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Evaluation of subsurface drainage on phosphorus losses and application of the SoilIceDB model in the Black Hawk Lake Watershed, Iowa

by

Conrad E. Brendel

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

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The student author and the program of study committee are solely responsible for the content of this thesis. The Graduate College will ensure this thesis is globally accessible and will not permit alterations after a degree is conferred.

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ABSTRACT

The Upper Midwestern United States is extensively tile drained and drainage provides a preferential pathway for nutrient losses from cropland. Phosphorus (P) in subsurface drainage is the focus of current research on agricultural nutrient losses, however, the effects of best management practice (BMP) implementation on drainage phosphorus losses is unclear. Phosphorus and suspended solids losses were monitored at five sites — two streams, two tile drain outlets, and a grassed waterway — located in three paired subwatersheds of Iowa’s Black Hawk Lake watershed. Subwatersheds ranged in size from 221.23-822.49 hectares and BMP implementation ranged from 22.5-87.5% of the subwatershed area. Specific water quality analytes examined include total phosphorus (TP), dissolved reactive phosphorus (DRP), total suspended solids (TSS), and volatile suspended solids (VSS). The results from the study reveal that drainage analyte losses can equal or exceed those of surface waters. Precipitation events accounted for the majority of analyte losses at each of the subwatersheds. An analysis of intra-event samples from the five monitoring sites showed that flow is the driving factor of event analyte concentrations. Results from the paired watersheds indicate that BMP implementation has a positive impact on P and suspended solids losses in both surface and drainage waters. This study also evaluated the performance of the new drainage phosphorus and sediment loss model, SoilIceDB, at the small catchment scale as well as its applicability to cropland outside of Scandinavia. Preliminary results suggest that with more extensive calibration, the model will be able to acceptably simulate drainage flow and DRP losses. Establishing relationships between BMP implementation and P losses and a successful model will assist water quality improvement projects and could identify areas for remediation and BMP implementation.
CHAPTER 1: INTRODUCTION

Artificial subsurface drainage is crucial to the success of row-crop agriculture in the Upper Midwestern United States. The use of drainage has transformed this region, previously covered in swamps and wetlands, into some of the world’s most fertile agricultural land (Du et al., 2005). Benefits of subsurface drainage include allowing for trafficable conditions for timely field operations in seasonally and perennially wet locations, preventing excessive soil water conditions, providing salinity control in irrigated areas, and increasing nutrient uptake of crops by creating a well-aerated root environment (Du et al., 2005, Fausey et al., 1987, Reeve and Fausey, 1974, Vos, 1987, Zucker and Brown, 1998). Although drainage provides many benefits to agricultural production, drainage also represents a major pathway for nutrient losses from agricultural lands.

While past research on drainage nutrient losses has been more focused on nitrogen, research has also shown that drainage is a significant source of phosphorus (P) losses. Studies have found that artificial drainage contributes P loads ranging from 17% to greater than 50% of the total P losses (Culley et al., 1983, Jamieson et al., 2003, Ruark et al., 2012, Smith et al., 2015, Tomer et al., 2010). Leaching through the soil matrix is one way in which P enters drainage waters. In locations where soil P concentrations are very high and the soil P sorption capacity is very low, P has a high potential to leach from the soil matrix into groundwater (Kleinman et al., 2007, Vadas et al., 2007). As soil moisture content decreases, the relative contribution of macropores to chemical transport and water movement increases (Shipitalo and Edwards, 1996). Macropore
flow occurs when surface soil layers are saturated during non-ponded conditions (Andreini and Steenhuis, 1990, Shipitalo and Edwards, 1996). By studying the timing of drainage P transport relative to drainage flow, Geohring et al. (2001) concluded that macropore flow is the primary transport mechanism of TP through soil. During ponded conditions, P enters drainage through surface intakes. This pathway has been found to deliver at least 75% of the drainage Total P (TP) load (Tomer et al., 2010). Substantial reductions of 66-78% in TP drainage loads have been observed when open surface inlets and tile risers were replaced with blind inlets (Feyereisen et al., 2015, Smith and Livingston, 2013).

Modeling of P transport in drainage is very limited. Radcliffe et al. (2015) reviewed the ability of P indices and eight models (ADAPT, APEX, DRAINMOD, HSPF, HYDRUS, ICECREAMDB, PLEASE, and SWAT) to estimate P losses in drainage waters. The authors consider P indices too simplistic to adequately model P fate and transport. ICECREAMDB is the only one of the eight models that was specifically designed to model P losses in artificially drained areas and they deem it the most promising, but emphasize that more testing is needed. The authors also highlight limitations of the remaining models; APEX and HSPF both indirectly simulate drainage and DRAINMOD and HYDRUS lack P routines. In addition, most models lack programming for important P transport pathways including leaching (SWAT), macropore transport (APEX, PLEASE, SWAT), and particulate P transport in runoff or through the soil matrix (HSPF, HYDRUS, PLEASE). In another study, Que et al. (2015) modeled the effects of tile drainage control on nutrient loads in the South Nation River basin in Canada using AnnAGNPS. Modeled P loading
changes agreed with observed changes less than 50% of the time and the authors suggest that the AnnAGNPS model could be improved by taking into account P transport in drainage.

Few studies have combined extensive field monitoring — including measuring drainage total suspended solids (TSS) and volatile suspended solids (VSS) concentrations and loads, splitting P loads into event and baseflow components, and measuring intra-event P concentrations — with P modeling of artificially drained fields. The goal of this study is to evaluate and predict the effects of subsurface drainage and different levels of best management practice (BMP) implementation on P export. Specific objectives include measuring intra-event P trends, comparing P in drainage vs. surface flow during a range of flow conditions, and testing the ICECREAMDB model in the Des Moines Lobe land-region in Iowa. An increased understanding of sources and contributions of P during varying flow conditions is needed for water quality improvement projects and a successful model could be used to prioritize areas for best management practice implementation.
CHAPTER 2: LITERATURE REVIEW

2.1. Regional Water Problem

Many Iowa surface waterbodies violate water quality standards. On the 2014 Iowa 303(d) list of impaired waters, 571 waterbodies were listed with a total of 754 impairments (IDNR, 2015). Water quality impairments due to nutrients are of increasing concern and the U.S. EPA is recommending that states adopt numeric criteria for total nitrogen, total phosphorus (TP), and clarity. Artificial subsurface drainage is vital to the success of row-crop agriculture in the Upper Midwestern United States; however, drainage provides a direct pathway for nutrients to enter surface waters. Nitrate is typically the focus of nutrients in drainage waters but recent studies have identified drainage phosphorus (P) loads greater than those required for eutrophication.

2.2. Phosphorus in Soils

Because of low atmospheric returns, P in soils is primarily derived from the soil’s parent material (Walker and Syers, 1976). Typically, however, most of the total soil P content is unavailable for biological utilization because it is bound in mineral particles, absorbed to mineral surfaces, or unavailable due to secondary mineral formation (occlusion) (Yang et al., 2013). Therefore, in order to increase crop production, farmers supplement soil with P amendments such as manure and commercial fertilizers. Common commercial phosphate fertilizers include Superphosphate (OSP), Concentrated Superphosphate (CSP), Monoammonium Phosphate (MAP), Diammonium Phosphate (DAP), Ammonium Polyphosphate (APP), and Rock Phosphate (Rehm et al., 2010).
Phosphorus in soils can be divided into three main pools: solution P, active P, and fixed P. Within these pools, P can be further subdivided into organic and inorganic forms. The solution P pool is continuously replenished and comprises a very small portion of the total soil P, usually only a fraction of a pound of P per acre. Within this pool, P is typically in the form of orthophosphate which is the only form of P which plants will uptake. In comparison, the active P pool is quite large and can contain several pounds to several hundred pounds of P per acre. The active P pool contains organic P that is easily mineralized and inorganic P that is adsorbed to small particles within the soil. Finally, the fixed P pool contains insoluble inorganic P compounds and organic P compounds that are resistant to mineralization. Overall, this pool has very little impact on soil fertility (Busman et al., 2009).

**2.3. Phosphorus in Water**

Phosphorus in water can occur in several forms: dissolved reactive phosphorus (DRP), dissolved organic phosphorus (DOP), and total particulate phosphorus (TPP). Combined, DRP, DOP, and TPP constitute the TP content (Beauchemin et al., 1998). Soil type, tile depth, soil and crop management, and weather affect the P losses in subsurface drainage (Culley et al., 1983). As a result, there is high variability in reported P losses. In a two year study of 27 tile-drained soils, Beauchemin et al. (1998) found that DRP, DOP, and TPP respectively account for 0-59%, 0-79%, and 2-96% of TP losses. In a similar study, Heckrath et al. (1995) found that DRP, DOP, and TPP respectively account for 66-86%, 4.5-11%, and 8-35% of TP.
2.4. Environmental Impact of Phosphorus

Non-point source pollution of P from agricultural land has a significant impact upon eutrophication in lakes and streams. Regions of high animal production, like Iowa, are at risk of high P losses because manure is often applied in excess of crop nutrient demands in order to dispose of the surplus waste (Stamm et al., 1998). Furthermore, manure is often applied at rates to meet crop N requirements, which can result in excess P. Phosphorus is an essential nutrient for crop growth, but P can also increase surface water productivity in freshwater systems. The most common effect of increased N and P concentrations in surface waters is an increase in the abundance of algae and aquatic plants (Smith, 2003). According to the Swedish classification system for eutrophication, TP concentrations of 25-50, 50-100, and greater than 100 µg/L correspond to Eutrophic I conditions, Eutrophic II conditions, and hypertrophic conditions, respectively (Willen, 2000). These values are an order of magnitude less than the soil solution P concentrations needed for plant growth (0.2-0.3 mg/L) (Tisdale et al., 1993); this highlights the importance of controlling P-losses to limit eutrophication. The U.S. EPA recommends that to control eutrophication, stream TP concentrations should not exceed 0.05 mg/L at points where streams enter lakes or reservoirs and should not exceed 0.1 mg/L in streams which do not discharge into lakes or reservoirs (Mueller et al., 1996). Controlling P-losses is also important because it is difficult to limit inputs of other nutrients such as nitrogen and carbon which are exchanged between air and water through processes like nitrogen fixation by blue-green algae (Daniel et al., 1998).
The impacts of eutrophication on surface waters are serious and widespread. Eutrophication impairs surface water use for drinking, industry, recreation, and fisheries due to excessive growth of algae and aquatic plants and the oxygen shortages caused by their senescence and decomposition (Daniel et al., 1998). Blue-green algae in drinking water supplies can have serious human health implications. Toxins in blue-green algae include hepatotoxic peptides, neurotoxic alkaloids, saxitoxin derivatives, a cytotoxic alkaloid, allergens, and lipopolysaccharides; these toxins can cause liver injury in dialysis patients, resulting in death. In addition, recreational exposure to blue-green algae can result in illnesses ranging from acute pneumonia and hepatoenteritis to skin irritation and gastroenteritis (Falconer, 1999).

2.5. Phosphorus Pathways

There are two primary inputs of P to surface waters: artificial subsurface drainage and surface runoff.

2.5.1. Phosphorus Through Subsurface Drainage

Once covered in swamps and wetlands, the use of artificial subsurface drainage has transformed the Upper Midwestern United States into some of the most fertile agricultural land in the world (Du et al., 2005). Excess water in the soil profile and ponded on the soil surface is removed by gravity flow through sub-surface drainage pipes, installed below the root zone, and above-ground surface intakes (Zucker and Brown, 1998). Subsurface drainage allows for trafficable conditions for timely field operations in seasonally and perennially wet locations, prevents excessive soil water conditions, provides salinity control in irrigated areas, and increases nutrient uptake of
crops by creating a well-aerated root environment (Du et al., 2005, Fausey et al., 1987, Reeve and Fausey, 1974, Vos, 1987, Zucker and Brown, 1998). It is estimated that 25% of the cropland in the United States and Canada could not be productive without artificial drainage (Pavelis, 1987, Skaggs et al., 1994).

Although drainage provides many documented benefits to crop production, it also serves as a significant source of P loading to surface waters. In Iowa, Tomer et al. (2010) used source-pathway separation and found that approximately half of the outlet P load was due to drainage on a unit-area basis. Phosphorus enters drainage waters by leaching through the soil matrix or by preferential flow.

In areas where the soil P sorption capacity is very low but soil P concentrations are very high, P has a high potential to leach from the soil matrix into groundwater (Kleinman et al., 2007, Vadas et al., 2007). Heckrath et al. (1995) studied the correlation between soil P concentrations and drainage water P concentrations and found that P is strongly retained in the plow layer (0-23cm depth) of soils with Olsen-P concentrations up to 60mg/kg of soil. However, when the soil Olsen-P concentration exceeds this, they observed sharp increases in drainage DRP and TP concentrations. Maguire and Sims (2002) also observed rapid increases of drainage DRP above certain soil P change points; they identified the following change points for Mehlich-3 extractable P, Mehlich-1 extractable P, Iron Strip Phosphorus (FeO-P), Water Soluble Phosphorus (WSP), and CaCl2-P respectively: 181, 81, 42.6, 8.6, and 1.59 mg/kg of soil. McDowell and Sharples (2001) observed a similar Mehlich-3 extractable P change point of 193 mg/kg of soil for DRP in arable soils. Furthermore, Culley et al. (1983) concluded that 34% of the total
drainage P load at their site was sediment associated and Heathwaite and Dils (2000) found a significant correlation (P<0.01) between suspended sediment concentration and P losses in drainage. Therefore, controlling soil P concentrations is important in minimizing the risk of P losses from topsoil through leaching.

Preferential flow pathways include flow through soil macropores and surface intakes for subsurface drainage. In preferential flow, P is primarily transported in the particulate fraction and is associated with the organic and colloidal forms of P (Heathwaite and Dils, 2000).

Macropore flow occurs in saturated soil surface layers during non-ponded conditions (Andreini and Steenhuis, 1990). Geohring et al. (2001) studied P transport through macropores and concluded that macropore flow is the primary transport mechanism of TP through soil based on the timing of P transport in drainage effluent relative to tile flow. Then, in a related column study, Geohring et al. (2001) found that soluble P may be transported through 1mm or larger macropores with negligible P sorption to pore walls and that no measurable P is transported in the absence of macropores. Furthermore, they concluded that high drainage P loads after manure application can be attributed to macropore transport. Heathwaite and Dils (2000) measured TP concentrations in macropores and observed a mean TP concentration of 1.2 mg/L in the upper 0-15cm of grassland soil with concentrations decreasing with increasing depth.
During ponded conditions, P enters drainage through surface intakes. Using hydrograph separation, Tomer et al. (2010) found that at least 75% of P in drainage waters was delivered from surface intakes. Feyereisen et al. (2015) found that soluble reactive phosphorus (SRP) in drainage water was reduced by 35% when surface intakes were replaced with gravel inlets and that TP loads were reduced by 66% when the surface intakes were replaced with blind inlets. Similarly, Smith and Livingston (2013) found that DRP and TP in drainage water were reduced by 68 and 65%, respectively, by replacing surface intake risers with blind inlets.

Studies have found that subsurface drainage contain TP loads ranging from 17% to greater than 50% of the total P losses, emphasizing the significance of drainage P contributions to surface waters (Culley et al., 1983, Jamieson et al., 2003, Ruark et al., 2012, Smith et al., 2015, Tomer et al., 2010). Furthermore, Smith et al. (2015) observed that 49% of the soluble phosphorus (SP) loading occurred through subsurface drainage and Ruark et al. (2012) estimated that 16-58% of dissolved P loads in Wisconsin were due to tile drainage.

