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Juan David Perez-Gutierrez

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By

Juan D. Pérez-Gutiérrez

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Water quality monitoring and modeling studies of on-farm water storage systems in a Mississippi Delta agricultural watershed

By

Juan D. Pérez-Gutiérrez

Approved:

Joel O. Paz
(Major Professor)

Mary Love M. Tagert
(Co-Major Professor)

Mohammad Sepehrifar
(Minor Professor)

Ronald L. Bingner
(Committee Member)

Ying Ouyang
(Committee Member)

Fei Yu
(Graduate Coordinator)

Jason M. Keith
Dean
College of Engineering

Name: Juan D. Pérez-Gutiérrez

Date of Degree: August 11, 2017

Institution: Mississippi State University

Major Field: Biological Engineering

Major Professor: Joel O. Paz

Title of Study: Water quality monitoring and modeling studies of on-farm water storage systems in a Mississippi Delta agricultural watershed

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Candidate for Degree of Doctor of Philosophy

Federal and state programs have encouraged farmers in the Mississippi Delta region to implement best management practices (BMPs) to promote soil and water conservation. An on-farm water storage (OFWS) system is a structural BMP that has several potential benefits, namely, the ability to capture and reuse rainwater and tailwater runoff, provide supplemental water for irrigation, reduce groundwater withdrawals, and improve downstream water quality. However, research demonstrating these benefits and providing new insights for downstream water quality improvement and nutrient-rich runoff management is limited. This dissertation addresses these research gaps by examining the ability of OFWS systems to mitigate off-site nutrient movement, analyzing the impacts of rainfall characteristics on the ability of OFWS systems to reduce NO_3-N, studying the hydrological and physical-chemical characteristics of the volume of water exiting an OFWS system, and using the AnnAGNPS model to simulate runoff, nutrient, and sediment loads entering a tailwater recovery ditch and identify the critical contributing areas of non-point source pollution.

Significant seasonal water quality improvements were observed at different locations throughout the OFWS system, and more importantly, highlight downstream nutrient reduction, particularly during winter and spring. However, recurrent and high intensity rainfall events can minimize the system's effectiveness in reducing downstream nutrient pollution. The NO_3-N concentrations observed in the ditch were strongly dependent on antecedent hydrological conditions with characteristics of next-to-last rainfall events playing a more influential role. The nutrient load was greater in winter, as this season produced the highest effluent discharge. Agricultural fields draining to the outlet of the system produced 7.1 kg NO_3-N $ha^{-1}yr^{-1}$ and 2.3 kg TP $ha^{-1}yr^{-1}$ that was discharged with outflow events. AnnAGNPS simulations showed that larger fields coupled with poorly drained soils resulted in higher runoff, and this condition mirrored the annual rainfall patterns. High nitrogen loss was due to fertilization of corn and winter wheat. TP and sediment loss patterns were similar and influenced by the hydrological condition. This study can be used by stakeholders and agencies to better identify where these systems can be implemented to improve water quality and offer a supplemental source of surface water.

DEDICATION

I dedicate this work to my family and friends living in Colombia and to people who believe that turning this world into a better one is possible through the application of science.

I also dedicate this work to Colombia, my country, and its soldiers and citizens fighting against Las FARC; we will beat the enemy: *Fe en la causa!*

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My love goes out to my Mom, who taught me the art of loving, being patient, and working hard, and to my beautiful partner and her daughter, who showed up in my life just on time.

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CHAPTER I

INTRODUCTION

Increased agricultural non-point source pollution (NPSP) originating from the Mississippi River Basin continues to be a major concern for much of the nation. To sustain crops and increase yields, farmers have found part of the solution in the use of fertilizers. The growing use of this agricultural input can be detrimental to aquatic ecosystems when a substantial portion of the fertilizers is transported from croplands to groundwater *via* percolation and to adjacent waterbodies *via* surface and irrigation runoff (Carpenter et al., 1998; Hollinger et al., 2001; Ongley, 1996; Rabalais et al., 2002b; Sims et al., 1998), often reaching coastal ecosystems (Nixon, 1995; Rabalais, 2002; Seitzinger et al., 2002). High nutrient loads result in the excessive growth of phytoplankton and macrophytes, which causes algal blooms where dissolved oxygen is consumed and depleted as bacteria decompose carbon in the dead plant material. The depletion of oxygen below 2 mg/L can cause hypoxia and consequently a shift in the benthic population and its related food chain, resulting in fish kills, loss of aquatic biodiversity, and many other adverse ecological effects (Carpenter et al., 1998; Smith et al., 1999).

This environmental concern has received much attention, resulting in the push for remedial measures which have been initiated by both stakeholders and the scientific community. One example of those measures is the implementation of best management practices (BMPs) at a field and watershed scale. Structural BMPs such as tail-water recovery (TWR) ditches and agricultural ponds (*i.e.,* on-farm water storage systems - OFWS) can collect and store surface runoff and irrigation tail water from farmed lands. This ability to capture and hold water suggests that OFWS systems have the potential of reducing nutrients exported from agricultural watersheds to receiving waterbodies. In addition to the nutrient reduction benefit, these systems are also gaining popularity for their water supply benefits in areas where irrigated agriculture is predominant and groundwater levels are declining. The dual benefit of reducing nutrient pollution and supplying irrigation water is thus important in areas such as the Lower Mississippi River Valley, where agriculture is intensified and strongly depends on irrigation. Farmers and landowners in this region are tasked with the issue of (1) reducing off-site movement of nutrients, which contributes to the hypoxic zone in the northern Gulf of Mexico, and (2) conserving water resources to slow declining groundwater levels in the Mississippi River Valley Alluvial Aquifer (MRVAA), which is the primary source of water for irrigation of crops. Consequently, OFWS systems have been implemented in different areas across the MRVAA, primarily in areas experiencing declines in groundwater levels. According to Rabalais et al. (2002a), an average of roughly 1 million metric tons per year of nitrate, 67% of which originates from agricultural sources, are released into the Gulf of Mexico, causing devastating ecological effects such as "the dead zone" due to hypoxia phenomenon. Similarly, phosphorus has also been suggested as a major contributor to the Gulf hypoxia problem (Sylvan et al., 2006; USEPA, 2007). In addition, the overuse of groundwater from the MRVAA is, on average, nearly 530 m^3 ha^{-1} yr^{-1} (Kebede et al., 2014; Massey, 2010; Wax et al., 2008).

Over the past decade, many researchers have investigated the role of ditches as an individual structural BMP because of their crucial function of linking agricultural watersheds to external ecosystems (Ahiablame et al., 2010; Herzon and Helenius, 2008). Dollinger et al. (2015) collated scientific contributions on the benefits of implementing ditches for agroecological management. Their study classified these benefits into waterlogging control, soil erosion prevention, water quality improvement, flood control, aquifer recharge, and biodiversity conservation. While several studies have addressed the role of ditches in nutrient movement, little attention has been paid to the combined effect of TWR ditches and on-farm reservoirs or their performance as a BMP on agricultural lands. Popp et al. (2004) cited increased profitability and reduced dependence on groundwater when using on-farm reservoirs and tail-water recovery systems in conjunction with other BMPs. Later, preliminary results from Carruth et al. (2014) and Pérez-Gutiérrez et al. (2015) indicated that OFWS could reduce nutrient runoff from farms and that the stored water could be used for irrigation needs. In a recent study, Moore et al. (2015) observed no statistical differences in water quality among sampling points in an intensively used on-farm storage reservoir and its surrounding ditches in the Northeast Arkansas Delta. While these investigations examined OFWS systems, there are still many questions regarding the nutrient removal effectiveness and seasonal water quality variation of OFWS, which would be helpful for making better agricultural management decisions. Therefore, it is important to monitor and analyze the water quality changes in these systems to improve our understanding of how this emerging BMP impacts the environment in terms of downstream nutrient control and water conservation.

The overall goal of this study is to provide new insights into the benefits of OFWS systems in Porter Bayou Watershed by addressing several key questions: (1) What is the seasonal efficiency of OFWS systems in reducing downstream nutrient pollution? (2) What is the effect of antecedent dry time and intensity of rainfall events on the OFWS water quality? (3) What is the volume of discharge water and associated nutrient load exiting an OFWS system? (4) What is the impact of contributing areas on the water, nutrient, and sediment loads entering an OFWS system?

Dissertation structure

This dissertation is a compilation of journal manuscripts submitted or intended for submission to refereed scientific journals. Each manuscript addresses a specific objective for our study site, a farm within the Porter Bayou watershed in Mississippi, US. Chapter 2 examines the seasonal water quality changes in an OFWS system by measuring several physical and chemical constituents at multiple sampling points throughout the system. Chapter 2 was published in the *Agricultural Water Management* journal. Chapter 3 investigates how rainfall characteristics are related to $NO_3 - N$ concentrations in a TWR ditch. Chapter 4 shows the impacts of the hydrological characteristics on the physical-chemical characteristics of effluent from an OFWS system. The objective of the chapter is to quantify the water discharge volume and its associated nutrient load leaving the OFWS system. Chapter 5 uses the AnnAGNPS model to quantify runoff, nutrient, and sediment loads entering a TWR ditch and identify the areas of the agricultural watershed with the highest load contribution to the ditch. Finally, Chapter 6 summarizes the major findings of this dissertation.

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CHAPTER II

SEASONAL WATER QUALITY CHANGES IN ON-FARM WATER STORAGE SYSTEMS IN A SOUTH-CENTRAL U.S. AGRICULTURAL WATERSHED

A paper published in the Agricultural Water Management journal

Juan D. Pérez-Gutiérrez, Joel O. Paz, Mary Love M. Tagert

Abstract

The objective of this study was to investigate the ability of on-farm water storage (OFWS) systems to mitigate off-site nutrient movement in a south-central U.S. agricultural watershed. We examined the seasonal water quality changes in an OFWS system by measuring several physical and chemical constituents at multiple sampling points throughout the system. Water quality sampling occurred every three weeks during the growing season and every six weeks during the dormant season from February 2012 to December 2014. The collected data were grouped into four seasons and then analyzed using boxplots along with the Wilcoxon and Kruskal-Wallis rank-sum tests for detecting changes in nutrient concentrations. Significant water quality changes were observed in the OFWS system by season and nutrient species, indicating a variation in downstream nutrient reduction with season. The in-ditch median removal efficiency, from the center of the tailwater recovery ditch to the outlet, was 54% during winter and 50% during spring for NO_3-N; 60% during spring for NH_3-N; 26% during autumn and 65% during winter for ortho-P; and 31% during winter and 10% during spring for TP. The in-pond

median concentration removal efficiency was ~77% during summer for NO_3-N, while the concentration remained stable during winter, spring and autumn; 53% from winter to spring and 58% from spring to summer for NH_3-N; 70% from winter to spring for ortho-P, while remaining stable during the other seasons; and 28% from winter to spring and 55% from spring to summer for TP. Our results support the hypothesis that OFWS systems could mitigate downstream nutrient-enrichment pollution, especially during spring. The results obtained from this study offer a better insight into the behavior of OFWS systems and help enhance the management of agroecosystems from an ecological and hydrological perspective for water quality pollution control and water resource conservation.

Introduction

It is widely accepted that agricultural practices have become a significant contributor of pollutants that adversely alter the natural cycle of nutrients, especially for nitrogen and phosphorus (Schlesinger, 1991; Vitousek et al., 1997; Smith et al., 1999). This alteration is derived in large part from the dramatic increase in the use of fertilizers needed to maintain agricultural profitability and higher yields, which is required to feed a growing global population (*i.e.*, roughly 77 million individuals per year according to the US Census Bureau, 2015). In 2012, the world consumption of fertilizers reached nearly 120 and 46.5 million tons of nitrogen and phosphorus per year, respectively (FAO, 2015). The increasing use of fertilizers could be detrimental to aquatic ecosystems as a substantial portion of the nutrient inputs is transported from croplands to groundwater *via* percolation and to adjacent waterbodies *via* surface and irrigation runoff (Ongley, 1996; Carpenter et al., 1998; Sims et al., 1998; Hollinger et al., 2001; Rabalais et al., 2002b), often reaching coastal ecosystems (Nixon, 1995; Rabalais, 2002; Seitzinger et al., 2002). The over enrichment of nutrients in waterbodies stimulates eutrophication, which is the most common leading factor in the deterioration of aquatic ecosystems. Additional nutrients result in the excessive growth of phytoplankton, macrophytes, and toxic algal blooms, and dissolved oxygen is consumed and depleted as bacteria decompose carbon in the dead plant material. The depletion of oxygen can cause hypoxia and as a consequence a shift in the benthic population and its related food chain, resulting in fish kills, loss of aquatic biodiversity, and many other adverse ecological effects (Carpenter et al., 1998; Smith, 1999; Smith, 2009).

Agriculture in the southern United States, specifically in the Mississippi Delta region (MDR), faces two major challenges to maintain a high level of productivity while preserving the surrounding ecosystem's health: (1) off-site movement of nutrients contributing to the development of the hypoxic zone in the northern Gulf of Mexico, especially during the spring season, and (2) the declining groundwater levels in the Mississippi River Valley Alluvial Aquifer (MRVAA). According to Rabalais et al. (2002a), an average of roughly 1 million metric tons per year of nitrate, 67% of which originates from agricultural sources, are released into the Gulf of Mexico, causing devastating ecological effects such as "the dead zone" due to hypoxia phenomenon. Similarly, phosphorus has also been suggested as a major contributor to the Gulf hypoxia problem (USEPA, 2007; Sylvan et al., 2006). In addition, the overuse of groundwater from the MRVAA is, on average, nearly 530 m^3 ha^{-1} yr^{-1} (Kebede et al., 2014; Massey, 2010; Wax et al., 2008). To address these complex environmental issues, stakeholders and the scientific community have been promoting control measures such as the implementation of Best Management Practices (BMPs) at the field and watershed scale. One of these BMPs is the on-farm water storage (OFWS) system, which has been gaining popularity in agriculturally-dominated regions such as the MDR. By combining tail-water recovery (TWR) ditches and on-farm reservoirs, an OFWS system becomes a structural BMP that collects and stores surface runoff and irrigation tail water from farmed lands. Therefore, OFWS systems have been suggested to have the potential of (1) reducing nutrients exported from agricultural watersheds to receiving waterbodies and (2) providing an alternative source of water for the irrigation of cropped fields, which require to be adequately investigated.

Over the past decade, many researchers have investigated the role of ditches as an individual structural BMP because of their crucial function of linking agricultural watersheds to external ecosystems (Herzon and Helenius, 2008; Ahiablame et al., 2010). Dollinger et al. (2015) collated the vast majority of scientific contributions focused on the benefits of implementing ditches for agroecological management. Their study classified these benefits into waterlogging control, soil erosion prevention, water quality improvement, flood control, aquifer recharge, and biodiversity conservation. While several studies have addressed the role of ditches in nutrient movement, little attention has been paid to the combined effect of TWR ditches and on-farm reservoirs (i.e., an OFWS system) or their performance as a BMP on agricultural lands. Popp et al. (2004) cited increased profitability and reduced dependence on groundwater when using on-farm reservoirs and tail-water recovery systems in conjunction with other BMPs. Later, preliminary results from Carruth et al. (2014) and Pérez-Gutiérrez et al. (2015) indicated that OFWS could reduce nutrient runoff from farms and also that the stored water could be used for irrigation needs. In a recent study, Moore et al. (2015) observed no statistical differences in water quality among sampling points in an intensively used on-farm storage reservoir and its surrounding ditches in the Northeast Arkansas Delta. While these investigations examined OFWS systems, there are still many questions regarding the system's nutrient removal effectiveness and seasonal water quality variation, which are necessary for making better agricultural management decisions. Therefore, it is important to monitor and analyze the water quality changes in these systems to improve our understanding of how this emerging BMP impacts the environment in terms of downstream nutrient control and water conservation.

The objective of this study was to investigate the mitigation of nutrient runoff from a south-central U.S. agricultural watershed implementing an OFWS system, by examining the spatial and temporal variations of water quality occurring at sampling points located throughout the system. With the goal of measuring downstream nutrient reduction, we tested the hypothesis of water-quality statistical differences between the sampling points by season, using a suitable non-parametric approach.

Materials and methods

Study area

The monitored OFWS system is implemented on a farm located in the central portion of the MDR within the headwater region of the Porter Bayou watershed (PBW; Figure 2.1), north of Indianola, Mississippi. The PBW extends from latitude 33°26'41" to 33°51'40" north and longitude 90°48'54" to 90°31'34" west, covering nearly 506.2 km^2, most of which are cultivated, producing mainly soybeans and corn (MDEQ, 2012). The topography of PBW is relatively flat with elevations ranging from 90 to 150 m. From

2012 to 2014, the observed total monthly precipitation ranged from about 200 to 600 mm and primarily occurred from early autumn to late spring (Figure 2.2), when the runoff was usually high. The monthly average temperature ranged from 16.7 °C during winter to 26.7 °C during summer (Figure 2.2). More information about the watershed can be found at MDEQ (2008; 2012).

The soils on the 110-ha fields surrounding the monitored system are comprised of several soil types namely, Alligator silty clay loam (24.1%), Forestdale silty clay loam (21.1%), Dowling overwash phase (17.9%), Forestdale silt loam (14.9%), and Dowling clay (13.9%). The soils are exposed during the dormant season, and a soybean-corn crop rotation with conventional and non-tillage practices covered the farm during the growing season for the monitoring period. Typically, nitrogen was applied during early spring, while phosphorus was applied during the fall.

Field sampling and analytical techniques

For water quality data acquisition, an edge-of-field monitoring network was established in 2012 in the OFWS system at our study site (Figure 2.1). The network consists of four sampling points within the system: (1) the inlet, M1; (2) TWR ditch, M2; (3) the outlet, M3; and (4) the pond, MP. Table 2.1 provides the main characteristics of the monitored OFWS system. Sample collection was conducted from March 2012 to December 2014 every three weeks during the growing season (March to October) and every six weeks during the dormant season.

Manual samples were collected in high density polyethylene bottles according to EPA Method 600/4-82-029 (USEPA, 1982). Samples were analyzed *in situ* for potential of hydrogen, pH (pH units); electrical conductivity, EC ($\mu S\ cm^{-1}$); dissolved oxygen, DO (mg L^{-1}); temperature, T (ºC); and *ex situ* for nitrate nitrogen, NO_3-N (mg L^{-1}); ammonia nitrogen, NH_3-N (mg L^{-1}); orthophosphate, ortho-P (mg L^{-1}); total phosphorus, TP (mg L^{-1}); total kjeldahl nitrogen, TKN (mg L^{-1}); and total suspended solids, TSS (mg L^{-1}). *In situ* parameters were measured using a Thermo Scientific Orion Star A329 Portable Multiparameter meter (Thermo Fisher Scientific Inc., Waltham, MA). Before conducting field measurements, all sensors (*i.e.*, Thermo Scientific Orion Ross Ultra pH/ATC Triode for pH, DuraProbe® 4 cell Conductivity Electrode Graphite for EC and T, and Thermo Scientific Orion RDO® Rugged Dissolved Oxygen Sensor for DO) were calibrated relative to their corresponding standard. Field conditions were recorded in a logbook, and samples were immediately stored at 4°C in an ice-filled cooler for transport to the Agricultural and Biological Engineering Water Quality Laboratory at Mississippi State University for analysis. Samples were analyzed for TSS using 0.7-µm particle size glass fiber filters and EPA Method 160.2 (USEPA, 1979). TNT plus™, a prepackaged vial chemistry technique (Hach® Loveland, CO), was used for nutrient analyses, and measurements were automatically read by the Hach® DR 2800™ portable spectrophotometer. For the ortho-P analysis, raw samples were filtered through 0.45-µm pore diameter binderless borosilicate glass microfiber filters. Table 2.2 summarizes the methods used for chemical analysis and their corresponding EPA compliance monitoring code. Samples were subsequently preserved by adding 2 mL of concentrated H_2SO_4 per liter of raw sample and immediately transferred to the Mississippi State Civil and Environmental Engineering Laboratory for TKN analysis, following EPA Method 351.4 (USEPA, 1979).

Water quality data analysis

Water quality data were grouped into four seasons: winter, Wi (December 22 – March 20); spring, Sp (March 21 – June 21); summer, Su (June 22 – September 22); and autumn, Au (September 23 – December 21). To detect water quality changes, we used graphical and statistical analyses. For the graphical analysis, box-and-whisker plots, or boxplots, were used at the seasonal scale for each sampling point and water quality constituent. This type of approach is useful for comparison between data sets and for a visual determination of whether data fit the assumptions of a statistical test procedure (USGS, 1989). A boxplot summarizes the distribution of data by displaying the median, the variability, the skewness, and the non-typical values. In this study, boxplots were set at 90th (the upper whisker), 75th (the upper quartile), 50th (the median), 25th (the lower quartile), and 10th (the lower whisker) percentiles. Outliers were considered those observations 1.5 times beyond the 25th and 75th percentiles. The statistical analyses were conducted to test the significance of the detected changes in the water quality at the sampling points throughout the OFWS system. In this study, these changes were examined using the median because it is a resistant measure of the center of frequency in the presence of outliers. Therefore, the nonparametric Wilcoxon (Wilcoxon, 1945) and Kruskal-Wallis (Kruskal and Wallis, 1952) rank-sum tests were applied. The former is a test for whether the medians of independent samples of two data sets are similar or not. The latter extends the Wilcoxon rank-sum principle to three or more data sets. These two methods are appropriate when normality assumptions are violated and censored data are present in the data distributions (Helsel, 2012). The p-values ≤ 0.1 were considered statistically significant. MATLAB® and the Statistical Toolbox™ (The MathWorks, Inc., Natick, MA) were used to perform all mathematical and statistical calculations.

Results and discussion

Seasonal variability of nitrogen species

Nitrate nitrogen

Figure 2.3a shows the seasonal variability of the OFWS nitrate nitrogen concentration. In general, we found that the median NO_3-N concentration increased from winter to spring, decreased from spring to summer, increased from summer to autumn, and remained fairly stable from autumn to winter. We also found that the summer concentrations were close to 0.23 mg L^{-1} and did not change spatially or temporally. These findings suggest that the changes in NO_3-N concentration might greatly depend on the current hydrologic and hydrodynamic characteristics of the system. The movement of pollutants through the ditch from late autumn to early spring may be governed by a combination of advective and diffusive transport processes, primarily driven by the rainfall occurring during that time frame. In contrast, diffusive processes could have dominated during summer and early autumn when precipitation was minimal. This shifting from semi-lotic to shallow lentic conditions, which ultimately would increase the system's residence time, in conjunction with the summer warmer temperatures may have stimulated biogeochemical transformations of nutrients (Lillebø et al., 2007). Such transformations might predominantly occur on the system's bottom sediments and biofilms (Peterson et al., 2001). A study conducted by Moore et al. (2015) in an on-farm storage system in the northeast Arkansas Delta noted no statistical differences in water quality between the ditches or between the two ditches and the reservoir, which compared well with the results that were obtained for summer in this study. In contrast, we found statistical evidence of higher NO_3-N concentrations during winter, spring, and autumn in the Mississippi OFWS system. Although the study conducted by Moore et al. (2015) was limited by the number of samples analyzed, we do not know why there is a discrepancy for three seasons. However, we can infer that the spatial similarities in summer NO_3-N concentrations for the Arkansas and Mississippi studies might be due to plant uptake and decreased NO_3-N concentrations because of little rainfall.

The inlet (M1 location) showed, on average, the same median NO_3-N concentration throughout the winter, spring, and autumn seasons (i.e., around 0.45 mg L^{-1}; $p - value = 0.34$). Despite such similarities, spring and autumn reported a larger variability (0.23 – 4.53 mg L^{-1} and 0.23 – 2.82 mg L^{-1}, respectively) than winter (0.023 – 1.71 mg L^{-1}). Summer concentrations showed little variation and remained close to the median (0.03 mg L^{-1}). Compared with the M1 location, median concentrations in the TWR ditch (M2) were (i) slightly over fourfold higher during winter and spring ($p - value < 0.01$), (ii) the same during summer ($p - value = 0.08$), and (iii) twofold larger during autumn ($p - value = 0.1$). At the M2 location, the 90th percentile concentration occurred during spring, which was slightly higher than 5 mg L^{-1}. These results suggest that higher amounts of NO_3-N may have entered the system *via* surface runoff from the fields that drain into the TWR ditch, especially during the rainy season. This result was expected as roughly 40% of the total annual precipitation across the study area occurred from March to June during the period of sampling (Figure 2.2), and the dominant soil series are classified as having a very high runoff potential under the runoff class property. In addition, according to Randall et al. (1997), under continuous corn and corn-soybean rotation, the evapotranspiration rate is limited relative to cover crop systems, leading to higher runoff along with major nutrient losses, conditions that could likely be mirrored in our system.

The M3 location, the outlet of the system, exhibited a similar median NO_3-N concentration (about 0.9 mg L^{-1}; $p - value = 0.58$) during winter, spring, and autumn. Although the NO_3-N concentration varied considerably through autumn, results showed the maximum concentration during spring exceeding that from autumn by a factor of 1.6. When compared with M2, the median NO_3-N concentration at M3 was reduced by 54% during winter (although it was statistically significant only at $p - value = 0.17$) and 50% during spring ($p - value < 0.01$), likely due to potential biological assimilation and denitrification processes (Peterson et al., 2001). Our results agree with those from Fu et al. (2014), who investigated the nutrient mitigation capacity of two agricultural ditches (constructed and traditional) in China. Results from that study reported removal efficiencies of 57% and 21% for the NO_3-N concentrations in a constructed and traditional ditch, respectively. Both ditches were hydro-geomorphically similar in length, width, and slope; however, the constructed ditch was enhanced with geogrid, geotextile, and fine and coarse gravel. Another similar study conducted by Littlejohn et al. (2014) found a 25% reduction in the median NO_3 load in a ditch containing low-grade weirs for nutrient removal. They attributed this low removal percentage to potential high nitrification rates overwhelming the NO_3-N concentration reduction in the system studied. In an earlier investigation, Moore et al. (2010) compared the nutrient reduction potential of a vegetated and non-vegetated agricultural ditch in the Mississippi Delta. Under a simulated storm event, they reported NO_3-N load reductions up to 74% and 78%

in the vegetated and non-vegetated ditch, respectively. In a more recent investigation conducted in planted mesocosms, Taylor et al. (2015) reported a 68% and 61% reduction in NO_3-N load in the vegetated and non-vegetated treatment, respectively. Several researchers have found similar nitrate removal efficiencies in wetlands that drain agricultural areas (Fink and Mitsch, 2007; Jordan et al., 2011; Reddy et al., 1982; Woltemade, 2000).

At the pond sampling point, MP, there were no significant differences among the median NO_3-N concentrations (slightly higher than 1 mg L^{-1}; $p - value = 0.98$) during winter, spring, and autumn. Summer NO_3-N concentrations remained fairly stable and close to the detection limit (0.23 mg L^{-1}). Consistent with results from Moore et al. (2015), we found that the median NO_3-N concentration in the pond during autumn was significantly higher (by a factor of ~4; $p - value < 0.01$) than during summer. While winter and spring NO_3-N concentrations showed similar variability at the MP location, higher concentrations were more frequent during winter (close to 2 mg L^{-1}). When comparing median concentrations over seasons, results from this study show that the pond's removal efficiency for NO_3-N was more than 77% during summer. We not only hypothesize that the sediment denitrification rates in the pond could be higher during summer, as noted in other studies (David et al., 2006), but we also believe that primary production might have been a contributing factor as well, controlling the in-pond inorganic nitrogen during the warmer months (Figure 2.2). Conversely, during the cooler months, these two biogeochemical sinks for NO_3-N might be minimal so that the NO_3-N remained primarily in the water column.

