Temporal and Spatial Variation of Nutrients and Sediment in the Marshes of Green Lake

by

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cfs	cubic feet per second
CHLA	chlorophyll A
CKM	County K Marsh
CTH A	County Trunk Highway A
CTH K	County Trunk Highway K
EC	electric conductivity
EPC_0	equilibrium phosphorus concentration
GLSD	Green Lake Sanitary District
mg/L	milligrams per liter
MGD	million gallons per day
MLR	multiple linear regression
NWIS	National Water Information System
pH	potential hydrogen
RWTP	Ripon Wastewater treatment Plant
SCE	Silver Creek Estuary
SS	suspended sediment
SWIMS	Surface Water Integrated Monitoring System
TDP	total dissolved phosphorus
TN	total nitrogen
TP	total phosphorus
TSS	total suspended solids
USGS	United States Geological Survey
WDNR	Wisconsin Department of Natural Resources

Abstract

Green Lake is the deepest natural inland lake in Wisconsin. Located in the Southeastern Wisconsin Till Plains ecoregion, its watershed is dominated by agriculture. Many of the lake's primary tributaries have excess total phosphorus (TP) and suspended solids (SS) which degrade water quality within the lake. Many of these impaired streams, however, flow through large marshes which presents an opportunity for nutrient and sediment reduction before flows reach the lake itself. The largest marsh, Silver Creek Estuary (SCE) has been a clear water macrophyte-dominated system since around 2006. Prior to carp exclusion and a reduction in an upstream TP point source, the marsh was turbid and phytoplankton-dominated. U.S. Geological Survey (USGS) monitoring data at the outlet of the marsh (1987 – 2017) were used to determine whether the shift from a turbid to clear state decreased TP and SS loading to Green Lake. USGS monitoring data at the primary marsh inlet and outlet (2012 – 2017) were used to quantify when the macrophyte-dominated marsh retained or was a source of TP and SS. Samples taken at the marsh inlet and outlet by the USGS and as part of this study (2006 – 2017) were also compared to determine if nutrient concentrations decreased through the marsh while it was macrophyte dominated.

In all seasons, TP concentrations at the outlet of SCE decreased following the equilibrium shift and the reduction of the upstream point source. The SS concentrations only decreased in spring and summer. The biological shift likely contributed to the SS reductions and TP reductions in the spring and summer. The TP reductions observed beyond the growing season were likely due to the TP reduction from the upstream point source. Over the five-year period (2012 –2017), the restored marsh retained both TP and SS. Seasonally, SS was exported in the fall, but more than that amount was retained in all other seasons resulting in the marsh

being a SS sink. In summer (2006 – 2017), TP, total dissolved P (TDP), SS, and total nitrogen (TN) concentrations all decreased from the inlet to the outlet of the marsh.

The second marsh, County K Marsh (CKM), is believed to be similar to SCE before restoration. Efforts to exclude carp and establish macrophytes began in 2015, with intentions to shift the marsh from phytoplankton to macrophyte-dominance. TP and SS loads from the outlets of the macrophyte-dominated (SCE) and phytoplankton-dominated (CKM) marshes were compared across seasons using USGS monitoring data from 2012 – 2017. Inlets and several locations within each marsh were also sampled regularly over two field seasons (July – October 2016 and April – October 2017) to assess how nutrient reductions differed within the macrophyte versus phytoplankton systems. TP, total suspended solids (TSS) and TN loads were also estimated for each marsh inlet and outlet and used to quantify marsh storage for each sampling event.

Flow-weighted mean TP and SS concentrations at the two marsh outlets were similar in the winter but increased more during the growing season in the phytoplankton-dominated marsh than in the macrophyte-dominated marsh. Fall TP concentrations in CKM remained high compared to SCE. Summer SS concentrations were also higher in CKM compared to SCE. Compared to all 12 sampling sites, both marshes had inputs with high concentrations of TP, TDP, TSS and TN. Interior sites of CKM, however, had higher TP and TSS concentrations than the inlets. TP, TDP, TSS and TN all decreased downstream through SCE. In CKM, only TN and TDP decreased from inlets to downstream. In total over 11 sampling events, CKM was estimated to have retained TN but not TP or TSS while SCE retained all three. Equilibrium phosphorus concentrations (EPC_o), the ambient P concentration that determines whether sediments absorb or release P, estimated for each marsh suggested that SCE sediment may release more P than

CKM sediment. On the day of sampling, water column DRP concentrations indicated sediments in both marshes were releasing P. Since the daily storage estimates did not suggest internal P loading for SCE, P released from sediment was likely deposited elsewhere in the marsh and not exported to Green Lake.

Similar winter nutrient and sediment concentrations measured in both marshes but very different summer concentrations suggest biological processes are instrumental during the growing season. While an equilibrium shift for CKM would not be accompanied by a stark reduction in an upstream TP source, as was the case for SCE, data suggest that excluding carp from and introducing more macrophytes to CKM could reduce TP and SS export to Green Lake.

Chapter 1 Introduction

Green Lake Background

Green Lake, Wisconsin's deepest natural and second most voluminous inland lake, is a valued asset of the Upper Fox River watershed. The lake has a surface area of 30.3 km^2, maximum depth of 72 m, mean depth of 31 m, and average residence time of about 25 years (Stauffer, 1985). The lake drains on its northwest edge into the Puchyan River, eventually flows through the Upper Fox River, Lake Winnebago, Green Bay and Lake Michigan. Green Lake's surrounding 240.6 km^2 watershed is primarily agriculture (63%), with corn (50% of the cropped area) and soybean (18%) as the most prevalent crops (USDA National Agricultural Statistics Service, 2016; Figure 1-1).

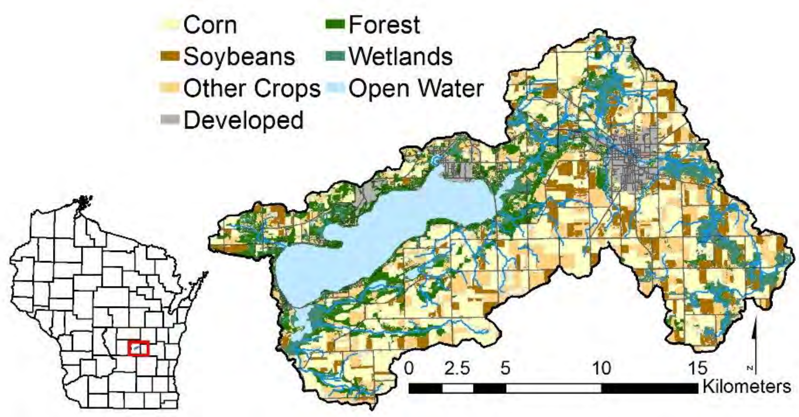

Figure 1-1: Green Lake, in central Wisconsin, has a surface area of 30.3 km^2. The surrounding 240.6 km^2 watershed is primarily agriculture, with the predominant crops of corn and soybeans.

Green Lake has a rich history of recreation and scientific study. The lake attracts many seasonal visitors and is especially popular for recreational boating and fishing. The two-story fishery of both cold- and warm-water species makes the lake a destination for central Wisconsin anglers. Scientific exploration of the lake dates back more than 120 years and includes documentation of deep water crustacea diversity and depth and temperature profiles from the 1890's (Marsh, 1891a; 1891b).

Shoreline residents identify closely with the lake and seek to maintain its high quality. As concerns of excess sediment and nutrients, low dissolved oxygen concentrations, urbanization, climate change, and aquatic invasive species have grown, members of the lake community have worked towards creating a comprehensive management plan. The Green Lake management team is comprised of the local Green Lake Sanitary District (GLSD), the local non-profit Green Lake Association (GLA), representatives from the Green Lake and Fond du Lac County Land Conservation Departments, staff from the Wisconsin Department of Natural Resources (WDNR), researchers from the United States Geological Survey (USGS) and others. In 2013, the group completed a Lake Management Plan (LMP) addressing the community's concerns (Sesing *et al.*, 2015). This partnership brought together city, county, state and federal efforts pertaining to Green Lake.

The LMP addressed issues of concern, such as lake phosphorus (P) and dissolved oxygen (DO) concentrations (Sesing *et al.*, 2015). In 2014, the WDNR listed Green Lake as impaired under section 303(d) of the Clean Water Act because DO concentrations were less than 5 mg/L and thought to be caused by high P concentrations (> 0.015 mg/L). In efforts to remove Green Lake from WDNR's impaired list, the LMP is focused extensively on reducing the amount of watershed P that enters tributaries and eventually reaches the lake. P reductions are expected to improve water quality by decreasing algae growth, thereby reducing large phytoplankton decomposition events that consume DO. Adequate DO is vital in supporting a biodiverse lake ecosystem.

Before European settlement, Green Lake would have been considered an oligotrophic, or low productivity, lake (Sesing *et al.*, 2015). However, land use change within the watershed has resulted in supplemental nutrient delivery to the lake, and it is now classified as mesotrophic (moderate nutrient conditions). While Green Lake itself is a mesotrophic system, shallow wetland systems draining into the lake have become very productive, or eutrophic. These systems have received large amounts of sediment and P from the watershed, which have led to excess invasive plant and algae growth. Four of the main tributaries to the lake are listed as impaired for degraded habitat because of sediment, total suspended solids, or total P (Figure 1-2). To address these impairment issues, the LMP set the following goals: 1) within 20 years, reduce phosphorus loading to the lake by 30% and 2) within 50 years, achieve conditions such that the lake will be classified as oligotrophic (Sesing *et al.*, 2015). Efforts to attain these goals include continued water quality monitoring, modeling pollutant sources, identifying areas with large pollutant contributions, and implementing best management practices (BMP's) in pertinent areas in an effort to reduce external P inputs.

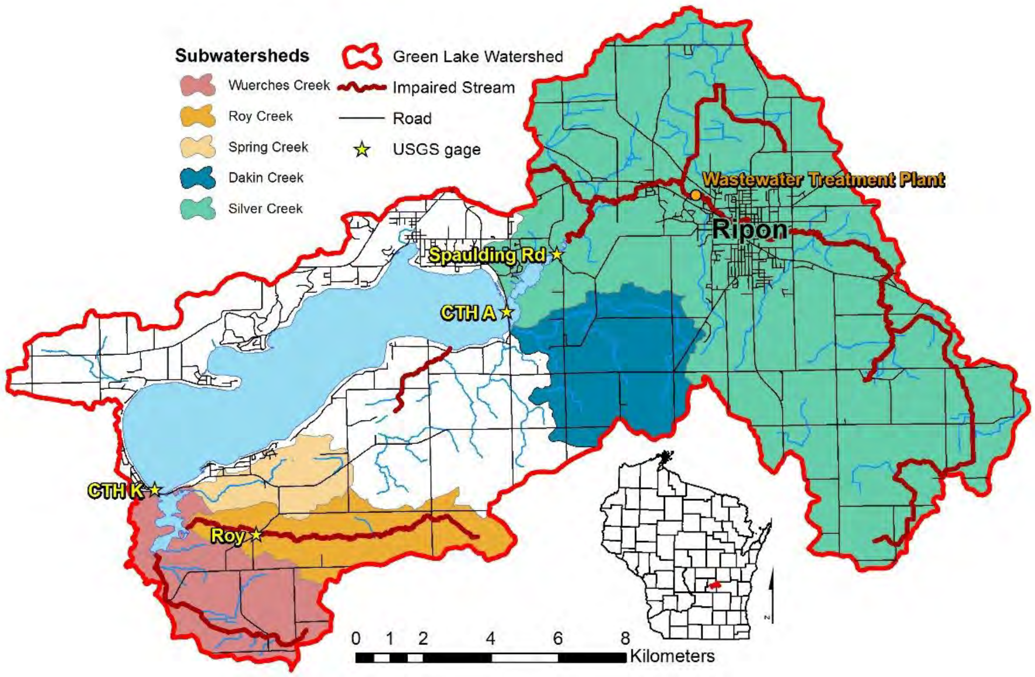

Figure 1-2: Subwatersheds of tributaries flowing through large marshes to Green Lake. Roy, Spring and Wuerches Creeks flow into County K Marsh (CKM) on the lake's southwestern end. Dakin and Silver Creek flow into Silver Creek Estuary (SCE) on the eastern end. Impaired streams are shown in dark red. Data from four USGS gages were used in this study. The Ripon Wastewater Treatment Plant (RWTP) is located upstream of SCE on Silver Creek.

Much of the Green Lake watershed drains through two wetland complexes, one on the southwestern end of the lake and the other on the eastern end (Figure 1-2). County K Marsh (CKM), on the southwestern edge, has three tributaries, two of which (Wuerches Creek and Roy Creek) are impaired. Silver Creek Estuary (SCE), on the eastern edge, has two tributaries, the larger of which (Silver Creek) is impaired. Understanding P loading through these two marsh systems is critical for evaluating overall loading to the lake. Watershed land use and anthropogenic activities contributing to nonpoint source pollution and point loads shape the water quality in tributary streams reaching these marshes. Ecosystem characteristics of the marshes themselves also influence the amount of P that enters downstream Green Lake.

Subwatershed Characteristics

Silver Creek is the largest tributary to Green Lake and has a watershed of 123 km^2 (Figure 1-3). Agriculture is the predominant land use (63%); however, the Silver Creek watershed has the largest amount of developed area (11%) compared to the other subwatersheds of SCE and CKM (Figure 1-4). The City of Ripon (population ~ 8,000) is located along Silver Creek, and its wastewater treatment plant discharges into the stream (Figure 1-5). Corn (32%) and soybean (15%) are the two most prevalent crops. Wetlands (17%), located predominantly along Silver Creek and its tributaries, are also prevalent.

Roy, Wuerches, and Dakin creek subwatersheds have similar areas (range of 14.2 – 18.4 km^2; Figure 1-3). The Dakin Creek subwatershed has the highest proportion of agriculture (85%), with corn (43%) and soybean (13%) as the most dominant crops (Figure 1-4). Roy and Wuerches subwatersheds have similar proportions of agriculture (75% and 72%, respectively), but Roy has a higher percentage of forest (17%) compared to Wuerches (11%). Corn is the most prevalent crop in the Roy Creek subwatershed (49% of the area). Corn covers 43% of the Wuerches watershed and is the dominant land use, with wetlands (13%) the second most dominant land use.

Spring Creek's watershed is the smallest of the subwatersheds draining through the marshes (7 km^2; Figure 1-3) and has the smallest percentage of agriculture (45%) and the largest percentage of forest (31%; Figure 1-4). Corn, the dominant crop, covers 27% of the watershed. Spring Creek flows from Spring Lake, an unimpaired spring-fed lake of 25 ha (Figure 1-6).

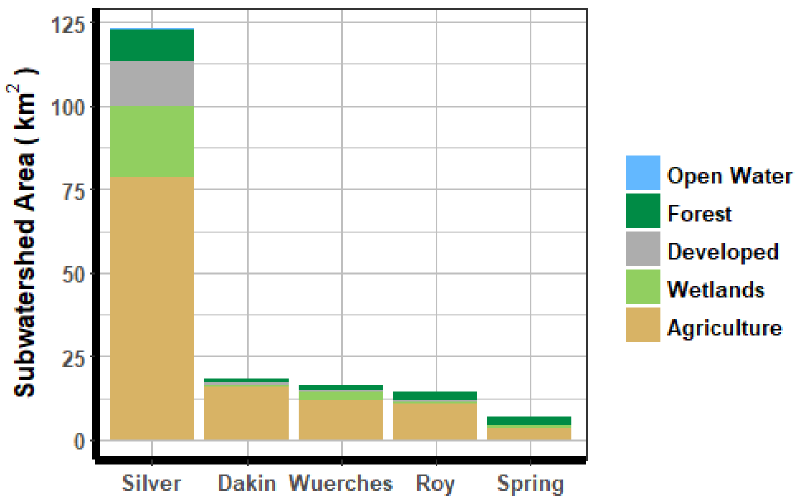

Figure 1-3: Land use area within each subwatershed for tributaries draining to SCE (Silver and Dakin) and CKM (Wuerches, Roy, and Spring). Agriculture is the predominant land use.

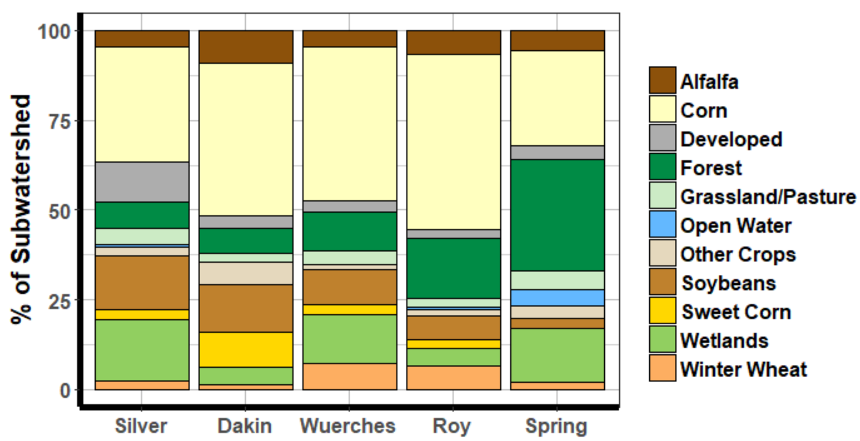

Figure 1-4: Percentage of land use within each subwatershed of SCE (Silver and Dakin) and CKM (Wuerches, Roy, and Spring).

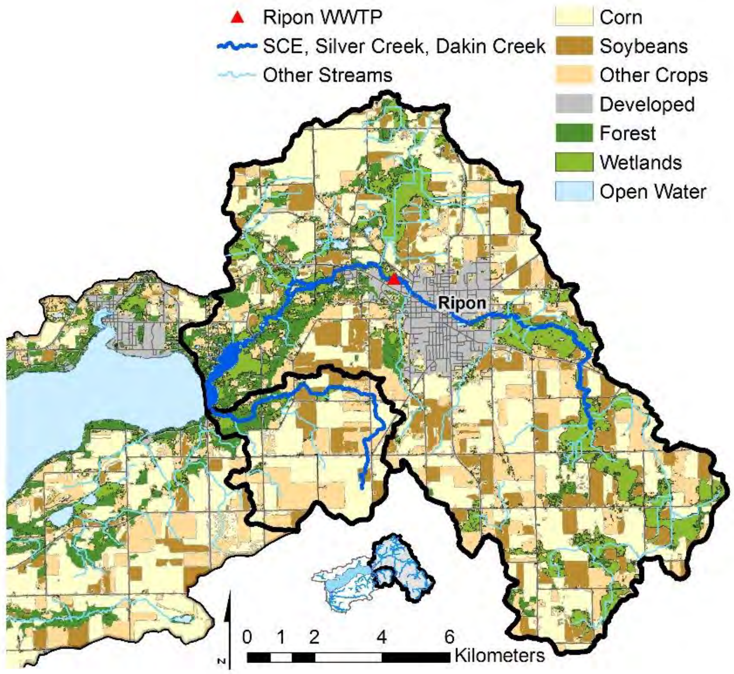

Figure 1-5: Silver Creek (northern black outline) and Dakin Creek (southern black outline) subwatershed land use. Ripon Wastewater Treatment Plant (WWTP) discharges into Silver Creek.

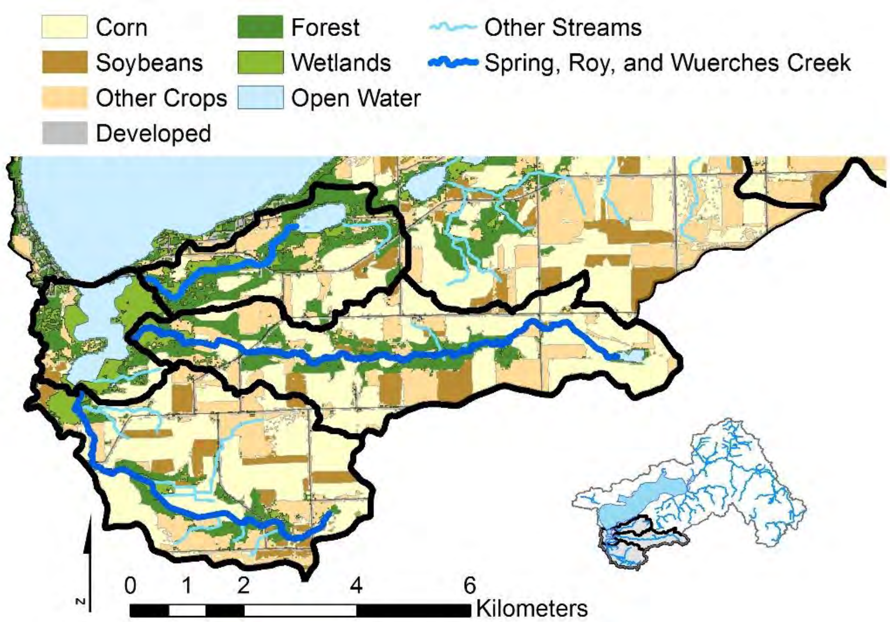

Figure 1-6: Land use in Spring (northern), Roy (central), and Wuerches (southern) creek subwatersheds of County K Marsh.

Most tributaries of SCE and CKM have similar pollutants and impairments. Silver Creek has been listed as impaired for sediments/total suspended solids since 1998 and for phosphorus since 2018 (Wisconsin Department of Natural Resources, 2018a). These pollutants contribute to elevated water temperature and degraded habitat within the creek. Upstream of the City of Ripon, Silver Creek is classified as a Class II trout stream. Roy Creek has been listed as impaired for sediment/total suspended solids since 2002 and for phosphorus since 2014 (Wisconsin Department of Natural Resources, 2017). These impairments lead to degraded habitat and biological communities within the stream. Wuerches Creek has been listed as impaired for sediments/total suspended solids since 1998 and phosphorus since 2008 (Wisconsin Department of Natural Resources, 2018b). These pollutants lead to low DO, elevated water temperatures, and degraded habitat in Wuerches Creek.

Total P (TP) and total suspended solids (TSS) loading from subwatersheds draining to Green Lake were modeled in 2015 using the Soil and Water Assessment Tool (SWAT). The SWAT model estimated that on an average annual basis, approximately 50% of the TP loading to Green Lake comes from the SCE watershed and 25% from the CKM watershed (Baumgart, 2015). Approximately 40% and 30% of TSS comes from SCE and CKM watersheds, respectively. While the SCE watershed contributes a larger portion of TSS and TP loading to Green Lake than the CKM watershed, Wuerches Creek (a subwatershed of CKM) was estimated to contribute the highest TSS and TP loads per unit area. Wuerches Creek was ranked highest of the 26 subwatersheds for TP export per hectare and second for TSS export per hectare (Figure 1-7 A and B). Roy Creek (also part of CKM) was ranked fifth for TP export per hectare. Dakin Creek, within the SCE watershed, was ranked eighth of 26 subwatersheds for TSS export per hectare and sixth for TP export per hectare. Silver Creek had four subwatersheds ranked within the top ten for TSS export per hectare, and two subwatersheds were ranked within the top ten for TP export per hectare.

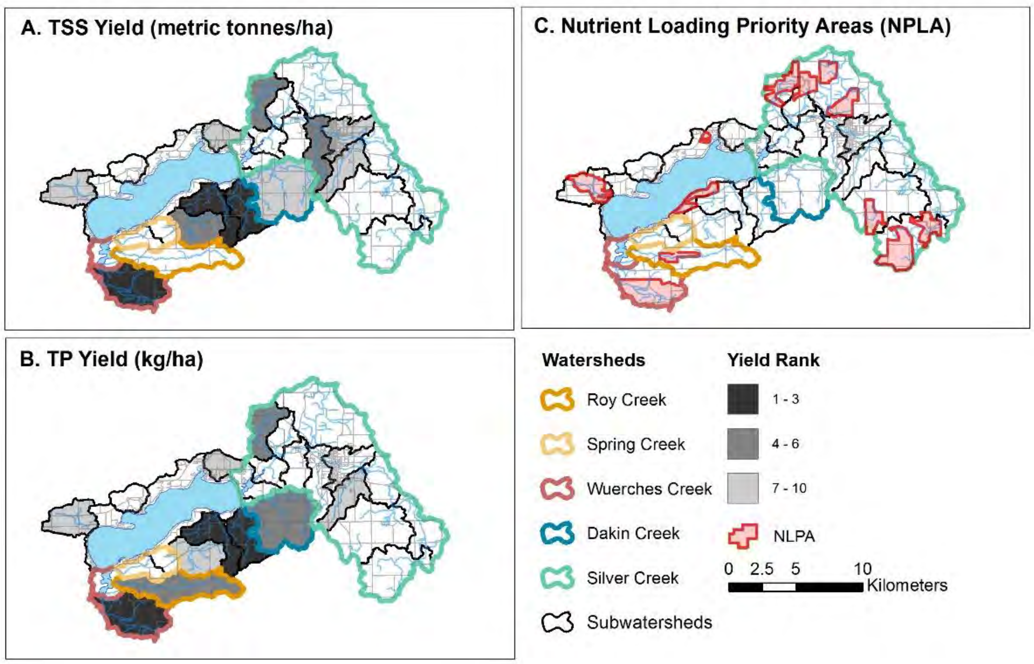

Figure 1-7: Subwatersheds ranked within the top 10 for (A) TSS and (B) TP export per hectare in SWAT model. (C) Nutrient loading priority areas identified through EVAAL model.

An Erosion Vulnerability Assessment for Agricultural Lands (EVAAL) was conducted in 2016 to aid in defining nutrient loading priority areas (NLPA; Delta Institute, 2016). These areas were determined by considering where TP and sediment loading was likely to be high and where few BMP’s had previously been implemented. Of the 12 NLPAs identified, seven were in the Silver Creek watershed (Figure 1-7 C). Two NLPAs were in the CKM watershed; one each in Wuerches and Roy subwatersheds. Recommendations for NLPAs included implementing agricultural BMP’s of cover crops, critical area planting (establishing permanent vegetation in areas vulnerable to erosion), no till and strip till, nutrient management plans, mulch till, filter strips, grassed waterways, and streambank and shoreline protection.

Across Wisconsin, streams with prevalent agriculture in the watershed often have high concentrations of P. Seventy-four percent of Wisconsin watersheds had streams with average annual TP exceeding 0.1 mg/L, and of those, 63% of the watersheds had extensive (> 36%) agricultural land use (Liu *et al.*, 2009). Wisconsin agricultural watersheds export more TP, TN, and TSS on an annual average basis compared to forested watersheds and display more interannual variability; in years with high precipitation, waters in agricultural basins had particularly high nutrient and sediment loading (Powers *et al.*, 2014). Distributed practices to reduce TP, TN and TSS export at the source within the upland watershed are important for protecting the downstream waterbodies. Powers *et al.* also report, though, that TN, TP and TSS yields were lower in watersheds with greater relative area of lakes or reservoirs and that presence of these waterbodies was associated with reduced interannual variability in TP and TSS export. Since Green Lake's watershed is primarily agriculture, retaining excess nutrients in the upland watershed or in upstream waterbodies (e.g. CKM or SCE) may be a valuable strategy for reducing loads to the lake.

Limiting Nutrients

While colloquially called estuaries, SCE and CKM can provide ecosystem services similar to shallow lakes and marshes. Nutrient inputs, particularly N and P, to these systems influence the dominant biological community and the amount of primary production. Some studies have shown that P, not N, controls the growth rate and biomass of algae in freshwater systems (e.g. Guildford and Hecky, 2000), while others have documented positive relationships between P and algal biomass and between N and algal biomass, though the relationship was stronger for P (e.g. Søndergaard *et al.*, 2017). The historic paradigm of N controlling biological growth in marine systems and P controlling in terrestrial systems has been expanded to include the importance of both nutrients in both systems (Lewis and Wurtsbaugh, 2008). While some still propose that reductions in P inputs remain the only scientifically proven method for addressing eutrophication in lakes (Schindler, 2012), others stress the need to address both P and N because inland waters high in N eventually flow downstream to reach N limited marine systems (Paerl, 2009).

Nutrient enrichment of waters in streams, wetlands, and lakes influence the resulting biological communities. Increasing stream N and P concentrations have been linked to lower fish and macroinvertebrate community assemblage metrics, suggesting the importance of nutrients in determining not only the growth of primary producers but also the make-up of secondary consumers in wadeable streams (Robertson *et al.*, 2006). Biotic indices in both wadeable and nonwadeable Wisconsin streams have been shown to correlate with TP and TN concentrations; however, the influence of TP is stronger than TN (Robertson *et al.*, 2006, 2008). In Wisconsin, TP concentration criteria for small streams is 0.075 mg/L (Johnson *et al.*, 2011). While there is currently no statewide standard for TN, the 25th percentile TN concentration (a percentile often used for developing nutrient criteria) in the Southeast Wisconsin Till Plains region is 2.02 mg/L (Wang *et al.*, 2007). Since additions of N and P to freshwater systems are correlated with increased primary production, it is valuable to understand the role of both nutrients in efforts to control eutrophication (Dodds and Smith, 2016; Søndergaard *et al.*, 2017).

Alternative Equilibria

Shallow lakes are characterized by their equilibrium state, which can be macrophyte or phytoplankton dominated (Scheffer *et al.*, 1993). A macrophyte (aquatic plant) regime is typically associated with clear water and suitable habitat for desirable fish and invertebrates. A phytoplankton (photosynthesizing algae) regime is associated with turbid conditions, nuisance algal blooms, and the subsequent oxygen depletion that can threaten the greater biological community. The equilibrium shifts from macrophyte to phytoplankton dominance as nutrients in the system increases, but under moderate nutrient concentrations, a system can exist in either equilibrium. In nutrient-rich shallow waters, such as CKM and SCE, lake managers have focused their efforts on returning to and maintaining a clear water state. This can be achieved by reducing nutrient inputs to the system and/or inducing another perturbation (e.g. reducing sediment disturbance) so that the system shifts away from high turbidity and phytoplankton dominance.

Alternative equilibria are possible under the same nutrient conditions because both regimes have positive feedback mechanisms influencing turbidity (Scheffer *et al.*, 1993). In a phytoplankton dominated (turbid) state, nutrients in the water column are consumed by phytoplankton, which increases their density and reduces light penetration. In a macrophyte-dominated (clear) system, vegetation utilize nutrients in the water column, reducing the quantity available for phytoplankton. This leads to increased water clarity and allows macrophytes to establish in deeper water and consume more nutrients. Vegetation can also stabilize sediment, which reduces turbidity and internal loading of nutrients, and provides habitat and protection for zooplankton that consume phytoplankton. If nutrient loading to a phytoplankton dominated system is reduced to intermediate levels, the system will only switch to macrophyte dominance if turbidity is sufficiently low. At high P concentrations, phytoplankton might have an advantage in nutrient utilization, but in shallow lakes (less than 1.9 m mean depth), macrophytes may remain dominant providing adequate light penetration (Mjelde and Faafeng, 1997). If substantial nutrient differences occur between the two equilibria, however, they should not be considered as multiple stables states because they are dominant in distinct environmental conditions (Capon *et al.*, 2015).

The introduction of carp is often associated with a system transitioning from a clear to turbid state (Bajer *et al.*, 2009). Carp degrade water quality by increasing turbidity, P and ammonia (Lougheed *et al.*, 1998; Weber and Brown, 2009; Fischer *et al.*, 2013). Carp are also associated with decreased macrophytes and benthic invertebrates. Few species of macrophytes, especially submergent ones, are able to persist under conditions of high turbidity and physical disturbance (Bajer *et al.*, 2009). Carp influence water bodies through bottom-up processes of increasing nutrient availability and turbidity and through top-down processes of benthic invertebrate predation (Weber and Brown, 2009). Carp mature early, grow quickly, and are tolerant of degraded water quality, allowing the species to become a dominant force in maintaining many turbid systems.

One management strategy to reduce turbidity is to decrease populations of bottom feeding fish (*i.e.* biomanipulation), such as carp (*Cyprinus carpio;* Scheffer *et al.*, 1993). Removing benthivorous carp can also reduce nutrient concentrations by decreasing excretion and activity that resuspends sediment and associated nutrients (Schrage and Downing, 2004). Fewer benthivorous fish in a waterbody also reduces the physical disturbance of macrophytes. Once turbidity decreases and favorable conditions exist, macrophyte expansion can occur quickly.

The accumulation and release of P from sediments influences the regulation of P availability to autotrophic organisms in waterbodies. Sediments in shallow eutrophic systems show seasonal patterns in P retention, often with positive P retention in the winter but negative retention in the summer (Søndergaard *et al.*, 2013). Temperature, redox conditions, pH, iron content, microbial activity, and macrophyte presence all influence the amount of P sediments can bind (Søndergaard *et al.*, 2003). Internal loading occurs when P previously held by sediments is released into the water column. This can also occur through wind-driven and other physical resuspension of sediments. Both shallow lakes and marshes exhibit varying temporal patterns of P retention and delayed responses to external P reductions (Søndergaard *et al.*, 2002; Fisher and Acreman, 2004; Robertson *et al.*, 2018).

Macrophytes consist of emergent, floating-leaf, free-floating, and submerged communities (Borman *et al.*, 2013). Emergent macrophytes often form clumps in shallow water and spread through a network of roots and rhizomes. Floating-leaf species occupy deeper waters, have rhizomes that can create dense networks and have leaves attached by long stalks that float on the surface. Free-floating macrophytes are small plants that float on the water's surface with roots dangling into the water column. Unattached to the sediment, these macrophytes drift with the wind and current. Submerged macrophytes, in deeper water, are most affected by light availability through the water column, can have both submersed and floating leaves, and can be rooted or unattached to the sediment.

Macrophytes play an important role in regulating P and N within a shallow water body through processes that affect both the water column and sediment. During the growing season, submerged macrophytes remove P and N from the water column and sediments in order to accumulate nutrients in their biomass (Nichols and Keeney, 1976; Barko and Smart, 1980; Gao *et al.*, 2009). Excretory losses from the shoots are not large, but P and N are released upon decay at the end of the season. Macrophyte senescence releases P and N into the water, increasing nutrient concentrations. Depending on the species, this release can happen early in the summer (*Potamogeton crispus*; curly leaf pondweed; Gao *et al.*, 2009), later in the summer

(*Myriophyllum spicatum*; Eurasian water milfoil; Landers, 1982), or in the fall (*Nymphaea odorata*; white water lily*;* Moore et al., 1994). *M. spicatum* is estimated to release 70% of P and 50% of N upon senescence (Landers, 1982). Macrophytes can also impact the amount of oxygen in the water column. Dense stands of floating-leaved macrophytes can reduce aeration, spurring anaerobic conditions and internal P release from sediments (Moore *et al.*, 1994). Therefore, macrophyte dominance is not always linked to improved water quality.

Carp densities and equilibrium state impact nutrient retention of shallow water bodies. Green Lake's history of abundant carp and high nutrient loading from the agricultural watershed has likely impacted the ability of its surrounding shallow areas (e.g. CKM and SCE) to provide desired TP, TN and TSS retention upstream of the lake. Therefore, lake managers continue to work towards nutrient reductions within the watershed, carp removal, and macrophytes dominance in CKM and SCE.

Marsh Characteristics

Silver Creek Estuary (SCE)

SCE has a surface area of 86 ha and is generally less than 1.2 m deep (Butterfield *et al.*, 2015). At the outlet of SCE (at CTH A) flow and P and suspended sediment (SS) loads have been monitored since 1987 by the USGS (Figure 1-2). At the inlet of Silver Creek (Spaulding Rd), the primary tributary, the USGS has monitored flow and P and SS loads since the end of 2011. Before a concerted effort around 2002 to exclude carp from SCE, the SCE was phytoplankton dominated and turbid. Since the reduction in carp density and a decrease in P from the upstream Ripon Wastewater Treatment Plant (RWTP), the SCE equilibrium has shifted to macrophyte dominance and supports prolific submergent, floating-leaf, and free-floating communities (Figure 1-8).

Figure 1-8: Submergent and free-floating macrophytes in SCE. June 15, 2017

A point-intercept rake plant survey was conducted in 2006 and macrophytes were recorded at 97% of sample locations. The macrophytes that occurred most frequently were *Ceratophyllum demersum* (coontail, 82%), *Lemna spp*. (duckweed, 78%), and invasive *M. spicatum* (77%; Sesing, 2006). Other common species included *N. odorata* (59%) and *Wolffia spp*. (watermeal, 48%). *M. spicatum* and *C. demersum* are submergent macrophytes, *N. odorata* is a floating-leaf macrophyte, and *Lemna spp.* and *Wolffia spp*. are free-floating macrophytes.

The community of aquatic plants was mapped in 2013 and floating-leafed macrophytes covered 44% of SCE and emergent species covered 3% (Butterfield *et al.*, 2015). Species were predominantly *N. odorata* (floating-leaf) and *Typha spp*. (cattail, emergent). In a July 2014 point-intercept rake survey, the most frequently encountered macrophyte was *C. demersum. Elodea canadensis* (common waterweed) was also prevalent, and *Myriophyllum sibiricum X spicatum* (hybrid water milfoil) was not as prevalent (7%). Overall, 87% of the sample points contained macrophytes.

The dominant species in SCE utilize a variety of strategies to obtain nutrients and overwinter (Borman *et al.*, 2013). *N. odorata* has rhizomes that overwinter in the sediment. New leaves emerge in the spring and eventually float on the water's surface. *E. canadensis* sprouts from shallow roots, overwinters (photosynthesizing at a reduced rate), and new shoots develop in spring. *M. spicatum* forms dense canopies which grow fast in low light and warm temperatures. The species sprouts from roots that overwinter and can start growing early in the spring, shading out native species. *Lemna spp.* and *Wolffia spp* are free-floating species with hanging roots to obtain their nutrient requirements. They drift with the current and are not dependent on sediment type, water depth, or water clarity. They overwinter as buds that drop to the sediment in the winter and rise to the surface in the spring. With sufficient nutrients, they can grow prolifically and be skimmed off the water's surface for harvesting. *C. demersum* is a submerged plant that lacks true roots, obtaining its required nutrients from the water column. The plant overwinters, photosynthesizing at a reduced rate and growth begins in the spring.

Previous SCE vegetation surveys were conducted in mid-summer, which is too late to monitor maximum presence of the invasive plant *P. crispus* (Butterfield et al., 2015). In early summer, this is a prevalent plant in SCE (Figure 1-9). *P. crispus* grows from a rhizome and sprouts from turions (Borman *et al.*, 2013). This macrophyte grows throughout the winter and sprouts summer leaves quickly in May. It senesces in mid-July, but before summer dormancy, turions are dropped to the sediment. These sprout in September and continue to grow during winter.

Figure 1-9: P. crispus in SCE. May 16, 2017.

County K Marsh (CKM)

CKM has a surface area of 106 ha and an average depth of 0.62 m (Cunningham, 2015). The outlet of CKM (at CTH K) has been monitored by the USGS since the end of 2012 for flow and P and SS loads (Figure 1-2). At the same time, the USGS installed a gage on one of the tributaries (Roy Creek) about a mile upstream of the marsh to monitor stream flow. No tributary inlets at the marsh have been continuously monitored.

CKM is currently a phytoplankton dominated system; however, restoration efforts to shift the equilibrium are underway. The 2014 mapping of the CKM aquatic plant community documented 2% cover of emergent species and 1.6% cover of floating leaf macrophytes (Butterfield *et al.*, 2015). Species included *N. odorata* and *Typha spp.* During a 2014 point-intercept rake survey, vegetation was found at two locations (2%). Managers of Green Lake prioritized CKM restoration and initiated carp removal and installation of native propagules. Carp are harvested by commercial fishers, fyke nets, and electroshocking. In May 2016, 40,000 pounds of carp were harvested commercially (Green Lake Sanitary District, 2016). A physical barrier at the CTH K bridge was first installed in 2015 and improved in 2016. In the summer of

2016, the barrier prevented most carp from reaching their desired spawning area, and large numbers of carp congregated on the lake side of the barrier. Fatigued and stressed from attempting to pass through the barrier, many carp died which lead to abundant floating dead carp along the lake shore (Green Lake Sanitary District, 2017). In summer 2017, more frequent and aggressive carp harvesting decreased the number of floating dead carp, and 27,500 native plant propagules - *Stuckenia pectinata* (sago pondweed), *Vallisneria americana* (wild celery) and *Sagittaria latifolia* (common arrowhead) - were planted throughout CKM in twelve exclosures (Green Lake Association, 2017a; Green Lake Association, 2017b; Figure 1-9).

Figure 1-10: S. pectinata in CKM exclosure. July 13, 2017.

Research Objectives

Long term monitoring has been conducted in CKM and SCE; however, a comprehensive analysis has not been conducted to assess their relative and seasonal impacts on P and sediment loading to Green Lake. Processes affecting nutrient retention in these systems are complex and depend on flow and nutrient inputs, as well as internal components, such as equilibrium state and sediment composition. The goal of this research was to compare water column and sediment conditions within these two wetland systems and to evaluate each marsh’s impact on water quality entering the lake. Better understanding of water quality within SCE and CKM can support previous and future modeling efforts and guide actions for reducing nutrients reaching Green Lake.

The first portion of this research (Chapter 2) focused on evaluation of long-term trends observed at SCE. The objectives were to:

1. Quantify changes in total phosphorus (TP) and suspended sediment (SS) loads from SCE coinciding with its transition from phytoplankton (turbid; 1987 – 2002) to macrophyte (clear; 2006 – 2017) dominance,
2. Evaluate whether SCE has served as a TP and SS sink or source since 2012, and
3. Quantify changes in TP, total dissolved phosphorus (TDP), SS, and total nitrogen (TN) concentrations from the inlet to outlet of SCE during the period of macrophyte dominance (2006 – 2017).

The second portion of this research (Chapter 3) compared conditions within the two marshes. The objectives were to:

1. Quantify differences in total phosphorus (TP) and suspended sediment (SS) loads from the two marshes since 2012,
2. Quantify and compare spatial and temporal changes in TP, total suspended solids (TSS), and TN concentrations and loads within the two marshes during the growing seasons of 2016 and 2017,
3. Evaluate whether each marsh acts as a sink or source of TP, TN, and TSS, and
4. Quantify P variation within and the potential for P release from sediments within the two marshes.

Chapter 2 Water Quality Comparison of SCE Before and After Macrophyte Dominance

Introduction

Silver Creek, the largest tributary to Green Lake, enters the lake through a marsh on the lake's eastern edge (Figure 2-1). Understanding the water quality patterns that occur in the marsh, and changes in the subsequent loading to Green Lake, may help explain water quality changes observed within the lake. Therefore, characterizing sediment and phosphorus loads from Silver Creek has been a priority of lake monitoring efforts since 1987 (Figure 2-2). Dakin Creek, a smaller tributary, also drains through the marsh.

Over the past 30 years, the Silver Creek Estuary (SCE) ecosystem has changed substantially. In the 1990's, the marsh was turbid and carp-dominated. Carp are typically associated with high turbidity and low aquatic plant diversity because their foraging disturbs the sediment (Lougheed et al. 1998; Bajer et al. 2009; Bajer & Sorensen 2015). A 1998 aquatic survey conducted by the Green Lake Sanitary District (GLSD) and the Wisconsin Department of Natural Resources (WDNR), showed no vegetation within the marsh. Local residents wanted improved water clarity, so in 2002, the GLSD built a gate to block spawning carp from entering the shallow marsh.

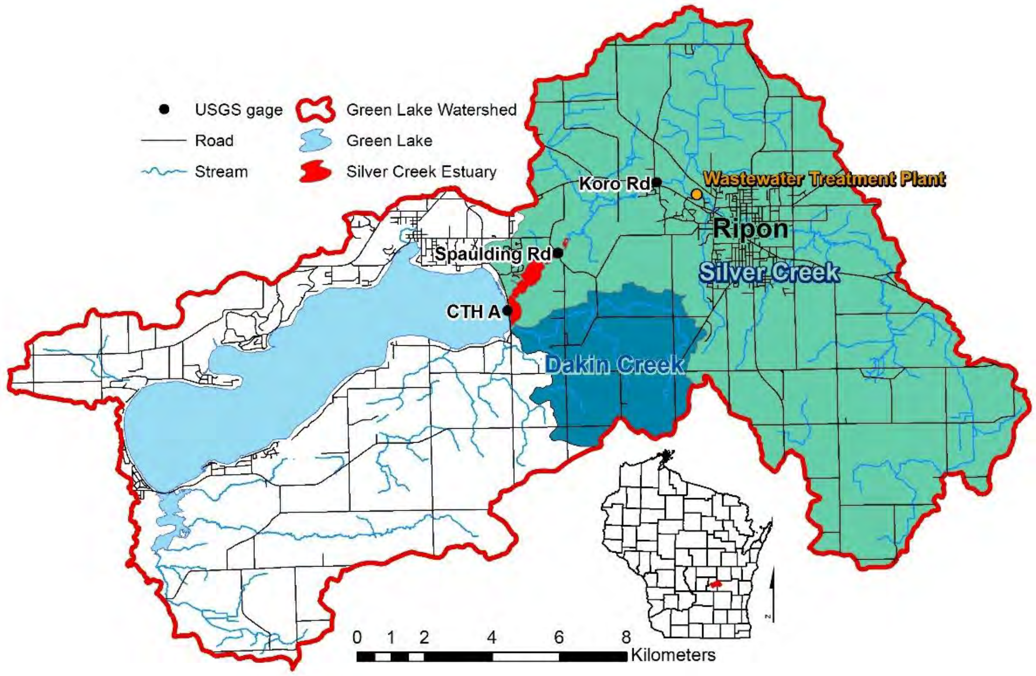

Figure 2-1: Subwatersheds draining to Silver Creek Estuary. Dakin Creek (blue subwatershed) and Silver Creek (teal subwatershed) both drain through the marsh which is highlighted in red. USGS gages are located at the inlet (Spaulding Rd) and at the outlet (CTH A). Historically (1987-1996), there was a gage at Koro Rd. Ripon Wastewater Treatment Plant is upstream of the gages and discharges into Silver Creek.

The GLSD expected that this change (carp removal from the ecosystem) would shift the marsh from a phytoplankton (turbid) stable state to a stable state dominated by macrophytes (clear with aquatic plants). Under intermediate nutrient conditions, shallow lakes can exist in one of two stable states – clear or turbid (alternative equilibria theory; Scheffer et al. 1993). The clear state is dominated by macrophytes, and the turbid state by phytoplankton (photosynthesizing bacteria and algae). Carp removal can spur this state transition because of the impact of their benthic foraging habits and excretion. Reducing suspended sediment and water column nutrients can create conditions, such as increased water clarity, that favor macrophytes (Weber and Brown, 2009).

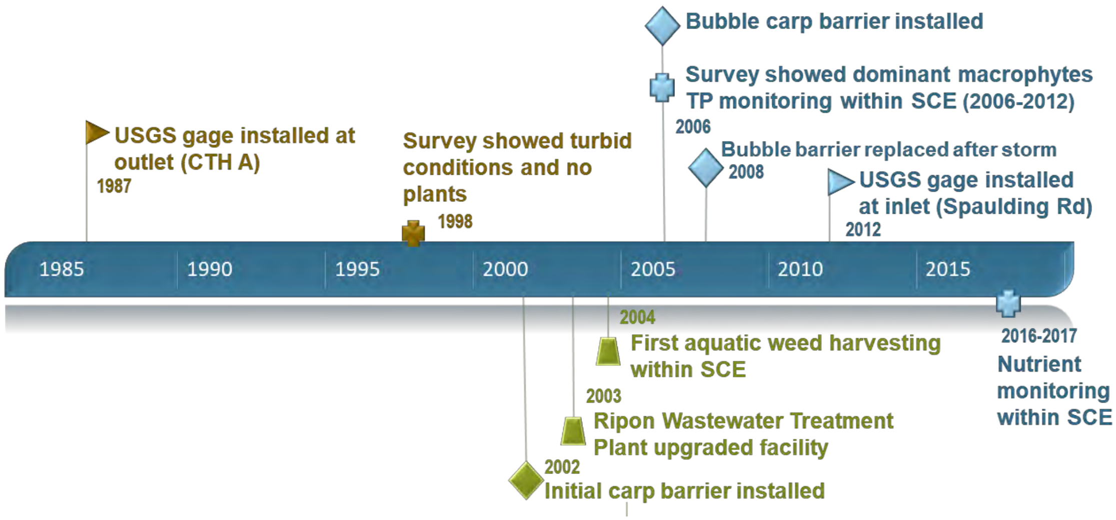

Figure 2-2: Timeline of monitoring and restoration management within Silver Creek Estuary from 1987-2017.

The original carp barrier blocked entry during the spawning season (May 1st - July 4th) and rolled up similar to a garage door to allow boat crossing. This design malfunctioned frequently and required daily debris removal. Therefore, a bubble barrier was installed in 2006 (Figure 2-2). The original gate, however, was successful in creating conditions within the marsh conducive for macrophyte colonization. By 2004, the GLSD began harvesting aquatic plants to create lanes for recreational boaters, and by 2006, they harvested 98 metric tonnes of vegetation from the marsh. A survey in July and August of 2006 showed macrophytes at 97% of 195 sample locations. *Ceratopyllum demersum* (coontail), *Lemna spp.* (duckweed*),* *Myriophyllum spicatum* (Eurasian water milfoil), and *Nymphaea odorata* (white water lily) all occurred at more than 50% of the sites. The marsh has since remained macrophyte-dominated, and the bubble barrier has continued to reduce carp entrance from the lake into the marsh during each spawning season. After a large storm in 2008 changed the morphology of the marsh outlet, the GLSD installed a redesigned bubble barrier that has been in place for the last decade (Figure 2-3).

Figure 2-3: Bubble barrier at CTH A bridge where Silver Creek Estuary enters Green Lake. Bubbles block fish entrance during carp spawning season, May-July. Photo taken by Stephanie Prellwitz.

Over the past 30 years, there have been changes in land use within the SCE watershed. Throughout this period, the SCE watershed has remained predominantly agricultural; however, the agricultural area has decreased slightly. According to land use data from the Wisconsin Initiative for Statewide Cooperation on Landscape Analysis and Data (WISCLAND) project collected in 1991-1993, the Dakin Creek subwatershed was 90% agricultural and the Silver Creek subwatershed was 77% agricultural (Wisconsin Department of Natural Resources, 2002). In 2016, Dakin Creek's subwatershed was 85% agricultural land use and Silver Creek's subwatershed was 64% (USDA National Agricultural Statistics Service, 2016). However, the proportion of corn increased within both watersheds; Dakin Creek's subwatershed increased from 26% to 43% corn cover and Silver Creek's subwatershed increased from 25% to 32% (Figure 2-4). Urban development increased from 6% to 11% in the Silver Creek subwatershed.

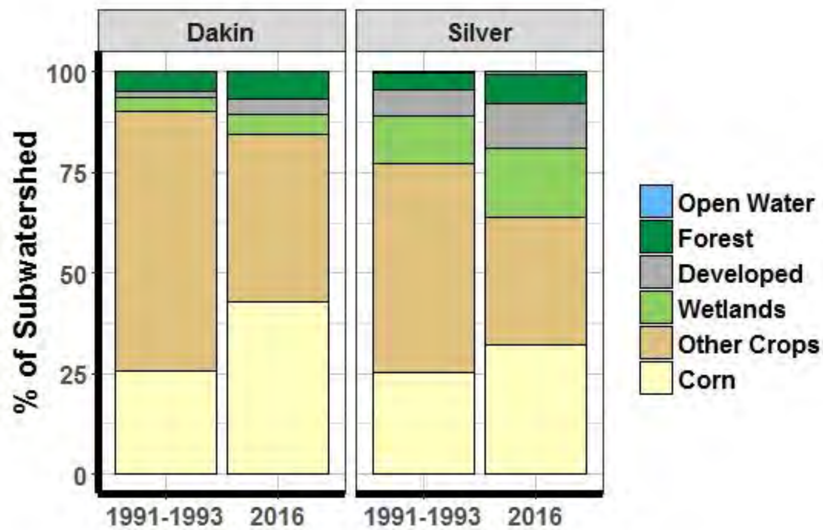

Figure 2-4: Land use in the Dakin and the Silver Creek subwatersheds in 1991-1993 (Wisconsin Department of Natural Resources, 2002) and 2016 (USDA National Agricultural Statistics Service, 2016).

The biological transition of the SCE marsh was assumed to be accompanied by a reduction in sediment and phosphorus loading to Green Lake; however, lake managers lacked a comprehensive water quality analysis. The 2002 carp exclusion also occurred just before a February 2003 upgrade at the Ripon Wastewater Treatment Plant (RWTP), an upstream TP point source (Figure 2-2; Ripon 2018). This upgrade reduced the amount of phosphorus discharged into Silver Creek, which could have likewise impacted the equilibrium of SCE 6 km downstream (Figure 2-1). While there is obvious evidence that the stable state of SCE shifted, the impact of this transition on the quality of water reaching Green Lake has not been evaluated in detail nor has the relative impact of restoration (carp exclusion and macrophyte establishment) versus RWTP upgrade.

The upgraded RWTP is designed for an average flow of 1.8 million gallons per day (MGD; 0.079 m^3/s) and has anoxic mixing zones, oxidations ditches, clarifiers, and tertiary sand filters, with UV disinfection concluding the treatment process (Sachs, 2017). Sludge resulting from the treatment process is land applied. Over the summer, sludge is injected into available land within the Silver Creek watershed. In the winter, the sludge is exported out of the Green Lake watershed. As a WDNR permitted point source, the plant must keep detailed records of its operation. The RWTP influent and effluent are monitored by collecting 24-hour flow proportional composite samples used for daily water quality analysis. The plant's permit limit for total phosphorus (TP) is a monthly average of 1.0 mg/L.

Changes upstream of the SCE outlet were likely to impact seasonal P loading to Green Lake differently. The RWTP upgrade to oxidation (aeration) ditches and biological phosphorus removal reduced the phosphorus load into Silver Creek year round, and thus reduced the overall phosphorus load potentially reaching Green Lake. The exclusion of carp, however, focused on altering conditions within the marsh during spawning season (late spring through early summer), and subsequent macrophytes changed in-marsh conditions during the growing season. Since most of the macrophytes are dormant or only have limited winter growth, their potential impact on P reductions would most likely be observed in summer. Higher TP loads occur in spring and summer compared to winter and fall, so reductions in the high loading seasons can effectively reduce maximum TP loads to Green Lake.

Separating the impact of each management action (the treatment plant upgrade and the carp exclusion) is important to understand changes in loads reaching Green Lake following macrophyte establishment within SCE. Since the restoration and subsequent monitoring of the SCE inlet at Spaulding Rd, it is also valuable to quantify how the marsh acts in retaining nutrients and whether that is seasonally dependent. Therefore, the objectives of this study were to:

1. Quantify changes in total phosphorus (TP) and suspended sediment (SS) loads from SCE coinciding with its transition from phytoplankton (turbid; 1987 – 2002) to macrophyte (clear; 2006 – 2017) dominance,
2. Evaluate whether SCE has served as a TP and SS sink or source since 2012, and
3. Quantify changes in TP, total dissolved phosphorus (TDP), SS, and total nitrogen (TN) concentrations from the inlet to outlet of SCE during the period of macrophyte dominance (2006 – 2017).

Methods

Compilation of data

USGS Loads

Sediment and nutrient data from the outlet of SCE were used to examine temporal changes in loading. The USGS installed a monitoring station at County Trunk Highway A (CTH A, 4073468) to monitor flow, TP and SS loads at the outlet of SCE in 1987 (Figure 2-1; Table 2-1). The USGS has automated samples collected and analyzed for TP (USGS parameter 00665) and SS (USGS parameter 80154) concentrations. Water velocity data were recorded with acoustic velocity meters. Over the 30 years, the USGS collected 1,820 TP samples and 1,749 SS samples, corresponding to an average frequency of about one sample every 6 days. The USGS used velocity and stage data to calculate a daily mean flow (USGS parameter 00060) at the site. Graphical Constituent Loading Analysis System (GCLAS) software was then used with daily flows and concentration data to calculate daily mean SS (USGS parameter 80155) and TP (USGS parameter 91050) loads (Koltun *et al.*, 2006). Water velocity at the gage was measured until 2011. Loads of SS were reported from 1987-2012 and 2014-2017. TP loads were available for the entire 30-year period since installation (1987-2017).

In late 2011, the USGS installed a gage at Spaulding Road (Spaulding Rd, 4073466) to monitor stream, SS, and TP loads at the primary inlet to SCE (Figure 2-1; Table 2-1). The same previously described sample collection and data analysis process were used by the USGS at CTH A (USGS collected samples, stream velocity, and stage levels and used GCLAS to calculate loads) was used at Spaulding Rd. The USGS collected 332 TP concentration grab samples and 312 SS concentration grab samples, corresponding to an average collection frequency of about 6 days. Flow measurements at the Spaulding Rd gage were used to estimate flow at CTH A starting in 2012.

*Table 2-1: Monthly data available for flow and suspended sediment total phosphorus loads at the inlet and outlet of SCE and from the Ripon Wastewater Treatment Plant, a point discharge into Silver Creek. *Flow is effluent from plant.*

Site	Flow	Suspended Sediment Load	Total Phosphorus Load
CTH A (Outlet)	Feb 1987 - Nov 2011, estimated Dec 2011 - Sept 2017	Feb 1987 - Aug 2012, Oct 2014 - Sept 2017	Feb 1987 - Sept 2017
RWTP	Feb 1987 - Sept 2017*		Feb 1987 - Sept 2017
Spaulding Rd (Inlet)	Dec 2011 - Sept 2017	Dec 2011 - Sept 2017	Dec 2011 - Sept 2017

All available monthly average flow (cfs), SS load (tons/day), and TP load (lb/day) from both gages were downloaded from the USGS National Water Information System (NWIS; USGS 2018). Monthly average loads posted through September 2016 were USGS approved. Monthly average values from October 2016 through September 2017 were calculated using provisional daily loads from NWIS since approved monthly values were not available at the beginning of this study. All data were converted into metric unit equivalents.

Ripon Wastewater Treatment Loads

Daily plant effluent flow (MGD) and phosphorus concentration (mg/L) were obtained from the RWTP for the same period as the CTH A record (1987-2017). The RWTP records included phosphate (PO_4^{3-}) concentrations for each daily sample of effluent. Daily phosphate concentrations were reported to WDNR as TP concentrations for permit compliance. These concentrations were averaged over each month to obtain a monthly average TP concentration. The daily TP concentration was multiplied by the daily effluent flow to estimate daily TP load (kg/day) from the plant. The daily TP loads were then averaged over each month (kg/day) for comparison to USGS calculated TP loads at CTH A and Spaulding Rd. Comparison of monthly average loads (kg/day) (as opposed to total monthly loads) eliminates variability associated with months of different duration. Daily flows were converted to m^3/s and averaged over each month. RWTP data were termed “Before” or “After” the February 2003 plant upgrade.

Precipitation

Daily precipitation depths were downloaded from the National Oceanic and Atmospheric Administration’s (NOAA) Climate Data online tool (https://www.ncdc.noaa.gov/cdo-web/datatools/findstation). From 1987-2006, data were obtained for a station in the City of Ripon (Ripon 5 NE; USC00477209). A closer station was established in 2007 (Markesan; USC00475096), and data from that location were used through 2017. Both stations were outside of the Green Lake watershed, but they were the closest and most complete available precipitation datasets. Daily precipitation were averaged over each month from 1987-2017 to match the frequency of the USGS load data. Monthly precipitation from the Ripon and Markesan stations were compared to data from the national PRISM dataset (prism.oregeonstate.edu; Appendix A).

The USGS, RWTP, and precipitation data were used to examine differences in TP and SS loads before, during, and after the transition of SCE from a phytoplankton to macrophyte-dominated system. The USGS monthly TP and SS loads were divided by the average monthly flow to estimate monthly flow-weighted mean TP and SS concentrations. The CTH A data were divided into three time periods, “Before” the transition to macrophytes (1987-2001), “During” (2002-2005), and “After” (2006-2017). During restoration encompassed the time between the installation of the first carp barrier at CTH A and the first SCE survey showing macrophyte dominance. Analysis focused on the difference from before to after the restoration. All data were further divided into four seasons: winter (December – February), spring (March – May), summer (June – August), and fall (September – November). Seasons generally coincided with patterns in precipitation and biological growth. Macrophyte growth in SCE occurs mostly during summer.

Monthly average retention times for SCE were estimated over the 30-year period to assess how quickly nutrients flushed through the system. The water volume for SCE was obtained from Onterra, LLC, a lake management planning consulting group that has conducted several projects within Green Lake (Personal communication, E. Heath, Onterra, LLC) and the retention time (T_R; days) for water within the SCE was estimated as:

$$T_R = \frac{V}{Qa} \qquad (1)$$

Where: V = volume of wetland (679,648 m^3)

Q = monthly average flow at CTH A (m^3/s)

a = 86400 (s/day)

Statistical Analysis for 1987-2017 data

Statistical analyses identified significant differences between groups of data (e.g. before and after the RWTP upgrade) and indicated what variables were significantly related to other variables. This facilitated identification of changes over time or variation among seasons. Explanatory variables provided context for observed changes.

All analyses used data representing the daily average for each month, hereafter referred to as the monthly average. The effects of relevant explanatory variables (both continuous and categorical) on monthly average flow, TP and SS loads, and flow-weighted concentrations (TP and SS) from the RWTP (did not include SS data) and the SCE outlet were analyzed. The R statistical software was used to create multiple linear regression (MLR) models (Team, 2018) and the *car* package was used to perform an analysis of covariance (ANCOVA) on the models (Fox, 2018). Explanatory variables were examined for both additive effects and interactions influencing the dependent variable. An interaction between two explanatory variables occurs when a change in the value of one variable depends on the value of the other; the two variables are not independent and the effect of one is better understood when the other variable is also considered. The number of observations for each time period (Before, During or After SCE restoration or RWTP upgrade) were unequal, so the unbalanced design was statistically evaluated using type III sum of squares in the ANCOVA. The method for type III sum of squares includes all model variables in adjustments for the sum of squares so the main effects of individual variables are adjusted for interactions (Hector *et al.*, 2010). If independent variables showed significant interactions, the effects of those variables individually were not interpreted. Models initially incorporated a variety of interactions, but interactions that were not statistically significant (α = 0.05) were removed from the final model. To improve data and model residual normality so that parametric methods could be used for statistical analysis, most continuous data were transformed using natural logarithms prior to statistical analysis (Fu and Wang, 2012).

Once an appropriate MLR model was determined for a given dependent variable, significant predictors and interactions were noted. Estimated marginal means (also known as least-squares means) were then computed for the model using the *emmeans* package for R (Lenth *et al.*, 2018). The MLR was used to create a modeled dataset which were then used to estimate means for specific groups of data. Marginal means for the dependent variable were estimated for quantiles of modeled flow based on precipitation and season. The marginal means were used to compare the statistical difference between the outcome variable before and after the upgrade and restoration. Before and after estimated values were analyzed using a Tukey pairwise comparison to obtain a p value for seasonally-grouped and all data.

For RWTP data analysis, MLRs were developed to examine factors impacting effluent flow, TP concentrations, and TP load (Table 2-2). Models were also developed for precipitation, flow, TP, and SS loads and TP and SS concentrations at the CTH A outlet (Table 2-3). Explanatory variables were similarly analyzed using ANCOVAs. Where noted, variables were naturally log-transformed to improve normality of the model residuals. Significant predictors ($p < 0.05$) from each ANCOVA were identified.

Table 2-2: Variables considered in MLR for monthly average RWTP effluent flow, TP concentration and TP load. Grey boxes mark variables included as part of an interaction. X marks variables evaluated for statistical significance in final MLR. Blanks were variables not considered in final MLR. Precipitation, when considered as a predictor, was naturally log-transformed to improve normality of model residuals.

		Dependent Variable		
		Effluent at RWTP (m^3/s)	**TP concentration at RWTP (mg/L)**	**TP Load at RWTP (kg/day)**
Transformation		ln	ln	ln
Predictor	Season (Winter, Spring, Summer, Fall)			
	RWTP Upgrade (Before, After)			
	Season and RWTP upgrade interaction	x	x	x
	Precipitation (mm/day)		x	x
	Season and Precipitation (mm/day) interaction	x		
	RWTP Upgrade and Precipitation (mm/day) interaction	x		
	Effluent at RWTP (m^3/s)			x

Table 2-3: Variables considered in MLR for SCE outlet monthly average data. Grey boxes mark variables included as part of an interaction. X marks variables evaluated for statistical significance in final MLR. Blanks were variables not considered in final MLR. All concentrations are flow-weighted mean concentrations.