2.5.2. Phosphorus in Surface Runoff

Surface runoff is another significant source of P loading to surface waters. In runoff, P is transported primarily in the dissolved fraction (Heathwaite and Dils, 2000). However, a rainfall simulation study by Flanagan and Foster (1989) found that peak storm intensity and time of occurrence of the peak storm intensity did not have a significant effect ($\alpha=0.1$) on soluble phosphate losses because of large variability in measured values and low runoff totals.
Relationships between Runoff P and P application and soil P were identified in Allen et al. (2006). Indoor rainfall simulations were conducted on 5 Midwest soils: Fayette, Harps, Marshall, Nicollet, and Tama. Monoammonium phosphate fertilizer (NH₄H₂PO₄) was applied at rates of 0, 50, 125, 300, and 600 mg P/kg and as the application rate increased, DRP concentration, bioavailable phosphorus (BAP) concentration, DRP/Total Runoff P (TPR) ratio, and BAP/TPR ratio increased linearly for all soils. For all soils, runoff DRP, BAP, and TPR increased linearly (p<0.05) as the soil P concentration was increased. Furthermore, runoff DRP and BAP increased linearly (p<0.05) with increasing estimated soil P saturation for all indices. No change points for runoff DRP or BAP were identified for any soil type. McDowell and Sharpley (2001) also found that surface runoff DRP concentrations increased with soil test P concentrations for Alvira, Berks, Calvin, Denbigh, and Watson soils. However, they identified change points in DRP in the first 250 mL of runoff at Mehlich-3 P concentrations above 185 mg/kg of soil and Olsen P concentrations above 35 mg/kg of soil. Furthermore, DRP concentrations were higher in the first 250mL of runoff than all of the surface runoff combined. Although Allen et al. (2006) identified increases in DRP and BAP for all soils, P losses were greater in certain types of soils than others; DRP and BAP losses in calcareous Harps soil had two times greater rates of increase (0.0037 mg/L DRP per mg/kg applied P & 0.0049 mg/L BAP per mg/kg applied P) than other soils.

Since P in runoff is not typically associated with the particulate fraction, tillage practices that control erosion do not necessarily prevent the loss of soluble P. Römkens et al. (1973) performed a rainfall simulation comparing soluble P losses among five
tillage-planting systems — coulter-plant (coulter), buffalo till-plant (till), chisel-plant (chisel), disk and coulter-plant (disk), and conventional-plant (conventional) — and found that runoff nutrient losses were greatest in coulter followed by chisel, till, disk, and conventional. However, measured soil losses for each system followed a nearly opposite order with the conventional system having the highest measured soil losses, followed by till, disk, coulter, and chisel. Among unfertilized plots, runoff sediment P concentrations were lowest in conventional with increasing concentrations measured in coulter, till, and chisel. In fertilized plots, sediment P concentrations were also lowest in conventional with increasing concentrations in disk, till, chisel, and much higher concentrations in coulter. Practices, such as chisel and coulter, produce soil ridges and crop residue which can trap larger silt and sand-sized particles, resulting in runoff sediment with relatively high clay fractions; this selective erosion explains the differences between runoff sediment P concentrations between practices. Römkens et al. (1973) identified a correlation coefficient of 0.96 for the correlation between the P concentration and clay content of runoff sediment of unfertilized plots. Overall, P losses in surface runoff can also be quite significant; Heathwaite and Dils (2000) observed a mean TP concentration of 1.1 mg/L in surface runoff which is much greater than the concentration classifying waters as hypertrophic.

2.6. Past Modeling of P in Drainage

The suitability of many models has been evaluated for the use of predicting P in drainage waters. Models can be categorized as either process based or empirically based; process based models use solutions to differential equations to estimate P losses
whereas empirically based models use statistics and regression equations to estimate P losses (Radcliffe et al., 2015). Many models have a combination of process based and empirically based components and are therefore categorized as mixed models. A summary of these models and their limitations follows:

2.6.1. ADAPT

The Agricultural Drainage and Pesticide Transport (ADAPT) model was developed by incorporating drainage and subirrigation algorithms from DRAINMOD into the GLEAMS model. ADAPT is a field scale, daily time-step, mixed model. The Green-Ampt Equation is used to estimate infiltration and the curve number method is used to estimate surface runoff. Subsurface hydrology is simulated using the DRAINMOD algorithms. ADAPT also includes a simple model of macropore flow; macropore flow volume is estimated as a function of clay content and the number of dry days during which the soil water supply does not meet the potential evapotranspiration demand (Chung et al., 1992).

ADAPT’s P routines are based on the Erosion-Productivity Impact Calculator (EPIC) model, however ADAPT is limited in its ability to model P losses because it only includes P routines for surface runoff. The forms of P simulated in ADAPT are soil mineral P, runoff dissolved P, and adsorbed P in runoff sediment. In order to effectively model P in drained agroecosystems, ADAPT must be improved by including P routines for drainage losses related to soil type, farming practices, and drainage water management (Radcliffe et al., 2015).
2.6.2. AnnAGNPS

The AnnAGNPS (Annualized Agricultural Non-Point Source Pollution) model is an evolution of the AGNPS model which was designed to simulate non-point pollution loads in agricultural areas. AnnAGNPS is a watershed scale, daily time-step, empirical model. In the model, watersheds are subdivided into small, homogeneous subwatersheds called cells which are connected by stream channels called reaches (Bingner et al., 2015). Within the model, the curve number method, Revised Universal Soil Loss Equation, and Hydrogeomorphic Universal Soil Loss Equation are used to estimate surface runoff, sediment delivery to the edge of the field, and sediment loads to the reach respectively. AnnAGNPS simulates tile drainage flow using the Hooghoudt’s equation (Que et al., 2015).

Overall, AnnAGNPS is limited in its ability to model drainage P losses in agroecosystems because it does not consider the transport and fate of sediment and P in surface and subsurface flow pathways. The model treats P transport from a surface pool perspective. While this is effective for many systems, several improvements could be made when assessing drainage management practices in flat, drainage dominated watersheds. Specifically, AnnAGNPS could benefit by including routines for subsurface transport and fate of P (Que et al., 2015).

2.6.3. APEX

The Agricultural Policy Environmental Extender (APEX) model is based on the EPIC model. While EPIC can only be executed for single fields, APEX can be executed for single fields as well as whole farms or watersheds subdivided by fields, soil types,
landscape positions, or subwatersheds. APEX is a mixed model which functions on a daily-yearly time step and can be used for long-term, continuous simulations. APEX also includes modeling of extensive management practices including nutrient management practices, tillage operations, conservation practices, alternative cropping systems, and management practices related to sediment, nutrient, and other pollutant losses through surface runoff. Surface runoff is estimated either by a modified curve number method or by the Green-Ampt infiltration equation. Although APEX does not simulate macropore flow, drainage flow is simulated as an increase of the natural lateral subsurface flow in the soil layer containing the drainage tile (Gassman et al., 2009).

APEX uses a modified version of EPIC’s three-pool model to simulate inorganic P cycling. Despite this capability, however, only two studies have used APEX to estimate P losses through drainage; the rest have focused on nitrogen. Francesconi et al. (2016) modeled SP in surface runoff and tile flow in a corn/soybean rotation in Indiana’s St. Joseph River watershed. Overall, the APEX model performance was poor but they found that using the model’s Langmuir (nonlinear) sorption option improved tile flow SP estimates by 30% during corn years, when P inputs were added. Ford et al. (2015) performed a sensitivity analysis on APEX and found that APEX adequately captures surface DRP concentration dynamics. Furthermore, they found that APEX was able to estimate median drainage DRP concentrations well at the monthly timescale; however, the model underestimated several high measured drainage DRP concentrations by nearly an order of magnitude for the monthly and annual timescales.
Overall, APEX is limited in its ability to model P losses in drained agroecosystems. The results of Ford et al. (2015) highlight negative feedback mechanisms in which decreases in P loads from one source cause increases in P loads from alternative sources. APEX performance could be improved by providing a more complete and soil-specific representation of chemical and hydrologic processes (Francesconi et al., 2016, Radcliffe et al., 2015). In addition, since the P partition coefficient is a single user-defined value for the entire watershed, APEX could over/underestimate the drainage P losses. Furthermore, in APEX, the P adsorption capacity is based only on the soil clay content whereas the capacity is actually influenced by several dynamic soil properties (Radcliffe et al., 2015). Finally, APEX could be improved by adding preferential flow routines (Ford et al., 2015, Radcliffe et al., 2015).

2.6.4. DRAINMOD

DRAINMOD is a field scale, process based model with a subhourly-daily to timestep, depending on the process being modeled. The model was originally designed to simulate the performance of agricultural drainage and related water management systems, but functionality has been expanded to include the hydrology, soil carbon, nitrogen dynamics, and vegetation growth for agricultural and forest ecosystems (Negm et al., 2014, Tian et al., 2012). Hydrologic processes are simulated in DRAINMOD by a one-dimensional model based on the water balance approach.

Currently, DRAINMOD does not simulate P fate and transport in drained fields. Two primary modifications must be made to DRAINMOD in order to model P in
drainage: modeling of flow and P transport through macropores and modeling of soil erosion and the associated particulate phosphorus (PP) losses (Radcliffe et al., 2015).

2.6.5. **HSPF**

The Hydrological Simulation Program – FORTRAN (HSPF) is a mixed model which is an evolution of the Pesticide Transport and Runoff (PTR) model and the Agricultural Runoff Management (ARM) model. HSPF was designed to simulate hydrologic and water quality processes on pervious and impervious land surfaces, in the soil profile, and in streams and well-mixed impoundments. The model operates at the watershed scale with an hourly-yearly timestep; landscape hydrologic processes are simulated at a user defined segment area comprised of land uses with similar hydrologic characteristics. Surface runoff is estimated using an empirical method based on the Stanford Watershed Model (Bicknell et al., 1996).

P fate and transport in HSPF is simulated using the AGCHEM module. Within AGCHEM, model parameters may be adjusted to simulate the transformation and movement of the different forms of soil P (Donigian et al., 1994). However, HSPF’s ability to model P losses in drained fields is limited because it does not model drainage flow and P loss directly nor does it simulate leaching of particulate P through the soil matrix (Radcliffe et al., 2015).

2.6.6. **HYDRUS**

HYDRUS is a fully process based, field scale model developed by the U.S. Salinity Laboratory. Three versions of the model exist: the freeware one-dimensional version
(HYDRUS-1D) and the commercial two- and three-dimensional versions (HYDRUS 2D/3D). The models run at a minute-year long time step. As a fully process based model, numerical solutions for the Richards equation, the advection dispersion equation, and the transport equation are used to solve for water flow, solute transport, and heat respectively. HYDRUS also includes many options to simulate preferential flow including the mobile-immobile model, the dual-porosity model, the dual-permeability model, and the dual-permeability model with the mobile-immobile model (Radcliffe et al., 2015).

Drainage flow is simulated by including a boundary condition at the bottom of HYDRUS-1D, at the side of HYDRUS 2D, or by using an internal node/boundary. The ability to model drainage P losses in HYDRUS is limited because the model does not simulate erosion and the associated PP losses and lacks specific P routines and management options. It is possible to simulate P in the soil profile in HYDRUS by specifying sorption equations, parameter values, and transformation rates but the large number of unique parameters and the difficulty in obtaining these values makes this unfeasible. Similarly, management practices can be simulated in HYDRUS by adjusting the hydraulic properties of the affected soil layers but these parameters would need to be respecified each time the management practices were changed (Radcliffe et al., 2015).

2.6.7. **PLEASE**

The Phosphorus LEAching from Soils to the Environment (PLEASE) model is a static, mixed model based on the kinetics of inorganic P in soils and the lateral water flow from soils to surface waters. The model is field scale and runs on a yearly time step.
The total annual lateral water flux from the field is estimated as the net annual precipitation surplus adjusted for the annual net seepage flux of water from the soil profile to groundwater (Schoumans et al., 2013).

PLEASE is able to estimate the load of P leaching from fields to surface waters, the dissolved inorganic P concentration in the soil solution, the amount of reversibly sorbed P in soil layers, and the TP concentration profile with depth. The advantage of the PLEASE model is that it was developed to simulate P losses in a region where P losses are important. However, disadvantages of PLEASE are that it does not estimate runoff losses of P or macropore flow and with its yearly time-step it cannot be used to model precipitation events (Radcliffe et al., 2015).

2.6.8. SWAT

The Soil and Water Assessment Tool (SWAT) is a watershed scale mixed model that operates at various time scales from minutes-years. SWAT consists of a combination of primarily process-based submodels for nutrient subroutines and empirical submodels for routines such as erosion and surface runoff. Tile drainage is simulated in SWAT at the hydrologic response unit scale by adjusting parameters for drainage depth, time needed to drain soil to field capacity, and the lag time for water to enter surface waters after entering the tile (Neitsch et al., 2002). The DRAINMOD model was recently incorporated into SWAT to produce the SWATDRAIN model; in this model, surface flow is simulated using SWAT and subsurface flow is simulated using DRAINMOD (Golmohammadi et al., 2016).
SWAT simulates P losses using mechanistic P routines. In order to improve P loss estimates, SWAT parameters such as PHOSKD (P soil partitioning coefficient) and PSP (P availability index) are commonly calibrated. However, because SWAT lacks P transport components in the leaching and preferential flow pathways it is limited in its ability to model P losses in drained agroecosystems (Radcliffe et al., 2015).

2.7. The ICECREAMDB Model

Radcliffe et al. (2015) deemed ICECREAMDB as the most promising model for simulating P losses in drainage because it minimizes the number of input parameters by combining mechanistic and empirical approaches.

2.7.1. ICECREAMDB

ICECREAMDB is a graphical front-end for the ICECREAM model that also includes options to structure outputs. ICECREAMDB is a management oriented P loss model that quantifies runoff, erosion, and P losses and has the capability to simulate P losses through drainage. Specifically, the model calculates losses of sediment, PP, and DP through surface, matrix, and macropore transport (Radcliffe et al., 2015). The model is designed for the field scale, but model results have been aggregated to simulate at the small watershed scale by using typical soil-crop-slope combinations (Rekolainen et al., 2002). ICECREAMDB runs at a daily time step with daily time series data as input. The model was not designed to simulate single storm events, so a minimum length of one year is required for input time series data (Barlund et al., 2008). ICECREAMDB is a mixed model and includes empirical submodels, such as USLE for erosion and degree days for crop development, and process based submodels such as P sorption/desorption
(Radcliffe et al., 2015). The SOIL model has been coupled to the ICECREAMDB model and outputs from SOIL are used as inputs for the P calculations in ICECREAMDB.

2.7.2. **SOIL**

The SOIL model calculates one dimensional water and heat dynamics in the soil profile (Jansson, 1994). The basis of the SOIL model is a soil profile divided into a finite number of layers. Water dynamics are calculated using the Richard’s equation and heat dynamics are calculated using the heat conduction equation (Radcliffe et al., 2015). Pools are included in the SOIL model for snow, intercepted water, and surface ponding to simulate processes at the upper soil boundary (Jansson, 1994). The surface ponding pool is created if rainfall intensity exceeds the soil infiltration capacity. Water is lost from the soil profile through either deep percolation or drainage flow; drainage losses are subtracted from the soil layer that contains the water table.

