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Assessing monitoring and modeling approaches to improve water quality in the Hickory Grove Lake

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Iowa State University

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Assessing monitoring and modeling approaches to improve water quality in the Hickory
Grove Lake

by

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A Dissertation submitted to the graduate faculty
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

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TABLE OF CONTENTS

	Page
LIST OF FIGURES	v
LIST OF TABLES	vii
NOMENCLATURE	viii
ACKNOWLEDGEMENTS	x
ABSTRACT	xii
CHAPTER 1 GENERAL INTRODUCTION	1
1.1 Introduction	1
1.2 Water Quality Legislation	3
1.3 Pervasive Contaminants in the Midwest Watersheds.....	4
1.4 Overview of the TMDL Program	7
1.5 Organization of the Dissertation	10
1.6 References	11
CHAPTER 2 PHOSPHORUS TRANSPORT PATHWAYS DURING RAINFALL- RUNOFF EVENTS IN A TILE DOMINATED AGRICULTURAL WATERSHED	15
2.1 Abstract	15
2.2 Introduction.....	16
2.3 Materials and Methods	19
2.3.1 Experimental site description.....	19
2.3.2 Hydrology and water quality monitoring	21
2.3.3 Chemical hydrograph separation	23
2.4 Results	24
2.4.1 Spring precipitation and phosphorus monitoring in study area	24
2.4.2 Spring rainfall event characteristics in 2013	26
2.4.3 Hydrograph for monitored events	27
2.4.4 Phosphorus dynamics during runoff events	29
2.4.5 Phosphorus fluxes during event 3	30
2.4.6 Chemical hydrograph separation	31
2.4.7 TP, DRP, and SBP loads and sources for event 3.....	33
2.5 Discussion	34
2.6 Conclusion	36
2.7 Acknowledgements	37
2.8 References	37

	Page
CHAPTER 3 NEAR SHORE BEACH VOLUME MODELING APPROACH FOR SETTING BEACH BACTERIA TMDLs: A CASE STUDY, HICKORY GROVE LAKE, IOWA	41
3.1 Abstract	41
3.2 Introduction.....	42
3.3 Materials and Methods	46
3.3.1 Study area	46
3.3.2 Source of fecal bacteria in the watershed	47
3.3.3 Water quality monitoring	48
3.4 Analysis of Monitoring Data	49
3.4.1 Effect of precipitation on <i>E. coli</i> concentrations at the lake beach.....	49
3.4.2 Comparison of <i>E. coli</i> concentrations at the lake inlet to the lake beach.....	50
3.4.3 Comparison of <i>E. coli</i> concentrations at the lake outlet to the lake beach.....	53
3.4.4 Spatial sampling	54
3.5 Near-shore Beach Volume Model	56
3.6 Beach Bacteria TMDL	57
3.7 Limitations of the Approach	59
3.8 Monitoring Recommendations and Analysis.....	60
3.9 Conclusions/Lessons Learned	61
3.10 Acknowledgements.....	62
3.11 References	62
CHAPTER 4 IDENTIFYING POTENTIAL LOCATIONS FOR GRASSED WATERWAYS USING TERRAIN ATTRIBUTES AND PRECISION CONSERVATION TECHNOLOGIES	66
4.1 Abstract	66
4.2 Introduction	67
4.3 Methods	70
4.3.1 Study area	70
4.3.2 Elevation data	72
4.3.3 Compound topographic index model	73
4.3.4 Grassed waterway design	74
4.3.5 Model evaluation	75
4.4 Results and Discussion	76
4.4.1 Plan curvature	76
4.4.2 Erosion feature identification	78
4.4.3 Impacts of CTI Threshold on grassed waterway identification	80
4.4.4 Design of grassed waterways	82
4.4.5 Implications of precision conservation	84

4.5 Conclusions	85
4.6 Acknowledgements	85
4.7 References	86
CHAPTER 5 CONCLUSIONS.....	89
5.1 Review of Introduction	89
5.2 Review of Phosphorus Transport Pathways	89
5.3 Review of a Novel Approach to Set Beach Bacteria TMDL	90
5.4 Review of GWW Identification using Topographic Attributes	90
5.4 Implications of the Study	91
5.4 Future Recommendations	92
APPENDIX GRASSED WATERWAY IDENTIFICATION AND DESIGN PROCESS PYTHON CODE	94

LIST OF FIGURES

	Page
Figure 1.1 World's water withdrawal and its use by sector.....	2
Figure 2.1 Map of Hickory Grove Lake Watershed showing water quality monitoring locations, tile drainage main line locations (field-scale tiles are not mapped), and surface intakes.....	20
Figure 2.2 Three-year record of TP and DRP concentrations at TO (a) and SO (b), Hickory Grove Lake watershed, Iowa.....	25
Figure 2.3 Hourly discharge measurements at TO and SO for monitored rainfall events.....	28
Figure 2.4 Box plots showing DRP and TP concentrations for the monitored events. Box plots showing median (line in the middle of box), 25th quartile (lower box limit) and 75th quartile (upper box limit).....	30
Figure 2.5 TP and DRP concentrations at TO and SO during event 3. Note that the scale and axes are different for DRP and TP.....	31
Figure 2.6 Chemical hydrograph separation of total discharge (Q) into surface intakes, subsurface-tile flow components at TO and surface runoff component at SO.....	32
Figure 2.7 Cumulative loads for TP and DRP at TO and SO during event 3.....	33
Figure 3.1 Location of the Hickory Grove Lake and the sampling locations in the watershed.....	47
Figure 3.2 Relationships between rainfall and <i>E. coli</i> concentrations at the Hickory Grove Lake beach from 2004 through 2012.....	50
Figure 3.3 <i>E. coli</i> concentrations at the Hickory Grove Lake beach and at the lake inlet during the years 2010, 2011, and 2012.....	52
Figure 3.4 Relationships between <i>E. coli</i> concentrations at the Hickory Grove Lake beach and at the lake inlet.....	53
Figure 3.5 Relationships between <i>E. coli</i> concentrations at the Hickory Grove Lake beach and at the lake outlet from 2010 through 2012.....	54

Figure 3.6 <i>E. coli</i> spatial sampling in Hickory Grove Lake on August 30th, 2011. Values shown on the map are the <i>E. coli</i> concentrations in cfu/100 ml	55
Figure 4.1 Location of Hickory Grove Lake watershed in Iowa showing grassed waterways and field boundaries	71
Figure 4.2 Plan curvature (m/100) for Field 9 and 29 overlain by grassed waterway boundaries	77
Figure 4.3 CTI model predictions of GWWs and their correspondence with existing GWW locations in the HGLW	79
Figure 4.4 CTI model performance statistics at various thresholds	80
Figure 4.5 Model predicted locations for GWWs in Field 9 at CTI threshold of 30	82
Figure 4.6 Comparison of observed and predicted GWW lengths.....	83

LIST OF TABLES

	Page
Table 2.1 Hydrological response to precipitation characteristics at TO and SO in the HGLW.....	26
Table 2.2 DRP and TP median, 25th, and 75th quartile concentrations measured at sites TO and SO for runoff events during 2010 and 2012	29
Table 2.3 Cumulative TP, DRP, and SBP loads (g/ha/event) during event 3 at TO and SO, Hickory Grove Lake, Iowa	34
Table 3.1 Near shore beach volume model parameter values used for setting the TMDL	58
Table 4.1 Grassed waterway attributes utilized in the design process.....	74
Table 4.2 Error matrix table to assess the predictive performance of CTI model ...	76

NOMENCLATURE

ArcGIS	Arc Geographic Information System
BMP	Best Management Practice
CFU	Colony Forming Units
CHS	Chemical Hydrograph Separation
CRP	Conservation Reserve Program
CTI	Compound Topographic Index
DEM	Digital Elevation Model
DRP	Dissolved Reactive Phosphorus
<i>E. coli</i>	Escherichia coli
EPA	Environmental Protection Agency
FIB	Fecal Indicator Bacteria
g	gram
GM	Geometric Mean
GWW	Grassed Waterway
h	Hour
ha	Hectare
HGLW	Hickory Grove Lake Watershed
HUC	Hydrologic Unit Code
IDLE	Interactive DeveLopment Environment
IDNR	Iowa Department of Natural Resources
KDHE	Kansas Department of Health & Environment

L	Liter
LA	Load Allocation
LDC	Load Duration Curve
LiDAR	Light Detection And Ranging
MOS	Margin Of Safety
mg	Milligram
MPN	Most Probable Number
NRCS	Natural Resources Conservation Service
NSBV	Near Shore Beach Volume
P	Phosphorus
PBIAS	Percent Bias
PLANC	Plan of Curvature
TO	Tile Outlet
TP	Total Phosphorus
TMDL	Total Maximum Daily Load
SBP	Sediment Bound Phosphorus
SO	Subwatershed Outlet
SSM	Single Sample mean
SWAT	Soil and Water Assessment Tool
µg	Microgram
USDA	United States Department of Agriculture
WLA	Waste Load Allocation
WQRL	Water Quality Research Laboratory

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ABSTRACT

Surface water quality regulated by agricultural pollution remains to be an important environmental concern around the world. Major contaminants from agriculture systems such as bacteria, sediment, and nutrients (nitrogen and phosphorus) continue to affect the designated use of a waterbody. As per the Clean Water Act legislation, water quality impairments must be addressed through the Total Maximum Daily Load (TMDL) approach. The TMDL program is a comprehensive and watershed-scale approach involving contaminant source identification and quantification, and conservation practice recommendation to reduce contaminant transport. The overall goal of this study was to improve the TMDL development process in achieving water quality goals and restoring impaired waterbodies. Specific objectives were to: (1) identify phosphorus transport pathways during rainfall-runoff events in a tile-dominated agricultural watershed; (2) demonstrate a novel approach in setting a bacteria TMDL for an impaired waterbody and; (3) determine potential locations for conservation practice placement at the watershed-scale to maximize reduction of sediment transport. The Hickory Grove Lake located in central Iowa, a waterbody impaired due to *E. coli* levels at the swimming beach was the focus of this study.

Phosphorus (P) transport pathways in the tile drained agricultural watershed were determined through intensive monitoring during runoff events and a chemical hydrograph separation (CHS) method. Rainfall events in Spring 2013 were monitored for flow, Dissolved Reactive Phosphorus (DRP) and Total Phosphorus (TP) concentrations at the tile outlet (TO) and subwatershed outlet (SO) in the Hickory Grove Lake Watershed (HGLW). The drainage areas of TO and SO are 879 ha and 852 ha, respectively. The discharge at TO comprises

runoff from surface intakes and flow from subsurface tile-drains, whereas discharge at SO comprises flow from TO and surface runoff during runoff events. The median TP concentrations during spring runoff events in 2013 at TO and SO were 0.89 mg/L and 1.13 mg/L, respectively. The TP and DRP levels at TO and SO during low flow and high flow conditions were similar. The highest P levels at TO and SO were observed during the rising limb of the hydrograph. Surface intakes accounted for 15.2% of the total discharge at SO and 23.6% of the total discharge at TO. It was also estimated that 28.2% of the TP load at SO originated from surface intakes. Due to surface intake contribution to subsurface tile-drains, similar P concentrations were observed in TO and SO. This study improves understanding of the P dynamics and transport pathways in tile drained agricultural watersheds. Therefore, contaminant source identification and quantification during TMDL development must acknowledge the underappreciated transport pathway of P (surface intake) in tile drained watersheds.

The Hickory Grove Lake beach was listed on Iowa's 303d list of impaired waters due to elevated *E. coli* concentrations, and therefore, a novel approach was proposed to develop a bacteria TMDL. Fecal bacteria monitoring data at the Hickory Grove Lake Inlet, Lake Outlet, and Lake Beach was used to develop linear regression relationships and understand the influence of fecal bacteria sources in the watershed on the Lake Beach *E. coli* levels. It was determined that fecal bacteria from the HGLW had very little effect on *E. coli* levels at the Lake Beach, instead fecal bacteria from waterfowl were regulating the *E. coli* levels at the beach. Spatial monitoring of the lake suggested that *E. coli* levels were elevated at the Lake Beach and at other locations where geese reside year-round. A TMDL developed using a Near-Shore Beach Volume model was set at 1.8×10^{11} cfu/day for the single sample mean

(SSM) target and 1.01×10^{11} cfu/day for the geometric mean target. The daily fecal bacteria load from as few as 5 resident geese were sufficient to cause *E. coli* levels at the Lake Beach to exceed the SSM standard. Therefore, efforts to achieve the bacteria TMDL must focus on deterring the resident geese at the lake.

Conservation practice recommendation and placement to mitigate contaminant transport is the next step after TMDL development. Spatial monitoring of the Hickory Grove Lake in November 2012 indicated that the east basin of the lake is now filled with sediment. The Light Detection and Ranging (LiDAR) data and precision conservation technologies were used in this study to identify potential locations for grassed waterway (GWW) placement in the HGLW to reduce sediment transport. The compound topographic index (CTI) model supplemented with 3 m LiDAR data was used to identify GWW locations. The CTI model identified all existing GWWs and recommended new locations for GWW placement at a CTI threshold of 30. The CTI model overestimated the lengths of existing GWWs suggesting a need to further extend the GWWs. The design recommendations of the predicted GWWs suggested that the total surface area required for predicted GWWs was 29.3 ha. The results of this study imply that LiDAR derived terrain attributes can be effectively used in identifying potential locations for GWWs.

The overall results of the complete study suggest that conventional TMDL development may not be appropriate for all impaired waterbodies; a novel and holistic approach is required depending on the contaminant source and its transport pathways, watershed characteristics, and hydrology of the watershed.

CHAPTER I. GENERAL INTRODUCTION

1.1 Introduction

“High quality water is more than the dream of the conservationists, more than a political slogan; high quality water, in the right quantity at the right place at the right time, is essential to health, recreation, and economic growth.” (EDMUND S. MUSKIE, U.S. Senator, speech, 1 March 1966)

Water is essential for the survival and functioning of all living organisms. In fact, history provides evidence that water played a major role in the advancement of all the great civilizations. Civilizations such as the Roman Empire, Egyptian Civilization, Indus-Valley Civilization, and the Omayyad Dynasty flourished around water. The efficient management of water resources has been and will be a driving factor of survival for any civilization or community. Water, plays an important role in every nation’s economic development, directly or indirectly affecting the agriculture, tourism, goods and energy production sectors. The Food and Agriculture Organization (FAO) estimates that only 2.5% of all the water on earth is fresh water, and only 1% of the earth’s fresh water is available for water withdrawal and human use (FAO, 2009). But the quality of the earth’s fresh water threatens to affect human health and the expansion of the industrial and agricultural sectors. Typically, water quality is influenced by the hydrological processes leading to runoff; biological processes within the waterbody; atmospheric processes such as deposition and evapotranspiration; leachate from the soils; and weathering of rock minerals. The above described processes can alter the physical, chemical and biological composition of a waterbody. As the hydrologic cycle transports water from one place to another,

anthropologic activities such as agriculture, mining, construction, and habitation influence the hydrologic system but do not, however, return the water in the same condition it was obtained.

Agriculture has been identified as the major cause of degradation of surface and groundwater resources (FAO, 1996). Agriculture dominates the water withdrawals on global basis (Figure 1.1); accounting for more than 70% of the total water withdrawal in Asia and 43% in North America (FAO, 2009). The total fresh water withdrawal in the U.S is 477 km³/y, and the agriculture sector accounts for 41% of it (FAO, 2009). The United States (U.S.) has over 442 million acres of cropland, accounting for 19.5% of the total land area (Lubowski et al., 2006). Major crops grown in the U.S. are corn, soybean, wheat, hay, cotton, and rice. The U.S. is the largest producer of corn in the world and in 2012 corn production in U.S. was estimated at 10.8 billion bushels (USDA, 2013). In contrast to these positive outputs, agricultural activities such as improper, excessive and poorly timed application of fertilizers, pesticides, and irrigation; improperly managed cover crops, livestock access to streams, runoff from animal feeding operations, and poorly timed pre-planting soil preparations can affect water quality.

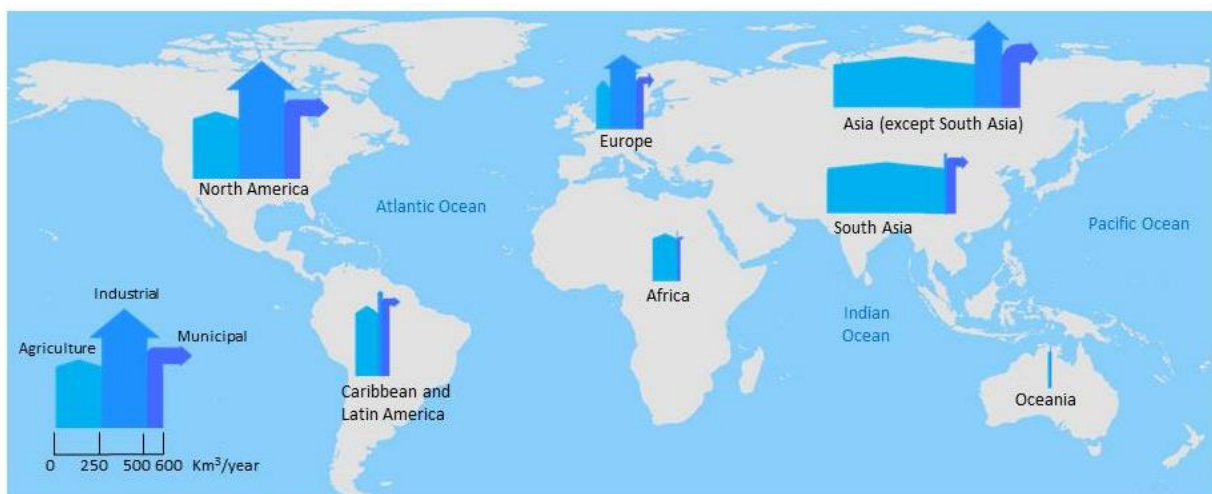


Figure 1.1 World's water withdrawal and its use by sector (Modified from: FAO, 2009)

1.2 Water Quality Legislation

Water quality monitoring is the first step towards effective management of water resources. Waterbodies around the world are being monitored temporally and spatially to assess their condition. The physical, chemical and biological characteristics of a water sample determine the quality of a waterbody. The characteristics of water samples are compared with water quality criteria to determine the quality, and criteria often vary with the intended or designated use of the waterbody. The designated use for each waterbody is determined by the use and value of the waterbody for municipal, aquatic and wildlife, agricultural, or industrial and navigational purposes. Water quality is affected by natural, wildlife and human influences.

In order to protect U.S. waters and prevent water pollution, the federal government passed the Water Pollution Control Act (WPCA) in 1948 to address water pollution (EPA, 2013). To better address continuing major water quality problems, in 1965 the Water Quality Act was passed providing funds to develop water quality planning programs and water quality standards for interstate waters (EPA, 2013). Major amendments were made to the WPCA in 1972 to develop water quality standards for all surface waters and regulate point sources/end-of-pipe discharges to surface waters through National Pollutant Discharge Elimination System (NPDES); the WPCA is now referred to as the Clean Water Act (CWA) (EPA, 2013). Additional amendments made to the CWA in 1977, 1983, and 1987 emphasized the importance of nonpoint sources of pollution. The condition of the waterbodies are assessed by monitoring for pollutants by the Environmental Protection Agency (EPA) or the local state water pollution control agencies under section 305(b) CWA. Section 305(b) requires each state to monitor, assess and report the quality of its waters to the EPA. The water quality assessment for each waterbody type indicated that 52% of assessed rivers and streams are impaired and 68% of the assessed lakes,

reservoirs and ponds are impaired (US Environmental Protection Agency, 2012a). A waterbody is considered impaired if it does not meet the designated use water quality criteria. The major causes of impairments are pathogens, sediment, and nutrients; and agriculture has been cited as a major source of these contaminants (US Environmental Protection Agency, 2012b).

1.3 Pervasive Contaminants in Midwest Watersheds

Before the 19th century much of the agricultural lands in Great Lakes and Corn Belt states were natural swamps and wetlands. Due to natural high water tables in these swamps and wetlands, artificial subsurface drainage is required to farm these lands. A subsurface drainage system removes excess water from the plant root zone to increase soil aeration and creates a healthier environment for plant growth. Subsurface drainage systems improve trafficability allowing easy movement of farm equipment in agricultural fields during cultivating, planting, fertilizing, and harvesting crops. Subsurface tile drainage is a widespread practice in agricultural lands of the Midwest US, transforming poorly drained soils and inoperable wetlands to highly productive agricultural lands. Many of the world's productive and sustainable agricultural soils are on drained soils (Gilliam et al., 1999). A subsurface drainage is a network of perforated tiles or PVC pipes installed at a depth of 3 - 4ft below the soil surface. The subsurface water drainage into perforated tiles is achieved by gravity and the drained surplus water from soils is routed laterally to streams/surface waters.