Ammonia nitrogen

Figure 2.3b shows the seasonal variability of ammonia nitrogen concentrations throughout the OFWS system. At M1, the variability of the NH_3-N concentration was relatively similar among seasons, except during autumn, when the variability covered a broader interval (0.015 – 0.213 mg L^{-1}, the 10th and the 90th percentile concentrations, respectively). The median NH_3-N concentrations showed no significant changes during winter, summer, and autumn (0.06 mg L^{-1}; $p - value = 0.96$). In contrast, the median NH_3-N concentration during spring was significantly lower ($p - value = 0.09$), which might be explained by dilution processes. A marginal decrease in the median NH_3-N concentration was observed from winter to spring (by a factor ~3), and a modest increase was seen later during summer and autumn (by a factor 2.8). The highest 90th percentile NH_3-N concentration occurred during autumn, reaching levels up to 0.213 mg L^{-1}. At M2, the highest median NH_3-N concentration occurred during spring (0.224 mg L^{-1}), followed by autumn and winter (around 0.1 mg L^{-1}). These results indicate that the main source of NH_3-N likely entered the system from the fields draining to the TWR ditch, similar to the pattern observed with NO_3-N concentrations. We also found a noticeable decline, as much as 79%, between spring and summer NH_3-N concentrations. During summer, the NH_3-N concentrations exhibited negligible variability over a narrow interval (0.015 – 0.098 mg L^{-1}, the lower and the upper quartile concentrations). As discussed by Dodds et al. (1991), ammonium is energetically preferable to nitrate in photosynthetic and heterotrophic assimilation. Thus, we hypothesized that the aggregated effect of biological assimilation, sorption to sediments, nitrification, and high solar radiation may have been responsible for the low NH_3-N concentrations observed in the OFWS system during late spring, summer, and early autumn. We also observed that the median concentration of NH_3-N increased by a factor of 2.2 from summer to autumn, which could be attributed to the sorbed and regenerated NH_3-N released to the water column from the stream bottom (Peterson et al., 2001) and to the remaining portion of fertilizers flushed by runoff from the field after harvest.

At M3, the outlet of the system, the median NH_3-N concentration was fairly constant (around 0.11 mg L^{-1}; $p - value = 0.46$) throughout winter, spring, and autumn; whereas, the summer median concentration was nearly twice as low. During the spring season, the NH_3-N concentration largely varied, reaching a maximum value of 0.423 mg L^{-1}. When compared with M2, the median NH_3-N concentration at M3 showed a reduction of 60% during spring ($p - value = 0.03$). This measured reduction was high relative to previous studies in a constructed ecological and traditional soil ditch in China (49% and 12% on average, respectively (Fu et al., 2014)). However, our result was lower than the 71% reduction efficiency reported by Moore et al. (2010) after a single simulated runoff event conducted on a non-vegetated ditch in the MDR. The 60% NH_3-N reduction efficiency in this study is also slightly lower than the estimated 67% reduction observed by Littlejohn et al. (2014) in a terraced ditch within the Lower Mississippi Alluvial Valley. Moore et al. (2010) also studied the NH_3-N reduction efficiency in a vegetated ditch, finding the same order of magnitude that we found in our study. When comparing the spring seasonal removal efficiencies for NO_3-N (50%) and NH_3-N (60%) in the TWR ditch, we had better results for NH_3-N. This particular finding is likely due to higher nitrification rates during spring and early summer. The resulting NO_3-N is more water-soluble and thus, more mobile and may have been readily transported downstream

(Bernot and Dodds, 2005). Bernot et al. (2006) also noted this finding when conducting studies in agriculturally influenced streams of the Midwestern US. By measuring the nutrient length uptake (S_w) (i.e., the mean distance a nutrient molecule will travel before being removed from the water column; Stream Solute Workshop, 1990), they found that the NH_3-N S_w was lower than the NO_3-N S_w. Peterson et al. (2001) reported similar findings by examining nitrogen dynamics in 12 headwater streams across the US.

At the MP location, we found similar variability during winter and spring; however, the median NH_3-N concentration during winter was twofold higher than in the spring, and the highest concentration rose slightly over 0.25 mg L^{-1} during the spring season. The median NH_3-N concentration was reduced by 53% from winter to spring and by 58% from spring to summer (although no statistical difference was detected from winter to spring, $p - value = 0.37$; or from spring to summer, $p - value = 0.34$); however, median NH_3-N concentrations were almost fivefold higher during autumn than summer ($p - value = 0.01$) in the pond. These results are consistent with the results observed for the NO_3-N concentration at the MP location confirming that the temperature might be a controlling factor for the in-pond nitrogen species.

Seasonal variability of phosphate species

Orthophosphate

Figure 2.4a shows the seasonal variability of the orthophosphate concentrations through the OFWS system. At the OFWS inlet (M1), the median ortho-P concentrations were low during summer (0.074 mg L^{-1}), high during spring (0.235 mg L^{-1}), and remained moderate during winter and autumn (around 0.2 mg L^{-1}; $p - value > 0.9$). The variability of the ortho-P concentrations increased from winter to spring when the concentration reached a peak value of 0.5 mg L^{-1}. The median concentration dropped by a factor of 3.2 from spring to summer ($p - value < 0.01$), followed by a 2.7 factor increase from summer to autumn ($p - value < 0.01$). The maximum median ortho-P concentration throughout the system (slightly more than 0.913 mg L^{-1}) occurred at M2 during winter. In addition, we observed a marked variability from 0.164 to 0.913 mg L^{-1} in the TWR ditch during the winter season. At M2, the median ortho-P concentration during spring was (i) less than the winter by a ratio of 1:3 ($p - value < 0.01$), and (ii) higher than the summer and autumn by a factor of ~2 ($p - value = 0.02$). These results suggest that in-ditch biotic processes were enhanced by warmer temperatures and light availability. Median ortho-P concentrations were stable around 0.1 mg L^{-1} ($p - value = 0.2$) during summer and autumn. In addition, the distribution of ortho-P concentrations during summer was within a narrow range (0.063 – 0.163 mg L^{-1}). During the autumn, concentrations were more variable (between 0.068 and 0.206 mg L^{-1}). When compared with M1, the median ortho-P concentration was reduced by 49% at M2 ($p - value = 0.05$) and by 26% at M3 ($p - value = 0.09$) during the autumn. Also, when compared with M2, we found that the median ortho-P concentration was reduced by 65% at M3 ($p - value = 0.03$) during the winter. Previous research in the MDR observed lower percentages of ortho-P reduction (i.e., ~14%) in a 500 m-length ditch (Littlejohn et al., 2014). The higher ortho-P removal efficiencies found in this study might be due to the greater in-ditch residence time and concentrations of Ca, Mg, Fe, and Al, which result in a higher rate of P adsorption and precipitation processes (Penn et al., 2007). In a vegetated drainage ditch, Kröger et al. (2008) found a 44% Dissolved Inorganic Phosphorus (DIP) reduction, which is similar to what was observed in the current study.

Other studies have observed much higher removal efficiencies for ortho-P. For instance, Moore et al. (2010) reported a 98% reduction, on average, in DIP load under a simulated storm event conducted on vegetated and non-vegetated drainage ditches in the MDR, which is also similar to the findings of Penn et al. (2007).

The median ortho-P concentration at the M3 location was similar during winter and spring (around 0.2 mg L^{-1}; $p-value = 0.88$), and marginally lower during summer and autumn (around 0.15 mg L^{-1}; $p-value = 0.33$). Median ortho-P concentrations were more variable at M3 during the spring season, when we observed the maximum 90^{th} percentile concentration of 0.46 mg L^{-1}. These findings reflect the impact of rainfall runoff from the agricultural landscape on the adjacent ditch (Figure 2.2).

While having similar low values close to the detection limit during spring, summer, and autumn, the median ortho-P concentration was higher during the winter season ($p-value < 0.01$) at the MP location. This suggests that the pond could act as a sink for P species (observed removal efficiency of more than 70%; $p-value = 0.03$) during the warmer seasons, likely due to sorption of P to the in-pond sediments and P biological uptake.

Total phosphorus

Figure 2.4b shows the seasonal variability of the total phosphorus concentrations through the monitored OFWS system. The inlet sampling point, M1, showed that the median TP concentration increased from winter to spring ($p-value = 0.12$), decreased from spring to summer ($p-value = 0.14$), and remained fairly stable throughout the summer and autumn ($p-value = 0.83$). The M1 TP concentrations largely varied in spring (from 0.113 - 0.82 mg L^{-1}), and while lower, concentrations were also variable during the winter (0.015 - 0.43 mg L^{-1}) and summer (0.107 - 0.523 mg L^{-1}). During autumn, TP concentrations ranged from 0.23 – 0.3 mg L^{-1}. At the M2 sampling point in the TWR ditch, we found that the highest median TP concentrations occurred during winter and spring (around 0.675 mg L^{-1}; $p - value = 0.62$), whereas the minimum median concentrations occurred during summer (0.228 mg L^{-1}) and autumn (0.295 mg L^{-1}). At M3, the outlet of the system, the median TP concentrations showed an increase by a factor of 1.6 from winter to spring, while the median TP concentrations during summer and autumn were almost twofold lower than the spring median concentration. The M3 sampling point had the highest variability of TP during the spring (0.347 – 1.6 mg L^{-1}; $p - value < 0.01$), which had a median concentration of 0.564 mg L^{-1}. The other three seasons exhibited almost the same median TP concentration around 0.3 mg L^{-1}($p - value = 0.16$), with no statistically significant differences. We found that from the M2 sampling point in the ditch to the M3 point at the outlet, the median TP concentration was reduced 31% during winter ($p - value = 0.03$) and 10% during spring ($p = 0.1$). Moore et al. (2010) reported 95% and 86% TP reduction when examining non-vegetated and vegetated ditches, respectively, in the MDR, which are greater than the reductions measured in this study. However, our results are in the 12% to 73% range of TP removal by plant uptake observed by Reddy and Debusk (1985) and Silvan et al. (2004). The TP reductions measured in this study are in line with the findings of Fu et al. (2014), who noted a removal efficiency of 26% in a constructed ditch in Tai Lake Basin, China. Fu et al. (2014) concluded that the physical settlement, plant uptake, and adsorption/desorption of P species could be the most important mechanisms for P removal.

At MP, the median TP concentration was reduced by 28% from winter to spring ($p - value = 0.04$), and by 55% from spring to summer ($p - value < 0.01$). The highest 90th percentile TP concentration of 1.5 mg L^{-1} in the pond was observed during winter. Also, no significant differences were detected between the median TP concentrations during spring and autumn (0.35 mg L^{-1}; $p - value = 0.75$). Again, the median TP concentration during summer was the lowest among seasons (0.16 mg L^{-1}). However, the median TP concentration was almost twofold higher during autumn than during summer ($p - value < 0.01$). In addition to the movement of P *via* soil erosion during runoff experienced at times of high rainfall in late autumn and winter, these results were expected as the P dynamics might have been enhanced by in-pond shallowness and higher solar radiation during late spring, summer, and early autumn. The results of our study indicate that the downstream nutrient reduction can vary with season.

Summary and conclusions

This study examined the water quality changes occurring in the OFWS system implemented at a farm within the PBW, an agricultural watershed in the MDR. Our results provide evidence of significant seasonal water quality changes among the different points monitored throughout the OFWS, and more importantly, highlight downstream nutrient reduction. Our study showed a 54% and 50% reduction in NO_3-N concentration in the TWR ditch during winter and spring, respectively. When comparing median concentrations over seasons, our results showed that the pond's removal efficiency for NO_3-N was more than 77% during summer. A 60% reduction in NH_3-N concentration was measured in the TWR ditch during spring, whereas NH_3-N removal percentages of 53% were observed from winter to spring and 58% from spring to summer in the pond. Orthophosphate concentrations in the ditch were reduced by 49% at M2 and 26% at M3 during autumn, as measured from M1. During winter, the ortho-P concentration was reduced by 65% from M2 to M3. The in-pond ortho-P concentrations removal efficiency was observed to be approximately 70% from winter to spring, remaining stable through the other seasons. Total phosphorus in the ditch, as measured from M2 to M3, was reduced by 31% and 10% during winter and spring, respectively. From winter to spring, the in-pond TP concentration was reduced by 28% and from spring to summer by 55%. The results of this study indicate that the downstream nutrient reduction can vary with season, with significant reductions possible during spring. This variation is of special interest when targeting the effect of nutrient runoff from agricultural fields into the Gulf of Mexico as the dead zone is mainly observed during spring. Our results provide support in favor of the hypothesis that OFWS systems could mitigate downstream nutrient-enrichment pollution. However, enhanced top-of-the-field agricultural management is required in combination with OFWS to decrease nutrient loading downstream, especially during soil-exposed periods when high nutrient runoff is likely to occur. This study provides better insight into the behavior of OFWS systems and helps to improve the management of agroecosystems for water quality pollution control and water resource conservation.

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Table 2.1 Hydro-geometric characteristics of the monitored OFWS system

Hydro-Geometric Feature	**Value**	**Units**
TWR Ditch (Trapezoidal shape)		
Length	818.8	m
Side slope	1.5:1	-
Channel bed slope	0	-
Bottm width	3.6	m
Flow depth	1.8	m
Freeboard	0.3	m
Storage volume	13,320	m^3
On-farm Reservoir		
Depth	2.4	m
Side slope	3:1	-
Surface area	4.45	ha
Bottom width	3.6	ha
Storage volume	114,700	m^3

Table 2.2 Analytical methods used in the nutrient-related chemical analyses at the laboratory scale

Constituent	Chemical Method	Hach® Code	TNT plus™ number	EPA Compliance Monitoring
NO_3-N	Dimethylphenol	10206 - Low Range: 0.23-13.5 mg L^{-1}	835	40 CFR 141
NH_3-N	Salicylate	10205 - Ultra Low Range: 0.015 to 2.0 mg L^{-1}	830	EPA 350.1
Ortho-P	Ascorbic Acid	10209 - Low Range: 0.05-1.5 mg L^{-1}	843	EPA 365.1, 365.3
TP		10210 - Low Range: 0.05-1.5 mg L^{-1}		

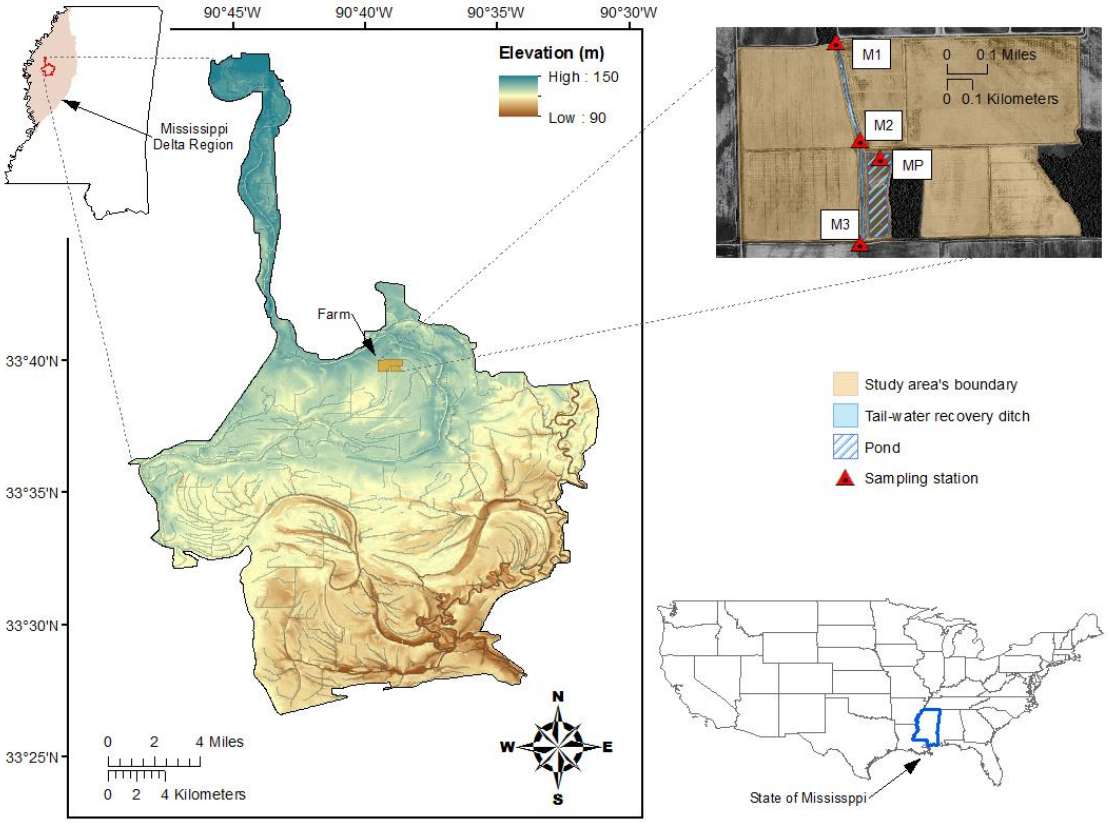

Figure 2.1 Location of the edge-of-field water quality monitoring network at the farm in the Porter Bayou watershed, relative to the Mississippi Delta Region.

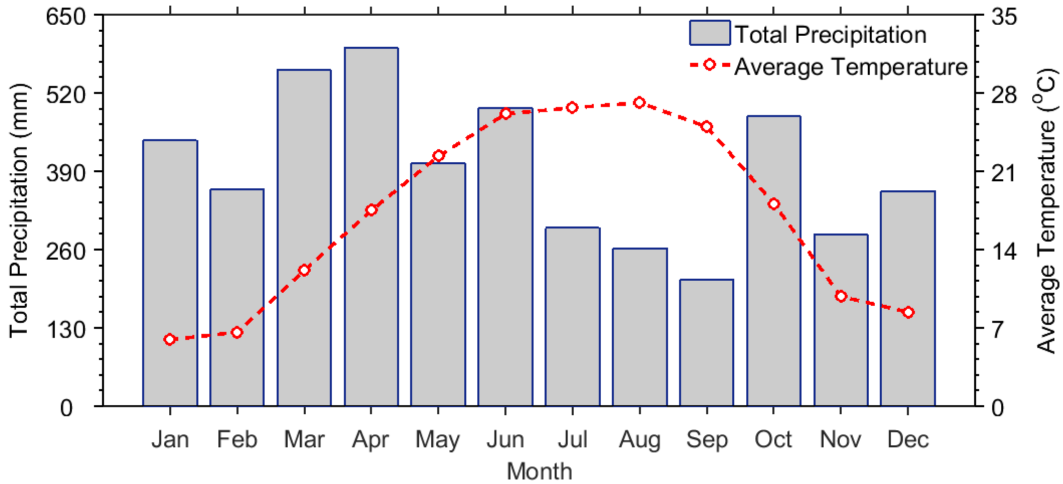

Figure 2.2 Total monthly precipitation and monthly average temperature observed during 2012 to 2014 at the study site. Source: PRISM Climate Group (http://www.prism.oregonstate.edu/explorer/)

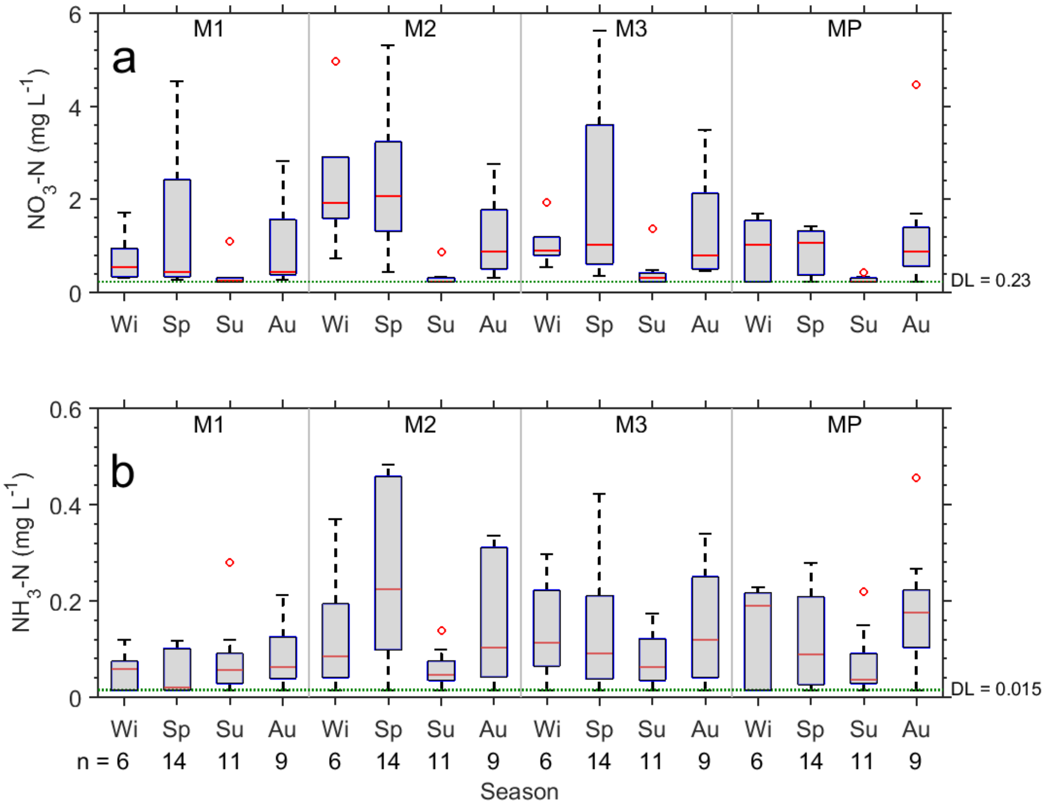

Figure 2.3 Seasonal variability of (a) NO_3-N and (b) NH_3-N in the monitored OFWS system.

M1: the inlet; M2: TWR ditch; M3: the outlet; MP: the pond; DL: detection limit; Wi: winter; Sp: spring; Su: summer; Au: autumn; Outliers are shown as red circles; The number of samples (n) for each grouped dataset is shown below the X axes.

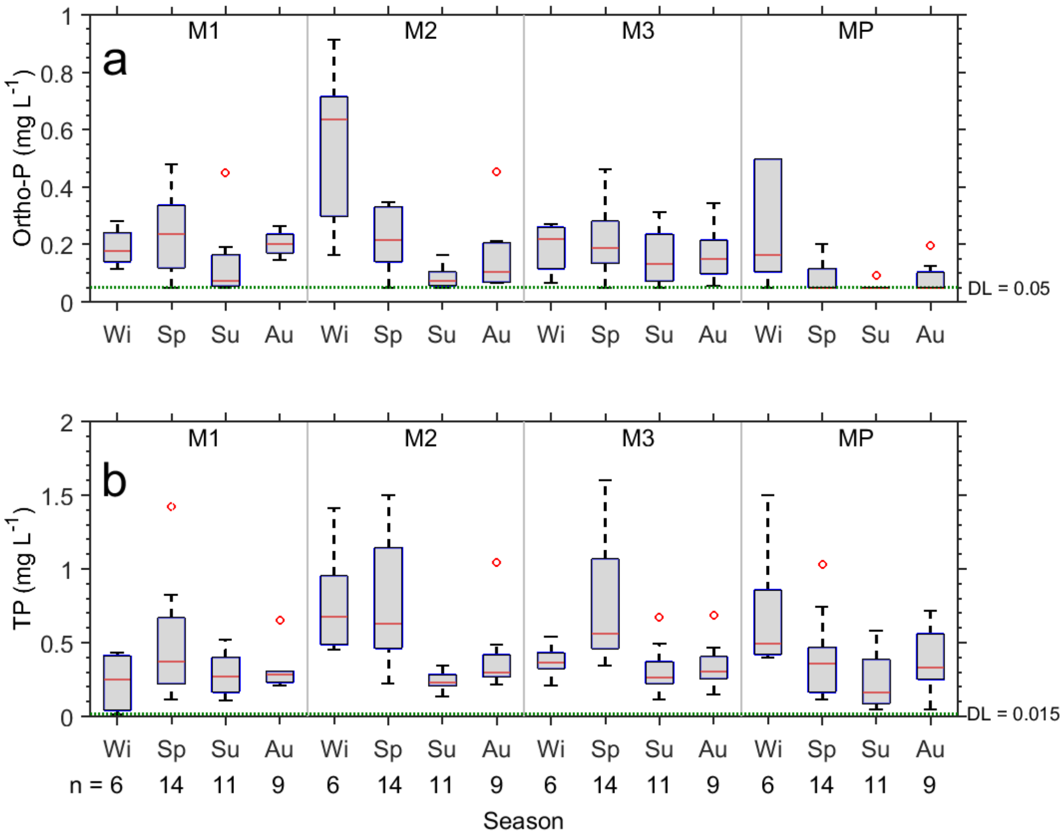

Figure 2.4 Seasonal variability of (a) Ortho-P and (b) TP in the monitored OFWS system.

M1: the inlet; M2: TWR ditch; M3: the outlet; MP: the pond; DL: detection limit; Wi: winter; Sp: spring; Su: summer; Au: autumn; Outliers are shown as red circles; The number of samples (n) for each grouped dataset is shown below the X axes.

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CHAPTER III

ANALYZING THE IMPACTS OF RAINFALL CHARACTERISTICS ON $NO_3 - N$ REDUCTION BENEFITS OF TAILWATER RECOVERY DITCHES

Juan D. Pérez-Gutiérrez, Joel O. Paz, Mary Love M. Tagert, Mohammad Sepehrifar

Abstract

Rainfall characteristics can become a major factor that influences the ability of best management practices to reduce nutrients lost from agriculture to receiving waterbodies. The goal of this study was to look at how rainfall characteristics impact $NO_3 - N$ concentration in a tailwater recovery ditch implemented at a farm within the Porter Bayou watershed in Mississippi, US. To accomplish this goal, we proposed a methodology that correlates rainfall characteristics (*i.e.* a combination of variables such as depth, intensity, frequency, and antecedent hydrological conditions before water sampling) and $NO_3 - N$ concentration monitored in the ditch. Subsequently, a hierarchical clustering approach was implemented to classify rainfall events in the context of the $NO_3 - N$ concentrations. Forty-six rainfall events that matched water sampling dates from May 2012 to March 2016 were selected and analyzed for linear dependence. The rainfall events were grouped into four classes by using the k-means clustering method. For this, the rainfall characteristics that significantly correlated with $NO_3 - N$ concentrations were used as input. Results indicate that the $NO_3 - N$ concentrations observed in the ditch were strongly dependent on antecedent hydrological

conditions within the study area, and specifically on the (1) duration of rainfall events before sampling and (2) characteristics of next-to-last rainfall events. The combined variables of total rainfall depth and frequency showed that classes I and III were likely to have the most impact on the in-ditch $NO_3 - N$ concentration. The effect of class I rainfall events on $NO_3 - N$ concentrations appears to be magnified under higher depth, intensity, duration, and a shorter time before next-to-last rainfall events. In addition, the influence of class-III rainfall events on the $NO_3 - N$ concentrations was driven mainly by high-frequency, low magnitude, and dry antecedent conditions. The results show that next-to-last rainfall events should be considered when understanding the nutrient reduction potential of TWR ditches. The rainfall classes identified by using the *k*-means clustering approach provide information which has significant implications for future design, operation, and management of TWR ditches for more efficient nutrient control strategies. Results of this investigation can help improve nutrient loss management in agricultural landscapes.

