
Image credit: Fox-Wolf Watershed Alliance

Winnebago Pool Lakes
Nutrient Technical Support

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PREPARED FOR

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EXECUTIVE SUMMARY

This technical assistance project supports the state of Wisconsin with nutrient reduction and implementation planning specific to understanding the role of nutrients and aquatic vegetation in the Winnebago Pool Lakes and determining the potential for restoration activities to improve water quality and reduce harmful algal blooms. The purpose of this project is to provide scientific basis for comprehensive lake system management to reduce harmful algal blooms in Lake Winnebago. The focus of this project is a conceptual model of the lake system that describes the drivers of algal and aquatic macrophyte dynamics along with an assessment of available computer modeling approaches that could be used to simulate changes in management and water quality in the Winnebago Pool Lakes.

Multiple disturbances have acted on Lake Winnebago and the Upper Pool Lakes and have led them to being in a persistent, turbid, phytoplankton-dominated state. Algal and aquatic macrophyte dynamics in the Winnebago Pool Lakes are driven by a combination of altered water level regime, wind and wave action, nutrient loading, benthivorous fish, and other physical disturbances (Figure ES-1):

- Altered water level regime: The United States Army Corps of Engineers (USACE) manages water levels in the Winnebago Pool Lakes to balance the needs of multiple users of the watershed’s water resources. However, this management strategy has disrupted the natural water levels of the lakes, both in terms of lake depth and the timing and rate of water level change (Figure ES-2), affecting macrophyte establishment and survival (Figure ES-3):
    - Water levels are currently drawn down over the winter to provide capacity to hold spring flood waters. When water levels are low, the lakes are prone to wave action, ice scour, and freezing of sediments, which damage plant roots and rhizomes. Natural water levels would have been higher in the late winter.
    - Water levels are currently raised gradually through the early spring, and Lake Winnebago is refilled after ice out, which typically occurs at the beginning of April. During wet seasons, there is a greater potential for flooding and damage to aquatic vegetation due to deeper water and rapid increases in water level. The high water levels reduce light availability for rooted macrophytes. Natural water levels would have peaked from spring snowmelt and then gradually decreased.
    - Water levels currently peak in June and are maintained relatively constant throughout the summer to support recreation. Natural water levels would have gradually declined over the summer, with minimum water levels typically occurring in August–September.
    - Water levels are lowered gradually in the fall to a winter target level. If large changes in water level occur during this period, vegetation can be damaged. Natural water levels would have gradually increased over the fall after the late summer water level minimum.

  While the water levels across the Winnebago Pool Lakes are on average deeper than historical water levels, the timing and rate of change of water level fluctuations appears to have more of an effect on plant establishment and survival than just water level depth. This is evidenced by the lack of aquatic macrophytes in the Upper Pool Lakes, where water depths are shallow enough to support aquatic vegetation, yet macrophytes do not exist where one would expect from water depth alone.
- Nutrient loads. High phosphorus loading is from both external and internal sources; internal loading represents approximately 50 percent of the growing season load to Lake Winnebago (Dale Robertson, USGS, personal communication). In-depth analysis of in-lake nutrients is being completed as part of TMDL development; this analysis was not available for this project. Because of the poor quality of the aquatic macrophyte habitat in the lake, primary production tends to be in the form of algal growth in the water column (i.e., phytoplankton) as opposed to rooted plants and associated periphyton. Watershed nitrate concentrations have increased in recent years, although the link between nitrogen concentrations and cyanobacteria in the Winnebago Pool Lakes is not known. Whereas phosphorus must be reduced to

lower algal and cyanobacteria growth and shift the lake back to the clear water phase, nitrogen loading can influence algal biomass and community composition on shorter, ecologically-meaningful time scales.

- Wind and wave action: Lake Winnebago and the Upper Pool Lakes are shallow lakes with a relatively large surface area. Wind energy has the potential to disturb and prevent re-establishment of aquatic macrophytes (Figure ES-3). Wind energy also disturbs lake sediments, leading to sediment resuspension and release of phosphorus.
- Benthivorous fish can directly damage rooted macrophytes and disturb bottom sediments, leading to nutrient release from the sediments.
- Other physical disturbances such as motorboats and snowmobiles can also directly damage rooted macrophytes and disturb bottom sediments, leading to nutrient release from the sediments. The extent to which these physical disturbances drive algal and aquatic macrophyte dynamics in the Winnebago Pool Lakes is unknown, although it is assumed that these disturbances play a relatively minor role relative to the other stressors.

The above stressors maintain the lake in a stable, turbid, phytoplankton-dominated state. The relative balance of nitrogen and phosphorus availability affects the algal species assemblage. When phosphorus concentrations are low relative to nitrogen, which typically occurs in the spring and early summer in Lake Winnebago, primary production in the system is likely limited by phosphorus. As phosphorus increases relative to nitrogen (lowered N:P), co-limitation by both nitrogen and phosphorus is likely. However, phosphorus is typically the limiting nutrient for cyanobacteria growth and production of cyanotoxins in lakes. Ultimately, phosphorus must be reduced to control algal growth (Schindler 2012) and shift the lake back to the clear water phase. Increases in light availability (e.g., through water level management) without reductions in phosphorus availability could lead to increases in cyanobacteria biomass and cyanotoxin production.

The low abundance of aquatic macrophytes decreases sediment stability, making the sediments more susceptible to disturbance by wind and wave action, benthivorous fish such as carp, and other physical disturbances. Nutrients are released from the sediments to the water column, further fueling algal growth. The high concentrations of algae in the water column increase the turbidity of the water and decrease light availability. Lower levels of light limit growth of rooted macrophytes; high turbidity in the lake is a result of sediment inputs from the watershed, wind and wave resuspension of bottom sediments, and high concentrations of phytoplankton. The timing and rate of change of water level fluctuations also limit plant establishment and survival. The multiple stressors reinforce the turbid state, making it difficult to shift the system to the clear-water, macrophyte-dominated state. To better understand the ecological interactions in the lake and to enhance the conceptual model, data for all of the primary ecological components (e.g., macrophytes, wind and wave action, algal growth, nutrient availability, water levels, etc.) are needed from the same time period. The existing monitoring program on the Winnebago Pool Lakes should be expanded to include additional data collection.

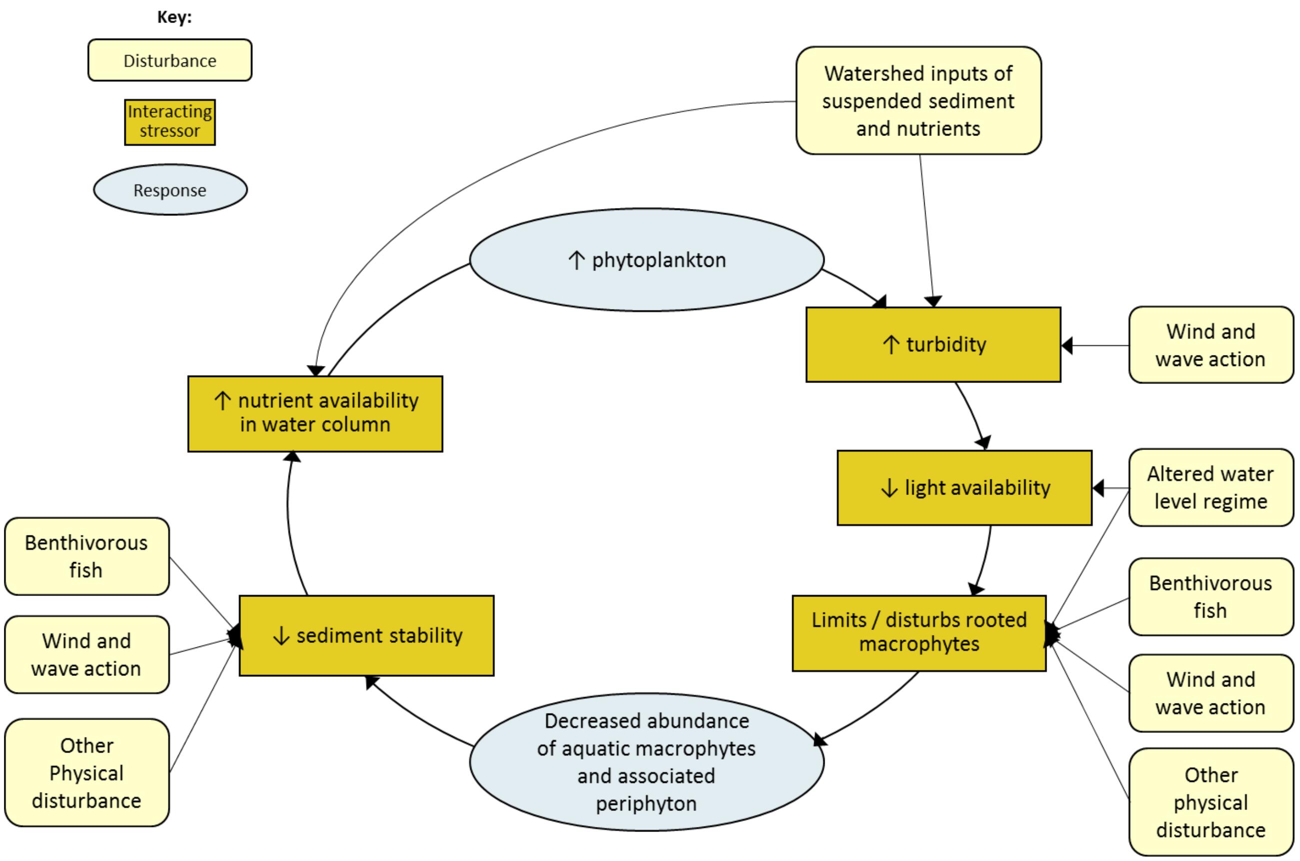

Figure ES-1. Conceptual model of algae and aquatic macrophytes in the Winnebago Pool Lakes.

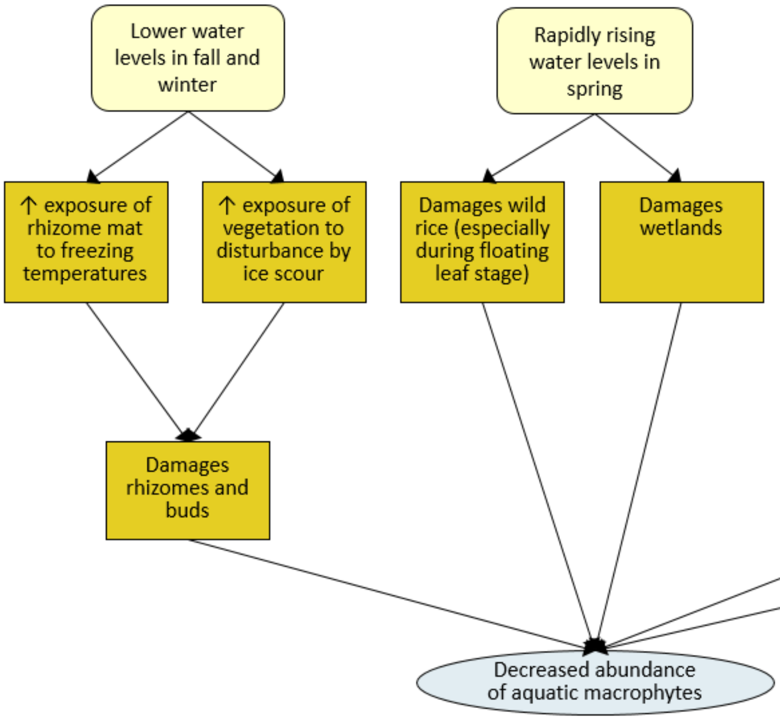

Figure ES-2. Altered water level regime component of conceptual model of algae and aquatic macrophytes in the Winnebago Pool Lakes.

See key in Figure ES-1

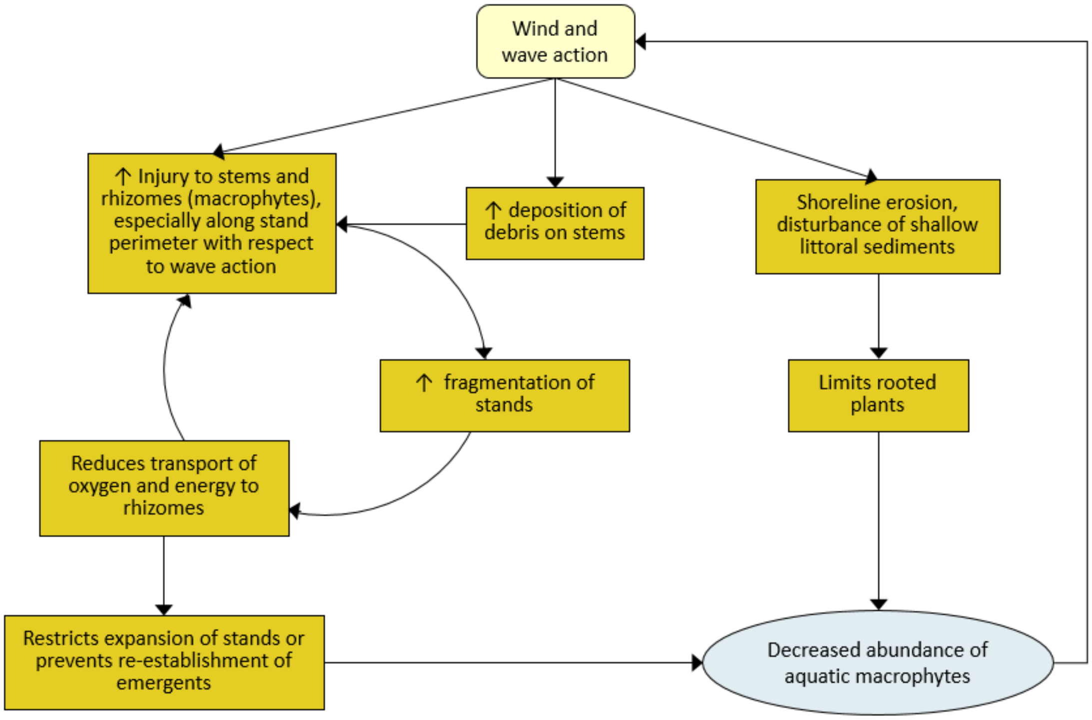

Figure ES-3. Wind and wave action component of conceptual model of algae and aquatic macrophytes in the Winnebago Pool Lakes.

See key in Figure ES-1

A computer model to further evaluate the Winnebago Pool Lakes can be a valuable tool to understanding the effect that management actions may have on water quality. A well-chosen computer model should have the ability to simulate the interactions and responses identified above in the conceptual model. A single model does not have to contain all of the required elements; multiple models can often be linked to provide the needed inputs and responses. However, PCLake, as described by Janse (2005), is the only model that has some proven track record in addressing stable state transitions in shallow lakes. As an alternate, a linked and customized Water Quality Analysis Simulation Program (WASP) and Environmental Fluid Dynamics Code (EFDC) model could be used to evaluate ecosystem conditions under both clear and turbid states that could form the basis for an external analysis of the likelihood of shift back to a clear state. Application of either model will require a comprehensive data collection effort.

The goal of the management options is to restore aquatic macrophytes in the Winnebago Pool Lakes to shift the lakes from a turbid state to a clear water state. Whereas this report focuses on in-lake restoration alternatives, reduction of external nutrient loads is often a prerequisite for a successful shift back to a clear state. External nutrient loading is often a “forward switch” in shallow lakes, shifting the lake from a clear water state to a turbid state, or maintaining the lake in a turbid state. To reverse the switch to a clear state, a much lower phosphorus concentration is needed, and therefore external nutrient reductions should be considered the highest priority. However, activities that address internal loading of nutrients can be conducted concurrently to address internal phosphorus reserved in the sediments. Restoration alternatives include altering the water level regime and reducing nutrient loads, wind and wave action, and benthivorous fish.

1 INTRODUCTION

1.1 PROJECT PURPOSE

This technical assistance project supports the state of Wisconsin with nutrient reduction and implementation planning specific to understanding the role of nutrients and aquatic vegetation in the Winnebago Pool Lakes and determining the potential for restoration activities to improve water quality and reduce harmful algal blooms. The purpose of this project is to provide scientific basis for comprehensive lake system management to reduce harmful algal blooms in Lake Winnebago. The focus of this project is a conceptual model of the lake system that describes the drivers of algal and aquatic macrophyte dynamics along with an assessment of available computer modeling approaches that could be used to simulate changes in management and water quality in the Winnebago Pool Lakes.

A parallel and complementary project is under way by the Wisconsin Department of Natural Resources (WDNR) and partners to develop phosphorus total maximum daily loads (TMDLs) for the Winnebago Pool Lakes. Because the TMDL study is evaluating watershed pollutant loading to the Winnebago Pool Lakes, the current report does not focus on watershed processes or pollutant sources.

1.2 LAKE WINNEBAGO POOL LAKES

Lake Winnebago, located in Fond du Lac, Winnebago, and Calumet counties, is the largest inland lake in Wisconsin spanning 131,939 acres (Figure 1). The lake is shallow for its surface area, with a maximum depth of only 21 feet. Lake Winnebago separates the upper and lower portions of the Fox River, which flows northeast through the state to discharge into Green Bay on Lake Michigan. The Wolf River is the second major tributary to Lake Winnebago. Watershed land cover is predominantly cultivated crops, hay and pasture, and forest (Figure 2). Major developed areas include Oshkosh to the west and Fond du Lac to the south.

The water level on Lake Winnebago has been controlled by two dams (federal dam at Menasha and a private dam at Neenah) since the 1850s when they were constructed. Originally installed to deepen Lake Winnebago for hydropower and commercial shipping purposes, the dams are now controlled by the United States Army Corps of Engineers (USACE) to accommodate flood storage and prevent large ice dams from damaging shoreline property. Under normal conditions, water levels are kept approximately three feet higher than historic levels.

The lake is a popular recreational and fishing spot for the area with numerous public boat landings and public beaches. The lake brings in fishermen from around the region for the abundance of lake sturgeon and other sport fish such as muskies, northern pike, and walleye. Other fish species include panfish, largemouth and smallmouth bass, and catfish.

Lake Winnebago is part of a larger system of water referred to as the Winnebago Pool which comprises Lake Winnebago and three smaller pools of water to its west: Lake Poygan, Lake Winneconne, and Lake Butte des Morts. These smaller pools connect the Wolf River to Lake Winnebago and are referred to as the “Upper Pool Lakes.” When the Upper Pool Lakes are discussed with Lake Winnebago, they are referred to as the “Winnebago Pool Lakes.”

Lake Poygan is the farthest west of the Winnebago Pool Lakes. It is located in Waushara County and spans 14,024 acres. Its maximum depth is eleven feet and its substrate is predominantly muck and sand. Wild rice and wild celery were historically abundant throughout the lake. The Lake Poygan Sportsmen’s Club is active in many restoration projects in the watershed.

Lake Winneconne is the smallest of the three Upper Pool Lakes (4,552 acres) and lies directly to the east of Lake Poygan. The lake is shallow with a maximum depth of nine feet. Lake Winneconne and Lake Poygan are separated by a small strip of vegetation today, but were historically separated by a large wetland with only a channel of the Wolf River connecting them. Lake Winneconne is classified as eutrophic and contains the same fish species as Lake Winnebago. In addition to being threatened by the invasive species affecting Lake Winnebago, water hyacinth has been verified as an invasive species in Lake Winneconne.