Subsurface drainage, also referred to as groundwater table management, has both positive and negative implications for surface water quality. Tile drains increase infiltration into the soil by removing the excess water in subsurface soils and reduces overland flow, thereby reducing surface runoff, peak flow rates, soil loss or erosion from agricultural fields and phosphorus attached to sediment. At the same time tile drains enhance nitrate leaching potential by providing

a direct pathway for nitrates from subsurface soils to surface waters. Nutrient enrichment in coastal zones is often associated with nitrogen and phosphorus runoff from agricultural watersheds. The over enrichment of nutrients in waterbodies accelerates growth of phytoplankton; as phytoplankton die, the natural bacterial degradation exhausts the dissolved oxygen in the waterbodies. This leads to low dissolved oxygen levels, coral reef damage, loss of aquatic vegetation, and fish kills. Globally, the most pervasive water quality problem is eutrophication, notably the “dead zone” in the Gulf of Mexico.

The National Academy of Sciences Committee on Causes and Management of Coastal Eutrophication has reported that nitrogen is the major cause of eutrophication in coastal marine waters in the U.S. (NRC, 2000). The Upper Mississippi River Basin (UMRB) which has an area of 493,900 km² and areal coverage in Minnesota, Wisconsin, Iowa, Missouri and Illinois is the major contributor of nitrate N to the Gulf of Mexico (US Environmental Protection Agency, 2007). It is estimated that tile drainage from the Upper and Central Mississippi Basins (which includes portions of Minnesota, Wisconsin, Iowa, Missouri, and Illinois) accounts for 39% of the N delivered to the Gulf of Mexico (Alexander et al., 1995). The subsurface drainage system releases nitrate-N loads approximately 10 times higher than surface runoff (Gilliam and Skaggs, 1986) and much of the nutrient loading to the Mississippi river and Gulf of Mexico occur during times of peak discharge (Royer et al., 2006). The average nitrate-N yield in the most heavily tile-drained areas of North America (from southern Minnesota into the Des Moines lobe of Iowa and north central Illinois, Indiana and Ohio) was more than 7.51 kg N ha⁻¹ for the period 1997 to 2006 (David et al., 2010).

Phosphorus, an essential element for plant life is added to croplands to increase the crop productivity. Phosphorus runoff from croplands accelerates eutrophication, causing depletion of

oxygen levels in fresh waters (Klatt et al., 2003). Soil erosion is identified as major contributor of phosphorus to surface waters. Ferber et al. (2004) reported that agricultural runoff contributes an average 436,000 tons of sediment per day enriched with phosphorus to the Mississippi river, which is being deposited into the Gulf of Mexico. Various studies have documented the importance of phosphorus in runoff from agricultural fields to streams (US Environmental Protection Agency, 1998; Sharpley et al., 1999; Hart et al., 2004). Phosphorus in runoff can be divided into organic and inorganic, particulate and dissolved forms. The proportions of dissolved and particulate forms of phosphorus in runoff vary with landuse dominance and transport mechanisms (overland flow, preferential flow, interflow, and subsurface drainage) in watersheds (Gentry et al., 2007). The pathways of phosphorus transfer are still poorly understood in tile dominated watersheds and various studies have reported the importance of phosphorus loss in agricultural drainage (Heckrath et al., 1995; Sims et al., 1998; Stampfli and Madramootoo, 2006; Gentry et al., 2007). Phosphorus loss in subsurface drainage systems in various forms must be quantified to evaluate the importance of subsurface flows on phosphorus loss with respect to surface runoff.

Pathogen impairments in waterbodies can occur either from wildlife or human influences such as leaky septic systems, waste water treatment plants, combined sewer overflows, or activities from agricultural animals such as runoff from grazing lands and confined animal feeding operations, livestock access to streams, or defecation of migratory birds (Paul et al., 2006; Teague et al., 2009). Studies have reported that presence of indicator organisms in high levels is related to the risks of human illness. Fecal indicator bacteria (FIB) such as fecal coliform, *E. coli* and enterococci are being used to detect the presence of pathogens. Elevated levels of FIB in waterbodies increases the risk of water-borne diseases (gastroenteritis and

cholera) (US Environmental Protection Agency, 1986). The Centre for Disease Control and Prevention reported that waterborne disease outbreaks were associated with recreational waters and drinking water in U.S. during 1978-2008 (CDC, 2011).

Nitrate-nitrogen, phosphorus and *E. coli* are the most prevalent contaminants in the Midwest watersheds (Tomer et al., 2010). These contaminant must be monitored at least weekly during normal flow conditions and frequently during runoff events to accurately assess the total contaminant loads received by downstream waterbodies. Funding high frequency water quality monitoring projects requires millions of dollars, and therefore alternatives for more economic water quality monitoring are needed. Also, identifying the major transport pathway of these contaminants will help to strategically design water quality monitoring networks.

In order to protect and restore the nation's waters and achieve the overall goals of the CWA, states are required to prepare and submit a list of impaired waters to the USEPA, which is referred to as 303(d) list (US Environmental Protection Agency, 2001). The impaired waterbodies are then ranked based on the severity of the pollution and designated uses of the waterbodies. The next step towards restoring these impaired waterbodies is to develop Total Maximum Daily Loads (TMDLs) for each impaired waterbody.

1.4 Overview of the TMDL Program

A TMDL is defined as the total amount of pollutant that a waterbody can receive and still meet its designated use water quality criteria. TMDLs can be expressed in either amount of pollutant entering the water body or other appropriate measures that relate to a State's water quality standard (US Environmental Protection Agency, 1991; NRC, 2001). The TMDL program was established by the U.S. Congress in the Clean Water Act of 1972, Section 303, and until the 1990s, the EPA and States focused on regulating point source pollution through NPDES

permitting (US Environmental Protection Agency, 2005). A TMDL is estimated by using the equation below:

$$\text{TMDL} = \Sigma\text{WLA} + \Sigma\text{LA} + \text{MOS}$$

Where, WLA represents the waste load allocation for point sources (e.g. wastewater treatment plants, CAFOs), LA represents the load allocation for nonpoint sources (e.g. agricultural runoff) and MOS is the margin of safety. The USEPA (US Environmental Protection Agency, 2001) has two approaches for determining the MOS in TMDLs: Implicit and Explicit. Implicit approaches involve conservative assumptions in setting numeric targets for water quality standards and explicit approaches involve the addition of a numeric safety factor to pollutant loading estimates.

The TMDL process involves waterbody monitoring to identify impairments, prioritizing the impaired waterbodies, developing the TMDL and watershed management plan, implementing the management plan, and assessing the effectiveness of the management plan. There are four major approaches for TMDL development: Modeling, Narrative, Data Driven/waterbody monitoring, and Alternative Approaches (such as phased, pilot watershed, reference watershed, and ecological assessment). TMDLs are developed using both water quality monitoring and mathematical models. Often watershed-scale models/modeling approaches are used to develop TMDLs due to the scarcity of monitored data. The TMDL program rely heavily on models to estimate the nutrient load reductions necessary to achieve the water quality standards and to estimate effectiveness of BMPs.

Among the many models used to develop TMDLs, Hydrologic Simulation Program-Fortran (HSPF) model, Soil and Water Assessment Tool (SWAT), and Water Quality Analysis Simulation Program (WASP) models are often used to develop nutrient and bacteria TMDLs and

to study the impacts of landuse and BMP alternatives on water quality (Santhi et al., 2001; Zhang and Yu, 2004; Benham et al., 2005; Benham et al., 2006; Franceschini and Tsai, 2008; Schilling and Wolter, 2009). The selection of an appropriate model to develop a TMDL depends on various factors such as the ability of the model to simulate pollutant transport, consideration of pollutant sources, simplicity of the model, and model input data requirements. The model selection for TMDL development depends on the frequency of water quality impairment. If the frequency of water quality impairment is predominantly during rainfall-runoff events, an event-based model can be used and if the impairment is during low-flow conditions, a continuous simulation model is recommended. Appropriate models are also selected based on their ability to simulate point/nonpoint source pollution.

TMDL development often involves using both watershed loading models and receiving waterbody models. The selection of watershed loading models depends on the sources of pollution, time scale of model simulation, algorithms used, and mainly on the project objectives. The receiving waterbody type also influences model selection. One-dimensional mixing models are preferred for rivers/streams that are laterally and horizontally well mixed; a two-dimensional models for lakes/reservoirs with vertical stratification; a three-dimensional models for estuaries/coastal waters is normally preferred where complex circulation patterns and temperature gradients dominate (US Environmental Protection Agency, 1997).

The validity of the TMDL developed using the models depend on how well the model is calibrated with monitored data. The high quality and frequency of monitored data is needed to properly link pollutant sources with observed water quality. A TMDL also includes developing a watershed management plan. This plan includes identifying and quantifying the sources of pollutants, recommend BMPs to reduce pollutant loadings, cost to implement the BMPs, time

line to implement and achieve water quality goals, and monitoring recommendations for implemented BMPs. Diebel et al. (2008) concluded that the key to effective watershed management involves identifying and targeting critical source areas that need BMP implementation to achieve overall water quality goals.

1.5 Organization of the Dissertation

This study focused on addressing the problems highlighted during the TMDL development process. This dissertation follows the steps involved in the TMDL process (Monitoring - Modeling - Implementation): chapter 2 focuses on monitoring approaches, chapter 3 focuses on TMDL development and finally chapter 4 focuses on assisting with the BMP implementation plan. Chapter 2 investigates the transport pathways of phosphorus in a subsurface-tile dominated agricultural watershed. Chapter 3 demonstrates a beach bacteria TMDL development method using a near-shore beach volume model. Chapter 4 focuses on mapping of potential locations for grassed waterways and grassed waterways design at the field scale. Chapter 5 follows on to general conclusions and future recommendations.

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CHAPTER 2. PHOSPHORUS TRANSPORT PATHWAYS DURING RAINFALL-RUNOFF EVENTS IN A TILE DOMINATED AGRICULTURAL WATERSHED

This paper to be submitted to Journal of Environmental Quality

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2.1 Abstract

Excess phosphorus (P) in surface waterbodies has been associated with algal productivity and dissolved oxygen depletion. Timing and role of hydrology unique to each landscape regulates the P export from the agricultural watersheds. It is therefore critical to understand the P transport pathways to minimize P losses through effective mitigation strategies. This study investigates the P concentrations, fluxes and transport pathways in tile drains and surface runoff at a high temporal resolution during a runoff event in Hickory Grove Lake Watershed, Iowa. The goals of this study are 1) to determine how the surface intakes regulate the P transport in tile drains during a runoff event; and (2) to quantify the dissolved reactive phosphorus (DRP) and total phosphorus (TP) transport in tile drain

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landscape dominated by surface intakes. The cropland dominated watershed is completely pattern tiled with subsurface drains; discharging combined tile flow from an 879 ha cropland at the tile outlet (TO). The surface runoff from the upland drainage area (852 ha) and the tile flow from TO converge at the subwatershed outlet (SO). The TO and SO were monitored for flow, DRP and TP levels during runoff events in Spring 2013. The median TP concentrations during spring runoff events in 2013 at TO and SO were 0.89 mg/L and 1.13 mg/L, respectively. Similar TP and DRP concentrations were observed at TO and SO during runoff events, while peak TP and DRP concentrations were observed at the beginning of runoff event. Chemical hydrograph separations indicated 23.6% of tile discharge from TO originated from surface intakes, while 28.2% of TP load at SO originated from surface intakes. The role of surface intakes in P transport to tile drains during runoff events stresses the underappreciated transport pathway of P in tile drained agricultural watersheds.

Keywords: Phosphorus, surface intakes, tile drain, transport pathway, runoff.

2.2 Introduction

Phosphorus (P) transport from agricultural fields to surface waters is an important environmental concern around the world (NRC, 2005). Excess P in fresh waterbodies has been attributed to eutrophication by the U.S. Environmental Protection Agency (USEPA) and impedes their use for recreation and domestic water supplies. Experimental studies on Louisiana coastal waters suggest that P plays an equal and important role with nitrogen in the occurrence of gulf hypoxia and the Upper Mississippi River Basin (UMRB) has been identified as one of the major contributors of P to the Gulf of Mexico (US Environmental Protection Agency, 2007; Sylvan et al., 2006). Studies have reported an increase in total phosphorus (TP) concentrations in streams in the U.S over the last decade (David and

Gentry, 2000; Sprague and Lorenz, 2009) and the increase in TP concentrations in streams correlates well with increased fertilizer use in the U.S (Economic Research Service, 2012). Therefore there has been much interest on understanding the processes regulating the P mobility to streams in the US Midwest (Tomer et al., 2010; Vidon et al., 2012; Royer et al., 2006).

Soils in agricultural systems replenished with P via fertilizer or manure are recognized as major sources of P to surface waters (Royer et al., 2006; McDowell and Wilcock, 2004). P in the soil environment can be found as dissolved reactive phosphorus (DRP) and particulate phosphorus (PP) and the sum of DRP and PP is referred to as TP. The DRP, which is the most bioavailable form of P generally accounts for less than 50% of TP in streams (Royer et al., 2006; McDowell and Wilcock, 2004) and 10 – 22% of TP during rainfall events (Vidon and Cuadra, 2011). It is well known that P sorbs to soil particles and that majority of P exports from agricultural systems is in PP form (Hansen et al., 2000; Kurz et al., 2005; Sharpley et al., 1999). Surface runoff has been identified as dominant P transport mechanism in agricultural watersheds and is a well-studied topic (Sharpley et al., 1994). Royer et al. (2006) observed that runoff events during late winter-early spring were associated with majority of the annual P export. For instance, Vidon et al. (2008) reported average TP concentrations of 0.12 mg/L over a 12-month period and in excess of 0.17 mg/L during events in agricultural streams in Indiana. TP concentrations in excess of 0.035 mg/L can contribute to eutrophication in surface waterbodies (Vollenweider and Krekes, 1982). Many agricultural watersheds in the U.S. Midwest are dominated by poorly drained soils, therefore subsurface drainage systems were installed to lower the water table and ensure good soil conditions for crop. But, these subsurface drainage systems have modified the

natural hydrology of agricultural watersheds by providing a transport mechanism for nutrients.

P movement into subsurface soils has received relatively little attention as it was perceived that P leaching is negligible to subsurface soils. Experimental studies on P leaching in soils that are not artificially drained observed TP concentrations of 0.1 mg/L and in excess of 0.2 mg/L during late spring in leachate (Turner and Haygarth, 2000). Research suggests that preferential flow pathways such as macropore flow (soil matrix flow) can be important in P transport to subsurface soils (Stone and Wilson, 2006; Geohring et al., 2001). Stone and Wilson (2006) used a hydrograph separation method and conservative mixing analysis to identify impacts of preferential flow in a tile drained agricultural field and estimated that preferential flow during storms contributed up to 51% of the total storm tile drain flow. Geohring et al. (2001) conducted a field study to evaluate P movement to tile drains and observed high P concentrations in tile flow after liquid dairy manure application, which was attributed to soil macropore flow. Recent research also suggests that subsurface drainage plays an important role in P loss to streams (Tomer et al., 2010; Chapman et al., 2001). The tile drained landscapes supplemented with surface drainage of potholes and roadside ditches can transport both DRP and PP in ponded surface runoff via tile drains to streams. Smith et al. (2008) emphasized that the hydrology is dominated by subsurface tile drainage in pothole landscapes and that nutrient loss is highly sensitive to direct drainage of potholes. Tomer et al. (2010) used hydrograph separation method to identify and quantify pollutant transport pathways and found that about 13% of the tile flow and about 75% of the TP in tile originated from surface intakes during a rainfall-runoff event.

Nevertheless, in spite of these studies investigating the importance of soil macropore and surface intakes on P transport to tile drains, the extent to which surface intakes regulate DRP and TP transport from the soil surface to tile drains during runoff events remains poorly quantified. Quantifying P transport during runoff events is very difficult in tile drained landscapes due to altered transport pathways. Thus the overall goal of this study was to determine the P transport pathways during high flow events in a tile dominated agricultural watershed, including the significance of surface intakes during runoff events on P transport. The specific objectives of this study are (1) to determine how the surface intakes regulate the P transport in tile drains during runoff events; and (2) to quantify the DRP and TP transport in tile drain landscape dominated by surface intakes. The DRP and TP concentrations were monitored at tile outlet and surface runoff outlet at high temporal resolution during three rainfall events in spring 2013. Identifying the significance of surface intakes during runoff events on P transport is needed for the development of strategies to control agricultural P pollution.

2.3 Materials and Methods

2.3.1 Experimental site description

The study was conducted in the Hickory Grove Lake Watershed (HGLW), located in the South Skunk River Basin (HUC ID: 07080105) and approximately 30 km east of Ames, IA (Figure 2.1). The watershed lies in the southern part of the Des Moines Lobe region, a landscape formed during the Wisconsin glacial period. As the glaciers retreated, the deposition of till over Wisconsin loess created deep, rich soils well suited for row crops; however, these soils characteristically have poor natural drainage. Over 65% of the soils in the HGLW are dominated by poorly drained loams, including poorly drained Canisteo-

Coland-Harps-Webster soils (43.5% of the drainage district area) and very poorly drained Okoboji soils (1.3% of the drainage district area) where ponding occurs frequently. As in much of the Upper Midwest, artificial drainage has been installed in this watershed to lower the water table, in order to remove excess water from the root zone to support crop growth. The subsurface tile drainage system and other drainage infrastructure is managed within the context of a drainage district (Figure 2.1) pre Iowa code (Story County, 2014). DRP and TP concentrations were monitored at the Tile Outlet (TO), which drains about 879 ha, and at the Subwatershed Outlet (SO) which is located about 30 m downstream from the TO site (Figure 2.1).

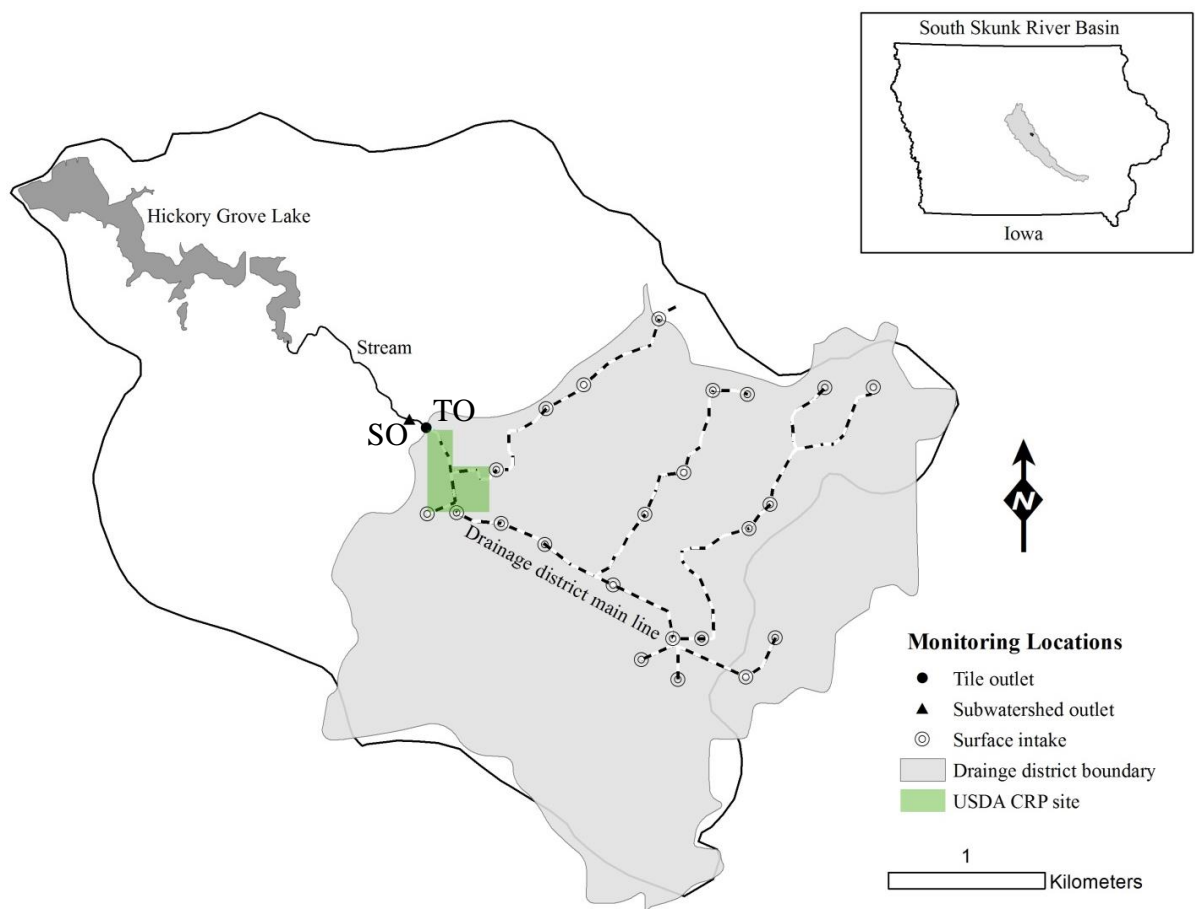


Figure 2.1 Map of Hickory Grove Lake Watershed showing water quality monitoring locations, tile drainage main line locations (field-scale tiles are not mapped), and surface intakes.

