{-1} and 189 day field-measured value of 142 kg ha^{-1} for soils under a corn-soybean rotation in Brookings, South Dakota (Carpenter-Boggs et al., 2010). The long-term (1970-2009) average annual RZWQM-simulated net mineralization rate was 168 kg N ha^{-1}. The simulated annual denitrification included contributions from both the root zone (top meter of the unsaturated zone) and groundwater to a depth of 3.9 m. Simulated denitrification in the root zone accounted for 88% of the 5-yr total with annual totals ranging from 10 kg ha^{-1} in 2006 (year of lowest annual precipitation and simulated/observed subsurface drainage) to 30 kg ha^{-1} in 2007 (year of greatest annual precipitation and second highest simulated subsurface drainage). The simulated 5-yr average denitrification rate of 21 kg ha^{-1} was nearly 3.5 times larger than the long-term (1970-2009) average of 6.1 kg ha^{-1} simulated by RZWQM (Qi et al., 2011) but comparable to the 16-yr (1989-2004) simulated average of 17.3 kg ha^{-1} for a corn-soybean plot subject to different fertilizer rates (Qi et al., 2012). The simulated 5-yr average field denitrification rate was also within the range reported for agricultural watersheds in the U.S. Midwest (10-23 kg ha^{-1}; Li et al., 2010).

Simulated crop nitrogen uptake was considered "satisfactory" when percent error (PE) was less than ±15% and relative root mean square error (RRMSE) was less than 30% (Qi et al., 2011). MIKE SHE performed close to a satisfactory level in simulating the 4-yr (2006-09) average crop nitrogen (NO_3-N + NH_4-N) uptake as the PE was -2% and RRMSE was 31%. The simulated 4-yr average crop nitrogen uptake of 174 kg ha^{-1} was close to the observed 4-yr average of 181 kg ha^{-1}, but corn nitrogen uptake was overestimated by 151% in 2007 and soybean nitrogen uptake was underestimated by 48% in 2008; RZWQM also overestimated corn uptake in 2007 and underestimated soybean nitrogen uptake in 2008, though to a lesser degree. In 2007, fertilizer application combined with lower than normal precipitation in Jun and Jul resulted in large simulated storages of NO_3-N in the unsaturated zone which allowed for particularly high crop uptake from Jul-Sep. In 2008, no fertilizer application combined with high precipitation and simulated subsurface drainage in Apr-Jun resulted in these three months accounting for 90% (65 kg ha^{-1}) of the simulated annual NO_3-N loss which depleted the store of soil NO_3-N.

While the simulation of annual NO_3-N loss in subsurface drainage was considered unsatisfactory by statistical measures (except for PBIAS), the simulation results were believable. The simulated 5-yr average NO_3-N loss of 36 kg ha^{-1} was comparable to the observed 5-yr average of 41 kg ha^{-1} while simulated NO_3-N losses in individual years ranged from 6 kg ha^{-1} in 2006 (year of lowest simulated subsurface drainage) to 73 kg ha^{-1} in 2008 (year of highest simulated subsurface drainage). Simulated NO_3-N loss was consistent with the simulated hydrology and within the range observed and/or modeled in other Iowa field studies (Baker et al., 1975; Bakhsh et al., 2002; Bakhsh et al., 2004; Thorp et al., 2007).

Simulated and observed annual NO_3-N loss and FWANC, along with the ranges observed for all corn-soybean plots during the study, are summarized in Table 5. Similar to the wide range in observed annual subsurface drainage among corn-soybean plots (Table 3), observed annual NO_3-N losses among plots were also highly variable. Simulated annual NO_3-N loss was comparable to the average NO_3-N loss in 2005, within the range observed in 2007 and 2008, and lower than, though close to, the observed minimum NO_3-N losses in 2006 and 2009. The years of 2006 and 2009 had the lowest annual precipitation and simulated/observed subsurface drainage during the study period. In 2006, 90% (16.8 kg ha^{-1}) of the observed average annual NO_3-N loss occurred in April and May, while the simulated NO_3-N loss during these two months was only 3.5 kg ha^{-1}. Given the simulated subsurface drainage in April and May were comparable to observations, simulated NO_3-N losses in these two months were likely lower than expected because of a low soil nitrogen supply and no fertilizer application in this year. Additionally, the overestimation of simulated subsurface drainage in March (58 mm vs none observed) resulted in one-third (2.0 kg ha^{-1}) of the simulated annual NO_3-N loss. In addition to the inability of the hydrologic model to retain soil water during freeze-thaw conditions like those typically observed in March, it is also possible the simplified representation of NH_4-N as an entirely soluble species in the soil nitrogen model may lead to overestimated nitrification and leaching during the cold season when it is known that NH_4-N remains attached to soil and nitrification is inhibited at soil temperatures below 10 ^{o}C (Sawyer, 2011). Additionally, MIKE SHE only allows soluble species to be specified as water quality sources, so anhydrous ammonia fertilizer application was represented as a soil input of soluble NH_4-N. This representation of anhydrous ammonia is also inaccurate, and these two limitations may result in greater than expected soluble NH_4-N loss that lessens the current and future store of soil NO_3-N.

Simulated NO_3-N loss in 2009 was lower than expected primarily because of underestimated subsurface drainage in Oct and Nov. In these two months, the average observed subsurface drainage and NO_3-N loss were 124 mm and 18.1 kg ha^{-1} (64% of the total annual NO_3-N loss), respectively, while only 11 mm of subsurface drainage and 0.2 kg ha^{-1} of NO_3-N loss were simulated. Difficulties measuring subsurface drainage in the fall of 2009 were noted in the study, however, which may have impacted the associated NO_3-N losses.

Trends in simulated FWANC were similar to simulated NO_3-N loss. The simulation of FWANC was unsatisfactory by statistical measures (except for PBIAS), which was also the case for the RZWQM simulation. As was the case for simulated NO_3-N loss, simulated and observed FWANC were comparable in 2005, 2007, and 2008, while simulated FWANC in 2006 and 2009 were underestimated by 69-76%. The 5-yr average simulated FWANC was 9.8 mg l^{-1}, 29% less than the 5-yr average observed FWANC of 13.9 mg l^{-1}. Excluding 2006 and 2009, the 3-yr average simulated FWANC was 13.6 mg l^{-1}, close to the 3-yr average observed FWANC of 13.2

mg l^{-1}. Unlike the large variability in observed annual subsurface drainage and NO_3-N loss among corn-soybean plots, FWANC was fairly consistent across study years at all plots, ranging from a minimum 5-yr average of 12.0 mg l^{-1} to a maximum 5-yr average of 14.0 mg l^{-1}. The underestimation of simulated FWANC in 2006 and 2009 is attributed to both overestimated subsurface drainage volume and underestimated NO_3-N loss.

Simulated Response of FWANC to Fertilizer Rate

The 5-yr MIKE SHE ECO Lab water quality simulation was repeated nine separate times at the nine fertilizer rates (0, 45, 56, 90, 112, 134, 168, 179, and 252 kg N ha^{-1}) applied in the field study from 1989-2004 (Lawlor et al., 2008) to evaluate the simulated response of FWANC to fertilizer rate. The relationship between fertilizer rate and FWANC simulated by MIKE SHE ECO Lab is shown in Figure 4 for each year and the 5-yr average of each fertilizer simulation, along with the observed (Lawlor et al., 2008) and RZWQM-simulated (Qi et al., 2012) average responses. MIKE SHE ECO Lab was not able to reproduce the exponential relationship between fertilizer rate and FWANC observed or simulated by RZWQM. While observed FWANC increased exponentially with fertilizer rate from 7.1 mg l^{-1} when no fertilizer was applied to 24.0 mg l^{-1} at a rate of 252 kg ha^{-1} (a similar range in FWANC was predicted by RZWQM, which was not statistically different ($\alpha = 0.05$) from the observations), MIKE SHE ECO Lab predicted a linear relationship between fertilizer rate and FWANC that did not capture the range of observed FWANC; the 5-yr average FWANC simulated by MIKE SHE ECO Lab ranged from 8.6 mg l^{-1} when no fertilizer was applied to 10.8 mg l^{-1} at 252 kg ha^{-1}. Additionally, the simulated FWANC in 2005, 2006, and 2009 remained fairly constant across fertilizer scenarios.

A review of the simulated nitrogen mass balances helped explain the simulated behavior and identify shortcomings of the current MIKE SHE ECO Lab soil nitrogen model. Comparing simulated nitrogen components from the minimum (0 kg ha^{-1}) and maximum (252 kg ha^{-1}) fertilizer scenarios, net mineralization remained the same (modeled as a function of organic nitrogen concentration, soil temperature, soil moisture, and the vertical extent of the unsaturated zone), and denitrification increased from a 5-yr average of 17 kg ha^{-1} to 25 kg ha^{-1}. While this represented a 44% increase, the masses represented a small fraction of the total nitrogen inputs. For comparison, RZWQM-simulated net mineralization increased by 12 kg ha^{-1} (13% increase) and denitrification increased from 7 kg ha^{-1} to 33 kg ha^{-1} between the minimum and maximum fertilizer scenarios (Qi et al., 2012). While NH_4-N inputs increased as more fertilizer was applied, the subsurface storages and leaching of NH_4-N remained nearly identical between scenarios, indicating increased crop uptake and/or nitrification. Hence, the major differences between fertilizer scenarios was attributed to the NO_3-N mass balance.

Figure 5 shows the simulated annual NO_3-N loss, combined crop uptake of NO_3-N and NH_4-N, and subsurface storages of NO_3-N in the unsaturated and saturated zones for the nine fertilizer scenarios. The simulated 5-yr average NO_3-N loss increased by 32% from 30 kg N ha^{-1} when no fertilizer was applied to 40 kg N ha^{-1} at the maximum rate. However, the majority of the fertilizer applied in each scenario was either taken up by crops or stored as NO_3-N in the unsaturated zone. Note the greater rates of increase in both simulated crop uptake and unsaturated zone NO_3-N storage in fertilizer years (odd years). Simulated annual NO_3-N loss in 2005, 2006, and 2009 remained fairly constant even as fertilizer rate increased because the majority of the simulated leaching occurred outside the growing season, primarily in winter and early spring (Feb-May) prior to corn planting and fertilizer application. Most of the fertilizer that was applied was either taken up by crops or stored in the unsaturated zone above the groundwater table. In 2007 and 2008, simulated NO_3-N loss did increase with fertilizer rate because the wet conditions resulted in simulated NO_3-N losses during the growing season which became more pronounced as fertilizer rate increased. This is illustrated by the greater amount and rate of increase of saturated zone storage of NO_3-N simulated in these two years compared to the others.

Simulated crop nitrogen uptake increased linearly with fertilizer rate (top right panel of Figure 5). While few studies were identified directly comparing crop nitrogen uptake to fertilizer rate, it is well documented that crop yield diminishes with fertilizer rate (Figure 6; Sawyer 2015; Greer and Pittelkow, 2018), while a relatively linear relationship between crop yield and crop nitrogen uptake for both corn and soybeans has been observed (Figure 7; Salvagiotti et al., 2008; Oliveira et al., 2018). Since crop nitrogen uptake is directly proportional to crop yield, uptake can be approximated as some linear fraction of yield. Therefore, we would expect the relationship between crop nitrogen uptake and fertilizer rate to also be non-linear in which crop nitrogen uptake diminishes with increasing fertilizer rate. In the current modeling framework, plant uptake of solutes (soluble NH_4-N and NO_3-N) is calculated with a simple empirical function by MIKE SHE (not calculated in ECO Lab). Plant uptake is treated as a solute sink and is directly proportional to root water uptake (transpiration), solute concentration in the particular unsaturated zone or saturated zone layer, and an empirical concentration factor that is specified for each solute but applies to all vegetation types. This simplified, pseudo first order approach assumes plant uptake is directly proportional to solute concentration (all other things being equal) and does not consider actual nutrient requirements needed for plant growth that are included in the plant growth processes of agricultural systems models. Hence, the simulated crop nitrogen uptake is directly proportional to fertilizer rate even though crop nitrogen demand may be met at low to moderate fertilizer rates. This shortcoming in the current plant uptake function inhibits a buildup of residual soil NO_3-N at higher fertilizer rates, which would have a greater potential for loss in subsurface drainage.

Shortcomings of the MIKE SHE plant uptake function partially explain the inability of MIKE SHE ECO Lab to simulate the correct trend and range in FWANC in response to fertilizer rate. No consideration of adsorbed NH_4-N in the ECO Lab soil nitrogen model and the ability to only apply soluble water quality sources in MIKE SHE are other limitations with the water quality model that may have a substantial influence on model performance. Until these issues are resolved, site and scenario specific calibration of the plant uptake function (empirical concentration factor) and other soil nitrogen parameters may be required to achieve reasonable NO_3-N leaching estimates.

Summary and Conclusions

A soil nitrogen model was developed with MIKE SHE ECO Lab to simulate nitrogen cycling and NO_3-N loss from a tile-drained, corn-soybean plot in north central Iowa from 2005-09. The nitrogen model includes particulate organic nitrogen, soluble NH_4-N, and NO_3-N and simulates the processes of net mineralization, nitrification, denitrification, and crop uptake, and includes soluble sources of nitrogen in precipitation and represented as fertilizer. The model was evaluated by comparing simulated water balance and nitrogen components with observations and RZWQM simulation results. Additionally, the simulated response of FWANC to nitrogen fertilizer rate was compared to observations from an earlier (1989-2004) study conducted at the research site.

Simulated annual water and nitrogen balances were in general agreement with observations from the field study, RZWQM results, and other literature estimates. Annual and monthly subsurface drainage were simulated satisfactorily by the model according to several statistical criteria, though subsurface drainage was consistently overestimated in March and underestimated in Oct. The 5-yr average simulated net mineralization, denitrification, crop nitrogen uptake, and NO_3-N loss in subsurface drainage compared favorably with field observations or results from the RZWQM simulation. While simulated annual NO_3-N loss was unsatisfactory by statistical measures, the NO_3-N losses were in general agreement with field observations and literature estimates. Simulated annual NO_3-N loss was within the range of observed values in the three wettest years of the study, while simulated NO_3-N loss was severely underestimated in the two driest years (2006 and 2009). Discrepancies in simulated annual NO_3-N loss in 2006 and 2009 were attributed primarily to a low supply of soil nitrogen in 2006 and an underestimation of simulated subsurface drainage in the fall of 2009 when the majority of the observed NO_3-N loss took place.

The MIKE SHE ECO Lab model was not able to predict the exponential response between fertilizer rate and FWANC observed in an earlier study and correctly simulated by RZWQM. Simulated FWANC increased linearly with fertilizer rate and the range of FWANC was much narrower and lower than observed. Simulated FWANC remained relatively constant across fertilizer rates in 2005, 2006, and 2009 due to a combination of simulated NO_3-N loss outside the growing season when fertilizer was simulated to have little effect, dry conditions during the growing season which enabled large amounts of crop uptake and a large storage of NO_3-N in the unsaturated zone, and biases in the simulated hydrology during March and Oct which influenced the amount and timing of simulated NO_3-N loss. In 2007 and 2008, simulated FWANC did increase with fertilizer rate because wet conditions resulted in simulated NO_3-N losses during the growing season which became more pronounced with fertilizer application rate. The inability of the MIKE SHE ECO Lab model to predict the correct response of FWANC to fertilizer rate is attributed in part to the MIKE SHE plant uptake function which did not capture the non-linear relationship expected between crop nitrogen uptake and fertilizer rate, as well as limitations with how NH_4-N and fertilizer sources are represented in the model.

Simulating hydrologic and nitrogen cycling for agricultural areas in Iowa with MIKE SHE ECO Lab is the first of its kind. The soil nitrogen model developed is simple and requires revision but does show promise for predicting annual nitrogen components in conventional agricultural fields. Limitations of the current soil nitrogen model include the exclusion of soil organic matter dynamics to derive mineralization estimates, absence of adsorption/desorption processes for NH_4-N which may result in overestimated nitrification and leaching during the cold season and a shortage of soil NO_3-N during the growing season, no consideration of legume fixation or volatilization losses, lack of plant growth considerations in the current MIKE SHE plant uptake function, and the ability to only define soluble water quality sources in MIKE SHE. These simplifications, along with errors in the simulated hydrology in March and Oct, are believed to be the primary reasons for the poor performance of simulated NO3-N loss in dry years and with fertilizer rate. Despite these shortcomings, simulated nitrogen components were comparable to annual observations and literature estimates for agricultural areas in the U.S. Midwest. The current soil nitrogen model will be used in Chapters 4-5 to derive stream nitrogen estimates in order to assess wetland NO_3-N removal.

Tables and Figures

Table 2. Calibrated parameters for the MIKE SHE ECO Lab soil nitrogen model.

Parameter	Component	Value	Source
Organic nitrogen concentration	Unsaturated zone	300 g m^{-3} soil: 0-0.3 m depth 100 g m^{-3} soil: 0.3-1 m depth	Calibrated; based on values from Gregorich and Anderson, 1985; Hansen et al., 1991; Ma et al., 2007; Thorp et al., 2007
First order rate constant for net mineralization at 20 °C		0.001 day^{-1}: 0-0.3 m depth $3.3x10^{-4}$ day^{-1}: 0.3-1 m depth	Calibrated; based on values from Imhoff et al., 1983; Hansen et al., 1991
First order rate constant for nitrification at 20 °C		0.1 day^{-1}: 0-1 m depth	Imhoff et al., 1983
First order rate constant for denitrification at 20 °C		0.004 day^{-1}: 0-1 m depth	Imhoff et al., 1983
First order rate constant for nitrification at 20 °C	Saturated zone	0.01 day^{-1}	Assumed
First order rate constant for denitrification at 20 °C		0.002 day^{-1}	Vervloet et al., 2018
Arrhenius temperature coefficient	Unsaturated and saturated zones	1.06	Bicknell et al., 1993
NH_4-N concentration in precipitation	MIKE SHE	0.5 mg l^{-1}	Qi et al., 2011
NO_3-N concentration in precipitation		1.3 mg l^{-1}	Qi et al., 2011
Aqueous NH_4-N fertilizer application depth		0.15-0.2 m depth	Sawyer, 2011; Vitosh, 2011
Empirical plant uptake factor		1 for both NH_4-N and NO_3-N	Calibrated

Table 3. Simulated and observed annual water balance components for a tile-drained, corn-soybean plot in north central Iowa from 2005-09.

Year	Crop	P	ET	Surface Runoff	Subsurface Storage Change	Subsurface Drainage			
						Sim	Obs (Avg)	Obs (Min)	Obs (Max)
		mm							
2005	Corn	846	539	3	40	263	258		
2006	Soybean	627	532	0	-66	161	124	65	215
2007	Corn	1050	520	16	86	428	488	104	892
2008	Soybean	926	445	4	7	471	492	226	1053
2009	Corn	684	451	1	-8	241	192	114	335
Avg		827	497	5	12	313	311	127	624
PBIAS						1%			
NSE						0.93			
RSR						0.26			

Table 4. Simulated and observed annual nitrogen components for a tile-drained, corn-soybean plot in north central Iowa from 2005-09.

Year	Crop	N Rate	Net Mineralization	Denitrification	Crop N Uptake		NO_3-N Loss	
					Sim	Obs	Sim	Obs
		kg N ha^{-1}						
2005	Corn	140	181	15	195		33	38
2006	Soybean	0	178	10	188	201	6	19
2007	Corn	140	172	30	219	145	55	66
2008	Soybean	0	159	19	90	174	73	56
2009	Corn	140	153	29	199	205	11	28
Avg			169	21	174	181	36	41
				PE	-4%	PBIAS	-14%	
				RRMSE	31%	NSE	0.42	
						RSR	0.76	

Table 5. Simulated and observed NO_3-N loss in subsurface drainage and FWANC for a tile-drained, corn-soybean plot in north central Iowa from 2005-09.

Year	NO_3-N Loss (kg N ha^{-1})				FWANC (mg l^{-1})			
	Sim	Obs (Avg)	Obs (Min)	Obs (Max)	Sim	Obs (Avg)	Obs (Min)	Obs (Max)
2005	33	38			12.4	14.7		
2006	6	19	9	31	3.7	15.1	13.7	14.4
2007	55	66	15	105	13.0	13.5	11.8	14.1
2008	73	56	27	101	15.4	11.3	9.6	11.8
2009	11	28	15	52	4.6	14.7	12.9	15.6
Avg	36	41	16	72	9.8	13.9	12.0	14.0
PBIAS	-14%				-29%			
NSE	0.42				-25.43			
RSR	0.76				5.14			

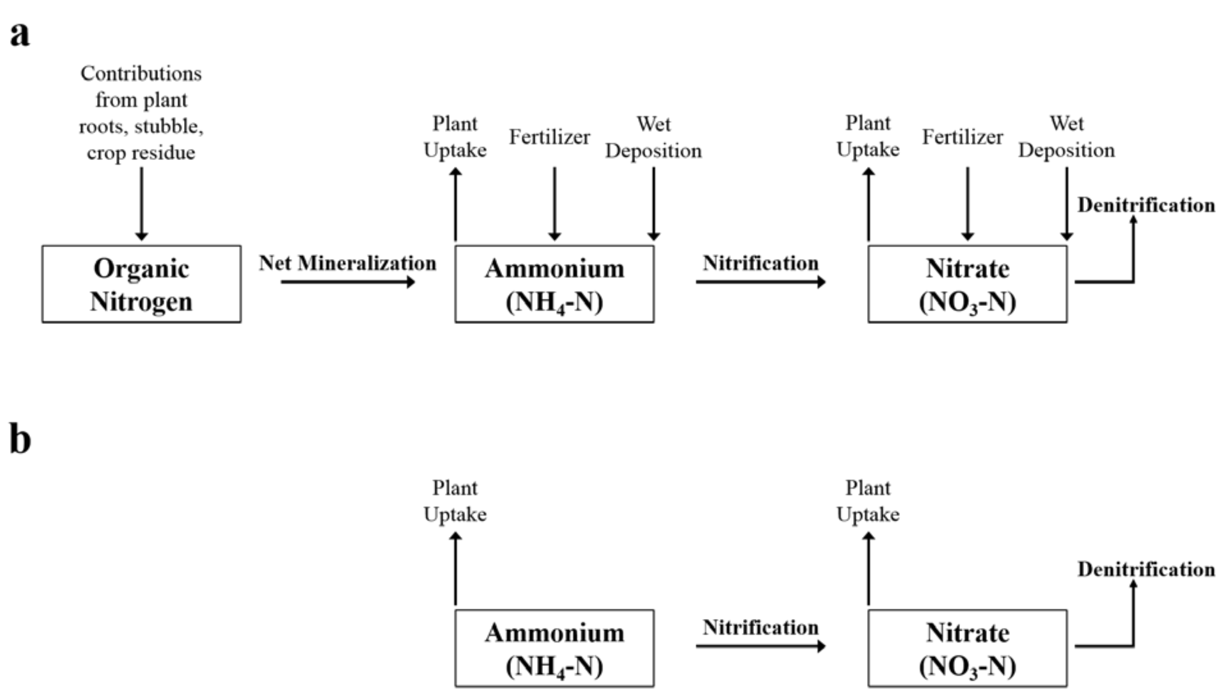

Figure 2. Soil nitrogen conceptual model (MIKE SHE ECO Lab). (a) Root zone model (top meter of unsaturated zone) (b) Groundwater model (saturated zone).

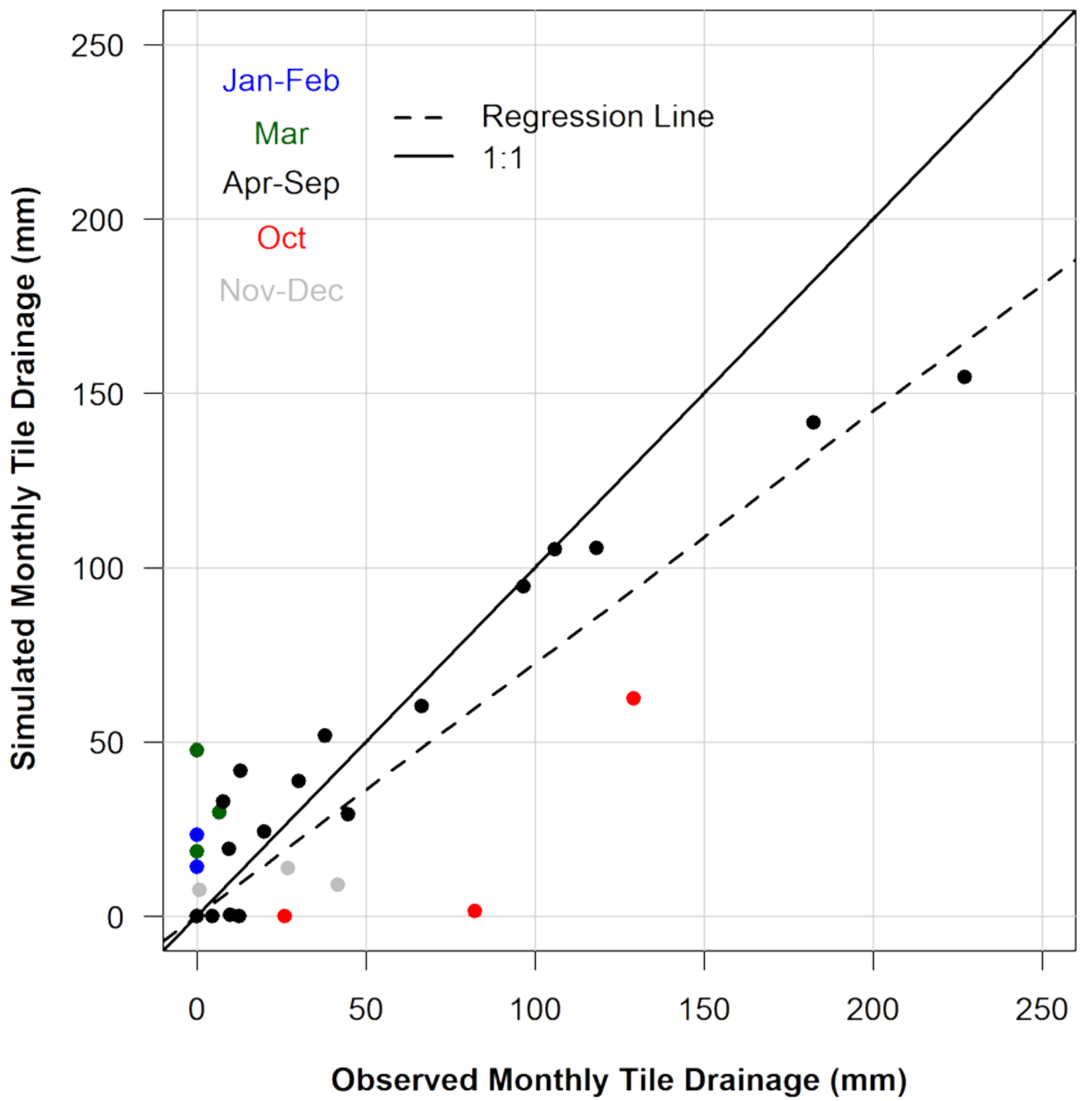

Figure 3. Comparison of simulated and observed monthly subsurface drainage from Apr 2006 to Dec 2009.

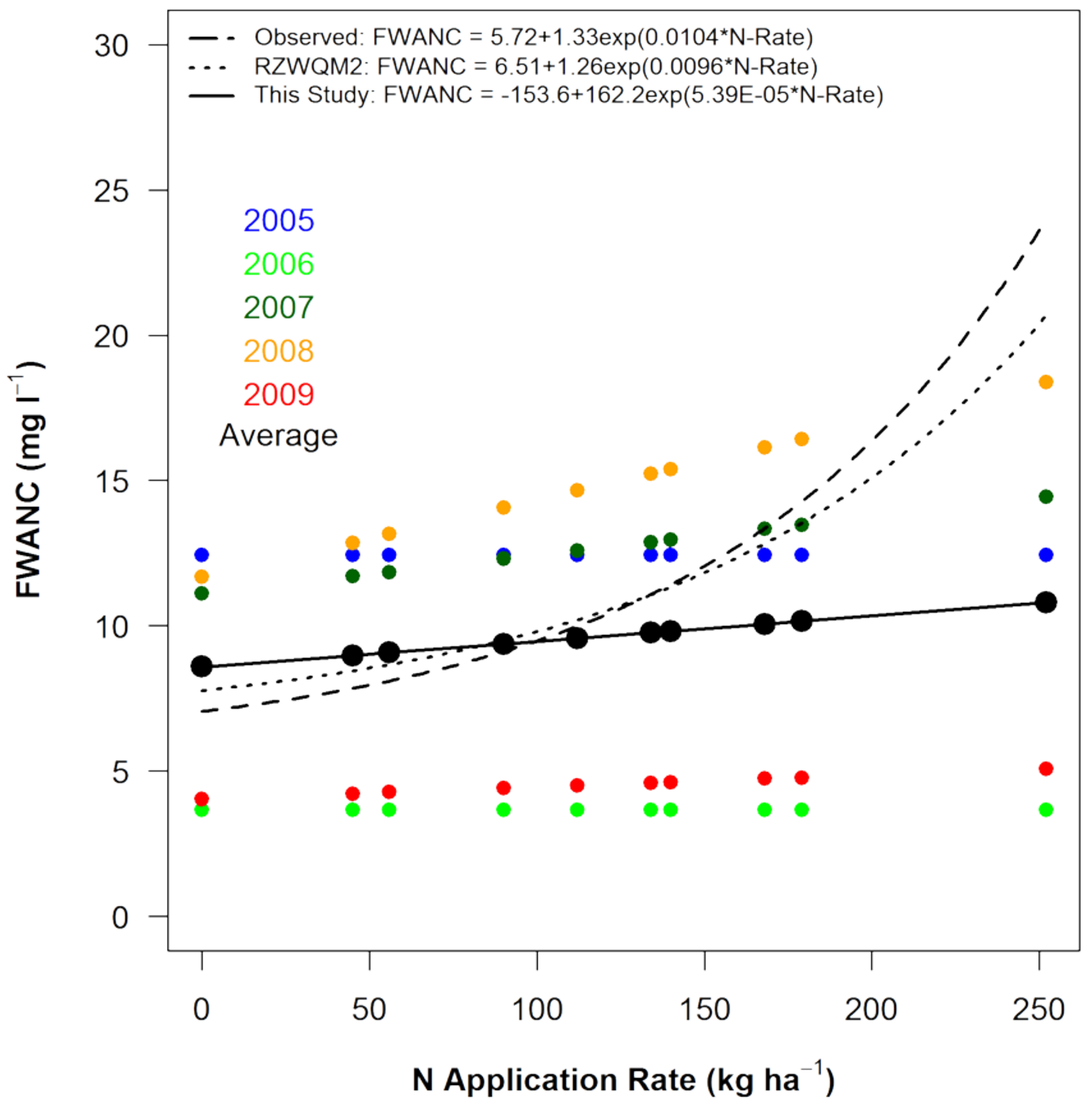

Figure 4. Effect of nitrogen fertilizer rate on annual FWANC in subsurface drainage simulated by MIKE SHE ECO Lab as compared to observations (Lawlor et al., 2008) and RZWQM-simulation results (Qi et al., 2012).

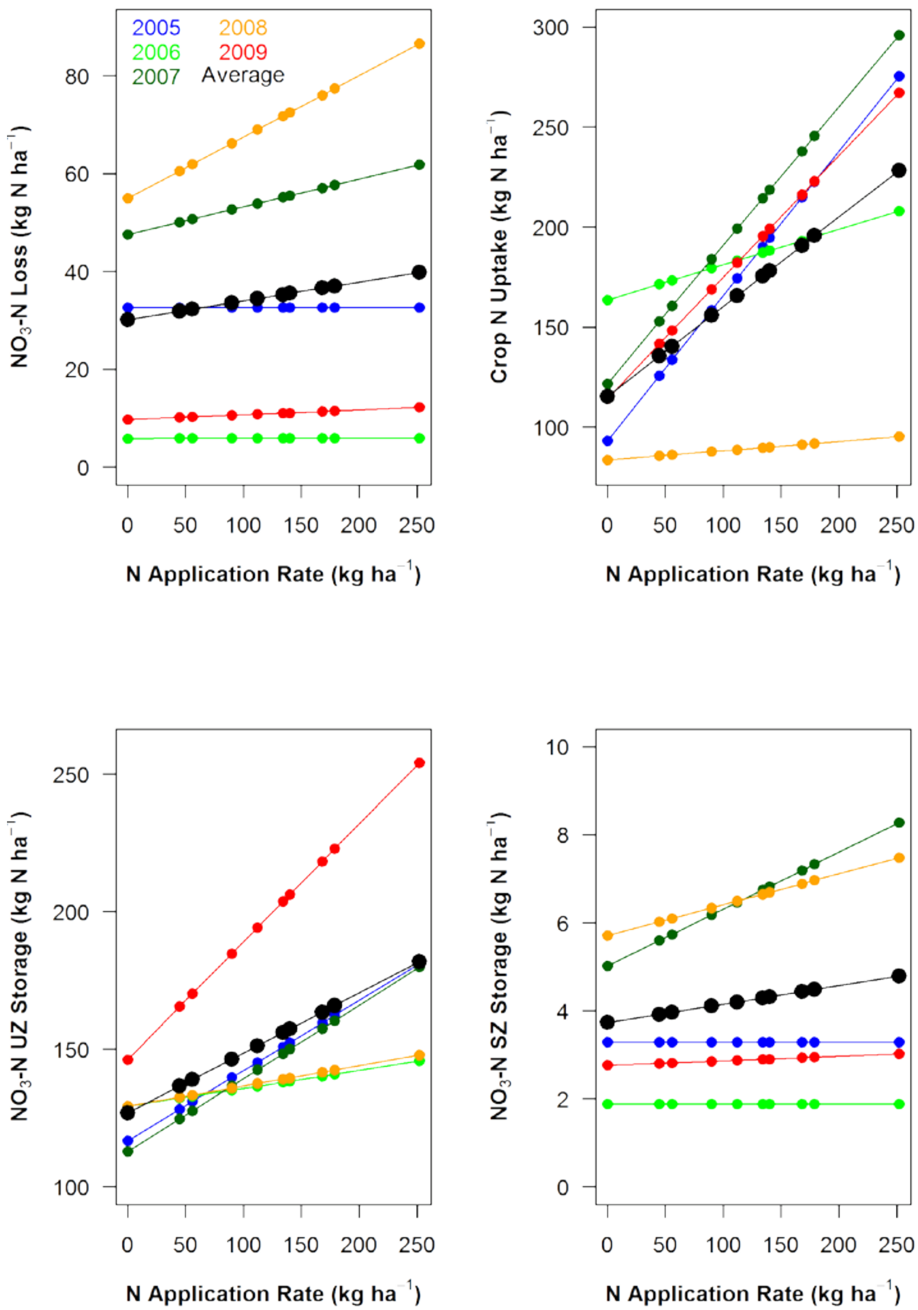

Figure 5. Simulated NO_3-N loss in subsurface drainage (top left), crop nitrogen uptake (top right), and subsurface storages of NO_3-N (bottom) for the different fertilizer scenarios.

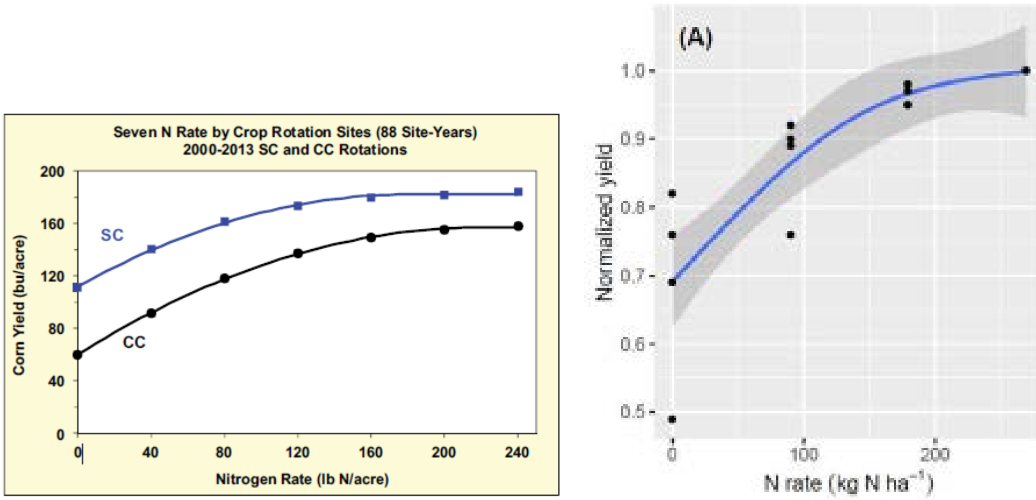

Figure 6. Observed relationships between crop yield and fertilizer rate. (Left) Soybean-corn (SC) and continuous corn (CC) rotations (Sawyer, 2015). (Right) Soybean-corn rotation (Greer and Pittelkow, 2018).

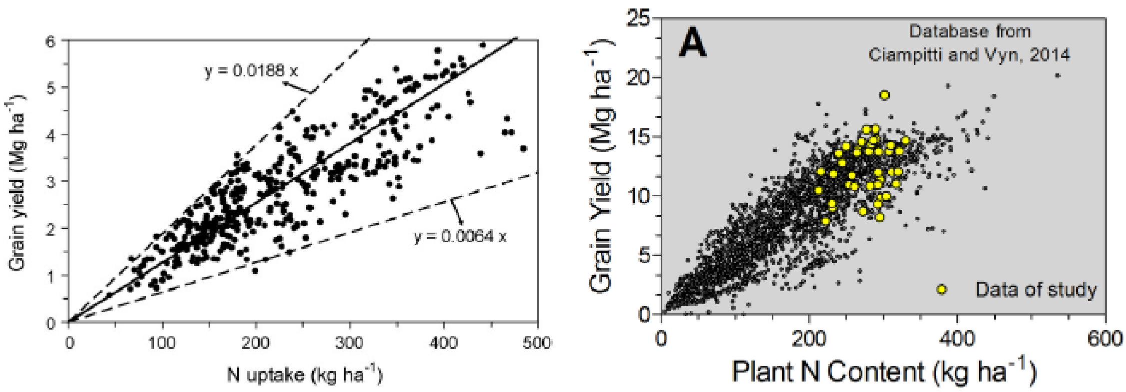

Figure 7. Observed relationships between crop yield and crop nitrogen uptake. (Left) For soybeans (Salvagiotti, et al., 2008). (Right) For corn (Oliveira et al., 2018).

3. ESTIMATING NITRATE-NITROGEN RETENTION IN A LARGE CONSTRUCTED WETLAND USING HIGH-FREQUENCY CONTINUOUS MONITORING AND HYDROLOGIC MODELING

This chapter describing nitrate retention patterns at Iowa's second largest CREP wetland was submitted to *Ecological Engineering* 10 July 2017 and accepted after a second revised submission 30 March 2018. https://doi.org/10.1016/j.ecoleng.2018.03.014

Abstract

Wetlands are an effective edge-of-field conservation practice for reducing agricultural nitrate-nitrogen (NO_3-N) loads, but their removal performance varies with hydrologic conditions and other factors difficult to capture with traditional grab sampling schemes. We quantified NO_3-N retention in a large Iowa constructed wetland using high-frequency (15-min) in situ NO_3-N sensors and a physically-based hydrologic model that estimated discharge. Monitoring from May-Nov over a 3-yr period (2014-16) indicated the wetland reduced incoming NO_3-N concentrations 49% and loads by an estimated 61 kg day^{-1} (0.48 g m^{-2} day^{-1} based on wetland area removal). Monthly and seasonal (May-Nov) wetland retention performance were significantly influenced by hydrologic conditions, as NO_3-N concentration reductions ranged from 23% in a year that received nearly 50% more seasonal precipitation than average (2016) to 59-65% in years that received average seasonal precipitation (2014-15). On a monthly basis, NO_3-N mass retention was highest in Jun when NO_3-N loading was highest, while retention efficiency – the percent of the incoming NO_3-N load retained by the wetland – was highest in Jul and Aug when water temperature and hydraulic residence time were higher. The high-frequency monitoring captured NO_3-N dynamics not possible with lower-frequency sampling. Extrapolating the May-Nov 3-yr average wetland NO_3-N retention estimated in this study to a much larger scale, over 5600 wetlands treating more than 60% of Iowa's area and totaling an

estimated $1.5 billion in design and construction would be required to reduce the state's baseline NO_3-N load by 45%, indicating the sizable investment in wetland construction and restoration needed to achieve Gulf of Mexico Hypoxia water quality goals.

Key Words: wetland; continuous nitrate sensor; nitrate-nitrogen retention; hydrologic modeling; Iowa

Introduction

Nitrogen export from agricultural areas in the Midwestern U.S. impairs aquatic ecosystem health at local and regional scales through eutrophication caused by nutrient enrichment (Hynes, 1969; USEPA, 2008, 2016). The U.S. EPA has called for a 45% reduction in annual nitrogen and phosphorus loading (relative to the 1980-1996 average annual load) in the Mississippi-Atchafalaya River Basin (MARB) to reduce the 5-yr average size of the summer hypoxic zone in the Gulf of Mexico to 5000 km^2 or less by 2035 (NTF, 2008, 2015). The Upper Mississippi River Basin (UMRB), which accounts for 15% of the MARB by area and comprises large portions of several Corn Belt states including Iowa, delivered 45% of the annual nitrate-nitrite nitrogen (NO_x-N) load in the MARB from 2000-15 (Aulenbach et al., 2007). Iowa and Illinois (9% of the MARB by area) account for an estimated 35% of the total nitrogen flux in the MARB and as much as double this in flood years (Goolsby et al., 1999). Therefore, implementing and quantifying the performance of conservation practices that target nutrient reduction in agricultural areas like Iowa is critical to achieving Gulf Hypoxia water quality goals.

Conservation practices are particularly needed to reduce nitrate-nitrogen (NO_3-N) export from tile-drained landscapes. Subsurface tile drainage removes excess soil water to improve crop yields but also hastens delivery of NO_3-N to streams (Robertson and Saad, 2011) and is a major source of NO_3-N loading to streams and rivers in the Corn Belt region (Schilling et al., 2012;

McLellan et al., 2015). An estimated 37% of the Corn Belt is tile-drained (Fausey et al., 1995), and average NO_3-N concentrations and yields (NO_3-N load divided by drainage area) in tile-drained corn-soybean systems often exceed 20 mg l^{-1} and 40 kg ha^{-1} yr^{-1}, respectively (Jaynes et al., 2001; Ikenberry et al., 2014).

Wetlands are one effective edge-of-field strategy for reducing NO_3-N loads in cropped, tile-drained landscapes (Kovacic et al., 2000; Crumpton et al., 2006; Groh et al., 2015). NO_3-N removal in wetlands occurs through denitrification and plant assimilation (Ingersoll and Baker, 1998). Denitrification can account for 60-80% of the NO_3-N removal in natural wetlands (Cooke, 1994; Crumpton and Goldsborough, 1998) and more than 90% in constructed wetlands (Xue et al., 1999; Lin et al., 2002). Denitrification is a temperature dependent process in which nitrate is reduced to nitrogen gas by anaerobic bacteria (Rolston, 1981). Unlike plant assimilation, denitrification is a dissimilatory process that permanently removes nitrogen (Jones et al., 2017c) and is favored in wetlands because of the anoxic sediments and organic carbon energy source provided by aquatic plants for anaerobic bacteria (Ingersoll and Baker, 1998). Wetland NO_3-N removal performance is determined by a number of factors, including hydrologic conditions (discharge and residence time), inflow NO_3-N concentration, and water temperature (Crumpton et al., 2006).