Introduction

Modern agriculture depends on fertilizers to boost yields and meet global food requirements. However, excessive use of fertilizers can lead to surface water impairment (Carpenter et al., 1998; Carpenter et al., 2011; Hale et al., 2015; Howarth et al., 2011; Sobota et al., 2013; Sprague et al., 2011; Tilman et al., 2002; Tilman and Clark, 2015). In the United States, specifically within the Mississippi River Basin, effects of additional fertilizers have been of great concern because of their contribution to the hypoxic "dead zone" development in the northern Gulf of Mexico (Porter et al., 2015; Rabalais, 2002; Rabalais et al., 2002a; Rabalais et al., 2002b; Rabotyagov et al., 2014; Turner et al., 2006). Action plans promoted by the Hypoxia Task Force, a federal/state partnership established in 1997, to reduce the size of the bottom-water hypoxic zone seem to have been insufficient, as the goal of reducing the dead zone size to less than 5,000 km^2 by 2015 was not achieved (USEPA, 2015). Instead, this area was on average three times larger (i.e., 15,478 km^2) from 2001 – 2015 than the targeted size, and up to 16,760 km^2 during mid-summer 2015 (Data Source: Nancy N. Rabalais, LUMCON, and R. Eugene Turner, LSU). Best management practices (BMPs), which complement such action plans, are implemented in agroecosystems to protect the environment from potential agriculturally-driven threats, and to preserve healthy soils and water resources (Khanal and Lal, 2015; Lemke et al., 2011; Liu et al., 2013; Yates et al., 2007). Such practices, which can be classified as structural or non-structural, vary with the type of pollutant targeted for control and the landscape characteristics where the BMP will be placed. Also, the success of BMPs to mitigate the loss of agricultural pollutants can be altered by hydrological conditions and physical field characteristics (Her et al., 2017; Rittenburg et al., 2015). Therefore, understanding the response of BMP effectiveness under hydrological conditions varying with time becomes a prerequisite for maximizing BMP performance.

Hydrological conditions, such as rainfall, can activate flow paths that connect agroecosystems to BMPs and affect the intent of such practices to control non-point source (NPS) pollution (Rittenburg et al., 2015). Many field-scale studies have investigated this influence under a range of environmental conditions and for several pollutants. For example, under an ambient-precipitation condition on a poorly drained Webster clay loam in Minnesota, Randall and Iragavarapu (1995) reported a 25-fold higher average loss of kg $NO_3 - N$ ha^{-1} from conventional tillage plots during the 3-yr wet period (1990 – 1992) as compared with the three-year dry period (1987 – 1989). Later, Kreuger (1998) found soil and weather conditions to be major influencing factors in the the loss of pesticides to stream water from an agricultural catchment characterized by the maritime climate in southern Sweden. Kleinman et al. (2006) simulated two rainfall intensities (2.9 and 7.0 cm h^{-1}) on an alfalfa (*Medicago sativa* L.) – orchardgrass (*Dactylis glomerata* L.) field in eastern USA. They reported that larger runoff volumes from the foot-slope position plots produced 60% and 181% higher losses of total P and N, respectively, than did runoff from the mid-slope plots under spring-time conditions. In a temperate area, Delpla et al. (2011) investigated the impact of 8 rainfall events on runoff water quality by an adapted monitoring over three maize cropped plots (one control plot with no amendment, and two plots fertilized with cattle manure and pig slurry). They showed that the export of dissolved organic carbon was proportional to the rainfall event intensity. Recently, Wang et al. (2015) investigated rainfall-induced nutrient losses after manure fertilization of three experimental plots (50%, 100%, 150%) from farmland in an alluvial plain, citing gradually increasing loads of total nitrogen and total phosphorus with fertilizer amount. While these studies expand our knowledge of the rainfall implications on water quality in agroecosystems, researchers lack an understanding of the effects of rainfall characteristics on the performance of BMPs. Addressing this gap in knowledge is critical to the future management of nutrient loss in farmlands.

A tailwater recovery (TWR) ditch is a structural BMP that has gained acceptance in recent years because of its primary use in agricultural landscapes for downstream nutrient reduction and water conservation (Pérez-Gutiérrez et al., 2017b). The ditch collects and stores surface and irrigation runoff from cropland and plays a fundamental role in trapping nutrients lost from agriculture to receiving waterbodies (Moore et al., 2015; Omer et al., 2016; Pérez-Gutiérrez et al., 2015; Pérez-Gutiérrez et al., 2017b). However, according to Pérez-Gutiérrez et al. (2017b), nutrient reduction efficiency in TWR ditches varied significantly with season. These results are likely due to the potential role that rainfall characteristics play on the ability of TWR ditches to control and trap nutrients. Therefore, assessments of how rainfall characteristics alter the nutrient reduction efficiency of TWR ditches are necessary for maximizing the environmental benefits of this BMP.

The goal of this study was to examine the influence of rainfall characteristics on the in-ditch nutrient concentration by analyzing nine variables describing the rainfall events before water quality sampling in the ditch. In particular, we investigated how rainfall characteristics are related to $NO_3 - N$ concentration in a TWR ditch implemented at a farm within the Porter Bayou watershed in Mississippi, US. The variables were input to a clustering approach to classify the rainfall events in the context of the measured $NO_3 - N$ concentrations. Results of this investigation should improve the management of nutrient loss in agricultural landscapes.

Materials and methods

Study area

The study area is in the Porter Bayou watershed (PBW; 33°26'39" – 33°51'38" N, 90°48'54" – 90°31'34" W), a 506.2 km^2 watershed located within the MDR (Figure 3.1). Rainy weather typically dominates this area during winter and spring, and dry weather during summer and early fall seasons. Mean annual rainfall was 1220.6 mm from

2013 to 2015 with an average air temperature of 17.7 °C. According to the Soil Survey Geographic database, the watershed is covered primarily by Forestdale (28.6%), Dowling (21.37%), Dundee (17.96%), Alligator (12.34%), Pearson (5.28), and Sharkey (5.05%) soil types, which in general are poorly drained soils. Land use in PBW is predominantly soybean, corn, and rice production.

Within the PBW, a TWR ditch at Pitts farm (Figure 3.1) was monitored for water quality and rainfall from May 2012 to March 2016. The ditch has two trapezoidal channels that come together to form a Y-shaped feature. Both channels have 1.5:1 side slope, 1.83 m maximum depth, and 0.305 m freeboard. The channel running from north to south is 430 m long with a total storage capacity of 8,140 m^3, whereas the channel running from northeast to southwest is 930 m in length with a total storage capacity of 16,920 m^3.

Water quality and weather data acquisition

An edge-of-field monitoring network was established at the study site in 2012 to collect water quality and weather data. The network consists of five water sampling points and one weather station defined as follows (Figure 3.1): (1) first inlet, P1; (2) mid-ditch, P2; (3) outlet, P3; (4) second inlet, P4; (5) pond, (PP); and (6) weather station, (WS). From May 2012 to March 2016, 1-L water grab samples were collected at the sampling sites every three weeks during the growing season (March to October) and every six weeks during the dormant season. Samples were placed in a cooler at 4 °C for transport from the field to the Agricultural and Biological Engineering Water Quality Laboratory at Mississippi State University. All water samples were analyzed within 24 h after collection for $NO_3 – N$ (mg L^{-1}) by the dimethylphenol method using TNT plus^{TM}, a prepackaged vial chemistry technique (Hach® Loveland, CO), and read by the HACH® DR 2800™ portable spectrophotometer. In this study, $NO_3 - N$ results for samples collected at P2 sampling point were used. Rainfall data were recorded automatically at 15-minute intervals by a WatchDog 2700 Weather Station (Spectrum® Technologies, Inc., Aurora, IL) deployed at the study site. The monitoring program established in this investigation followed the quality assurance recommendations from USEPA (2002).

Data analysis

To examine the influence of rainfall characteristics on in-ditch $NO_3 - N$ concentration, we investigated time series rainfall data recorded and water quality data collected. Also, cumulative rainfall was used to identify the rain events that occurred before each sampling event. In this study, individual storm events were defined by 2 hours without rainfall. This defined period will help understand the effect of consecutive rainfall events on the in-ditch $NO_3 - N$ concentration. Five rainfall characteristics describing rainfall events before sampling and four characteristics of next-to-last rainfall events (Table 3.1) were established and used with the $NO_3 - N$ concentration to conduct an analysis of linear dependence utilizing Kendall's tau correlation coefficient. *P*-values equal to or smaller than 0.05 were considered statistically significant.

To identify what class or categories of rainfall events were linked with certain levels of monitored NO_3-N concentration, a hierarchical clustering approach was conducted using the results of the correlation analyses as input. In simple terms, a clustering method partitions a data set based on similarities of the data. The method has been widely used in various scientific fields (Anderberg, 2014) including hydrology, especially for rainfall classification analyses (dos Santos et al., 2016; Fang et al., 2012;

Peng and Wang, 2012; Wei et al., 2007). In this study, and following the methods of Wei et al. (2007), the *k*-means clustering was applied to the dataset, and trials were performed until the most suitable clusters appeared. Significant differences between resulting clusters (hereafter classes) were tested using the Wilcoxon rank-sum method, and *p*-values equal to or smaller than 0.05 were considered statistically significant. The Shapiro-Wilk test was used to verify the assumption of normality of the data set. Also, when performing the *k*-means clustering method, the surrogate for water quality data that fell below the detection limit was one-half the detection limit. All the data analyses were performed using Matlab® and the Statistics and Machine Learning Toolbox™ (MathWorks Inc, 2015).

Results and discussion

Rainfall and water quality time series

The temporal changes in rainfall and $NO_3 – N$ concentration monitored in the ditch are shown in Figure 3.2. During the study period, the hourly rainfall recorded by the weather station was, on average, 0.15 mm and as high as 56.4 mm, and the total measured rainfall was 4653 mm. In general, more rainfall was observed during winter and spring seasons (from December to May). The mean of the 46 $NO_3 – N$ observations was 1.02 mg L^{-1} and ranged from 0.23 mg L^{-1} to 4.23 mg L^{-1}. Overall, $NO_3 – N$ concentrations were higher during the pre-growing (February – May) and post-harvest periods (September – November). Not surprisingly, $NO_3 – N$ concentrations spiked during periods of abundant rainfall and declined when rain virtually ceased. For example, the observed $NO_3 – N$ concentration rose from 0.294 mg L^{-1} on November 29, 2012 to 3.43 mg L^{-1} on March 7, 2013, which corresponds to more than a ten-fold increase.

Afterward, the concentration declined to 1.9 mg L^{-1} on April 9, 2013, and then increased up to 4.02 mg L^{-1} on May 6, 2013. During this short period, successive rainfall events resulted in 771 mm of rain which, in addition to possible interaction with fertilizer applications during this time period, might have caused the high $NO_3 - N$ levels observed in the ditch. Conversely, periods during which very little rainfall occurred, e.g. 26 mm from June 25, 2015 to October 23, 2015, the $NO_3 - N$ concentration was consistently low (approximately 0.23 mg L^{-1}). Recent studies have shown similar patterns of increased N in streams when climate transitions from drought to wet conditions (Loecke et al., 2017; Van Metre et al., 2016). However, in our study, an exception to this pattern was found on December 5, 2013 when a peak in $NO_3 - N$ concentration (3.9 mg L^{-1}) was observed during a period without rain from October 16, 2013 to December 13, 2013. This unusual event might be due to rainfall occurring north of the system (undetected by the weather station), causing upstream nutrient loads to be transported through the downstream drainage network. In summary, the magnitude of in-ditch $NO_3 - N$ concentrations was consistently related to changes in the slope of the cumulative precipitation line as shown in Figure 3.2. These findings, therefore, indicate that the observed $NO_3 - N$ concentration in the ditch was directly influenced by the local rainfall characteristics.

Classification of rainfall events based on their influences on $NO_3 - N$ concentrations

Using the rainfall data collected from May 2012 to March 2016, 46 out of 620 rainfall events were identified as being linked to the $NO_3 - N$ concentration of the samples monitored in the ditch. These events were the input records of the linear dependence analysis using Kendall's tau correlation coefficient. The resulting correlation matrix of the nine rainfall characteristics and $NO_3 - N$ concentrations is shown in Table

3.2. The in-ditch $NO_3 - N$ concentration was found to be significantly correlated with six out of nine rainfall characteristics. Of the six significantly correlated rainfall characteristics, two corresponded to the rain event before water quality sampling (*i.e.* RDuEP and TBRS), and the remaining four were associated with the next-to-last rainfall event (*i.e.* DNRE, INRE, DuNRE, TBNRE). As expected, RDuEP showed a direct association with $NO_3 - N$ ($r = 0.248$; $p - value = 0.028$), indicating that the duration of rainfall events before sampling might play a more important role on the in-ditch $NO_3 - N$ concentration rather than magnitude-related characteristics of these rain events. Further, this finding is supported by failing to reject the null hypothesis of zero correlation coefficient between $NO_3 - N$ concentration and RDEP ($r = 0.164$; $p - value = 0.126$) and RIEP ($r = 0.022$; $p - value = 0.847$), respectively. The inverse association found between $NO_3 - N$ and TBRS ($r = -0.289$; $p - value = 0.006$) was also expected. The negative correlation means that the more time elapsed between a rainfall event and the subsequent water quality sampling event, the lower the $NO_3 - N$ concentration measured in the ditch. This finding highlights the fact that residence time is a critical controlling factor in nutrient reduction through the TWR ditch, although this variable was not directly measured in the present study. On the other hand, we found that $NO_3 - N$ concentration positively correlated with depth (DNRE; $r = 0.287$; $p - value = 0.007$), intensity (INRE; $r = 0.234$; $p - value = 0.029$), and duration (DuNRE; $r = 0.320$; $p - value = 0.004$) of next-to-last rainfall events. These results indicate that characteristics of successive rainfall events might have greatly influenced the observed level of $NO_3 - N$ in the ditch. Also, this influence was supported by the negative correlation found between $NO_3 - N$ and TBNRE ($r = -0.281$; $p - value = 0.009$), meaning that higher $NO_3 - N$ concentrations were found as the dry time before the next-to-last rainfall event was shorter. Therefore, these findings suggest that the $NO_3 - N$ concentrations observed in the ditch were strongly dependent on antecedent hydrological conditions within the study area with (1) the duration of rainfall events before sampling and (2) the characteristics of next-to-last rainfall events playing a more influential role.

The 46 rain events linked to water quality sampling were grouped into four classes using the *k*-means clustering method (Table 3.3), for which the six rainfall characteristics that significantly correlated with $NO_3 - N$ concentrations were used as input. Regarding precipitation frequency ranked from highest to lowest frequency of occurrence, we found class III > class I > class II > class IV. For total rainfall depth, class II had the highest amount followed by class III, class I, and class IV. The combined variables of total rainfall depth and frequency showed that classes I and III were likely to have the most impact on the in-ditch water quality. Congreves et al. (2016) reported similar results with higher $NO_3 - N$ losses associated with greater total precipitation and more frequent and intense precipitation events. This impact can be further analyzed by showing the inter-month distribution of the different classes during the monitoring period as displayed in Figure 3.3. Class-I rainfall events mostly occurred during the spring, especially during the pre-growing season, whereas class-III events were more prevalent during the growing season and fall.

Class-I rainfall events were associated with higher $NO_3 - N$ concentrations that had a median concentration (median = 1.78 mg L^{-1}, $W = 455$, $p - value = 4.35 \times 10^{-6}$) more than 5 orders of magnitude greater than the $NO_3 - N$ concentrations linked with class-III events (median = 0.33 mg L^{-1}). These two classes did not show significant differences in RDEP ($W = 352$, $p - value = 0.174$), RIEP ($W = 313$, $p - value = 0.752$), RDuEP ($W = 360.5$, $p - value = 0.095$), TBRS ($W = 257$, $p - value = 0.246$), and TBTR ($W = 264.5$, $p - value = 0.337$). Instead, the classes did significantly differ in the characteristics describing next-to-last rainfall events. When compared with class-III events, class-I were significantly higher in depth (DNRE, $W = 477.5$, $p - value = 4.12 \times 10^{-8}$), intensity (INRE, $W = 443$, $p - value = 2.81 \times 10^{-5}$) and duration (DuNRE, $W = 463$, $p - value = 7.07 \times 10^{-7}$) of next-to-last rainfall events. In addition, class-I events had shorter TBNRE ($W = 220$, $p - value = 0.014$). In general, these results suggest that the effect of class-I rainfall events on in-ditch $NO_3 - N$ concentrations appears to be magnified under higher depth, intensity, duration, and shorter dry time before next-to-last rainfall events.

Results show that the $NO_3 - N$ level in the TWR ditch was highly sensitive to successive rainfall events occurring during the spring and fall seasons and seems to be interacting with the timing of fertilizer application. These findings are significant because they reflect the combined effect of spring and fall fertilizer applications and substantial amounts of rain falling over fertilized and exposed soils classified as poorly drained and in very high potential runoff class. According to our data set, 60% of the total annual rainfall occurred during the spring (35% from February to May) and fall (25% from September to November) seasons, when farmers usually applied fertilizer over the fields. Nitrogen application during fall is in the form of ammonia (NH_3), which is likely converted into NO_3 *via* nitrification through winter and spring seasons. The final product of this oxidation process is nitrate, which is readily moved off soil rather than its counterpart ammonia (Kyveryga et al., 2004; Sahrawat, 1989, 2008; Subbarao et al., 2006). Consequently, a significant portion of the nitrogen applied during fall might have been washed off the field and into the ditch during spring by class-I rainfall events as the primary triggering factor. The influence of these class-I rainfall events appeared to be dictated by higher depth (DNRE), intensity (INRE), and duration (DuNRE) of next-to-last rainfall events. Also, this condition may have been exacerbated by supplemental nitrogen applied during spring, typically in the form of nitrate, to replenish what was lost from the fall application. Consecutive antecedent rainfall might bring the soils to field capacity relatively fast such that rain events before sampling may not need high magnitudes to trigger increased levels of $NO_3 - N$ in the ditch. Instead, the effects of class-I rainfall events appeared to have been magnified by antecedent rainfall conditions.

Lower levels of $NO_3 - N$ were observed in the ditch during late summer and fall and were associated with class-III rainfall events. Our study showed that the possible influence of class-III rainfall events on in-ditch water quality was driven mainly by high-frequency, low magnitude, and dry antecedent conditions. During summer, the study area is covered with growing crops planted in rows actively uptaking $NO_3 - N$, which might account for decreased $NO_3 - N$ concentrations mobilized with overland flow and soil erosion, and result in lower levels of in-ditch $NO_3 - N$. This summer period also experiences higher evapotranspiration rates (up to 200 mm; (Pérez-Gutiérrez et al., 2017a)), likely increasing the soil storage capacity such less runoff may occur (Randall and Mulla, 2001). Also, irrigation events occurring late summer and early fall might have influenced water quality in the ditch, although the methods used in this study did not enable us to determine such an effect.

Rainfall events classified as II and IV were not as frequent, so thus not tested for significant differences between or among classes. The only three class-II events were linked to a median $NO_3 - N$ concentration of 0.74 mg L^{-1} and accounted for 211.8 mm of rainfall, which was the highest total rainfall depth among classes. Rainfall events classified as II represented storms with very high depth (RDEP), high intensity (RIEP), and moderate duration (RDuEP). One event with 42.67 mm of rainfall was grouped as class-IV and linked to 4.23 mg L^{-1} of $NO_3 - N$ in the ditch. The one class-IV rainfall event was characterized by high depth (RDEP), low intensity (RIEP), and very long duration (RDuEP). Both class-II and -IV were consistently associated with very low magnitudes of next-to-last rainfall characteristics (*i.e.* DNRE, INRE, and DuNRE), which show that their associated effects might have been independent of antecedent hydrological conditions. Class-II and -IV rainfall events should be further analyzed as very high amounts of $NO_3 - N$ could be transported during the dormant season due to longer duration of these two classes of events with no apparent dependence on next-to-last rainfall events (Figure 3.4a). However, the impact of the class-II and -IV rainfall events on the $NO_3 - N$ concentrations in the ditch should not be discounted due to the limited number of occurrences of water samples collected. The impact of the class-II and -IV rainfall events, which ultimately are associated with extreme hydrological events, on TWR and structural BMPs is an open topic that needs to be investigated.

Summary and conclusions

Understanding the effects of rainfall characteristics on the performance of BMPs is important for improving their ability to control excess nutrients transferred from agroecosystems into surface waters. Most of the studies that have addressed this research need have either been performed under limited and controlled conditions, field-based or with a rainfall simulation without identifying the relative effect of rainfall characteristics on the BMP performance. This study examined how rainfall characteristics affect the NO_3-N concentration in a TWR ditch implemented at a farm within the Porter Bayou watershed in Mississippi, US. By accomplishing this goal, two central insights are provided by this study: (a) potential interaction between timing of fertilizer application and rainfall characteristics and (b) the role of next-to-last rainfall characteristics on the level of in-ditch NO_3-N. The time series analysis of rainfall and NO_3-N concentration, which included time series data for cumulative precipitation, was essential to link the observed levels of in-ditch NO_3-N with rainfall characteristics. Also, the analysis showed that the potential interaction between fertilizer application timing and successive rainfall events is a determining factor to the in-ditch water quality. The linear dependence analysis provided insight into the implications of antecedent hydrological conditions on the nutrient reduction ability of the ditch. The rainfall classes identified by using the *k*-means clustering approach provide information on the rainfall characteristics that should be considered for future design, operation, and management of TWR ditches to achieve more efficient nutrient control strategies. The methodology used in the present study can be of practical relevance if additional research is done at different spatial scales and for various individual or combined BMPs and pollutants. Results of this investigation can help reduce downstream nutrient loss in agricultural landscapes.

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Table 3.1 Description of rainfall characteristics established for analysis in this study

Characteristic	Acronym	Description	Units
Rainfall events before sampling	RDEP	Rainfall depth of the event before sampling	mm
	RIEP	Rainfall intensity of the event before sampling	mm h^{-1}
	RDuEP	Rainfall duration of the event before sampling	days
	TBRS	Time between previous rainfall and sampling events	days
	TBTR	Time between two rainfall events before sampling	days
Next-to-last rainfall event before sampling	DNRE	Rainfall depth of the event before sampling	mm
	INRE	Rainfall intensity of the event before sampling	mm h^{-1}
	DuNRE	Rainfall duration of the event before sampling	days
	TBNRE	Time between previous rainfall and sampling events	days

Table 3.2 Kendall's tau correlation coefficients between rainfall characteristics and NO_3-N concentrations measured at P2 sampling site in the tailwater recovery ditch.

Variables	Water quality	Rainfall characteristics								
	NO_3 - N	RDEP	RIEP	RDuEP	TBRS	TBTR	DNRE	INRE	DuNRE	TBNRE
NO_3 - N	1.00									
RDEP	0.16	1.00								
RIEP	0.02	**0.72****	1.00							
RDuEP	**0.25***	**0.67****	**0.37****	1.00						
TBRS	**-0.29****	-0.03	0.02	0.02	1.00					
TBTR	-0.09	0.05	0.04	0.02	0.02	1.00				
DNRE	**0.29****	0.07	0.00	0.18	0.09	-0.07	1.00			
INRE	**0.23***	0.07	0.01	0.15	0.06	-0.07	**0.80****	1.00		
DuNRE	**0.32****	0.05	-0.02	0.12	0.11	-0.05	**0.66****	**0.43****	1.00	
TBNRE	**-0.28****	0.08	0.16	0.02	0.18	0.15	-0.09	-0.02	-0.17	1.00

The pairwise linear correlation coefficients between variables are presented below the diagonal.
Boldface font indicates a correlation significantly different from zero (*$p < 0.05$ and **$p < 0.01$)
NO_3 – N: nitrate nitrogen concentration (mg L^{-1}); RDEP: rainfall depth of the event prior to sampling (mm); RIEP: rainfall intensity of the event prior to sampling (mm h^{-1}); RDuEP: rainfall duration of the event prior to sampling (days); TBRS: time between previous rainfall and sampling events (days); TBTR: time between two rainfall events before sampling (days); DNRE: depth of next-to-last rainfall event (mm); INRE: intensity of next-to-last rainfall event (mm h^{-1}); DuNRE: duration of next-to-last rainfall event (days); TBNRE: time before next-to-last rainfall event (days).

Table 3.3 Description of classes obtained using the *k*-means clustering method for linking rainfall characteristics and NO_3-N concentration measured at P2 sampling site in the tailwater recovery ditch studied.

Class	Frequency	Total rainfall depth	Summary statistics	Water quality	Rainfall characteristics								
				NO_3 - N	RDEP	RIEP	RDuEP	TBRS	TBTR	DNRE	INRE	DuNRE	TBNRE
	times	mm		mg L^{-1}	mm	mm h^{-1}	days	days	days	mm	mm h^{-1}	days	days
I	14	108.708	Mean	2.15	7.76	2.21	0.15	3.73	1.48	15.80	2.98	0.24	0.88
			Median	1.78^a	5.08^a	1.25^a	0.12^a	2.96^a	0.33^a	13.84^a	2.57^a	0.167^a	0.06^a
			Maximum	4.02	21.08	11.68	0.33	12.17	7.42	43.94	7.03	0.75	6.42
			Minimum	0.23	0.25	0.25	0.04	0.54	0.04	2.29	0.76	0.08	0.04
			SD	1.21	7.63	3.00	0.10	3.00	2.38	11.46	1.85	0.17	1.74
II	3*	211.836	Mean	0.61	70.61	8.62	0.33	4.39	0.25	13.40	3.45	0.08	2.46
			Median	0.74	77.98	9.75	0.33	2.42	0.25	0.25	0.25	0.04	0.38
			Maximum	0.86	94.49	10.50	0.38	10.04	0.33	39.70	9.84	0.17	6.75
			Minimum	0.23	39.37	5.62	0.29	0.71	0.17	0.25	0.25	0.04	0.25
			SD	0.34	28.29	2.63	0.04	4.97	0.08	22.77	5.54	0.07	3.72
III	28	110.9985	Mean	0.59	3.96	1.95	0.09	6.01	4.02	2.05	1.10	0.08	3.60
			Median	0.33^b	1.78^a	0.95^a	0.04^a	4.25^a	0.81^a	1.27^b	0.48^b	0.08^b	0.77^b
			Maximum	1.63	15.49	13.72	0.42	15.75	19.92	8.38	7.11	0.21	18.38
			Minimum	0.23	0.25	0.25	0.04	0.04	0.04	0.25	0.25	0.04	0.04
			SD	0.43	4.57	2.76	0.08	4.79	6.12	2.34	1.52	0.05	5.11
IV	1*	42.67	**	4.23	42.67	1.09	1.63	0.88	9.96	1.78	0.89	0.08	0.04

Medians in columns followed by the same lowercase letter are not significantly different between classes ($p > 0.05$)
* Not applicable for test of hypothesis, and ** for summary statistics because of the low number of observations
NO_3 - N: nitrate nitrogen concentration (mg L^{-1}); RDEP: rainfall depth of the event prior to sampling (mm); RIEP: rainfall intensity of the event prior to sampling (mm h^{-1}); RDuEP: rainfall duration of the event prior to sampling (days); TBRS: time between previous rain and sampling events (days); TBTR: time between two rainfall events before sampling (days); DNRE: depth of next-to-last rainfall event (mm); INRE: intensity of next-to-last rainfall event (mm h^{-1}); DuNRE: duration of next-to-last rainfall event (days); TBNRE: time before next-to-last rainfall event (days).