Dependent Variable		**Precipitation (mm/day)**	**Flow at CTH A (m^3/s)**	**TP Load at CTH A (kg/day)**	**TP concentration at CTH A (mg/L)**	**SS Load at CTH A (metric tonnes/day)**	**SS concentration at CTH A (mg/L)**
Predictor	**Transformation**	ln	ln	ln	ln	ln	ln
Season (Winter, Spring, Summer, Fall)							
SCE Restoration (Before, During, After)							
Season and SCE Restoration Interaction		x	x	x	x	x	x
RWTP Upgrade (Before, After)				x	x	x	x
TP (mg/L) at RWTP	ln				x		
TP Load (kg/day) from RWTP	ln			x			
Precipitation (mm/day)	ln						
Season and Precipitation (mm/day) Interaction	ln(Precipitation)		x	x	x	x	x
Effluent at RWTP (m^3/s)	ln						
Effluent at RWTP (m^3/s) and Season Interaction	ln(Effluent)		x				
Flow at CTH A (m^3/s)	ln						
Season and Flow at CTH A (m^3/s)	ln(Flow)			x	x	x	x
SS (mg/L) at CTH A					x		
SS Load at CTH A (metric tonnes/day)	ln			x			

SCE Inlet versus Outlet Loads

For the period when both inlet loads to and outlet loads from SCE were available (December 2011 – September 2017), the monthly average TP and SS load at the outlet (CTH A) was subtracted from the monthly average load at the inlet (Spaulding Rd). These differences were used to identify periods when SCE acted as a TP and SS sink (positive difference between Spaulding Rd minus CTH A) or source (negative difference between Spaulding Rd minus CTH A) to downstream Green Lake.

Contributions from Dakin Creek, the other input to SCE, were not estimated and not included in the models. Dakin Creek's watershed (18.4 km^2) covers 13% of the SCE watershed (141.4 km^2), so the majority of flow to the marsh is from the monitored inlet at Spaulding Rd (Figure 2-1; Figure 2-5 A). Agricultural land use in Dakin Creek's subwatershed (85%) is higher than in Silver Creek's subwatershed (64%) and developed areas in Dakin Creek's subwatershed (4%) are lower than in Silver Creek's subwatershed (11%; Figure 2-5 B). Dakin Creek's subwatershed also has a larger percentage of Other Crops (29%) compared to Silver Creek's subwatershed (17%; Figure 1-4 in Chapter 1). Sweet corn (9.7%), alfalfa (9%), and peas (4%) make up a larger percentage of area in the Dakin Creek subwatershed than in the Silver Creek watershed (2.6%, 4.7%, and 1.6%, respectively).

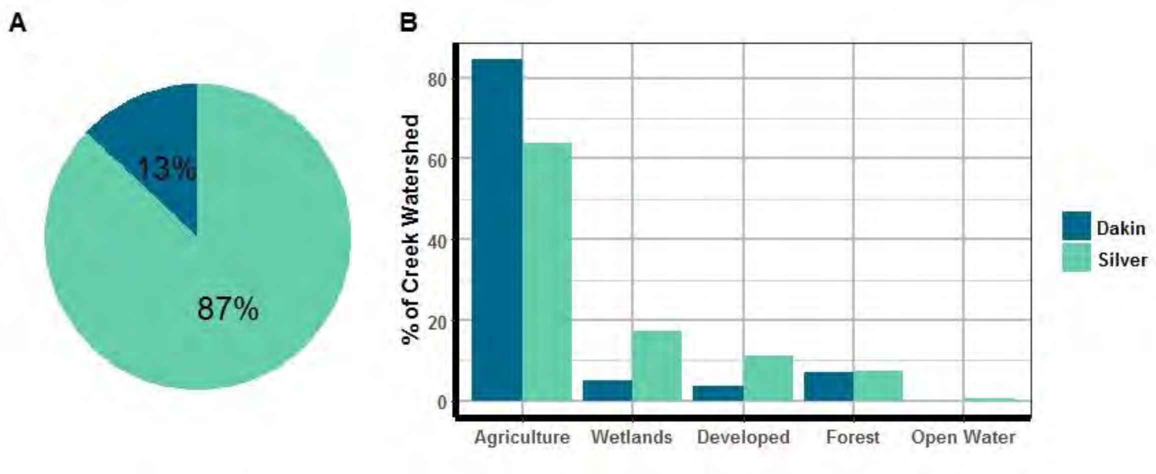

Figure 2-5: (A) Dakin and Silver Creek subwatersheds as percent area of total SCE watershed. (B) 2016 Land use percentage within Dakin and Silver creek subwatersheds.

The SS and TP flow-weighted mean concentrations were used to calculate a mean removal efficiency for each month. Removal efficiency (RE) was calculated as:

$$RE\ (in\ \%) = \frac{I_c - O_c}{I_c} \times 100 \qquad (2)$$

where: I_c = Inlet flow weighted mean concentration at Spaulding Rd (mg/L)

O_c = Outlet flow weighted mean concentration at CTH A (mg/L)

Data from Spaulding Rd and CTH A were analyzed using MLRs and ANCOVAs as previously described. Various explanatory variables were examined as significant predictors of loads, flow-weighted mean concentrations, and removal efficiencies (Table 2-4).

Table 2-4: Variables considered in MLR for SCE Spaulding Rd (inlet) and CTH A (outlet) monthly average data. Grey boxes mark variables included as part of an interaction. X marks variables evaluated for statistical significance in final MLR. Blanks were variables not considered in final MLR. All concentrations are flow-weighted mean concentrations.

		Dependent Variable					
		TP Load at Spaulding Rd and CTH A (kg/day)	**TP concentration at Spaulding Rd and CTH A (mg/L)**	**TP Removal Efficiency (%)**	**SS Load at CTH A and Spaulding Rd (metric tonnes/day)**	**SS concentration at CTH A and Spaulding Rd (mg/L)**	**SS Removal Efficiency (%)**
Predictor	**Transformation**	ln	ln		ln	ln	
Season (Winter, Spring, Summer, Fall)				x			x
Site (Spaulding Rd, CTH A)							
Season and Site Interaction		x	x		x	x	
Year (factor)		x	x	x			
TP at RWTP (mg/L)	ln		x				
TP Load from RWTP (kg/day)	ln	x					
Precipitation (mm/day)	ln			x	x	x	x
Site and Precipitation (mm/day) Interaction	ln(Precipitation)	x	x				
TP at Spaulding Rd (mg/L)				x			
SS at Spaulding Rd (mg/L)							x
Flow at Spaulding Rd (m^3/s)	ln		x	x			x
Season and Flow at Spaulding Rd (m^3/s) Interaction	ln(Flow)	x			x	x	

Sample Concentration Data

USGS sampling data

Sample concentrations within SCE over the past 11 years were analyzed. TP, total dissolved phosphorus (TDP; USGS parameter 00666), SS, and total nitrogen (TN; USGS parameter 00600) concentrations for samples collected by the USGS from 2005-2017 at CTH A, Spaulding Rd, Site 1 and Site 2 (Figure 2-6) were downloaded from the WDNR Surface Water Integrated Monitoring System (SWIMS; WDNR 2018) and NWIS.

Figure 2-6: USGS monitoring sites are located at Spaulding Rd (inlet) and CTH A (outlet) to monitor flow through Silver Creek Estuary. Water quality samples were collected at those two monitoring sites and at Sites 1, 2, and 3.

All water quality parameters were measured at Spaulding Rd (inlet) and CTH A (outlet). At interior Sites 1 and 2, only TP was measured. Table 2-5 summarizes the frequency of sampling for each parameter at each site. TP was the most frequently measured water quality parameter within SCE (n= 1137), followed by SS (n = 993), TDP (n = 114), and TN (n = 61). Water quality was sampled at CTH A on the most occasions (n = 726), followed by Spaulding Rd (n = 403). At CTH A, the largest number of samples were for TP (n = 669) and SS (n = 649). At Spaulding Rd, TP (n = 397) and SS (n = 344) were also the most frequently sampled water quality parameters.

From January 2006 – September 2011, TP and SS sampling were most frequent at CTH A (outlet). This period is referred to as Period 1. Sampling focused on capturing concentrations during changing flows, with 44% of both the TP and SS samples taken within 24 hours of another sample (Table 2-5). Monitoring during frequently changing conditions under high flows enabled the USGS to estimate SS and TP loads associated with storm events. During Period 1, CTH A (outlet) was sampled on 548 days and Spaulding Rd (inlet) was sampled on 64 days. After November 2011 (referred to as Period 2), with the installation of the Spaulding Rd gage, TP and SS sampling focused on capturing conditions at the new gage. Over 50% of the Spaulding Rd TP and SS samples were collected within 24 hours of another sample while only 5% of the CTH A TP samples and 2% of the CTH A SS samples met that condition. During Period 2, Spaulding Rd was sampled on 333 days and CTH A was sampled on 121 days. TP sampling at interior Sites 1 and 2 occurred monthly during summer from 2006 – 2012.

TN and TDP samples at Spaulding Rd and CTH A were collected at more regular intervals, with few samples (TDP: 14-17%, TN: 3-26%) collected within 24 hours of another (Table 2-5). TDP samples were collected intermittently, with no sampling in SCE between September 2009 and February 2012. TN sampling began in February 2014 and occurred approximately monthly.

Table 2-5: Grab sampling scheme for USGS water quality analyses at Spaulding Rd (Sp Rd; inlet), CTH A (outlet), and at two interior sites (Site 1 and 2) within SCE from December 2005 – October 2017.

Water Quality Parameter	Site	Time Period	Samples within 24 hours of each other (%)	Frequency of Other Samples	n
Total Phosphorus (TP)	Sp Rd (Inlet)	December 2005 - September 2011	2%	· No sampling from October 2009 - March 2010, October 2010 - March 2011 · Average frequency: 52 days in winter, 25 days in spring, 22 days in summer, and 30 days in fall	64
		November 2011 - October 2017	51%	· Average frequency: 14 days in winter, 6 days in spring, 16 days in summer, and 28 days in fall	333
	Site 1	May 2006 - September 2012	3%	· Sampled monthly June - September	36
	Site 2	May 2006 - September 2012	3%	· Sampled monthly June - September	35
	CTH A (Outlet)	January 2006 - September 2011	44%	· Sampling in all seasons but most frequent in spring · No sampling from October 2009 - February 2010 · Average frequency: 20 days in winter, 4 days in spring, 6 days in summer, and 9 days in fall	548
		November 2011 - October 2017	5%	· No sampling from October 2013 - February 2014 · Average frequency: 38 days in winter, 14 days in spring, 14 days in summer, and 23 days in fall	121
Total Dissolved Phosphorus (TDP)	Sp Rd (Inlet)	July 2006 - June 2007	14%	· Approximately once a season	7
		February 2012 - June 2017	17%	· No sampling in April 2012 - January 2013, July 2015 - February 2016 · Approximately monthly samples	42
	CTH A (Outlet)	January 2006 - August 2009	14%	· No sampling July 2007 - February 2008, May 2008 - April 2009 · Approximately every other month	21
		February 2012 - September 2017	7%	· No sampling April 2012 - March 2013, September 2013 - February 2014, July 2015 - February 2016 · Approximately monthly samples	44

Water Quality Parameter	Site	Time Period	Samples within 24 hours of each other (%)	Frequency of Other Samples	n
Suspended Sediment (SS)	Sp Rd (Inlet)	December 2005 - May 2009	3%	· No sampling fall 2008 · Approximately monthly samples	31
		November 2011 - October 2017	52%	· No sampling in summer 2012, fall 2014, summer 2017 · Only 12 samples in the fall · Average frequency: 13 days in winter, 7 days in spring, 12 days in summer, 18 days in fall	313
	CTH A (Outlet)	January 2006 - September 2011	44%	· Sampling in all seasons but most frequent in spring · Average frequency: 22 days in winter, 4 days in spring, 6 days in summer, and 9 days in fall	554
		November 2011 - October 2017	2%	· No sampling from summer 2012, fall 2015 · Average frequency: 45 days in winter, 16 days in spring, 17 days in summer, and 23 days in fall	95
Total Nitrogen (TN)	Sp Rd (Inlet)	February 2014 - April 2017	26%	· No sampling from July 2015 - February 2016 · Approximately monthly samples	27
	CTH A (Outlet)	March 2014 - September 2017	3%	· 31 samples in March - September · No sampling July 2015 - February 2016, April - July 2016 · Approximately monthly samples	34

Additional sampling data

In addition to samples collected by the USGS, monthly grab samples (n = 11) were collected from July – October 2016 and April – October 2017 at Spaulding Rd and at Sites 1, 2, and 3 within SCE (Figure 2-6). Locations were accessed by kayak. For data analysis, Site 3 concentrations were considered to be representative of the SCE outlet because of proximity to the CTH A monitoring site. Samples to be analyzed for TP and TN were acidified with H_2SO_4 in the field, and all samples were preserved with ice and then refrigerated (at 4° C) until delivered to the Wisconsin State Laboratory of Hygiene for nutrient testing. Samples were analyzed for TP (method EPA 353.1), TDP (method EPA 365.1), and TN (method EPA 353.2). For analysis, these samples were termed “BSE data.”

Data Analysis

Additional explanatory data were added to the sample concentrations to aid in identifying trends. The time of sampling was matched with the flow at Spaulding Rd which was downloaded from NWIS. Flow time stamps since gage installation in November 16, 2011 were rounded to the closest five minutes, as were the time stamps for each sample. Each sample was then matched to the flow of the same time stamp. Prior to installation of the Spaulding Rd gage, daily average stream flows at CTH A were downloaded from NWIS. Because of the effects of seiches in Green Lake, USGS reported five-minute interval flows were often negative and so USGS daily averages were used in this analysis. For samples taken on days with a negative daily average stream flow, no flow was assigned. Daily flows at CTH A were multiplied by 0.92 to estimate flow at Spaulding Rd (conversion factor determined by the USGS). The Spaulding Rd flow associated with each sample time stamp were also coded into four categories: low (up to 25th percentile), lower middle (25th – 50th percentile), upper middle (50th – 75th percentile) and high (above the 75th percentile).

Samples were matched with the daily precipitation and the amount of precipitation over the preceding three and seven days. Each duration of cumulative precipitation (daily, previous three days, previous 7 days) was coded into five categories: no precipitation, low (up to 25th percentile), lower middle (25th – 50th percentile), upper middle (50th—75th percentile) and high (above the 75th percentile). All data were further divided into four seasons: winter (December – February), spring (March – May), summer (June – August), and fall (September – November).

Removal efficiencies for TN, TP, TDP, and SS were calculated (Equation 2) for Spaulding Rd and CTH A samples collected within a few days of each other. TP and SS removal efficiencies were calculated for grab samples taken on the same day at the inlet and outlet. Since fewer TN and TDP samples were collected, inlet and outlet measurements were paired if they were collected within three days of each other and there had not been a greater than 23 mm of precipitation during the time between sampling days. If more than one sample was taken on a given day for a parameter, the average concentration for that day was used to calculate the removal efficiency.

Various nutrients can limit aquatic growth. Within shallow lakes systems, P is often a limiting nutrient. However, increases in P can lead to other factors (e.g. light or N) being a stronger restrictor of growth in aquatic ecosystems (Kolzau *et al.*, 2014). In order to characterize the likelihood of this occurring in SCE, a molar N to P ratio was calculated for samples that were analyzed for both TN and TP (the ratio of TN concentration to TP concentration was multiplied by 2.21).

The effects of relevant explanatory variables (both continuous and categorical) on the dependent variables of nutrient concentrations and removal efficiencies were analyzed. Since the number of observations within each category were unequal, data were analyzed appropriately for an unbalanced design. The R statistical software was used to create MLRs and to perform an ANCOVA for each model. Explanatory variables were examined for both additive effects and interactions. Effects of explanatory variables were assessed using type III sum of squares (Hector *et al.*, 2010). If explanatory variables showed significant interactions, the effects of those individual variables were not interpreted. Significant predictors ($p < 0.05$) were noted. If necessary, data were transformed using natural logarithms prior to statistical analysis to improve normality of model residuals.

Linear regression models were developed for nutrient concentrations, removal efficiencies (RE), and for observed TDP:TP concentration ratio and TN:TP molar ratio (Table 2-6). Categorical predictors included season, site, sampling scheme (Period 1 or Period 2), year, and precipitation during the past one, three, and seven days (none, low, lower middle, upper middle, and high). Continuous predictors included daily precipitation, cumulative precipitation over the preceding three days, streamflow, and nutrient concentrations. Interactions between categorical and continuous predictors were also considered.

Table 2-6: Variables considered in MLR for SCE grab sample water quality data. Grey boxes mark variables included as part of an interaction. X marks variables evaluated for statistical significance in final MLR. Blanks were variables not considered in final MLR.

Dependent Variable		**TP Spaulding Rd and CTH A (mg/L)**	**TP four sites (mg/L)**	**TP RE (%)**	**TDP (mg/L)**	**TDP:TP ratio**	**TDP RE (%)**	**SS (mg/L)**	**SS RE (%)**	**TN (mg/L)**	**TN RE (%)**	**TN:TP ratio**
Predictor	**Transform-ation**	ln	ln		ln			ln				ln
Season (Winter, Spring, Summer, Fall)				x			x				x	
Site (Inlet, Site 1, Site 2, Outlet)												x
Season and Site Interaction		x	x		x	x		x		x		
Sampling (Period 1 or 2)		x							x			
Sampling Period and Site Interaction								x				
Sampling Period and Season Interaction								x				
TP at Spaulding Rd (mg/L)	ln			x			x					
TP (mg/L)	ln				x	x				x		
Season and TP (mg/L) Interaction	ln(TP)											x

Dependent Variable		TP Spaulding Rd and CTH A (mg/L)	TP four sites (mg/L)	TP RE (%)	TDP (mg/L)	TDP:TP ratio	TDP RE (%)	SS (mg/L)	SS RE (%)	TN (mg/L)	TN RE (%)	TN:TP ratio
TDP at Spaulding Rd (mg/L)							x					
Year (factor)			x				x					
Season and Year Interaction		x										
SS (mg/L)						x						
SS at Spaulding Rd (mg/L)	ln								x			
TN at Spaulding Rd (mg/L)											x	
Daily Precipitation (None, low, lower middle, upper middle, high)		x	x									
Daily Precipitation (mm)											x	
Cumulative Precipitation over prior 3 days (No, low, lower middle, upper middle, high)				x	x					x		
Cumulative Precipitation over prior 3 days (mm)												x
Cumulative 7 day Precipitation (No, low, lower middle, upper middle, high)						x	x	x	x			
Flow at Spaulding Rd (m^3/s)	ln		x				x				x	
Year and Flow at Spaulding Rd (m^3/s) Interaction	ln(Flow)	x		x	x	x		x				
Site and Flow at Spaulding Rd (m^3/s) Interaction	ln(Flow)									x		x
Season and Flow at Spaulding Rd (m^3/s) Interaction								x	x			

Results

RWTP loads to Silver Creek

Monthly average effluent flows were slightly greater (2%) after the upgrade at the RWTP (overall median 0.059 m^3/s) compared to before (overall median 0.058 m^3; Table 2-7). Before the RWTP upgrade, median seasonal monthly average flows ranged from 0.051 - 0.064

m^3/s and followed the order winter < fall < summer < spring (Figure 2-7 A; Table 2-7). Following the upgrade, seasonal median effluent flows remained constant in the winter (0.051 m^3/s) and followed the same order as before the upgrade, with the remaining seasonal medians ranging from 0.057 -- 0.073 m^3/s. The largest increase of all seasons was in the spring median (14%). These seasonal and overall increases in flows were not statistically significant (p ranged from 0.053-0.725; Table 2-7).

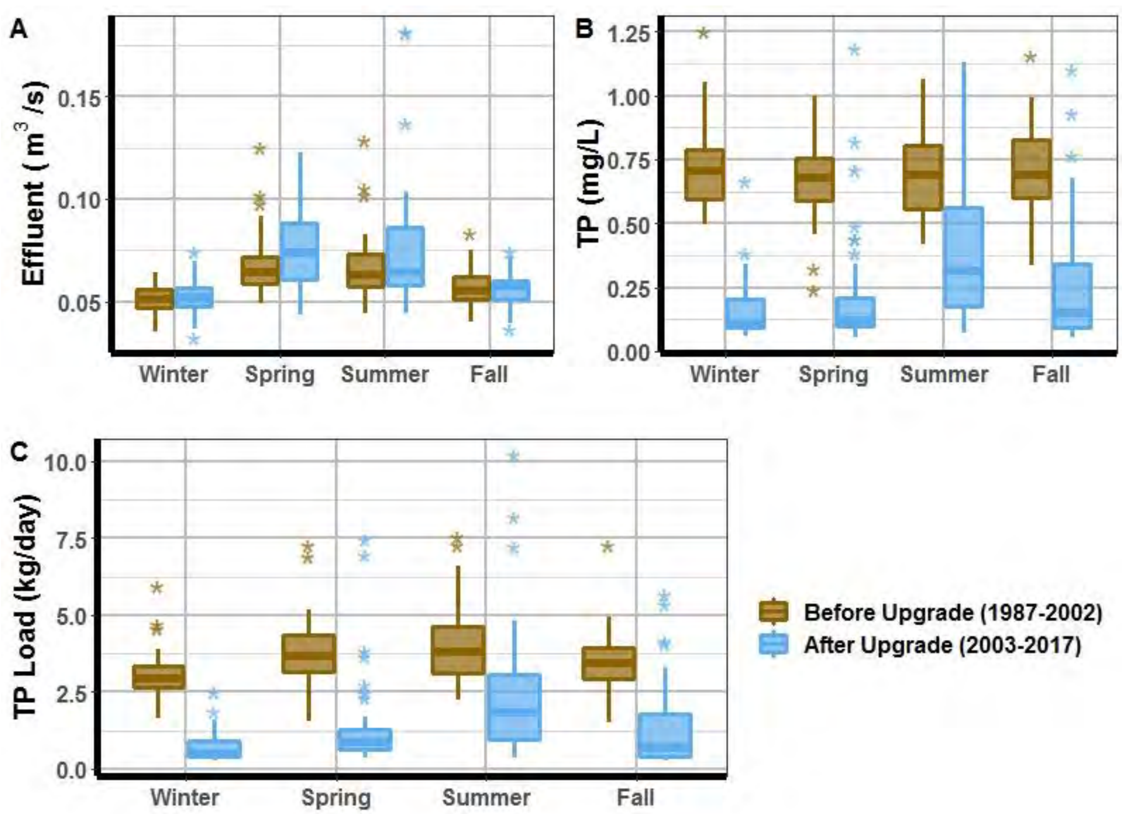

Figure 2-7: (A) Average monthly RWTP effluent flows before (1987-2003) and after (2003-2017) the RWTP upgrade. (B) Average monthly TP concentration in RWTP effluent before and after upgrade. (C) Average monthly TP load before and after the upgrade calculated using effluent and TP concentrations from RWTP. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

*Table 2-7: Difference between median observations of average monthly effluent flow, TP concentrations, and TP loads before and after the RWTP upgrade. Number of observations are also noted. Bolded values are significant, p <0.05. (^ p values calculated without using interactions, * n = 48 for median TP concentrations in winter and spring before the upgrade)*

	Upgrade	Winter	Spring	Summer	Fall	All Seasons
Median Effluent (m^3/s)	**Before**	0.051	0.064	0.062	0.054	0.058
	After	0.051	0.073	0.064	0.057	0.059
	Difference (% Change)	0 (0%)	0.009 (14%)	0.002 (3%)	0.003 (6%)	0.001 (2%)
	p value	0.256	0.100	0.232	0.725	0.053
Median TP (mg/L)	**Before**	0.70	0.68	0.69	0.69	0.69
	After	0.11	0.12	0.31	0.15	0.16
	Difference (% Reduction)	0.59 (84%)	0.55 (82%)	0.38 (55%)	0.55 (78%)	0.52 (76%)
	p value	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Median TP Load (kg/day)	**Before**	2.93	3.66	3.82	3.43	3.40
	After	0.52	0.89	1.85	0.67	0.89
	Difference (% Reduction)	2.41 (82%)	2.77 (76%)	1.97 (51%)	2.76 (80%)	2.51 (74%)
	p value	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
n	**Before***	47	47	48	48	190
	After	43	45	45	43	176

Monthly average RWTP effluent flow was significantly impacted by the interaction between season and precipitation ($p < 0.001$) and between upgrade period and precipitation ($p = 0.046$), indicating that the impact of precipitation across seasons and upgrade period was not constant (Figure 2-8; Appendix C Table C-1). Increasing precipitation corresponded to increases in flow that were greater in the spring and summer compared to fall and winter ($p < 0.001$; Figure 2-8 A). After the upgrade, increasing precipitation corresponded to greater increases in flow compared to before the upgrade ($p = 0.046$; Figure 2-8 B). Median precipitation increased from 1.63 mm/day before the upgrade to 2.01 mm/day after the upgrade.

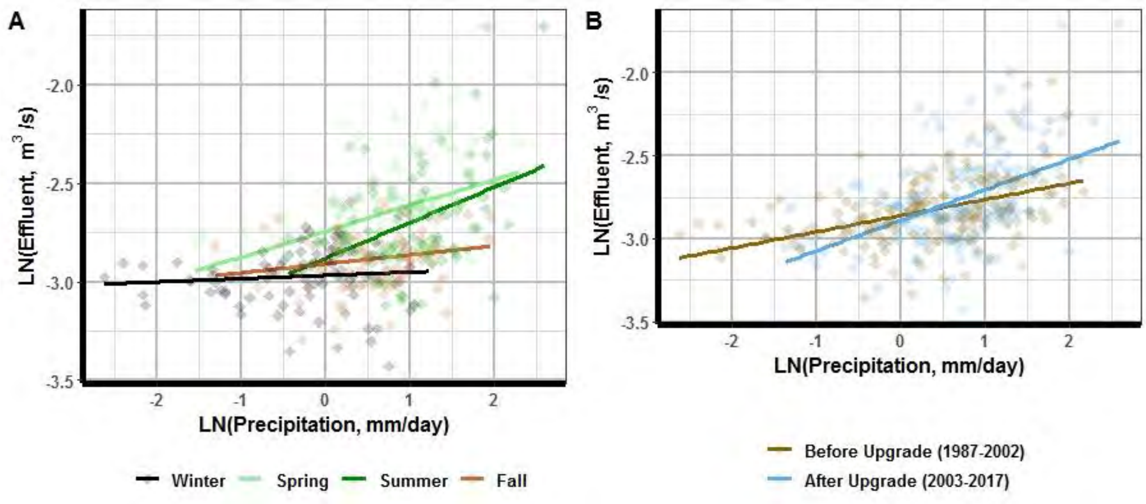

Figure 2-8: Measured and estimated RWTP effluent versus monthly average precipitation from 1987-2017. Both variables are natural log transformed. Linear models span the precipitation range of the respective sub-dataset. (A) Significant interaction occurred between season and precipitation ($p < 0.001$). (B) Significant interaction occurred between upgrade status and precipitation ($p = 0.046$).

Monthly average TP concentrations decreased (76%) after the upgrade at the RWTP (overall median 0.16 mg/L) compared to concentrations before the upgrade (overall median 0.69 mg/L; Table 2-7). Before the RWTP upgrade, median effluent TP concentrations in all seasons were close to 0.7 mg/L (Figure 2-7 B). Following the upgrade, however, seasonal median TP concentrations decreased, ranging from 0.11 – 0.31 mg/L and followed the order winter < spring < fall < summer (Table 2-7).

TP concentrations were significantly related to monthly precipitation ($p = 0.001$) and the interaction between season and RWTP upgrade ($p < 0.001$; Appendix C Table C-1). Increased precipitation corresponded to slightly lower TP concentrations in the effluent. Before the upgrade, TP concentrations were similar in all seasons ($p > 0.43$). Following the upgrade, all seasonal TP concentrations were lower than they had been before ($p < 0.001$), but summer TP concentrations remained significantly higher than those in the other seasons ($p < 0.001$; Figure

2-7 B). TP concentrations following the upgrade decreased the least in the summer (55%; Table 2-7) compared to the other seasons.

Monthly average TP loads decreased (74%) after the upgrade at the RWTP (overall median 0.89 kg/day) compared to before (overall median 3.40 kg/day; Table 2-7). When grouped by SCE restoration period, the TP median before restoration was 3.46 kg/day and 0.67 kg/day afterwards (81% reduction). Before the RWTP upgrade, median seasonal monthly average TP loads ranged from 2.93 – 3.82 kg/day and followed the order winter < fall < spring < summer (Figure 2-7 C; Table 2-7). Following the upgrade, seasonal median TP loads ranged from 0.52 – 1.85 kg/day and followed the same order as before the upgrade. The largest decreases in seasonal median TP load occurred during spring (2.77 kg/day) and fall (2.76 kg/day), and the smallest reduction occurred in summer (1.97 kg/day; Table 2-7). Among seasons, the greatest percent reduction in median TP load (82%) was in winter. When data were grouped by SCE restoration period, the seasonal median decreased 2.94 kg/day in spring, 2.90 kg/day in fall, 2.49 kg/day in winter, and 2.25 kg/day in summer.

TP loads were significantly related to effluent flow ($p < 0.001$), and the interactions between season and upgrade ($p < 0.001$) and precipitation and upgrade ($p = 0.004$; Appendix C Table C-1). As with TP concentrations, TP loads decreased significantly in all seasons following the upgrade ($p < 0.001$), but high summer variability resulted in significantly higher TP loads compared to other seasons in the period after the upgrade ($p < 0.001$; Figure 2-7 C). As with effluent flow, TP loads increased more in relation to precipitation after the upgrade than before (Figure 2-9).

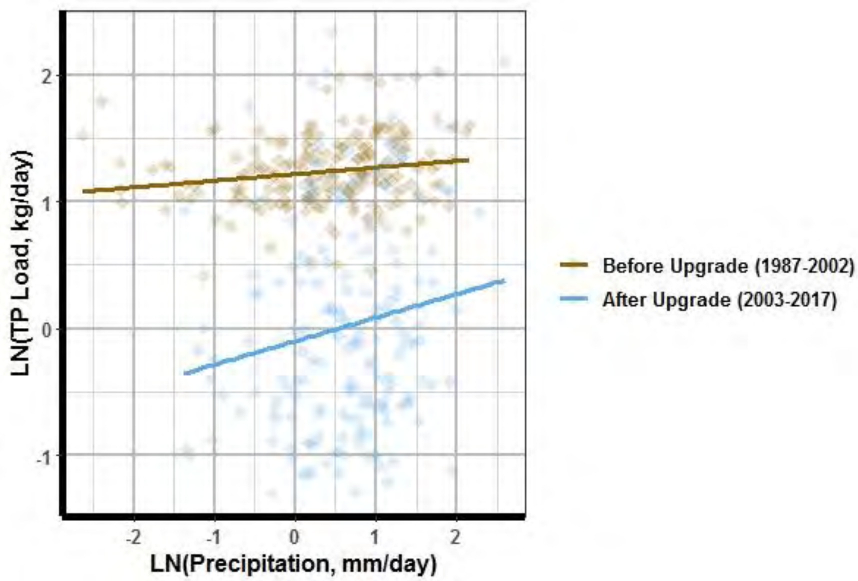

Figure 2-9: Measured and estimated RWTP TP load versus monthly average precipitation from 1987 – 2017. Shows significant interaction between upgrade status and precipitation (p = 0.004). Both variables are natural log transformed. Linear models span the precipitation range of the respective sub-dataset.

Loads from SCE over 30 years

Precipitation and Flow

Monthly average precipitation increased (28%) after the restoration of SCE (overall median 2.10 mm/day) compared to before (overall median 1.64 mm/day; $p < 0.001$; Table 2-8). From 1987 – 2001, before SCE restoration, seasonal monthly average precipitation ranged from 0.71 – 2.81 mm/day and followed the order winter < fall < spring < summer (Figure 2-10 A; Table 2-8). Since 2006, after the restoration, the seasonal median precipitation ranged from 1.18 - 3.13 mm/day, following the same order as before but increasing in all seasons. Precipitation was significantly impacted by the interaction between season and restoration ($p = 0.033$; Appendix B Table B-2). In winter ($p < 0.001$) and spring ($p = 0.024$), the increase in precipitation before and after restoration was significant, but it was not for summer ($p = 0.98$) and fall ($p = 0.81$). There were similar significant increases in winter and spring but not in summer and fall for an analysis using a different precipitation dataset (Appendix A Table A-1).

Monthly average flow at the outlet of SCE increased, but not significantly, from before (overall median 0.75 m^3/s) to after (overall median 0.80 m^3/s) restoration ($p = 0.962$; Table 2-8). Both before and after restoration, seasonal median flows followed the order fall < winter < summer < spring (Figure 2-10 B). Before restoration, seasonal flow ranged from a median of 0.51 – 1.46 m^3/s (Table 2-8). Following the upgrade, all seasonal medians increased and ranged from 0.59 – 1.93 m^3/s. The largest increase (32%) in median flow from before to after restoration occurred in the spring; however, no differences were significant ($p = 0.276 – 0.962$). The median spring flow was approximately 3 times larger than the median fall flow both before and after restoration. In June 2008, the monthly average flow at CTH A was 9.21 m^3/s, the highest during the Before and After periods (the highest average flow of 9.89 m^3/s occurred during restoration in June 2004). The highest monthly average precipitation (13.3 mm/day) over the entire 30 years also occurred in June 2008. Flow at the SCE outlet was significantly related to the interactions between the volume of effluent from RWTP and season ($p = 0.001$) and between season and precipitation ($p = 0.002$; Appendix B Table B-2). The restoration period was not a significant predictor ($p = 0.13$). Flow increased more quickly with increasing RWTP effluent in summer compared to spring and winter ($p < 0.007$; Figure 2-11 A). In summer, flow increased with increasing precipitation significantly more than it did in spring ($p = 0.029$; Figure 2-11 B).

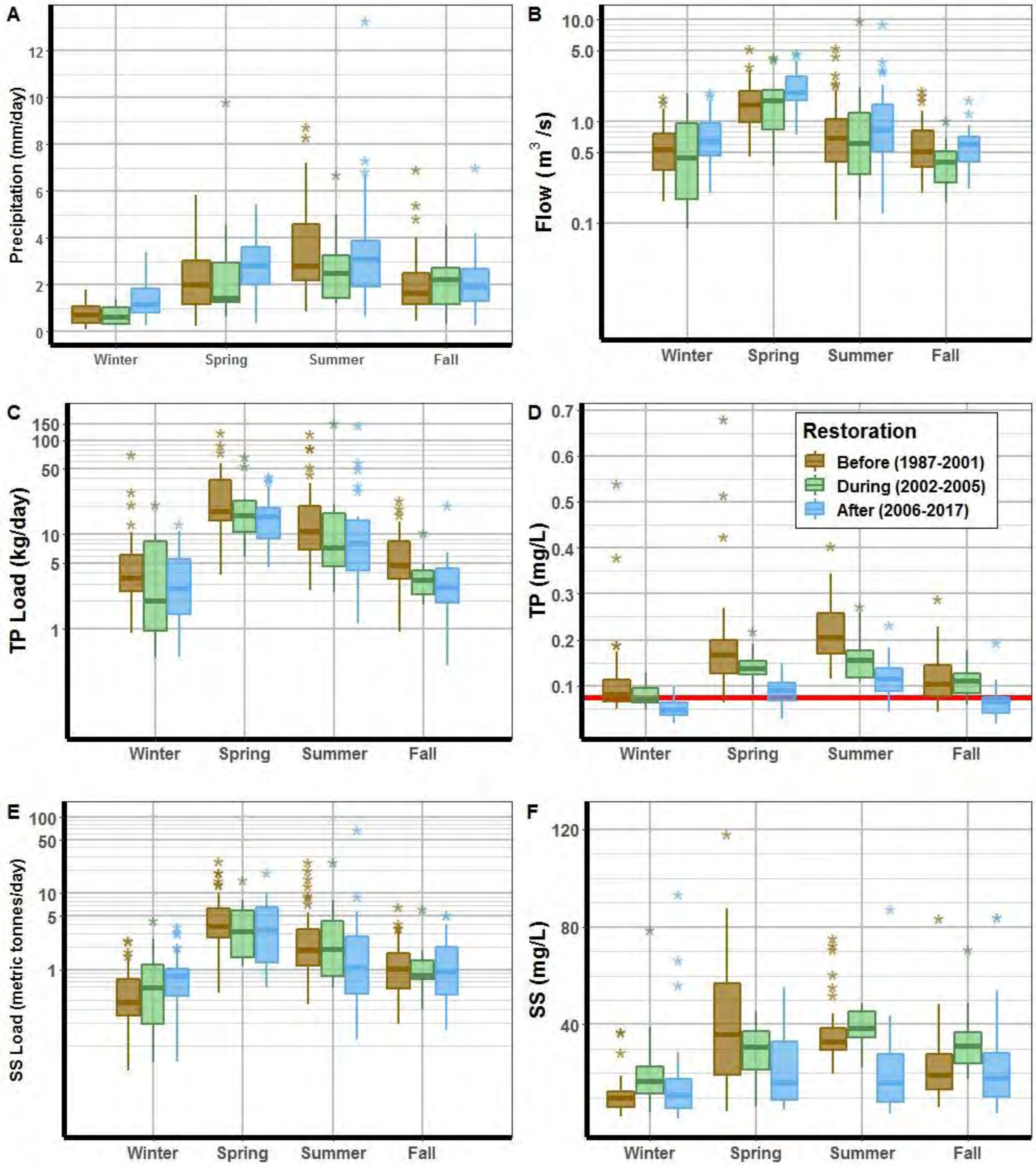

Figure 2-10: (A) Daily average precipitation for each month before (1987-2001), during (2002-2006) and after (2007-2017) SCE restoration from a phytoplankton to macrophyte-dominant state. (B) Monthly average flow at the outlet of SCE, CTH A. (C) Monthly average TP load for each month at CTH A. (D) Monthly average flow-weighted TP concentrations (TP load divided by flow) at CTH A. Red line indicates TP water quality criteria of 0.075 mg/L. (E) Monthly average SS load at CTH A. (F) Monthly average flow-weighted SS concentrations (SS load divided by flow) at CTH A. Does not include 204.8 SS mg/L in May 1989. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

Table 2-8: Difference between median values of average monthly precipitation and flow at the SCE outlet, CTH A, before and after SCE restoration. Number of observations are also noted. Bolded values are significant, p <0.05.

	Restoration	Winter	Spring	Summer	Fall	All Seasons
Precipitation (mm/day)	**Before**	0.71	2.02	2.81	1.64	1.64
	After	1.18	2.79	3.13	1.92	2.10
	Difference (% Increase)	0.47 (67%)	0.77 (38%)	0.32 (11%)	0.28 (17%)	0.46 (28%)
	p value	**<0.001**	**0.024**	0.980	0.810	**<0.001**
Flow at CTH A (m^3/s)	**Before**	0.53	1.46	0.69	0.51	0.75
	After	0.64	1.93	0.83	0.59	0.80
	Difference (% Increase)	0.12 (22%)	0.47 (32%)	0.15 (21%)	0.08 (17%)	0.05 (7%)
	p value	0.276	0.962	0.280	0.940	0.962
n	**Before**	44	45	45	45	179
	During	12	12	12	12	48
	After	35	36	36	34	141

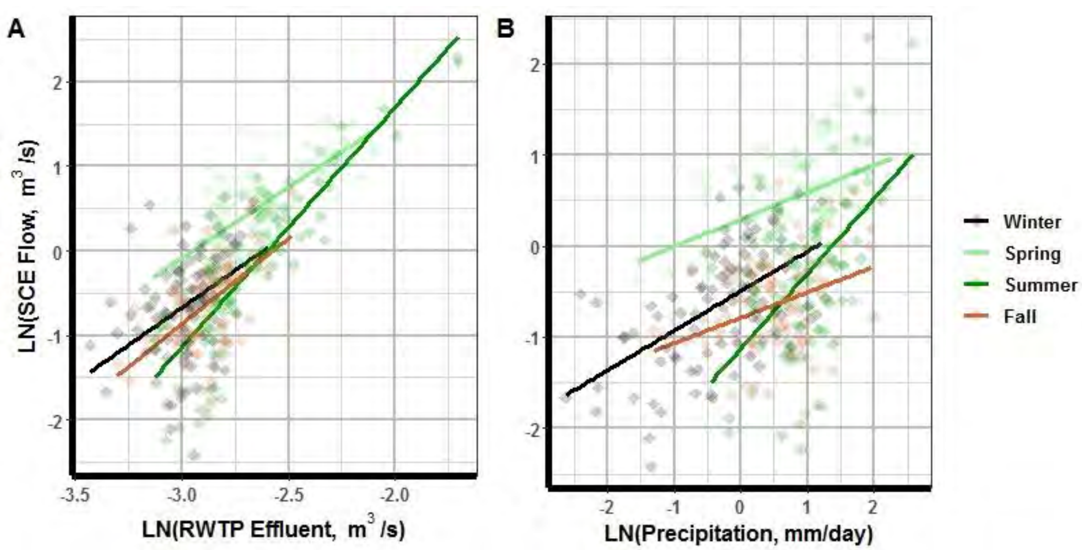

Figure 2-11: Measured and estimated monthly average SCE outlet flow versus (A) monthly average RWTP effluent and (B) monthly average precipitation. All variables are natural log transformed. Linear models span the range of the respective sub-dataset.

Total Phosphorus

Following SCE's restoration to macrophyte dominance in 2006, the median TP load at CTH A was significantly reduced from an overall median of 8.16 kg/day before restoration to an overall median of 5.5 kg/day after (33% decrease; $p < 0.001$; Table 2-9). Both before and after restoration, median TP loads followed the order winter < fall < summer < spring, ranging from 3.47 – 17.7 kg/day before restoration and 2.69 – 15.4 kg/day afterwards. Seasonal TP loads decreased in all four seasons following restoration ($p < 0.001$; Table 2-9; Figure 2-10 C). Spring had the largest median TP load and the smallest percent reduction (13%) from before to after restoration. The seasonal median load decreased by the largest percentage (41%; 1.97 kg/day) in the fall, but reductions in median TP mass were greatest in summer (2.99 kg/day) and spring (2.3 kg/day) before and after restoration. The TP load was significantly related to restoration period ($p < 0.001$), SS load at CTH A ($p < 0.004$), and interactions between season and precipitation ($p = 0.004$) and season and flow ($p = 0.01$; Appendix B Table B-3). Neither the TP load from the upstream RWTP ($p = 0.42$) nor its upgrade status ($p = 0.31$) were significant predictors of TP load at CTH A. TP loads were lower in winter and fall compared to spring and summer and increasing precipitation was associated with increasing TP loads, especially in summer (Figure 2-12 A). Increasing flow was associated with increasing TP load, and spring and summer had higher TP loads than winter and fall (Figure 2-12 B).

Table 2-9: Differences between seasonal medians of daily TP load and TP concentration before and after the SCE restoration. P values show statistical significance of the difference between before and after the restoration for data from each season. Number of observations also shown.

	Restoration	Winter	Spring	Summer	Fall	All Seasons
Median TP Load (kg/day)	**Before**	3.47	17.70	10.90	4.76	8.16
	After	2.69	15.40	7.95	2.79	5.50
	Difference (% Reduction)	0.77 (22%)	2.30 (13%)	2.99 (27%)	1.97 (41%)	2.66 (33%)
	p value	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Median Flow-weighted Mean TP Concentration (mg/L)	**Before**	0.081	0.167	0.207	0.104	0.140
	After	0.048	0.089	0.114	0.064	0.074
	Difference (% Reduction)	0.033 (40%)	0.077 (46%)	0.092 (45%)	0.040 (38%)	0.066 (47%)
	p value	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
n	**Before**	44	45	45	45	179
	During	12	12	12	12	48
	After	35	36	36	34	141

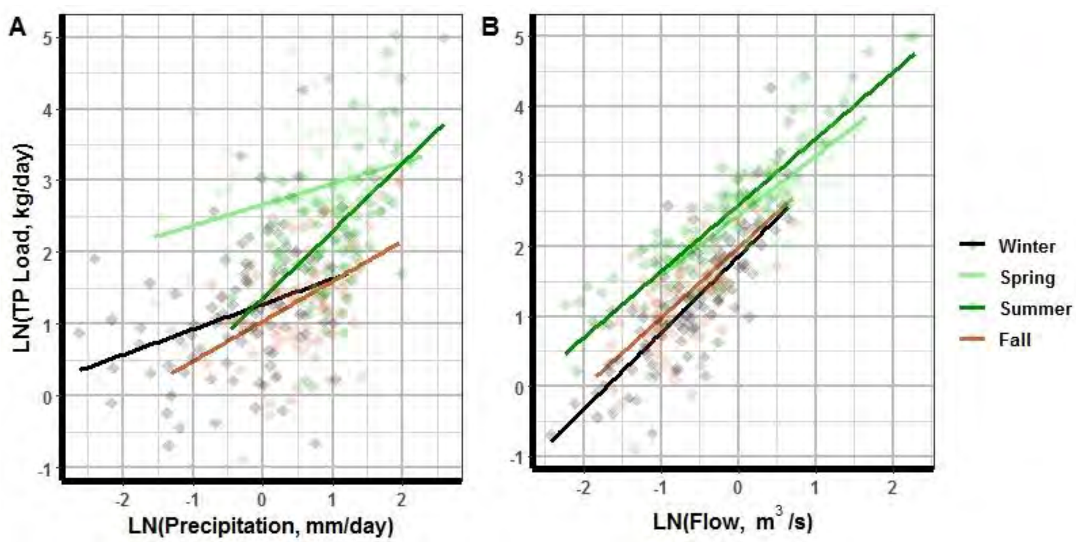

Figure 2-12: Measured and estimated monthly average TP load at SCE outlet versus (A) monthly average precipitation and (B) monthly average flow at SCE outlet. All variables are natural log transformed. Linear models span the horizontal range of the respective sub-dataset. Varying slopes and intercepts for linear models show the significant interactions between explanatory variable (plot A p = 0.004, plot B p = 0.01).

The overall difference in median TP load at CTH A before and after restoration (2.66 kg/day) was similar to the reduction in the median TP load at the RWTP following the upgrade (2.49 kg/day). However, in the winter, spring, and fall, the differences between the median TP load before and after the RWTP upgrade were greater than the differences between the median TP load measured at CTH A before and after SCE restoration. Summer was the only season in which the decrease in the median TP load before and after restoration at CTH A (2.99 kg/day) was greater than the reduction observed from the RWTP upgrade (1.97 kg/day).

Monthly average retention times in SCE were used to examine changes in TP load. Over the 30-year time period, SCE retention time varied from a monthly average of 88 days (January 2003) to 0.8 days (19 hours; June 2004). Monthly average retention times did not change significantly before (median = 10.5 days) and after (median = 9.9 days) restoration ($p = 0.07$). Over the 30 years, median seasonal retention times ranged from 4.6 – 14.4 days and followed the order spring < summer < winter and fall. Spring, the period with highest stream flows, had significantly shorter retention times than fall, winter, and summer ($p < 0.001$; Figure 2-13 A). Average monthly TP load was negatively correlated with SCE retention time, with higher spring and summer TP loads associated with shorter retention times and lower TP loads in fall and winter when retention times were longer (Figure 2-13 B).

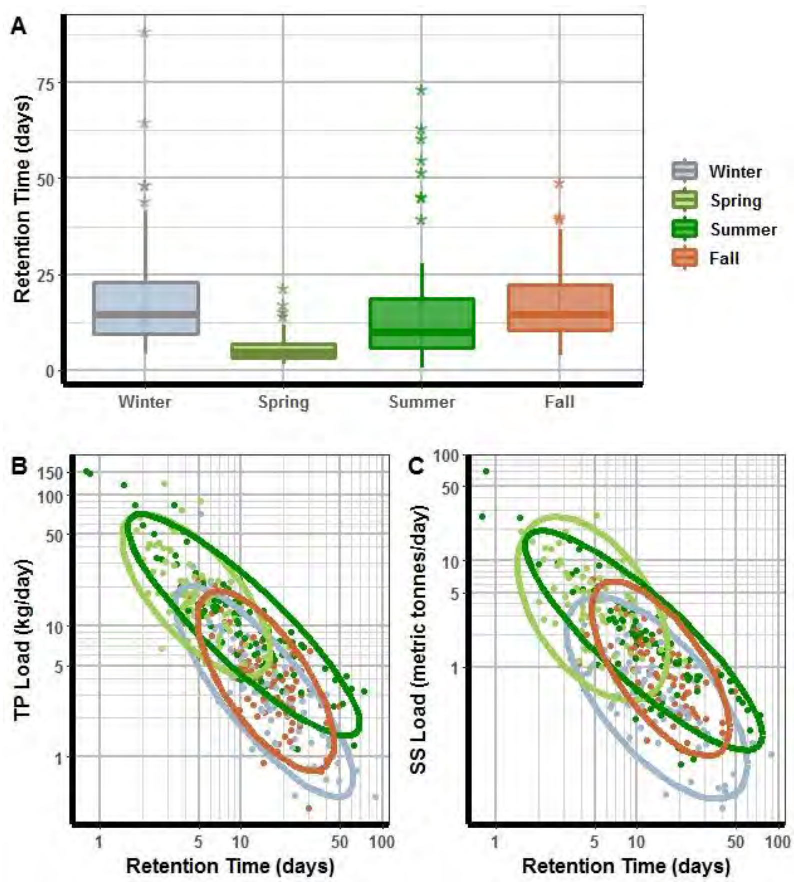

Figure 2-13: (A) Monthly average SCE retention time using data from 1987-2017 separated by season. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually. (B) Monthly average SCE retention time versus the daily average TP load for each month using data from 1987-2017. (B) Monthly average SS load from SCE outlet versus monthly average retention time using data from 1987-2017. Ellipses show a 95% confidence interval of a multivariate t-distribution.

Percent reductions in flow-weighted mean TP concentrations at CTH A were larger than reductions in TP load. Overall, the median flow-weighted mean TP concentration decreased by 47% (0.066 mg/L; $p < 0.001$) from before to after SCE's transition to macrophyte dominance (Table 2-9). As with TP load, seasonal median flow-weighted mean TP concentrations maintained their relative rank from before to after restoration but followed the order of winter < fall < spring < summer (Figure 2-10 D). Before the restoration, the seasonal median flow-weighted TP concentrations ranged from 0.081 – 0.207 mg/L and afterwards, ranged from 0.048 – 0.114 mg/L. The ratio of summer to winter median flow-weighted mean TP concentrations decreased from 2.55 before restoration to 2.36 afterwards. The flow-weighted mean TP concentration was significantly related to restoration period ($p < 0.001$), flow-weighted mean SS concentration at CTH A ($p = 0.004$), and the interaction between season and precipitation ($p = 0.004$) and between season and flow at CTH A ($p = 0.01$; Appendix B Table B-3). Neither the concentration of TP from the RWTP ($p = 0.297$) nor its upgrade status ($p = 0.278$) were significant predictors of flow-weighted mean TP concentration at CTH A. Before restoration, spring and summer had significantly higher TP concentrations than fall ($p < 0.001$), and summer concentrations were significantly higher than winter ($p < 0.014$). After restoration spring and summer had significantly higher TP concentrations than both fall and winter ($p < 0.022$). Increasing precipitation was associated with significantly higher fall TP concentrations compared to other seasons ($p = 0.043$; Figure 2-14 A). As flow increased, flow-weighted mean TP concentrations became significantly higher in summer and spring compared to fall and winter ($p < 0.02$; Figure 2-14 B).

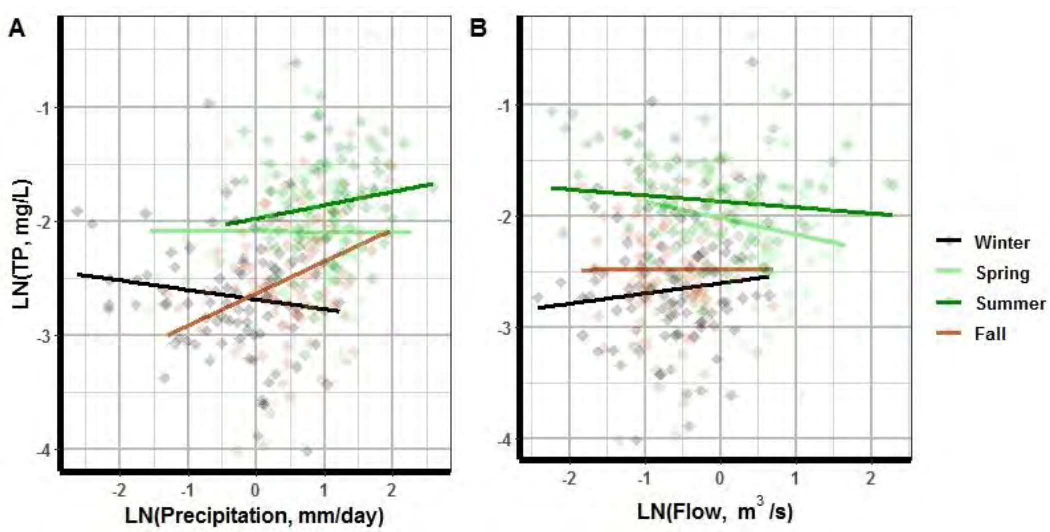

Figure 2-14: Measured and estimated monthly flow-weighted mean TP concentrations at SCE outlet versus (A) monthly average precipitation and (B) monthly average flow at SCE outlet. All variables are natural log transformed. Linear models span the horizontal range of their respective sub-dataset. Varying slopes and intercepts of linear models show the significant interactions between explanatory variables (plot A p = 0.004, plot B p = 0.01).

Total Phosphorus Retention in SCE

The inlet to SCE (Silver Creek at the Spaulding Rd gage) was not monitored before the restoration; therefore, no data are available to determine if SCE was retaining TP prior to restoration. However, comparing TP loads from the RWTP upstream and at the marsh outlet at CTH A indicates that there was limited TP retention between the RWTP and Green Lake. Before the RWTP upgrade and the SCE restoration, TP load from the RWTP exceeded that of CTH A in 32 of the 177 months (18%). These predominantly occurred in the winter (n = 16) and fall (n = 14; Figure 2-15). Following the upgrade and restoration, TP loads from the RWTP only infrequently (5 out of 141 months) exceeded that of the SCE outlet. When this did occur, it was in summer (n = 2) and fall (n = 3). Before the upgrade and restoration, this only occurred once in summer.

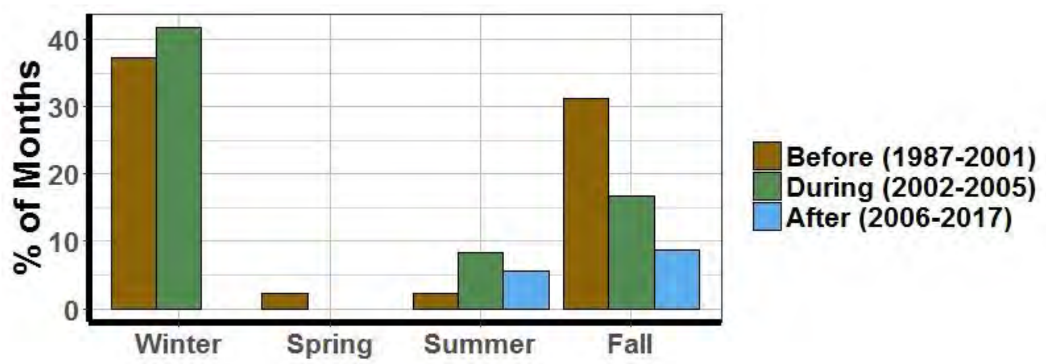

Figure 2-15: Percent of months in each season when RWTP monthly average TP load exceeded monthly average TP load at SCE outlet. Data colored by SCE restoration period. The RWTP upgrade occurred in 2003, during SCE restoration.

The difference between TP load at RWTP and SCE varied seasonally. Over the 30-year period, average TP load at RWTP and at CTH A were closer during winter and fall (Figure 2-16). Based on average monthly TP loads over the 30 years, TP loads from the RWTP constitutes an average of 50% of the TP load recorded at CTH A in fall and 40% in winter. However, the percentages in spring and summer decrease to 12% and 18%, respectively. Since 2011, average TP loads from the RWTP were most similar to those at Spaulding Rd (SCE inlet) in fall; RWTP TP loads constitute an average of 52% of the Spaulding Rd TP load in the fall but a smaller portion during winter (33%), spring (19%), and summer (26%).

Since December 2011, the RWTP monthly average TP load accounted for an overall median of 11% of the monthly average TP load at Spaulding Rd. Seasonally, the RWTP TP load accounted for between 5% (spring median) and 19% (summer median) of the TP load at Spaulding Rd. The monthly average TP load at the RWTP was never greater than downstream at Spaulding Rd.

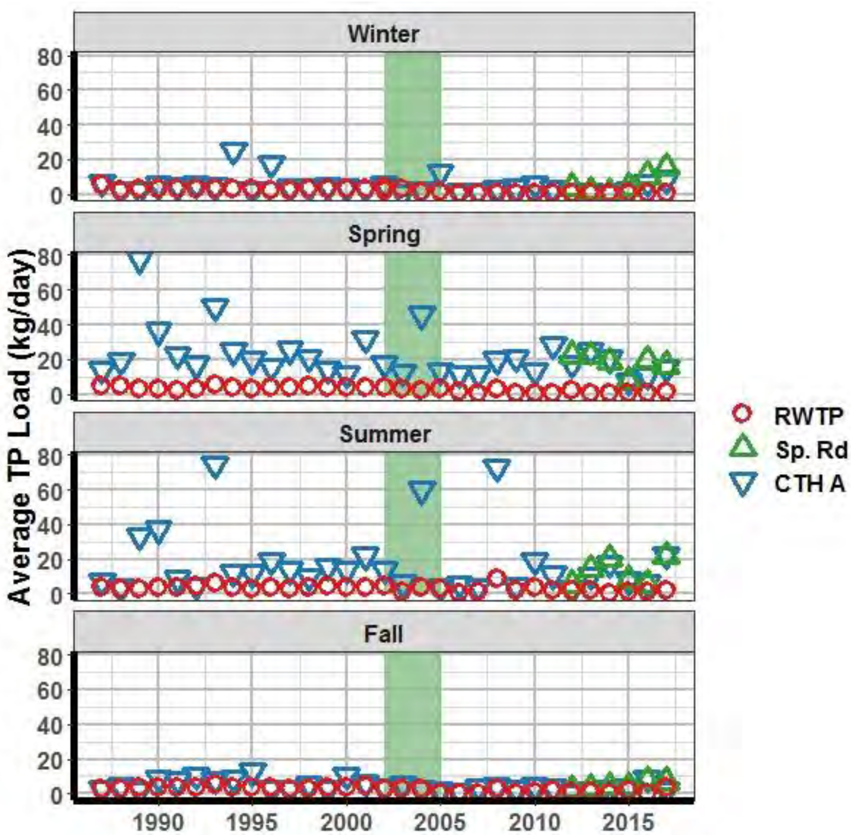

Figure 2-16: Average seasonal TP load at RWTP and CTH A (SCE outlet) from 1987-2017. Restoration period (2002-2005) shaded in green. TP loads at the Spaulding Rd (Sp. Rd; SCE inlet) begin in winter 2011.

Differences in TP loads and flow-weighted mean concentrations at Spaulding Rd and CTH A were compared from December 2011 thru September 2017. Overall, median TP loads were significantly higher at Spaulding Rd (6.3 kg/day) than at CTH A (5.09 kg/day; $p < 0.001$; Table 2-10). Except for spring, seasonal TP loads were significantly higher at Spaulding Rd than at CTH A ($p < 0.037$; Figure 2-17 A). In winter, median TP load was 32% (1.22 kg/day) lower at CTH A than at Spaulding Rd. During fall, median TP load was 43% (1.67 kg/day) lower at CTH A compared to Spaulding Rd. TP loads were significantly related to year ($p < 0.001$), and the interactions between site and precipitation ($p = 0.016$), season and precipitation ($p = 0.02$), and flow and season ($p = 0.019$; Appendix B Table B-4). TP loads were significantly lower in 2012 than in 2013 – 2015 ($p < 0.001$). The interaction between site and precipitation showed that as precipitation increased, the difference between TP load at Spaulding Rd and CTH A decreased (Figure 2-18 A). The interaction between precipitation and season showed decreasing fall TP loads as precipitation decreased (Figure 2-18 B). The interaction between flow and season showed decreases in flow and TP load to be most associated with fall and winter (Figure 2-18 C).

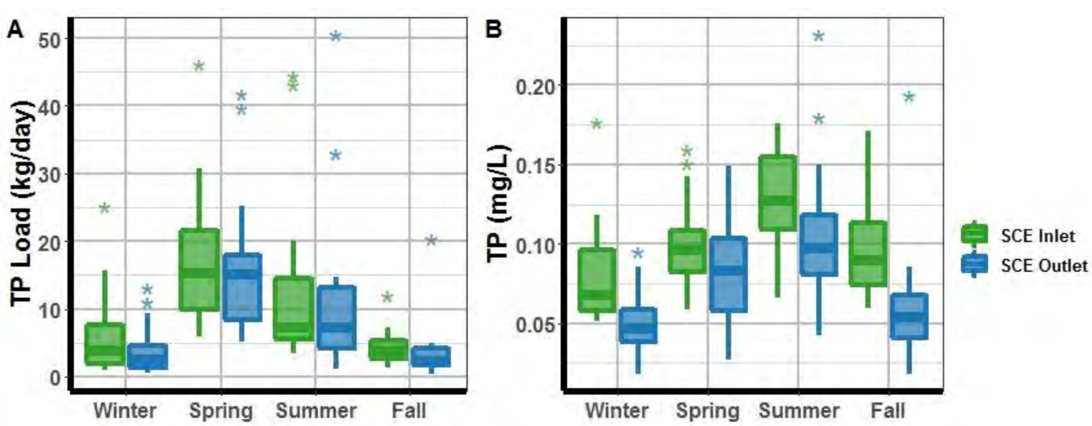

Figure 2-17: (A) Monthly average TP load at SCE inlet (Spaulding Rd) and outlet (CTH A) for data from December 2011 – September 2017. (B) Flow-weighted mean monthly TP concentrations at SCE inlet and outlet over same time period. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

Table 2-10: Seasonal median TP loads and flow-weighted mean TP concentrations at the inlet (Spaulding Rd) and outlet (CTH A) of SCE using data from December 2011 to September 2017. Number of observations also shown. P values indicate statistical difference between the inlet and outlet with significant p values (<0.05) bolded.