NO_3-N reduction benefits of both natural and constructed wetlands receiving agricultural runoff have been quantified by others (Table 6). These studies cover a wide range of influent conditions and indicate wetland NO_3-N retention can be highly variable. In general, however, it appears annual NO_3-N retention rates of 0.1-0.3 g m^{-2} day^{-1} (300-1100 kg ha^{-1} yr^{-1}) and removal efficiencies of 20-70% are feasible for natural and constructed agricultural wetlands on an annual basis, with retention varying seasonally (Xue et al., 1999; Hunt et al., 1999). Water quality monitoring in these studies typically consisted of weekly or monthly grab sampling which was sometimes combined with more frequent (e.g. hourly) automated storm sampling.

Development of robust and accurate real-time, continuous measurement devices (Pellerin et al., 2013; Rode et al., 2016) has provided opportunities for greater insight into streamflow-nutrient dynamics. Pellerin et al. (2014) compared high-frequency and regression-based NO_3-N load estimates on the lower Mississippi River over a 2-yr period and showed that regression-based techniques tended to underestimate NO_3-N loads during the spring months critical to Gulf Hypoxia formation and overestimate loads during the rest of the year. Bieroza et al. (2014) used high-frequency nitrogen and phosphorous monitoring to reveal greater (18-30%) loading patterns than those estimated from low-frequency grab sampling in two agricultural streams of England over a 17-month period. Previously undetected nutrient loading patterns at seasonal, individual event, and diurnal time scales were illustrated by hourly monitoring in northwestern Tasmania (Bende-Michl et al., 2013). High-frequency, continuous sampling in the Thames River Watershed (U.K.) revealed both increasing and decreasing NO_3-N concentrations during storm events and that the delivery mode was dependent on antecedent moisture and nutrient supply (Wade et al., 2012). Diurnal and seasonal NO_3-N patterns in the Mississippi River have been assessed using continuous NO_3-N sensors (Bark, 2010). High-frequency NO_3-N monitoring was used to decipher retention and dilution processes in a low order agricultural stream in eastern Iowa (Jones et al., 2017c) and to quantify transport and supply limitations in a tile-drained catchment of central Iowa (Jones et al., 2017a). Reynolds et al. (2016) subsampled 15-min NO_3-N measurements at 17 sites across Iowa to quantify uncertainties connected to conventional, labor-intensive sampling schemes and concluded that manual and automated sampling often do not capture the spatial and temporal variability of NO_3-N concentrations the way continuous monitoring can. These studies demonstrate the advantages of high-frequency, continuous monitoring for quantifying seasonal and storm event stream nutrient flux and improved nutrient load estimation.

In this study, we used high-frequency (15-min) in situ NO_3-N sensors coupled with a hydrologic model to evaluate NO_3-N concentration, load, and retention patterns in a large Iowa constructed wetland from May-Nov over a 3-yr period (2014-16). Automated water quality sensors upstream and downstream of the wetland measured NO_3-N concentration and other water quality variables, and a hydrologic model was developed to estimate discharge for computing loads. The objectives of this study were to 1) quantify the average daily NO_3-N retention and removal efficiency of the wetland on a seasonal (May-Nov) and monthly basis; 2) quantify benefits of the high-frequency monitoring data; and 3) estimate the extent of wetland implementation needed in Iowa to achieve a 45% reduction in NO_3-N loading. This study is unique in its use of continuous monitoring and modeling to estimate wetland NO_3-N retention and reveals the significant amount of information that is lost through lower frequency sampling.

Materials and Methods

Study Area

The constructed wetland is located on Slough Creek, a low-order agricultural stream in north central Iowa (Figure 8). The wetland drains 15.8 km^2 and is located in the Iowan Surface landform region, an erosional surface with extensive tile drainage formed during the Wisconsin glaciation (20,000 years B.P.) (Prior, 1991). Surficial (top 1-2 m) soils in the watershed are primarily silty clay loam (55%) and loam (31%) (NRCS and IGS, 2006), and the deeper subsurface contains mostly clay, limestone, and shale (Prior, 1991; IGS, 2017). The watershed has low relief (average land slope of 2.4%), and land cover is dominated by corn (50%) and soybeans (32%) (USDA, 2017), which are grown by natural rainfall (no irrigation). While the extent of subsurface drainage is unknown, 60% of the watershed is estimated to require tile drainage for optimal crop productivity based on soils and slope information (IGS, 2008). Long-term average (1981-2016) annual precipitation is about 900 mm, with approximately 74% falling between May and Nov (PRISM, 2017). Slough Creek flows into Spring Creek 4.3 km downstream of the wetland. Spring Creek is a tributary of the Cedar River, a source of municipal drinking water supply for Iowa's second largest city, Cedar Rapids (pop. 130,000 (U.S. Census Bureau, 2017)).

Design and construction of the Slough Creek wetland were funded through the Iowa Conservation Reserve Enhancement Program (CREP), a federal- and state-sponsored cost-share program that provides financial incentives to landowners to restore and construct wetlands to reduce NO_3-N loads in tile-drained, agricultural watersheds (Crumpton et al., 2006). It ranks second in drainage area (15.8 km^2) and third in pool area (9.9 ha) among the 83 CREP-sponsored wetlands in Iowa (IDALS, 2016). Per the engineering design plans, the normal pool storage is 41,700 m^3, average pool depth (normal pool storage divided by normal pool area) is 0.42 m, and maximum pool depth is 0.9 m. Outflow is governed by a 61 m long broad-crested weir and a 1.2 m diameter water control structure (YCA, 2012). Outflow from the water control structure can be actively managed by setting stop logs in the structure to the desired water level height but was kept at a constant level in this study corresponding to the normal pool (weir) elevation. These large outflow structures, particularly the broad-crested weir, do little to throttle down incoming flows, resulting in a nearly constant pool area and storage throughout the year. Engineering design was completed in Jun 2012 and design/construction costs were $251,000 (IDALS, 2016).

Review of historical aerial imagery indicated vegetation distribution and density in the wetland has increased considerably since construction (Supplementary Material (SM), Figure SM1). Aerial imagery from shortly after construction (Aug 2013) captured a low pool condition and showed that vegetation density and distribution were low. Vegetation was sparse throughout but most concentrated along the center of the wetland near the former stream channel (Figure 8) while recently excavated areas had little or no vegetation. The estimated vegetation extent was 60-70% of the normal pool area. Imagery from Sep 2014 captured a high pool condition making vegetation patterns more difficult to identify, but vegetation extent appeared to have increased while vegetation density was still low in multiple areas. By Sep 2015, vegetation had been established relatively uniformly in nearly all (100%) of the wetland, though greener and perhaps more dense vegetation was observed at the downstream end. Presumably, vegetation patterns in 2016 were similar to 2015, but imagery from 2016 captured a high pool condition and vegetation distribution was difficult to identify.

Continuous water quality sensors upstream (inlet) and downstream (outlet) of the wetland on Slough Creek track influent and effluent NO_3-N concentration, water temperature, dissolved oxygen (DO), pH, and specific conductance every 15-min (Figure 8). The sensors are 1.8 km apart and the drainage area of the inlet sensor is 12.3 km^2. The inlet sensor is co-located with a continuous water level gauge. Although Slough Creek is the primary inflow to the wetland, the wetland also intercepts five subsurface tiles that drain agricultural land between sensors (YCA, 2012).

Water Monitoring

Sensor measurements were collected 8 May to 20 Nov in 2014, 1 May to 20 Nov in 2015, and 6 May to 17 Nov in 2016. Sensors were retrieved during winter to prevent ice damage.

Instrument malfunction, maintenance, and weather and power issues can result in missing data from time to time, and we only considered calendar days when average daily NO_3-N concentrations were available at both sensors (Jones et al., 2017c). The number of such days were 141 (2014; 72% of deployment period), 204 (2015; 100% of deployment period), and 182 (2016; 93% of deployment period). Data gaps ranged from 5-15 days in 2014 and 2-7 days in 2016.

NO_x-N was quantified using the Hach Nitratax SC plus, 5 mm path length in the range of 0.1-25 mg l^{-1}. The Nitratax did not report separate concentrations of nitrate-nitrogen and nitrite-nitrogen. Because NO_3-N is the primary form of NO_x-N in agricultural streams (Crumpton and Goldsborough, 1998; Kovacic et al., 2000; Lin et al., 2002; Pellerin et al., 2013), NO_x-N is referred to hereafter as NO_3-N. Manufacturer reported instrument precision was 0.1 mg l^{-1} and accuracy was ±3% (Hach Company, 2011). Each 15-min NO_3-N concentration reported by the sensor was the average of three readings taken at the time of the measurement. The Nitratax worked on the principles of UV light absorption. As the concentration of NO_3-N increased, absorption of UV at 210 nm also increased. A built-in photometer measured the primary beam, while a second beam of UV light provided a reference standard and corrected for interference caused by turbidity and dissolved organic material which can increase light absorbance. This correction enabled the sensor to accurately measure unfiltered river water. The optic windows of the device were automatically cleaned with a wiper mechanism before each measurement. The UV cell was immersed into the stream extending outward several centimeters from a perforated polyvinyl chloride housing (Jones et al., 2017b).

Each sensor was serviced and calibrated before field deployment by the Hach Service Department as per the manufacturer’s specifications. Once deployed, monthly field visits were conducted to perform maintenance and collect grab samples that were analyzed in a certified laboratory for NO_3-N within 24 hours of collection. Grab samples were collected in containers preserved with sulfuric acid and refrigerated until analyzed. Laboratory NO_3-N samples were analyzed using the cadmium reduction method and Lachat flow injection analysis (LAC10-107-074-1J; Diamond 1998). Grab sampling was performed over a range of NO_3-N concentrations (5.4-16.4 mg l^{-1} at the inlet and 0.7-14.1 mg l^{-1} at the outlet), water temperatures (8.3-18.5 ^oC at the inlet and 8.9-27.7 ^oC at the outlet), and flow conditions (water stage measurements in Slough Creek at the inlet sensor varied by more than 0.6 m). Good agreement was observed between inlet ($R^2 = 0.99$, slope = 1.05, n = 22, $p<0.001$) and outlet ($R^2 = 0.99$, slope = 1.12, n = 21, $p<0.001$) in situ and laboratory NO_3-N samples.

Water temperature, DO, pH, and specific conductance were measured with a Hydrolab HL4 multiparameter water quality sensor (OTT Hydromet). Water temperature was measured with an electronic thermometer contained within the Hydrolab and was capable of measuring within the range of -5 to 50 ^oC. Of these four parameters, water temperature and DO were considered in future analyses because of their notable influence on denitrification (Xue et al., 1999; Lin et al., 2002). Furthermore, much of the DO data, especially at the outlet, was deemed unreliable due to instrument difficulties and was excluded from the results, but its influence on NO_3-N removal was discussed qualitatively.

Data collected by the sensors did not contain corrections for fouling, calibration drift, temperature, or other factors that can impact data quality. Data found to be erroneous or questionable during the review process were omitted from the final approved dataset. Examples of rejected data included measurements outside the operational range of the device, data values that occurred when the ratio of reference measurement extinction to the measurement extinction was low (an indicator of high turbidity), and data collected during times when the sensor malfunctioned or site conditions suggested the measurement was invalid based on field observations. Additional details on the quality assurance/quality control (QA/QC) protocols in place for the sensor data can be found in Jones et al. (2017b).

Discharge Estimation

Streamflow measurements to calculate NO_3-N loading were not available at the Slough Creek wetland, so discharge was estimated using the hydrologic model MIKE SHE. MIKE SHE is a physically-based, spatially distributed, surface-subsurface hydrologic model that partitions precipitation into the various components of the water cycle including evapotranspiration, surface runoff, infiltration, and groundwater flow (DHI, 2016). MIKE SHE was coupled to MIKE 11, a one-dimensional (1D) hydraulic model, to model channel flow in the Slough Creek stream network. MIKE SHE-MIKE 11 has been used for a variety of hydrologic and water quality applications in agricultural watersheds and for wetland evaluation (Thompson et al., 2004; Loinaz et al., 2013; Zhou et al., 2013; House et al., 2016). The model formulation is described in greater detail by Graham and Butts (2005).

The MIKE SHE model of the Slough Creek watershed was developed on a 30 m grid and contained 19030 surface nodes. The model was developed with publicly-available meteorologic, topographic, land use, and soils datasets, and model input parameters were derived from literature. The model was supplied with spatially distributed (approximately 4 km pixels) Stage IV hourly radar rainfall estimates and daily reference evapotranspiration calculated at Nashua, Iowa (34 km southeast of the wetland) using a modified Penman-Monteith approach (IEM, 2016). Seven land use classes (agricultural, grass/pasture, forest, wetlands, water, developed, and barren) were defined from the 2016 USDA Cropland Data Layer (USDA, 2017) for surface roughness and evapotranspiration parameterization (Te Chow, 1959; Doorenbos and Pruitt, 1977; Engman, 1986; Breuer et al., 2003). The subsurface model contained three layers that defined the surficial soil textures (0-2 m deep), underlying glacial till (2-10 m deep), and deeper limestone geology (10-20 m deep) (Prior, 1991). Agricultural tile drainage, macropore flow, and snowmelt were all modeled. The MIKE 11 model consisted of four stream branches totaling 10.3 km with cross sections spaced approximately every 100 m through the wetland and every 200 m elsewhere. Cross sections were extracted from a 2 m hydro-enforced DEM (James and Gelder, 2016) that was modified to take into account the wetland design geometry and storage. Finally, two weir hydraulic structures were defined at the wetland outlet that passively regulated wetland outflow (YCA, 2012).

The hydrologic modeling was performed in a two-phase process to minimize the effect of initial conditions on modeled discharge (Ajami et al., 2014). First, a recursive simulation was run using the precipitation and evapotranspiration forcing datasets from a single year (2010) until a pseudo steady-state equilibrium was achieved. Equilibrium was achieved when the watershed-integrated surface and subsurface storage changes between consecutive years were less than 0.01 mm (<0.001% of the annual precipitation). Following, a continuous simulation was run from 2010-16 using the end result from the recursive simulation as the initial condition. Model output was assessed by comparing modeled annual (Jan-Dec) and seasonal (May-Nov) water balance results and flow statistics to regional estimates and a nearby USGS discharge gauge (Little Cedar River near Ionia, USGS 05458000) and timing of modeled 15-min discharge to water level measurements at the inlet sensor.

Data Analysis

Wetland NO_3-N retention was estimated from May-Nov over the 3-yr study period using daily NO_3-N loads. Daily NO_3-N loads were calculated by multiplying average daily discharge by average daily NO_3-N concentration. Average daily inlet and outlet NO_3-N concentrations were calculated from the measured 15-min data when more than six hours of data were available in a given day at both sensors. Average daily discharge was calculated from the modeled 15-min discharge time series and wetland outflow was used to calculate both inlet and outlet NO_3-N loading. NO_3-N load reductions due to wetland retention processes were calculated by subtracting daily outlet loads from daily inlet loads. Daily NO_3-N load reduction estimates were divided by the average daily wetland pool area calculated from the hydrologic model so normalized areal mass retention rate estimates (g NO_3-N m^{-2} day^{-1}) could be compared to other wetland studies. Wetland NO_3-N removal efficiency was calculated by dividing the daily load reduction by the daily inlet load. NO_3-N retention was assessed at the monthly and seasonal (May-Nov) time scales by averaging the daily data over that time period (e.g. average daily NO_3-N load reduction for May 2014).

Wetland discharge was expressed using two commonly used quantities: hydraulic loading rate (HLR) and hydraulic residence time (HRT). HLR is an areal-based flow metric that is directly proportional to discharge intensity and was calculated by dividing the average daily outlet discharge by the average daily wetland pool area calculated from the hydrologic model. HRT is a time-based flow metric that describes the average length of time a parcel of water remains in the wetland and was estimated by dividing the average daily outlet discharge by the model-derived normal pool volume. HLR and HRT are inversely related.

The approach utilized in this study to estimate NO_3-N loading and wetland retention made several simplifying assumptions. Average daily wetland outlet discharge was used to calculate both inlet and outlet NO_3-N loading to account for the added hydraulic loading from the five unmonitored tiles that discharged directly into the wetland downstream of the inlet sensor. The main assumptions of this approach were:

1. The five tiles that discharged into the wetland had the same average daily NO_3-N concentration as at the Slough Creek inlet sensor. Although the tile concentrations were probably higher than the Slough Creek stream concentrations (Schilling et al., 2012), they were likely similar given both the tiles and the stream drain essentially the same type of land (tile-drained, row crop areas). Additionally, the flow contribution to the wetland from these tiles compared to the Slough Creek stream was believed to be small. The drainage area of the inlet sensor represented 78% of the entire wetland drainage area and the modeled inlet total discharge volume accounted for a similar percentage (76-77%) of the modeled outlet total discharge volume from May-Nov in any given year during the study period.
2. Flow attenuation through the wetland was minimal. Assuming inlet and outlet discharge were equal was reasonable given CREP wetlands are not designed for flood control as a primary benefit as evidenced by the large outflow weir at this particular site.
3. Seepage and evaporation losses from the wetland were minor. Wetland seepage losses were expected to be small given the low permeability of the underlying glacial drift and loamy till found in the Iowan Surface (Prior, 1991; Schilling and Wolter, 2005). Wetland evaporation was estimated from the hydrologic model and represented less than 5% of the modeled outlet total discharge volume from May-Nov in any given

year during the study period and, therefore, was assumed to have a minimal impact on the daily loading calculations.

4. HRTs in the wetland were on the order of one day or less since no time lag was applied to the outlet NO_3-N concentrations to account for travel time effects through the wetland. While estimated HRTs in the wetland did often exceed one day during the study period, the focus of this paper was on characterizing NO_3-N retention not in any one storm event but at the monthly and seasonal (May-Nov) time scales. Concentration buffering and mixing within specific events that typically lasted several hours to several days was expected to have little effect on the NO_3-N loading and retention estimates at these longer time scales.

To compare NO_3-N dynamics captured across a range of monitoring frequencies, data from May-Nov of 2016 was used as a case study. The high-frequency (15-min) NO_3-N concentrations were resampled at coarser time scales to mimic common grab sampling schemes. Daily, weekly (1st, 8th, 15th, and 22nd of each month), biweekly (1st and 5th of each month), and monthly (15th of each month) grab sampling schemes were imitated by selecting the 15-min NO_3-N concentration reported at 9 am on the appropriate sampling date (Bieroza et al., 2014). Linear interpolation was used to fill in missing inlet and outlet NO_3-N concentrations to create continuous 15-min datasets. Inlet NO_3-N concentrations were missing a total of 1.5 hours (0.03% of the sensor deployment period), while outlet NO_3-N concentrations were missing a total of 17.4 days (8.9% of the sensor deployment period). The majority of missing outlet NO_3-N concentrations occurred during three separate time periods spanning 2.9-8.1 consecutive days in Jun-Aug. High-frequency NO_3-N loads were calculated using 15-min NO_3-N concentration and the corresponding inlet or outlet 15-min modeled discharge. Grab sample loads were calculated using two methods. The first method (method #1) calculated the loads using the grab sample NO_3-N concentration and the average daily, weekly, biweekly, or monthly discharge computed from the 15-min data. The second method (method #2) calculated the loads similar to the high-frequency loads by using 15-min discharge and 15-min NO_3-N concentration obtained from linear interpolation of the grab sample NO_3-N concentrations. NO_3-N dynamics were compared over the entire monitoring period (6 May to 17 Nov) and during a period of frequent and intense precipitation (18 Aug to 17 Sep).

Results

Precipitation

Precipitation was important to characterize because of its large influence on discharge, NO_3-N loading, and NO_3-N retention performance (Mitsch and Gosselink, 2015). May-Nov accounted for 68% (2014) to 80% (2016) of the annual precipitation, similar to the long term average (74%). May-Nov precipitation totals in 2014 and 2015 were within ±1% of the long term average (666 mm; 1981-2016), while the May-Nov total in 2016 was 46% above average. Despite similar May-Nov totals, monthly precipitation was substantially more variable in 2014 (from 80% below average in Jul to 91% above average in Jun) than 2015. Finally, 2016 was a historically wet year. September was exceptionally wet (143% above average) and precipitation from Jun-Oct was 71% above average.

Discharge

The continuous hydrologic simulation of the Slough Creek wetland watershed produced water balance results representative of the region, providing some confidence that the discharge estimates were reasonable given the lack of field monitoring data. Water balance results were calculated at the wetland outlet. On an annual (Jan-Dec) basis, 23% (2015) to 38% (2016) of the precipitation was converted to streamflow (called the runoff coefficient) by the model, comparable to long term averages for streams in the Iowan Surface (23%-34%; Schilling and Libra, 2003; Schilling and Wolter, 2005). Evapotranspiration accounted for 54% (2016) to 72% (2015) of the annual precipitation, also similar to regional estimates (60-70%; Sanford and Selnick, 2013). However, modeled baseflow (both groundwater and tile flow) accounted for a greater percent of annual streamflow (81-92%; called the baseflow index) than regional estimates (60-70%; Schilling and Libra, 2003; Schilling and Wolter, 2005). Differences were attributed to the model construct (all agricultural land was assumed to be tiled in the model, which inherently increased the baseflow index) as well as differences in scale between the Slough Creek catchment (15.8 km^2) and the USGS gauges (800-16860 km^2) used to derive the regional estimates.

The hydrologic model performance on a seasonal (May-Nov) basis also compared favorably with a nearby USGS discharge gauge (Table 7). The Little Cedar River near Ionia (USGS 05458000) is 27 km southeast of the Slough Creek wetland and has a drainage area of 793 km^2. Like the annual comparisons, May-Nov runoff coefficients were similar between the Slough Creek model and the Little Cedar, while the baseflow index for Slough Creek was about 20% greater than that for the Little Cedar. Normalized by watershed area, mean daily flows were comparable (1-2 mm day^{-1}) and peak flows were within a factor of two.

Monthly runoff coefficients from May-Nov during the 3-yr period (n = 21 months) for the Slough Creek model were also similar, though slightly lower, to the Little Cedar River (R^2 = 0.83, slope = 0.76, p<0.001; SM, Figure SM2), suggesting the model's partitioning of precipitation into streamflow was reasonable at the monthly time scale as well. Finally, a qualitative comparison of the modeled 15-min discharge and measured water levels at the watershed inlet suggested the aggregated watershed response and timing was reasonable on an event basis as well (SM, Figures SM3-SM5).

Water Quality

Summary results reflect daily averages when NO_3-N concentrations were obtained at both sensors. Daily precipitation and water temperature (measured), outlet discharge and HRT estimates (modeled), NO_3-N concentration (measured), and NO_3-N loading estimates at the Slough Creek wetland during the study period are shown in Figure 9 (plots showing variability in daily data are included in SM, Figures SM6-SM8).

Water temperature was consistently lower at the inlet than the outlet from May-Sep and comparable in Oct and Nov of each year (Figure 9). The May-Nov 3-yr average water temperature was 13.6 ^{o}C at the inlet and 18.7 ^{o}C at the outlet. Daily extremes ranged from near freezing to 20 ^{o}C at the inlet and 28 ^{o}C at the outlet.

Modeled outlet daily discharges were similar in 2014-15 and elevated from Jun-Nov in 2016 (Figure 9). Daily discharge spanned nearly three orders of magnitude in response to variable precipitation intensity and duration. Consequently, HRTs were also highly variable with daily estimates ranging from 0.2-70.8 days. HRT was considerably lower in 2016 than 2014-15 and was generally lowest in Jun and highest in Jul/Aug in each year. May-Nov average HRT varied from 3.6 days (2016) to 11.9 days (2014). The May-Nov 3-yr average and median HRTs were 8.0 and 5.1 days, respectively.

Daily inlet NO_3-N concentrations were consistently higher than outlet concentrations (Figure 9). Inlet concentrations ranged from 4.8-17.4 mg l^{-1}, while outlet concentrations ranged from 0.4-16.7 mg l^{-1}. Outlet concentrations were greater than inlet concentrations on 20 days of 527 total sensor days (4%) during the 3-yr study. Outlet concentrations ranged from 19% higher to 93% lower than inlet concentrations. Inlet concentrations exceeded the 10 mg l^{-1} regulatory drinking water standard on 63% of sensor days, while outlet concentrations exceeded 10 mg l^{-1} on only 16% of sensor days. In 2015 when precipitation and discharge were low to moderate and HRT was high, inlet concentrations exceeded 10 mg l^{-1} on 94 days (46% of sensor days) while outlet concentrations exceeded 10 mg l^{-1} on only one day. Conversely, in 2016 when precipitation and discharge were considerably higher and HRT was lower, inlet concentrations exceeded 10 mg l^{-1} on 163 days (90% of sensor days) and outlet concentrations exceeded 10 mg l^{-1} on 69 days (38% of sensor days).

Following similar patterns to discharge, daily NO_3-N loading spanned nearly three orders of magnitude (Figure 9). Daily inlet loads ranged from 4-3693 kg day^{-1}, while daily outlet loads ranged from 1-3518 kg day^{-1}. Daily load reductions ranged from -479 to 656 kg day^{-1}. Above average precipitation resulted in sustained higher discharges and lower HRTs in 2016 as compared to 2014-15, and wetland retention and removal efficiency systematically diminished from Jun to Nov.

At the monthly time step, hydrologic conditions and NO_3-N patterns were variable but certain trends were evident. Monthly values ranged from 0.02-0.36 m day^{-1} for HLR and 1.7-28.1 days for HRT. Inlet NO_3-N concentrations ranged from 6.6-16.1 mg l^{-1} and inlet loads ranged from 19-665 kg day^{-1}. NO_3-N load reductions ranged from near zero to 169 kg day^{-1} (1.34 g m^{-2} day^{-1}), and removal efficiencies ranged from near zero to 81%. Mass retention was typically higher in months of greater precipitation, hydraulic loading, and inlet NO_3-N concentration and loading. Removal efficiency was typically higher in months of higher HRT and water temperature and lower NO_3-N loading. Mass retention was highest in Jul 2016 when precipitation, hydraulic loading, and inlet NO_3-N concentration and loading were within the top

5 of all months during the study period and HRT was the lowest of all months. Removal efficiency was highest in Aug 2014 and Jul 2015. In Aug 2014, HRT was the highest of all months, inlet and outlet water temperature were 4th highest, and inlet NO_3-N concentration and loading were second lowest. In Jul 2015, outlet water temperature was the 2nd highest of all months.

Seasonal (May-Nov) average NO_3-N fluxes over the 3-yr study period are summarized in Table 8. Average daily NO_3-N concentrations were available at both sensors a total of 527 days during the 3-yr study period, representing 88% of the time the sensors were deployed. Seasonal NO_3-N flux and retention were clearly influenced by variable hydrologic conditions. The greatest seasonal average NO_3-N concentrations and loads and lowest removal efficiency (23%) occurred in 2016 when HRT was the lowest (3.6 days). Wetland NO_3-N load reductions (67 kg day^{-1}) and removal efficiency (65%) were greatest in 2015 when hydrologic conditions were moderate and less variable and HLR was the lowest (0.07 m day^{-1}). The May-Nov 3-yr average inlet NO_3-N concentration exceeded the 10 mg l^{-1} drinking water standard and the average concentration reduction was 49%. The May-Nov 3-yr average load reduction was 61 kg day^{-1} (0.48 g m^{-2} day^{-1}).

Comparison between Low and High-Frequency Monitoring

For May-Nov 2016, the central behavior in NO_3-N concentration was preserved among the different sampling frequencies, but the high-frequency monitoring captured a much greater range (minimum and maximum values) in NO_3-N concentrations (Figure 10). Mean and median concentrations of the different sampling frequencies were similar. Grab sample mean and median inlet NO_3-N concentrations were within 3% of the high-frequency estimates, while grab sample mean and median outlet NO_3-N concentrations were within 7% of the high-frequency estimates.

The greatest deviations were observed at the monthly time scale. The majority of the grab sample frequencies also captured the skew in the inlet (negative) and outlet (positive) NO_3-N concentration distributions, though the skew weakened as the monitoring frequency decreased. Skew is significant because it reflects the temporal nature of NO_3-N upstream and downstream of the wetland. Upstream of the wetland, NO_3-N concentrations were high the majority of the time but did occasionally lessen due to surface runoff dilution during storm events or during extended dry periods when little soil NO_3-N had been mobilized. Downstream of the wetland, NO_3-N concentrations were low the majority of the time due to wetland retention processes but did occasionally elevate during intense storm events. High-frequency monitoring captured the greatest range of NO_3-N concentrations (2.1-17.6 mg l^{-1} at the inlet and 2.9-17.0 mg l^{-1} at the outlet) which systematically decreased as the sampling frequency decreased.

All grab sample NO_3-N load estimates were greater than the high-frequency load estimates (Table 9). For the period of 6 May to 17 Nov, inlet grab sample loads were 5-15% greater than the high-frequency loads, and outlet grab sample loads were 0-14% (4-14% greater when daily grab sample loads were excluded) greater than the high frequency loads.

An example of the NO_3-N concentration dynamics captured with high-frequency monitoring during a 31-day period of frequent precipitation (18 Aug to 17 Sep) in 2016 is shown in Figure 11. NO_3-N concentrations rapidly decreased during storm events and gradually increased over several days after events. This behavior was most dramatic at the wetland inlet. During baseflow conditions, the high-frequency monitoring revealed wetland retention processes were active. For example, during the dry period from 29 Aug to 6 Sep, inlet NO_3-N concentration varied minimally (11.5-12.1 mg l^{-1}) while outlet NO_3-N concentration steadily decreased over time from 10.7 mg l^{-1} to 6.5 mg l^{-1}. Consequently, the instantaneous NO_3-N concentration reduction increased from approximately 10% to over 40% as the estimated HRT in the wetland gradually increased from 1.8 days to 4.5 days. Diurnal variations in inlet and outlet NO_3-N concentrations were also noticeably different. During the same low flow period (29 Aug to 6 Sep), inlet NO_3-N concentrations varied by up to 2% (0.2 mg l^{-1}) over a single day, while outlet NO_3-N concentrations varied by as much as 28% (2.0 mg l^{-1}) over a single day. Despite the high diurnal variability in outlet NO_3-N concentrations, no clear patterns related to time of day were evident. Finally, the high-frequency load estimates suggested a limited number of days contributed a disproportionate amount of the load during this 31-day period and that load estimation on an event-basis using grab samples can be quite inaccurate. Ten of the 31 days (32%) accounted for over 50% of the total inlet and outlet high-frequency loads, respectively. Daily grab sample loads were similar to the high-frequency estimates (4-6% greater), while all coarser grab sample loads were noticeably higher (8-32%) (Table 9).

Discussion

Comparison of Estimated NO_3-N Loading and Wetland Removal to Other Studies

Direct comparison of NO_3-N loading and retention estimated for the Slough Creek wetland with other studies was challenging. Aside from variable influent conditions, watershed characteristics, and wetland design considerations, the studies often encompassed different monitoring periods, monitoring frequencies, and seasonal conditions (e.g. winter conditions in Iowa compared to North Carolina). Most of the reference wetland studies reported annual averages that included cold season (Dec-Apr) monitoring not conducted in this study. In the U.S. Midwest, winter NO_3-N loading is typically low (Ikenberry et al., 2014), and colder temperatures, less sunlight, and often times lower NO_3-N concentrations inhibit biological retention processes. Therefore, we expected our seasonal NO_3-N load and retention estimates to be lower than the annual averages reported in other studies. Additionally, while our study lacked cold season monitoring, it is important to note that hypoxic zone extent in the Gulf of Mexico is of most concern in summer when NO_3-N loading in the MARB is greatest (Pellerin et al., 2014).

Slough Creek inlet NO_3-N concentrations and yields were greater than, though comparable to, other studies conducted in the Iowan Surface. The Slough Creek May-Nov 3-yr average inlet NO_3-N concentration, flow-weighted average (FWA) NO_3-N concentration (total NO_3-N load divided by total discharge volume on sensor days), and yield were 10.6 mg l^{-1}, 11.2 mg l^{-1}, and 11.3 kg km^{-2} day^{-1}, respectively. One Iowan Surface study of 0.4-ha tile-drained corn-soybean plots reported a 6-yr FWA NO_3-N concentration of 10.3 mg l^{-1} (Bakhsh et al., 2002), less than 10% different from the Slough Creek estimate. The 6-yr average annual NO_3-N yield in that study was 4.4 kg km^{-2} day^{-1}, considerably lower than the Slough Creek estimate, but annual yields for individual plots were as large as 12.6 kg km^{-2} day^{-1} (Bakhsh et al., 2002). Schilling and Wolter (2005) estimated the long term average NO_3-N yield for the Iowan Surface to be 6.1 kg km^{-2} day^{-1} based on multiple linear regression models developed for streamflow and NO_3-N concentration from datasets prior to 2000, also lower than the Slough Creek average. Jones et al. (2017b) used an extensive network of in situ NO_3-N sensors to evaluate NO_3-N patterns in Iowa in 2016 and calculated an average concentration of 10.7 mg l^{-1} and yield of 11.2 kg km^{-2} day^{-1} for the Iowan Surface, very similar to the Slough Creek wetland inlet values.

Estimated NO_3-N mass retention in the Slough Creek wetland was also greater than most other wetland studies (Table 6). The May-Nov 3-yr average NO_3-N mass retention estimated in this study was 0.48 g m^{-2} day^{-1} while the highest average annual retention reported for other wetland studies in Table 6 was 0.33 g m^{-2} day^{-1}, which was also for several Iowa wetlands (Crumpton et al., 2006). However, the average inlet NO_3-N loading (1.41 g m^{-2} day^{-1}) and NO_3-

N concentration (10.6 mg l^{-1}) estimated in this study were also higher than most of the other wetland studies as well. When seasonal or monthly NO_3-N retention rates were reported in other studies, retention was noticeably higher during warmer periods (Hunt et al., 1999; Xue et al., 1999). Retention rates as high as 1.26 g m^{-2} day^{-1} were reported for several wetland microcosms with similar NO_3-N loading rates to the Slough Creek wetland and retention rates as high as 2.8-5.0 g m^{-2} day^{-1} have been achieved in other wetland studies (Lin et al., 2002). Additionally, we acknowledge that our retention estimates may be overestimated because wetland seepage was assumed to be minimal (NO_3-N mass retention = inlet load – outlet in-stream load – seepage export). While expected to be small, some NO_3-N export due to seepage likely did occur which would have lowered our estimated removal rates. Removal efficiencies estimated for the Slough Creek wetland (seasonal averages of 23-65%) were consistent with other wetland studies (Table 6), and the May-Nov 3-yr average removal efficiency (49%) was close to the average annual removal efficiency (52%) reported for wetlands in the Iowa Nutrient Reduction Strategy (IDALS et al., 2014).

Discharge was one of the greatest sources of uncertainty in our wetland retention estimates. While all model parameters were within reasonable ranges and multiple measures were taken to ensure modeled discharge was reasonable in timing, magnitude, and volume, no discharge measurements were available for a more direct model validation. To quantify the impact of the model uncertainty on the estimated retention rates, the hydrologic model was rerun with adjusted precipitation, which has been shown to have a greater impact on modeled discharge than the uncertainty in model parameter estimation (Quintero et al., 2012), and the NO_3-N loading and retention were recalculated. The precipitation was scaled by a conservative factor of ±15% after review of May-Nov rainfall totals indicated the radar rainfall estimates used in the modeling were within 10% of the nearest NOAA rain gauge (GHCND:USC00136305) 10 km away in Osage, Iowa. As expected, a 15% change in precipitation had a profound effect on the modeled discharge, and consequently, NO_3-N loading and retention. A 15% decrease in precipitation reduced the seasonal modeled discharge volumes, NO_3-N loading, and NO_3-N retention estimates by approximately 40-45%. A 15% increase in precipitation increased the seasonal modeled discharge volumes, NO_3-N loading, and NO_3-N retention estimates by approximately 45-55%. While we believe these uncertainty estimates are quite conservative, they highlight the importance of field measurements (e.g. discharge in this study) to more robustly validate hydrologic models, the driving influence of rainfall on discharge and NO_3-N loading, and the need for careful consideration and thorough evaluation given to model construct and parameter inputs by hydrologic modelers.

Mean Monthly Variation in Wetland NO_3-N Retention

On a mean monthly basis, wetland NO_3-N mass retention and removal efficiency peaked at different times (Table 10). Monthly mass retention was highest in Jun when precipitation, hydraulic loading, and NO_3-N concentration and loading were also highest; however, the removal efficiency in Jun was 18% lower than in Aug, the month of highest removal efficiency. In Aug, water temperature and HRT were among the highest of all months and inlet NO_3-N concentration was lowest; however, Aug mass retention was 53% lower than Jun mass retention. Jul provided the optimal combination of both retention metrics, as mass retention and removal efficiency were only slightly less than the Jun and Aug estimates, respectively. In Jul, NO_3-N loading was less than Jun primarily because of lower hydraulic loading. Increased HRT, combined with similar water temperatures to Aug, presumably provided more time for enhanced biological processing. These findings were consistent with variations in mass retention and removal efficiency due to seasonal changes in NO_3-N loading, discharge, and water temperature reported in other wetland studies (Hunt et al., 1999; Kovacic et al., 2000; Garcia-Garcia et al., 2009).

Removal Efficiency and Hydraulic Loading Rate

Crumpton et al. (2006) suggested that annual removal efficiency could be predicted solely by HLR based on 34 years of data from 12 wetlands in Ohio, Illinois, and Iowa. To our knowledge, the HLR-removal efficiency relationship has not been thoroughly assessed at finer time scales, so the Slough Creek monthly results were compared to the annual results from Crumpton et al. (2006) to evaluate if monthly removal efficiency could be predicted exclusively by HLR.

Figure 12 illustrates that both Slough Creek wetland seasonal and monthly removal efficiency decayed exponentially with increasing HLR, but there was much greater variability in the monthly values. There was little connection between HLR and removal efficiency by month; however, the seasonal removal efficiencies more clearly tracked with HLR values. Compared to Crumpton et al. (2006), larger HLRs were estimated for the Slough Creek wetland and its regression line systematically predicted a greater removal efficiency than the wetlands of that study. However, the Slough Creek regression line (power law) did not include the influence of Nov 2016 (negative removal efficiency) which would have shifted the curve down a considerable amount. This exercise suggested that while a good predictor of annual wetland performance, HLR was not a very good predictor of monthly wetland performance. The large month-to-month variability in the HLR-removal efficiency relation was to be expected considering the substantial effects that both temperature and incoming NO_3-N concentration have on denitrification rates (Xue et al., 1999). Clearly these can vary independently of HLR and were likely driving the monthly scatter illustrated in Figure 12. Refining prediction and assessment of wetland performance at shorter (i.e. seasonal and monthly) time scales will by extension require water quantity and quality monitoring at finer time scales as well.

Value of High-Frequency, Continuous Monitoring

High-frequency (15-min) monitoring captured a greater range of NO_3-N concentrations than the grab sampling schemes, diurnal variations in concentration, concentration dynamics connected to storm events, and the marked variability in wetland retention performance during low- and high-flow conditions.

Diurnal variations in NO_3-N were substantially greater at the outlet than the inlet. During a 9-day baseflow period in Aug and Sep in 2016, 15-min inlet NO_3-N concentrations varied by less than 2% over a single day and 5% over the entire period while outlet NO_3-N concentrations varied by up to 28% over a single day and 64% over the entire period. The magnitude of these diurnal fluctuations was associated with the degree of biological activity dependent on light conditions, temperature, and DO levels (Nimick et al., 2011). Hence, these data illustrated NO_3-N retention processes (plant assimilation and denitrification) in the stream were minimal relative to the wetland. Although only a 9-day period was evaluated, the lack of a recognizable inlet NO_3-N diurnal signal suggested ammonium (NH_4-N), which was not measured in this study, was a minimal source of NO_3-N in Slough Creek, at least during late summer. If NH_4-N were present in considerable amounts, inlet NO_3-N levels would have likely increased during the day when nitrification rates are higher due to greater water temperature, DO, and pH (Nimick et al., 2011). Low levels of in-stream NH_4-N suggested by the high-frequency data were consistent with our belief that NH_4-N sources (municipal and industrial wastewater, manure from livestock) in the watershed were low since population was low and row crops were the predominant land use. The noticeably higher diurnal variability in outlet NO_3-N concentration compared to inlet NO_3-N concentration with no discernible diel pattern observed at either location warrants further investigation. Measured inlet and outlet DO were lowest in the late summer, so we hypothesize anoxic conditions in the wetland during this time may have enhanced denitrification during both day and night and altered the typical diurnal NO_3-N signal attributed solely to assimilation by plants and primary producers (Nimick et al., 2011; Halliday et al., 2012).

NO_3-N concentration-discharge relationships can help reveal nutrient delivery mechanisms and dominant hydrological pathways during storm events (Oeurng et al., 2010; Bieroza et al., 2014). NO_3-N dilution during storm events (Figure 11) reflected a quick system response indicative of diluted surface runoff (Holz, 2010), while the more gradual increase in NO_3-N concentration after the storm event peak suggested soil NO_3-N was mobilized from upstream agricultural areas and delivered to the stream network through slower subsurface pathways (groundwater leaching and tile drainage) (Oeurng et al., 2010). This explanation of NO_3-N transport pathways in Slough Creek seems probable given the catchment's fine soil texture composition which can result in flashy, surface runoff-driven behavior during storm events and intensively tile-drained nature which is the major hydrologic pathway for subsurface water to streams during most of the year. Wetland removal efficiency declined substantially during storm events, with retention processes more evident during low flow periods. When considering constructed wetland design, management, and performance in small catchments like Slough Creek where the typical hydrologic response time is on the order of hours to a few days, it is particularly important to characterize NO_3-N delivery processes and dynamics at fine time scales.

We also believe that the high-frequency monitoring provided more accurate NO_3-N load estimates. Both the high-frequency (avg: 17.0 kg km^{-2} day^{-1}) and grab sample (avg of daily, weekly, biweekly, and monthly frequencies: 18.5 kg km^{-2} day^{-1}) NO_3-N load estimates for May-Nov 2016 were higher than expected but the high-frequency estimates were in better agreement with the average annual NO_3-N yield estimated for the Iowan Surface in 2016 (11.2 kg km^{-2} day^{-1}; Jones et al., 2017b). Daily grab sample NO_3-N loads were within 6% of the high-frequency NO_3-N loads, while the weekly, biweekly, and monthly grab sample loads were an average of 10% (6 May to 17 Nov) and 27% (18 Aug to 17 Sep) greater than the high-frequency loads. Pellerin et al. (2014) compared lower Mississippi River NO_3-N load estimates calculated from high-frequency NO_3-N sensors with regression-based techniques and also reported greater deviations at finer time scales. In that study, regression-based load estimates were only 3.5% less than the high-frequency loads for the entire 2-yr study period, but monthly deviations of 20-40% were common (Pellerin et al., 2014). Routine low-frequency (weekly to monthly) monitoring captures a narrower range of nutrient concentrations and undersamples during extreme hydrologic conditions when a disproportionate amount of the load is delivered, leading to significant errors in nutrient load estimation (Rozemeijer et al., 2010; Reynolds et al., 2016). High-frequency monitoring has the potential to provide more accurate load estimates at a significantly lower cost on a per sample basis than grab sampling (Pellerin et al., 2014), which has important implications for conservation practice implementation, nutrient reduction strategies, and Gulf Hypoxia. Low-frequency monitoring may still be valuable for quantifying mean and long term NO_3-N concentration trends or when resources for higher frequency monitoring are not available (Bieroza et al., 2014; Reynolds et al., 2016).