2.7.3. **SoilIceDB**

The SoilIceDB model is a system which runs the ICECREAMDB model on the SOIL model output. In ICECREAMDB, the water in the SOIL surface ponding pool is divided between surface runoff and macropore flow with the fraction of the pool allocated to surface runoff set as the fraction of the soil surface area in which soil pores are sealed due to raindrop impact, compaction by machinery, or by frozen water in winter (Radcliffe et al., 2015). ICECREAMDB simulates preferential flow and transport in macropores through a short-circuit pathway corresponding to the macropore domain; another flow and transport pathway exists for the micropore domain. Water, suspended particles, and P entering macropores are transported directly to drainage flow without
interaction with the micropore domain. ICECREAMDB includes a specific sediment pool at the soil surface; sediment losses in macropore transport come from this pool (Larsson et al., 2007). The maximum size of the pool is dependent upon the soil clay content and sediment is added to the pool through soil tillage, freezing, and thawing (Radcliffe et al., 2015). In order to simulate drainage outflow, ICECREAMDB includes a groundwater reservoir composed of two compartments: one receiving water and P from the macropore domain and one receiving water and P from the micropore domain. Drainage flow commences when the storage capacity of the compartments is exceeded (Larsson et al., 2007).

Like ADAPT and APEX, ICECREAMDB’s P routines are based on the EPIC model. Outputs from SOIL for surface runoff, matrix flow, macropore flow, drainage flow, evapotranspiration, snow depth, and soil temperature are used as inputs for the P processes within ICECREAMDB (Radcliffe et al., 2015). In ICECREAMDB, soil P is divided into 3 inorganic pools (stable P, active P, labile P) and 3 organic pools (litter, humus, manure). Partitioning of P between the 3 inorganic pools is a function of pH, percent base saturation, and clay content. All solid P pools are assumed to contribute to P bound to suspended particles (Larsson et al., 2007, Radcliffe et al., 2015). Convective mass transfer is used to calculate leaching from the soil matrix and transport of DP between soil layers. The P content of eroded sediments is calculated using an enrichment ratio based on the total soil P content. Then, the P content of sediment lost through macropore transport is set as the fraction of TP in the soil (Radcliffe et al., 2015).
2.7.4. Previous Applications of ICECREAMDB

Applications of the ICECREAMDB model to artificially drained fields have been limited to Sweden (Blombäck and Persson, 2009, Larsson et al., 2007, Liu et al., 2012). Previously, ICECREAM has been used to model surface runoff in Finland (Rekolainen and Posch, 1993, Rekolainen et al., 2002, Tattari et al., 2001). The ability to model drainage was added to ICECREAMDB by Larsson et al. (2007) and simulation results were compared to measured flow and drainage losses of SP, PP, and DRP. They found that the model reasonably estimated the episodic losses but that some short-term fluctuations were not captured. Blombäck and Persson (2009) evaluated the generality of the soil-related parameterization and assessed the location dependence of the model. They found that the standard parameterizations used in ICECREAMDB captured the beginnings and endings of run-off events but that the size of the peak flows and the total run-off volume were underestimated. In contrast, the standard parameterizations overestimated TP and DP. When site-specific parameterization was used, runoff was further underestimated while the overestimation of TP decreased. Overall, they concluded that the site-specific parameterization did not improve the simulation results over the standard parameterizations. Liu et al. (2012) used ICECREAMDB to identify P leaching risks in drained fields receiving long-term fertilization regimes. They found that the model overestimated TP leaching by a factor of 5-9 over measured data and concluded that the model must be further developed to include P sorption and desorption processes. Radcliffe et al. (2015) reviewed the model performance and concluded that overall, ICECREAMDB performs satisfactorily in estimating DRP losses in
soils with normal sorption capacity and in identifying the timing of macropore transport; however, modeling of DRP is difficult in soils with very high/low sorption capacity and the estimation of the magnitude of water flows and PP losses can be improved. In addition, ICECREAMDB’s new approach for partitioning between surface runoff and macropore must be evaluated against measured data and the model must be tested on its ability to estimate the transport of P during spring snow melts (Radcliffe et al., 2015).
CHAPTER 3: EVALUATION OF SUBSURFACE DRAINAGE AND
BEST MANAGEMENT PRACTICE IMPLEMENTATION ON
PHOSPHORUS LOSSES IN THE BLACK HAWK LAKE
WATERSHED, IOWA

3.1. Introduction

Hypoxia in the Gulf of Mexico is directly related to nutrient losses from
agriculture in the Upper Midwest. From 1985-2014, the size of the hypoxic zone in the
northern Gulf of Mexico averaged 13,650 square kilometers (LUMCON, 2017). Model
simulations have indicated that agriculture is responsible for over 70% of the nitrogen
and phosphorus delivered to the Gulf of Mexico (Alexander et al., 2007). Iowa is the
second-highest contributor of nitrogen (11.3% of the total flux) and the third-highest
contributor of phosphorus (9.8% of the total flux) to the Gulf of Mexico (Alexander et
al., 2007). The effects of these nutrient losses are visible not only in the Gulf, but also in
Iowa’s surface waters; of the 225 impairments listed for the 118 lake/reservoir
waterbodies on the 2014 Iowa 303(d) list, 64 are due to algae and 53 are due to
turbidity, both of which can be indicators of excess nutrients (IDNR, 2015). Historically,
productivity in coastal waters is limited by nitrogen (N) while phosphorus (P) is the
priority nutrient limiting upstream freshwater productivity. However, it is difficult to
control eutrophication by limiting a single nutrient because changes in anthropogenic
activities have resulted in an imbalance in N and P loading to waters (Paerl, 2009).

While past research on agricultural nutrient losses has typically focused on N,
current research is concentrated on P losses through drainage. Once a region covered in
swamps and wetlands, the use of drainage has transformed the Upper Midwest into some of the world’s most productive agricultural land (Du et al., 2005). Artificial drainage is necessary for crop production on approximately 25% of the croplands in the U.S. and Canada with some states and provinces requiring drainage on over half of their croplands (Skaggs et al., 1994). Subsurface drainage allows for trafficable conditions for timely field operations in seasonally and perennially wet locations, prevents excessive soil water conditions, provides salinity control in irrigated areas, and increases nutrient uptake of crops by creating a well-aerated root environment (Du et al., 2005, Fausey et al., 1987, Reeve and Fausey, 1974, Vos, 1987, Zucker and Brown, 1998). However, drainage P losses can be quite significant; Studies have found that artificial drainage contributes P loads ranging from 17% to greater than 50% of the total P losses (Culley et al., 1983, Jamieson et al., 2003, Ruark et al., 2012, Smith et al., 2015, Tomer et al., 2010).

Phosphorus enters drainage waters through two primary pathways: leaching through the soil matrix and preferential flow through macropores and surface intakes to tile lines. Soils with high P concentrations and low P sorption capacity have a high potential for P to leach from the soil matrix into runoff and groundwater (Kleinman et al., 2007, Vadas et al., 2007). However, as soil moisture content decreases, the relative contribution of macropores to chemical transport and water movement increases (Shipitalo and Edwards, 1996). Flow through soil macropores occurs when surface soil layers are saturated during non-ponded conditions (Andreini and Steenhuis, 1990, Shipitalo and Edwards, 1996). Geohring et al. (2001) studied the timing of drainage P
transport relative to drainage flow, and concluded that macropore flow is the primary transport pathway of TP through soil. When ponding occurs on the soil surface, P may also enter drainage through surface intakes. This pathway has been found to deliver at least 75% of the drainage Total P (TP) load (Tomer et al., 2010).

Although previous studies have quantified P losses, the impacts of best management practices (BMPs) on P losses are equivocal. Phosphorus BMPs include animal waste systems, barnyard runoff management, conservation tillage, contour strip cropping, crop rotation, filter strips, nutrient management plans, and riparian forest buffers. In general, TP and particulate phosphorus (PP) losses are well controlled by these BMPs; average loss reductions for each BMP range from 30-62% for TP and 33-84% for PP. However, up to 22% increases in TP losses have been measured with conservation tillage implemented. The effects of BMPs on dissolved phosphorus (DP) losses are more varied. Average DP loss reductions range from 26-62% for the barnyard runoff management, contour strip cropping, crop rotation, filter strips, nutrient management plans, and riparian forest buffers BMPs. However, animal waste systems have an average increase in DP losses of 13% and conservation tillage has an average increase in DP losses of 167%. Increases in DP losses have also been measured for filter strips and nutrient management plans (Gitau et al., 2005). The increases in DP losses due to animal waste systems and nutrient management plans are likely due to the implementation of nitrogen-based nutrient management plans which allow for manure application based on crop N requirements; typically, manure N:P ratios are less than those needed by crops so over-application of P occurs (Brannan et al., 2000). These
findings highlight the need for more data regarding the effectiveness of BMPs on phosphorus losses. Furthermore, the impact of BMPs on drainage P losses must be evaluated.

The objective of this study is to evaluate the effects of BMP implementation on P export in drainage waters. Specifically, this study expands upon the work of previous studies by comparing the P concentrations and loads in drainage and surface flow and the intra-event P losses in watersheds with different levels of BMP implementation during a variety of flow conditions. An increased understanding of the effects of BMP implementation on P losses during different flow conditions will help with water quality improvement projects and could identify areas for remediation and BMP implementation.

3.2. Materials & Methods

The Black Hawk Lake (BHL) Watershed is located in Carroll and Sac Counties, Iowa, along the western edge of the Des Moines lobe. This landform region is characterized by a gently rolling landscape with abundant moraines and prairie potholes. Many of these potholes have been drained with underground tile lines to facilitate agriculture; the watershed has a drainage area of 5,324 hectares (excluding the lake) and landuse distribution of 52.7% corn, 21.9% soybeans, 6.7% grass/hay/pasture, 5.8% water/wetland (excluding the lake), 1.9% timber, and 11% other. A unique feature of the watershed is that it drains from south to north. The lake’s designated use is primary contact recreation and it serves as an important recreational resource in the region. Black Hawk Lake is Iowa’s southern-most glacial lake and has a surface area of
373 hectares with 18.3 kilometers of shoreline. As a result of its glacial origins, the lake is quite shallow with a maximum depth of 4.6 meters and an average depth of 1.8 meters. In 2004, the lake was added to the 303(d) Impaired Waters Listing for algae and turbidity and in 2008 it was also added to the 303(d) list for indicator bacteria. A Total Maximum Daily Load (TMDL) for the lake’s algae and pH impairments was approved by the U.S. EPA in 2012 and the lake was removed from the 303(d) list.

3.2.1. Monitoring Sites

Monitoring occurs in three of the lake’s 15 subwatersheds: subwatersheds 8, 11, and 12. The Iowa Department of Natural Resources (IDNR) has 16 additional monitoring sites located throughout the watershed. These sites are shown in Figure 1. Surface flow samples are collected at subwatersheds 8, 11, and 12 and drainage flow samples are collected at subwatersheds 8 and 12.

Soil types are similar among the three subwatersheds. The dominant soil type is Clarion loam, which covers 45.0-47.9% of the individual subwatershed areas. Clarion soils have typical clay and sand contents of 18-28% and 30-50%, respectively. Nicollet loam is also prevalent in the subwatersheds and accounts for 17.0-21.4% of the subwatershed areas. The A horizon of Nicollet soils have typical clay and sand contents of 24-35% and 20-35%, respectively. However, the B horizon is dominated by clay. Webster clay loam is also common in each of the subwatersheds and accounts for 7.7-17.6% of their respective areas. Typical clay and sand contents of Webster soils are 29-39% and 16-33%, respectively. Finally, Coland clay loam accounts for 17.5% of the area of subwatershed 8. Coland soils have typical clay and sand contents of 22-35% and 15-
40%, respectively. Although all are formed in glacial till, Clarion soils are moderately well
drained, Nicollet soils are somewhat poorly drained, and Webster soils are poorly
drained. Coland soils are formed in alluvium and are poorly drained (USDA, 2017).

Figure 1: ISU and IDNR Monitoring Locations in Black Hawk Lake subwatersheds

Subwatershed 8 is the largest of the three monitored subwatersheds and has an
area of 822.49 hectares. The subwatershed is extensively tile drained and a 91cm (36-
inch) diameter drainage district tile discharges just upstream of the subwatershed
outlet. Samples are collected from the tile outlet as well as from a grassed waterway which discharges in the same location. The subwatershed has few BMPs installed; BMP implementation only covers 22.5% of the subwatershed area. However, there are a few terraces and grassed waterways installed and several fields have nutrient management plans implemented.

Subwatershed 11 has an area of 229.44 hectares. Surface flow samples are collected from the middle of a concrete culvert. Flow through the culvert occurs nearly year-round, which indicates a possible upstream drainage source. Like subwatershed 8, this subwatershed has few implemented BMPs; BMPs only cover 30.0% of the subwatershed area. However, some fields in its southwestern corner have a nutrient management plan implemented in conjunction with no till or terracing. A CREP wetland was installed downstream of the culvert monitoring location between the summer of 2013 and the summer of 2014.

With an area of 221.23 hectares, subwatershed 12 is similar in size to subwatershed 11; however, BMP implementation occurs on 87.5% of the subwatershed. Installed BMPs include terraces, CRP filters and wetlands, and nutrient management plans. Like subwatershed 8, this subwatershed contains a segment of drainage district tile. Sample collection occurs on both sides of a concrete culvert; surface flow samples are collected upstream of the culvert and drainage flow samples are collected from a 40cm (15.5-inch) diameter tile located on the downstream side of the culvert. Upon exiting the culvert, the surface flow must vertically drop several feet before continuing; this effectively prevents the surface and drainage flow from mixing.
3.2.2. Site Analysis

Water yields and drainage ratios were calculated for each of the three BHL subwatersheds. Water yields for each year of the study were calculated by dividing the total flow out of the subwatershed by the subwatershed area. Drainage ratios were then calculated by dividing the subwatershed water yield by the total subwatershed precipitation depth.

The percentage of each subwatershed that would benefit from subsurface drainage was estimated using a Geographic Information System (GIS). The 2012 Iowa Cropland Data coverage developed by the United States Department of Agriculture (USDA) was used to determine the area of each subwatershed with row crop land cover (USDA, 2012). Then, the Soils of Iowa coverage from the IDNR was used to determine the area of each subwatershed with soils with somewhat poorly drained, poorly drained, and very poorly drained soils (IDNR, 2003). Using the two coverages, all areas with row crop land cover and soils and soils with poor drainage were assumed to benefit from drainage. Estimated values were very similar in each of the subwatersheds; Subwatershed 8 had the greatest estimated subwatershed area benefitting from drainage, 45.3% of the subwatershed area, followed by subwatershed 11 (44.3%) and subwatershed 12 (39.7%). Overall, the entire subwatershed areas were assumed to contribute to drainage flow because subsurface flow can occur between areas benefitting from drainage and areas not benefitting from drainage.
3.2.3. Sample Collection

Weekly flow-weighted (WFW) and event flow-weighted (EFW) composite samples were collected using ISCO 6600-Series automated samplers from March-November of 2015 and 2016. In addition, flow, velocity, and level were measured using ISCO 750 Area Velocity Flow Modules or ISCO 720 Pressure Transducer Modules. Level measurements using the pressure transducer modules were verified with Solinst Levelloggers. ISCO 674 Rain Gauges are installed in each subwatershed to collect precipitation data every 5 minutes. Daily precipitation data was also obtained using the PRISM Data Explorer (PRISM, 2017). Grab samples were collected on each site visit during the March-November sampling season; in 2015 site visits occurred weekly whereas visits in 2016 occurred semimonthly. In 2016, intra-event samples were collected every four hours during precipitation events occurring on April 19th and April 27th and every three hours during events occurring on April 30th and June 14th.