Lake Butte des Morts is the third and closest of the pool lakes to Lake Winnebago. It is located in Winnebago County and spans 8,581 acres. It is the largest and deepest of the Upper Pool Lakes but still remains shallow with a sandy, mucky substrate and a maximum depth of nine feet. The pool is fed by the Wolf River. Aquatic emergent plant species are more abundant here than in the other two Upper Pool Lakes. Major species include hardstem bulrush, phragmites, and stiff arrowhead. Small amounts of wild rice have also been reported. Water clarity is low. Fish species and invasive species present in Lake Winnebago are also present here. The Butte des Morts Conservation Club is active in the area and promotes conservation of wetland habitat in the Butte des Morts watershed.

Local community groups active in protecting the Lake Winnebago area include the Fox–Wolf Watershed Alliance and the Lake Winnebago Quality Improvement Association of Fond du Lac County.

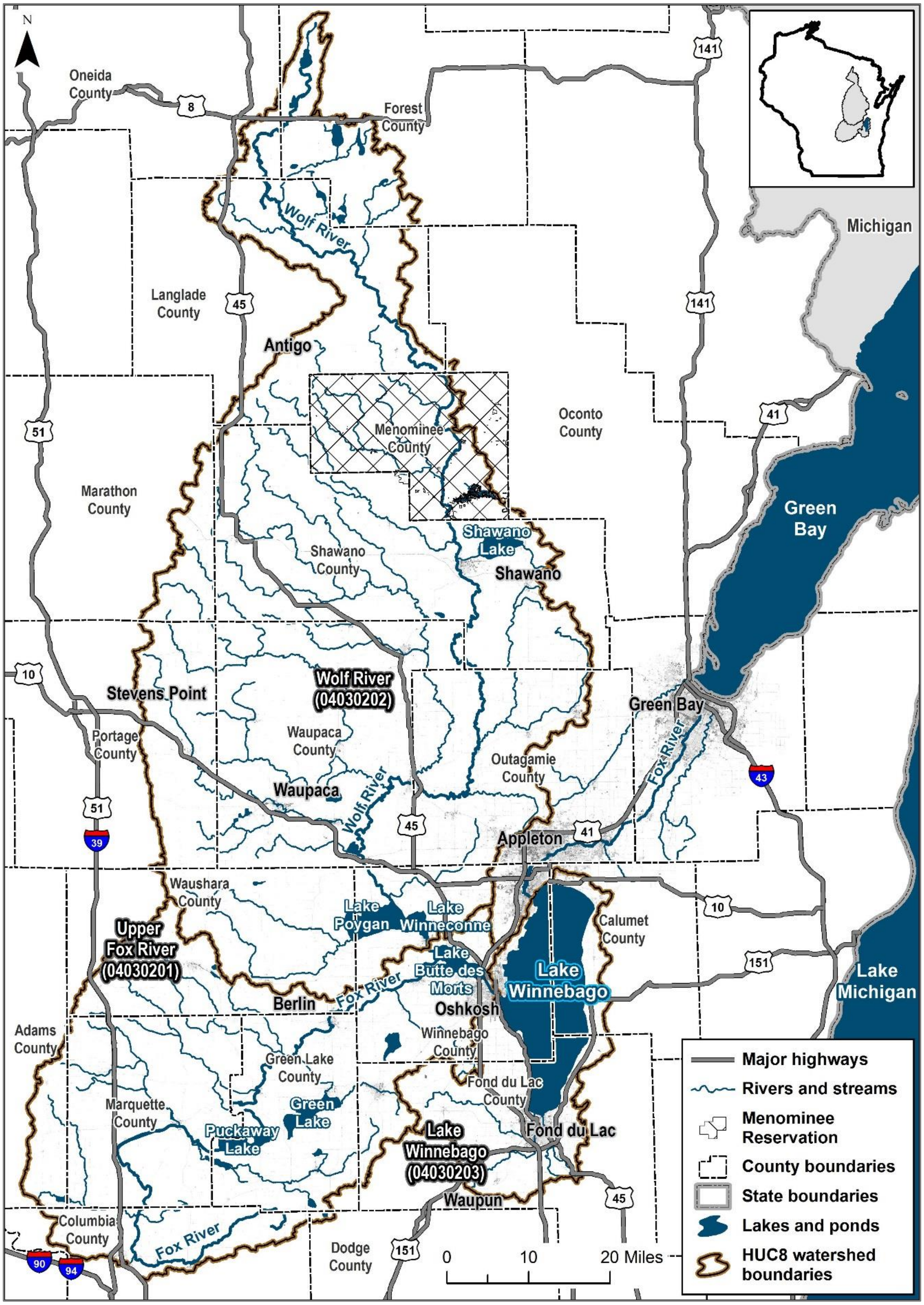

Figure 1. Lake Winnebago watershed.

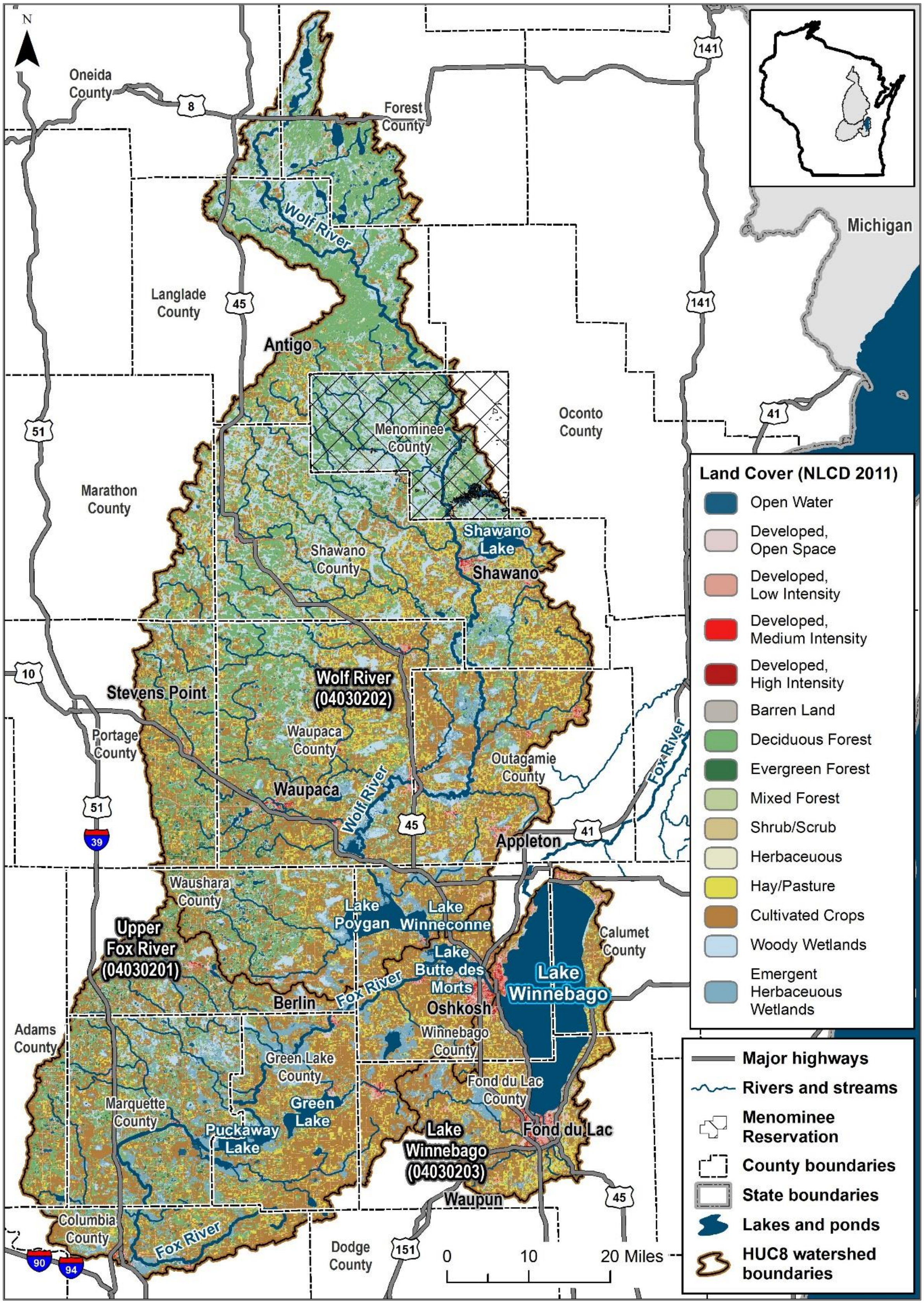

Figure 2. Land cover in the Lake Winnebago watershed.

1.3 EXISTING PLANS

Many reports and studies have addressed the Winnebago Pool Lakes and its watershed. The following reports, listed in chronological order, provide background information and information on planning efforts in the watershed.

- *Executive Summary: Water Pollution Studies, Fox River Valley, Wisconsin* (Fox Valley Water Quality Planning Agency 1978) includes a section on the trophic status of the Winnebago Pool. Using summer phosphorus and chlorophyll-*a* concentrations, a regression model estimated that a 55 percent reduction in summer phosphorus concentration in the lake is needed to achieve good recreational potential in the lake, and a 76 percent reduction in concentration is needed for high recreational potential. The report recommends investigating several techniques to decrease turbidity, stabilize lake sediments, and remove a portion of the available phosphorus: Decrease spring and early summer lake water levels to reduce peak flows that destroy or prevent reestablishment of aquatic vegetation; artificially propagate reed canes in the Upper Pool Lakes; continue/increase fish management such as rough fish removal.
- The goal of the *Poygan Marsh Wildlife Area Master Plan Concept Element* (WDNR 1981) is to "manage a state-owned wildlife area for the production of wetland wildlife as well as to provide public hunting, trapping, fishing and other compatible recreation and education." The Poygan Marsh Wildlife Area is located along the southwest shore of Lake Poygan.
- The *Lower Fox River/Winnebago Pool Long-Range Plan: Summary and Recommendations* (East Central Wisconsin Regional Planning Commission and the Lower Fox River/Winnebago Pool Long-Range Plan Task Force 1989) "provides a development strategy to realize the historic, recreational and commercial potential of the Lower Fox River Corridor extending from Green Bay through the Winnebago Pool. Its intent is to enlist local, state, federal and private cooperation in establishing the corridor as one of national significance." The recommendations focus on operation of the lock and dam system, preservation and promotion of historic significance, enhancement and expansion of recreational opportunities, and development and expansion of commercial/recreational resources. Proposed improvements in the Winnebago Pool Lakes include navigation improvements (i.e., dredging), a proposed visitor center, Oshkosh waterfront redevelopment, a proposed boat launching facility, and land and water use limits.
- The *Winnebago Comprehensive Management plan* (WDNR 1989) was developed "to restore, improve, and maintain the ecological diversity and quality, and beneficial uses of the fish, wildlife and water resources of the Winnebago system." The plan identifies resource use and management needs, defines objectives, and presents management options. The objectives and management options are grouped into fish and wildlife populations and habitat, water quality, and resource administration and use. The water quality objectives include four nutrient and sediment pollution objectives:
  - Decrease phosphorus loading by 33 percent to achieve an average lake total phosphorus concentration of 90 µg/L
  - Increase summer water clarity to an average Secchi depth of 1.0 meters in Lake Winnebago and 1.7 meters in critical habitat areas in the Upper Pool Lakes
  - Decrease algal densities such that average chlorophyll is less than 35 µg/L
  - Decrease suspended solids to achieve a concentration of 10 to 12 mg/L in critical habitat areas

  The management options proposed to address these nutrient and sediment pollution objectives are primarily watershed BMPs. Other water quality objectives address Fox River fish kills, water quality monitoring, and water quality standards.
- *The State of the Upper Fox River Basin* (WDNR 2001) provides an overview of the Upper Fox River Basin, including ecoregions, geomorphology, soils, groundwater, wetlands, land use, surface water, and hydrologic modifications. The WDNR's Upper Fox River Basin Water Team's responsibilities include fisheries and habitat, water regulation and zoning, watershed management, and drinking and groundwater. The report provides priority action items for each of these responsibilities. Information about individual water bodies is also provided.

- *Rapid Watershed Assessment Lake Winnebago Watershed* (NRCS 2007) provides "initial estimates of where conservation investments would best address the concerns of landowners, conservation districts, and other community organizations and stakeholders." The assessment presents an inventory of data that includes land elevation, land use, annual precipitation, soils, and resource concerns.
- *Winnebago County 2011–2020 Land and Water Resource Management Plan* goal is "to restore, improve, and protect ecological diversity and quality and to promote beneficial uses of its land, water, and related resources" (Winnebago County Land and Water Conservation Department 2010). Resource concerns were identified from existing plans and through input from agency committees, a citizen's advisory committee, and a public hearing. Nine objectives and accompanying goals are identified, which target agricultural runoff, stormwater runoff, habitat loss, land management, invasive species, and climate change impacts. The following goals relate directly to the Winnebago Pool Lakes:
    - Increase shoreland and wetland restoration projects
    - Support stabilizing water levels to increase lake and wetland aquatic and plant habitat resiliency
    - Support the adoption of ecologically responsible seasonal water level management on the Winnebago System
    - Create awareness of the benefits of these plant communities to the resource

The Winnebago Waterways program is active in the Lake Winnebago watershed. It is coordinated by the Fox–Wolf River Watershed Alliance with the purpose to engage local residents in improving and protecting water quality within the Winnebago Pool Lakes and watershed. Two recent reports were completed for the program in an effort to update the WDNR's *Winnebago Comprehensive Management plan* (WDNR 1989) and further engage residents in the planning process. By doing so, the Winnebago Waterways program hopes to restore eligibility for additional sources of funding. Outcomes of this technical assistance document can be used to support and enhance future planning efforts as the program further develops.

- *Weigh In on the Winnebago Waterways: A Coordinated Public Engagement Effort for the Lake Winnebago System* (Biodiversity Project 2013) presents results from a public engagement effort to identify how people use the Winnebago Pool Lakes and what users' top issues are. The top three issues identified were algae blooms, polluted runoff, and invasive species. The effort also included the compilation of inventories, plans, and data related to the Winnebago Pool Lakes.
- *Final Report: Winnebago Waterways Phase II* (Bluestem Communications 2014) summarizes activities of the second phase of the *Winnebago Waterways* effort:
    - A communications and outreach strategy was developed to engage key public constituencies
    - Development of a Winnebago Waterways Central Hub to share information with constituencies was explored and recommendations made
    - Cooperative management options were evaluated, and a nonprofit organizational structure was recommended
    - Data were compiled to fill data gaps identified in phase 1 of *Winnebago Waterways*; a status update on the data gaps is provided

2 SHALLOW LAKE ECOLOGY

The WDNR classifies Lake Winnebago as a shallow lake with a mixed water column, and the Upper Pool Lakes also exhibit characteristics of shallow lakes. Shallow lakes typically do not develop a stable thermal stratification throughout the growing season. Instead, instances of temporary thermal stratification can develop multiple times throughout the growing season, interspersed with periods when the water column mixes. Mixing of the water column is driven by wind energy and occurs when the wind energy is sufficient to overcome the stability of stratification. Wind and wave action can disturb the lake bottom sediments, leading to release of nutrient-laden sediments into the water column and directly disturbing rooted macrophytes.

The relative influence of the multiple drivers of primary production in shallow lakes differs from the relationships in deeper lakes. Drivers and controls of primary production in lakes include light availability, water temperature, dissolved oxygen, nutrient availability, and grazing by zooplankton, other invertebrates, and fish. In shallow lakes, light, temperature, and dissolved oxygen are typically sufficient throughout the water column to support primary production, which therefore is prominent not only at the surface and around the shoreline as in a deep lake, but also throughout the bottom sediments. In a deep lake, a large part of the lake volume does not support plant or algal growth due to low light, temperature, and oxygen. Therefore, the biological components (e.g., plants, algae, invertebrates, and fish) in a deep lake have less of an influence on algal and plant growth than do the physical (e.g., light and temperature) and chemical (e.g., dissolved oxygen) components.

Due to the strong influence of biotic factors on primary production in shallow lakes, shallow lakes typically exhibit one of two alternative stable states: the clear, macrophyte-dominated state, and the turbid, phytoplankton-dominated state (Figure 3). In the clear water state, rooted macrophytes (and associated periphyton) stabilize the bottom sediments and sequester nutrients, and phytoplankton growth is minimal. The clear water state is typically more stable when nutrient concentrations are lower (Zimmer et al. 2009); as nutrient concentrations increase, the stability of the clear water state decreases and the lake becomes more vulnerable to slight perturbations that can shift the lake to the turbid state (Scheffer 2001). In the turbid state, phytoplankton growth is high, leading to turbid waters that limit light availability to rooted macrophytes. A positive feedback cycle stabilizes the turbid state—high phytoplankton growth reduces light availability, which suppresses growth of rooted macrophytes, which reduces sediment stability and increases release of nutrients to the water column, which further fuels algal growth, etc.

The clear water state is generally regarded as more desirable than the turbid state due to higher water clarity and higher biological diversity. Many factors can lead to the shift of a lake from a clear water state to a turbid state, including increased nutrient load, damage to macrophytes (e.g., from wind and wave action, physical harvesting, herbicides, and boat activity), and loss of invertebrate grazers such as zooplankton. Shifting the lake from a turbid to a clear water state is often more difficult than the shift to the turbid state; options include manipulation of the fish community to remove zooplanktivorous fish and/or to increase density of piscivores. Reduction of external nutrient loads is often a prerequisite for a successful shift back to a clear state.