Because the sensors are continuous, automated, and remotely connected to a cellular network, data collection is considerably easier than manual grab sampling or automated ISCO sampling which both require frequent field visits to collect or retrieve data. The 15-min data is made publically available online through the Iowa Water Quality Information System (http://iwqis.iowawis.org/) for viewing and troubleshooting in near real-time. Aside from the increased ease of data collection, the high-frequency monitoring paired with the ability to actively control outflow from the Slough Creek wetland through a water level control structure provide the opportunity for more active management of the wetland pool to optimize NO_3-N retention on a daily basis. Under this scenario, the wetland could be drawn down during low flow and loading conditions to allow detention and processing of large NO_3-N loads delivered by rain events, thereby increasing wetland efficiency. Active management of the Slough Creek wetland was not performed during this study, but the wetland was drawn down during the summer of 2017 to encourage greater establishment of emergent vegetation and to begin to take a more involved role in pool management. Monitoring of water quality and NO_3-N retention continued during the draw down and will provide quantitative insight into the wetland retention performance under low pool conditions and help inform future CREP wetland management.

Other Considerations for Wetland NO_3-N Removal

In addition to hydrologic conditions, NO_3-N flux, and water temperature, wetland NO_3-N removal also depends on DO and organic carbon. Both these variables are critical for denitrification, the primary NO_3-N removal mechanism in constructed wetlands (Xue et al., 1999; Lin et al., 2002), as anaerobic bacteria require an electron donor (organic carbon) in order to derive energy from the reduction of NO_3-N (Jones and Kult, 2016). Hence, low DO levels and organic carbon sources (most often aquatic vegetation in wetlands) are prerequisites for denitrification to occur. Wetland NO_3-N removal can be hindered when either requirement is not adequately met (Hunt et al., 1999; Lin et al., 2002). While detailed vegetation surveys were not performed at the Slough Creek wetland, aerial imagery qualitatively suggested low amounts of wetland vegetation, particularly in the first two years of the study, may have resulted in low organic carbon levels and reduced rates of denitrification. Given inlet NO_3-N concentrations were consistently high throughout the study period and moderate hydrologic conditions allowed for increased processing time in 2014-15, it seems feasible that organic carbon may have been a limiting factor on NO_3-N removal during these two years. While greater NO_3-N loading estimated in 2016 would theoretically allow for more denitrification to occur which would by extension require a greater organic carbon demand, we suspect denitrification was limited more so in this particular year by extreme hydrologic conditions that reduced the HRT in the wetland, allowing significantly less time for denitrification to occur.

Trends from the measured DO data indicated that inlet DO (4-16 mg l^{-1}) was typically higher than outlet DO, and we believe the wetland was indeed anoxic (DO < 2 mg l^{-1}) at certain times, particularly in late summer (Aug) when both inlet and outlet DO were lowest. Denitrification rates were probably enhanced during this time, as evidenced by greater NO_3-N removal efficiencies. However, much of the remaining time DO levels greater than 2 mg l^{-1} likely restricted denitrification strictly to the wetland sediment. Improved hydraulic design and a more active role in pool management could help to increase the amount of water-sediment interaction in the wetland, thereby providing greater potential for denitrification and increased removal efficiencies throughout the year.

Achieving the 45% NO_3-N Load Reduction Goal in Iowa Using Restored or Constructed Wetlands

NO_3-N retention estimates for the Slough Creek wetland were extrapolated to the state of Iowa to provide an indication of the extent of wetland restoration and construction needed to reduce the state's NO_3-N load by 45%. Using the average daily baseline NO_3-N load estimated for Iowa (7.63×10^5 kg day^{-1}; IDALS et al., 2014) and the May-Nov 3-yr average NO_3-N mass retention estimated for the Slough Creek wetland (61 kg day^{-1}), 5638 wetlands of similar retention performance to the Slough Creek wetland would be required to reduce the state's baseline NO_3-N load by 45%. Design and construction costs would total an estimated $1.5 billion, or equivalently, about $87 million annually at a 4% interest rate over a 30-yr period (the minimum easement required for CREP wetlands). These wetlands would treat 61% of the state's area (77% of agricultural areas). The INRS estimated that a 22% NO_3-N load reduction could be achieved with wetlands treating 45% of agricultural areas (IDALS et al., 2014); extrapolating this analysis to a 45% load reduction suggests wetlands would need to treat 73% of Iowa's area (92% of agricultural areas), greater than though similar to the Slough Creek-derived estimates. Similar analyses have been performed by others. Mitsch et al. (2005) estimated that creation or restoration of 22000 km^2 of wetlands would be needed to reduce the nitrogen load to the Gulf of Mexico by 40%, and Crumpton et al. (2006) estimated 2100-4500 km^2 of wetlands would be required to reduce NO_3-N loading by 30% in the Upper Mississippi and Ohio River Basins. All these studies clearly indicate the sizeable investment in wetland restoration and construction, not to mention wetland management and monitoring, needed to achieve quantifiable Gulf Hypoxia water quality goals.

Conclusions

In this study, we analyzed NO_3-N concentrations and estimated mass retention and removal efficiency in one of Iowa's largest constructed wetlands using high-frequency continuous monitoring and a physically-based hydrologic model. Wetland retention processes (presumably denitrification and to a lesser extent plant assimilation) reduced NO_3-N concentrations 49% and loads 61 kg day^{-1} (0.48 g m^{-2} day^{-1}) from May-Nov over a 3-yr period. The Slough Creek wetland retention estimates were greater than reported in other wetland studies (Table 6), which was attributed primarily to differences in monitoring schemes (high frequency monitoring vs grab sampling), monitoring period (annual vs seasonal), and uncertainty associated with the modeled discharge in this study. On a monthly basis, mass retention was highest in Jun when NO_3-N loading was greatest and removal efficiency was greatest in Jul and Aug when greater HRTs and water temperatures provided more conducive conditions for enhanced processing. Correlation between wetland NO_3-N mass retention and removal efficiency and hydrologic variables (notably discharge and HRT), water temperature, and NO_3-N supply was consistent with other studies (Kovacic et al., 2000; Mitsch et al., 2005; Garcia-Garcia et al., 2009).

Results from this study and others (Table 6) emphasize that wetland NO_3-N retention and removal efficiency can be highly variable in response to hydrologic conditions, NO_3-N concentration and loading, temperature, and other factors. As a result, nutrient reduction plans that outline feasible ways to achieve Gulf Hypoxia water quality goals to inform policy and potentially influence management decisions utilize nutrient reduction estimates for wetlands and other conservation practices that are highly uncertain. For example, the INRS assumes an average annual NO_3-N reduction of 52% for wetlands in its nutrient reduction scenario analyses, but the reported annual variability is from 11-92% (IDALS et al., 2014). This high uncertainty underscores the need for robust water monitoring data to improve prediction of wetland removal performance taking into account relevant environmental variables known to influence performance. Robust water monitoring is particularly important during the spring and summer months when hypoxia in the Gulf of Mexico is of greatest concern.

Additionally, future studies like these should include field measurements of discharge to better validate hydrologic models and make an effort to perform water monitoring over the entire year, even if at a reduced frequency during certain times due to adverse field conditions or instrument constraints. Doing so should in theory improve accuracy and reduce uncertainty in the hydrologic model performance and nutrient load estimates and provide a more complete picture of annual wetland removal performance and how retention varies seasonally. Annual water monitoring, even if at reduced frequencies during certain times of the year, is important given reducing the *summer* hypoxic zone in the Gulf of Mexico to 5000 km^2 by 2035 is based on reducing *annual* nutrient loading in the MARB by 45%.

Extrapolating the May-Nov 3-yr average NO_3-N retention estimate for the Slough Creek wetland to the state of Iowa, over 5600 wetlands of similar retention performance to the Slough Creek wetland would be required to reduce the state's baseline NO_3-N load by 45%, costing an estimated $1.5 billion in design and construction. While simplified, this analysis and others like it demonstrate the significant investment in wetland restoration and construction needed to achieve quantifiable Gulf Hypoxia water quality goals. Continued water monitoring and modeling is needed to track progress, inform policy, and narrow the uncertainty in conservation practice performance.

Acknowledgments

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Tables and Figures

Table 6. Summary of mean NO_3-N retention characteristics in several natural and constructed wetlands treating predominantly agricultural runoff. Standard deviations listed in parentheses, when available. NO_3-N loading and retention are normalized by wetland pool area. (HLR: hydraulic loading rate).

Wetland Location and Type	Study Years	Monitoring Period	HLR (m day^{-1})	Inlet NO_3-N Concentration (mg l^{-1})	Inlet NO_3-N Loading (g m^{-2} day^{-1})	NO_3-N Mass Retention (g m^{-2} day^{-1})	NO_3-N Mass Removal Efficiency	Reference
Slough Creek wetland, constructed	2014-16	May-Nov	0.13 (0.20)	10.6 (2.7)	1.41 (2.19)	0.48 (0.67)	49% (27%)	This study
Iowa, constructed (3)	2004, 2006	Annual	0.08 (0.09)	13.9 (2.9)	1.01 (1.08)	0.33 (0.22)	57% (23%)	Crumpton et al., 2006
Illinois, constructed	1995-97	Jan-Feb (4 ºC) May-Jun (25 ºC)		8.4 10.5	0.21 0.80	0.05 0.28	25% 40%	[1]Xue et al., 1999
Illinois, constructed (3)	1995-97	Annual	0.03-0.04	7.5-14.5	0.50	0.19	38%	[1]Kovacic et al., 2000
Illinois, constructed (2)	2012-13	Annual (2012, drought) Annual (2013, flood)	0.01 0.03	11.2-15.5	0.06 0.35	0.04 0.19	60% 55%	[1]Groh et al., 2015
Ohio, constructed river diversion	1994-2003	Annual	0.09 (0.02)	1-8	0.30 (0.12)	0.11 (0.06)	35% (2%)	[1]Mitsch et al., 2005
Ohio, constructed river diversion	1994-2013	Annual	0.11 (0.004)		0.27 (0.01)	0.04 (0.007)	15.6%	[1]Mitsch et al., 2014
North Carolina, natural	1993-96	Annual May-Nov	$3.0x10^{-7}$ ($4.5x10^{-7}$) $1.1x10^{-7}$ ($1.6x10^{-7}$)	6.6 (1.2)	0.62 (0.50) 0.48 (0.30)	0.32 (.20) 0.37 (0.17)	51% (28%) 76% (13%)	Hunt et al., 1999
Spain, natural (2)	2007-08	Annual		24.5 (6.1)	0.33 (0.28)	0.22 (0.17)	72% (15%)	Garcia-Garcia et al., 2009

Table 7. Seasonal (May-Nov) water balance comparison between the Slough Creek wetland outlet (modeled) and the Little Cedar River near Ionia (measured; USGS 05458000) during the 3-yr study period. (P: precipitation, ET: evapotranspiration; Q_b: baseflow volume; Q: total streamflow volume; Q_{mean}: mean daily streamflow; Q_{peak}: peak mean daily streamflow).

	Slough Creek Wetland Outlet (modeled)					Little Cedar River near Ionia (measured; USGS 05458000)				
Year	P (mm)	Q/P	Q_b/Q	Q_{mean} (mm day^{-1})	Q_{peak} (mm day^{-1})	P (mm)	Q/P	[1]Q_b/Q	Q_{mean} (mm day^{-1})	Q_{peak} (mm day^{-1})
2014	668	0.26	0.82	0.8	17.6	686	0.29	0.68	0.9	8.8
2015	663	0.19	0.93	0.6	4.5	685	0.27	0.70	0.9	9.4
2016	973	0.35	0.77	1.7	15.9	967	0.44	0.55	2.0	21.8
Avg	768	0.27	0.84	1.0		780	0.33	0.64	1.3	

Table 8. Seasonal (May-Nov) summary of average daily hydrologic conditions, NO_3-N flux, and wetland NO_3-N retention performance. Numbers in parentheses are standard deviations of daily averages except for precipitation, which lists the ratio of the seasonal total to the long-term average seasonal total (1981-2016). (P: total precipitation from May-Nov; HLR: wetland hydraulic loading rate; HRT: wetland hydraulic residence time).

			Hydrology			NO_3-N Concentration (mg l^{-1})			NO_3-N Loading (kg day^{-1})			
Year	Sensor Days	Percent of Sensor Deployment Period	P (mm)	HLR (m day^{-1})	HRT (days)	Inlet	Outlet	Red.	Inlet	Outlet	Red.	Retention (g m^{-2} day^{-1})
2014	141	72%	668 (1.00)	0.09 (0.21)	11.9 (13.9)	10.2 (2.7)	4.7 (3.8)	59% (24%)	133 (365)	86 (331)	47 (71)	0.37 (0.56)
2015	204	100%	663 (1.00)	0.07 (0.07)	9.2 (7.1)	9.4 (2.4)	3.3 (1.8)	65% (15%)	95 (106)	28 (35)	67 (86)	0.53 (0.68)
2016	182	93%	973 (1.46)	0.22 (0.25)	3.6 (4.0)	12.2 (2.4)	9.1 (2.1)	23% (18%)	309 (285)	244 (284)	65 (91)	0.51 (0.72)
Avg	176	88%	768 (1.15)	0.13 (0.20)	8.0 (9.4)	10.6 (2.7)	5.7 (3.7)	49% (27%)	179 (278)	118 (258)	61 (85)	0.48 (0.67)

Table 9. Relative differences between high-frequency (15-min) and grab sample NO_3-N load estimates for two periods in 2016. Percent differences were calculated as the difference between the grab sample and high-frequency loads divided by the high-frequency load.

	6 May – 17 Nov (195 days)				18 Aug – 17 Sep (31 days)			
Artificial Grab Sampling Scheme	Inlet Load Difference (%)		Outlet Load Difference (%)		Inlet Load Difference (%)		Outlet Load Difference (%)	
	Method #1	Method #2	Method #1	Method #2	Method #1	Method #2	Method #1	Method #2
Daily	5	6	0	0	4	6	4	6
Weekly	7	7	4	4	31	28	28	24
Biweekly	15	14	14	12	28	32	29	26
Monthly	12	12	12	13	10	30	8	25

Table 10. Monthly means of hydrologic, water temperature, and NO_3-N characteristics at the Slough Creek wetland during the 3-yr study period (2014-16). Standard deviations listed in parentheses. (P: precipitation, HLR: wetland hydraulic loading rate; HRT: wetland hydraulic residence time).

	Hydrology			Temperature (°C)		NO_3-N Concentration (mg l^{-1})			NO_3-N Loading (kg day^{-1})			
Month	P (mm)	HLR (m day^{-1})	HRT (days)	Inlet	Outlet	Inlet	Outlet	Red.	Inlet	Outlet	Red.	Retention (g m^{-2} day^{-1})
May	84 (9)	0.05 (0.03)	10.3 (5.1)	11.1 (1.6)	18.7 (3.8)	12.5 (1.6)	7.2 (2.8)	44% (17%)	73 (48)	40 (31)	33 (29)	0.26 (0.23)
Jun	188 (51)	0.22 (0.33)	3.2 (1.9)	14.7 (1.5)	21.9 (1.7)	14.1 (1.8)	8.4 (4.2)	43% (25%)	408 (575)	290 (581)	118 (147)	0.93 (1.16)
Jul	120 (63)	0.15 (0.14)	5.7 (6.0)	16.6 (1.3)	23.9 (1.5)	12.1 (2.2)	5.2 (4.1)	60% (26%)	244 (226)	128 (159)	116 (108)	0.92 (0.85)
Aug	145 (43)	0.12 (0.21)	13.8 (17.0)	17.1 (1.0)	22.7 (1.9)	8.6 (2.6)	3.8 (2.9)	61% (22%)	143 (208)	87 (166)	56 (55)	0.44 (0.44)
Sep	133 (61)	0.19 (0.28)	4.9 (3.5)	15.8 (1.6)	19.2 (2.7)	9.1 (1.5)	4.5 (3.4)	52% (32%)	202 (221)	147 (222)	54 (61)	0.43 (0.49)
Oct	56 (18)	0.09 (0.10)	9.4 (9.0)	11.8 (1.8)	11.2 (2.1)	9.2 (1.9)	5.8 (3.0)	39% (24%)	110 (118)	86 (126)	24 (28)	0.19 (0.22)
Nov	46 (28)	0.07 (0.04)	6.2 (2.6)	8.4 (2.9)	8.8 (2.8)	9.6 (1.0)	5.6 (3.0)	42% (28%)	90 (54)	62 (61)	28 (28)	0.22 (0.22)

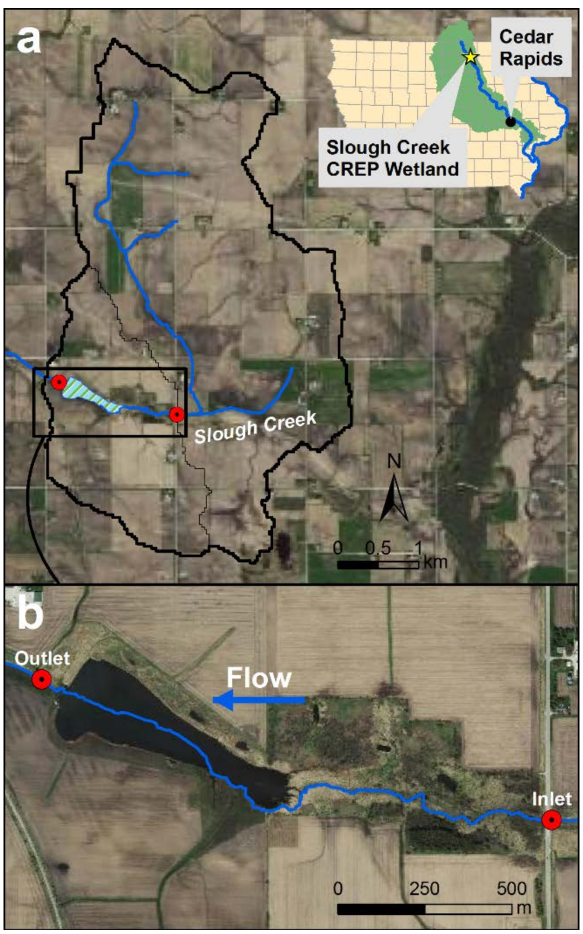

Figure 8. Slough Creek wetland overview. (a) Watershed draining to the wetland (b) Continuous water quality sensors located at the inlet and outlet of the wetland.

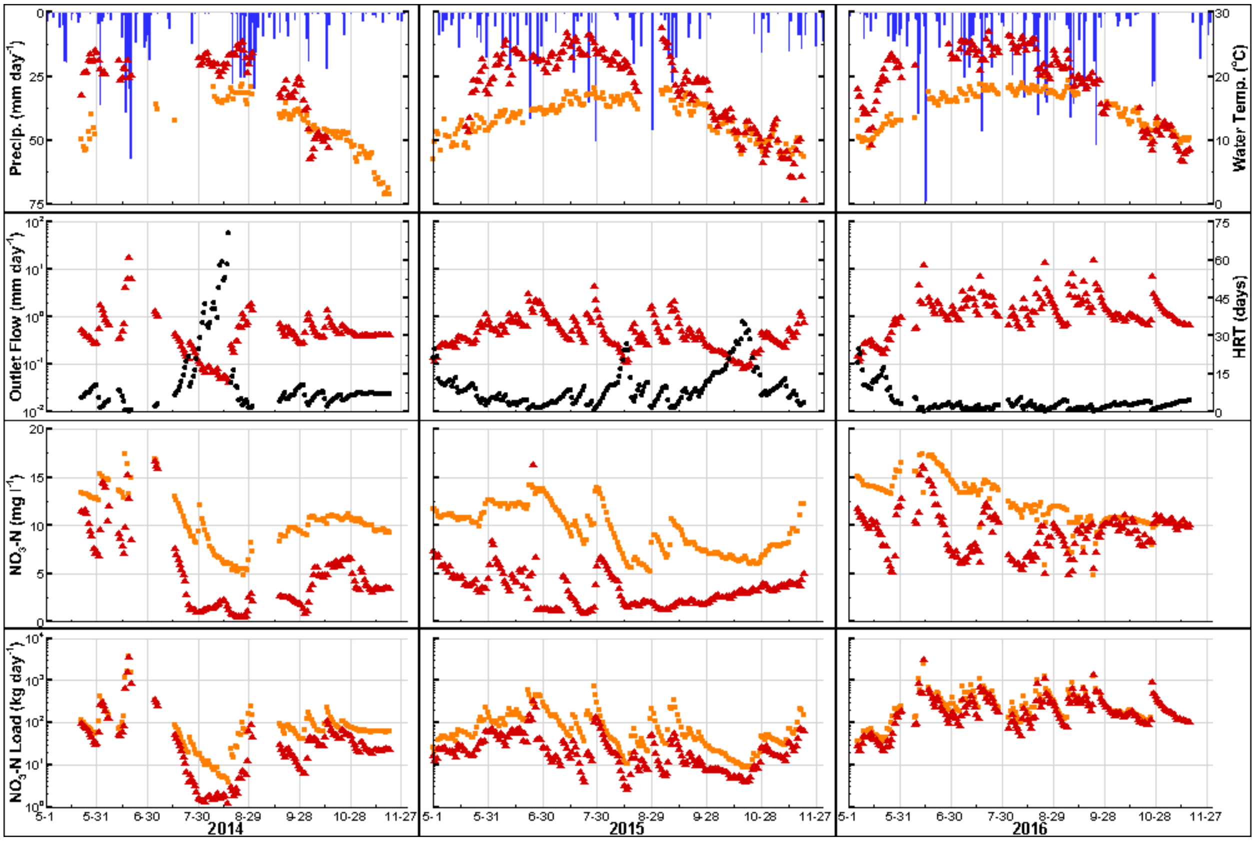

Figure 9. Daily hydrologic and water quality conditions at the Slough Creek wetland for May-Nov during the 3-yr study period (2014-16). Each data point represents a daily average inlet (orange squares) or outlet (red triangles) condition. The top row shows daily precipitation (blue) and water temperature (measured); the second row from the top shows outlet flow normalized by drainage area (modeled) and hydraulic residence time (HRT, modeled; black circles); the third row from the top shows NO_3-N concentration (measured); the bottom row shows NO_3-N loading estimates. Each column defines a particular year.

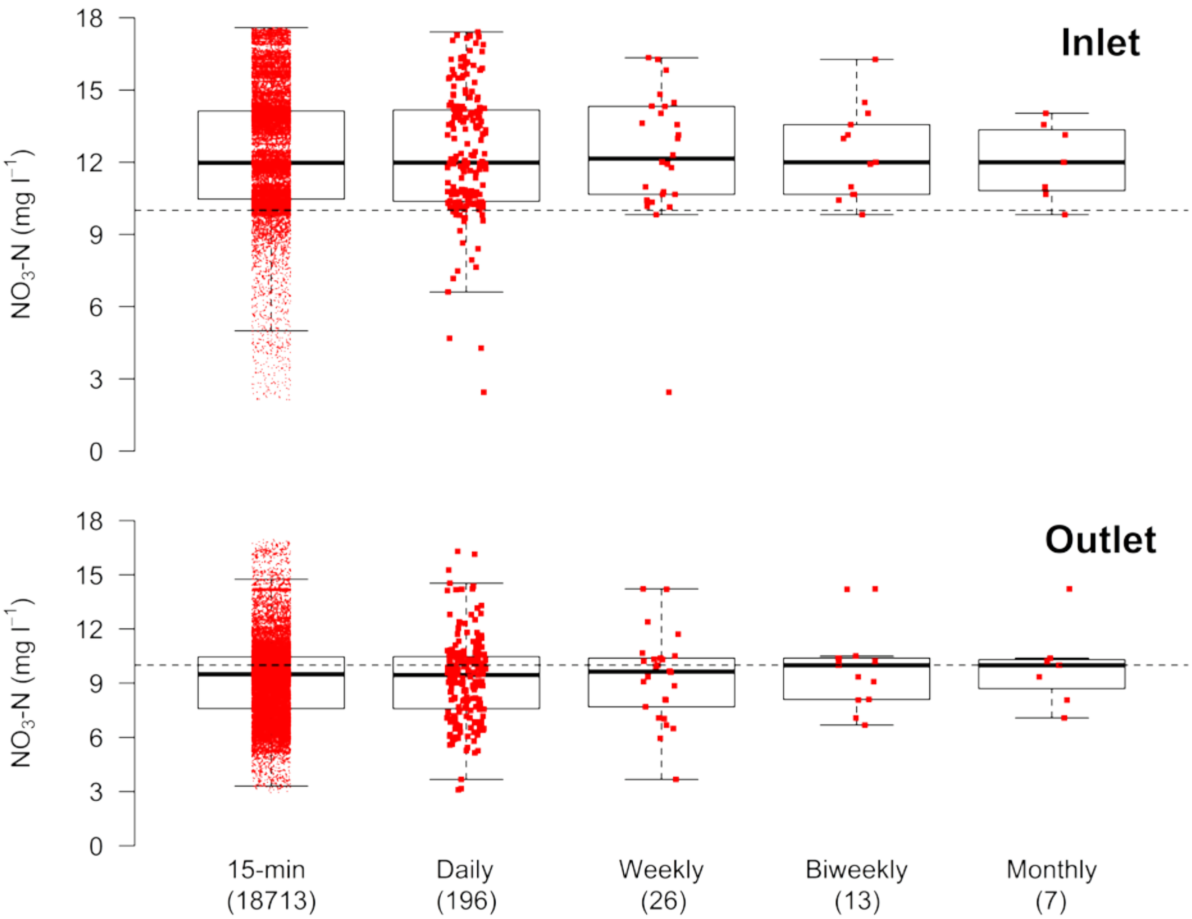

Figure 10. Variability in inlet (top) and outlet (bottom) NO_3-N concentrations captured by different monitoring frequencies for 6 May to 17 Nov in 2016. Sample size (n) of each monitoring frequency shown in parentheses.

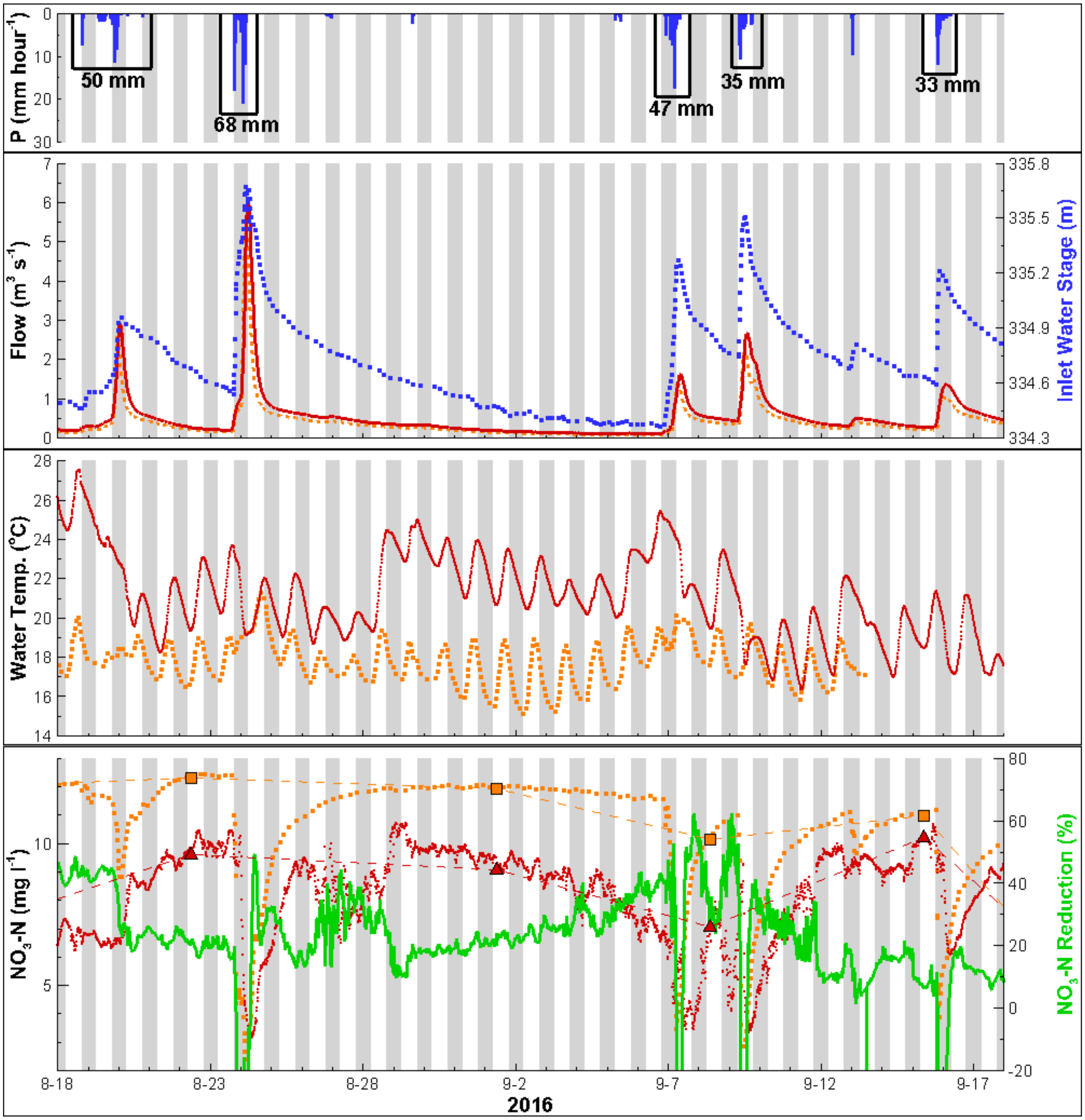

Figure 11. Example of NO_3-N retention patterns captured with high-frequency (15-min) monitoring during a 31-day period of intense and frequent precipitation in Aug and Sep of 2016. Inlet (orange dashes) and outlet (red dots) conditions are shown. The top panel shows hourly Stage IV radar rainfall estimates; the second panel from the top shows 15-min discharge (modeled) and hourly inlet water stage (blue dashes; measured); the third panel from the top shows 15-min water temperature (measured); the bottom panel shows 15-min NO_3-N concentrations (measured) and the instantaneous 15-min NO_3-N concentration reduction (green; measured). NO_3-N concentrations denoted by large squares (inlet) or triangles (outlet) connected by dashed lines mimic weekly grab sampling. Gray shaded areas indicate night time (sunset to sunrise).

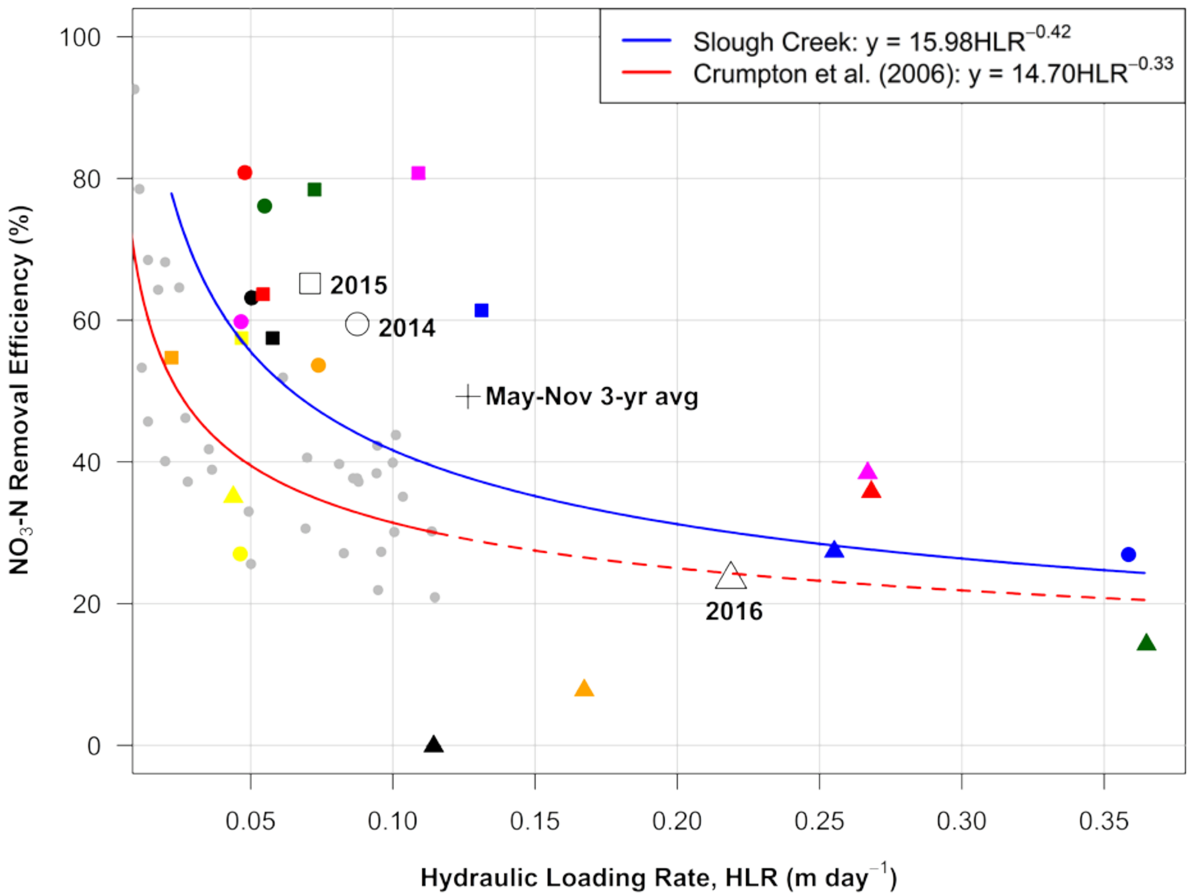

Figure 12. Comparison of removal efficiency-hydraulic loading rate relations between the Slough Creek wetland and Crumpton et al. (2006). The Slough Creek regression line (blue) is based on monthly values from May-Nov whereas the Crumpton et al. (2006) regression line (red) is based on annual averages. Each colored point defines the average daily removal efficiency and hydraulic loading rate for a given month at the Slough Creek wetland during the 3-yr study (n = 21 months). A negative removal efficiency was estimated in Nov 2016 (black triangle) and was removed from the regression analysis for the Slough Creek wetland. Symbols denote the year (2014: circle; 2015: square; 2016: triangle) and colors denote the month (May: yellow; Jun: blue; Jul: magenta; Aug: red; Sep: green; Oct: orange; Nov: black). May-Nov averages are also shown. The Crumpton et al. (2006) removal efficiency-hydraulic loading rate relation was derived from annual data (n = 34) of several Ohio, Illinois, and Iowa wetlands (gray dots).

4. PREDICTING NITRATE REMOVAL FROM IOWA'S SECOND LARGEST CREP WETLAND

Abstract

A simple stream nitrogen model was developed in MIKE 11 ECO Lab and used to predict nitrate-nitrogen (NO_3-N) removal from Iowa's second largest CREP wetland. Wetland simulations implementing the stream nitrogen model and imposed water datasets from upstream of the wetland (inlet) produced satisfactory warm season (May-Nov) predictions of NO_3-N concentration and water temperature downstream of the wetland (outlet) in 3/4 and 4/4 study years, respectively, at the daily and monthly time scales. Simulated outlet NO_3-N was most sensitive to the denitrification first order rate constant during low flow periods, and temperature dependent kinetics were necessary to accurately predict the enhanced NO_3-N removal observed during summer. The denitrification rate constant and temperature had low influence during storm events.

In the continuous, integrated watershed simulations, satisfactory warm season predictions of inlet and outlet NO_3-N concentration and outlet water temperature were achieved in all four study years at the monthly time scale. Simulated inlet NO_3-N concentrations were within the range of observations and exhibited similar seasonal patterns, recession slopes, and storm dilution trends, while increases in NO_3-N following storm events were overestimated. Temperature had a pronounced effect on the simulated wetland NO_3-N removal over an annual cycle. Monthly wetland NO_3-N removal predicted with temperature dependent kinetics exhibited strong seasonality while wetland NO_3-N removal predicted with first order kinetics only was considerably higher in colder months (Oct-May) and more uniform throughout the year. Annual NO_3-N loading was reduced by an average of 10% (50 g m^{-2} wetland yr^{-1}) with temperature sensitivity and 19% (94 g m^{-2} wetland yr^{-1}) without. The mass retention estimates are higher than

expected and the percent removal estimates are lower than expected as a result of overestimated NO_3-N loading. The NO_3-N loading inaccuracies stem from a probable overestimation in simulated tile flow and highlight the fundamental importance of accurate hydrologic predictions before consideration of water quality processes.

Introduction

Wetlands provide various environmental and societal benefits including habitat for plants, wildlife, and fish, recreation, flood control, carbon sequestration, and the improvement of water quality (Crumpton, 2001; Zedler, 2003; Mitsch et al., 2014; Thomas et al., 2016). In the U.S. Midwest and other agriculturally intensive areas, wetlands are a particularly important conservation practice for nutrient retention to lessen the environmental impacts associated with nitrate-nitrogen (NO_3-N) loss in tile drainage (Baker et al., 1975; Crumpton and Goldsborough, 1998). Monitoring studies have demonstrated that wetlands can reduce agricultural NO_3-N loads by 20-70% (Drake et al., 2018) via denitrification and to a lesser extent plant uptake (Xue et al., 1998; Lin et al. 2002). Wetland restoration and construction is especially important in Iowa, which has lost at least 95% of its wetland areas that existed prior to European settlement (Miller et al., 2009), given its disproportionate contribution to the NO_3-N load entering the Gulf of Mexico (Jones et al., 2018).

Water quality modeling at the watershed scale has typically been performed with conceptual, lumped parameter watershed models. Watershed models of this type decompose the landscape into subbasins and river reaches defined by spatially averaged (lumped) flow and water quality parameters, use conceptual approaches to represent subsurface flow processes, and generally rely on more empirically-based methods to describe flow and water quality processes. Ikenberry et al. (2017) used the Soil and Water Assessment Tool (SWAT) to accurately simulate daily wetland outflow and NO_3-N removal in two small Iowa constructed wetlands (drainage areas < 3.1 km^2). The model performed well at predicting daily wetland outflow and NO_3-N removal when measured inflow and NO_3-N concentration were imposed, while performance was limited by the accuracy in predicting inlet conditions with the full watershed simulations. SWAT has also been used to simulate nitrogen fate and transport in large Iowa watersheds (9400-16175 km^2), but these studies only assessed changes to land use and agricultural management (Jha et al., 2007; Schilling and Wolter, 2009). The Hydrological Simulation Program-Fortran (HSPF), another lumped parameter, continuous watershed model that can operate on a subdaily time step, was applied in the early 1980s to both small (52 km^2) and large Iowa (7240 km^2) watersheds to evaluate the impacts of conservation tillage and contour farming on annual runoff and nutrient balances (Donigian et al., 1983). Following calibration, the model simulated annual nutrient balances and stream nutrient concentrations in the expected range. More recently, Allen et al. (2015) performed a continuous 65-yr simulation of water quality in a 1696 km^2 eastern Iowa agricultural watershed using the water quality parameters developed from the 1983 studies; simulated stream NO_3-N concentrations tended to be several times higher than monthly grab samples but were reasonable given no calibration was performed. Finally, another conceptual hydrologic model (THREW) simulated the annual nitrogen balance and stream nitrogen concentrations in a sufficient manner over a 10-yr period for a 1400 km^2, tile-drained watershed in south-central Illinois but predictions were made at coarse spatial scales (74 km^2 average subbasin size; Li et al., 2010).

MIKE SHE-MIKE 11 is a physically-based, spatially distributed watershed model that has been used previously to evaluate agricultural wetlands (Thompson et al. 2004; Clilverd et al. 2016). However, most of these studies evaluated the hydrologic impacts of wetlands with little attention given to water quality (Cui et al. 2005; Christierson et al., 2015). In part because of the large number of inputs required and large computational expense, nutrient fate and transport simulations with physically-based watershed models have been limited. The objectives of this study were to: (1) develop a stream nitrogen model using MIKE 11 ECO Lab to predict wetland NO_3-N removal; (2) evaluate the stream nitrogen model through wetland simulations using imposed inlet conditions to predict outlet NO_3-N concentrations and removal at Iowa's second largest CREP wetland; and (3) conduct integrated watershed simulations using the ECO Lab soil nitrogen and stream nitrogen models to predict wetland NO_3-N dynamics.

Materials and Methods

MIKE SHE-MIKE 11 ECO Lab

MIKE SHE is a physically-based, spatially distributed watershed model capable of continuous simulations and was overviewed in Chapter 2. MIKE 11 is a one-dimensional hydraulic model that simulates flows and water levels in streams, rivers, and lakes according to the fully dynamic Saint Venant equations. Solute transport is simulated by the advection-dispersion equation (DHI, 2017). Since MIKE 11 is a one-dimensional model, predicted water velocities (used to compute discharge) and solute concentrations represent mean, depth-averaged estimates for the entire cross section (DHI, 2017). MIKE 11 can be coupled to MIKE SHE to perform watershed simulations describing how surface and subsurface water fluxes (calculated by MIKE SHE) are transported through the river network (calculated by MIKE 11).

ECO Lab is another MIKE module composed of user-defined state variables and process equations for ecological modeling applications. ECO Lab can be coupled to the advection-dispersion module in MIKE 11 like was done with MIKE SHE (Chapter 2). Part of this chapter describes the development and testing of a stream nitrogen model using MIKE 11 ECO Lab.

MIKE 11 ECO Lab Stream Nitrogen Model

The ECO Lab stream nitrogen model describes the primary processes affecting the balance of inorganic nitrogen in riverine settings (Figure 13). The model includes soluble ammonium (NH_4-N) and nitrate (NO_3-N) and the processes nitrification and denitrification. Because most microbial processes are temperature dependent (Rolston, 1981), water temperature is also modeled. Nitrification and denitrification are modeled using first order, temperature dependent kinetics, which assumes process rates are solely limited by nitrogen concentration and temperature; other factors known to influence process rates, notably the effects organic carbon and dissolved oxygen can have on denitrification (Xue et al., 1999; Lin et al., 2002; Jones and Kult, 2016), are not considered. First order kinetics are most valid for low nitrogen concentrations; this assumption may often times be violated in agricultural landscapes that are subject to large NO_3-N losses resulting in stream NO_3-N concentrations that often exceed 10-15 mg l^{-1} (Schilling and Libra, 2000; Ikenberry et al., 2014; Jones et al., 2018). Despite these limitations, this approach is widely used in practice and was adopted in this study because of its simplicity and need for only two parameters (the first order rate constant at 20 ^oC and the Arrhenius temperature coefficient) to be specified for the stream nitrification and denitrification processes.