3.2.4. Sample Analysis

The weekly and event-flow weighted samples were analyzed for Dissolved Reactive Phosphorus (DRP), Total Phosphorus (TP), Total Suspended Solids (TSS) and Volatile Suspended Solids (VSS). In order to test for DRP, a 20mL representative sample was first passed through a 0.45-micron filter. Then, the samples were analyzed for DRP on a Seal Analytical AQ2 using Method EPA-118-A Rev. 5. When testing for TP, organic phosphorus in the water samples was converted to orthophosphate by persulfate digestion. Following digestions, the samples were analyzed for TP on the AQ2 using Method EPA-119-A Rev. 6. To ensure quality control, a spiked sample was created and
analyzed for every tenth sample; the TP concentration of the spiked sample should equal the TP concentration of the sample plus the TP concentration of the standard. Samples were analyzed for TSS and VSS using Method 2540-D and 2540-E, respectively, from Standard Methods for the Examination of Water and Wastewater, 22nd edition (Rice et al., 2012).

The intra-event samples from the four 2016 events were analyzed for TP, TSS, and Total Dissolved Phosphorus (TDP). As before, samples were analyzed for TP and TSS using Method EPA-119-A Rev. 6 and Method 2540-D, respectively. In order to test for TDP, a 20mL representative sample was first passed through a 0.45-micron filter. Then, the samples were analyzed for TDP on the AQ2 using Method EPA-119-A Rev. 6. Total Particulate Phosphorus (TPP) was calculated as the difference between the TP and TDP concentrations.

Linear regression analyses were performed to identify any correlations between flow and flow-weighted TP, DRP, TSS, and VSS concentrations at sites T8, S11, S12, and T12. Analyses were not performed for site S8 because flow only occurred during events. Outputs from the linear regression analyses included coefficients of determination ($r^2$) and slope.

Linear regression analyses were also performed between flow and intra-event TP, TDP, TPP, and TSS concentrations for all five BHL monitoring sites. Furthermore, an analysis of variance (ANOVA) was performed to identify any significant differences between the peak intra-event analyte concentrations at the five BHL monitoring sites or
at the three surface sites (S8, S11, & S12) versus the two tile sites (T8 & T12). Differences were considered significant for \( p \)-values less than or equal to 0.05.

3.3. Results

3.3.1. Background/Hydrology

The BHL watershed experienced greater precipitation in 2015 than in 2016. Precipitation recorded at the three sites ranged from 108.5-112.5 cm during the 2015 monitoring period and 78.6-80.4 cm during the 2016 monitoring period. In 2015, there were 15 recorded precipitation events which had an average depth of 3.4 cm and an average intensity of 0.54 cm/h. Similarly, in 2016 there were 16 recorded events which had an average of depth of 2.9 cm and an average intensity of 0.52 cm/h. Although the average precipitation depths and intensities are similar for the two years, there were more, large storm events in 2015; 26.7% of events in 2015 had a precipitation depth of greater than 5 cm as compared to only 12.5% in 2016. In addition, the timing of the events differed across the two years. In 2015, 66.7% of events occurred after August 15th versus 18.8% for 2016. Average water yields for the three BHL subwatersheds were 20.9 cm for subwatershed 8, 12.8 cm for subwatershed 11, and 26.6 cm for subwatershed 12. The corresponding ratio of water yield to precipitation, referred to as drainage ratio, was calculated for each of the subwatersheds. Average drainage ratios for the three BHL subwatersheds were 21.5% for subwatershed 8, 13.6% for subwatershed 11, and 29.4% for subwatershed 12.
3.3.2. Flow Exceedance Curves

Flow exceedance curves were created for each of the three subwatersheds and are presented in Figure 2. Daily average flow for subwatersheds 8 and 12 were calculated as the sum of the daily average flows at their respective surface and tile components. Since the subwatersheds are different sizes, the daily average flow rate was normalized by watershed area (Figure 3). Daily average unit-area flow was highest in subwatershed 12 followed by subwatersheds 8 and 11. Subwatershed 12 had the most high-flow conditions with 5.0% of days exceeding an average flow of 0.486 m$^3$/s. In comparison, discharges corresponding to a 5.0% exceedance probability were 0.215 m$^3$/s at subwatershed 8 and 0.068 m$^3$/s at subwatershed 11. The most low-flow conditions were observed in Subwatershed 11 with no flow observed on 10.8% of days. In contrast, no flow was only observed on 4.4% of days in subwatershed 12 and 3.7% of days in subwatershed 8.
Unit-area flow exceedance curves were also created for the daily average flow per hectare measured at the two tile sites (T8 & T12) and two stream sites (S11 & S12); exceedance curves were not created for the grassed waterway (S8) because flow only occurs during large events. At the tile and stream sites, the daily average unit-area flow calculated by dividing the daily average flow by the subwatershed area. The unit-area flow exceedance curves are presented in Figures 4A-D and were overlaid with the TP concentration, TSS concentration, and the corresponding flow rates from each of the WFW and EFW samples. In addition, the EPA recommended TP limit of 0.05 mg P/L for streams discharging into lakes was overlaid on Figures 4 A & B (Mueller et al., 1996). Unit-area flow exceedance curves overlaid with the DRP and VSS concentrations are included as Supplementary Figures 1 A-D.
Figure 3: Unit-area flow exceedance curves for March to November in Subwatersheds 8, 11, and 12.

From Figure 4B, it is apparent that both unit-area flow and TSS concentrations are similar at T12 and T8. However, Figure 4A shows that TP concentrations are lower at T12 than at T8. Overall, 59.7% of T8 samples exceeded the EPA recommended TP limit of 0.05 mg P/L whereas only 9.8% of T12 samples exceeded the limit. Figures 4C and D show that site S12 experiences more high flow conditions than S11, but that overall, both sites have similar unit-area flow and TP and TSS concentrations. At S11, 48.3% and 63.0% of S11 and S12 samples, respectively, exceeded TP concentrations of 0.05 mg P/L. Overall, no significant concentration response to flow trends ($r^2=0.00-0.18$) was observed for TP, DRP, TSS, or VSS at any of the four sites. However, 68.8% of regression lines between analyte concentration and flow had a positive slope.
Figure 4: Flow per area exceedance curves for March to November for the tile monitoring sites with (A) total phosphorus concentrations and (B) total suspended solids concentrations; flow exceedance curves for March to November for the stream monitoring sites with (C) total phosphorus concentrations and (D) total suspended solids concentrations.
3.3.3. **Weekly Flow-Weighted Concentrations**

TP, DRP, TSS, and VSS concentrations were measured for the WFW composite samples from sites T8, S11, S12, and T12. The maximum and minimum concentration data for each site and analyte in 2015 and 2016 is summarized in Table 1. Non-detections are represented in the table as ND. Figure 5 presents the ranking of the 2-year median WFW concentrations for each site and analyte. The four sites are ranked from one to four with one representing the site with the lowest median concentration and four representing the site with the highest median concentration. The data labels indicate the value of the 2-year median concentration.

<table>
<thead>
<tr>
<th>Site</th>
<th>Max (mg P/L)</th>
<th>Min (mg P/L)</th>
<th>Max (mg P/L)</th>
<th>Min (mg P/L)</th>
<th>Max (mg P/L)</th>
<th>Min (mg P/L)</th>
<th>Max (mg P/L)</th>
<th>Min (mg P/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>T8</td>
<td>0.491</td>
<td>0.002</td>
<td>0.428</td>
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<td>0.293</td>
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Overall, the median WFW TP concentration was 0.042 mg P/L across the four sites during the 2-year study period. In 2015, the greatest TP concentrations (exceedance probabilities less than 10%) were observed from May through September whereas concentrations in 2016 were greatest from April through October. TP concentrations at the stream sites (S11 & S12) and the tile sites (T8 & T12) ranged from 0.002-1.419 mg P/L and non-detect to 0.491 mg P/L, respectively. Overall, 42.0% of WFW samples exceeded the EPA recommended limit of 0.05 mg P/L for streams discharging into lakes. Site T12 had the lowest median WFW TP concentration for the two years (0.008 mg P/L) followed by S11 (0.035 mg P/L), T8 (0.045 mg P/L), and S12 (0.059 mg P/L). In both 2015 and 2016, the TP concentration of the WFW samples from site T12 never exceeded 0.05 mg P/L whereas the value was exceeded by 43% of the samples at S11, 49% at T8, and 61% at S12 during the same time period.
In 2015 and 2016, the median WFW DRP concentration was 0.002 mg P/L across the four sites. WFW DRP concentrations in 2015 displayed less seasonality than the TP concentrations during the same time period, but in 2016 DRP concentrations were greatest (exceedance probabilities less than 10%) from May through October. During the study period, DRP concentrations at the stream sites ranged from non-detect to 0.475 mg P/L and tile DRP concentrations ranged from non-detect to 0.191 mg P/L. Sites S12 and T12 had the lowest median WFW DRP concentration for the two years (0.001 mg P/L) followed by S11 (0.003 mg P/L) and T8 (0.012 mg P/L).

Over the two-year sample period, WFW TSS concentrations at the stream sites ranged from 0.5-852.0 mg/L. At the tile sites, TSS concentrations ranged from non-detect to 119.0 mg/L. Overall, the median WFW TSS concentration was 8.0 mg/L for the four sites. No seasonal trends in TSS concentrations were observed. The tile outlet monitoring locations (T8 & T12) had lower 2-year median WFW TSS concentrations than the stream sites (S11 & S12); median concentrations were 3.8 mg/L at T12, 6.3 mg/L at T8, 12.0 mg/L at S12, and 19.0 mg/L at S11.

Like the WFW TSS concentrations, no seasonal trends were observed for WFW VSS concentrations during the sample period (Fig. 2). Overall, the median WFW VSS concentration was 5.1 mg/L across the five sites and both years. VSS concentrations at the stream sites ranged from non-detect to 736.0 mg/L and tile VSS concentrations ranged from non-detect to 91 mg/L. Following the same ranking as the TSS concentrations, the lowest 2-year median WFW VSS concentration was observed at T12 (3.0 mg/L) followed by T8 (4.7 mg/L), S12 (5.0 mg/L), and S11 (14.7 mg/L).
3.3.4. Event Flow Weighted Concentrations

TP, DRP, TSS, and VSS concentrations were also measured for the EFW samples from sites S8, T8, S11, S12, and T12. Table 2 presents the maximum and minimum concentration data for each analyte and site for 2015 and 2016. Non-detects are represented in the table as ND. The ranking of the 2-year median EFW concentrations for each analyte and site is presented in Figure 6. The five sites are ranked from one to five with one representing the site with the lowest median concentration and five representing the site with the highest median concentration. The data labels indicate the value of the 2-year median concentration.

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<td>Min</td>
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<td>296</td>
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Over the two-year study period, the EFW TP concentration ranged from 0.002-3.118 mg P/L at the stream sites (S11 & S12), from 0.002-4.709 mg P/L at the tile sites (T8 & T12), and from 1.277-1.865 mg P/L at the grassed waterway (S8). Overall, the median EFW TP concentration across the five sites during the two-year period was 0.091 mg P/L and 71.3% of the EFW samples exceeded the EPA recommended limit of 0.05 mg P/L for streams discharging into lakes. The lowest two-year median EFW TP concentration was observed at T12 (0.034 mg P/L) followed by S12 (0.055 mg P/L), T8 (0.138 mg P/L), S11 (0.164 mg P/L), and S8 (1.425 mg P/L). In a slightly different order, T12 had the fewest EFW samples exceeding 0.05 mg P/L (32%) followed by S12 (70%), S11 (79%), T8 (89%) and S8 (100%).

During the study period, the median EFW DRP concentration across the five sites was 0.026 mg P/L. EFW DRP concentrations ranged from non-detect to 0.316 mg P/L at the stream sites, from non-detect to 1.177 mg P/L at the tile sites, and from 0.727-1.549 mg P/L at the grassed waterway. WFW DRP concentrations were greatest for all five
sites in 2015 with maximum concentrations ranging from 0.067-1.549 mg P/L. Site T12 also had the lowest two-year median EFW DRP concentration (0.005 mg P/L) followed by S12 (0.006 mg P/L), S11 (0.028 mg P/L), T8 (0.048 mg P/L), and S8 (1.038 mg P/L).

EFW TSS concentrations ranged from 1.3-2026.0 mg/L at the stream sites, non-detect to 496.0 mg/L at the tile sites, and 24.0-296.0 mg/L at the grassed waterway. The overall EFW TSS median concentration for the five sites in 2015 and 2016 was 18.0 mg/L. The tile outlet monitoring locations (T8 & T12) had lower 2-year median EFW TSS concentrations than those of the stream monitoring locations (S11 & S12) and the grassed waterway (S8); the median concentration was 6.3 mg/L at T12, 11.5 mg/L at T8, 20.2 mg/L at S12, 34.8 mg/L at S11, and 69.0 mg/L at S8.

Overall, the median EFW VSS concentration for the five sites during the study period was 16.0 mg/L. During this time, EFW VSS concentrations ranged from non-detect to 1812.0 mg/L at the stream sites, from non-detect to 350.0 mg/L at the tile sites, and 16.0-228.0 mg/L at the grassed waterway. The lowest 2-year median EFW VSS concentrations were observed at the tile outlet monitoring locations, T12 (5.3 mg/L) and T8 (8.7 mg/L). Median EFW VSS concentrations were higher at S12 (16.0 mg/L), S11 (37.0 mg/L), and S8 (56.0 mg/L).
3.3.5. Cumulative Loads

Cumulative precipitation depth and cumulative TP, DRP, TSS, and VSS loads were calculated for each of the five sites in 2015 and 2016 and are displayed in Figures 7 and 8.

TP and DRP export in 2015 from subwatershed 8 (S8 & T8) followed the trends in cumulative precipitation with 20.9% of the TP export and 17.8% of the DRP occurring during August. In contrast, at subwatershed 11, 63.9% of the total TP export and 82.2% of the total DRP export in 2015 occurred between 6/22/15 and 6/30/15. Likewise, at subwatershed 12, 68.7% of the cumulative TP export in 2015 occurred between 4/25/15 and 5/1/15 with 99.8% of the subwatershed export during this period occurring at site S12. Overall, this time period accounted for 62.1% of the two-year cumulative TP export from S12. However, the one week period only accounted for 12.7% of the total 2015 DRP export from subwatershed 12. The temporal patterns of TP and DRP export in 2016 were more similar among the sites than in 2015; export occurred primarily during events in May and October. In subwatershed 11, 46.6% and 86.1% of the total 2016 subwatershed TP and DRP export, respectively, occurred from 4/28/16-5/3/16. However, substantial export also occurred during events in August and November for subwatershed 12. In both years of the study period, DRP export from the stream sites (S11 & S12) was minimal compared to the corresponding TP export during events occurring in late summer and fall.