The HGLW (total 1629 ha) is dominated by corn (*Zea mays* L.) and soybean [*Glycine max* L. (Merr.)] row crops and about 84% of the watershed is under cropland with corn - soybean crop rotations. Field and farmer surveys conducted in Spring 2012 indicated that typical fertilizers in the study area were Diammonium phosphate, Anhydrous Ammonia and Urea Ammonium Nitrate (UAN - 32). Diammonium phosphate was applied to soybean crops prior to planting at a rate of 85 kg/ha and nitrogen (Anhydrous ammonia/UAN-32) was applied to about 200 ha in the south part of the watershed in the fall after soybean harvest and in the spring prior to corn planting at a rate of 156 kg/ha in the remaining fields. The watershed is fairly flat with median slopes less than 2%. Due to the low topographic relief, conservation practices which have been implemented in the watershed implemented reduce sediment and pollutant transport are limited to grassed waterways and reduced tillage (conservation, strip-till and no-till).

2.3.2 Hydrology and water quality monitoring

As part of Hickory Grove Lake Water Quality Improvement Project, the watershed was monitored for discharge and nutrients (nitrate, ammonium, dissolved reactive phosphorus, total nitrogen and total phosphorus) from April 2010 to November 2012 by the Water Quality Research Laboratory, Iowa State University (Andrews et al., 2012). Therefore, this study relied on both previous monitoring data and the data collected during rainfall-runoff events in spring 2013 to evaluate the P transport pathways in this watershed at sites TO and SO (Figure 2.1). The drainage district located in the watershed is dominated by poorly drained Canisteo-Coland-Harps-Webster soils (43.5% of the drainage district area) and very poorly drained Okoboji soils (1.3% of the drainage district area) where ponding

occurs frequently. About 15 ha of land upstream of the TO was enrolled in USDA's Conservation Reserve Program with perennial grass cover (Figure 2.1).

A drainage district is an underground network that connects field-scale tile lines in the drainage area to the main line which typically outlets to a surface water body (Figure 2.1). Tile lines in the watershed vary between 12.7 cm and 25.4 cm in diameter, and are located approximately 120 cm below the surface (Aaron Andrews, Personal Communication). Individual tile lines in the watershed are connected to the drainage district main line (91 cm in diameter), which outlets at the site TO. A drainage district is located in east section of the watershed and drains about half the watershed (879 ha), and the drainage district outlet serves as the headwater for the stream that flows in to Hickory Grove Lake. Surface runoff from the upland drainage area and tile flow from the TO converge at the subwatershed outlet (SO).

Two portable samplers (ISCO Model 6712, Teledyne ISCO, Lincoln, NE) were deployed at TO and SO. Flow measurements were collected using an area velocity flow module (Model 750, Teledyne ISCO, Lincoln, NE) that measures velocity and depth of the flow. Discharge was calculated by multiplying the cross-sectional area of the channel by the velocity of the stream. Rainfall was monitored at the TO from April 2011 using a tipping bucket rain gauge (Model 674, Teledyne ISCO, Lincoln, NE). The portable sampler was used to collect discrete water samples during rainfall-runoff events at 1 h interval for water quality analysis. The samplers were triggered before the beginning of the runoff event to collect flow measurements and water samples. In addition to event sampling during runoff events in May 2013, weekly grab samples were collected at these two locations during the recreational period in 2010, 2011 and 2012 as part of Hickory Grove Lake Water Quality Improvement

Project (Andrews et al., 2012). These water samples were analyzed for total phosphorus (TP) and dissolved reactive phosphorus (DRP). The DRP concentrations were determined by the automated flow injection ascorbic acid method using an Automated Ion Analyzer system (Lachat Quickchem 8000, Lachat Instruments, Loveland, CO) with a detection limit of 0.001 mg/L, and TP concentrations were analyzed with a spectrophotometer (HACH DR 2800, HACH, Loveland, CO) using an acid persulfate digestion method with a detection limit of 0.1 mg/L (APHA-AWWA-WPCF, 1998). Water samples were collected within 24 h and stored at 4°C prior to nutrient analysis.

2.3.3 *Chemical hydrograph separation*

Retreating glaciers during the Wisconsin glacial period created closed depressions or prairie potholes (with no drainage outlets) across the Des Moines Lobe region. Surface intakes are often installed in these closed depressional areas and roadside ditches to route ponded surface runoff via subsurface drainage system to streams and lakes. Surface intakes are usually 6" perforated pipes protruding above the ground surface which provide a direct conduit to the subsurface tile drainage system. Figure 2.1 shows the locations of surface intakes in the HGLW. Because of the prevalence of surface intakes in the watershed, flow at the TO during rainfall-runoff events includes soil macropore/matrix flow and surface runoff intercepted by surface intakes; whereas flow at the SO includes flow from the TO and surface runoff from the upstream drainage area. The unique setup of the watershed allowed monitoring for tile flow at TO (drainage area 879 ha) and combined surface runoff and tile flow at SO (drainage area 852 ha). During non-rainfall days flow at the SO is almost equal to flow from TO. To determine the contribution of surface intakes in total flow at TO and SO, a

chemical hydrograph separation method was used. According to the conservation of mass, the total discharge and total P load at TO for an event can be expressed as

$$Q_e = Q_t + Q_i \quad (1)$$

$$Q_e C_e = Q_t C_t + Q_i C_i \quad (2)$$

Where Q is total stream discharge, C is phosphorus concentration and subscripts e, t, and i refer to event, tile and surface intake flow and phosphorus concentrations. The surface intake flow (Q_i) can be obtained by substituting Eq. (1) in Eq. (2) and solving for Q_i :

$$Q_i = Q_e \left[\frac{C_e - C_t}{C_i - C_t} \right] \quad (3)$$

Eq. (3) has three unknowns Q_i , C_t , and C_i ; an estimate of C_t can be obtained by analyzing P concentrations during low flow conditions at the TO when the tile flow is 100% of the flow at the TO. A range of values for P concentrations in surface runoff will be assigned for C_i and Eq. (3) will be solved for Q_i . A common trend observed in surface runoff hydrographs is that P concentrations increase during the rising limb of the hydrograph and decrease during the falling limb of the hydrograph. The hydrograph separation output is dependent on the estimated values of C_i and therefore the sensitivity of the output to the C_i will be assessed.

2.4 Results

2.4.1 Spring precipitation and phosphorus monitoring in the study area

10-yr Stage IV NEXRAD precipitation data (2004-2013) was used to inform spring precipitation characteristics in the HGLW (IEM, 2013). From 2004 to 2013, the annual precipitation averaged 830 mm and the spring precipitation averaged 326 mm. Spring precipitation was calculated as the total precipitation between March 21st and June 20th in a

given year. Precipitation in 2011 and 2012 was below the 10-yr average annual precipitation and the year 2012 (annual precipitation of 532 mm) was a drought year in Iowa. The spring precipitation in 2013 was about 487 mm, which was 49% higher than the 10-yr average spring precipitation. The wet spring in 2013 (about 55% of annual total precipitation) was preceded by two-year drought weather.

Figure 2.2 shows the DRP and TP concentrations at TO and SO during the 3 yr (2010 – 2012) study period. The 3 yr P monitoring data included weekly grab sampling and event sampling by the portable samplers during the recreation season. The TP and DRP were detected in all the grab samples collected from TO during the 3 yr monitoring period. During runoff events in the 3 yr monitoring period,

the TP concentrations at the SO exceeded 1 mg/L, whereas the TP

concentrations at TO reached 0.91 mg/L. The median TP concentrations at TO and SO were 0.07 mg/L and 0.09 mg/L, respectively, whereas the median DRP concentrations at TO and SO were 0.02 mg/L and 0.04 mg/L, respectively. The EPA water quality criterion for

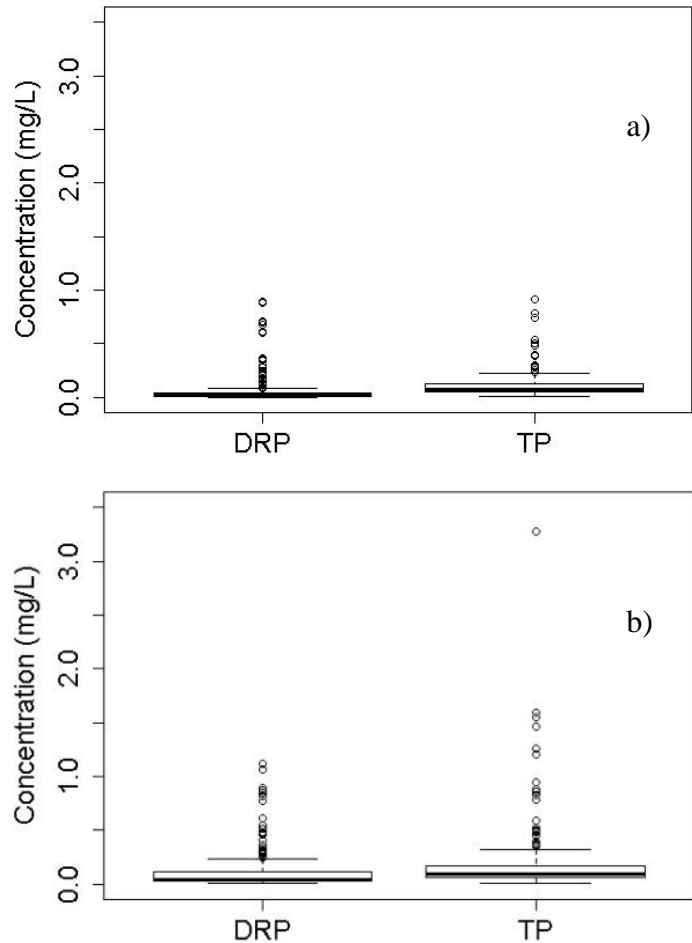


Figure 2.2. Three-year record of TP and DRP concentrations at TO (a) and SO (b), Hickory Grove Lake watershed, Iowa.

phosphates in streams discharging to lakes and reservoirs is 0.05 mg/L to control algae blooms (US Environmental Protection Agency, 1986), which was exceeded by the median TP concentrations during the 3 yr monitoring period at the TO and SO. The peak TP concentration (3.28 mg/L) and peak DRP concentration (1.11 mg/L) was observed at SO in a water sample collected during a runoff event in summer 2011. The 75th quartile of DRP concentrations at the TO and SO are 0.04 and 0.11 mg/L, respectively. The 75th quartile of TP concentrations at the TO and SO are 0.12 and 0.17 mg/L, respectively.

2.4.2 Spring rainfall event characteristics in 2013

A total of four rainfall events in spring 2013 were monitored for flow and P concentrations at TO and SO between May 1 and May 31, 2013. The 4 rainfall events studied ranged from 30.0 mm to 51.5 mm of precipitation. In this study, the analysis of surface runoff and tile flow response was focused on event 3 as this was the only event with a complete dataset (flow and nutrient data. Event 2 resulted in relatively low tile flow in response to the total precipitation input of 21.6 mm, whereas event 4 produced high surface runoff in response to the total precipitation of 51.5mm (2.03 in). Flow was not monitored at SO for event 4, due to extreme flow conditions at SO that dislodged the area-velocity module and suction line of the portable sampler. Table 2.1 shows the event characteristics and hydrological responses to monitored rainfall events.

Table 2.1 Hydrological response to precipitation characteristics at TO and SO in the HGLW.

Event flow	Date	Total rainfall (mm)	1-day antecedent rainfall (mm)	Mean flow (mm/h)		Maximum flow (mm/h)		Tile ratio*
				TO	SO	TO	SO	
1	9-May	39.1	0.0	0.172	0.231	0.328	0.489	0.76
2	19-May	21.6	0.0	0.046	0.089	0.102	0.199	0.53
3	20-May	30.0	21.6	0.184	0.290	0.699	1.774	0.65
4	25-May	51.5	0.76	0.269	-	0.529	-	-

* Tile flow ratio is calculated as the total flow at TO divided by the total flow at SO for an event

Mean and maximum flows at TO and SO were highest for events 3 and 4 and lowest for events 2 and 1 (Table 2.1). The mean and maximum flows were higher at SO compared to TO for monitored events (events 1, 2, and 3). The mean flows varied from 0.089 – 0.290 mm/h and maximum flows varied from 0.489 – 1.774 mm/h at SO for monitored events (events 1, 2, and 3). The mean flow at SO was twice as high as mean flow at TO for event 2. For event 3, the maximum flow at TO and SO were twice as high in event 1. Maximum 1 h rainfall intensity was observed for event 3 (29.9 mm/h) which had a 1-day antecedent rainfall of 21.6 mm (0.8 in). Even though events 1 and 4 had higher rainfall amounts than event 3, maximum flow (peak runoff rate) was observed for event 3 due to the 1-day antecedent rainfall and peak rainfall intensity. Regardless of the mean and maximum flows, the highest tile flow ratio was observed for event 1, while the lowest tile ratio was observed for event 2. The tile flow ratios for the monitored events varied from 0.53 - 0.76. The high tile flow ratio indicates that most of the flow at SO during an event was from TO. For example, the tile flow ratio for event 3 was 0.65, which indicates that 65% of the total flow at SO was from TO and the remaining 35% was surface runoff from the upland drainage area.

2.4.3 Hydrograph for monitored events

Tile flow and surface runoff responses to the 4 monitored rainfall events at TO and SO are shown in figure 2.3. The TO acts as the headwater for SO, therefore during low flow conditions, flow at SO was mostly flow from TO. The minute differences in flow between TO and SO during low flow conditions can be attributed to sensitivity of the area-velocity sensor of ISCO. Flow at TO in the hours at the beginning of each event ranged from 0.01 - 0.05 mm/h and flow at SO ranged from 0.11 - 0.38 mm/h. The total precipitation for event 1

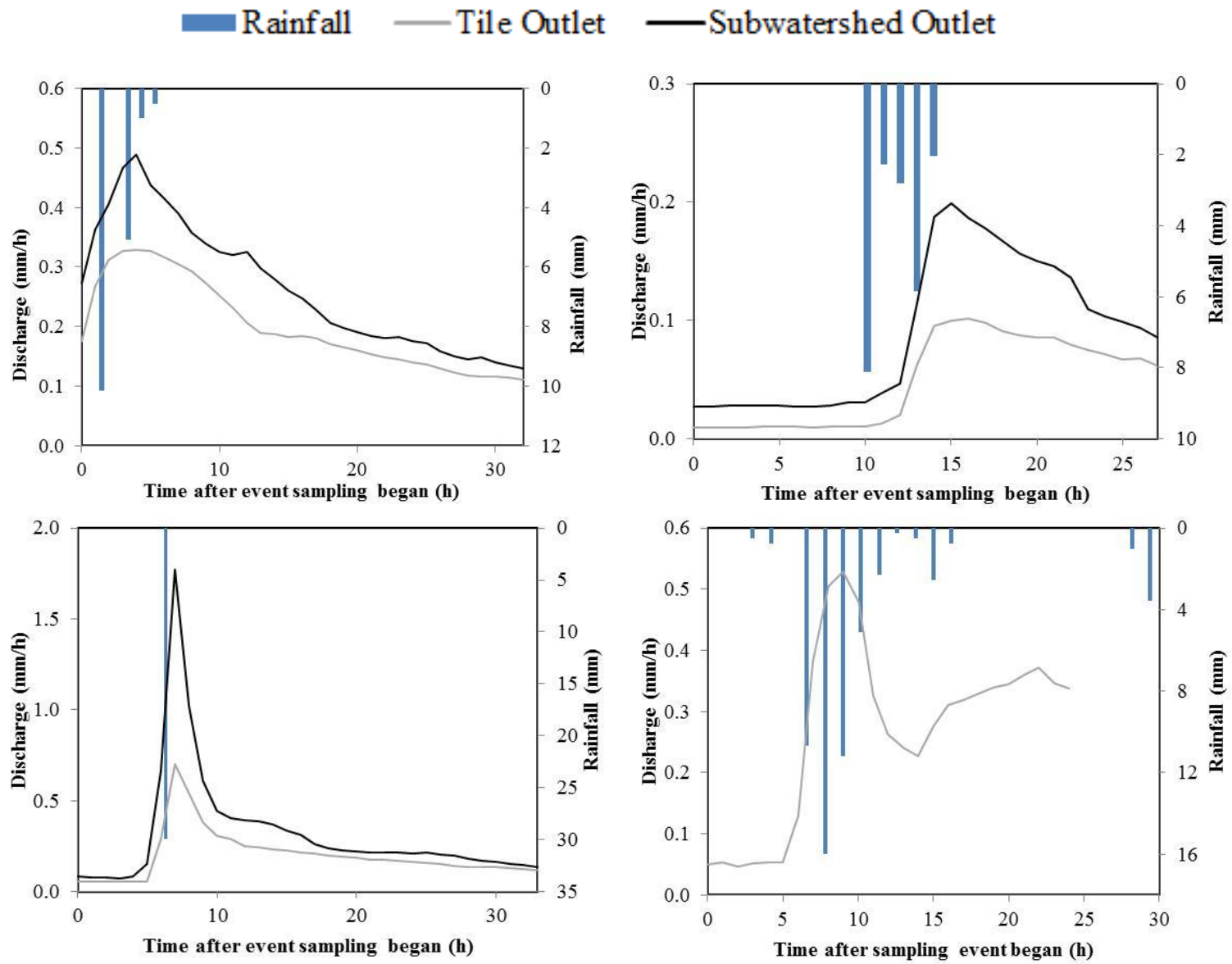


Figure 2.3 Hourly discharge measurements at TO and SO for monitored rainfall events

was 39.1 mm (1.54 in) which began at 22:50 h on 8 May 2013 (Figure 2.3). A steady increase in flow at TO and SO was observed during event 1 and the runoff at SO peaked 10 h after the rainfall began. For event 3, rainfall began at 19:30 h on 20 May 2013 or 5 h after the ISCOs were started to collect water samples, an immediate increase in flow at TO and SO was evident. Surface runoff contribution began right away (saturation excess overland flow) as there was a 1-day antecedent rainfall of 21.6 mm (0.8 in). Flow at SO and TO peaked at 7 h after the sample collection started and decreased until about 30 h. The time of concentration at SO for event 3 was less than 2 h. The peak flow rate for event 3 was 1.774 mm/h, the highest in all four monitored events. Based on 3 yr monitored data (2010 - 2012), the peak flow rate for event 3 was the second highest peak flow observed at SO. The highest peak flow rate at SO (2.11 mm/h) was observed during a flooding event on 11 August 2010. The peak flows at TO generally occurred with or immediately before the peak flow at SO for all the events monitored.

2.4.4 Phosphorus dynamics during runoff events

Distributions of DPR and TP concentrations are shown in Figure 2.4 while the (median, 25th and 75th quartile concentrations) are listed in Table 2.2.

Table 2.2. DRP and TP median, 25th, and 75th quartile concentrations measured at sites TO and SO for runoff events during 2010 and 2012.

Indicator	TO		SO	
	DRP	TP	DRP	TP
Median	0.18	0.89	0.20	1.13
25 th quartile	0.14	0.77	0.16	0.91
75 th quartile	0.66	2.64	0.67	2.70

The median, 25th quartile, and 75th quartile DRP and TP concentrations during the events at TO and SO were similar. The peak TP and DRP concentrations (4.9 mg/L and 0.91

mg/L) were observed at SO during event 3. The TP and DRP concentrations were more variable at SO in event 3 than in events 1, 2, and 4.

The sediment bound phosphorus (SBP) concentration was estimated as the difference between TP and DRP (APHA-AWWA-WPCF, 1998). The median SBP concentration for the monitored events at TO and SO were 0.76 mg/L and 0.90 mg/L. The SBP concentrations at SO for the monitored events varied between

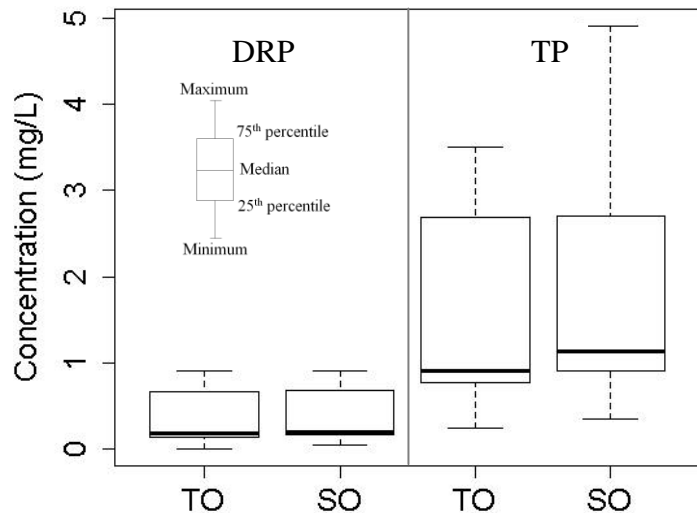


Figure 2.4 Box plots showing DRP and TP concentrations for the monitored events. Box plots show median (line in the middle of box), 25th quartile (lower box limit) and 75th quartile (upper box limit).