The MIKE 11 stream temperature model simulates the atmospheric-water heat balance, which includes net shortwave solar radiation, net atmospheric longwave radiation, water longwave radiation, evaporation (latent heat flux), and conduction (sensible heat flux) (Chapra, 1997). The same stream temperature model, along with heat exchange from groundwater and tile drainage sources simulated by MIKE SHE, was implemented previously in MIKE 11 ECO Lab (Loinaz et al., 2013). Because the current version of MIKE 11 ECO Lab no longer stores the separate MIKE SHE flow contributions (surface runoff, groundwater, and tile drainage) at each

MIKE 11 calculation node, thermal contributions from baseflow were not able to be included in the current study. More details on the atmospheric-water heat balance modeled in MIKE 11 ECO Lab can be found in Chapra (1997) and Loinaz et al. (2013).

MIKE SHE-MIKE 11 Model Setup

The MIKE 11 ECO Lab stream nitrogen model was evaluated using water quality measurements at the Slough Creek CREP wetland (Chapter 3; Drake et al., 2018). Wetland simulations were performed using the Slough Creek MIKE 11 model defined between the inlet and outlet water quality sensors. Simulated discharge and measured NO_3-N concentration and water temperature at the inlet water quality sensor were imposed as boundary conditions and simulated NO_3-N concentration and water temperature at the outlet water quality sensor were compared to observations. Following, integrated watershed simulations for the entire Slough Creek wetland watershed were performed to predict inlet NO_3-N and wetland NO_3-N removal. The integrated watershed simulations included both the MIKE SHE ECO Lab soil nitrogen model (Chapter 2) and the MIKE 11 in-stream nitrogen model to simulate the entire nitrogen cycle.

The Slough Creek MIKE SHE-MIKE 11 model (Chapter 3; Drake et al., 2018) was modified in several ways to better reflect conventional agricultural management required for the water quality simulations. Corn and soybean areas were differentiated and separate evapotranspiration parameters were assigned to each crop (Allen et al., 1998). A conventional corn-soybean rotation was assumed based on the 2016 land use from the Cropland Data Layer (Figure 14); corn and soybeans were assumed to be planted on 15 May and 22 May, respectively, each year and harvest occurred in mid Oct to early Nov (Al-Kaisi 2000; Qi et al., 2011). A single spring application of nitrogen fertilizer was applied to corn two weeks after planting (May 29) at a rate of 228 kg ha^{-1}; the most recent (2012) estimate for Floyd and Mitchell counties (Brakebill, et al., 2017).

Several calibration changes were made to improve the simulated hydrology and water quality. Simulated monthly runoff appeared to be overestimated from Nov-Feb and underestimated in Mar, so a time varying degree-day melt coefficient was introduced (Mockus, 2004). The number of computational layers in the saturated zone was increased from three to six to describe simulated groundwater concentrations in more detail and each layer was defined with a variable (rather than uniform) depth to improve the stability of the simulated groundwater table. Finally, in the original model nearly all the baseflow (>90%) was derived from drain flow which caused the simulated NO_3-N concentrations to be too flashy. To partially correct this, the horizontal hydraulic conductivities in the saturated zone were increased from ten to 100 times the vertical hydraulic conductivities to increase the lateral flow and the groundwater contribution to streamflow. While this change did not significantly alter the total baseflow simulated by the model, it did reduce the simulated tile drainage flow by 13% and increased the simulated groundwater flow by 17%. Simulated NO_3-N concentration patterns were more comparable to observations as a result.

Hydrologic and Water Quality Simulations

High-frequency (15-min) measurements of NO_3-N concentration upstream (inlet) and downstream (outlet) of the wetland were available during the warm season (Apr/May through Oct/Nov) over a 4-yr (2014-17) period. This 4-yr period served as the evaluation period for the hydrologic and water quality simulations.

The Slough Creek watershed MIKE SHE-MIKE 11 model was ran continuously from Jan 2010 through Sep 2017 (Stage IV radar rainfall estimates not available after Sep 2017).

Following the same approach used in Chapter 3 (Drake et al., 2018), simulated water balance components and discharge from 2014-17 were compared to literature estimates and Little Cedar observations (USGS 05458000, 793 km^2).

The wetland simulations were performed with MIKE 11 ECO Lab using the MIKE 11 hydraulic model defined between the inlet and outlet water quality sensors (Figure 15). As discharge was not regularly monitored at the wetland, simulated discharge along with measured NO_3-N concentration and water temperature at the inlet sensor were imposed as boundary conditions in the hydraulic model. Simulated and observed NO_3-N concentration were compared at the outlet sensor to assess the efficacy of the MIKE 11 ECO Lab model in predicting wetland NO_3-N removal. The wetland simulations were also ran without temperature to assess its influence on simulated NO_3-N removal. A sensitivity analysis of the first order rate constant for denitrification at 20 ^oC (k_{den}) was performed by running simulations with the lower (0.05 day^{-1}), upper (0.3 day^{-1}), and recommended (0.1 day^{-1}) values (Jorgensen, 1979) to illustrate the feasible range and potential uncertainty in simulated NO_3-N concentrations. Ikenberry et al. (2017) performed a similar analysis with SWAT and noted simulated NO_3-N concentrations were far less sensitive to the Arrhenius temperature coefficient, so the recommended value for denitrification (1.16) was maintained for all the simulations (Jorgensen, 1979).

Finally, integrated watershed nitrogen fate and transport simulations were conducted to make NO_3-N predictions at the inlet sensor and assess the simulated wetland NO_3-N removal. The integrated watershed simulation utilized both the MIKE SHE ECO Lab soil nitrogen model (Chapter 2) and the MIKE 11 ECO Lab in-stream nitrogen model (this chapter) to simulate nitrogen fate and transport in a comprehensive manner. The parameters for the soil nitrogen model were taken from Chapter 2 and the recommended parameters for nitrification and denitrification from Jorgensen (1979) were used for the stream nitrogen model. The stream nitrogen model simulated nitrification throughout the entire MIKE 11 network while denitrification was only included in the wetland. This approach has been used in other modeling studies (Nieuwenhoven, 2015) and was felt to be reasonable given biological retention processes in the Slough Creek stream are expected to be small compared to the wetland (Drake et al., 2018).

Results and Discussion

Simulated Hydrology

The simulated hydrology for the Slough Creek watershed was reasonable across different time scales. Simulated annual water balance components for the Slough Creek outlet from 2014-17 are representative of Iowan Surface streams and discharge estimates are comparable to the Little Cedar reference gauge (Table 11). The 3-yr (2014-16) average annual precipitation was 1048 mm, 17% higher than the 30-yr (1981-2010) annual average (898 mm; PRISM, 2018). The 3-yr annual average simulated evapotranspiration of 63% was comparable to regional estimates of 60-70% (Sanford and Selnick, 2013). Annual runoff coefficients (Q/P) from 2014-16 ranged from 24-43%, comparable to those observed for the Little Cedar (30-45%). The fraction of streamflow derived from baseflow (groundwater and tile drainage), called the baseflow index (BFI), was higher in the Slough Creek simulation (3-yr avg: 84%) than estimated for the Little Cedar (3-yr avg: 65%). The overestimated baseflow contribution is primarily attributed to the simulated tile drainage, which alone accounted for 58% of the average annual streamflow volume. Because information on tile drainage extent is generally not available, all row cropped (corn and soybean) areas (83% of the model domain) in the model were assumed to be tiled. While this is a reasonable assumption for the Iowan Surface, the modeling impact is significant.

The MIKE SHE tile drainage module works in such a way that whenever the simulated groundwater table is within one meter of the surface (the assumed tile drain depth) in designated tiled areas, a fraction of the groundwater is instantaneously routed to the nearest MIKE 11 river branch. Finally, mean daily discharge was comparable between the Slough Creek simulation and the Little Cedar, while peak mean daily streamflows were more variable.

Simulated monthly runoff from Jan 2014 to Sep 2017 (n = 45 months) for the Slough Creek outlet was also comparable to observed monthly runoff for the Little Cedar (Figure 16). Except for Sep 2016 (far right green dot in the plot), there is generally no systematic underestimation or overestimation of monthly runoff, though simulated monthly runoff for Apr-Jun (blue dots) was sometimes higher than observed for the Little Cedar. The anomaly in Sep 2016 is partially attributed to rainfall differences; the basin average monthly precipitation in Sep 2016 for the Little Cedar was 25% greater than Slough Creek, while the Little Cedar observed monthly runoff was 61% greater than the Slough Creek simulated monthly runoff.

The simulated average monthly runoff from Jan 2014 to Sep 2017 for the Slough Creek watershed is also comparable to the Little Cedar (Figure 17). Seasonal patterns in monthly runoff were reasonably simulated as monthly runoff increased from Jan-Apr, decreased in May, peaked in Jun, and then tended to decrease over the remainder of the year. The lower than expected simulated monthly runoff in Sep is largely influenced by the exceptionally wet Sep in 2016; excluding this year, the average Sep monthly runoff reduced to 10-11 mm for both Slough Creek and the Little Cedar. Overall, the simulated Slough Creek runoff was 2.3% lower than the observed Little Cedar runoff.

The frequency and magnitude of normalized mean daily discharge simulated for the Slough Creek outlet was also favorable with Little Cedar observations. Figure 18 shows the flow duration curves computed for the Slough Creek outlet and the Little Cedar. Both flow duration curves are similar to each other over most exceedance probabilities. At low flows that are matched or exceeded more than 90% of the time (middle panel), daily discharge simulated for Slough Creek decreases more rapidly than observed for the Little Cedar; this is attributed to the greater sensitivity of the smaller Slough Creek catchment to seasonal variations in tile flow. At higher flows that are matched or exceeded less than 10% of the time, the flow duration curves are similar. At extremely low exceedance probabilities, daily discharge simulated for Slough Creek is higher than the Little Cedar presumably because of the flashier nature associated with smaller catchments.

Finally, the response and timing of simulated inlet and outlet discharge compare favorably with inlet water stage measurements during precipitation events (Figure 19). While the recession slopes of the simulated hydrographs are steeper than expected, the shape of the rising limb and timing of peak discharge agree well with water stage measurements. Simulated outlet discharge is slightly delayed and an average 29% greater than the simulated inlet discharge, reflecting the 29% increase in drainage area between the inlet (12.3 km^2) and outlet (15.8 km^2) sensors.

Wetland Simulations

Wetland NO_3-N simulations were ran from 8 May to 20 Nov in 2014, 19 May to 20 Nov in 2015, 6 May to 17 Nov in 2016, and 12 Apr to 6 Oct in 2017. These time periods corresponded to when 15-min water temperature and NO_3-N concentration measurements were mostly available at both water quality sensors. Data gaps in inlet concentration or temperature were estimated with linear interpolation. The wetland simulations used the simulated inlet discharge as the lone hydraulic forcing; simulated flow contributions to the wetland downstream of the inlet sensor were not included. This was deemed reasonable for the purpose of evaluating the MIKE 11 in-stream nitrogen model given the Slough Creek stream at the inlet sensor captures water from 78% of the wetland drainage area and the simulated inlet discharge represented a similar fraction of the simulated outlet discharge. Performance of the MIKE 11 ECO Lab in-stream nitrogen model was assessed qualitatively with time series plots and quantitatively using statistical criteria adapted from Moriasi et al. (2015). Although the performance criteria developed by Moriasi et al. (2015) are intended for simulated discharge and NO_3-N load, the metrics were applied to NO_3-N concentration and water temperature in this study. The water quality simulations were considered "satisfactory" if Nash-Sutcliffe Efficiency (NSE) exceeded 0.35, percent bias (PBIAS) was less than 30%, and the coefficient of determination (R^2) was greater than 0.30 for daily and monthly data.

The wetland simulations of water temperature during the warm season were satisfactory in all four study years at the 15-min, daily, and monthly time scales (Figure 20 and Table 12). With imposed inlet temperature observations, the MIKE 11 stream temperature model was able to reasonably reproduce the diurnal and seasonal patterns in water temperature at the outlet sensor, though the simulated diurnal amplitude tended to be overestimated. The overestimation in diurnal amplitude, particularly in July and Aug, is attributed to several factors related to the wetland simulation setup, inability to include baseflow heat exchange in the MIKE 11 stream temperature model, and errors in cross section geometry upstream of the wetland. By not including the simulated flow contributions to the wetland downstream of the inlet sensor in the wetland simulations, an estimated 22% of the total simulated baseflow at the outlet sensor was ignored, which would lower the simulated outlet water temperatures to some degree. However, the change is expected to be relatively small given the previous MIKE SHE-MIKE 11 ECO Lab stream temperature study estimated that a 10% reduction in groundwater flow would increase stream temperature by an average of 0.3 ^{o}C and a maximum of 1.5 ^{o}C (Loinaz et al., 2013).

Probably more influential on the water temperature simulation is errors in cross section geometry. The MIKE 11 cross sections used in this study were extracted from a 2-m DEM. While the DEM was altered in the vicinity of the wetland to reflect the engineering design plans, the upstream cross sections were not. Figure 21 shows an example of differences between field-surveyed cross sections and cross sections estimated from a 1-m LIDAR DEM near the inlet sensor; the DEM cross sections tend to smooth out the cross section shape and underestimate channel depth compared to the surveyed cross sections. Since water temperature is strongly influenced by hydraulic geometry and is inversely related to water depth in the MIKE 11 model, cross section inaccuracies upstream of the wetland could have a significant impact on the modeled water temperature. It is worth noting that diurnal temperature variations were also overestimated in the other MIKE SHE-MIKE 11 stream temperature study (Loinaz et al., 2013).

Outlet NO_3-N concentration was simulated in a poorer manner than water temperature but the simulated range encompassed the observations most of the time (Figure 22 and Table 13). Figure 22 shows the imposed inlet NO_3-N concentration (orange), observed outlet NO_3-N concentration (red), and sensitivity of simulated outlet NO_3-N concentration to both k_{den} and temperature. Simulated outlet NO_3-N concentration was most sensitive to k_{den} when inlet NO_3-N concentration gradually decreased (associated with lower discharge periods) and least sensitive to k_{den} when inlet NO_3-N concentration sharply increased (associated with higher discharge periods connected to storm events). During times of receding inlet NO_3-N, lower discharges presumably increased the residence time in the wetland and provided more conducive conditions for enhanced processing, hence the influence of k_{den} was more apparent.

The sensitivity of simulated NO_3-N concentration to water temperature was also apparent. Simulated outlet NO_3-N concentration was most sensitive to temperature in July/Aug and Oct/Nov when wetland water temperatures deviated from 20 ^oC the most. The temperature dependent outlet NO_3-N concentration predictions are lower in July/Aug when warmer water temperatures should theoretically enhance wetland NO_3-N removal and higher in Oct/Nov when colder water temperatures should theoretically inhibit NO_3-N removal. The influence of temperature is minor during precipitation events. Temperature dependent NO_3-N predictions were in better agreement with observations in July/Aug while the importance of temperature was less obvious in other months. Overall, the temperature dependent model predicted outlet NO_3-N concentrations slightly better during the 4-yr study period (Table 13); the temperature dependent NO_3-N predictions were satisfactory in three of four years (2014, 2016, and 2017), while the predictions without temperature were satisfactory in two of four years (2014 and 2017).

An example of simulated and observed NO_3-N dynamics during a month of frequent precipitation (Sep 2016) is shown in Figure 23. Monthly rainfall in Sep 2016 was 143% above average (Drake et al., 2018). During storm events, dilution from lower concentrated rainfall and surface runoff decrease inlet and outlet NO_3-N concentrations; after events, NO_3-N concentrations return to near pre-event levels as soil NO_3-N is mobilized and leached from upstream areas. During this period, the simulated increases (mobilization) and decreases (dilution) in outlet NO_3-N are reasonable, though concentrations are under predicted, particularly in the second half of Sep when hydrologic conditions greatly diminished wetland NO_3-N removal. During periods of high discharge, the sensitivity of simulated outlet NO_3-N concentration to k_{den} and temperature is low. The diurnal signal observed in the outlet NO_3-N concentrations is not captured by the predictions or observed in the inlet NO_3-N concentrations, which suggests biological retention processes in the Slough Creek stream network are minimal compared to the wetland. Finally, outlet water temperature is simulated reasonably well during this time period and its effect on the outlet NO_3-N predictions is as expected – lower (higher) concentrations are predicted when the water temperature is above (below) 20 ^oC.

Wetland Simulation Summary and Estimated NO_3-N Loading

Seasonal average simulated and observed outlet water temperature, outlet NO_3-N concentration, and estimated inlet and outlet NO_3-N loading are summarized in Table 14. Seasonal (May~Nov) average outlet water temperature was consistently around 18-19 ^oC – an average of 5 ^oC greater than inlet water temperature – and the seasonal average outlet temperature predictions deviated by less than 0.7 ^oC in each year. Seasonal average outlet NO_3-N concentration was more variable, from 3.1 mg l^{-1} in 2015 to 9.1 mg l^{-1} in 2016. The seasonal average NO_3-N predictions with and without temperature dependence were similar and an average of 6-7% greater than the 4-yr observed average of 5.5 mg l^{-1}. Both predictions overestimated the lowest and highest average outlet NO_3-N concentrations in 2015 and 2016, respectively.

NO_3-N loading estimates were calculated at the inlet sensor using the 15-min simulated inlet discharge and measured inlet NO_3-N concentration and at the outlet sensor using the simulated outlet discharge from the wetland simulation (the advected inlet discharge) and simulated or measured outlet NO_3-N concentration. Wetland NO_3-N removal was estimated as the difference between inlet and outlet loading. NO_3-N loading followed similar patterns to discharge. Average inlet loading during the evaluation period ranged from 75-252 kg day^{-1}. The 4-yr average inlet load of 156 kg day^{-1} corresponds to an average watershed yield of 12.7 kg km^{-2} day^{-1}, higher than the Iowan Surface long term average annual estimate of 6.1 kg km^{-2} day^{-1}

(Schilling and Wolter, 2005) but comparable to the 2016 annual estimate of 11.2 kg km^{-2} day^{-1} (Jones et al., 2018). The estimated seasonal wetland NO_3-N removal ranged from an average of 29-53 kg day^{-1}. The 4-yr average wetland removal of 39 kg day^{-1} corresponds to an average NO_3-N retention of 0.31 g m^{-2} wetland day^{-1}, within the range observed for natural and constructed agricultural wetlands (0.1-0.4 g m^{-2} day^{-1}; Drake et al., 2018). Seasonal percent NO_3-N removal ranged from 14% in 2016 (year with highest average inlet load) to 70% in 2015 (year with lowest average inlet load); the 4-yr average percent removal was 25%, also within the range commonly observed for agricultural wetlands (20-70%; Drake et al. 2018). Seasonal predictions of wetland removal with and without temperature dependence were similar to each other; both predictions were lower than expected in 2015 because of overestimated NO_3-N concentration in late Jun, early July, and mid Sep and higher than expected in 2016 because of underestimated NO_3-N concentrations in Oct and Nov (Figure 22). These estimates are intended to provide a general indication of NO_3-N removal at the Slough Creek wetland during the warm season but are incomplete since the wetland simulations did not include 22% of the simulated wetland outflow.

Watershed Simulations

Simulated Flood Reduction Impact of the Slough Creek Wetland

The integrated watershed simulations suggest the Slough Creek wetland is expected to have a low flood reduction impact. Figure 24 compares the simulated monthly peak discharges at the outlet sensor for watershed simulations with and without the wetland included in the MIKE 11 model for Jan 2014 to Sep 2017 (n = 45 months). Interestingly, simulated monthly peak discharges with the wetland included are greater in 37 of the 45 months, particularly at larger peak flows. The Slough Creek wetland is expected to have a small impact on reducing peak discharges given CREP wetlands are designed primarily for water quality improvement and generally lack the hydraulic characteristics necessary to be effective flood control structures; they have shallow pool areas to encourage plant establishment and enhance nutrient processing, large outflow structures that do little to throttle down incoming flows (e.g. 61 m weir at the Slough Creek CREP wetland), and, consequently, minimal flood storage to temporarily retain runoff during storm events. However, the wetland is rarely expected to have an adverse impact on flooding as the predictions would suggest. This unexpected result is primarily attributed to the MIKE 11 model setup. While seepage losses were simulated by MIKE 11, open water evaporation was not; although evaporation from the Slough Creek stream network is expected to be small compared to the watershed evapotranspiration, evaporative losses from the wetland could noticeably lower the pool elevation in the summer months, the time when the largest peak discharges often occur (IFC, 2014). To maintain stability of the predicted nitrogen concentrations, the MIKE 11 model also required imposing a small constant inflow (0.001 m^3 s^{-1}) at the upstream end of the four stream branches to ensure the hydraulic model had some water all the time; as a result, the simulated water level in the wetland was nearly always at or above normal pool and the simulated wetland outflow never dropped below 0.004 m^3 s^{-1}; as a result, the simulated flow attenuation through the wetland was minimal. This compromise of the hydrologic simulation to improve the water quality simulation performance illustrates one of the challenges associated with coupled hydrologic and water quality modeling and deserves further attention in future work.

Simulated Annual Nitrogen Balance

The simulated annual nitrogen balance for the Slough Creek wetland watershed is comparable to literature estimates, though plant nitrogen uptake and NO_3-N loss are likely underestimated and overestimated, respectively (Table 15). The annual values in Table 15 reflect watershed-integrated totals calculated by MIKE SHE and do not pertain to a specific location. In addition to annual fertilizer inputs (228 kg ha^{-1} applied to corn each spring), an average 13 kg ha^{-1} of NO_3-N and 5 kg ha_{-1} of NH_4-N were added annually through precipitation. Annual net mineralization ranged from 140-172 kg ha^{-1}, within the range of 100-170 kg ha^{-1} identified in field monitoring and modeling studies of corn-soybean systems (Carpenter-Boggs et al., 2000; Thorp et al., 2007; Qi et al., 2011; Qi et al. 2012). Simulated field denitrification was composed of contributions from the top meter of the unsaturated zone (meant to mimic the root zone) and the saturated zone (shallow groundwater within 11 m of the surface). Simulated annual denitrification in the top meter of the unsaturated zone represented 5-12% of annual fertilizer inputs, which are typical estimates for agricultural watersheds (Li et al., 2010; Qi et al., 2012). Annual plant nitrogen (NO_3-N + NH_4-N) uptake ranged from 130-188 kg ha^{-1} which is probably underestimated. While these estimates reflect average annual uptake rates for all vegetation (grass, forest, and row crops) in the watershed model, the vast majority of the watershed vegetation is in row crops (91%). Comparable crop uptake was observed for corn-soybean systems at lower (140 kg ha^{-1}) fertilizer rates (Qi et al., 2011) and crop uptake in excess of 200 kg ha^{-1} has been simulated by RZWQM at similar fertilizer rates to the one used in this study (Thorp et al., 2007).

As a result of the probable underestimation in simulated plant nitrogen uptake, simulated annual NO_3-N losses in MIKE SHE were likely overestimated. The 4-yr average simulated NO_3-N loss delivered from MIKE SHE to MIKE 11 through surface runoff, groundwater, and tile drainage was 45 kg ha^{-1}; while not unreasonable, the 4-yr average simulated NO_3-N loss is higher than the long term average annual (22 kg ha^{-1}; Schilling and Wolter, 2005) and 2016 (41 kg ha^{-1}; Jones et al., 2018) in-stream NO_3-N loads estimated for the Iowan Surface. The overestimated NO_3-N loss is primarily attributed to an overestimated contribution of simulated tile drainage flow. The 4-yr average contributions of simulated tile drainage to streamflow and NO_3-N loss were 58% and 74%, respectively. These estimates are likely too high given baseflow (groundwater and tile drainage) is estimated to account for approximately 60-70% of annual streamflow for most Iowan Surface streams (Table 11; Schilling and Wolter, 2005); additionally, one study of an Iowa watershed partially located in the Iowan Surface estimated using measurements of streamflow and NO_3-N concentration that tile drainage accounted for 15-43% of streamflow volume and 50-80% of stream NO_3-N load from April to Nov during one year (Arenas Amado et al., 2017). The overestimation tile flow is partially attributed to the assumption that all row cropped areas in the model are tiled in lieu of more detailed information available on tile spatial extent.

Inlet Predictions

Inlet water temperature was simulated unsatisfactorily in the integrated watershed simulations (Figure 25). While seasonal patterns and the mean behavior of the inlet temperature predictions somewhat resemble the observations, the diurnal variations are significantly overestimated. The poor inlet temperature predictions are attributed to poor (underestimated) predictions of water depth in the Slough Creek stream network due primarily to errors in the DEM-extracted cross section geometries upstream of the wetland (Figure 21). Since the cross sections upstream of the wetland likely did not capture the narrow channel form, simulated water depths tended to be very shallow; as a result, the temperature predictions were sometimes erroneously high or low due to the low thermal inertia (buffering capacity) of the simulated water body. This behavior is apparent in the inlet water temperature predictions. While the poor inlet temperature predictions are primarily attributed to errors in cross section geometry, we recognize the difficulty associated with making reasonable hydrologic predictions everywhere in a watershed and acknowledge that errors in simulated discharge may also contribute to the poor temperature predictions at the inlet sensor.

Predicted inlet NO_3-N concentrations derived from the contributing watershed were comparable to observations (Figure 26). While not perfect, the simulated range and pattern of inlet NO_3-N concentrations are similar to the 15-min observations, particularly given the watershed simulations were continuous (Jan-Dec) and the uncertainties associated with the simulated hydrology and assumed agricultural management. Simulated inlet NO_3-N concentrations exhibit similar seasonal patterns (higher concentrations in Jun and lower concentrations in Aug), storm NO_3-N dynamics, and recession slopes to the observations. Simulated inlet NO_3-N concentrations were often underestimated in Apr and May each year, particularly in 2014 and 2016. This is likely the result of overestimated runoff in Apr and May (Figure 17) as well as potentially earlier fertilizer application than assumed in the model (end of May). It is also interesting to note that despite the poor inlet temperature predictions, the effect on the simulated inlet NO_3-N concentrations was minimal. While in-stream denitrification was only simulated in the wetland, in-stream nitrification was simulated in the entire Slough Creek stream network. Temperature had a minimal impact on the simulated in-stream NO_3-N concentrations in part because simulated in-stream NH_4-N concentrations were generally low but primarily because the residence time in the Slough Creek stream network was very small compared to the wetland. Because water is not detained in the stream network like it is in the wetland, simulated in-stream processing is low and advective transport is high. The model results confirm this as simulated nitrification in the wetland was noticeably greater than elsewhere in the Slough Creek stream network.

The most notable discrepancy of the inlet NO_3-N predictions is the overestimated response of NO_3-N following storm events. This behavior is attributed to the overestimated contribution of simulated tile drainage. Figure 27 illustrates the influence of the MIKE SHE tile drainage module on simulated discharge and NO_3-N concentration at the inlet sensor during July 2016. When the tile module is removed from the model, the simulated discharge is flashier – the storm peak discharges are larger and the recession slopes are steeper. These discharge dynamics reflect the fact that the simulated surface runoff increased by nearly a factor of four from 16% to 58% of the average annual streamflow when the tile module was removed. As a result, the simulated dilution of NO_3-N during storm events is overestimated while the subsequent increase in NO_3-N is more reasonable (the opposite pattern observed when the tile module was included). Future efforts should focus on revising the simulated hydrology to find a more suitable compromise between these two limits.

Outlet Predictions

Predicted outlet water temperature was in significantly better agreement with observations than the inlet predictions (Figure 28). Unlike the inlet predictions, the simulated outlet temperature diurnal variations were reasonable most of the time. The improved performance in the stream temperature predictions at the outlet sensor is due to the more accurate cross section geometry in the vicinity of the wetland and, more importantly, the fact that water is always detained in the wetland, ensuring simulated water depths are reasonable (0.3~0.6 m). The integrated watershed simulations indicate that the stream temperature model produces reliable predictions when simulated water levels in the MIKE 11 network are sufficiently deep.

Simulated outlet NO_3-N concentrations were comparable to measurements, indicating the NO_3-N removal predicted by the in-stream nitrogen model was reasonable (Figure 29). As with the inlet NO_3-N predictions, the NO_3-N response following storm events was overestimated, particularly during the persistent wet period in Sep 2016. Stream temperature has a notable influence on the simulated wetland denitrification. Temperature dependent NO_3-N predictions are generally higher during the colder months (April, May, Oct, and Nov) and lower during the warmer summer months (July and Aug). Temperature appears to be an important factor for accurately predicting wetland NO_3-N concentrations in summer, while the importance in other months is less straight forward to discern.

Mean Monthly NO_3-N Concentration and Wetland NO_3-N Removal

Simulated and observed mean monthly inlet and outlet NO_3-N concentrations during the 4-yr study period are shown in Figure 30. Since temperature had minimal influence on the simulated 15-min inlet NO_3-N concentration time series (Figure 26), the monthly means with and without temperature sensitivity are nearly identical. On a mean monthly basis, simulated inlet NO_3-N concentration exhibited similar seasonal patterns to the observations, but concentrations were underestimated in six of the eight monitoring months (Apr-Nov) by an average of 14%. The percent difference between simulated and observed mean monthly concentrations was greatest in May (24% underestimation) and smallest in July (3% overestimation).

Mean monthly outlet NO3-N concentrations were highest from Apr-Jun and lowest in Aug and Sep. Temperature had a noticeable effect on the outlet NO_3-N predictions. Unlike the inlet predictions, simulated outlet NO_3-N concentrations were often overestimated during the monitoring months; the temperature dependent predictions overestimated NO_3-N concentrations in all eight monitoring months by an average of 40% while the simulations without temperature overestimated NO_3-N concentrations in five of the eight monitoring months by an average of 61%. The temperature dependent outlet NO_3-N predictions display clear seasonality while the predictions without temperature were less predictable.

NO_3-N loading and wetland retention were also estimated from the integrated watershed simulation results. For the baseline watershed simulation without the wetland, annual NO_3-N loading at the outlet sensor ranged from 25 kg ha^{-1} in 2015 to 63 kg ha^{-1} in 2016 and the 4-yr average load was 41 kg ha^{-1} based on watershed area. As discussed previously, while these load estimates are not unfeasible, they are higher than expected, particularly in 2016 (Schilling and Wolter, 2005; Jones et al., 2018). With the wetland included in the watershed simulations, annual outlet NO_3-N loads were reduced by 8-15% (3-5 kg ha^{-1}) when temperature was simulated and 15-27% (6-10 kg ha^{-1}) when temperature was not simulated. The 4-yr average annual wetland NO_3-N retention was 50 g m^{-2} wetland yr^{-1} when temperature was simulated and 94 g m^{-2} wetland yr^{-1} when temperature was not simulated. Both wetland retention NO_3-N estimates are higher than the reported range in other wetland studies (10-40 g m^{-2} wetland yr^{-1}; Mitsch et al., 2014), though higher retention rates have been observed for other Midwest wetlands receiving larger NO_3-N loads (Kovacic et al., 2000; Crumpton et al., 2006).

Temperature had a noticeable effect on the simulated mean monthly wetland NO_3-N retention (Figure 31). With temperature dependent kinetics, both wetland NO_3-N mass removal and percent removal show clear seasonality. Simulated NO_3-N mass removal was essentially zero in January, gradually increased to peak rates in June and July, and decreased thereafter; percent removal followed a similar trend but peak rates occurred in July and Aug. Without temperature, wetland NO_3-N retention was more uniform throughout the year and significantly greater mass and percent removal were predicted during the colder months of Oct-May; the highest mean monthly mass removal was predicted in Apr and the highest mean monthly percent removal was predicted in Jan, respectively. Based on the better performance of predicted outlet NO_3-N concentrations during summer with temperature dependent kinetics, temperature is believed to be important for more accurately describing NO3-N retention patterns during the growing season. Outside the growing season, the simulated effect of temperature in this study may be too strong since wetland denitrification can still occur during cold periods (e.g. winter) if other necessary conditions (NO3-N concentrations, organic carbon, and dissolved oxygen) are met (Kovacic et al., 2000).

Summary and Conclusions

A simple in-stream nitrogen model was developed in MIKE 11 ECO Lab and tested on Iowa's second largest CREP wetland. Wetland simulations using imposed influent flow and water quality yielded satisfactory simulations of wetland water temperature at the 15-min, daily, and monthly time scales during the warm season (Apr/May – Oct/Nov) in all four years of the study period; satisfactory simulations of wetland NO_3-N concentration at the daily and monthly time scales were achieved in three of four years when first order, temperature dependent kinetics were assumed and two of four years when only first order kinetics were used to describe wetland NO_3-N removal. Wetland NO_3-N concentrations in July and Aug were better predicted with temperature dependent kinetics.

Integrated, continuous watershed nitrogen fate and transport simulations predicted inlet NO_3-N concentrations and assess wetland NO_3-N removal. The simulated annual nitrogen balance for the wetland watershed during the 4-yr evaluation period was comparable to literature estimates, though plant nitrogen uptake was likely underestimated leading to overestimated NO_3-N losses, particularly in tile drainage. Given the uncertainties associated with the simulated hydrology and agricultural management, inlet NO_3-N concentrations were simulated reasonably well; the predictions were within the range of observations, exhibited similar seasonal patterns and recession slopes, and reproduced some of the observed storm NO_3-N dynamics, though the NO_3-N response following storm events was overestimated. Outlet NO_3-N concentrations were predicted in a reasonable manner as well, verifying the in-stream nitrogen model can provide reliable predictions of NO_3-N. Overall, the simulations of inlet and outlet NO_3-N concentration and outlet water temperature during the primary monitoring period (May-Nov) were satisfactory at the monthly time scale in all four years.

The effect of temperature on simulated NO_3-N was also evaluated extensively in this study. The MIKE 11 stream temperature model provided reliable estimates of water temperature when a sufficient depth of water was simulated, as was consistently the case in the wetland. Despite poor inlet temperature predictions due primarily to errors in cross section geometry, the effect on the corresponding NO_3-N predictions was minimal due to the low residence time in the stream network for processing compared to the wetland. Temperature had a noticeable effect on the simulated monthly NO_3-N retention patterns. Temperature appears to be important to include when predicting NO_3-N removal during the growing season, while its importance outside the growing season may be less influential and deserves further study. The average annual wetland NO_3-N removal predicted with temperature dependent kinetics from the integrated watershed simulations was 4 kg ha^{-1} yr^{-1} based on watershed area (50 g m^{-2} wetland yr^{-1}) during the 4-yr study period, corresponding to an average annual percent removal of 10%, but revisions to several components of the hydrologic and water quality models are necessary to improve these initial estimates.

Tables and Figures

Table 11. Simulated annual water balance for the Slough Creek CREP wetland watershed.

	Slough Creek Wetland Outlet Modeled: 15.8 km^2						Little Cedar River near Ionia Observed (USGS 05458000): 793 km^2				
Year	P	ET/P	Q/P	BFI	Q_{mean}	Q_{peak}	P	Q/P	BFI	Q_{mean}	Q_{peak}
	mm				mm day^{-1}		mm			mm day^{-1}	
2014	992	0.67	0.27	0.80	0.8	20.6	993	0.30	0.64	0.8	8.8
2015	924	0.70	0.24	0.93	0.6	7.5	1005	0.30	0.69	0.8	9.4
2016	1227	0.52	0.43	0.80	1.5	22.4	1322	0.45	0.61	1.6	21.8
2017	798	0.73	0.49	0.81	1.5	32.9	795	0.44	0.71	1.3	10.1
3-yr Avg (2014-16)	1048	0.63	0.31	0.84	1.0		1107	0.35	0.65	1.1	

*2017: Jan-Sep only due to unavailability of Stage IV radar rainfall estimates during remainder of year
P: precipitation; ET: evapotranspiration; BFI: baseflow index; Qmean: mean daily discharge; Qpeak: peak mean daily discharge

Table 12. Simulated outlet water temperature performance from the wetland simulations.

	2014	2015	2016	2017
		NSE		
15-min	0.81	0.87	0.90	0.75
Daily	0.90	0.88	0.91	0.74
Monthly	0.89	0.92	0.94	0.85
		PBIAS		
15-min	-0.1	0.3	-0.5	3.6
Daily	0.1	0.4	-0.5	4.1
Monthly	0.9	0.0	-2.1	3.0
		R^2		
15-min	0.99	0.99	0.99	0.98
Daily	0.99	0.99	0.99	0.98
Monthly	0.99	0.99	0.99	0.99

NSE: Nash-Sutcliffe Efficiency

PBIAS: percent bias (%; positive indicates model overestimation)

R^2: coefficient of determination

Table 13. Simulated outlet NO_3-N concentration performance from the wetland simulations.

	Simulated				Simulated: no temperature			
	2014	2015	2016	2017	2014	2015	2016	2017
				NSE				
15-min	0.56	-1.42	0.29	0.90	0.86	-2.15	0.00	0.83
Daily	0.62	-1.69	0.35	0.92	0.87	-2.57	0.08	0.85
Monthly	0.51	-3.82	0.44	0.96	0.91	-7.23	-0.94	0.88
				PBIAS				
15-min	13.4	32.4	-5.2	5.3	-13.7	42.6	-2.9	15.4
Daily	10.6	32.4	-5.5	4.9	-14.2	42.7	-2.9	15.6
Monthly	11.6	37.2	-5.3	3.4	-11.8	38.4	-3.4	12.3
				R^2				
15-min	0.84	0.65	0.96	0.97	0.96	0.60	0.95	0.95
Daily	0.86	0.68	0.97	0.98	0.97	0.63	0.95	0.95
Monthly	0.85	0.86	0.99	0.99	0.99	0.71	0.96	0.96

NSE: Nash-Sutcliffe Efficiency

PBIAS: percent bias (%; positive indicates model overestimation)

R^2: coefficient of determination

Table 14. Seasonal (May-Nov) summary of simulated and observed outlet water temperature, NO_3-N concentration, and estimated NO_3-N loading from the Slough Creek wetland simulations.

Year	Evaluation Period	Water Temp (°C)		NO_3-N Concentration (mg l^{-1})			NO_3-N Loading (kg day^{-1})			
		Obs	Sim	Obs	Sim	Sim: no temp	Inlet	[1]Removal	[2]Sim Removal	[2]Sim Removal: no temp
2014	8 May - 20 Nov	18.5	18.5	5.2	5.9	4.5	135	29	29	34
2015	19 May - 20 Nov	18.1	18.2	3.1	4.1	4.4	75	53	34	30
2016	6 May - 17 Nov	19.1	19.0	9.1	8.6	8.8	252	35	59	50
2017	12 Apr - 6 Oct	18.9	19.6	4.7	5.2	5.7	164	40	28	31
4-yr Avg		18.7	18.8	5.5	5.9	5.8	156	39	38	36

[1]Removal estimated using the outlet NO_3-N load calculated with simulated discharge and measured NO_3-N concentration at the outlet sensor
[2]Removal estimated using the outlet NO_3-N load calculated with simulated discharge and simulated NO_3-N concentration at the outlet sensor

Table 15. Simulated annual nitrogen balance for the Slough Creek wetland watershed.

Year	Net Min.	Den.		Plant N Uptake	NO_3-N Loss				FWANC
		UZ	SZ		Surface Runoff	Groundwater	Tile Drainage	Total	
					kg ha^{-1}				mg l^{-1}
2014	148	14	21	161	3	7	25	36	13.3
2015	172	16	19	188	1	6	20	27	12.2
2016	157	27	38	130	3	13	53	69	13.0
2017	140	12	31	163	3	9	36	48	12.2
4-yr Avg	154	17	27	160	3	9	33	45	12.7

*Only Jan-Sep is considered in 2017 due to lack of Stage IV radar rainfall estimates
Net Min: net mineralization; Den: denitrification; UZ: unsaturated zone; SZ: saturated zone; FWANC: flow-weighted average NO_3-N concentration

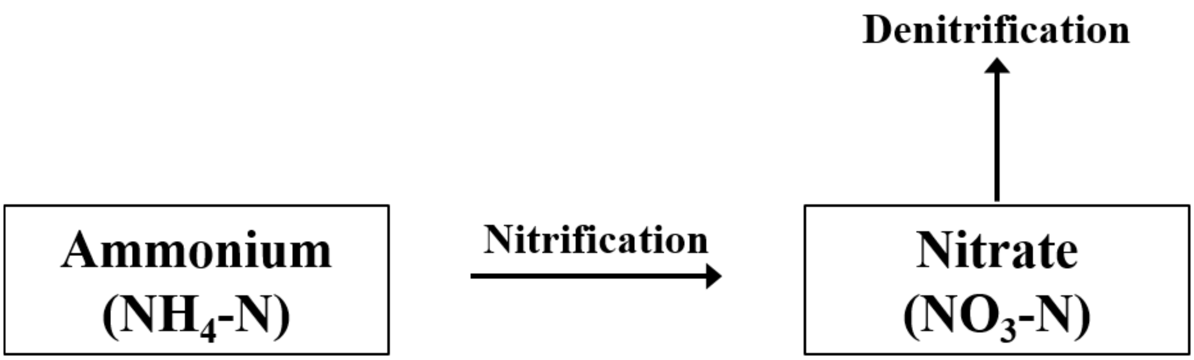

Figure 13. Stream nitrogen conceptual model (MIKE 11 ECO Lab).

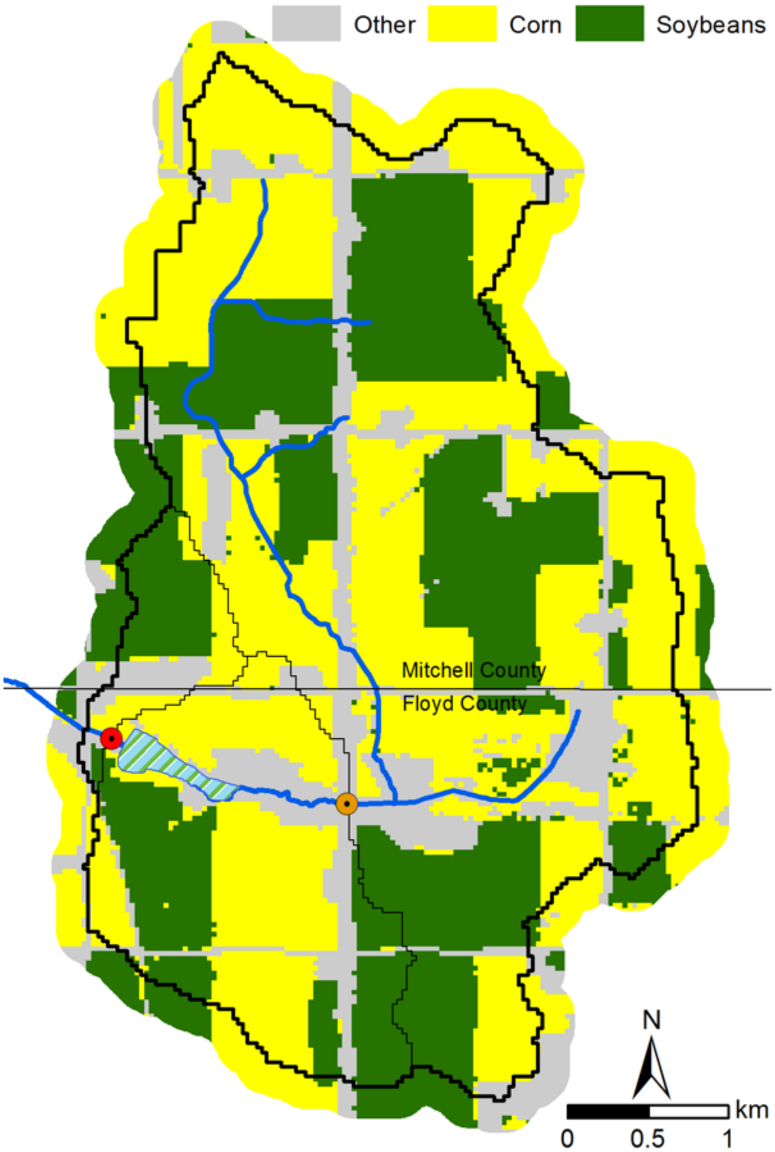

Figure 14. Slough Creek watershed row crop land use (2016) used as the basis for the assumed agricultural management.