		Winter	Spring	Summer	Fall	All Seasons
Median TP Load (kg/day)	**Inlet**	3.76	15.20	7.16	3.90	6.30
	Outlet	2.54	15.00	7.23	2.23	5.09
	Difference (% Reduction)	1.22 (32%)	0.2 (1%)	-0.07 (1% increase)	1.67 (43%)	1.2 (19%)
	p value	**0.015**	0.075	**0.037**	**< 0.001**	**<0.001**
Median Flow-weighted Mean TP Concentration (mg/L)	**Inlet**	0.067	0.096	0.127	0.089	0.095
	Outlet	0.046	0.083	0.097	0.054	0.066
	Difference (% Reduction)	0.021 (31%)	0.013 (13%)	0.03 (24%)	0.035 (39%)	0.029 (31%)
	p value	**0.003**	**0.015**	**0.006**	**<0.001**	**<0.001**
n		18	18	18	16	70

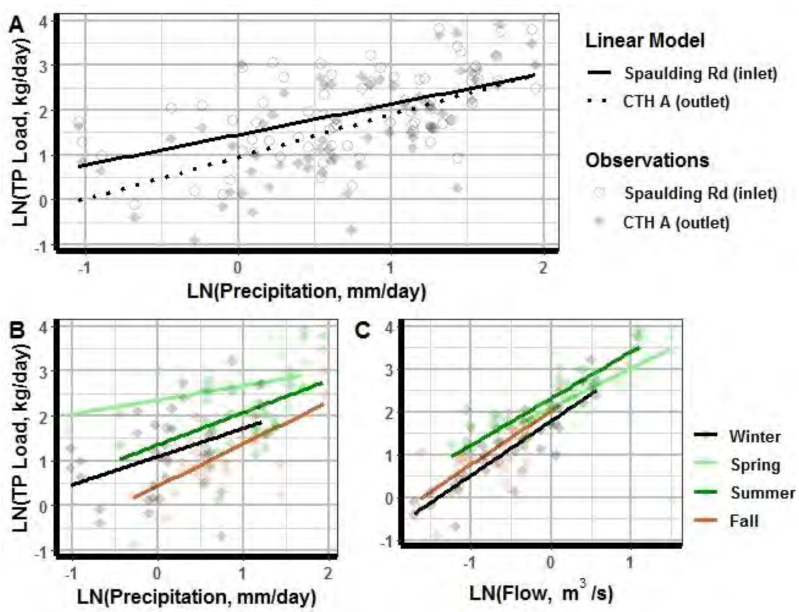

Figure 2-18: Measured and estimated monthly average TP load at Spaulding Rd (inlet) and CTH A (outlet) versus explanatory variables. All variables are natural log transformed. Linear models span the horizontal range of their respective sub-dataset. (A) TP load versus precipitation showing the significant interaction between precipitation and site (p = 0.016). (B) TP load versus precipitation showing the significant interaction between precipitation and season (p = 0.02). (C) TP load versus flow at Spaulding Rd showing the significant interaction between flow and season (p = 0.019).

The TP load at Spaulding Rd was greater than at CTH A during 75% of the monitored months (December 2011 – September 2017; Figure 2-19). In total, for months with Spaulding Rd TP loading greater that CTH A TP loading, SCE retained 4,964 kg of P. But in the other months, when Spaulding Rd TP load was less than that at CTH A, a total of 1,571 kg, of TP was released. This resulted in a net total 3,393 kg of TP retained over the monitoring period. From December 2011 to September 2017, this equates to an average 1.59 kg/day difference between TP load at Spaulding Rd and CTH A. On average, all seasons had positive retention, but the largest retention occurred in the winter (average of 2.41 kg/day) and the lowest during fall (average 0.88 kg/day). TP retention ranged from -11.1 kg/day in April 2013 to 15 kg/day in April 2016. TP retention was significantly explained by TP load at Spaulding Rd ($p = 0.007$) and monthly average precipitation ($p = 0.04$; Appendix B Table B-4). As TP load at Spaulding Rd increased, so did the variability in retention but positive retention was more dominant (Figure 2-20 A). As precipitation increased, TP retention within in SCE became more variable and slightly decreased (Figure 2-20 B).

Figure 2-19: TP retention equals monthly average TP load at Spaulding Rd (SCE inlet) minus monthly average TP load at CTH A (SCE outlet). Positive retention signifies SCE stored TP. For negative quantities, it exported additional TP than what flowed in. Gradation indicates precipitation category. Average monthly precipitation over the period was divided into four categories (low, lower middle, upper middle, and high) by quartiles.

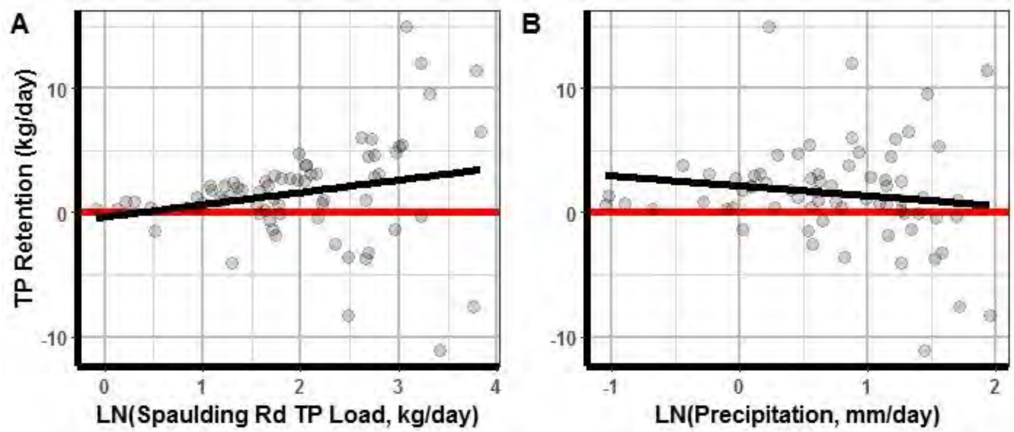

Figure 2-20: Significant predictors of SCE TP retention. TP retention equals monthly average TP load at Spaulding Rd (SCE inlet) minus monthly average TP load at CTH A (SCE outlet). Positive retention signifies SCE stored TP. For negative quantities, it exported additional TP than what flowed in. (A) TP retention versus monthly average TP load at Spaulding Rd. (B) TP retention versus monthly average precipitation. Both variables on the horizontal axes are natural log transformed.

Monthly flow-weighted mean TP concentrations decreased significantly from Spaulding Rd to CTH A ($p < 0.001$; Table 2-10). The overall median concentration at CTH A (0.066 mg/L) was 31% lower than the median TP concentration at Spaulding Rd (0.095 mg/L). Seasonal median flow-weighted mean TP concentrations followed the same relative order at both Spaulding Rd and CTH A: winter < fall < spring < summer. Differences between Spaulding Rd and CTH A were greatest in the fall; median TP concentration at Spaulding Rd (0.089 mg/L) decreased 39% to a median of 0.054 mg/L at CTH A (Figure 2-17 B; Table 2-10). The difference between median TP concentration at the Spaulding Rd and CTH A was smallest in the spring; median TP concentration at Spaulding Rd (0.096 mg/L) decreased 13% to a median of 0.083 mg/L at CTH A (Figure 2-17 B; Table 2-10). Flow-weighted mean TP concentration was significantly related to year ($p = 0.001$), TP concentration at the RWTP ($p = 0.023$), and the interactions between season and precipitation ($p = 0.045$) and site and precipitation ($p =0.017$; Appendix B Table B-4). As precipitation increased, the difference between TP concentration at

Spaulding Rd and CTH A decreased (Figure 2-21 A). Precipitation had less of an effect on TP concentrations in the spring than in other seasons (Figure 2-21 B).

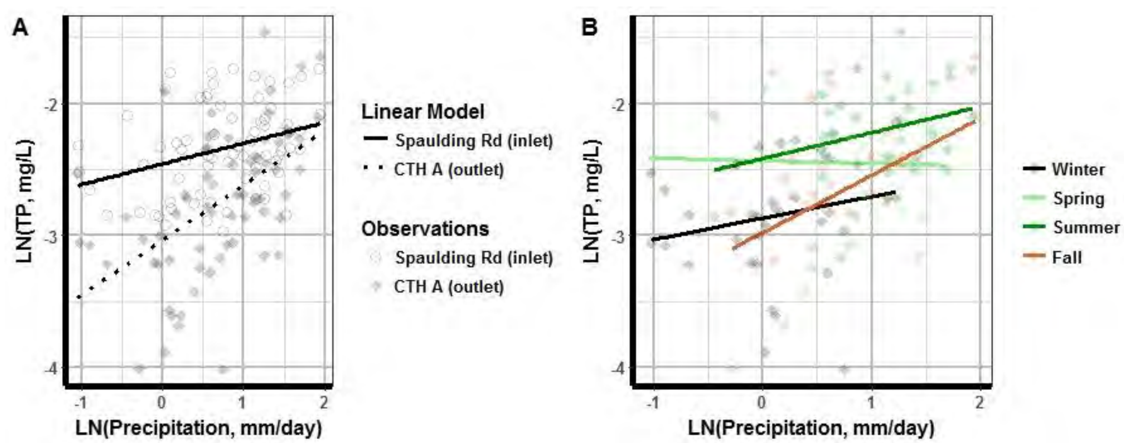

Figure 2-21: Measured and estimated flow-weighted mean TP concentrations at Spaulding Rd (inlet) and CTH A (outlet) versus monthly average precipitation. All variables are natural log transformed. Linear models span the horizontal range of their respective sub-dataset. (A) Shows interaction between site and precipitation (p = 0.017). (B) Shows interaction between season and precipitation (p = 0.045).

The overall median TP removal efficiency from Spaulding Rd to CTH A was 32%. In winter and fall, median TP removal efficiencies were approximately 40% but decreased to approximately 20% in the spring and summer. TP removal efficiency was significantly related to the monthly average precipitation ($p < 0.001$), the year ($p = 0.002$), and flow-weighted mean TP concentration at Spaulding Rd ($p = 0.045$; Appendix B Table B-5). TP concentrations at Spaulding Rd had stronger relations to removal efficiencies during winter and summer compared to fall and spring, but the differences were not significant (Figure 2-22 A). Removal efficiencies in 2012 were particularly high and significantly different from removal efficiencies between 2013 and 2016 ($p = 0.011$). Average monthly precipitation was the most significant predictor of TP removal efficiency; months with higher average precipitation corresponded to lower TP removal efficiency within SCE (Figure 2-22 B).

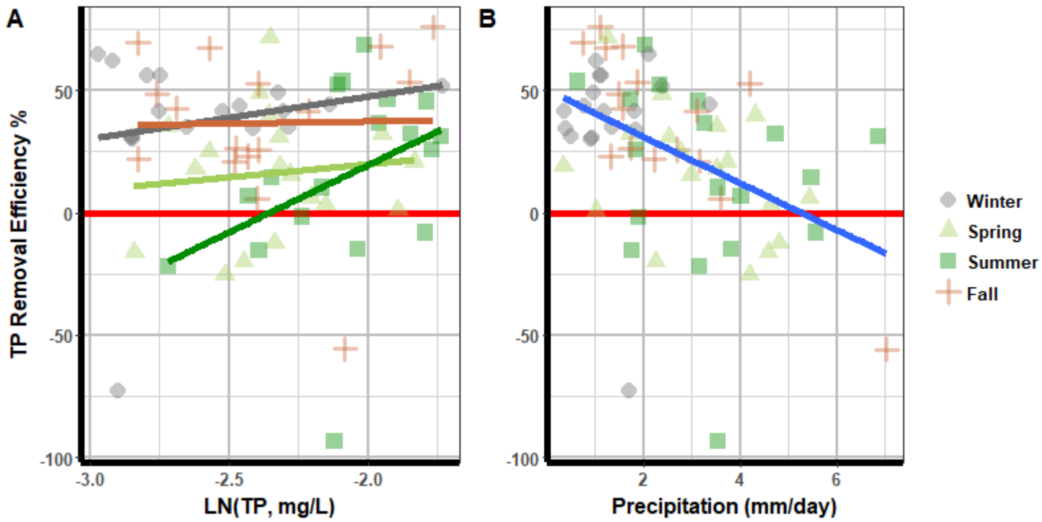

Figure 2-22: Monthly removal efficiency of TP from Spaulding Rd to CTH A using flow-weighted average monthly TP concentrations from 2011-2017. (A) Removal efficiency versus natural log of TP concentration at Spaulding Rd (p = 0.045). (B) Monthly removal efficiency versus the average daily precipitation for that month (p < 0.001).

Suspended Sediment

Similar to TP, monthly average SS loads at CTH A decreased significantly from before (overall median = 1.38 metric tonnes/day) to after (overall median 1.07 metric tonnes/day) restoration ($p < 0.001$; Table 2-11). Both before and after restoration, seasonal median SS loads followed the order winter < fall < summer < spring. Seasonal median SS loads ranged from 0.38 to 3.72 metric tonnes/day before restoration and from 0.83 to 3.26 metric tonnes/day after (Figure 2-10 E; Table 2-11). The seasonal median SS load was lower for all seasons following restoration, except for winter, which had a median 0.45 metric tonnes/day higher following restoration compared to before. These before and after restoration differences were significant in spring ($p < 0.001$) and summer ($p < 0.001$) but not in fall ($p = 0.05$) and winter ($p = 0.60$), and the greatest reduction occurred in summer (0.73 metric tonnes/day).

SS load was significantly related to interactions between season and restoration period ($p < 0.001$), season and precipitation ($p < 0.001$), and season and flow at CTH A ($p = 0.020$; Appendix B Table B-6). The RWTP upgrade status was not a significant predictor of SS load (p =

0.062). Increasing precipitation and flow were both associated with increasing SS load (Figure 2-23). As precipitation increased, spring had significantly higher SS load compared to the other seasons ($p = 0.008$), and winter had significantly less modeled SS load compared to the other seasons across the quartiles for precipitation ($p < 0.001$; Figure 2-23 A). In relation to differences in flow, winter had significantly lower SS load compared to the other seasons ($p = 0.002$), and as flow decreased to the modeled 25th percentile, spring had significantly higher SS load compared to both winter and fall ($p = 0.009$; Figure 2-23 B). SS load generally decreased as retention times increased (Figure 2-13 C), and retention times were shorter and SS loads higher in spring and summer compared to winter and fall.

Table 2-11: Seasonal median observations of SS load and flow-weighted SS concentration before and after the SCE restoration at CTH A (outlet of SCE). Number of observations also shown. Significant p values (<0.05) are bolded.

	Restoration	Winter	Spring	Summer	Fall	All Seasons
Median SS Load (metric tonnes/day)	**Before**	0.38	3.72	1.80	1.03	1.38
	After	0.83	3.26	1.07	0.92	1.07
	Difference (% Reduction)	-0.45 (20% increase)	0.46 (12%)	0.73 (40%)	0.11 (11%)	0.31 (22%)
	p value	0.600	**<0.001**	**<0.001**	0.053	**<0.001**
Median of Flow-weighted Mean SS Concentration (mg/L)	**Before**	9.85	35.80	32.80	19.40	25.50
	After	11.00	16.20	16.30	17.90	14.30
	Difference (% Reduction)	-1.15 (12% increase)	19.6 (55%)	16.5 (50%)	1.5 (8%)	11.2 (44%)
	p value	0.600	**<0.001**	**<0.001**	0.053	**<0.001**
n	**Before**	44	45	45	45	179
	During	12	12	12	12	48
	After	29	30	30	27	116

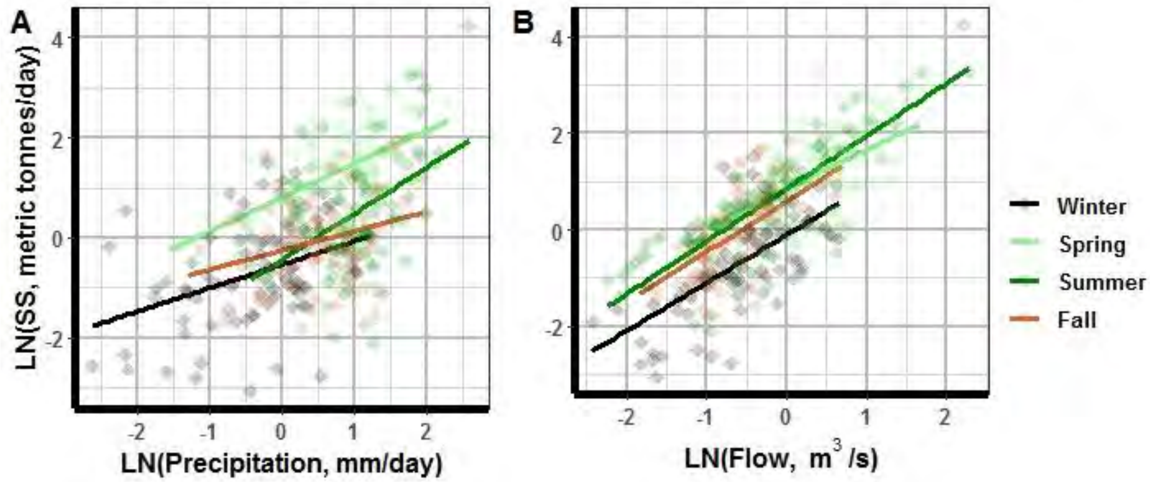

Figure 2-23: Measured and estimated monthly average SS load at SCE outlet versus (A) monthly average precipitation and (B) monthly average flow at SCE outlet. All variables are natural log transformed. Linear models span the horizontal range of the respective sub-dataset. Varying slopes and intercepts for linear models show the significant interactions between explanatory variables ($p < 0.001$ in plot A and $p = 0.02$ in plot B).

Monthly average flow-weighted SS concentrations at CTH A decreased from before (overall median = 25.5 mg/L) to after (overall median = 14.3 mg/L) SCE restoration ($p < 0.001$; Table 2-11). Before restoration, seasonal medians ranged from 9.85 – 35.8 mg/L and followed the order winter < fall < summer < spring (Figure 2-10 F). After restoration, seasonal medians ranged from 11 - 17.9 mg/L and followed the order winter < spring < summer < fall. The difference between median SS concentrations before and after restoration was smaller in winter and fall compared to spring and summer. As with SS load, SS concentrations decreased significantly following restoration in spring ($p < 0.001$) and summer ($p < 0.001$) but not in winter ($p = 0.60$) and fall ($p = 0.05$; Table 2-11). The seasonal median concentrations decreased by about 50 % in both spring and summer following restoration.

The model predicting SS load was also used to predict flow-weighted SS concentrations. While this was the most suitable model with the explanatory variables available, it was not as effective in explaining SS concentrations (adjusted $r^2 = 0.37$) as SS load (adjusted $r^2 = 0.72$). The flow-weighted mean SS concentration was significantly related to the interactions between season and restoration ($p < 0.001$), season and precipitation ($p < 0.001$), and season and flow ($p = 0.02$; Appendix B Table B-6). As precipitation increased, SS concentration increased significantly more in spring compared to the other seasons ($p < 0.008$; Figure 2-24 A). During winter, SS concentrations neither varied with precipitation nor flow and were consistently lower than the other seasons for a given amount of precipitation ($p < 0.001$) or flow ($p < 0.002$; Figure 2-24). As flow decreased, SS concentrations increased in spring and were significantly higher than both fall and winter ($p = 0.009$; Figure 2-24 B).

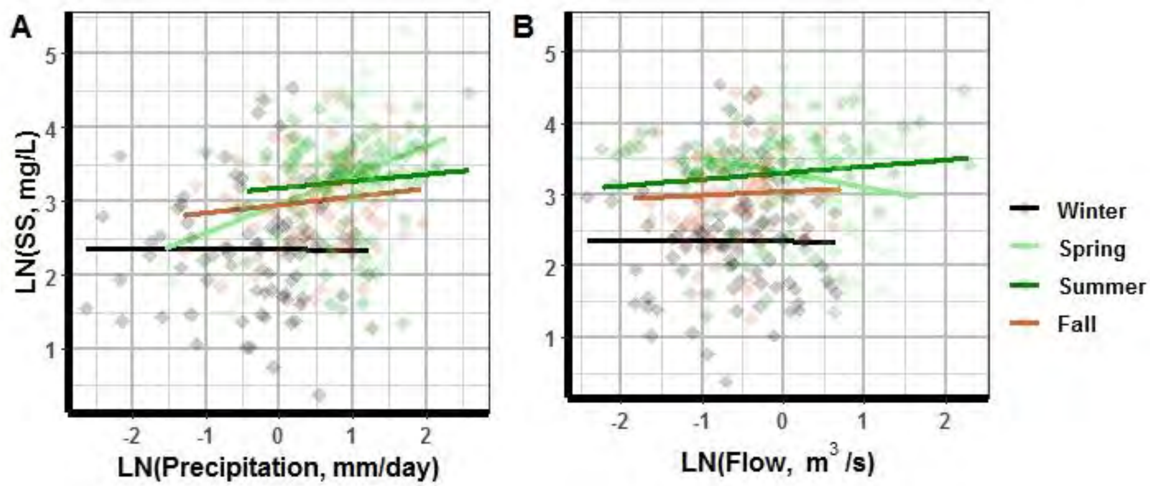

Figure 2-24: Measured and estimated monthly average flow-weighted SS concentration at SCE outlet versus (A) monthly average precipitation and (B) monthly average flow at SCE outlet. All variables are natural log transformed. Linear models span the horizontal range of the respective sub-dataset. Varying slopes and intercepts for linear models show the significant interactions between explanatory variables (p < 0.001 in plot A and p = 0.02 in plot B).

Flow-weighted mean TP and SS concentrations both had distinct decreases in spring and summer (Figure 2-25). In winter and fall, however, decreases in TP concentrations were not accompanied by significant decreases in SS concentrations. After the restoration, SS summer concentrations decreased substantially. The average summer SS concentration decrease by 64% from the summer of 2006 to 2017 (22.6 mg/l to 8.02 mg/l). In comparison, summer average TP concentrations only decreased by 19% (0.13 mg/l to 0.11 mg/L) over that same time period.

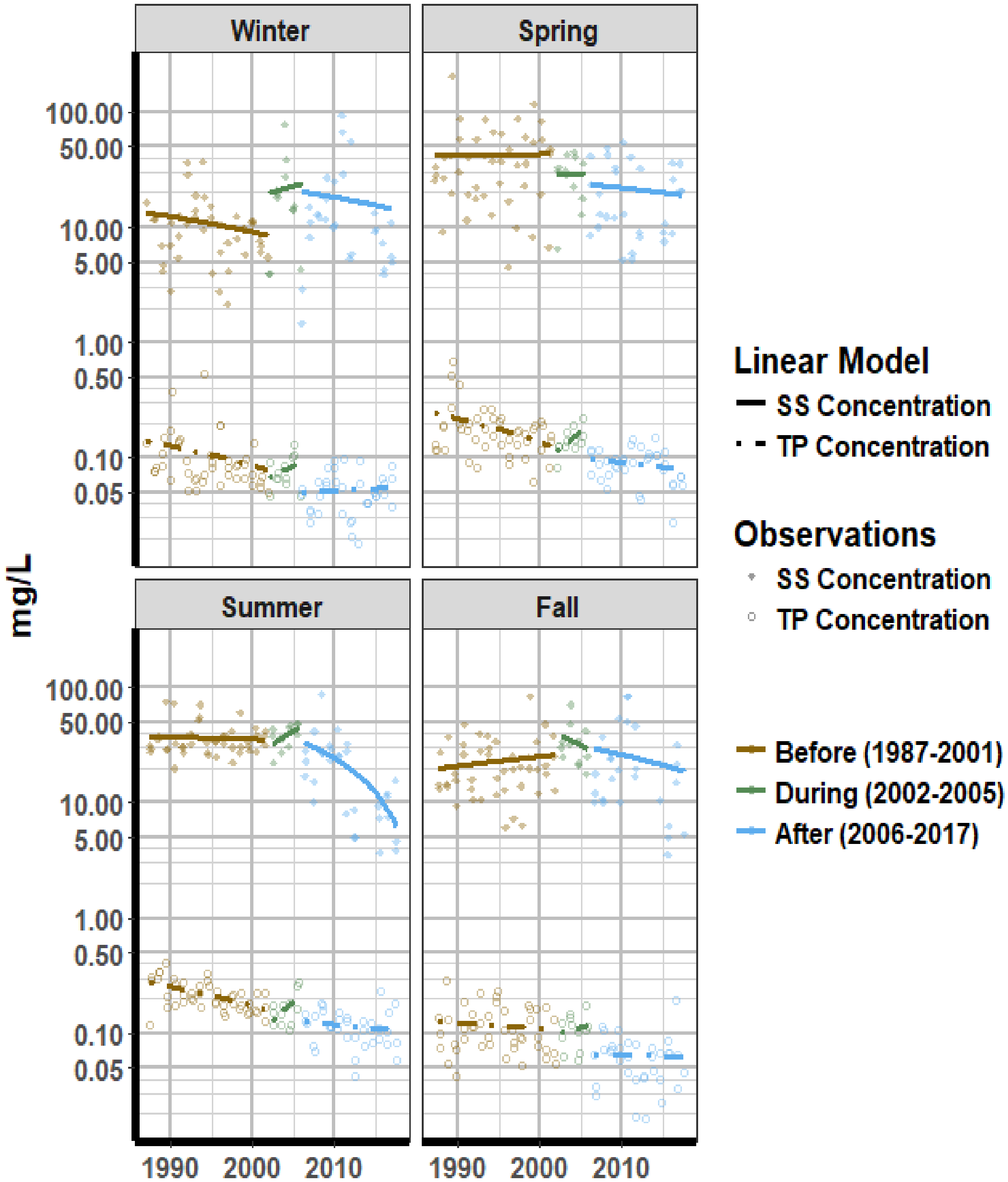

Figure 2-25: Estimated and measured flow-weighted monthly mean TP and SS concentrations at CTH A (SCE outlet) from February 1987 – September 2017. Linear model represents seasonal trend over restoration period (Before, During, After).

Suspended Sediment Retention in SCE

Seasonal average SS loads at both CTH A, over 30 years, and at Spaulding Rd, over 6 years, were greater overall in spring and summer than in winter and fall (Figure 2-26). Since

Spaulding Rd has been monitored, there have not been any seasons with high (>10 metric tonnes/day) SS load at CTH A. The largest average SS load (27.2 metric tonnes/day) measured at CTH A occurred in the summer of 2008, which was also the period with the highest average rain (6.2 mm/day). Seasonal average SS load at Spaulding Rd exceeded 10 metric tonnes/day only once (spring 2017; 10.5 metric tonnes/day) which was a season with average precipitation of 3.3 mm/day.

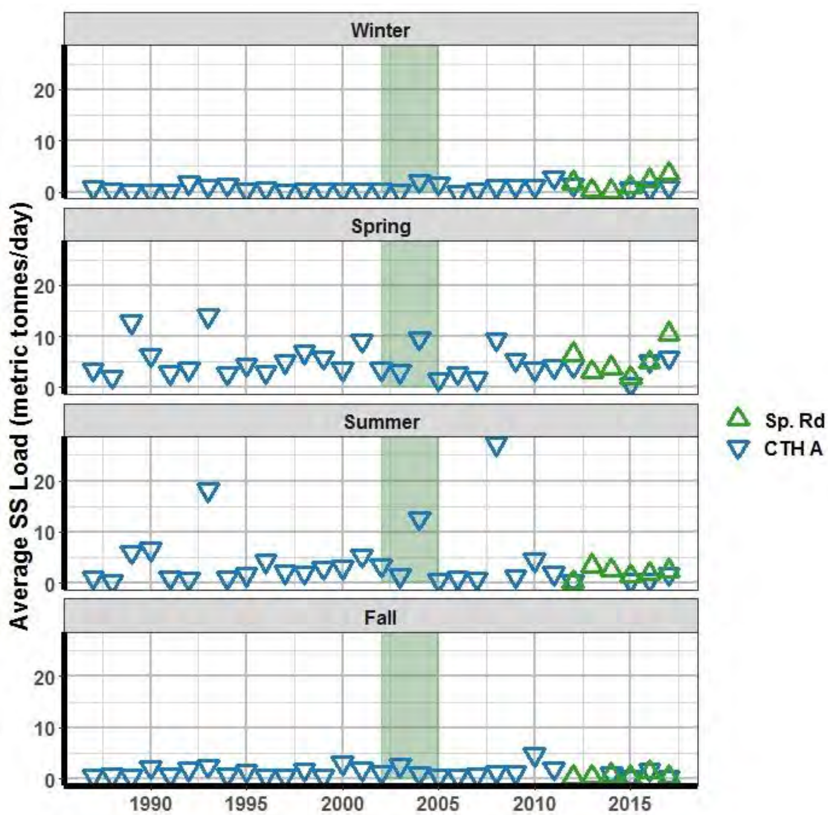

Figure 2-26: Seasonal average SS load at CTH A (SCE outlet; 1987-2017) and Spaulding Rd (SCE inlet; 2011-2017). Restoration period (2002-2005) is shaded in green.

Overall from 2011 to 2017, monthly average SS load at Spaulding Rd (median = 1.64 metric tonnes/day) was significantly greater than at CTH A (median = 0.83 metric tonnes/day; $p < 0.001$; Table 2-12). At Spaulding Rd, seasonal median SS load ranged from 0.56 – 4.85 metric tonnes/day and followed the order fall < summer < winter < spring (Figure 2-27 A). At CTH A, seasonal median SS load ranged from 0.49 – 3.18 metric tonnes/day and followed the order summer < winter < fall < spring. The relationship between SS load at Spaulding Rd and CTH A varied seasonally. The SS load at Spaulding Rd was significantly higher than at CTH A in all seasons but fall. In fall, the median SS load at CTH A (0.71 metric tonnes/day) exceeded that at Spaulding Rd (0.56 metric tonnes/day), but the difference was not significant ($p = 0.75$). The SS load was significantly related to site ($p < 0.012$) and an interaction between season and flow at Spaulding Rd ($p = 0.008$; Appendix B Table B-7). With increasing flow, spring SS load increased more than during winter ($p = 0.034$; Figure 2-28 A).

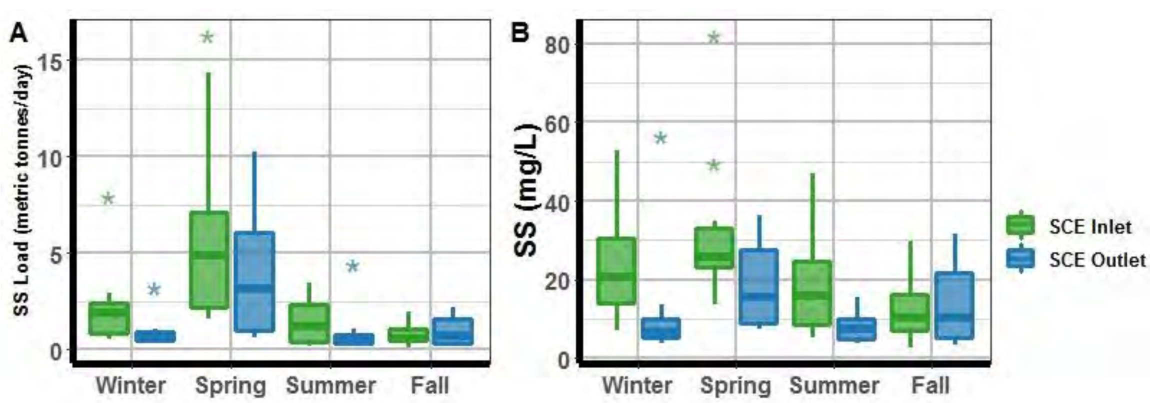

Figure 2-27: (A) Monthly average SS load and (B) flow-weighted TP concentrations at the inlet and outlet of SCE using data from December 2011 – September 2017. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

Table 2-12: Seasonal median SS load and flow-weighted SS concentrations at the SCE inlet (Spaulding Rd) and outlet (CTH A) for data from December 2011 – September 2017. P values indicate statistical differences between inlet and outlet. Significant p values (<0.05) are bolded.

		Winter	Spring	Summer	Fall	All Seasons
Median SS Load (metric tonnes/day)	**Inlet**	1.86	4.85	1.20	0.56	1.64
	Outlet	0.68	3.18	0.49	0.71	0.83
	Difference (% Reduction)	1.18 (63%)	1.67 (34%)	0.71 (59%)	-0.15 (28% increase)	0.81 (49%)
	p value	**<0.001**	**0.042**	**0.012**	0.750	**<0.001**
Median Flow-weighted Mean SS Concentration (mg/L)	**Inlet**	20.5	25.9	15.9	10.4	18.6
	Outlet	6.6	15.7	7.5	10.5	9.0
	Difference (% Reduction)	13.9 (68%)	10.2 (39%)	8.4 (53%)	-0.1 (1% increase)	9.6 (51%)
	p value	**<0.001**	**0.020**	**0.005**	0.980	**<0.001**
n		12	12	12	9	45

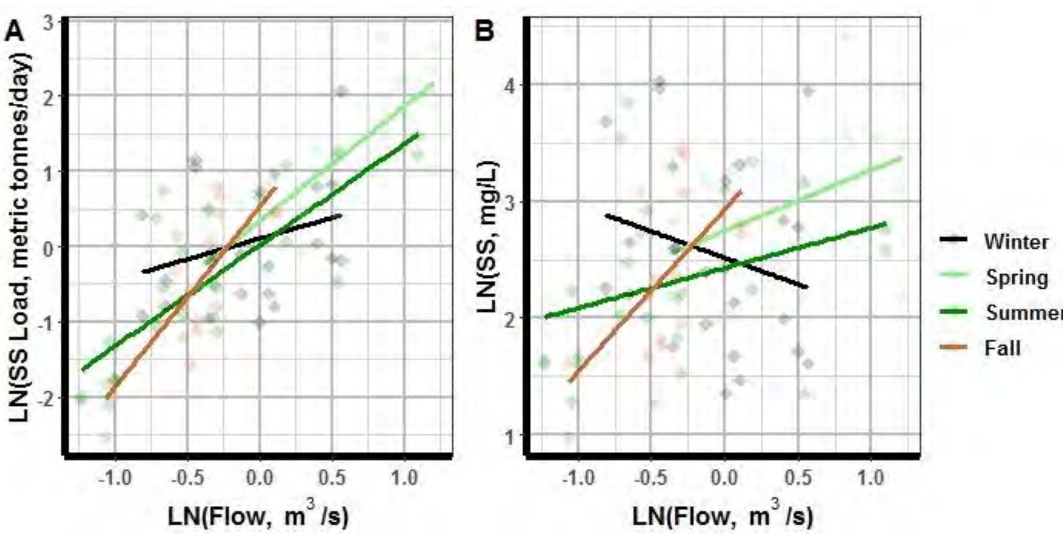

Figure 2-28: (A) Measured and estimated monthly average SS load at SCE inlet and outlet versus monthly average flow at SCE inlet (2011-2017). (B) Measured and estimated monthly average flow-weighted SS concentration at SCE inlet and outlet versus monthly average flow at SCE inlet (2011-2017). Variables are natural log transformed. Linear models span the horizontal range of the respective sub-dataset. Varying slopes and intercepts for linear models show the significant interaction between explanatory variables, season and flow (p = 0.018 in plot A and p = 0.008 in B).

Differences between monthly average SS load at the inlet and outlet were positive (indicating SS retention) in 70% of the observation period (Figure 2-29). In months when Spaulding Rd SS load exceeded that at CTH A, SCE retained a total of 1,646 metric tonnes. In months when Spaulding Rd SS load was less than that at CTH A, the marsh released 235 metric tonnes. Over the monitored period, the SS load was 3,729 metric tonnes at Spaulding Rd, and 2,336 metric tonnes at CTH A. Therefore, a net total of 1,393 metric tonnes of SS were retained. This is equivalent to an average daily retention of 1.02 metric tonnes of SS. Approximately half, a total of 723 metric tonnes, was retained in the spring. The largest monthly average SS retentions within SCE occurred in February and March of 2017 (USGS provisional data). Fall had an overall export of 30 metric tonnes of SS but that was heavily influenced by the export of 53 metric tonnes in November 2016 (USGS provisional data). In 44 of the 45 monitored months, SCE retained either TP, SS or both (Figure 2-30 A). In 24 of those 44 months (53%), SCE retained both SS and TP. There was only one month (June 2017; Figure 2-30) when SCE did not retain either SS or TP. SS retention within SCE was only positively related to SS load at Spaulding Rd ($p = 0.002$; Figure 2-31). Season ($p = 0.76$), precipitation ($p = 0.95$), and flow at Spaulding Rd ($p = 0.27$) were all not significant in explaining SS retention.

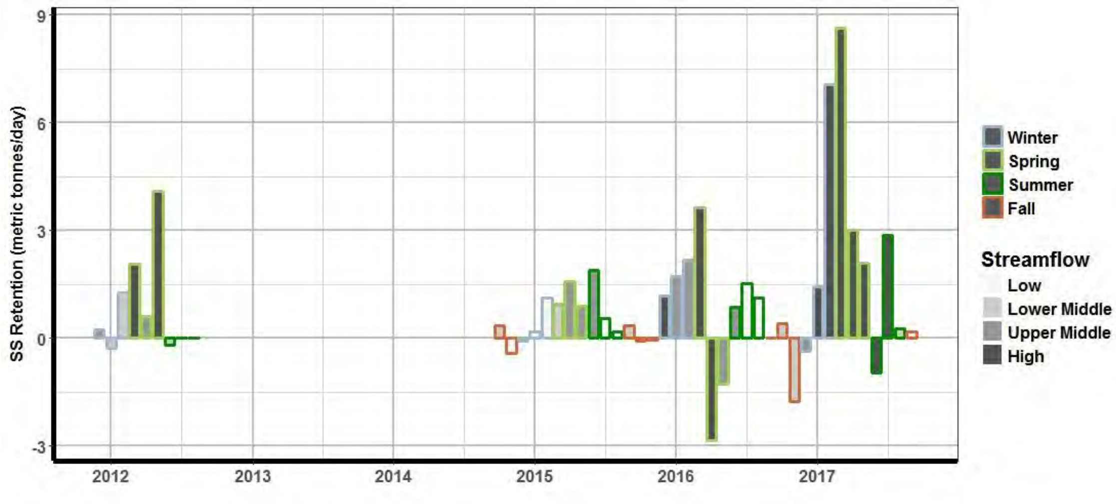

Figure 2-29: SS retention equals monthly average SS load Spaulding RD (SCE inlet) minus monthly average SS load at CTH A (SCE outlet). Positive retention signifies SCE stored SS. For negative quantities, it exported additional SS than what flowed in. Gradation shows relative monthly average streamflow at Spaulding Rd. Average monthly streamflow for period was divided into four categories (low, lower middle, upper middle, and high) by quartiles.

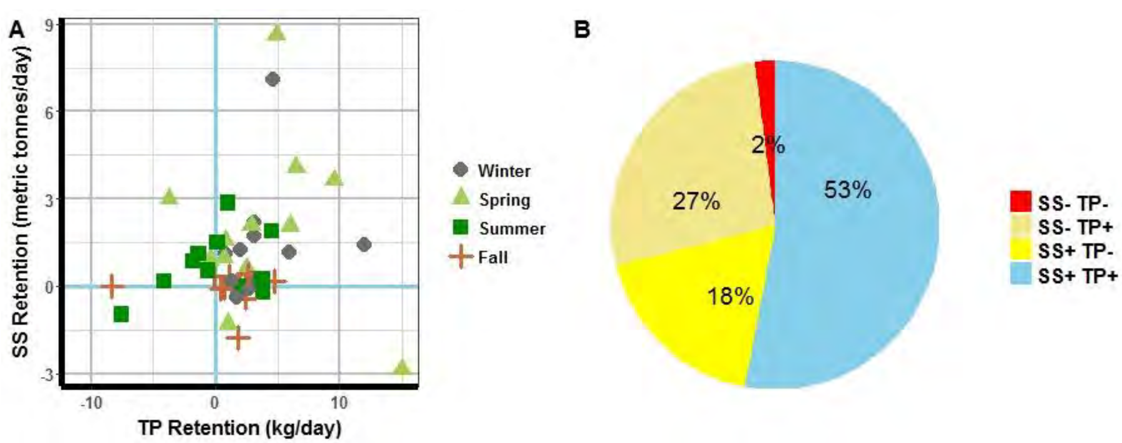

Figure 2-30: (A) Plot of monthly average TP retention versus monthly average SS retention within SCE. Retention equals monthly average load at Spaulding Rd (SCE inlet) minus monthly average load at CTH A (SCE outlet). (B) Percentage of observations within each quadrant of (A) showing positive or negative retention of TP or SS within SCE.

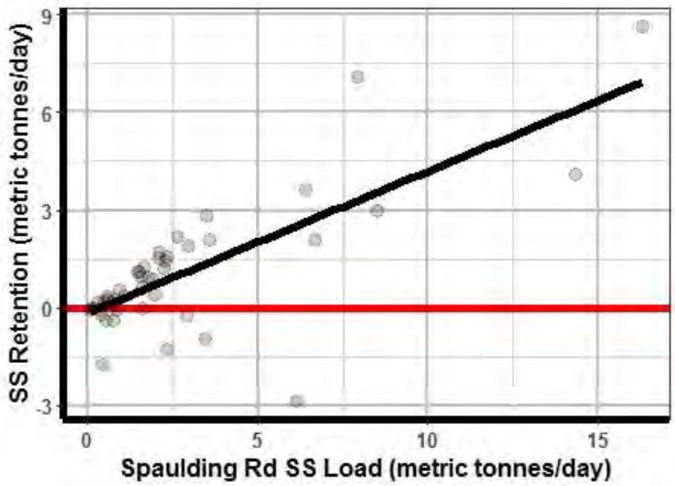

Figure 2-31: Monthly average SS load at Spaulding Rd versus monthly average SS retention within SCE. Retention equals monthly average SS load at Spaulding Rd (SCE inlet) minus monthly average SS load at CTH A (SCE outlet).

Monthly average flow-weighted mean SS concentrations significantly decreased from Spaulding Rd (overall median = 18.6 mg/L) to CTH A (overall median = 9 mg/L; $p < 0.001$; Table 2-12). At Spaulding Rd, seasonal median SS concentrations ranged from 10.4 - 25.9 mg/L and followed the order fall < summer < winter < spring. At CTH A, the seasonal median SS concentrations ranged from 6.6 - 15.7 mg/L and followed the order winter < summer < fall < spring (Figure 2-27 B). In all seasons except fall, the seasonal median SS concentration at CTH A was significantly lower than at Spaulding Rd (winter, $p < 0.001$; spring, $p = 0.02$; summer, $p = 0.005$). In fall, there was no significant difference in SS concentrations between the two sites ($p = 098$). Flow-weighted mean SS concentration was significantly related to site ($p < 0.005$) and the interaction between season and flow at Spaulding Rd ($p = 0.008$; Appendix B Table B-7). As flow increased, winter SS concentrations decreased below the spring SS concentrations ($p = 0.034$; Figure 2-28 B).

Overall, the median removal efficiency of SS from Spaulding Rd to CTH A was 40%. The seasonal median SS removal efficiency was lower in the fall (8%) compared to the other three seasons (ranged from 47-61%; Figure 2-32 A). In November 2016, the difference between flow-weighted mean SS concentrations at Spaulding Rd (7 mg/L) and CTH A (31.7 mg/L) was large and resulted in an export of SS (a removal efficiency of -353%; Figure 2-29; not shown in Figure 2-32). Flow-weighted SS concentration at Spaulding Rd was the only significant predictor ($p < 0.001$) of removal efficiency (as SS concentration increased, removal efficiency increased; Figure 2-32 B; Appendix B Table B-7). Insignificant predictors were flow at Spaulding Rd ($p = 0.204$), precipitation ($p = 0.184$), and season ($p = 0.302$; Figure 2-32 A). The outlier from November 2016 was not used in the prediction model.

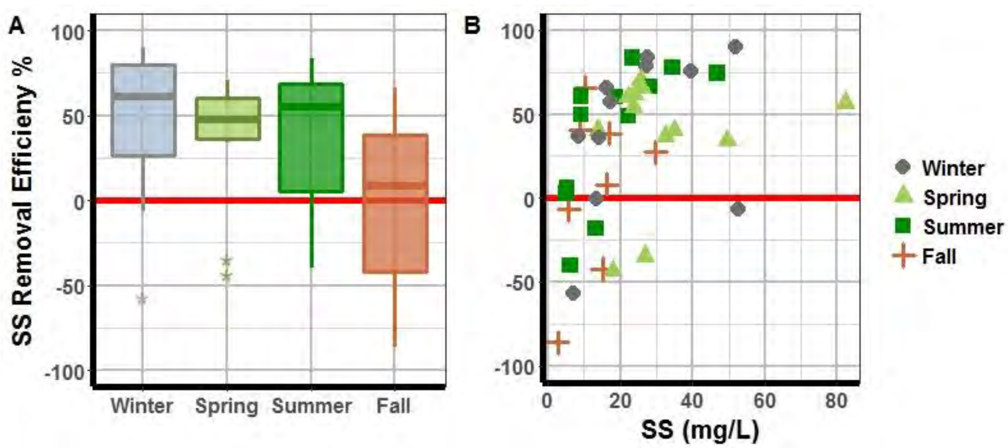

Figure 2-32: Removal efficiency of SS from the input to output of SCE using monthly average flow-weighted SS concentrations. Positive removal signifies SCE retained SS, and for negative quantities, SCE exported SS to Green Lake. Both plots exclude the outlier of -353% SS removal efficiency in November 2016. (A) Removal efficiency grouped by season. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually. (B) Monthly removal efficiency plotted versus the monthly average flow-weighted SS concentration at Spaulding Rd.

Sample Concentrations within Silver Creek Estuary

Phosphorus

Total Phosphorus

Samples were collected periodically at Spaulding Rd six years before the gage was installed and have continued to be collected since. Over the period of December 2005 – October 2017, the overall median TP concentration in the samples collected at Spaulding Rd (0.129 mg/L) was higher than the overall median TP concentration for samples at CTH A (0.108 mg/L; Table 2-13). This was a significant decrease in TP concentration ($p < 0.001$). Seasonal median TP concentrations at Spaulding Rd ranged from 0.102 - 0.174 mg/L and followed the order spring < winter < fall < summer (Figure 2-33). At CTH A, seasonal median TP concentrations ranged from 0.046 - 0.153 mg/L and followed the order winter < fall < spring < summer. For all seasons, the decrease in concentrations from Spaulding Rd to CTH A was significant ($p \leq 0.001$) even though the median TP concentration during spring at Spaulding Rd (0.102 mg/L) and CTH A (0.103 mg/L) was very close. The largest difference in seasonal median TP concentration at Spaulding Rd (0.113 mg/L) and CTH A (0.067 mg/L) occurred in winter.

The TP concentrations at the inlet and outlet were significantly related to relative amount of daily rain (i.e. the amount of rain categorized as “no rain”, “low”, “lower middle”, “upper middle”, and “high”; $p = 0.032$), sampling period ($p < 0.001$), and the interactions between flow at the inlet and year ($p < 0.001$), season and site ($p =0.007$) and season and year ($p < 0.001$; Appendix B Table B-8). During Period 1 of the USGS sampling (December 2005 – September 2011), TP concentrations were monitored more frequently at CTH A than at Spaulding Rd. In Period 2 (November 2011 to October 2017), once the Spaulding Rd gage was installed, sampling was more frequent at that location. Over the sampling period, samples were collected slightly more often during increasing flow (Figure 2-34 A). At Spaulding Rd, winter sampling occurred during increasing flow, especially over the winters of 2015-2016 and 2016-2017. The TP concentrations decreased slightly over the sampling period in all seasons except during winter at both Spaulding Rd and CTH A (Figure 2-25 B). Winter TP concentrations increased more over time at Spaulding Rd than at CTH A. Increased TP concentrations were most associated with increasing flow in winter (Figure 2-34 C).

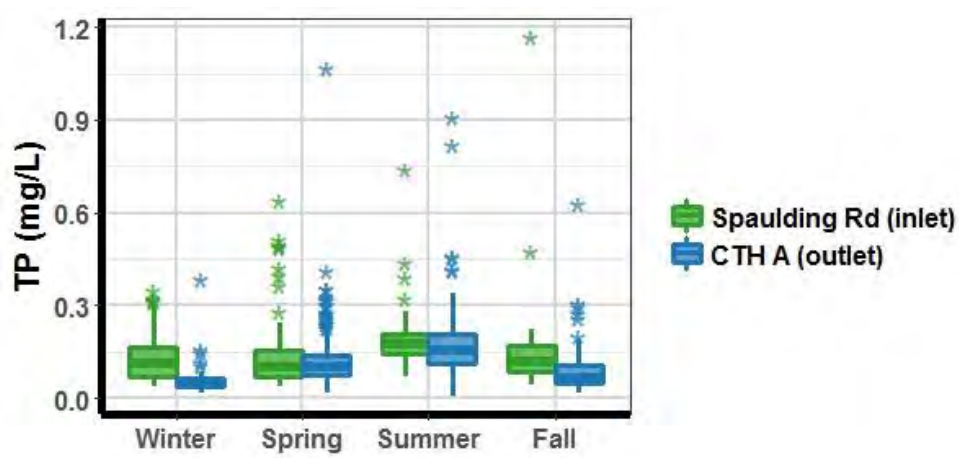

Figure 2-33: Seasonal TP concentrations from grab samples at the Spaulding Rd (inlet) and CTH A (outlet) of SCE from December 2005 – October 2017). The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

Table 2-13: Sample median TP concentrations at the inlet and outlet of SCE in Period 1 (December 2005 – September 2011) and Period 2 (October 2011 – October 2017). Reductions in seasonal median TP concentrations from the inlet to outlet and number of samples also shown.

	Site	Winter	Spring	Summer	Fall	All Seasons
Median TP (mg/L)	**Spaulding Rd (inlet)**	0.113	0.102	0.174	0.117	0.129
	CTH A (outlet)	0.046	0.103	0.153	0.066	0.108
	Difference (% Reduction)	0.067 (59%)	-0.001 (1% increase)	0.021 (12%)	0.051 (44%)	0.021 (16%)
	p value	**<0.001**	**0.001**	**<0.001**	**<0.001**	**<0.001**
n	**Spaulding Rd (inlet)**	97	181	108	28	414
	CTH A (outlet)	51	368	217	101	737

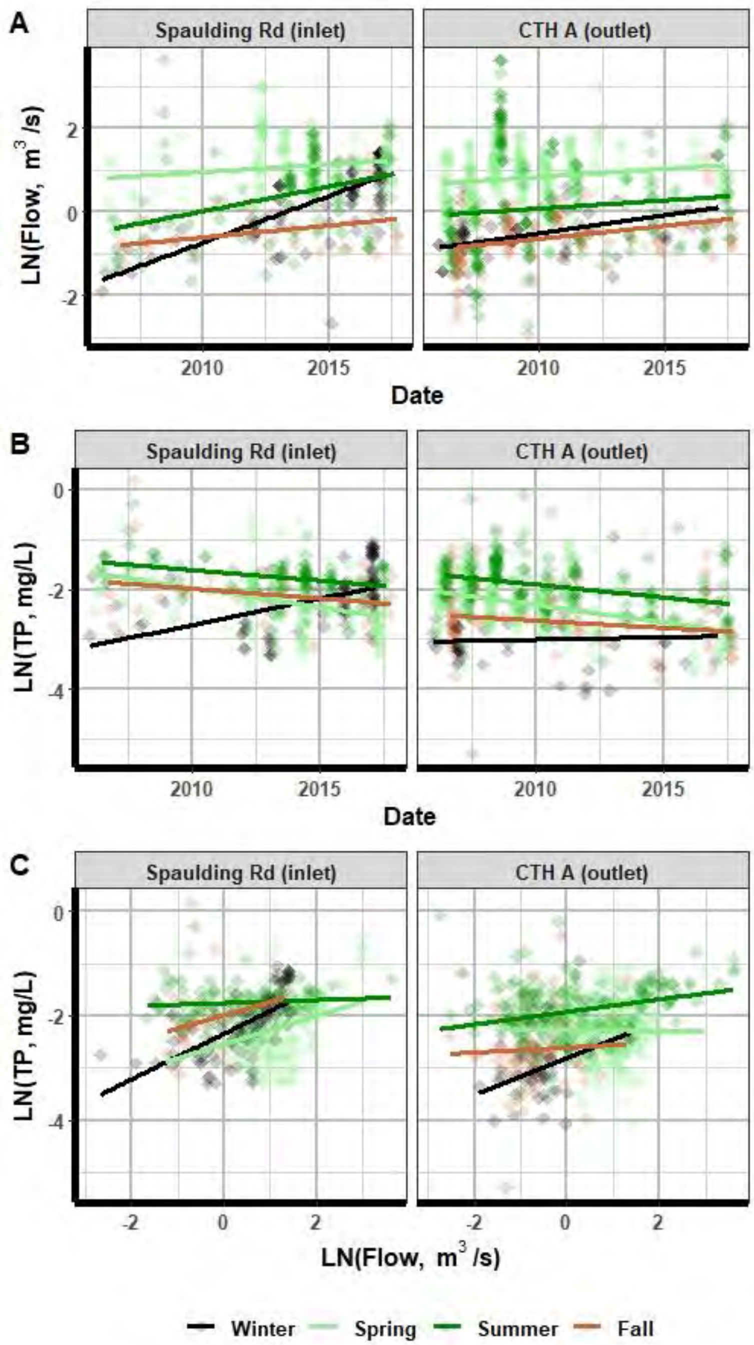

Figure 2-34: (A) Seasonal trends in measured and estimated flow at Spaulding Rd (inlet) when TP samples were collected at the inlet and outlet from December 2005 – October 2017. The year and flow interaction was a significant predictor of TP concentrations ($p < 0.001$) (B) Seasonal trends in measured and estimated sample TP concentrations collected from December 2005 – October 2017. The season and year interaction was a significant predictor of TP concentrations ($p < 0.001$). (C) Seasonal trends in measured and estimated sample TP concentrations versus flow at the inlet. Both flow and TP concentrations are natural log transformed.

For samples collected on the same day at each of the four locations within SCE, overall median TP concentrations decreased significantly from upstream Spaulding Rd (0.15 mg/L) to downstream CTH A (0.092 mg/L; $p < 0.001$; Table 2-14). At Spaulding Rd, Site 1, and Site 2, seasonal median TP concentration followed the order spring < fall < summer (Figure 2-35). At CTH A, seasonal median TP concentrations followed the order fall < spring < summer. TP concentrations significantly decreased from Spaulding Rd to CTH A in summer and fall ($p < 0.001$) but not in spring ($p = 0.775$). The largest percent decrease (50%) in median TP concentration from the Spaulding Rd (0.119 mg/L) to CTH A (0.059 mg/L) occurred in fall and the smallest (18%) occurred in the spring (0.078 mg/L at Spaulding Rd and 0.064 mg/L at CTH A). The largest difference in median concentration between Spaulding Rd and CTH A occurred in the summer (0.076 mg/L). TP concentrations were significantly related to season ($p < 0.001$), relative amount of daily precipitation ($p = 0.026$), site ($p < 0.001$), year ($p < 0.001$), and flow at Spaulding Rd ($p = 0.003$; Appendix B Table B-8). The TP concentrations were significantly lower in spring compared to summer ($p < 0.001$) and at Site 2 and CTH A compared to Spaulding Rd ($p = 0.008$). The TP concentrations were significantly higher following a large amount of precipitation in the past day (above the 75th percentile) compared to no rain ($p < 0.007$), and TP concentrations increased with increasing flow ($p = 0.003$).

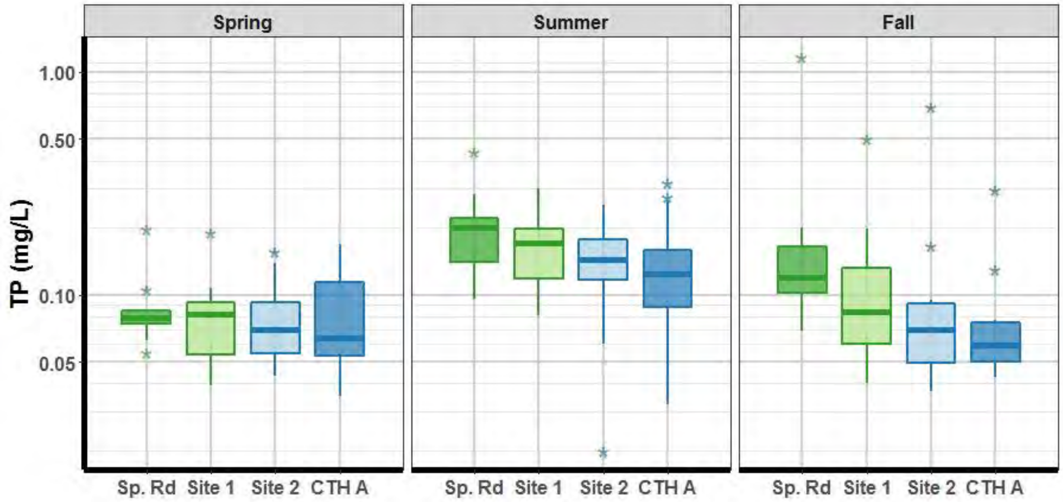

Figure 2-35: TP concentrations from days with samples collected at Spaulding Rd (Sp. Rd; inlet), CTH A (outlet) and two interior sites of SCE (USGS: May 2006 – September 2012, BSE: July 2016 – October 2017). The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individual.

Table 2-14: Seasonal median TP concentrations and number of TP grab samples collected on concurrent days at four sites within SCE (USGS: May 2006 – September 2012, BSE: July 2016 – October 2017). Median difference is the reduction from the inlet to outlet.

	Site	Spring	Summer	Fall	All Seasons
Median TP (mg/L)	**Spaulding Rd (inlet)**	0.078	0.200	0.119	0.150
	Site 1	0.082	0.171	0.084	0.123
	Site 2	0.069	0.144	0.070	0.118
	CTH A (outlet)	0.064	0.124	0.059	0.092
	Difference (% Reduction)	0.014 (18%)	0.076 (38%)	0.060 (50%)	0.021 (16%)
	p value	0.775	**<0.001**	**<0.001**	**<0.001**
n	**Spaulding Rd (inlet)**	10	29	11	50
	Site 1	9	27	11	47
	Site 2	8	27	11	46
	CTH A (outlet)	11	30	12	53

On 137 days, samples were collected at both Spaulding Rd and CTH A; removal efficiencies were calculated for these days. Overall median TP removal efficiency was 36%. Winter (n= 19) and fall (n=23) had seasonal median TP removal efficiencies of 45%. Lower removal efficiencies were observed in spring (median = 34%; n = 47) and summer (median = 32%; n = 48). TP removal efficiency was significantly related to the categorical amount of rain in the preceding three days (p = 0.01), TP concentration at Spaulding Rd (p = 0.001), and the interaction between year and flow at Spaulding Rd (p = 0.023; Appendix B Table B-8). Season was not a significant predictor (p = 0.124). The TP removal efficiency increased as TP concentrations at Spaulding Rd increased (Figure 2-36 A). Samples taken on days with more than 19.55 mm of precipitation in the past three days (categories Upper Middle and High) had statistically lower removal efficiencies than samples taken with no rain over the preceding three days (p = 0.013; Figure 2-36 B).

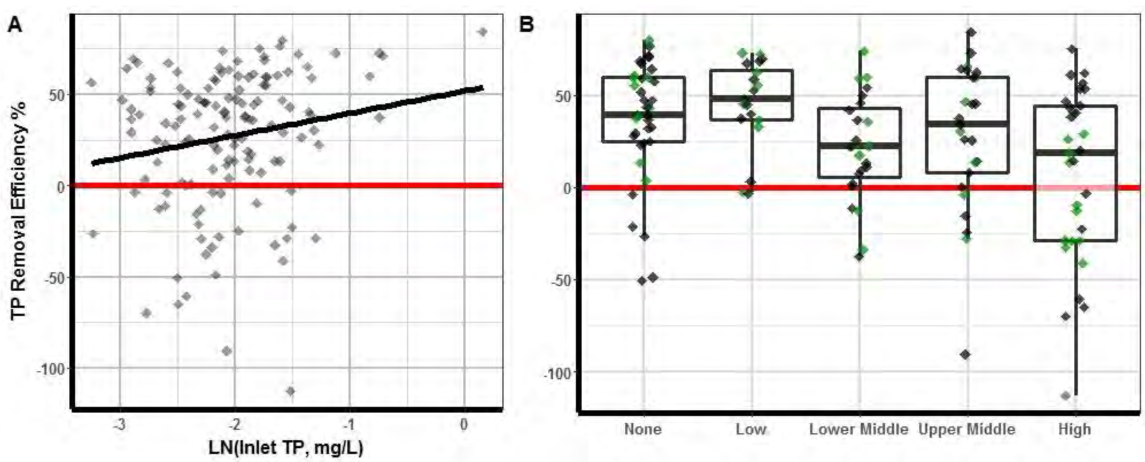

Figure 2-36: TP removal efficiency for grab samples collected on the same day at Spaulding Rd and CTH A. Data were collected February 2006 – October 2017 by USGS and BSE. (A) TP removal efficiency versus TP concentrations at the inlet (Spaulding Rd). Spaulding Rd concentrations were a significant predictor of removal efficiencies (p = 0.001).TP concentrations are natural log transformed (B) TP removal efficiency grouped by categorical amount of precipitation over the preceding three days, another significant predictor (p = 0.01). Summer data are green, all other seasons are black. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individual. None n = 37, Low n = 20, Lower Middle n = 24, Upper Middle n = 25, High n = 31.

Total Dissolved Phosphorus

Overall, median TDP concentrations significantly decreased from Spaulding Rd (0.068 mg/L) to CTH A (0.034 mg/L; $p = 0.027$; Table 2-15). At Spaulding Rd, seasonal median TDP concentrations ranged from 0.043 – 0.114 mg/L and followed the order winter < spring < fall < summer (Figure 2-37). At CTH A, the seasonal medians ranged from 0.018 – 0.042 mg/L and followed the order fall < winter < spring < summer. For all seasons, the median TDP concentration was higher at Spaulding Rd than at CTH A; however, this difference was only significant in summer ($p < 0.001$). TDP was significantly related to site ($p < 0.001$), season ($p = 0.045$), TP concentration ($p < 0.001$), and the interaction between year and flow at Spaulding Rd ($p = 0.008$; Appendix B Table B-9). TDP concentrations increased with increasing TP concentrations (Figure 2-38 A) and with increasing flow (Figure 2-38 B).

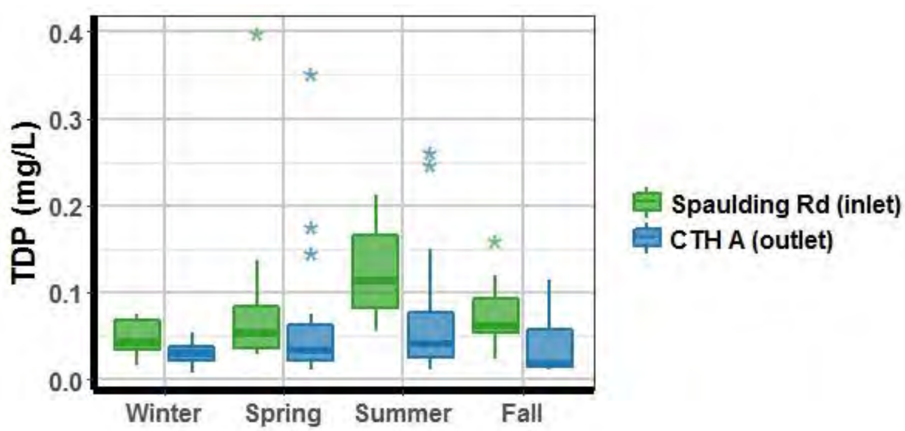

Figure 2-37: TDP concentrations at Spaulding Rd and CTH A grouped by season. Samples were collected from January 2006 – October 2017 by the USGS and BSE. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individual.

Table 2-15: Median TDP concentrations at the inlet and outlet of SCE grouped by season. Samples were collected from January 2006 – October 2017 by the USGS and BSE. Statistical difference between inlet and outlet samples shown. Significant differences (p <0.05) are bolded. Number of TDP samples also shown.