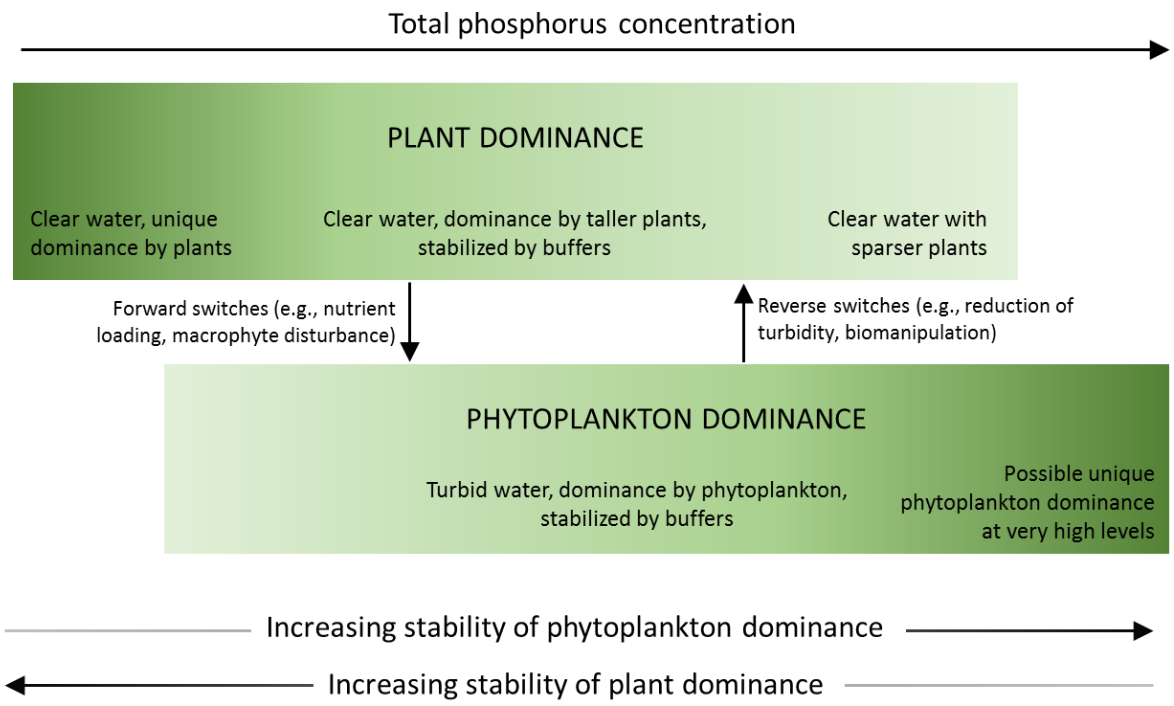

Figure 3. General theory of alternative stable states in shallow lake systems. Adapted from Moss et al. (1996).

3 WINNEBAGO POOL LAKES CONDITIONS

3.1 WATER QUALITY

Water quality is a key component of comprehensive lake management focused on reducing algae and harmful algal blooms. Poor water quality in Lake Winnebago has been documented at least as far back as the 1970s, when the United States Environmental Protection Agency (USEPA), in cooperation with the Wisconsin Department of Natural Resources and the Wisconsin National Guard, evaluated the water quality in Lake Winnebago as part of the National Eutrophication Survey (USEPA 1975). The lake was described as eutrophic, with algal blooms observed in June and August of 1972 and dense aquatic vegetation in the littoral zone. Nitrogen was found to be limiting to algal growth in November 1972.

Algae blooms were identified as a top concern among users of the Winnebago Pool Lakes in *Weigh In on the Winnebago Waterways: A Coordinated Public Engagement Effort for the Lake Winnebago System* (Biodiversity Project 2013), and all four of the Winnebago Pool Lakes are on Wisconsin's 303(d) impaired waters list due to excess algal growth.

Cyanobacteria blooms, also known as blue-green algae blooms, have been observed on the lake. The Winnebago County Health Department has issued water quality advisories for Lake Winnebago based on visual and laboratory confirmation of cyanobacteria blooms by the WDNR[1]. Beversdorf et al. (2017) and McDermott et al. (1995) document the presence of cyanobacterial compounds in Lake Winnebago, and Trabeau (2004) suggests that cyanobacteria could play a role in the observed declines in the zooplankton *Daphnia pulicaria* abundance in the lake. Drought years can increase the likelihood of cyanobacteria blooms in Lake Winnebago (Ted Johnson, WDNR, personal communication).

3.1.1 Water Quality Data Inventory

Water quality data on the Winnebago Pool Lakes were compiled from the following sources:

- Water quality data (1989–2016) from the three long term lake stations on Lake Winnebago (Figure 4)—*2 miles from Neenah* (713243), *3 miles from Oshkosh* (713245), and *deep hole, south end* (713244). Provided by WDNR.
- Water quality data (1973–2016) from the Fox River at Oshkosh (713056) and the Fox River at the Lake Winnebago outlet (713002). Provided by WDNR.
- Water quality data (1977–2016) from multiple sites in the Upper Pool Lakes. Provided by WDNR.
- Data for the two occasions in 2011 and 2012 for which water samples were collected for the Wisconsin HAB Surveillance Program in response to illness reports. Provided by WDNR.
- 2016 and 2017 Lake Winnebago surface and intake depth monitoring data of the four water utility intakes. Provided by WDNR.

Other sources of data on the Winnebago Pool Lakes exist, but were not evaluated for this effort:

- Continuous monitoring data are collected by the *Aquatic Environmental Microbiology and Chemistry* group ("The Miller Lab") at the University of Wisconsin Milwaukee's Zilber School of Public Health. Data (2012–2014) on algal pigments (i.e., chlorophyll and phycocyanin), dissolved oxygen, photosynthetically active radiation (PAR), water temperature, and wind are available online for academic, research, education, and other not-for-profit professional purposes.

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[1] For example, https://www.co.winnebago.wi.us/public-health/news/2014/7/3/lake-winnebago-water-quality-advisory-blue-green-algae

- Bart De Stasio from Lawrence University has collected data on the Winnebago Pool Lakes; the data were not available for this report.
- Xiao et al. (2017) developed an approach to forecasting harmful algal blooms that was verified in part using Lake Winnebago chlorophyll-*a* fluorescence, which was continuously monitored using a fluorometer mounted to a buoy platform. The paper does not specifically evaluate chlorophyll-*a* or cyanobacteria concentrations in Lake Winnebago but is included here to document the existence of continuously monitored (on a 5-minute interval) chlorophyll-*a* fluorescence in Lake Winnebago.

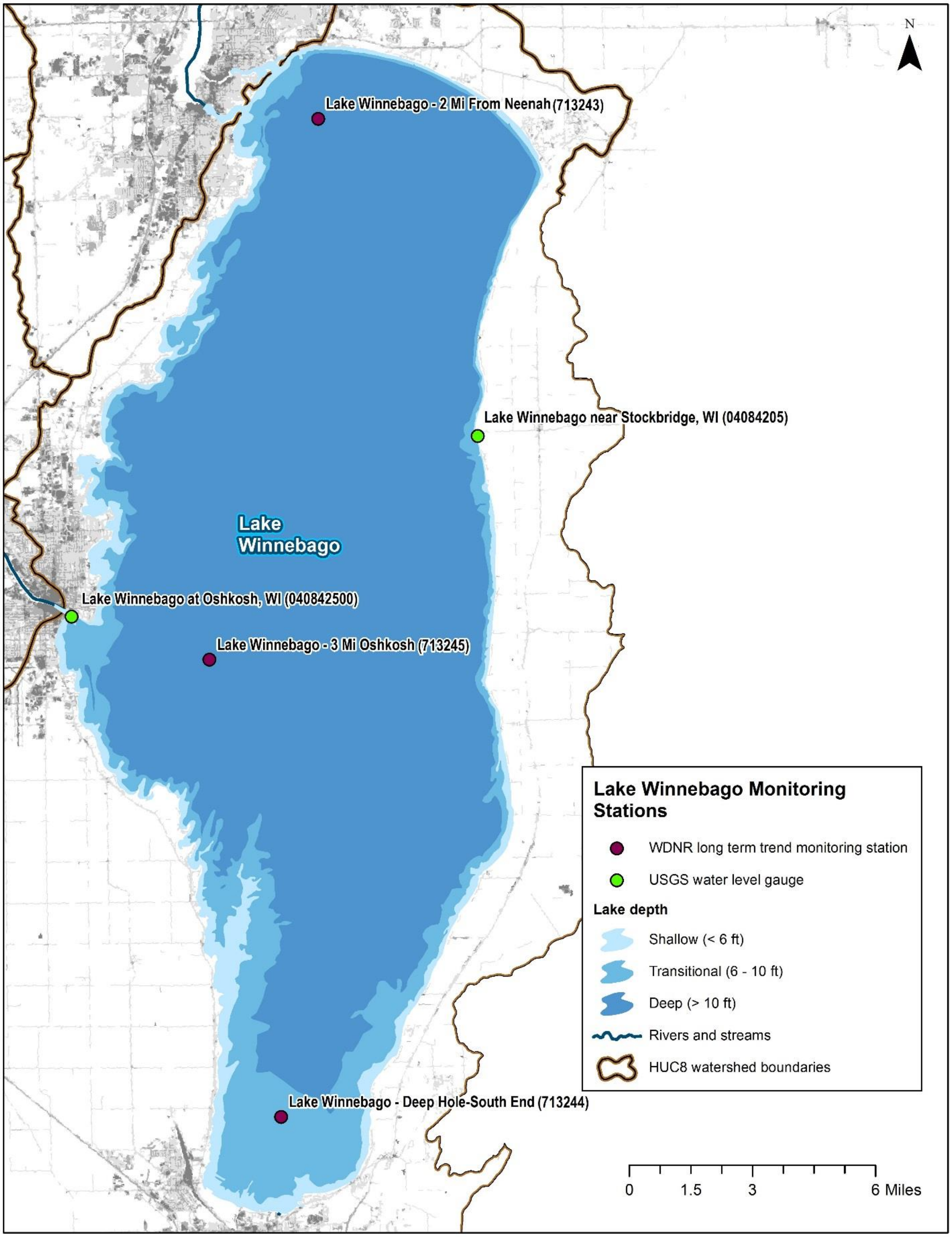
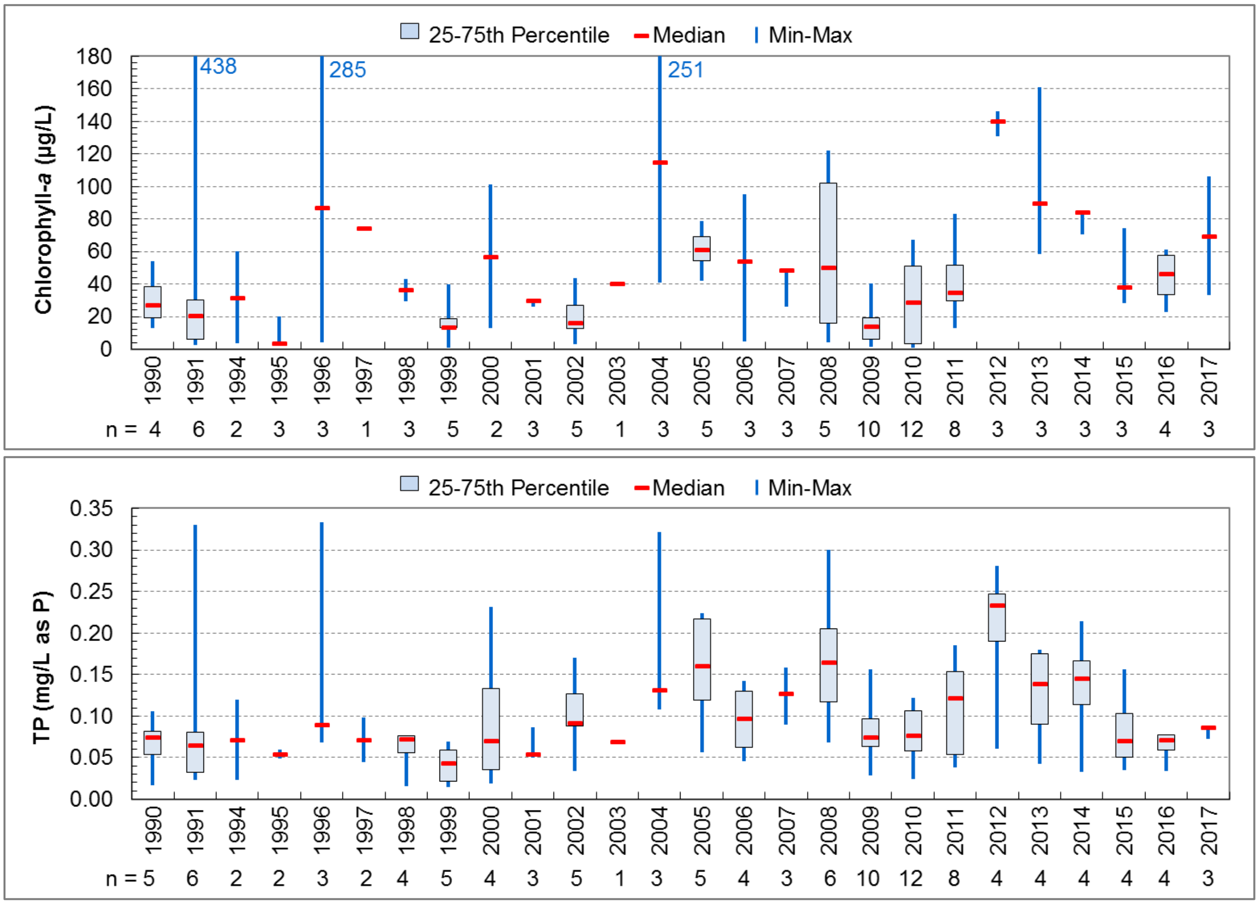


Figure 4. Monitoring stations on Lake Winnebago.

3.1.2 Water Quality Data Assessment

Lake Winnebago

In Lake Winnebago, growth of suspended algae, as measured by chlorophyll-*a* concentration, has fluctuated over the years, as have phosphorus and nitrogen concentrations (Figure 5). The low numbers of samples taken each year preclude in-depth analysis of water quality. Because cyanobacteria typically have less chlorophyll-*a* per unit biomass than true algae due to other pigments in the cyanobacteria, the use of chlorophyll as an indicator of phytoplankton biomass is limited. Data from the northern-most site (2 miles from Neenah, site #713243) are shown in Figure 5; patterns are similar at the other two monitoring stations. Wisconsin's recreational impairment thresholds for the Winnebago Pool Lakes are 0.04 mg/L total phosphorus and 20 µg/L chlorophyll-*a* (greater than 30 percent of days in the sampling season with nuisance algal blooms as indicated by chlorophyll-*a* concentrations greater than 20 µg/L; WDNR 2017).



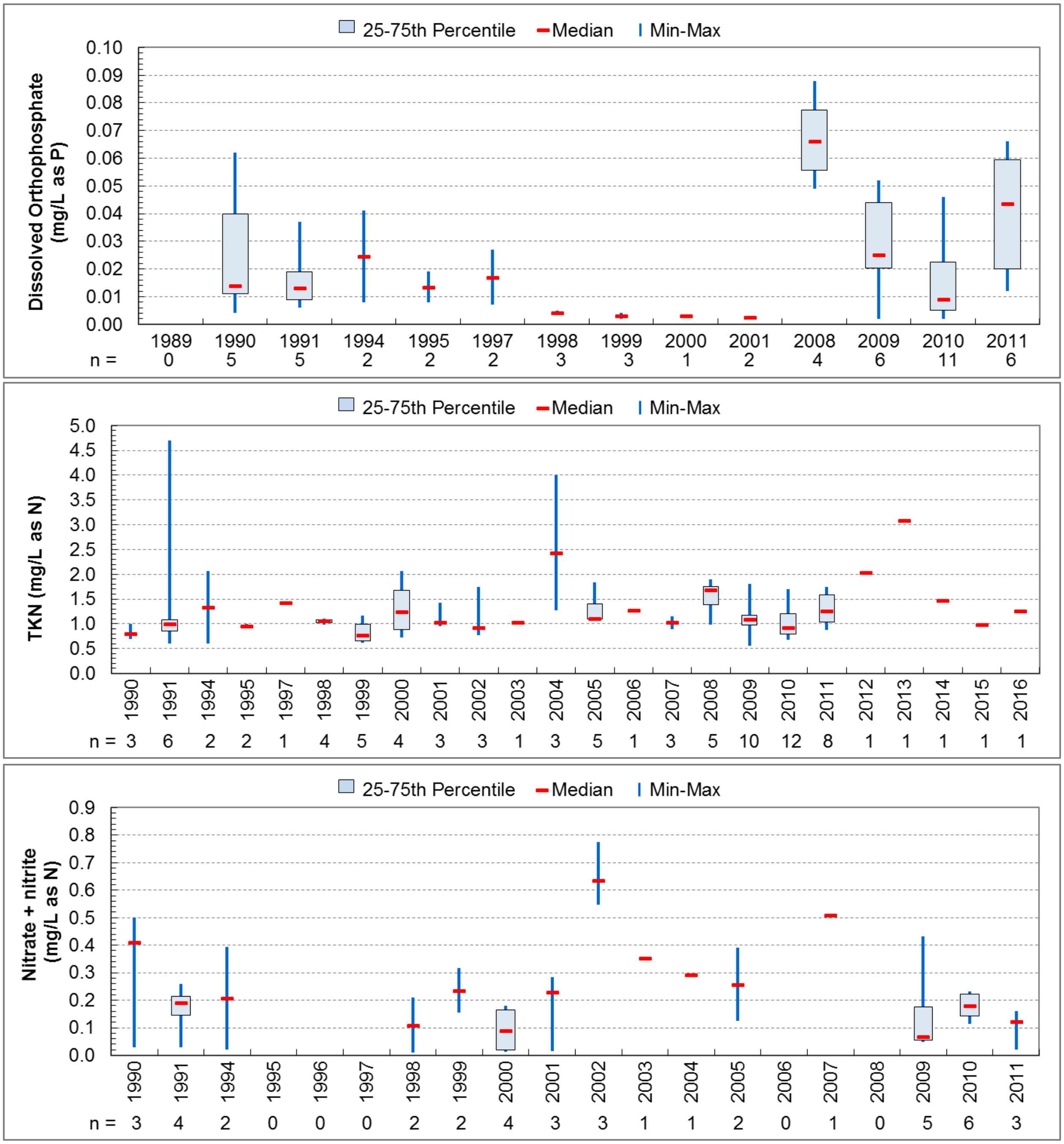


Figure 5. Chlorophyll-a, total phosphorus, dissolved orthophosphate, total Kjeldahl nitrogen, and nitrate plus nitrite nitrogen concentrations by year, Lake Winnebago, 2 miles from Neenah (site 713243).

Over the course of the growing season, chlorophyll-*a,* phosphorus, and total Kjeldahl nitrogen (TKN) concentrations typically increase, with the highest concentrations on average in September (Figure 6 through Figure 9). Measurements of total phosphorus (TP) and total Kjeldahl nitrogen are unfiltered, whole water samples and include the nutrient content of the algae themselves. Therefore, high concentrations in the late summer when chlorophyll is also high is likely a reflection of the algal growth. Nitrate concentrations, on the other hand, decrease on average from April through August (Figure 10). The relative amount of nitrogen to phosphorus also decreases, both in terms of total nutrients (Figure 11) and dissolved inorganic nutrients (Figure 12). More intensive data collection occurred from 2009 through 2011 for TMDL development. When data from just these years (2009–2011) are evaluated, similar seasonal trends as those seen in the longer term data set (Figure 6 through Figure 12) are observed.

These seasonal patterns are often seen in shallow lakes. Seasonal increases in phosphorus often occur in eutrophic shallow lakes, typically as a result of internal loading (Søndergaard et al. 2003). As nitrogen is assimilated by primary producers over the course of the growing season, and warmer water temperatures lead to increased rates of denitrification, the nitrogen pool typically is not replenished due to minimal inputs from the watershed and internal sources (Moss et al. 2013). These processes can lead to reduced inorganic nitrogen concentrations later in the growing season.