0.33 mg/L and 4.29 mg/L. In general, the majority (> 80%) of the TP concentration observed in water samples collected during monitored runoff events was SBP. Despite the USDA CRP site in the surface runoff pathway to SO, water samples collected at SO during the event showed suspended sediments (Figure 2.1).

2.4.5 Phosphorus fluxes during event 3

The TP concentrations at SO during event 3 varied between 0.28 mg/L and 4.90 mg/L and the DRP concentrations at SO during event 3 varied between 0.04 mg/L and 0.68 mg/L (Figure 2.5). The TP concentrations at TO varied between 0.24 mg/L and 3.12 mg/L. Similar TP levels (0.2 mg/L to 3.74 mg/L) were observed during a runoff event in September 2006 in tile discharge from an Iowa watershed located in the Des Moines Lobe (Tomer et al., 2010). The TP and DRP concentrations increased with flow at TO and SO and the TP and

DRP concentrations generally returned to pre-event concentrations as the flow decreased. The highest TP and DRP concentrations at TO and SO were observed at the beginning/during the rising limb of the hydrograph and the TP and DRP concentration varied as a function of flow. The DRP concentration patterns

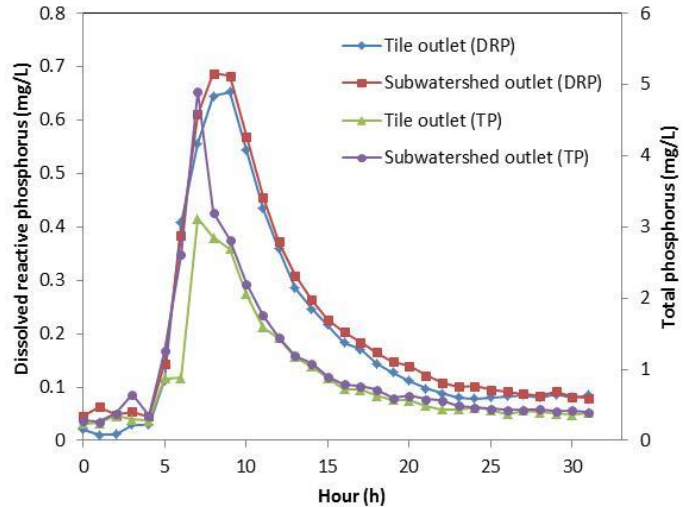


Figure 2.5 TP and DRP concentrations at TO and SO during event 3. Note that the scale and axes are different for DRP and TP.

were very similar to TP concentration patterns at TO and SO for all the events monitored.

Pre-event TP concentrations at TO and SO were 0.24 mg/L and 0.28 mg/L, respectively, but the TP concentrations during the recession period of the hydrograph at TO and SO were 0.39 mg/L and 0.40 mg/L, respectively. The increase in TP concentrations during the recession can be attributed to higher concentrations of suspended sediment. The DRP concentrations at TO were very similar to DRP concentrations at SO for the entire event. Similarly, the TP concentrations at TO were very similar to TP concentrations at SO except during the peak TP. The similar TP and DRP concentrations at TO and SO during the runoff event may be surprising, but the flow at TO included flow from surface intakes and subsurface-tile flow. The flow at SO include total flow from TO and surface runoff from the upland drainage area.

2.4.6 Chemical hydrograph separation

In this study, chemical hydrograph separation (CHS) was used on event 3 to estimate the contribution of surface intakes to the total tile discharge at TO and to the total discharge originating from the upland drainage area at SO. The TP levels at TO and SO were used in

the CHS method. The TP in surface runoff was assumed to vary between 4.0 mg/L and 4.9 mg/L, based on the TP monitoring at SO during event 3. It was also assumed that TP in subsurface-tile flow was 0.24 mg/L. The subsurface-tile flow in TO was calculated as the difference between total discharge at TO and estimated discharge from surface intakes and surface runoff at SO was estimated as the difference between total discharge at SO and total discharge at TO.

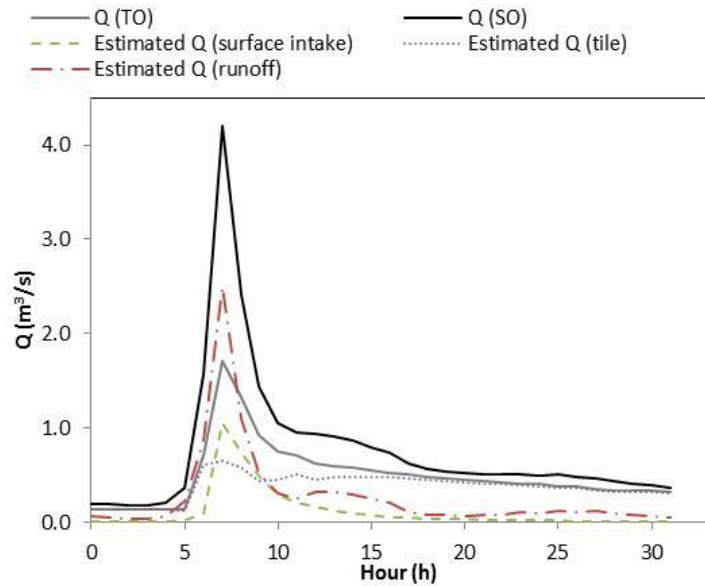


Figure 2.6 Chemical hydrograph separation of total discharge (Q) into surface intakes, subsurface-tile flow components at TO and surface runoff component at SO.

The results indicated that surface intakes accounted for 23.6% of the total discharge at TO and 15.2% of the total discharge at SO (Figure 2.6). About 50% of the total discharge at SO may have originated from subsurface-tile flow. Varying the TP level in subsurface-tile flow between 0.24 mg/L and 0.39 mg/L indicated that between 13.6% and 15.2% of the total discharge at SO may have originated from surface intakes. The CHS methods on an Iowa watershed located in Des Moines Lobe yielded estimates that conduit flow contributed 16.5% total discharge at watershed outlet (Schilling and Helmers, 2008). The flow monitoring at SO and TO during event 3 indicated that 64.5% of the total discharge at SO originated from TO and 35.5% of the total discharge at SO originated from surface runoff. The tile drainage contribution at the SO (64.5% -monitored) was in close agreement with a CHS method

estimate of 68.6% reported by Tomer et al. (2010) for total event discharge at a watershed outlet originating from tiles.

The CHS method estimates that the surface intake contribution at TO didn't start until 6 h and continued till 17 h, whereas the surface runoff at SO started at 5 h and ceased at 17 h (Figure 2.6). The hydrograph separations were relatively insensitive to TP concentrations in tile drains compared to TP concentrations in surface runoff. Varying the TP concentration in TO from 0.24 mg/L to 0.39 mg/L only varied the surface intake contribution at TO from 23.6% to 21.2%. Decreasing the TP concentration in surface runoff to less than 4.0 mg/L led to unrealistic subsurface-tile flows based on graphical hydrograph interpretation (the estimated subsurface-tile flow decreased tremendously to less than pre-event flow rates during the peak flow conditions at TO).

2.4.7 TP, DRP, and SBP loads and sources for event 3

TP, DRP and SBP loads for event 3 at TO and SO, and multiple sources involved in P export are presented in Table 2.3. A sharp increase in P loads at the beginning of the event followed by a steady load for the remainder of the event was observed (Figure 2.7). A similar pattern for P loads was observed during events in a tile drained watershed in

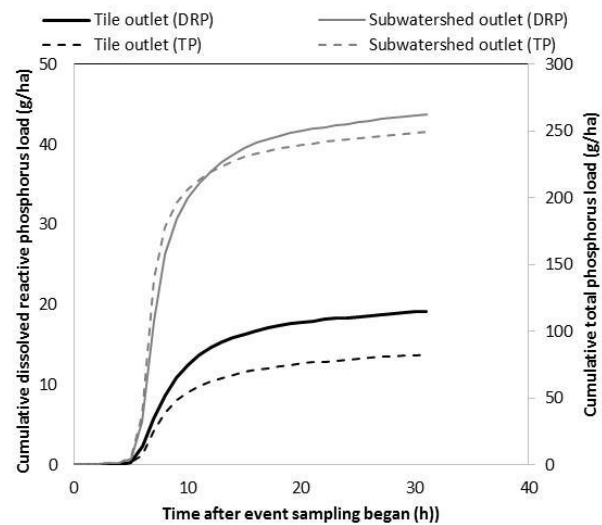


Figure 2.7 Cumulative loads of TP and DRP at TO and SO during the event 3.

Illinois (Vidon et al., 2012). Cumulative load plots (Figure 2.7) show that the majority (>60%) of the TP load and DRP load at TO occurred within 5 h after the surface runoff

began. About 83% of the TP load at SO occurred within 5 h after the surface runoff began while the surface runoff for event 3 lasted 12 h. The P loads stabilized after 15 h. This event transported very high amounts of TP down to the Hickory Grove Lake. The majority of the P (> 75%) transported at TO and SO and even in other sources was through SBP.

Surface intakes generated about 85% of the TP load and 94% of the DRP load observed at TO. Tomer et al. (2010) observed that surface intakes contributed to at least 75% of the TP load from the tile outlet monitored in their study. Even in surface intakes, the majority of P transported was in the SBP form, which indicates suspended sediment transport through surface intakes as evident in water samples from TO. The water samples collected from the TO during the event had poor clarity. A field survey after the event revealed sediment deposition around surface intakes. Overall, the surface intakes contributed 28.2% of the TP at SO. The TO and surface runoff from the upland drainage area accounted for 33.3% and 66.7%, respectively of the TP load at SO.

Table 2.3 Cumulative TP, DRP, and SBP loads (g/ha/event) during event 3 at TO and SO, Hickory Grove Lake, Iowa.

Location	Source	Load			%SBP in TP
		TP	DRP	SBP	
TO	Total	83.2	19.3	63.9	76.8
	Surface Intake	70.6	18.1	52.5	74.4
	Subsurface-tile	12.6	1.2	11.4	90.5
SO	Total	250.2	43.8	196.4	78.5
	TO	83.2	19.3	63.9	76.8
	Surface Runoff	167.0	24.5	132.5	79.3

2.5 Discussion

This study is the first of its kind to present P fluxes during spring rainfall events in a watershed dominated by subsurface-tile drainage management with surface intakes. The unique setup of the HGLW allowed monitoring for P fluxes in tile flow and surface runoff

from similar drainage areas. This offers an opportunity to address key issues about the P transport pathways in tile drained watersheds during early spring, when majority of the annual P transport occurs (Royer et al., 2006). In terms of precipitation, May 2013 (254.5 mm) was the wettest month of the year and twice the 10-yr monthly average (114 mm). Rainfall events in May 2013 (flashy and steady events) are representative of the spring storms in Iowa. Event 3 stands aside from other 3 monitored events as an extremely flashy event. The antecedent moisture conditions (for event 3) were associated with peak flow rates and high P concentrations at TO and SO. This particular event was a significant storm that was mainly influenced by antecedent moisture conditions, however it is possible tillage and late summer crop planting operations might have also impacted the P loads.

The TP concentrations reported in this study during specific rainfall events are consistent with those observed previously at tile outlets in other Iowa watersheds. Tomer et al. (2010) reported TP concentrations ranging from 0 mg/L to 3.8 mg/L in two tile outlets in Tipton Creek, Iowa. Schilling and Helmers (2008) reported that the TP concentrations in a stream during baseflow period was 0.09 mg/L, and 1.3 mg/L during an event in tile drained watershed located in Central Iowa. A state-wide monitoring study conducted by IDNR to develop a nutrient budget reported that the average TP concentrations in Iowa streams vary from 0.14 mg/L to 2.22 mg/L (IDNR, 2004). Stream TP concentrations are influenced by overland flow and streambank erosion and therefore are not directly comparable to tile drain TP concentrations (Royer et al., 2006). The average TP concentration at TO during monitored events in 2013 was 1.57 mg/L and 0.11 mg/L during the 3 yr monitoring study, which are consistent with P concentrations observed in other tile drain monitoring in Iowa.

Most of the existing P monitoring studies focus on either tile drainage outlets or streams, but rarely both. The monitoring and CHS results of this study aid in understanding the P fluxes, patterns and transport pathways to address mitigating strategies. The majority of the P runoff occurred at the beginning of the surface runoff, and surface intakes played a significant role next to surface runoff in P transport through tile drain during event 3. The water samples collected from TO during each event had suspended solids and poor clarity indicating sediment movement through surface intakes into tile drains. A survey conducted in the watershed after the event showed sediment deposition around surface intakes.

The nutrient criteria developed for Class A Iowa lakes indicate that mean TP concentrations in lake during summer must be less than or equal to 0.035 mg/L (IDNR, 2008). The 10-yr average TP concentrations in the Hickory Grove Lake are 0.06 mg/L values, which was twice the recommended nutrient criteria (ILIS, 2013). A Water Quality Improvement Plan developed to address eutrophic conditions in the lake suggested annual P load reductions (22%) from watershed to the lake (Unpublished report). The watershed water quality implementation plan must also include strategies to reduce P transport via surface intakes.

2.6 Conclusions

Reducing P export from the agricultural watersheds is a key step toward protecting water quality and aquatic life in streams and lakes. The monitoring results of this study suggest that P concentrations in tile drains occurred at levels that could contribute to eutrophication in Hickory Grove Lake. Combined with previous monitoring work in the watershed, P levels were detected in samples collected from tile drainage water (TO) year round. Water quality monitoring indicated that the majority of the P transported in tile drains

and in surface runoff was in the sediment bound form. Although surface runoff remains the dominant transport pathway of P, surface intakes provide an alternate route for P movement in a watershed dominated by subsurface-tile drainage management. Short-term, intense precipitation events have the potential to transport significant amounts of P to streams. Intensive P monitoring and hydrograph separations indicated that P transport in tile drains is primarily regulated by surface intakes during runoff events. This stresses the need to better understand the impact of surface intakes on P transport via tile drains. In-field erosion control practices combined with buffered surface intakes could reduce P movement in to tile-drains. Aside from implications for selection of best management practices, this study also provides critical insight to P dynamics and transport pathways during runoff events in tile drained watersheds.

2.7 Acknowledgements

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CHAPTER 3. NEAR SHORE BEACH VOLUME MODELING
APPROACH FOR SETTING BEACH BACTERIA TMDLs: A CASE
STUDY, HICKORY GROVE LAKE, IOWA

This paper has been submitted to Applied Engineering in Agriculture

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3.1 Abstract

A novel approach to set bacteria Total Maximum Daily Load (TMDL) using a Near-Shore Beach Volume (NSBV) model is described along with recommendations for design of a monitoring network to support this method. Sources of fecal bacteria in the Hickory Grove Lake watershed include unpermitted septic systems, manure applications in the watershed, livestock access to streams, and wildlife. The Lake Inlet, Lake Outlet and Lake Beach were monitored for *E. coli* concentrations from 2010 – 2012, this monitoring data was used to assess relationships between watershed bacteria loads to the beach impairments. Fecal bacteria from waterfowl are identified as the major source to the Lake Beach causing the water quality impairment. The bacteria TMDL for the Hickory Grove Lake Beach was set at 1.87×10^{11} cfu/day for the single sample maximum target and 1.01×10^{11} cfu/day for the geometric mean target, which correlates to the presence of fewer than 5 resident geese. Monitoring recommendations to support this approach include weekly beach water quality monitoring and post-event sampling; periodic spatial sampling of

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the lake; weekly and post-event grab sampling of the water quality at the lake inlet mixing zones; and weekly and post-event grab sampling of the water quality at the lake outlet.

Keywords: Fecal bacteria, *E. coli*, TMDL, beach impairment.

3.2 Introduction

Inland lake beaches are a popular source of recreation throughout the United States and when these beaches are impaired due to poor water quality, major economic losses can occur in local communities. Pathogens are identified as the primary cause of surface water impairments in the United States; an estimated 157,151 miles of rivers and streams and 272,391 acres of lakes, reservoirs and ponds are classified as impaired by the U.S. Environmental Protection Agency (USEPA) due to elevated pathogen levels (US Environmental Protection Agency, 2012a). A waterbody is classified as impaired if it does not meet its designated uses and one of the primary designated uses of the inland lake beaches is contact recreation (swimming, bathing, water skiing, and water play by children). The USEPA has developed stringent water quality standards to protect humans from exposure to pathogen contaminated waters. Fecal indicator bacteria (FIB) such as the fecal coliforms, Enterococci, and *Escherichia coli* are primarily used to detect fecal contamination as they are the preferred indicator of pathogens in waters (US Environmental Protection Agency, 1986). Excessive quantities of FIB in surface waters are associated with increased risk of bacteria-induced illnesses in humans (Frenzel and Couvillion, 2002).

The goal of the Clean Water Act (CWA), passed by Congress in 1972, is to “restore and maintain the chemical, physical, and biological integrity of the Nation’s waters”. Initially, the CWA focused on point sources of pollution, but after discovering the magnitude of nonpoint source pollution, the CWA was amended in 1987 to better encompass reduction of nonpoint source pollution. The CWA requires states to identify impaired water bodies and

develop pollutant-specific Total Maximum Daily Loads (TMDL). A TMDL is the sum of the pollutant loads from point sources, nonpoint sources and a margin of safety (US Environmental Protection Agency, 2012b). The TMDL development often involves identifying all point and nonpoint sources of the pollutant causing the water quality impairment, quantifying the pollutant contribution from each source, and determining the pollutant reductions necessary from each source to achieve water quality standards. Point sources of FIB can include discharges from wastewater treatment plants, storm sewers, or confined feeding operations; and failing septic systems (Teague et al., 2009; Paul et al., 2006). Nonpoint sources of FIB can include runoff from agricultural croplands amended with manures, livestock pastures, urban landscapes; and fecal deposition by livestock, wildlife, and waterfowl (Paul et al., 2006).

Previous studies have demonstrated various methods to set pathogen TMDLs for surface waters including load duration curves (LDC) (KDHE, 2012; US Environmental Protection Agency, 2007a), watershed scale water-quality modeling (Chin et al., 2009; Parajuli et al., 2009), risk based load reduction (Chin, 2009), and combined Bayesian statistics and LDC (Shen and Zhao, 2010). The limitations in each approach and watershed specific conditions such as the sources of pathogens, watershed characteristics and hydrology have resulted in many different approaches for pathogen TMDL development. Watershed scale water quality models such as the Soil and Water Assessment Tool (SWAT) and Hydrologic Simulation Program-Fortran (HSPF) using either empirical or process-based equations, are often used to quantify and develop bacteria TMDLs (Benham et al., 2005; 2006; Baffaut and Sadeghi, 2010). These complex models require large amounts of watershed-specific input data, including measured water quality data for model calibration

and validation. Benham et al. (2005) applied the HSPF model to develop bacteria TMDL in Linville Creek Watershed, VA and proposed four bacteria source load reduction scenarios. The authors suggested that a detailed bacteria source characterization is necessary to enhance the accuracy and the applicability of a TMDL study (Benham et al., 2005). The complexity of predicting fate and transport of bacteria makes it challenging to apply the watershed-scale models for setting TMDLs. Wu et al. (2006) stated that a high level of uncertainty exists in TMDLs developed using watershed scale water quality models. The LDC approach recommended by the U.S. EPA for TMDL development is a simpler approach and requires flow data, numerical water quality standards and observed water quality data. The load allocations and the load reductions necessary to achieve the water quality goal at different flow regimes can be identified using LDCs. However, this approach is limited in that it does not consider factors such as fate and transport mechanisms, decay rates, and bacteria resuspension into streams from bottom sediments. Water quality models and LDC methods are often used to establish bacteria TMDLs for flowing streams (IDNR, 2010a; IDNR, 2010b), but developing a bacteria TMDL for a standing waterbody such as a lake is a challenging task.

Typically water samples are collected from lake beaches and a lake is classified as impaired if the FIB concentration in the sample exceeds the applicable water quality standard. Developing a TMDL for lake beaches requires clear understanding of bacteria movement in the waterbody (advection and diffusion), decay rates (due to natural decay, predation, damage by ultraviolet (UV) radiation), resuspension rates, lake water mixing coefficients in horizontal and vertical directions, and other unknown factors (Jin et al., 2003; Bowie et al., 1985). Previous studies have emphasized the importance of these processes.