	Site	Winter	Spring	Summer	Fall	All Seasons
Median TDP (mg/L)	**Spaulding Rd (inlet)**	0.043	0.054	0.114	0.063	0.068
	CTH A (outlet)	0.030	0.034	0.042	0.018	0.034
	Difference (% Reduction)	0.013 (30%)	0.020 (37%)	0.072 (63%)	0.045 (71%)	0.034 (50%)
	p value	0.859	0.335	**<0.001**	0.153	**0.027**
n	**Spaulding Rd (inlet)**	8	23	19	10	60
	CTH A (outlet)	8	25	32	11	76

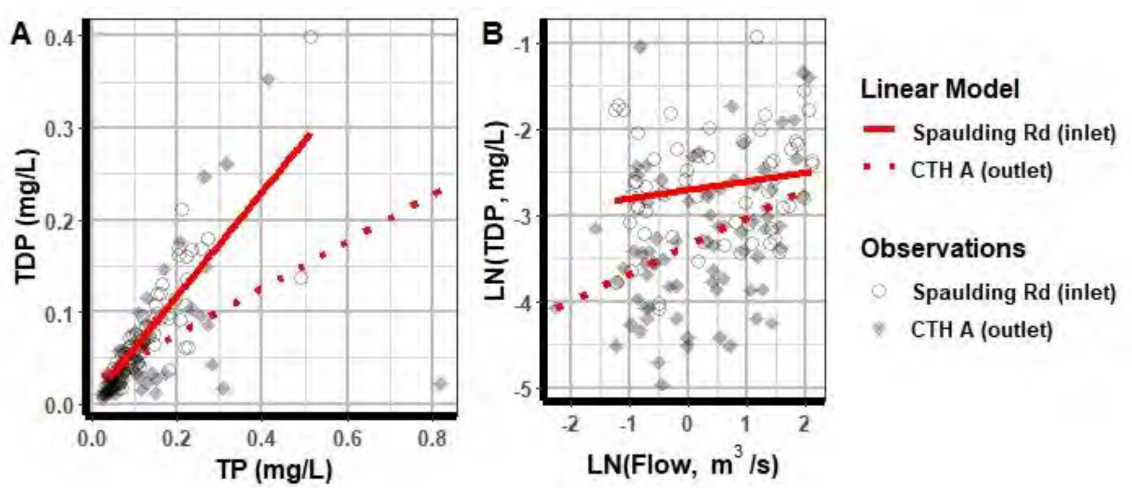

Figure 2-38: (A) Sample TDP concentrations versus TP concentrations at Spaulding Rd and CTH A. (B) TDP concentration at Spaulding Rd and CTH A versus flow at Spaulding Rd. Both variables are naturally log transformed.

The ratio of TDP:TP varied from 0.03 to 1, with a median of 0.55, for the 135 samples when both TDP and TP were measured. The median TDP:TP at Spaulding Rd was 0.60 (n= 60), and the median at CTH A was 0.48 (n = 75). The two samples with the smallest TDP:TP (0.03 and 0.06) corresponded to the largest SS concentrations (489 and 240 mg/L, respectively). Overall, TDP:TP decreased significantly from Spaulding Rd to CTH A ($p = 0.001$). Seasonally, the decrease was significant in spring ($p = 0.017$) and summer ($p = 0.015$). SS concentration was a significant predictor of TDP:TP ($p = 0.001$; Figure 2-39; Appendix B Table B-9) based on the subset of data analyzed for SS, TDP and TP (Spaulding Rd, $n = 40$ and CTH A, $n = 53$). TDP:TP was also significantly related to categorical precipitation within the past seven days ($p < 0.001$), site ($p < 0.001$), and the interaction between year and flow at Spaulding Rd ($p < 0.001$). TP concentration ($p = 0.28$) and season ($p = 0.88$) were not significant predictors. TDP:TP increased with increasing precipitation over the preceding seven days and with increasing flow.

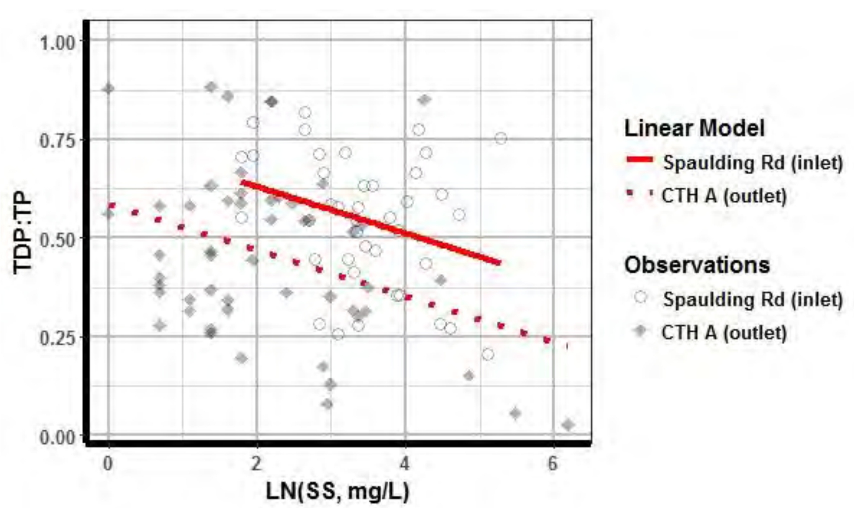

Figure 2-39: Measured and estimated TDP:TP ratio versus SS concentration. SS concentrations are natural log transformed. Samples were collected from January 2006 – October 2017 by the USGS and BSE. Linear models extend to range of SS in sub-dataset. SS was a significant factor in predicting TDP:TP within SCE (p = 0.001)

TDP concentrations were measured at both Spaulding Rd and CTH A on 33 occasions and had an overall median TDP removal efficiency of 43%. Seasonal median removal efficiency followed the order of spring (30%, n=11) < winter (36%, n = 5) < fall (46%, n = 9) < (summer 63%; n = 11; Figure 2-40 A). All winter TDP removal efficiencies were positive, but the sample size was small. The other seasons had at least one occasion with negative removal. TDP removal efficiency was significantly related to TDP concentration at Spaulding Rd ($p = 0.019$), and year (p

= 0.05) but not by season (p = 0.161), streamflow (p = 0.057), or the categorical amount of precipitation over the preceding seven days (p = 0.861; Appendix B Table B-9). In summer, TDP removal efficiencies increased with increasing TDP concentrations at Spaulding Rd (Figure 2-40 B). This did not occur in the other seasons, but the interaction was not significant (p = 0.58, removed from model).

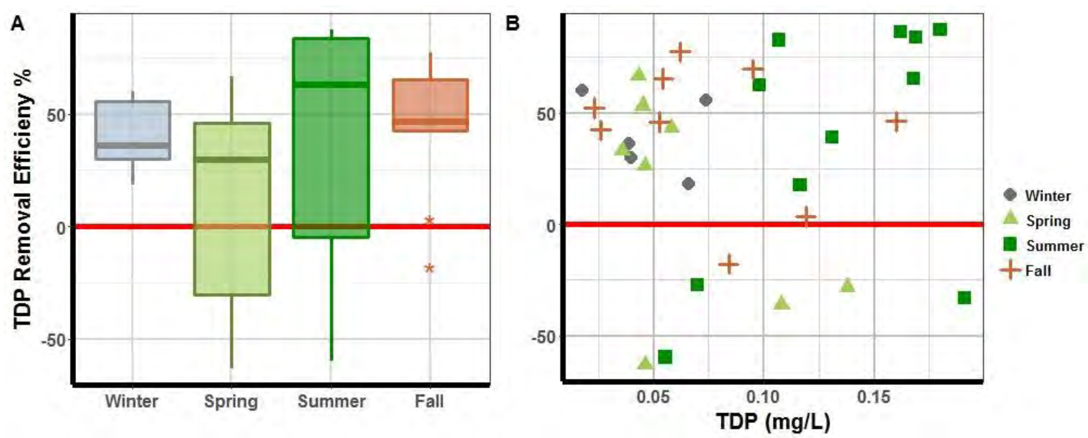

Figure 2-40: TDP removal efficiency from Spaulding Rd (inlet) to CTH A (outlet) of SCE for grab samples collected within a period of three days. Data were collected July 2006 – October 2017 by USGS and BSE. (A) TDP removal efficiency grouped by season. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individual. (B) TDP removal efficiency versus TDP concentration at the inlet.

Suspended Sediment

During Period 1 (December 2005 – September 2011), the overall median SS concentration at Spaulding Rd was 37 mg/L and at CTH A was 34 mg/L (Table 2-16). The seasonal medians at Spaulding Rd ranged from 29 – 61 mg/L and followed the order spring < summer < winter = fall (Figure 2-41). At CTH A, which was sampled more frequently, the seasonal medians ranged from 29 – 49 mg/L and followed the order spring < winter < summer < fall. During Period 1, no statistical differences were observed between Spaulding Rd and CTH A, overall or in any season. The largest difference (44%) between the seasonal median at Spaulding

Rd (61 mg/L) and CTH A (34 mg/L) occurred in winter, but the difference was not significant (p = 0.076).

During Period 2 (November 2011 – October 2017), all comparable (same season and same site) overall and seasonal medians were lower than for Period 1. The overall median at Spaulding Rd (24 mg/L) was significantly greater than at CTH A (6 mg/L; $p < 0.001$; Table 2-16). Seasonal medians at Spaulding Rd, which was sampled more frequently, ranged from 16 – 28 mg/L and followed the order fall < winter < spring < summer. At CTH A, seasonal medians ranged from 4 – 8 mg/L and followed the order summer < winter < fall < spring. The percent reductions from Spaulding Rd to CTH A ranged from 55% (fall) to 86% (summer). During Period 2, SS concentrations at Spaulding Rd and CTH A were significantly different overall ($p < 0.001$) and in each season ($p < 0.001$ and $p = 0.046$).

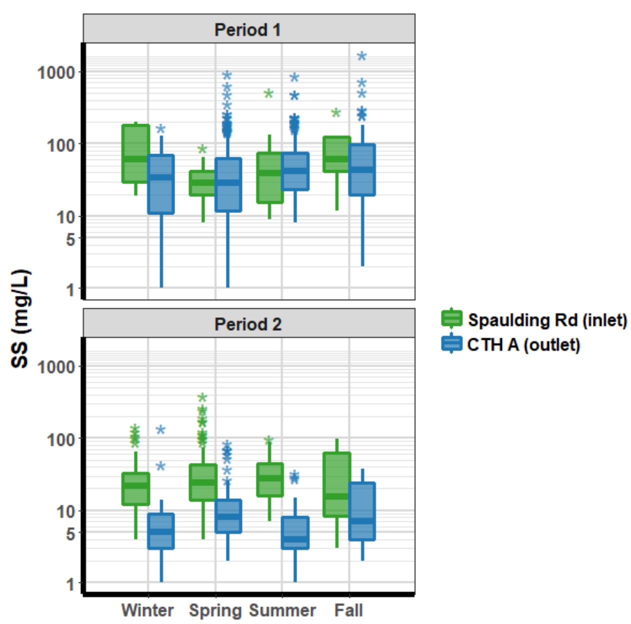

Figure 2-41: SS concentrations in grab samples collected at the inlet and outlet of SCE. Period 1 sampling occurred from December 2005 – September 2011. Period 2 sampling occurred from October 2011 – October 2017. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individual.

Table 2-16: Median SS concentrations and number of samples at the inlet and outlet of SCE. Period 1 was December 2005 – September 2011, and Period 2 was November 2011 – October 2017.

		Site	Winter	Spring	Summer	Fall	All Seasons
Period 1	**Median SS (mg/L)**	**Spaulding Rd (inlet)**	61	29	39	61	37
		CTH A (outlet)	34	29	41	44	34
		Difference (% Reduction)	27 (44%)	0 (0%)	-2 (5% increase)	17 (28%)	3 (8%)
		p value	0.076	0.977	0.282	0.784	0.293
	n	**Spaulding Rd (inlet)**	8	11	8	4	31
		CTH A (outlet)	35	304	143	72	554
Period 2	**Median SS (mg/L)**	**Spaulding Rd (inlet)**	22	24	28	16	24
		CTH A (outlet)	5	8	4	7	6
		Difference (% Reduction)	17 (77%)	16 (67%)	24 (86%)	9 (55%)	18 (75%)
		p value	**<0.001**	**<0.001**	**<0.001**	**0.046**	**<0.001**
	n	**Spaulding Rd (inlet)**	87	156	58	12	313
		CTH A (outlet)	13	38	29	15	95

SS concentrations were predicted by categorical amount of precipitation in the previous seven days ($p = 0.009$), and the interactions between flow at Spaulding Rd and year ($p < 0.001$), flow at Spaulding Rd and season ($p = 0.014$), the sampling period and season ($p = 0.024$), sampling period and site (p < 0.001), and season and site (p = 0.011; Appendix B Table B-10). Increasing precipitation was associated with increasing SS concentrations. During Period 1, increasing flow corresponded to decreasing spring SS concentrations (Figure 2-42). This pattern was particularly evident in 2007, 2009 and 2010. However, during Period 2 when Spaulding Rd was more frequently sampled, increasing flow corresponded to increasing spring SS concentrations at Spaulding Rd. In 2012, this pattern was observed at the both Spaulding Rd and CTH A. Increasing SS concentrations were a predictor of increasing TP concentrations; however, the relationship was weakest in winter compared to the other seasons (Figure 2-43).

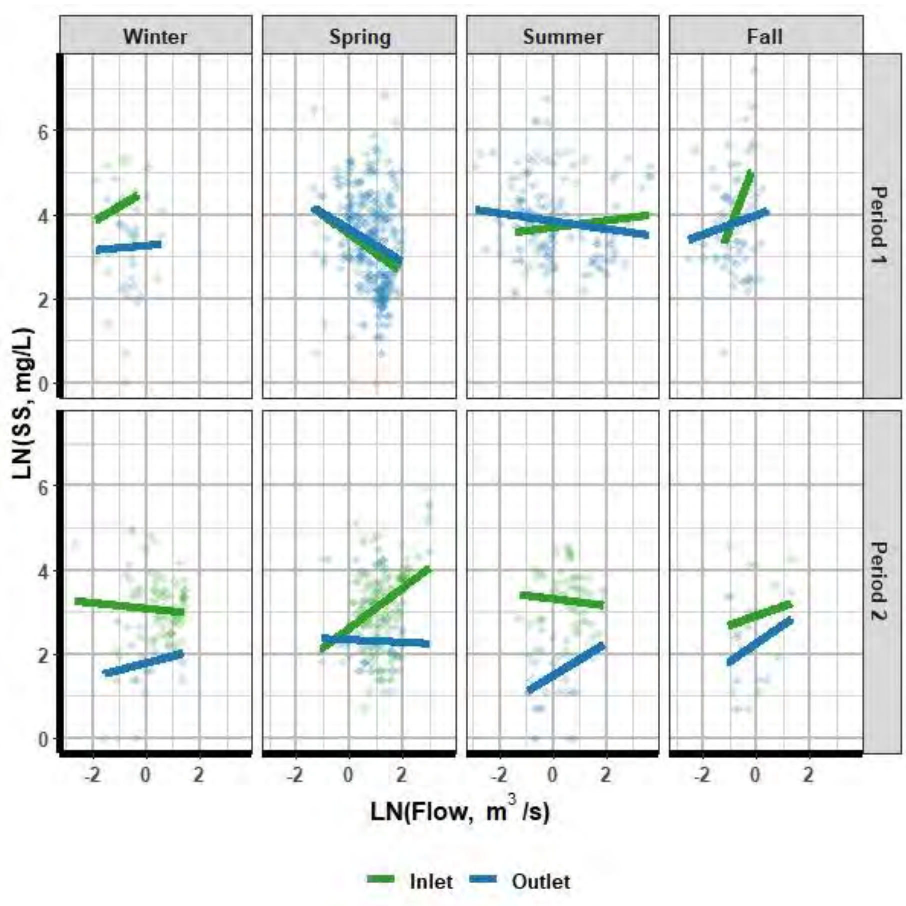

Figure 2-42: Seasonal inlet flow (Spaulding Rd) versus measured and estimated sample SS concentrations at the SCE inlet and outlet in sampling Period 1 (December 2005 – September 2011) and Period 2 (October 2011 – October 2017). All variables are natural log transformed. Linear models extend the range of sub-dataset in the x direction.

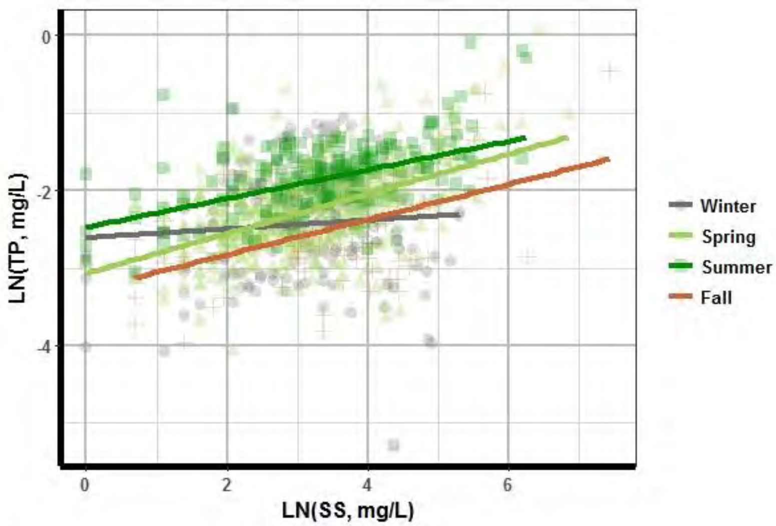

Figure 2-43: Seasonal trends in measured and estimated SS concentrations versus TP concentrations for samples at Spaulding Rd and CTH A. All variables are natural log transformed. Linear models extend the range of sub-dataset in the horizontal direction.

The overall median SS removal efficiency for the 89 days when concentrations were sampled at both Spaulding Rd and CTH A was 65%. Seasonal median SS removal efficiencies ranged from 53% in spring (n = 39) to 77% in fall (n = 11; Figure 2-44 A). An outlier sample collected in October 2016 had a net loss of SS (a negative SS removal efficiency of -443%; not shown in Figure 2-44). The sample followed a large amount of rain (56.9 mm over the previous three days) and the Spaulding Rd and CTH A SS concentrations were 7 mg/L and 38 mg/L, respectively. SS removal efficiency was significantly related to sampling period ($p < 0.001$), SS concentration at Spaulding Rd ($p < 0.001$), and the interaction between flow at Spaulding Rd and season ($p = 0.003$; Figure 2-44 B; Appendix B Table B-10). Categorical amount of rain over the previous seven days was not a significant predictor of SS removal efficiency ($p = 0.064$).

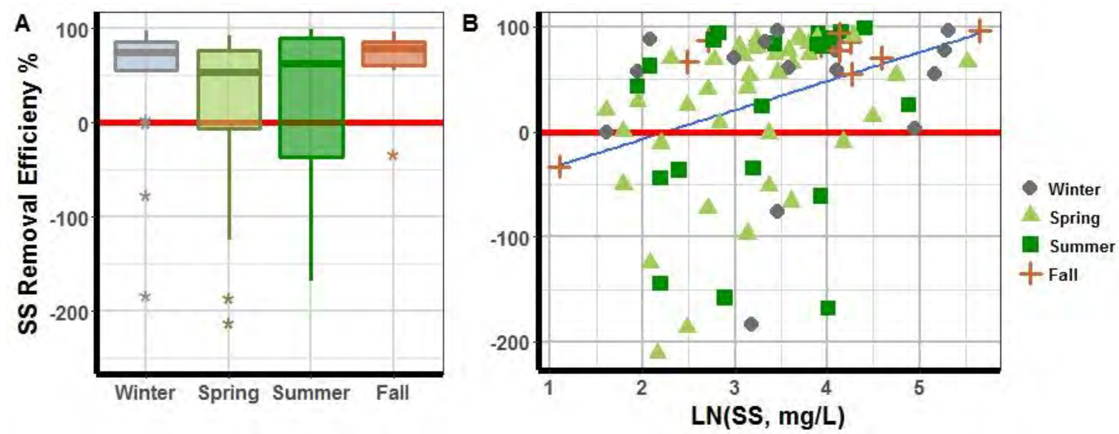

Figure 2-44: Removal efficiency of SS from Spaulding Rd to CTH A using results from grab samples collected on the same day. Data were collected by USGS from February 2006 – May 2017. Neither graph shows outlier of -443% from October 27, 2016. (A) SS removal efficiency grouped by season (winter n = 18, spring n = 39, summer n = 21, fall n = 11). The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individual. (B) SS removal efficiency versus natural log SS concentration at Spaulding Rd.

Total Nitrogen

The overall median TN concentration decreased from 3.1 mg/L at Spaulding Rd to 2.1 mg/L at CTH A (Table 2-17); however, the difference was not significant ($p = 0.06$). Seasonal median TN concentrations at Spaulding Rd ranged from 2.2 – 4.2 mg/L and followed the order spring < summer < fall < winter (Figure 2-45). At CTH A, seasonal median TN concentrations ranged from 1.6 – 4 mg/L and followed the order fall < summer < spring < winter. Decreases in TN concentrations from Spaulding Rd to CTH A were significant in summer ($p < 0.001$) and spring ($p = 0.034$; Figure 2-45; Table 2-17). Median TN concentration also decreased in winter and fall; however, these seasons had smaller sample sizes and the differences were not significant ($p = 0.092$ and $p = 0.154$ for fall and winter, respectively).

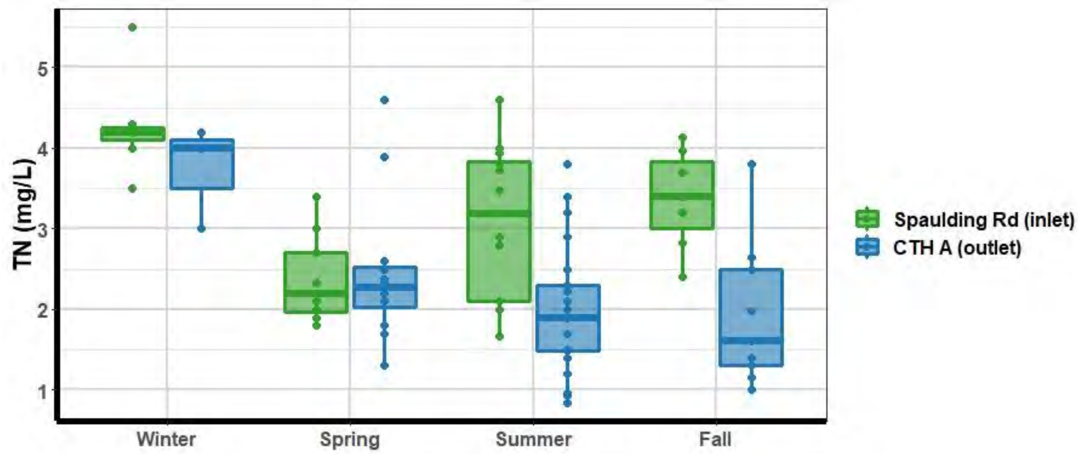

Figure 2-45: TN concentrations measured at the inlet and outlet of SCE. Data collected by USGS and BSE from February 2014 – October 2017. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR. All points are plotted individually.

Table 2-17: Median TN concentrations and number of observations for data collected at the inlet and outlet of SCE from February 2014 – October 2017. Significant p values (<0.05) are bolded. ^ p value calculated without using significant interaction.

	Site	Winter	Spring	Summer	Fall	All Seasons
Median TN (mg/L)	**Spaulding Rd (inlet)**	4.2	2.2	3.2	3.4	3.1
	CTH A (outlet)	4.0	2.3	1.9	1.6	2.1
	Difference (% Reduction)	0.2 (5%)	-0.1 (5% increase)	1.3 (41%)	1.8 (53%)	1 (32%)
	p value	0.154	**0.034**	**<0.001**	0.092	0.060
n	**Spaulding Rd (inlet)**	7	12	12	7	38
	CTH A (outlet)	3	12	21	9	45

TN concentrations were significantly related to TP concentrations ($p < 0.001$) and the interaction between season and site ($p < 0.001$), and site and flow at Spaulding Rd ($p < 0.001$;

Appendix B Table B-11). TN concentrations were not significantly related to the categorical amount of precipitation over the preceding three days (p = 0.22). TN concentrations at Spaulding Rd decreased as flow increased; however, at CTH A, TN concentrations increased as flow increased. When compared to SCE retention time, Spaulding Rd TN concentrations increased with longer retention times (decreasing flow) but CTH A TN concentration decreased (Figure 2-46).

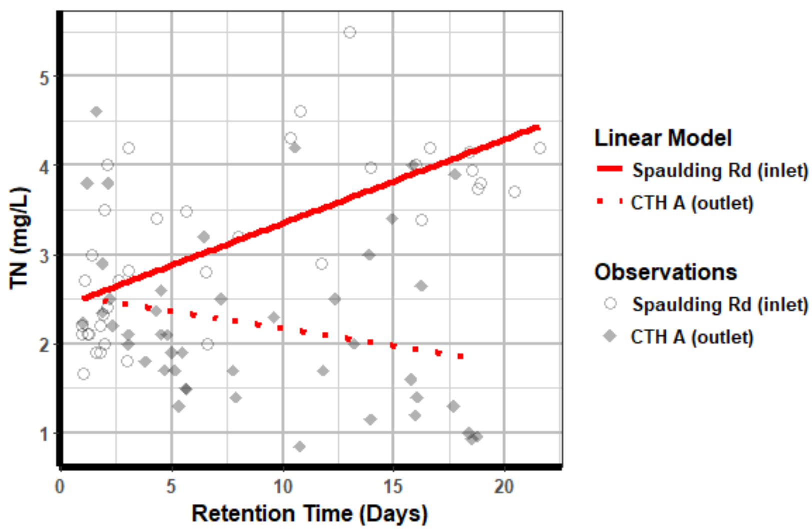

Figure 2-46: Measured and estimated TN concentrations versus SCE retention time. Linear models based on samples collected from February 2014 – October 2017 by the USGS and BSE at Spaulding Rd and CTH A. Linear models extend to range of retention time in sub-dataset.

The median TN removal efficiency for 21 coinciding samples was 29%. The seasonal median TN removal efficiencies ranged from -9% to 70% and followed the order spring (n = 4) < winter (24%; n = 3) < fall (39%; n = 7) < summer (n = 7). The removal efficiency was significantly related to TN concentrations at the Spaulding Rd (p = 0.009; Figure 2-47 A), amount of precipitation on the sampling day (p = 0.002; Figure 2-47 B), season (p = 0.02), and categorical amount of rain in the preceding three days (p = 0.027; Appendix B Table B-11).

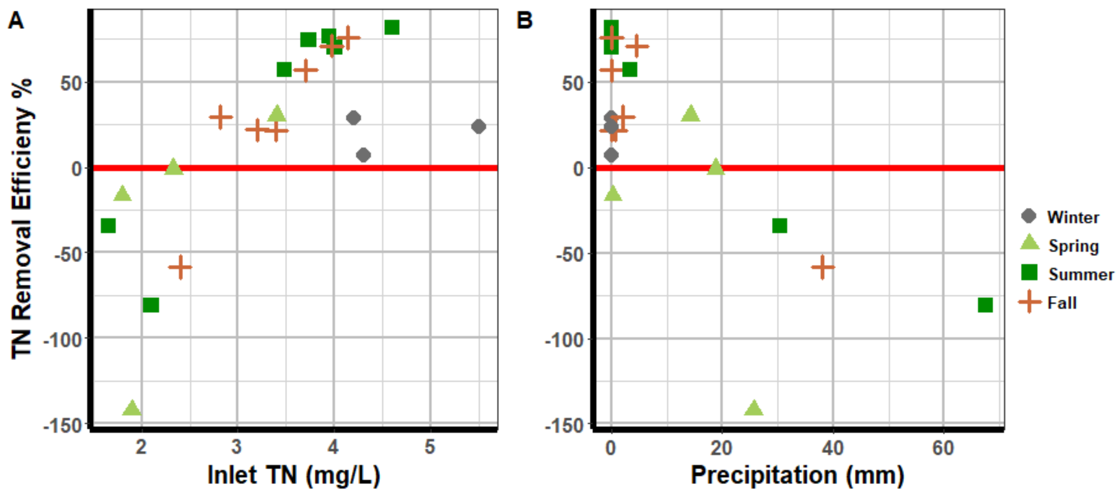

Figure 2-47: Removal efficiency of TN from inlet to outlet of SCE using data collected from May 2014 – October 2017 by USGS and BSE. (A) TN removal efficiency versus TN concentrations at SCE inlet. (B) TN removal efficiency versus precipitation on day of sampling.

The overall median TN:TP molar ratio for 105 samples was 56. The seasonal median TN:TP ratios ranged from 45.2 – 128 and followed the order summer < fall < spring < winter (Table 2-18). Once in spring and on nine occasions in summer, the TN:TP ratio was lower than 22.6 (Figure 2-48 A). Below a ratio of 22.6, TP no longer typically limits biological growth (Guildford and Hecky, 2000). Below a ratio of 7, TN typically limits biological activity (Abell *et al.*, 2010). The minimum TN:TP ratio observed was 15.7 on July 26, 2016. TN:TP ratio was significantly related to cumulative precipitation in the preceding three days ($p = 0.006$; Figure 2-48 A), and the interactions between site and flow at Spaulding Rd ($p < 0.001$) and season and TP concentrations ($p = 0.023$; Appendix B Table B-11). TN:TP increased with decreasing TP concentrations. TN:TP ratios at Spaulding Rd and Site 1 decreased with increasing flow, but this was not observed at Site 2 and CTH A (Figure 2-48 B).

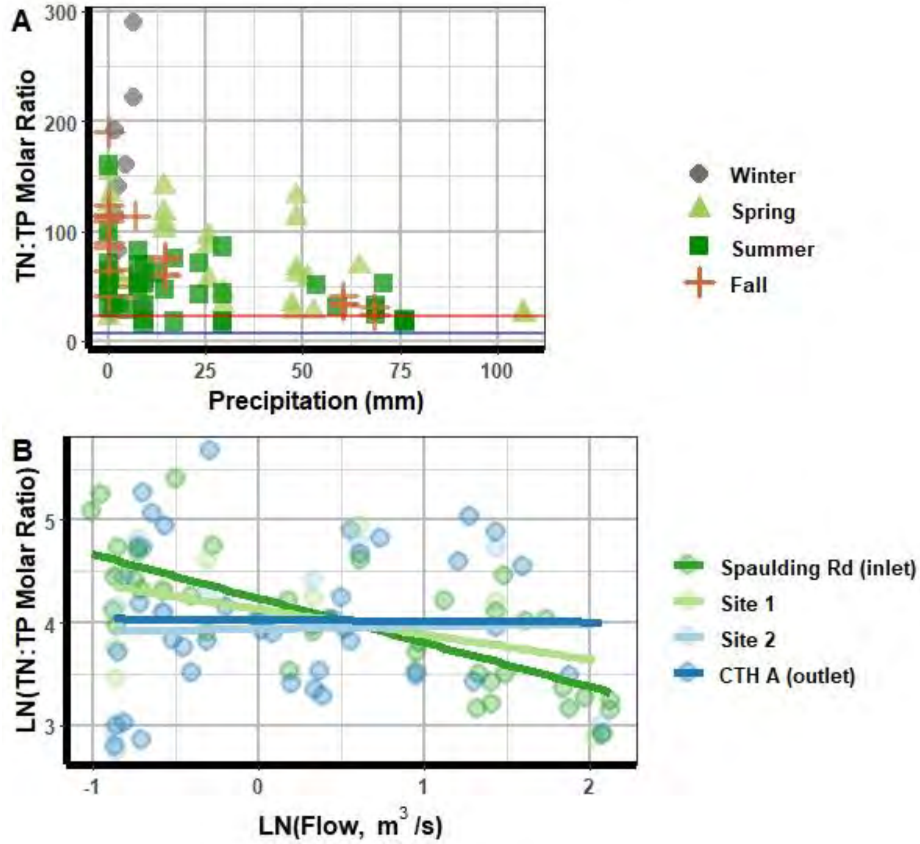

Figure 2-48: TN:TP molar ratios for grab samples collected in SCE. Samples were collected at the inlet (n=38), Site 1 (n = 11), Site 2 (n=11), and the outlet (n = 46) of SCE by the USGS and BSE from February 2014 – October 2017. (A) Seasonal TN:TP molar ratio versus cumulative three-day precipitation preceding sampling. Above the red line N:P = 22.6, P is limiting (Guildford and Hecky, 2000) and below the blue line TN:TP = 7, N is limiting (Abell et al., 2010). (B) Measured and estimated TN:TP molar ratio versus flow at Spaulding Rd. Both variables are natural log transformed. Differing slopes show the significant interaction between site and flow ($p < 0.001$)

Table 2-18: Median TP concentrations, TN concentrations, TN:TP molar ratios, and number of observations for samples collected in SCE. Observations are grouped by season. Samples were collected at the inlet (n=38), Site 1 (n = 11), Site 2 (n=11), and the outlet (n = 46) of SCE by the USGS and BSE from February 2014 – October 2017.

	Winter	Spring	Summer	Fall	All Seasons
Median TN (mg/L)	4.20	2.20	2.10	2.57	2.40
Median TP (mg/L)	0.070	0.075	0.118	0.076	0.096
Median TN:TP Molar Ratio	128.0	67.0	45.2	64.9	56.0
n	10	28	43	24	105

Discussion

Objective 1: Quantifying Changes in SS and TP Before and After Equilibrium Shift

Thirty years of monitoring at the SCE outlet show that water quality has improved over the study period. The TP and SS loads vary on monthly and seasonal scales; however, both TP and SS loads decreased in the summer and spring, and TP loads also decreased in fall and winter. The combined impacts of the RWTP upgrade and equilibrium shift (from phytoplankton to macrophyte-dominance) in the marsh are likely responsible for the decreases in TP and SS in the spring and summer, whereas the RWTP upgrade alone likely had more of an influence on TP decreases in the winter and fall.

Influence of RWTP Upgrade on Total Phosphorus loads

For all seasons, RWTP TP load decreased, and the plant's contribution became a smaller proportion of the TP load at CTH A following macrophyte restoration compared to before. Before the upgrade, RWTP TP load was equivalent to a majority of the TP load at CTH A in the winter and fall, but this was substantially reduced after the upgrade. In spring and summer, RWTP loads were proportionally a less dominant component of the TP loads at CTH A. The RWTP upgrade resulted in a 74% reduction in point source TP load to Silver Creek. The RWTP effluent flows were similar before and after the upgrade; however, TP concentrations decreased by 76%, resulting in lower TP loads to Silver Creek. In summer, the median TP load after the upgrade was approximately 50% of the median before the upgrade, whereas in winter, spring, and fall, the median TP loads after the upgrade were ~20 to 25% of what they were prior to the upgrade. Before the SCE restoration, RWTP median TP loads in winter and fall were 86% and 73%, respectively, of the seasonal median values downstream at CTH A. Following the restoration, these percentages decreased to 18% in winter and 21% in fall. In spring and summer, the seasonal median TP loads at the RWTP were 21% and 35%, respectively, of those at the outlet before the restoration, and 5% and 20%, respectively, after restoration. Even in summer, when seasonal median TP load from RWTP was largest and the seasonal reduction from before to after restoration was the smallest, the proportion of RWTP TP loads to CTH A TP loads decreased 15% following the two management actions.

Before the SCE restoration, when RWTP TP loads were high, loads at CTH A were occasionally less than the plant TP load, most commonly in the winter and fall. Before SCE restoration, the monthly average RWTP TP load exceeded that at CTH A in 32 of 177 months (18%), indicating that there were times when Silver Creek and the marsh retained TP (i.e. acted as a sink). This occurred in 16 of the 43 winter months (37%) and 14 of the 45 fall months (31%).

At times, TP reaching SCE exceeded TP released at the upstream RWTP based on data from a historic gage on Silver Creek (Koro Rd; Figure 2-1). From February 1987 to August 1996, the Silver Creek TP load measured 1.8 km downstream of the RWTP (Koro Rd site) was similar to or greater than TP load from the plant (data not shown). Monthly TP loads at Koro Rd were occasionally smaller than those from the plant (the largest decrease in monthly average TP between the two locations was 0.53 kg/day); however, the median monthly average TP increased 5.07 kg/day between RWTP and Koro Rd. This indicates that TP load from the RWTP was a low estimate for Silver Creek TP further downstream. Over the monitoring period at Koro Rd, there was net retention of TP between that gage and the CTH A outlet only in the fall (average retention of 1.5 kg/day). In winter, there was almost no difference between TP load at the two sites (average increase of 0.002 kg/day from Koro Rd to CTH A). In spring and summer, however, TP increased between Koro Rd and CTH A (spring average increase 7.8 kg/day; summer average increase 0.9 kg/day). Overall, for the 10-year period, there was no net retention but rather an increase in TP between the Koro Rd and CTH A (overall average increase of 1.95 kg/day). Similarly, there was no net retention of SS between the two sites (overall average increase of 0.99 metric tonnes/day), and seasonally, a net SS retention between the two sites only occurred in winter (average of 0.15 metric tonnes/day). This additional TP and SS either entered Silver Creek downstream of Koro Rd, entered SCE through the other tributary (Dakin Creek) or came from an internal source within SCE.

Without monitoring at the primary SCE inlet (Spaulding Rd) before restoration, the loads at the Koro Rd site and RWTP provide the only data for examining whether Silver Creek and the marsh retained TP and SS upstream of Green Lake. In winter and fall, the section of Silver Creek between the RWTP and CTH A more often retained TP and SS. In spring and summer, that section of creek and SCE exported TP and SS. Following the restoration from 2006 – 2011, there was no upstream monitoring of TP loads other than inputs from the RWTP. During this period, the RWTP TP load exceeded the CTH A load in only 2 of 71 months (3%). This suggests fewer instances of TP retention downstream of the RWTP compared to before the restoration (32 of 177; 18%). Once monitoring began at the SCE inlet at Spaulding Rd, however, further evidence was available to assess retention upstream of CTH A.

Since installation of the Spaulding Rd gage, TP load from RWTP to Spaulding Rd increased in all 70 months; there was no evidence of net TP retention between RWTP and the SCE inlet at Spaulding Rd. The TP load at the RWTP exceeded that at CTH A in only three of 70 months. Although on those three occasions TP load increased from RWTP to Spaulding Rd, the decrease from Spaulding Rd to CTH A indicated retention within SCE (i.e. marsh was a TP sink). Spaulding Rd monitoring thus provides evidence that when the RWTP TP load exceeded that at CTH A, the marsh itself, and not the upstream creek, retained TP.

The RWTP is a point source of TP input to Silver Creek, SCE, and Green Lake, but it is no longer a dominant contributor in the watershed. Since 2012, RWTP TP load represented about 22% (median) of the Spaulding Rd TP load in the fall to 5% (median) in the spring. The monthly average TP load at RWTP was a significant predictor of TP load at Spaulding Rd ($p = 0.04$), but season and flow were more significant ($p = 0.03$). Increases in winter flow led to noticeable increases in TP load at Spaulding Rd. While RWTP continues to be a source of TP to Silver Creek, most of the TP reaching the SCE inlet at Spaulding Rd appears to be from other nonpoint sources, such as agricultural and urban runoff.

Median TP load reductions at RWTP and CTH A following restoration were similar but showed different seasonal variation. The overall difference in median TP load from RWTP before and after SCE restoration was 2.79 kg/day. This difference was slightly higher than the overall decrease in the median TP load at CTH A over the same period (2.66 kg/day). In all seasons except summer, the reduction in RWTP median TP load before and after SCE restoration was greater than the reduction at CTH A. If RWTP TP load changes alone were responsible for reductions at CTH A, then we might expect that TP load at CTH A would decrease the least in summer (similar to the 2.25 kg/day decrease at RWTP) and more in the other seasons (average decrease in seasonal medians was 2.78 kg/day at RWTP). However, this was not the case. The TP load reductions downstream at CTH A following restoration were greatest in the summer (2.99 kg/day) and lower in the other seasons (average decrease in seasonal medians was 1.68 kg/day). While the RWTP upgrade reduced TP loading to Silver Creek, seasonal patterns of change were different at the RWTP compared to CTH A. This suggests that changes in TP loading from SCE were not solely explained by the RWTP upgrade and that the phytoplankton to macrophyte equilibrium shift in the marsh also had an impact.

Changes in Loading at CTH A

At CTH A, differences between TP load before and after restoration were greater than for SS. Each season's median TP load decreased from the period before restoration to after (decrease ranged from 0.77 – 2.99 kg/day), as did each season's median flow-weighted mean TP concentration (0.033 – 0.092 mg/L). For SS load, significant seasonal reductions from before to after restoration occurred only in spring and summer. There were no significant changes in SS load between the two time periods during winter or fall. The largest and most significant reduction (40%) in SS load occurred in the summer. Flow-weighted mean SS concentration also significantly decreased in the spring (55%) and summer (50%) following restoration but not in winter (12% increase) and fall (8%; Figure 2-25).

Significant reductions in both TP and SS concentrations and loads during the spring and summer are expected because carp reductions and macrophyte establishment improve both water quality measures. Seventy-five percent of studies evaluating carp ecosystems found that carp introduction increased water column P and N and 91% of studies found that turbidity increased (Weber and Brown, 2009). These increases were associated with carp benthic foraging and excretion and with decomposition of destroyed macrophytes. Carp removal has been linked to decreasing total suspended solids concentration, increasing springtime water clarity and decreasing P concentrations, likely as a result of reduced sediment resuspension and decreased excretion (Schrage and Downing, 2004; Bajer and Sorensen, 2015b). Reduced sediment disturbance and increased zooplankton grazing of phytoplankton following carp removal promotes macrophyte establishment via reduced physical disturbance of roots and increased light penetration through the water column (Schrage and Downing, 2004). Macrophytes decrease TP and SS by consuming nutrients and stabilizing sediment. Macrophytes that established following SCE restoration, such as *C. demersum* and *M. spicatum*, effectively remove P from the water column (Gao *et al.*, 2009). The physical presence of aquatic macrophytes also helps to reduce sediment resuspension by reducing turbulence, trapping particulate matter, stabilizing sediment, and increasing sedimentation rates (Barko *et al.*, 1991). Additionally, summer SS flow-weighted mean concentrations have continued to decrease since restoration, suggesting that impacts of macrophytes on SS are cumulative over time (Figure 2-25).

Since SS load and flow-weighted mean concentrations did not change significantly in winter and fall, processes other than carp removal and macrophyte establishment influenced the significant TP reductions observed during those seasons. Carp seek more hospitable conditions in deeper water during the cold winter, unlikely to overwinter in the shallow marsh (Penne and Pierce, 2008). The carp barrier was also only deployed during carp spawning (May 1st - July 4th) and did not target reducing carp densities in other seasons. Therefore, carp utilization of the shallow marsh in fall and winter was not likely changed between the before and after restoration periods and would not contribute to decreasing TP loading over those seasons. Likewise, macrophyte growth would not be a factor in changing TP concentrations during the winter. Cold temperatures during winter reduce biological activity, such as decomposition and growth of macrophytes. When temperatures are cold, senescing macrophytes accumulate as sedimentary organic matter rather than decompose which would increase water column TP concentrations (Carpenter, 1980). Furthermore, equilibrium shifts and phytoplankton reductions have been associated with reductions in summer TP concentrations and not with changes in winter concentrations (Søndergaard *et al.*, 2002).

Reductions in TP at CTH A could be linked to changes in precipitation, changes in watershed land use or management practice implementation, reductions at the RWTP, or the equilibrium shift within the marsh. Precipitation increased from before to after restoration (significantly in winter and spring) which would not favor TP reductions. Flow also increased in all seasons but not significantly. Since both TP and SS loads were positively associated with increasing precipitation, no significant change in winter SS load, a significant decrease in spring SS load, and significant decrease in TP loads in both seasons (winter and spring) were unexpected. Precipitation changes from before to after restoration, therefore, are unlikely to explain TP decreases at CTH A.

Better land management practices upstream are also not likely a factor in TP reductions because according to the Green Lake County Land Conservation Department (LCD), only two best management practice (BMP) cost sharing projects were implemented in SCE watershed following SCE restoration (personal communication Stephanie Prellwitz, Executive Director Green Lake Association). These land stabilization projects treated an area of 0.6 km^2. Funding for BMPs has been prioritized elsewhere in the Green Lake watershed since 2006. From 1987-1996, the Green Lake County LCD worked with 18 different farmers to implement a total of 33 BMPs in the SCE watershed. The most frequently implemented projects were grassed waterways (7), critical area stabilizations (5), reducing tillage (4), and controlling barnyard runoff (4). Since these BMPs covered only a small portion of the SCE watershed, their impact on TP reductions at SCE was likely to be small.

Land use changes within the SCE watershed could also be linked to TP reductions at CTH A. Between the 1990’s and 2016, agricultural land decreased from 79% to 67% of the watershed area, and developed areas increased from 6% to 10%. While developed land might also be associated with high TP loading, it generally contributes less per unit area than agricultural areas (Liu *et al.*, 2009). The decrease in agriculture, therefore, could contribute to the TP reductions observed following SCE restoration. Over this time period, corn, an intensively fertilized crop, increased from 25% land cover to 33% in the SCE watershed. Corn is often rotated with soybeans, but as corn profitability increases, rotating becomes less frequent. In 2016, there was roughly twice as much corn as soybeans (15% land cover) in the SCE watershed, suggesting farmers were implementing a corn-corn-soybean rotation. The increase in corn from the 1990's to 2016 could be associated with farmers changing from a corn-soybean to a corn-corn-soybean rotation which has been shown to increase TP loss (Mbonimpa *et al.*, 2012). Intensifying corn production could therefore mitigate some of the TP reductions associated with decreased agricultural areas. Taken together, precipitation increases, few documented BMPs, and increased corn production do not provide evidence to support the TP reductions observed at CTH A following SCE restoration. The influence of the RWTP upgrade and the equilibrium shift of the marsh are more likely responsible for the decreased TP loading to Green Lake via SCE.

Before the restoration, winter and fall median RWTP TP load made up a large proportion of at the TP load at CTH A. This proportion decreased following the restoration. Therefore, reductions in winter and fall TP load at CTH A following restoration may be more closely linked to the point source reduction upstream at the RWTP. The TP reductions at RWTP were smallest in summer, but at CTH A reductions were high in summer. This points to both the equilibrium shift and RWTP reduction decreasing TP load at CTH A during the growing season, but the RWTP having a predominant influence on TP reduction during the non-growing season.

Objective 2: SCE as a Sink or Source for TP and SS

Overall, SCE acted as a sink for both TP and SS from 2012 – 2017. In total, the marsh retained 3,393 kg of TP and 1,393 metric tonnes of sediment (Table 2-19). On average, the marsh retained 1.59 kg/day of TP and 1.02 metric tonnes/day of SS. The marsh was a TP sink more often in winter (94%) and fall (87%) months compared to spring and summer months (both 61%). For SS, the marsh was a sink most often in spring (83%) and winter (75%), and less often in summer (67%) and fall (56%). While there were occasional months when SCE did not retain either SS or TP, there was only one month (in summer following heavy rain) when neither SS nor TP were retained. The TP retention was significantly explained by precipitation and the amount of TP load at Spaulding Rd. The TP retention decreased with increasing precipitation and increased with increasing TP load at Spaulding Rd. Retention of SS was significantly explained by SS load at Spaulding Rd (with a positive relationship) but was not significantly impacted by precipitation. Seasons and flow were not significant explanatory variables for either TP or SS retention in SCE. Models for predicting both TP and SS retention explained little of the variation in retention (adjusted $r^2 < 0.3$) demonstrating that additional information is needed to explain what influenced retention. Even though retention amounts were difficult to predict, there was predominately positive retention of both TP and SS within the marsh, supporting the improved water quality downstream entering Green Lake.

Table 2-19: SCE total SS and TP retention for monitoring period at Spaulding Rd and CTH A (TP: December 2011 – September 2017. SS: December 2011 – August 2012 and October 2014 – September 2017).

Season	SS Retention (metric tonnes)	TP Retention (kg)
Winter	454	1315
Spring	723	972
Summer	246	676
Fall	-30	430
Total	**1393**	**3393**

While there was net retention of TP in all seasons, there was net export of SS in fall and net retention of SS in the remaining seasons. Fall had a smaller sample size (n = 9) compared to the other seasons (n = 12), however, and one month (November 2016) had large SS export (1.7 metric tonnes/day) compared to the other fall months (average SS retention of 0.09 metric tonnes/day; Figure 2-29). The fall month with the second highest SS export (November 2014) only released 23% of the SS that had been exported in November 2016. The November 2016 export therefore constituted a large portion of the fall net SS export over the monitoring period. Despite this, SS retention in winter, spring and summer exceeded SS export in fall. SS loads decreased significantly between Spaulding Rd and CTH A in winter, spring, and summer. Overall, the median SS removal efficiency for SCE was 41%. The marsh was an SS sink retaining an average of 1.02 metric tonnes/day over the study period.

The SS and TP load calculations did not include loading from Dakin Creek, the other inlet to the marsh, and therefore these retention estimates within SCE are likely conservative. For example, spring TP load was not significantly different between Spaulding Rd and CTH A. However, the flow-weighted mean TP concentration at CTH A, which accounted for additional streamflow added by Dakin Creek, was significantly lower compared to Spaulding Rd. This suggests that the TP load from Dakin Creek was also retained in the marsh. The 3,393 kg of TP retained was 15% of the TP load entering SCE at Spaulding Rd during the monitoring period. Thus, a conservative estimate for this period was that SCE retained the TP load from Dakin Creek in addition to the 15% from Spaulding Rd. Overall, the median TP removal efficiency from Spaulding Rd to CTH A was 32%. Since this accounts for additional flow from Dakin Creek and exceeds the estimate only based on TP load from Spaulding Rd, it provides further evidence that more than 15% of Spaulding Rd TP load is retained in SCE. The 32% removal efficiency is also an underestimate because it does not include TP inputs from Dakin Creek, only additional flow. The overall SS retention of 1393 metric tonnes in SCE is likely also conservative because no SS additions were assumed for Dakin Creek. Therefore, 37% of the SS that entered through Spaulding Rd was retained in SCE over this period, in addition to the SS contribution from Dakin Creek. Quantification of nutrient inputs from Dakin Creek would be valuable to the monitoring program to more accurately evaluate nutrient retention within the marsh.

Assessed at a monthly scale over a five-year period (2012 – 2017), SCE retained both SS and TP in 53% of months and exported both SS and TP during only one month (2%). The only net export was SS in fall, but that amount was just 2% of the total SS amount retained (Table 2-19). In summary, SCE acted as a sink for both TP and SS, effectively reducing watershed TP and SS loads reaching Green Lake.

Objective 3: Changes in Water Quality from Spaulding Rd to CTH A

Total Phosphorus and Total Dissolved Phosphorus

The TP decreased as water flowed downstream through SCE in all seasons. During the growing season, TP concentrations at CTH A were higher than in the non-growing season. The TP concentrations were significantly lower in all seasons at CTH A than at Spaulding Rd. Between the two sites, the differences in median TP concentrations were largest in winter (0.067 mg/L) and fall (0.051 mg/L), and the winter and fall seasonal medians at CTH A were both below the stream TP impairment level of 0.075 mg/L used by WDNR. In the spring, the median TP concentrations were slightly greater at CTH A compared to Spaulding Rd (a difference of 0.001 mg/L), but statistical analysis of the MLR and modeled predicted means for TP concentrations identified a significant decrease from Spaulding Rd to CTH A. In summer, the median TP concentration was 0.021 mg/L lower at CTH A than at Spaulding Rd with model predictions also significantly lower at CTH A. Median TP concentrations were above the TP impairment level at both sites in spring and summer. When sampling occurred at four locations throughout the marsh on the same day, TP concentrations decreased significantly from Spaulding Rd to CTH A in summer and fall, and seasonal median TP concentrations decreased moving from upstream to downstream at the four sites.

Increasing TP concentrations were associated with increasing flow at both Spaulding Rd and CTH A, especially in the winter and most obviously during winter 2016-2017 at Spaulding Rd. During that winter, samples were collected when flows were an average of four times greater than in other winters. The average TP concentration for the 2016-2017 winter was also twice the average observed during the other winters. While 30 years of monitoring at CTH A show that winter is not often a time of high TP loading, the winter of 2016-2017 was an exception. Since TP loads at Spaulding Rd and CTH A were positively related to flow, upstream watershed efforts to reduce runoff and stormflows would help decrease TP loading to Green Lake. In-stream efforts for erosion control, such as bank stabilization, could also decrease the erosive effect of large stream flows and in turn could reduce TP loading from SCE.

The TP removal efficiency within the marsh decreased as precipitation increased. TP removal efficiencies were not significantly impacted by season, and median removal efficiency (36%) was similar to the median removal efficiency based on monthly average flow-weighted concentrations (32%). Excluding spring, TP removal efficiencies only decreased below zero when there had been precipitation in the previous three days. In spring, negative removal efficiencies only occurred when there had been precipitation over the previous seven days and/or the Spaulding Rd TP concentration was relatively low. As the amount of rain over the previous three days increased, there were more instances of negative removal efficiencies, especially in summer. This indicates that under dry weather conditions, major summer internal TP loading from sediments was not observed. In fall, the only instance of SCE acting as a source of TP was after 68 mm of precipitation occurred over the previous three days. Even though decomposing macrophytes release P into the water column (Carpenter and Adams, 1979; Landers, 1982), under low precipitation conditions, this did not lead to SCE being a fall source of TP to Green Lake.

Reductions in SCE summer TDP are likely attributed to macrophytes throughout the marsh. Only in summer, were TDP concentrations significantly lower at CTH A than at Spaulding Rd. The summer median TDP concentration at Spaulding Rd (0.114 mg/L) was higher than the impairment level for TP (0.075 mg/L). At CTH A, the summer TP median was still above that impairment level, but the dissolved component decreased to 0.042 mg/L. The portion of TDP:TP significantly decreased from Spaulding Rd to CTH A in spring and summer, indicating that dissolved P entering the marsh was either taken up by macrophytes or otherwise transformed to particulate P. On the occasions when there were high (> 0.15 mg/L) summer TDP concentrations at Spaulding Rd, TDP removal efficiencies exceeded 50% in four of the five occasions. This consumption of TDP is likely associated with macrophytes. Water column TP and TDP concentrations both can decrease when *C. demersum* is present (Dai, Wu, *et al.*, 2012). Since TDP concentrations were only significantly reduced by SCE in summer, dominant biological processes could be the cause. Some of the macrophytes overwinter, photosynthesizing at a reduced rate (Borman *et al.*, 2013), which consumes TDP more slowly. The significant decrease in summer TP, TDP, and TDP:TP ratio demonstrates that in addition to retaining summer P in the system, macrophytes also uptake TDP.

Suspended Sediment

The SS concentrations during the first period of SS sampling (December 2005 – September 2011) were not significantly different between Spaulding Rd and CTH A. However, during the second period (November 2011 – October 2017), CTH A concentrations were lower than at Spaulding Rd. This may be a result of the sampling scheme (high flows were sampled at Spaulding Rd more frequently than at CTH A) or changing processes in the marsh. The relationship between SS concentration and flow in spring was different during the two sampling periods. During the first period, increasing spring flows were associated with decreasing SS concentrations, though this relationship was determined mostly by samples collected at CTH A (n = 304 and Spaulding Rd n = 11). In the second period, the reverse trend was observed, with increasing flow associated with increasing SS concentrations, though this relationship was more influenced by Spaulding Rd (n = 156 and CTH A = 38). Combined over both sampling periods, spring SS concentrations increased with flow at Spaulding Rd but decreased with flow at CTH A, suggesting that with increasing spring flows, SCE serves to retain SS. This is supported by spring having the most retention of the four seasons over 2012 – 2017 (Table 2-19).

The median SS removal efficiency was 65% for all sampling events. The lowest seasonal median occurred in spring (53%), the season with the largest sample size. For the large SS concentrations at Spaulding Rd (> 100 mg/L), however, the removal efficiencies were all positive, again showing the contribution of SCE in reducing SS loads to Green Lake during times of high SS concentrations at Spaulding Rd.

Total Nitrogen

The TN concentrations overall were high but decreased from Spaulding Rd (median 3.1 mg/L) to CTH A (2.1 mg/L). When TN was sampled most frequently, during summer, it had the most significant decrease between the two locations compared to the other seasons. The summer median TN concentration (1.9 mg/L) at CTH A was higher than the median of other nonwadeable Wisconsin rivers (1.286 mg/L; Robertson *et al.*, 2008). Wisconsin does not have a nutrient criteria for nitrogen, but the 25th percentile in the Southeastern Wisconsin Till Plains for wadeable streams is 2.02 mg/L (Robertson *et al.*, 1998; Wang *et al.*, 2007). This concentration could be used to develop a criterion. The median TN concentration at Spaulding Rd exceeded 2.02 mg/L but was closest to reaching this benchmark in spring (median 2.2 mg/L). Median CTH A concentrations decreased below the reference value in summer and fall. At both sites, the highest median TN concentrations were observed in winter, though the sample size was small (n = 3).

The median TN removal efficiency was 29%. The largest decrease in TN concentration from Spaulding Rd to CTH A occurred when there had been little rain, flow was low, and retention times in the marsh were longer. The TN concentrations decreased at Spaulding Rd and increased at CTH A with increasing precipitation and flow and as retention time decreased. As a result, TN removal efficiency decreased as precipitation increased. Once SCE retention time decreased below three days, TN removal efficiency was negative (release of N).

During the summer, TN is consumed by macrophytes. The SCE has higher than 20% *C. demersum* cover, the amount shown to reduce TN and TP in the water column (Dai, Jia, *et al.*, 2012), likely leading to water quality improvements as a result of increased macrophyte density. While this study does not include TN concentrations before the SCE restoration, others have shown that restoration efforts including *C. demersum* have decreased TP and TN compared to conditions before plant installation (Wang *et al.*, 2018). In SCE, there was little evidence of macrophyte decomposition contributing to TN release. There was only one fall occasion when

TN removal efficiency was negative, and this was after a precipitation event (68 mm over the preceding 3 days).

SCE is enriched with both P and N, but the marsh is still likely P limited. The TN:TP molar ratio never dropped below 7, the upper bound for biological activity being limited by TN (Abell *et al.*, 2010). However, TN:TP ratios did decrease in the summer, which is when TN would be most likely to act as a limiting nutrient in a shallow lake (Søndergaard *et al.*, 2017).

Changes in water quality from Spaulding Rd to CTH A result from a number of physical and biological processes. Reductions in summer nutrient concentrations downstream through the marsh are likely due to the increased domination of macrophytes (Eugelink, 1998; Gao *et al.*, 2009; Dai, Wu, *et al.*, 2012; Tang *et al.*, 2017). Surveys in 2013 and 2014 showed extensive floating-leaved plant communities (44% of SCE surface cover) and submerged communities (at approximately 90% of surveyed points; Butterfield *et al.*, 2015). In the 2014 plant survey, submerged *Ceratophyllum demersum* was the most frequently encountered macrophyte in SCE. *C. demersum*, which lacks true roots, has a high surface area to volume ratio which makes it efficient in removing P from the water column (Gao *et al.*, 2009). Site specific sampling within the marsh shows that water column particulate P in areas with *C. demersum* decreases compared to areas without the macrophyte, suggesting the reduction in sediment resuspension (Dai, Wu, *et al.*, 2012). *Elodea Canadensis,* which was also prevalent in the 2014 SCE survey, is able to take up P from both its leaves and roots and translocate P from its shoots to roots (Eugelink, 1998).

Conclusion

Thirty years of monitoring show that SS and TP entering Green Lake from SCE has decreased as the result of a TP point source reduction upstream and an equilibrium shift within the marsh from phytoplankton to macrophyte dominance. Quantifying the impact of the upstream point reduction is difficult, however, because the inlet of the marsh was not monitored prior to the equilibrium shift. The importance of each management action likely differed seasonally. Since the upstream RWTP contributed a large proportion of TP loading relative to the marsh outlet in fall and winter, reductions observed at the marsh outlet in those seasons were likely a result of the RWTP upgrade. During the spring and summer, however, the combination of the RWTP point source reduction and the restoration success contributed to reductions in TP load. Reductions in SS were also observed in spring and summer suggesting that the carp removal and macrophytes establishment improved water quality reaching downstream Green Lake. Since no significant SS reductions were observed in fall and winter, the TP reductions during those seasons were unassociated with the equilibrium shift and therefore most likely linked to reduction in the RWTP load.

Additional monitoring following restoration indicated that the marsh acted as a sink for both TP and TSS, with concentrations decreasing through the marsh. From 2012 – 2017, flow-weighted mean TP and SS concentrations generally decreased from Spaulding Rd (the primary inlet) to CTH A (outlet). On occasion, however, the marsh did act as a source of TP and/or SS to Green Lake. Large TP retention mostly occurred when TP loads at Spaulding Rd were large and TP retention increased with less precipitation. For SS, SCE was an overall sink during the entire monitoring period, but seasonally, SS was exported from the marsh in fall. The SS retention was only predicted by SS load at Spaulding Rd, with larger loads leading to more retention within the marsh. Generally from 2012 –2017, the marsh retained either TP, SS or both and both were exported during only one month (June 2017).

SCE is now a macrophyte dominant marsh and removes P, SS and N before waters reach Green Lake. However, the system continues to be enriched with high levels of TN and TP, and there is still need for further nutrient reduction so that lake managers can meet their long-term LMP objective of returning Green Lake to an oligotrophic state. Continued monitoring at Spaulding Rd and CTH A will help assess progress towards the LMP goals. Further identification of opportunities for P reduction will likely be required.

Chapter 3 Comparison of Silver Creek Estuary to County K Marsh

Introduction

The majority of total phosphorus (TP) and suspended sediment (SS) loading to Green Lake enters through two marsh systems, Silver Creek Estuary (SCE) on the eastern end of the lake and County K Marsh (CKM) on the western end (Figure 3-1; Baumgart, 2015). The surface area of CKM (106 ha) is about one third larger than SCE (83 ha): the SCE watershed (123 km^2) is about 3.8 times larger than the area draining through CKM (38 km^2). The majority of land use within both watersheds is agricultural (68% in CKM and 66% in SCE) and corn is the predominant crop (Figure 3-2; USDA National Agricultural Statistics Service, 2016).

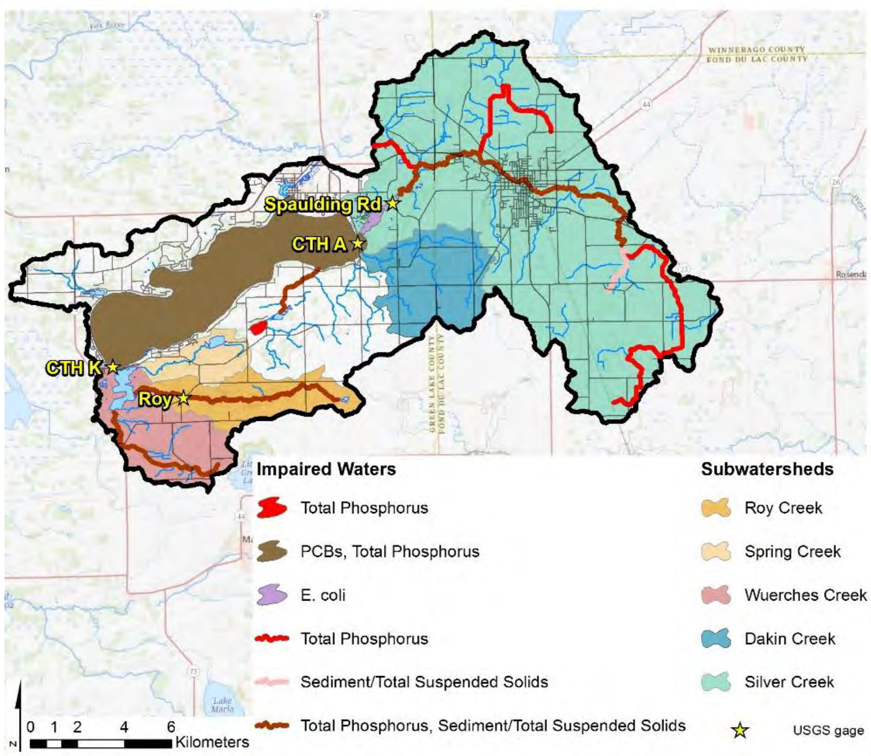

Figure 3-1: Subwatersheds draining through County K Marsh (Wuerches, Roy, and Spring Creeks) on the western edge of Green Lake and Silver Creek Estuary (Silver and Dakin Creeks) on the eastern edge. The 303(d) listed impaired waterbodies and USGS monitoring gages also shown.