TP exported from each of the subwatersheds was greater in 2015 than in 2016. During the two-year study period, subwatershed 8 (Sites S8 & T8) had the largest TP
load (800.1 kg) of the three subwatersheds with 80.4% of the total export occurring in 2015. Subwatershed 11 had a similar TP export (768.4 kg) as subwatershed 8 with 89.5% of the total load occurring in 2015. With a two-year TP load of 447.6 kg, subwatershed 12 (Sites S12 & T12) had the smallest TP export among the three subwatersheds. Like the other subwatersheds, however, 86.2% of the TP exported from subwatershed 12 occurred in 2015. The percent of the TP exported through the tile outlets in subwatersheds 8 and 12 during the two-year period were very different; T8 contributed 69.8% of the TP load in subwatershed 8 whereas T12 contributed only 4.8% of the TP load in subwatershed 12.

DRP exports were also greater in 2015 than in 2016 for each of the subwatersheds. Subwatershed 8 had the greatest two-year cumulative DRP load (374.6 kg) of the subwatersheds. Although subwatersheds 8 and 11 had similar two-year cumulative TP exports, subwatershed 11 had a cumulative export of 98.3 kg, an order of magnitude lower than that of subwatershed 8. Subwatershed 12 had the lowest cumulative DRP load (79.1 kg) of the three subwatersheds. Despite the differences in cumulative loads, the percent of the two-year cumulative DRP load occurring during 2015 was similar for each of the watersheds; 88.1% of the total DRP export occurred in 2015 for subwatershed 8, 87.4% for subwatershed 11, and 89.6% for subwatershed 12. Overall, 59.2% of the two-year cumulative DRP load in subwatershed 8 was delivered through the tile and 13.3% of the DRP load in subwatershed 12 was delivered through the tile. During the study period, the percent of the total TP load occurring as DRP was
higher for the tile outlet locations (28.3-70.0%) and grassed waterway (53.5-68.3%) than the stream locations (10.5-16.9%).

During the study period, the export of TSS and VSS from subwatersheds 8 and 12 closely followed the pattern of cumulative precipitation. At subwatershed 11, however the majority of the TSS and VSS export occurred during short periods of time with minimal export observed in the following months. In 2015, 61.8% of the total TSS load and 61.6% of the total VSS load was exported between 6/22/15 and 6/30/15. Similarly, in 2016, 77.6% of the total TSS load and 87.7% of the total VSS load was exported between 4/26/16 and 5/1/16. Furthermore, minimal TSS and VSS export were observed at S11 in July of 2015 and June-August of 2016 despite the occurrence of precipitation events.

Cumulative TSS loads varied greatly among the three subwatersheds. Subwatershed 11 had a cumulative TSS load of 792,177 kg during the study period with 94.7% of the load occurring in 2015. The TSS loads from subwatersheds 8 and 12 were an order of magnitude lower than that of subwatershed 11; subwatershed 8 had a total load of 75,750 kg with 74.6% occurring in 2015 and subwatershed 12 had a total load of 32,670 kg with 70.7% occurring in 2015. During the two-year period, T8 contributed 82.6% of the total TSS export from subwatershed 8 while T12 only contributed 18.0% of the export from subwatershed 12.

Over the two-year study period, a total of 704,129 kg of VSS was exported from subwatershed 11, 55,519 kg from subwatershed 8, and 24,134 kg from subwatershed
12. Cumulative VSS exports followed similar temporal patterns to those of TSS for each of the subwatersheds. The VSS loads in 2015 accounted for 95.3% of the two-year cumulative load for subwatershed 11, 71.2% for subwatershed 8, and 67.5% for subwatershed 12. Furthermore, the percent of the VSS export occurring through the tile outlets was nearly identical to that of the TSS export. From 2015-2016, T8 accounted for 82.6% of the VSS load from subwatershed 8 and T12 accounted for 17.5% of the load from subwatershed 12. The percent of the total TSS load occurring as VSS was highest at S11 (88.9%) but was similar among the other four sites and ranged between 72.0-74.3%.
Figure 7: 2015 cumulative loads of total phosphorus (TP), dissolved reactive phosphorus (DRP), and cumulative precipitation at the monitoring locations in (A) subwatershed 8, (B) subwatershed 11, and (C) subwatershed 12; 2015 cumulative loads of total suspended solids (TSS), volatile suspended solids (VSS), and cumulative precipitation at the monitoring locations in (D) subwatershed 8, (E) subwatershed 11, and (F) subwatershed 12.
Figure 8: 2016 cumulative loads of total phosphorus (TP), dissolved reactive phosphorus (DRP), and cumulative precipitation at the monitoring locations in (A) subwatershed 8, (B) subwatershed 11, and (C) subwatershed 12; 2016 cumulative loads of total suspended solids (TSS), volatile suspended solids (VSS), and cumulative precipitation at the monitoring locations in (D) subwatershed 8, (E) subwatershed 11, and (F) subwatershed 12.
3.3.6. Unit-Area Loads

The cumulative loads of TP and DRP for each of the five monitoring locations were normalized by subwatershed area to calculate unit-area loads. In addition, the cumulative subwatershed TP and DRP loads were divided by the subwatershed area to produce their respective subwatershed unit-area loads. Furthermore, the unit-area loads were subdivided by their WFW and EFW components; the ratio of the EFW component to the total unit-area load is referred to as the event contribution. Likewise, the ratio of the unit-area load of the tile sites (T8 & T12) to the total unit-area load of their corresponding subwatersheds is termed tile contribution. The TP and DRP unit area loads are displayed in Figure 9.

In both 2015 and 2016, subwatershed 11 had the greatest TP unit-area load (0.353-2.997 kg P/ha) of the three subwatersheds and subwatershed 8 had the smallest load (0.191-0.782 kg P/ha). However, subwatershed 8 had the greatest DRP unit-area load (0.054-0.401 kg P/ha) for each year. Overall, events accounted for the majority of the TP unit-area loads in each of the subwatersheds. In 2015, TP unit-area load event contributions were 79.1%, 92.7%, and 83.4% for subwatersheds 8, 11, and 12, respectively. Event contributions to the TP unit-area loads in 2016 were less those in 2015 at each of the subwatersheds; in 2016, event contributions for the TP unit-area loads in subwatersheds 8, 11, and 12 were 67.5%, 68.3%, and 46.7%, respectively. Events also accounted for the majority of the DRP unit-area loads at all three subwatersheds with greater event contributions in 2016 than in 2015; in 2015, the event contributions at the three subwatersheds ranged from 55.8-93.8% whereas in
2016 they ranged from 89.9-94.3%. During the study, S11 had the greatest TP and DRP unit-area loads of the five monitoring locations in both 2015 and 2016.

![Figure 9](image1.png)

**Figure 9:** Unit-area loads of total phosphorus (TP) and dissolved reactive phosphorus (DRP) in 2015 (A&B) and 2016 (C&D) by subwatershed and monitoring location.

Unit-area loads were also calculated for TSS and VSS for each of the three subwatersheds and five monitoring locations. The TSS and VSS unit-area loads are displayed in Figure 10. In both 2015 and 2016, subwatershed 11 had TSS and VSS unit-area loads 1-2 orders of magnitude greater than those of subwatersheds 8 and 12. The majority of the TSS and VSS export from each of the subwatersheds occurred in 2015; the percent of the cumulative two-year TSS unit-area load exported in 2015 averaged
80.0% for the three subwatersheds. Similarly, the percent of the cumulative two-year VSS unit-area loads exported in 2015 averaged 78.0% for the three subwatersheds.

Although TSS and VSS export was higher in 2015 than in 2016, event contributions were similar during both years at the three subwatersheds. Across the three subwatersheds, event contributions in 2015 ranged from 60.2-92.5% and 64.2-92.3% for the TSS and VSS unit-area loads, respectively, whereas event contributions in 2016 ranged from 55.7-81.5% and 60.4-91.8% of the TSS and VSS unit-area loads, respectively. For both years of the study, S11 had the greatest TSS and VSS unit-area loads of the five monitoring locations.
3.3.7. Intra-Event Analysis

TP, TDP, TPP, and TSS concentrations were measured from each intra-event sample collected during the April 19th, April 27th, April 30th, and June 14th events in 2016. Time series of the analyte concentrations at each monitoring location were produced for each of the four events and similar trends were observed. Time series for the April 30th event are provided because it was the only event in which an event response was observed at each of the five monitoring locations. The flow, TP, TDP, and TPP time series for the event are depicted in Figure 11 and the TSS and TPP time series are shown in Figure 10: Unit-area loads of total suspended solids (TSS) and volatile suspended solids (VSS) in 2015 (A&B) and 2016 (C&D) by subwatershed and monitoring location.
Figure 12; the time series for the April 19th, April 27th, and June 14th events are provided as Supplemental Figures 2-7.

During the four events, the TP, TDP, and TPP concentrations generally followed the trends in flow and peaks in the analyte concentrations corresponded to peaks in the hydrograph. Strong, positive correlations were observed between analyte concentrations and flow at the surface monitoring locations (S8, S11, & S12); adjusted $r^2$ values ranged from 0.71-0.82 for TP, 0.59-0.87 for TDP, and 0.54-0.70 for TPP. Intra-event tile flow remained fairly constant after the initial hydrograph peak. Consequently, correlations between analyte concentrations and flow were weaker for the tile locations (T8 & T12) than at the surface sites; adjusted $r^2$ values ranged from 0.15-0.33 for TP, 0.24-0.28 for TDP, and 0.01-0.28 for TPP. For the April 27th, April 30th, and June 14th events, TPP concentrations exceeded those of TDP during the rising limb of the hydrograph and TDP concentrations were greater than TPP concentrations as the hydrograph returned to baseflow conditions. However, low TPP concentrations were measured during the April 19th event and TDP concentrations were consistently greater than the TPP concentrations. During the four events, peak concentrations observed at the three surface sites ranged from 0.139-3.114 mg P/L for TP, 0.081-0.802 mg P/L for TDP, and 0.029-2.709 mg P/L for TPP. Peak concentrations at the two tile sites ranged from 0.094-1.217 mg P/L for TP, 0.047-0.655 mg P/L for TDP, and 0.046-0.655 mg P/L for TPP. After performing an analysis of variance (ANOVA) between the five monitoring locations, no significant differences were identified between the peak TP ($p=0.2128$) and TPP ($p=0.1432$) concentrations. An ANOVA was also performed between the peak
concentrations at the surface vs tile monitoring locations, but again no significant differences were identified between the peak TP ($p=0.4628$) and TPP ($p=0.3643$) concentrations. ANOVAs performed on the peak TDP concentrations found no significant differences ($p=0.9851$) between the surface vs tile monitoring locations but significant differences ($p=0.0313$) between the five monitoring locations.

Like the other analytes, peaks in TSS concentrations at T8, S11, S12, and T12 corresponded to peaks in flow during the four events. Strong correlations were observed between TSS concentration and flow at the stream sites, S11 (adj. $r^2 = 0.60$) and S12 (adj. $r^2 = 0.46$), but weak correlations were observed at the tile outlets, T8 (adj. $r^2 = -0.05$) and T12 (adj. $r^2 = 0.18$). However, strong correlations between TPP and TSS concentrations occurred at each of the four sites; adjusted $r^2$ values ranged from 0.95-0.99. TSS concentrations in the grassed waterway, S8, remained high at the end of the hydrograph. Consequently, at S8, no correlation was observed between TSS concentration and flow (adj. $r^2 = -0.07$) and a weak correlation was observed between TSS and TPP concentration (adj. $r^2 = 0.35$). Peak TSS concentrations ranged from 23.3-2104.0 mg/L at the three surface sites and from 27.3-336.0 mg/L at the two tile outlets. An ANOVA was performed on the peak TSS concentrations but no significant differences ($p=0.2245$) were identified between the five sites or between the surface vs tile sites ($p=0.3798$).
Figure 11: Flow and intra-event concentrations of total phosphorus (TP), total dissolved phosphorus (TDP), and total particulate phosphorus (TPP) during event on 4/30/16 at the (A) grassed waterway and (B) tile in subwatershed 8, (C) stream in subwatershed 11, and (D) stream and (E) tile in subwatershed 12.
Figure 12: Intra-event concentrations of total suspended solids (TSS) and total particulate phosphorus (TPP) during event on 4/30/16 at the (A) grassed waterway and (B) tile in subwatershed 8, (C) stream in subwatershed 11, and (D) stream and (E) tile in subwatershed 12.
3.4. Discussion

3.4.1. Comparison of Monitoring Sites

During the study, the recorded precipitation was similar at each of the three BHL subwatersheds and ranged from 108.5-112.5 cm during the 2015 monitoring period and from 78.6-80.4 cm during the 2016 monitoring period. On average, the BHL subwatersheds experienced 38.5% more precipitation in 2015 than in 2016. As evidence of this, a greater number of large events were observed in 2015 than in 2016. During the first year of the study, 26.7% of events exceeded a depth of 5 cm compared to only 12.5% of events in the second. Events in 2015 also occurred later in the year than those in 2016. In 2015, 66.7% of events occurred after August 15th versus 18.8% for 2016.

Despite similar precipitation, average water yields and drainage ratios varied between the three subwatersheds. The average water yield was 20.9 cm for subwatershed 8, 12.8 cm for subwatershed 11, and 30.3 cm for subwatershed 12 (Table 3). Overall, average water yields at subwatersheds 8 and 12 were consistent with the value of 24.7 cm/year observed by Ikenberry et al. (2014) and the 26.3 cm/year water yield estimated for the entire Des Moines Lobe (IDALS, 2012). However, flow data is only available for the BHL subwatersheds from March-November for each year of the study. Therefore, the comparatively lower water yields in subwatershed 11 could be reflective of the shorter monitoring period. Average drainage ratios for the three BHL subwatersheds were 21.5% for subwatershed 8, 13.6% for subwatershed 11, and 29.4% for subwatershed 12 (Table 3). Reported drainage ratios in undrained fields range from 15-27% and drainage ratios in drained fields and watersheds range from 31-88%.
(Eastman et al., 2010, Ikenberry et al., 2014, King et al., 2015). However, each of these studies calculated drainage ratios based on complete annual data. Since data in the BHL subwatersheds is only available from March-November, the low drainage ratios in the BHL subwatersheds could also be explained by the differences in monitoring periods.

Although the water yields and drainage ratios varied between the BHL subwatersheds, flow patterns were similar. The flow per area exceedance curves for each of the BHL subwatersheds (Figure 3) had similar shapes, indicating comparable flow responses. Overall, the curves are very flat; at each of the subwatersheds, the difference in the unit-area flow at 20% and 70% exceedance probabilities was 0.0001 m$^3$/s/ha. This translates to a difference in flow of only 0.08, 0.02, and 0.03 m$^3$/s at subwatersheds 8, 11, and 12 respectively and shows that flow is sustained at a generally constant rate throughout the monitoring period. This sustained flow is indicative of a subsurface drainage influence (Schilling and Helmers, 2008). Regarding water yields and drainage ratios, subwatershed 8 typically had the greatest daily average flow followed by subwatersheds 12 and 11. From 0-10% exceedance probabilities, the curves are steep and indicate the high flows during short time periods associated with precipitation events. The highest-flow conditions were observed at subwatershed 12; the daily average discharge corresponding to a 5.0% exceedance probability was 0.503 m$^3$/s at subwatershed 12 compared to 0.215 m$^3$/s at subwatershed 8 and 0.068 m$^3$/s at subwatershed 11. In contrast, subwatershed 11 experienced the most low-flow conditions and no flow was observed on 10.8% of days; no flow was experienced on 4.4% of days in subwatershed 12 and 3.7% of days in subwatershed 8. From the flow per
area exceedance curves in Figures 4A-D, it is apparent that flow observed at the two tile outlets is very similar and flow observed at the stream sites is very similar.