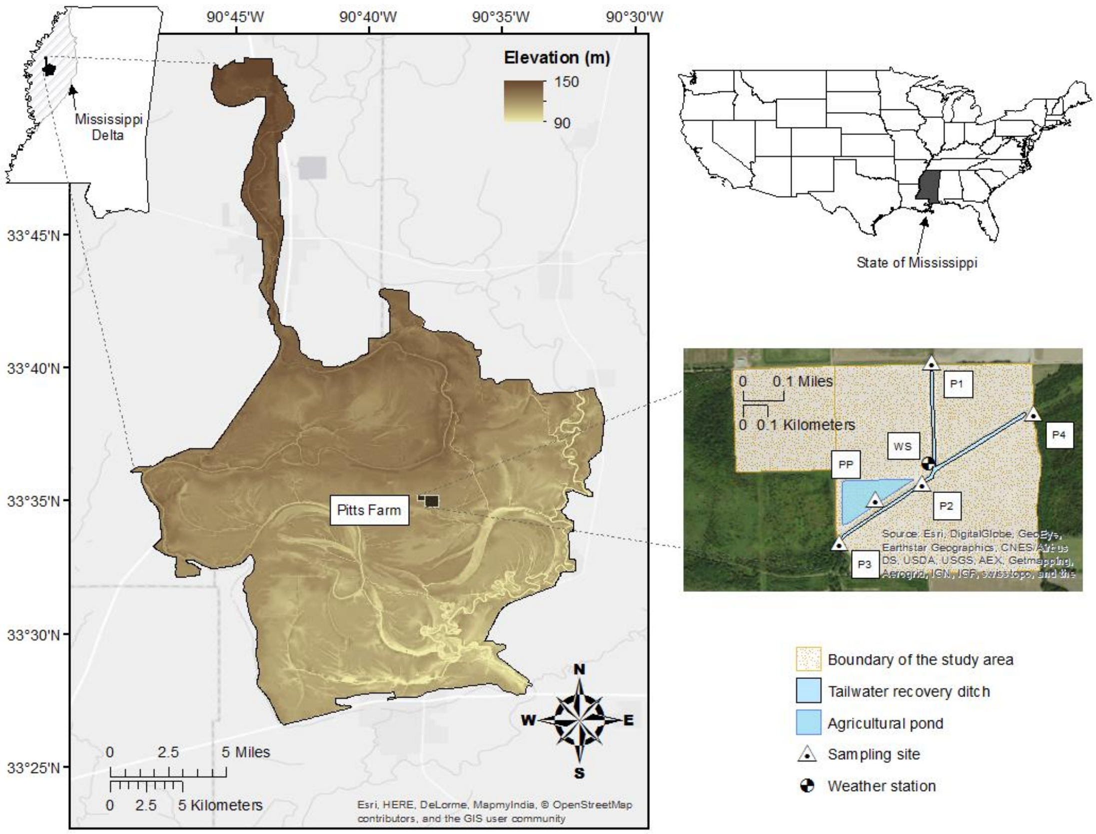

Figure 3.1 Map of the study area showing the Porter Bayou watershed in the Mississippi Delta region (left) and the sampling sites (right) along the tail water recovery ditch.

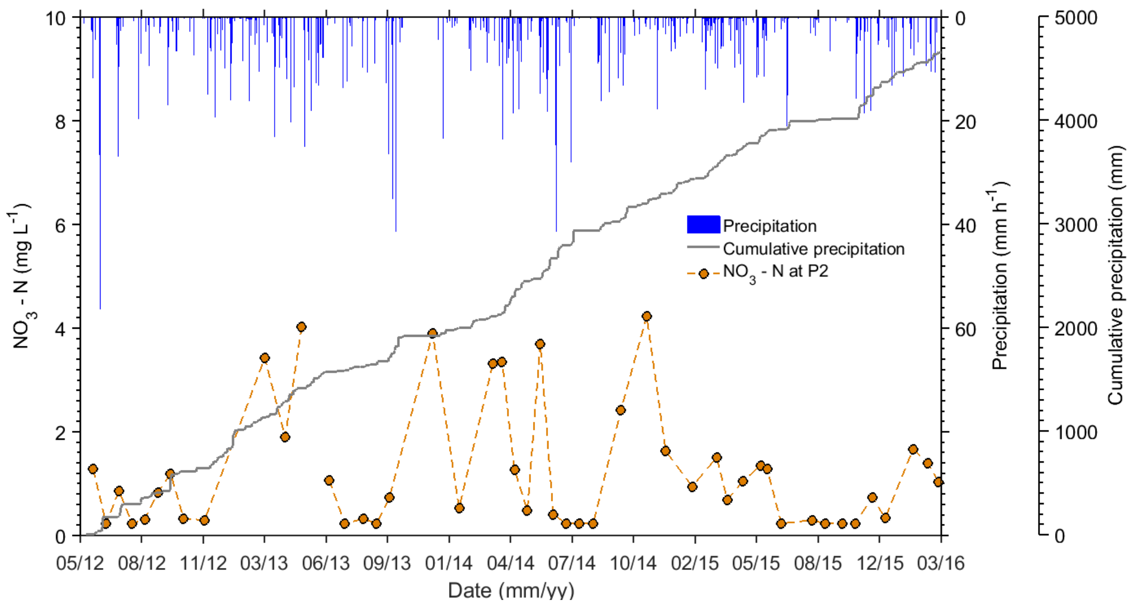

Figure 3.2 Time series of precipitation (inverted blue bars; mm h^{-1}) and in-ditch NO_3 – N concentration (dashed ocher line with circle markers; mg L^{-1}) .

This chart also includes the cumulative precipitation (continuous gray line; mm) recorded by the weather station deployed at the study site.

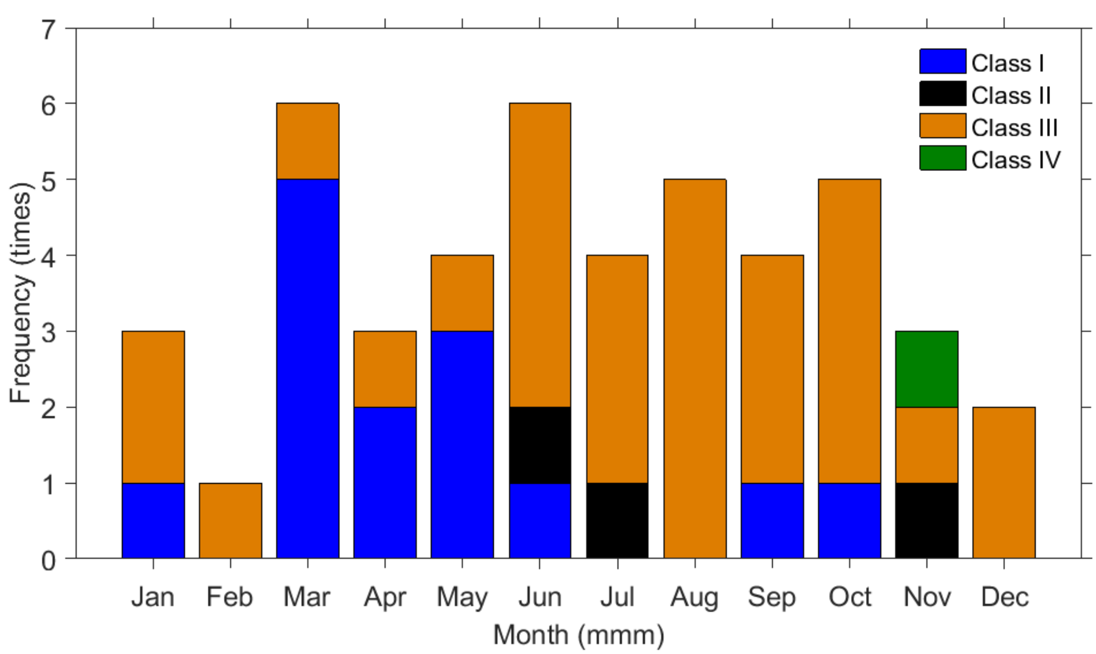

Figure 3.3 Inter-month distribution of the different rainfall classes during the monitoring period.

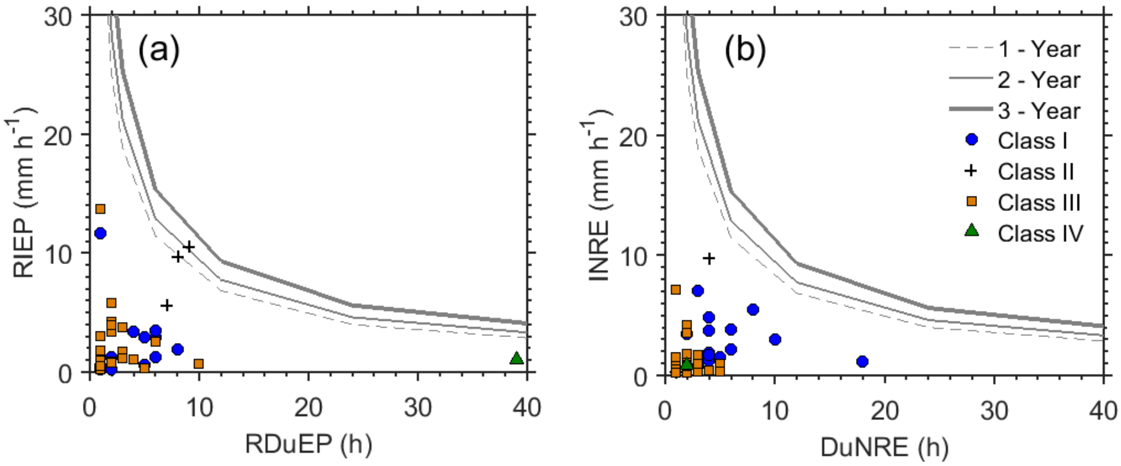

Figure 3.4 Rainfall events about intensity-frequency-duration (IDF) plot for 1 –, 2 –, and 3 – year return period.

(a) RIEP: rainfall intensity of the event before sampling (mm h^{-1}) Vs. RDuEP: rainfall duration of the event before sampling (days). (b) INRE: intensity of next-to-last rainfall event (mm h^{-1}) Vs. DuNRE: duration of next-to-last rainfall event (days).

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CHAPTER IV

TOWARDS AN IMPROVED UNDERSTANDING OF ON-FARM WATER STORAGE SYSTEMS IN MISSISSIPPI: DISCHARGE OF OUTFLOW EVENTS AND THEIR ASSOCIATED NUTRIENT LOAD

Juan D. Pérez-Gutiérrez, Joel O. Paz, Mary Love M. Tagert, Ying Ouyang

Abstract

On-farm Water Storage (OFWS) systems can mitigate downstream nutrient pollution originating from agricultural landscapes. However, little attention has been placed on measuring the volume of discharge water and the transported nutrient load that exits these systems. Thus, essential information about the hydrological and physical-chemical characteristics of the OFWS discharge is absent in scientific literature. The objective of this study was to quantify the volume of discharge water and the associated nutrient load leaving an OFWS system implemented on a farm located in Porter Bayou Watershed, Mississippi. Discharge water was recorded every five min from December 22, 2015 to December 21, 2016. Sample collection at the outlet pipe of the system began in March 2012 and ended in May 2017 and occurred every three weeks during the growing season (March to October) and every six weeks during the dormant season. Our results show that the volume of water discharged in winter, spring, and fall dictated the nutrient load exiting the system during the one-year discharge monitoring period. Although concentrations were higher during spring, the estimated nutrient load was

greater in winter because this was when the highest water volume was discharged during the one-year period. The typical estimated $NO_3 - N$ and TP yield resulted in 5.38 kg ha^{-1} and 1.79 kg ha^{-1} during winter; 0.84 kg ha^{-1} and 0.22 kg ha^{-1} during spring; 0.16 kg ha^{-1} and 0.13 kg ha^{-1} during summer; and 0.71 kg ha^{-1} and 0.17 kg ha^{-1} during fall, respectively. Agricultural fields draining to the outlet of the system produced 7.1 kg $NO_3 - N$ ha^{-1} yr^{-1} and 2.3 kg TP ha^{-1} yr^{-1} that were discharged with outflow events. Water management operations should be established during chronic outflow events in winter to help reduce downstream nutrient loads. In the future, this water management operation should be implemented not only at the on-farm scale but also between-farms and among-farms. Results of this study provided new information on the benefits of OFWS systems for nutrient reduction and water storage and how the systems respond to hydrological variability in agricultural watersheds of Mississippi. The insights should enable continued agricultural sustainability and ecosystem health in the region.

Introduction

Water resources and their associated ecosystems are highly sensitive to additional nutrients used extensively by modern agriculture. When transferred into waterbodies by agricultural runoff or groundwater flow, excess nutrients can degrade aquatic ecosystems by largely contributing to eutrophication (Carpenter et al., 1998; Dodds and Smith, 2016; Dodds, 2006; Howarth et al., 2011; Nixon, 1995; Rabalais, 2002; Seitzinger et al., 2002; Smith, 2016; Smith et al., 1999). To address the challenge of maintaining the health of surrounding ecosystems while assuring agricultural profitability and higher yields, control measures such as the voluntary implementation of best management practices (BMPs) and conservation practices (CP) could be adopted (Barry and Foy, 2016;

Chaubey et al., 2010; Dodd and Sharpley, 2016; Her et al., 2017; Her et al., 2016; Osmond, 2010; Shao et al., 2017; Tomer and Locke, 2011; Tomer et al., 2014). Such practices vary with the type of agricultural pollutant to be treated and the landscape characteristics over which the practice will be placed, and they can be classified as structural or non-structural. Regardless of its category, BMPs and CPs are expected to balance economic feasibility with environmental benefits, which determines the final decision of whether or not to implement a particular practice (Ritter and Shirmohammadi, 2000). Therefore, sufficient knowledge regarding the net environmental and economic benefits of BMPs and CPs is critical for sustaining agricultural profitability and ecosystem health.

Agriculture in the United States produces $394.6 billion in sales (USDA, 2012) and covers slightly over 40% of the territory (FAO, 2015). To help protect the environment from agricultural impacts across the nation, the U.S. Department of Agriculture (USDA) promotes the voluntary implementation of BMPs and CPs. The USDA also provides financial incentives to farmers and landowners willing to adopt these practices over their fields (Tomer and Locke, 2011). Thus, a variety of practices have been established over the last several years, and scientific assessments have been made to estimate their effects and benefits (Duriancik et al., 2008; Osmond, 2010; Tomer and Locke, 2011; Tomer et al., 2014). Despite the progress shown on how BMPs and CPs can improve water quality, there are still challenges that need to be addressed, including how to enhance these practices and how they can benefit the environment at the field and edge-of-field scale (Tomer et al., 2014). This fact has not only stimulated research, but has also led to the emergence of a new generation of BMPs, such as low-grade weirs, slotted inlet pipes, and the two-stage ditch (Kröger et al., 2015), and on-farm water storage (OFWS) systems (Moore et al., 2015; Omer et al., 2016; Ouyang et al., 2017; Pérez-Gutiérrez et al., 2015; Pérez-Gutiérrez et al., 2016a; Pérez-Gutiérrez et al., 2016b; Pérez-Gutiérrez et al., 2017b). Although these relatively new structural BMPs have been implemented and their popularity is increasing, our understanding of the net benefits that these practices offer to the environment is limited. This understanding is of central importance for the planning and evaluation of conservation initiatives and to support better management decisions.

In the Mississippi Delta, OFWS systems typically combine tail-water recovery (TWR) ditches and agricultural ponds and are implemented for downstream nutrient reduction and water conservation. The surface runoff and irrigation tail-water is collected by ditches, and most of this water is pumped to ponds to be stored for future irrigation needs. The remainder of the in-ditch water evaporates, infiltrates, or flows out of the system. Recent research has focused on the spatial and temporal water quality changes occurring throughout OFWS systems to quantify and document their nutrient reduction benefits. Pérez-Gutiérrez et al. (2017b) found that the nutrient removal efficiency of an OFWS system can vary with season, and significant reductions were observed during spring. These findings are of substantial interest as the hypoxia zone observed in the Gulf of Mexico is greatly driven by agricultural runoff originating from the Mississippi River Basin, mainly in the warmer seasons (Dodds, 2006; Porter et al., 2015; Rabalais, 2002; Rabalais et al., 2002a; Rabalais et al., 2002b; Rabotyagov et al., 2014; Sprague et al., 2011; Turner et al., 2006; USEPA, 2007, 2015). Although the findings from Pérez-Gutiérrez et al. (2017b) have shown that OFWS systems can mitigate downstream nutrient pollution originating from agricultural landscapes, less attention has been placed on measuring the volume of discharge water and associated nutrient load that exits these systems. Thus, essential information about the hydrological and physical-chemical characteristics of the discharge water is absent in the scientific literature. This information is critical for assessing the potential impact of water discharged from OFWS systems on downstream water quality. Therefore, research addressing this lack of information is needed to better understand the net environmental benefits of using OFWS systems in agricultural areas. The main objective of the study was to quantify the discharge water and associated nutrient load leaving an OFWS system implemented at a farm located in Porter Bayou Watershed, Mississippi. An analysis of the discharge from the OFWS outlet and the associated nutrient concentration distribution is presented in this study.

Materials and methods

Site characteristics

The study was conducted on an OFWS system implemented on an agricultural field of 1.1 km^2 size situated north of Indianola, Mississippi within the Delta region of Mississippi (Figure 4.1). The field is located within the Porter Bayou watershed (PBW), an intensively farmed watershed of 506.2 km^2 that extends from latitude 33°26'39" to 33°51'38" north and longitude 90°48'54" to 90°31'34" west. The land use is predominantly crop production for soybeans and corn (MDEQ, 2008, 2012), and slightly more than 70% of the watershed is covered by soil types such as Forestdale, Dowling, Alligator, Sharkey, Brittain, and Waverly, which are classified as poorly drained and have a very high potential runoff class (Table 4.1). In addition, the PBW is relatively flat with surface elevations ranging from 90 to 150 m (Figure 4.1). Between 2010 to 2015, the mean temperature ranged from -8.3 °C during winter to 33.2 °C during summer (Figure 4.2a). The average seasonal precipitation observed during summer and fall was 247.8 mm and 395 mm, respectively (Figure 4.2b). Spring and fall seasons have similar rainfall characteristics and have important implications on the water budget in the study area. As expected during the colder seasons, evapotranspiration (ET) was lower than precipitation (Figure 4.2b), resulting in surplus water and greater potential for the OFWS system to capture and store water during these periods. The opposite is true during the warmer periods, when the ET rate is higher than precipitation.

The OFWS system, which was built according to NRCS (2011) specifications, consists of a trapezoidal-shape TWR ditch and an elongated agricultural pond with a combined storage volume of 128,020 m^3. The ditch is 818.8 m long with an average depth of 1.8 m, and water flows through the TWR ditch from north to south; the pond is 2.4 m deep and has a surface area of 4.45 ha. Water drains from 2.14 km^2 of fields, runs off into the TWR ditch through a system of pipes, and exits the system through the single outlet pipe (OP) set at 1.2 m above the canal bed. Additional information about the OFWS system can be found in Pérez-Gutiérrez et al. (2017b).

Measurement of discharge water and precipitation

A Stingray 2.0 Portable Level-Velocity Logger (Instruments Direct®, Kennesaw, GA,) was installed at the OP of the OFWS system (Figure 4.1) and used to continuously record discharge every 5 min from December 22, 2015 to December 21, 2016 (henceforth designated as discharge monitoring period – DMP). To test differences in hydrological characteristics of the outflows between seasons, the nonparametric Wilcoxon (Wilcoxon,

1945) and Kruskal-Wallis (Kruskal and Wallis, 1952) rank-sum tests were applied. The former is a test for whether the medians of independent samples of two data sets are similar or not. The latter extends the Wilcoxon rank-sum principle to three or more data sets. The *p*-values equal to or smaller than 0.05 were considered statistically significant. The Shapiro-Wilk test was used to verify the assumption of normality of the data set. Precipitation data were recorded automatically throughout the DMP at 15-minute intervals by a WatchDog 2700 Weather Station (Spectrum® Technologies, Inc., Aurora, IL) located 9.2 km southeast of the OP.

Quantification of nutrient load

The water quality monitoring period (henceforth designated as WQMP) began in March 2012 and ended in May 2017. Water samples were collected at the OP every three weeks during the growing season (March to October) and every six weeks during the dormant season. Each sample was analyzed *ex situ* for nitrate nitrogen, NO_3-N (mg L^{-1}); ammonia nitrogen, NH_3-N (mg L^{-1}); orthophosphate, ortho-P (mg L^{-1}); and total phosphorus, TP (mg L^{-1}) within 24 hours after sample collection. A prepackaged vial chemistry technique, TNT plus™ (Hach® Loveland, CO), was used for the analyses. The analytical methods employed for water quality data acquisition are described by Pérez-Gutiérrez et al. (2017b). Appropriate quality assurance and quality control measures were followed using USEPA (2002) recommendations. The distribution of nutrient concentrations in water samples was analyzed to quantify the nutrient load associated with the water discharged from the OFWS system. Significant differences between seasonal groups were tested using the nonparametric Wilcoxon and Kruskal-Wallis rank-sum tests. The *p*-values equal to or smaller than 0.05 were considered statistically significant, and the Shapiro-Wilk test was used to verify the assumption of normality of the data set.

Once the distribution of nutrient concentrations was obtained, the nutrient load was computed by using the Eq. (4.1):

$$W = V \cdot C \tag{4.1}$$

where W is the computed nutrient load in kg, V is the total volume of discharge water in m^3 exiting the OFWS system, and C is the nutrient concentration in mg L^{-1} by season at any percentile of interest. The value of C in Eq. (4.1) was established to be the 10^{th}, 25^{th}, 50^{th}, 75^{th}, and 90^{th} percentile of the corresponding distribution of nutrient concentrations. Therefore, several estimates of W were calculated, and Matlab® and the Statistical Toolbox™ (The Matworks, Inc., Natick, MA) were used to perform all mathematical and statistical calculations.

Results and discussion

Precipitation and water discharge

The total precipitation during the DMP was 1,336 mm, most of which occurred during winter (33%), followed by summer (29%), and spring (23%) (Figure 4.3). The remaining 15% of the total amount of rain fell during fall. Largerer rainfall events were observed during the summer season, which experienced three out of the five highest daily rainfall records (91.7 mm on July 27, 2016; 67.8 mm on August 17, 2016; and 53.34 mm on July 9, 2016). During winter, two successive strong events also were observed on March 9 and 10, 2016, with 83.3 mm and 55.1 mm of rain, respectively. These events contributed the most to the total amount of rainfall observed during winter and summer. The resulting amount of total rainfall during these two seasons exceeded the 6-year seasonal average for winter and summer (Figure 4.2b) by 61% and 57.4%, respectively. Spring total rainfall did not follow this pattern and remained below the 5-year seasonal average by 18.5%. Similar to the spring pattern, the fall rainfall observed during the DMP was half the fall 6-year average. Despite the differences in total precipitation between seasons, the number of rainy days was similar throughout most of the DMP (Figure 4.3c). However, there were nine fewer rainy days, on average, during fall.

Discharge varied with rainfall and season throughout the DMP (Figure 4.4). In general, the differences in the observed discharge and its hydrological characteristics reflected the differences in the seasonal total rainfall and its distribution. Therefore, rainfall depth and distribution had important implications on the observed outflow events. High rainfall amounts during winter and summer resulted in 66% and 15%, respectively, of the total volume of discharge (736,929 m^3) from the system during the DMP (Figure 4a and c). The spring and fall seasons combined accounted for only 19% of the total outflow volume (Figure 4.4b and d), indicating predominantly drier conditions during these seasons. Overall, the estimated annual discharge volume (736,929 m^3) was equivalent to 25% (or 344.3 mm) of the total annual precipitation measured at the study site. This annual discharge was within the range found in a study conducted in Iowa addressing nitrate losses in subsurface drainage (Jaynes et al., 2001).

Rainfall, discharge, and the total volume that exited the system for each season are shown in Figure 4.4. In terms of the seasonal volume of water that exited the OFWS system, the order was: winter > summer > spring > fall. Eight outflow events during the winter resulted in a total volume of 490,329 m^3, which represents 53% of the total depth of winter precipitation (Figure 4.4a). The total outflow volume in the winter was greater than summer, spring, and fall by a factor of 4.6, 6.6, and 7.4, respectively. During the spring season, seven outflow events resulted in 73,852 m^3 of water that flowed out of the system (Figure 4.4b) and represented 11% of the total depth of spring rainfall. Summer outflow events resulted in 106,724 m^3 of outflow, which is equivalent to 13% of the total rainfall observed during summer (Figure 4.4c). During the fall season, the measured water discharge was 66,024 m^3 and represented only 15.4% of the total amount of rain that fell in this season. Fall outflow events were not used for statistical comparisons because of the small number of events that were observed.

A graphical summary of the hydrological characteristics of the outflow events by season is shown in Figure 4.5. During winter, the median value of the discharged volume was found to be 26,744 m^3 and typically flowed throughout the OP for four days (Figure 4.5a and b). Storm events with median rainfall of 29 mm were responsible for producing the outflows during winter (Figure 4.5c). The typical peak discharge increased up to 0.27 $m^3\ s^{-1}$, 11 hours after the outflow event began (Figures 4.5d and 5e). Median peak discharge was found to be 0.12 $m^3\ s^{-1}$ and was similar among seasons ($X^2 = 1.89$; $p =$ 0.39). During spring, the median total rainfall of each storm event that produced an outflow was similar to the median observed during winter (27.4 mm; W = 69; $p - value =$ 0.61). The duration of outflow events in the spring season was one day shorter than with winter events. However, duration of outflow events for both seasons was not significantly different ($W = 72$; $p - value = 0.4$). Overall, the peak discharge for spring was 62% lower when compared with winter records and was observed typically after 10.6 hours. Five outflow events were observed during the summer, and they had a median volume of 12,466 m^3 and flowed for about 3.5 days (Figure 4.5a and 4.5b). Total rainfall amounts before the start of outflow events during the summer were higher than those observed during winter and spring, although the differences were significant only at $p - value$ = 0.65 (X^2 = 0.88). This indicates the role that the growing season plays on runoff generation, and therefore, on water exiting the system. When crops cover the ground, the amount of rainfall that translates into runoff decreases; infiltration increases resulting in lower runoff (Eldridge and Koen, 1993; Ludwig et al., 2005) and subsequently, lower discharge from the system.