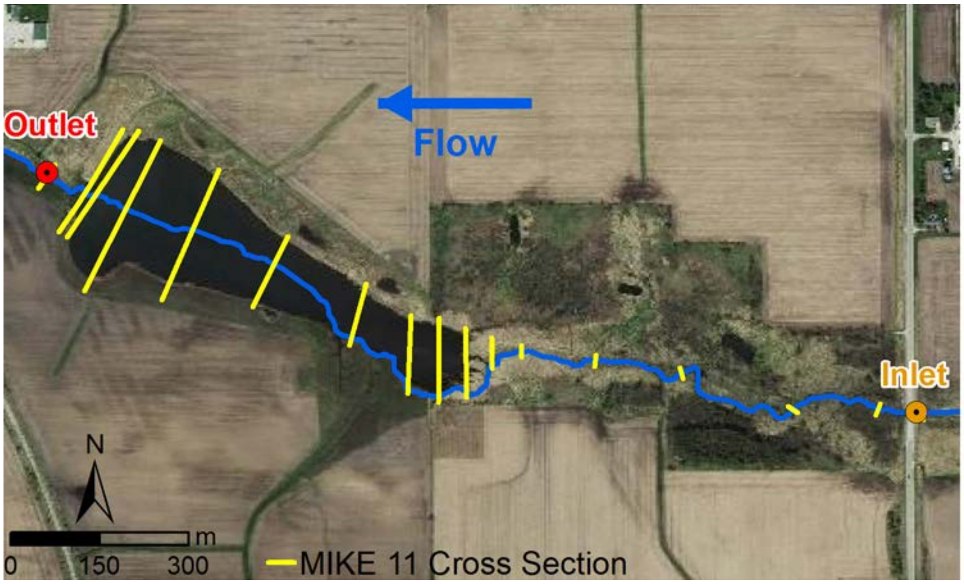

Figure 15. MIKE 11 setup for the wetland simulations.

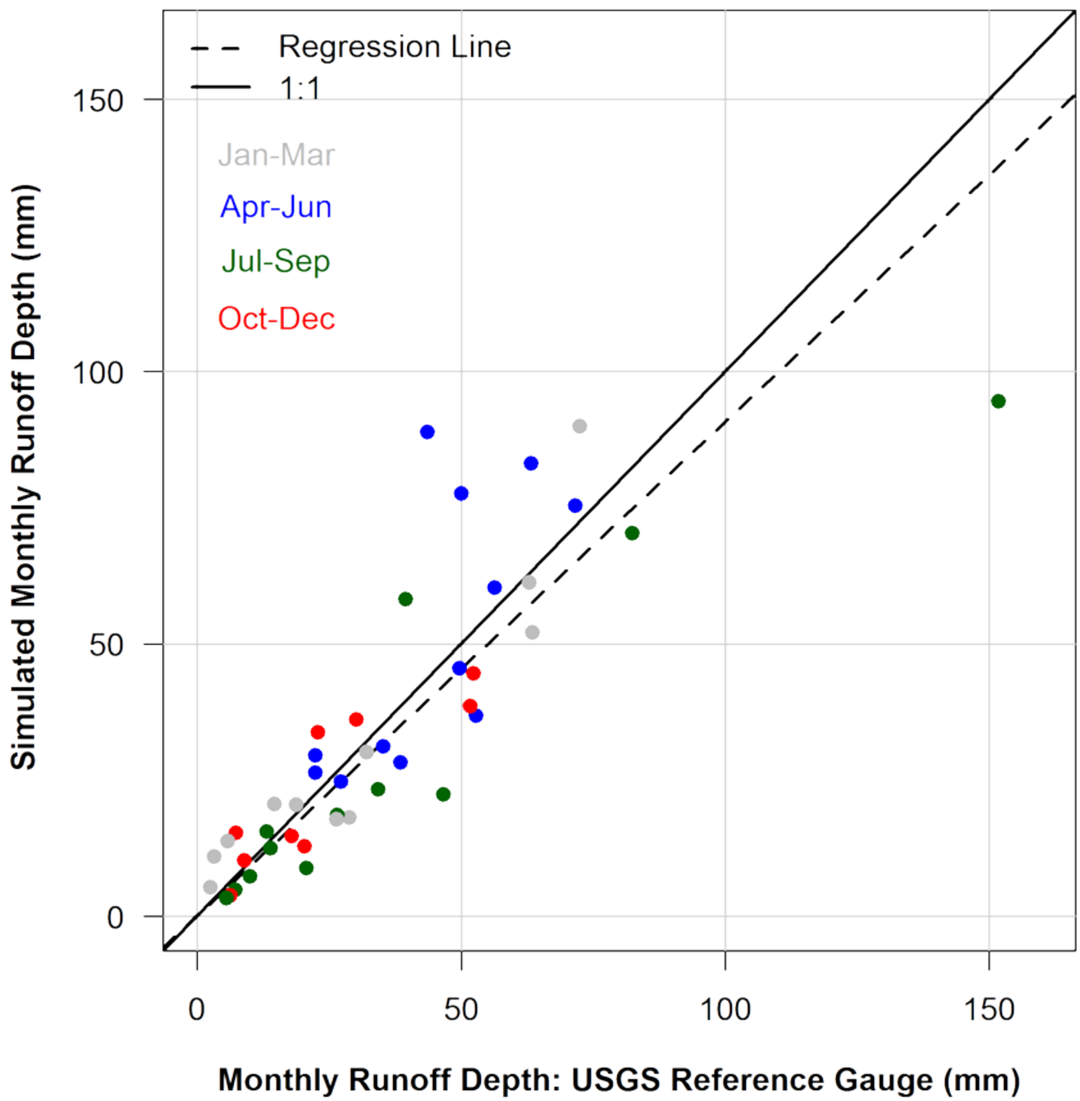

Figure 16. Simulated monthly runoff for Slough Creek compared to Little Cedar (USGS 05458000) observations (Jan 2014-Sep 2017).

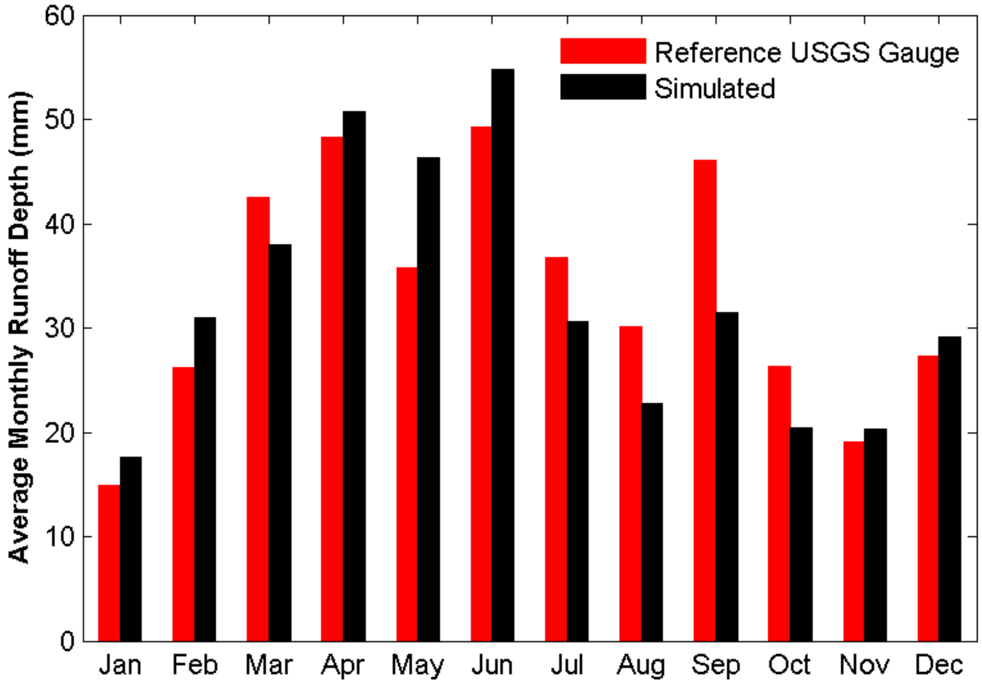

Figure 17. Simulated average monthly runoff for Slough Creek compared to Little Cedar (USGS 05458000) observations (Jan 2014-Sep 2017).

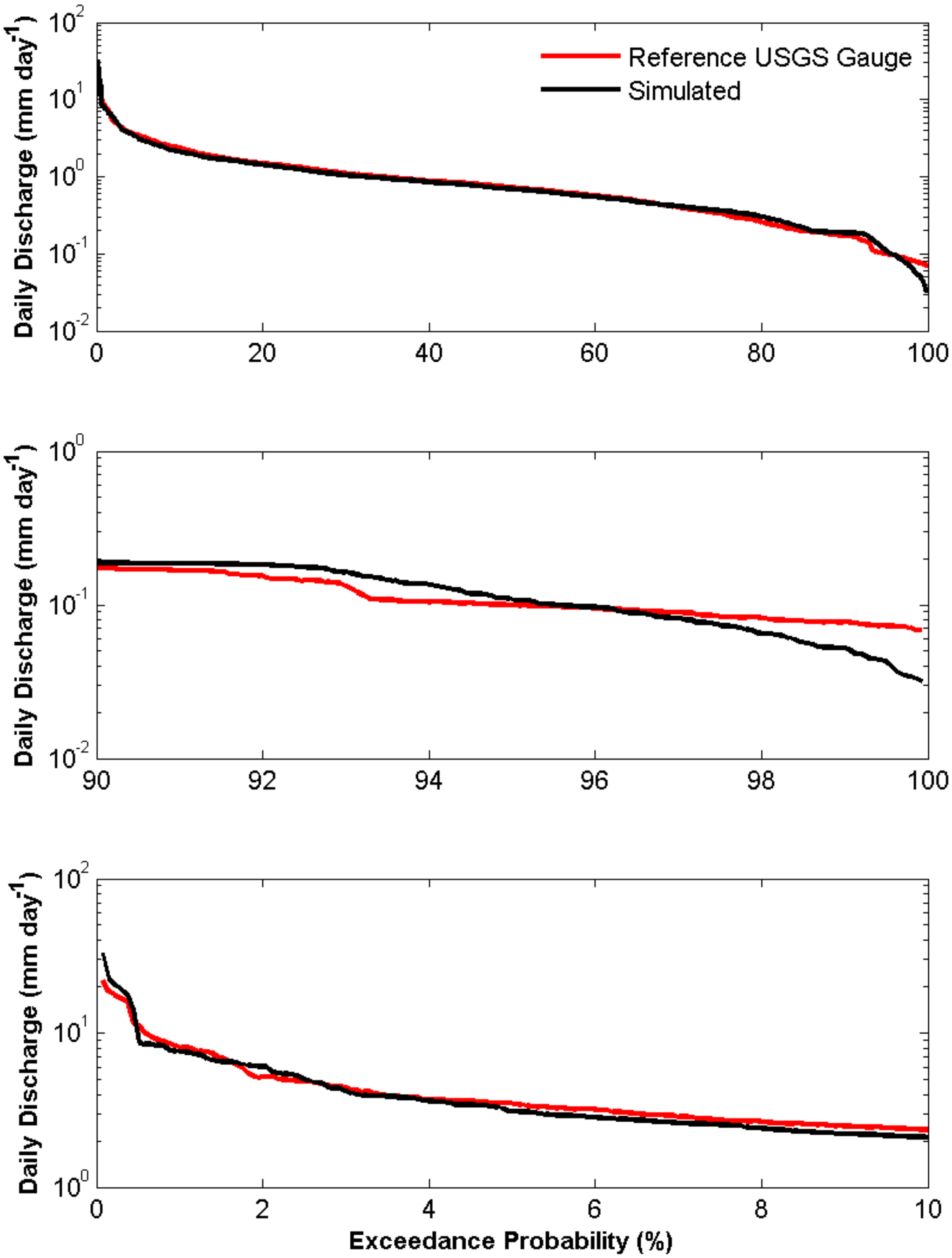

Figure 18. Flow duration curves of mean daily discharge normalized by drainage area for Slough Creek (simulated) and the Little Cedar (observed; USGS 05458000) (Jan 2014-Sep 2017).

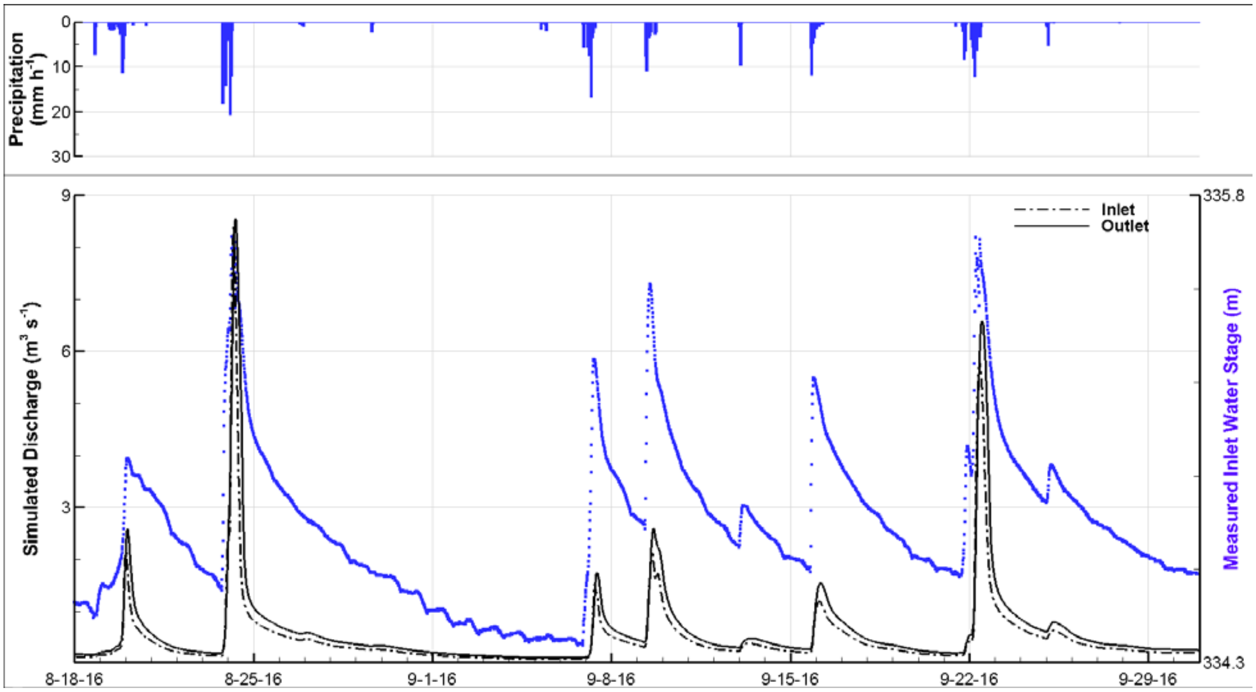

Figure 19. Example of simulated 15-min discharge dynamics at the Slough Creek inlet and outlet during a period of frequent precipitation in Aug/Sep 2016. Inlet stream stage measurements (15-min) are also shown.

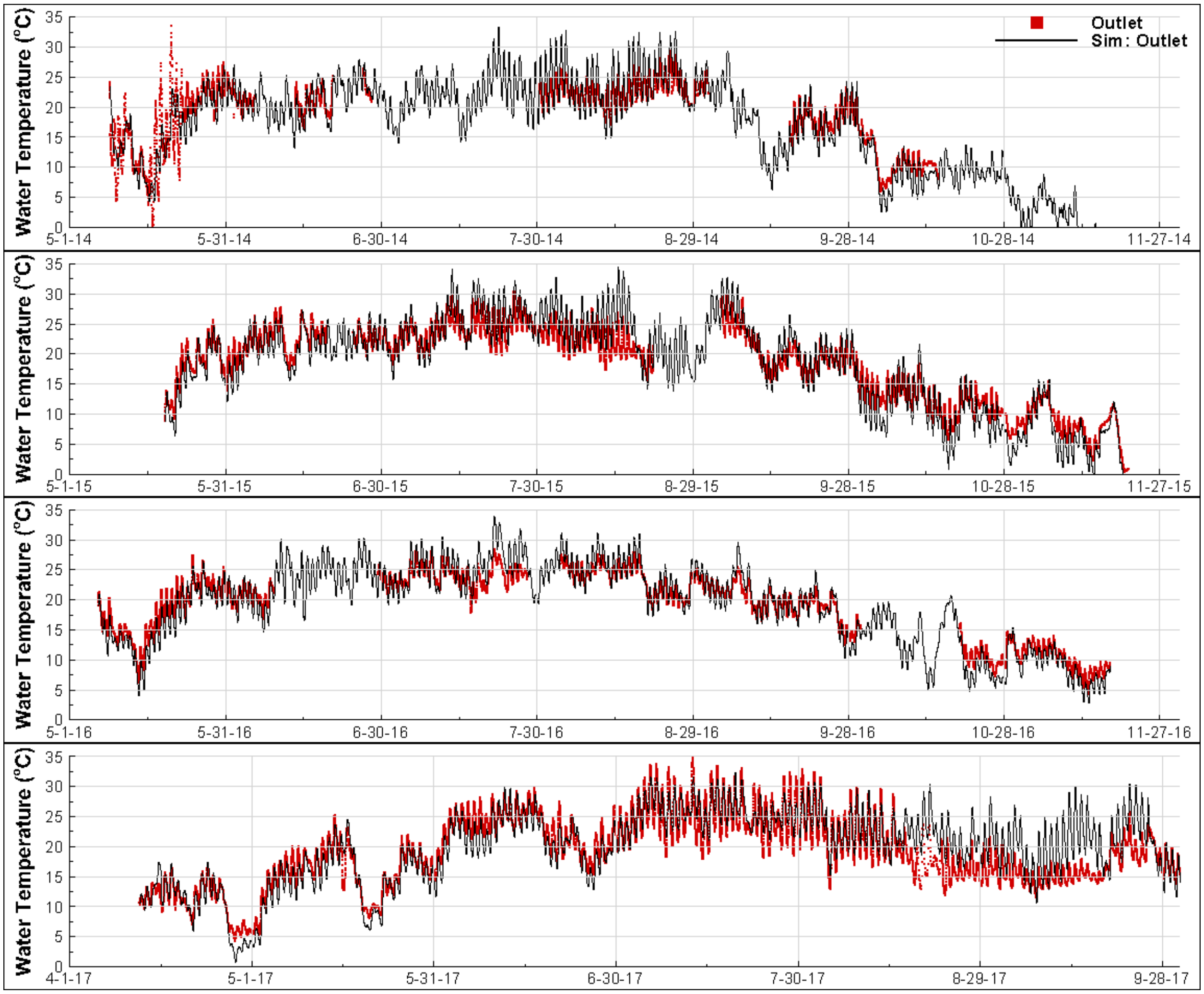

Figure 20. Wetland simulations of outlet water temperature compared to observations (15-min).

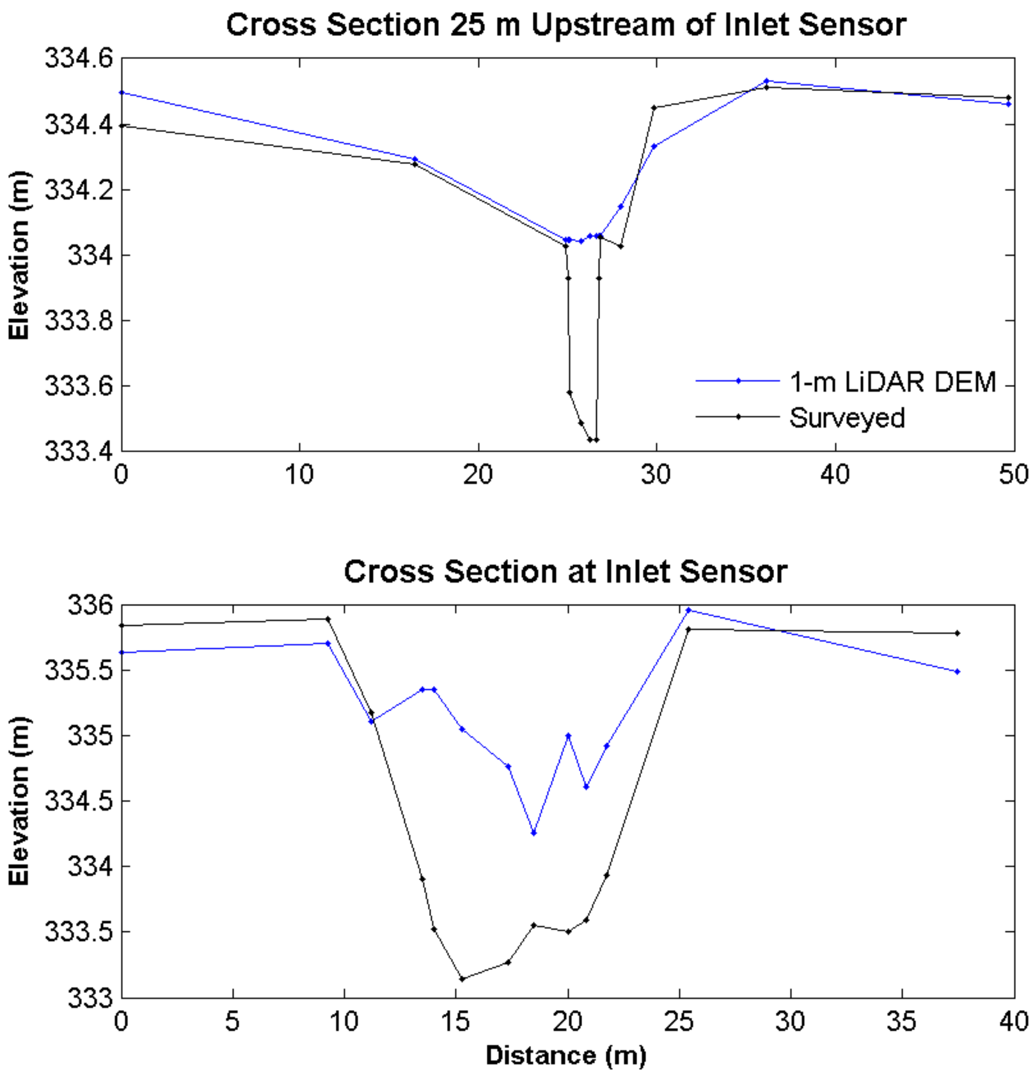

Figure 21. Comparison of surveyed and DEM-extracted cross sections near the Slough Creek wetland inlet water quality sensor.

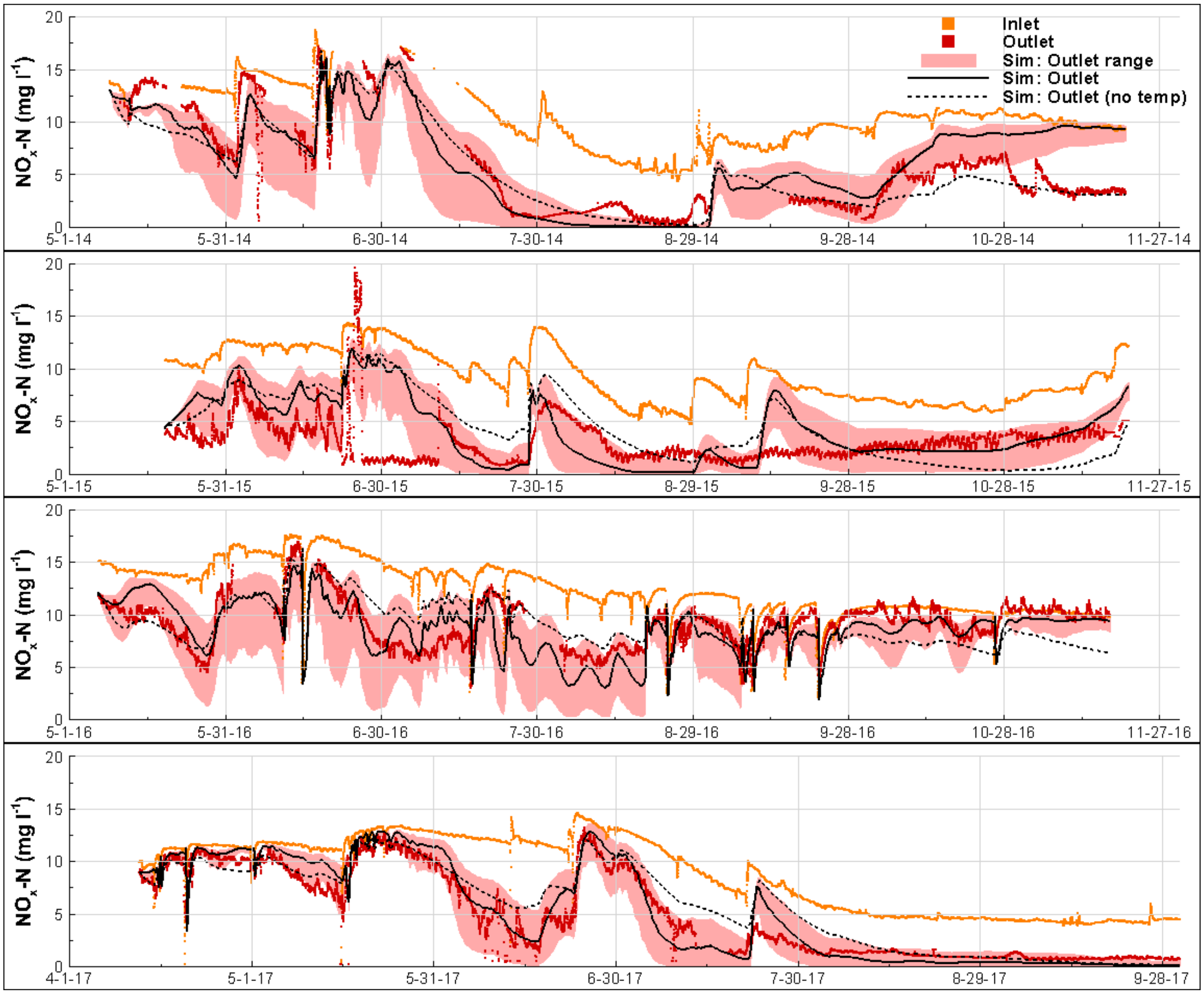

Figure 22. Wetland simulations of outlet NO_3-N concentration compared to observations (15-min). The sensitivity of the first order denitrification rate constant and temperature on the NO_3-N predictions, along with the imposed inlet NO_3-N concentrations, are also shown.

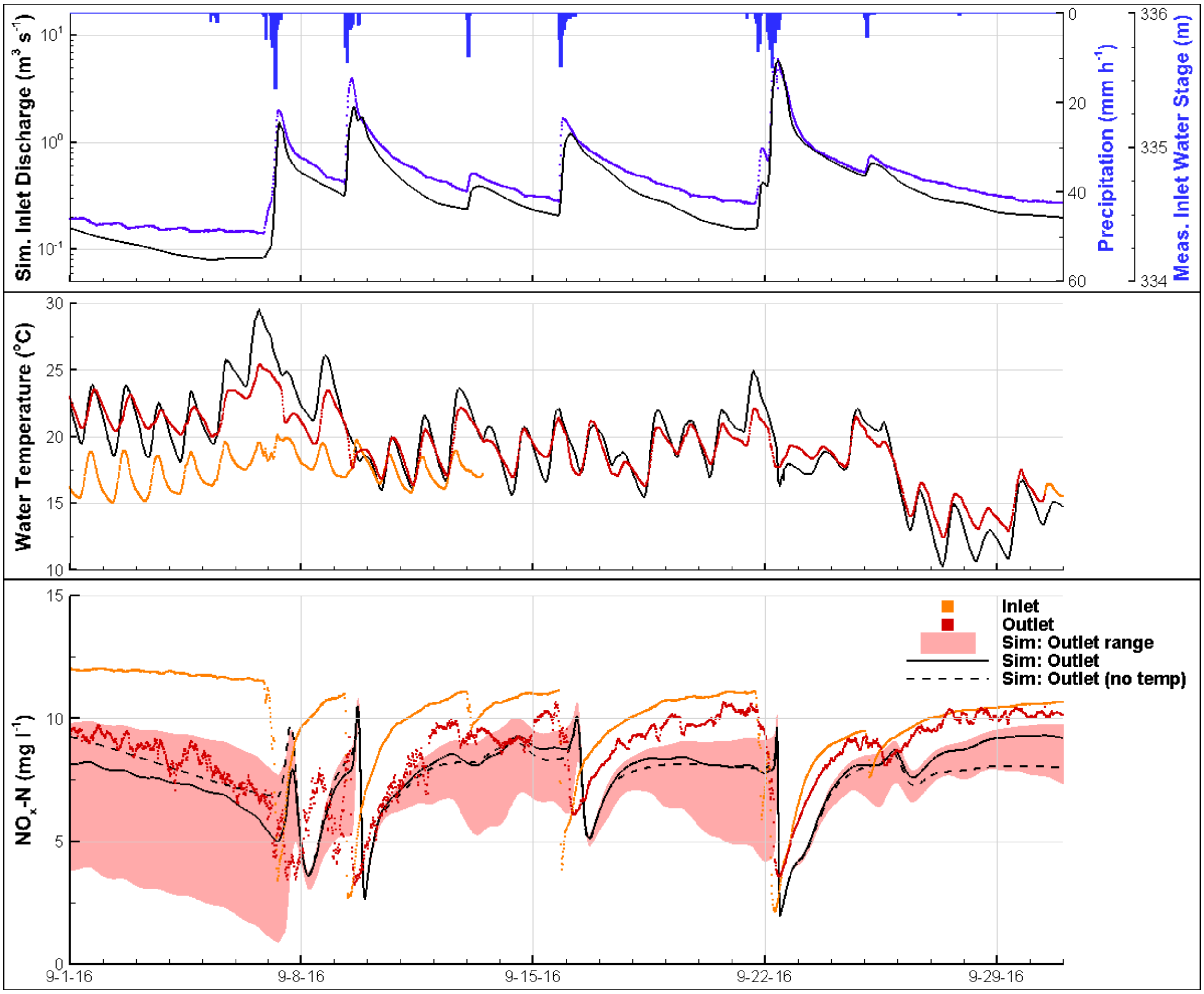

Figure 23. Example of simulated and observed NO3-N and temperature dynamics from the wetland simulations during a period of frequent precipitation in Sep 2016.

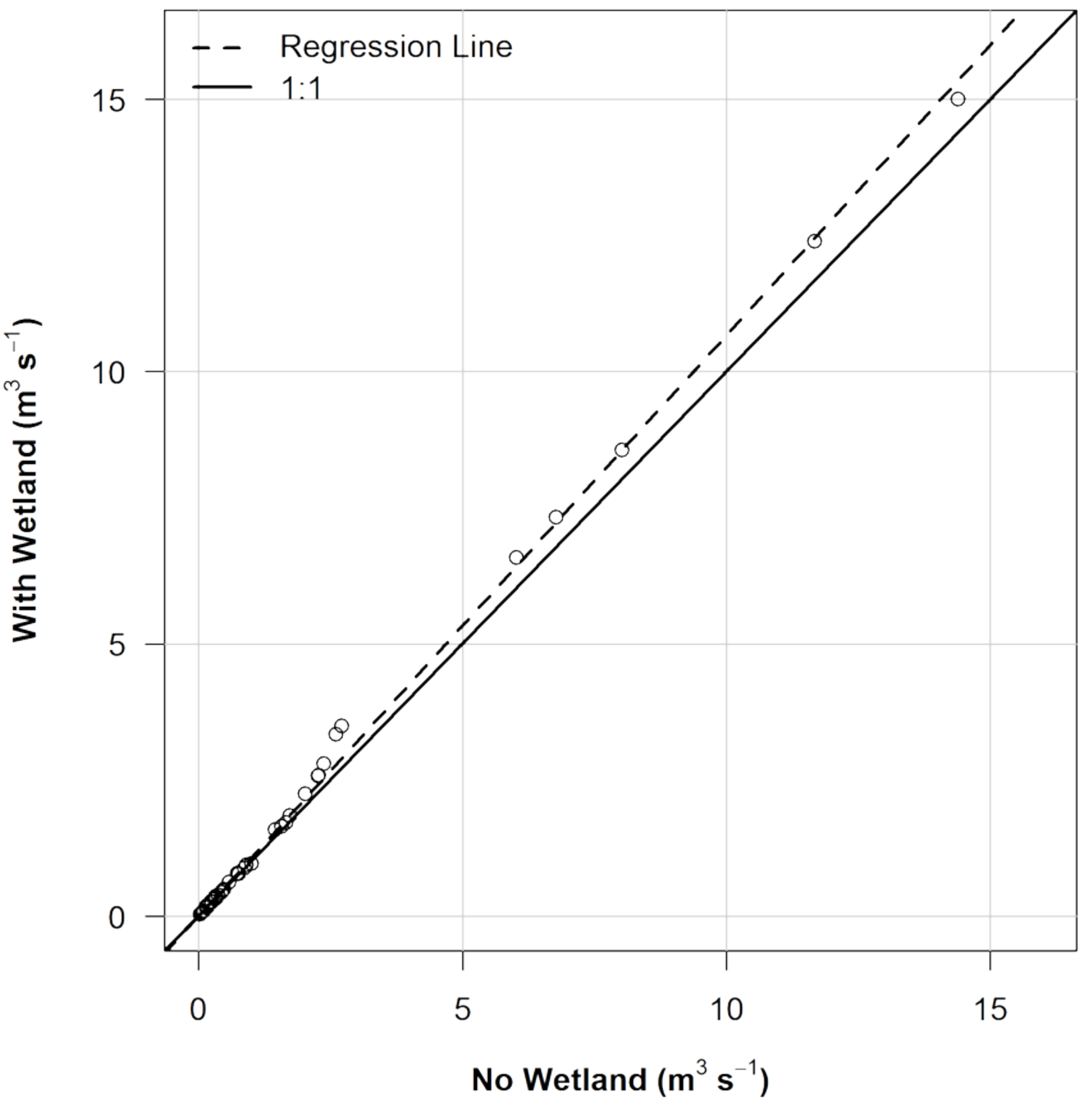

Figure 24. Comparison of simulated peak monthly discharges with and without the Slough Creek wetland included in the MIKE 11 model (Jan 2014-Sep 2017).

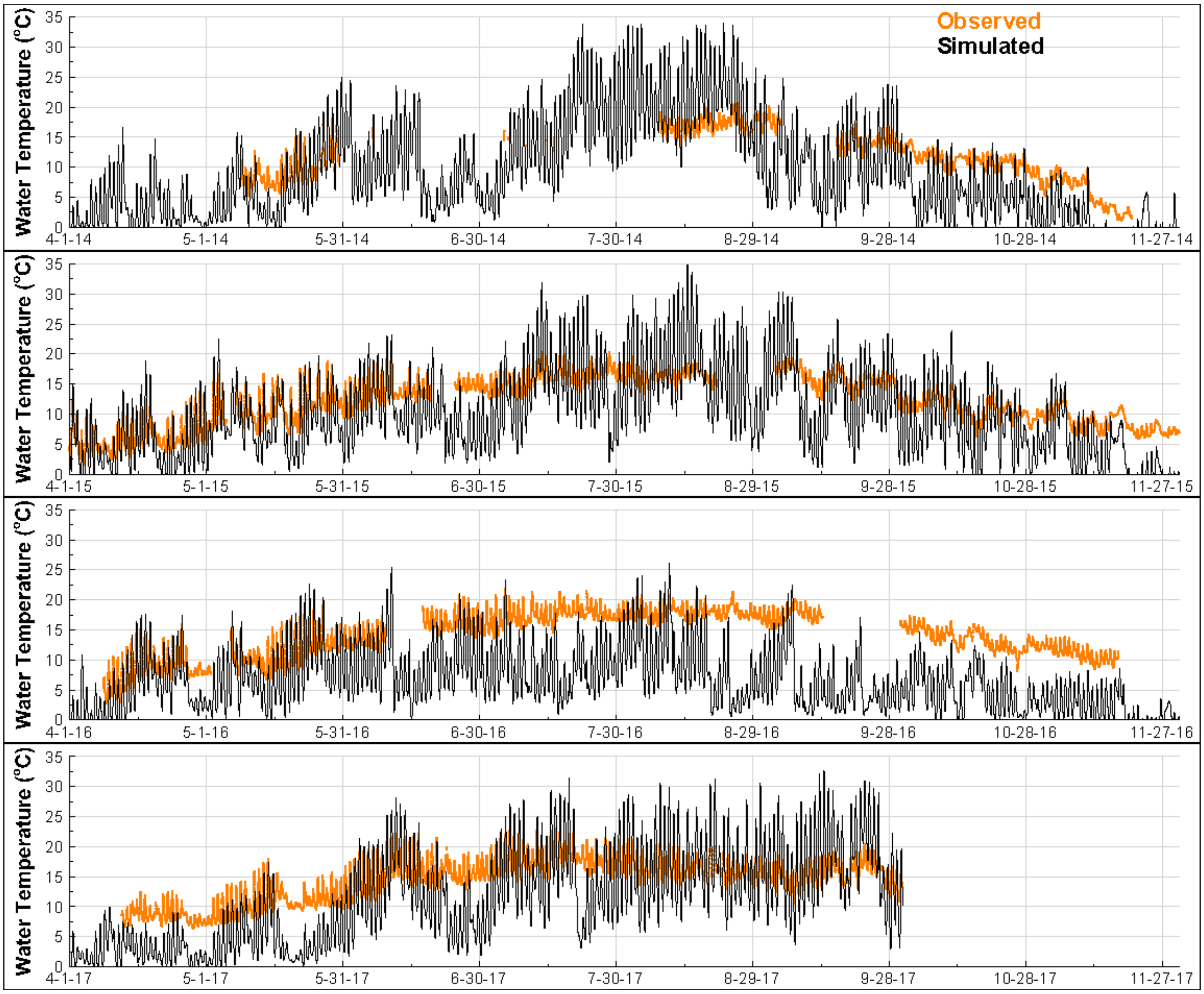

Figure 25. Watershed simulation of inlet water temperature compared to observations (15-min).

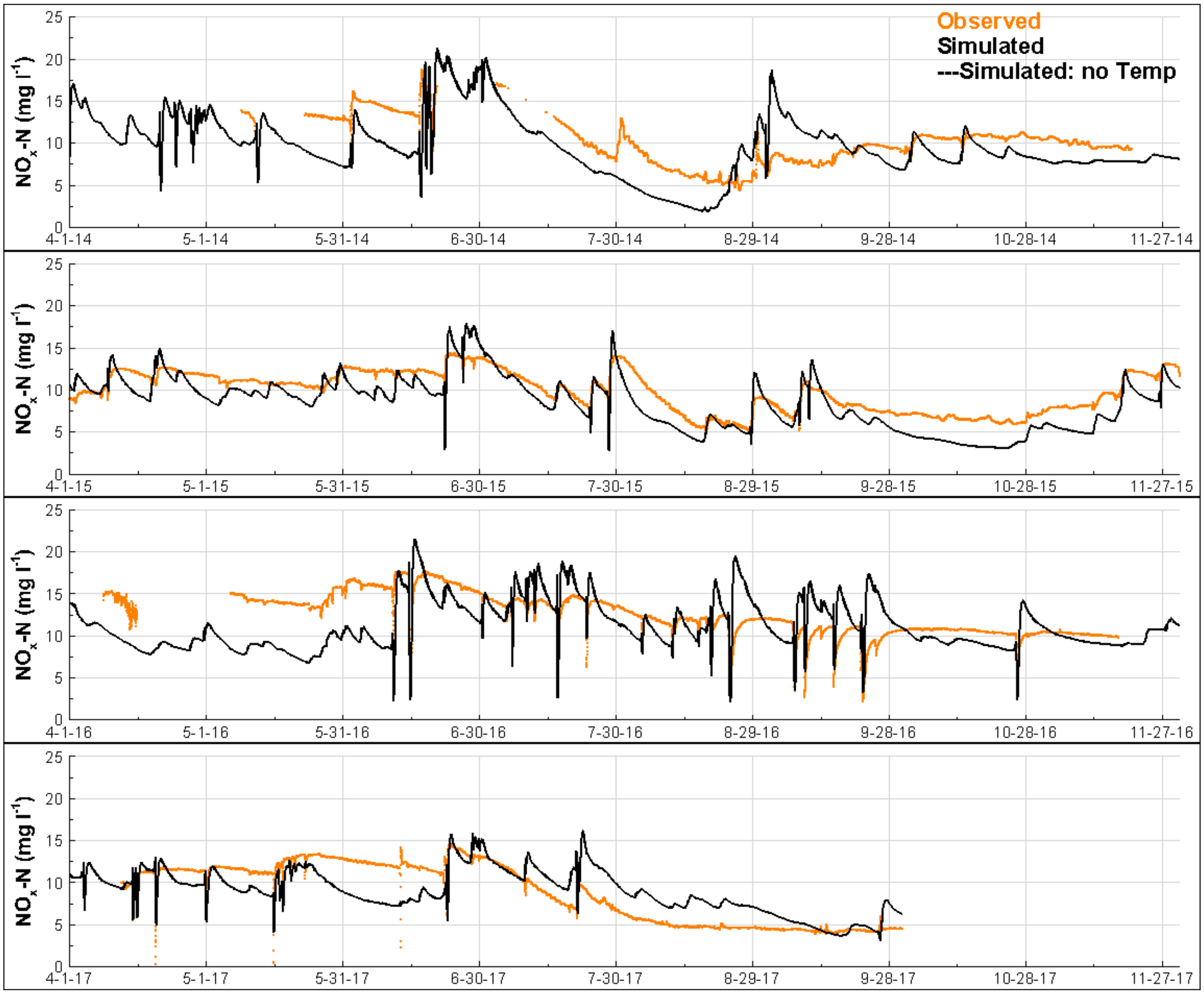

Figure 26. Watershed simulation of inlet NO_3-N concentration compared to observations (15-min).