Finally, soils are similar at each of the BHL subwatersheds. The prevailing soil type is Clarion loam which covers an average of 46.7% of the individual subwatershed areas. Nicollet loam and Webster clay loam are also dominant in the subwatersheds and constitute an average of 19.1% and 12.3%, respectively, of the subwatersheds.

A summary of the properties of each of the subwatersheds is presented as Table 3.

### Table 3: Subwatershed Properties

<table>
<thead>
<tr>
<th>Property:</th>
<th>Subwatershed 8</th>
<th>Subwatershed 11</th>
<th>Subwatershed 12</th>
</tr>
</thead>
<tbody>
<tr>
<td>Subwatershed Area (ha)</td>
<td>822.49</td>
<td>229.44</td>
<td>221.23</td>
</tr>
<tr>
<td>BMP Implementation (% of Subwatershed)</td>
<td>22.5</td>
<td>30.0</td>
<td>87.5</td>
</tr>
<tr>
<td>Row Crop Land Cover (% of Subwatershed)</td>
<td>87.0</td>
<td>93.6</td>
<td>73.7</td>
</tr>
<tr>
<td>Average 2015-2016 Precipitation (cm)</td>
<td>95.7</td>
<td>93.5</td>
<td>94.5</td>
</tr>
<tr>
<td>Average 2015-2016 Water Yield (cm)</td>
<td>20.9</td>
<td>12.8</td>
<td>26.6</td>
</tr>
<tr>
<td>Average 2015-2016 Drainage Ratio (%)</td>
<td>21.5</td>
<td>13.6</td>
<td>29.4</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Major Subwatershed Soil Types:</th>
<th>% of Subwatershed</th>
</tr>
</thead>
<tbody>
<tr>
<td>Canisteo Clay Loam</td>
<td>0.1</td>
</tr>
<tr>
<td>Clarion Loam</td>
<td>45.0</td>
</tr>
<tr>
<td>Coland Clay Loam</td>
<td>17.5</td>
</tr>
<tr>
<td>Nicollet Loam</td>
<td>17.0</td>
</tr>
<tr>
<td>Webster Clay Loam</td>
<td>7.7</td>
</tr>
<tr>
<td>Area Benefitting from Drainage</td>
<td>45.3</td>
</tr>
</tbody>
</table>
3.4.2. Hydrologic Patterns

Corresponding to the annual precipitation depths, greater loads were observed in 2015. Averaged between the three subwatersheds, loads in 2015 accounted for 85.4% of the two-year cumulative subwatershed TP export, 88.4% of the DRP export, 80.0% of the TSS export, and 78.0% of the VSS export. Previous studies have also observed greater TP, DRP, and suspended sediment loads during wetter years (Barr, 2016, Eastman et al., 2010, Gentry et al., 2007, Lamb and Toniolo, 2016). For all three subwatersheds, events accounted for the majority of the total TP, DRP, TSS, and VSS unit-area loads for the study period; average event contributions among the three subwatersheds ranged from 76.9-90.2% for TP, 59.3-93.9% for DRP, 58.9-91.9% for TSS, and 64.3-92.3% for VSS. Few other studies have divided loads into their event and baseflow contributions. However, a report by Barr (2016) found that events contributed approximately 29% of the total suspended sediment load measured at two locations on the Big River in Missouri and a study by Gentry et al. (2007) found that approximately 40% of the annual TP load in three east-central Illinois rivers occurred during one 7-day event.

As shown by Gentry et al. (2007), a single event can overwhelm the annual contaminant loads. In the first year of the BHL study, 63.9% of the total TP export, 82.2% of the total DRP export, 61.8% of the total TSS export, and 61.6% of the total VSS export from subwatershed 11 occurred during a nine-day period between June 22\textsuperscript{nd} and June 30\textsuperscript{th}. Precipitation during this event totaled 8.9 cm with an average daily total of 1.0 cm. During the same year, 68.7% of the cumulative TP export from subwatershed 12
occurred during a seven-day period between April 25th and May 1st. This single event accounted for 62.1% of the two-year cumulative TP export from S12, the stream site in subwatershed 12. However, precipitation during this event only totaled 1.1 cm. In the second year of the study, 46.6% of the total TP export, 86.1% of the total DRP export, 77.6% of the total TSS export, and 87.7% of the total VSS export from subwatershed 11 occurred during a 6-day period between April 26th and May 1st. During this period, subwatershed 11 received a total of 7.1 cm of precipitation with a daily average of 1.2 cm. The two events in subwatershed 11 likely represent worst-case scenarios for suspended solid losses and the associated P losses. Prolonged periods of precipitation overwhelm the soils infiltration capacity and cause surface runoff. In subwatershed 11, crops are planted to the stream edge and there is poor streambank stabilization. Due to the agricultural practices and lack of BMP implementation, agricultural runoff in subwatershed 11 is easily transported into the stream. Combined with extreme precipitation events, the P and solids losses can be enormous. For example, during the June 22nd-30th event in 2015, the unit-area TSS loss from subwatershed 11 was approximately 1,870 kg/ha or 0.83 tons/acre. The extreme TP losses from the April 25th, 2015 event in subwatershed 12 are more enigmatic. Although the event accounted for 68.7% of the TP exported from the subwatershed during the 2015 monitoring period, the event only accounted for 1.9% of the DRP export for the monitoring season. This indicates that the high TP losses were particulate associated. However, the TSS and VSS loads during the event were not abnormal. Photos taken at the field site before and after the event show a large increase in algae which is indicative of an increase in P
loading. Therefore, the suspended solids during the event must have been highly P-enriched. This conclusion is corroborated by McDowell et al. (2004) which found that particulate P enrichment ratios increased with decreasing erosion. These findings show how precipitation, agricultural practices, and BMPs can interact to influence P and suspended solids losses.

Overall, no significant concentration response to flow trends \( (r^2=0.00-0.18) \) were observed for TP, DRP, TSS, or VSS at any of the tile or stream sites (T8, S11, S12, T12). Previous studies have found varying levels of correlation between concentration and flow. A study by King et al. (2015) found that drainage TP and DRP concentrations were highest when flow exceeded the 75th percentile of measured flow rates. However, a study by Kinley et al. (2007) observed few significant correlations between flow rate and TP and soluble reactive phosphorus concentrations in drainage samples collected from 39 Canadian fields. In their study, Kinley et al. (2007) observed positive slopes for 73% and 60% of regression lines between flow and TP and soluble reactive phosphorus concentrations. Likewise, 68.8% of regression lines between BHL analyte concentrations and flow had a positive slope. This suggests that, in general, analyte concentrations increase with flow but with varying rates of increase. Thus, land management is likely playing a greater role than flow in P and suspended solids losses.

As evident in Figures 4A & 3C, TP concentrations exceeded the EPA recommended limit of 0.05 mg P/L for streams discharging into lakes during all flow conditions. During the two-year study, 42.0% of the WFW samples and 71.3% of the EFW samples from the five monitoring sites (S8, T8, S11, S12, T12) exceeded 0.05 mg
P/L. A study performed by King et al. (2015) measured TP and DRP concentrations in drainage water and a watershed outlet and found that over 90% of their samples exceeded Environment Canada’s recommended limit of 0.03 mg/L. Overall, 60.7% of WFW samples and 81.6% of EFW samples from the BHL monitoring sites exceeded 0.03 mg/L during the two-year period. TP and DRP concentrations exceeding both the EPA and Environment Canada recommended limits have been observed in other agricultural watersheds drained fields in the United States and Canada (King et al., 2015, Kinley et al., 2007, Logan et al., 1980). The samples from the BHL watershed raise concern that TP concentrations are frequently over 10 times and up to 94 times greater than the 0.05 mg P/L EPA recommended limit. Typically, P losses from fields are not of economic importance to farmers and they apply manure in excess of crop nutrient demands to meet crop N requirements or to dispose of surplus waste (King et al., 2015, Sharpley et al., 1999, Stamm et al., 1998). Previous studies have observed elevated P concentrations in surface and drainage waters during events after manure application (Geohring et al., 2001, Smith et al., 2001, Van Es et al., 2004). However, since BHL P concentrations are elevated during the entire monitoring period and not just during events, land management is likely more influential than manure application on P concentrations in the BHL watershed.

3.4.3. Surface vs. Tile

Overall, of the five monitoring locations, the grassed waterway (S8) had the highest median EFW concentration for all analytes. Although grassed waterways have been shown to control erosion while transporting runoff offsite, they are typically
ineffective at removing dissolved nutrients (Shipitalo and Edwards, 1996). A study conducted by Fiener and Auerswald (2009) measured DRP concentrations in runoff from watersheds with and without hydrodynamically rough grassed waterways implemented. They concluded that grassed waterways had little impact on DRP concentration and that the grassed waterways would only reduce the DRP load corresponding to the decrease in total runoff. Samples from the BHL grassed waterway could only be collected when flow was experienced flow during extreme precipitation events. Therefore, the median analyte concentrations could be highest at the grassed waterway because samples were being collected only at the peak of the hydrograph whereas samples were collected for the entire hydrograph at the other four sites. Despite having the highest median EFW concentrations for all analytes, the grassed waterway contributions to the total subwatershed loads were small because it rarely experienced flow; in 2015, the grassed waterway contributed an average of 32.2% of the total subwatershed analyte loads and in 2016, the grassed waterway only contributed an average of 0.9% of the total subwatershed analyte loads.

Site S11 had the highest TP and DRP unit-area loads of the five monitoring sites in both 2015 and 2016. Overall, TP unit-area loads ranged from 0.231-2.997 kg P/ha at the stream sites (S11 & S12) and from 0.048-0.489 kg P/ha for TP at the tile sites (T8 & T12). DRP unit-area loads ranged from 0.024-0.374 kg P/ha at the stream sites and from 0.013-0.216 kg P/ha at the tile sites. These results show that subsurface drainage is an import pathway in P transport and that drainage P losses can be significant. Previous studies have also observed comparable unit-area P losses between drainage and surface
waters. Reported TP unit-area losses in drainage and surface waters range from 0.28-1.5 kg/ha and 0.09-2.12 kg/ha, respectively, and reported drainage DRP unit-area losses range from 0.22-0.84 kg/ha (Eastman et al., 2010, Gentry et al., 2007, King et al., 2015).

In these studies, drainage was found to contribute between 24.0-90.7% of the total TP export and 33-69% of the total DRP export. Tile contributions to the cumulative TP and DRP losses in BHL watershed are consistent with the reported values; tile contributions to TP losses ranged from 62.5-99.4% at subwatershed 8 and 2.8-17.2% at subwatershed 12 and tile contributions to DRP losses ranged from 53.8-98.9% in subwatershed 8 and 10.8-34.7% in subwatershed 12.

In both years of the study, S11 had the highest TSS and VSS unit-area loads of the five monitoring sites by one or more orders of magnitude. At the stream sites, unit-area loads ranged from 27.8-3,269.3 kg/ha for TSS and 22.8-2,926.0 kg/ha for VSS. In contrast, the unit-area loads at the tile sites ranged from 11.2-52.9 kg/ha for TSS, and 6.4-36.5 kg/ha for VSS. TSS and VSS loads in surface waters are generally greater than those in drainage because of the presence of stream bank/bed erosion. A study by Kronvang et al. (1997) found that 66-89% of the total catchment losses of suspended sediment were from stream bank/bed erosion versus only 11-15% from subsurface drainage. However, the results from the BHL watershed demonstrate that drainage can also be an important pathway in sediment transport.

During the monitoring period, the tile sites had higher proportions of DRP in the TP load than the stream sites. At the two tile sites, DRP accounted for 28.3-70.0% of the TP load versus 10.5-16.9% for the two stream locations. At the grassed waterway, DRP
accounted for the majority (53.5-68.3%) of the TP load. The DRP:TP ratios measured at the BHL stream sites are much lower than those observed by Gentry et al. (2007). In their study, they found that DRP loads accounted for 35-73% of the total TP load at their stream sites. These results indicate that the dominant forms of P loss in the BHL watershed are particulate phosphorus at the surface sites and dissolved phosphorus at the tile sites. Although the BHL tile sites have greater DRP:TP ratios than the stream sites, the TSS and VSS unit-area loads were similar. Therefore, the sediment transported via drainage is less P enriched than the sediment transported by runoff. In runoff, eroded particles are P enriched because of the preferential transport of light, P-sorptive fine particles (McDowell et al., 2004). In the BHL watershed, there are few surface intakes to tile drainage. Consequently, suspended solids in drainage waters in the BHL watershed are likely due to preferential flow of topsoil through soil macropores (Grant et al., 1996, Laubel et al., 1999).

3.4.4. Impact of Conservation Practices

Subwatershed 12 has the greatest BMP implementation of the three monitored subwatersheds. Within the subwatershed, BMPs including terraces, CRP filters and wetlands, and nutrient management plans cover 87.5% of the total subwatershed area. In contrast, BMP implementation only occurs on 22.5% of subwatershed 8 and 30.0% of subwatershed 11. Comparisons are made between water quality at the tile outlets (T8 & T12) in subwatersheds 8 and 12 and between the surface sites (S11 & S12) in subwatersheds 11 and 12.
The positive impact of implemented BMPs on surface water quality is incontroversible. Site S12 had lower median WFW DRP, TSS, and VSS concentrations than site S11. However, S12 also had the highest median WFW TP concentration (0.059 mg P/L) of the five monitoring sites and 61% of samples exceeded the EPA recommended TP limit of 0.05 mg P/L. In comparison, S11 had a median WFW TP concentration of 0.035 mg P/L and 43% of samples exceeded 0.05 mg P/L. Because the DRP, TSS, and VSS concentrations are lower at S11 than S12, the elevated TP concentrations at S12 could indicate that soils in subwatershed 12 are P-enriched compared to the soils in subwatershed 11. Despite this, S12 had lower EFW median concentrations than S11 for all analytes and, overall, S12 had lower unit-area loads than S11 for each analyte and each year. These results indicate that the terraces, CRP filters and wetlands, and nutrient management plans implemented in subwatershed 12 also have a positive effect on P and suspended solids losses in surface waters, especially during events. Respectively, filter strips and nutrient management plans have been found to reduce TP losses by an average of 56% and 47% (Gitau et al., 2005). In subwatershed 11, crops are planted up to the edge of the stream and there is poor stream bank stabilization. Combined with the lack of BMP implementation, these factors explain the high suspended solids losses from site S11. Since particulate P dominates the P losses at the BHL stream sites, these factors also explain the high P losses from S11.