To evaluate the discharge response when the ditch was full, a regression analysis for discharge volume and peak discharge against the total precipitation was conducted (Figure 4.6). During winter, the discharged volume increased 1,610 times with each unit of the total amount of precipitation event ($F_{1,6}$ = 295.95; $p - value$ < 0.001; Figure 4.6a). An outflow event that occurred mid-March accounted for 66% of the total volume of water discharged during winter. This event was triggered by a 209 mm-rainfall storm measured during 6 days; it reached a peak discharge of 0.6 $m^3\ s^{-1}$ within nearly 1.5 days and drained for about 13 days. This finding shows the substantial negative effects of one single rainfall event of 35 mm d^{-1} of intensity on downstream water quality. Therefore, management practices must be implemented to minimize the impacts of high intensity rainfall events, especially when there is minimal ground cover during winter. During spring, the volume of discharged water increased by a factor of 331.4 ($F_{1,5}$ = 11.2; $p - value$ = 0.02; Figure 4.6b) for each unit of total rainfall measured. Similarly, the summer discharge volume increased 322 times with each unit of the total amount of precipitation event ($F_{1,6}$ = 7.87; $p - value$ = 0.07; Figure 4.6c). The increase factor of peak discharge relative to each unit of total amount of rainfall by event did not vary among seasons

(Figure 4.6 d-f). The time to peak discharge was significantly shorter during summer (the median = 1.62 h, $X^2 = 9.67$, $p - value = 0.008$; Figure 4.5e) when compared with winter and spring seasons. This finding indicates that when the ditch was full, the discharge of excess water downstream occurred faster during the summer than during winter and spring. Also during summer, the stream network in the study area routes less water and most of the canals are dry. Therefore, very low hydraulic head downstream from the OP allows discharge of excess water to be released faster. In addition, warmer temperatures and lower pollutant concentrations reduced the density of water, facilitating its motion. The opposite might occur during winter and spring when the stream network is more active and usually full of water due to high amounts of rainfall that produce runoff. This condition increases the hydraulic head of canals downstream of the OP so that discharge water is drained more slowly.

Quantification of nutrient load

Nutrient concentrations of samples collected during outflow events from March 2012 to May 2017 exhibited seasonal variability. Median $NO_3 - N$ and TP concentrations were similar in winter ($NO_3 - N = 2.35$ mg L^{-1}; TP = 0.78 mg L^{-1}), spring ($NO_3 - N =$ 2.44 mg L^{-1}; TP = 0.65 mg L^{-1}), and fall ($NO_3 - N = 2.31$ mg L^{-1}; TP = 0.55 mg L^{-1}) ($NO_3 - N$: $X^2 = 0.15$, $p - value = 0.93$; TP: $X^2 = 0.84$, $p - value = 0.66$). However, when compared with these three seasons, summer $NO_3 - N$ and TP were 86% and 61% lower, respectively, although significant only at $p = 0.2$ ($NO_3 - N$: $X^2 = 5.22$; TP: $X^2 = 4.72$). The highest concentrations and largest variability occurred during spring and fall as a response to the interaction among pre-growing season fertilizer application, and depth and distribution of rainfall.

Results of this study are in agreement with the findings by Aryal and Reba (2017) who reported similar $NO_3 - N$ concentrations at the outlet of two watersheds in Northeast Arkansas for spring and fall. They also found that the summer season produced the lowest $NO_3 - N$ concentration. However, our observations during summer were about 50% lower than the values reported by Aryal and Reba (2017). TP concentrations measured at the two outlet sites in Northeast Arkansas were slightly lower than the average median TP concentration reported for winter, spring, and fall in our study. Yuan et al. (2013) estimated phosphorus losses from two agricultural watersheds in the same Mississippi Delta region and reported that the highest TP concentrations were possible during spring which is in line with our findings. Likewise, a study conducted in China assessing spatial and temporal variations of nitrogen and phosphorus losses noted the same pattern during spring (Chen et al., 2016) due to the fertilizer application matching wet periods.

Differences in seasonal nutrient concentrations and volume of effluent dictated the nutrient load exiting the system during the one-year DMP (Table 4.2). Although concentrations in the winter were lower than spring, the estimated nutrient load was greater in winter because the highest discharge during the DMP was produced during the winter. An exception to this pattern was observed during summer, when the volume of water was greater and the nutrient concentrations were lower when compared to spring and fall. This condition resulted in a lower nutrient load produced during the summer than during the spring and fall. The estimated load during spring and fall were comparable, whereas the range was larger during spring. Nutrient loads during summer were as low as the 10^{th} and 25^{th} percentile loads estimated for spring and fall, respectively. The estimated $NO_3 - N$ and TP yield resulted in 5.38 kg ha^{-1} and 1.79 kg ha^{-1} during winter, 0.84 kg ha^{-1} and 0.22 kg ha^{-1} during spring, 0.16 kg ha^{-1} and 0.13 kg ha^{-1} during summer, and 0.71 kg ha^{-1} and 0.17 kg ha^{-1} during fall.

The estimated annual nutrient loads of 1,520 kg $NO_3 - N$ and 495 kg TP that exited the system were determined by adding all computed seasonal nutrient loads. In terms of the seasonal contribution of nutrient loads to the annual estimation, winter was ranked first with 76% and 77.3% for $NO_3 - N$ and TP, respectively, followed by spring (12% and 9.7%), fall (10% and 7.3%), and summer (2% and 5.7%). The entire drainage area yielded 7.1 kg ha^{-1} $NO_3 - N$ and 2.3 kg ha^{-1} TP during the one-year DMP. Recent research conducted on similar agricultural fields in the Delta region of Arkansas reported 9.6 kg ha^{-1} and 8.6 kg ha^{-1} annual $NO_3 - N$ yield from two fields, 5,340 ha and 2,335 ha large (Aryal and Reba, 2017), which was higher than our $NO_3 - N$ yield estimation. The Arkansas study, however, found a lower TP yield (1.2 kg ha^{-1} and 2.1 kg ha^{-1}) as compared to our study. Past research showed yields of 8.25 kg $NO_3 - N$ ha^{-1} yr^{-1} and 2.93 kg TP ha^{-1} yr^{-1} in a Virginia agricultural watershed covering 214 ha (Inamdar et al., 2001). These yields compare well with our findings especially because the contributing area is roughly the same size as the one monitored in this study. The Inamdar et al. (2001) investigation assessed the nutrient load produced by a watershed after implementing agronomic BMPs such as strip cropping, conservation tillage, nutrient and integrated pest management, along with structural BMPs including vegetative filter strips, grade stabilization and drop structures. The similarity between the findings of this study and Inamdar et al. (2001) suggests that implementing an OFWS system *alone* might have comparable effects on agricultural watershed nutrient exports as with a combination of

BMPs. Our results showed higher TP loadings than what was observed from agricultural fields implementing subsurface tiles in Illinois (0.2 kg ha^{-1} to 1.3 kg ha^{-1} (Gentry et al., 2007)) and Indiana (0.34 kg ha^{-1} (Smith et al., 2015)). However, our results were substantially lower if compared to the estimated $NO_3 - N$ yield of 31.2 kg ha^{-1} yr^{-1} and 21.3 kg ha^{-1} yr^{-1} at two watersheds in the Midwestern United States (Kalkhoff et al., 2016).

More nutrients were transported during winter compared to the rest of seasons due to higher total rainfall inputs during this season. The poorly drained soils in the watershed are exposed to weather conditions during winter. Therefore, higher amounts of rain translate into greater runoff, which in turn readily reaches the OFWS system limiting its storage capacity. In addition, evapotranspiration was minimal leading to more overland flow production after rainfall events. Thus, the system was typically filled during the winter season, which implies that minimal rainfall was needed to produce sufficient runoff and outflow events. This condition was observed during the first seven outflow events measured during winter. However, when the system was full, one single strong rainfall event can also result in an extreme outflow event like the one observed in March, which accounted for 66% of the total volume of water discharged during winter. These extreme events could drain water for about a half of the month. This longer outflow hydrograph recession was likely due to high downstream hydraulic head which prevented water from easily moving downstream. In addition, and to a lesser extent, colder temperatures and higher pollutant concentrations increased the density of water making its motion slower. Long-lasting outflow events have a chronic effect on water quality and aquatic ecosystems. Managing operations should be established during chronic outflow events to help reduce downstream nutrient loads. For example, pumping water out of the TWR ditch to an on-farm reservoir could be implemented to help the ditches perform better under extreme outflow conditions during winter. Another alternative is to install a hydraulic structure at the outlet to control the outflow discharge as needed. In the future, this water management operation should be implemented not only at the on-farm scale but also between-farms and among-farms. Adequate management of seasonal water availability at a larger scale will benefit downstream waterbodies and producers.

Summary and conclusions

During recent years, OFWS systems have been increasingly implemented in Mississippi agricultural fields (Pérez-Gutiérrez et al., 2017b) and other farmed regions within the Lower Mississippi River Valley (Moore et al., 2015) because of their nutrient reduction and water supply benefits. While most studies that have investigated these systems have focused on quantifying nutrient reduction effectiveness, less focus has been placed on providing essential information about the hydrological and physical-chemical characteristics of the volume of water exiting these systems. Our study provides important information and seasonal analysis of the discharge water monitored over the course of a year (December 22, 2015 to December 21, 2016) and its associated nutrient load monitored from March 2012 to May 2017. Overall, although nutrient concentrations were higher during spring, the winter season contributed the most to the total annual estimated $NO_3 - N$ and TP load resulting in 76% and 77.3%, respectively, followed in order of magnitude by spring (12% and 9.7%), fall (10% and 7.3%), and summer (2% and 5.7%). In addition, effluent from the OFWS system was different among seasons with respect to the volume, frequency, peak, and time to peak discharge. These characteristics were strongly dependent on the seasonality of depth and distribution of rainfall. Winter and summer experienced stronger rainfall events resulting in the highest amount of rain among seasons. In fact, both seasons exceeded the six-average total rainfall events of winter and summer. Spring and fall seasons produced less water that exited the system. During spring, rainfall matched the five-year average for the number of expected rainfall events, while fall was 50% lower. Higher peak discharges with longer time peaks were predominant during winter, which resulted in a larger nutrient load transported to downstream waterbodies. Meanwhile, spring outflows had lower peak discharge and long time to peak, opposite to winter outflow events. Summer and fall season outflows peaked earlier than winter and spring events. The potential impact on downstream water quality and aquatic ecosystems is associated with the transition from dry to wet seasons and the alteration derived from varied outflow events by each season. This study uniquely describes hydrographs of outflow events for OFWS systems in combination with water quality analysis. Therefore, this study offers new insights for downstream water quality improvement as well as management of harvested nutrient-rich runoff that are critical to sustaining agricultural profitability and ecosystem health in the region.

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Table 4.1 Basic description and properties of soil types listed for the Porter Bayou Watershed according to the Soil Survey Geographic (SSURGO) database (Soil Survey Staff, 2016).

Soil name	Cover percentage of watershed area	Natural drainage class	Runoff class	Hydrologic soil group
Forestdale	28.65%	Poorly drained	Very high	D
Dowling (sharkey)	21.37%	Very poorly drained	Very high	D
Dundee	17.96%	Somewhat poorly drained (17.83%) Well drained (0.13%)	Low (16.01%) Medium (1.82%)	C (17.83%) B (0.13%)
Alligator	12.34%	Poorly drained	Very high	D
Pearson (dundee)	5.28%	Somewhat poorly drained	Low Medium	C
Sharkey	5.05%	Poorly drained	High (4.40%) Very high (0.65%)	D
Dubbs	3.23%	Well drained	Low (2.91%) Medium (0.32%)	B
Brittain (forestdale)	2.74%	Poorly drained	Very high	D
Water	0.94%	NA	NA	NA
Dexter (dubbs)	0.47%	Well drained	Low	B
Waverly (sharkey)	0.41%	Poorly drained	Very high	D
No data	0.35%	NA	NA	NA
Bosket (dubbs)	0.276%	Well drained	Low (0.272%) Medium (0.004%)	B
Brittain (amagon)	0.25%	Poorly drained	Very high	C/D
Dundee-clack	0.22%	Somewhat poorly drained	Low (0.11%) Medium (0.10%)	C
Swamp	0.21%	Very poorly drained	Very high	NA
Souva (amagon)	0.09%	Poorly drained	Very high	C/D
Beulah	0.07%	Somewhat excessively drained	Very low	A
Waverly (rosebloom)	0.04%	Poorly drained	Very high	B/D

Table 4.1 (Continued)

Marsh	0.03%	NA	NA	NA
Clack (crevasse)	0.02%	Excessively drained	Very low	A
Souva (sharkey)	0.01%	Poorly drained	Very high	D
Borrow	0.01%	No data	No data	No data
Gravel	0.00%	NA	NA	NA

Table 4.2 Seasonal NO_3 – N and TP load estimated at the outlet.

Season	Percentile	Nutrient concentration*		Volume of discharge#	Estimated nutrient load	
		NO_3 - N	TP		NO_3 - N	TP
		mg L^{-1}	mg L^{-1}	m^3	kg	kg
	90th	3.12	1.41		1,530	691
	75th	2.89	1.11		1,417	542
Winter	50th	2.35	0.78	490,329	1,152	382
	25th	1.99	0.67		976	329
	10th	1.94	0.59		951	289
	90th	9.22	1.60		681	118
	75th	4.37	1.20		323	89
Spring	50th	2.44	0.65	73,852	180	48
	25th	0.81	0.48		60	35
	10th	0.61	0.16		45	12
	90th	-	-		-	-
	75th	0.42	0.30		45	32
Summer	50th	0.33	0.26	106,724	35	28
	25th	0.23	0.22		25	23
	10th	-	-		-	-
	90th	4.68	1.50		309	99
	75th	4.09	1.26		270	83
Fall	50th	2.31	0.55	66,024	153	36
	25th	0.96	0.37		63	24
	10th	0.51	0.31		33	20

*Concentrations based on samples collected from March 2012 - May 2017. #Measured from December 22, 2015 - December 21, 2016

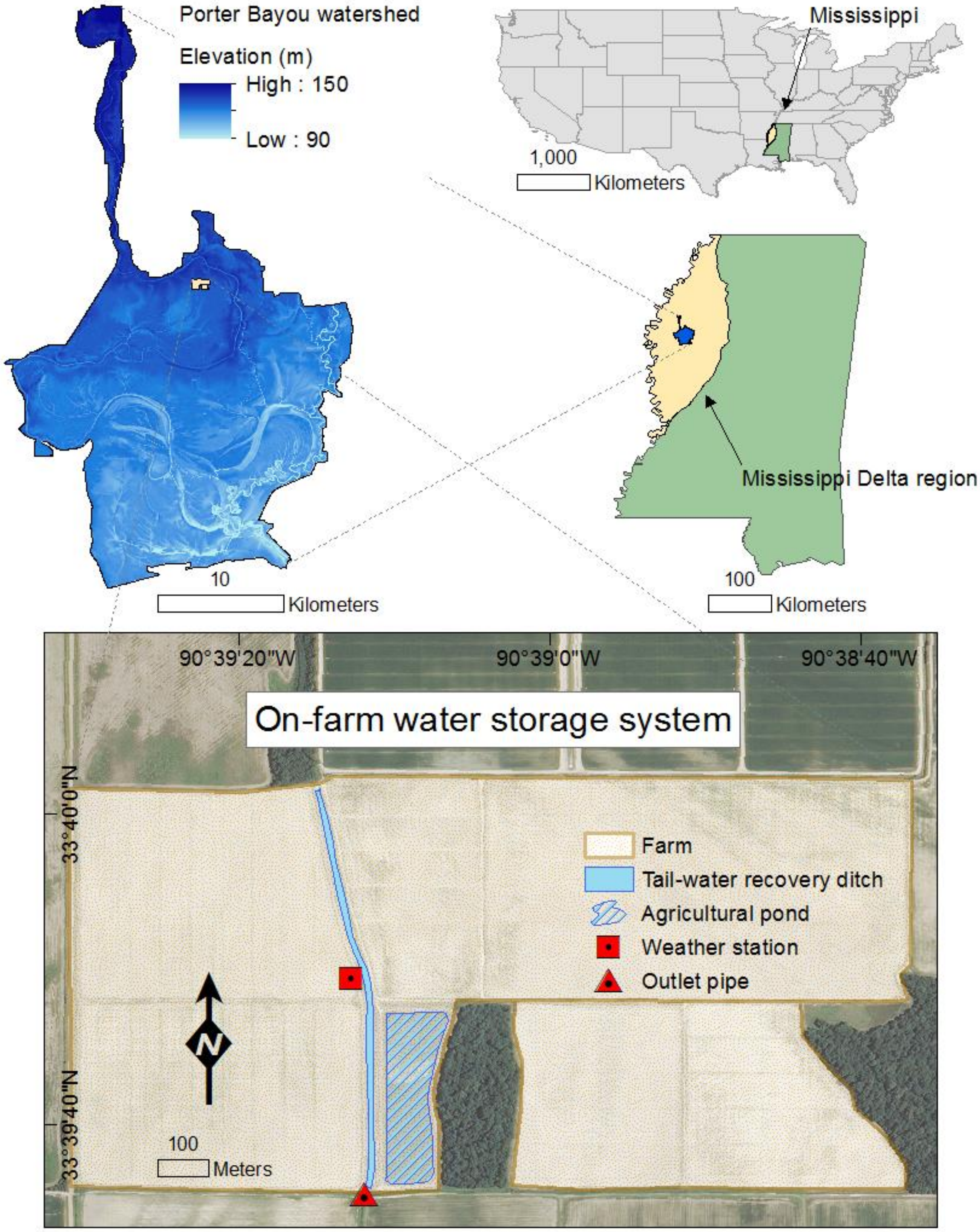

Figure 4.1 Map of the study area showing the farm implementing the on-farm water storage system investigated in the Porter Bayou watershed.

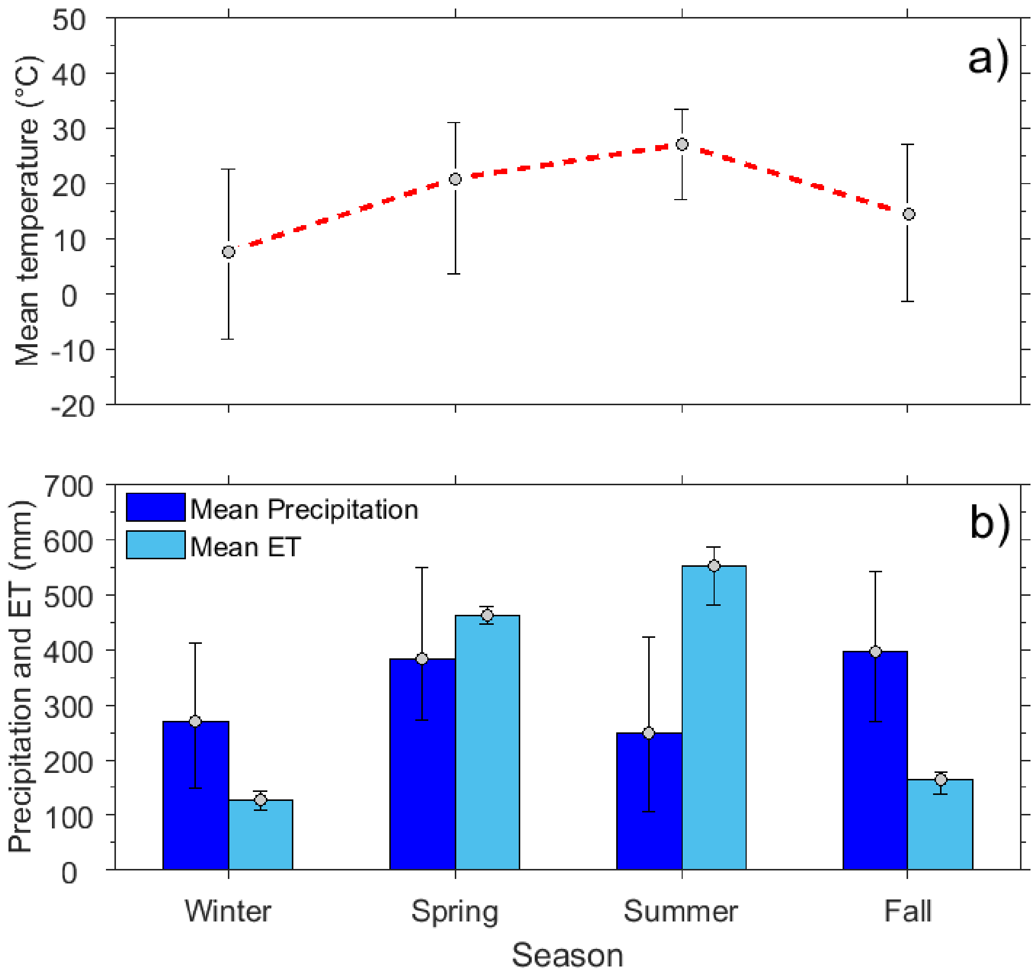

Figure 4.2 Seasonal 6-year average (a) temperature and (b) total precipitation and ET from 2010 to 2015 at the study site.

Error bars indicate maximum and minimum values. ET was estimated using Priestley-Taylor method (Priestley and Taylor, 1972).

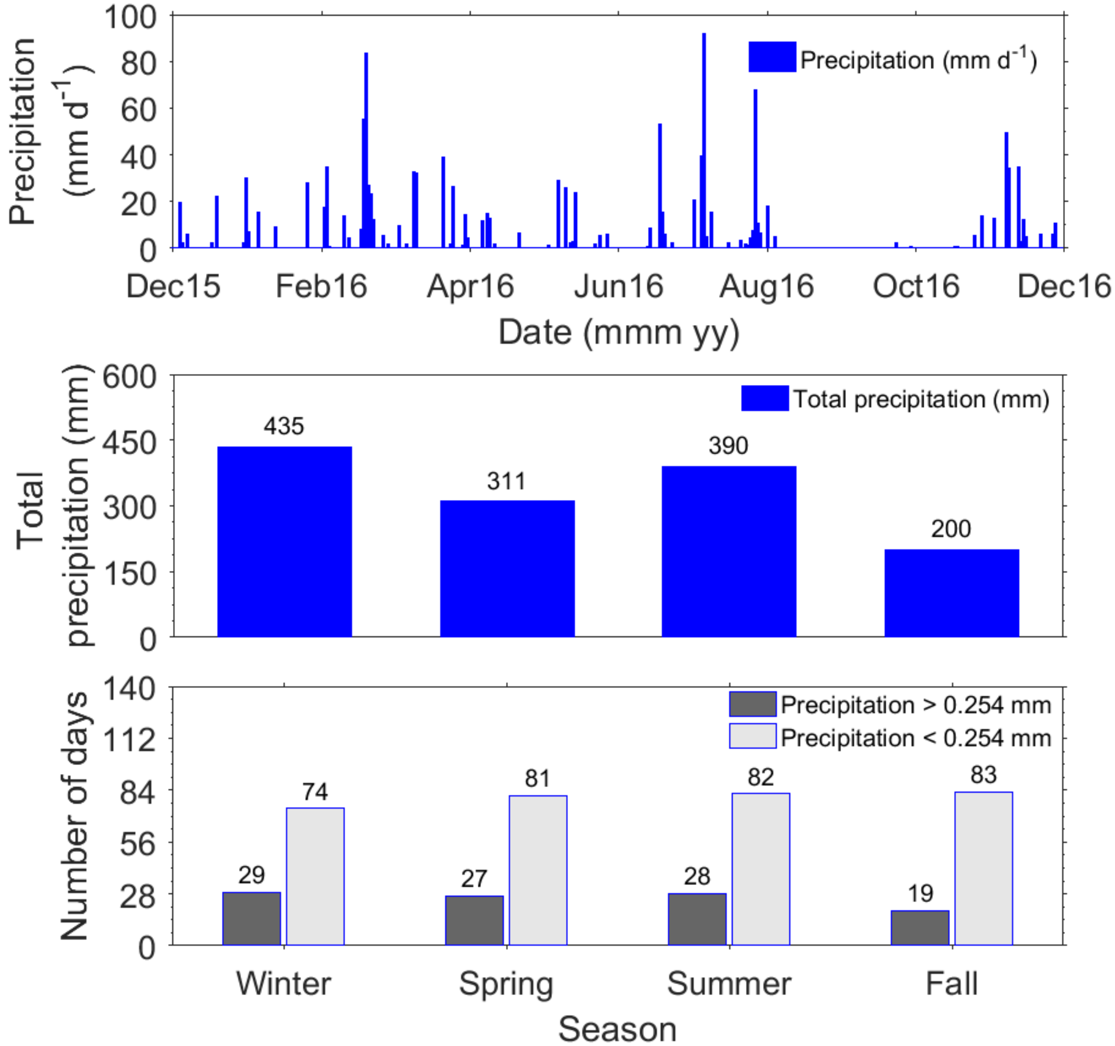

Figure 4.3 Time series of (a) precipitation recorded during the DMP (b) and inter-season distribution of the total precipitation and (c) number of rainy days.

Winter: December 22 – March 20, spring: March 21 – June 21, summer: June 22 – September 22, and fall: September 23 – December 21).

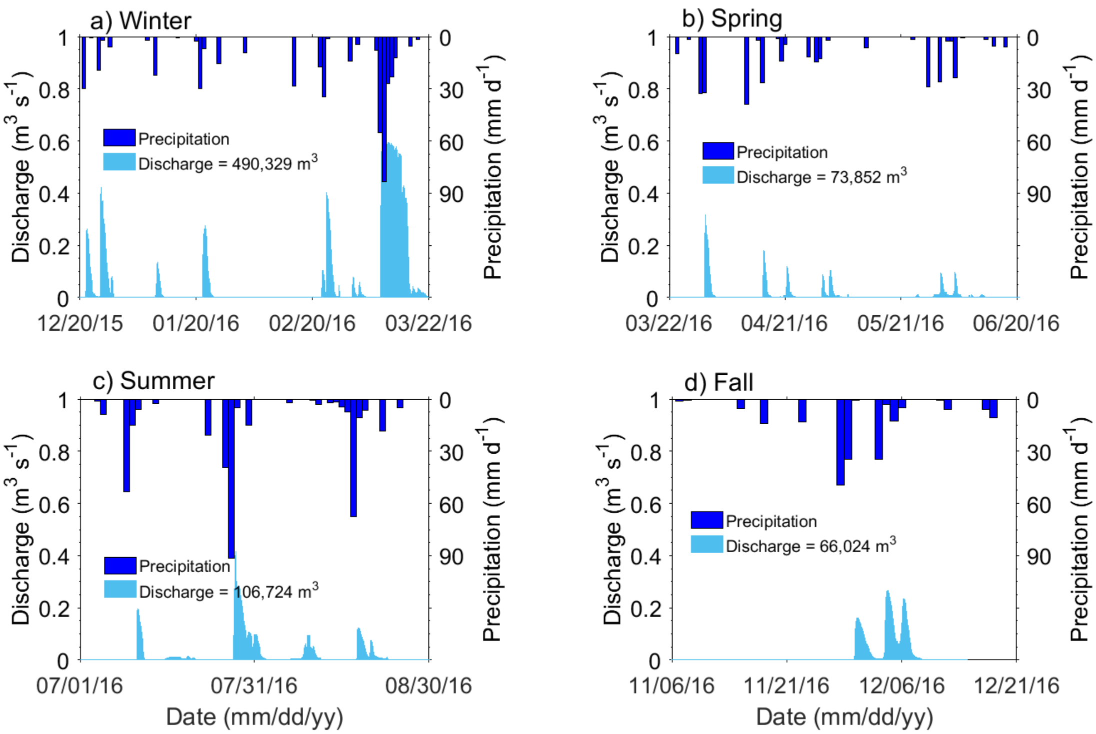

Figure 4.4 Seasonal time series of precipitation (inverted blue bars) and discharge (light-blue area plot) observed at the outlet.