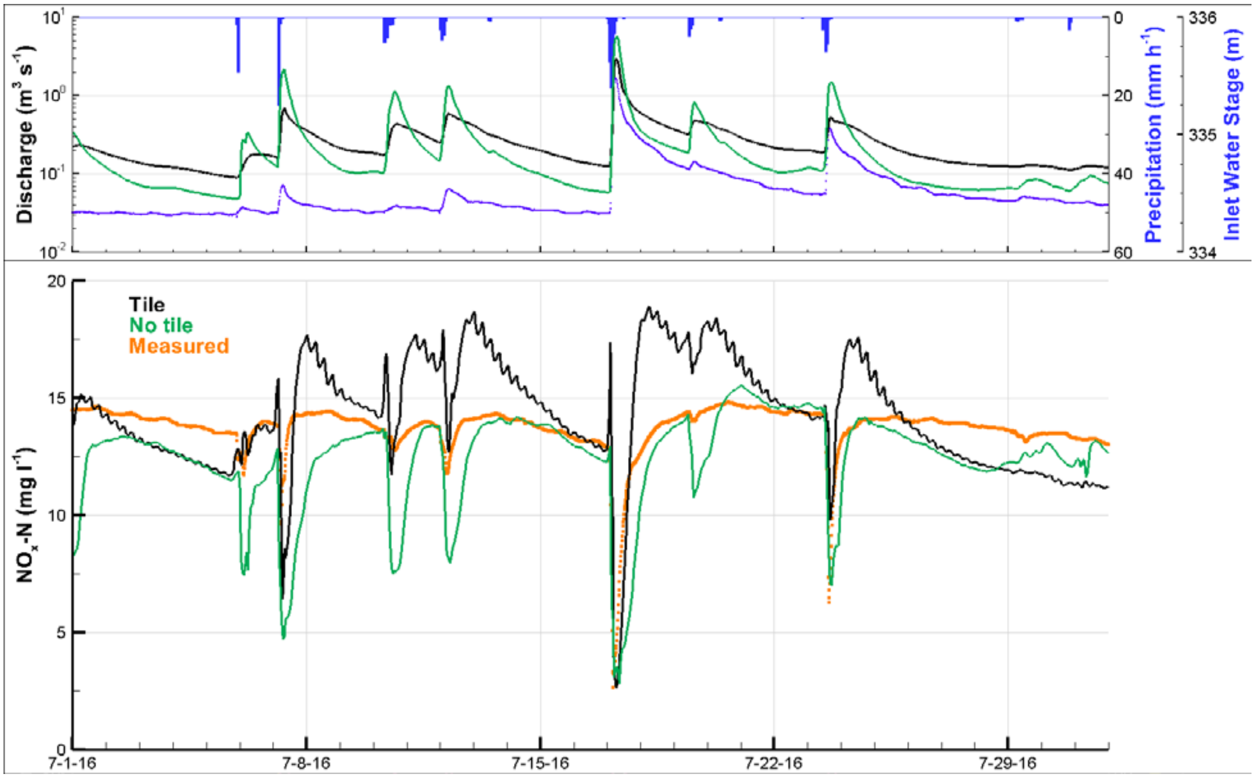

Figure 27. Example showing the influence of the MIKE SHE tile drainage module on simulated NO_3-N dynamics at the Slough Creek wetland inlet sensor.

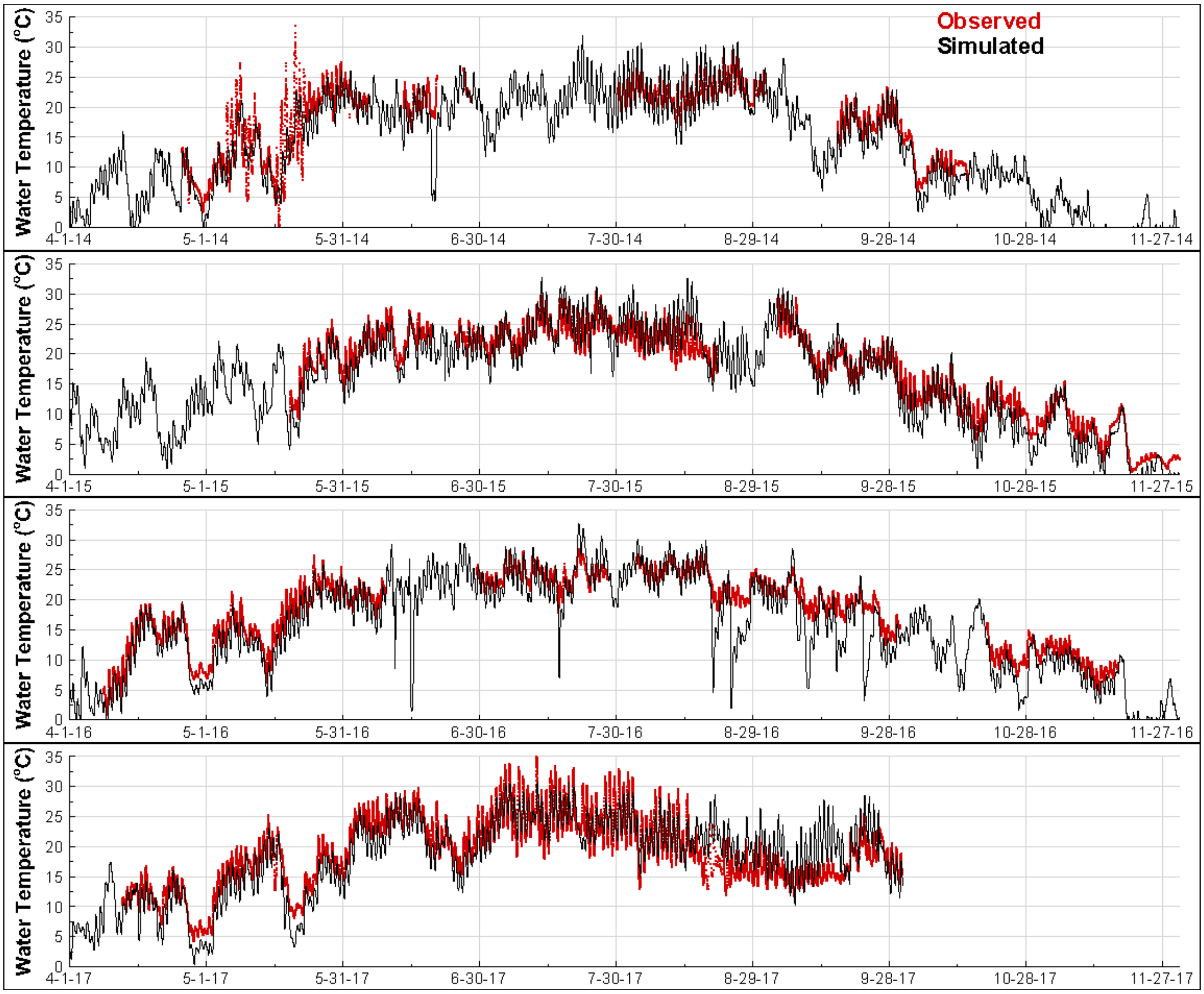

Figure 28. Watershed simulation of outlet water temperature compared to observations (15-min).

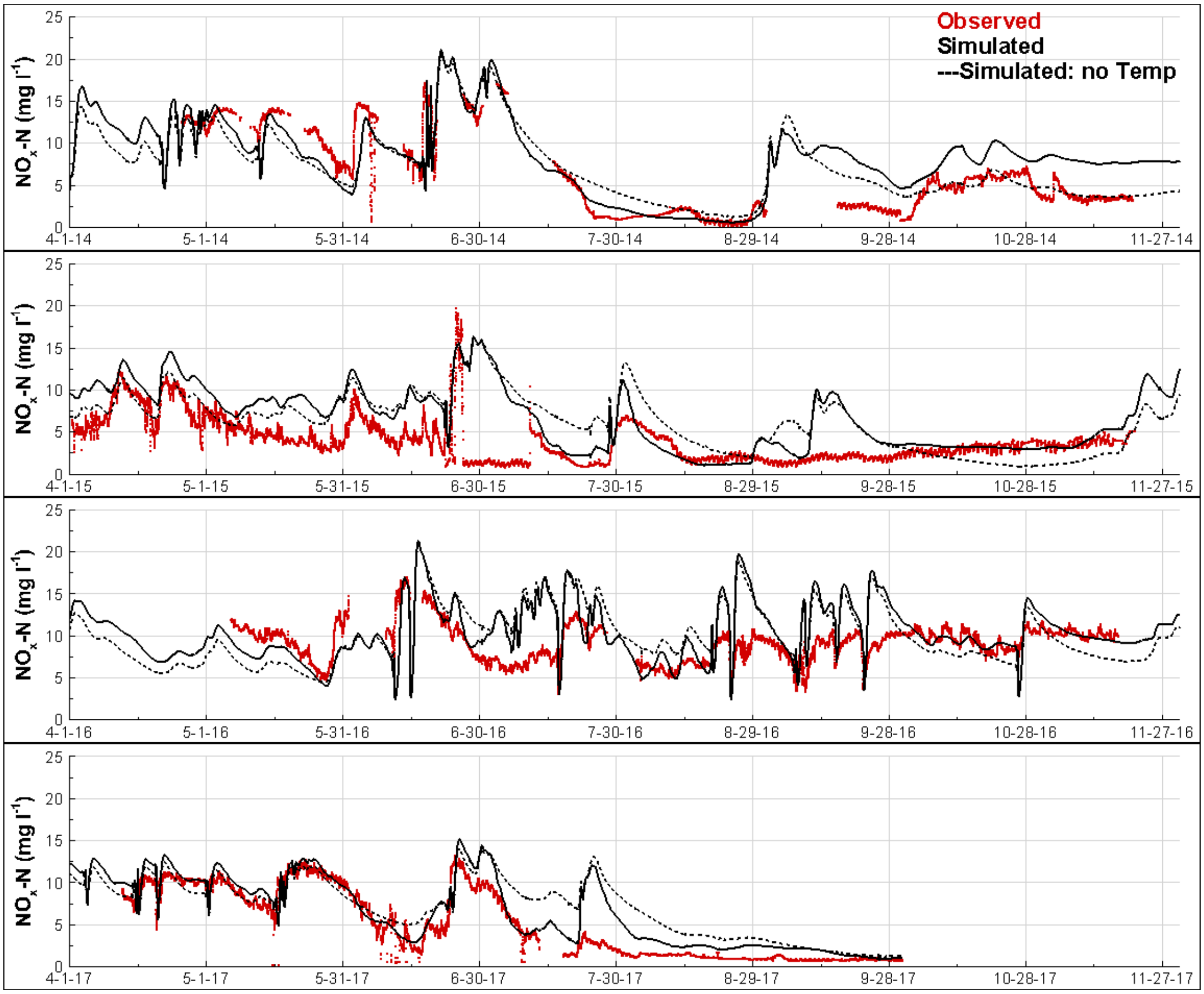

Figure 29. Watershed simulation of outlet NO_3-N concentration compared to observations (15-min).

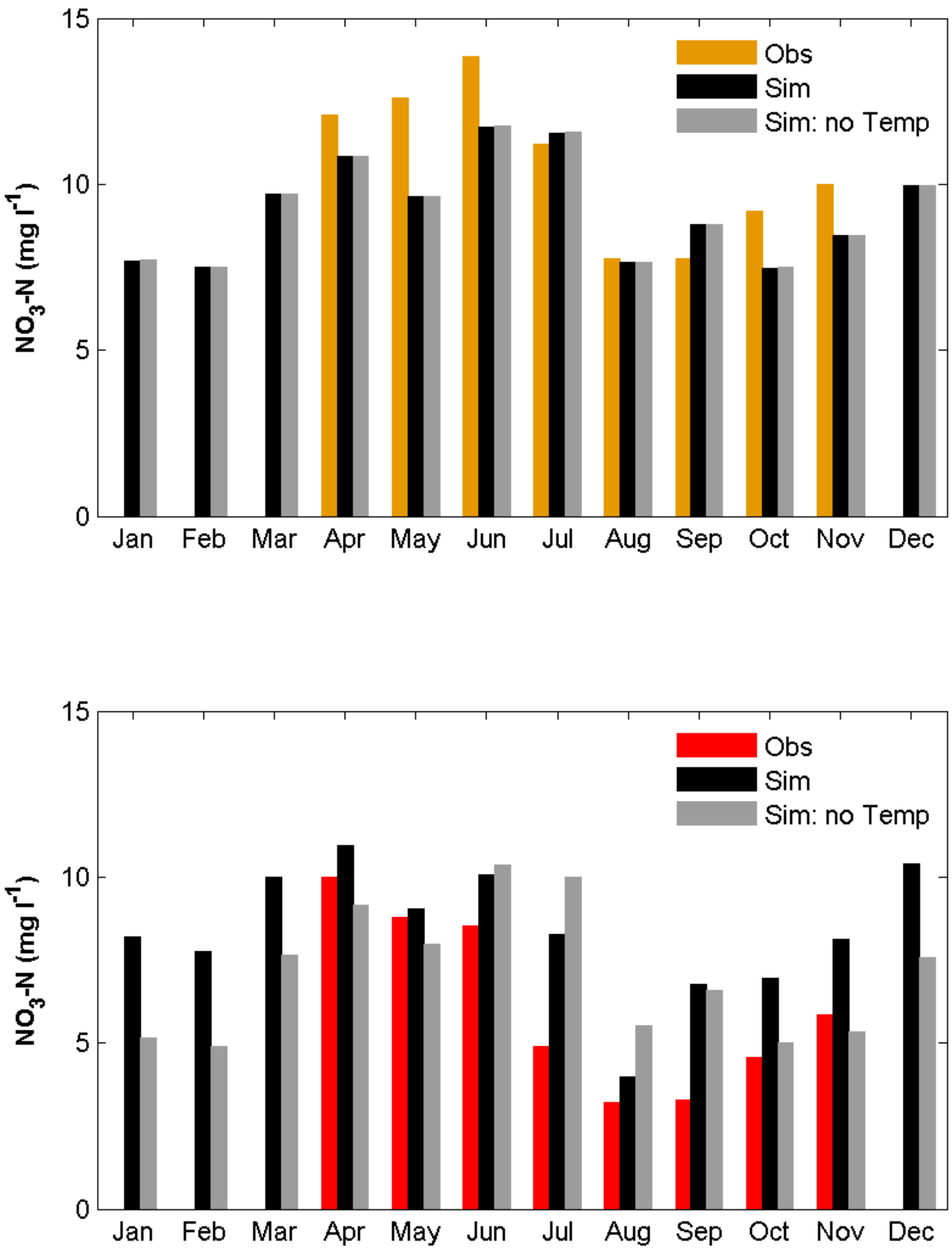

Figure 30. Simulated and observed average monthly NO_3-N concentration at the Slough Creek wetland inlet and outlet sensors (Jan 2014-Sep 2017).

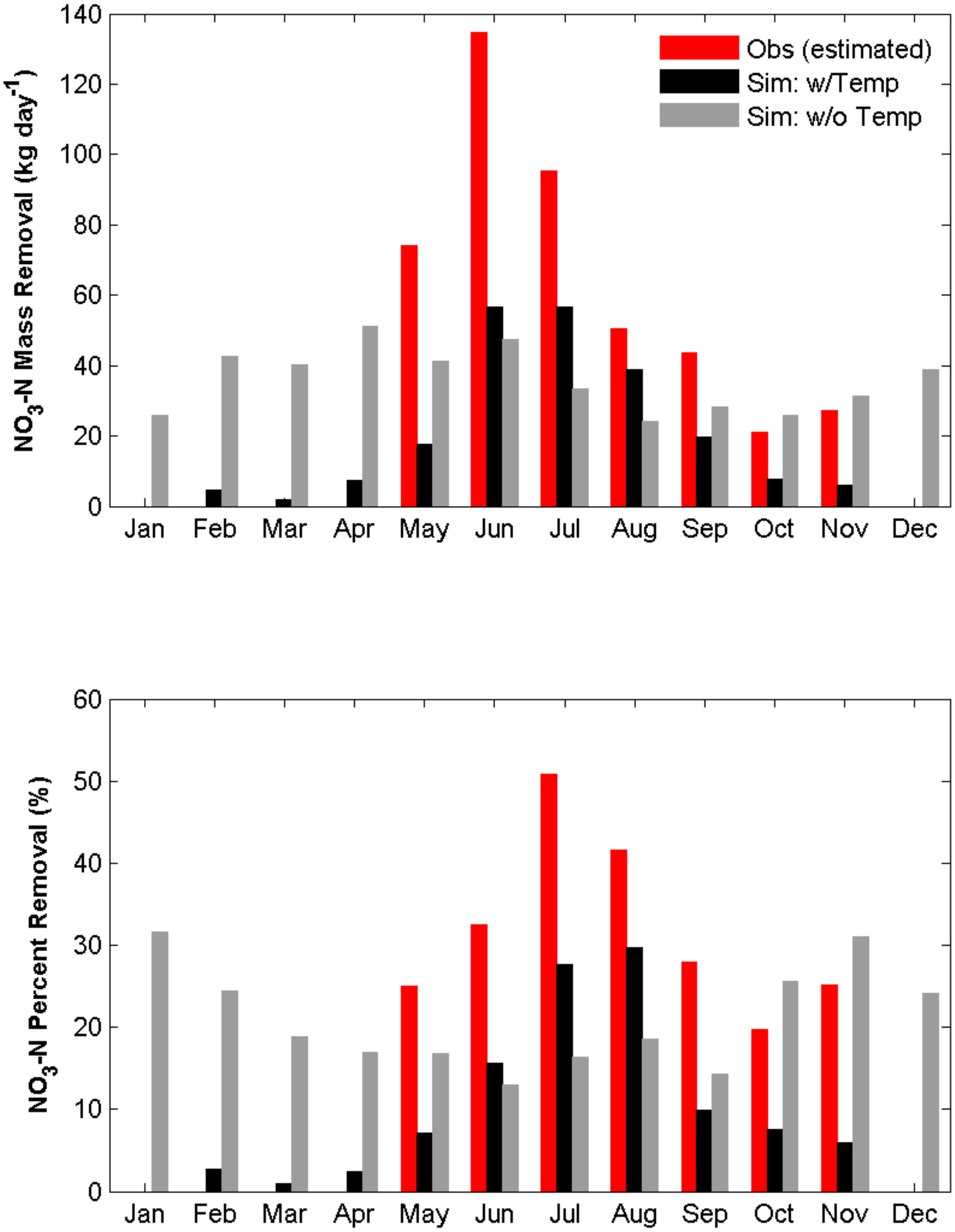

Figure 31. Influence of temperature on simulated average monthly NO_3-N mass and percent removal at the Slough Creek wetland (Jan 2014-Sep 2017).

5. WATERSHED SIMULATIONS TO EVALUATE MULTIPLE WETLANDS FOR FLOW AND NITRATE REDUCTION

Abstract

As part of the Iowa Watersheds Project, $1.5 million were spent on flood mitigation and water quality improvement practices in the agricultural Beaver Creek Watershed (BCW) in north central Iowa, culminating in the implementation of six constructed wetlands by Aug 2016. These six wetlands were evaluated for flow and NO_3-N reduction under variable hydrologic conditions during a 4-yr period using continuous watershed simulations performed with MIKE SHE-MIKE 11 ECO Lab. Nitrogen fate and transport simulations were performed using relatively simple, user-defined soil and stream nitrogen models developed and tested previously on an agricultural research plot and a well monitored constructed wetland 18 km from the BCW. Simulated hydrologic and nitrogen components were evaluated in a systematic manner across different time scales to identify model strengths and weaknesses. During the first 3 years (2014-16) of the study period prior to IWP practice implementation, reliable predictions of stream NO_3-N concentration dynamics were made under average hydrologic conditions, while predictions were less reliable under more extreme (wetter) hydrologic conditions. Approximately 80% of the simulated stream annual NO_3-N load occurred from Apr-Nov (monitoring period of the water quality sensors). During a 13-month period following IWP practice implementation (Sep 2016 – Sep 2017), simulated average peak monthly discharge reductions ranged from 3-43% at the wetlands that dissipated to near 0% at the BCW outlet. Estimated wetland NO_3-N load reductions ranged from 7-25% at the project locations to 2% at the BCW outlet. The lower than expected flow and NO_3-N reduction estimates are attributed to errors in simulated hydrology, uncertainties in the assumed agricultural management, and limitations of the nitrogen model setup. This study is one of the first of its kind to assess agricultural conservation practices (wetlands) for flow and NO_3-N

reduction in a watershed context using MIKE SHE-MIKE 11 ECO Lab. While the modeling framework shows promise in several regards, revisions to both the hydrologic and water quality model setups are needed to improve the reliability of the discharge and NO_3-N predictions.

Introduction

The purpose of this chapter was to evaluate six recently constructed wetlands in a 45 km2 agricultural watershed in north central Iowa for flow and NO_3-N reduction with MIKE SHE-MIKE 11 ECO Lab. As part of the recently completed Iowa Watersheds Project (IWP; 2012-16), a federally funded initiative to reduce the severity of flooding in selected Iowa watersheds (Weber et al., 2017), six dual purpose flood mitigation and nutrient removal wetlands totaling $1.5 million were built during the summer of 2015 and 2016 in the Beaver Creek watershed (BCW), a small agricultural catchment in the Cedar River Basin. The flood reduction benefits of these six structures, along with three previously constructed wetlands, were quantified previously with the physically-based hydrologic model HydroGeoSphere (Thomas et al., 2016). Hydrologic simulations ran with variable synthetic design storm rainfall totals, antecedent moisture conditions, and initial project storage conditions indicated that the nine flood detention structures could reduce peak discharges at the BCW outlet by 3-17%. The current study seeks to complement and expand upon these previous efforts to estimate the early flow and NO_3-N reductions provided by the six IWP wetlands using continuous simulations performed with another physically-based hydrologic and water quality model, MIKE SHE-MIKE 11 ECO Lab.

Materials and Methods

Modeling Methodology

In a similar manner to the integrated watershed simulations performed previously for the Slough Creek wetland watershed, continuous watershed hydrologic and nitrogen fate and transport simulations were performed for the BCW with MIKE SHE-MIKE 11 ECO Lab to estimate the flow and NO_3-N load reduction benefits offered by the six IWP constructed wetlands. The same hydrologic and water quality modeling approach as used for the Slough Creek CREP wetland watershed (Chapter 4) was applied to the BCW. Given the close proximity of both watersheds (18 km apart), spatial datasets and model inputs used in the BCW simulation were similar to those used in the Slough Creek study. Flow and NO_3-N reductions were estimated at each wetland and further downstream locations. Simulated stream NO_3-N were compared to observations from two continuous high-frequency (15-min) water quality sensors located on the Beaver Creek main stem in the upper half of the watershed and near the outlet during the warm season (Apr-Nov) from 2014-17. The first three study years were used to evaluate the baseline MIKE SHE-MIKE 11 ECO Lab model prior to IWP wetland implementation and the last study year (1 Sep 2016 – 30 Sep 2017) was used to evaluate the potential impact of the six IWP constructed wetlands.

Study Area

The Beaver Creek Watershed (BCW) is a 45 km^2 agricultural HUC-12 watershed (070802010901) in north central Iowa (Figure 32). Average annual (1981-2010) precipitation is 902 mm, with an average of 73% falling as rain between Apr and Sep (PRISM, 2018). The BCW is located in the Iowan Surface landform region, an erosional surface of low relief and extensive tile drainage (Prior, 1991; Jones et al., 2018). Total watershed relief is approximately 60 m, the average basin slope is 3.7%, and approximately 60% of the area is estimated to require tile drainage based on soils and slope information. Corn (46%) and soybean (25%) production comprise 71% of the land use with grass/pasture (15%), deciduous forest (8%), and developed areas (8%) defining the majority of the remaining area. Loam (68%), clay loam (14%), and silty clay loam (16%) are the primary surficial (top two meters) soils in the upland areas, near the stream network, and in the transitional zones, respectively. The surficial geology of the region is characterized by a thin layer of glacial till overlying permeable limestone bedrock (Griffith et al. 1994). Beaver Creek discharges into the Little Cedar River near Bassett, Iowa.

The BCW contains nine documented constructed wetlands designed for flood mitigation and/or nutrient reduction purposes (Figure 33; Table 16). All nine detention structures are located in the upper (northern) half of the catchment where agricultural intensity is highest and relief is lowest. The projects were designed with a normal (permanent) pool ranging from 0.7-3.2 ha to hold some water all the time. When water levels exceed the normal pool elevation, water is discharged through some type of weir or orifice structure until the emergency spillway is reached, located approximately one meter above the normal pool. At the emergency spillway, discharge over a weir occurs at a much greater rate to avoid overtopping or compromising the structure. The flood storage, defined as the pool storage between the normal pool and emergency spillway elevations, temporarily retains runoff during storm events. The available flood storage paired with the normal pool hydraulics determine the degree of flow attenuation achieved during high runoff periods. The normal pool and flood storage volumes listed in Table 16 are expressed as equivalent uniform depths by dividing by the upstream drainage area; in this context, the flood storage conceptually describes the depth of runoff from the contributing area that can be temporarily retained during storm events.

The six constructed wetlands funded through the Iowa Watersheds Project (IWP) were designed to achieve dual flood mitigation and nutrient reduction benefits and were completed between the fall of 2015 and the summer of 2016. The projects drain a total of 534 ha (12% of the BCW area) comprised of 85% row crops. The wetland hydraulics for sites 2-6 are similar.

Normal pool hydraulics are governed by a combination weir-orifice riser structure ranging from 0.8-2.3 m in diameter while discharge over the emergency spillway is governed by a 12-15 m weir. Site 1 is similar but contains a 26 m weir at the normal pool elevation. Normal pool storages range from 5-11 mm with approximately 2-3 times as much available flood storage. In excess of $1.5 million were spent for flood reduction and water quality improvement in the BCW through the IWP, with individual projects ranging from approximately $180,000 to $400,000 to design and complete.

Three other detention structures (Sites A-C) were built several years earlier through the Iowa Conservation Reserve Enhancement Program (CREP) and private funds. The two CREP wetlands were designed for water quality improvement in a similar manner to the Slough Creek CREP wetland; wetland hydraulics are governed by 8-13 m weirs at both the normal pool and emergency spillway elevations, and, therefore, are expected to have a small flood reduction benefit. Site C has similar hydraulics to the IWP structures and is the smallest of all nine projects in terms of both pool area and drainage area. Sites A-C drain a similar amount and type of land (528 ha; 81% row crops) as the IWP projects. In total, the nine detention projects retain and process runoff from 24% (1062 ha) of the BCW.

As part of the IWP, three stream gauges were installed in the spring of 2014 to track flow and water quality conditions along the Beaver Creek main stem (Figure 33). Stream water levels (no discharge) are monitored at each gauge while the uppermost (Beaver03) and lowermost (Beaver01) sites also track NO_3-N concentration and other water quality parameters (Jones et al., 2018). Stream levels are monitored continuously throughout the year while water quality is monitored continuously during the warm season (Apr-Nov); sensor measurements are reported every 15-60 min.

MIKE SHE-MIKE 11 Model Setup

The BCW MIKE SHE-MIKE 11 model was developed in a similar manner to the Slough Creek model with the spatial datasets shown in Figure 32. The MIKE SHE model calculates surface and subsurface water fluxes using 50 m grid cells, corresponding to 18,600 calculation nodes in each surface and subsurface layer. The model domain extends to a depth of 12 m. Infiltration and soil water retention in the unsaturated zone is described by the Van Genuchten method using 33 computational layers ranging in thickness from 5 cm near the surface to 1 m at depths below 8 m. Groundwater dynamics in the saturated zone are calculated using six computational layers. Because the top saturated zone layer is meant to encompass the typical range of water table fluctuations, this layer is variable in thickness from 1 m in persistently saturated areas (e.g. near the stream network) to 7 m in drier, upland areas; the next four computational layers have a uniform thickness of 1 m, and the final bottom computational layer is also variably defined to extend to a uniform depth of 12 m below the surface. The model was forced with hourly Stage IV radar rainfall estimates (Figure 32) and daily estimates of potential evapotranspiration (IEM, 2016).

The MIKE 11 model contains 12 branches totaling 41 stream km, of which the Beaver Creek main stem accounts for over half the distance. Cross sections extracted from a 2 m hydro-enforced DEM are spaced approximately every 400 m along the main stem and 200-300 m along the smaller branches. In the vicinity of the projects, the cross section spacing was reduced to approximately 50 m. Figure 34 shows the nine wetlands and the MIKE 11 cross sections used to describe wetland flow and water levels. Site A also includes the MIKE SHE 50 m mesh to give a sense for the scale that surface and subsurface water fluxes are calculated at by MIKE SHE. At sites 2-6 where combined weir-orifice riser structures were installed, wetland hydraulics were estimated using tabulated stage-discharge data from the engineering design plans; at the remaining sites, wetland hydraulics were estimated using broad-crested weir and culvert (pipe or orifice flow) structure options available with MIKE 11. As with the Slough Creek MIKE 11 model, a small (0.001 $m^3 s^{-1}$) constant inflow was imposed at the upstream end of each branch to maintain numerical stability of the MIKE 11 hydrodynamic and water quality simulations.

Watershed Simulations

Continuous, integrated watershed simulations were conducted to evaluate the six IWP constructed wetlands for flow and NO_3-N reduction under variable hydrologic conditions. Hydrologic and water quality simulations were performed over an 8-yr period from 2010-17, with the first 4 years serving as a warm up period and the last four years (Jan 2014 – Sep 2017) used for model evaluation corresponding to when both measured stream NO_3-N concentration and Stage IV radar rainfall estimates were available. Since the three non-IWP projects (Sites A-C) were built prior to water quality sensor deployment and because the primary focus of this study was to evaluate the flow and NO_3-N impacts of the six IWP wetlands, projects A-C were included in the baseline MIKE SHE-MIKE 11 model. The baseline simulation was used to evaluate the water quality performance of the MIKE SHE-MIKE 11 ECO Lab model prior to IWP project implementation (Jan 2014 – Aug 2016) and used as a reference to assess the impact of the IWP wetlands. The IWP simulations were initiated at the end of June 2016 using flow and water quality conditions from the baseline simulation as initial conditions and model performance was evaluated from 1 Sep 2016 to 30 Sep 2017. Simulated hydrology and water quality were evaluated in a similar manner to the Slough Creek study. Simulated water balance components were compared to literature estimates and simulated discharge was compared to the Little Cedar USGS discharge gauge for reference. For water quality, the simulated annual nitrogen balance was compared to literature estimates and predicted stream NO_3-N concentrations were compared to measurements at the two water quality sensors. For the IWP simulation, flow and NO_3-N reductions were evaluated locally (at the wetlands) and at downstream locations to assess the watershed impact of these six detention structures.

Results and Discussion

Simulated Hydrology

The simulated hydrology for the BCW was assessed in a systematic manner from coarser (annual) to finer (hourly) time scales to ensure water balance components were reasonable followed by discharge dynamics. For the evaluation of simulated water balance components and discharge, the baseline simulation was used as the six IWP wetlands are not expected to significantly alter water balance components or the flow regime at the BCW outlet. Similar to the Slough Creek results, the simulated annual water balance for the BCW is reasonable but baseflow is likely overestimated (Table 17). The 3-yr (2014-16) average annual precipitation was 1056 mm, 17% above the 30-yr (1981-2010) average (902 mm; PRISM, 2018). While simulated annual discharge and evapotranspiration were reasonable, the simulated baseflow index (3-yr avg: 85%) is overestimated by 10-20% which results from an overestimated contribution of simulated tile drainage (3-yr avg: 65%) to streamflow, respectively (Thomas, 2015; Arenas et al., 2017).

Simulated monthly runoff from Jan 2014-Sep 2017 along the Beaver Creek main stem at the two water quality sensors, Beaver03 (top panel) and Beaver01 (bottom panel), were similar to the Little Cedar (Figure 35). Besides lower than expected runoff at both locations in Sep 2016 (far right green dots) when precipitation was about three times the long term average for both Beaver Creek and the Little Cedar, simulated monthly runoff totals are fairly evenly distributed about the 1:1 line. Greater than expected simulated monthly runoff from Apr-Jun sometimes occurred, This may be attributed to earlier row crop planting than assumed in the model (mid-May) which would have allowed for more evapotranspiration losses earlier in the growing season.

Simulated average monthly runoff at Beaver03 and Beaver01 was similar to the Little Cedar (Figure 36). Simulated runoff at both locations was overestimated in Jan and underestimated in Mar, indicative cold season dynamics and snowmelt dynamics were poorly simulated. The lower than expected Sep runoff is attributed to the particularly wet Sep 2016 (Figure 35); when Sep 2016 was excluded, the average simulated (13 mm) and observed (11 mm) monthly runoff depths for Sep were similar. Simulated runoff from Apr-Jul accounted for approximately 45% of annual runoff at both sites compared to 42% for the Little Cedar. Overall, simulated runoff at Beaver03 (398 mm) and Beaver01 (385 mm) were 1% lower and 4% lower, respectively, than the Little Cedar (402 mm).

Normalized mean daily discharges simulated at Beaver03 and Beaver01 were lower than, similar to, and higher than the Little Cedar at low (bottom panel), moderate (top panel), and high (middle panel) exceedance probabilities, respectively (Figure 37). Simulated low flows are larger than expected due to the overestimated baseflow contribution and to a lesser degree the imposed discharge in each branch necessary for numerical stability. The discrepancy (underestimation) at higher flows is smaller and extremely high discharges (less than 1% probability of exceedance) are in fairly good agreement with the Little Cedar.

Finally, the timing of simulated discharge and stream stage measurements at Beaver03 and Beaver01 were comparable at event time scales (Figure 38). While the magnitude of simulated discharge could not be evaluated, the favorable agreement in simulated hydrograph timing and shape to stream stage measurements gives an initial indication that storm discharge dynamics are reasonably well simulated.

Baseline Simulation

The simulated annual nitrogen balance for the BCW exhibited both similarities to and deviations from literature estimates for agricultural watersheds (Table 18). The 4-yr average net mineralization of 160 kg ha^{-1} is within the range identified in several field monitoring and modeling studies for corn-soybean systems (100-170 kg ha^{-1}; Carpenter-Boggs et al., 2000; Thorp et al., 2007; Qi et al., 2011; Qi et al., 2012). Annual denitrification in the top meter of the unsaturated zone ranged from 7-18 kg ha^{-1}, similar to other estimates for agricultural watersheds (10-23 kg ha^{-1}; Li et al., 2010), and simulated denitrification in the saturated zone (12 m deep) was 2-3 times larger. The 4-yr average simulated plant nitrogen uptake of 153 kg ha^{-1} is likely underestimated (Thorp et al., 2007; Thorp et al., 2008; Qi et al., 2011), though this estimate is the average for all vegetation types which includes approximately 75% row crops and 25% grass/pasture and forest. As a result of the lower than expected plant nitrogen uptake, simulated NO_3-N loss is higher than expected. Simulated tile drainage contributed an average of 75% to annual NO_3-N loss (as compared to an average 64% annual contribution to streamflow), and the 4-yr average NO_3-N loss of 45 kg ha^{-1} is higher than typical estimates for Iowan surface streams (20-25 kg ha^{-1}; Schilling and Wolter, 2005) but similar to the 2016 Iowan Surface estimate of 41 kg ha^{-1} (Jones et al., 2018). While simulated NO_3-N loss is likely overestimated, it is worth reemphasizing that precipitation was above average in three of the four study years; annual precipitation was 11% above average in 2014, 41% above average in 2016, and the Jan-Sep total in 2017 was 1% above the annual average.

Stream NO_3-N predictions from the baseline simulation are compared to observations at Beaver03 in Figure 39. Simulated stream concentrations at Beaver03 from Apr-Nov are within a factor of two of the observations during most of the monitoring period in each year. Concentrations are systematically under predicted from Apr-Jun, while agreement is better later in the growing season (July-Nov). The under predictions from Apr-Jun are partially attributed to overestimated winter runoff and uncertainties with the assumed agricultural management. While monthly runoff from Apr-Jun was reasonably well simulated (an average 7% lower than the Little Cedar from 2014-16), runoff in Jan-Feb was overestimated by an average of 68% from 2014-16. While runoff in Jan and Feb represented a small fraction of annual streamflow in both the simulation (9%) and for the Little Cedar (5%), it may have depleted the simulated soil NO_3-N pool earlier than expected. The agricultural management assumed for the water quality simulation is another source of uncertainty. The simulated agricultural management assumed a corn-soybean rotation based on the 2016 land use (Figure 33) and a single application of nitrogen fertilizer to corn at 233 kg ha^{-1} at the end of May each year. Review of the BCW annual land use from 2014-17 suggested that more continuous corn may be grown, which would imply fertilizer application to a greater fraction of the BCW row crop land on an annual basis. Fertilizer application may also occur earlier in Apr or May and even prior to corn planting (Thorp et al., 2007). To account for this, the fertilizer date was moved from 29 May to 15 Apr. This change increased simulated stream NO_3-N concentrations in Apr and May but only by 1-2 mg l^{-1} at Beaver03.

While there is clear room for improvement, the water quality simulation is doing some things well. While simulated NO_3-N concentrations are mostly underestimated, the recession slopes are similar to the observations. Dilution and increases in NO_3-N concentration connected to storm events are also comparable to observations, though the model still suffers from overestimated NO_3-N concentrations following storm events as a result of overestimated tile flow. The better agreement in 2015, a near average precipitation year, suggests water quality model predictions are more reliable in years of average hydrologic conditions while the struggles in 2014 and 2016 indicate the model is less reliable in wetter years. While improving simulated NO_3-N dynamics under extreme hydrologic conditions is an ongoing goal, the current model performance is in general agreement with other watershed modeling studies that attempted nitrogen fate and transport simulations (Hansen et al, 2009; Vervloet et al., 2018). These studies often note the difficulty in capturing inter-annual dynamics and model performance is typically best under moderate, near average hydrologic conditions.

Stream NO_3-N concentration was better predicted at the downstream Beaver01 water quality sensor (Figure 40). Simulated concentrations from Apr-Jun are under predicted to a lesser degree. Agricultural intensity (percentage of land use in row crops) decreases moving downstream in the BCW, so it is likely errors and uncertainties associated with the agricultural management are less impactful at downstream locations as the result of spatial averaging. Additionally, while the magnitude of simulated concentrations may deviate from the observations, the seasonal patterns are generally preserved.

Simulated and observed average monthly NO_3-N concentrations from Jan 2014 – Aug 2016 (n = 32 months) at Beaver03 and Beaver01 are shown in Figure 41. As expected based on Figures 39-40, simulated concentrations are noticeably underestimated in Apr-Jun and more reasonable from July-Nov. Simulated concentrations from Apr-Jun are underestimated by an average of 42% at Beaver03 and 24% at Beaver01. In comparison, the average deviation in simulated concentrations for July-Nov was a 5% overestimation at Beaver03 and a 16%

underestimation at Beaver01. Overall, the mean seasonal (Apr-Nov) observed concentrations at Beaver03 and Beaver01 were 9.3 and 10.6 mg l^{-1}, respectively, while the mean seasonal simulated concentrations at Beaver03 and Beaver01 were 7.0 and 8.6 mg l^{-1}, respectively.

Finally, the estimated NO_3-N loading at the two water quality sensors was comparable to other Iowan Surface streams. Annual NO_3-N load estimates from 2014-16 at Beaver03 ranged from 14 kg ha^{-1} (2015) to 47 kg ha^{-1} (2016) based on watershed area, and the 3-yr average annual load was 29 kg ha^{-1}. Similarly, annual load estimates from 2014-16 at Beaver01 ranged from 16 kg ha^{-1} (2015) to 54 kg ha^{-1} (2016), and the 3-yr average annual load was 33 kg ha^{-1}. The 3-yr average load estimates are higher than the 22 kg ha^{-1} long term average for the Iowan Surface but less than the 41 kg ha^{-1} load estimated for the Iowan Surface in 2016 (Jones et al., 2018).

Simulated average monthly NO_3-N loads from 2014-16 at Beaver03 and Beaver01 are shown in Figure 42. Simulated monthly loading ranged from approximately 1-5 kg ha^{-1} based on watershed area and exhibited patterns indicative of both monthly discharge and concentration. Monthly loading was greatest in Jun, the month of greatest simulated discharge and NO_3-N concentration, and lowest in Jan and Feb. Average loading in Sep was uncharacteristically high due to higher than normal discharge in Sep 2016. Close to 80% of the estimated average annual load occurred during the water quality monitoring period of Apr-Nov (approximately 67% of the year).

IWP Simulation

Constructed wetlands 1-6 were added to the MIKE 11 model for the IWP simulation to assess their influence on flow and NO_3-N reduction from 1 Sep 2016 to 30 Sep 2017 (13 months).

Figure 43 compares the peak monthly discharges from the baseline and IWP simulations at each IWP wetland location during the evaluation period (n = 13 months). Peak monthly discharges from the IWP simulation are lower than the baseline simulation if the data points are below the 1:1 line. As expected, the six constructed wetlands provided a measurable flood reduction benefit. Average monthly peak discharge reductions over the 13 month period ranged from 3% at Site 1 to 43% at Site 3. Reductions were more apparent at higher peak discharges while lower peak discharge predictions were virtually the same between the baseline and IWP simulations. The average peak discharge reduction was lowest at Site 1, the site with the largest drainage area and largest hydraulic structure (26 m weir) at its normal pool elevation. The average peak discharge reduction was greatest at Site 3, the site located in series with and downstream of Site 2. Hence, the flow attenuation provided by Site 2 allowed even greater reductions to occur downstream at Site 3. Finally, as one might expect, including evaporation in MIKE 11 (black open circles) had a minimal influence on the simulated peak monthly discharges.

The estimated peak discharge reductions at the project locations are smaller than those estimated in another Beaver Creek modeling study that used HSPF (IFC, 2016). The current study with MIKE SHE-MIKE 11 estimated monthly peak discharge reductions at the six wetlands during a 13 month period (Sep 2016 – Sep 2017) following project implementation. In the HSPF study, annual peak discharge reductions were calculated from a continuous, 65-yr (1948-2013) simulation. The HSPF study estimated average annual peak discharge reductions of 75% and 20% at Sites 3 and 6, respectively. While different time periods and model setups make direct comparisons difficult, the greater peak discharge reductions estimated by HSPF, particularly at Site 3, suggest the MIKE SHE-MIKE 11 peak discharge reductions are underestimated. This may likely be the case given the imposed discharge in each MIKE 11 branch lessens the flood reduction potential since simulated wetland water levels are nearly always at the normal pool and a small amount wetland outflow is always occurring.