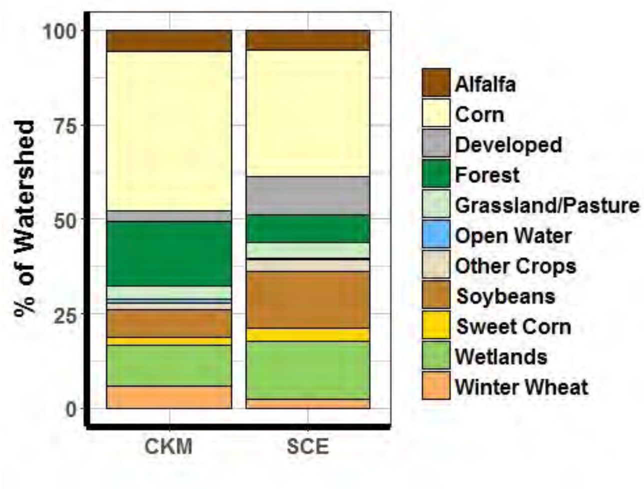

Figure 3-2: 2016 land use in watersheds of CKM and SCE (USDA National Agricultural Statistics Service, 2016).

Historically, both marsh systems have had high densities of carp and have been dominated by phytoplankton. Following carp exclusion around 2002 and a reduction in TP load from an upstream point source in 2003, SCE transitioned states into a macrophyte dominated system (Chapter 2; Figure 3-3). Efforts to shift the equilibrium of CKM from phytoplankton to macrophytes are underway following the installation of a carp barrier in 2015 (upgraded in 2016; Figure 3-4). Exclosures containing native macrophyte propagules were placed into the marsh to determine the effects of carp (Figure 3-5).

Figure 3-3: Lemna spp. (duckweed), free-floating macrophytes, cover the surface of SCE. Submergent macrophytes occupy the water column below. Photo taken June 15, 2017.

Figure 3-4: The barrier at the CTH K bridge at the outlet of CKM keep carp out of the shallow marsh during spawning season. Photo taken May 5, 2016 by Stephanie Prellwitz.

Figure 3-5: Exclosure in CKM containing Stuckenia petinata (sago pondweed). Photo taken September 15, 2017.

Both marshes have tributaries impaired by TP and suspended sediment (Figure 3-1). To compare marshes, the United States Geological Survey (USGS) monitored flow, suspended sediment (SS) and TP loads from 2012 – 2017 at the outlet of each marsh (County Trunk Highway K for CKM and County Trunk Highway A for SCE) and at one point upstream of each marsh (Figure 3-1). Loads at the Silver Creek inlet to SCE were monitored at Spaulding Rd, and loads in Roy Creek were monitored about 2.5 kilometers upstream of the CKM inlet. To build upon the long-term USGS monitoring, there was need for additional data at the other marsh inlet tributaries and at sites within both marshes to further compare water quality between the two systems.

Lakes and reservoirs across Wisconsin, especially in agricultural watersheds, play an important role in reducing export of total nitrogen (TN), TP and SS through river systems (Powers *et al.*, 2014). These lentic waterbodies reduce maximum yields, demonstrating their importance for retention during loading events. In order to reduce loads reaching Green Lake, it is important to understand whether, and under what conditions, the lentic areas of CKM and SCE retain TN, TP, and SS.

Patterns of TP retention in shallow lakes change seasonally and are influenced by the dominant biological equilibrium state – phytoplankton or macrophytes (Søndergaard *et al.*, 2013). Even after external sources of P are reduced, shallow lakes can exhibit delayed reductions in TP export because of accumulated TP in sediments (Robertson *et al.*, 2018). Shallow lake sediments can serve as a TP source (*i.e.* internal P loading). Many physical, chemical, and biological processes, such as wind mixing, oxygen concentrations, pH, geology, and trophic state, contribute to the quantity of P transported from the benthic environment to the water column (Orihel *et al.*, 2017; Robertson *et al.*, 2018). Rising temperatures increase benthic microbial activity which consumes oxygen and releases P bound to sediment iron (Liu *et al.*, 2018). Anoxic conditions spur redox cycling of P, increasing TP release within a shallow lake. As a result, even systems with reduced TP inputs can exhibit net TP export over the summer (Søndergaard *et al.*, 2013). However, macrophyte roots can add oxygen to sediments, which increases their ability to retain P and reduces overall internal P loading within a shallow water body (Kasan *et al.*, 2016)

Given the prominent agricultural land use within the CKM and SCE watersheds, accumulated P laden sediment could contribute to internal loading under favorable conditions. Quantifying the potential for internal loading can guide decisions on how to reduce TP export to Green Lake, either through source reductions upstream of the marshes or targeting processes within the marshes.

Differences in loading to the lake from each of the marshes may be a result of larger inputs into one marsh compared to the other or a result of different internal processes stemming from the marsh's turbid or clear condition. The comparison of nutrient loading into and out of both marshes and evaluation of seasonal loads and relative contributions of TP, TN, and SS to Green Lake was needed. Restoration efforts for CKM are underway and, therefore, quantifying loads under current turbid conditions will establish a baseline to compare future conditions if an alternate clear water equilibrium is reached. Thus, objectives of this research were to:

1. Quantify differences in total phosphorus (TP) and suspended sediment (SS) loads from the two marshes since 2012,
2. Quantify and compare spatial and temporal changes in TP, total suspended solids (TSS), and TN concentrations and loads within the two marshes during the growing seasons of 2016 and 2017,
3. Evaluate whether each marsh acts as a sink or source of TP, TN, and TSS, and
4. Quantify P variation within and the potential for P release from sediments within the two marshes.

Methods Monthly Average USGS Data

Nutrient loading data at the outlet of each marsh from October 2012 – August 2017 were obtained from USGS gages at County Trunk Highway A (CTH A) and County Trunk Highway K (CTH K; Figure 3-1). These data included flow, and TP and SS concentrations and loads at the outlets of CKM (CTH K; USGS site 40734605) and SCE (CTH A; USGS site 4073468). Stream flow at the SCE outlet was not measured directly but rather estimated by the USGS using data from the upstream gage at Spaulding Road (4073466). The SS loads at the SCE outlet was only available from October 2014 – August 2017. The USGS collected samples and analyzed them for TP (USGS parameter 00665) and SS (USGS parameter 80154) concentrations. Water stage data (USGS parameter 00065) were recorded using a gage. Over this time period, the USGS collected 294 TP samples and 283 SS samples at the CKM outlet, corresponding to an average frequency of about one sample every six days. At the SCE outlet, the USGS collected 100 TP samples (average of once every 18 days) and 62 SS samples (average of once every 17 days). The USGS used stage data to estimate mean daily stream flow (USGS parameter 00060) at each site. The Graphical Constituent Loading Analysis System (GCLAS) software was then used with daily stream flow and concentration data to calculate daily mean SS (USGS parameter 80155) and TP (USGS parameter 91050) loads (Koltun *et al.*, 2006).

For both gages, all available monthly average stream flow (cfs), SS load (tons/day), and TP load (lb/day) were downloaded from the USGS National Water Information System (NWIS; USGS, 2018). At the SCE outlet, monthly average stream flows were used through September 2015. Monthly average stream flows from October 2015 – August 2017 for the SCE outlet used monthly average stream flows from Spaulding Rd multiplied by a factor of 1.087 (as determined by USGS). Approved monthly average SS and TP loads at CTH K and CTH A were available through August 2016. Monthly values from September 2016 to August 2017 were calculated using provisional daily data from NWIS since approved monthly values were not available when the analysis began. All data were converted into metric unit equivalents.

Daily precipitation data (October 2012 – August 2017) were downloaded from the National Oceanic and Atmospheric Administration's (NOAA) Climate Data online tool (https://www.ncdc.noaa.gov/cdo-web/datatools/findstation) for a station in Markesan (USC00475096). The station was located outside of the Green Lake watershed but had the closest and most complete available precipitation dataset. Daily precipitation depths were averaged over each month from 2012 – 2017 to match the frequency of the USGS load data.

The TP and SS loads were compared seasonally and as a function of stream flow. All data were divided into four seasons: winter (December – February), spring (March – May), summer (June – August), and fall (September – November). The USGS average monthly TP and SS loads were divided by corresponding average monthly stream flows to estimate monthly flow-weighted mean TP and SS concentrations for each marsh. Monthly average retention times for each marsh were estimated to assess how quickly nutrients and sediments are flushed through each system. The water volume for each marsh was obtained from Onterra, LLC, a lake management planning consulting group that previously conducted several projects within Green Lake (Personal communication, E. Heath, Onterra, LLC). The volumes of SCE and CKM were 679,648 m^3 and 1,094,105 m^3, respectively. The monthly average retention time (T_R; days) for water was estimated as:

$$T_R = \frac{V}{Qa} \qquad (1)$$

Where: V = Volume of marsh (m^3)

Q = Monthly average flow at marsh outlet (m^3/s)

a = 86400 (s/day)

Multiple linear regressions (MLRs) were developed to examine factors that impact precipitation, flow, TP and SS loads, and TP and SS flow-weighted mean concentrations at the marsh outlets (Table 3-1). Explanatory variables were analyzed using ANCOVAs. Significant predictors (α = 0.05) from each ANCOVA were identified. Identifying explanatory variables of each water quality parameter allowed relationships among variables to be determined and helped explain observed differences. The MLRs were also used to identify significant variation between groups of modeled data for the two marshes.

Statistical analysis used data representing the daily average for each month, hereafter referred to as the monthly average. The effects of relevant explanatory variables (both continuous and categorical) on monthly average flow and TP and SS loads and flow-weighted mean concentrations (TP and SS) at the marsh outlets were analyzed. The R statistical software was used to create MLR models (Team, 2018), and the *car* package was used to perform an analysis of covariance (ANCOVA) on the models (Fox, 2018). Explanatory variables within each MLR were examined for both additive effects and interactions influencing the dependent variable. An interaction between two explanatory variables occurs when a change in the value of one variable depends on the value of the other; the two variables are not independent and the effect of one is better understood when the other variable is also considered. The number of observations within each season were unequal, so the unbalanced design was statistically evaluated using type III sum of squares in the ANCOVA. The method for type III sum of squares includes all model variables in adjustments for the sum of squares so main effects of individual variables are adjusted for interactions (Hector *et al.*, 2010). If independent variables showed significant interactions, the effects of those variables individually were not interpreted, only their interactive effects together. Models initially incorporated a variety of interactions, but interactions that were not statistically significant (α = 0.05) were removed from the final model. To improve data and model residual normality so that parametric methods could be used for statistical analysis, most continuous data were transformed using natural logarithms prior to statistical analysis (Fu and Wang, 2012).

Once an appropriate MLR model was established for a given dependent variable (one that included only significant predictors/interactions and predictors/interactions of specific interest), significance of each were noted. Estimated marginal means (also known as least-squares means) were then computed for the model using the *emmeans* package for R (Lenth *et al.*, 2018). Marginal means for the dependent variable were estimated for quantiles of modeled stream flow based on precipitation and season. The marginal means were used to compare the statistical difference between the outcome variable based on the two marshes and based on season. Values were analyzed using a Tukey pairwise comparison to obtain a p value for seasonally-grouped and all data.

*Table 3-1: Variables considered in MLR for monthly average precipitation, flow, TP and SS loads and TP and SS flow-weighted mean concentrations at the outlets of CKM and SCE. Grey boxes mark variables evaluated as part of a significant interaction. X marks variables evaluated for statistical significance in final MLR. Blanks were variables not considered in final MLR. Site refers to either CKM or SCE outlet. * Flow and and retention time used the same predictive model.*

Outcome		**Precipitation (mm/day)**	**Flow* (m^3/s)**	**TP Load (kg/day)**	**TP (mg/L)**	**SS Load (metric tonnes/day)**	**SS (mg/L)**
Predictor	**Transformation**	ln	ln	ln	ln	ln	ln
Season (Winter, Spring, Summer, Fall)		x					
Date (monthly continuous scale)		x		x	x	x	x
Season and Date Interaction			x				
Precipitation (mm/day)	ln		x	x	x		
Site							
Season and Site Interaction			x	x	x	x	x
Flow (m^3/s)	ln			x	x	x	
SS Load (metric tonnes/day)	ln						
Site and SS Load (metric tonnes/day) Interaction	ln			x			
SS (mg/L)	ln						
Site and SS (mg/L) Interaction	ln				x		

Water Quality Sampling

Samples were collected at marsh inlets and at sites within the marshes during the growing seasons of 2016 and 2017 (Figure 3-6). Samples were analyzed for TP, total dissolved phosphorus (TDP), total nitrogen (TN), total suspended solids (TSS), pH, electric conductivity (EC), and chlorophyll A (CHLA).

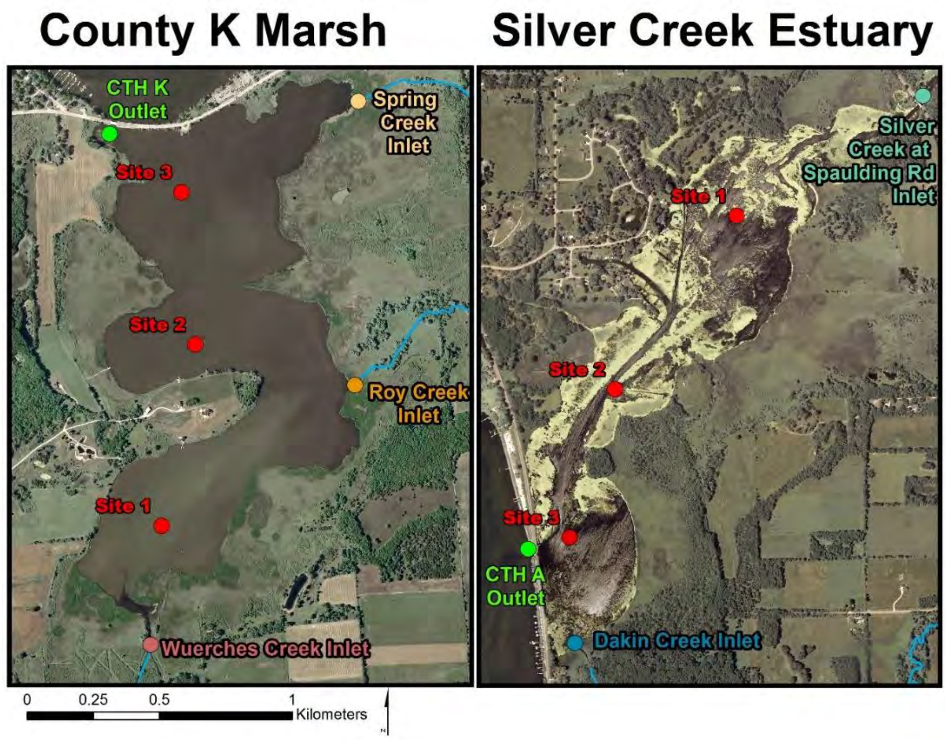

Figure 3-6: Sampling locations for grab samples collected in 2016 and 2017. Since Site 3 in SCE was close to the CTH A outlet, grab samples were only collected at Site 3 and not at the CTH A bridge.

Monthly grab samples were collected from July – October 2016 and April – October 2017 at seven locations within CKM and at five locations in SCE; locations were accessed by kayak (Figure 3-6; Figure 3-7). The SCE Site 3 concentrations were considered representative of the SCE outlet because of proximity to the CTH A monitoring site. Samples were collected near the end of each month in 2016 and in the middle of each month in 2017 and represented a variety of high and low flow events. Grab samples were collected at a depth of approximately 0.5 m from the surface. Shortly after water collection, subsamples tested for TDP were filtered through a 0.45 µm syringe filter. Samples analyzed for TP and TN were acidified with H_2SO_4 in the field. All samples were preserved with ice and then refrigerated (near 4° C) until delivered to the Wisconsin State Laboratory of Hygiene for nutrient testing. Samples were analyzed for TP (method EPA 353.1), TDP (method EPA 365.1), TN (method EPA 353.2), CHLA (method EPA 445), and TSS (method SM2450D). Samples at Sites 1 and 2 in both marshes in September 2016 and at Sites 1 – 3 in June – August 2017 were measured for CHLA. From June – October 2017, samples from all locations were measured for EC and pH at the Water Quality Laboratory in the Biological Systems Engineering Department at the University of Wisconsin – Madison.

Figure 3-7: Collecting grab samples by kayak. Photo taken October 8, 2017.

Daily Load Calculations

For each of the 11 grab sampling events, the concentrations at the inlets and outlets of each marsh were used to estimate daily loads of TP, TN, and TSS. USGS average daily flows at the Spaulding Rd, CTH K, and Roy Creek Rd (4073458) gages for each day of sampling were used to estimate an area-based equivalent flow for ungauged subwatersheds.

The Green Lake watershed was divided into subwatersheds for a Soil and Water Assessment Tool (SWAT) model developed by Baumgart (2015). . Those delineations served as the watersheds for each inlet to CKM and SCE, except for watersheds where the SWAT defined outlet did not match the location of a USGS gage (Roy Creek Rd and Spaulding Rd).

SWAT annual water yields at the three inlets to CKM (obtained from Paul Baumgart, Assistant Scientist at the University of Wisconsin – Green Bay) were used in scaling gaged flows from the USGS Roy creek gage. For Roy Creek, a subwatershed to the gage was delineated (red outline in Figure 3-8). The USGS flows measured at the Roy Creek gage were then multiplied by an area ratio of 1.16 (ratio of Roy Creek watershed to Roy Gage watershed) to estimate the total flow contribution for the full Roy Creek tributary. The SWAT subwatershed annual water yield for Spring and Wuerches Creeks were divided by the annual water yield of Roy Creek (ratios in Table 3-2). The estimated Roy Creek daily flows (modified from the original gage flow) were multiplied by the appropriate annual water yield ratio to estimate the total daily flow for Spring and Wuerches Creeks.

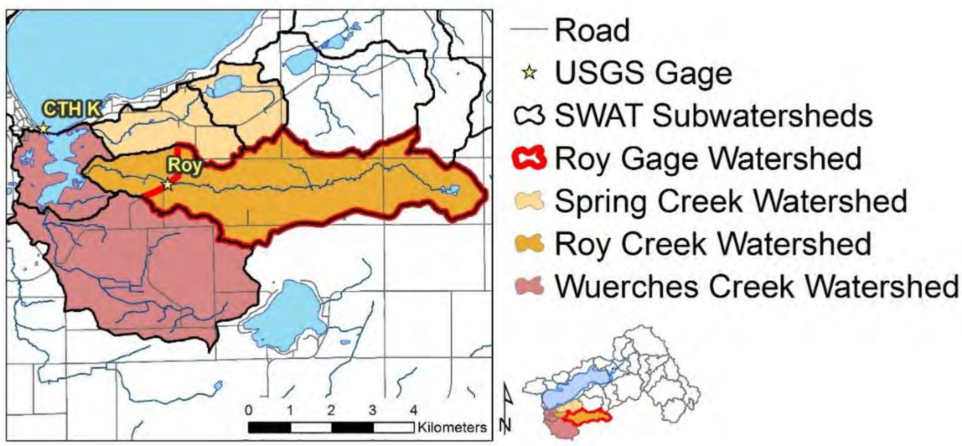

Figure 3-8: County K Marsh subwatersheds used for calculating daily flows via a ratio based on watershed size and modeled annual water yield.

Table 3-2: Annual water yields and annual water yield ratios used to estimate flows to CKM.

Subwatershed	Annual Water Yield (m^3)	Ratio to Roy Creek
Spring Creek	1,769,509	0.48
Wuerches Creek	4,487,391	1.22
Roy Creek	3,675,649	1.00

USGS gaged flows were only available for Spaulding Rd. However, SCE also received flow from Silver C. 1a and Dakin Creek subwatersheds (SWAT defined; Figure 3-9). To determine the amount of flow originating downstream of Spaulding Rd, Silver C. 1a subwatershed was modified (expanded) to include area up to the Spaulding Rd gage and to include area in the southeast corner of SCE that was unassigned in the original SWAT model (red outline in Figure 3-9). The annual water yield for Silver Creek 1a was multiplied by the ratio of the new area to the original area to determine the total annual water yield from SCE itself (Table 3-3).

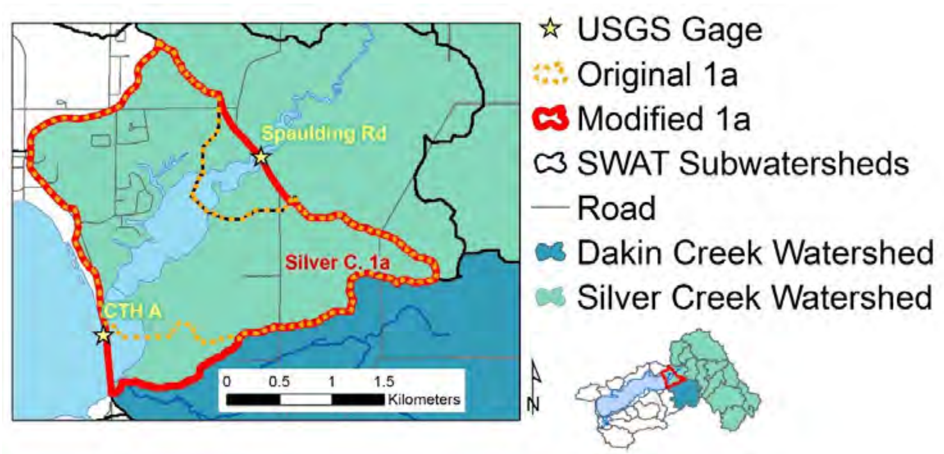

Figure 3-9: SCE subwatersheds modified from the original SWAT delineation.

Table 3-3: The area ratio of the modified to original subwatershed used to calculate a modified annual water yield.

Subwatershed	Area ratio	Original Annual Water Yield (m^3)	Modified Total Annual Water Yield (m^3)
Silver C. 1a	1.208	991,271	1,197,675

In 2012, USGS determined that the subwatershed for the Spaulding Rd gage accounted for 92% of the flow observed at the CTH A SCE outlet. The remaining 8% of flow, therefore, came from Silver C. 1a and Dakin Creek subwatersheds. The ratio of the modified SWAT annual water yield from Silver C. 1a subwatershed to SWAT annual water yield for Dakin Creek subwatershed was 0.3; Silver C. 1a contributed the equivalent of 30% of what Dakin Creek contributed. Using this ratio results in Dakin Creek contributing 6% of the total flow of SCE outlet and subwatershed Silver C. 1a contributing 2% (Table 3-4). The USGS daily flow at the Spaulding Rd gage was multiplied by the appropriate ratio to determine daily flow at Dakin Creek and at the SCE outlet (Table 3-4; Figure 3-10).

Table 3-4: Flow comparisons for SCE inlet and outlet subwatersheds

Subwatershed	% Outlet Flow	Ratio to Spaulding Gage Flow
Spaulding Rd Gage	92	1
Dakin Creek	6	0.065
Silver Creek Estuary	100	1.08

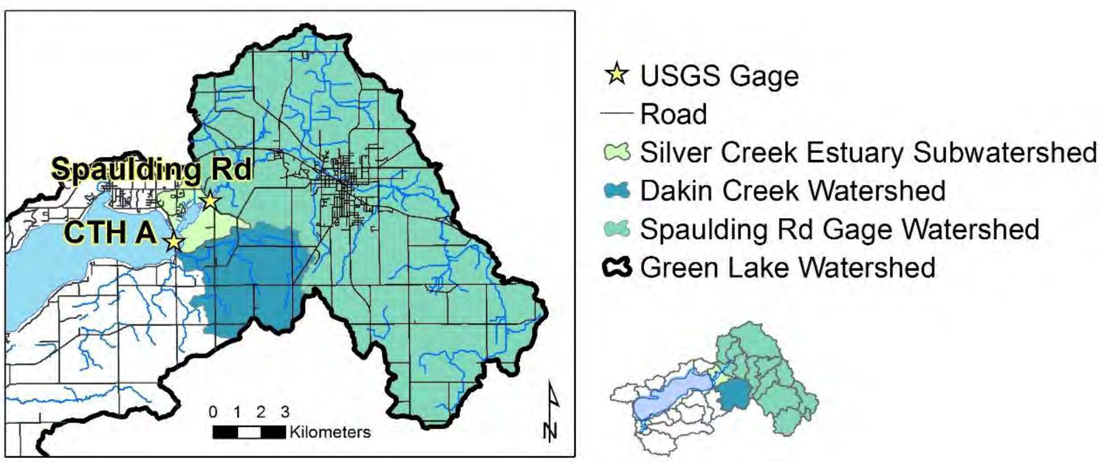

Figure 3-10: SCE subwatersheds used for calculating daily flows via a ratio based on watershed size and SWAT modeled annual water yield.

Daily-average-scaled flows were multiplied by the nutrient concentration at the respective inlets and outlets to estimate daily loads. The TP, TN, and TSS daily loads at the inlet and outlet were compared to determine if each marsh acted as a sink or source of nutrients to Green Lake. Removal efficiency (RE) within each marsh for each sampling event was calculated as:

$$RE\ (in\ \%) = \frac{I_l - O_l}{I_l} \times 100 \qquad (2)$$

where: I_l = Sum of inlet daily loads for each sampling day (TP and TN: kg/day; TSS: metric tonnes/day)

O_l = Outlet daily loads for each sampling day (TP and TN: kg/day, TSS; metric tonnes/day)

Sediment Cores

Since carp and wind disturb sediment, and because sediment P release can be a large source of internal P loading, differences in sediment texture, TP concentration, and potential P release were compared for sediments collected at different locations within each marsh. Sediment cores were collected once at the six interior sites in CKM and SCE (Figure 3-6). Locations were accessed by motor boat, and at each sampling site, a water grab sample was collected at the sediment/water interface and the water depth was recorded. Water samples were analyzed for dissolved reactive P (DRP). Four to five 4.7 cm diameter sediment cores were collected at each site (Figure 3-11). Each core was taken on a different side of the boat to avoid sampling disturbed sediment. Surface sediments from 0 – 2.5 cm depth from all cores at each site were combined in a Ziplock bag. Sediments from 2.5 – 5 cm depth were combined in a separate bag. Samples at each site were analyzed for extractable P using a dilute salt extraction. Four additional cores, with sediment from 0 – 5 cm depth, were collected at Site 2 in each marsh (Figure 3-6). These samples were combined at each site for equilibrium P concentration at zero sorption (EPC_0) analysis.

Figure 3-11: Collecting sediment samples in SCE from motor boat. Photo taken July 17, 2017.

Sediment samples were sifted through a 2 mm sieve and a portion of each sample was oven dried (60°C) overnight to obtain a dry weight equivalent. This dry weight equivalent was later used in the calculation for potential P release following a dilute salt extraction. A portion of each dried sample was sent to the Wisconsin State Laboratory of Hygiene for TP analysis (method SW846 6010B). The analyses for texture, the dilute salt P extraction, and equilibrium phosphorus concentration were performed at the Water Quality Laboratory of the Biological Systems Engineering Department at University of Wisconsin – Madison.

Texture Analysis

Particle size was determined for dried and sieved (2 mm sieve) sediments (depth 0 – 5 cm) from each interior site. 20 – 40 g of dried sediment was mixed with 100 ml of a 5% Sodium metaphosphate ($Na_6(PO_3)_6$) solution in a 250 ml bottle. Bottles were placed in a rotating shaker at a speed of 15 rpm for 16 hours. Sample contents were combined with 1000 ml of deionized water in a glass cylinder. The contents of the cylinder were mixed with a plunger for 30 seconds to obtain uniform suspension. The plunger was removed and a stopwatch started. At 30 seconds, a hydrometer was inserted into the column of water, and a reading was taken at 40 seconds to determine the amount of silt and clay suspended in solution. A second hydrometer reading was taken at 6 hours and 52 minutes to determine the amount of clay in suspension. For each hydrometer reading, the level of a hydrometer and temperature in a blank of only deionized water were recorded.

Sample hydrometer readings were corrected based on temperature as follows:

$$H_{ct} = (H_t - 0.2(T_{bt} - 67)) - H_{bt} \qquad (3)$$

where: H_{ct} = Corrected hydrometer reading at time t (g/L)

H_t = Sample hydrometer reading at time t (g/L)

T_{bt} = Temperature of blank at time t (°F)

H_{bt} = Blank hydrometer reading at time t (g/L)

The particle distribution was then determined as:

$$\%\ Clay = \frac{H_{c2} \times 100}{M} \qquad (4)$$

where: H_{c2} = Corrected hydrometer reading at 6 hours and 52 minutes (g/L)

M = Mass of sample (g)

and

$$\%\ Silt = \frac{H_{c1} \times 100}{M} - \%\ Clay \qquad (5)$$

where: H_{c1} = Corrected hydrometer reading at 40 seconds (g/L)

and

$$\% \, Sand = 100 - \% \, Silt - \% \, Clay \qquad (6)$$

These particle fractions were used to determine soil textures using the USDA textural triangle (method from UW – Madison Soil and Plant Analysis Lab; personal communication Z. Zopp).

Dilute Salt Extraction

To evaluate the potential for sediment to release readily available P, a dilute salt extraction was performed under laboratory conditions on subsamples of sediment from each marsh. These initial P release values can indicate specific sites or depths of sediment within each marsh more susceptible to P release. A dilute salt extraction was performed on the top (0 – 2.5 cm) and bottom (2.5 – 5 cm) sediment samples from all six interior sites. The most readily available P, that which could quickly desorb into solution, was measured after a short period of mixing (Self-Davis *et al.*, 2009). The wet equivalent of 1.0 g of dry sediment was added to a 60 ml bottle. Twenty-five ml of a 0.002 M $CaCl_2$ solution and 3 drops of chloroform were added to inhibit microbial growth. A solution of 0.002 M $CaCl_2$, with an electrical conductivity of 500 µS/cm, was used because it was similar to the average electrical conductivity of the water samples, 570 µS/cm. Extractions were performed in triplicate. The solutions were vortexed for 10 seconds at 1450 rpm and then placed in an orbital mechanic shaker at 200 rmp for one hour. Samples were poured into 50 ml tubes and centrifuged at 2000 x *g* for 12 minutes. The supernatant was filtered (0.45 µm) and analyzed for dissolved reactive P (DRP) in an AQ2 discrete autoanalyzer using SEAL AQ2 Method EPA-118-A Rev. 5.

The dilute salt-extractable P (mg/kg sediment) was estimated as:

$$\frac{CV}{M} \qquad (7)$$

where: C = Concentration of P in solution following an hour of extraction (mg/L)

V = Volume of extractant (L)

M = Mass of sediment (kg)

Equilibrium Phosphorus Concentration

The equilibrium phosphorus concentration (EPC_0) defines the P concentration at which sediment P release and adsorption are in equilibrium. Sediments are expected to release P into the water column if water column concentrations are below the equilibrium concentration and adsorb P if concentrations are above. Identifying the ambient P concentration at which sediments are likely to release P is helpful in understanding conditions under which sediments are likely to contribute to internal P loading.

Subsamples of the 0 – 5 cm depth cores (Site 2 in each marsh) were analyzed for equilibrium phosphorus concentration (EPC_0) using a U.S. Department of Agriculture method (Graetz and Nair, 2009). Sorption capacity was estimated using 0.002 M $CaCl_2$ solutions with P concentrations of 0, 0.25, 0.5, 1, and 2.5 mg/L. A solution of 0.002 M $CaCl_2$, with an electrical conductivity of 500 µS/cm, was used because it was similar to the average electrical conductivity of the water samples, 570 µS/cm. The wet equivalent of 1.0 g of dry sediment was added to a 60 ml bottle. Twenty-five ml of a $CaCl_2$ P solution and 3 drops of chloroform were added to inhibit microbial growth. All samples were analyzed in triplicate. A blank bottle (no replication) containing only the 0.002 M $CaCl_2$ solution was analyzed to check for contamination. The solutions were vortexed for 10 seconds at 1450 rpm and then placed in an orbital mechanic shaker at 200 rmp for 24 hours to equilibrate. Samples were then poured into 50 ml tubes and centrifuged at 2000 x *g* for 12 minutes. The supernatant was filtered (0.45 µm) and analyzed for dissolved reactive P (DRP) in an AQ2 discrete autoanalyzer using SEAL AQ2 Method EPA-118-A Rev. 5.

The DRP concentrations in solution were used in the two-step method of the Langmuir equation to determine equilibrium phosphorus concentration (Zhang *et al.*, 2009):

$$S' = \frac{(C_i - C) \times V}{M} \qquad (8)$$

where: S' = Amount of P removed from solution and adsorbed to sediment following a 24 hr equilibration period (mg/kg sediment)

C_i = Initial P concentration in solution (mg/L)

C = P concentration in solution after 24 hr equilibration (mg/L)

V = Volume of aqueous solution (L)

M = Mass of soil (kg)

and

$$S' = KC - S_0 \qquad (9)$$

where: K = Slope of best linear fit of S' plotted versus C; P equilibrium buffering capacity or bonding energy constant (L/kg sediment)

S_0 = Negative y-intercept for best linear fit of S' versus C; original P bound to solids (mg/kg sediment)

When the amount of P removed from solution (S') is equal to zero, the equilibrium DRP concentration (EPC_0; mg/L) of the solution is:

$$EPC_0 = S_0/K \qquad (10)$$

The DRP in the solution will be adsorbed by the sediments when the DRP solution concentration is greater than EPC_0. Below the EPC_0 solution concentration, P desorption from the sediments will occur, increasing DRP levels in solution. The average S' values of the three replicates for each solution concentration were plotted versus C to determine a linear isotherm and an estimated EPC_0 for each marsh.

Sediment samples from SCE with initial P concentrations of 0 and 0.25 mg/L were re-analyzed because initial results were non-linear. These results were combined with the original results and averaged for each initial P concentration (0 and 0.25 mg/L).

Results

Average Monthly Loads 2012 – 2017

Precipitation, Stream Flow, and Retention Time

Since central Wisconsin has precipitation and stream flow patterns that vary seasonally, quantifying this variation was important to understand subsequent seasonal fluctuations in TP and SS loads. The median monthly average precipitation from October 2012 – August 2017 was 2.25 mm/day (Table 3-5). Seasonal median precipitation followed the order of winter < fall < spring < summer and ranged from 1.18 – 3.53 mm/day (Figure 3-12 B). The wettest months were September 2016 (average precipitation of 7.03 mm/day) and June 2014 (average of 6.85 mm/day; Figure 3-12 A). The driest months were January (0.36 mm/day) and March (0.35 mm/day) 2015. Precipitation was significantly predicted by season ($p < 0.001$; Appendix C Table C-1); winter had less precipitation than the other seasons ($p < 0.001$). There was no trend in precipitation ($p = 0.09$).

*Table 3-5: Seasonal median precipitation, flow, and marsh retention time for monthly average data from October 2012 – August 2017. CKM:SCE ratio is the ratio of the two marsh medians. Significant differences between modeled means of the two marshes (p < 0.05) are bolded. *n = 13 for fall SCE Outlet stream discharge. **n = 58 for all seasons SCE Outlet flow.*

		Winter	Spring	Summer	Fall	All Seasons
Median Precipitation (mm/day)		1.18	2.98	3.53	2.04	2.25
Median Flow (m^3/s)	**CKM Outlet**	0.26	0.59	0.32	0.26	0.31
	SCE Outlet	0.58	1.92	1.09	0.60	0.84
	CKM:SCE Ratio	0.45	0.31	0.29	0.43	0.38
	p value	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Median Retention Time (day)	**CKM Outlet**	49.5	21.4	40.0	49.0	40.3
	SCE Outlet	13.7	4.1	7.2	13.0	9.4
	CKM:SCE Ratio	3.61	5.22	5.56	3.77	4.29
	p value	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
n		15	15	15	14*	59**

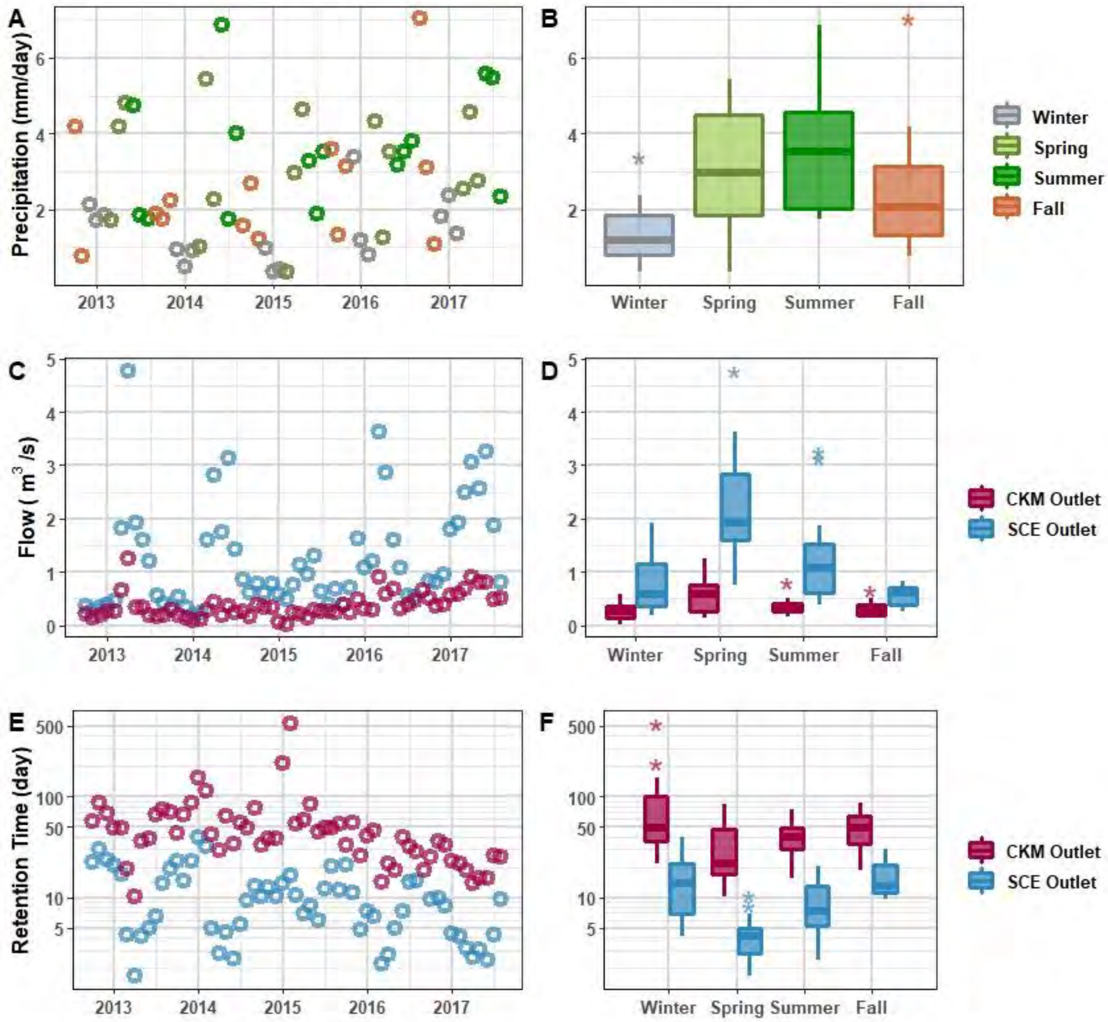

Figure 3-12: (A) Time series of monthly average precipitation from October 2012 – August 2017. Markesan, WI (B) Boxplot of monthly average precipitation grouped by season. (C) Time series of monthly average flow at the outlet of CKM and SCE October 2012 – August 2017. Provisional data was used after September 2016. (D) Boxplot of same monthly average flow grouped by season. (E) Time series of monthly average retention time for CKM and SCE from October 2012 – August 2017. Provisional data was used after September 2016. (G) Boxplot of same monthly average retention times grouped by season. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

The median monthly average stream flow at the CKM outlet from October 2012 – August 2017 was 0.31 m^3/s (Table 3-5). Median seasonal stream flow at the CKM outlet followed the order of winter = fall < summer < spring and ranged from 0.26 – 0.59 m^3/s (Figure 3-12 D). The median monthly average stream flow at the SCE outlet was 0.84 m^3/s (Table 3-5). Median seasonal stream flow at the SCE outlet followed the order of winter < fall < summer < spring and ranged from 0.58 – 1.92 m^3/day (Figure 3-12 D). The highest monthly average stream flows (1.26 m^3/s and 4.77 m^3/s at the CKM and SCE outlets, respectively) occurred April 2013 (Figure 3-12 C). In spring and summer, the CKM median flows were about 30% of SCE median stream discharges (Table 3-5). This ratio increased to about 45% in the winter and fall. Significant explanatory variables of flow were precipitation ($p < 0.001$) and the interactions between season and date ($p = 0.009$) and season and site ($p = 0.027$; Appendix C Table C-1). Overall, flow increased with precipitation. Flows increased more over time in fall and winter compared to summer and spring (Figure 3-13). These increases were expected to explain some variation in TP and SS loads.

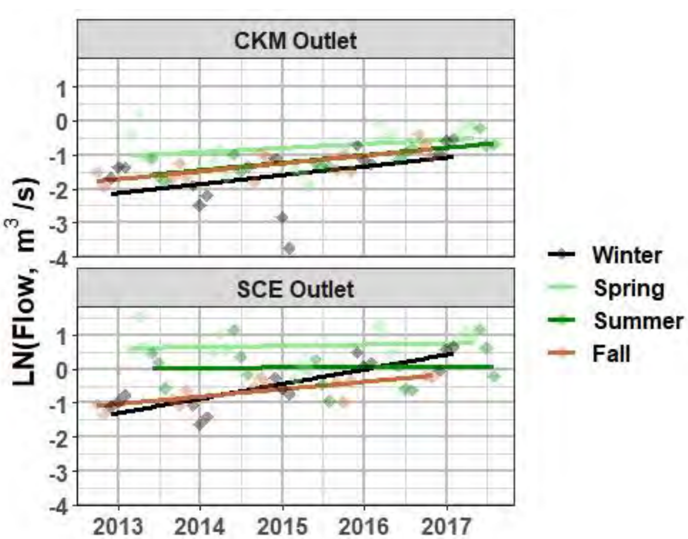

Figure 3-13: Naturally log transformed flow at the outlets of CKM and SCE versus date.

Retention times ranged from 1.6 – 39.8 days in SCE, which were shorter than CKM retention times (range of 10.1 – 533 days; Figure 3-12 E). The overall median CKM retention time (40.3 days) was more than 4 times longer than the median SCE retention time (9.4 days;

Table 3-5). Median retention times were similar in winter and fall for both marshes (49 days and 13 days in CKM and SCE, respectively; Figure 3-12 F). Median retention times were shortest in spring (21.4 days and 4.1 days in CKM an SCE, respectively). For all seasons, retention times were significantly longer in CKM compared to SCE ($p < 0.001$). Significant explanatory variables of retention time were precipitation ($p < 0.001$) and the interactions between season and date ($p = 0.009$) and season and site ($p = 0.027$; Appendix C Table C-1). These mirrored significant explanatory variables of stream flow because volume was held constant. Longer retention times lead to more opportunity for settling of sediment and uptake of TP. However, resuspension of sediment by wind can also occur when water flows slowly through a shallow system. These differences in retention times were expected to help explain some of the variation observed in SS and TP loads.

Total Phosphorus Loads

The median monthly average TP load at the CKM outlet from October 2012 – August 2017 was 2.65 kg/day (Table 3-6). Median seasonal CKM outlet TP load followed the order of winter < fall < summer < spring and ranged from 1.16 – 6.5 kg/day (Figure 3-14 B). Seasonality in loads can be a result of both biological processes within the marshes and seasonal variation in streamflow and wind. At the CKM outlet, the highest TP load (30.7 kg/day) occurred in May 2017. The median monthly average TP load at the SCE outlet was 6.00 kg/day (Table 3-6). Seasonal median SCE TP load followed the order of fall < winter < summer < spring and ranged from 2.23 – 15.3 kg/day (Figure 3-14 B). The highest TP load for the SCE outlet (50.6 kg/day) occurred in June 2017 (Figure 3-14 A). The SCE seasonal median TP load was greater than the CKM seasonal median TP load in all seasons except fall (Table 3-6). Loads from SCE were expected to be larger because of its larger flow and contributing watershed area. Significant explanatory variables of TP loads were precipitation (p = 0.001), season (p = 0.005), stream flow (p <0.001), and the interaction between site and SS load (p < 0.001; Appendix C Table C-2). The TP load increased with precipitation and stream flow. Overall, modeled predicted mean TP loads (from the MLR which identified significant explanatory variables such as stream flow) were significantly greater for CKM (p = 0.03) than for SCE (even though the median TP load was greater for SCE than CKM). Seasonally, CKM predicted mean TP loads were only significantly greater than SCE in the fall (p = 0.001; Table 3-6). Increasing SS loads were more associated with an increase in TP loads at the CKM outlet than at the SCE outlet (Figure 3-15 A).

Table 3-6: Median TP outlet load and median flow-weighted mean TP concentration at the outlets of CKM and SCE. CKM:SCE ratio is the ratio of the two medians for the given variable. Significant differences between modeled means of the two marshes (p < 0.05) are bolded. ^ predicted means for CKM were significantly larger than SCE. **n = 13 for fall SCE Outlet flow-weighted TP concentration. **n = 58 for all seasons SCE Outlet flow-weighted TP concentration.*

		Winter	**Spring**	**Summer**	**Fall**	**All Seasons**
Median TP Load (kg/day)	**CKM Outlet**	1.16	6.50	5.32	2.41	2.65
	SCE Outlet	3.10	15.30	7.78	2.23	6.00
	CKM:SCE Ratio	0.37	0.42	0.68	1.08	0.44
	p value	0.733	0.258	0.167	**0.001^**	**0.03^**
Median Flow-weighted Mean TP Concentration (mg/L)	**CKM Outlet**	0.057	0.127	0.211	0.092	0.118
	SCE Outlet	0.048	0.086	0.105	0.062	0.068
	CKM:SCE Ratio	1.19	1.48	2.01	1.48	1.74
	p value	0.063	**0.009^**	**0.016^**	**<0.001^**	**<0.001^**
n		15	15	15	14*	59**

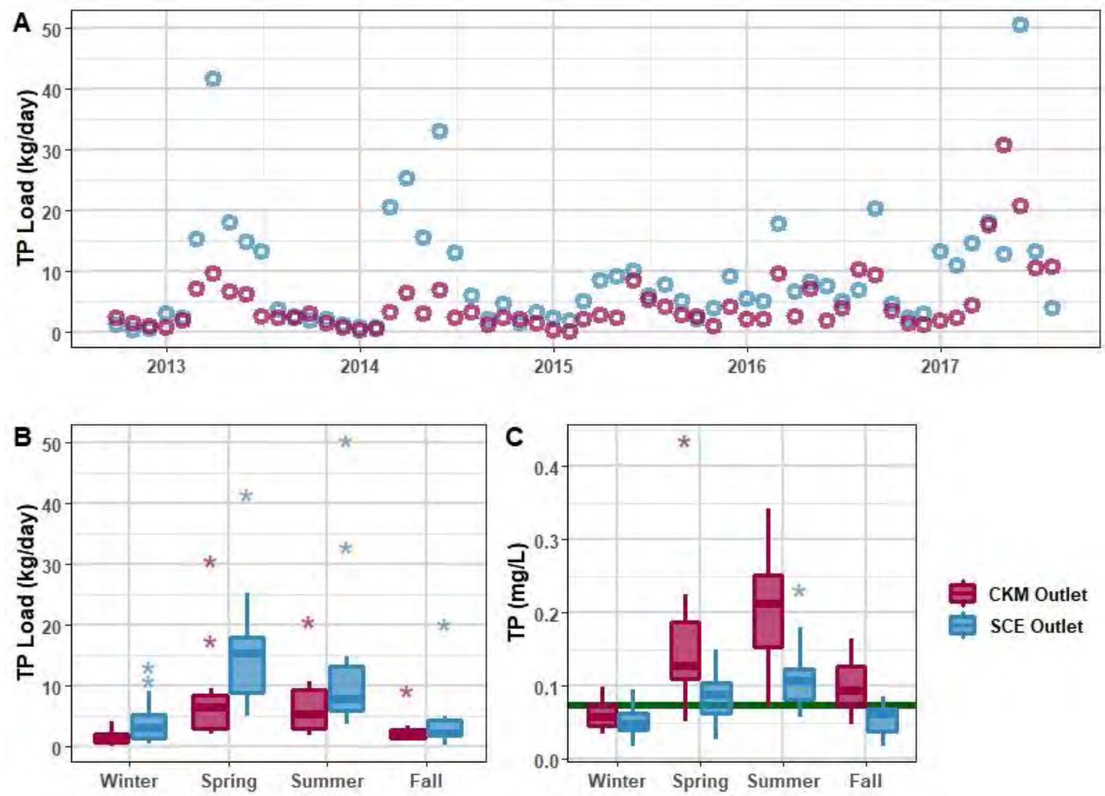

Figure 3-14: (A) Time series of monthly average TP load at the outlets of CKM and SCE October 2012 – August 2017. (B) Monthly average TP load grouped by season. (C) Monthly average flow-weighted TP concentration at the outlets of CKM and SCE. Green line is Wisconsin water quality standard of 0.075 mg/L. Monthly average data from October 2012 – August 2017. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

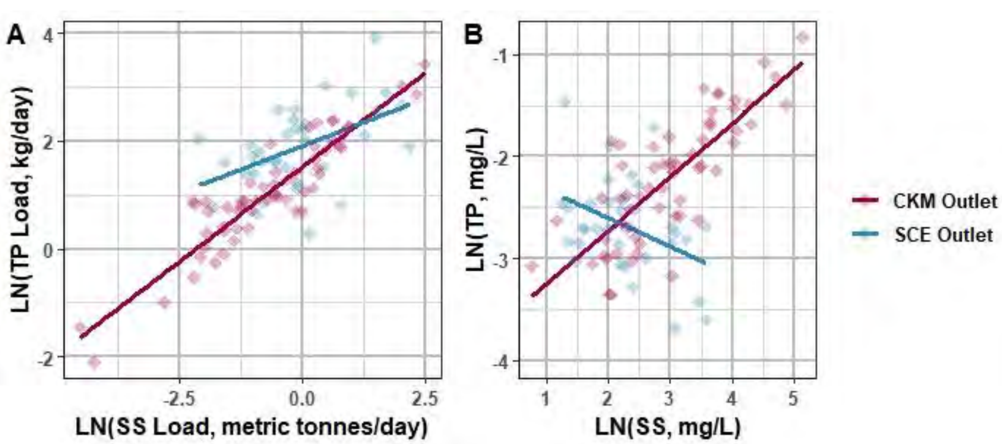

Figure 3-15: (A) Monthly average TP load versus SS load at the outlets of CKM and SCE. (B) Monthly average flow weighted TP concentrations versus monthly average SS concentration at the outlets of CKM and SCE. All variables are naturally log transformed.

The median monthly average flow-weighted TP concentration at the CKM outlet from October 2012 – August 2017 was 0.118 mg/L (Table 3-6). Seasonal median flow-weighted mean TP concentrations followed the order winter < fall < spring < summer and ranged from 0.057 – 0.211 mg/L (Figure 3-14 C). Median concentrations were highest in the growing season. The median monthly average flow-weighted TP concentration at the SCE outlet was 0.068 mg/L (Table 3-6). Seasonal median flow-weighted mean TP concentrations followed the order of winter < fall < spring < summer and ranged from 0.048 – 0.105 mg/L (Figure 3-14 C). The CKM seasonal median flow-weighted TP concentrations were higher than the SCE seasonal median flow-weighted TP concentrations in all seasons. This difference was significant in all seasons except winter ($p = 0.063$). Significant explanatory variables of flow-weighted TP concentration were precipitation ($p = 0.013$), season ($p = 0.008$), and the interaction between site and flow-weighted SS concentration ($p < 0.001$; Appendix C Table C-2). Flow-weighted TP concentration increased with precipitation and stream flow. Winter had significantly lower TP concentrations than summer ($p = 0.025$) in both marshes, and in SCE, fall concentrations were also significantly lower than summer ($p = 0.004$). Increasing flow-weighted SS concentrations were associated with increasing flow-weighted TP concentrations at the CKM outlet but not at the SCE outlet (Figure 3-15 B).

Suspended Sediment Loads

The median monthly average SS load at the CKM outlet from October 2012 – August 2017 was 0.56 metric tonnes/day (Table 3-7). Median seasonal SS load followed the order winter < fall < summer < spring and ranged from 0.2 – 1.38 metric tonnes/day (Figure 3-16 B). The median monthly average SS load at the SCE outlet from October 2014 – August 2017 was 0.86 metric tonnes/day (Table 3-7). Median seasonal SS load followed the order of summer <

winter < fall < spring and ranged from 0.58 – 3.59 metric tonnes/day (Figure 3-16 B). At the CKM outlet, the highest SS load (12.06 metric tonnes/day) occurred in May 2017 (Figure 3-16 A). The highest SS load for the SCE outlet (9 metric tonnes/day) occurred in April 2016. The SCE seasonal median SS load was higher than the CKM seasonal median SS loads in all seasons except summer. Significant explanatory variables of SS load were stream flow ($p < 0.001$) and the interaction between site and season ($p = 0.001$; Appendix C Table C-3). Date was not a significant explanatory variable ($p = 0.21$). The SS load increased with flow. Altogether, predicted mean SS loads (from the MLR based on explanatory variables such as stream discharge) were significantly higher for CKM than for SCE ($p < 0.001$), but seasonally, this difference was only significant in spring and summer ($p < 0.001$; Table 3-7). Even though the spring CKM median and overall CKM median SS loads were less than those for SCE, the modeled means for spring CKM and overall CKM SS loads were significantly higher than those for SCE. After taking into account the larger flow for SCE, the differences in SS load between the two systems was significant, and the load for CKM was higher. For CKM, SS loads were significantly higher in spring and summer compared to fall and winter ($p = 0.001$). At the SCE outlet, however, there were no significant differences among seasons in SS loads.

*Table 3-7: Median SS load and median flow-weighted mean SS concentrations at the outlet of CKM and SCE. CKM:SCE ratio is the ratio of the two medians for the given variable. Significant differences between modeled means of the two marshes ($p < 0.05$) are bolded. ^ predicted mean for CKM were significantly larger than SCE. *n = 7 for fall SCE Outlet flow-weighted SS concentration. **n = 34 for all seasons SCE Outlet flow-weighted SS concentration.*

		Winter	Spring	Summer	Fall	All Seasons
Median SS Load (metric tonnes/day)	**CKM Outlet**	0.20	1.38	1.30	0.30	0.56
	SCE Outlet	0.83	3.59	0.58	0.93	0.86
	CKM:SCE Ratio	0.24	0.38	2.24	0.32	0.65
	p value	0.133	**<0.001^**	**<0.001^**	0.218	**<0.001^**
Median Flow-weighted Mean SS Concentration (mg/L)	**CKM Outlet**	8.48	33.60	39.60	14.50	19.80
	SCE Outlet	7.38	20.50	7.59	10.50	9.26
	CKM:SCE Ratio	1.15	1.64	5.22	1.38	2.14
	p value	0.290	**0.001^**	**<0.001^**	0.339	**<0.001^**
n	**CKM Outlet**	15	15	15	14	59
	SCE Outlet	9	9	9	8*	35**

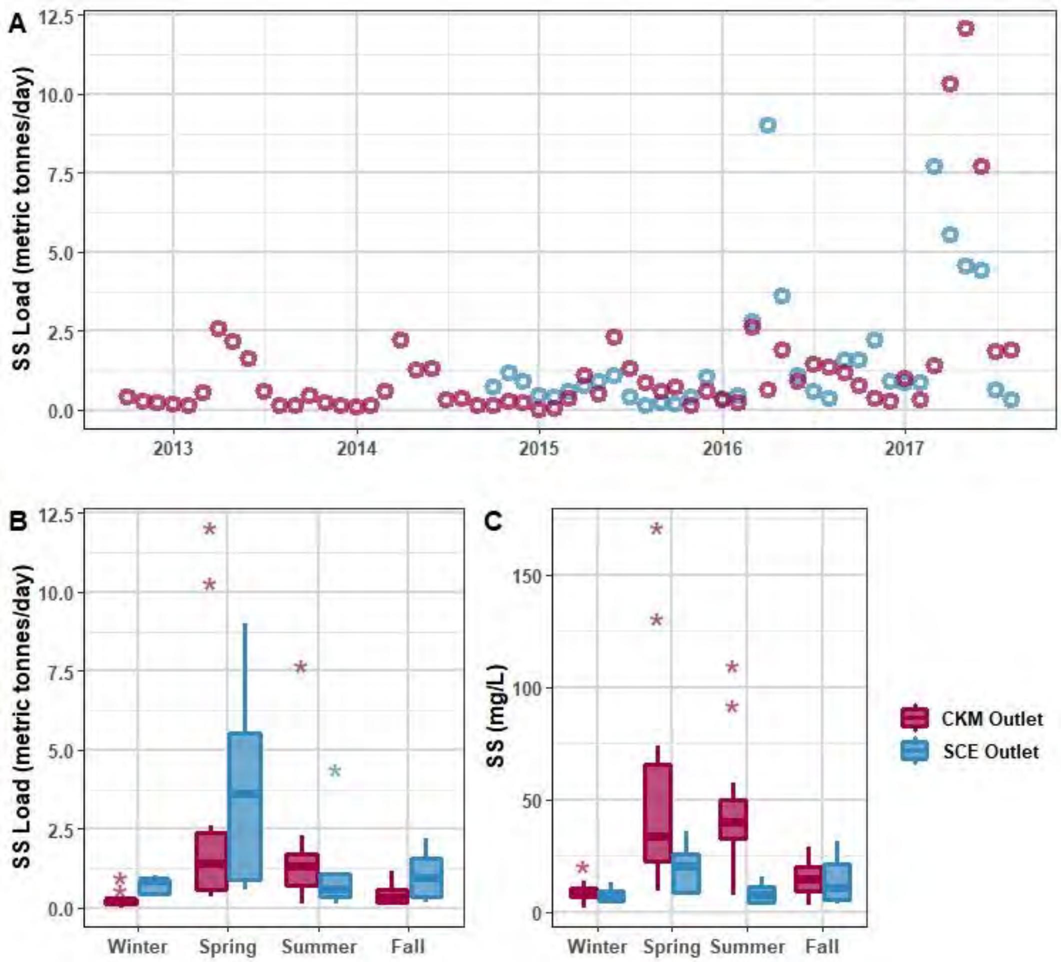

Figure 3-16: (A) Time series of monthly average SS load at outlets of CKM (October 2012 – August 2017) and SCE (October 2014 – August 2017). (B) Monthly average SS load grouped by season. (C) Monthly average flow-weighted SS concentration at the outlets of CKM and SCE. Monthly average data from October 2012 – August 2017 for CKM and October 2014 – August 2017 for SCE. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

The median monthly average flow-weighted SS concentration at the CKM outlet from October 2012 – August 2017 was 19.8 mg/L (Table 3-7). Median seasonal flow-weighted SS concentrations followed the order winter < fall < spring < summer and ranged from 8.48 – 39.6 mg/L (Figure 3-16 C). The median monthly average flow-weighted SS concentration at the SCE outlet from October 2014 – August 2017 was 9.26 mg/L (Table 3-7). Median seasonal flow-weighted SS concentrations followed the order winter < summer < fall < spring and ranged from

7.38 – 20.5 mg/L (Figure 3-16 C). The CKM seasonal median SS flow-weighted concentration was higher than the SCE seasonal median SS flow-weighted concentration in all seasons, but the difference was only significant in spring and summer (p = 0.001; Table 3-7). Significant explanatory variables of flow-weighted SS concentration were date (p = 0.04) and the interaction between site and season (p < 0.001; Appendix C Table C-3). Flow-weighted SS concentration increased slightly with time (Figure 3-17). For CKM, spring and summer flow-weighted SS concentrations were significantly higher than concentrations in winter and fall (p <0.001). For SCE, flow-weighted SS concentrations were only significantly higher in spring compared to summer and winter (p = 0.035).

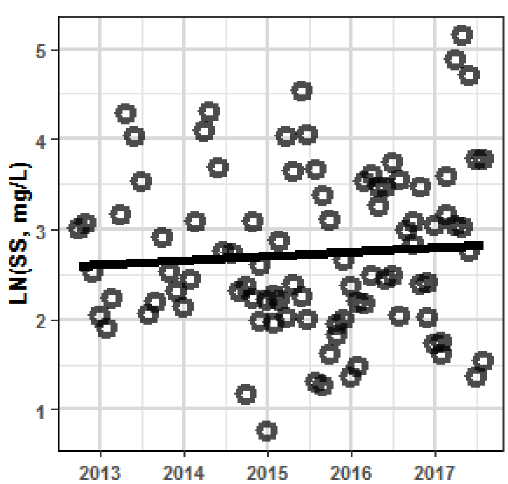

Figure 3-17: Natural log of flow-weighted SS concentration versus time for marsh outlet data (both CKM and SCE). Both marshes are graphed together because there was no significant interaction between time and marsh (p = 0.9).

Seasonal averages for flow-weighted mean SS and TP concentrations in both marshes were similar in winter (Figure 3-18). In summer, average concentrations of SS and TP in SCE were lower than in CKM. For CKM, concentrations in both spring and summer of 2017 were higher compared to 2016. At the CKM outlet, the seasonal average of the flow-weighted SS concentrations was highest in spring 2017 (108.4 mg/L; Figure 3-18 A). While the highest seasonal average of CKM outlet flow-weighted TP concentrations occurred in summer 2017 (0.27 mg/L), spring 2017 had the second highest seasonal average concentration (0.24 mg/L; Figure 3-18 B). In 2017, the average CKM spring flow-weighted SS concentration was four times higher than in 2016 (25.8 mg/L). For the SCE outlet, however, the average flow-weighted SS concentrations were similar in the springs of 2016 and 2017 (23.37 mg/L and 25.67 mg/L, respectively).

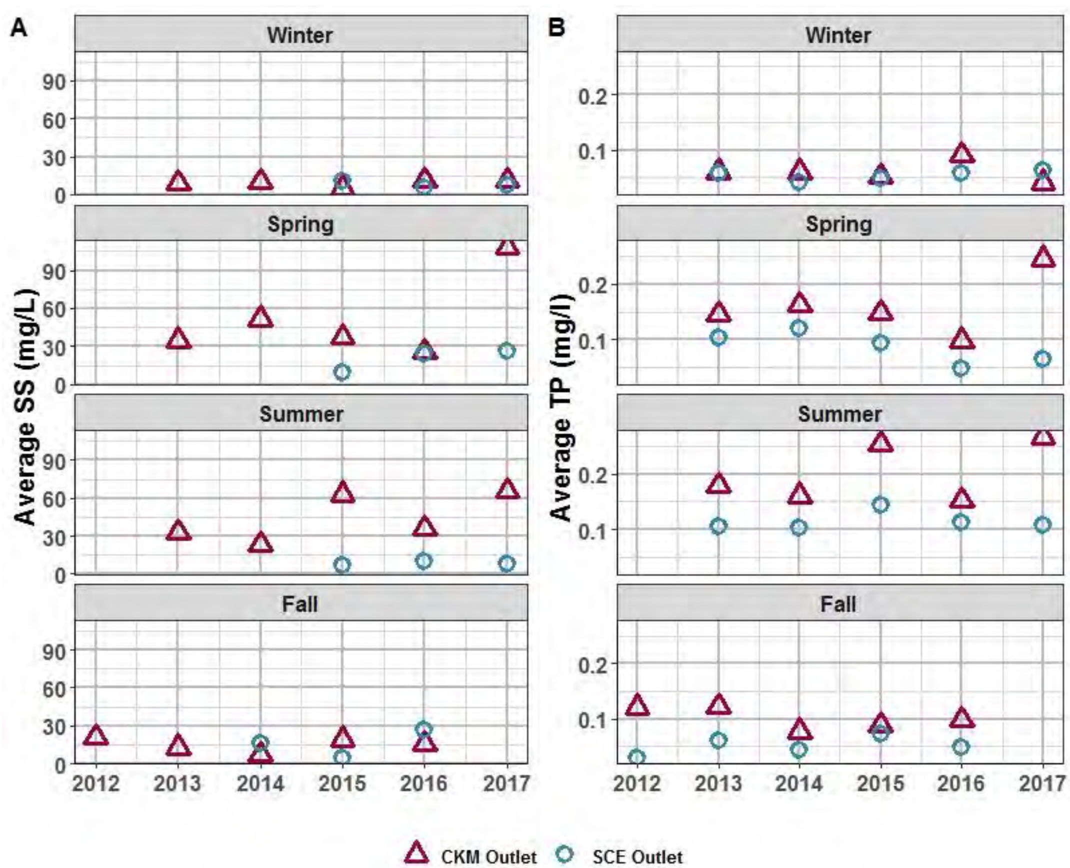

Figure 3-18: Seasonal average flow-weighted (B) SS concentration and (A) TP concentration at the outlet of CKM and SCE from 2012 – 2017.