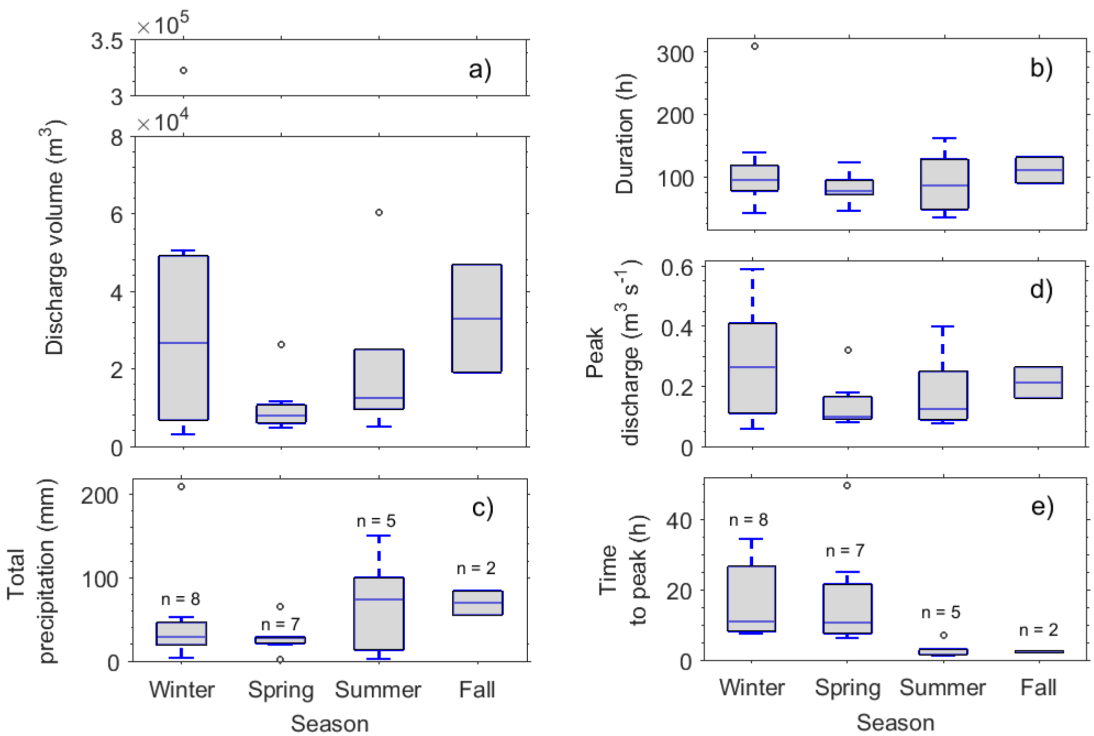

Figure 4.5 Boxplots of the hydrological characteristics of the outflow events by season.

(a) discharge volume by outflow event; (b) hydrograph duration of the outflow event; (c) total precipitation of the rainfall event before outflow initiation; (d) hydrograph peak discharge; (e) time to hydrograph peak discharge.

Boxplots were set at 90^{th} (the upper whisker), 75^{th} (the upper quartile), 50^{th} (the median), 25^{th} (the lower quartile), and 10^{th} (the lower whisker) percentiles. Outliers were considered those observations 1.5 times beyond the 25^{th} and 75^{th} percentile and are shown as grey circles. The number of samples (n) for each grouped dataset is shown above the upper whisker in the bottom subplots. Note: Fall outflow events were not used for statistical comparisons because of the small number of events observed.

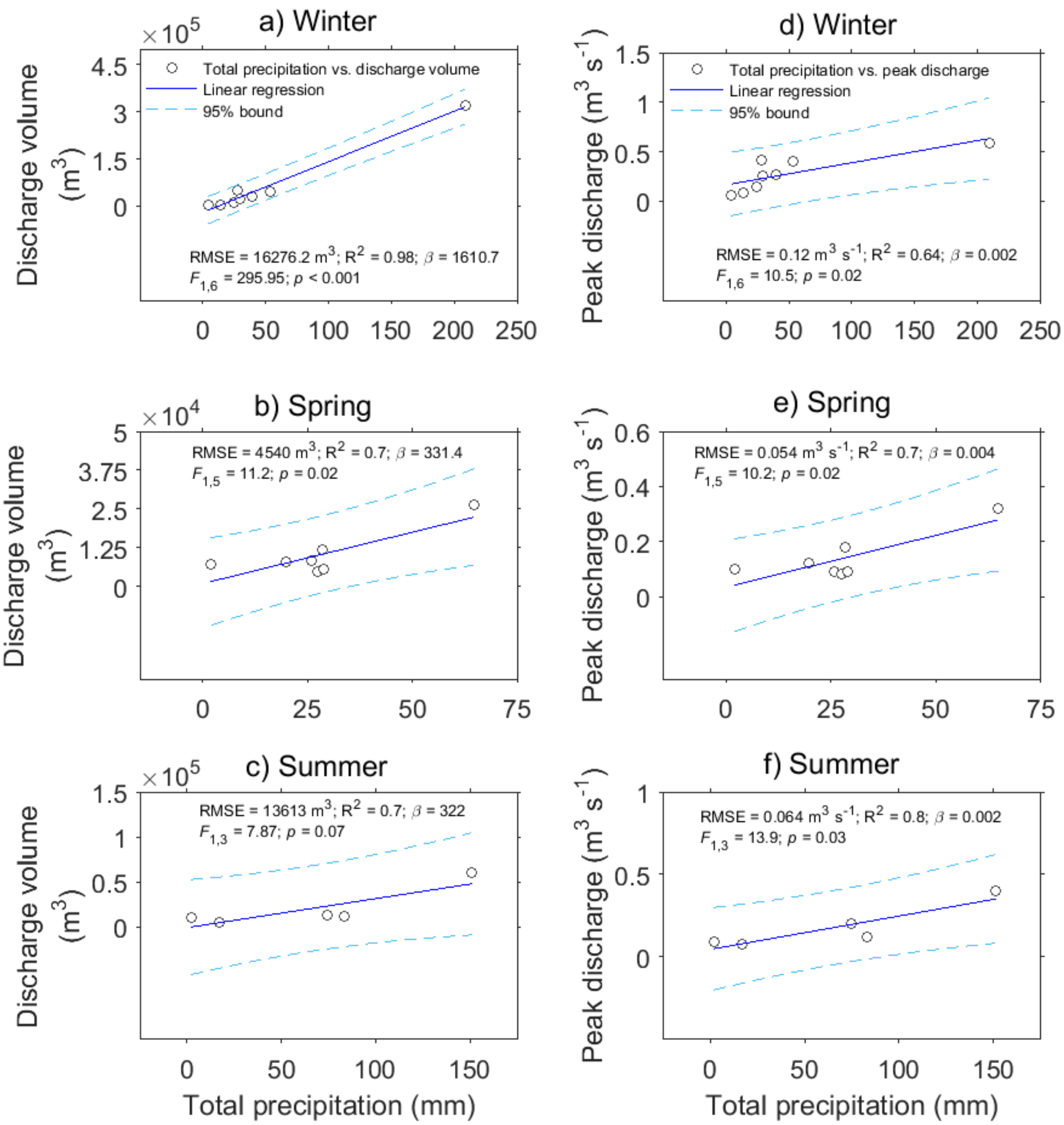

Figure 4.6 Seasonal regression analysis for three hydrograph characteristics of the outflow events by season.

Regression does not include fall season data because of the low number of outflow events observed during this period.

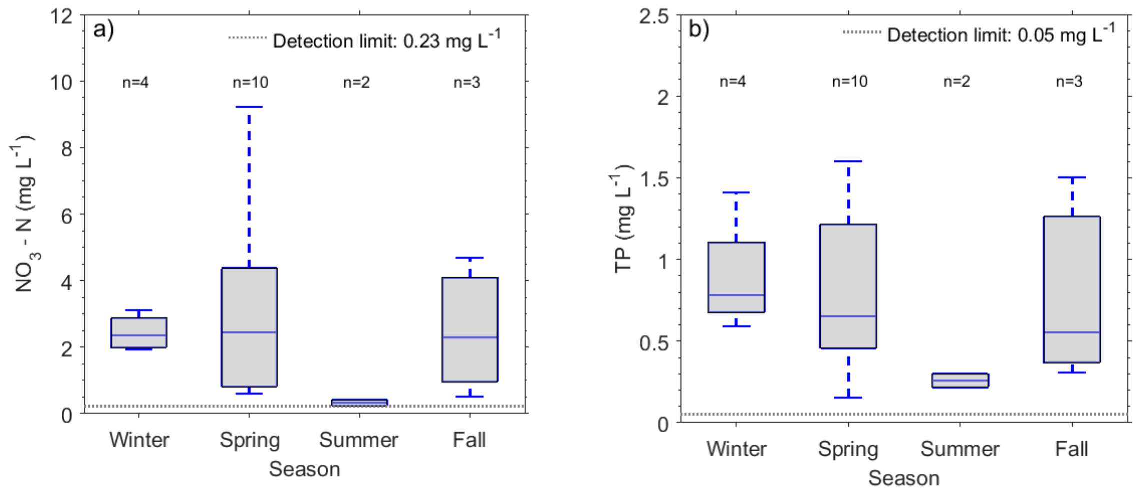

Figure 4.7 Boxplots of the seasonal variability of (a) NO_3 – N and (b) TP concentrations at the outlet.

Boxplots were set at 90th (the upper whisker), 75th (the upper quartile), 50th (the median), 25th (the lower quartile), and 10th (the lower whisker) percentiles. Outliers were considered those observations 1.5 times beyond the 25th and 75th percentile and are shown as grey circles. The number of samples (n) for each grouped dataset is shown above the upper whisker.

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CHAPTER V

USING AnnAGNPS TO SIMULATE RUNOFF, NUTRIENT, AND SEDIMENT LOADS IN AN AGRICULTURAL CATCHMENT WITH AN ON-FARM WATER STORAGE SYSTEM

Juan D. Pérez-Gutiérrez, Joel O. Paz, Mary Love M. Tagert, Lindsey M.W. Yasarer, Ronald L. Bingner

Abstract

A tailwater recovery (TWR) ditch is a structural best management practice that can improve downstream water quality by significantly reducing nutrient loads from agricultural watersheds. Although research has highlighted in-ditch nutrient reductions, less attention has been placed on estimating the water, nutrient, and sediment loads entering TWR ditches. This lack therefore is limiting our ability to understand the impact of implementing these systems in agricultural watersheds. In this study, runoff, nutrient, and sediment loads entering a TWR ditch in an agricultural catchment within the Porter Bayou watershed in Mississippi were quantified, and the main contributing sources were identified using the Annualized Agricultural Non-Point Source (AnnAGNPS) model simulations. The model was set to simulate runoff, nutrient, and sediment loads from 2010 to 2016, establishing the first two years as a warm-up period and the subsequent five years as the period for analysis of the loads entering the TWR. Simulations showed that fields with larger areas coupled with hydrologic soil group C or D resulted in higher

runoff, and that this condition mirrored the annual rainfall patterns. The volume of runoff exceeded the TWR ditch storage volume by roughly 110 times, mostly during the winter and spring seasons. Results showed that nitrogen load was sensitive to fertilizer application. Therefore, during years when corn and winter wheat were planted, nitrogen load increased compared to other years because these crops need nitrogen fertilization to grow. The TP and sediment load patterns were similar and influenced by the hydrological condition over time. Simulating different management scenarios indicated that planting winter wheat in the agricultural catchment can benefit water quality by reducing export of TP and sediment loads. If winter wheat were planted in the priority subwatersheds (Scenario 1), reductions of TP and sediments loads were about 19% and 13% at M1, respectively. Although planting winter wheat in all fields (Scenario 2) may not be feasible, this scenario would result in substantial reductions in TP and sediment loads from the contributing areas draining to M1 (TP: 39%; sediment: 24.2%) and M2 (TP: 63%; sediment: 45%) at the ditch. Scenario 2 also showed that 188,100 m^3 of runoff can be reduced from fields draining to the TWR ditch. While planting winter wheat can reduce runoff, TP, and sediment loads, this management practice can also result in higher nitrogen loads from overland flow because winter wheat requires nitrogen fertilizer. Quantification of the water, nutrient, and sediment loading constitutes an essential step towards an improved understanding of the benefits of TWR ditches on availability and quality of water when implemented in agricultural watersheds. Results of this study provide both stakeholders and resource management agencies with critical information that is needed to better identify where these systems should be implemented to improve water quality and offer a supplemental source of surface water for irrigation.

Introduction

The global population has been projected to increase between 9.6 and 12.3 billion by 2100 (Gerland et al., 2014). As a result, agriculture will have to double its productivity to feed people (Tilman et al., 2011; Tilman et al., 2002). Meeting this increasing food demand has led to the intensification of agriculture, which translates into greater use of chemical inputs and pressure on soil and water resources. In the form of non-point source pollution (NPSP), agricultural activities transfer excess fertilizers into aquatic ecosystems causing devastating ecological and economical effects (Carpenter et al., 2011; Ladapo and Aminu, 2017; Rabalais et al., 2002a; Rabotyagov et al., 2014; Withers et al., 2014). Therefore, a balance between increasing yields and mitigating adverse environmental impacts is pivotal to the future of agriculture.

The implementation of best management practices (BMPs) has been recognized to significantly reduce NPSP from croplands to downstream waterbodies (Osmond, 2010; Tomer and Locke, 2011; Tomer et al., 2014). However, much effort is still devoted to quantifying the effectiveness of such practices, especially newer BMPs, and the benefits that they offer to the environment. One of the challenges in measuring the effect of BMPs is that their performance varies spatially and temporally due to heterogeneity of the landscape and seasonality of hydrological factors (Her et al., 2017). The challenge is even greater when attempting to evaluate the effects of several structural and/or non-structural BMPs combined (Arabi et al., 2008; Lizotte et al., 2017b; Meals, 1987), address the shift in BMP performance over time (Bracmort et al., 2006), and understand lag times in water quality response (Meals et al., 2010). To evaluate benefits and quantify effectiveness of BMPs while accounting for these challenges, the use of watershed models can be a feasible alternative (Abdelwahab et al., 2016; Bracmort et al., 2006; Lam et al., 2011; Lizotte et al., 2017a; Parajuli et al., 2009; Santhi et al., 2014; Yuan et al., 2001; Zhang and Zhang, 2011). However, due to the diversity of BMPs that can be implemented over agricultural fields, information describing the water quality benefits for each BMP and the combined use of different BMPs is limited.

Structural BMPs such as tail-water recovery (TWR) ditches and agricultural ponds (i.e., on-farm water storage systems - OFWS) can improve downstream water quality by significantly reducing nutrient loads from agricultural watersheds (Moore et al., 2015; Pérez-Gutiérrez et al., 2015; Pérez-Gutiérrez et al., 2017b). Although financial assistance has been provided through the U.S. Department of Agriculture for the potential nutrient reduction benefits, these systems are also gaining popularity for their water supply benefits in areas where farmers need access to surface water for irrigation. This dual benefit of reducing nutrient pollution and supplying water for irrigation is important in areas such as the Lower Mississippi River Valley, where agricultural production strongly depends on irrigation. Farmers and landowners in this region are tasked with the issue of (1) reducing off-site movement of nutrients, which contributes to the hypoxic zone in the northern Gulf of Mexico, and (2) conserving water resources to slow declining groundwater levels in the Mississippi River Valley Alluvial Aquifer (MRVAA), which is the primary source of water for irrigation of crops. Consequently, OFWS systems have been implemented across the MRVAA, especially in areas with more severe groundwater declines. Although research has highlighted in-ditch and in-pond nutrient reductions, less attention has been placed on estimating the water and nutrient loads entering and exiting OFWS systems. Evaluating the OFWS drainage area will provide a better understanding of the impacts of implementing these systems in agricultural watersheds.

The Annualized Agricultural Non-Point Source (AnnAGNPS) model (Bingner and Theurer, 2001; Geter and Theurer, 1998) is a watershed model that has been designed to evaluate the impact of agricultural management practices on hydrological and water quality responses in watersheds. Most studies using the AnnAGNPS model demonstrate its performance after model calibration with field-observed data (Baginska et al., 2003; Chahor et al., 2014; Kliment et al., 2008; Licciardello et al., 2007; Parajuli et al., 2009; Polyakov et al., 2007; Sarangi et al., 2007; Shamshad et al., 2008; Zema et al., 2012). However, AnnAGNPS has also been used for similar purposes without conducting calibration with measured data. A study used the model with no calibrated parameters to estimate runoff and sediment in an agricultural watershed within the Mississippi Delta Region (Yuan et al., 2001). They concluded that the model had an adequate ability to simulate monthly and annual runoff and sediment yield with no calibration process. This finding is of significant interest because it is difficult and costly to secure hydrological and water quality observed data extensive enough to conduct calibration processes in watershed-scale modeling studies. To date, there is no study assessing the benefits of tail-water recovery ditches using watershed scale models. This is, at least in part, because these ditches have multiple inlets so that measured runoff data is difficult to obtain, and most of the time it is impractical and expensive. However, this information is critical to stakeholders and action agencies to better identify where these systems can be implemented to improve water quality and relieve pumping pressure on groundwater.

In this study, the AnnAGNPS model was implemented to simulate runoff, sediment, and nutrient load and to identify the main contributing areas draining to a TWR ditch established as part of an OFWS system located within Porter Bayou Watershed, Mississippi. Quantification of the water, nutrient, and sediment loading constitutes an essential step in understanding the water quality and quantity benefits of OFWS systems when implemented in agricultural watersheds, as well as how management of these systems might be altered to improve performance.

Materials and methods

Study site

This investigation was conducted in the Porter Bayou watershed (PBW; 33°26'39" – 33°51'38"N, 90°48'54" – 90°31'34"W) in the Mississippi Delta region (MDR), an intensively farmed area located in northwest Mississippi (Figure 5.1). The simulated watershed is within the PBW located north of Indianola, MS in Sunflower county, and includes Metcalf farm and the surrounding area that drains into the OFWS system outlet at M3 (Figure 5.1). The simulated watershed has a total drainage area of 214.04 ha. The major crops grown from 2012 to 2016 were soybean, corn, and rice (Table 5.1). Winter wheat was planted during 2013 and 2014 in four and two fields, respectively. The watershed is dominated by the following soil types: Forestdale, Tensas, Dundee, Pearson, and Dowling, which are primarily poorly drained soils and prone to produce high runoff. In addition, the watershed is relatively flat with surface elevations ranging from 130 to 135 m. The average temperature ranged from -8.3 ºC in winter 2014 to 32.2 ºC in summer 2012 (Figure 5.2a). The lowest total seasonal precipitation was 99 mm and observed in summer 2015, while the highest was 579 mm and recorded in spring

2014 (Figure 5.2b). Average annual precipitation was 1,308 mm, and 2013 and 2014 exceeded this average by 242 mm and 114 mm, respectively. Meanwhile, 2012 and 2015 were drier years with total annual rainfall of 1,100 mm and 1,133 mm, respectively.

The OFWS system investigated consists of a trapezoidal-shape TWR ditch and an elongated agricultural pond, which have a combined storage volume of 128,020 m^3 (TWR ditch: 13,320 m^3; Pond: 114,700 m^3). Water flows from north to south through the ditch, which is 818.8 m long and 1.8 m deep on average; the pond is 2.4 m deep with a surface area of 4.45 ha. Runoff is routed to the single outlet pipe (33°39'35.6" N, 90°39'11.9" W) set at 1.2 m above the canal bed (Figure 5.1). The system was designed according to NRCS (2011) guidelines, and more information about its characteristics can be found at Pérez-Gutiérrez et al. (2017b).

Model description

AnnAGNPS is a physical-process model developed to simulate runoff, sediment, nutrient, and pesticide yields at a daily time step in small watersheds. The model divides the watershed into subwatersheds based on homogeneous physical characteristics such as soil type, land use, and land management. AnnAGNPS is a continuous-simulation model and has been primarily developed to evaluate the impacts of different agricultural management conditions on watersheds. As with other physical-process watershed-scale models, the major input data are climate, land characterization, field operations, chemical characteristics, and feedlot operations. A detailed description of the model can be found in Bingner et al. (1998); Bosch et al. (1998); Cronshey and Theurer (1998); Geter and Theurer (1998); Theurer and Cronshey (1998).

Model input

A detailed field survey was conducted to identify field boundaries and collect elevation data required by the model. Eighteen fields were identified as subwatersheds (or cells) and associated reaches were defined for routing runoff to the outlet in the AnnAGNPS model (Figure 5.1). Because all fields were land leveled, they were defined as homogeneous drainage areas or subwatersheds. Delineation of the watershed was done manually with the aid of Google™ earth and geographic information system (GIS) technologies. Parameters describing the subwatersheds such as area, average elevation, and average land slope were determined from the field survey (Table 5.1). Parameters representing the time of concentration and travel time were computed from data provided by the field reconnaissance following USDA-SCS (1986) methods, modified by Theurer and Cronshey (1998). Soil data and physical properties were obtained from the Soil Survey Geographic (SSURGO) database (Soil Survey Staff, 2016). Although 15 types of soil were identified for the watershed, the dominant soil type was determined for each subwatershed as required by the model, using GIS operations (Table 5.1). Crop planting dates were obtained from the report of commodities farm provided by the Sunflower county USDA – NRCS office. A crop management schedule was assigned to each field according to the typical operations conducted in the MDR (Table 5.2). Irrigation was included in the crop management schedule, starting in late May or June and ending in August. The SCS curve number is an important model parameter used to estimate runoff. Table 5.3 shows the curve numbers used in the model, based on different land use categories and hydrologic soil types in the watershed. Weather data were recorded automatically at 15-minute intervals from March, 2012 to December, 2016 by a

WatchDog 2700 Weather Station (Spectrum® Technologies, Inc., Aurora, IL) located 9.2 km southeast of the outlet (M3). The weather data were subsequently processed to create daily time scale files as required by the AnnAGNPS model. Data from the PRISM Climate Group (http://www.prism.oregonstate.edu/explorer/) were used to fill any gaps in rainfall records. Other missing weather data such as daily maximum and minimum temperature, precipitation, dew point, wind velocity, and solar radiation were patched using data from the Moorhead Climate station, which is located 22 km south of the study site and managed by the National Oceanic and Atmospheric Administration.

AnnAGNPS was used to simulate runoff, sediment, and nutrient loads from 2010 to 2016. The first two years of simulation were established as a warm-up period, while the next five years were used for analysis of the loads entering the TWR. As described by Bosch et al. (1998), AnnAGNPS outputs are predefined by the user for the watershed source of interest (subwatersheds, reaches, among others). The model produces event-based output as well as monthly and annual summaries of hydrologic and water quality parameters. This study focused only on runoff, nitrate nitrogen ($NO_3 - N$), and total phosphorus (TP) load estimations. Although the model subdivided sediment into 5 particle size classes (clay, silt, sand, small aggregate, and large aggregate), only clay and silt were combined to represent sediment load in the simulated watershed. The model partitions nitrogen and phosphorus load into sediment-bound and dissolved fractions. Attached phosphorus, however, is additionally subdivided into inorganic and organic fraction. In this study, the AnnAGNPS dissolved nitrogen and TP outputs were used to represent $NO_3 - N$ and TP, respectively.

Results and discussion

Spatial variation

The average annual runoff production in the simulated watershed as estimated by the AnnAGNPS model is shown in Figure 5.3. During the 5-year simulation period (2012 – 2016), the model estimated an average annual runoff of 1,370,053 m^3 that drained into the main outlet, of which 11% is yielded by irrigation runoff. The five highest runoff-producing fields were C17 (9.73%), C9 (9.31%), C4 (8.78%), C6B (7.74%), and C3 (7.47%). These five fields cover a large drainage area (C17: 22.54 ha, C9: 18.7 ha, C4: 17, C6B: 15.74 ha, and C3: 16.85 ha; 90.83 ha out of the total area of 214.04 ha) when compared to other cells in the watershed, which may have explained the high runoff production. Nitrogen load transported throughout the watershed to the outlet resulted in 623.2 kg yr^{-1} (Figure 5.4). Five fields contributed 56.2% of the average annual nitrogen load, and the order was: C17 (18.41%) > C21 (16.15%) > C11 (8.29%) > C16 (7.03%) > C6A (6.36%). The average annual TP load from 2012-2016 in each subwatershed is shown in Figure 5.5, and the average annual TP load for the modeled watershed resulted in 256 kg yr^{-1} over this time period. Five fields contributed 66.2% of the average annual TP load in the following order: C9 (21.7%) > C3 (18.66%) > C17 (16.16%) > C6A (5.09%) > C21 (4.06%). The average annual sediment load resulted in 312.8 tons yr^{-1}, and five fields contributed 62% of the average annual sediment load (Figure 5.6) in the following order: C17 (36.18%) > C6B (7.47%) > C5 (6.45%) > C11 (6.03%) > C21 (5.84%). A summary of the impact of each subwatershed on water quantity and quality in the simulated watershed is shown in Table 5.4.

Subwatershed C17 was ranked first in runoff generation, sediment load, and nitrogen load and third in TP load. This is the subwatershed with the largest area and the second highest average slope among all subwatersheds. Soils in this field, classified as hydrologic soil group C, are shallow and have below-average infiltration, and thus have moderately high runoff potential. Furthermore, crops in this field were more diverse and varied between rice, corn, soybean, and winter wheat with assigned curve numbers ranging from 83 to 90. Higher curve numbers translate into higher runoff, and this effect is magnified over larger fields when rainfall occurs. Runoff transported higher loads of nitrogen when corn and winter wheat were planted because these two crops required 150 kg ha^{-1} and 120 kg ha^{-1} of soluble nitrogen fertilizer, respectively. However, winter wheat might have reduced TP load during the winter. Therefore, the combined effect of landscape characteristics and fertilizer application played a more influential role in the nutrient load from the simulated watershed. Subwatershed C9 was ranked 2nd with respect to the area, and was classified as hydrologic soil group D. In terms of runoff generation, nitrogen load, and TP load, C9 was ranked 2nd, 8th, and 1st, respectively. This subwatershed was planted in a soybean-rice rotation during four years and then planted with corn, which was fertilized with 150 kg ha^{-1} of soluble nitrogen and 13 kg ha^{-1} of phosphorus. Thus, the area of the subwatershed seemed to be an important factor in the TP load contribution in the simulated watershed.