For example, the UV component of solar radiation greatly impacted the *E. coli* concentrations in south Lake Michigan where reductions between 34-99% were observed throughout a day (Whitman et al., 2004). Rehmann and Soupir (2009) found that incorporating interactions between streambed sediment and the water column improved estimates of *E. coli* concentrations in streams. Similarly, Jin et al. (2003) developed a mathematical model to estimate fecal bacteria concentrations at a beach and associated the underestimation of the fecal bacteria concentrations to resuspension and survival of microorganisms in the lake bottom sediments. Foreshore and beach sediments have been identified as important bacteria sources for lake beaches (Alm et al., 2003; Whitman and Nevers, 2003, Yamahara et al., 2007; Zhongfu et al., 2010). Resuspension of bacteria laden beach sands during recreation or due to wave action can act as an additional net source of bacteria to the local beach waters. Therefore, a new approach which incorporates all the factors affecting the bacteria movement, decay, and growth in lake waters is needed to develop reliable bacteria TMDLs for inland lake beaches.

This study presents an alternative approach to establish a bacteria TMDL for lake beaches under the condition where watershed bacteria contributions are limited and there is an alternative source of bacteria at the beach. To justify this new approach we assessed the impacts of watershed fecal bacteria inputs to the lake on the fecal bacteria concentrations at the lake beach; used a near-shore beach volume model to develop a bacteria TMDL and determine load reductions needed to achieve water quality goals; and provided monitoring recommendations to support application of this approach.

3.3 Materials and Methods

3.3.1 Study area

The study area is the 98-acre manmade Hickory Grove Lake (Figure 3.1), which is located in Hickory Grove Park, Story County, Iowa, a popular recreational area that serves more than 70,000 visitors each year. The lake is a component of the Iowa Ambient Watershed Monitoring and Assessment Program, which is administered by the Iowa Department of Natural Resources (IDNR) – Iowa Geological and Water Survey to assess the condition of Iowa's surface and groundwater resources. The lake is designated for primary contact recreation, aquatic life and human health uses, and was listed on the 2008 303(d) Impaired Waters Listing for elevated bacteria levels. The lake drains an area of 1,629 ha which is dominated by cultivated land cropped in corn and soybean rotations (83.3%), followed by urban (6.4%), pasture (3.2%), water (2.6%), forest (2.2%), rangeland (1.7%), and wetland (0.6%) land uses. The outflow of the lake flows into East Indian Creek, which is a tributary of the Skunk River.

A total area of 879 ha within the watershed is managed with subsurface tile drains, which is used to remove excess water from subsurface soils to improve crop productivity. The subsurface tile drainage system and other drainage infrastructure is managed within the context of drainage district (Figure 3.1) per regulations established in Iowa Law (Story County, 2014). The subsurface tile drain network is one of the major flow paths to the lake and is estimated to represent approximately 75% of the inflow to the lake. Therefore, the outlet of the tile drain system here after is referred to as the lake inlet. A large detention basin is located on the east side of the lake which intercepts flow entering the lake from upland areas and allows for settling of sediment as well as decay and settling of bacteria.

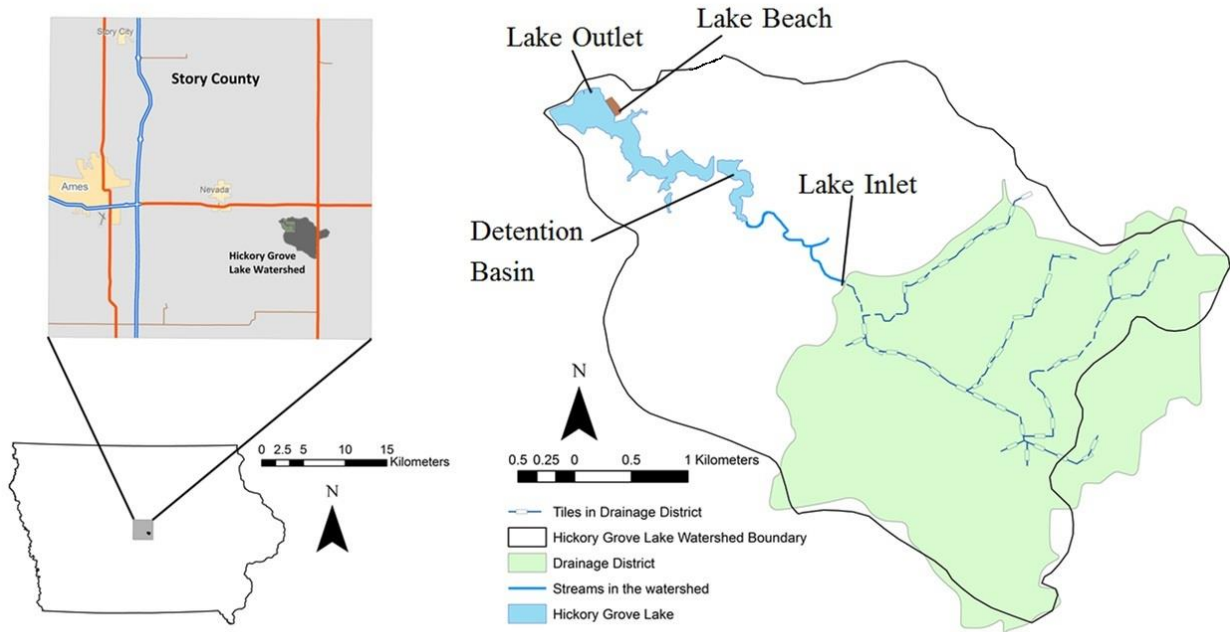


Figure 3.1 Location of the Hickory Grove Lake and the sampling locations in the watershed

3.3.2 Source of fecal bacteria in the watershed

The water quality in Hickory Grove Lake is dominated by non-point source pollution as there are no permitted point sources in the watershed that discharge to the lake. While the lake is listed on the 303d list due to elevated bacteria levels, few areas in the watershed receive manure application. Field surveys conducted by the watershed coordinator confirmed that poultry manure is applied to fields (64.8 ha) to the north of the lake approximately every 2-3 years (Aaron Andrews, Watershed coordinator, personal communication, April 2012). Runoff from these manure amended fields can potentially transfer fecal bacteria to the lake.

Assessment of livestock populations in the watershed identified 10 – 12 cattle upstream of the lake inlet with continuous access to the stream. Studies have shown that livestock exclusion and fencing streams reduces FIB loadings by 46% to 52% (Meals, 2001) and FIB concentrations in waters by 57% to 66% (Line, 2003). Other animal sources include

resident wildlife and migrating birds during spring and fall which may also contribute to increased bacteria levels in the lake. The park ranger estimated that there are approximately 50 resident geese present at the lake during the recreation season (April – September) and that geese numbers range from 1500 to 2000 during migration season (Dustin Eighmy, Hickory Grove Park Ranger, personal communication, March 2011). The bacteria load from geese is approximated to be $4.9E+10$ fecal coliform organisms per goose per day (US Environmental Protection Agency, 2001).

Optical brighteners tests were conducted on samples collected from the lake inlet during low flow conditions to assess the potential bacteria contributions from the rural septic systems. Elevated optical brightener levels were detected during one of the tests (optical brighteners concentration – $19.1 \mu\text{g/L}$ and *E. coli* concentration – 135 MPN/100 ml on 08/04/2011), confirming the presence of a human source of fecal contamination. The Story County Environmental Health Department evaluated the septic systems in the watershed and identified 8 unpermitted septic systems located within the watershed boundary. The fecal bacteria load from the unpermitted septic systems is estimated to be $4.87E+12$ cfu/year, assuming 2 persons/household, a septic failure rate of 50%, and *E. coli* concentration of $6.3E+5$ cfu/100 ml and a discharge rate of 70 gallons/person/day (Horsley and Witten, 1996).

3.3.3 Water quality monitoring

The *E. coli* concentrations at the Hickory Grove Lake beach were monitored weekly during the recreational period by the Watershed Monitoring and Assessment Section of the Iowa DNR from 2004 to 2012. The Water Quality Research Laboratory (WQRL) at Iowa State University monitored *E. coli* concentrations at the lake inlet and the lake outlet every week during the recreational season from 2010 to 2012. In addition to weekly grab sampling,

grab samples were also collected at the lake inlet and the lake outlet during rainfall-runoff events and samples were analyzed for *E. coli* concentrations. The lake water and bottom sediments were also spatially sampled to determine *E. coli* hotspots in the lake. This monitoring design allowed for comparison of *E. coli* concentrations at the lake beach to concentrations at the lake inlet and outlet during storm and base-flow conditions.

3.4 Analysis of Monitoring Data

Linear regressions were used to describe the relationships between the *E. coli* concentrations at the lake beach with *E. coli* concentrations at the Lake Inlet and Outlet, and rainfall amounts. These assessments were performed to evaluate the effects of event surface runoff and tile flow on the *E. coli* concentrations at the lake beach and to identify the significance of watershed bacteria loads on bacteria concentrations at the beach.

3.4.1 Effect of precipitation on E. coli concentrations at the lake beach

The average annual precipitation in the Hickory Grove Lake Watershed is approximately 939 mm. The year 2010 was a wet year with a total annual precipitation about 1082 mm and years 2011 and 2012 were dry years with total annual precipitations 751 mm and 559 mm, respectively. The subsurface flow from the lake inlet flowed year round in 2010, whereas in 2011 and 2012, the flow at the drainage district outlet (upstream from the lake inlet) ceased by early August in 2011 and late July 2012. Relationships between beach bacteria levels and precipitation amounts were examined from 2004 to 2012 to help determine if the upland watershed areas are contributing to beach bacteria levels. Figure 3.2 shows relationships between rainfall amounts and *E. coli* concentrations at the Lake Beach; for correlation purposes bacteria levels were only used if precipitation occurred the day prior to or the day of sample collection. A conservative estimate of the time of concentration for

the Hickory Grove Lake Watershed was 3.5 h, therefore a 24-h antecedent rainfall amount would be optimal to estimate the effects of watershed contributions on beach bacteria concentrations. An R^2 value of 0.0053 was observed between beach bacteria concentrations and rainfall amounts indicating no obvious trends. High *E. coli* concentrations at the beach were observed during periods of little or no rainfall and vice versa.

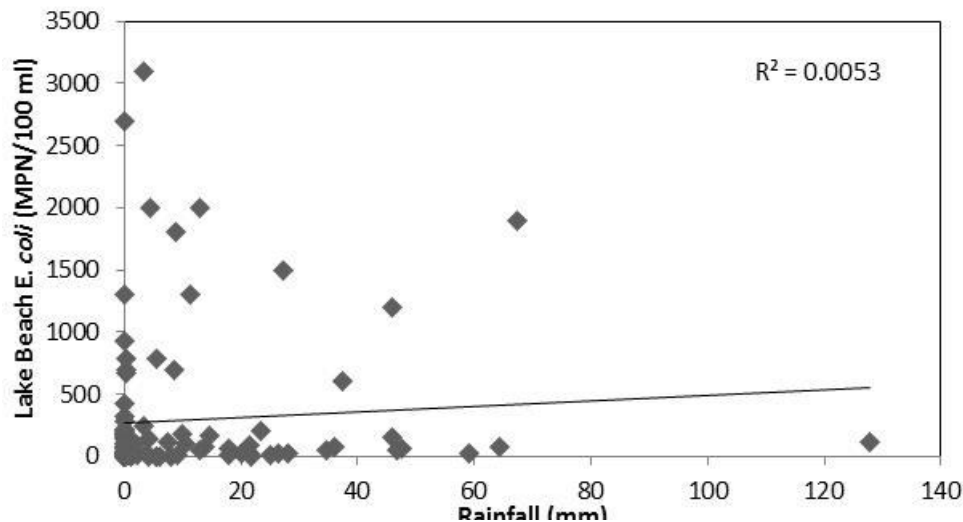


Figure 3.2 Relationships between rainfall and *E. coli* concentrations at the Hickory Grove Lake beach from 2004 through 2012.

3.4.2 Comparison of *E. coli* concentrations at the lake inlet to the lake beach

Figure 3.3 shows the *E. coli* concentrations measured at the lake inlet and at the lake beach from 2010 to 2012. The IDNR single sample mean (SSM) and geometric mean (GM) standards for recreational use lakes are 235 cfu/100 ml and 126 cfu/100 ml, respectively. The Hickory Grove Lake is considered impaired if the *E. coli* concentrations at the lake beach exceed the IDNR GM standard represented by the solid line in figure 3.3. The *E. coli* concentrations at the Lake Beach exceeded the IDNR GM standards several times during the recreational season between 2010 and 2012 which led to beach closures. This analysis again demonstrates the lack of agreement between the *E. coli* concentrations entering the lake and

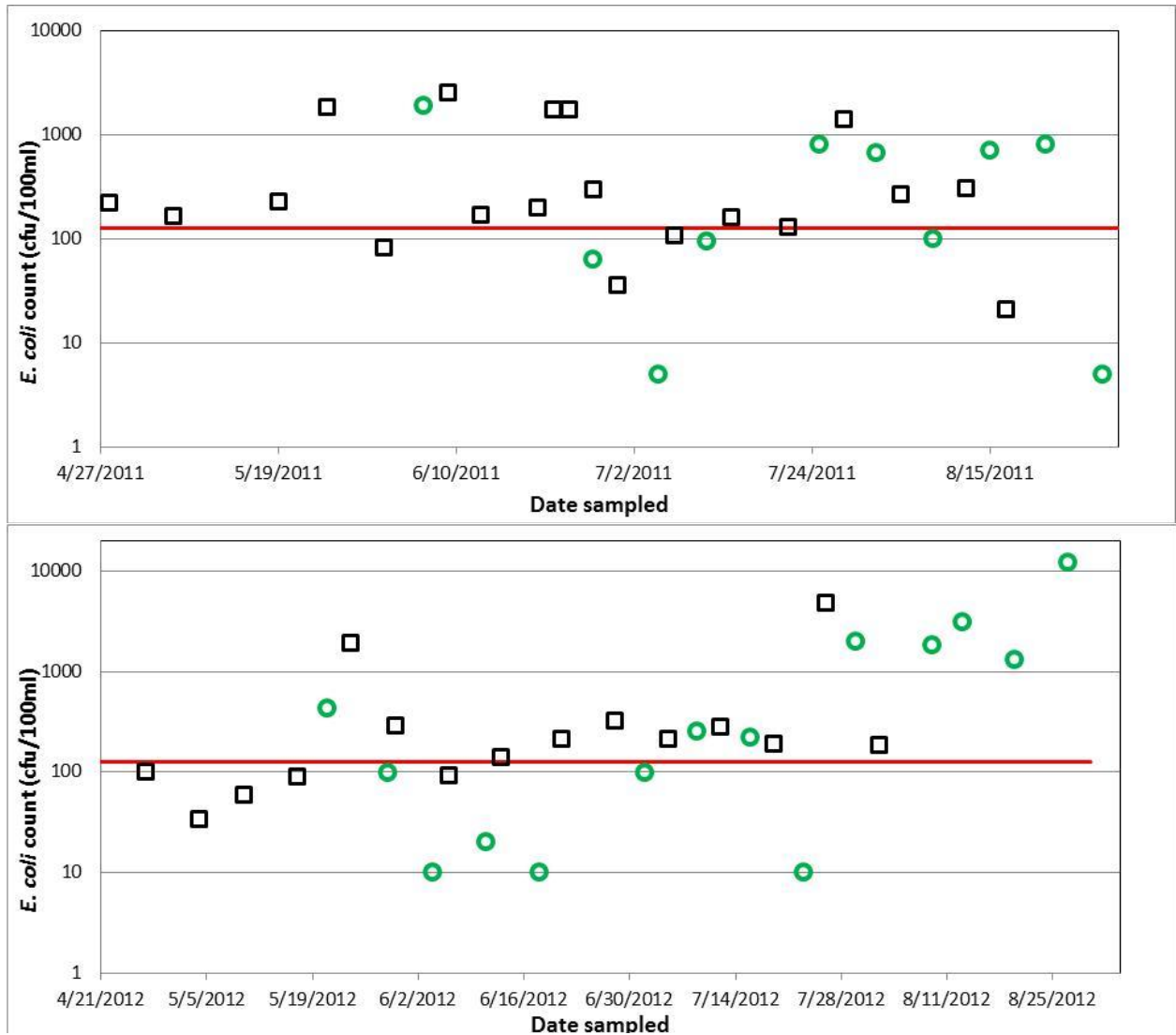


Figure 3.3 *E. coli* concentrations at the Hickory Grove Lake beach and at the lake inlet during the years 2010, 2011, and 2012.

To confirm the observed lack of relationships, the dataset was examined for linear correlations between the *E. coli* concentrations at the lake beach and at the lake inlet over three years, from 2010 to 2012. As shown in figure 3.4 there was no clear and consistent relationship between *E. coli* concentrations at the two locations. This figure excludes data points when flow was not present at the lake inlet.

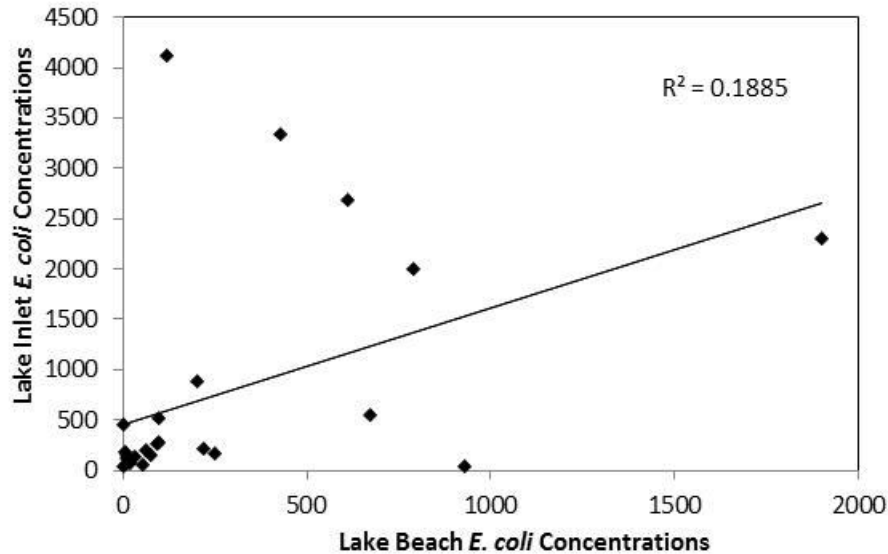


Figure 3.4 Relationships between *E. coli* concentrations at the Hickory Grove lake beach and at the lake inlet.

During the relatively dry years of 2011 and 2012 an interesting trend was noted. The drainage district tile line discharging to the lake ceased in early August and late July of 2011 and 2012, respectively. The samples collected from the Lake Beach exceeded the water quality standards during the dry periods. For example, the *E. coli* concentration at the Lake Beach was 12,000 cfu/100 ml in third week of August 2012, but all contributing flow from the watershed had ceased by last week of July, 2012 (Figure 3.3).

3.4.3 Comparison of *E. coli* concentrations at the lake outlet to the lake beach

The *E. coli* concentrations at the lake outlet and the lake beach were also analyzed to examine any relationships between the two sampling locations. The lake outlet is an uncontrolled spillway where the discharge from the lake is controlled by the depth of water in the lake. The lake outlet sampling location was approximately 400 m northwest of the lake beach. Figure 3.5 shows the *E. coli* concentrations at the lake beach and at the lake outlet from 2010 through 2012. An R-squared value of 0.0033 indicates poor correlation between

location had averaged 16 cfu/100 ml. The sediment samples collected from the detention basin did not have detectable levels of *E. coli*.

The regression relationships between the lake beach *E. coli* concentration and the rainfall amounts, the lake inlet and lake outlet *E. coli* concentrations; and the lake outlet and the lake beach *E. coli* concentrations were used to support the hypothesis that migratory birds are the primary source of bacteria leading to beach water quality impairments. In all cases, poor relationships were observed indicating watershed bacteria inputs do not significantly affect the bacteria concentrations at the Hickory Grove Lake beach. The spatial sampling of the lake also indicated that bacteria were concentrated at the beach and other locations were resident geese frequently populate. Direct inputs from resident geese at the beach were identified as the major source contributing to beach bacteria impairments rather than watershed bacteria loads to the lake; therefore, a beach bacteria TMDL was developed using a near-shore beach volume model.

3.5 Near-shore Beach Volume Model

Chapra (1997) developed a process-based equation to estimate bacteria concentrations in a waterbody during steady-state conditions. The bacteria concentrations in a waterbody were determined based on decay rates, bacteria loading rates, and diffusion rates of bacteria. This model assumes that the diffusion of organisms is equal in all directions.

$$C = \frac{w}{\pi H E} K_0 \sqrt{\frac{k r^2}{E}} \quad (1)$$

where C = concentration of FIB (mass/volume), W = rate of FIB loading (mass/time), r = radius/distance from the beach (length), H = depth of the lake corresponding with distance from the beach (length), E = diffusion of microorganisms in a waterbody

(length²/time), k = decay rate of microorganisms (1/time), and K_0 = first-order modified Bessel function of the second kind.