The seasonal patterns in nutrient concentrations and nitrogen to phosphorus ratios suggest that the lake shifts from a predominantly phosphorus limited system in the spring and early summer to a system that is likely either co-limited by phosphorus and nitrogen and/or light limited in the late summer and early fall. Co-limitation by phosphorus and nitrogen is common in freshwater systems (Sterner 2008), and light limitation is a possibility due to the high concentration of phytoplankton (Dale Robertson, USGS, personal communication). Further review of the results of the algal assays on water from Lake Winnebago in November 1972 (USEPA 1975) suggests that algal growth was co-limited by nitrogen and phosphorus. Nitrogen limitation in the fall is common in shallow lakes, likely due to seasonal shifts in denitrification and late summer phosphorus release from the sediments (Moss et al. 2013), leading to high phosphorus concentrations relative to nitrogen. Nitrogen limitation can favor certain species of cyanobacteria that fix their own nitrogen and therefore can use atmospheric nitrogen. Nitrogen fixation, however, uses more energy than uptake of more bioavailable nitrogen forms and can limit algal bloom size and toxicity (Great Lakes HABs Collaboratory n.d.).

These seasonal patterns of nutrient and chlorophyll concentrations are reflected in Lake Winnebago (3 miles from Oshkosh, site 713245) in 2011 (Figure 13). Total phosphorus, total Kjeldahl nitrogen, and chlorophyll concentrations peaked in September, and nitrate concentrations and nitrogen to phosphorus ratios declined over the course of the growing season.

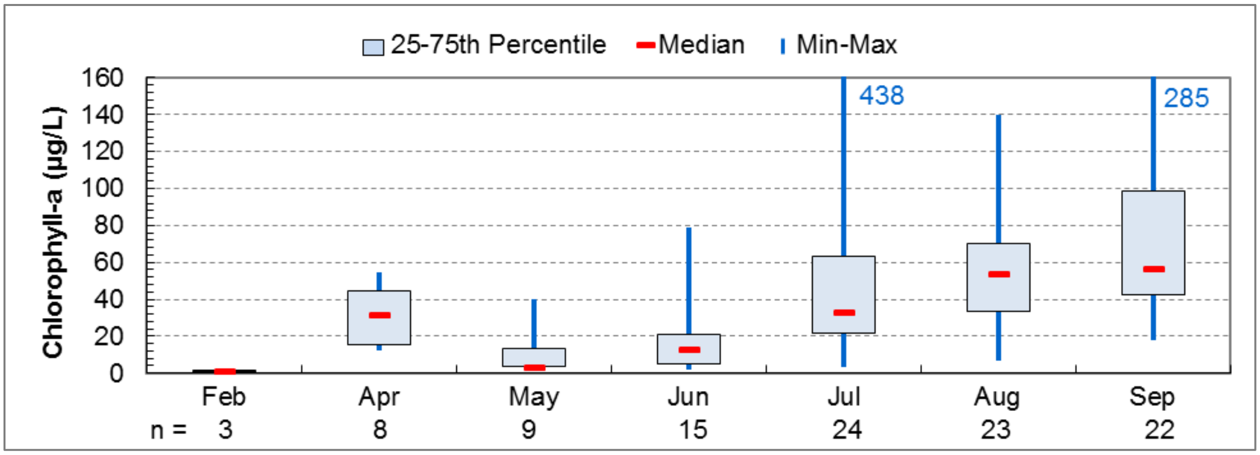


Figure 6. Chlorophyll-a concentrations by month, 1990–2017, Lake Winnebago, 2 miles from Neenah (site 713243).

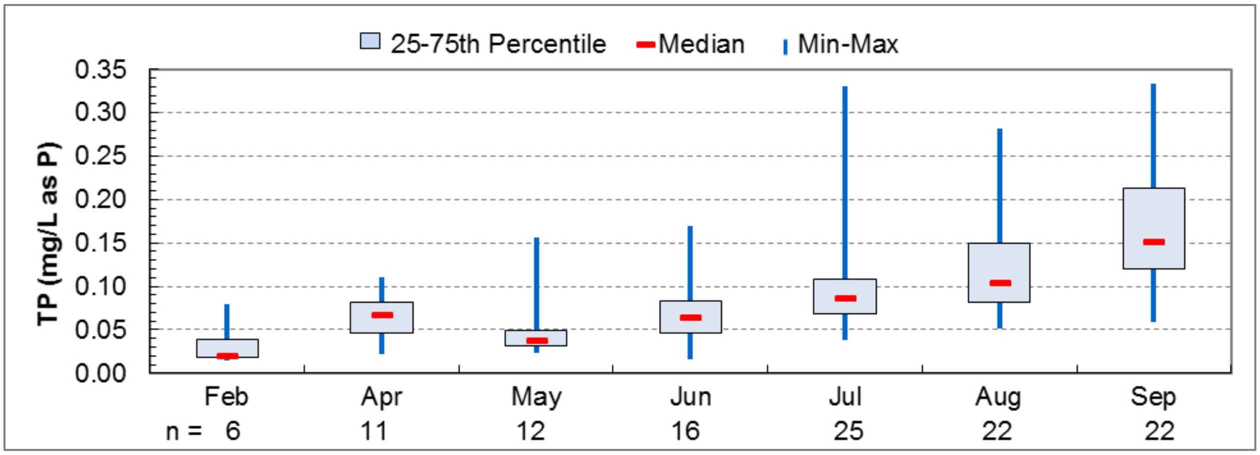


Figure 7. Total phosphorus concentrations by month, 1990–2017, Lake Winnebago, 2 miles from Neenah (site 713243).

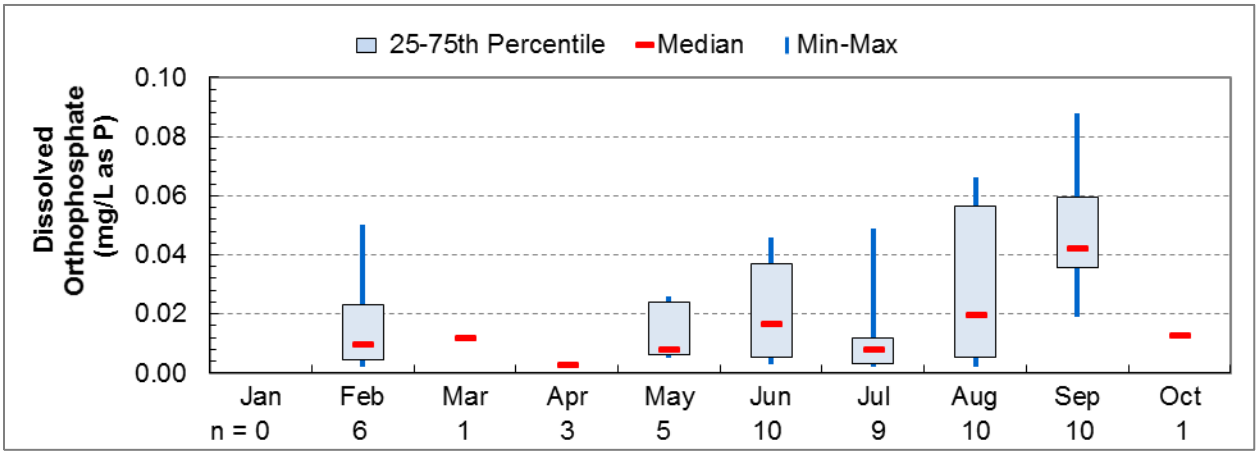


Figure 8. Dissolved orthophosphate concentrations by month, 1990–2011, Lake Winnebago, 2 miles from Neenah (site 713243).

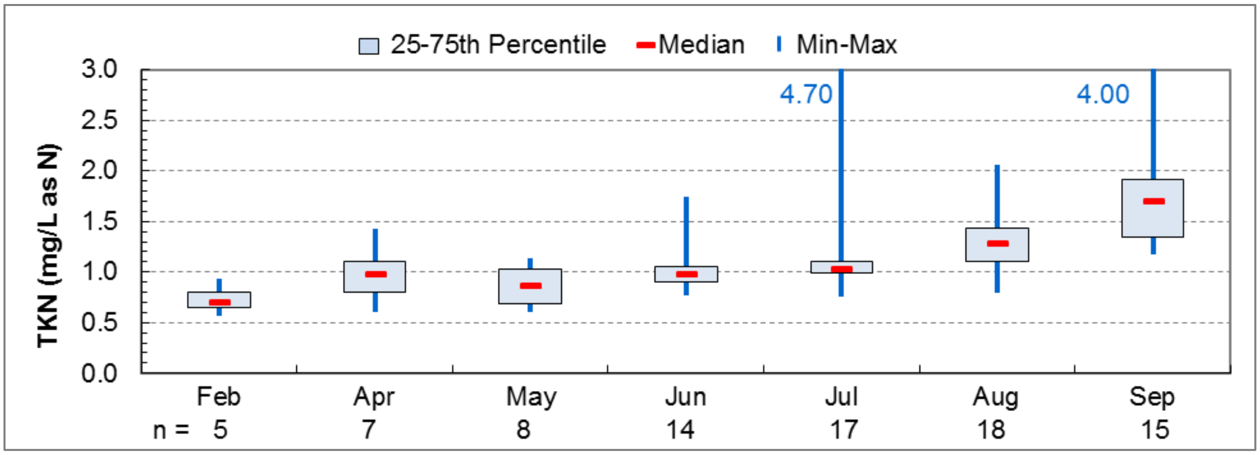


Figure 9. TKN concentrations by month, 1990–2016, Lake Winnebago, 2 miles from Neenah (site 713243).

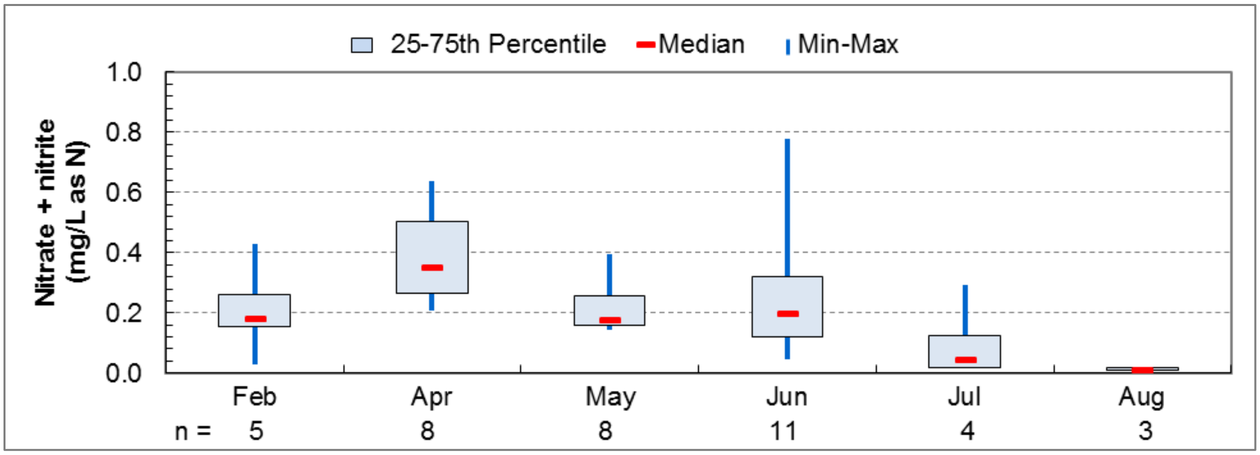


Figure 10. Nitrate plus nitrite concentrations by month, 1990–2011, Lake Winnebago, 2 miles from Neenah (site 713243).

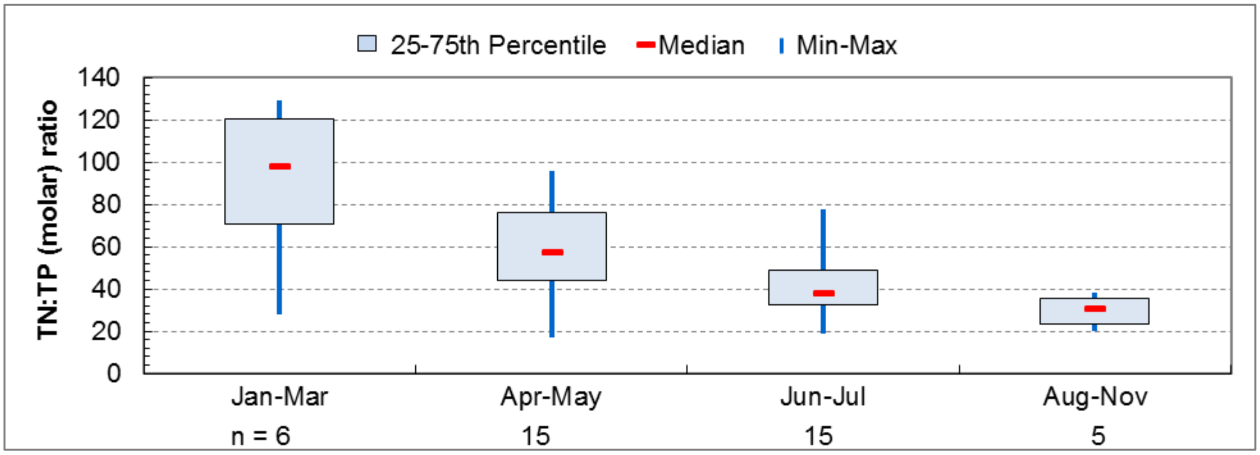


Figure 11. Total nitrogen to total phosphorus ratios, 1990–2016, Lake Winnebago, 2 miles from Neenah (site 713243).

Total nitrogen calculated as the sum of TKN and nitrate plus nitrite nitrogen.

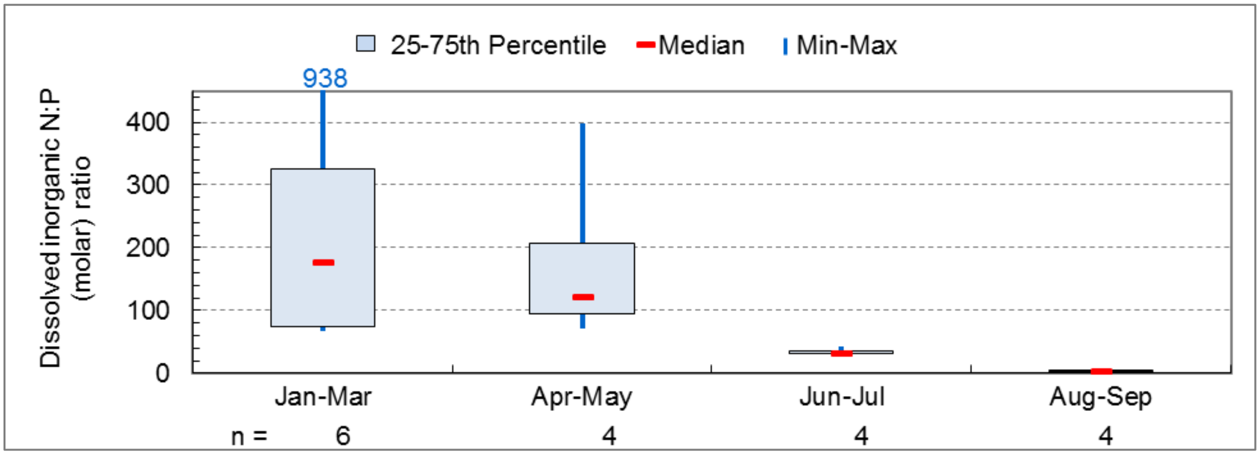


Figure 12. Dissolved inorganic nitrogen to phosphorus ratios, 1990–2011, Lake Winnebago, 3 miles from Oshkosh (site 713245).

Dissolved inorganic nitrogen calculated as the sum of ammonia and nitrate plus nitrite nitrogen.

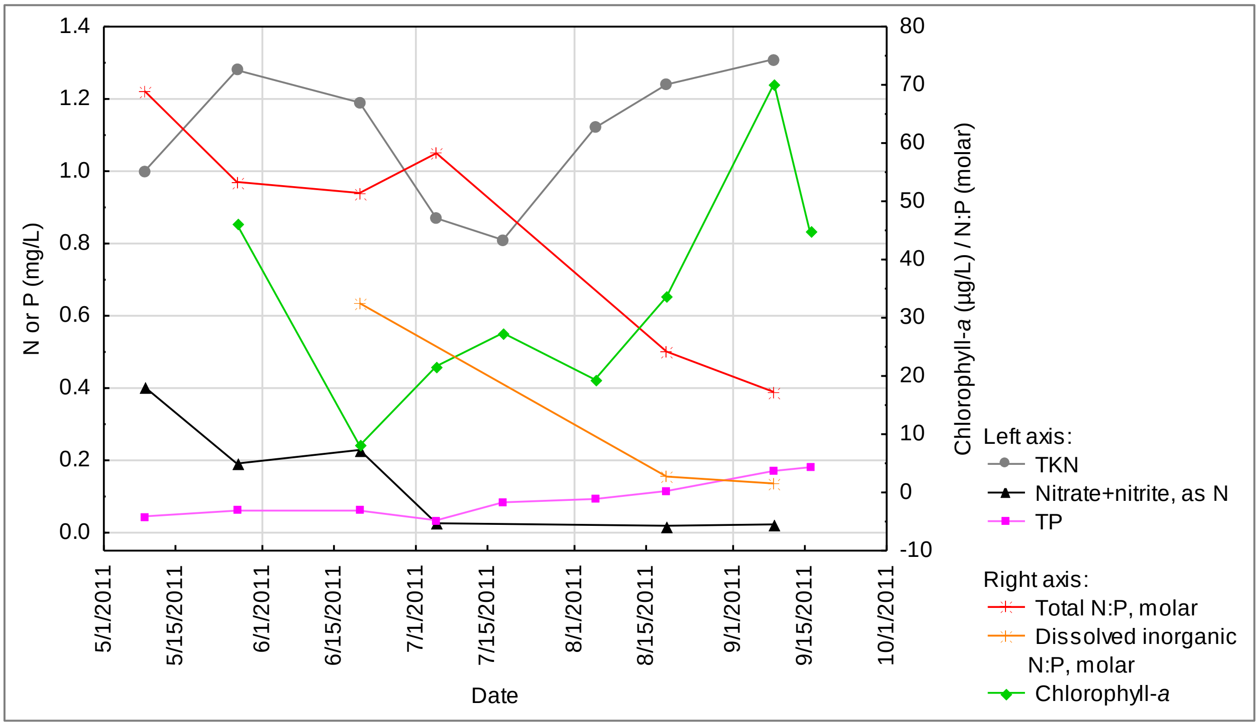


Figure 13. Seasonal changes in water quality in Lake Winnebago, 3 miles from Oshkosh (site 713245), 2011.

Chlorophyll concentrations at the three monitoring sites ranged from less than 1 µg/L to over 400 µg/L and on average were highest when phosphorus and total Kjeldahl nitrogen were high (Figure 14 and Figure 15) and when the relative amount of nitrogen to phosphorus was low (Figure 16). The four chlorophyll-*a* observations that are greater than 100 µg/L were from July and August, when nitrogen is low relative to earlier in the growing season. High chlorophyll concentrations do not necessarily indicate that cyanobacteria or cyanotoxins were high. Analysis of algal growth in Lake Winnebago is being completed for the Lake Winnebago TMDL project.