Subwatershed C4 field was ranked 3rd and 13th, respectively, for magnitude of area and average land slope, and classified as hydrologic soil group D. Regarding water quantity and quality, C4 field was ranked 3rd in runoff generation, 15th in nitrogen load, 9th in TP load, and 7th in sediment load. Soybean was planted in this field throughout the five years of simulation. Subwatershed C6B was ranked 5^{th} and 3^{rd} with respect to the magnitude of area and average land slope, and was classified as hydrologic soil group D. In terms of runoff generation, nitrogen load, TP load, and sediment load, C6B was ranked 4^{th}, 6^{th}, 7^{th}, and 2^{nd}. This field was planted primarily with soybean except during 2013, when it was planted with corn. In addition, C3 was ranked 4^{th} and 9^{th} with respect to the magnitude of area and average land slope. In terms of runoff generation, nitrogen load, TP load, and sediment load, C3 was 5^{th}, 12^{th}, 2^{nd}, and 10^{th}. The C3 subwatershed was classified as hydrologic soil group D and simulated as turn area which explains the high TP load attached to sediments and transported by runoff. Finally, subwatershed C21 was ranked 10^{th} and 7^{th} with respect to the magnitude of area and average land slope, and classified as hydrologic soil group D. Regarding water quantity and quality, C21was ranked 9^{th} in runoff generation, 2^{nd} in nitrogen load, 5^{th} in TP load, and 5^{th} in sediment load. Similar to C17, subwatershed C21 was also planted with corn and winter wheat during two consecutive years in 2013 and 2014.

A list of the ten fields that had the highest impact on water quantity and quality in the simulated watershed is shown in Table 5.4. Five out of ten subwatersheds (C17, C6B, C21, C9, and C3) were ranked at least first or second with respect to runoff production, $NO_3 - N$ load, TP load, and sediment load.

Temporal variation

The total annual runoff production and nutrient and sediment load from the simulated watershed, as estimated by the AnnAGNPS model, are shown in Figure 5.7. During the 5-year simulation period (2012 – 2016), an average annual total runoff of 1,465,678 m^3 drained to the main outlet. This volume exceeds the TWR ditch storage volume by roughly 110 times, which highlights the magnitude of surface water availability in the simulated watershed. In terms of magnitude of runoff produced by year, the order was 2013 > 2014 > 2016 > 2015 > 2012 (Figure 5.7a). This order followed the same pattern observed for the total precipitation by year. Overall, fields draining upstream from M1 generated the highest volume of runoff at a rate of 952,578 $m^3 yr^{-1}$, which resulted in roughly 65% of the total runoff volume produced annually. Fields draining into the TWR ditch between M1 and M2 contributed 13.3% of the annual runoff production. Meanwhile, the fields draining into the ditch between M2 and M3 contributed 21.7% of the average annual runoff.

The changes in total annual nitrogen for each TWR ditch segment are shown in Figure 5.7b. Nitrate nitrogen was highest in 2015, followed in order of magnitude by 2014, 2013, 2016, and 2012. Overall, the area that drains into M1 was responsible for a greater percentage of the nitrogen load in the TWR ditch. Load entering the TWR ditch from surrounding fields downstream of M1 did not substantially contribute to the total load estimated at the outlet. In fall 2014, the subwatersheds with the highest contributing nitrate loads, C17 and C21, were planted with winter wheat which was fertilized with soluble nitrogen. Available nitrogen in soil after the fertilizer application was likely transported by runoff from winter and spring rainfall in 2015, which might explain the higher nitrogen load during this year. In addition, soybeans were planted on 84% of the simulated watershed during 2014. After soybeans are harvested, 33% of the nitrogen that was used by the plant is left over in the soil (IPNI, 2017), available for potential transformation mediated by microorganisms, and then might be transported off fields by runoff.

Total annual TP and sediment loads were higher in 2013, followed in order of magnitude by 2014, 2012, 2016, and 2015 (Figure 5.7c and 5.7d). The changes in TP and sediment loads were similar and can be attributed to the fact that phosphorus is usually transported as sediment-bound phosphorus. Similar to the pattern observed for $NO_3 - N$, the bulk of TP and sediments were from the area that drains into M1. In addition, loads entering the ditch between M2 and M3 were higher than the loads entering the system between M1 and M2.

In 2013, significant amounts of precipitation were recorded during the dormant season (winter, fall, and spring). This condition favored the production of runoff from fields with exposed soil during the dormant season. It is highly likely that erosion due to high runoff resulted in greater loads of TP and sediment in 2013. In addition, 41.4% of the watershed was planted with corn during the growing season, mainly in five (C17, C6B, C21, C6A, C5) of the highest runoff contributing fields in the simulated watershed (Table 5.4). Winter wheat covered 12.2% of the simulated watershed after the growing season in 2013. Both corn and winter wheat were fertilized with soluble nitrogen, which is reflected by the high nitrogen loads that were simulated by the model in 2013. Most of the load entered the ditch through the M1 outlet, while major load contributions from fields located west and east of the TWR ditch occurred between M2 and M3. In 2014, rainfall was highest during spring, roughly equally distributed between summer and fall, and minimal in winter. Runoff production, TP and sediment load mirrored the rainfall and runoff production pattern resulting in higher loads in 2014, although slightly lower than those observed during 2013. In contrast, the nitrogen load during 2014 was higher than during 2013. One possible explanation to this finding is that most of the fertilizer applied during spring and fall was transported off the field as most of the rain fell during these two seasons. In addition, during 2013 and 2014, 41.5% of the simulated watershed was planted with corn and 16.6% with winter wheat. Planting winter wheat after a year when corn was planted seemed to have resulted in increased nitrogen load transported with runoff. Rainfall was higher during winter and summer of 2016, and minimal during spring and fall. In 2016, fields were planted with corn (34.7%; fields C9, C11, C14, C16, C18, C19, and C20), soybean (26.5%; fields C4, C5, C6A, C6B), and rice (22.7%; fields C12, C17, C21). Runoff produced in 2012 and 2015 were the lowest among the 5-year simulation period, which may be attributed to smaller amounts of rainfall observed in 2012 and 2015 compared to other years. The water savings potential of the TWR ditch was estimated combining the simulated runoff and discharge water measured at the ditch outlet available for 2016 (Figure 5.8). Of the 1,526,105 m^3 of produced runoff, 56.5% (862,581 m^3) was saved by the ditch during 2016.

Table 5.5 presents the average annual runoff, nutrient and sediment loads entering the TWR ditch per unit area. The values represent the aggregate contributions divided by the area of the fields draining into a specific reach, and provide a unique picture of the impact of each source area regardless of its size. Annual runoff volume per unit area entering the ditch at segment M1 – M2 was slightly higher than the other reaches by about 600 m^3 ha^{-1} yr^{-1}, despite the fact that M1 – M2 had the smallest drainage area. This difference is attributed to the runoff potential of the fields draining into this reach (M1 – M2). Subwatershed C4 is one of the three fields with the highest area in the simulated watershed, and soil in this field has the highest runoff potential and is in hydrological soil group D. In addition, reaches transporting the produced runoff flow directly into the ditch. This condition minimizes the loads of water due to infiltration and evaporation resulting in a greater volume of water draining into the ditch.

$NO_3 - N$ load export per unit area was higher at M1 as most of the fields that were planted with corn and winter wheat during the simulation period are located upstream of M1. TP load entering the ditch between M2 and M3 tended to be higher. Subwatershed C3, one of the three fields that drains between M2 and M3, was the only field that was simulated as turn area. Typically, turn areas have compacted soils and do not have vegetation. This means that the soil in this field is highly susceptible to erosion and off-site movement of nutrients by rainfall-runoff, which might explain the higher TP load simulated by AnnAGNPS at this reach (M2 – M3). Sediment loading did not show much variation through the TWR ditch segments, indicating that the sediment load might be equally distributed over the total area contributing to the TWR ditch outlets.

Impact of additional agricultural management operations

In order to examine the impacts of management practices on water quality and quantity, two scenarios were implemented in AnnAGNPS, namely Scenario 1: planting winter wheat in priority subwatersheds, and Scenario 2: planting winter wheat in each subwatershed. The current management practices were set as the baseline scenario. Subwatersheds C17, C6B, C21, C9, and C3 were designated as priority watersheds. The impacts of management practices on estimated runoff, sediment and nutrient loads at different locations within the TWR ditch (M1, M2, and M3) are shown in Figure 5.9. Targeted implementation of management practices (Scenario 1) had a bigger effect in reducing loads at M1 than either M2 or M3. In particular, TP and sediment loads at M1 were reduced by 19.3% and 12.6%, respectively. However, under scenario 1, $NO_3 - N$

loading increased substantially at all locations primarily because winter wheat was fertilized with soluble nitrogen at a rate of 120 kg ha^{-1}. Therefore, large amounts of the nitrogen applied to fields may have been transporteded off by runoff.

Scenario 2 resulted in an even more substantial reduction than Scenario 1 in terms of TP and sediment loads. If winter wheat were planted in all subwatersheds, TP loads from the areas upstream of M1 will be reduced by approximately 38%. Reductions in TP loads at M2 and M3 were about 63% and 25%, respectively. Sediment loads reduction followed a similar pattern as for TP with a higher percentage reduction at M2 (45%) and M1 (24.2%) than at M3 (22%). The all winter wheat scenario also showed reduction in runoff produced by the simulated watershed. In total, 188,100 m^3 (97,900 m^3 at M1; 29,700 m^3 at M2; and 60,500 m^3 at M3) of runoff were reduced from fields draining to the TWR ditch. Despite the positive effects brought by planting winter wheat in the simulated watershed, AnnAGNPS simulations showed very high increases in the NO_3 – N load entering the ditch at M1 due to the nitrogen fertilization necessary to grow winter wheat. Although cover crops may reduce NO_3 – N leaching to groundwater, NO_3 – N load may increase with runoff due to biomass leaching during rainfall events (Miller et al., 1994). This effect might be exacerbated if soluble nitrogen is applied over poorly drained soils with a high runoff potential.

Summary and conclusions

The AnnANGPS model was used to quantify the water, sediment, and nutrient loading entering a TWR ditch implemented as part of an OFWS system in an agricultural watershed within the PBW, Mississippi. Simulations showed that fields with larger areas coupled with hydrologic soil group C or D resulted in higher runoff, and this condition mirrored the annual rainfall patterns. The volume of runoff exceeded the TWR ditch storage volume by roughly 110 times, mostly during the winter and spring seasons. Therefore, these seasons offer the highest potential for capturing excess water in the OFWS system. Results showed that the fields with larger areas also produced the highest total nutrient and sediment loads. AnnAGNPS simulations showed that nitrogen load was sensitive to fertilizer application. Therefore, during years when corn and winter wheat were planted, nitrogen loading increased compared to other years as these crops need nitrogen fertilization to grow. The TP and sediment loading patterns were similar and influenced by the hydrological temporal condition.

Simulations of different management scenarios indicated that planting winter wheat in the simulated watershed can benefit water quality by reducing the export of TP and sediment loads. However, winter wheat requires nitrogen fertilizer which can result in higher nitrogen laods washed off by runoff. In particular, if winter wheat were planted in the priority subwatersheds (Scenario 1), reductions of TP and sediments loads were about 19% and 13%, respectively, at M1. Although planting winter wheat in all fields (Scenario 2) may not be feasible, this scenario would result in substantial TP and sediment load reduction from the contributing areas draining to M1 (TP: 3%; sediment: 24%) and M2 (TP: 63%; sediment: 45%) at the ditch. Scenario 2 also showed that 188,100 m^3 of runoff can be reduced from fields draining to the TWR ditch.

Results of this study provide both stakeholders and agencies critical information needed to better identify where these systems can be implemented to improve water quality and relieve pumping pressure on groundwater in the Lower Mississippi River Alluvial Valley. In addition, this study suggests that agricultural watersheds in the MDR might produce substantial amounts of runoff, which could be an important source of water for irrigation if adequately managed. While managing the water availability during winter and spring, nutrient reduction benefits of OFWS systems can be maximized.

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Table 5.1 Basic characteristics and land use including planting dates of the subwatersheds.

Subwatershed ID	Area (ha)	Average elevation (ft)	Average land slope	Soil type	Hydrologic soil group	Land use				
						2012	2013	2014	2015	2016
C3	16.85	130	0.0012	Db - Dowling silty clay loam	D	TRNAR	TRNAR	TRNAR	TRNAR	TRNAR
C4	17.00	131	0.0010	Fm - Forestdale silty clay loam	D	Soybean May 11	Soybean Mar 21	Soybean May 11	Soybean May 4	Soybean May 9
C5	9.40	131	0.0011	Am - Dundee silt loam	C	Soybean May 10	Corn Mar 21	Soybean May 11	Soybean May 4	Soybean May 9
C6A	14.67	131	0.0018	Am - Dundee silt loam	C	Soybean May 10	Corn Mar 21	Soybean May 11	Soybean May 4	Soybean May 9
C6B	15.74	131	0.0018	Fb - Forestdale silt loam	D	Soybean May 10	Corn Mar 21	Soybean May 11	Soybean May 4	Soybean May 9
C7	13.63	134	0.0020	Fm - Forestdale silty clay loam	D	Pasture	Pasture	Pasture	Pasture	Pasture
C8	2.34	132	0.0006	Db - Dowling clay	D	Forest	Forest	Forest	Forest	Forest
C9	18.70	132	0.0008	Db - Dowling clay	D	Soybean Jun 12	Rice May 26	Soybean Jun 20	Rice May 3	Corn Apr 25
C11	12.22	132	0.0017	Fb - Forestdale silt loam	D	Soybean Apr 24	Soybean Jun 10 WW Oct 25	Soybean May 20	Soybean Apr 9	Corn Apr 25
C12	13.41	133	0.0004	Fb - Forestdale silt loam	D	Rice Apr 13	Corn Apr 18	Soybean May 8	Soybean Apr 30	Rice Mar 30
C13	1.16	133	0.0016	Fb - Forestdale silt loam	D	Forest	Forest	Forest	Forest	Forest
C14	13.89	133	0.0005	Fb - Forestdale silt loam	D	Soybean Apr 23	Soybean May 27	Soybean May 6	Soybean Apr 28	Corn Apr 25

Table 5.1 (Continued)

C16	8.89	134	0.0012	Dk - Silty clay	D	Soybean Apr 24	Soybean Jun 10 WW Oct 25	Soybean May 20	Soybean Apr 9	Corn Apr 25
C17	22.54	135	0.0019	Pa - Pearson silt loam	C	Rice Apr 13	Corn Apr 18	Soybean May 6 WW Oct 25	Soybean Jun 12	Rice Mar 30
C18	6.95	134	0.0011	Fb - Forestdale silt loam	D	Soybean Apr 24	Soybean May 18 WW Oct 25	Soybean May 6	Soybean Apr 30	Corn Apr 25
C19	6.66	135	0.0012	Dk - Silty clay	D	Soybean Apr 24	Soybean Jun 10	Soybean May 20	Soybean Apr 9	Corn Apr 25
C20	7.00	134	0.0004	Dk - Silty clay	D	Soybean Apr 24	Soybean May 18 WW Oct 25	Soybean May 7	Soybean Apr 30	Corn Apr 25
C21	12.99	134	0.0013	Dk - Silty clay	D	Soybean Jun 4	Corn Apr 18	Soybean May 18 WW Oct 25	Soybean Jun 12	Rice Mar 30

TRNAR: Turn area; WW: winter wheat

Table 5.2 Typical crop management operation for crops planted in agricultural watersheds within the MDR used in this study.

Cropland	Activity	Application rate
Soybean	Bedder	-
	Plant	-
	Harvest	-
	Disk	-
Corn	Bedder	-
	Sprayer (Pre)	-
	Plant	-
	Fertilizer	150 kg ha^{-1} (Soluble nitrogen)
	Fertilizer	13 kg ha^{-1} (Phosphorus)
	Sprayer (Post)	-
	Sprayer (Insecticide)	-
	Harvest	-
Rice	Sprayer (Pre)	-
	Plant	-
	Harvest	-
	Disk	-
Wheat	Plant	-
	Fertilizer	120 kg ha^{-1} (Soluble nitrogen)
	Harvest	-
	Burn stubble	-

Table 5.3 SCS curve numbers selected for runoff estimation relative to cropland at the agricultural watershed. Source: USDA-SCS (1985).

Cropland		Land cover class	Hydrologic soil type	
			C	D
Soybean	Plant	Soybean straight row (Poor)	88	91
	Harvest	Fallow + crop residue (Poor)	90	93
Corn	Plant	Rowcrop with residue	85	89
Rice	Plant	Rowcrop with residue	85	89
Wheat	Plant	Small grain straight row + crop residue (Poor)	83	86

Table 5.4 Rankings of 10 subwatersheds based on their impact on water quantity and quality in the simulated watershed.

Subwatershed	Rank				
	Runoff production	$NO_3 - N$ load	TP load	Sediment load	Total
C17	1	1	3	1	6
C6B	4	6	7	2	19
C21	9	2	5	5	21
C6A	8	5	4	6	23
C9	2	8	1	13	24
C11	10	3	10	4	27
C3	5	12	2	10	29
C16	11	4	8	9	32
C5	13	11	6	3	33
C4	3	15	9	7	34

Rows with highlighted records indicate that the individual rank resulted in either 1st or 2nd. Subwatersheds with highlighted rows were designated as priority fields.

Table 5.5 Production of average annual runoff and load of nutrient and sediments entering the TWR ditch.

Category	Unit	TWR channel reach		
		M1	M1 - M2	M2 - M3
Runoff	$m^3\ ha^{-1}\ yr^{-1}$	6,785	7,364	6,743
NO_3 - N	$kg\ ha^{-1}\ yr^{-1}$	4.05	0.96	1.7
TP	$kg\ ha^{-1}\ yr^{-1}$	1.19	0.77	1.56
Sediment	$ton\ ha^{-1}\ yr^{-1}$	1.65	1.54	1.28
Area	ha	140.38	26.40	47.26

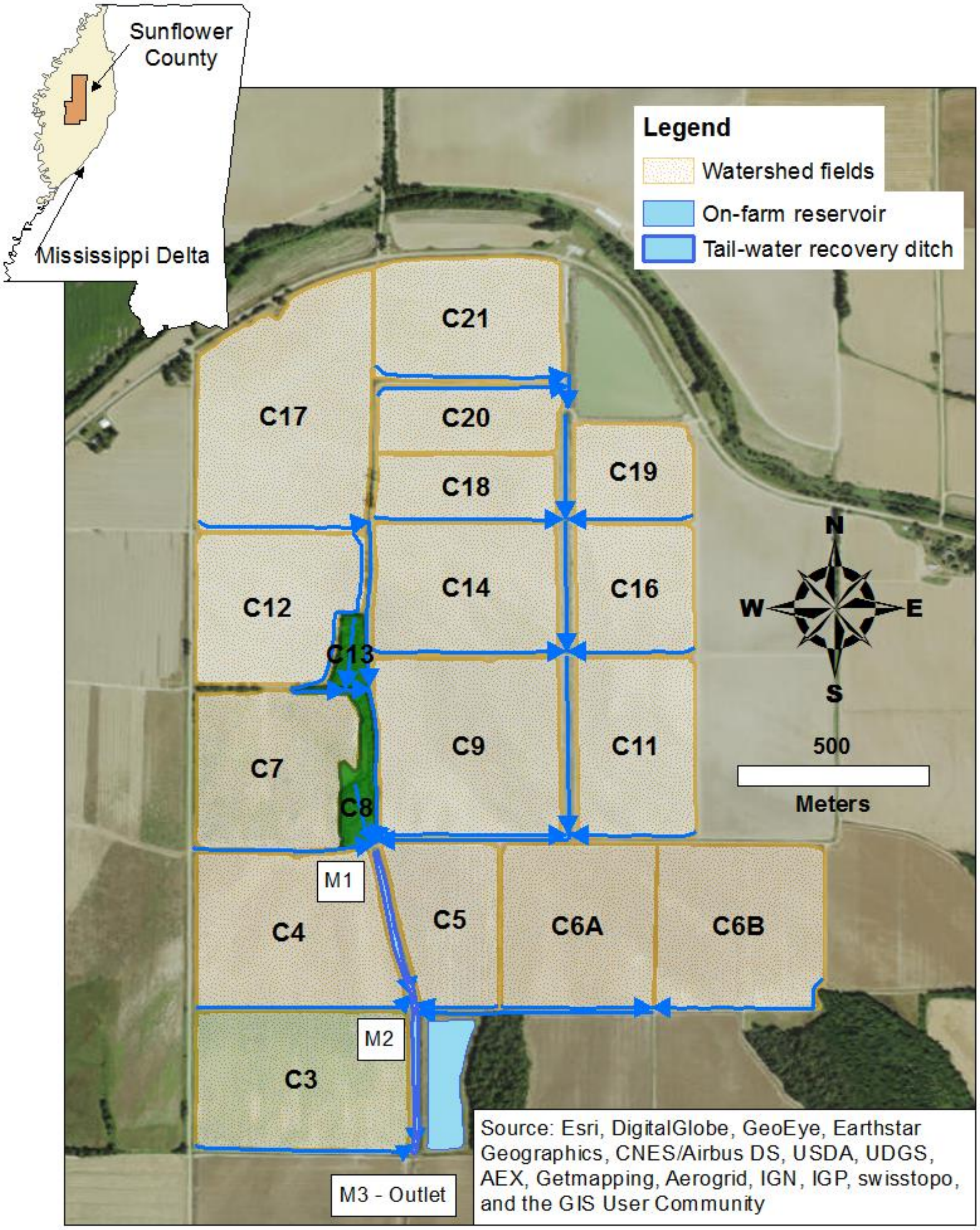

Figure 5.1 Map of the study area showing the simulated agricultural watershed implementing the on-farm water storage system investigated in the Mississippi Delta region.

Blue arrows represent runoff flow direction towards the outlet. (M1: TWR inlet; M2: TWR mid-canal; M3: TWR outlet; Labels within fields indicates the subwatershed identification used by the AnnAGNPS model).

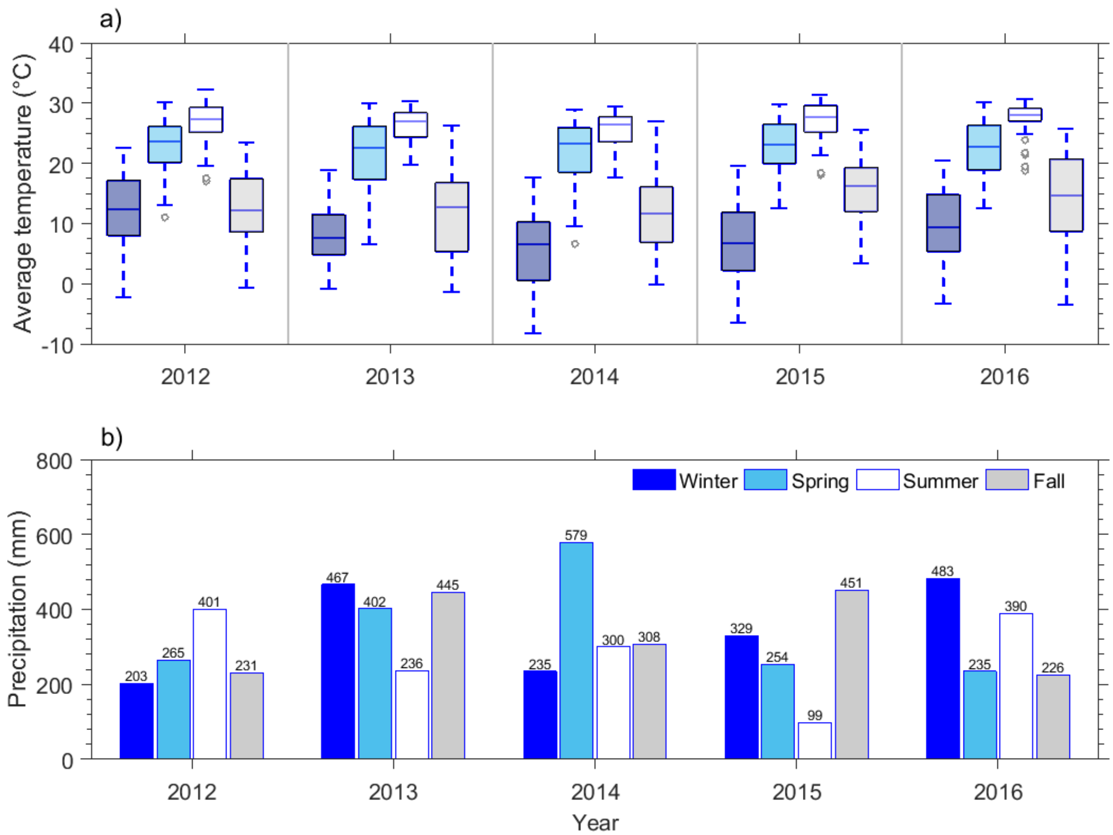

Figure 5.2 Average temperature and total precipitation by season during 2012 - 2016 at the study site. (a) boxplots of the average temperature. (b) bar chart showing the total precipitation.

Boxplots were set at 90th (the upper whisker), 75th (the upper quartile), 50th (the median), 25th (the lower quartile), and 10th (the lower whisker) percentiles. Outliers were considered those observations 1.5 times beyond the 25th and 75th percentile and are shown as grey circles.