3.6 Beach Bacteria TMDL

The Near-Shore Beach Volume (NSBV) model was used to estimate the maximum allowable bacteria load to the Hickory Grove Lake Beach. This method was previously applied at the George Wyth Lake, Iowa, a bacteria impaired beach, and the TMDL was approved by the U.S. EPA in December of 2008. The *E. coli* load to the lake beach was estimated by taking into account the number of geese at the beach, the time spent by geese at the beach, and the defecation by geese while on the waterbody. It was estimated that there are approximately 50 resident geese at the park during the recreational season (Dustin Eighmy, Hickory Grove Park Ranger, Personal Communication, March 2011). The daily bacteria load is approximately $4.9E+10$ fecal coliform organisms per goose per day (US Environmental Protection Agency, 2001). The IDNR estimates that the ratio of *E. coli* to fecal coliform is 0.92:1, based on the concentrations observed in waterbodies in Iowa (IDNR, 2008). Therefore, the bacteria load generated by the resident geese was estimated to be $4.51E+10$ *E. coli* organisms per day per goose. Geese spend the majority of their time in or near the lake and defecate while on water, therefore it was assumed that at least 50% of the bacteria load generated by the geese is received by the lake (IDNR, 2008). Another assumption made in this study is that the geese spend equal time at four locations in the lake and therefore only one-quarter of the estimated *E. coli* load is received by the beach waters (Dustin Eighmy, Hickory Grove Park Ranger, Personal Communication, March 2011). The total bacteria load received by the beach waters was calculated as $2.82E+11$ *E. coli* organisms per 50 geese per day.

Table 3.1 NBSV parameter values used for setting the TMDL

Parameter	Value/range	Units
<i>H</i>	10	ft
<i>E</i>	930 to 9.3E+8	ft ² /day
<i>k</i>	1.6	1/day
<i>r</i>	100	ft
<i>C</i>	2.9E+9	cfu/ac-ft

Table 3.1 shows the parameters used in setting the bacteria TMDL. Bowie et al. (1985) summarized decay rates for fecal coliform and *E. coli* at various temperatures for different waterbodies in the U.S. The decay rates varied from 0 to 2 per day for various streams, estuaries, and lakes. An average decay rate of 1.6 per day was used in this study (IDNR, 2008). The diffusion rate of microorganisms (*E*) varies widely with type of waterbody, temperature of waterbody, and source of microorganisms. Therefore, in order to account for high uncertainty in *E*, randomly selected values within the specified range were used to characterize the allowable bacteria loads at the lake beach. Monte Carlo simulations were performed on the NBSV model (1000 simulations) by varying the diffusion rate of *E. coli*, the diffusion rate was varied until the simulated beach *E. coli* concentrations were within the observed lake beach *E. coli* concentrations. The beach volume was estimated as 4,243 m³ (3.44 ac-ft) using 91.4 m (300 ft) as the beach length, 30.5 m (100 ft) as the beach width (floating buoys were located at 100 ft into the lake from the beach), and the depth of the lake at the buoys was assumed to be 3.05 m (10 ft) with the lake beach at a 45 degree slope. The daily allowable maximum bacteria load from geese was estimated as 1.87×10^{11} cfu/day, and the geometric mean bacteria load was estimated as 1.01×10^{11} cfu/day. The allowable bacteria loads represent the median loads from the 1000 simulations performed.

In this study, an implicit Margin of Safety approach was used in the TMDL development. This approach includes using conservative estimates of model parameters, bacteria loads received by the lake, and Monte Carlo simulations to reduce the uncertainties in the model output. The LA from the NSBV model for the SSM target is 1.87×10^{11} cfu/day and for the GM the target is 1.01×10^{11} cfu/day. The SSM and GM target loads were approximately equal to the daily loads generated by four and two resident geese, respectively. As few as 5 resident geese can elevate the *E. coli* concentrations at the beach above the water quality standard.

3.7 Limitations of the Approach

The method proposed here is recommended as an alternative for setting beach bacteria TMDLs in situations when the bacteria levels at an impaired beach do not appear to be related to watershed bacteria loads. Watershed activities may still have some impact on lake water quality, but in this case study, the overwhelming load to the beach from resident geese created a local hot spot with elevated *E. coli* concentrations which were not observed at the watershed outlet or other locations within the waterbody. A secondary compliance location could be set at the lake inlet and a TMDL could be developed for this location to address watershed bacteria inputs to the lake. A use attainability analysis would be required to identify the appropriate use of the inlet waters and identify the relevant water quality standards.

The near shore beach model requires several assumptions to define parameters which are not well known for Iowa lakes. Particularly the diffusion rate (E) is poorly defined in both the reference which introduces the model (Chapra, 1997) and other model applications in Iowa. Here we limited the range of acceptable values to those resulting in bacteria

concentrations observed at the Hickory Grove Lake beach within the past three years based on IDNR weekly sampling. If more intensive spatial sampling at the beach area were to be adopted, a weighted concentration could be calculated and similarly used to set the range of acceptable diffusion rates. Another parameter which introduces high variability into the recommended load reductions is the definition of the near shore beach volume. A smaller beach volume might be more representative of the recreational waters entered by children but a larger volume (for example the distance from the shore to the buoys) encompasses the entire swimming zone. A smaller beach volume is the most conservative estimate for protection of public health. Monitoring of beach sediments indicates elevated *E. coli* concentrations occur in the beach sand (Hartz et al., 2007) and resuspension of these sediment-attached bacteria is completely neglected in this model. However, limited information on resuspension of beach sediments into swimming areas is currently available and defining the parameters needed to predict this process is difficult and could introduce additional uncertainty into the model.

3.8 Monitoring Recommendations and Analysis

Prior to applying the NSBV model approach for establishing a beach bacteria TMDL, the monitoring system should be first designed to clarify if there is a relationship between lake inlet bacteria concentrations (representing the watershed contributions) and the beach bacteria levels. This can be accomplished by collecting water samples at key locations and following storm events. Watershed time of concentration and the lake retention time should be considered when identifying when post-event sampling occurs. Monitoring recommendations are as follows:

- Weekly beach water quality monitoring and post-event sampling;

- Periodic spatial sampling of the lake;
- Weekly and post-event grab sampling of the water quality at the lake inlet mixing zones;
and
- Weekly and post-event grab sampling of the water quality at the lake outlet.

3.9 Conclusions/Lessons Learned

Identifying the sources of contamination and the degree of contaminant loads received by the waterbody are the first steps in the TMDL development process. The near shore beach volume model approach proposed here is a viable alternative for setting load reductions at bacteria impaired beaches, where the predominant source of *E. coli* is waterfowl. Efforts to deter resident geese from lake beaches include controlled goose hunt programs, geese relocation programs, mylar tape around the beach to deter geese, harrowing the beach to expose the existing bacteria to UV radiation, a PTO driven grooming machine to remove goose droppings, sonic deterrents, and green lasers.

This approach would allow Clean Water Act Section 319 implementation funds to target the localized contributions from geese and increase the likelihood of achieving water quality improvements. The recommended modifications to current monitoring approaches could expedite the TMDL development process, but design of the monitoring system would be specific to the waterbody in which it is being applied. The NSBV model may not apply to larger or more complex watersheds, where bacteria concentrations at the lake beach are affected by the watershed bacteria loads.

Research indicates that bacteria adsorb to particles (Hartz et al., 2007), and their fate and transport from upland areas could be associated with sediment. Therefore, to obtain more accurate bacteria TMDLs future research is recommended to improve understanding of the

relationships between sediment and bacteria transport in streams, and modify the existing NSBV equation to include the resuspension of bacteria attached to beach sediments into beach waters. Additional research is also needed to better define the diffusion parameter. If this approach were to be widely adopted by states, regional guidelines for model parameter selection based on lake properties would be useful.

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CHAPTER 4. IDENTIFYING POTENTIAL LOCATIONS FOR GRASSED WATERWAYS USING TERRAIN ATTRIBUTES AND PRECISION CONSERVATION TECHNOLOGIES

This paper to be submitted to Journal of Soil and Water Conservation
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4.1 Abstract

Grassed waterways (GWWs) are an effective conservation practice to prevent ephemeral gully erosion resulting from channelized surface runoff in agricultural fields. Field reconnaissance to identify areas of channelized erosion within a watershed can be time consuming and labor intensive. Recent advancements in precision conservation and light detection and ranging (LiDAR) technologies can provide valuable information on environmental sensitive areas causing soil degradation. The objective of this study was to demonstrate that a compound topographic index (CTI) model supplemented with LiDAR data can be used to identify potential GWW locations and design recommendations. The LiDAR digital elevation model with a spatial resolution of 3 m was used to derive slope, drainage area and plan curvature terrain attributes. The GWW identification and design process was automated in ArcGIS Python environment. The terrain attribute plan curvature

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identified erosion channels, but discontinuity in the model output was observed. The CTI model was calibrated to a field with GWWs installed under the USDA Natural Resources Conservation Service guidelines which yielded a threshold of 30. The model (CTI = 30) was able to identify all the existing GWWs (14) in the watershed. Field surveys were conducted in the watershed, and areas exhibiting evidence of channelized erosion were identified by the model for GWW placement. Furthermore, the CTI model overestimated (PBIAS = -23.34%) the lengths of predicted GWWs suggesting a need to further extend the existing GWWs. The total surface area of the predicted GWWs was 9.8 ha in the study watershed with depth of GWWs reaching 1 m. The design process provides an estimate of land to be set aside for conservation practices. The terrain analysis was effective in targeting conservation practice placement and improves the accuracy of field assessments.

Keywords Grassed waterways, compound topographic index model, LiDAR.

4.2 Introduction

Soil degradation continues to be a major challenge in achieving food security and sustainability in the 21st century (Delgado et al., 2011a). Soil erosion from agricultural lands by water occurs primarily by three processes: sheet erosion, rill erosion and gully erosion. Foster (1986) introduced a term called ‘ephemeral gullies’ to describe erosion resulting from concentrated overland flow in agricultural fields. Ephemeral gullies are temporary erosion features that are larger than rills but can be easily obscured by tillage (Foster, 1986). Ephemeral gully erosion can contribute significantly to the total soil losses in agricultural watersheds and leads to soil degradation. In the U.S., ephemeral gullies typically account for 30% of total soil loss (Bennett et al., 2000). Studies have identified ephemeral gully erosion

as a significant, if not the dominant, source of sediment in agricultural watersheds (Poesen et al., 2003; Gordon et al., 2008).

Grassed waterways (GWWs) are effective conservation practices used to prevent ephemeral gully formation along natural drainage ways (Atkins and Coyle, 1977). GWWs reduce ephemeral gully erosion and agrochemical export to surface waters (Briggs et al., 1999; Chow et al., 1999). Chow et al. (1999), for example, found that GWWs combined with terraces reduced runoff by an average of 86% and soil erosion by 95%. In a laboratory experiment, GWWs were found to reduce herbicide loss in runoff by an average of 56% (Briggs et al., 1999). Moreover, there has also been extensive hydrologic modeling efforts to evaluate the effectiveness of GWWs on runoff and its constituents (Fiener and Auerswald, 2006; Dermisis et al., 2010). Dermisis et al. (2010) investigated the effects of GWWs length on runoff and erosion reductions in an agricultural watershed in Iowa using the Water Erosion Prediction Project (WEPP) Model. The sediment yield reductions were found to be directly related to the GWW length and the peak runoff rate was identified as the dominant factor influencing the effectiveness of GWW on sediment yield reduction (Dermisis et al., 2010). These modeling studies have determined that GWWs characteristics including length, bottom cross-section and the roughness of vegetation govern runoff and sediment reduction efficiency. Moreover, the placement of the GWW is crucial for effective soil and water conservation; however, current methods to identify the eroded channels resulting from concentrated water flow are limited.

Recent advancements in the resolution of digital elevation data can be used to more accurately identify environmentally sensitive areas for precision conservation. Delgado et al. (2011b) conducted a comprehensive review of works demonstrating the use of precision

conservation technologies in soil and water conservation. Light Detection and Ranging (LiDAR) data provides very precise terrain attributes that can be used to identify critical source areas for targeted conservation practices (Galzki et al., 2011). Mueller et al. (2005) used terrain attributes, remote sensing and soil electrical conductivity data to develop erosion probability maps. Critical source areas can be described as the portions of the landscape that have a 'disproportionate and significant effect on water quality or soil degradation'. Researchers have used precision conservation techniques with high resolution elevation data to identify critical source areas for best management practice (BMP) placement (Tomer et al., 2003; Luck et al., 2010). More recently, Tomer et al. (2013) demonstrated that terrain attributes obtained from LiDAR (1m) data can be used to identify critical areas for wetland placement in a HUC-12 watershed.

The primary function of effective GWWs are to prevent ephemeral gully erosion; therefore, locating the areas prone to erosion is necessary to identify potential locations for GWWs (Fiener and Auerswald, 2003). Various studies have developed methods to detect ephemeral gully locations using terrain attributes and topographic threshold values (Moore et al., 1988; Thorne et al., 1986; Parker et al., 2007; Daggupati et al., 2013). Pike et al. (2009) developed erosion probability maps with 4 m elevation data using logistic regression and neural networks for GWWs placement in Kentucky fields. For example, Thorne et al. (1986) developed the Compound Topographic Index (CTI) model and detected ephemeral gully locations in Mississippi fields using soil-specific threshold values. The CTI is a product of slope, upstream drainage area, and plan curvature, and ephemeral gully locations were detected when the CTI exceeded the topographic threshold. Daggupati et al. (2013) investigated four topographic index models to predict ephemeral gully locations in Kansas

and concluded that Slope Area and CTI models predicted the occurrence of ephemeral gullies better than other models. While terrain attributes have been used to locate potential ephemeral gullies, topographic models have not yet been evaluated as tools for predicting placement of GWWs for implementation.

Identifying critical source areas for BMP implementation is crucial for developing an effective watershed management plan. To achieve overall watershed water quality goals, conservation practices must be located where they are most effective. The goal of this study was to evaluate the CTI model in identifying potential locations for GWW placement. The specific objectives were to (1) demonstrate the CTI model in identifying potential locations for GWWs at watershed-scale, and 2) design the GWWs at field-scale according to the United States Department of Agriculture-Natural Resources Conservation Service (USDA-NRCS) guidelines. The results are expected to be useful to conservationists and farmers for prioritizing BMP sites to meet watershed scale water quality goals, receive incentive payments through USDA Conservation Reserve Program (CRP) and determining the area lost to practice implementation.

4.3 Methods

4.3.1 Study area

This study was conducted in the Hickory Grove Lake Watershed (HGLW) located in the Des Moines Lobe region of Iowa (Figure 4.1). The watershed has an area of 1629 ha and drains into an approximately 40 ha lake, which results in a high watershed to lake area ratio of 40:1 and excess nutrient and sediment delivery to the lake. Landuse in the watershed is dominated by agriculture, with 84.7% of the watershed cropped with rotations of corn (*Zea mays* L.) and soybean (*Glycine max* [L.] Merr.). The watershed is low-relief with median

slopes less than 2% except near stream corridors. The watershed is composed primarily of poorly drained soils formed in clayey lacustrine sediments deposited from the Des Moines glacial lobe. The agricultural fields in the watershed are supported by artificial subsurface drainage system where surface intakes are common in fields and in roadside ditches. Subsurface tile drainage was installed in most of the cropland in the watershed to make agricultural production feasible. The watershed covers 34 agricultural fields with areas ranging from 9 ha to 116 ha.

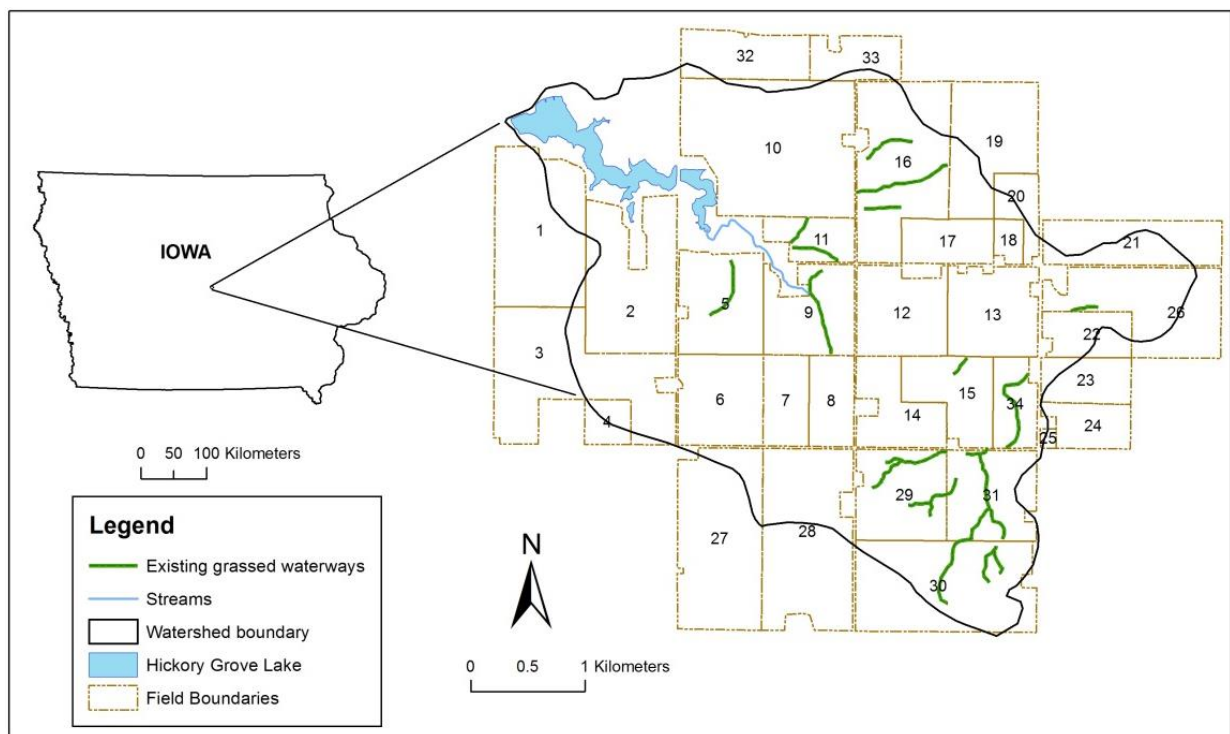


Figure 4.1 Location of Hickory Grove Lake Watershed in Iowa showing grassed waterways and field boundaries

Groundtruthing and USDA-National Agricultural Statistics Survey (NASS) cropland data (USDA-NASS, 2009) collected in 2009 were used to identify and digitize existing GWWs in the watershed. The watershed has 14 GWWs with lengths varying from 193 m to 886 m and the widths of the waterways vary from 10 m to 34 m. The total area covered by

the existing GWWs in the watershed is 16.3 ha. The GWWs in Field 30 (Figure 4.1) were designed and installed according to the USDA Natural Resources Conservation Service (NRCS) specifications. Ground truthing and farmer surveys conducted in the watershed indicated that the GWWs in Fields 5, 16, and 31 are undersized and require frequent regrading (every couple of years) due to erosion in each respective GWW.

4.3.2 Elevation data

A Digital Elevation Model (DEM) in raster format containing elevation data was used to provide terrain attributes for the topographic models. The LiDAR data for the study area was collected by the Iowa Department of Natural Resources in 2008 under the State of Iowa's Light Detection and Ranging program. A DEM with a horizontal resolution of 3 m was generated by aggregating the 1 m LiDAR data to smooth out sharp edges in the data. The primary terrain attributes of slope, upstream drainage area, and plan curvature were derived from the 3 m DEM using Arc Geographic Information System (ArcGIS) software (ESRI, 2012). These terrain attributes have previously been used to study topographic features in various landscapes (Galzki, et al., 2011). The 'pit-filling' operation was conducted on the 3 m DEM before calculating terrain attributes to remove small depressions that would cause water impoundments. This was also necessary to enforce flow pathway conveyance to downstream waters due to the impoundments caused by road structures.

The slope was calculated using the finite difference slope estimation method (Gallant and Wilson, 1996). The hydrologic flow model commonly referred to as the D8 method was used to calculate flow direction and flow accumulation in ArcGIS (Jenson and Domingue, 1988). The upstream drainage area or flow accumulation refers to the total upland area that drains into any single cell and this function can be used to determine the drainage area

boundary. The plan curvature is a measure of flow convergence or divergence across the surface and also determines the local flow geometry and the degree of concentration of the runoff (Zeverbergen and Thorne, 1987). These three terrain attributes were combined into a mathematical formula to characterize the spatial variability of the stream network occurring on the landscapes.