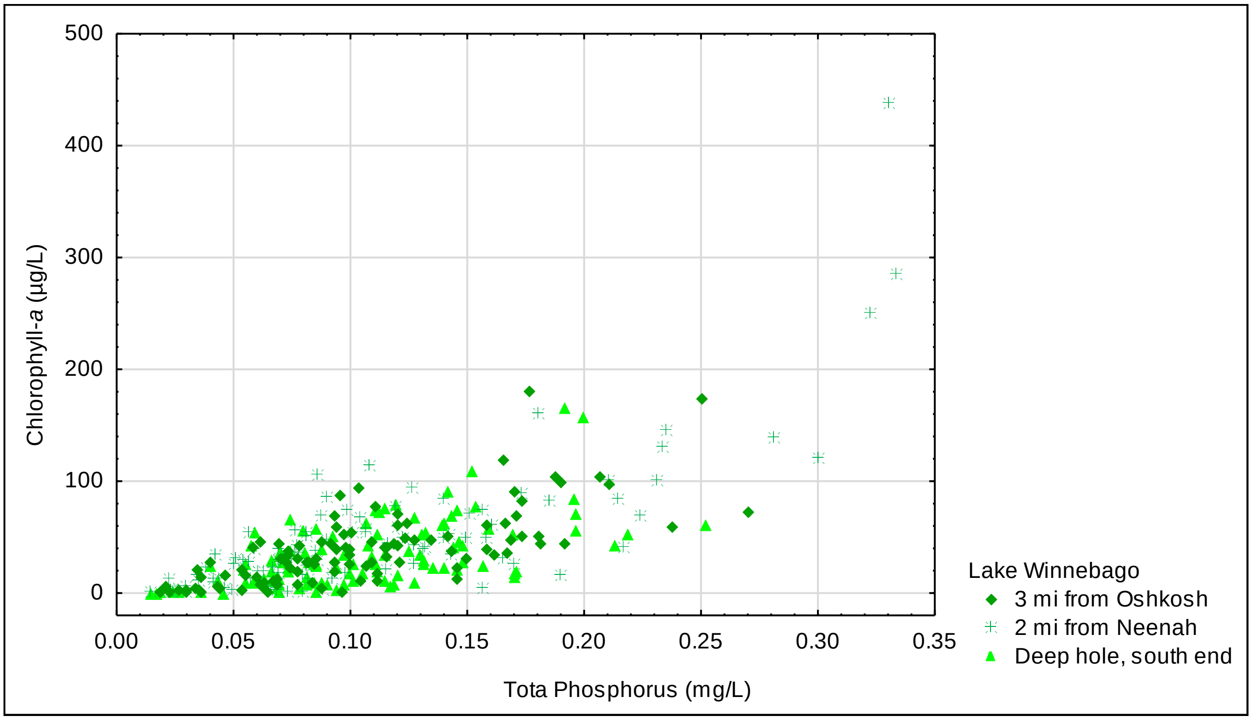


Figure 14. Chlorophyll concentration as a function of total phosphorus.

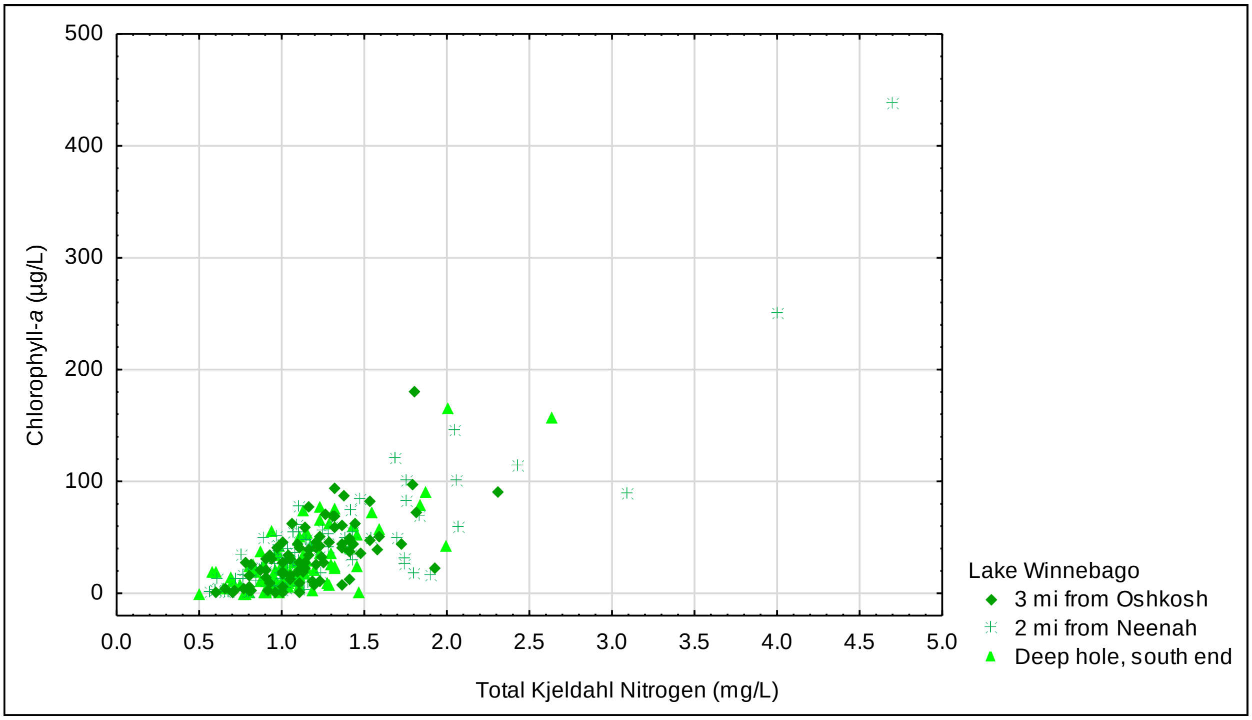


Figure 15. Chlorophyll concentration as a function of total Kjeldahl nitrogen.

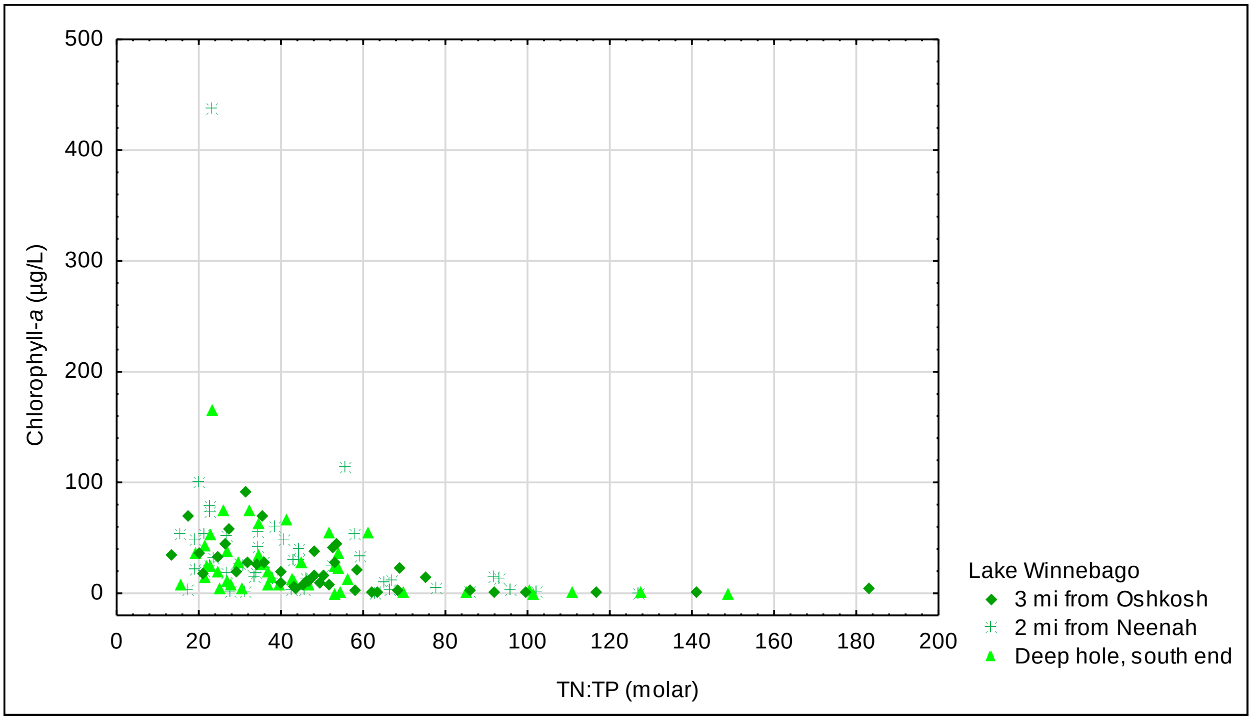


Figure 16. Chlorophyll concentration as a function of TN:TP.

Concentrations of microcystin (a toxin produced by cyanobacteria primarily of the genus *Microcystis*) were monitored at several locations in 2016 and 2017. Concentrations vary and at times exceeded the USEPA's draft microcystin criteria for recreational uses, 4 µg/L (USEPA 2016; Figure 17). The higher microcystin concentrations were observed in August and September. Additionally, water quality samples were collected as part of the Harmful Algal Bloom Surveillance Program, an interagency program that monitors water bodies if an illness is reported. The samples were collected in July 2011 and June 2012. Chlorophyll-*a* concentrations were 2,290 and 235 µg/L, respectively, and the concentrations of various forms of microcystin ranged from non-detect to 45 µg/L for microcystin-LR.

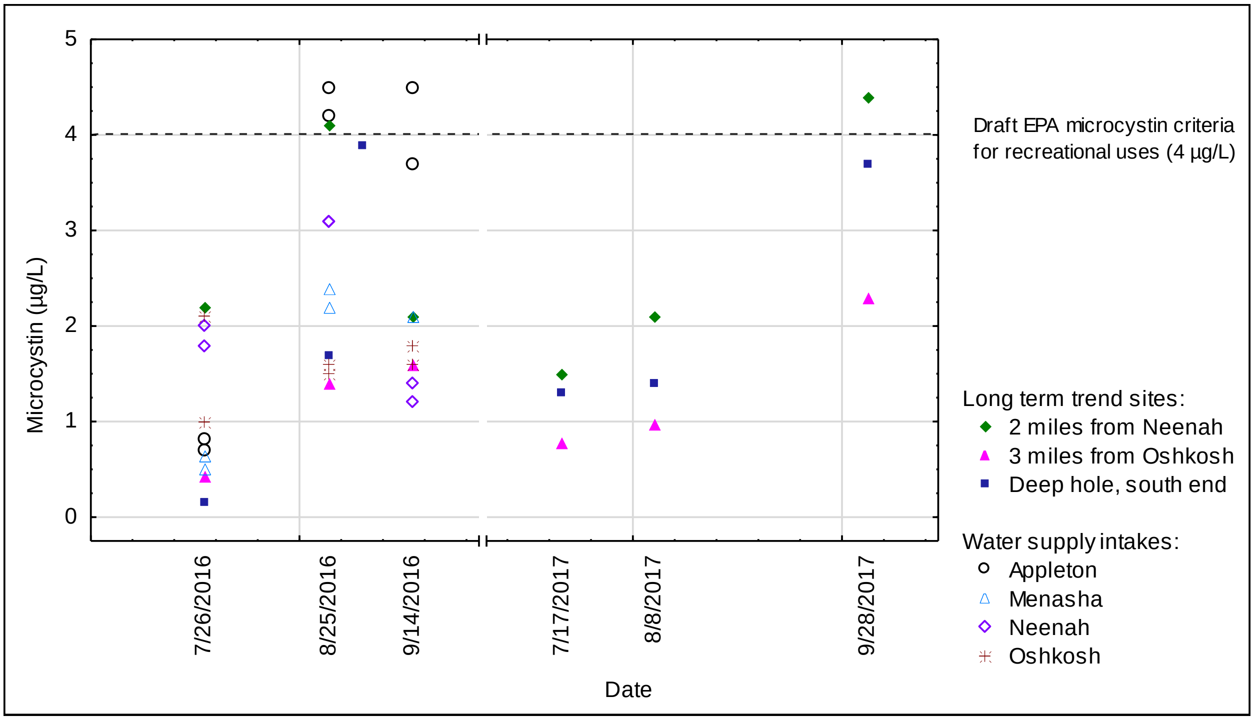


Figure 17. Microcystin concentrations in Lake Winnebago, 2016–2017. Data provided by WDNR.

Upper Pool Lakes

Water quality data are also available for the Upper Pool Lakes (Figure 18 through Figure 20). Data were collected from multiple sites, ranging from one sample date per year to up to ten samples from individual sites in certain years. The years with the highest number of samples throughout the growing season are 2009 through 2011. The relative chlorophyll-*a* concentrations from upstream to downstream (Lake Poygan, Winneconne, Butte des Morts, and Winnebago) do not show a consistent pattern from year to year (Figure 21), suggesting that algal growth in each lake is more influenced by local conditions than by upstream algal densities.

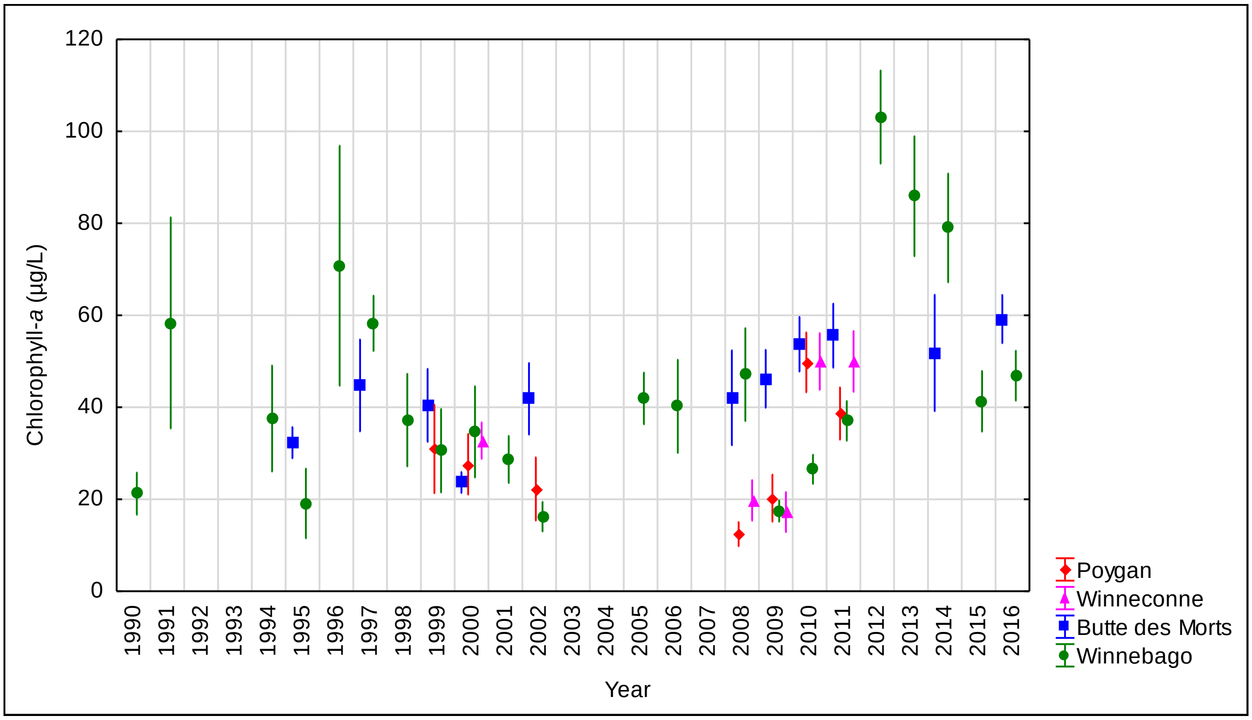


Figure 18. Chlorophyll-a concentrations (means +/- standard error) by year, Lake Winnebago and Upper Pool Lakes.

Data from multiple sites are aggregated by lake. Years with fewer than four sample dates in a lake are not shown. Lake Winnebago recreational impairment threshold is 25 µg/L.

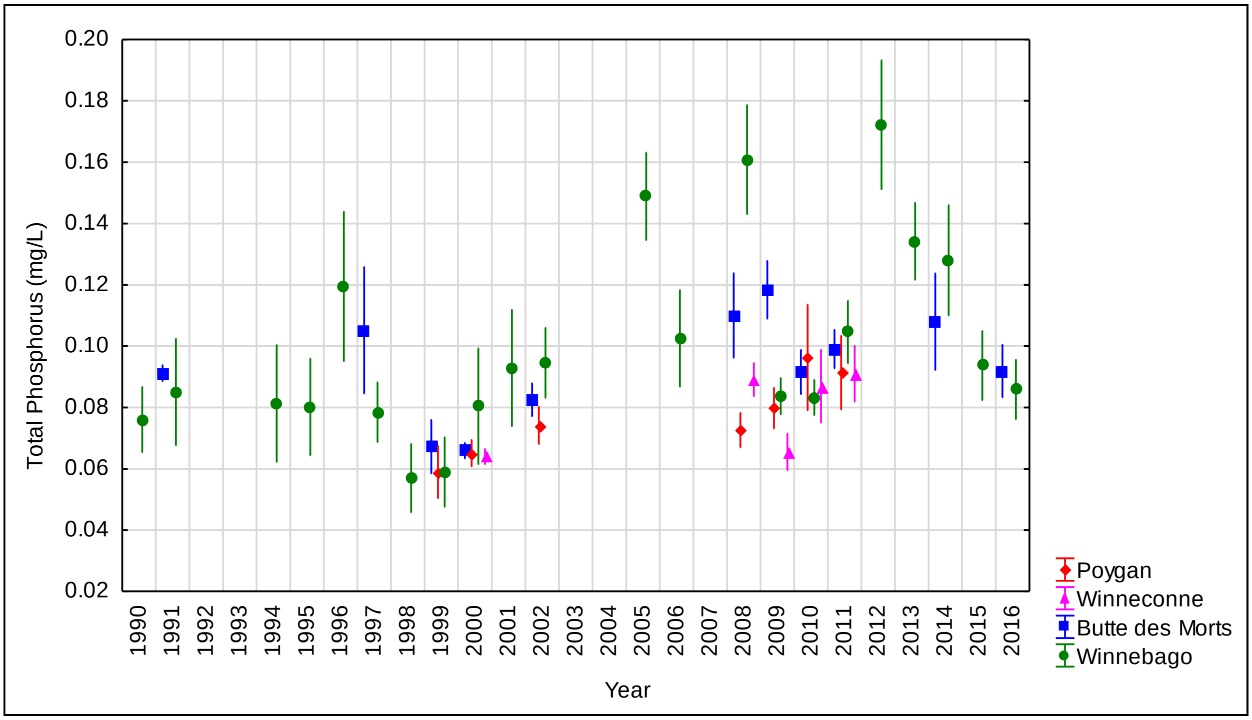


Figure 19. Total phosphorus concentrations (means +/- standard error) by year, Lake Winnebago and Upper Pool Lakes.

Data from multiple sites are aggregated by lake. Years with fewer than four sample dates in a lake are not shown. Lake Winnebago recreational impairment threshold is 0.04 mg/L.