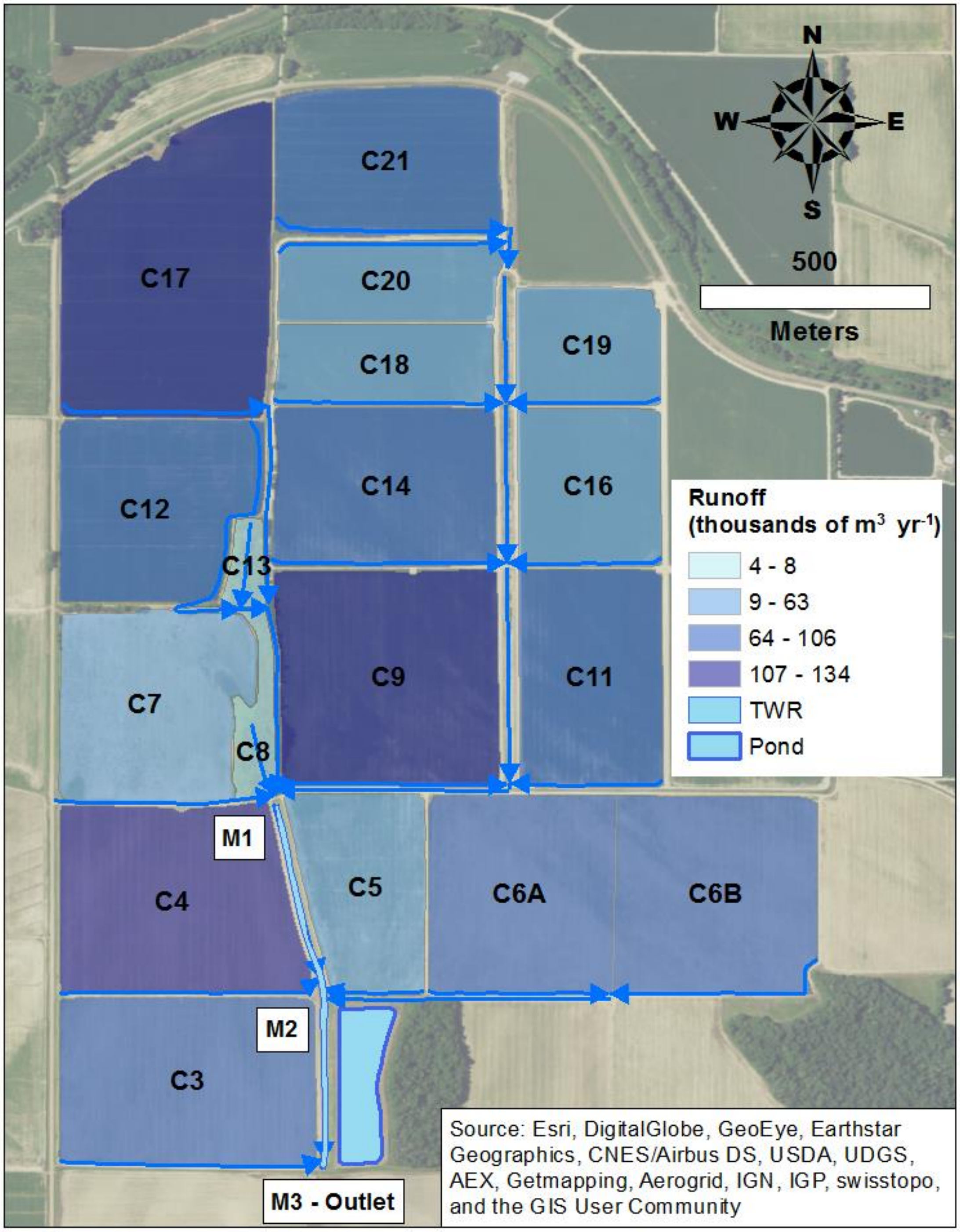

Figure 5.3 Map of the study area showing the average annual runoff production in the simulated watershed.

Blue arrows represent runoff flow direction towards the outlet. (M1: TWR inlet; M2: TWR mid-canal; M3: TWR outlet; Labels within fields indicates the subwatershed identification used by the AnnAGNPS model).

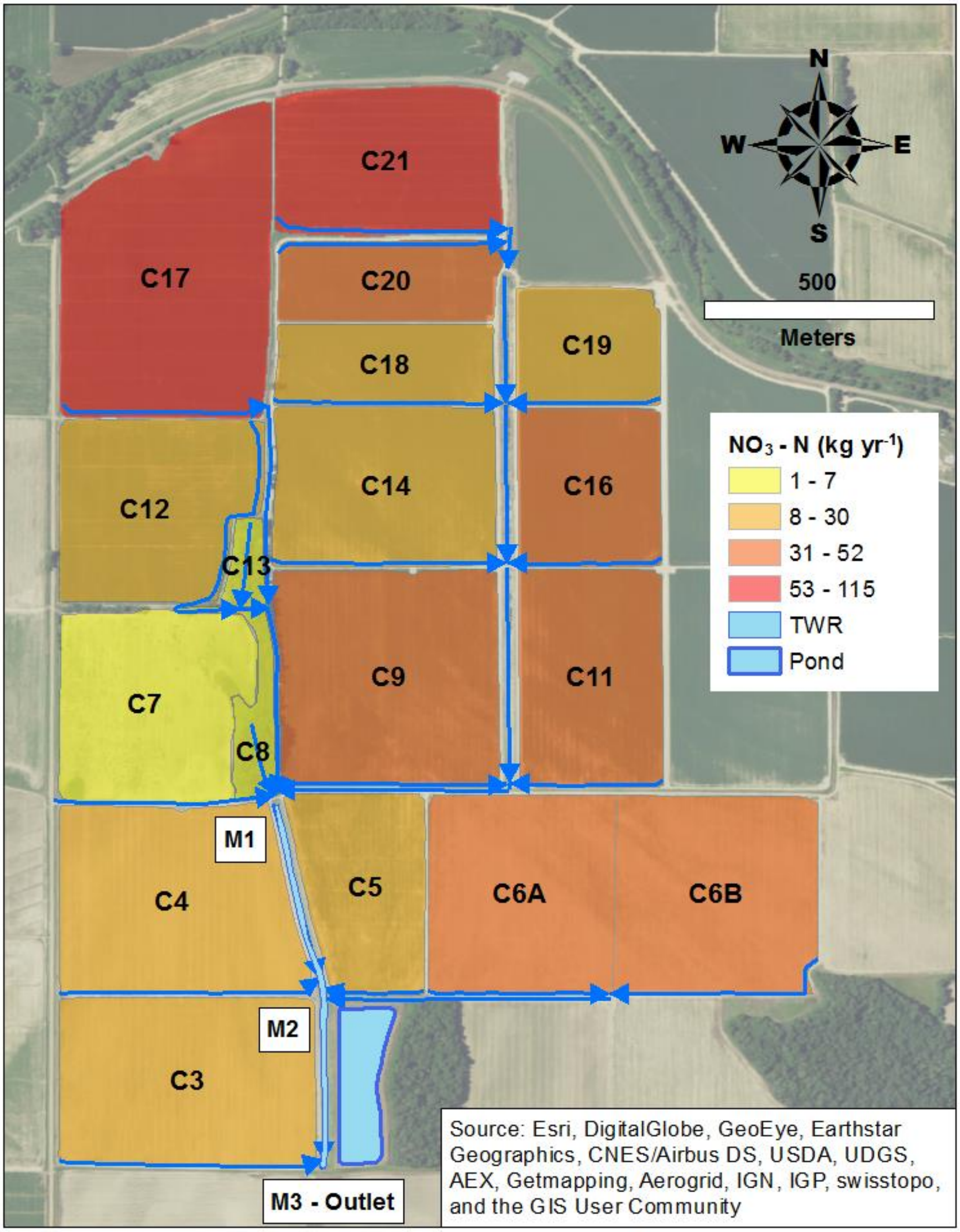

Figure 5.4 Map of the study area showing the average annual nitrogen load in the simulated watershed.

Blue arrows represent runoff flow direction towards the outlet. (M1: TWR inlet; M2: TWR mid-canal; M3: TWR outlet; Labels within fields indicates the subwatershed identification used by the AnnAGNPS model).

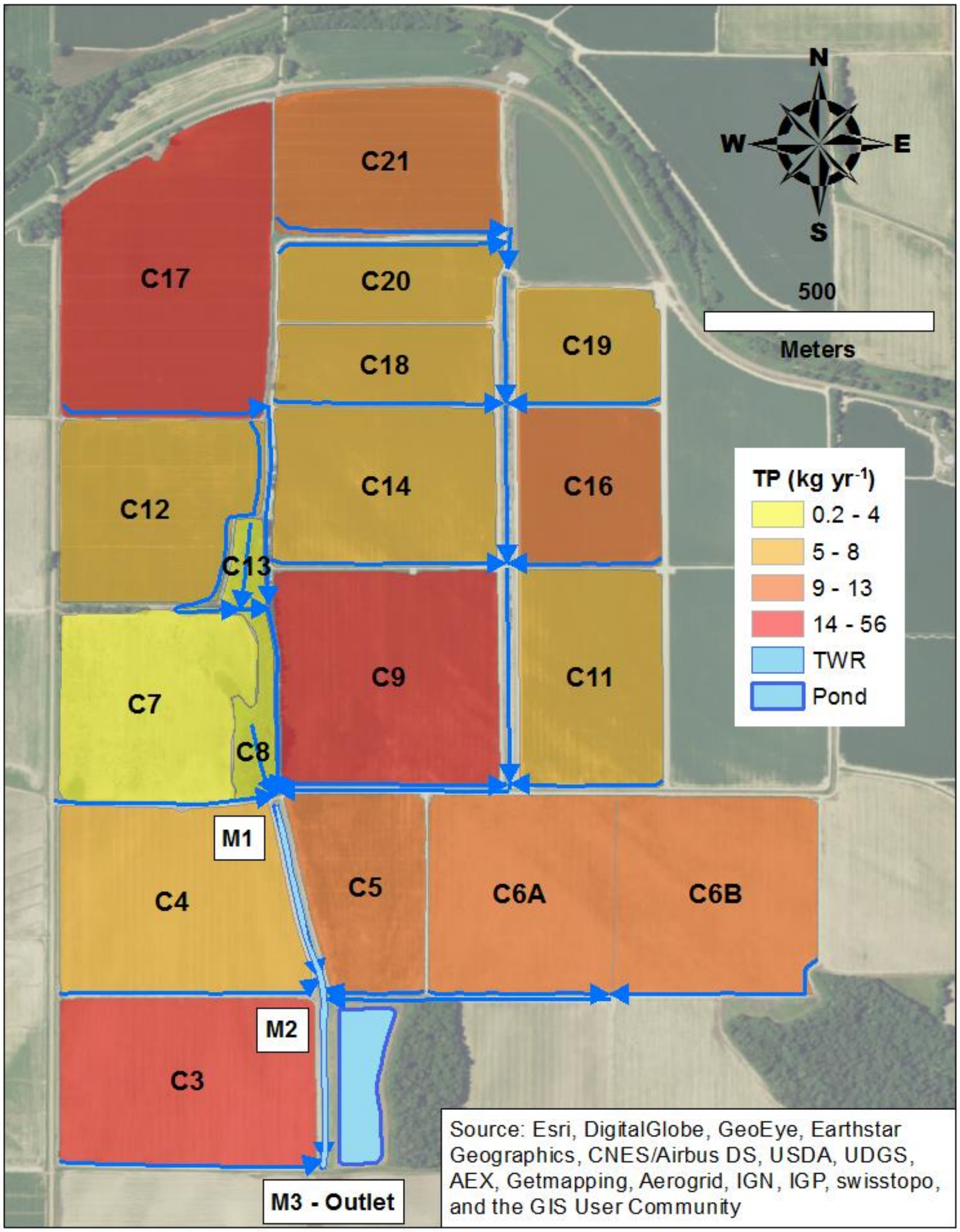

Figure 5.5 Map of the study area showing the average annual TP load in the simulated watershed.

Blue arrows represent runoff flow direction towards the outlet. (M1: TWR inlet; M2: TWR mid-canal; M3: TWR outlet; Labels within fields indicates the subwatershed identification used by the AnnAGNPS model).

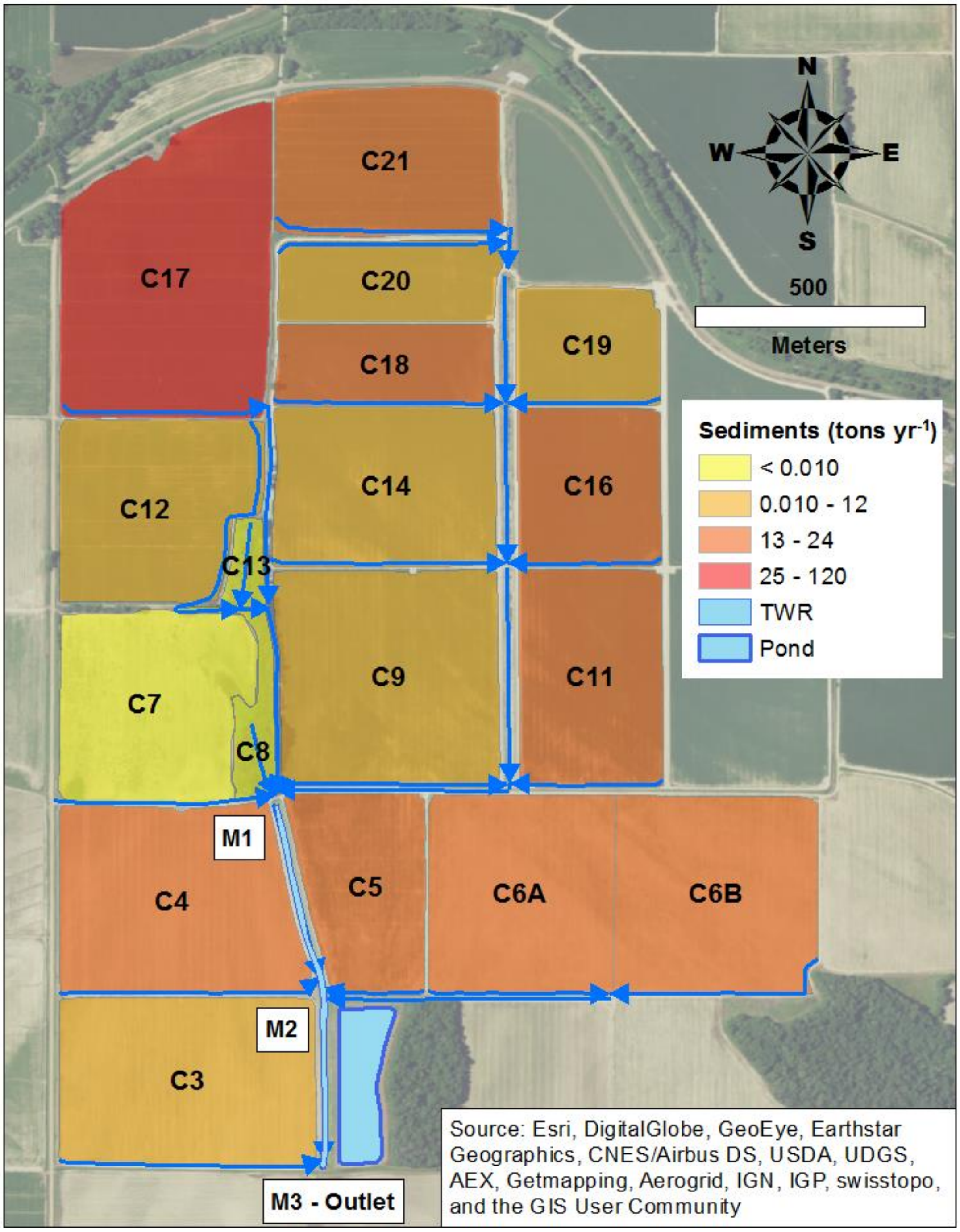

Figure 5.6 Map of the study area showing the average annual sediment load in the simulated watershed.

Blue arrows represent runoff flow direction towards the outlet. (M1: TWR inlet; M2: TWR mid-canal; M3: TWR outlet; Labels within fields indicates the subwatershed identification used by the AnnAGNPS model).

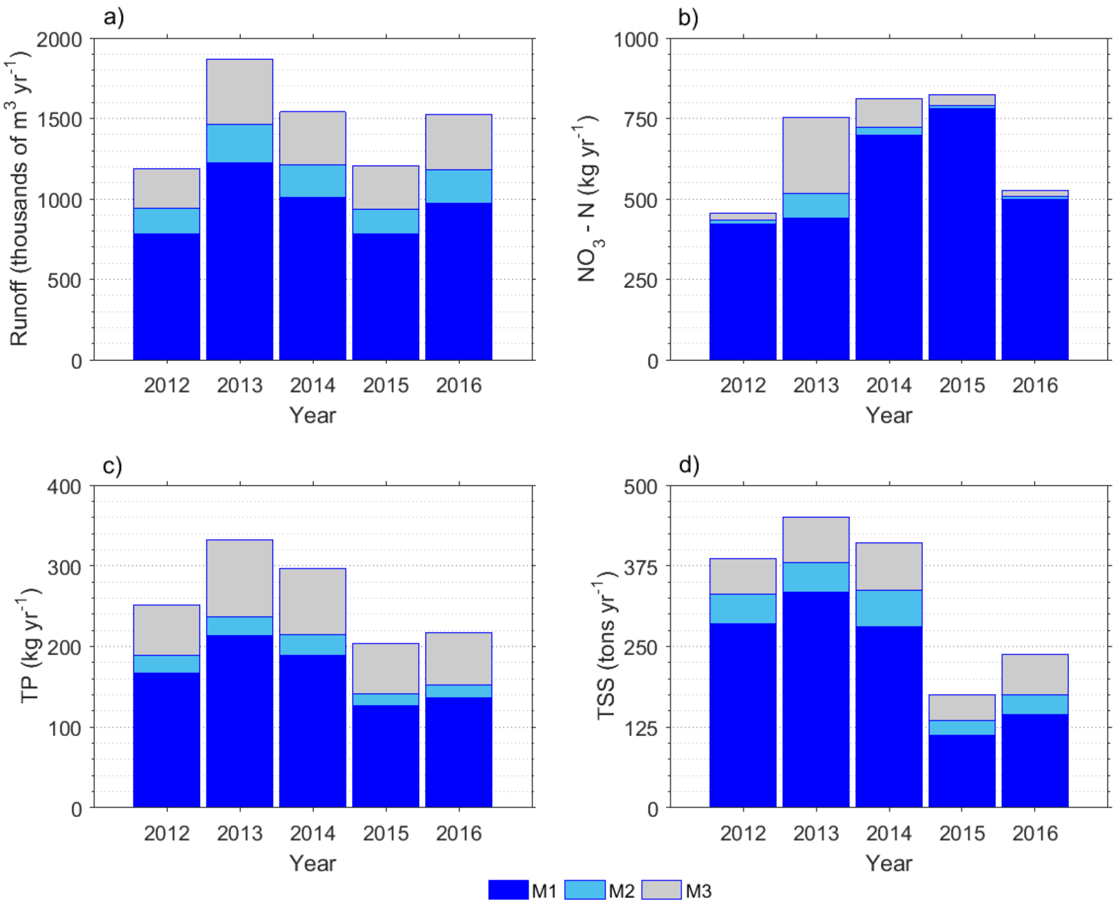

Figure 5.7 Comparison of (a) total annual runoff production, (b) total annual nitrogen load, (c) total phosphorus (TP) load, (d) and sediment load in the simulated watershed.

M1: TWR inlet; M2: TWR mid-canal; M3: TWR outlet.

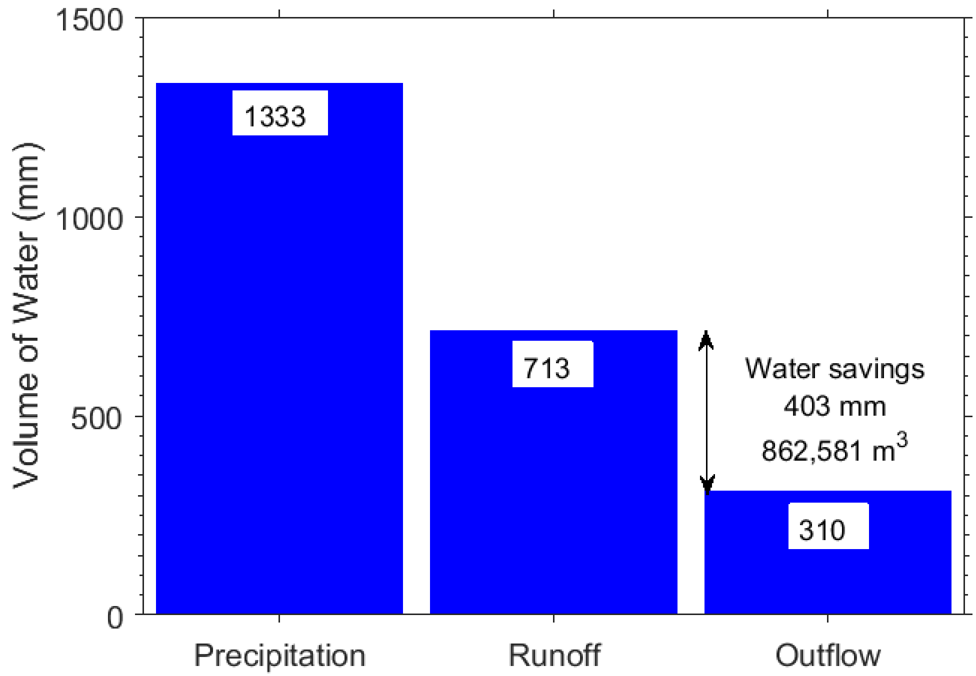

Figure 5.8 OFWS system water savings potential estimated for 2016.

Precipitation: Total precipitation; Runoff: AnnAGNPS simulated runoff; Outflow: water discharge measured at the outlet pipe (Pérez-Gutiérrez et al., 2017a)

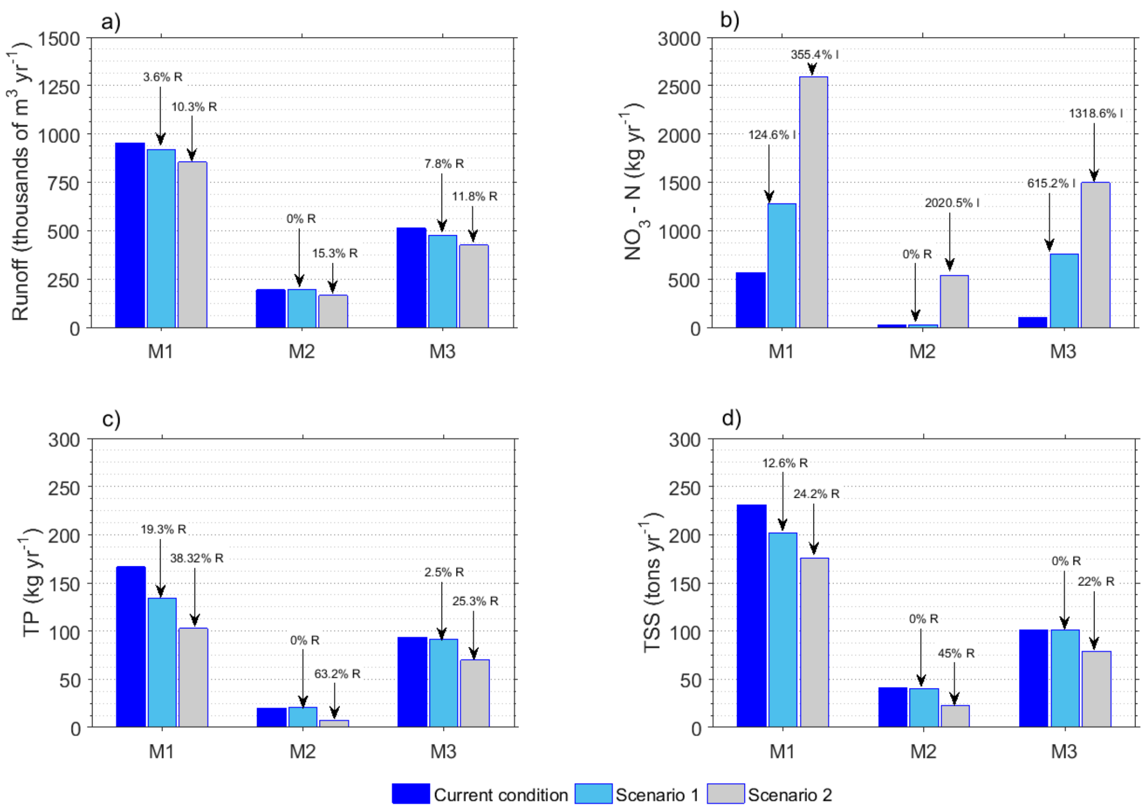

Figure 5.9 Impacts of two additional agricultural management operations in the simulated watershed on the average of (a) total annual runoff production, (b) NO_3 – N load, (c) TP load, and (d) sediment load.

Text arrows indicate the reduction or increase percentage relative to the current condition scenario; R: reduction; I: increase; WW: winter wheat; M1: TWR inlet; M2: TWR mid-canal; M3: TWR outlet; Scenario 1: planting winter wheat in priority subwatersheds; Scenario 2: planting winter wheat in each subwatershed.

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CHAPTER VI

GENERAL CONCLUSIONS

The major conclusions of the water quality monitoring and modeling studies of OFWS systems in Porter Bayou Watershed are:

1. Significant water quality changes were observed in the monitored OFWS system by season and nutrient species. The in-ditch median removal efficiency, from the center of the tailwater recovery (TWR) ditch to the outlet, was 54% during winter and 50% during spring for NO3-N; 60% during spring for NH_3-N; 26% during autumn and 65% during winter for ortho-P; and 31% during winter and 10% during spring for TP. The in-pond median concentration removal efficiency was ~77% during summer for NO_3-N as the concentration remained stable during winter, spring and autumn; 53% from winter to spring and 58% from spring to summer for NH_3-N; 70% from winter to spring for ortho-P, while remaining stable during the rest of seasons; and 28% from winter to spring and 55% from spring to summer for TP. The results favor the hypothesis that OFWS systems could mitigate downstream nutrient-enrichment pollution, especially during spring. More importantly, this study provides a better insight into the behavior of OFWS systems and help enhance the management of agroecosystems from an ecological and hydrological perspective for water quality pollution control and water resource conservation.

2. $NO_3 - N$ concentrations observed in the ditch were strongly dependent on antecedent hydrological conditions within the study area with (1) duration of rainfall events before sampling and (2) characteristics of next-to-last rainfall events playing a more influential role. Results indicated that next-to-last rainfall events should be accounted for when understanding the nutrient reduction potential of TWR ditches. The rainfall classes identified by using the *k*-means clustering approach provided information which has significant implications for future design, operation, and management of TWR ditches for more efficient nutrient control strategies.

3. Season analysis of discharge water and nutrient load from OFWS system indicates that winter season contributed the most to the total annual estimated $NO_3 - N$ and TP load, followed in order of magnitude by spring, fall, and summer. In addition, effluent from the OFWS system was strongly dependent on the seasonality of depth and distribution of rainfall. Higher peak discharges with longer time peaks were predominant during winter which resulted in more nutrient load transported to downstream waterbodies. The potential impact on downstream water quality and aquatic ecosystems is associated with the transition from wet to dry seasons and the alteration derived from varied outflows events by each hydrological period.

4. The AnnAGNPS model was implemented to simulate runoff, sediment, and nutrient load and to identify the main contributing areas from a small catchment with a TWR ditch. Simulations showed that fields with larger areas coupled with poorly drained soils resulted in higher runoff, and that this condition mirrored the annual rainfall patterns. The volume of runoff exceeded the TWR ditch storage volume by roughly 110 times, mostly during the winter and spring seasons. During years when corn and winter

wheat were planted, nitrogen load increased compared to other years as these crops need nitrogen fertilization to grow. TP and sediment load patterns were similar and influenced by the hydrological temporal condition. Comparison of different management scenarios indicate that planting winter wheat in the simulated watershed can benefit water quality by reducing export of TP and sediment loads. However, this management practice can result in higher nitrogen load washed off by overland flow because winter wheat requires nitrogen fertilizer. Quantification of the water, nutrient, and sediment loading constitutes an essential step towards an improved understanding of the benefits of TWR ditches on availability and quality of water when implemented in agricultural watersheds.

Recommendations for future research

The water quality and quantity monitoring and modeling approach used in this study provided essential insights into the OFWS systems benefits. However, high-frequency nutrient measurements are required to better understand the biogeochemistry in OFWS systems. In addition, water quality high-frequency measurements would provide critical data to perform calibration, validation, sensitivity and uncertainty analysis when a modeling tool is used to simulate the impacts of OFWS systems on agricultural watershed hydrology. It is also critical to work hand in hand with farmers and producers operating OFWS systems to understand the challenges in managing these systems.