Grab Sample Concentrations

Precipitation over the seven days preceding each of the 11 grab sample collection dates ranged from 0 – 76 mm (**Error! Reference source not found.**Table 3-8). Days with more than 60 mm of precipitation over the preceding seven days were termed ‘storm events’. The other eight sampling days were ‘non-storm events’. Maps showing the spatial distribution of grab sample concentrations are located in Appendix D (Figure D-1 thru Figure D-8).

*Table 3-8: Dates for grab sampling in CKM and SCE. Cumulative precipitation over the preceding day, 3 days, and 7 days are shown. *Sampling following at least 60 mm of precipitation over the preceding 7 days are bolded as storm events.*

Sampling Date	Precipitation over preceding		
	Day (mm)	3 Days (mm)	7 Days (mm)
7/26/2016	0	9	39
8/29/2016	0	29	40
9/23/2016*	**2**	**60**	**71**
10/24/2016	0	0	0
4/16/2017*	**19**	**48**	**62**
5/16/2017	14	14	20
6/15/2017*	**30**	**76**	**76**
7/13/2017	3	8	26
8/15/2017	0	0	9
9/15/2017	0	0	0
10/8/2017	5	14	19

Phosphorus

The TP concentrations ranged from a minimum of 0.025 mg/L at the Spring Creek inlet to CKM to a maximum of 0.709 mg/L at the Wuerches Creek inlet to CKM (Appendix E Table E-1). Of the inlets to CKM, Wuerches had the highest median TP concentration (0.101 mg/L) and Spring Creek had the lowest (0.03 mg/L; Figure 3-19 A). For the SCE inlets, Spaulding Rd had a higher median (0.119 mg/L) than Dakin Creek (0.063 mg/L).

The median TP concentration within SCE was 0.074 mg/L, just below the TP impairment level of 0.075 mg/L (Figure 3-19 A). In contrast, the median TP concentration for the interior CKM sites (Sites 1-3 and the outlet) was 0.207 mg/L and only two of the 44 samples were below the TP impairment level of 0.075 mg/L. The TP concentrations increased most evidently with precipitation at the Wuerches and Roy Creek inlets (Figure 3-20). Within CKM, median TP concentrations decreased from 0.253 mg/L at Site 1 to 0.132 mg/L at the outlet (Figure 3-21; Appendix E Table E-2), but the median TP outlet concentration remained higher than the median inlet concentration (0.080 mg/L; Figure 3-21 A). Within SCE, the median TP concentration also decreased from Site 1 (0.099 mg/L) to Site 3/outlet (0.054 mg/L). Both Sites 2 (0.062 mg/L) and 3/outlet had median TP concentrations lower than the inlet median (0.088 mg/L).

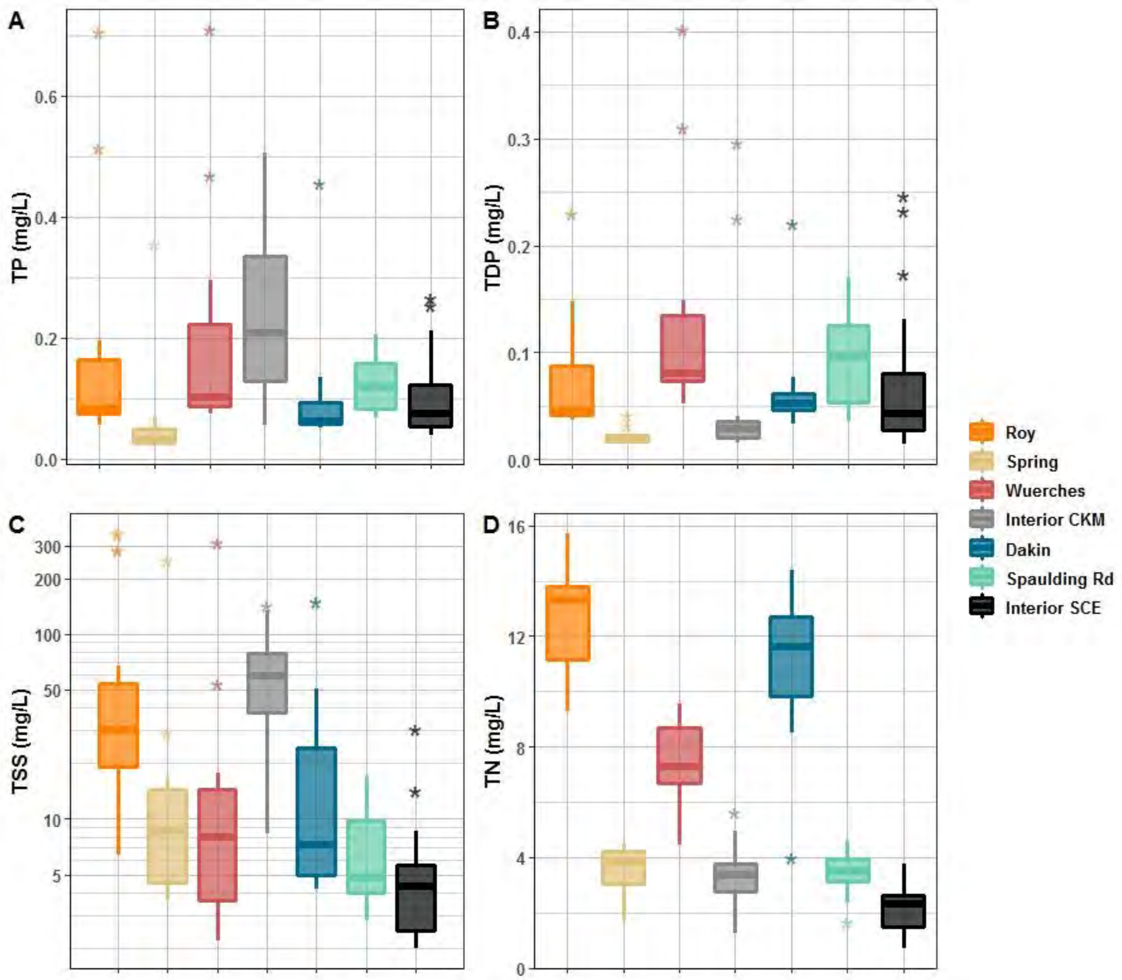

Figure 3-19: (A) Total phosphorus, (B) total dissolved phosphorus, (C) total suspended solids, and (D) total nitrogen concentrations in grab samples collected at inlet streams and interior locations of County K Marsh and Silver Creek Estuary. Samples collected monthly July – October 2016 and April – October 2017. Each inlet is displayed separately and the interior marsh sites (Sites 1, 2, 3 and outlet) are grouped. Grouped inlets and separate within marsh sites shown separately in Figure 3-21. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

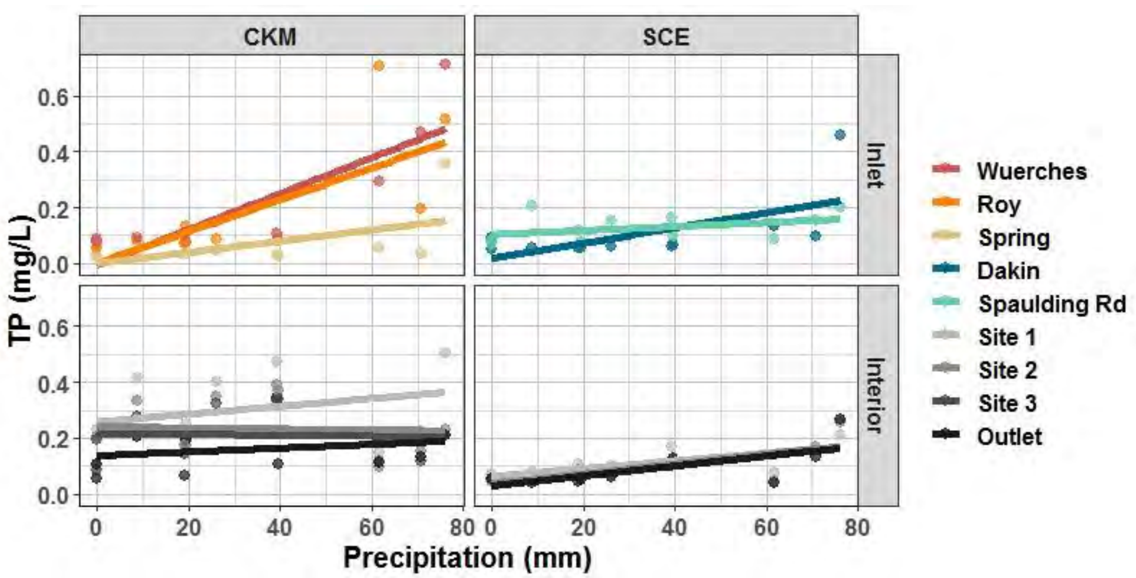

Figure 3-20: Best linear fit for grab sample TP concentrations versus cumulative precipitation over the preceding seven days. Results separated by marsh and by inlet or interior location.

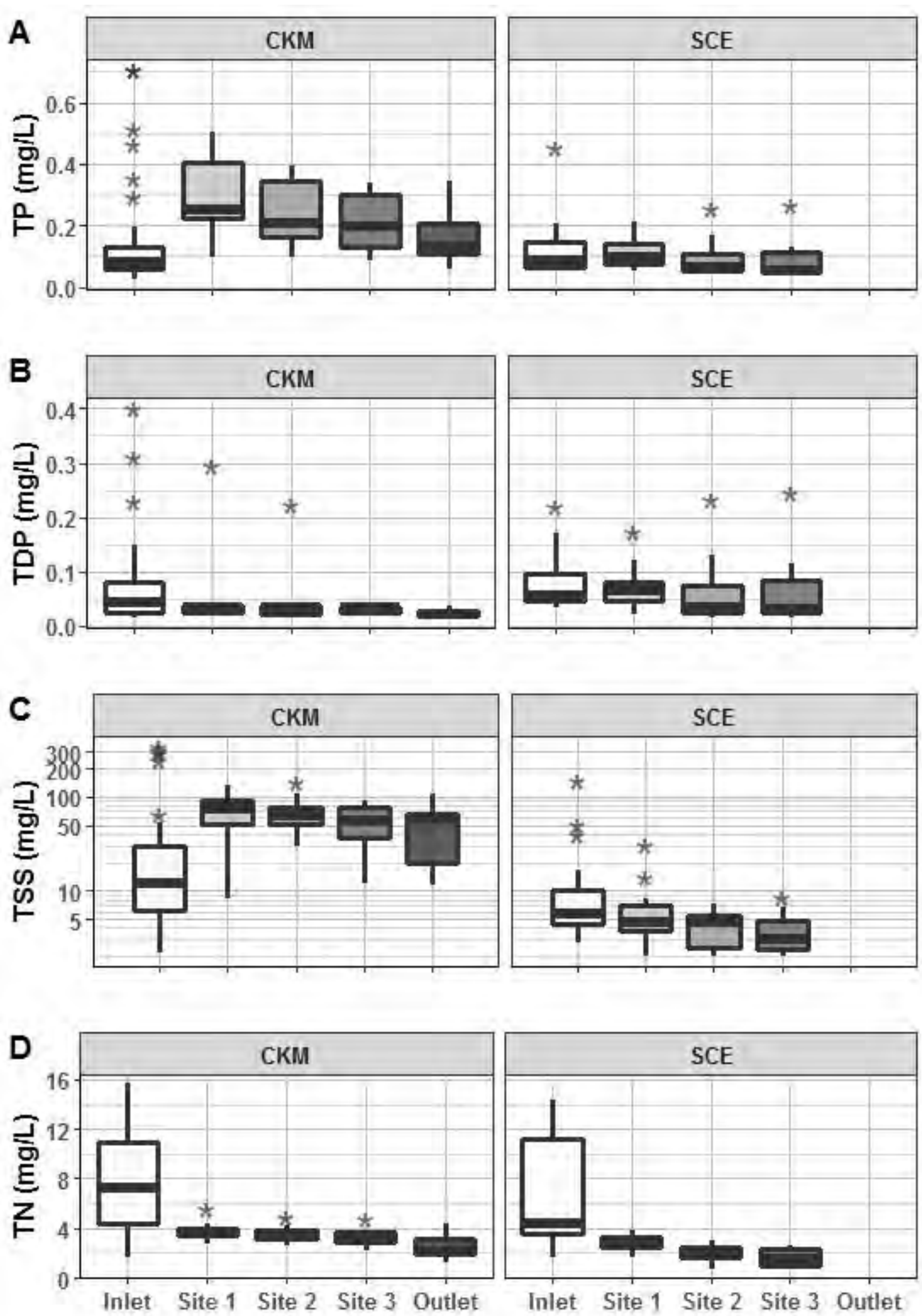

Figure 3-21: (A) Total phosphorus, (B) total dissolved phosphorus, (C) total suspended solids concentrations, and (D) total nitrogen in grab samples collected at inlet streams and interior locations of County K Marsh and Silver Creek Estuary. Samples collected monthly July – October 2016 and April – October 2017. Inlet concentrations are grouped, and the interior marsh sites are plotted separately. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

Of the inlets to CKM, the highest TDP median (0.08 mg/L) was at Wuerches Creek and the lowest was at Spring Creek (0.02 mg/L; Figure 3-19 B). For the SCE inlets, the median at Spaulding Rd (0.095 mg/L) was higher than at Dakin Creek (0.052 mg/L). Increased precipitation during the seven days prior to sampling corresponded to increased TDP, most evidently at Wuerches, Roy, and Dakin Creeks (Figure 3-22).

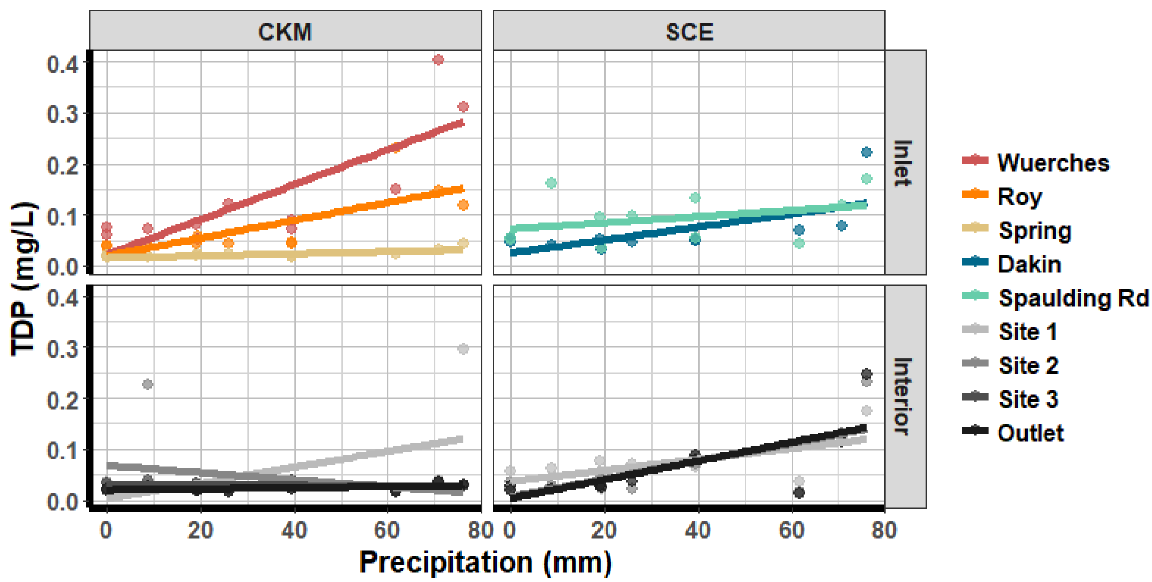

Figure 3-22: Best linear fit for grab sample TDP concentrations versus cumulative precipitation over the preceding seven days. Results separated by marsh and by inlet or interior location.

At the interior sites, median TDP concentration at Site 1 was higher in SCE (0.065 mg/L) than in CKM (0.033 mg/L; Figure 3-21 B; Appendix E Table E-2). Median TDP concentrations were similar at the three sites within CKM but were slightly lower at the outlet (0.021 mg/L). In SCE, the median TDP concentration decreased from Site 1 (0.065 mg/L) to 3/outlet (0.029 mg/L).

Median TDP:TP for the interior sites of CKM was 0.14 and of SCE was 0.6. TDP:TP decreased with increasing precipitation for Wuerches, Roy, Spring, and Dakin Creeks. The highest TDP:TP median was at Wuerches Creek (0.8), and Spaulding Rd and Dakin Creek were slightly lower (0.76).

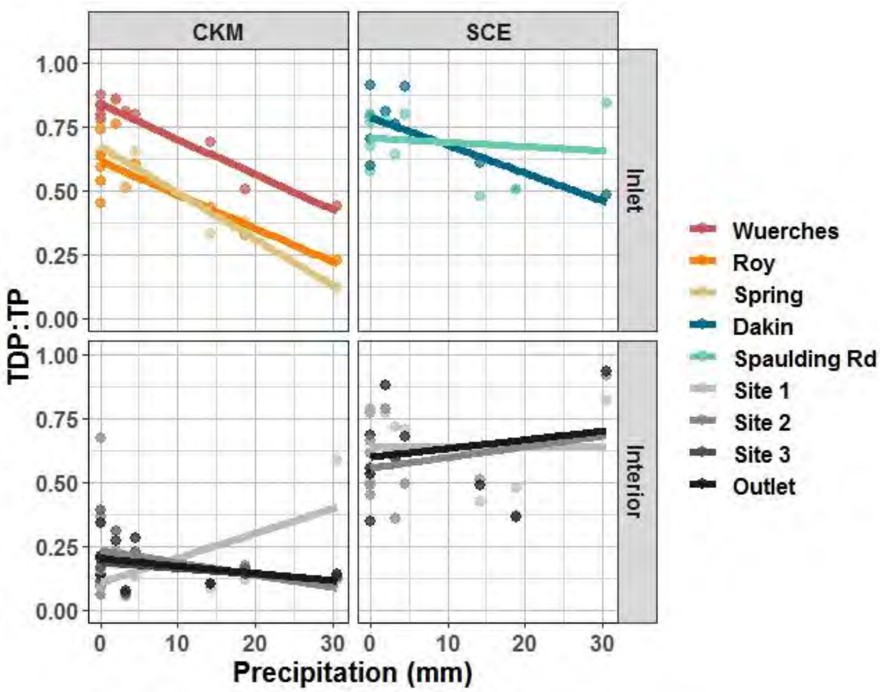

Figure 3-23: Best linear fit for grab sample TDP to TP ratio versus sampling day precipitation. Results separated by marsh and by inlet or interior location.

Total Suspended Solids

TSS concentrations ranged from below the level of detection within SCE to a maximum of 348 mg/L at Roy Creek (Appendix E Table E-1). Of the inlets to CKM, Roy Creek had the highest TSS median concentration (30.4 mg/L; Figure 3-19 C). Median TSS concentrations were similar for Spring and Wuerches Creeks (8 mg/L). For the SCE inlets, Dakin Creek had a higher median (7.2 mg/L) than Spaulding Rd (4.8 mg/L).

Within CKM, the interior sites had median TSS concentrations almost five times higher (≥ 56 mg/L) than at the inlets (11.8 mg/L; Figure 3-21 C, Appendix E Table E-2). Within SCE, median concentrations at all interior sites (≤ 4.67 mg/L) were less than the median concentration of the inlets (5.6 mg/L). The TSS was more strongly correlated to precipitation at the inlets to CKM and at Dakin Creek than at the other sampling sites (Figure 3-24 A). There was also stronger correlation between TP and TSS at these sites (Figure 3-24 B). The TP increased with TSS more strongly within CKM than within SCE (Figure 3-24 B).

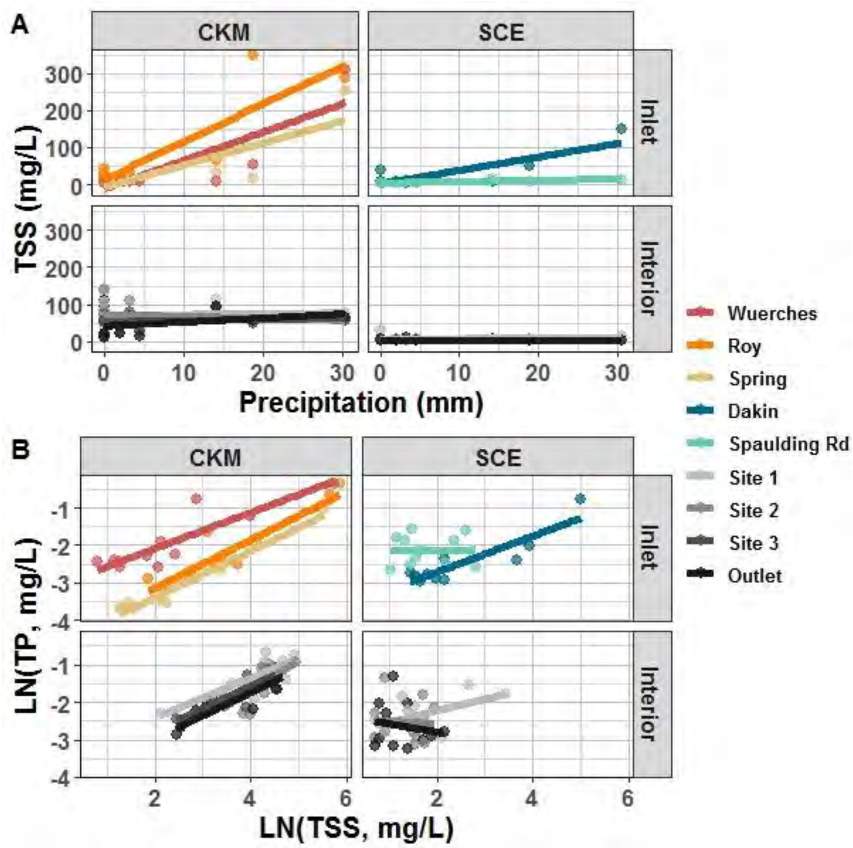

Figure 3-24: (A) Best linear fit for grab sample TSS concentrations versus sampling day precipitation. (B) Best linear fit for TSS concentrations versus TP concentrations in grab samples taken at inlet and interior sites of CKM and SCE. Both parameters are naturally log transformed. Results separated by marsh and by inlet or interior location.

Total Nitrogen

TN concentrations ranged from a minimum of 0.71 mg/L within SCE to a maximum of 15.7 mg/L in Roy Creek (Appendix E Table E-1). Of the inlets to CKM, Roy Creek had the highest median TN concentration (13.3 mg/L) and Spring Creek had the lowest (3.83 mg/L; Figure 3-19 D). Of the SCE inlets, median TN concentration was higher at Dakin Creek (11.6 mg/L) than at Spaulding Rd (3.48 mg/L).

Within CKM, median TN concentration decreased from Site 1 (3.6 mg/L) to the outlet (2.14 mg/L; Appendix D Figure 3-21 D, Appendix E Table E-2). Similarly, within SCE, median TN decreased from 2.64 mg/L at Site 1 to 1.49 mg/L at Site 3/outlet. The TN concentration at the inlets decreased with increasing precipitation (Figure 3-25). Within each marsh, TN concentrations upstream and downstream became more similar as precipitation increased.

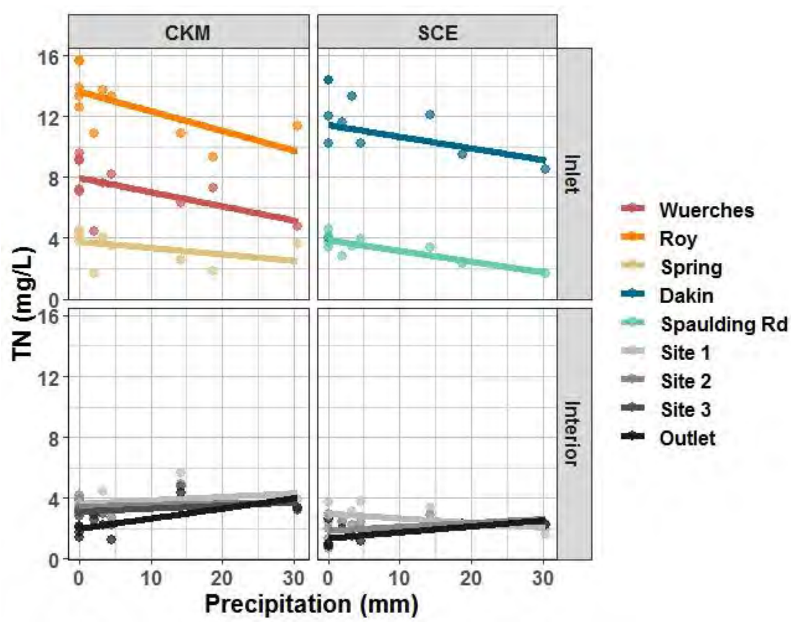

Figure 3-25: Best linear fit for TN concentrations in grab samples versus sampling day precipitation. Results are separated by marsh and by inlet or interior sites.

When there was little precipitation, TN to TP molar ratios were higher at the inlets (except at Spaulding Rd) compared to the interior sites of the marshes (Figure 3-26). Dakin and Roy Creeks had the highest median TN:TP ratios of all the sites (Dakin = 389, Roy = 347), with maxima exceeded 600 in October 2016. Spaulding Rd and the interior sites of SCE had similar median TN:TP ratios (60), while interior sites of CKM had a lower median TN:TP ratio of 35. TN:TP ratios generally decreased with increasing precipitation, though the relationship was weakest for interior sites of CKM. The TN:TP ratios at the inlets decreased below the P limiting range (ratio of 22.6; Guildford & Hecky, 2000) only with the most precipitation over the previous seven days.

The lowest average TN:TP within SCE (18.8) occurred after the June 2017 storm, and both marshes had average TN:TP ratios below 22.6 in July 2016 (Figure 3-27). For CKM, TN:TP ratios were also below 22.6 in July and August 2017. Within CKM, the two highest average TN:TP ratios occurred in October 2016 (88) and April 2017 (68). Within SCE, the two highest average TN:TP ratios occurred in October 2016 (117) and May 2017 (121). The minimum average TN:TP ratio observed (13) was in CKM in July 2016. For CKM, lower TN:TP ratios were associated with lower proportions of TDP, whereas the opposite was observed in SCE (Figure 3-28).

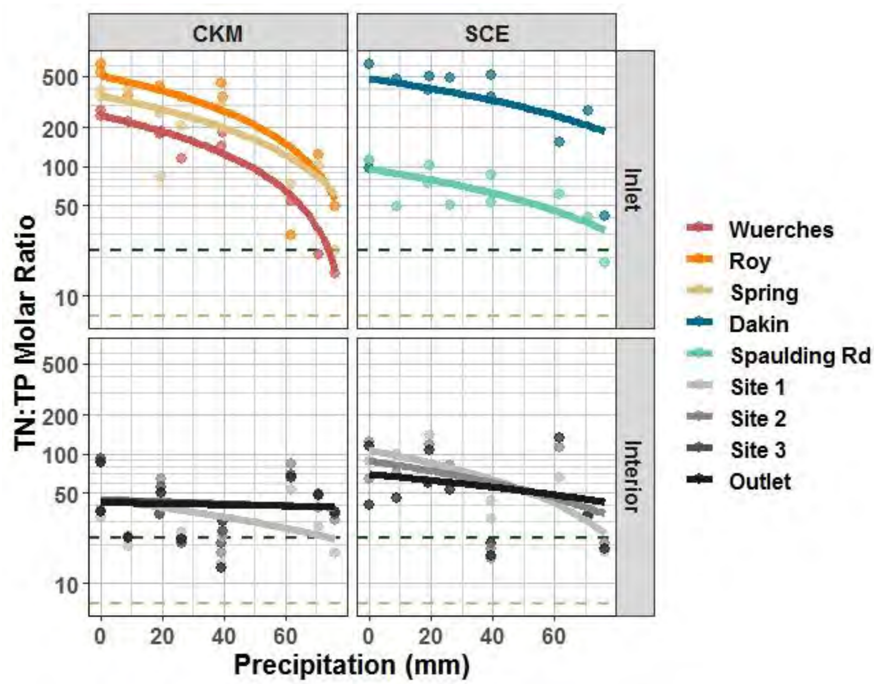

Figure 3-26: Best linear fit of TN to TP molar ratio for grab samples versus cumulative precipitation over the preceding 7 days. Results are separated by marsh and by inlet versus interior sites. Above the dark green dashed line (ratio of 22.6; Guildford & Hecky, 2000), P is likely to be limiting. Below the light green dashed line (ratio of 7; Abell, Özkundakci, & Hamilton, 2010), N is likely to be limiting.

*Figure 3-27: Average TN:TP molar ratio of interior sites within each marsh plotted for each sampling event in chronological order. * = sampling followed precipitation event. Above the dark green dashed line (ratio of 22.6; Guildford & Hecky, 2000), P is likely to be limiting. Below the light green dashed line (ratio of 7; Abell, Özkundakci, & Hamilton, 2010), N is likely to be limiting.*

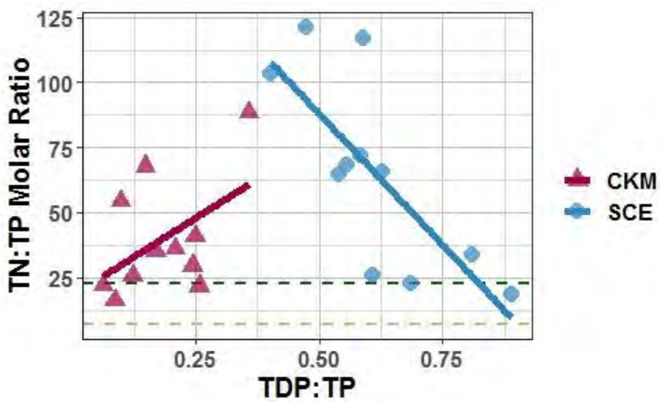

Figure 3-28: Best linear fit for average TN:TP molar ratio of interior sites within each marsh versus average TDP:TP ratio. Above the dark green dashed line (ratio of 22.6; Guildford & Hecky, 2000), P is likely to be limiting. Below the light green dashed line (ratio of 7; Abell, Özkundakci, & Hamilton, 2010), N is likely to be limiting.

Chlorophyll A

Within CKM, chlorophyll A (CHLA) ranged from 10.4 to 355 µg/L (median 153 µg/L), and from 0.7 to 24.6 µg/L (median 3.5 µg/L) in SCE. The highest concentrations of CHLA were measured in CKM in July 2017. Precipitation in September 2016 (71 mm) and June 2017 (76 mm) was associated with decreased CHLA concentrations in CKM (Figure 3-29 A). In CKM, CHLA concentrations were positively related to TP, TSS, and TN (Figure 3-29 B – D). In SCE, CHLA concentrations were relatively constant over the measured range of TP, TSS, and TN.

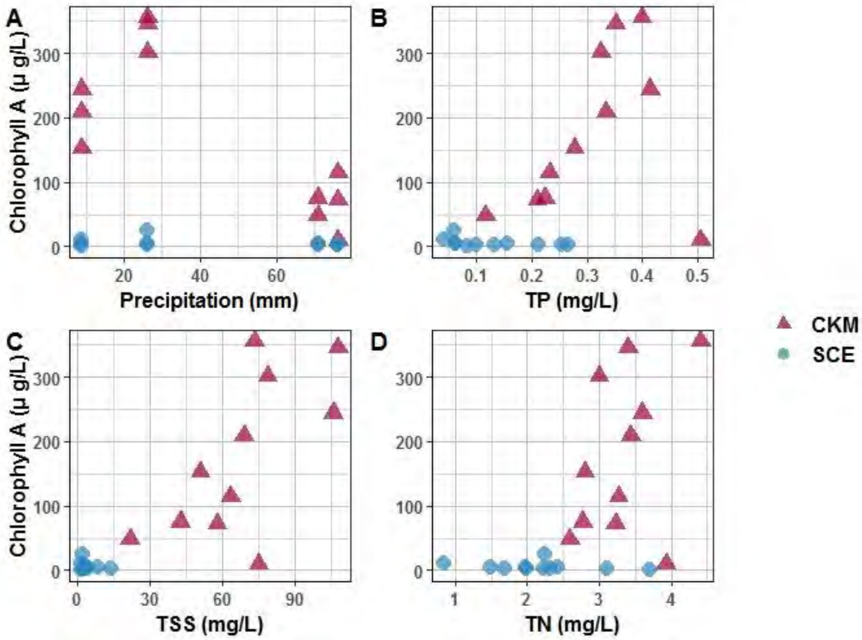

Figure 3-29: Chlorophyll A concentrations versus (A) cumulative precipitation over the preceding seven days, (B) TP concentration, (C) TSS concentration, and (D) TN concentration. Samples were collected at Sites 1 – 3 of CKM and SCE in September 2016 (but not at Site 2), and in June, July, and August 2017.

Estimated Daily Loads for In-Marsh Sampling Days

For each sampling event throughout the marshes, TP, TSS, and TN daily loads were estimated at each marsh inlet and outlet. The total estimated inlet loads were summed for each marsh and compared to the respective outlet loads to determine if the marsh retained nutrients and sediments on the sampling days.

Generally, the highest loads were observed following precipitation events. The TP, TSS, and TN loads at the SCE inlets were highest during June 2017 (Figure 3-30). Loads to SCE were generally higher than loads to CKM with the following exceptions: TP inlet loads to CKM (37.2 kg/day) exceeded SCE inlet loads (34.0 kg/day) during April 2017 (Figure 3-30 A), and TSS loads to CKM were higher in April (14.3 metric tonnes/day) and June (17.3 metric tonnes/day) 2017 compared to SCE inlet loads (April 4.2 metric tonnes/day; June 14.8 metric tonnes/day; Figure 3-30 B). For these April and June sampling events, CKM inlet loads of TP and TSS exceeded the

CKM outlet loads. The highest load of TN (1,547 kg/day) occurred at the outlet of SCE during the June 2017 sampling event (Figure 3-30 C).

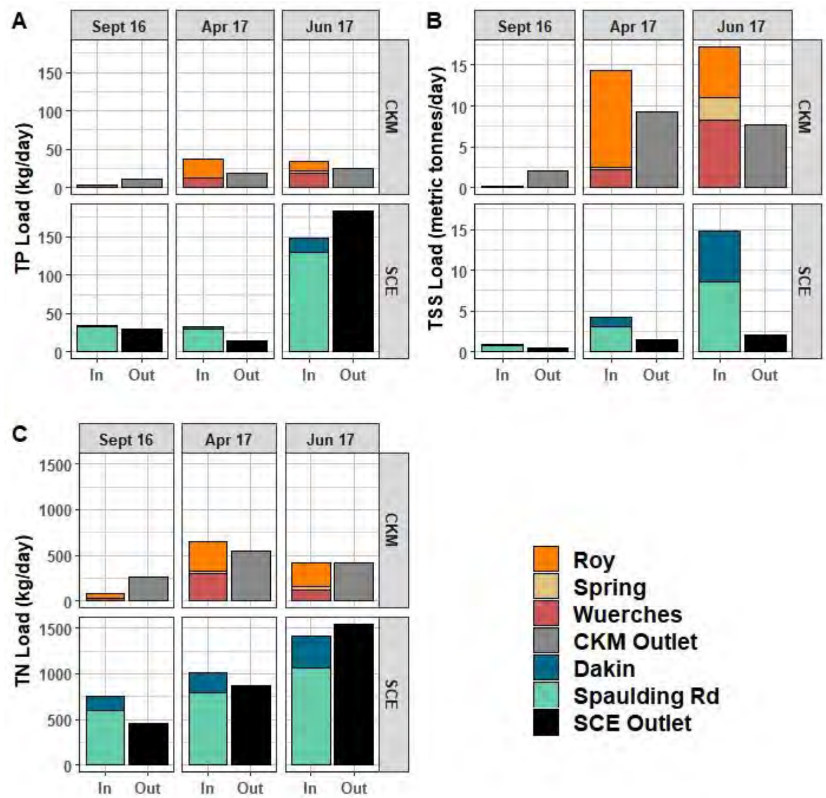

Figure 3-30: Daily (A) TP, (B) TSS, and (C) TN loads from sampling after storm events (greater than 60 mm of precipitation over the preceding 7 days) at the inlets and outlet of each marsh.

For the non-storm events, the total CKM inlet TP and TSS loads were consistently lower than outlet loads (Figure 3-31 A-B), whereas in SCE, the total inlet TP and TSS loads were consistently higher than outlet loads (except for TP in August 2016). The total TN loads at the inlets were higher than at the outlets in both marshes for most sampling events (Figure 3-31 C). Combining storm and non-storm events, the majority of TP (91%), TSS (69%) and TN (80%) loads to SCE entered through Spaulding Rd (Figure 3-32). The highest proportion of loads entered CKM through Roy Creek (47% of TP, 59% of TSS, and 54% of TN) and the smallest proportion entered through Spring Creek (7% of TP, 10% of TSS, and 7% of TN).

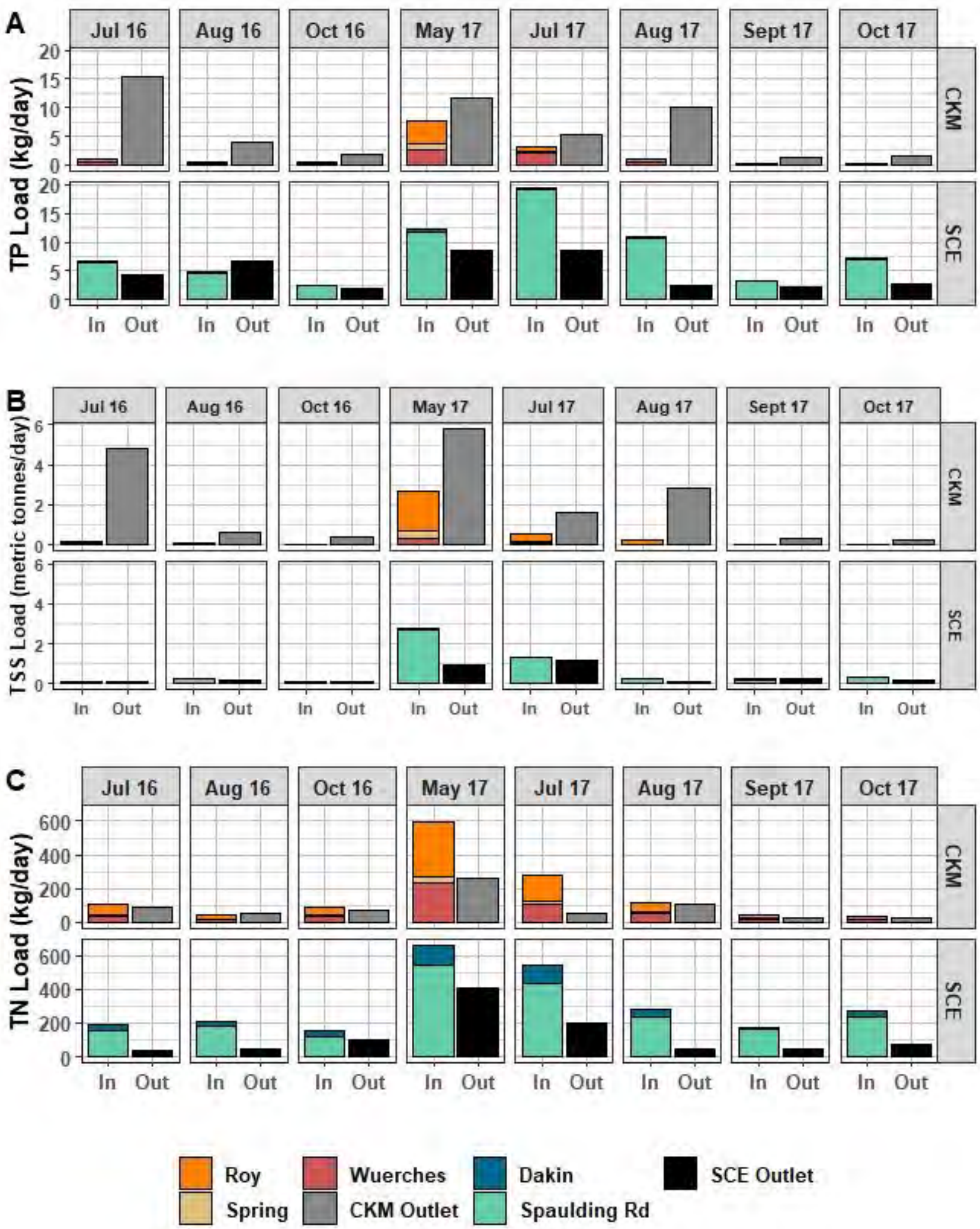

Figure 3-31: Daily (A) TP, (B) TSS, and (C) TN loads for sampling events that did not follow a storm event (less than 60 mm of precipitation over the preceding 7 days). Daily loads were calculated at the inlets and outlet of each marsh.

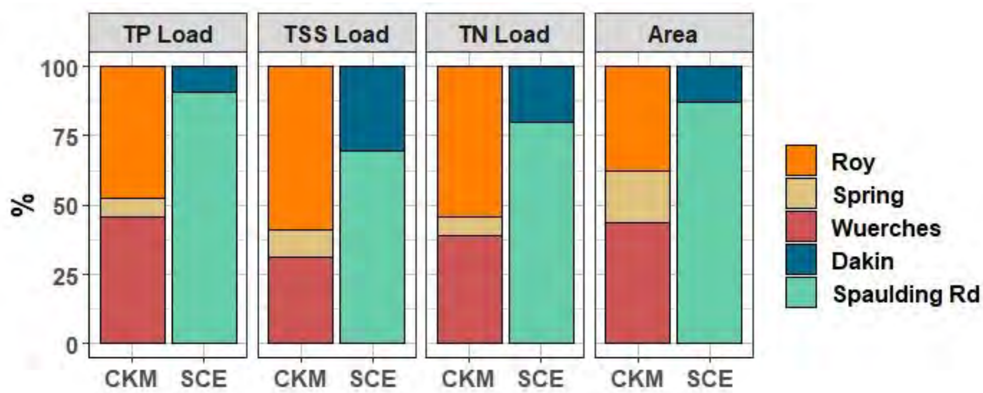

Figure 3-32: Percent contribution from each inlet for summed TP, TSS, and TN loads over the 11 sampling events at each marsh. Proportional areas for each inlet subwatershed are shown for each marsh in the last panel.

Over the 11 sampling events in CKM, there was export of TP (-17.45 kg), minimal export of TSS (-0.1 metric tonnes), and storage of TN (551. 95 kg; Table 3-9). However, SCE retained TP (15.2 kg), TSS (18.23 metric tonnes), and TN (1,822.43 kg; Table 3-9). Although SCE stored TP on the majority of sampling events, the highest TP export (-36 kg/day; Appendix F Figure F-1 A) followed the storm event in June 2017. However, the greatest amount of TP retained in SCE (17.5 kg/day) also followed a storm (April 2017). Within CKM, TP and TSS were retained only during the April and June 2017 storm events (Appendix F Figure F-1 A and B). Overall, TN was retained in both marshes, but each marsh exported TN during one storm sampling event (September 2016 for CKM and the June 2017 sampling for SCE; Appendix F Figure F-1 C).

Removal efficiencies (RE) for the 11sampling days were more variable for CKM compared to SCE. On occasion, for both TP and TSS, REs for CKM were below -1500% (Table 3-10) while REs for SCE did not reach below -40%. Negative removal efficiencies reflect a net loss from the marsh. Median TP and TSS REs for CKM were negative (-251% and -710%, respectively) whereas for SCE, both were positive (32% and 41%, respectively). SCE also had higher maximum

REs for both TP and TSS (78% and 86%, respectively) compared to CKM (51% and 55%, respectively). The maximum REs for TN were similar for both marshes (81% for CKM and 83% for SCE); however, the median values were higher for SCE (63%) compared to CKM (16%).

Table 3-9: Cumulative storage of TP, TN, and TSS within each marsh for the 11 sampling events.

Wetland	TP Storage (kg)	TN Storage (kg)	TSS Storage (metric tonnes)
CKM	-17.5	552.0	-0.1
SCE	15.2	1822.4	18.2

Table 3-10: Removal efficiencies (RE) for TP, TSS, and TN loads through CKM and SCE on sampling days. Minimum, median, and maximum REs shown for each parameter.

Parameter	Wetland	RE %		
		Minimum	Median	Maximum
TP	CKM	-2677	-710	55
	SCE	-6	41	86
TSS	CKM	-1599	-251	51
	SCE	-39	32	78
TN	CKM	-198	16	81
	SCE	-9	63	83

Sediment Analysis

Phosphorus in Cores and at Sediment/Water Interface

Sediment TP concentrations for samples collected on July 17, 2017 (a few days after the July 2017 inlet and outlet water quality sampling) were higher in SCE (average 1,146 TP mg/kg sediment for 0 – 2.5 cm and 2.5 – 5 cm samples) compared to CKM (average 851 TP mg/kg sediment for 0 – 2.5 cm and 2.5 – 5 cm samples). Concentration differences between the two core depths were greater for SCE (0 – 2.5 cm average 1,223 TP mg/kg sediment; 2.5 – 5 cm average 1,069 TP mg/kg sediment) compared to CKM sediments (0 – 2.5 cm average 854 TP mg/kg sediment; 2.5 – 5 cm had average 848 TP mg/kg sediment; Figure 3-33 A). Average DRP concentrations at the sediment-water interface were higher in SCE (0.023 mg/L) compared to CKM (0.002 mg/L; Figure 3-33 B). Sites 1 and 2 in SCE had the highest sediment TP concentrations and the highest DRP concentrations at the sediment/water interface.

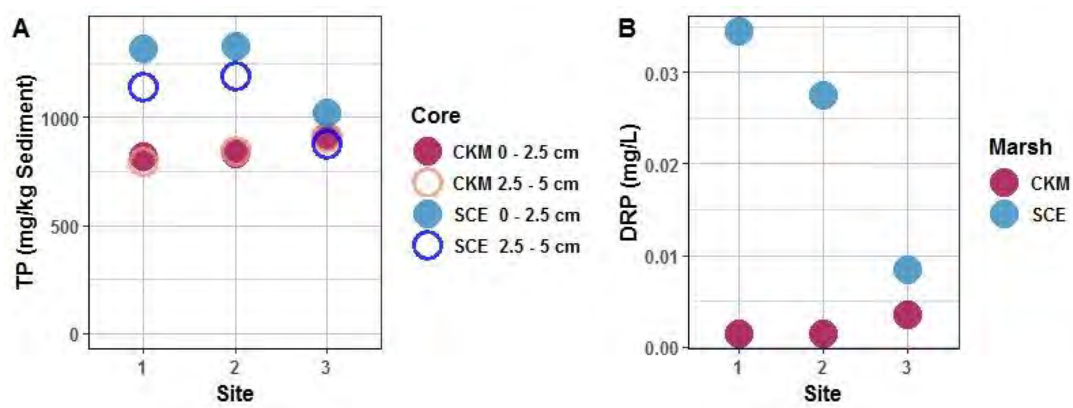

Figure 3-33: (A) Sediment TP in cores taken at Sites 1 – 3 in each marsh. Cores were separated at a depth interval of 2.5 cm. (B) DRP concentrations at the sediment/water interface at Sites 1 – 3 in bother marshes. All cores and samples collected July 17, 2017.

Texture Analysis

Marsh sediment texture analysis quantified the amount of clay, an important factor impacting P adsorption of sediment. Sediment texture at Sites 1—3 in both marshes for combined 0 – 5 cm depth cores was classified as silt loam except at Site 3 in CKM, which was loam (Table 3-12). For the silt loams, clay content ranged from 10 – 16%, silt range from 52 – 64%, and sand ranged from 20 – 34%. The loam sample at Site 3 in CKM had higher clay (21%) and lower silt (46%) content compared to the other locations.

Table 3-11: USDA soil texture analysis results of sediment collected at Sites 1 – 3 in both marshes July 17, 2017.

Wetland	Site	% Clay	% Silt	% Sand	Classification
CKM	1	16	64	20	Silt Loam
	2	15	52	33	Silt Loam
	3	21	46	33	Loam
SCE	1	10	61	29	Silt Loam
	2	10	58	31	Silt Loam
	3	11	55	34	Silt Loam

Dilute Salt Extraction

The potential P release from sediment was examined in the laboratory. The median dilute salt-extractable P was 0.512 mg P /kg sediment for CKM sediments, which was higher than the median of SCE sediments (0.375 mg P /kg sediment). The difference in extractable P between the top (0 – 2.5 cm) and bottom (2.5 – 5 cm) layers of sediment was greater in SCE (median for top was 0.076 mg P /kg sediment greater than for bottom) compared to CKM (median for top was 0.051 mg P /kg sediment less than for bottom; Figure 3-34). The bottom sediments of SCE had the lowest median dilute salt extractable P (0.35 mg P/kg sediment) compared to the top layer and to CKM (Table 3-12).

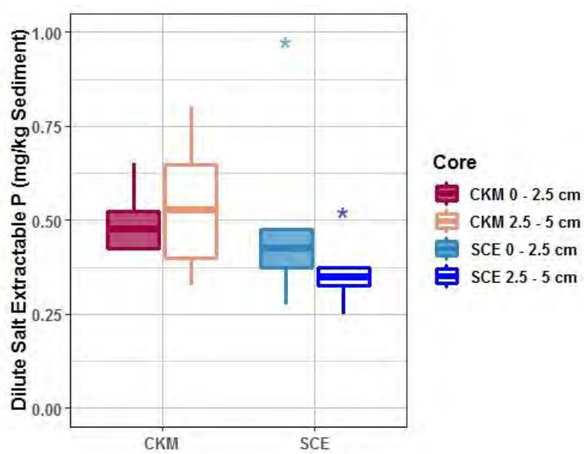

Figure 3-34: Dilute salt extractable P from sediment collected at Sites 1 – 3 in both marshes on July 17, 2017. Top layer of sediment was a depth of 0 – 2.5 cm. The bottom layer was 2.5 – 5 cm depth. Each box represents 9 observations. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

Table 3-12: Median dilute extractable P from sediments in the top and bottom layer of cores collected at Sites 1 – 3 in both marshes July 17, 2017.

Wetland	Core	Median Dilute Salt Extractable P (mg/kg sediment)
CKM	Top (0 - 2.5 cm)	0.475
	Bottom (2.5 - 5 cm)	0.525
SCE	Top (0 - 2.5 cm)	0.425
	Bottom (2.5 - 5 cm)	0.350

Equilibrium Phosphorus Concentration

The equilibrium point at which sediments were likely to adsorb or release P was determined for sediment from each marsh. The estimated EPC_0 for CKM was 0.0126 mg P/L (Figure 3-35 A), and SCE sediment had an estimated EPC_0 of 0.0306 mg P/L (Figure 3-35 B).

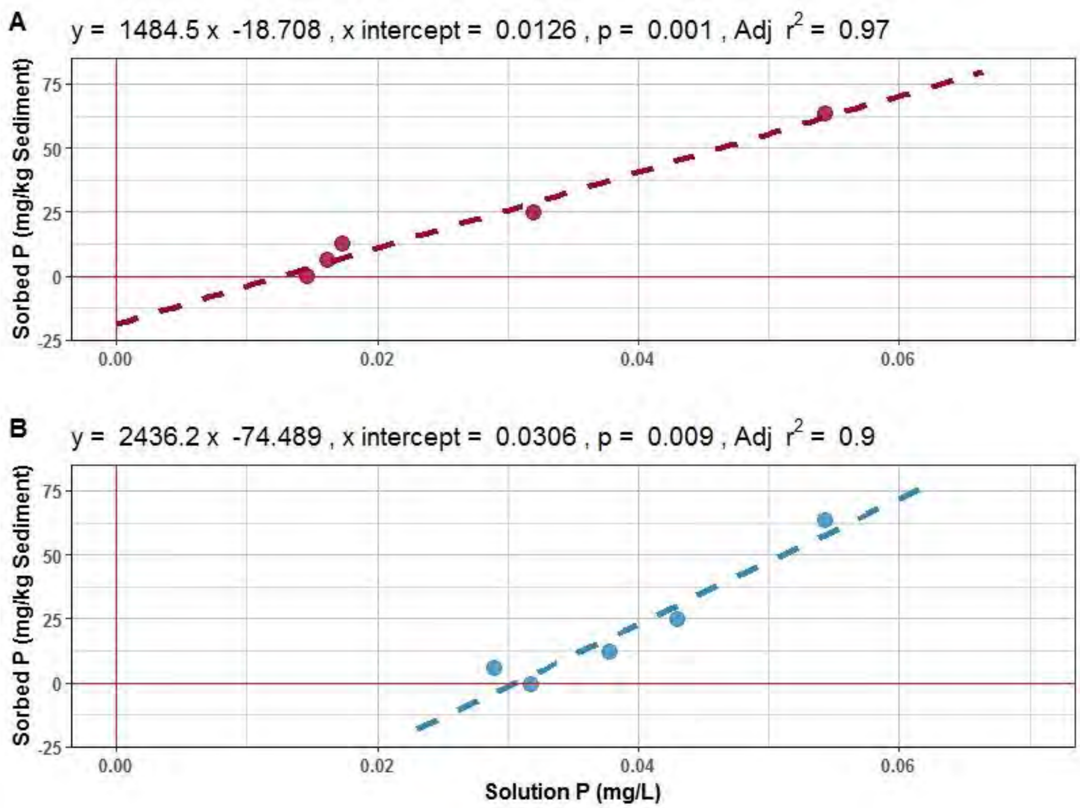

Figure 3-35: P removed from solution to sediment (sorbed P) versus P concentration in solution for sediment at Site 2 in (A) CKM and (B) SCE. EPCo is defined as zero P sorbing to sediment, where the best fit linear regression crosses the x axis (y = 0 P mg/kg sediment). Points are averages of replicates (either 3 or 6) with the same initial P concentration.

Discussion

Objective 1: Quantify differences in total phosphorus (TP) and suspended sediment (SS) loads from the two marshes since 2012

Greater flow through SCE and shorter marsh retention times typically led to more TP and SS entering Green Lake through SCE than CKM. The SCE outlet stream flow was two to three times greater than the CKM outlet stream flow in all seasons, and, at both sites, the highest seasonal median stream flow occurred during the spring. Retention times were three to five times longer in CKM than in SCE, and, in both marshes, the shortest retention times were associated with the high spring flows. While stream flows from SCE were consistently and significantly greater than CKM, median fall TP loads was less at SCE compared to CKM, and this was true for SS loads in summer. Median flow-weighted TP and SS concentrations, therefore, followed different patterns than flow.

At the CKM outlet, the only significant seasonal difference in flow-weighted mean TP concentration was between summer (higher) and winter (lower). At the SCE outlet, summer flow-weighted mean TP concentrations were significantly higher than both winter and fall. At both marsh outlets, median flow-weighted mean TP concentrations were highest in summer. High summer TP concentrations in SCE returned to a lower nutrient state earlier (in fall), whereas CKM's decrease was delayed until winter. Median flow-weighted mean TP concentrations in each season were up to two times greater for CKM compared to SCE; these differences were significant in all seasons except winter. During winter, there was no significant difference in flow-weighted mean TP concentration between the two marshes. Conditions diverged during the growing season (TP concentrations in CKM increased more than in SCE) and the return to lower TP concentrations occurred during different seasons within each marsh.

Higher flow-weighted fall TP concentrations in CKM compared to SCE may be linked to differences in the rate of P cycling through a phytoplankton versus a macrophyte-dominated system. The TP concentrations from the macrophyte-dominant SCE did not peak following autumn senescence; low SCE fall TP concentrations were similar to winter concentrations. Macrophyte decay has led to fall TP pulses in other studies (Landers, 1982) but, at the monthly-scale examined here, it was not observed. Instead, fall TP concentrations at the SCE outlet decreased compared to summer and may be a result of macrophyte TP uptake and storage for use over the winter and the following spring. Differences in TP concentrations between CKM and SCE in the fall may also be linked to greater P availability in phytoplankton versus macrophyte mass. Proportional release of bioavailable P from phytoplankton debris is greater than the bioavailable proportion released from macrophyte debris (Feng *et al.*, 2018). This could lead to faster recycling of P in the phytoplankton system, continued cycling into the fall, and sustained summer-like P concentrations. In the macrophyte-dominated system, the combination of P translocated to macrophyte roots for overwintering and the low proportion of bioavailable P released through decomposition may have led to the lower fall TP concentrations than the phytoplankton-dominated system. TP concentrations could have also been impacted by processes influencing sediment resuspension (e.g. wind).

Similar to flow-weighted TP concentrations, flow-weighted SS concentrations were comparable at both marsh outlets in the winter but became higher at CKM compared to SCE during the growing season. The high summer SS concentrations for CKM could be linked to both resuspended sediment and phytoplankton biomass. Despite the smaller stream flows into CKM than SCE, median summer SS loads at the outlet from CKM was twice that at SCE, with CKM median summer flow-weighted mean SS concentration five-fold greater than SCE. Median SS concentration in clear water SCE was highest in the spring and decreased by 37% in the summer. Conversely, turbid CKM had the highest median SS concentration in the summer, which was an 18% increase compared to the spring. CKM seasonal flow-weighted SS concentrations were significantly higher than SCE in spring and summer. At the SCE outlet, flow-weighted SS concentrations were only significantly higher in spring compared to both winter and summer. At the CKM outlet, flow-weighted SS concentrations were significantly higher in spring and summer than in winter and fall. High summer SS concentrations were therefore unique to the wind and phytoplankton-dominated system. The SS concentration includes any substance floating in the water column that contributes to turbidity (e.g. phytoplankton and sediment). The median CHLA (a proxy for phytoplankton biomass) concentration (collected in 2016 and 2017) was 43 times higher in CKM compared to SCE which points to phytoplankton as a major contributor to high summer SS concentrations in CKM. Since CHLA was not measured over the five-year period, future monitoring could help explore relationships between SS and CHLA within CKM. Restoration of CKM could include efforts to reduce wind resuspension of sediment. Further study of the phytoplankton community could also quantify population densities and link CHLA and phytoplankton fluctuations. Variation in SS unaccompanied by similar variation in phytoplankton may illustrate reduction in wind resuspension of sediment.

Resuspension of sediment can be driven through physical disturbance, such as wind action and carp bioturbation. Carp, which densely occupied CKM prior to 2015 exclusion efforts, resuspend sediment and increase turbidity and likely contributed to the elevated SS and nutrient concentrations in the spring and summer. Following exclusion efforts, summer SS concentrations in CKM were not consistently lower than those observed in 2013 and 2014. Furthermore, flow-weighted mean SS concentrations increased from 2012 to 2017 and were linked to increases in stream flow. Continued high summer SS concentrations therefore demonstrate sustained resuspension of sediment within the marsh in addition to dense phytoplankton. Macrophytes reduce summer SS concentrations by promoting sedimentation and reducing wind resuspension of sediment (Barko *et al.*, 1991). Since CKM is not macrophyte-dominated, factors contributing to sediment stabilization are less available to reduce summer SS concentrations. Even though carp disturbance has decreased in CKM following the installation of the barrier at the marsh outlet, without mechanisms for sediment stabilization, wind can continue to resuspend sediment. Decreases in SS concentrations will likely be delayed until sediment disturbance by both carp and wind are reduced.

Carp within CKM may contribute to the high summer TP concentrations; however, since exclusions efforts began, there has not been consistent evidence of reductions in summer TP concentrations in CKM. Prevalent carp are associated with internal TP loading (Weber and Brown, 2009), and the singular act of removing benthivorous fish can decrease TP concentrations in the water column (e.g. Søndergaard *et al.*, 1990). In June and July 2016, following the installation of the carp barrier, the CKM outlet had its lowest flow-weighted TP concentrations of the monitoring period (0.073 mg/L and 0.118 mg/L, respectively). While this showed promise of less internal loading and lower summer TP concentrations, the average flow-weighted TP concentration in summer 2017 was 0.27 mg/L (provisional data). Increases in precipitation from 2016 to 2017 likely contributed to the increase in TP from 2016 to 2017. Average summer precipitation in 2017 was almost 30% higher (4.5 mm/day) compared to 2016 (3.5 mm/day). Even with the reduction in carp, interannual variability can contribute to increases in P loading from the marsh to Green Lake.

Seasonal differences in TP and SS concentrations are indicative of different equilibrium states between the two marshes. A notable difference between the two marshes was that while flow-weighted TP concentrations increased in the summer at both outlets, flow-weighted summer SS concentrations increased at CKM but not at SCE. There was a close positive relationship between TP and SS in turbid CKM but not in clear SCE, suggesting that periods with high TP concentrations were comprised of more dissolved rather than particulate P in SCE. In SCE, increasing stream flow had a much smaller effect on SS concentrations than what was observed in CKM. Macrophytes stabilize sediments with their roots and increase water clarity, enabling benthic algae to hold sediments in place (Zhang *et al.*, 2017). Low summer SS and CHLA concentrations in SCE suggest that there is little sediment disturbance and small populations of phytoplankton. If CKM were to become macrophyte-dominated, a reduction in summer SS concentrations would be expected especially since CKM retention time is longer than in SCE.

While TP and SS flow-weighted mean concentrations at the two marsh outlets were similar in winter but different during the growing season, export data alone cannot fully explain how inputs to the marshes or within marsh functions may differ. Quantification of inlet and outlet concentrations and loads were therefore instrumental in better understanding of how internal processes differ between the two marshes.

Objective 2: Quantify and compare spatial and temporal changes in P, TSS, and TN within the two marshes during the growing seasons of 2016 and 2017

Both marsh systems had high TP concentrations at some inlets. Measured TP concentrations aligned with the impairments listed by WDNR: Roy Creek, Wuerches Creek, and Silver Creek at Spaulding Rd all had median TP concentrations above the stream water quality standard (0.075 mg/L). Dakin Creek only exceeded the criterion in samples following storm events and in September 2017. Spring Creek, which had the lowest overall median TP concentration, only exceeded the standard following the largest storm event. The Spring Creek watershed has the most forested and least agricultural area and had only one occasion of high TP concentration. This was in contrast to the multiple observations of high TP concentrations at other tributaries whose watersheds are more agricultural. Observations align with forested watersheds typically exporting less TP than agricultural watersheds (Robertson *et al.*, 2008; Powers *et al.*, 2014). Tributary inputs of nutrients and sediment into the marshes for the 11 sampling events were disproportionate to subwatershed area. In CKM, Roy Creek was the dominant source of TP, TSS, and TN. Conversely, Spring Creek contributed proportionally less TP, TSS, and TN to CKM than would be expected considering its land area. For SCE, Dakin Creek was the source of more TSS and TN than would be expected based on land area.

Storm events impacted the TDP:TP ratio at tributary inlets, with more lower ratios during storm events. The TDP:TP medians for all tributaries exceeded 0.5. For Wuerches Creek and Spaulding Rd, the median TDP concentrations were above the water quality standard established for TP. TDP:TP at inlets decreased for samples taken following storm events, except at Spaulding Rd when the largest TDP:TP ratio was observed after the largest storm event. In general, storm events led to more P entering the marshes and a higher proportion of that P was particulate compared to baseflow conditions. In the smaller tributary watersheds (excluding Spaulding Rd's), baseflow conditions, with higher proportions of TDP, transitioned to more particulate P following storm events. This suggests that instream processes producing TDP were dominated by particulate P in runoff following storm events (Verheyen *et al.*, 2015). Accumulated sediments in these tributaries could contribute to TDP release during baseflow conditions.