Downstream flood reduction benefits of the six wetlands were also evaluated at the six index point locations (Figure 44). As expected, peak monthly discharge reductions at the six index points were lower than at the wetland locations and systematically decreased moving further downstream as a lesser percentage of the drainage area had runoff controlled by wetlands. The average monthly peak discharge reduction calculated along the Beaver Creek main stem shortly downstream of Site 1 was negative (-2%), indicating peak discharges were actually higher in the IWP simulation than the baseline simulation. Similar behavior was seen with the Slough Creek simulations; the current MIKE 11 setup struggles to simulate any flow attenuation when large hydraulic structures are defined because simulated water levels in the wetland are always at or slightly above normal pool, and as a result, storm discharge pulses can actually generate higher flows than if the wetlands were absent. On the Beaver Creek tributary containing Sites 2-5, the average peak reduction decreased from 43% at Site 3 to 19% downstream near its confluence with Beaver Creek. Similarly, the average peak discharge reduction on the tributary containing Site 6 decreased from 14% at the wetland to 4% near its confluence with Beaver Creek. The average peak reduction systematically decreased along the Beaver Creek main stem from 4% at Beaver03 to near 0% at Beaver01.

Similar to the wetland peak flow reduction comparison, the estimated peak flow reductions at the six index points are substantially less than those estimated in the Beaver Creek HSPF study. In the HSPF study, average annual peak discharge reductions of 35%, 20%, 25%, 15%, and 3% were estimated near index points 2, 4, 3, 5, and 6, respectively. Given including evaporation in MIKE 11 had a minimal impact on the simulated peak monthly flows, the lower than expected peak discharge reductions are believed to result primarily from the small imposed discharge in each MIKE 11 stream branch and its substantial influence when detention structures are included in the MIKE 11 network. This issue deserves further attention in order to improve the existing flood mitigation estimates of the IWP wetlands.

Like the peak flow reduction analysis, the localized and downstream water quality impacts of the six constructed wetlands were also evaluated. Figure 45 compares simulated hourly NO_3-N concentrations upstream (inlet) and downstream (outlet) of the nine projects to weekly grab sample concentrations collected between 13 Jun and 1 Aug in 2017. In general, poor agreement is observed between the simulated concentrations and grab sample observations at both the inlet and the outlet, which is not terribly surprising given the uncertainties associated with the hydrologic and water quality simulations, the spatial scales being assessed relative to the scale of the watershed model, and the "snap-shot" nature of the grab sample concentrations. While simulated inlet concentrations are underestimated most of the time, the inlet grab sample concentrations at sites 1, 2, and 5 reflect average concentrations in tile drainage water that discharge directly into the wetland, which are expected to be higher than concentrations in the contributing stream (if one exists). Better agreement is observed in simulated and observed outlet concentrations, though the outlet predictions are still not very reliable. While the NO_3-N predictive capability of the watershed model is limited at these smaller scales, the simulated wetland NO_3-N removal and its variability in response to hydrologic conditions is believable.

Simulated stream NO_3-N concentrations are compared to observations at Beaver03 in Figure 46. For reference, simulated concentrations from the baseline simulation (black line), IWP simulation (green line), and IWP simulation without MIKE 11 temperature dependence are all shown. Although NO_3-N concentrations from the IWP simulations are lower than the baseline simulation, the differences are quite small (1-2 mg l^{-1}), suggesting NO_3-N removal from the six IWP wetlands had a small influence on stream NO_3-N at Beaver03 during this time period. While this conclusion may be reasonable for some time period following project implementation (July/Aug 2016) as wetland vegetation and organic carbon sources necessary for denitrification were being established, it is difficult to assert past Jun 2017 given the IWP simulations failed to reproduce the substantial decline in NO_3-N concentration observed in late summer 2017. The observed NO_3-N recession from July-Sep in 2017 is likely attributed to stream and/or wetland NO_3-N retention processes that were further enhanced by warm water temperatures. Future efforts will seek to more accurately assess the contribution of the IWP wetlands to the NO_3-N decline observed in late summer 2017.

Simulated stream NO_3-N concentration at Beaver01 exhibited similar patterns to the observations but was overestimated during most of the evaluation period (Figure 47). As with the Beaver01 predictions, the differences between the baseline and IWP simulations were relatively small (1-2 mg l^{-1}), although smaller deviations are expected downstream at Beaver01 is less pronounced. While the timing and recession slopes of simulated NO_3-N are reasonable, the increase in NO_3-N concentration following storm events is overestimated by a large amount, again believed to result from the overestimated contribution of simulated tile drainage to streamflow. Observed NO_3-N concentrations only decline to approximately 7 mg l^{-1} in July-Sep in 2017, suggesting the greater decline in NO_3-N concentration at Beaver01 was primarily attributed to wetland, rather than stream, NO_3-N retention processes.

Estimates of monthly wetland NO_3-N removal (kg ha^{-1} watershed) from Sep 2016 (S) through Sep 2017 (S) are shown in Figure 48. For each wetland, the drainage area, total NO_3-N mass removal, and corresponding percent removal during the 13-month period are listed. Simulated NO_3-N mass and percent removal at each wetland is directly tied to temperature; mass and percent removal were lower in colder months (Oct-Apr) and higher in warmer months (Jun-Sep) when NO_3-N loading was also greater. Mass removal was greatest in Jun/July 2017 and lowest during winter (Dec-Jan), while percent removal was greatest in Aug/Sep 2017 and lowest during winter. Total wetland NO_3-N mass removal ranged from 4.3-22 kg ha-1; percent removal ranged from 7.1% at Site 1, the site with the largest drainage area, to nearly 25% at Site 4, the site tied with the second smallest drainage area.

Estimates of monthly NO_3-N load reductions at the six index points exhibited similar characteristics to the wetland NO_3-N removal (Figure 49). The estimated mass and percent removal downstream of wetlands 2-5 (Index Point 2) were 7.5 kg ha^{-1} and 7.4%, respectively. NO3-N percent removal systematically decreased along the Beaver Creek main stem from 3% at Beaver03 to 1.8% at Beaver01. The NO_3-N retention estimates are believed to be most reliable during the growing season (May-Sep) when temperature dependence is important and less reliable outside the growing season when wetland NO_3-N removal can still occur despite cold temperatures. Hence, the estimated NO_3-N removal estimates during this 13-month period are possibly underestimated as a result of lower than expected retention during colder months.

Summary and Conclusions

As part of the Iowa Watersheds Project (IWP), six dual purpose flood mitigation and nutrient removal wetlands costing $1.5 million were built in the agricultural Beaver Creek Watershed (BCW) in north central Iowa in the summers of 2015 and 2016. The flood reduction impacts of the six wetlands were previously evaluated on an event basis for different design storm return periods (Thomas et al., 2016). The purpose of this study was to expand upon these initial efforts to evaluate the six IWP wetlands for flow and NO3-N reduction under variable hydrologic conditions using continuous watershed simulations.

The hydrologic and nitrogen fate and transport simulations were performed with MIKE SHE-MIKE 11 ECO Lab. Hydrologic and water quality model performance was assessed in a systematic manner across different time scales considering both simulated water and nitrogen mass balances and finer scale dynamics. During the first three years of the study period prior to IWP practice implementation (baseline simulation), simulated hydrologic and nitrogen mass balances were fairly representative of the region, while simulated stream NO3-N concentration was underestimated early in the growing season (Apr-Jun) and more reasonable later in the growing season (July-Nov). Differences between model predictions and observations are attributed to errors in the simulated hydrology (overestimated baseflow), uncertainties associated with the assumed agricultural management, and limitations with the water quality model setup. The months of Apr-Nov (water quality monitoring period) were estimated to account for close to 80% of the annual NO3-N stream load in the BCW.

The IWP simulation incorporating the six constructed wetlands was performed over a 13-month period (Sep 2016 – Sep 2017) and revealed strengths and weaknesses of the simulated hydrology and water quality. The six IWP wetlands provided measurable flood reduction benefits locally while the downstream impacts were more modest. Average peak monthly discharge reductions ranged from 3-43% at the wetlands and from 19% shortly downstream of five of the six projects to near 0% at the BCW outlet. While peak discharge reduction trends were as expected (greater near the projects and lower at further downstream locations), the percent reductions were noticeably lower than those estimated in another Beaver Creek study. The underestimation is attributed to certain components of the MIKE 11 model setup that were implemented to ensure reasonable water quality predictions. Future work should consider alternative methods on how to incorporate the wetland detention structures in MIKE 11 without sacrificing hydrologic performance.

Similarly, the simulated NO_3-N reduction impact of the IWP wetlands was greatest locally and decreased moving downstream. While simulated inlet and outlet NO_3-N concentrations at each wetland deviated from grab sample observations, the simulated wetland NO3-N removal trends were as expected. The wetlands reduced incoming NO_3-N loads by an estimated 7-25%. While the estimated wetland NO3-N mass removal was not insignificant, the percent reductions are likely underestimated as a result of overestimated NO3-N loading. Downstream load reductions ranged from 7% at the confluence of the tributary containing five of the six IWP wetlands to 2% near the BCW outlet. Stream concentration predictions at the two water quality sensors exhibited similar patterns to observations but were overestimated during most of the monitoring period and failed to capture the decline in NO_3-N concentration shortly downstream of the projects in late summer 2017. The observed decrease in NO_3-N recession was primarily attributed to wetland NO3-N removal, and future efforts will seek to improve the water quality predictions to assess the relative contribution of the IWP wetlands on NO_3-N reduction during this time period.

Tables and Figures

Table 16. Summary characteristics for the nine constructed wetlands in the Beaver Creek Watershed.

ID	Funding[1]	Constructed	Drainage Area (ha)	Row Crops	Wetland:Watershed Ratio[2]	Normal Pool Storage[3] (mm)	Flood Storage[4] (mm)	Cost[5]
1	IWP	Fall 2015	214	87%	1.5%	7	15	$ 405,914
2	IWP	July/Aug 2016	61	80%	1.4%	9	34	$ 220,524
3	IWP	July/Aug 2016	75	79%	0.9%	6	25	$ 203,936
4	IWP	July/Aug 2016	61	80%	2.0%	9	33	$ 186,076
5	IWP	July/Aug 2016	43	81%	2.7%	11	20	$ 180,212
6	IWP	July/Aug 2016	141	90%	1.1%	5	16	$ 342,695
Total			534	85%	1.6%	8	23	$ 1,539,359
A	CREP	2009	250	79%	1.1%	6	16	
B	CREP	2008	252	83%	0.9%	6	10	
C	Private	Fall 2013/Spring 2014	27	80%	4.1%	27	18	
Total			528	81%	1.2%	7	13	
Grand Total			1062	83%	1.4%	7	18	

[1]IWP: Iowa Watersheds Project; CREP: Iowa Conservation Reserve Enhancement Program

[2]Ratio of wetland normal pool area to watershed area
[3]Normal pool storage (m^3) normalized by drainage area
[4]Pool storage (m^3) between normal pool and emergency spillway elevations normalized by drainage area
[5]Includes site specific construction and easement costs and average cost of engineering/administration/misc. expenses for all six projects

Table 17. Simulated annual water balance for the Beaver Creek Watershed.

	Beaver Creek Watershed Modeled (070802010901): 45 km^2						Little Cedar River near Ionia Observed (USGS 05458000): 793 km^2				
Year	P	ET/P	Q/P	BFI	Q_{mean}	Q_{peak}	P	Q/P	BFI	Q_{mean}	Q_{peak}
	mm				mm day^{-1}		mm			mm day^{-1}	
2014	1003	0.69	0.30	0.81	0.9	22.4	993	0.30	0.64	0.8	8.8
2015	892	0.74	0.20	0.95	0.5	3.8	1005	0.30	0.69	0.8	9.4
2016	1273	0.53	0.40	0.79	1.4	23.2	1322	0.45	0.61	1.6	21.8
2017	911	0.67	0.48	0.81	1.6	33.4	795	0.44	0.71	1.3	10.1
3-yr Avg (2014-16)	1056	0.65	0.30	0.85	0.9		1107	0.35	0.65	1.1	

*2017: Jan-Sep only due to unavailability of Stage IV radar rainfall estimates during remainder of year
P: precipitation; ET: evapotranspiration; BFI: baseflow index; Q_{mean}: mean daily discharge; Q_{peak}: peak mean daily discharge

Table 18. Simulated annual nitrogen balance for the Beaver Creek Watershed.

Year	Net Min.	Den.		Plant N Uptake	NO_3-N Loss				FWANC
		UZ	SZ		Surface Runoff	Groundwater	Tile Drainage	Total	
					kg ha^{-1}				mg l^{-1}
2014	149	9	26	152	2	6	26	34	11.3
2015	176	7	19	191	0	4	15	19	10.5
2016	170	18	38	138	3	12	51	67	13.1
2017	143	8	35	133	4	12	43	59	13.5
4-yr Avg	160	11	30	153	2	9	34	45	12.1

*Only Jan-Sep is considered in 2017 due to lack of Stage IV radar rainfall estimates
Net Min: net mineralization; Den: denitrification; UZ: unsaturated zone; SZ: saturated zone; FWANC: flow-weighted average NO3-N concentration

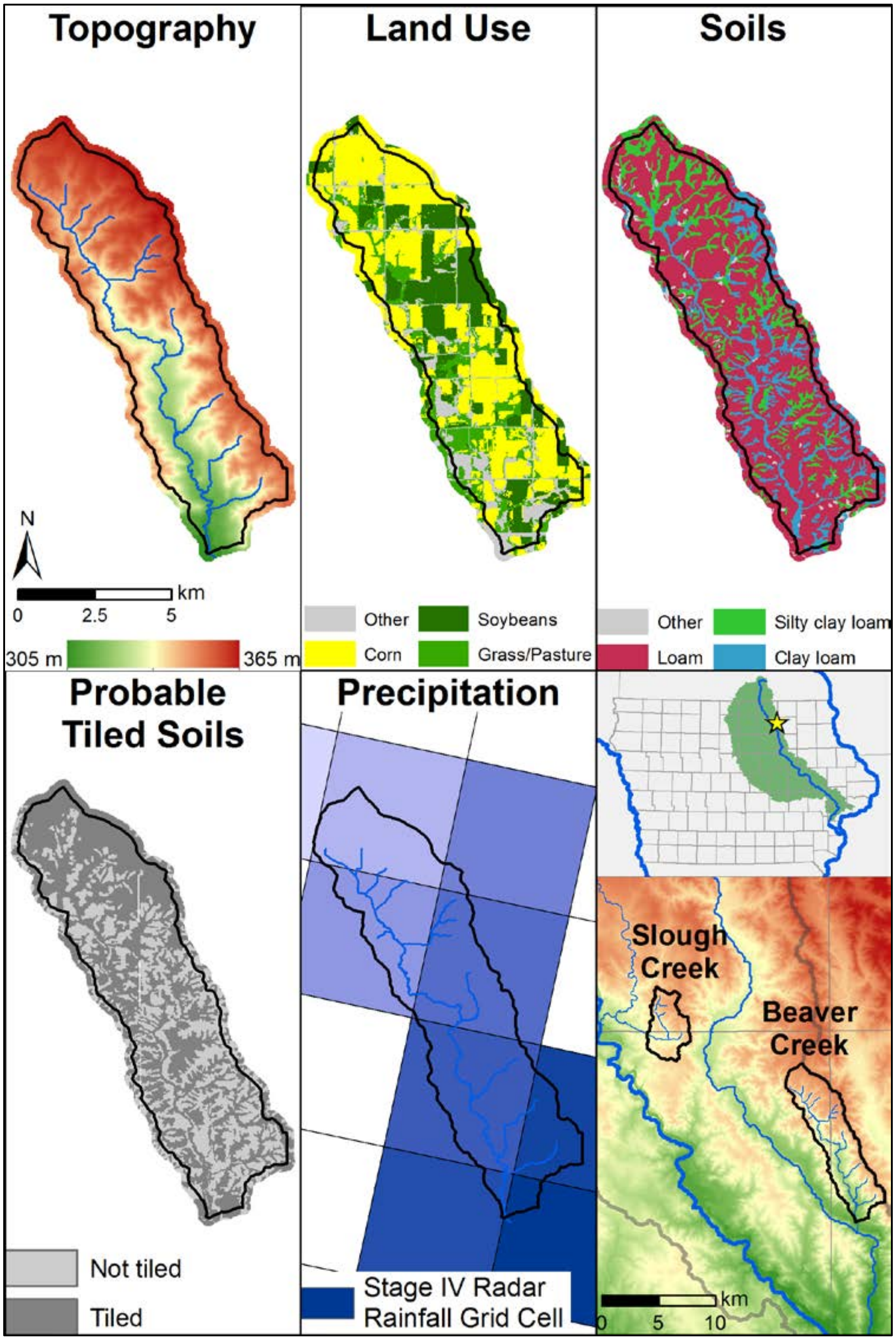

Figure 32. Beaver Creek Watershed overview and spatial datasets used to build the MIKE SHE-MIKE 11 hydrologic model.

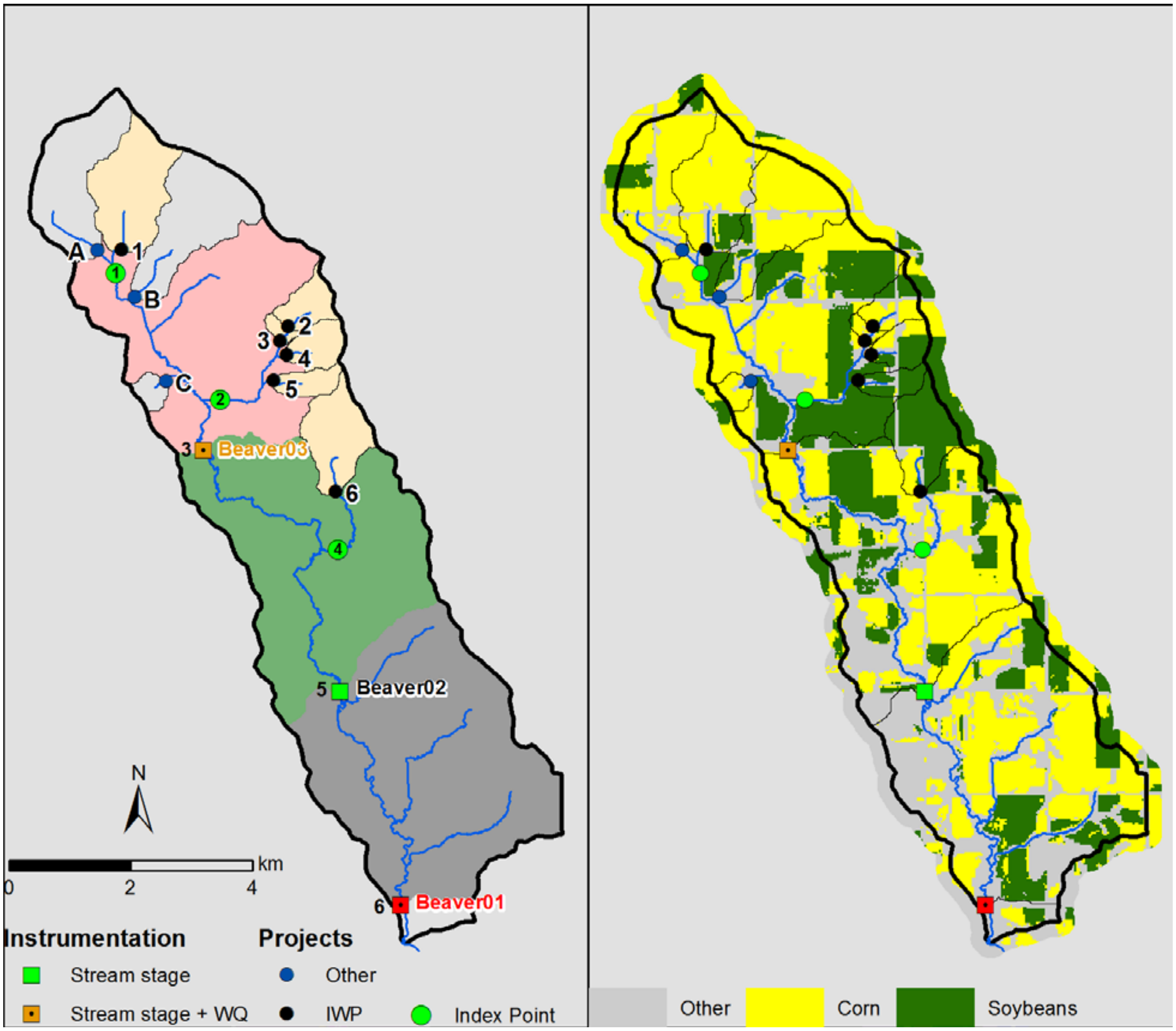

Figure 33. Beaver Creek Watershed wetlands, instrumentation, drainage areas, row crop intensity, and index points used to assess downstream impacts.

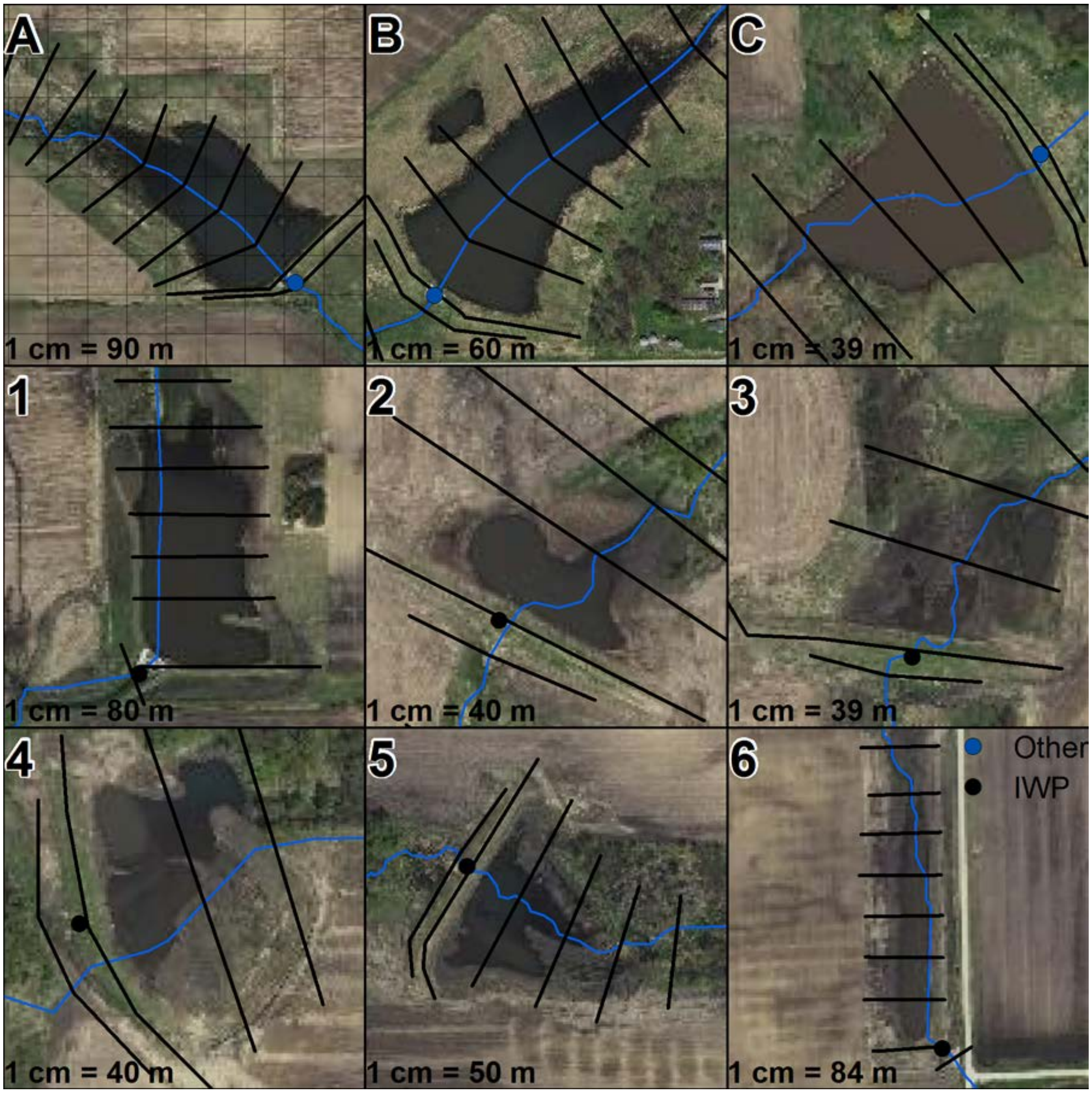

Figure 34. Aerial imagery of the nine constructed wetlands (Apr 2017) and how they are represented in the MIKE 11 model. Black lines denote cross sections and filled circles denote the location of hydraulic structures in the MIKE 11 model.

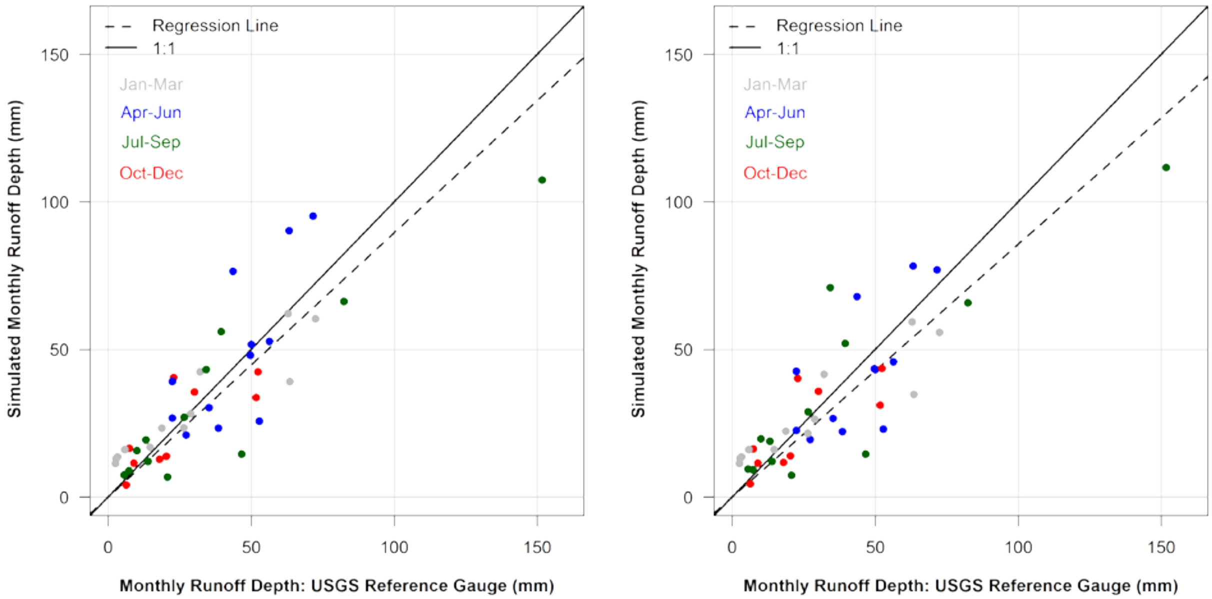

Figure 35. Simulated monthly runoff at Beaver03 (left) and Beaver01 (right) compared to Little Cedar (USGS 05458000) observations (Jan 2014-Sep 2017).

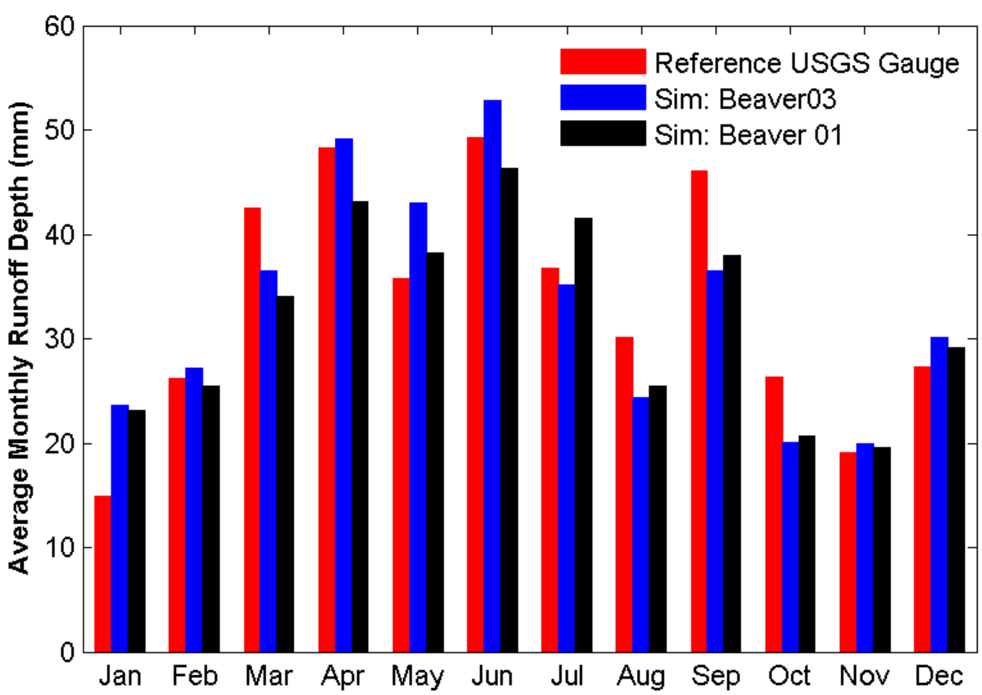

Figure 36. Simulated average monthly runoff at Beaver03 and Beaver01 compared to Little Cedar (USGS 05458000) observations (Jan 2014-Sep 2017).

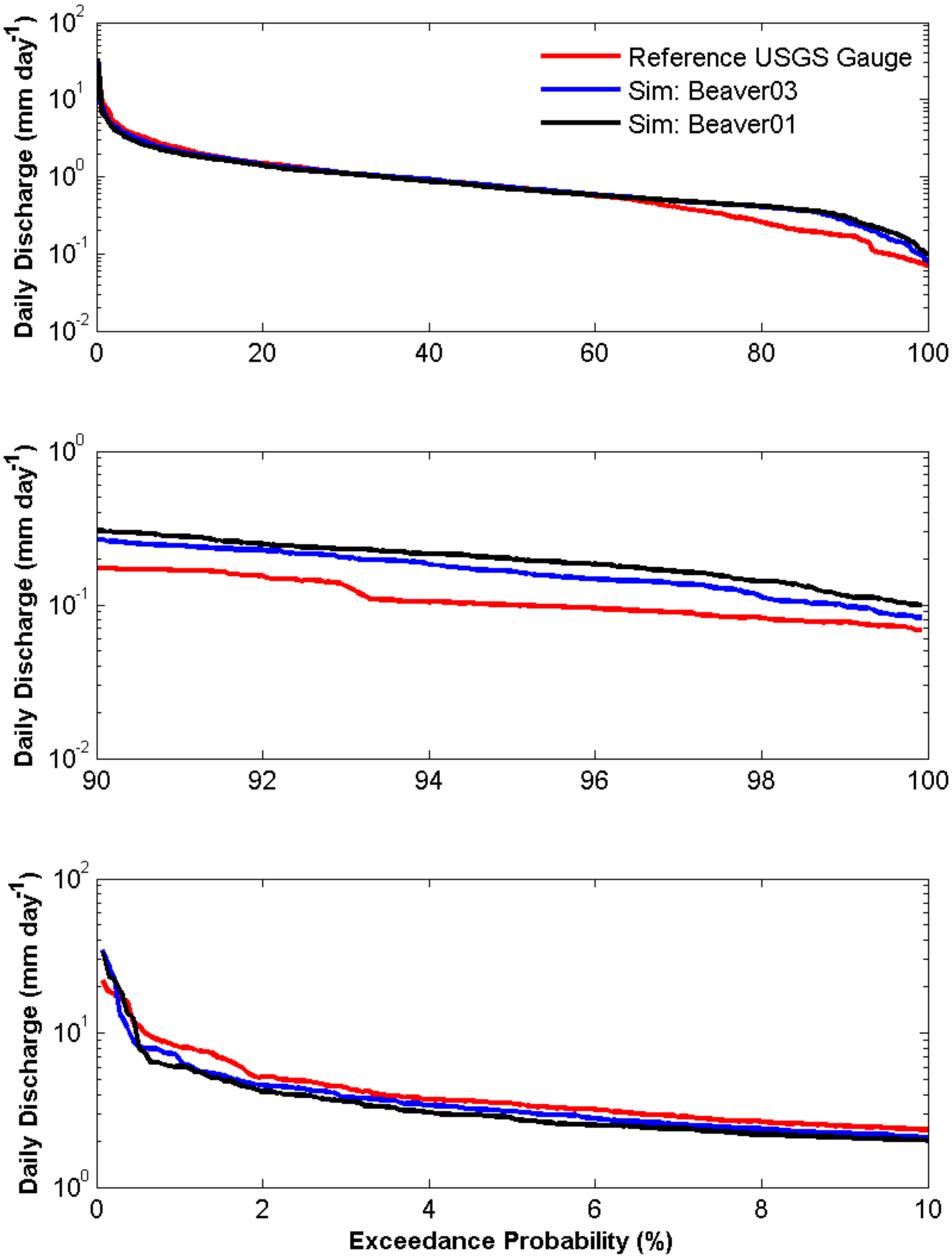

Figure 37. Flow duration curves of mean daily discharge normalized by drainage area for Beaver03 (simulated), Beaver01 (simulated), and the Little Cedar (observed; USGS 05458000) (Jan 2014-Sep 2017).

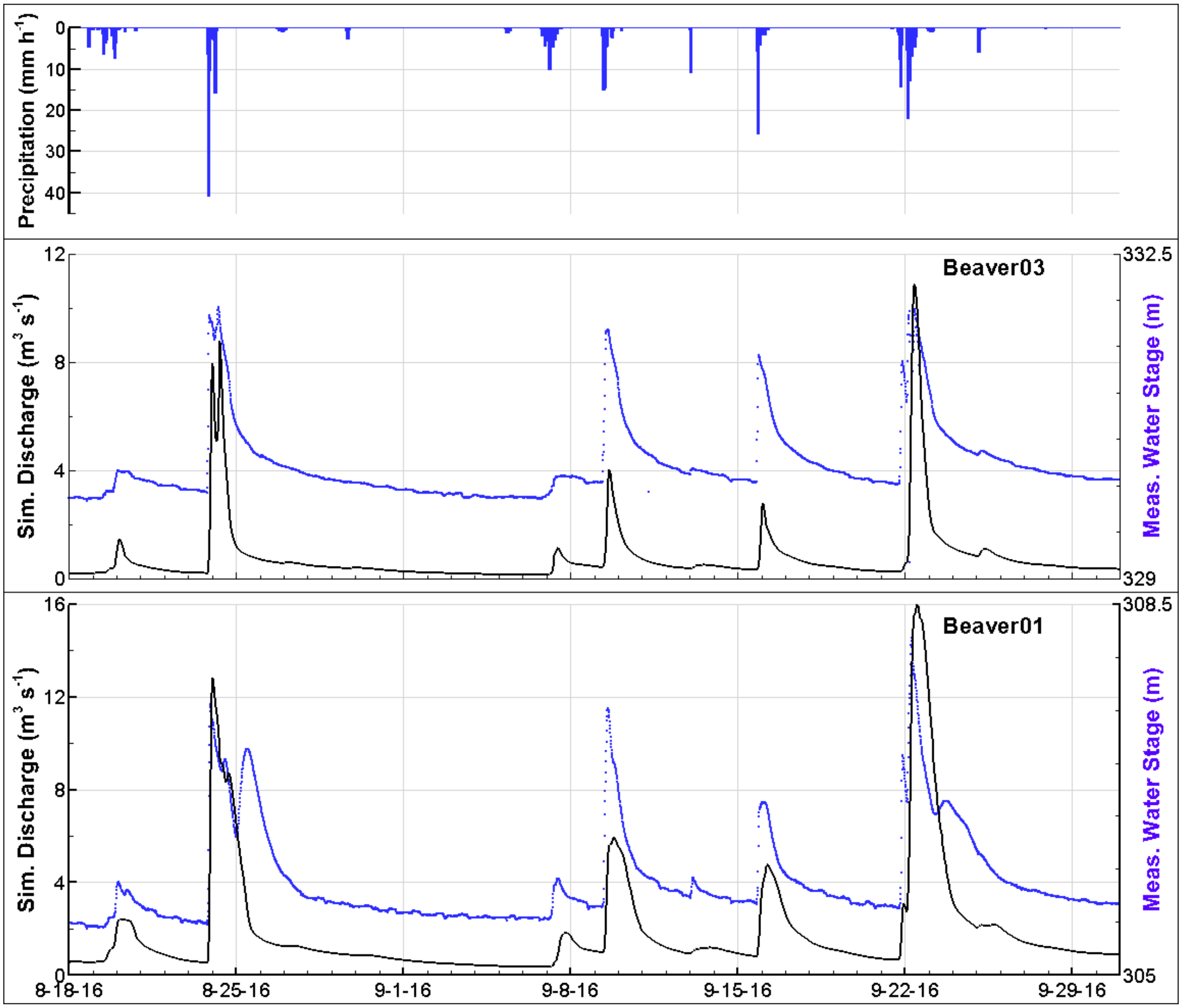

Figure 38. Example of simulated 15-min discharge dynamics at Beaver03 and Beaver01 during a period of frequent precipitation in Aug/Sep 2016. Stream stage measurements (15-min) at Beaver03 and Beaver01 are also shown.

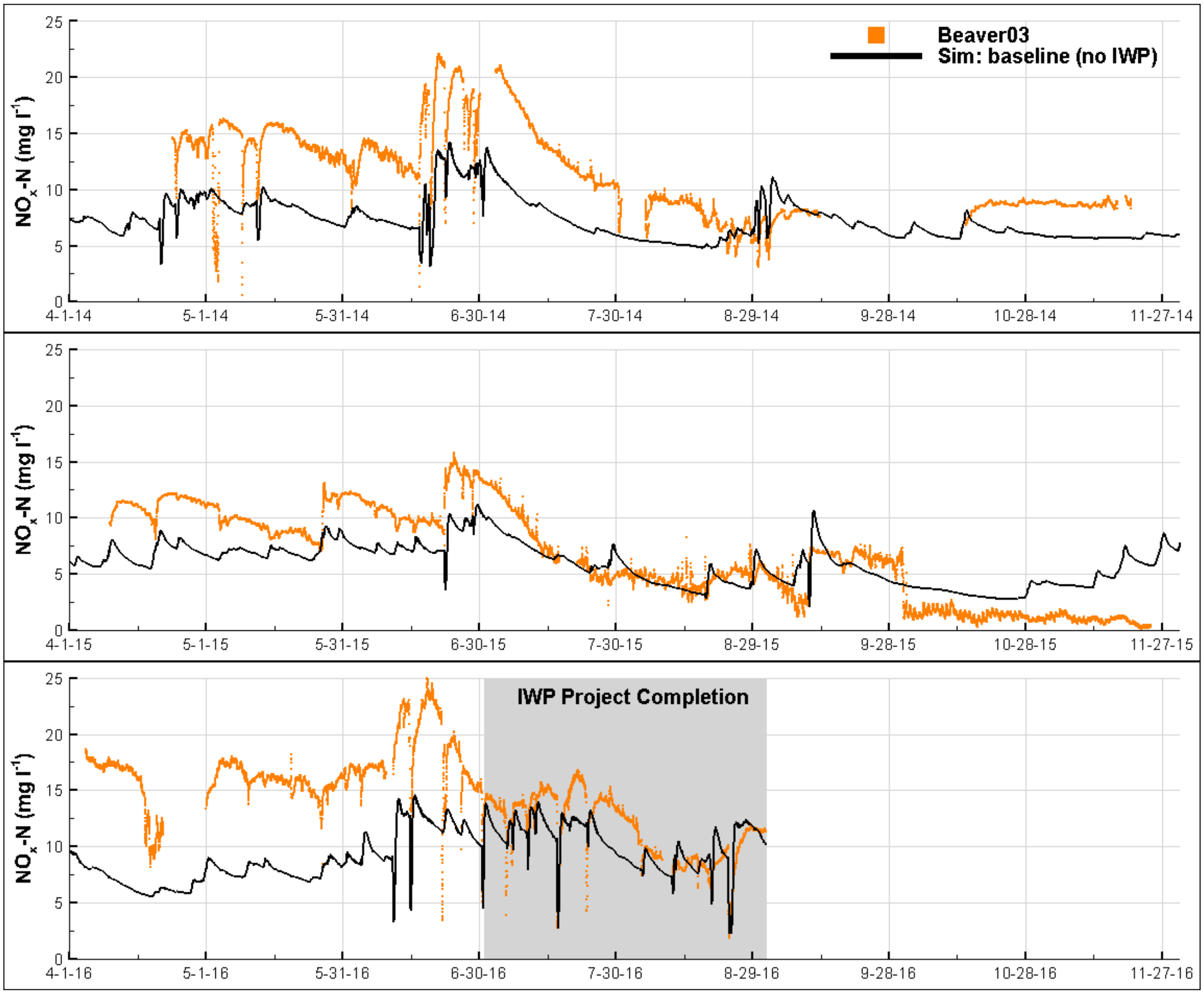

Figure 39. Baseline simulation of NO_3-N concentration compared to observations at Beaver03 (hourly).

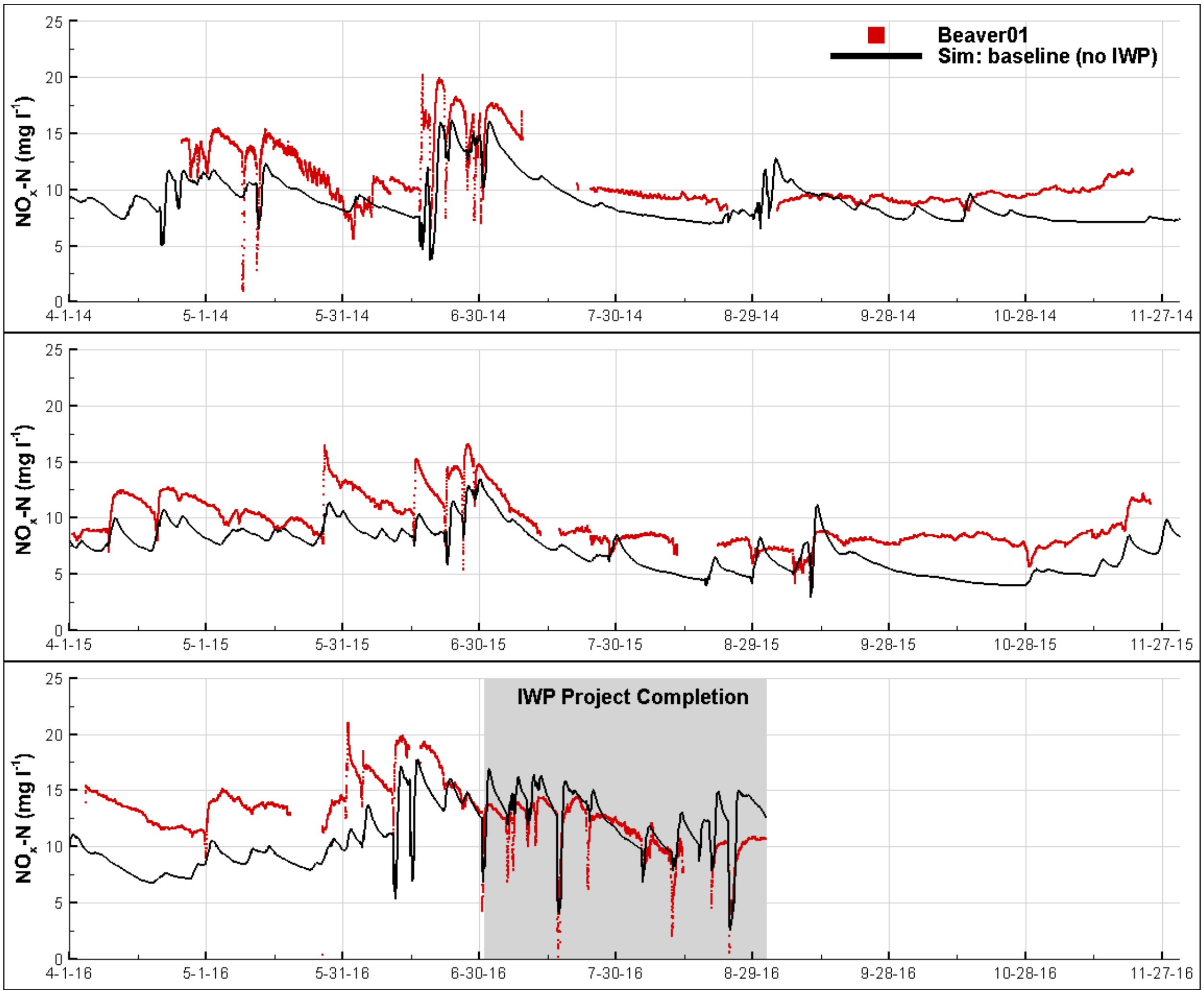

Figure 40. Baseline simulation of NO_3-N concentration compared to observations at Beaver01 (hourly).