4.3.3 *Compound topographic index model*

The Compound Topographic Index (CTI) model has been previously used to predict the location of ephemeral gullies (Daggupati et al., 2013; Parker et al., 2007; Thorne et al., 1986). The CTI was calculated based on the formula

$$CTI = A \cdot S \cdot PLANC$$

Where S is the slope of the surface (m/m), A is the upstream drainage area (m²) and PLANC is the plan curvature (m/100). The upstream drainage area along with the slope of the surface indicates the stream power, a proxy for flow intensity to predict sediment carrying capacity. The plan curvature indicates the concave and convex shape of the pixel across the surface, identifies the rapid change in slope on the surface and is perpendicular the direction of the slope. The major factors controlling concentrated flow in an agricultural field were represented in the CTI model. The output from the CTI model is a DEM with a topographic index value assigned to each pixel and the eroded channels or potential locations for GWWs were identified when the CTI value exceeded the topographic threshold value.

The CTI method requires considerable trial and error approach to determine the most appropriate threshold specific to each study site as the likelihood of an ephemeral gully formation is not dependent on terrain data alone. For example, Daggupati et al. (2013) used CTI = 62 to locate ephemeral gullies in two Kansas fields; whereas Parker et al. (2007)

determined that the critical CTI thresholds varied from 7 to 62 to locate ephemeral gullies for 10 sites in Mississippi. Therefore, a pragmatic approach was adopted to identify critical CTI threshold for the HGLW. First, the CTI threshold was determined for field 30 in the southern part of the watershed which already has GWWs designed and installed according to the USDA-NRCS recommendations. Then, the CTI threshold was used to identify potential locations for implementing GWWs. The CTI model was calibrated to Field 30.

4.3.4 Grassed waterway design

An automated process to design GWWs according to the effective stress methodology (USDA-NRCS, 2007) was built in Interactive DeveLopment Environment (IDLE) for python (v 2.7) interfaced with ArcGIS (ESRI, 2012). This process requires GWW attributes listed in Table 4.1 and drainage area characteristics specific for each GWW. The designed GWWs were tested under long grass and short grass conditions to examine if the runoff velocities during the design storm are within the permissible velocities of vegetation in the waterway. The Scientific Python (SciPy) computing environment in Python was used to determine the

Table 4.1. Grassed waterway attributes utilized in the design process

Indices	GWW attributes	Indices	Methods
Shape	Parabolic	Runoff	SCS curve number (USDA-SCS, 1990)
Vegetation	Kentucky bluegrass	Peak runoff rate	SCS-TR 55 (USDA-SCS, 1986)
Retardance curve index	5.60	GWW design	USDA NRCS effective stress (Temple et al., 1987)
Permissible velocity (m/s)	1.5		
Design storm event	10-y 24-h		

optimum depth and top width for GWWs. The GWWs were designed in a way that they would be able to convey the overland runoff for a 10-y 24-h storm event from the entire upstream drainage area.

4.3.5 Model Evaluation

The error matrix approach was used to summarize the agreement between the model predicted GWW features and ground truth data in identifying GWW features, which was previously used in studies identifying the locations of ephemeral gullies (Daggupati et al., 2013; Gutierrez et al., 2009). The error matrix uses a binary scale (1: GWW present, 0: GWW absent) to estimate the number of correct and incorrect predictions. Table 4.2 shows the matrix used in this study. The entries in the error matrix are defined as follows: ‘a’ indicates the number of instances the model predicted GWWs in an agricultural field when the GWWs are actually present, ‘b’ (false positive or error of commission) indicates the number of instances the model predicted the GWWs in a field when the GWWs are absent, ‘c’ (false negative or error of omission) indicates the number of instances the model did not predict GWWs when the GWWs are present and ‘d’ indicates the number of instances the model predicted and observed GWWs are absent. False negative values indicate poor model performance, whereas false positive values do not necessarily indicate poor model performance as the model may identify new locations for GWWs that are not present in the watershed. The false positive rate was calculated as the number of false positives divided by the total GWW features predicted by the model, whereas the false negative rate was calculated as the number of false negatives divided by total GWWs present in the watershed.

Table 4.2 Error matrix table to assess the predictive performance of CTI model

Model Predicted	Ground Truth		Total features present
	Presence of GWWs	Absence of GWWs	
Presence of GWWs	a	b	(a+b)
Absence of GWWs	c	d	(c+d)

The classification rate was used to determine the accuracy of the model in predicting the existing GWWs, which involved distinguishing the model predicted pixels that are inside and outside a GWW boundary. The classification rates were calculated as the number of pixels inside a GWW divided by the total number of pixels identified as potential erosion regions in a field. If the entire model predicted pixels are inside a GWW feature, it would be considered correctly classified. The percent bias (PBIAS), which indicates overestimation or underestimation bias was used to evaluate the performance of model predictions on GWW lengths (Gupta et al., 1999).

4.4 Results and Discussion

4.4.1 Plan curvature

Plan curvature has been previously described as the most useful terrain attribute in the CTI model to identify the rapid changes in the slope and areas of flow convergence in the landform topography (Parker et al., 2007; Pike et al., 2009). A negative plan curvature indicates the surface is concave in shape, a positive plan curvature indicates the surface is convex in shape, a zero value indicates the surface is flat. Negative plan curvature values were generally associated with boundaries of potentially erodible areas. The concave shape of a pixel in perpendicular to the flow direction allows the flow to converge across the surface. Figure 4.2 shows the plan curvature for two fields 29 and 9 in the study area with existing waterway boundaries. Most (78% and 67%) of the negative plan curvature pixels in

Field 29 and Field 9, respectively, were within the GWWs boundaries and followed the existing GWW shape.

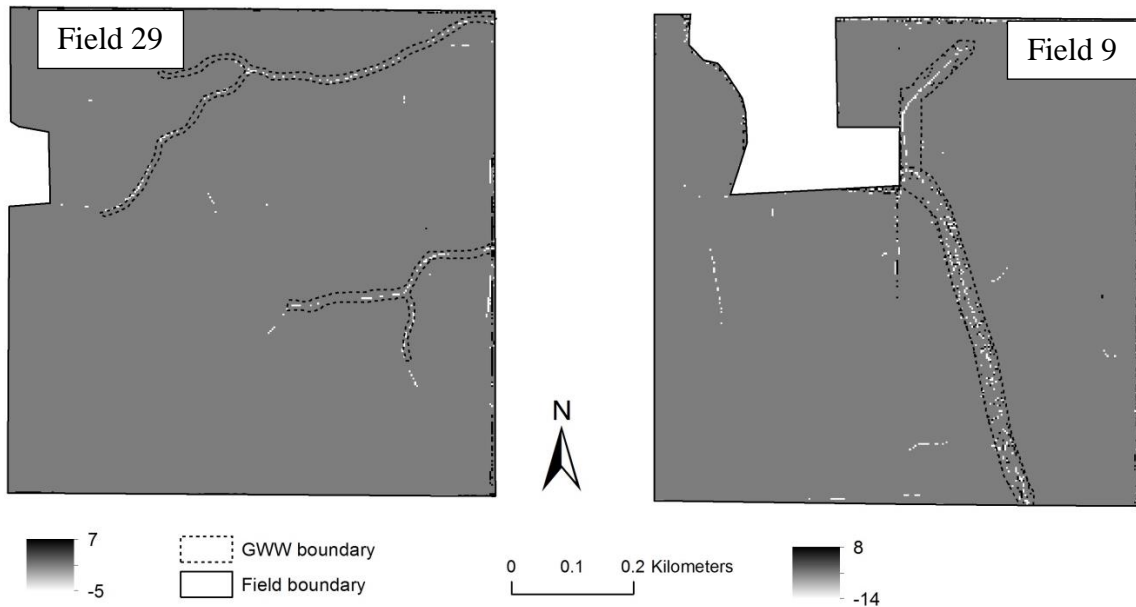


Figure 4.2 Plan curvature ($m/100$) for Fields 9 and 29 overlain by grassed waterway

Clearly, the plan curvature located the erosion channels in the HGLW as confirmed by the visual interpretation of the plan curvature maps with GWW boundaries. Closer observations, however, revealed that the negative plan curvature pixels were not spatially connected, which could be due to areas of reduced flow convergence or flat areas in the GWWs. The plan curvature also identified the concave surface in roadside ditches as indicated by the cluster of white pixels at the North and East boundaries of Field 9. The field boundaries in the watershed were manually digitized using USDA-NASS cropland data layer (horizontal resolution of 30 m) and due to the low resolution of the aerial image, portions of the roadside ditches were included in the field during the digitization process. Therefore, agricultural fields along the roads include concave surface pixels from the roadside ditches. However, plan curvature data alone was not adequate in identifying the erosion channels and

plan curvature must be integrated with slope and upstream drainage area to produce efficient erosion probability maps.

4.4.2 Erosion feature identification

Figure 4.3 shows the predicted locations of GWWs (CTI = 30) compared with existing GWWs in the watershed. Boundaries of the existing GWWs and the model predictions for Field 34 are shown in figure 3 (zoomed-in figure). Closer observation of the model output showed that some areas were discontinuous and were not spatially connected with existing GWWs. This was most likely due to the influence of plan curvature on the model output. The spatially disoriented pixels were connected to obtain a smooth trajectory in the model output using a snapping procedure developed by Daggupati et al. (2013) with a snapping distance of 3 m.

The false positive and false negatives in the error matrix table (Table 1) were used to analyze the performance of the CTI model. Visual interpretation of the CTI model output showed that the model accurately predicted all 14 existing GWWs in the watershed. The error matrix output showed that the model has zero false negatives, indicating excellent model performance at CTI = 30. The CTI model also predicted several GWW locations across the watershed indicating potential for eroding channels, which can be referred to as false positives. The model output suggested fields requiring new GWWs, especially Field 10 which is draining directly into the lake, and that the existing GWWs in the Field 29 should have extended further into the field. The model may have overpredicted the need for GWW in Fields 6 and 7 as there was very little evidence of erosion during field observations in fall 2013. The CTI values for cells in an existing GWW were found to be significantly larger than randomly selected cells outside the waterway.

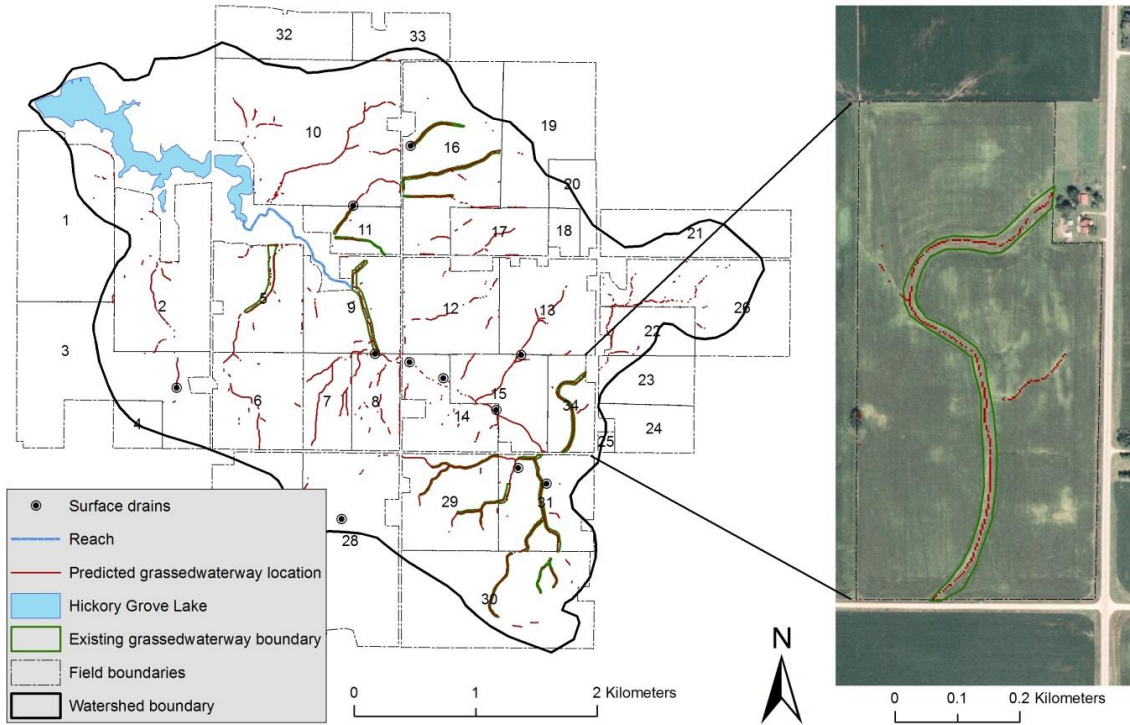


Figure 4.3 CTI model predicted GWWs and their correspondence with existing GWW locations in the HGLW

The CTI model also identified some GWW locations that are not spatially connected which could be due to the influence of plan curvature identifying areas of reduced flow convergence. Parker et al. (2007) evaluated the significance of the inclusion of plan curvature in the CTI model and concluded that despite the discontinuity in the model output, better ephemeral gully predictions were observed with CTI than the Slope-Area approach (product of drainage area and slope).

Field reconnaissance in the watershed identified several surface intakes in fields that drain GWWs (Figure 4.3). Although it is not recommended to install surface drains in GWWs, the grass surrounding the surface drain would reduce suspended particles in the first flush and help reduce runoff volume. Luck et al. (2010) observed that surface drains were

being used to control erosion in a Kentucky field, and therefore the locations of surface drains may further be utilized in prioritizing GWW placement.

4.4.3 Impacts of CTI threshold on grassed waterway identification

The calibrated CTI threshold (CTI = 30) was then used to detect GWW locations in the HGLW. A range of threshold values (CTI = 2.5 to 200) were examined to understand the effects of the calibrated CTI threshold on GWW identification in the HGLW. Figure 4.4 shows the model performance rates at various thresholds in the HGLW. The main goal of this study is to help conservationists locate areas with greater potential for soil erosion and reduce the need for in-field surveys. With this goal in mind, efforts were focused on reducing the false negative rate rather than the false positive rate in model output. An ideal model would have zero false negatives.

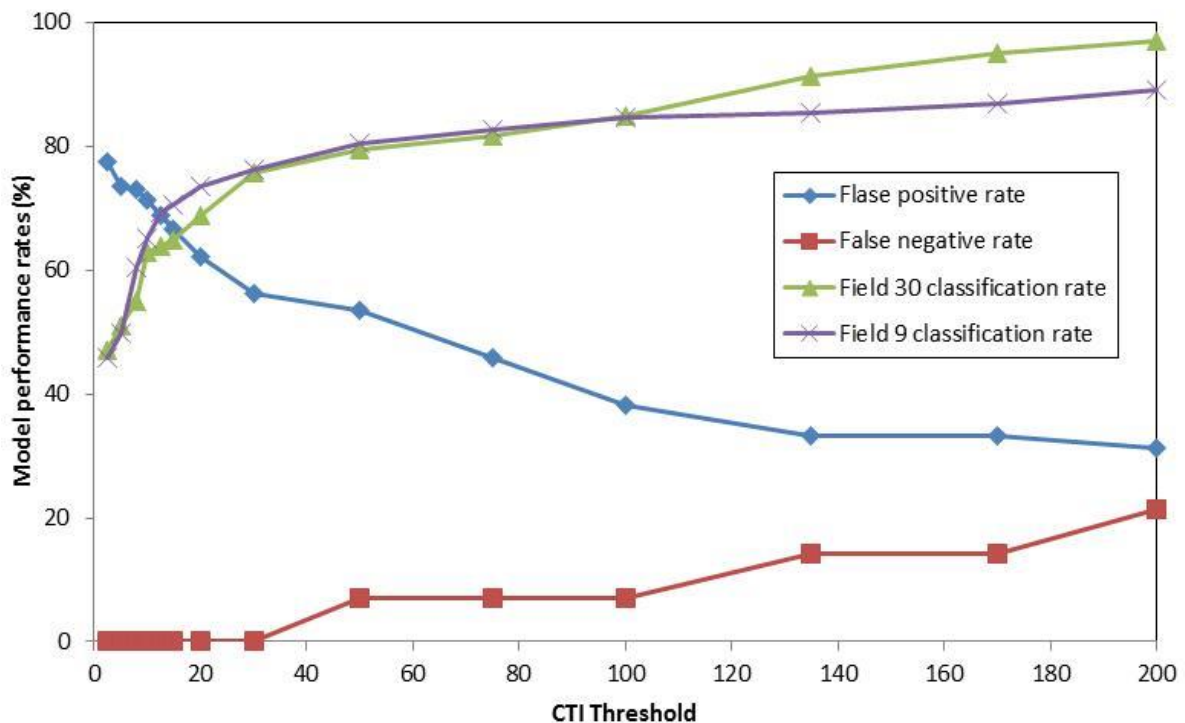


Figure 4.4 CTI model performance statistics at various thresholds

The CTI model has zero false negative rates for thresholds up to 30 and thereafter increased slightly as the threshold increased. At CTI = 50, the false negative rate is 7.14% as the model did not detect one of the GWWs in Field 11. A CTI threshold of 50 did not predict any GWWs in Field 30, but did identify GWW locations downstream of this field. The CTI model was unable to detect GWWs in Fields 16 and 30 with thresholds greater than 130; this may be due to the lower upslope drainage areas in these fields. Therefore, a CTI threshold of 30 appears to be the best fit for the HGLW because of the zero false negative rate and low false positive rate. These thresholds may vary with watershed topography, therefore the model predicted must be validated with groundtruth data. The peak false negative rate of (21%) was observed at the CTI threshold of 200. The false negative rate showed an increasing trend with increasing CTI thresholds whereas false positive rates showed a decreasing trend. Daggupati et al. (2013) observed similar trends in false positive and false negative rates in locating ephemeral gullies in two Kansas fields. The false positive rate for the HGLW at CTI = 30 was 56.25%, which indicates that more than half the model predictions were nonexistent in the watershed. The false positive rates in the HGLW varied between 77% and 31% for thresholds (0 to 200). The high false positive rates do not imply poor CTI model performance, but rather indicate priority areas for additional GWWs to be implemented in the watershed to control soil loss.

Model output revealed that several pixels outside the existing GWW boundaries were identified as locations for potential GWWs, in particular Field 30 where GWWs were designed and installed according to USDA-NRCS specifications. This overestimation of spatially disoriented areas was observed in almost all of the fields (Figure 4.5). To analyze the performance of the CTI model in predicting spatially disoriented areas, classification

rates were calculated for Field 9 and Field 30 at different thresholds (Figure 4.4). Field 9 in particular was chosen for this analysis because of its larger drainage area (722 ha) than any other field in the watershed which could explain the influence of terrain attribute (drainage area) on the model output. The classification rates for Field 9 and Field 30 varied from 46% to 89% and 47% to 97%, respectively, for thresholds

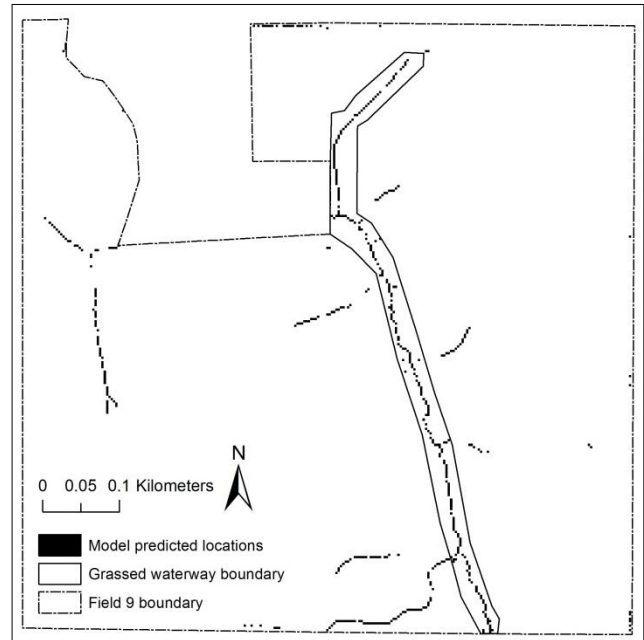


Figure 4.5 Model predicted locations for GWs in Field 9 at CTI threshold of 30

between 2.5 and 200. A classification rate of more than 75% was observed for Fields 9 and 30 at a CTI threshold of 30. Fields 9 and 30 had similar classification rates until the CTI threshold reached 100 and thereafter Field 9 had a slightly lower classification rate than Field 30. This was most likely attributed to the fact that Field 9 had a larger upslope drainage area. The spatially disoriented areas (pixels outside the existing GWs) where the model predicted the need for GWs lowered the classification rate.