The relationship between TP and TSS at Spaulding Rd was weaker than that observed at the other smaller tributary inlets (Dakin, Spring, Roy, and Wuerches Creeks). This was especially the case following storm events when TSS at Spaulding Rd did not vary as much as at the other inlets. The relatively low concentrations of TSS entering SCE through Spaulding Rd could be associated with its large water volume and widened stream channel, reducing its sediment delivery potential from the watershed. This geomorphology enables the creek to slow down and deposit sediment particles upstream of Spaulding Rd. Also unique to the watershed, is the upstream RWTP which contributes most of its P to Silver Creek as TDP. These factors combine so that even following the two largest storm events, the TDP:TP ratio at Spaulding Rd remained above 0.75. Since Spaulding Rd has the largest flow of the inlets and constitutes the majority of flow through SCE, TSS concentrations within the marsh responded to precipitation similarly. The presence of predominantly dissolved P at Spaulding Rd and within SCE therefore explains the weak relationship observed between TP and TSS.

The CKM interior sites had larger median TP concentrations than the inlets, suggesting internal loading within the marsh is important. The interior sites in CKM also had higher median TP concentrations than SCE. There was evidence of internal loading being less important within SCE; median TP concentrations at Site 2 and Site 3/Outlet were less than the median TP concentrations at the inlets. The median TP concentration at the CKM outlet was higher than at the SCE outlet (difference of 0.078 mg/L), but the median TDP concentration at the SCE outlet was slightly higher than at the CKM outlet (difference of 0.008 mg/L). Much more of the P within SCE was dissolved compared to in CKM. For CKM, TDP entering through the tributaries was mostly transformed into particulate P, via phytoplankton uptake, by the time it left the marsh.

The relationships among TP, TDP, TSS, and CHLA align with those expected given the dominant biological organisms within each marsh. In CKM, P is primarily contained in suspended phytoplankton, a component measured through TSS, CHLA, and particulate P concentrations. While CHLA concentrations represent trends in phytoplankton, a detailed examination of phytoplankton species and density would further define the relationship between CHLA and TSS. Especially in eutrophic systems, variable CHLA within different phytoplankton species hinders direct quantification of CHLA as a portion of TSS (Kasprzak *et al.*, 2008). In macrophyte-dominant SCE, P is retained in the larger organisms (*i.e.* aquatic plants), not captured as TSS, CHLA or particulate P suspended in the water column, resulting in the majority of suspended P being in the dissolved phase. The prevalence of phytoplankton in CKM resulted in higher particulate P, TSS and CHLA concentrations compared to SCE. Within SCE, the range in CHLA was much smaller than in CKM and increases in CHLA were not dependent on TSS or TP, even though TP concentration within SCE did vary throughout sampling.

The TDP entering CKM was assimilated into particulate P, and internal loading contributed to higher TP concentrations at interior sites compared to the inlets. Release of P from benthic sediments into the water column can be through physical processes, such as resuspension (solid particles) and diffusion (aqueous species; Orihel *et al.*, 2017; Robertson et al., 2018). The relatively low TDP concentrations in CKM suggests that bioavailable dissolved P diffusing from CKM sediments was quickly incorporated into phytoplankton mass. Increases in sediment organic phosphate-mineralizing bacteria and CHLA concentrations have been documented even as water column soluble reactive phosphorus (a subset of TDP) did not increase (Song *et al.*, 2009). The rapid cycling of P from the sediments into phytoplankton mass, therefore, results in mostly particulate P being exported from CKM into Green Lake. While P in phytoplankton is less accessible than TDP, organic P in phytoplankton debris (after senescence) is more bioavailable than what is in macrophyte debris (Feng *et al.*, 2018). This availability facilitates continued P cycling and sustained phytoplankton blooms in Green Lake.

In SCE, the difference in the dissolved P between the inlet and interior sites was not as distinct as for CKM, where the median TDP:TP ratio only decreased from 0.76 at the inlets to 0.6 at the interior sites. For CKM, the median ratio decreased from 0.63 to 0.14. Therefore, while some dissolved P was transformed within SCE, the majority of TP entering Green Lake from Silver Creek remained in the dissolved form. As previously noted, the median TDP concentrations at the SCE and CKM outlets were similar (difference of 0.008 mg/L), but the range of TDP concentration at the SCE outlet (maximum 0.247 mg/L) was much greater than at the CKM outlet (maximum 0.037 mg/L). This suggests continuous TDP uptake within CKM throughout the sampling period and evidence of P limitation for phytoplankton growth (Kolzau *et al.*, 2014). Unconsumed TDP in SCE exhibited that, typically, TDP passed through the marsh and was not utilized for biological growth within the marsh.

The TN concentrations at both inlet and interior sites of both marshes were elevated in comparison to the proposed reference level for other Wisconsin streams. For wadeable streams across Wisconsin, median concentrations May through October were highest in June at 2.11 mg/L (Robertson *et al.*, 2006). For Wisconsin nonwadeable rivers, median TN concentrations May through October were approximately 1.3 mg/L (Robertson *et al.*, 2008). A concentration of 2.02 mg/L TN was suggested as a potential reference level for the Southeast Wisconsin Till Plains Ecoregion (Wang *et al.*, 2007). Compared to other areas of Wisconsin, this ecoregion has more prevalent agriculture and has higher concentrations of TN and other nutrients. These high concentrations are associated with tolerant fish species, low indices of biotic integrity, and poor measures of macroinvertebrate diversity. In this study, the highest TN concentrations were observed in watersheds with the most agricultural land use. Roy and Dakin Creeks had the highest median TN concentrations (13.3 and 11.6 mg/L, respectively) of all the sampled sites. The Dakin Creek subwatershed is the most agricultural (85%), and Roy Creek's subwatershed is the second (75%). Corn is the most prevalent crop in both subwatersheds. Within the two marshes, CKM had a higher median TN concentration than SCE (3.37 and 2.32 mg/L, respectively). Only the SCE outlet had a median TN concentration below the recommended reference concentration, suggesting that elevated TN levels at the other sampling sites could be contributing to degraded habitat conditions for aquatic organisms.

At the inlets, the highest TN concentrations were generally associated with baseflow conditions (little precipitation and dominant groundwater discharge). Following storm events, these concentrations were diluted by the addition of surface runoff. Wisconsin has a nitrate drinking water standard of 10 mg/L because of health concerns, especially for infants. Since TN (not nitrate) concentrations were measured for this study, the median levels exceeding 10 mg/L at Dakin and Roy Creeks may or may not indicate contamination in local groundwater. However, since the highest concentrations were observed when groundwater discharge was likely prominent, it would be useful to quantify the particular nitrogen species comprising TN at these inlets. Since residents of these watersheds obtain water from private wells, regular nitrate testing of residential wells would determine whether drinking water precautions are needed. Additional testing of nitrates under stream baseflow conditions could also document the quality of shallow groundwater.

The TN:TP ratios at the inlets and within the marshes generally stayed within a range suggesting P should limit productivity. The TN:TP ratios at the inlets to the marshes varied with the amount of precipitation, whereas TN:TP ratios within the marshes varied more by season, than with flow. Seasonal changes in conditions that limit growth have been observed elsewhere in shallow lakes, with P being the limiting nutrient in spring followed by a shift to N or light limiting growth later in the summer (Kolzau *et al.*, 2014). Seasonal shifts in the growth-limiting nutrient are associated with temperature dependent rates of denitrification and P release from sediments. Increases in temperature intensify denitrification (reducing the amount of N available to organisms) and spur bacterial activity that releases P from sediments. This combination of N reduction and P proliferation can shift growth within the water column from P to N limitation. High TN concentrations in Dakin and Roy Creeks resulted in the highest TN:TP ratios of all inlets. At the inlets, TN:TP ratios generally decreased following precipitation. After the largest storm, TN:TP ratios at Spaulding Rd and Wuerches Creek decreased below the threshold for growth limitation by P (22.6; Guildford and Hecky, 2000). The TN:TP ratios at Spaulding Rd were lower than at the other inlets and similar to interior SCE sites. During 7 of the 11 sampling events, interior SCE sites had higher average TN:TP ratios compared to interior sites of CKM. During both summers, ratios in CKM dropped below 22.6, suggesting growth within the marsh may not have been only P limited. The SCE only reached these conditions in the summer of 2016 and following the storm event in June 2017.

No TN:TP molar ratios were measured below 7 (an estimate for N limitation; Abell *et al.*, 2010), and consumption of TDP within the marshes suggest P (and not N) is the limiting nutrient. Within CKM, lower TDP:TP ratios were associated with lower TN:TP ratios, which indicates continued consumption of TDP under conditions trending towards N limitation. Within SCE, however, the two sampling events (September and June storms) with the highest TDP:TP ratios were associated with relatively low TN:TP ratios. On these occasions, average TDP concentrations within SCE exceeded 0.1 mg/L. Within CKM, average TDP concentrations did not exceed 0.1 mg/L even though concentrations in Wuerches and Roy creeks were of that magnitude. During the September and June storm events, TDP:TP showed little decrease as water flowed through the marsh from Spaulding Rd to sites downstream, suggesting that TDP was not rapidly consumed within SCE when flows increased and marsh retention times decreased. Even though SCE TDP concentrations were high during these storm events, TN:TP ratios did not decrease to those associated with N limitation. During these storm events in SCE, TDP may have exceeded biological demand, but it appears that P would still likely be the limiting nutrient (of N and P) for both wetlands the majority of time. When the lowest TN:TP ratios occurred in summer, the interior marsh sites only reached the interval within which either N or P could become potentially limiting. For the inlets, reaching this interval condition only followed precipitation events. Therefore, implementing P reduction efforts upstream remains an effective practice for limiting biological growth within both wetlands and reducing P reaching Green Lake.

Objective 3: Evaluate whether each marsh acts as a sink or source of TN, TP, and TSS

Marsh TP retention following storm events (defined as > 60 mm over the preceding 7 days; 3 out of 11 sampling days) and non-storm events exhibited different patterns for CKM and SCE. For the 11 days in 2016 and 2017, CKM retained TP (acted as a sink) on two occasions, both of which followed storms. For the other sampling events, CKM was a source of TP to Green Lake (Table 3-13). In contrast, SCE acted as a sink of TP for seven of the eight non-storm events and two of the three storm events. The storm event for which SCE exported TP, however, was the largest net TP export for both wetlands over any of the sampling days (36 kg/day). Overall, CKM exported 17.5 kg of TP and SCE retained 15.2 kg of TP.

Table 3-13: TP, TSS, and TN storage for each marsh during non-storm, storm, and all 11 sampling events. Positive storage indicates the marsh retained that mass and negative storage signifies export of that mass.

Wetland	Event	TP Storage (kg)	TSS Storage (metric tonnes)	TN Storage (kg)
CKM	Non-Storm	-36.9	-12.9	621.2
	Storm	19.5	12.8	-69.2
	All	-17.5	-0.1	552.0
SCE	Non-Storm	30.1	2.3	1521.3
	Storm	-14.9	15.9	301.1
	All	15.2	18.2	1822.4

Similar to TP, CKM and SCE exhibited opposite trends in TSS retention for non-storm events. For all non-storm events, CKM was a source of TSS (Table 3-13). Conversely, SCE was a sink during seven of the eight events. For the non-storm event when SCE was a TSS source, net export was minimal (0.01 metric tonnes/day). The largest amount of TSS retention for both marshes occurred during storm events. For CKM, the cumulative retention of the April and June 2017 storm events equaled the approximate amount exported during non-storm events (resulting in a net export of 0.1 metric tonnes for the 11 sampling events).

The TSS retained during the days following storm events was not associated with phytoplankton, whereas the TSS exported following the non-storm events was probably associated with phytoplankton. Within CKM, CHLA concentrations (phytoplankton prevalence) decreased more rapidly than TSS concentrations during storm events. CHLA samples, were four times larger during the non-storm sampling (average 268 µg/L) than during the storm sampling (average 64 µg/L). The TSS, however, was only 1.5 times larger during the non-storm sampling (average 81 mg/L) compared to storm sampling (average 49 mg/L) events. Even though the TSS export for CKM balanced to almost zero over the 11 sampling events, the solids exported during the non-storm events differed from that retained during the storm events. Within SCE, CHLA concentrations similarly decreased during the storm events while TSS concentrations increased. CHLA concentrations during non-storm events (average 8 µg/L) were higher compared to storm events (average 3 µg/L). The TSS concentrations during non-storm events (average 4 mg/L) were slightly lower compared to storm events (average 5 mg/L), suggesting the higher storm TSS was also not associated with CHLA. Non-CHLA particles, such as sediment, were retained during the storm events whereas during the non-storm events, TSS associated with CHLA (phytoplankton) were exported to Green Lake.

For non-storm days both marshes generally retained TN. SCE acted as a TN sink for all non-storm sampling days. CKM was a TN sink on all but one non-storm event, August 2016 when CKM exported 8.6 kg/day. The 2017 August sampling event, however, showed retention of 13.2

kg/day. Each marsh had one storm event during which it exported TN. The September 2016 storm event had the largest export of TN of all the events, with CKM exporting 174 kg. However, in total, both marshes retained more TN than was exported. Over the 11 events, SCE was a sink for more than three times the TN (1822 kg) compared to CKM (552 kg; Table 3-13).

The TP, TSS and TN loads through both marshes were greater following storm events than during non-storm conditions. Some of the largest quantities of sediment and nutrient storage or export were observed during storm sampling compared to the other sampling occasions. For CKM, export of TP during the non-storm events exceeded the storage estimated for the two storm sampling events. Thus, for this analysis, internal P loading within CKM under baseflow conditions was not compensated by P retained during high flow events. For the times considered in this study, CKM appeared to contribute additional P to the CKM load entering Green Lake. For SCE, individual storm-events either retained or exported TP, but retention for the other non-storm events exceeded what was exported during storms. Overall, SCE acted as a sink for TP and showed only one instance of potential internal P loading under baseflow conditions. The April and June 2017 storm events had the highest TP, TN, and TSS inlet loads of the 11 sampling events (at SCE, TP for the September 2016 storm event was also high). For these events, both marshes retained sediment and reduced TSS loads into Green Lake. High TN export (> 10 kg) only occurred during storm events. However, not all storms led to TN export from the marshes. For SCE, the marsh retained both TN and TP in the April storm event but neither did in the June event. Lentic bodies of water have been shown to reduce export and variability of TP, TN and TSS yields for watersheds (Powers *et al.*, 2014). While TP, TN and TSS were not consistently retained, longer periods of monitoring at all marsh inlets and outlets could demonstrate otherwise. Previous analysis of loading at the primary inlet and outlet of SCE demonstrated that the marsh served as an overall sink of both TP and TSS on an annual basis (Chapter 2). CKM, however, does not have a comparable dataset.

In this study, retention of sediment and nutrients during storm events was only estimated through one grab sample at each inlet and outlet and an average daily flow. Multiple nutrient and sediment samples and flow estimates at more frequent points along the storm hydrograph would better capture the full range of loads into and out of each marsh. More data is needed for loads from SCE's Dakin Creek and for all CKM inlets. More detailed analysis of continuous TP, TN and SS loads at inlets and outlets over the growing season would illustrate the balance between retention/export following storms versus retention/export the remainder of the time within each marsh.

While there was evidence of both marshes retaining TP, TSS, TN during highflow conditions, SCE continued to retain all three and CKM only retained TN during non-storm events. Despite inlet concentrations being similar for both marshes, the internal processes determined whether the marsh retained or exported under baseflow conditions. Therefore, reducing the internal loading of P and TSS within CKM would reduce the contribution to Green Lake. A better understanding of within marsh processes, specifically sediment influences, are therefore valuable in attempts to shift CKM export of TP and TSS to retention.

Objective 4: Quantify P variation within and the potential for P release from sediments within the two marshes.

CKM, which cumulatively acted as a source of TP to Green Lake over the 11 sampling days (as well as during each individual non-storm event), showed evidence of internal loading under baseflow conditions. On the other hand, SCE acted as a sink for TP during most of the individual non-storm sampling events and cumulatively over the 11 sampling events, suggesting internal loading was less important for that marsh. Internal loading increases during the summer months because rising temperatures lead to more active microbial activity and P release from the sediments (Liu *et al.*, 2018). Wind resuspension and carp activity could also be factors contributing to internal P loading during this time. Therefore, the July 2017 sediment cores helped to understand differences between the two marshes during a warm period of expected internal loading. The TP balance estimated for each marsh using data form the July 2017 water quality sampling indicated that SCE retained TP and CKM released TP. Sediment samples were collected shortly after July 2017 water quality sampling event.

Dissolved P concentrations were generally higher in SCE compared to CKM which initially suggests there could be more P release from SCE sediments compared to CKM. DRP at the sediment/water interface (collected just prior to sediment cores) was an order of magnitude higher in SCE (average 0.024 mg/L) compared to CKM (average 0.002 mg/L). While DRP was not measured for the surface samples collected on the 11 occasions, TDP was analyzed, and it was also consistently higher in SCE (median 0.043 mg/L) compared to CKM (median 0.029 mg/L). Since P release from sediments includes processes of desorption and dissolution, internal P loading is expected to increase dissolved P (Orihel *et al.*, 2017). Higher DRP in the SCE water column compared to that in CKM could suggest more internal loading from SCE. For the July grab sampling prior to sediment collection, however, TP was higher in CKM compared to SCE, and TDP:TP was much higher in SCE (average 0.56) compared to CKM (0.06). CKM at that time also had much higher CHLA (average 334 μg/L), a measure of phytoplankton, compared to SCE (average 11 μg/L). Therefore, during July sampling, in CKM dissolved P was low and high levels of TP were associated with phytoplankton. Since CKM was estimated to have exported P during the July sampling event, this suggests P released from sediments was quickly assimilated into phytoplankton, resulting in low DRP levels within the marsh (Song *et al.*, 2009). Therefore, water column DRP measurements alone are not a suitable indicator of P release rate for marsh sediments.

Sediment TP was lower in CKM and more homogeneous with depth compared to SCE. In CKM, both the top (0 – 2.5 cm) and bottom (2.5 – 5 cm) layers of sediment had similar TP concentrations (average of 854 mg/kg sediment and 848 mg/kg sediment, respectively). This could be evidence of sediment disturbance, either by carp bioturbation or wind resuspension. On the other hand, TP concentrations differed slightly more between the two layers in SCE (top average of 1,223 mg/kg sediment; bottom average of 1,069 mg/kg sediment). This could be evidence of greater sediment stability within SCE and deposition of TP comprising the surface sediments.

Similar to sediment TP, dilute salt extractable P was more distinct between the top and bottom sediment layers in SCE compared to CKM. The interquartile range of dilute extractable P for SCE top sediments (0.37 – 0.47 mg/kg sediment) did not overlap with that of bottom sediments (0.32 – 0.37 mg/kg sediment). Whereas for CKM, the interquartile range for the dilute extractable P for the top sediments (0.43 – 0.53 mg/kg sediment) was encompassed within the interquartile range for the bottom sediments (0.40 – 0.65 mg/kg sediment). The median for dilute extractable P was therefore greatest for CKM top and bottom sediments (0.51 mg/kg sediment), then top sediments from SCE (0.42 mg/kg sediment), and lowest for the bottom SCE sediments (0.35 mg/kg sediment). The higher value for dilute extractable P for CKM suggests that CKM sediments could potentially release more P than SCE sediments.

The EPC_0 analysis indicated that CKM sediments continued to adsorb P from the water column at a lower concentration than sediments in SCE. The CKM sediments were estimated to absorb P if water column DRP concentrations exceeded 0.013 mg/L. Below that concentration, sediments were expected to release P. This concentration was higher than the average DRP concentration measured at Site 2 on the day of sediment collection (0.0015 mg/L), which supports the theory that DRP released from sediment was quickly taken up by phytoplankton. Water column concentrations below the EPC_0 spurred continual P release from the sediments, resulting in the internal P loading observed a few days prior in the July sampling.

The SCE estimated EPC_0 was higher than that for CKM, which was expected given the higher sediment/water interface DRP concentrations compared to CKM. The SCE sediments were predicted to absorb P from the water column if DRP concentrations exceeded 0.031 mg/L. The observed DRP concentration at Site 2 (0.028 mg/L) was below that level, but the difference between the two measurements was much smaller than the difference observed in CKM. Since SCE water column DRP concentrations were below the EPC_0, this suggests that sediments were releasing P. Overall, however, P inlet and outlet loads for the July sampling event indicated SCE was retaining P. If sediments were releasing P at Site 2, perhaps P was being absorbed elsewhere in SCE sediments or particulate P was settling out of suspension and not exiting the marsh. The DRP concentration at Site 1 (0.0345 mg/L) was higher compared to Site 2 and above the $EPC_{0.}$ Sediments upstream of Site 2 may have absorbed sufficient P to make SCE a P sink. The DRP concentration downstream at Site 3 (0.0085 mg/L), which was below the estimated EPC_0 and would contribute to sediment P release, also supported the theory that other processes, such as macrophyte P uptake, may have been more dominant than sediment P release. While SCE sediment was estimated to be a P source at a higher DRP concentration than CKM, the monthly P load balance (data from 2012-2017 presented in Chapter 2) showed SCE to regularly be a P sink under baseflow conditions. Thus, it does not appear that the high EPC_0, compared to CKM, resulted in internal P loading that outweighed net P retention elsewhere within the marsh.

Differences in P release from the two marsh sediments could be linked to their distinct physical or chemical constituents. Within CKM, there was more clay (average 17%) compared to SCE (average 11%). Sediment clay content is associated with possible sites for P adsorption and desorption resulting in higher P adsorption capacity as clay content increases (Agudelo *et al.*, 2011). Sediment TP, however, was higher in SCE (average 1146 mg/kg sediment) compared to CKM (851 mg/kg sediment) even though the higher CKM clay content would be expected to provide more P adsorption. Release of P from sediment is also dependent on conditions not measured in this study, such as organic matter content and quantities of oxygen, calcium, iron, manganese and aluminum (Hongthanat *et al.*, 2016; Orihel *et al.*, 2017). Further study examining the fractionation (*e.g.* loosely sorbed-P, Ca/Mg-P and FE/AL-P) of P within each marsh's sediment would show whether relative proportions of each pool of P differed between the two marshes. Elsewhere, vegetated sediments have been shown to have more inorganic forms of P and iron than non-vegetated sediments, which reduces the ability of P to be released from the vegetated sediments and promotes P storage (Kasan *et al.*, 2016).

A better understanding of physical processes resuspending sediment within each marsh could inform the portion of internal P loading expected from particulate sediment resuspension versus sediment dissolved P release. Physical disturbance of marsh sediment impacts the amount of P and SS resuspended in the water column. While carp disturbance has decreased as a result of carp exclusion and harvesting, there is still likely to be more bioturbation within CKM compared to SCE. This study, though, did not attempt to quantify the differences in bioturbation between the two marshes. Differences in fetch and wind patterns are similarly expected to be more disruptive in CKM compared to SCE because few macrophytes are present to stabilize sediment. But again, no quantitative measurements of wind velocity or shear stress at the sediment/water interface were collected to compare conditions between the two systems. Resuspension of particulate P and SS within either marsh system would augment any dissolved P released from sediment and would constitute a portion of P exported from each marsh. But these processes are expected to be more prominent within CKM compared to SCE.

This study only captured sediment TP content and the EPC_0 at one point in time. Seasonally, sediment TP and the proportion of P in various extractable pools shifts (Song *et al.*, 2009) resulting in a different EPC_0 at different points in time (Kasan *et al.*, 2016). Spring and summer have been shown to have a lower EPC_0 compared to fall and winter. This could lead to sediments acting as a sink for P during the non-growing season but as a source during the remainder of the year. Active bacterial communities releasing both inorganic and organic phosphate could have also depleted TP in CKM sediments compared to SCE sediments. Additional study of seasonal sediment P fractionation, sediment bacteria prevalence, and the resulting shifts to the EPC_0 could determine whether internal loading contributes to more sediment TP depletion throughout the summer in CKM than in SCE.

Conclusion

High nutrient concentrations were recorded entering both marshes. The inlets with high TP concentrations were Roy and Wuerches Creek for CKM and Silver Creek at Spaulding Rd for SCE. The highest TN concentrations were observed at the inlets of Dakin Creek and Roy Creek. In proportion to its subwatershed area, the TP, TSS, and TN loads from Roy Creek to CKM were more than expected. Dakin Creek exported more TN and TSS to SCE that would be expected for its proportional subwatershed area as well.

In most seasons, more TP and SS entered Green Lake through SCE than through CKM. Relative to flow, however, higher TP and SS concentrations reached the lake through CKM rather than SCE. In the winter, both marshes had similar concentrations of TP and SS. But as the growing season progressed, concentrations increased more in CKM than they did in SCE. In the fall, TP concentrations decreased in SCE more rapidly than in CKM. These differences in fall TP illustrated how phytoplankton continue to cycle P maintaining high marsh TP concentrations later in the growing season compared to a macrophyte-dominant system. Summer SS concentrations were also much higher in the turbid phytoplankton-dominant CKM compared to clear SCE. CHLA concentrations were also much higher in CKM compared to SCE which meant that phytoplankton likely contributed substantially to the high SS measurements.

While CKM had higher summer TP concentrations than SCE, TDP in both marshes was similar. TDP:TP ratios were much higher in SCE compared to CKM suggesting that TDP entering CKM through the inlets was quickly consumed by phytoplankton. During non-storm sampling, CKM was found to export TP and TSS whereas SCE retained both. This pointed to consistent internal TP loading within CKM but not SCE. Following storm events, both marshes retained TSS and CKM retained TP. SCE showed both retention and export of TP for individual storm events. Even though SCE sediments had favorable conditions for P release, according to the estimated EPC_0, monthly sampling did not suggest that the marsh was exporting P. Further analysis of sediment P release and macrophyte P uptake could better quantify P cycling within SCE.

Over all 11 sampling events, SCE retained TP, TSS and TN. CKM was a sink for TN and a source of TP. TSS retained in CKM during the storm events was equal to what was exported during the non-storm events. While these events do not encompass a full seasonal or yearly estimation of nutrient and sediment retention within each marsh, the results do demonstrate how the two marshes differ at a specific point in time.

Chapter 4 Green Lake's Phytoplankton versus Macrophyte-Dominated Marshes

Changes in SCE over 30 years

Upgrades at the RWTP in the early 2000's significantly reduced the amount of P discharged to Silver Creek 6 km upstream of the SCE inlet. This change occurred at about the same time as a major restoration effort in SCE. The P reductions at the RWTP ranged from about 2 kg/day in summer to 2.8 kg/day in the spring. In summer, the RWTP decreased its TP load by 50%, with approximately 80% reductions during the remaining seasons.

Downstream at the outlet of SCE, TP was significantly reduced following the point source reduction and the transition of the marsh from phytoplankton to macrophyte dominance. The TP load reductions ranged from 0.8 kg/day in winter to 3 kg/day in summer. The largest reductions were in fall (40%) and summer (25%). Flow-weighted mean TP concentrations at the SCE outlet decreased approximately 40% in each season from before to after restoration. Shifting the equilibrium state from phytoplankton to macrophyte-dominant is generally associated with TP reduction during the growing season. The fact that TP also decreased in winter is likely the impact of reduction from the upstream RWTP. Therefore, the point source reduction and biological changes both reduced TP loads to Green Lake.

After restoration, SS load reductions at the SCE outlet were significant in spring (0.46 metric tonnes/day) and summer (0.73 metric tonnes/day), and flow-weighted mean SS concentrations decreased in those seasons by 50%. Fall and winter SS flow-weighted mean concentrations were relatively unaffected by the equilibrium shift (8% reduction and 12% increase, respectively). Since macrophytes are expected to stabilize sediment and reduce phytoplankton, their impact on reducing SS occurs during the growing season. Therefore, there was little reason to expect that SS concentrations would change in the winter following restoration, which aligned with what was observed.

From 2012 – 2017, the restored macrophyte-dominated SCE was a net sink for both TP and SS. Year-round, TP was retained, and flow-weighted mean TP concentrations significantly decreased from the inlet to outlet of SCE in all seasons. There was net export of SS in fall but retention in all other seasons. Fall was also the only season when flow-weighted SS concentrations did not significantly decrease from inlet to outlet. Overall, SCE reduced the amount of TP and SS entering Green Lake by 15% and 37%, respectively, compared to the amount entering at Spaulding Rd. Since storage was only quantified as the difference between the amount estimated at the Spaulding Rd gage and the CTH A gage, TP and SS contributions from SCE's other tributary, Dakin Creek, were not accounted for. Storage is therefore underestimated, and if inputs from Dakin Creek were accounted for, TP and SS retention would have increased to 33% and 50%, respectively (Dakin inputs from SWAT model; Baumgart, 2015). Since the amount of SS exported during fall was relatively small compared to the amounts retained in other seasons, all seasons could have retained SS after factoring in additions from Dakin Creek.

In summer, water quality improved within macrophyte-dominated SCE as shown by samples collected from 2006 – 2017. Summer concentration decreases from the SCE inlet at Spaulding Rd to the outlet were documented for TP, TDP, SS and TN. When inlet concentrations increased, removal efficiencies generally increased regardless of season for TP, SS, TN, and in summer for TDP. Removal efficiencies for TN decreased as marsh retention times decreased, such that during times of high flow following precipitation events, the marsh was less likely to retain TN.

Overall, TP and SS concentrations and loads decreased following the transition from phytoplankton to macrophyte-dominance. The valuable ecosystem services now provided by the marsh are critical for decreasing TP and SS loads from the large Silver Creek watershed. Evaluating land management within the watershed is also important for reducing TP and SS loads reaching SCE. Decreasing loads to SCE will help ensure that the marsh remains in its macrophyte-dominated state, improving water quality before reaching Green Lake.

Marsh Comparison of CKM vs SCE

Because the 2012 – 2017 flows from SCE were higher than from CKM, the amount of TP and SS entering Green Lake was generally larger from SCE than from CKM. For both marshes, TP loads were highest in spring, generally with two to three times more TP entering the lake through SCE than CKM. This ratio decreased in the fall when about one third of the time, TP loads leaving CKM were greater than those from SCE. Loads of SS were likewise greatest in the spring, with about 1.5 times more SS entering the lake through SCE than CKM. Summer SS loads from CKM were similar to spring loads, while loads from SCE decreased. In eight out of the nine summer months monitored during the five-year period (SS was not monitored in all years), SS loads from CKM exceeded those from SCE. On a watershed area basis, CKM had higher yields for both TP and TSS than SCE (Appendix G Figure G-1). Seasonally, fall had the largest discrepancy for TP, with yields from CKM four times greater than from SCE (Appendix G Table G-1). SS yields differed most in the summer with eight times greater yield from CKM than SCE.

Flow-weighted mean TP and SS concentrations were similar in both marshes during the winter but diverged in the growing season. CKM had higher TP concentrations than SCE in all seasons except winter and higher SS concentrations than SCE in spring and summer. In both marshes, the highest TP concentrations were observed in summer. For CKM, only winter had significantly lower TP concentrations compared to summer. For SCE, TP concentrations were lower in both fall and winter than in summer. The phytoplankton-dominated CKM, therefore, continued to discharge higher concentrations of TP to Green Lake later into the season compared to macrophyte-dominated SCE. Marsh TP concentrations returned to lower winter concentrations about three months earlier in SCE compared to CKM. The SS concentrations increased from winter to spring in both marshes but decreased in summer for SCE yet remained high in CKM. Summer SS concentrations in turbid CKM were about eight times higher than in clear SCE. Phytoplankton was associated with high fall TP concentrations and high summer SS concentrations, whereas macrophytes were not. This similarity in winter conditions but divergent growing season conditions suggests the importance of internal processes driven by the dominant biological community.

The inlets of both marshes had high TN, P and TSS concentrations. Silver Creek at Spaulding Rd and Wuerches Creek contributed high concentrations of TP (median above 0.075 mg/L) and TDP (median above 0.075 mg/L) into SCE. The TN concentrations at Roy Creek, followed by Dakin Creek, were the highest of all the inlets. Dakin Creek contributed more TN and TSS than expected based on its relative subwatershed area. Roy Creek also had high TP and TSS concentrations and contributed more TP, TSS, and TN than expected based on subwatershed area. The TN concentrations decreased from inlets to interior locations for both marshes, and SCE concentrations were generally lower than those in CKM. Since both marshes had inputs with high TN, P and TSS, there was not strong evidence that CKM had lower water quality solely because of inlet concentrations.

Compared to their respective inlets, TP concentrations generally decreased within SCE but increased within CKM. Even though TP increased within CKM, TDP concentrations remained low, with ratios of TDP:TP generally below 0.25. This suggests that internal sources of P within CKM discharged as particulate P, either as sediment that was physically disturbed or as phytoplankton that quickly assimilated TDP released from sediments. High concentrations of TP and TDP entering SCE at Spaulding Rd were generally reduced from upstream to downstream. In SCE, the TDP:TP was higher than in CKM, regularly above 0.5. When compared to CKM, TDP concentrations within SCE were higher and more varied. Since TDP is bioavailable, lower concentrations suggest rapid assimilation into biological organisms. In SCE, high TDP concentrations were observed following the largest storm events. These instances with shorter marsh retention times (SCE storm median retention time 2 days; non-storm retention time 13 days) suggest insufficient time for biological uptake of TDP prior to entering Green Lake. While CKM had high TP concentrations throughout the marsh, the consistently low TDP concentrations suggest prevalent biological uptake. The longer retention times, in addition to prevalent phytoplankton, in CKM compared to SCE could contribute to such favorable conditions for TDP reduction. While TDP decreased from inlets to within CKM, the opposite increase in TP from inlets to within CKM suggests the importance of internal P sources (e.g. P released from sediments). In SCE, the dominant TP reduction mechanisms could be related to macrophyte uptake or sediment P sorption.

As with TP, TSS concentrations generally decreased from inlets to within SCE but increased from inlets to within CKM. Median TSS concentrations in turbid CKM were twelve times higher than in SCE. Concentrations of CHLA were likewise much higher in CKM (median CHLA was 37 times greater than SCE). In CKM, TSS and CHLA were lower following storm events than during non-storm periods. The relative CHLA decrease was larger, however, suggesting that storm-driven TSS was not dominated by phytoplankton (as measured though CHLA). The large difference in TSS and CHLA show that SCE is a clear, macrophyte-dominated ecosystem and CKM is a turbid, phytoplankton-dominated system.

Within both marshes, there was seasonal variation in the TN:TP molar ratio. The ratio was generally higher in SCE than CKM, and therefore SCE was likely to be more P limited. In SCE the lowest TN:TP ratios were in summer 2016 and after the 2017 summer storm event. In CKM the ratios were lowest during both summers; TN:TP ratios decreased from April to July and increased from August to October. The lowest ratios reached an interval where either N or P could potentially be a liming nutrient. Since neither marsh reached conditions of exclusive N limitation, continuing watershed P reduction remains valuable.

Over the 11 intra-marsh events sampled, CKM was a source of P to Green Lake, whereas SCE was a sink for P. This was most notable during non-storm events. Internal P loading within CKM could have been the result of both resuspension of sediment (e.g. from wind or carp) and P release from sediment. Since SCE generally retained P during the non-storm events, there was no strong evidence for internal P loading within the marsh being a large P contributor to the lake. In CKM, P retention was only estimated following some storm events. In SCE, both retention and export were estimated for individual storm events. The net TP retention in CKM during storms was insufficient to offset releases during the non-storm events. However, the net retention of TP within SCE during non-storm events offset TP exported during the largest storm event.

The SCE consistently retained TSS, which differed from near zero net retention for CKM. For non-storm events, as with TP, SCE retained TSS and CKM exported TSS. The SCE also retained TSS during storm events. The CKM only retained TSS following the two largest storm events, but the quantity almost matched the amount exported during the other sampling days. Under a variety of conditions, SCE reduced the amount of TSS reaching the lake, whereas CKM only did so following large storm events.

Both marshes decreased the amount of TN reaching Green Lake. During non-storm events, both marshes retained TN, with SCE generally retaining more than CKM. Each marsh exported TN during one storm event. The amount exported, however, was less than the amount retained over the 10 other sampling days. While both marshes retained TN, SCE retained about three times more than CKM.

Storage estimates based on the 11 sampling days for TP, TSS and TN compared how the two marshes differed in retention at the same limited points in time. Combined storage during the 11 non-storm and storm sampling events should not be extrapolated to represent marsh storage over longer periods of time (e.g. full storm hydrograph, monthly). Seasonal or long-term storage will depend on the distribution of storm vs non-storm events during the time period evaluated and will likely differ substantially from the estimates using the 11 sampling events this study.

SCE sediments were not expected to be a net source of P to the lake because SCE acted as a TP sink during non-storm events. However, results from dilute salt extractions and EPC_0 estimates support SCE sediments being a gross exporter of P. Slightly more DRP was released from CKM sediment than from SCE sediment using a dilute-salt solution extraction. However, the EPC_0 for SCE sediment was higher than CKM meaning that SCE sediment would release DRP into the water column at a higher ambient DRP concentration than CKM. On the day of sediment collection, the DRP sample collected at the SCE coring site was below the estimated

EPC_0, suggesting sediment P release. The TDP concentrations in SCE (collected during the 11 sampling events) were also occasionally lower than the estimated EPC_0 value, suggesting SCE sediments could have been releasing P into the water column. Since overall load estimates showed SCE retaining P, any P that was released from sediment was likely retained within the marsh through different processes (e.g. macrophyte uptake). The CKM samples from the day of sediment collection also had DRP concentrations below the EPC_0, supporting P release from sediments. The EPC_0 estimates suggest that sediments from both wetlands released P, but the overall P mass balance suggests that this source of additional P was only exported to Green Lake from CKM.

While a CKM equilibrium shift may not be accompanied by a stark reduction in inlet TP loads, as occurred in SCE with the biological shift and the historic reduction of RWTP TP loads, there is potential for decreased growing season TP concentrations following macrophyte dominance. Median TP concentrations in pre-restoration SCE were generally higher than current CKM concentrations (Figure 4-1). But summer concentrations are similar. A shift to macrophyte dominance in CKM would likely decrease summer TP concentrations, but lacking a large TP reduction at the inlets to the marsh, the decrease may not match what was observed in SCE. Reducing loads to CKM and lowering TP concentrations within the marsh would make a shift to macrophyte-dominance more likely to occur.

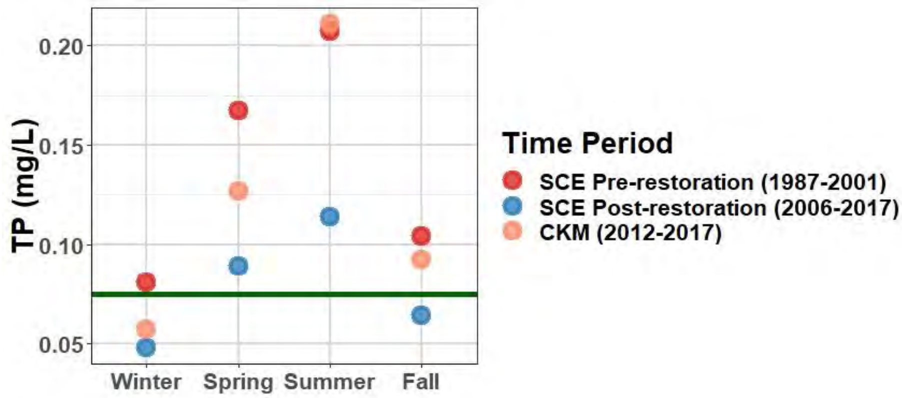

Figure 4-1: Seasonal median flow-weighted TP concentrations for each marsh outlet during each time period. The green line indicates the TP water quality criteria of 0.075 mg/L.

The CKM had higher median flow-weighted summer SS concentrations than pre-restoration SCE (Figure 4-2). In the other seasons, however, current SS concentrations in CKM were higher than the historic SCE median concentration. CKM was only more turbid than phytoplankton-dominant SCE (pre-restoration; 1987-2001) in summer. Since SCE concentrations have since decreased to below current CKM concentrations in spring and fall, CKM could experience similar decreases. Comparing data from the entire decade post SCE restoration (2006-2017) to only the second half (2012-2017) shows that SS concentrations have continued to decrease over time, especially in summer and fall. As macrophytes inhabited SCE over time, the water became clearer and SS decreased during the growing season. This cumulative impact demonstrates that the equilibrium shift continued to change marsh conditions since initial restoration a decade ago. Water quality improvements observed immediately after restoration, therefore, are not illustrative of the full impacts to come.

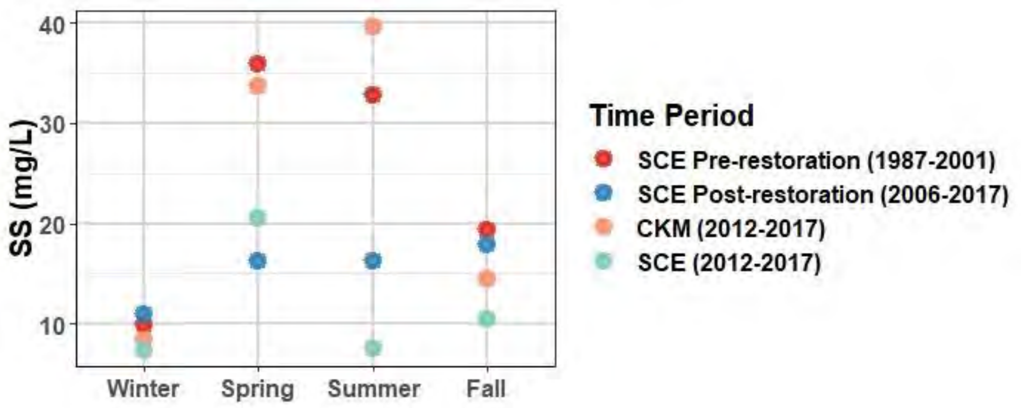

Figure 4-2: Seasonal median flow-weighted SS concentrations for each marsh during each time period. SCE pre-restoration and CKM were phytoplankton-dominated. SCE post-restoration and SCE (2012-2017) were both macrophyte-dominated.

Current similarities between the two marshes in winter TP and SS concentrations, yet differences during the remainder of the year, suggest that TP and SS concentrations could decrease with an equilibrium shift within CKM. Continued efforts for removing carp and establishing macrophytes are warranted. Reductions, however, may not be immediate. If supplemented by watershed load reductions, conditions within CKM could transition to favor macrophyte dominance. This shift would then likely reduce TP and SS load to Green Lake during the growing season.

Works Cited

Abell, J. M., Özkundakci, D. and Hamilton, D. P. (2010) Nitrogen and Phosphorus Limitation of Phytoplankton Growth in New Zealand Lakes: Implications for Eutrophication Control, *Ecosystems*, 13(7), pp. 966–977. doi: 10.1007/s10021-010-9367-9.

Agudelo, S. C. *et al.* (2011) Phosphorus Adsorption and Desorption Potential of Stream Sediments and Field Soils in Agricultural Watersheds, *Journal of Environment Quality*, 40(1), pp. 144–152. doi: 10.2134/jeq2010.0153.

Bajer, P. G. and Sorensen, P. W. (2015a) Effects of common carp on phosphorus concentrations, water clarity, and vegetation density: a whole system experiment in a thermally stratified lake, *Hydrobiologia*, 746(1), pp. 303–311. doi: 10.1007/s10750-014-1937-y.

Bajer, P. G. and Sorensen, P. W. (2015b) Effects of common carp on phosphorus concentrations , water clarity , and vegetation density : a whole system experiment in a thermally stratified lake, *Hydrobiologia*, 746, pp. 303–311. doi: 10.1007/s10750-014-1937-y.

Bajer, P., Sullivan, G. and Sorensen, P. (2009) Effects of a rapidly increasing population of common carp on vegetative cover and waterfowl in a recently restored Midwestern shallow lake, *Hydrobiologia*, 632(1), pp. 235–245. doi: 10.1007/s10750-009-9844-3.

Barko, J. W., Gunnison, D. and Carpenter, S. R. (1991) Sediment interactions with submersed macrophyte growth and community dynamics, *Aquatic Botany*, 41(1–3), pp. 41–65. doi: 10.1016/0304-3770(91)90038-7.

Barko, J. W. and Smart, R. M. (1980) Sediment based nutrition of submersed freshwater macrophytes, *Freshwater Biology*, 10, pp. 229–238.

Baumgart, P. (2015) *Application of the Soil and Water Assessement Tool (SWAT) to Evaluate Non-Point Source Phosphorus and TSS Loads in the Big Green Lake Watershed, Wisconsin*.

Borman, S., Korth, R. and Temte, J. (2013) *Through the Looking Glass*. Second. Edited by D. Snyder. Stevens Point, WI: Wisconsin Lakes Partnership.

Butterfield, B. *et al.* (2015) *Green Lake Aquatic Plant Community Assessment Green Lake County, Wisconsin January 2015*. De Pere, WI.

Capon, S. J. *et al.* (2015) Regime shifts, thresholds and multiple stable states in freshwater ecosystems; a critical appraisal of the evidence, *Science of the Total Environment*. Elsevier B.V., 534, pp. 122–130. doi: 10.1016/j.scitotenv.2015.02.045.

Carpenter, S. R. (1980) Enrichment of Lake Wingra , Wisconsin , by Submersed Macrophyte Decay, *Ecology*, 61(5), pp. 1145–1155.

Carpenter, S. R. and Adams, M. S. (1979) Effects of nutrients and temperature on decomposition of Myriophyllum spicatum L. in a hard-water eutrophic lake, *Limnology and Oceanography*, 24(3), pp. 520–528. doi: 10.4319/lo.1979.24.3.0520.

Cunningham, P. (2015) *Vegetation Analysis Report*.

Dai, Y., Wu, S., *et al.* (2012) Effects of Ceratophyllum demersum L. restoration on phosphorus balance at water–sediment interface, *Ecological Engineering*, 44, pp. 128–132. doi: http://dx.doi.org/10.1016/j.ecoleng.2012.04.003.

Dai, Y., Jia, C., *et al.* (2012) Effects of the submerged macrophyte Ceratophyllum demersum L. on restoration of a eutrophic waterbody and its optimal coverage, *Ecological Engineering*. Elsevier B.V., 40, pp. 113–116. doi: 10.1016/j.ecoleng.2011.12.023.

Delta Institute (2016) *The Green Lake Watershed Phosphorus Prioritization Tool*.

Dodds, W. K. and Smith, V. H. (2016) Nitrogen, phosphorus, and eutrophication in streams, *Inland Waters*, 6(2), pp. 155–164. doi: 10.5268/IW-6.2.909.

Eugelink, A. H. (1998) Phosphorus Uptake and Active Growth of Elodea canadensis Michx. and Elodea nuttallii (Planch.) St. John, *Water Science and Technology*, 37(3), pp. 59–65.

Feng, W. *et al.* (2018) Simulated bioavailability of phosphorus from aquatic macrophytes and phytoplankton by aqueous suspension and incubation with alkaline phosphatase, *Science of the Total Environment*. Elsevier B.V., 616–617, pp. 1431–1439. doi: 10.1016/j.scitotenv.2017.10.172.

Fischer, J. R., Krogman, R. M. and Quist, M. C. (2013) Influences of native and non-native benthivorous fishes on aquatic ecosystem degradation, *Hydrobiologia*, 711(1), pp. 187–199. doi: 10.1007/s10750-013-1483-z.

Fisher, J. and Acreman, M. C. (2004) Wetland nutrient removal : a review of the evidence, *Hydrology and Earth Systems Sciences*, 8(4), pp. 673–685.

Fox, J. (2018) *Package 'car'*. Available at: https://cran.r-project.org/web/packages/car/car.pdf.

Fu, L. and Wang, Y.-G. (2012) Statistical Tools for Analyzing Water Quality Data, in *Water Quality Monitoring and Assessment*, pp. 143–168. Available at: https://www.intechopen.com/books/water-quality-monitoring-and-assessment/statistical-tools-for-analyzing-water-quality-data.

Gao, J. *et al.* (2009) Phosphorus removal from water of eutrophic Lake Donghu by five submerged macrophytes, *Desalination*. Elsevier B.V., 242(1–3), pp. 193–204. doi: 10.1016/j.desal.2008.04.006.

Graetz, D. A. and Nair, V. D. (2009) Phosphorus Sorption Isotherm Determination, in Kovar, J. L. and Pierzynski, G. M. (eds) *Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters*. Second. SERA-IEG 17, pp. 33–37.

Green Lake Association (2017a) *2017 Annual Report & Membership Directory*.

Green Lake Association (2017b) Carpe-Diem!, *Times & Tides*, 44(1), pp. 3–4.

Green Lake Sanitary District (2016) *K Estuary Fyke Netting for Spring/Summer of 2016*.

Green Lake Sanitary District (2017) *Lake Management Planning (LMP) Target Carp Removal*, *Green Lake Sanitary District Newsletter*. Available at: http://www.glakesd.com/wp-content/uploads/2014/08/GLSDNewsletter_Spring2017.pdf (Accessed: 26 July 2018).

Guildford, S. J. and Hecky, R. E. (2000) Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: Is there a common relationship?, *Limnology and Oceanography*, 45(6), pp. 1213–1223. doi: 10.4319/lo.2000.45.6.1213.

Hector, A., von Felten, S. and Schmid, B. (2010) Analysis of variance with unbalanced data: An update for ecology & evolution, *Journal of Animal Ecology*, 79(2), pp. 308–316. doi: 10.1111/j.1365-2656.2009.01634.x.

Hongthanat, N. *et al.* (2016) Phosphorus source — sink relationships of stream sediments in the Rathbun Lake watershed in southern Iowa , USA, *Environmental Monitoring and Assessment*. Environmental Monitoring and Assessment, 188(453), pp. 2–14. doi: 10.1007/s10661-016-5437-6.

Johnson, T., Nickel, A. and Evensen, E. (2011) *An Assessment of Hill, Roy, Silver, and Wuerches Creeks (303d Impaired waters)*.

Kasan, N. A. *et al.* (2016) Study of Seasonal Phosphorus Dynamics in Vegetated and Non-vegetated Wetland Sediment Affected by Long-term Agricultural Productions, *Journal of Applied Sciences*. Faisalabad :, pp. 252–261. doi: 10.3923/jas.2016.252.261.

Kasprzak, P. *et al.* (2008) Chlorophyll a concentration across a trophic gradient of lakes: An estimator of phytoplankton biomass?, *Limnologica*, 38(3–4), pp. 327–338. doi: 10.1016/j.limno.2008.07.002.

Koltun, G. F. *et al.* (2006) User's Manual for the Graphical Constituent Loading Analysis System (GCLAS), in *U.S. Geological Survey Techniques and Methods, Book 4*. Reston, Virginia, pp. 1–51.

Kolzau, S. *et al.* (2014) Seasonal patterns of Nitrogen and Phosphorus limitation in four German Lakes and the predictability of limitation status from ambient nutrient concentrations, *PLoS ONE*, 9(4). doi: 10.1371/journal.pone.0096065.

Landers, D. H. (1982) Effects of naturally senecing aquatic macrophytes on nutrient chemistry and chlorophyss a of surrounding waters:, *Linmology Oceanogrpahy*, 27(3), pp. 428–439.

Lenth, R. *et al.* (2018) *Package 'emmeans'*. Available at: https://cran.r-project.org/web/packages/emmeans/emmeans.pdf.

Lewis, W. M. and Wurtsbaugh, W. A. (2008) Control of Lacustrine Phytoplankton by Nutrients: Erosion of the Phosphorus Paradigm, *International Review of Hydrobiology*, 93(4–5), pp. 446–465. Available at: http://dx.doi.org/10.1002/iroh.200811065.

Liu, Q. *et al.* (2018) Effects of temperature on phosphorus mobilization in sediments in microcosm experiment and in the field, *Applied Geochemistry*. Elsevier Ltd, 88, pp. 158–166. doi: 10.1016/j.apgeochem.2017.07.018.

Liu, Z., Li, Y. and Li, Z. (2009) Surface water quality and land use in Wisconsin, USA – a GIS approach, *Journal of Integrative Environmental Sciences*, 6(1), pp. 69–89. doi: 10.1080/15693430802696442.

Lougheed, V., Crosbie, B. and Chow-fraser, P. (1998) Predictions on the effect of common carp (Cyprinus carpio) exclusion on water quality , zooplankton , and submergent macrophyte submergent macrophytes in a Great Lakes wetland, *Canadian Journal of Fisheris and Aquatic*

Sciences, 55(May), pp. 1189–1197. doi: 10.1139/cjfas-55-5-1189.

Marsh, C. D. (1891a) Notes on Depth and Temperature of Green Lake, *Transactions of the Wisconsin Academy of Sciences, Arts and Letters*, VIII, p. 214–Plate VI.

Marsh, C. D. (1891b) On the Deep Water Crustacea of Green Lake, *Transactions of the Wisconsin Academy of Sciences, Arts and Letters*, VII, pp. 211–213.

Mbonimpa, E. G. *et al.* (2012) SWAT Model Application to Assess the Impact of Intensive Corn-Farming on Runoff, Sediments and Phosphorus Loss from Agricultural Watershed in Wisonsin, *Journal of Water Resources and Protection*, 4(July), pp. 423–431. doi: 10.4236/jwarp.2012.47049.

Mjelde, M. and Faafeng, B. A. (1997) Ceratophyllum demersum hampers phytoplankton development in some small Norwegian lakes over a wide range of phosphorus concentrations and geographical latitude, *Freshwater Biology*, 37(2), pp. 355–365. doi: 10.1046/j.1365-2427.1997.00159.x.

Moore, B. C., Funk, W. H. and Anderson, E. (1994) Water quality, fishery, and biologic characteristics in a shallow, eutrophic lake with dense macrophyte populations, *Lake and Reservoir Management*, 8(2), pp. 175–188. doi: 10.1080/07438149409354469.

Nichols, D. S. and Keeney, D. R. (1976) Nitrogen nutrition of Myriophyllum spicatum : uptake and translocation of 15 N by shoots and roots, *Freshwater Biology*, 6, pp. 145–154.

Orihel, D. M. *et al.* (2017) Internal phosphorus loading in Canadian fresh waters: a critical review and data analysis, *Canadian Journal of Fisheries and Aquatic Sciences*, 74, pp. 2005–2029. doi: 10.1139/cjfas-2016-0500.

Paerl, H. W. (2009) Controlling eutrophication along the freshwater-Marine continuum: Dual nutrient (N and P) reductions are essential, *Estuaries and Coasts*, 32(4), pp. 593–601. doi: 10.1007/s12237-009-9158-8.

Penne, C. R. and Pierce, C. L. (2008) Seasonal Distribution, Aggregation, and Habitat Selection of Common Carp in Clear Lake, Iowa, *Transactions of the American Fisheries Society*, 137(4), pp. 1050–1062. doi: 10.1577/T07-112.1.

Powers, S. M., Robertson, D. M. and Stanley, E. H. (2014) Effects of lakes and reservoirs on annual river nitrogen, phosphorus, and sediment export in agricultural and forested landscapes, *Hydrological Processes*, 28(24), pp. 5919–5937. doi: 10.1002/hyp.10083.

Ripon, C. of (2018) *About the Wastewater Treatment Facility*. Available at: http://www.cityofripon.com/?SEC=64BC4A07-B2B4-472A-AC2A-DAE700E04655 (Accessed: 15 May 2018).

Robertson, D. *et al.* (1998) - Dynamics in Phosphorus Retention in Wetlands Upstream of Delavan Lake, Wisconsin, - *Lake and Reservoir Management*, 14(4), pp. 466–477. doi: - 10.1080/07438149809354353.

Robertson, D. M. *et al.* (2006) *Nutrient Concentrations and Their Relations to the Biotic Integrity of Wadeable Rivers in Wisconsin: Professional Paper 1722*.

Robertson, D. M. *et al.* (2018) Water-Quality Response to Changes in Phosphorus Loading of the Winnebago Pool Lakes, Wisconsin, with Special Emphasis on the Effects of Internal Loading in a Chain of Shallow Lakes, *USGS - Scientific Investigations Report 2018–5099*, p. 58. doi: 10.1088/0960-1317/18/5/055004.

Robertson, D. M., Weigel, B. M. and Graczyk, D. J. (2008) *Nutrient Concentrations and Their Relations to the Biotic Integrity of Nonwadeable Rivers in Wisconsin: Professional Paper 1754*.

Sachs, R. (2017) *Modified Permit Fact Sheet*.

Scheffer, M. *et al.* (1993) Alternative equilibria in shallow lakes, *Trends in Ecology and Evolution*, 8(8), pp. 275–279. doi: 10.1016/0169-5347(93)90254-M.

Schindler, D. W. (2012) The dilemma of controlling cultural eutrophication of lakes, *Proceedings of the Royal Society B: Biological Sciences*, 279(1746), pp. 4322–4333. doi: 10.1098/rspb.2012.1032.

Schrage, L. J. and Downing, J. A. (2004) Pathways of increased water clarity after fish removal from Ventura Marsh; a shallow, eutrophic wetland, *Hydrobiologia*, 511, pp. 215–231. doi: 10.1023/B:HYDR.0000014065.82229.c2.

Self-Davis, M. L., Moore, P. A. and Joern, B. C. (2009) Water- or Dilute Salt-Extractable Phosphorus in Soil, in Kovar, J. and Pierzynski, G. (eds) *Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters*. Second, pp. 22–24. Available at: http://www.sera17.ext.vt.edu/Documents/P_Methods2ndEdition2009.pdf.

Sesing, M. (2006) *Silver Creek Plant Survey*.

Sesing, M. *et al.* (2015) *A Lake Management Plan for Green Lake, Green Lake, Wisconsin*.

Søndergaard, M. *et al.* (1990) Phytoplankton biomass reduction after planktivorous fish reduction in a shallow , eutrophic lake : a combined effect of reduced internal P-loading and increased zooplankton grazing, *Hydrobiologia*, (1978), pp. 229–240.

Søndergaard, M. *et al.* (2002) Seasonal dynamics in the concentrations and retention of phosphorus in shallow Danish lakes after reduced loading, *Aquatic Ecosystem Health & Management*, 5(1), pp. 19–29. Available at: http://dx.doi.org/10.1080/14634980260199936.

Søndergaard, M. *et al.* (2017) Nitrogen or phosphorus limitation in lakes and its impact on phytoplankton biomass and submerged macrophyte cover, *Hydrobiologia*. Springer International Publishing, 795(1), pp. 35–48. doi: 10.1007/s10750-017-3110-x.

Søndergaard, M., Bjerring, R. and Jeppesen, E. (2013) Persistent internal phosphorus loading during summer in shallow eutrophic lakes, *Hydrobiologia*, 710(1), pp. 95–107. doi: 10.1007/s10750-012-1091-3.

Søndergaard, M., Jensen, J. and Jeppesen, E. (2003) Role of sediment and internal loading of phosphorus in shallow lakes, *Hydrobiologia*, 506–509(1–3), pp. 135–145. doi: 10.1023/B:HYDR.0000008611.12704.dd.

Song, C. *et al.* (2009) Seasonal variations in chlorophyll a concentrations in relation to potentials of sediment phosphate release by different mechanisms in a large Chinese shallow eutrophic lake (Lake Taihu), *Geomicrobiology Journal*, 26(7), pp. 508–515. doi: 10.1080/01490450903061119.

Stauffer, R. E. (1985) Nutrient Internal Cycling and the Trophic Regulation of Green Lake , Wisconsin, *Limnology and Oceanography*, 30(2), pp. 347–363.

Tang, Y. *et al.* (2017) Aquatic macrophytes can be used for wastewater polishing but not for purification in constructed wetlands, *Biogeosciences*, 14(4), pp. 755–766. doi: 10.5194/bg-14-755-2017.

Team, R. C. (2018) R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. Available at: https://www.r-project.org/.

USDA National Agricultural Statistics Service (2016) *CropsScape - Cropland Data Layer*. Available at: https://nassgeodata.gmu.edu/CropScape/ (Accessed: 27 June 2018).

USGS (2018) *National Water Information System: Mapper*.

Verheyen, D. *et al.* (2015) Dissolved phosphorus transport from soil to surface water in catchments with different land use, *Ambio*, 44(2), pp. 228–240. doi: 10.1007/s13280-014-0617-5.

Wang, L. *et al.* (2018) Effects of a Combined Biological Restoration Technology on Nitrogen and Phosphorus Removal from Eutrophic Water, *Polish Journal of Environmental Studies*, 27(5), pp. 2293–2301. doi: 10.15244/pjoes/77609.

Wang, L., Robertson, D. M. and Garrison, P. J. (2007) Linkages Between Nutrients and Assemblages of Macroinvertebrates and Fish in Wadeable Streams : Implication to Nutrient Criteria Development, *Environmental Management*, 39, pp. 194–212. doi: 10.1007/s00267-006-0135-8.

WDNR (2018) *Surface Water Integrated Monitoring System*. Available at: https://dnrx.wisconsin.gov/swims/login.jsp (Accessed: 5 May 2018).

Weber, M. J. and Brown, M. L. (2009) Effects of Common Carp on Aquatic Ecosystems 80 Years after 'Carp as a Dominant': Ecological Insights for Fisheries Management, *Reviews in Fisheries Science*, 17(4), pp. 524–537. doi: 10.1080/10641260903189243.

Wisconsin Department of Natural Resources (2002) *Wiscland 1.0*. Available at: https://dnr.wi.gov/maps/gis/datalandcover.html (Accessed: 11 July 2018).

Wisconsin Department of Natural Resources (2017) *Impaired Water - Roy Creek (Roy Creek)*. Available at: https://dnr.wi.gov/water/impairedDetail.aspx?key=11030 (Accessed: 3 August 2018).

Wisconsin Department of Natural Resources (2018a) *Impaired Water - Silver Creek (Silver Creek)*. Available at: https://dnr.wi.gov/water/impairedDetail.aspx?key=11028 (Accessed: 3 August 2018).

Wisconsin Department of Natural Resources (2018b) *Impaired Water - Unnamed (Wuerches*

Creek). Available at: https://dnr.wi.gov/water/impairedDetail.aspx?key=359163 (Accessed: 3 August 2018).

Zhang, W. *et al.* (2009) Evaluation of Two Langmuir Models for Phosphorus Sorption of Phosphorus-Enriched Soils in New York for Environmental Applications, *Soil Science*, 174(10), pp. 523–530. doi: 10.1097/SS.0b013e3181be9a4c.

Zhang, X. *et al.* (2017) Biomanipulation-induced reduction of sediment phosphorus release in a tropical shallow lake, *Hydrobiologia*. Springer International Publishing, 794(1), pp. 49–57. doi: 10.1007/s10750-016-3079-x.

Appendix A Monthly Precipitation Data

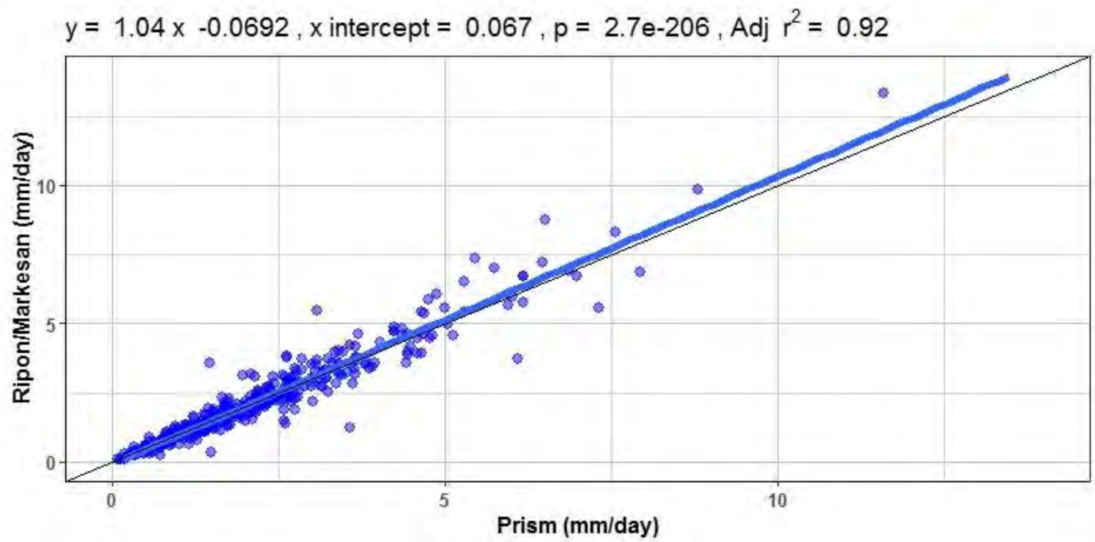

Figure A-1: Monthly average precipitation data at Ripon (1987 -2006) and Markesan (2007-2017) stations versus Prism (1987-2017) data. Equation for best fit linear trend also shown. PRISM (Parameter-elevation Relationships on Independent Slopes Model) data is for a 4km grid over SCE. Monthly data downloaded from prism.oregonstate.edu.