The implementation of BMPs in the BHL watershed also appear to have a positive impact on tile water quality. In subwatershed 12, the tile outlet (T12) had the lowest median WFW and EFW concentrations of the five monitoring locations for all
analytes. In fact, the TP concentrations of the T12 samples never exceeded the 0.05 mg P/L EPA recommended TP limit for streams discharging into lakes for the WFW samples and only exceeded the limit 32% of the time for the EFW samples. In contrast, at T8, 49% and 89% of WFW and EFW samples, respectively, exceeded TP concentrations of 0.05 mg P/L. During each year of the study, T12 also had lowest TP, DRP, TSS, and VSS unit-area loads of the four tile and stream sites. Furthermore, the percent of the two-year cumulative TP load exported through the tile pathway was quite different between subwatersheds 8 and 12. In subwatershed 8, 69.8% of the total TP exported was through T8 while in subwatershed 12, only 4.8% of the total TP exported was through T12. The results from subwatershed 8 indicate that subsurface drainage is a significant P transport pathway while the results from subwatershed 12 highlight the positive benefits of BMP implementation. Previous studies have also observed substantial drainage TP contributions ranging from 17% to over 50% of the total P losses (Culley et al., 1983, Enright and Madramootoo, 2004, Jamieson et al., 2003, King et al., 2015, Ruark et al., 2012, Smith et al., 2015, Tomer et al., 2010). Since unit-area flow is similar at T12 and T8 but T12 has lower analyte concentrations and loads, it is apparent that the BMPs in subwatershed 12 have a positive effect on drainage P and suspended solid losses. This finding is contrary to that of Lemke et al. (2011) which concluded that increased implementation rates for grassed waterways, stream buffers, and strip-tillage were not adequate to overcome the nutrient export from subsurface drainage. Furthermore, soil compositions of the three BHL subwatersheds were similar. Therefore,
land management practices likely have a greater influence than soil type on drainage P and suspended solids losses in the BHL watershed.

At the subwatershed scale, subwatershed 12 had the lowest cumulative TP, DRP, TSS, and VSS loads of the three BHL subwatersheds. However, when loads were normalized by area, unit-area loads for TP, TSS, and VSS were lowest each year at subwatershed 8. Despite this, subwatershed 8 had the highest DRP unit-area loads for both years of the study. Since the subwatershed unit-area loads were calculated as the cumulative subwatershed export divided by the total subwatershed area, the low TP, TSS, and VSS unit-area loads at subwatershed 8 are explained by its large size relative to subwatersheds 11 and 12. Subwatershed 8’s high DRP unit-area loads are reflective of the dominance of the tile outlet over the grassed waterway. The unit-area loads in subwatershed 12 were less than those of the similarly sized subwatershed 11 for each analyte and each year of the study. Furthermore, event contributions to the unit-area loads in subwatershed 11 averaged 88.4% versus only 65.0% in subwatershed 12. The comparison between subwatershed 11 and 12 emphasizes the benefits of BMP implementation at the subwatershed scale and especially the impact of BMPs on reducing the high losses associated with events.

3.4.5. Intra-Event

Intra-event samples were collected during events on April 19th, April 27th, April 30th, and June 14th in 2016 and were analyzed for TP, TDP, and TPP. During each of the events, the TP, TDP, and TPP concentrations generally followed the flow trends with peaks in concentrations corresponding to peaks in the hydrograph. At the grassed
waterway (S8) and the two stream (S11 & S12) monitoring locations, strong, positive correlations were observed between analyte concentrations and flow; adjusted \( r^2 \) values ranged from 0.71-0.82 for TP, 0.59-0.87 for TDP, and 0.54-0.70 for TPP. During the events, flow did not decrease after peaking at both of the tile locations, indicating that the soil has reached its infiltration capacity. Peaks in the tile sample analyte concentrations coincided with peaks in the hydrograph. However, analyte concentrations decreased after peaking despite the elevated tile flow. Subsequently, correlations between analyte concentrations and flow at the tile locations (T8 & T12) were weaker than those observed at the surface sites; adjusted \( r^2 \) values ranged from 0.15-0.33 for TP, 0.24-0.28 for TDP, and 0.01-0.28 for TPP. A study by Kinley et al. (2007) found significant correlations between flow rate and drainage TP concentrations at only 30% of their fields but that 73% of fields had positive slopes for the regression lines. They concluded that concentration generally increases with flow but with varying rates of increase. Overall, the intra-event time-series imply that flow is the driving force behind event analyte concentrations and that the analytes are flushed through the drainage system.

From the ANOVA results, no significant differences were identified between the intra-event peak TP or TPP concentrations at the five sites \((p=0.2128, \ p=0.1432)\) or between the surface vs tile sites \((p=0.4628, \ p=0.3643)\). Peak TP and TPP concentrations ranged from 0.139-3.114 mg P/L and 0.029-2.709 mg P/L, respectively, for the surface sites and from 0.094-1.217 mg P/L and 0.046-0.655 mg P/L, respectively, for the tile sites. Although no significant differences between peak intra-event TP or TPP
concentrations were observed in the BHL watershed, previous studies have observed P concentrations in surface runoff to be up to 10.9 times greater than those in drainage (Eastman et al., 2010, Enright and Madramootoo, 2004). Peak TDP concentrations during the four events ranged from 0.081-0.802 mg P/L for the three surface sites and from 0.047-0.655 mg P/L for the two tile sites. The ANOVA performed on the peak TDP concentrations at the surface vs tile sites also identified no significant differences (p=0.9851). However, the ANOVA on the peak TDP concentrations at each site indicated that the peak TDP concentrations at the grassed waterway (S8) were significantly higher (p=0.0313) than those of the other four sites.

The intra-event samples were also analyzed for TSS for each of the four events. At the two stream and the two tile monitoring locations, peaks in TSS concentrations coincided with peaks in the hydrograph. At the grassed waterway, however, TSS concentrations remained high at the end of the hydrograph. After peaking, tile TSS concentrations decreased despite continued elevated flow. Consequently, correlations between TSS concentration and flow were strong at the stream sites (adj. $r^2=0.46-0.60$) and weak at the tile outlets (adj. $r^2=-0.05-0.18$) and the grassed waterway (adj. $r^2=-0.07$). A study by Lamb and Toniolo (2016) also found significant positive correlations ($r^2=0.76-0.97$) between discharge and suspended sediment concentration in three rivers. During the four events, peak TSS concentrations at the three surface and two tile sites ranged from 23.3-2104.0 mg/L and 27.3-336.0 mg/L, respectively. No significant differences were identified between the peak TSS concentrations between the five sites.
(\(p=0.2245\)) or between the surface vs tile sites (\(p=0.3798\)). These results indicate that subsurface drainage is also an important pathway in suspended solids transport.

Correlations between intra-event TPP and TSS concentrations were strong at the stream sites (adj. \(r^2=0.92-0.97\)) and the tile outlets (adj. \(r^2=0.89-0.97\)) but low at the grassed waterway (adj. \(r^2=0.02\)). It is well documented that P losses are sediment associated. For example, Heathwaite and Dils (2000) found a significant correlation (\(p<0.01\)) between suspended sediment concentration and P losses in drainage and Grant et al. (1996) found significant correlations (\(p<0.001\)) between particulate matter and particulate phosphorus concentrations. Furthermore, Culley et al. (1983) concluded that 34% of the total drainage P load at their site was sediment associated. Compared to surface soil, eroded particles are P enriched because light, P-sorptive fines are more likely to be transported than coarser particles (McDowell et al., 2004). During the April 27th, April 30th, and June 14th events, TPP concentrations exceeded those of TDP during the rising limb of the hydrograph but TPP concentrations were less than TDP concentrations during the falling limb of the hydrograph. This indicates that initial P losses during events is particulate associated. Previous studies have also found that soil has a greater potential to supply flow with P at the start of an event compared to the end of an event (McDowell and Sharpley, 2002, McDowell and Sharpley, 2002). Thus, suspended solids and the associated particulate P are flushed through the drainage system.
CHAPTER 4: SIMULATING DRAINAGE PHOSPHORUS AND SUSPENDED SOLIDS LOSSES IN THE BLACK HAWK LAKE WATERSHED, IOWA USING THE SOILICEDB MODEL

4.1. Introduction

Modeling of P losses through drainage is quite limited. However, a review on drainage P modeling by Radcliffe et al. (2015) deemed ICECREAMDB as the most promising model to simulate these losses because it minimizes the number of input parameters by combining mechanistic and empirical approaches. ICECREAM and its graphical front-end, ICECREAMDB, have been used to simulate P losses in surface runoff in Finland and P losses in drainage in Sweden (Blombäck and Persson, 2009, Larsson et al., 2007, Liu et al., 2012, Rekolainen and Posch, 1993, Rekolainen et al., 2002, Tattari et al., 2001). Larsson et al. (2007) used ICECREAMDB to simulate drainage P losses from seven experimental plots in south-west Sweden. After comparing the simulated losses from ICECREAMDB with the measured drainage losses of SP, PP, and DRP, they concluded that the model performed acceptably when estimating losses during events but that some short-term fluctuations were not captured. Blombäck and Persson (2009) evaluated the effects of site-specific parameterization of ICECREAMDB’s soil-related parameters and concluded that when the standard parameter values were used, the model overestimated TP and DP losses. In contrast, the magnitude of the peak flows and total runoff volume were underestimated. When site-specific values were used, runoff volume was further underestimated but the overestimation of TP was reduced. Liu et al. (2012) used ICECREAMDB to identify the P leaching risks in drained fields with long-term fertilization regimes. They concluded that ICECREAMDB must be further developed to
include P sorption and desorption processes because the model overestimated measured TP leaching by a factor of 5-9. Finally, Radcliffe et al. (2015) reviewed the ICECREAMDB model performance and concluded that overall, the model performs satisfactorily in estimating DRP losses in soils with normal sorption capacity and in identifying the timing of macropore transport. However, they noted that modeling of DRP in soils with very high/low sorption capacity and the estimation of the magnitude of water flows and PP losses could be improved. ICECREAMDB has received several recent updates. First, the model has implemented a new approach for partitioning between surface runoff and macropore flow. Second, the SOIL model has been coupled to the P and erosion routines in ICECREAM, replacing the curve number method and cascade approach for water flow calculations (Radcliffe et al., 2015). Both of these updates require additional evaluation against measured data.

The objective of this study is to simulate the drainage P losses in the BHL watershed using ICECREAMDB. Specifically, this study provides an evaluation of the performance of the new updates to the ICECREAMDB model as well as its applicability to cropland outside of Scandinavia. A successful model could identify areas for remediation and BMP implementation.
4.2. Materials & Methods

4.2.1. SoilIceDB Model

SoilIceDB is a system which runs the ICECREAMDB model on the SOIL model output. Since both of these models are described in detail elsewhere, only a brief overview of the models is included here.

The SOIL model calculates one dimensional water and heat dynamics in the soil profile (Jansson, 1994). The basis of the SOIL is a soil profile divided into a finite number of layers. Water dynamics are calculated using the Richard’s equation and heat dynamics are calculated using the heat conduction equation (Radcliffe et al., 2015). Pools are included in the SOIL model for snow, intercepted water, and surface ponding to simulate processes at the upper soil boundary (Jansson, 1994). The outputs from SOIL are used as inputs for the ICECREAMDB model.

ICECREAMDB is the graphical front-end for the ICECREAM model, a management oriented phosphorus loss model that quantifies runoff, erosion, and P losses and has the capability to simulate P losses through drainage. Specifically, the model calculates losses of sediment, PP, and DP through surface, matrix, and macropore transport (Radcliffe et al., 2015). Although it was designed for the field scale, ICECREAM has been used for small catchment modeling by aggregating the results of the model using typical soil-crop-slope combinations. (Rekolainen et al., 2002). The model has a daily time step and uses daily time series data as input. ICECREAMDB is a mixed model and includes
empirical submodels, such as USLE for erosion and degree days for crop development, and process based submodels such as P sorption/desorption (Radcliffe et al., 2015).

4.2.2. Model Development

A study by Blombäck and Persson (2009) concluded that site-specific parameterization of the soil-related parameters did not improve the simulation results over the default parametrization. Therefore, simulations using the SoilIceDB model were performed using Iowa input data and the default soil-related parameters in order to obtain baseline performance results.

The SoilIceDB model reads inputs from seven different databases: Climate, SOIL Management, SOIL Parameters, SoilIceDB Settings, ICECREAM Management, ICECREAM Parameters, and ICECREAM Variables. The SOIL Parameters, ICECREAM Parameters, and ICECREAM Variables were unmodified from those included in the SoilIceDB package. The SoilIceDB model was validated using data from 2015-2016.

The SoilIceDB climate database requires daily temperature, humidity, wind speed, precipitation, and global radiation data. Historical climate data for the BHL watershed from the Carroll, Iowa weather station was obtained from the National Aeronautics and Space Administration (NASA) Near Real-time Daily Global Radiation and Meteorology data explorer and from the National Oceanic and Atmospheric Administration (NOAA) Climate data explorer (NASA, 2017, NOAA, 2017). The latitude, longitude, and altitude inputs were set according to those of the Carroll weather station. The SoilIceDB vegetation start and end dates were set as April 22 and
November 17\textsuperscript{th}, respectively, which are typical planting and harvesting dates for corn in Iowa (NASS, 1997). Using measurements from Kung (1964), the respective albedo values for snow, soil, and vegetation were set to 0.50, 0.10, and 0.16. All other climate parameters were left unchanged.

In the SOIL Management database, tile drain depth and spacing were set to the typical values for Nicollet soil in Iowa, 0.91 and 22.86 meters, respectively (Melvin et al., 2012). The soil type was set as Sandy Clay Loam. All other parameters were left unchanged.

The ICECREAM Management database provides the input cropping system, field dimensions, and slope classifications. A continuous corn system was used since fields in subwatersheds 8 and 12 were planted with corn in both 2015 and 2016. The field operations and fertilizer application rates were adapted from Ikenberry (2016) are summarized in Table 4. Fertilizers included anhydrous ammonia, urea ammonium nitrate (UAN), and diammonium phosphate (DAP). Broadcast fertilizers (UAN and DAP) were simulated at a depth of 0.9mm and incorporated into the soil with the seedbed implement as per the ICECREAM user manual. The SoilIceDB model is designed for field scales but we are testing its ability to model small catchments. Therefore, the input field widths and lengths representing the two BHL subwatersheds were approximated by solving for the field widths and lengths that provided the total subwatershed area while maintaining the subwatershed width:length ratios. Field width and length in meters for subwatershed 8 were 1,846 and 4,455, respectively. Subwatershed 12 is smaller and the field width and length in meters were 1,173 and 1,886, respectively. Input slope values
were calculated for each subwatershed by taking the weighted average of the slopes for each area within the estimated drainage extents. The weighted average slopes for subwatersheds 8 and 12 were 1.9% and 0.8%, respectively. The soil type was set to Sandy Clay Loam. All other input parameters were set to the default values.