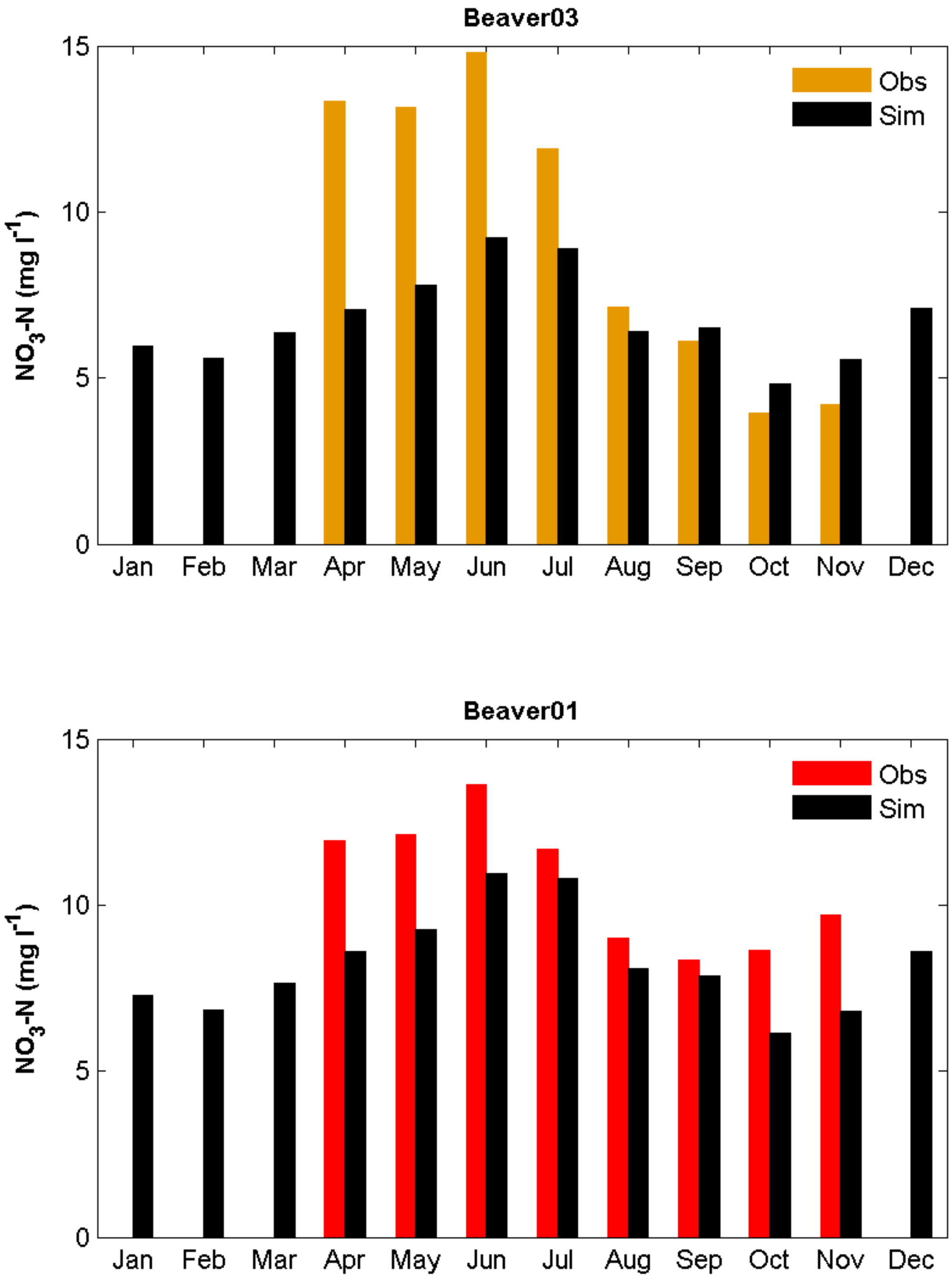

Figure 41. Simulated and observed average monthly NO_3-N concentration at Beaver03 and Beaver01 from the baseline simulation.

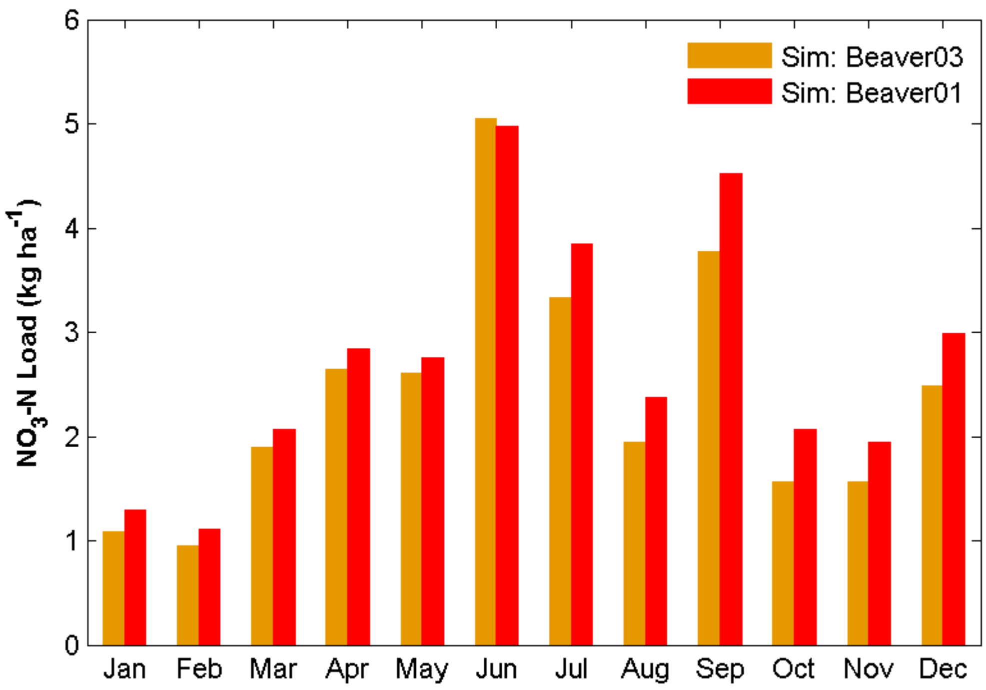

Figure 42. Simulated average monthly NO_3-N loading at Beaver03 and Beaver01 from the baseline simulation.

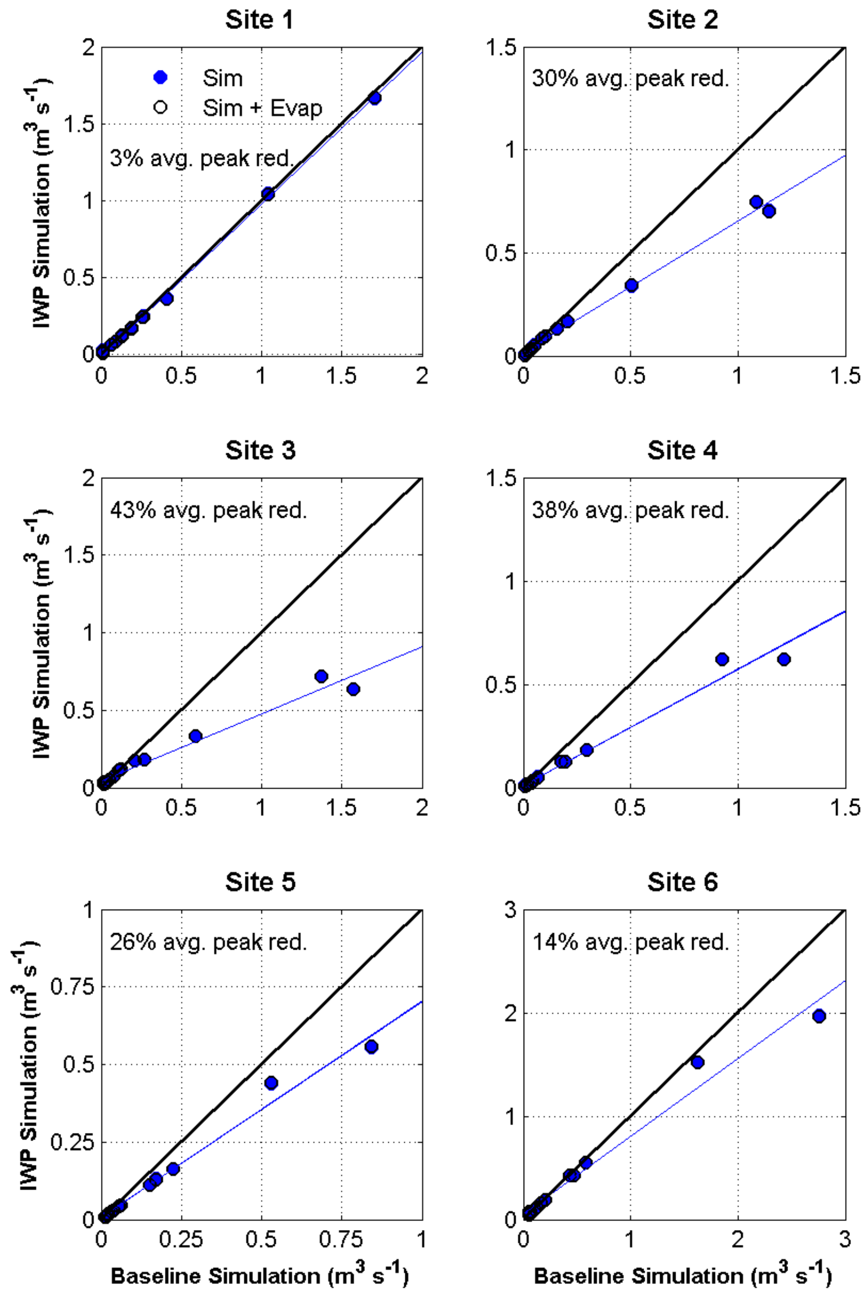

Figure 43. Comparison of simulated peak monthly discharges from the baseline and IWP simulations at the six IWP wetlands (Sep 2016-Sep 2017, n = 13 months).

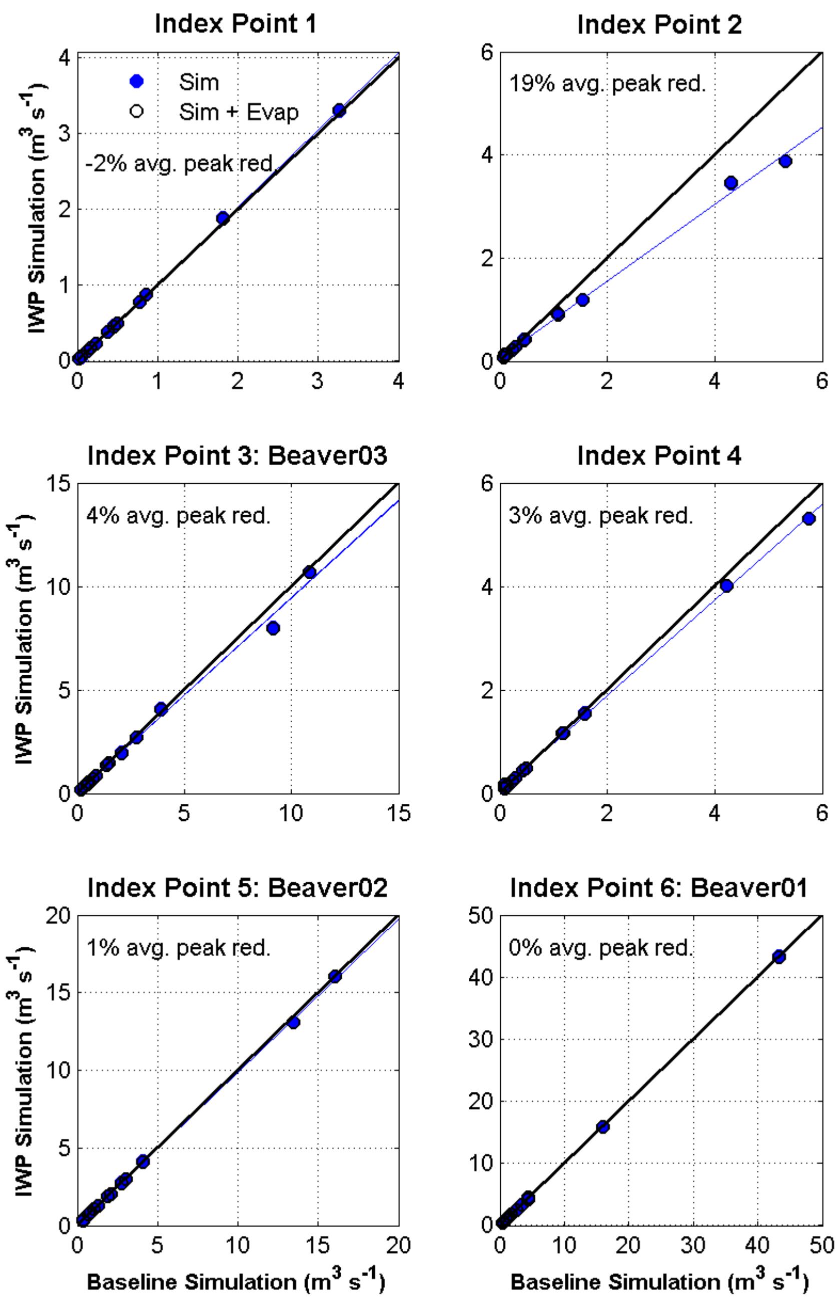

Figure 44. Comparison of simulated peak monthly discharges from the baseline and IWP simulations at the six downstream index points (Sep 2016-Sep 2017, n = 13 months).

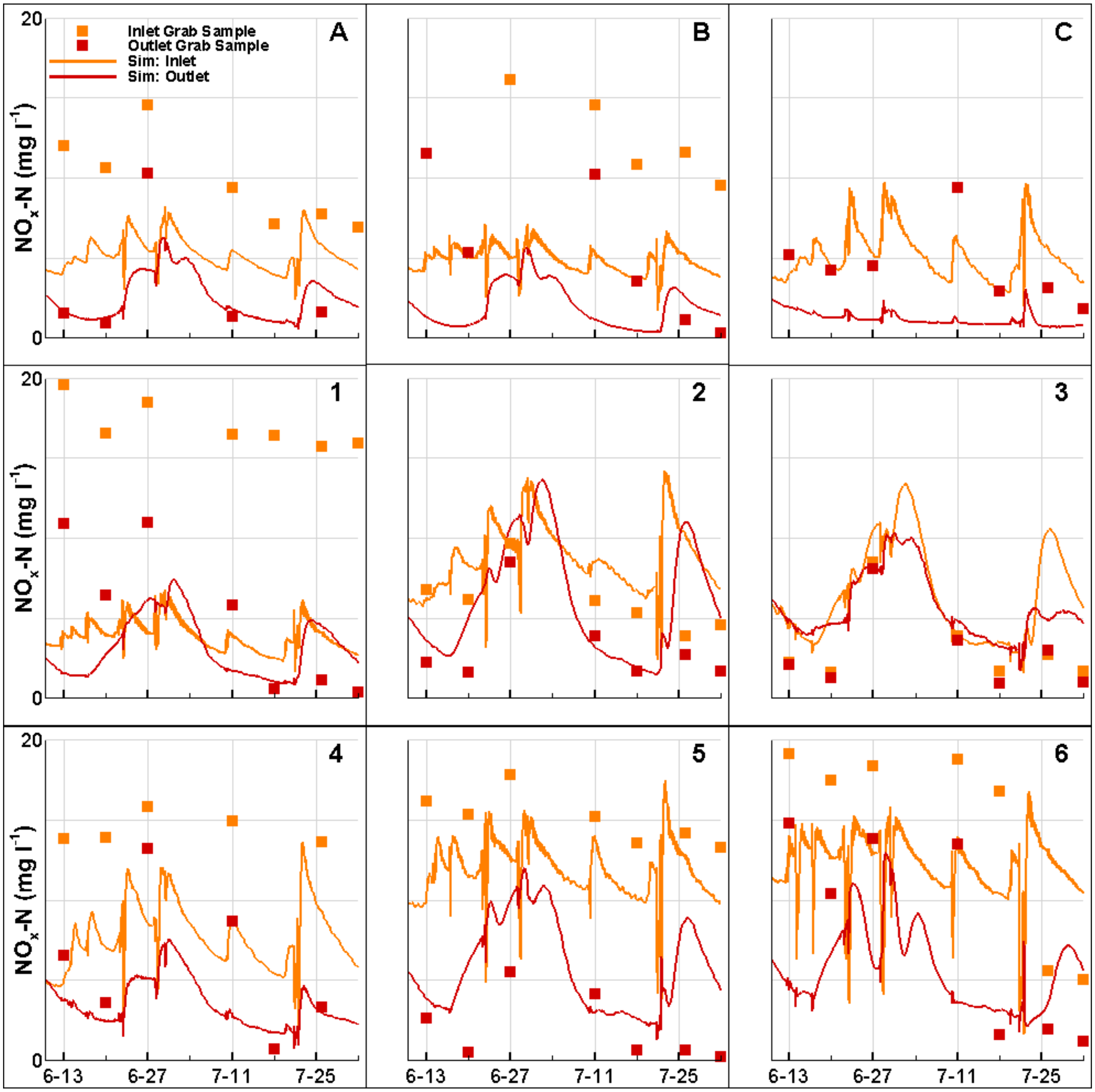

Figure 45. Comparison of simulated hourly NO_3-N concentrations at the nine Beaver Creek wetlands to weekly grab samples from 13 Jun to 1 Aug in 2017.

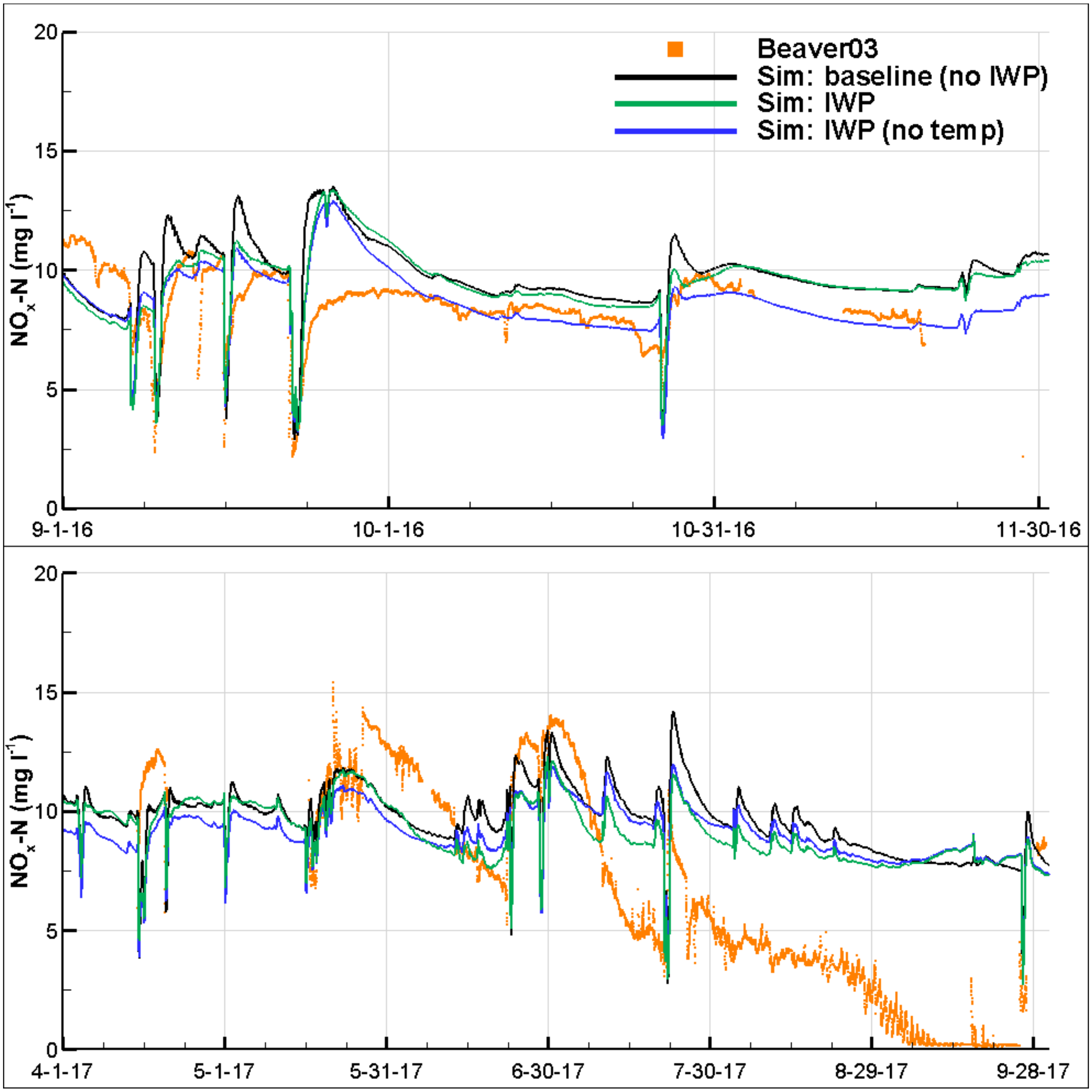

Figure 46. IWP simulation of NO_3-N concentration compared to observations at Beaver03 (hourly). Predictions from the baseline simulation (black) are also shown.

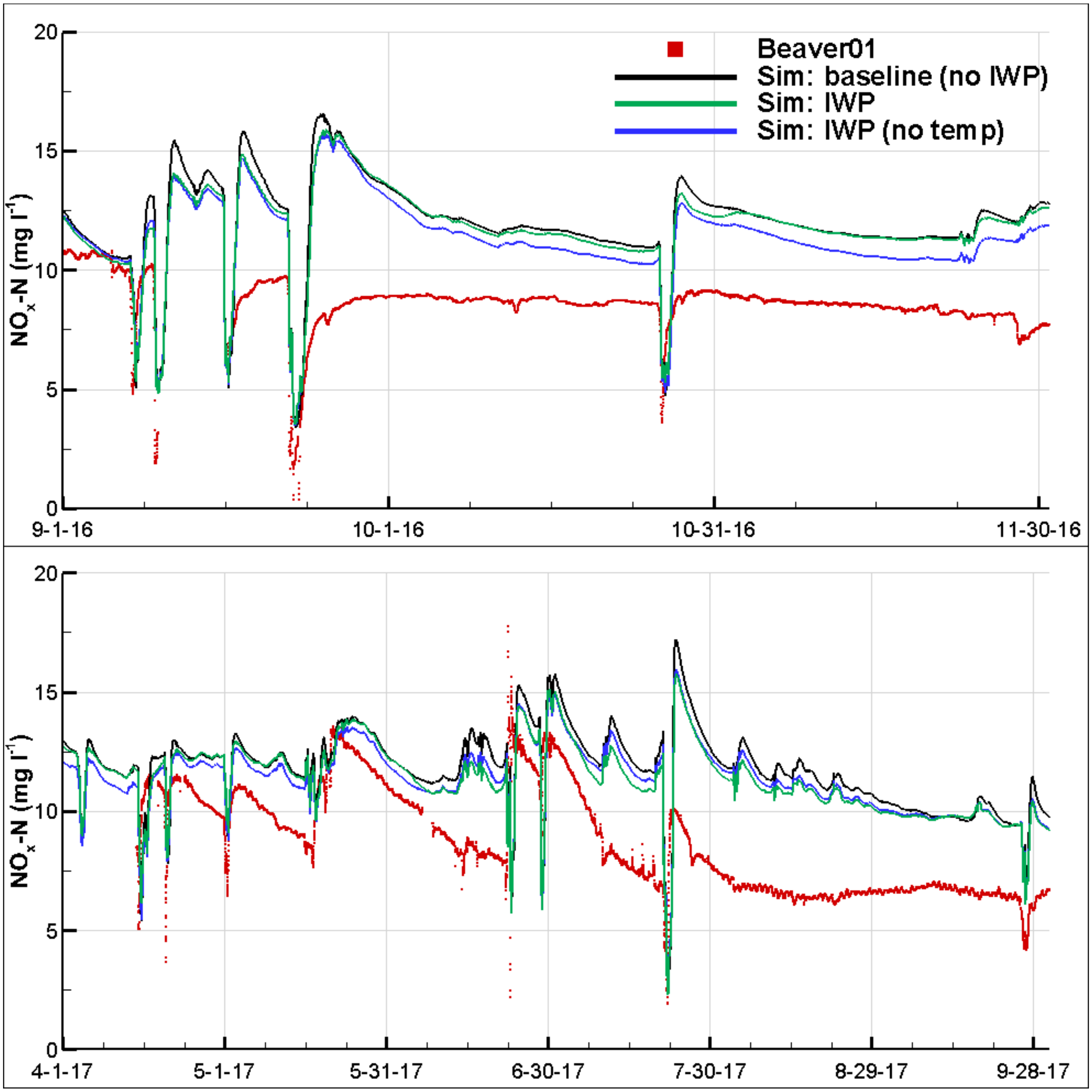

Figure 47. IWP simulation of NO_3-N concentration compared to observations at Beaver01 (hourly). Predictions from the baseline simulation (black) are also shown.

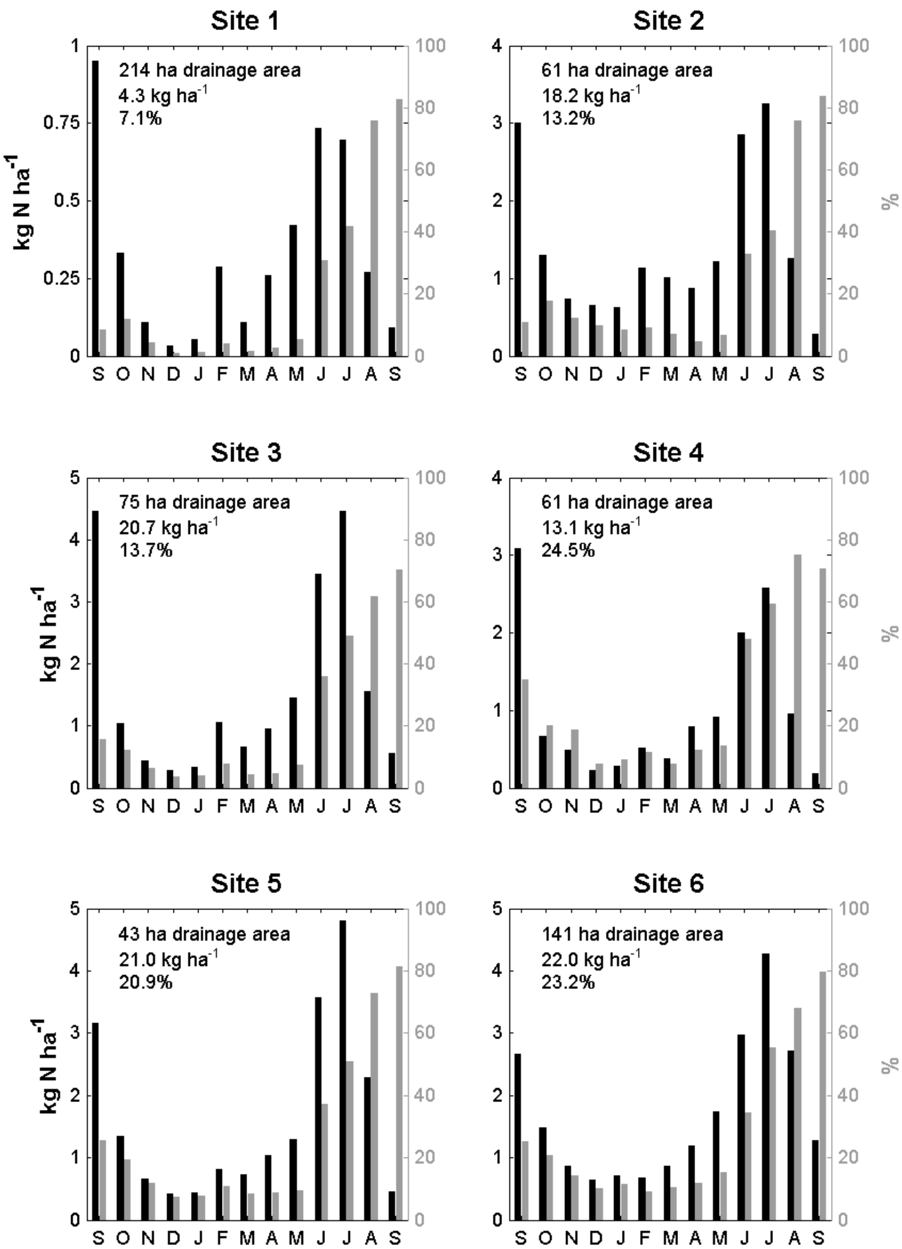

Figure 48. Simulated monthly NO_3-N mass and percent removal at the six IWP wetlands during the IWP simulation (Sep 2016-Sep 2017, n = 13 months). For each wetland, the drainage area, total NO_3-N load reduction (normalized by watershed area) during the 13 month period, and corresponding percent NO_3-N removal are listed.

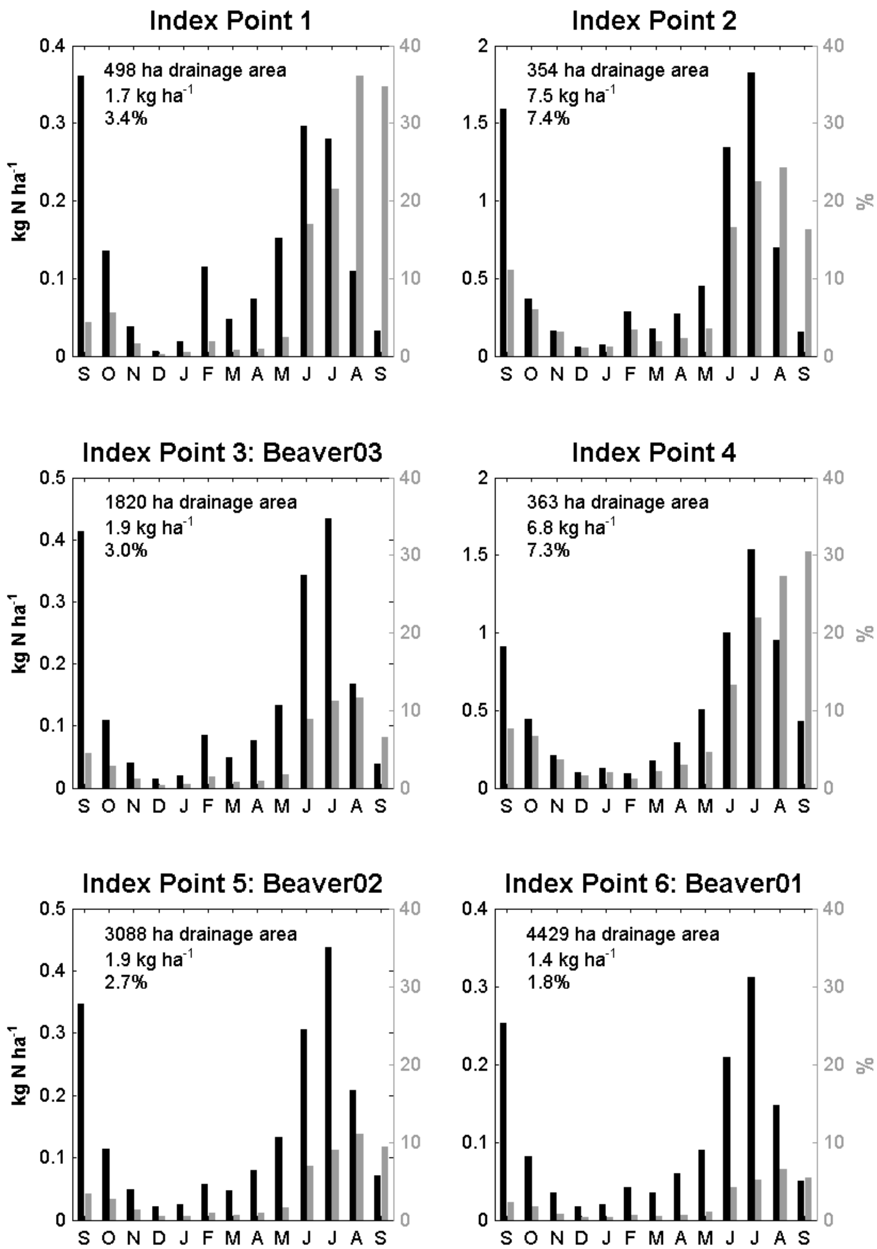

Figure 49. Simulated monthly NO_3-N load reductions at the six downstream index points during the IWP simulation (Sep 2016-Sep 2017, n = 13 months). For each index point, the drainage area, total NO_3-N load reduction (normalized by watershed area) during the 13 month period, and corresponding percent NO_3-N removal are listed.

6. SUMMARY AND CONCLUSIONS

Reducing agricultural nutrient loading, particularly in Iowa and other Corn Belt states, is critical to achieving Gulf Hypoxia water quality goals. This study evaluated nitrogen fate and transport in agricultural areas of Iowa at different spatial scales using both monitoring and modeling. Rare high-frequency, continuous water quality monitoring and hydrologic modeling were used to assess wetland NO_3-N dynamics and removal under variable hydrologic conditions. Field data from an agricultural research plot and the high-frequency water monitoring data were used to guide and inform numerical simulations of nitrogen fate and transport at the field, wetland, and watershed scales. The development of simple soil and stream nitrogen models implemented in the simulations showed potential for predicting nitrogen mass balance components, stream NO_3-N dynamics, and wetland NO_3-N removal over multiple years, but further improvements are needed to make accurate predictions under extreme hydrologic conditions and at larger watershed scales.

Field Scale Simulations

A 5-yr continuous simulation of a 0.06-ha tile drained corn-soybean plot illustrated strengths and weaknesses of the soil nitrogen model. Simulated annual nitrogen components and NO_3-N loss in subsurface drainage were comparable to observations and literature estimates in normal to wet years, while NO_3-N leaching was severely underestimated in the two driest study years. The simulated response of NO3-N loss in subsurface drainage to fertilizer rate was unsatisfactory compared to field data and RZWQM simulation results; the poor performance was primarily attributed to limitations with the built-in MIKE SHE plant uptake function, ability to only apply soluble water quality sources (e.g. fertilizer) in MIKE SHE, and not including adsorption-desorption processes for NH_4-N in the MIKE SHE ECO Lab soil nitrogen model.

High-Frequency, Continuous Water Quality Monitoring

High resolution (15-min) water quality monitoring data provided a unique opportunity to assess wetland NO_3-N removal under variable hydrologic conditions. This study provided valuable insights into wetland NO_3-N retention patterns at different time scales, the influence of hydrologic conditions and water temperature on wetland removal performance, storm NO3-N dynamics, and the connection between NO3-N diurnal variability and biological processes. The wetland retention estimates were put in context with other wetland studies and larger scale impacts of wetland implementation were estimated.

Wetland Simulations

The high-frequency water quality monitoring guided numerical simulations of wetland NO_3-N removal. Controlled wetland simulations using simple first order, temperature dependent kinetics were able to adequately predict warm season (May-Nov) wetland NO_3-N dynamics in three of four study years. Water temperature was also predicted in a satisfactory manner in all four study years. A sensitivity analysis illustrated the denitrification first order rate constant was most influential during low flow conditions when wetland residence times were high and that temperature was important for predicting summer NO_3-N removal.

Watershed Simulations

Integrated watershed simulations predicted nitrogen cycling in a comprehensive manner. To our knowledge, this represented one of the first attempts to evaluate wetlands for flow and NO_3-N reduction using continuous watershed simulations of nitrogen fate and transport with a physically based hydrologic model. Given the simplicity of the nitrogen model and assumptions of the agricultural management, stream and wetland NO_3-N dynamics were simulated in a reasonable manner under average hydrologic conditions while performance weakened with more extreme hydrologic conditions. Water temperature simulations were satisfactory in the wetland and unsatisfactory in the stream, indicating accurate cross section geometry and sufficient water depth are needed for reliable predictions. Errors in simulated hydrology were magnified in the water quality simulations, underscoring the fundamental importance of accurately simulated water balance components. Stream NO_3-N concentrations after storm events and NO_3-N loading were overestimated, indicating revisions to the hydrologic and water quality model setups are necessary to improve the reliability of the flood and NO_3-N reduction estimates offered by multiple wetlands at the watershed scale.

Limitations and Areas of Future Work

Study limitations and areas of future work include:

1. Hydrologic model
    a. Tile drainage. The contribution of tile drainage to streamflow in the watershed simulations is likely too high, resulting in overestimated increases in stream NO_3-N concentration following storm events and, as a result, overestimated NO_3-N loading. To correct this, initial efforts should treat the MIKE SHE drain time constant as a calibration parameter and its current value (10^{-7} s^{-1}) should be reduced until a more reasonable baseflow index is simulated.
2. Soil nitrogen model
    a. Nitrogen process parameterizations. This study assumed simple first order kinetics with adjustments for temperature and soil moisture. Agricultural systems models often assume zero order or Michaelis-Menten kinetics with reduction factors for multiple variables including temperature, soil moisture, and pH.

b. Soil organic matter dynamics. This study did not perform soil organic matter modeling to derive mineralization estimates. Including a more detailed representation of soil mineralization processes like those defined in agricultural systems models would be an interesting exercise to assess its influence and importance on NO_3-N leaching and stream NO_3-N predictions.

c. NH_4-N. This study did not consider particulate NH_4-N. This is believed to be a major limitation given NH_4-N is expected to remain sorbed to soil until nitrification occurs. Additionally, pH should be considered since both adsorption-desorption processes and the partitioning of NH_3-N and NH_4-N are pH-dependent.

3. Stream nitrogen model

a. Nitrogen process parameterizations. First order, temperature dependent kinetics were assumed for nitrification and denitrification. No consideration of other factors, notably dissolved oxygen and organic carbon, were considered in the denitrification estimates, which may be important during the cold season.

b. Temperature. While temperature dependent kinetics were important for predicting wetland NO_3-N removal in summer, its influence inhibited the simulated removal significantly outside the growing season. As denitrification in wetlands can still occur under cold conditions (Kovacic et al., 2000), the simulated influence of temperature outside the growing season may be too strong and other factors, notably organic carbon availability and dissolved oxygen, may be more important limiting factors.

4. MIKE SHE Limitations

a. Nitrogen fertilizer. Because MIKE SHE only allows soluble water quality sources, nitrogen fertilizer was represented as a source of soluble NH_4-N. This led to a flushing of soluble NH_4-N from the soil profile shortly after application that would have otherwise stayed in particulate form for an extended period or been nitrified at soil temperatures above 10 ^oC. The consistent underestimation in stream NO_3-N early in the growing season (Apr-Jun) is partially attributed to this MIKE SHE limitation.

b. ECO Lab. MIKE SHE ECO Lab is primarily intended for solute fate and transport applications. While particulate species can be included in MIKE SHE ECO Lab, it is much more cumbersome to track their mass balance compared to the automatically generated mass balances available for soluble species in MIKE SHE.

c. Plant Uptake Function. The built-in MIKE SHE plant uptake function simulated a linear response between fertilizer rate and crop nitrogen uptake whereas crop nitrogen uptake is expected to diminish at higher fertilizer rates. As a result, residual soil NO_3-N was lower than expected at higher fertilizer rates and NO_3-N losses in subsurface drainage were lower than expected. Future work should consider implementing a plant uptake function in ECO Lab directly that better describes the nonlinear relationship between crop nitrogen uptake and fertilizer rate.

Conclusions

This study used monitoring and modeling to study nitrogen fate and transport at the field, wetland, and watershed scales. Unique high-frequency, continuous water quality monitoring was used to assess wetland NO_3-N removal patterns and dynamics. This study represented one of the first attempts to perform continuous watershed simulations of nitrogen fate and transport to evaluate wetlands for flow and NO_3-N reduction using a physically-based, spatially distributed hydrologic-biogeochemical model. Consistent with findings from previous modeling studies, annual nitrogen components and NO_3-N dynamics were simulated reasonably well under average hydrologic conditions, while simulated NO_3-N dynamics weakened under extreme (wet) hydrologic conditions. Revisions to both the hydrologic and water quality model setups are necessary to improve the reliability and accuracy of simulated discharge, stream NO_3-N dynamics, and wetland NO_3-N removal estimates.

Revisions to current constructed wetland design standards could allow a greater flood reduction benefit to be achieved by these water detention structures. While wetland percent NO_3-N removal was shown through monitoring and modeling to diminish during higher discharge periods, a sizable flood reduction benefit could still be achieved by modifying the normal pool hydraulics. Replacement of a large weir, like the one at the Slough Creek CREP wetland or Site 1 in the Beaver Creek Watershed, with a smaller water control structure resembling a pipe or orifice at the normal pool elevation would more efficiently throttle down incoming flows and better utilize the available flood storage. Additionally, a water control structure, like the one at the Slough Creek CREP wetland and installed at Sites 2-5 in the Beaver Creek Watershed, allows the opportunity to actively rather than passively manage wetland outflows. More active management of the wetland pool could theoretically allow NO_3-N removal and flow attenuation to be optimized on a daily basis. Changes to the wetland normal pool hydraulics would require the dam embankment to be designed adequately to withstand temporary but significant hydrostatic forces above the normal pool elevation, an emergency spillway to ensure the dam was not overtopped during high flow events, and likely an increased buffer area around the perimeter of the wetland to ensure adjacent and upstream agricultural areas are not inundated during storm events.

As the ultimate goal of this research and other like work is to quantify progress of Gulf Hypoxia water quality goals and help guide future conservation practice implementation, continued investment in science-based water research, water monitoring, and water modeling is necessary.

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