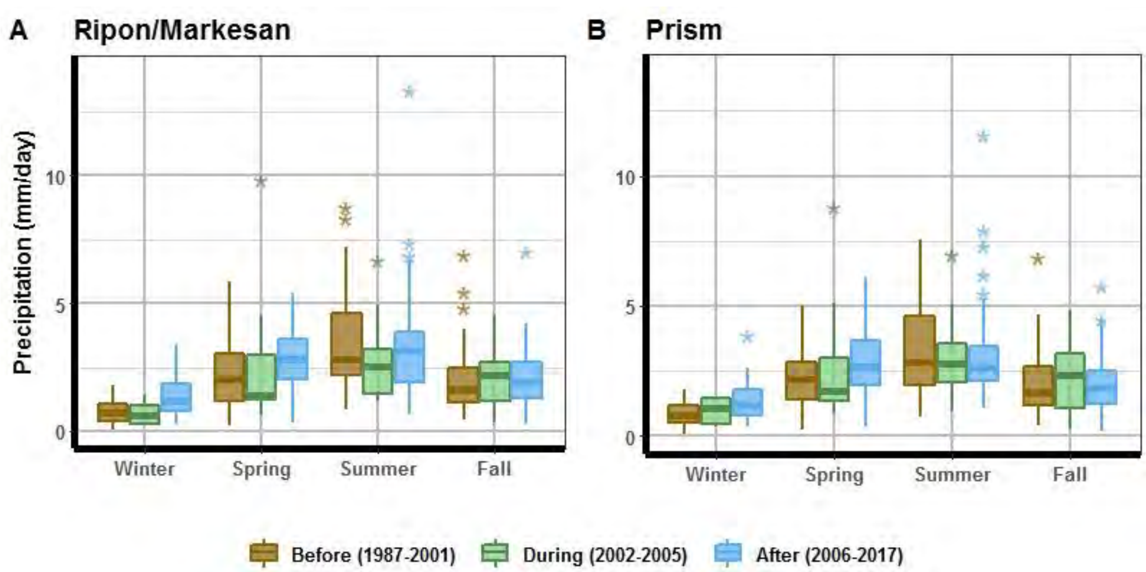

Figure A-2: Daily average precipitation for each month before (1987-2001), during (2002-2006) and after (2007-2017) SCE restoration from a phytoplankton to macrophyte-dominant state.(A) Monthly average data from Ripon station (187-2006) and Markesan station (2007-2017). (B) Monthly average data from Prism dataset. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

Table A-1: Difference between median observations of average monthly precipitation before and after the SCE restoration. Number of observations are also noted. P values shown for estimated means of dataset. Bolded values are significant, p <0.05. Precipitation from PRISM dataset.

	Restoration	**Winter**	**Spring**	**Summer**	**Fall**	**All Seasons**
Precipitation (mm/day)	**Before**	0.77	2.16	2.83	1.70	1.70
	After	1.16	2.69	2.61	1.83	2.17
	Difference (% Increase)	0.39 (51%)	0.53 (25%)	0.22 (8% Decrease)	0.13 (8%)	0.47 (28%)
	p value	**<0.001**	**0.050**	1.000	1.000	**<0.005**
n	**Before**	44	45	45	45	179
	During	12	12	12	12	48
	After	35	36	36	34	141

Appendix B ANCOVA model results

*Table B-1: ANCOVA model results for RWTP effluent, TP concentrations, and TP load. Precipitation was naturally log-transformed for use as a predictor. Significance of type III interactions shown. IS = interaction significant ($p < 0.05$); NS = considered but not significant ($p > 0.05$); *** = $p < 0.001$; ** = $p < 0.01$; * = $p < 0.05$; blank = not considered in model.*

Outcome		Effluent at RWTP (m^3/s)	Effluent TP Concentration at RWTP (mg/L)	Effluent TP Load at RWTP (kg/day)
Transformation		ln	ln	ln
Predictor	Season (Winter, Spring, Summer, Fall)	IS	IS	IS
	RWTP Upgrade (Before, After)	IS	IS	IS
	Season and RWTP upgrade interaction	NS	***	***
	Precipitation (mm/day)	IS	**	**
	Season and Precipitation (mm/day) interaction	***		
	RWTP Upgrade and Precipitation (mm/day) interaction	*		
	Effluent at RWTP (m^3/s)			***
Adjusted r^2		0.38	0.63	0.64

*Table B-2: ANCOVA model results for tends in monthly average precipitation and flow at the outlet of SCE, CTH A. Significance of type III interactions shown. IS = interaction significant ($p < 0.05$); NS = considered but not significant ($p > 0.05$); *** = $p < 0.001$; ** = $p < 0.01$; * = $p < 0.05$; blank = not considered in model.*

Outcome		**Precipitation (mm/day)**	**Flow at CTH A (m^3/s)**
Predictor	**Transformation**	ln	ln
Season (Winter, Spring, Summer, Fall)		IS	IS
SCE Restoration (Before, During, After)		IS	NS
Season and SCE Restoration Interaction		*	NS
Precipitation (mm/day)	ln		IS
Season and Precipitation (mm/day) Interaction			**
Effluent (m^3/s) at RWTP			IS
Effluent (m^3/s) at RWTP and Season Interaction	ln(Effluent)		**
Adjusted r^2		0.37	0.72

*Table B-3: ANCOVA results for multiple linear regression models to predict the effect of explanatory variables on tends in TP load and flow-weighted mean TP concentrations at the SCE outlet, CTH A. Significance of type III interactions shown. NS = considered but not significant (p > 0.05); *** = p < 0.001; ** = p < 0.01; * = p < 0.05; blank = not considered in model.*

Outcome		**TP Load at CTH A (kg/day)**	**TP at CTH A (mg/L)**
Predictor	**Transformation**	ln	ln
Season (Winter, Spring, Summer, Fall)		IS	IS
SCE Restoration (Before, During, After)		***	***
Season and SCE Restoration Interaction		NS	NS
RWTP Upgrade (Before, After)		NS	NS
TP at RWTP (mg/L)	ln		NS
TP Load (kg/day) from RWTP	ln	NS	
Precipitation (mm/day)	ln	IS	IS
Season and Precipitation (mm/day) Interaction	ln(Precipitation)	**	**
Flow at CTH A (m^3/s)	ln	IS	IS
Season and Flow at CTH A (m^3/s) Interaction	ln(Flow)	**	*
SS Loads at CTH A (metric tonnes/day)	ln	**	
SS at CTH A (mg/L)	ln		**
Adjusted r^2		0.86	0.57

*Table B-4: ANCOVA results for multiple linear regression models to predict the effect of explanatory variables on tends in monthly average TP load and flow-weighted mean TP concentrations at the inlet (Spaulding Rd) and outlet (CTH A) of SCE from December 2011 – September 2017. Significance of type III interactions shown. NS = considered but not significant (p > 0.05); *** = p < 0.001; ** = p < 0.01; * = p < 0.05; blank = not considered in model.*

Outcome		**TP Load at Spaulding Rd and CTH A (kg/day)**	**TP at Spaulding Rd and CTH A (mg/L)**	**TP Retention (kg/day)**
Predictor	**Transformation**	ln	ln	
Season (Winter, Spring, Summer, Fall)		IS	IS	NS
Site (Inlet, Outlet)		IS	IS	
Season and Site Interaction		NS	NS	
Year (factor)		***	**	
TP at RWTP (mg/L)	ln		*	
TP Load at RWTP (kg/day)	ln	NS		
TP at Spaulding Rd (kg/day)	ln			**
Precipitation (mm/day)	ln	IS	IS	*
Site and Precipitation (mm/day) Interaction	ln(Precipitation)	*	*	
Flow at Spaulding Rd (m^3/s)	ln	IS	NS	
Season and Flow at Spaulding Rd (m^3/s) Interaction	ln(Flow)	*		
Season and Precipitation (mm/day) Interaction	ln(Precipitation)	*	*	
Retention Time (days)				NS
Adjusted r^2		0.89	0.54	0.1

*Table B-5: ANCOVA results for multiple linear regression models to predict the effect of explanatory variables on monthly average TP removal efficiency within SCE. Significance of type III interactions shown. NS = considered but not significant (p > 0.05); *** = p < 0.001; ** = p < 0.01; * = p < 0.05; blank = not considered in model.*

Outcome		**TP Removal Efficiency (%)**
Predictor	**Transformation**	
Season (Winter, Spring, Summer, Fall)		NS
Precipitation (mm/day)	ln	***
Year		**
Flow at Spaulding Rd (m^3/s)	ln	NS
TP at Spaulding Rd (mg/L)	ln	*
Adjusted r^2		0.35

*Table B-6: SCE SS load and flow-weighted mean SS concentration ANCOVA results for multiple linear regression models. Significance of type III interactions shown. NS = considered but not significant (p > 0.05); *** = p < 0.001; ** = p < 0.01; * = p < 0.05; blank = not considered in model.*

Outcome		**SS Load at CTH A (metric tonnes/day)**	**SS at CTH A (mg/L)**
Predictor	**Transformation**	ln	ln
Season (Winter, Spring, Summer, Fall)		IS	IS
SCE Restoration (Before, During, After)		IS	IS
Season and SCE Restoration		***	***
RWTP Upgrade (Before, After)		NS	NS
Precipitation (mm/day)	ln	IS	IS
Season and Precipitation (mm/day) Interaction	ln(Precipitation)	***	***
Flow at CTH A (m^3/s)	ln	IS	IS
Season and Flow at CTH A (m^3/s) Interaction	ln(Flow)	*	*
Adjusted r^2		0.72	0.37

*Table B-7: Inlet and outlet SCE SS load and flow-weighted mean SS concentration ANCOVA results for multiple linear regression models. Significance of type III interactions shown. NS = considered but not significant (p > 0.05); *** = p < 0.001; ** = p < 0.01; * = p < 0.05; blank = not considered in model.*

Outcome		**SS Load at CTH A and Spaulding Rd (metric tonnes/day)**	**SS at CTH A and Spaulding Rd (mg/L)**	**SS Retention (metric tonnes/day)**
Predictor	**Transformation**	ln	ln	
Season (Winter, Spring, Summer, Fall)		IS	IS	NS
Site		*	**	
Site and Season Interaction		NS	NS	
Precipitation (mm/day)	ln	NS	NS	NS
Flow at Spaulding Rd (m^3/s)	ln	IS	IS	NS
Season and Flow at Spaulding Rd (m^3/s) Interaction	ln (Flow)	**	**	
SS Load at Spaulding Rd (metric tonnes/day)				**
Adjusted r^2		0.74	0.4	0.29

*Table B-8: ANCOVA results for multiple linear regression models for TP concentrations from samples collected in SCE. Significance of type III interactions shown. NS = considered but not significant (p > 0.05); *** = p < 0.001; ** = p < 0.01; * = p < 0.05; blank = not considered in model.*

Outcome		**TP Spaulding Rd and CTH A (mg/L)**	**TP four sites (mg/L)**	**TP Removal Efficiency (%)**
Predictor	**Transformation**	ln	ln	
Season (Winter, Spring, Summer, Fall)		IS	***	NS
Site (Inlet, Site 1, Site 2, Outlet)		IS	***	
Season and Site Interaction		**	NS	
Sampling Scheme (Period 1 or 2)		***		
TP at Spaulding Rd (mg/L)	ln			**
Year (factor)		IS	***	IS
Season and Year Interaction		***		
Daily Precipitation (None, low, lower middle, upper middle, high)		*	*	
Cumulative 3 Day Precipitation (None, low, lower middle, upper middle, high)				**
Flow at Spaulding Rd (m^3/s)	ln	IS	**	IS
Year and Flow at Spaulding Rd (m^3/s) Interaction	ln(Flow)	***		*
Adjusted r^2		0.45	0.48	0.38

*Table B-9: ANCOVA results for multiple linear regression models for TDP concentrations samples collected in SCE, for TDP:TP ratios of samples when SS was also measured, and for TDP removal efficiency from the SCE inlet to outlet. Significance of type III interactions shown. NS = considered but not significant (p > 0.05); *** = p < 0.001; ** = p < 0.01; * = p < 0.05; blank = not considered in model.*

Outcome		**TDP (mg/L)**	**TDP:TP**	**TDP Removal Efficiency (%)**
Predictor	**Transformation**	ln		
Season (Winter, Spring, Summer, Fall)		*	NS	NS
Site (Inlet, Outlet)		***	*	
Site and Season Interaction		NS	NS	
TP at Spaulding Rd (mg/L)	ln			
TP (mg/L)	ln	***	NS	
TDP:TP at Spaulding Rd				
TDP at Spaulding Rd (mg/L)	ln			NS
SS (mg/L)	ln		**	
Year (factor)		IS	IS	*
Cumulative 3 day Precipitation (No, low, lower middle, upper middle, high)		NS		
Cumulative 7 day Precipitation (No, low, lower middle, upper middle, high)			**	NS
Flow at Spaulding Rd (m^3/s)	ln	IS	IS	*
Year and Flow at Inlet (m^3/s) Interaction	ln(Flow)	**	**	
Adjusted r^2		0.72	0.57	0.43

*Table B-10: ANCOVA results for multiple linear regression models for SS concentration samples collected in SCE and for SS removal efficiency from the SCE inlet to outlet. Significance of type III interactions shown. NS = considered but not significant ($p > 0.05$); *** = $p < 0.001$; ** = $p < 0.01$; * = $p < 0.05$; blank = not considered in model.*

Outcome		**SS (mg/L)**	**SS Removal Efficiency (%)**
Predictor	**Transformation**	ln	
Season (Winter, Spring, Summer, Fall)		IS	IS
Site (Inlet, Outlet)		IS	
Season and Site Interaction		*	
Sampling Period (1 or 2)		IS	***
Sampling Period and Site Interaction		***	
Sampling Period and Season Interaction		*	
SS at Spaulding Rd (mg/L)	ln		***
Flow at Spaulding Rd (m^3/s)	ln	IS	IS
Season and Flow at Spaulding Rd (m^3/s) Interaction	ln(Flow)	*	**
Year		IS	
Year and Flow at Spaulding Rd (m^3/s) Interaction	ln(Flow)	***	
Cumulative 7 day Precipitation (No, low, lower middle, upper middle, high)		**	NS
Adjusted r^2		0.32	0.49

*Table B-11: ANCOVA results for multiple linear regression models for TN concentration of samples collected in SCE and for TN removal efficiency from the SCE inlet to outlet. Significance of type III interactions shown. NS = considered but not significant ($p > 0.05$); *** = $p < 0.001$; ** = $p < 0.01$; * = $p < 0.05$; blank = not considered in model.*

Outcome		**TN (mg/L)**	**TN Removal Efficiency (%)**	**TN:TP ratio**
Predictor	**Transformation**			ln
Season (Winter, Spring, Summer, Fall)		IS	*	IS
Site (Inlet, Site 1, Site 2, Outlet)		IS		IS
Site and Season Interaction		*		
TP (mg/L)	ln	***		IS
TN (mg/L) at Spaulding Rd			*	
Daily Precipitation (mm)			**	
Cumulative Precipitation over prior 3 days (No, low, lower middle, upper middle, high)		NS		
Cumulative Precipitation over prior 3 days (mm)			*	**
Flow at Spaulding Rd (m^3/s)	ln	IS	NS	IS
Site and Flow at Inlet (m^3/s) Interaction	ln(Flow)	***		***
Season and TP (mg/L) Interaction	ln(TP)			*
Adjusted r^2		0.66	0.8	0.82

Appendix C CKM and SCE Comparison ANCOVA Tables

*Table C-1: ANCOVA model results for monthly average precipitation, flow, and retention time. Significance of type III interactions shown. IS = interaction significant (p < 0.05); NS = considered but not significant (p > 0.05); *** = p < 0.001; ** = p < 0.01; * = p < 0.05; blank = not considered in model.*

Outcome		**Precipitation (mm/day)**	**Flow (m^3/s)**	**Retention Time (day)**
Predictor	**Transformation**	ln	ln	ln
Season (Winter, Spring, Summer, Fall)		***	IS	IS
Date		NS	IS	IS
Precipitation (mm/day)	ln		***	***
Site			IS	IS
Season and Date Interaction			**	**
Season and Site Interaction			*	*
Adjusted r^2		0.29	0.74	0.82

*Table C-2: ANCOVA model results for monthly average TP load and flow-weighted concentration. Significance of type III interactions shown. IS = interaction significant (p < 0.05); NS = considered but not significant (p > 0.05); *** = p < 0.001; ** = p < 0.01; * = p < 0.05; blank = not considered in model.*

Outcome		**TP Load (kg/day)**	**TP (mg/L)**
Predictor	**Transformation**	ln	ln
Season (Winter, Spring, Summer, Fall)		**	**
Date		NS	NS
Precipitation (mm/day)	ln	**	*
Site		IS	IS
Season and Site Interaction		NS	NS
Flow (m^3/s)	ln	***	NS
SS Load (metric tonnes/day)	ln	IS	
Site and SS Load (metric tonnes/day) Interaction	ln	***	
SS (mg/L)	ln		IS
Site and SS (mg/L) Interaction	ln		***
Adjusted r^2		0.88	0.71

*Table C-3: ANCOVA model results for monthly average SS load and flow-weighted concentration. Significance of type III interactions shown. IS = interaction significant ($p < 0.05$); NS = considered but not significant ($p > 0.05$); *** = $p < 0.001$; ** = $p < 0.01$; * = $p < 0.05$; blank = not considered in model.*

Outcome		**SS Load (metric tonnes/day)**	**SS (mg/L)**
Predictor	**Transformation**	ln	ln
Season (Winter, Spring, Summer, Fall)		IS	IS
Date		NS	*
Site		IS	IS
Season and Site Interaction		**	***
Flow (m^3/s)	ln	***	
Adjusted r^2		0.76	0.51

Appendix D Spatial Visualization of Sample Concentrations

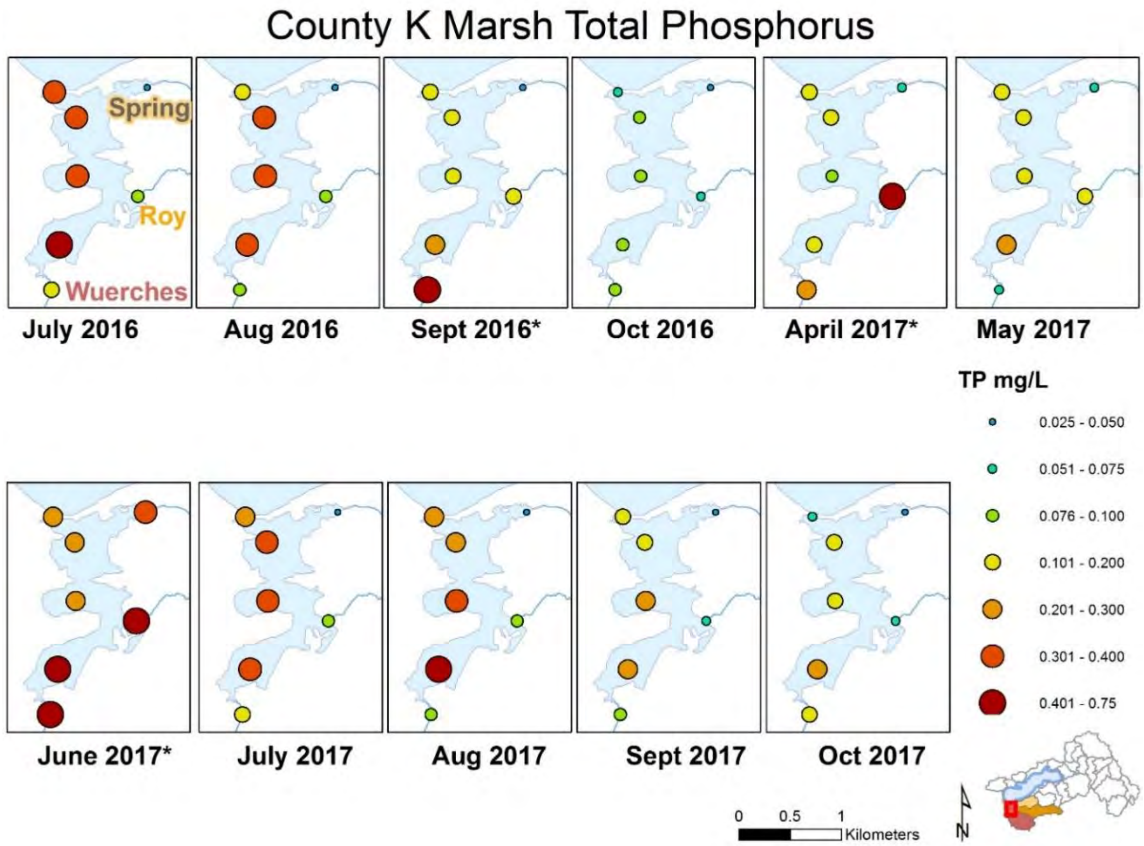

*Figure D-1: Grab sample TP concentrations in CKM on 11 sampling days. * = sampling following a storm event*

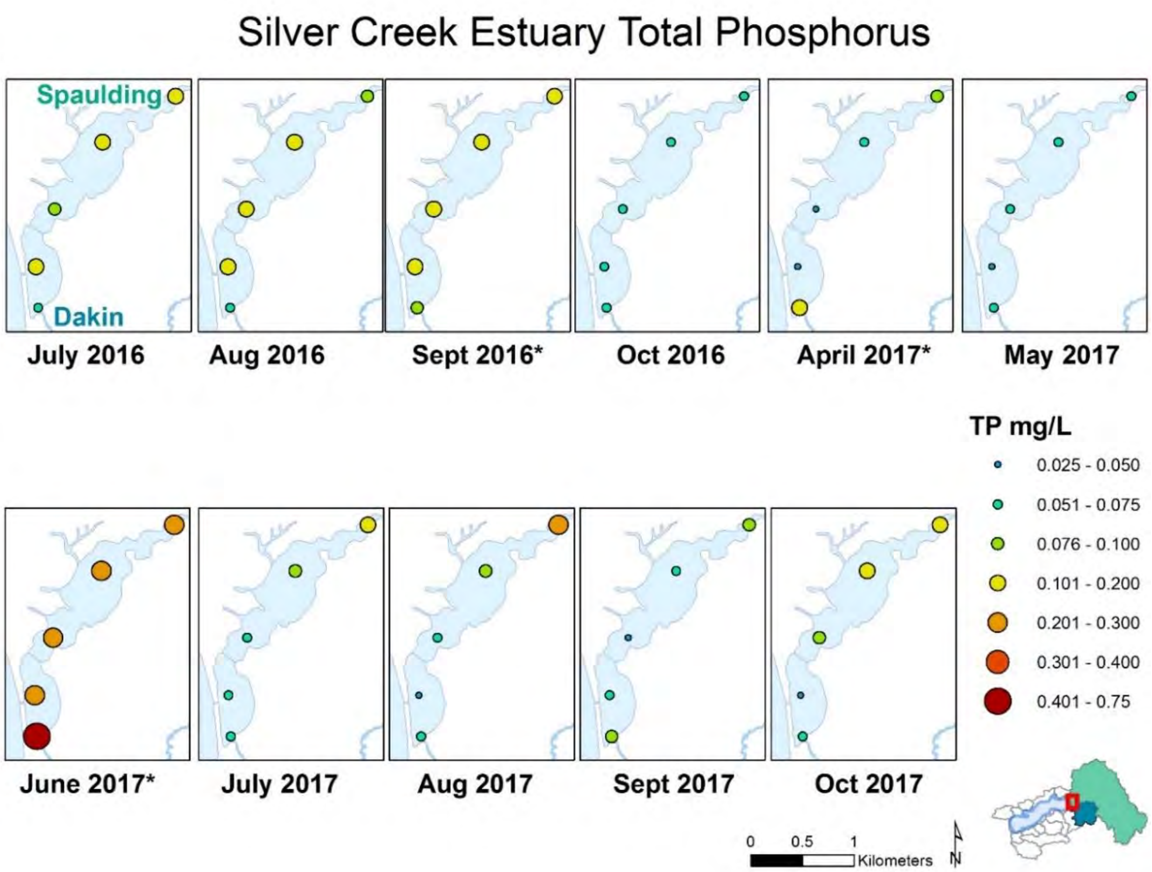

*Figure D-2: Grab sample TP concentrations in SCE on 11 sampling days. * = sampling following a storm event*

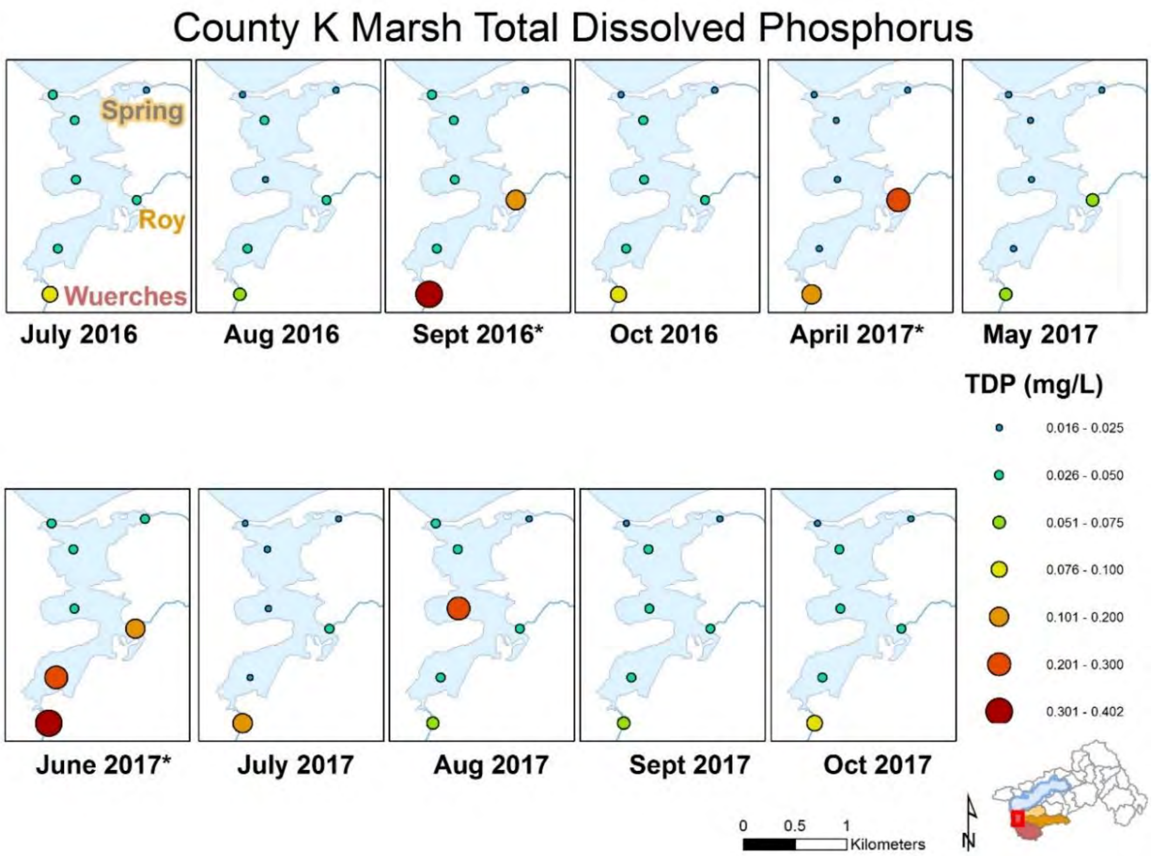

*Figure D-3: Grab sample TDP concentrations in CKM on 11 sampling days. * = sampling following a storm event*

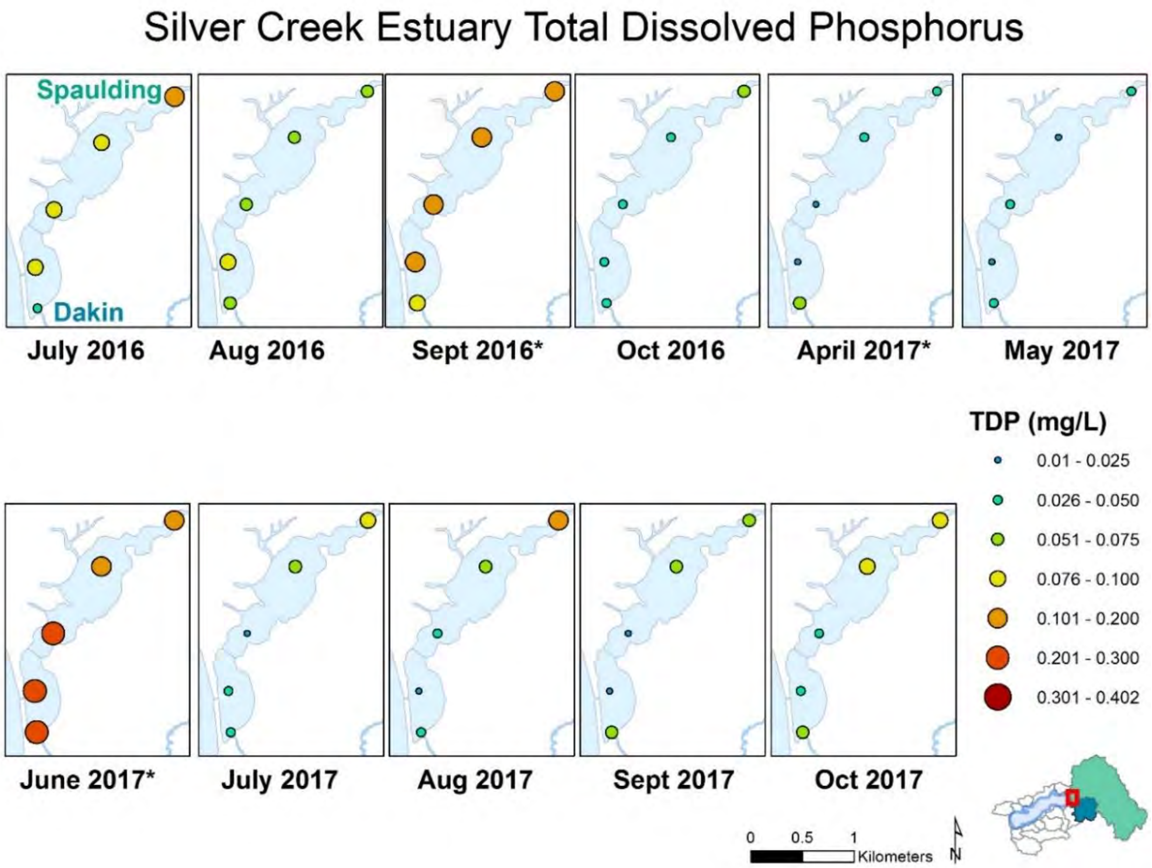

*Figure D-4: Grab sample TDP concentrations in SCE on 11 sampling days. * = sampling following a storm event*

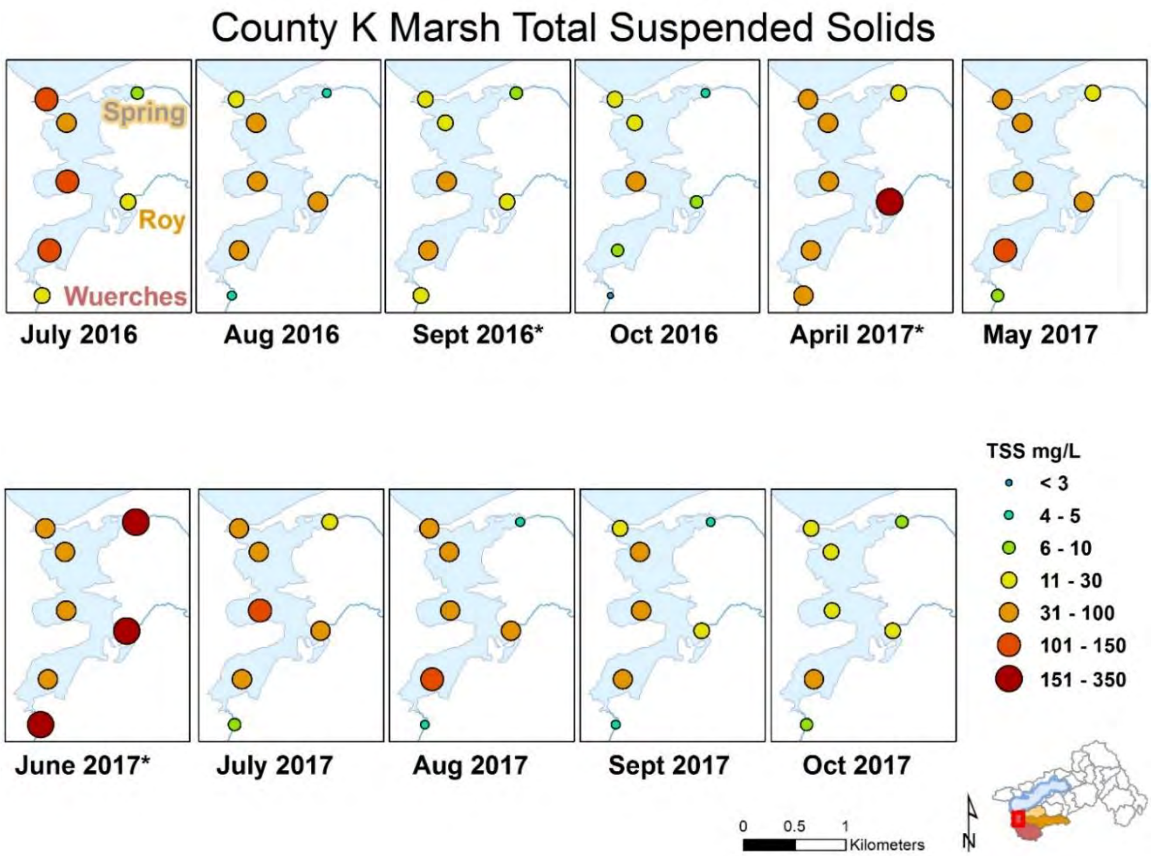

*Figure D-5: Grab sample TSS concentrations in CKM on 11 sampling days. * = sampling following a storm event*

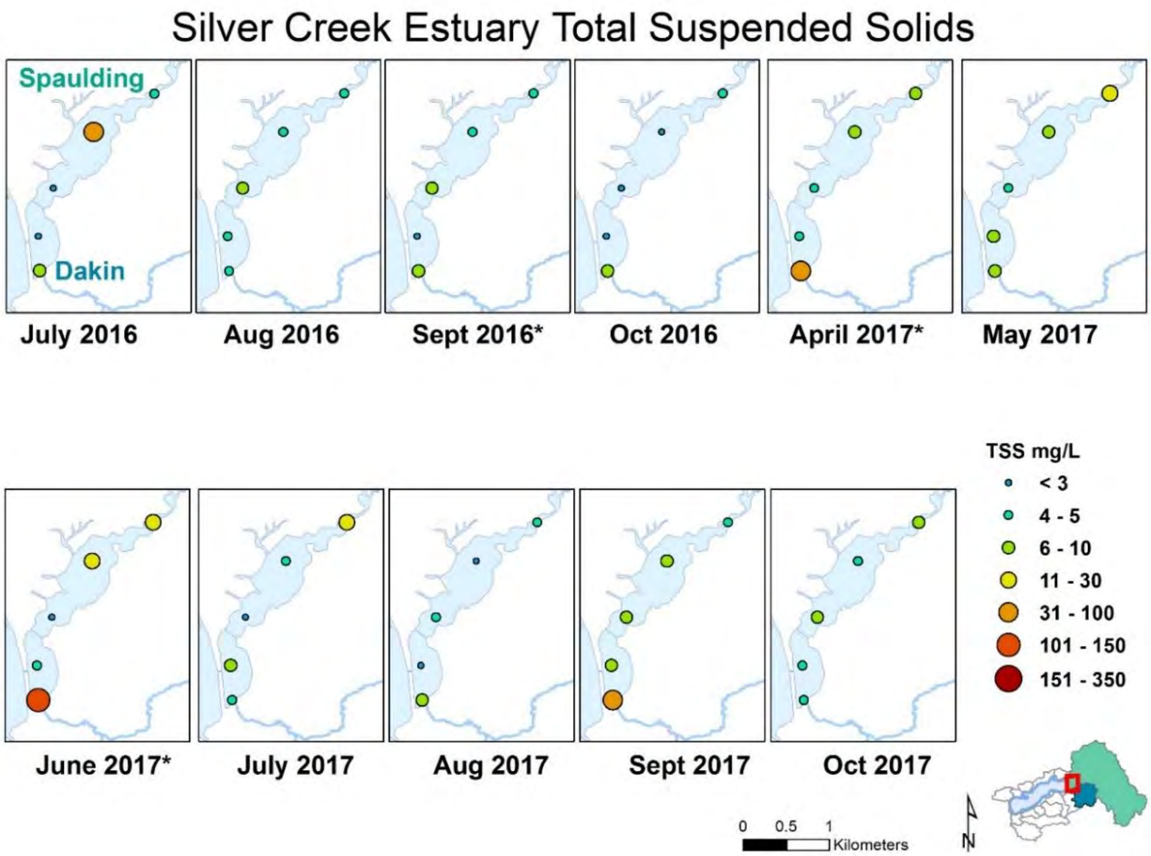

*Figure D-6: Grab sample TSS concentrations in SCE on 11 sampling days. * = sampling following a storm event*

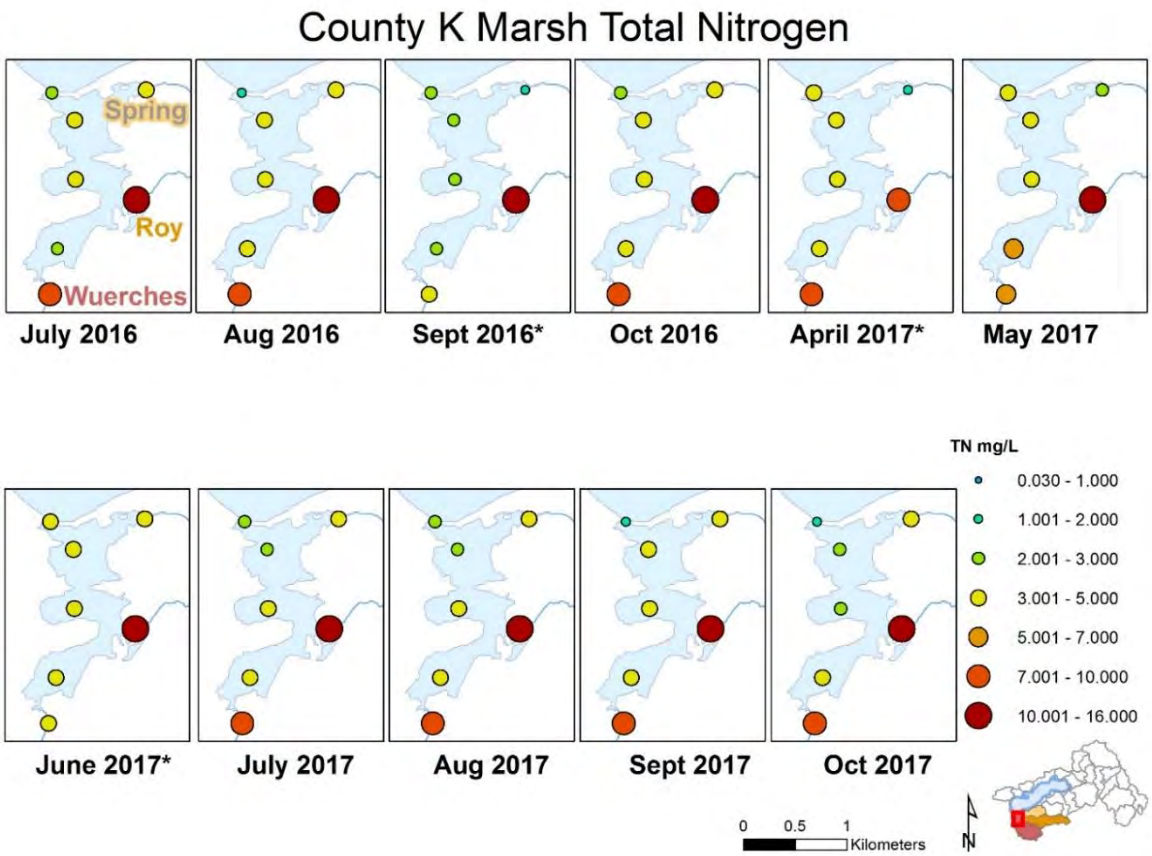

*Figure D-7: Grab sample TN concentrations in CKM on 11 sampling days. * = sampling following a storm event*

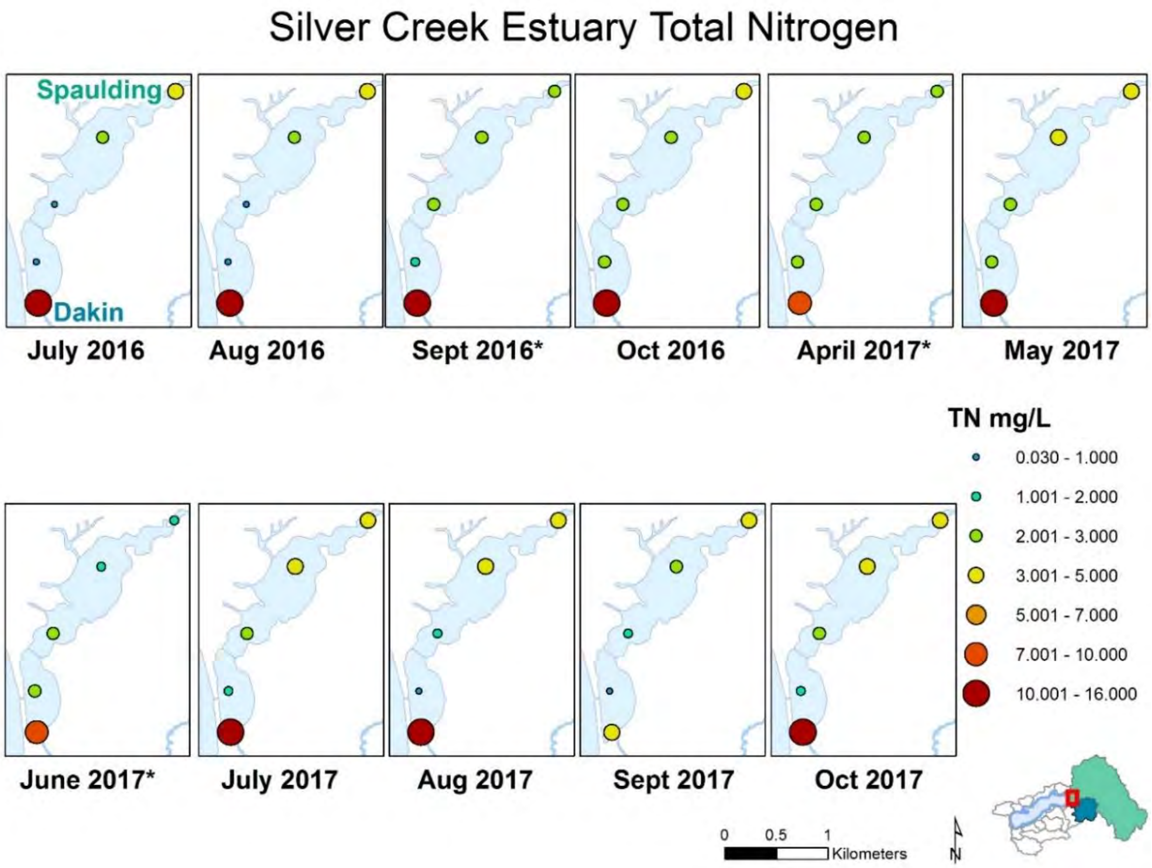

*Figure D-8: Grab sample TN concentrations in SCE on 11 sampling days. * = sampling following a storm event*

Appendix E Summary of Sample Concentrations

Table E-1: Minimum, maximum and median concentrations of TP, TDP, TN, and TSS from grab samples collected at inlet streams (separate) and interior sites (grouped) of CKM and SCE.

Para-meter		Roy	Spring	Wuerches	Interior CKM	Dakin	Spaulding Rd	Interior SCE
TP (mg/L)	Minimum	0.056	0.025	0.075	0.056	0.051	0.069	0.040
	Max-imum	0.706	0.356	0.709	0.505	0.456	0.205	0.265
	Median	0.082	0.030	0.101	0.207	0.063	0.119	0.074
TDP (mg/L)	Minimum	0.037	0.016	0.052	0.016	0.033	0.036	0.014
	Max-imum	0.230	0.042	0.402	0.296	0.221	0.170	0.247
	Median	0.044	0.020	0.080	0.029	0.052	0.095	0.043
TSS (mg/L)	Minimum	6.40	3.60	2.20	8.40	4.20	2.80	< 2
	Max-imum	348.00	252.00	312.00	142.00	149.00	16.80	30.90
	Median	30.40	8.60	8.00	59.75	7.20	4.80	4.29
TN (mg/L)	Minimum	9.300	1.70	4.45	1.27	4.01	1.66	0.71
	Max-imum	15.700	4.51	9.57	5.64	14.40	4.60	3.78
	Median	13.300	3.83	7.29	3.37	11.60	3.48	2.32
n		11	11	11	44	11	11	33

Table E-2: Minimum, maximum and median concentrations of TP, TDP, TN, and TSS from grab samples collected at inlets (grouped) and interior sites (separate) of CKM and SCE.

Wetland	Parameter		Inlet	Site 1	Site 2	Site3	Outlet
CKM	TP (mg/L)	Minimum	0.025	0.099	0.098	0.086	0.056
		Maximum	0.709	0.505	0.393	0.338	0.345
		Median	0.080	0.253	0.206	0.197	0.132
	TDP (mg/L)	Minimum	0.016	0.018	0.017	0.019	0.016
		Maximum	0.402	0.296	0.225	0.038	0.037
		Median	0.044	0.033	0.033	0.032	0.021
	TSS (mg/L)	Minimum	2.2	8.4	30	11.8	11.6
		Maximum	348	134	142	93	108
		Median	11.8	73.3	63.6	56	57
	TN (mg/L)	Minimum	1.7	2.77	2.63	2.22	1.27
		Maximum	15.7	5.64	4.96	4.74	4.38
		Median	7.29	3.6	3.41	3.18	2.14
	n		33	11	11	11	11
SCE	TP (mg/L)	Minimum	0.051	0.051	0.043	0.040	
		Maximum	0.456	0.212	0.254	0.265	
		Median	0.088	0.099	0.062	0.054	
	TDP (mg/L)	Minimum	0.033	0.023	0.016	0.014	
		Maximum	0.221	0.174	0.233	0.247	
		Median	0.054	0.065	0.035	0.029	
	TSS (mg/L)	Minimum	2.8	< 2	< 2	< 2	
		Maximum	149	30.9	7.14	8.64	
		Median	5.6	4.6	4.67	3	
	TN (mg/L)	Minimum	1.66	1.69	0.711	0.847	
		Maximum	14.4	3.78	2.95	2.65	
		Median	4.37	2.64	2.24	1.49	
	n		22	11	11	11	

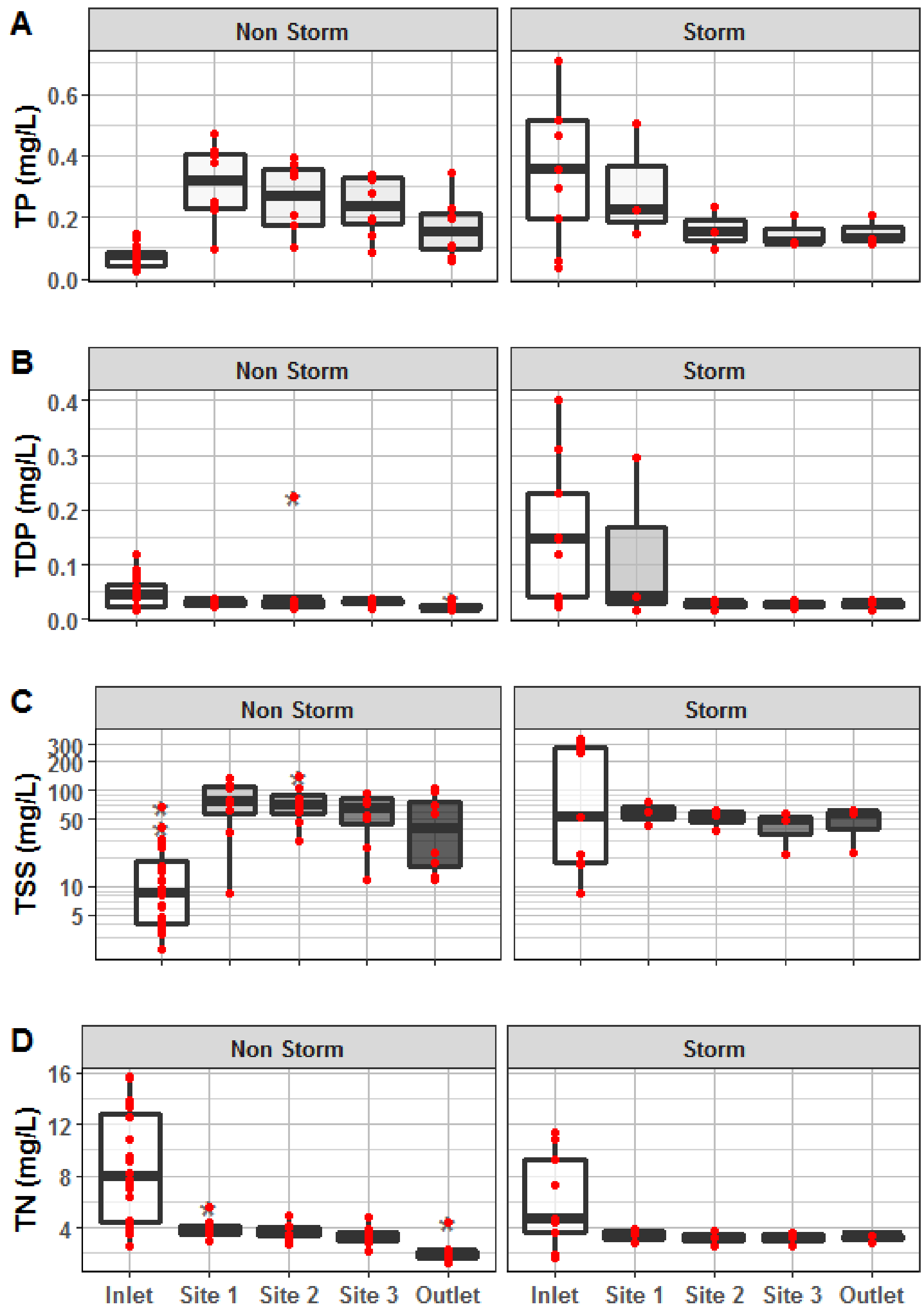

Figure E-1: Concentrations for (A) TP, (B) TDP, (C) TSS, and (D) TN at CKM inlet sites (grouped) and interior sites (separate). Sampling events following 60 mm of precipitation within the preceding 7 days are separated as storm events. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

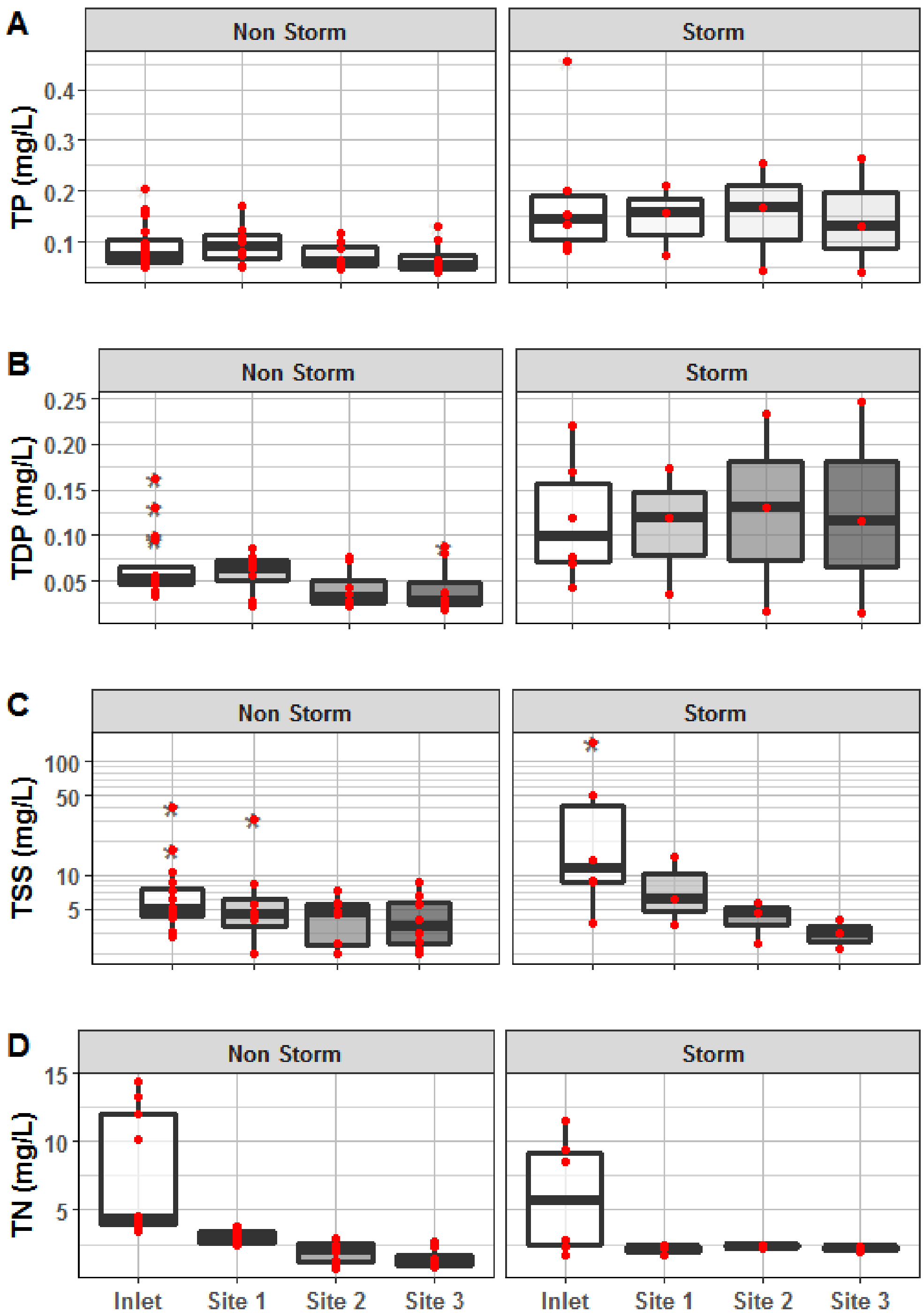

Figure E-2: Concentrations for (A) TP, (B) TDP, (C) TSS, and (D) TN at SCE inlet sites (grouped) and interior sites (separate). Sampling events following 60 mm of precipitation within the preceding 7 days are separated as storm events. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

Electric Conductivity and pH

Electric conductivity (EC) and pH were measured at all sites from June – October 2017. The EC ranged from 345 – 1001 µS/m. The lowest EC levels observed followed the precipitation event in June 2017 (range 345 – 682 µS/m). The EC at interior sites of CKM (median 482 µS/m) was generally lower than at the inlet sites (median 741 µS/m). The EC within SCE (median 794 µS/m) was higher than within CKM and similar to SCE inlet conditions (median 755 µS/m; **Error! Reference source not found.**).

Overall, pH ranged from 7.1 – 8.6, and the lowest pH conditions were observed following the June precipitation event (7.9 – 7.1). Similar to EC, the difference between inlet and interior sites was greater in CKM (inlet median = 7.7, interior median = 8.2) than in SCE (inlet median = 7.9, interior median = 8.0).

Table E-3: Median EC and pH data for inlet and interior sites of the two marshes. Data collected monthly June – October 2017 at all 12 sites.

Parameter	Marsh	Inlet	Interior
Median EC (µS/m)	CKM	741	482
	SCE	755	794
Median pH (S.U.)	CKM	7.67	8.17
	SCE	7.88	7.99

Appendix F Marsh Storage for each Sampling Event

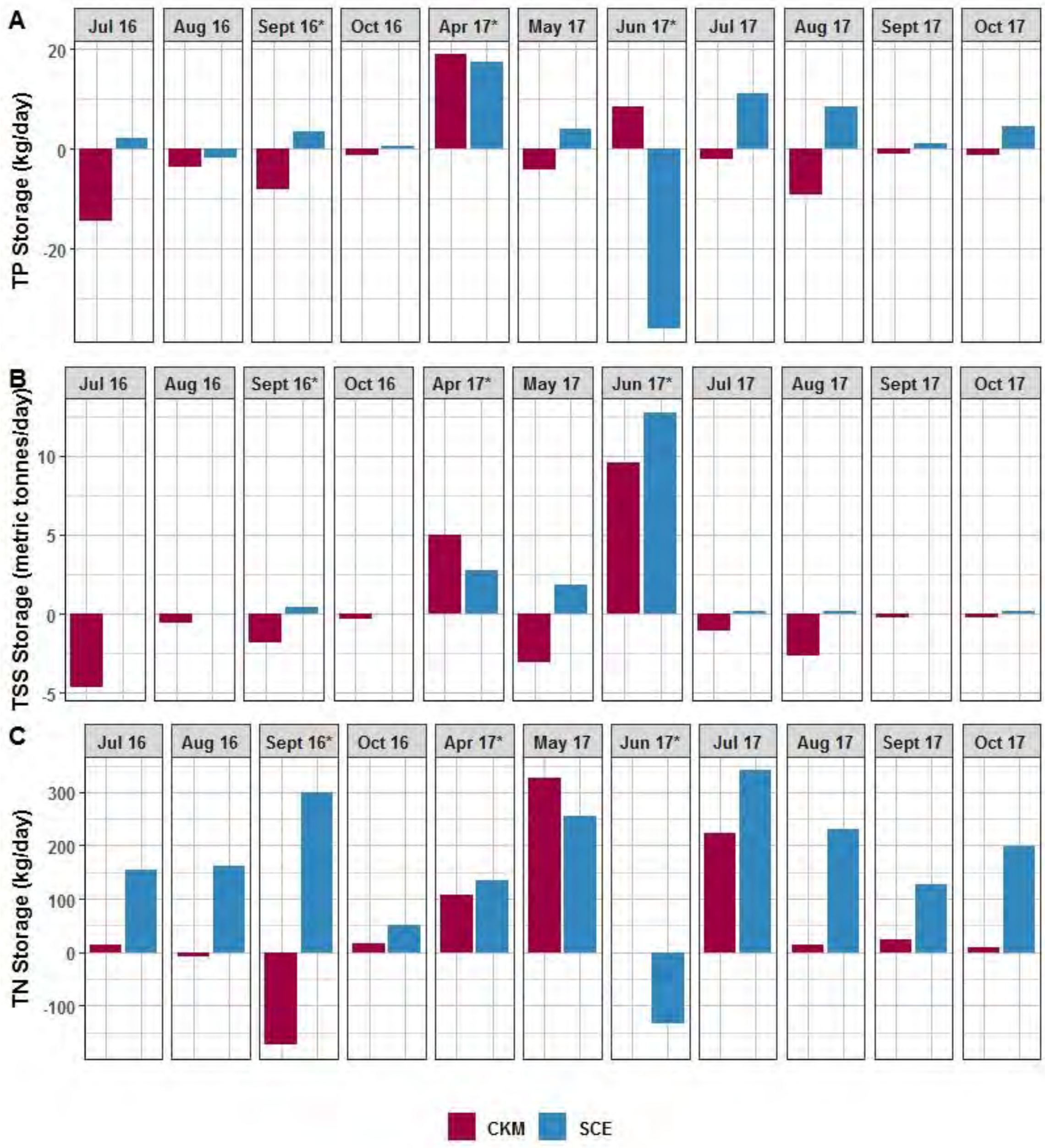

*Figure F-1: Daily storage of (A) TP, (B) TSS, and (C) TN on sampling days in both marshes. Storage was calculated by subtracting outlet loads from inlet loads. Positive storage indicates that the marsh retained nutrients on that day. If storage is negative, the marsh was a source of nutrients on that day. * Sampling occurred after a storm event (>60 mm of precipitation over the preceding 7 days).*

Appendix G Monthly Yields for SCE and CKM

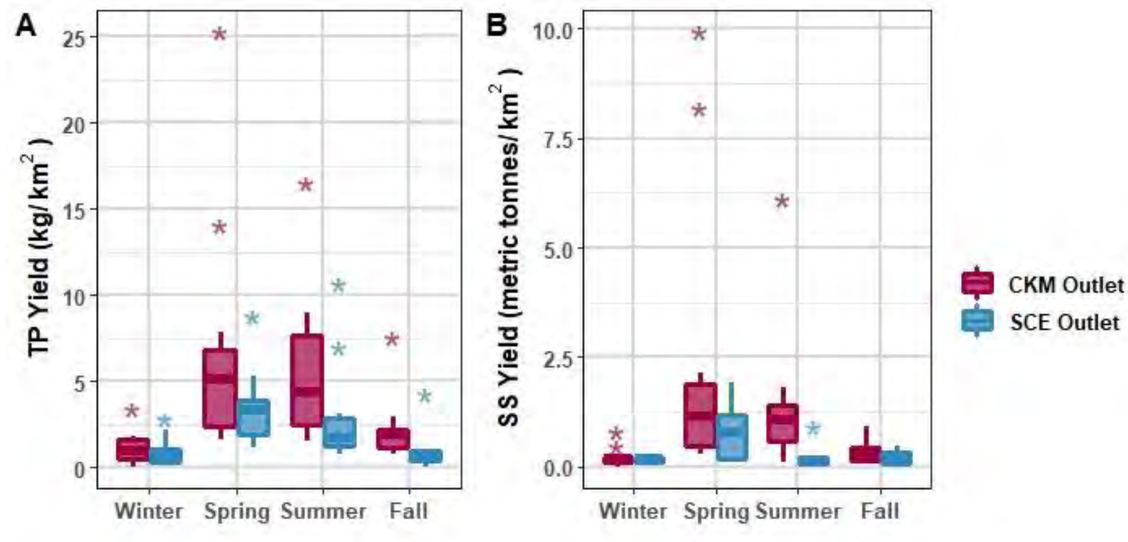

Figure G-1: (A) TP yield as total monthly TP load per square kilometer for the watersheds of CKM and SCE. TP yields are separated by season. (B) SS yield as total monthly SS load per square kilometer for the watersheds of CKM and SCE. The center line of each box is the median. The boundaries are the 25th and 75th percentiles or interquartile range (IQR). The lines extend to include data within 1.5 x IQR and remaining data are plotted individually.

Table G-1: Seasonal median TP and SS yields for CKM and SCE outlets. Data from October 2012 – August 2017. CKM:SCE ratio is the ratio of the two marsh medians. Significant differences between modeled means of the two marshes ($p < 0.05$) are bolded. ^ CKM was significantly higher.

		Winter	Spring	Summer	Fall	All Seasons
Median Monthly TP Yield (kg/km^2)	**CKM Outlet**	0.96	5.18	4.39	1.95	2.12
	SCE Outlet	0.68	3.36	1.71	0.48	1.31
	CKM:SCE Ratio	1.41	1.54	2.57	4.06	1.61
	p value	**<0.001^**	**<0.001^**	**<0.001^**	**<0.001^**	**<0.001^**
Median Monthly SS Yield (metric tonnes/km^2)	**CKM Outlet**	0.16	1.13	1.07	0.24	0.47
	SCE Outlet	0.16	0.79	0.13	0.20	0.19
	CKM:SCE Ratio	1.00	1.44	8.42	1.19	2.46
	p value	**<0.001^**	**<0.001^**	**<0.001^**	**<0.001^**	**<0.001^**