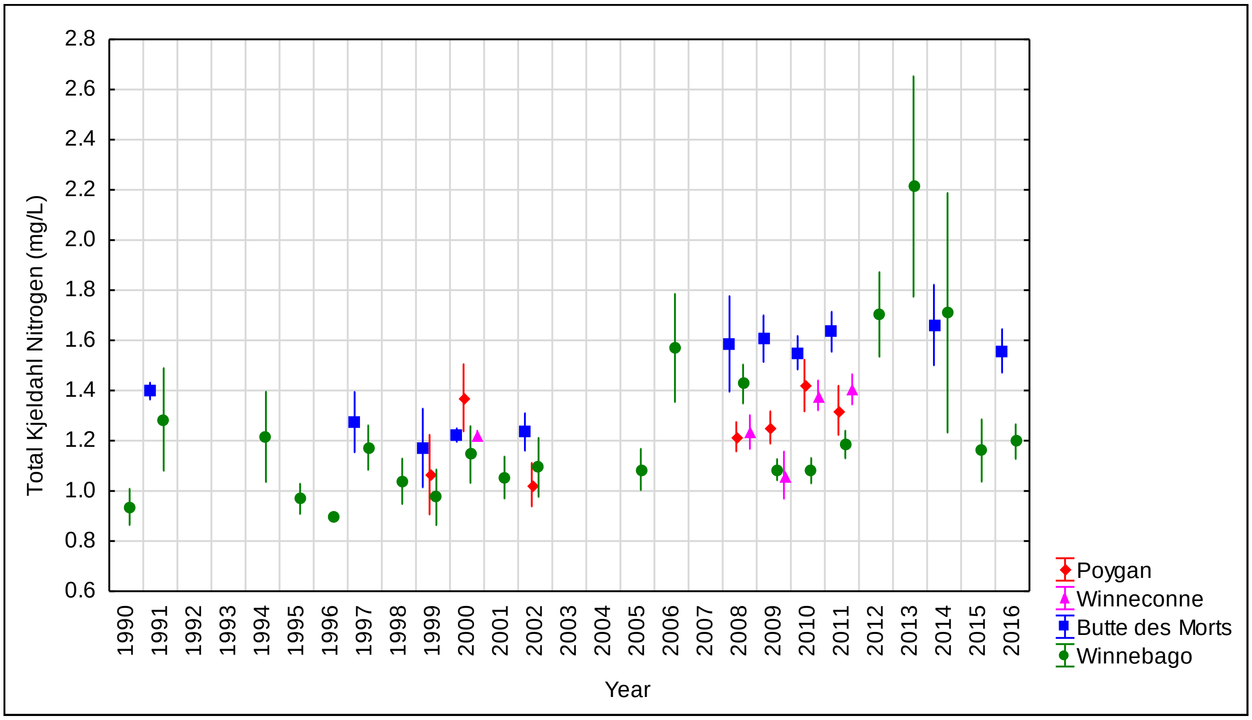


Figure 20. Total Kjeldahl nitrogen concentrations (means +/- standard error) by year, Lake Winnebago and Upper Pool Lakes.

Data from multiple sites are aggregated by lake. Years with fewer than four sample dates in a lake are not shown.

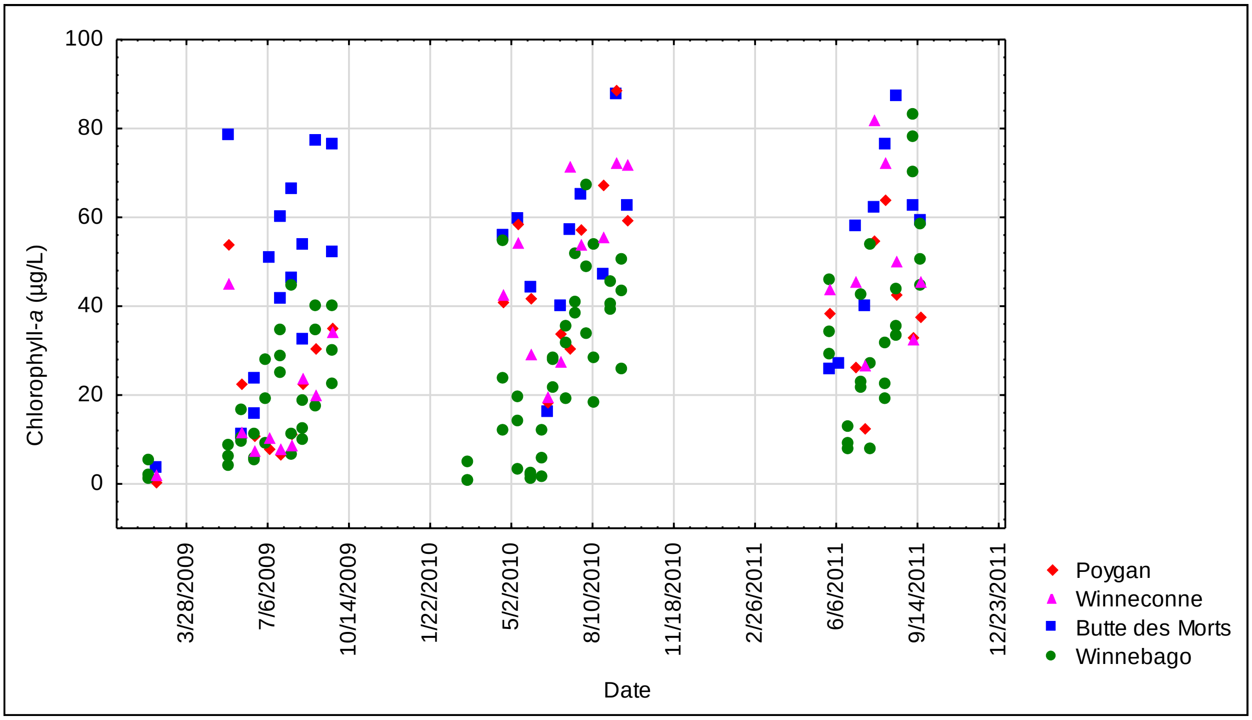


Figure 21. Chlorophyll-a concentrations, 2009–2011, Lake Winnebago and Upper Pool Lakes.

Data from multiple sites are aggregated by lake. Lake Winnebago recreational impairment threshold is 25 µg/L.

Fox River: Lake Winnebago Inflow and Outflow

Water quality has been monitored on the Fox River at Oshkosh (inflow to Lake Winnebago) and the Fox River at the Lake Winnebago outlet since 1995 and 1974, respectively. Nitrate concentrations at both sites have increased over the period of record (Figure 22; Kendall Tau trend analysis on annual medians where annual sample size >5; $p<0.05$). There are no statistically significant trends in the other water quality parameters (i.e., TKN, phosphate, and TP) monitored at these two sites.

WDNR also provided analysis of water quality data from four river stations with long term data—Wolf River at New London, Fox River at Berlin, Fox River at Oshkosh, and Fox River at Neenah. Figures from the long-term trend analysis can be found in Appendix B.

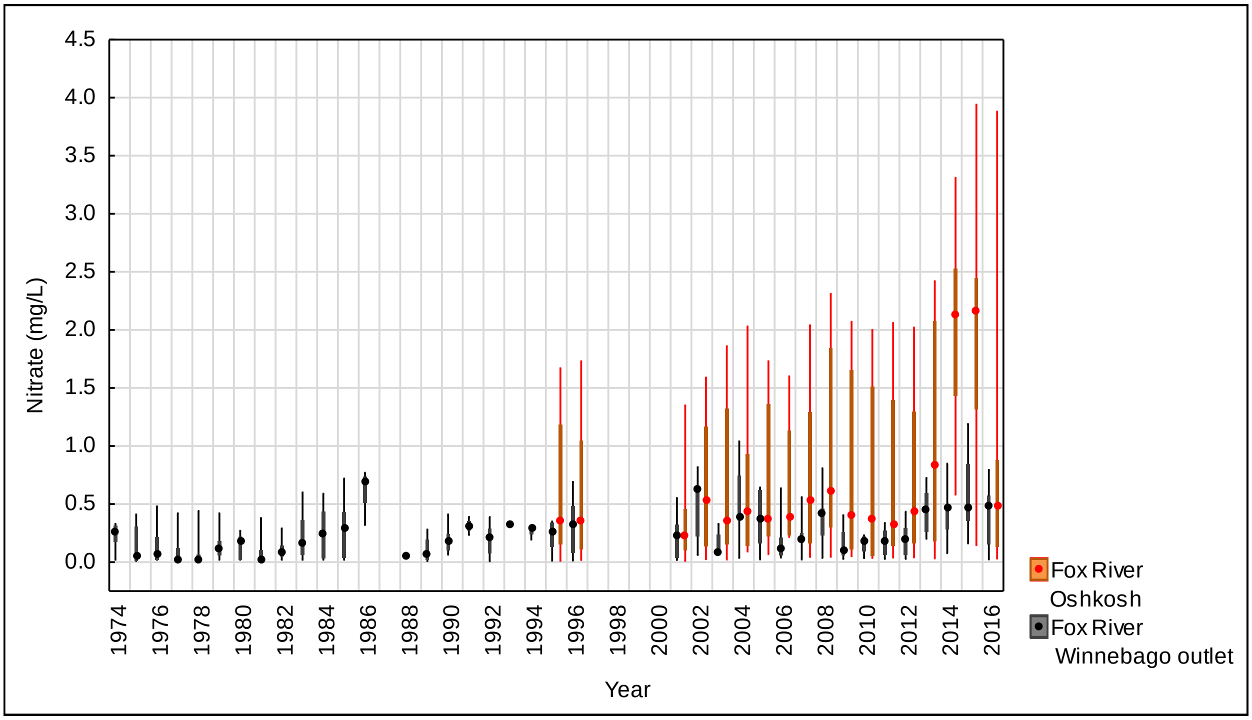


Figure 22. Nitrate concentrations by year, Fox River–A Main St. Bridge Oshkosh (713056) and Fox River–Lake Winnebago Outlet (713002).

Dots represent annual medians, boxes represent 25th to 75th percentiles, and lines represent the minimum and maximum.

3.2 AQUATIC VEGETATION

Aquatic vegetation, especially the extent of rooted macrophytes, can be used to describe the ecological health of a shallow lake. As described in section 2, shallow lakes that lack rooted macrophytes typically contain high levels of planktonic algae. Limited information is available on the overall health and extent of emergent and submergent aquatic macrophytes in the Winnebago Pool Lakes (Johnson 2017). The majority of the historical macrophyte data are from the Upper Pool Lakes. This section summarizes the historical information available on aquatic vegetation and provides a review and summary of recent aquatic vegetation data. A summary of the effect of dam construction on Lake Winnebago and the transition from a macrophyte-dominated system to an algal-dominated lake system is also described.

3.2.1 Historical Information

Several early studies (Harriman 1968, Harriman 1970, Fox–Wolf Lakes Task Force 1974, Linde 1976a, Linde 1980) present information on aquatic macrophytes in the Upper Pool Lakes. Kahl (1993) describes the early history of aquatic vegetation in the Upper Pool Lakes. After the dams at the Neenah and Menasha outlets of Lake Winnebago were built in the 1850s, the system transitioned from a riverine marsh ecosystem to a turbid, open-water lake system in three phases:

1. 1850s–1920s: The damming of Lake Winnebago increased summer water levels by an average of 2 feet (Figure 23). These higher water levels eliminated emergent macrophytes from the deeper parts of the lakes, and created large, floating bogs. Wave and ice action caused the floating bogs to disintegrate, creating smaller floating islands that were easily swept away. The area of open water expanded, increasing wave action and exacerbating the disintegration of the remaining floating bogs.
2. 1930s–1950s: Water levels increased by another 0.5 foot. Floating bog formation and disintegration continued, with other emergent and submergent macrophytes replacing the bogs.
3. 1960s: Macrophyte decline accelerated, probably due to extreme flooding and water turbidity, “especially resulting from nutrient loading from municipal waste water, agricultural lands, unstable shorelines, lake shore developments, carp and freshwater drum, and wave action” (Kahl 1993). The result was a system of large, turbid, open-water lakes.

After the macrophyte declines in the 1960s, the lakes remained turbid in part because macrophytes no longer stabilized lake sediments or dissipated wave action. Nutrients that otherwise would have been taken up by macrophytes were now available for phytoplankton growth. Severe flooding in 1969 and 1973 likely further reduced macrophyte abundance (Kahl 1993) and further stabilized the turbid state of the lakes.

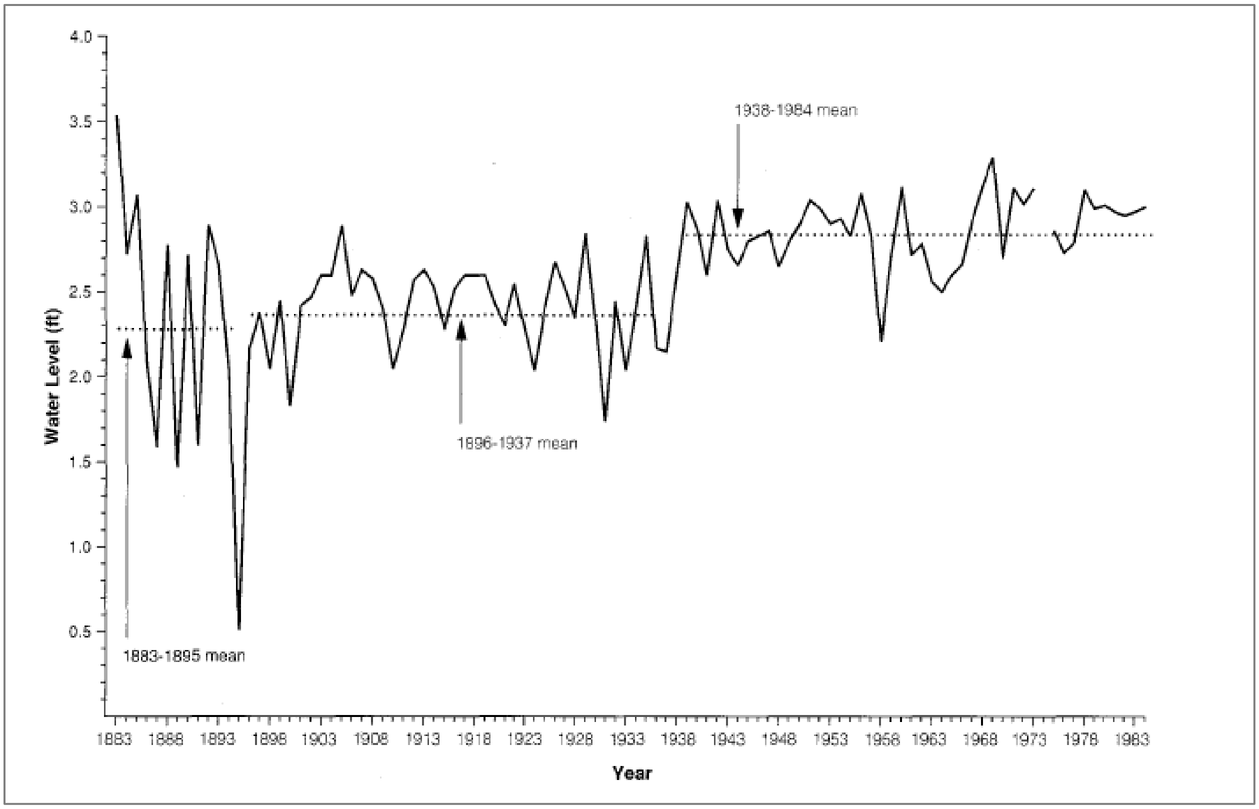


Figure 23. Long-term changes from natural stage and annual fluctuations in mean late-spring and summer (May–Aug) water levels of the Winnebago Pool Lakes as measured at Oshkosh, 1883–1984 (from Kahl 1993, modified from Linde 1975).

The WDNR Bureau of Research conducted a study from 1974–1983 on macrophyte changes and the factors responsible for the observed declines in macrophytes in the Upper Pool Lakes. Kahl 1993 summarizes:

> The primary factors limiting overall abundance of macrophytes during this study [1974–1982] likely included high spring–summer water levels, abnormal timing and magnitude of water level fluctuations, and turbidity. Consistently high water levels in May and June of 1975–84 probably controlled abundance of most emergent macrophytes system-wide. Rapidly rising water levels during the floating-leaf stage throughout June and early July apparently determined system-wide abundance of wild rice. A revised water level management plan implemented in 1982 failed to reduce late spring and early summer water levels.
>
> Low light availability (restricted by water turbidity and epiphyte communities) apparently was the ultimate limiting factor determining long-term, system-wide abundance of submerged macrophytes... However, because of consistently high turbidity throughout the study, late spring and early summer water levels determined the amount of lake bottom within the photic zone, and thus the annual abundance of submerged macrophytes. Primary sources of turbidity for Lake Butte des Morts included the Fox River, the Wolf River at Winneconne, lesser tributaries, and in-lake phytoplankton populations. For Lake Poygan, in-lake sources and lesser tributaries accounted for most turbidity.
>
> Sediments and undesirable fish—primarily carp and freshwater drum—may be more important sources of nutrients than external sources leading to high phytoplankton and epiphyte communities. Wave action and undesirable fish probably have a greater impact on submerged macrophytes in the UWPL [Upper Winnebago Pool Lakes] by contributing to turbidity than through direct physical damage to plants. Injury

> to new shoots and rhizomes by wave action, boats, and undesirable fish may restrict expansion of established stands or prevent re-establishment of perennial emergents in some locations. Furthermore, wave action severely erodes unprotected shorelines, adjacent marshes, and shallow littoral sediments.

Management recommendations in *Aquatic Macrophyte Ecology in the Upper Winnebago Pool Lakes, Wisconsin* (Kahl 1993) include establishing a spring–summer target water level that is under 2.5 feet at the Oshkosh gage; allowing seasonal and annual fluctuations around this water level to mimic a natural hydrologic cycle; implementing large-scale breakwater projects to reduce turbidity; and evaluating macrophyte propagule harvest and planting approaches.

Kahl (2004) presents a cursory evaluation of submerged aquatic vegetation in Lake Poygan and Lake Butte des Morts in 1986 through 1994. The mean rake frequency and frequency of occurrence decreased after 1991 in both lakes. For example, dense stands of wild celery were observed in Lake Poygan in 1986–1989; the stands disappeared after 1991.

Several research projects have investigated the aquatic macrophyte assemblage in Lake Poygan. Dense stands of common reed (*Phragmites australis*) in Lake Poygan and the other Upper Pool Lakes occupy the outer edges where emergent marshes consisting primarily of cattails (*Typha* spp.) formerly existed. Fragmentation and losses of these common reed stands between 1937 and 1997 is documented in Gabriel and Bodensteiner (2002) and Bodensteiner and Gabriel (2003). In Lake Poygan, losses in a patch of vegetation typically have occurred from the outer edges inward, leading to an irregular outline. The larger patch is broken into smaller patches, and the smallest, most vulnerable patches are progressively eliminated. This positive feedback cycle then exposes even more vegetation to wind and wave action, resulting in further losses of vegetation.