4.4.4 Design of grassed waterways

The design specifications were calculated for each identified GWW by the CTI model using the effective stress method (USDA, 2007). The time of concentration for the predicted GWWs in the HGLW ranged from 0.8 h to 3.6 h, with GWWs in Field 9 having the highest time of concentration in the watershed. The lengths of existing GWWs in the watershed ranged from 193 m to 918 m whereas the lengths of predicted GWW ranged from

245 m to 1530 m (Figure 4.6). The depth of GWWs was capped at 0.30 m (1 ft) to allow farm equipment to pass through the waterway for maintenance. The top width of predicted GWWs ranged from 3.2 m to 87.3 m. The total surface area of the

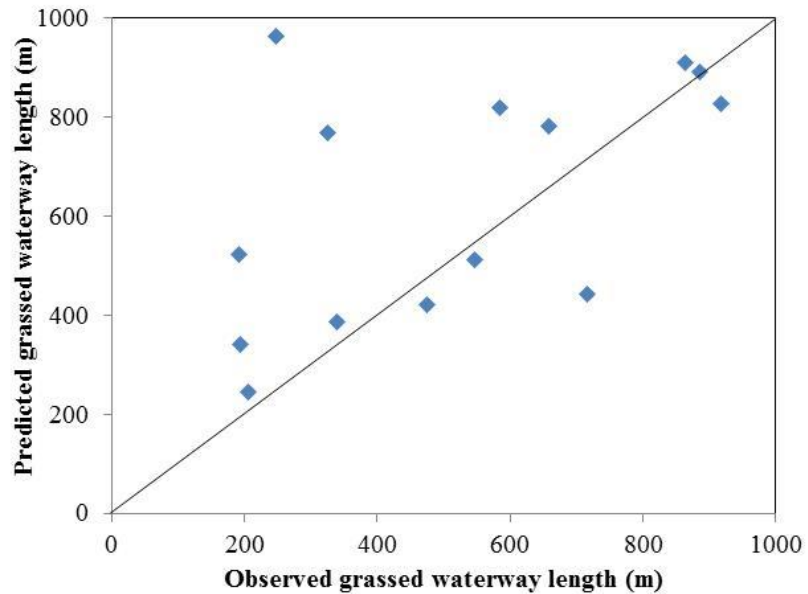


Figure 4.6 Comparison of observed and predicted grassed waterway lengths

predicted GWWs was 29.3 ha, which provides an estimate to the farmer/producer on how much land must be removed from agricultural production to reduce soil degradation. Figure 4.6 compares the lengths of existing GWW with the lengths of predicted GWWs. The CTI model at threshold of 30 resulted in very good agreement with a PBIAS value of -23.34, indicating that the CTI model has overestimated the GWW lengths. The overestimation could be the CTI model suggesting a need for extending the existing waterways.

Field surveys in the watershed indicated that GWWs in Fields 5, 16, and 31 were installed by the producer and need to be regraded every couple of years to control soil erosion. The producers would have been able to better control soil erosion, had there been access to field-specific USDA NRCS GWW design specifications. Producers can also receive CRP payments from the USDA FSA for GWW maintenance and installation in environmentally sensitive areas.

4.4.5 Implications of precision conservation

Results of this study have several implications for conservation planning and site-specific management. GWWs are clearly a key practice in reducing ephemeral gully erosion, and there is a need to develop erosion probability maps and design conservation practices to achieve water quality goals. This study illustrates an approach that can be used to identify potential sites where GWW placement will be most effective at a watershed scale. The results of this study implied that LiDAR derived terrain attributes can be effectively used in locating the majority of ephemeral gullies and can serve as an invaluable guide for conservationists. The LiDAR derived terrain attributes were previously used in locating ephemeral gullies (Daggupati et al., 2013), identifying channel erosion (Pike et al., 2009), mapping sink holes (Weibel, 2007), and identify locations for wetlands placement to reduce watershed nitrate loads (Tomer et al., 2013).

The erosion probability maps obtained from terrain analysis can guide farmers/producers to erosion features that have disproportionate effects on water quality to be targeted with conservation practices. The terrain attribute data provides a wealth of information to guide field surveyors to critical source areas. The LiDAR data would be extremely useful in low relief areas in defining flow paths compared to moderate and high relief areas. A field reconnaissance survey conducted in Seven Mile Creek watershed, MN (approximately 10,000 ha) to identify fine-scale gullies and side inlets costed \$9500; and would cost about \$100,000 to conduct a similar county wide survey (Galzki et al., 2011). Field surveys guided by the terrain analysis could identify the majority of the erosion features in a short period of time and greatly reduce the cost and need for field surveys. Ephemeral gully formation depends on factors such as tillage practices, cover crops, and magnitude of a

rainfall event which were ignored in the CTI model. Further research is needed to develop methods that include terrain and soil attributes to determine erosion prone areas in a field.

4.5 Conclusions

Results of this study suggest that LiDAR derived terrain attributes were able to accurately identify potential locations for GWWs in the HGLW. Both farmers and USDA NRCS conservationists may benefit from the output maps in locating environmentally sensitive areas. The CTI model has excellent predictive capacity in locating GWWs using 3m LiDAR data. Although the model calibration might limit the application of this method to other geographic areas with no field sampled data. Model thresholds played a major role in identifying GWW locations, and model thresholds may vary among physiographic regions. Further research is needed to determine the effects of model thresholds in moderate and very low relief areas. There is also a need to determine whether including the crop management practices, soil properties, or precipitation characteristics in terrain-methodology used in this study may improve the GWW location identification. The design of identified GWWs provided an estimate of the land to be removed from agricultural production if the farmer/producer wanted to enroll in CRP incentives program. Targeting conservation practice placement can mitigate the soil degradation and maximize their benefits on water quality.

4.6 Acknowledgements

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CHAPTER 5. GENERAL CONCLUSIONS

5.1 Review of Introduction

Protection of water resources from further degradation due to sedimentation, excess nutrient transport, and fecal bacteria deposition can be achieved only through a holistic watershed management approach. The TMDL program is being implemented with the goal of restoring surface water quality by addressing point and nonpoint source pollution. However, watershed characteristics, type of contaminant and its properties, timing of the contaminant export, and source of the contaminant pose various difficulties in setting TMDLs. To overcome these difficulties, new approaches are needed to improve the TMDL process and the important components (monitoring, modeling, and implementation). The goal of this study was to improve the TMDL development process in achieving water quality goals and restoring impaired waterbodies.

5.2 Review of Phosphorus Transport Pathways

The first project involved identifying and quantifying the transport pathways of P in an agricultural watershed completely under subsurface tile-drainage. This was accomplished by short-term, intensive monitoring for flow, TP, and DRP during runoff events in spring 2013 and using CHS method at TO and SO. The unique setup of the watershed allowed differentiating the subsurface tile-drain and surface runoff contributions at the watershed scale for the two similar drainage areas. More than 60% of the flow observed at SO during monitored runoff events was from TO. Most of the P transported from TO and SO during runoff events was in the SBP form and surface runoff carried the majority of the P load at the

SO. The upland drainage area at TO is dominated by surface intakes and have played an important role in P transport to subsurface tile-drains during runoff events.

5.3 Review of a Novel Approach to Set Beach Bacteria TMDL

This project involved identifying sources of fecal bacteria causing bacteria impairments at the Hickory Grove Lake Beach and developing a beach bacteria TMDL. Three years of monitoring data from the HGLW and at the Lake Beach was used to identify the major source of fecal bacteria causing bacteria impairments at the lake. It was identified that watershed fecal bacteria loads have minimal effect on Lake Beach impairments and that fecal bacteria loadings from waterfowl residing at the lake were the major source for bacteria impairment. A NBSV model which uses bacteria and Lake Beach properties was used to estimate the bacteria loads at the beach. Source identification used in this study allowed for developing a viable alternative beach bacteria TMDL, instead of developing a conventional watershed bacteria TMDL. Periodic spatial sampling of the lake is also recommended to identify the fecal bacteria hotspots in the lake.

5.4 Review of GWW Identification Using Topographic Attributes

The last component of a TMDL is implementation, which involves recommending conservation practices in achieving the TMDL and overall water quality goals. High resolution elevation data combined with precision conservation technologies were used to identify existing GWWs and recommend new GWW locations in the HGLW. The plan curvature attribute identified all erosion channels as confirmed by the visual interpretation of field survey data. The CTI model predicted that some of the existing GWWs need to be further extended into the fields. One limitation of this method is that thresholds used to identify GWWs may vary with watershed topography and require ground truth data for

model calibration. The use of terrain attributes in identifying fine erosion channels and design specifications of identified GWWs would be extremely useful to farmers/conservationists during field surveys.

5.5 Implications of the Study

The hydrology of the watershed played an important role in contaminant transport and must be understood when developing a TMDL. Subsurface tile-drainage management supplemented with surface intakes is extensively implemented in Des Moines Lobe region of Iowa, and while surface intakes in this study area proved to be an important transport pathway for P. This study can be potentially useful to improve water quality models and identifying conservation practices for P reduction. Many existing water quality models do not include surface intakes as a P routing mechanism, while predicting P concentrations exported to waters. This information is important in improving the understanding of P transport in subsurface drained watersheds, and will be useful in TMDL development.

This study showed that monitoring data can be used to narrow down the source of the impairment while developing a beach bacteria TMDL. Spatial monitoring of a waterbody for fecal bacteria conducted in this study, is very rare, and is needed to validate the waterfowl influence on beach *E. coli* concentrations. Current methodologies for bacteria TMDL development, however, primarily rely on watershed scale bacteria models; little effort is ongoing to improve existing dynamic waterbody models simulating fate of fecal bacteria. However, this study demonstrates an approach for setting a beach bacteria TMDL when the source of the impairment was waterfowl.

Finer resolution topographic data is needed to locate erosion channels at the field-scale. The GIS tool developed here can be potentially useful in locating GWWs in other

watersheds. TMDLs supplemented with conservation practice placement and design specifications are very useful to conservationists during field surveys and developing implementation plan for the TMDL. Overall, the results of this study illustrate that a single, conventional TMDL approach may not be viable for all watersheds and contaminants; therefore a distinct, contaminant and watershed-specific TMDL approaches are needed.

5.5 Future Recommendations

- The study area was completely under subsurface tile-drained supplemented with surface intakes, but the CHS method assumes that P concentrations in subsurface tile-drains remain constant for the entire runoff period. Therefore, it should be validated in a subsurface tile-drain field with similar P application rates which has no surface intake contribution.
- Instead of using hydrograph separation methods to estimate P loads from surface intakes, techniques to monitor flow from surface intakes in to subsurface tile-drains could be explored. Further monitoring of annual P loads from surface intakes, seasonal fluctuations in P concentrations in surface intakes and in runoff in Midwest watersheds is needed to support strategies to help mitigate regional water problems as well as the seasonal hypoxic zone that forms in the Gulf of Mexico.
- Fecal bacteria resuspension from beach sediments was completely neglected while developing the beach bacteria TMDL, and studies have demonstrated bacteria resuspension to be an important factor. Therefore, models that include resuspension and well defined bacterial parameters are needed for accurate TMDLs.
- In this study, the CTI model was tested in a low-relief, agricultural watershed. Future work is recommended to evaluate other watersheds with wide-ranging topography.

- The CTI model requires field surveyed data to calibrate the model; therefore, physical based models that base erosion prediction on soil shear stress, surface slope, rainfall, peak runoff, crop growth stage, and crop cover need to be developed and tested.

APPENDIX. GRASSED WATERWAY IDENTIFICATION AND DESIGN PROCESS PYTHON CODE

Grassed waterway identification

Import Modules

```
import arcpy
```

```
from arcpy import env
```

```
from arcpy.sa import *
```

Import Extensions and Set the Overwrite Output to True

```
arcpy.CheckOutExtension ("spatial")
```

```
arcpy.env.overwriteOutput = True
```

```
map_document = "Template.mxd"
```

Get the Input Data

```
Input_DEM = 'C:\\Rohith\\Thesis\\Grassed Waterway GIS TOOL\\Python Scripts\\GIS  
Project\\Inputs\\hg_30_dem'
```

```
Input_Field = 'C:\\Rohith\\Thesis\\Grassed Waterway GIS TOOL\\Python Scripts\\GIS  
Project\\Inputs\\field2.shp'
```

```
Output_Path = 'C:\\Rohith\\Thesis\\Grassed Waterway GIS TOOL\\Python Scripts\\GIS  
Project\\Output\\30 meter\\Field2\\T=30'
```

Fill the DEM

```
arcpy.AddMessage ('Filling DEM...')
```

```
FILLDEM = arcpy.gp.Fill_sa (Input_DEM, Output_Path + "\\FILLDEM','")
```

Find the Flow Direction

```
arcpy.AddMessage ('Calculating Flow Direction...')
```

```

FLOWDIR = arcpy.gp.FlowDirection_sa (FILLDEM, Output_Path + "\\FLOWDIR',
"NORMAL", "")

# Find the Flow Accumulation

arcpy.AddMessage ('Calculating Flow Accumulation...')

FLOWACC = arcpy.gp.FlowAccumulation_sa (FLOWDIR, Output_Path + "\\FLOWACC',
"", "INTEGER")

# Find the Major Flow Paths in Fields

arcpy.AddMessage ('Calculating the major flow paths...')

inRaster = Raster (Output_Path + "\\FLOWACC')

inTrueRaster = Raster (Output_Path + "\\FLOWACC')

whereClause = "VALUE >= 2000"

STREAMS = Con (inRaster, inTrueRaster, "", whereClause)

STREAMS.save (Output_Path + "\\STREAMS')

# Find the Drainage Area

arcpy.AddMessage ('Calculating the Drainage Area...')

DrainageArea = arcpy.gp.Times_sa (FLOWACC, 100, Output_Path + "\\DRAINAREA')

# Find the Slope in the Area of Interest

arcpy.AddMessage ('Calculating Slope...')

SLOPE = arcpy.gp.Slope_sa (FILLDEM, Output_Path + "\\SLOPE', "PERCENT_RISE", "")

# Find the % Slope Values

arcpy.AddMessage ('Converting the Slope values into % values...')

SLOPE_PERCENT = arcpy.gp.Divide_sa (SLOPE, 10000, Output_Path + "\\SLOPE%')

# Find the Slope*Area

```

```

arcpy.AddMessage ('Calculating the SA values...')

TEMPSA = arcpy.gp.Times_sa (DrainageArea, SLOPE_PERCENT, Output_Path +
'\TEMPSA')

# Find the Curvature of DEM

arcpy.AddMessage ('Calculating the Curvature of the DEM...')

Planc = Output_Path + '\planc'

TEMPCURVATURE = arcpy.gp.Curvature_sa (FILLDEM, Output_Path + '\TempCurve',
0.01, "", Planc)

# Convert the negative CTI values

NegPlanc = arcpy.gp.Negate_sa (Planc, Output_Path + '\NegPlanc')

# Find the Erosion Prone Areas

arcpy.AddMessage ('Finding the Erosion Prone Areas...')

TEMPCTI = arcpy.gp.Times_sa (TEMPSA, NegPlanc, Output_Path + '\TEMPCTI')

inRaster = Raster (Output_Path + '\TEMPCTI')

inTrueRaster = Raster (Output_Path + '\TEMPCTI')

whereClause = "VALUE >= 2000"

CTI = Con (inRaster, inTrueRaster, "", whereClause)

CTI.save (Output_Path + '\CTI')

inRaster = Raster (Output_Path + '\CTI')

inMask = Raster (Output_Path + '\STREAMS')

EPA = ExtractByMask (inRaster, inMask)

EPA.save (Output_Path + '\EPA')

inRaster = Raster(Output_Path + '\EPA')

```

```

inMask = Input_Field

FieldEPA = ExtractByMask (inRaster, inMask)

FieldEPA.save (Output_Path + '\\FieldEPA')

inRaster = Raster (Output_Path + '\\TEMPSA')

inTrueRaster = Raster (Output_Path + '\\TEMPSA')

whereClause = "VALUE >= 2000"

SA = Con (inRaster, inTrueRaster, "", whereClause)

SA.save (Output_Path + '\\SA')

```

Grassed waterway design

Import System Modules

```
import numpy
```

```
import math
```

```
import arcpy
```

```
import scipy.optimize as so
```

```
from numpy import power
```

```
from numpy import exp
```

```
from numpy import log
```

```
from numpy import sqrt
```

```
arcpy.env.overwriteOutput = True
```

Define Functions to calculate the dimensions of grassed waterways

```
def InitialAbstraction (P):
```

```
    S = 71.64
```



```
Q = power ((float (P) - float (0.2*S)), 2) / (float (P) + float ((0.8*S)))
```

```
return Q
```

```
def TimeofConcentration (L, s):
```

```
CN = 78
```

```
TC = power (float (L), 0.8) * power (((1000/int (CN)) - 9), 0.7) / (1900*sqrt (float (s)))
```

```
return TC
```

```
def PeakRunoffRate (C, I, A):
```

```
Qpeak = float (C) * float (I) * float (A) / 3.6
```

```
return Qpeak
```

```
def LongGrassCase (r,*func_args):
```

```
(Vp, s, C, C1) = func_args
```

```
n = float (exp(C1*((0.0133*((log(Vp*r))*log(Vp*r)))-(0.0322*log(Vp*r))+0.145)-4.16))
```

```
Eq1 = (Vp*r) - (C*power (r, 5/3.0)*sqrt(s)/n)
```

```
return Eq1
```

```
def Dimensions(d,*func_args):
```

```
(A,P) = func_args
```

```
Eq2 = 27*A*A + (32*power (d, 4)) - (18*d*P*A)
```

```
return Eq2
```

```
def ShortGrassCase (V, *func_args):
```

```
(s,C,C1,R) = func_args
```

```
Eq3 = (C*power(R, 5/3)*sqrt(s)/VR)-exp(C1*((0.0133*((log(V*R))*log(V*R)))-
```

```
(0.0322*log(V*R))+0.145)-4.16))
```

```
return Eq3
```

Obtain Input Parameters

```
Precipitation = arcpy.GetParameterAsText (0)
```

```
FlowLength = arcpy.GetParameterAsText (1)
```

```
Slope = arcpy.GetParameterAsText (2)
```

```
Intensity = arcpy.GetParameterAsText (3)
```

```
Area = arcpy.GetParameterAsText (4)
```

```
Vp = arcpy.GetParameterAsText (5)
```

```
C = arcpy.GetParameterAsText (6)
```

```
C1 = arcpy.GetParameterAsText (7)
```

Estimate the Runoff, TimeofConcentration, and PeakRunoffRate

```
Output = open ('myfile.txt', 'w')
```

```
arcpy.AddMessage ('Calculating Runoff...')
```

```
Runoff = float (InitialAbstraction(Precipitation))
```

```
arcpy.AddMessage ('Calculating Time of Concentration')
```

```
TimeofConcentration = float (TimeofConcentration(FlowLength,Slope))
```

```
arcpy.AddMessage ('Calculating Peak Runoff Rate...')
```

```
RunoffCoeff = float (Runoff)/float (Precipitation)
```

```
Qp = float (PeakRunoffRate (RunoffCoeff, Intensity, Area))
```

Estimate the Dimensions for GWW for Long Grass

```
arcpy.AddMessage ('Calculating the GWW dimensions for Long Grass...')
```

```
Rinit = 3.5
```

```
func_args = (Vp,Slope,C,C1)
```

```
full_output = 1
```

```

arcpy.AddMessage ('Ran until funcargs..')

optimize_results = so.fsolve (LongGrassCase, Rinit, func_args, full_output = 1)

R = optimize_results [0][0]

'Area of Grassed Waterway: Ag'

Ag = Qp/Vp

'Perimeter of Grassed Waterway: P'

P = Ag/R

func_args = (Ag, P)

Dinit = 3.1

full_output = 1

OptimizeDim = so.fsolve (Dimensions,Dinit, func_args, full_output = 1)

D = optimizeDim [0][0]

T = 3*Ag/(2.0*D)

# Check the Estimated Dimensions with permissible velocity

arcpy.AddMessage ('The Velocity in the Channel in Short Grass Conditions is being Tested
for Exceeding Permissible Velocity')

Vinit = 2.5

func_args = (s, C, C1, R)

full_output = 1

optimizeVel = so.fsolve (ShortGrassCase, Vinit, func_args, full_output = 1)

V = optimizeVel [0][0]

if V > Vp:

    arcpy.AddMessage ('The velocity in the channel exceeds the permissible velocity')

```

else:

```
    arcpy.AddMessage ('The estimated dimensions of the channel can hold the peak runoff  
rate')
```

```
# Write the Output in a Text File
```

```
Output.write ('The total runoff from the field is: ' + str (Runoff) + 'mm.\n')
```

```
Output.write ('The Runoff Coefficient for the rainfall is: '+str (RunoffCoeff) + '.\n')
```

```
# Output.write ('The Suggested Hydraulic Radius for the GWW is : '+str(R)+' meters.')
```

```
Output.write ('The time of concentration for the outlet is: ' + str (TimeofConcentration) + '  
h.'+ '\n')
```

```
Output.write ('The Peak Runoff Rate is: ' + str(Qp )+ ' cms.\n')
```

```
Output.close ()
```