Table 4: Field Operations for ICECREAM model

<table>
<thead>
<tr>
<th>Year</th>
<th>Month</th>
<th>Day</th>
<th>Action</th>
<th>N (kg/ha)</th>
<th>P (kg/ha)</th>
<th>Depth (mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>4</td>
<td>15</td>
<td>Add Fertilizer (Anhyd.)</td>
<td>120.54</td>
<td>0</td>
<td>178</td>
</tr>
<tr>
<td>1</td>
<td>4</td>
<td>15</td>
<td>Add Fertilizer (UAN)</td>
<td>47.85</td>
<td>0</td>
<td>0.9</td>
</tr>
<tr>
<td>1</td>
<td>4</td>
<td>15</td>
<td>Use Implement (Seedbed)</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>1</td>
<td>5</td>
<td>1</td>
<td>Plant Corn</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>1</td>
<td>9</td>
<td>30</td>
<td>Harvest Corn</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>1</td>
<td>10</td>
<td>25</td>
<td>Add Fertilizer (Anhyd.)</td>
<td>56.58</td>
<td>0</td>
<td>178</td>
</tr>
<tr>
<td>2</td>
<td>4</td>
<td>15</td>
<td>Add Fertilizer (Anhyd.)</td>
<td>109.88</td>
<td>0</td>
<td>178</td>
</tr>
<tr>
<td>2</td>
<td>4</td>
<td>15</td>
<td>Add Fertilizer (DAP)</td>
<td>58.68</td>
<td>65.45</td>
<td>0.9</td>
</tr>
<tr>
<td>2</td>
<td>5</td>
<td>1</td>
<td>Plant Corn</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>2</td>
<td>9</td>
<td>30</td>
<td>Harvest Corn</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>2</td>
<td>10</td>
<td>25</td>
<td>Add Fertilizer (Anhyd.)</td>
<td>56.58</td>
<td>0</td>
<td>178</td>
</tr>
</tbody>
</table>

4.2.3. Model Evaluation

The SoilIceDB model was setup to simulate the drainage flow, TP losses, DRP, and TSS losses from the two BHL drainage outlets. Simulations were run for 2008-2016 so there would be a 7-year spin-up period before the model was evaluated against the 2015 and 2016 observed data. Simulation results were summarized at the daily, monthly, and yearly timescales.
To evaluate the SoilIceDB model performance, Nash-Sutcliffe efficiency (NSE) and percent bias (PBIAS) analyses were conducted on the daily, monthly, and yearly simulated flow and DRP, TP, and TSS losses. NSE is a normalized statistic which indicates how well the simulated versus observed data fit the 1:1 line. Values range from $-\infty$ to 1.0 with values between 0.0 and 1.0 indicating an acceptable model performance. NSE values less than 0.0 indicate that the mean observed value is a better predictor than the simulated value. PBIAS measures the average tendency of simulated values to be greater than or less than their corresponding measured values. Low magnitude PBIAS values indicate accurate model performance and the optimal PBIAS value is 0.0. Positive PBIAS values represent an underestimation bias whereas negative PBIAS values represent an overestimation bias (Moriasi et al., 2007). Since BHL samples were not collected every day, the daily NSE and PBIAS analyses only compared the losses for each measured sample to the simulated losses for the day the sample was taken. The measured monthly losses were assumed to be the difference between the cumulative loss for the last sample collected during the month and the cumulative loss for the last sample collected during the previous month. Finally, the yearly simulated losses were calculated as the sum of the simulated losses for each of the months in the March-November monitoring period.
4.3. Results

The SoilIceDB model was used to simulate the flow and DRP, TP, and TSS losses at the two BHL tile outlets. The model evaluation statistics are summarized in Table 5.

Table 5: SoilIceDB Model Evaluation Statistics

<table>
<thead>
<tr>
<th>Site</th>
<th>Flow</th>
<th>DRP</th>
<th>TP</th>
<th>TSS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Timescale</td>
<td>NSE</td>
<td>PBIAS</td>
<td>NSE</td>
<td>PBIAS</td>
</tr>
<tr>
<td>Site</td>
<td>Flow</td>
<td>DRP</td>
<td>TP</td>
<td>TSS</td>
</tr>
<tr>
<td>Timescale</td>
<td>NSE</td>
<td>PBIAS</td>
<td>NSE</td>
<td>PBIAS</td>
</tr>
<tr>
<td>T8</td>
<td>T12</td>
<td>T8</td>
<td>T12</td>
<td>T8</td>
</tr>
<tr>
<td>Day</td>
<td>Month</td>
<td>Year</td>
<td>Day</td>
<td>Month</td>
</tr>
</tbody>
</table>

4.3.1. Flow

Flow simulations were evaluated at both the monthly and yearly timescales; flow was not evaluated at the daily time scale because the ISCO sampler software does not provide an option to summarize flow data at the daily timescale.

Overall, the model overestimated flow at the BHL tile outlets; PBIAS values ranged from -128.9 to -105.3. In Figures 13 A&B, it is evident that the model overestimates the total
flow and that the peaks in measured and simulated flow do not always coincide. NSE values were greater at the yearly timescale (NSE= -0.4 and -0.2) than at the monthly timescale (NSE= -1.0 and -1.9). Based on the results of the NSE and PBIAS analyses, the model was best at simulating flow at the yearly timescale, but did not acceptably simulate the flow at either time scale for both sites.

Figure 13: Monthly measured vs. simulated flow at the tile outlets in (A) subwatershed 8 and (B) subwatershed 12.
4.3.2. DRP

Overall, the model performed the best at simulating DRP losses at the daily timescale. For T8, the NSE and PBIAS values at the different timescales ranged from -1.9 to -0.1 and from 37.9-94.1, respectively. Likewise, for T12, NSE and PBIAS values for the different timescales ranged from -8.7 to -0.2 and from -206.9 to 11.3, respectively. The NSE values indicate that the mean observed values are a better predictor than the simulated values and the generally positive PBIAS values indicate that the SoilIceDB model has an underestimation bias. This is evident in the daily time series presented in Figures 14 A&B; observed peaks in DRP losses are generally simulated but the magnitudes of the peaks are underestimated. However, the model had a strong overestimation bias at the monthly timescale at T12.

4.3.3. TP

The model performed much better at simulating TP losses at T8 than at T12. However, for both sites, the model performed very poorly at the yearly time scale. For the T8 simulation, NSE and PBIAS values for the daily and monthly timesteps ranged from -15.2 to -1.1 and -173.8 to 69.5, respectively. In contrast, for the T12 simulation, NSE and PBIAS values ranged from -1,296.2 to -26.3 and from -1,715.8 to -57.8, respectively. At the yearly timescale, however, NSE and PBIAS values for the two sites ranged from -2.31x10^{10} to -29,337.4 and from -238,102.5 to -2,719.6, respectively. Based on the NSE and PBIAS analyses, the model typically overestimates TP losses and performs best at simulating TP losses at the daily timescale. As apparent in Figure 15 A,
however, at the daily timescale, the model underestimated TP losses at T8. Overall, the model did not acceptably model TP losses and the mean observed TP losses are better predictors than the simulated values.

Figure 14: Daily measured vs. simulated DRP losses at the tile outlets in (A) subwatershed 8 and (B) subwatershed 12.
4.3.4. TSS

The model performed unacceptably at simulating TSS losses in the two BHL tile outlets at each of the timescales. Overall, the model was best at simulating the TSS losses at the daily timescale. For T8, the NSE and PBIAS values were -39.6 and -101.2, respectively. Likewise, for T12, the NSE and PBIAS values were -5.3 and 33.3,
respectively. From Figures 16 A&B, it is evident that the model typically simulates that no TSS is lost through the tiles but then highly overestimates the sediment losses whenever it does simulate TSS losses.

![A) Daily measured vs. simulated TSS losses at the tile outlets in (A) subwatershed 8 and (B) subwatershed 12.](image)

At the monthly and yearly timescales, PBIAS values ranged from -1,612.3 to 33.3 and indicate that the model severely overestimates sediment losses from the two tile
outlets. With values of -835.2 or less, the results from the NSE analyses at the monthly and yearly timescales also indicate a poor model performance and that the mean observed TSS losses are better predictors than the simulated results.

4.4. Discussion

In order to develop baseline results on the SoilIceDB model performance, the model was setup with Iowa input data but run with default parameters developed for Sweden. Based on NSE and PBIAS analyses, the model did not produce acceptable simulations of the flow, DRP, TP, or TSS losses at the two BHL tile outlets.

The SoilIceDB model overestimated flow for the simulations at both sites and peaks in the measured and simulated flow did not always coincide. This finding is contrary to that of Blombäck and Persson (2009) which found that the magnitude of the peak flows were underestimated. Because flow has a significant influence on P and suspended solids losses, it is troublesome that the timing of peaks in the simulated flow does not match those of the measured flow. However, the magnitude of the NSE values at the yearly time scale is low (NSE= -0.4 and -0.2), so with further calibration, the model may be able to simulate flow acceptably.

Overall, the model was best at simulating DRP losses. Peaks in the simulated DRP loss time series generally coincided with peaks in the observed time series, but the model underestimated the magnitude of the peaks. In contrast, a study by Blombäck and Persson (2009) found that the ICECREAM model overestimated dissolved P losses. The NSE analysis of the simulated DRP losses produced NSE values from -8.7 to -0.2 and
indicate that the mean observed values are a better predictor than the simulated values. However, the low magnitude of the NSE values at the daily time step (NSE= -0.2) and the findings of provide optimism that, with a more extensive calibration, the model will be able to acceptably simulate DRP losses.

At both sites, the model performed poorly at simulating TP and TSS losses. Generally, the model overestimated TP losses. This finding is consistent with previous studies which have found that the ICECREAM model overestimates TP losses up to a factor of 5-9 over measured data (Blombäck and Persson, 2009, Liu et al., 2012). However, at the daily timescale, the model underestimated the TP losses at T8. When simulating TSS losses, the model typically estimated zero losses but when it did simulate losses, the values were extreme overestimates of the measured values. Consequently, the model extremely overestimated the episodic TSS losses but did not capture short-term fluctuations. In fact, for one sample, the model overestimated the TSS losses by a factor of 569.

Based on these preliminary baseline results, it appears that the new updates to the ICECREAMDB runoff/macropore flow partitioning and the introduction of the SOIL model have caused the model to underestimate flow and DRP losses instead of overestimating them as observed by Blombäck and Persson (2009). However, the model still significantly overestimates TP losses. Although neither of the SoilIceDB simulations produced acceptable results, the low magnitude of the NSE values for the flow and DRP simulations provide optimism that the model can be adapted to work on cropland outside of Scandinavia.
CHAPTER 5: CONCLUSIONS & RECOMMENDATIONS

5.1. Conclusions

This study expanded on the work of previous studies by comparing P and suspended solids export from both event and weekly flow-weighted samples and evaluating the effects of BMP implementation on drainage water quality. In addition, this study provided an evaluation of the performance of the new SOIL and ICECREAMDB model combination, SoilIceDB, as well as an evaluation of its suitability to cropland outside of Scandinavia.

The results from the BHL watershed show that high P concentrations are prevalent for all flow conditions in both surface and drainage waters. However, a comparison between surface and drainage monitoring sites demonstrated that drainage TP, DRP, TSS, and VSS losses are significant. In the BHL watershed, P losses in surface waters were dominated by particulate P whereas losses in drainage were dominated by dissolved P. During each year of the study, a single event overwhelmed the annual P and suspended solids losses from subwatershed 11. These findings highlight the importance of single events and show how the combination of precipitation, agricultural practices, and BMPs can influence P and suspended solids losses. An analysis of intra-event samples showed that flow is the driving factor behind analyte concentrations during events. Consequently, future water quality improvement efforts must focus on reducing event losses of particulate P in surface runoff and dissolved P in drainage waters. Fortunately, the paired watershed comparison demonstrated that conservation practices are effective in reducing P and suspended solids losses in both surface waters
and subsurface drainage. The preliminary results from the SoilIceDB simulations indicate that with more calibration, the model may acceptably simulate flow and DRP losses in the U.S. and at the small catchment scale. The results of this study will be especially helpful in identifying strategies to reduce agricultural nutrient losses and in future drainage P modeling efforts.

5.2. Recommendations

Future research can expand upon this study in several ways. The study by Gentry et al. (2007) highlighted the impact of the spring snowmelt on P losses. Since the BHL watershed monitoring period was March-November, the study could have missed significant P losses during spring snowmelts. Collecting grab samples during snowmelt events or extending the monitoring season with the ISCO samplers would provide insight on the effects of the implemented BMPs on winter P and suspended solids losses.

The Gentry et al. (2007) study also emphasized the impact of manure application practices on P losses. Previous studies have observed high P losses in drainage and surface waters after manure application (Geohring et al., 2001, Smith et al., 2001, Van Es et al., 2004). Stakeholders in subwatershed 12 have said that no manure is applied to the fields within the subwatershed. However, the extent of manure application in subwatersheds 8 and 11 is unknown. Although land management is likely more influential than manure application on P losses in the BHL watershed, manure application data such as extent, rate, and timing for the three BHL subwatersheds would
facilitate an analysis of the impact of BMPs on the responses of surface and tile sites to manure application.

The unit-area loss comparisons of this study are dependent on the assumption that the entire subwatershed contributes to drainage flow. However, a more comprehensive estimate would provide a more accurate comparison between the unit-area losses at the BHL monitoring sites. Most current drainage extent estimates are at the county or state scale, which aren’t detailed enough for small catchment scale monitoring or modeling purposes. The United States Geological Survey (USGS) recently created a raster GIS coverage of estimated subsurface drainage extent (Nakagaki and Wieczorek, 2016). However, in Iowa, the coverage assumes that cropland on both poorly and moderately drained soils are drained because Iowa’s reported drainage extent was greater than the area of cropland on poorly drained soils. In conclusion, the development of an accurate, high-resolution drainage extent coverage would benefit all catchment scale modeling efforts of extensively drained areas.

Finally, more calibration must be performed on the SoilIceDB model. Although Blombäck and Persson (2009) concluded that using site-specific soil parameterization did not improve simulation results over the default soil parameterization, using detailed soil data for the BHL subwatersheds could help to improve the simulation results. Furthermore, since the model was designed for the field scale, model performance at the small catchment scale could be improved by dividing the BHL subwatersheds into areas of similar soil-crop-slope combinations and aggregating the results.
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Supplementary Figure 1: Flow per area exceedance curves for March to November for the tile monitoring sites with (A) dissolved reactive phosphorus concentrations and (B) volatile suspended solids concentrations; flow exceedance curves for March to November for the stream monitoring sites with (C) dissolved reactive phosphorus concentrations and (D) volatile suspended solids concentrations.
Supplementary Figure 2: Flow and intra-event concentrations of total phosphorus (TP), total dissolved phosphorus (TDP), and total particulate phosphorus (TPP) during event on 4/19/16 at the (A) tile in subwatershed 8 and (B) stream in subwatershed 12.

Supplementary Figure 3: Intra-event concentrations of total suspended solids (TSS) and total particulate phosphorus (TPP) during event on 4/19/16 at the (A) tile in subwatershed 8 and (B) stream in subwatershed 12.
Supplementary Figure 4: Flow and intra-event concentrations of total phosphorus (TP), total dissolved phosphorus (TDP), and total particulate phosphorus (TPP) during event on 4/27/16 at the (A) tile in subwatershed 8, (B) stream in subwatershed 11, and (C) stream and (D) tile in subwatershed 12.
Supplementary Figure 5: Intra-event concentrations of total suspended solids (TSS) and total particulate phosphorus (TPP) during event on 4/27/16 at the (A) tile in subwatershed 8, (B) stream in subwatershed 11, and (C) stream and (D) tile in subwatershed 12.
Supplementary Figure 6: Flow and intra-event concentrations of total phosphorus (TP), total dissolved phosphorus (TDP), and total particulate phosphorus (TPP) during event on 6/14/16 at the (A) tile in subwatershed 8 and (B) stream and (C) tile in subwatershed 12.
Supplementary Figure 7: Intra-event concentrations of total suspended solids (TSS) and total particulate phosphorus (TPP) during event on 6/14/16 at the (A) tile in subwatershed 8 and (B) stream and (C) tile in subwatershed 12.