Bodensteiner and Gabriel (2003) also evaluated the relationship between the extent of common reed stands in 1937 and 1997 and various environmental variables. As the duration of low water levels and cold temperatures increases, the area of the reed stands decreases. Low water levels in the fall expose more vegetation to increased wind and wave action, freezing temperatures, and disturbance by ice. In the spring and early summer, higher water levels and high turbidity in watershed runoff reduce the availability of light for macrophytes, effectively shortening the growing season. The stress that common reed stands experience in the fall and winter, coupled with high water levels in the spring, could have been the primary mechanisms leading to decreases in the coverage of common reed stands (Bodensteiner and Gabriel 2003).

Gabriel and Bodensteiner (2011) characterized the ecosystem functions of mid-lake common reed stands in Lake Poygan in 1999. Common reed is often an invasive species in North America, and studies typically document the negative effects of common reed on a lake or wetland’s ecosystem function. However, this study found that common reed stands provide multiple ecological benefits in Lake Poygan, including nutrient uptake, fish habitat, and a decrease of wind and wave activity on the leeward side of the stands. This shelter from the wind allows for increased areas of other types of vegetation on the leeward side of a common reed stand compared to the windward side, particularly of emergent and floating-leaved species.

3.2.2 Recent Vegetation Studies

In 2008 and 2009, WDNR completed critical habitat surveys on Lake Butte des Morts, Lake Poygan, and Lake Winneconne and developed draft documentation. Multiple sites were selected in and along the shorelines of each lake, and the following types of information were collected: wildlife observations, critical habitat attributes, plant survey, shoreline inventory, coarse woody habitat survey, and management recommendations. The draft reports developed were *Lake Butte des Morts Critical Habitat Designations*, *Lake Poygan Critical Habitat Designations*, and *Lake Winneconne Critical Habitat Designations*. High frequencies of macrophyte occurrence were observed at several sites along the shoreline of Lake Butte des Morts and the area to the west of Lake Poygan (Figure 24). Low frequencies of macrophytes were observed in some of the shallow areas of the Upper Pool Lakes where higher frequencies would be expected based solely on the shallow water depths.

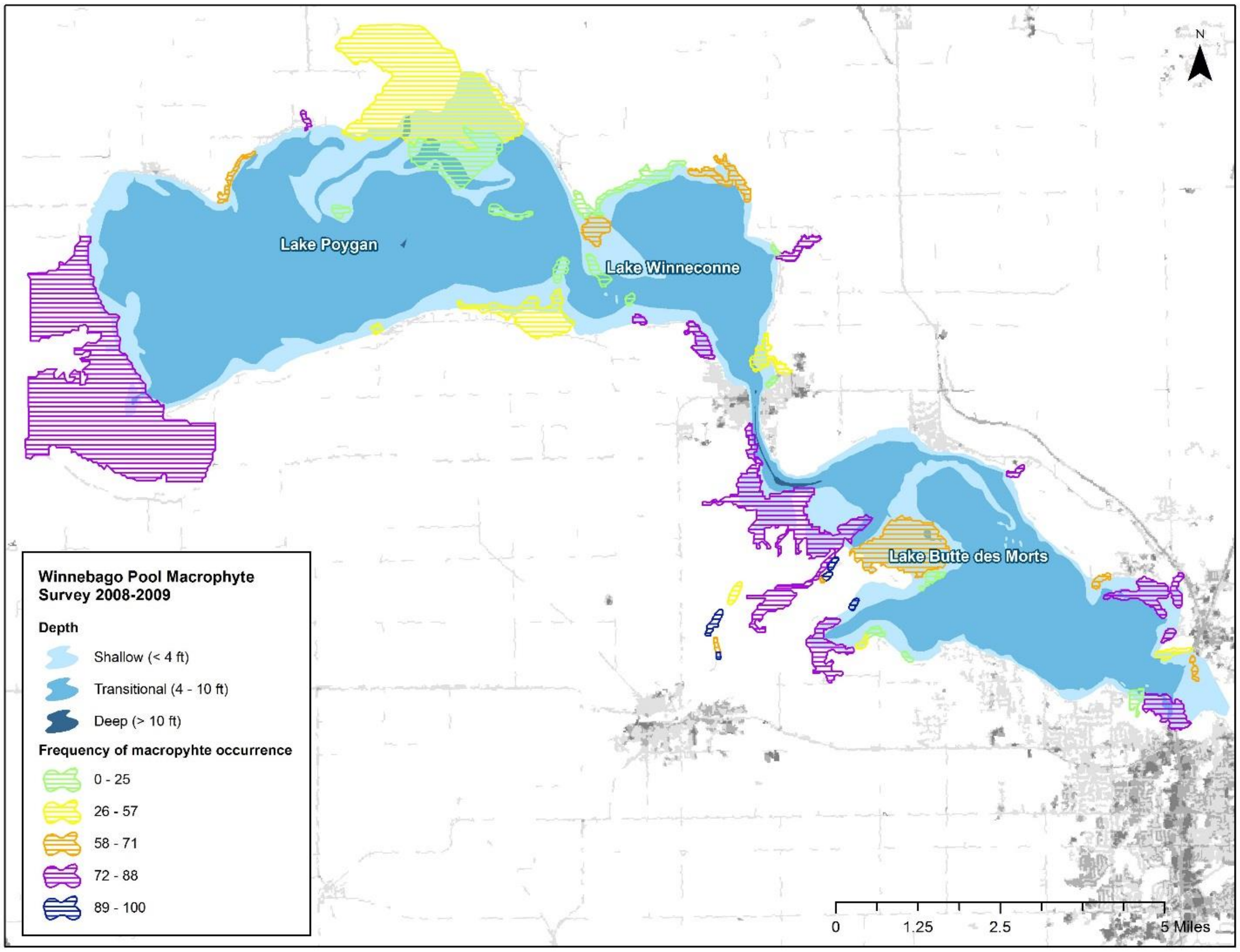


Figure 24. Frequency of occurrence at sites shallower than the maximum depth of plants, Upper Pool Lakes.

Data compiled from draft reports and files provided by WDNR. Depth categories are based on the WDNR's 2017 site index survey (Johnson 2017).

In a comparison of natural and armored shorelines at five paired locations along the Upper Pool Lakes, Gabriel and Bodensteiner (2012) found that natural shorelines had a greater abundance of floating leaved plants than armored shorelines. However, the patterns in vegetation were not consistent among the paired study sites. The authors suggest that differences in plant assemblages might be caused by substrate characteristics, which differ between natural and armored shorelines—natural shorelines had higher proportions of fine and organic material.

In 2017, the WDNR along with stakeholders developed a quantitative sampling approach to assess aquatic macrophytes in representative areas throughout the Winnebago Pool Lakes. The method is repeatable and thus can be used to assess trends in vegetation and is relatively quick given the large size of the system (Johnson 2017). The approach is a modified point intercept survey (Ted Johnson, personal communication). The point intercept method is a grid-based approach that records frequency of occurrence of aquatic macrophytes by measuring the proportion of survey points that intercept vegetation. 119 index sites were selected in the Winnebago Pool Lakes using randomly selected locations within a grid-based system. At each site, the plants were sampled and species identified.

The maximum depth where aquatic macrophytes were observed was greatest in Lake Butte des Morts (9.5 feet) and Lake Winnebago (7.5 feet) and lowest in Lake Poygan (3 feet; Table 1). Of the 66 sites in the entire system that are within the maximum rooting depth, vegetation was found at 52 percent of the sites. A greater percentage of sites within the maximum rooting depth in Lake Winnebago had vegetation compared to the Upper Pool Lakes (Table 1, Figure 25, and Figure 26).

Over half of the species found in the survey are turbidity tolerant, and approximately 25 percent are emergent, have floating leaves, or are a shallow water species and therefore depend less on water clarity. White-stem pondweed and freshwater sponge, which are both intolerant of pollution and turbidity, were both found in Lake Winnebago, indicating that at least portions of the lake have conditions that can support high quality species.

Table 1. Select summary statistics of Winnebago Pool Lakes aquatic macrophyte survey conducted in 2017 (data from Johnson 2017)

Summary Statistics	Winnebago	Butte des Morts	Winneconne	Poygan	Total
Total number of sites visited	61	18	16	24	119
Total number of sites with vegetation	25	5	2	2	34
Total number of sites shallower than maximum rooting depth of plants	33	18	9	6	66
Frequency of occurrence at sites shallower than maximum depth of plants (%)	76	28	22	33	--
Simpson's diversity index [a]	0.78	0.82	0.75	0.78	--
Maximum depth of plants (ft)	7.5	9.5	4.0	3.0	-
Average number of all species per site (shallower than max depth)	1.7	0.7	0.4	1.2	--
Average number of native species per site (shallower than max depth)	1.7	0.6	0.3	1.0	--

a. Simpson's diversity index ranges from 0 to 1 and represents the probability that two individuals randomly selected from a sample will belong to different species. A higher score indicates higher species diversity.

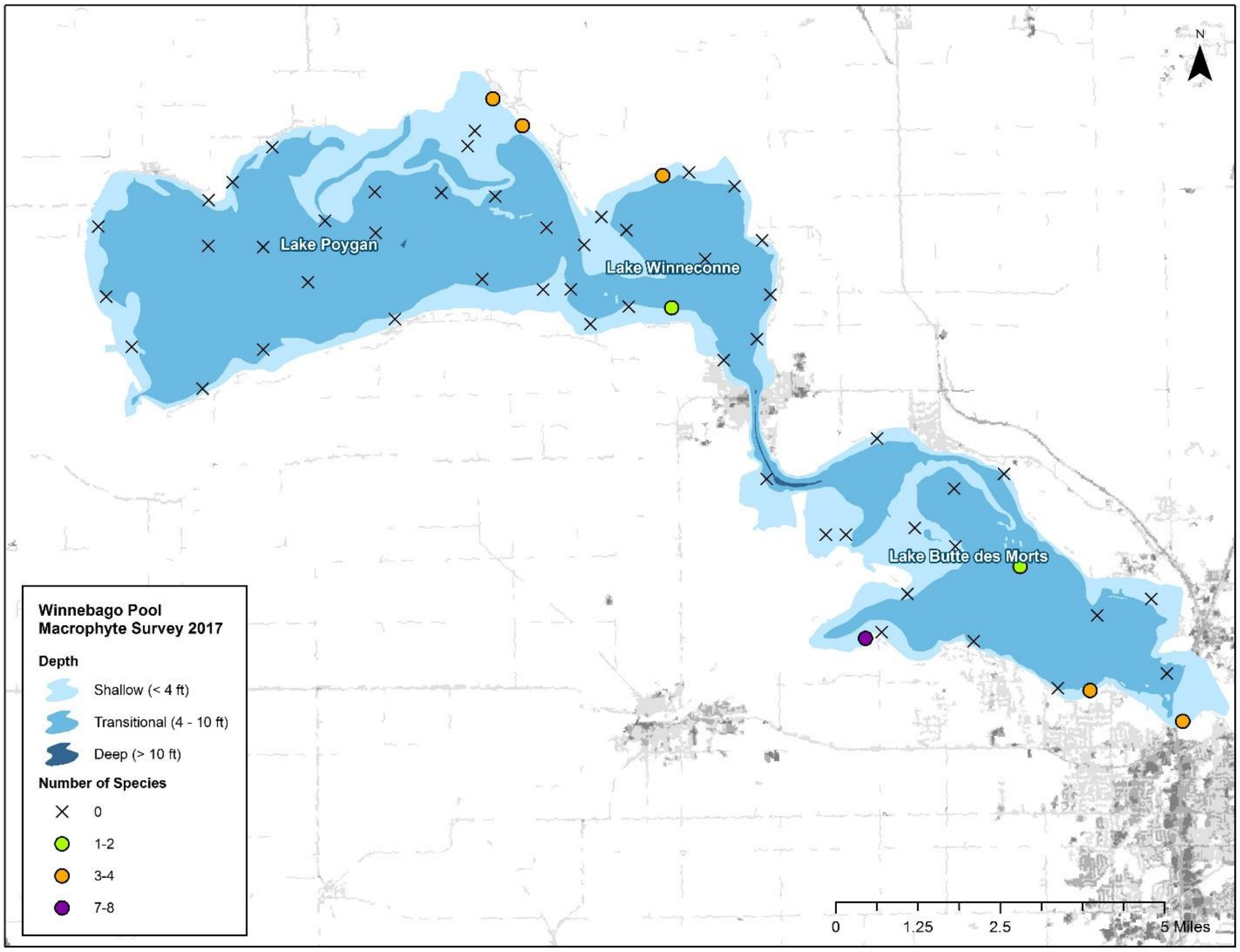


Figure 25. Number of species per site, Upper Pool Lakes.

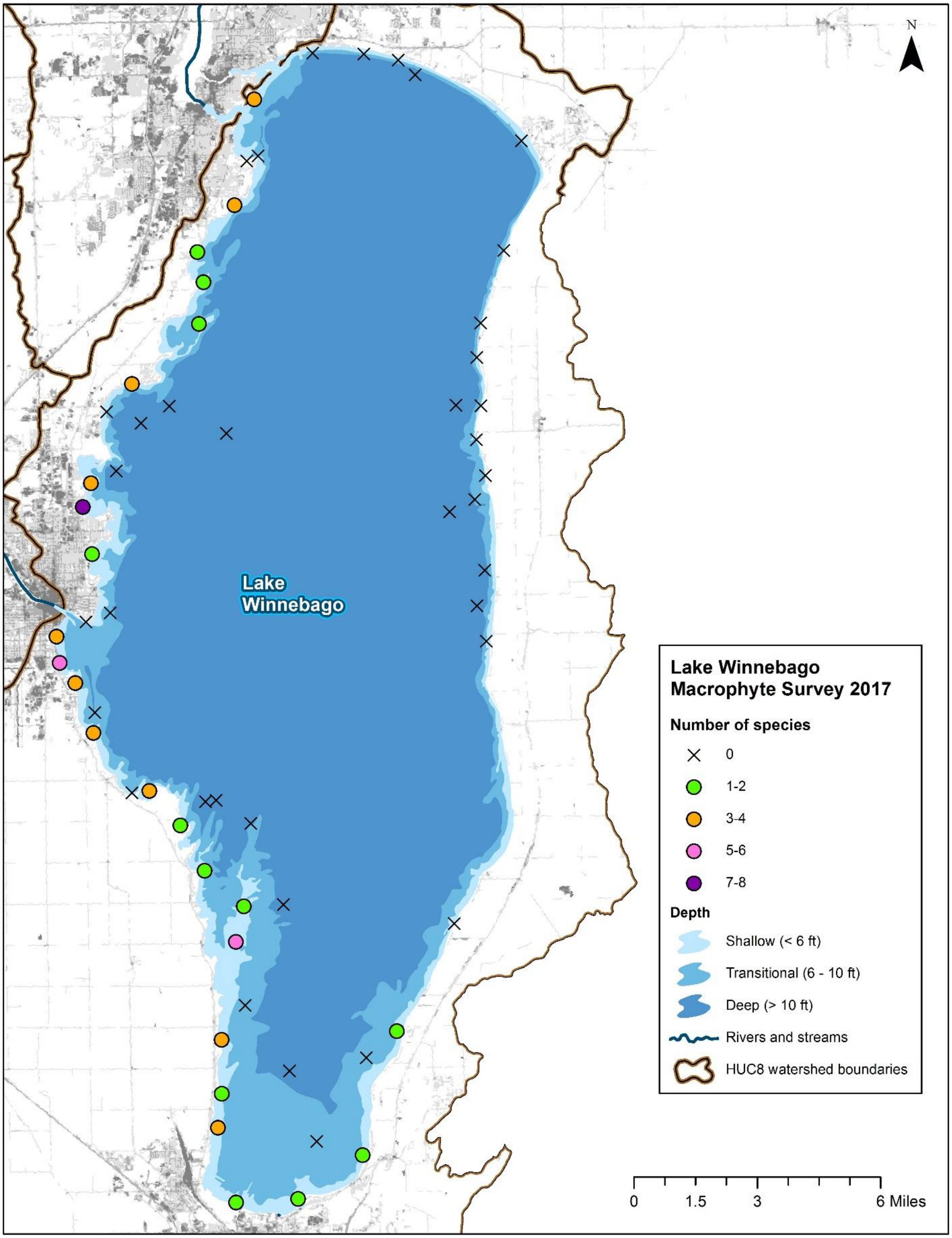


Figure 26. Number of species per site, Lake Winnebago.

While there is a lack of macrophytes in many parts of the Winnebago Pool Lakes, certain bays in Lake Winnebago experience dense growth of invasive macrophytes. Kessenich (2012) documents the status of aquatic macrophyte growth in four bays on the west side of Lake Winnebago, as well as the factors that influence macrophyte growth such as light penetration through the water column and algal abundance. The study was in response to complaints of excessive weed growth from recreational boaters and fishermen. Macrophytes were surveyed in Neenah Bay, Cowling Bay, North Asylum Bay, and Decorah Beach Bay in 2011. The macrophyte surveys found disturbed flora communities, with *Myriophyllum spicatum* (Eurasian watermilfoil), a highly invasive species, at high frequencies in three of the four bays—Neenah, Cowling, and Decorah Beach Bays. *Ceratophyllum demersum* (coontail) was found at high frequencies in Neenah Bay and Cowling Bay. Curly leaf pondweed (*Potamogeton crispus*) was not found at high frequencies; however, the bays were surveyed after this invasive species typically dies off. The study found that approximately 3 percent of the surface area of Lake Winnebago has depths shallower than the average maximum depth of plant occurrence (i.e., 3.79 meters), and approximately 4 percent of the lake has depths that are completely within the euphotic zone. (A lake's euphotic zone is the layer at the surface that receives enough light for plant or algae growth, which is typically approximately one percent of the light that is available at the lake surface.)

The patterns of macrophyte abundance were relatively similar among Neenah, Cowling, and Decorah Beach Bays; North Asylum Bay stood out as having lower macrophyte abundance, higher planktonic chlorophyll concentrations, and shallower light penetration. The report suggests that the lower macrophyte abundances in North Asylum Bay could be due to high wave exposure, high chlorophyll concentrations, and/or lower light penetration in the water column. Wind data superimposed on aerial photographs of the bays indicate that North Asylum Bay was more exposed to wind and wave action than the other three bays.

Artificial channels created in near-shore areas are shallow with little water circulation, and macrophytes often choke these channels. Mismanagement of native aquatic plants (e.g., herbicide overuse) has resulted in growth of Eurasian watermilfoil, which leads to dense monospecific stands of the invasive species; these stands become an even worse problem than the native plant growth that existed before herbicide treatment. The property owners that haven't used herbicides maintain a native aquatic macrophyte assemblage with fewer problems with aquatic recreation (Ted Johnson, WDNR, personal communication).

In the *Miller's Bay, Lake Winnebago, Draft Aquatic Plant Management Plan*, Hoyman and Heath (2009) document excessive growth of aquatic macrophytes in Miller's Bay, also on the west shore of Lake Winnebago. Miller's Bay supports intensive recreational use, and in 2002 excessive growth of aquatic macrophytes was first noticed. This growth became a nuisance in approximately 2004. Plant harvesting was undertaken in 2005 and 2006, but the plants returned to pre-harvest levels within 9 days. In 2007 and 2008, the bay was treated with herbicides, which reduced plant biomass and restored navigation to the southern portion of the bay.

In 2008, the city of Oshkosh initiated an aquatic macrophyte management planning project for Miller's Bay, and a point intercept survey was completed in 2008. Vegetation in the bay was dominated by coontail and Eurasian watermilfoil, with a low species diversity. Curly leaf pondweed was also present in June 2008; however, the plants did not form a dense canopy, and few plants with turions (reproductive buds) were found. The macrophyte assemblage in the bay indicates a disturbed system. Disturbances in the bay include powered watercraft and shoreline development.

The draft plan (Hoyman and Heath 2009) summarizes the two primary issues identified in Miller's Bay: 1) poor quality terrestrial and aquatic habitat, and 2) nuisance vegetation that severely impacts recreational use. The following actions for nuisance plant control are evaluated in the plan: no action, dredging, mechanical harvesting, and chemical herbicides. However, recommendations are not provided in the draft plan.

The "Neenah Channel" is located at the Lake Winnebago outlet that flows through the city of Neenah. The *Neenah Channel Aquatic Plant Management Plan* (Lake and Pond Solutions Co. 2015) reviews the aquatic macrophytes in the channel and describes management options. The city of Neenah has been harvesting "nuisance" aquatic macrophytes in the channel since 2012. A 2016 aquatic plan survey found that the most common species were Eurasian watermilfoil, water celery, and water stargrass. Other species observed were curly leaf pondweed, coontail, and elodea. Elimination of the invasive species is not realistic. To allow for navigational and recreational access, the report states that management should target localized areas of heavy growth and wind-deposited uprooted plants.

3.3 FISH AND WILDLIFE

The fish and wildlife present in a shallow lake are a reflection of the lake's water quality. Algae and aquatic macrophytes can be sources of food to fish and wildlife and can serve as refuge from predation. Additionally, benthivorous fish can directly impact aquatic macrophytes through physical disturbance of lake sediments. *Water Quality in the Lake Winnebago Pool* (WDNR 2004) describes the Winnebago Pool Lakes fishery as a warm water sport fish community supported by the eutrophic conditions in the lake. The Winnebago Pool is known for its walleye (*Stizostedion vitreum vitreum*) fishery, which is one of the best in the United States. The lake has the largest self-sustaining population of lake sturgeon (*Acipenser fulvescens*) in the world (WDNR 2004). Other species that are important components of the fishery include freshwater drum, locally known as sheephead (*Aplodinotus grunniens*), and white bass (*Morone chrysops*). According to the WDNR's Surface Water Data Viewer, other abundant fish are panfish and catfish. Largemouth and smallmouth bass are not as abundant but are still common in the lake, and muskellunge and northern pike are present.

The following presents information on individual fish species in the Winnebago Pool Lakes.

Lake Sturgeon (*Acipenser fulvescens*)

The sturgeon population consists of approximately 40,000 adults, which are harvested annually during a special winter spearing season (WDNR 2004). Bruch (1999) characterized the lake sturgeon population in Lake Winnebago and summarized the history of sturgeon management and the impact that each change in management has had on the sturgeon population. Population assessments and harvest analyses have provided valuable input into management decisions over the years (Bruch 1999).

Winnebago System Sturgeon Spawning Assessments 2010–2016 (Koenigs 2017a) presents results of spawning stock assessments of lake sturgeon in the Winnebago Pool Lakes from 2010–2016. The assessments involved marking fish for estimates of abundance and harvest, monitoring size structure, evaluating growth and mortality, evaluating movement, and determining river and spawning site fidelity of adult lake sturgeon. The majority of spawning sturgeon were captured from the Wolf River. The size structure did not vary appreciably among years, and on average 16 percent of the fish captured were greater than 70 inches, which typically indicates a fish of at least 100 pounds.

2016 Winnebago System Sturgeon Spearing Season, Post-Season Synopsis (Koenigs 2017b) presents the annual numbers of harvested lake sturgeon, lake water clarity, and Lake Winnebago shanty counts for 2002–2016. The percentage of harvested fish that weighed at least 100 pounds was higher in the 2006–2014 spearing seasons (4.8 percent) compared to the long term (1955–2005) average of less than 1 percent. In 2015 and 2016, contributions of these large fish decreased to 2 percent. Variability in fish condition (i.e., fish weight divided by its expected weight based on length) is influenced by the abundance of forage, and the primary prey of lake sturgeon are Chironomid lake fly larvae (also known as redworms) and gizzard shad. Water clarity is the biggest predictor of spearing success.

Using gut content and stable isotope analysis, Stelzer et al. (2008) determined the primary winter prey of lake sturgeon in Lake Winnebago. Gizzard shad (*Dorosoma cepedianum*) and the chironomid larvae *Chironomus plumosus* were each primary components of the sturgeon's winter diet. Anderson et al. (2012) measured secondary production of chironomids in Lake Winnebago in 2008 and 2009 and found that chironomid production was sufficient to support the lake sturgeon population in Lake Winnebago. The chironomid production rates measured in Lake Winnebago are high relative to many North American lakes; the authors suggest that this could be due to the eutrophic conditions in the lake.

Walleye (*Stizostedion vitreum vitreum*)

The objectives of the trawling assessment reported in *2016 Lake Winnebago Bottom Trawling Assessment Report* (Nickel 2017a) were to provide information on year class strength of game and nongame fish species, monitor population trends of game and nongame fish species, and monitor trends in the forage base. The same methods have been used to sample Lake Winnebago annually during the first week of August, September, and October since 1986. Strong year classes of walleye are often due to high spring water levels that create favorable conditions in walleye spawning habitat. In addition to providing spawning adults access to spawning habitat, the high water levels also keep eggs aerated and flush out newly hatched fry. Walleye fry feed on zooplankton, and therefore adequate zooplankton in the Upper Pool Lakes are needed. Walleye young-of-year abundances vary annually based on growing conditions. A relatively high young-of-year walleye catch rate was observed in a 2016 trawl survey and was attributed to high spring water levels in the Wolf River (Nickel 2017b).

Sauger (*Stizostedion canadense*)

Priegel (1969) characterized the life history of the sauger, which is an abundant and important game fish species in Lake Winnebago. Sauger were stocked in Lake Winnebago from 2001–2010, and habitat improvement projects were completed to rebuild the adult sauger population and support natural reproduction (Nickel 2017a). Young-of-year sauger increased during the years of the improvement program, but numbers declined after stocking ended in 2010. Some natural reproduction does occur in the lake. It is expected that, without stocking, the sauger population will persist in Lake Winnebago at densities similar to before stocking began in 2001 (Nickel 2017a).

Yellow Perch

2015–2016 Winnebago System Yellow Perch Stock Assessments, Post-Season Synopsis (Koenigs 2017c) describes results from two types of yellow perch surveys. Panfish, including yellow perch, depend on aquatic vegetation for spawning habitat, food production, and refuge from predators. Lake Winnebago bottom trawling surveys show an increase in yellow perch recruitment from 2001–2011, influenced by higher water clarity and an increase in vegetation. Recruitment decreased in 2012–2015, accompanied by a decrease in aquatic vegetation. Fyke netting in Lake Winnebago and Lake Butte des Morts from 2012–2016 also indicates declining perch abundance since 2012. The data suggest that the yellow perch fishery in the Winnebago Pool Lakes is a boom–bust fishery that depends on strong year classes; abundance and fishing success are high for two to three years following a strong year class. Restoring rooted aquatic macrophytes is needed to support the yellow perch fishery in the Winnebago Pool Lakes.

Black Crappie (*Pomoxis nigromaculatus*)

2016 Lake Winnebago Bottom Trawling Assessment Report (Nickel 2017a) also includes information on black crappie in Lake Winnebago. Vegetation is a primary driver of crappie year class strength. Crappies build nests, and vegetation provides material for nest production in addition to cover. Increasing water clarity and increased vegetation in the mid to late 2000s coincided with strong 2009 and 2010 crappie hatches. The young-of-year crappie catch from 2016 was the highest catch rate on record, and anecdotal reports suggest that vegetation growth was “some of the best vegetation growth” observed since the late 2000s.

Carp

Common carp are benthivorous fish that feed in lake sediments and disturb rooted aquatic vegetation. The loss of vegetation leads to unstable sediments, and the physical disturbance of sediments can release phosphorus into the water column. While abundance of carp is not quantified in the WDNR’s fisheries surveys, in Lake Butte de Morts commercial fishermen removed 750,000 pounds of rough fish, including carp, in November 2013 and 41,000 and 350,000 pounds on two different days in November

2014 (Nickel 2014). Based on observational information and the amount of fish removed through commercial seining, carp likely affect the water quality in the Upper Pool Lakes.

Forage Fish

Trout fish, gizzard shad, and freshwater drum are important components of the diet of many gamefish and panfish species, as discussed in *2016 Lake Winnebago Bottom Trawling Assessment Report* (Nickel 2017a). After declines in trout perch in 2010–2012, trout perch rebounded in 2016. Gizzard shad often have boom and bust recruitment cycles. In years following strong gizzard shad hatches, certain gamefish species often become “tight lipped,” making them more difficult for anglers to catch. Freshwater drum year classes do not vary as much, which provides stability to the gamefish forage base.

Changes in the water quality of the lake could lead to changes in the composition of the lake’s fishery (WDNR 2004). Certain species such as freshwater drum, common carp, and some sucker species prefer turbid, nutrient rich water; these species could decline if water quality in the lake improves. Increases in aquatic vegetation would promote fish species such as panfish that prefer denser plant growth.

Water Quality in the Lake Winnebago Pool (WDNR 2004) identifies the most problematic invasive species in the Winnebago Pool Lakes as common carp, zebra mussels (*Dreissena polymorpha*), and Eurasian watermilfoil (*Myriophyllum spicatum*). Kessenich (2012) discusses the potential implications of zebra mussels in Lake Winnebago. Zebra mussels were first observed in the lake in 1998, and Kessenich suggests that chlorophyll concentrations in the lake decreased after 1998, potentially as a result of the filter feeding of zebra mussels. In other large, shallow lake systems, zebra mussels have altered the aquatic macrophyte assemblage. For example, in Oneida Lake, NY, water clarity increased in the decades after the colonization of zebra mussels. An increase in diversity and frequency of submerged macrophytes followed the increase in clarity (Zhu et al. 2006). In Lake Michigan, zebra mussels have altered the cycling of phosphorus within the lake such that the nearshore zone has become a net sink for phosphorus, and the pelagic zone has become a significant source of phosphorus to the nearshore zone (Bootsma et al. 2012). A meta-analysis of studies and monitoring data from lakes and rivers across North America and Eurasia indicates that invasive mussels can “shift aquatic food webs and energy flow from pelagic–profundal to benthic–littoral energy pathways” (Higgins and Vander Zanden 2010). In Lake Winnebago, the limited data preclude an analysis of the effect of zebra mussels on rooted aquatic macrophytes. Bart De Stasio from Lawrence University conducted a study on the changes in the zooplankton community following invasion by zebra mussels in Lake Winnebago. However, the data and documentation were not available for this report.

As summarized in the *Aquatic Invasive Species Strategic Plan for the Winnebago Pool Lakes* (Winnebago Pool Lakes Aquatic Invasive Species Advisory Group 2008), other aquatic invasive species include curly leaf pondweed, purple loosestrife, reed canary grass, and flowering rush, in addition to viral hemorrhagic septicemia, which is a disease caused by an invasive virus. Invasive species that were not present in the lake in the 2000s but have the potential to enter the Winnebago Pool Lakes include white perch (*Morone americana*) and the round goby (*Neogobius melanostomus*).

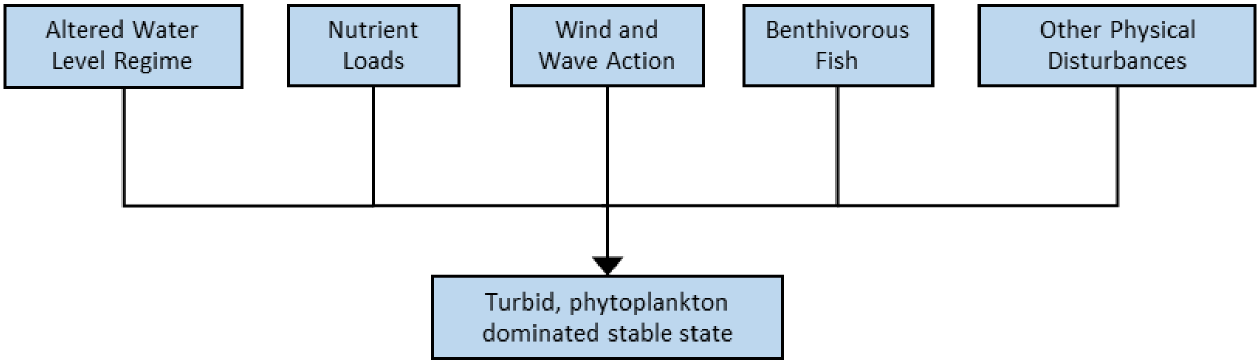
The Fauna of the Lake Winnebago Region (Baker 1924) describes in detail the mollusks that inhabit the different habitats in Lake Winnebago. Other animals and macrophytes are also discussed, although in less detail. The report contains descriptive information about the lake and its surroundings, including notes on the water transparency in the summer of 1920, “the turbidity of the water was great and transparency was reduced to a minimum. A white disc could be seen only to a depth of a few inches in the open lake.” The maximum depth of the lake was 6.38 meters at the time, and plants were abundant, occurring commonly to a depth of 2 meters. This observation of abundant macrophytes in highly turbid waters runs contrary to the current understanding of shallow lake ecology (section 2). The report was conducted during a transitional period of the Winnebago Pool Lakes when higher water levels eliminated emergent macrophytes from the deeper parts of the lakes (section 3.2.1).

Other wildlife mentioned in the literature include canvasback ducks (Kahl 1991). In the 1950s and early 1960s, the Upper Pool Lakes hosted peak fall populations of canvasback ducks of 8,000 to 77,000. After habitat quality in the lakes declined, the ducks stopped using these areas as staging habitats. The report suggests that the Upper Pool Lakes have potential for management and restoration of staging habitat for canvasback ducks. A follow-up study, *A Cursory Evaluation of Canvasback Migrational Staging Habitat in South and East Central Wisconsin, 1985–1994* (Kahl 2004), collected data on canvasback populations during peak migration periods on 25 Wisconsin lakes, including the Winnebago Pool lakes. Other data collection included water quality measurements on the three Upper Pool Lakes and surveys of submerged aquatic vegetation on Lakes Butte des Morts and Poygan. Lakes in the study attracted fewer migrational staging populations of canvasback ducks during the study period compared to prior surveys. Lake Poygan attracted more canvasbacks than the other lakes in the study and had the largest known wild celery beds.

4 WINNEBAGO POOL LAKES STRESSORS

The ecological balance in shallow lakes can be shifted by natural and anthropogenic disturbances. Multiple disturbances have acted on Lake Winnebago and the Upper Pool Lakes and have led them to being in a persistent, turbid, phytoplankton-dominated state. Altered water level regime, increased nutrient loads from external and internal sources, high wind and wave action, benthivorous fish activity, and other physical disturbances all exert pressures on the lakes.

Other factors such as aquatic invasive species and climate change might also affect the ecological interactions in the Winnebago Pool Lakes. For example, zebra mussels have the potential to increase water clarity and to shift the energy flow in lakes from the pelagic zone to the littoral zone (section 3.3). Curly leaf pondweed affects phosphorus cycling in lakes, often leading to a release of phosphorus in the lake when curly leaf pondweed senesces in early summer. The current role of these stressors in the Winnebago Pool Lakes is not known and is not included in the conceptual model. Ongoing and future research on the effects of aquatic invasive species and other potential stressors on water quality in the Winnebago Pool Lakes will provide insight into the role of these stressors in the system.



Shallow lakes that are in a turbid, phytoplankton-dominated state are prone to algal blooms, in which the amount of algae in the water column increases rapidly. Additionally, certain environmental conditions can favor the growth of cyanobacteria, which are also known as blue-green algae. Cyanobacteria are unique in that they can produce their own energy through photosynthesis that produces oxygen, like true algae. Many species of cyanobacteria are also able to fix atmospheric nitrogen, giving them a competitive advantage over non-nitrogen fixing species when nitrogen is in short supply. Some species of cyanobacteria produce toxins such as microcystins and cylindrospermopsins, which are hepatotoxins (liver toxins), and anatoxins, which are neurotoxins.

This section presents a summary of the existing information on the ecological interactions in the Winnebago Pool Lakes and the potential stressors that drive algal and macrophyte dynamics in the lake.

4.1 ALTERED WATER LEVEL REGIME

Dams located in Neenah and Menasha on the outlet of Lake Winnebago were built in the 1850s. As a result, water levels initially were raised by approximately 2.5 feet; subsequent modifications to the dams raised water levels by an additional 0.5 foot (WDNR 2004). Water level alterations at these two dams control the lake level in both Lake Winnebago and the Upper Pool Lakes.

The US Army Corps of Engineers (USACE) has regulated water levels in the Winnebago Pool Lakes since construction of the dams. *Building Strong®, Lake Winnebago: Fox-Wolf River Basin* (USACE n.d.), a fact sheet produced by the USACE, summarizes the history and current approach to water level regulation in the Winnebago Pool Lakes. A high water level limit on Lake Winnebago was defined in the 1886 Marshall Order, which was established to maintain water levels below flood stage. Flood stage was defined as 3.45 feet above the Oshkosh Datum, a local datum that is referenced to the crest of the Menasha Dam. The Marshall Order was modified in 1920 to address navigational needs, and a lower water level limit of 1.68 feet Oshkosh Datum was established for the navigation season[2].

Water levels are manipulated within the confines of the Marshall Order and to balance the needs of multiple users. All water levels presented here are with respect to the Oshkosh Datum[3].

- Winter drawdown (represented in Figure 27 by segment e–a): Water levels are slowly drawn down over the winter (early January through March 1) to provide capacity to hold spring flood waters. A March 1 lake level target is set annually and is typically 1.68 feet, the lower water level limit in the Marshall Order. The March 1 lake level target might be higher depending on winter and spring conditions from that year, with the goal of achieving the June 1 lake level target of 3.0 feet. For example, in years of lower than average snowfall, spring runoff would typically also be lower than average, and the March 1 target water level would be increased to ensure that the June 1 target is reached.
- Between drawdown and ice-out (segment a–b in Figure 27): The water level is typically raised slowly and gradually through the early spring. Large water level increases are avoided because they can cause ice damage to wetlands and shoreline structures.
- Spring refill (segment b–c in Figure 27): The lake is refilled after ice out, which typically occurs at the beginning of April. During refill, the lake water levels are maintained within a target range, shown by the 0.3-foot operating band in Figure 27. During dry seasons, the water levels can be increased more quickly, using the upper targets, to ensure that the June 1 target is met. During wet seasons, there is a greater potential for flooding and damage to aquatic vegetation, in which case the lower refill water level targets would be used.
- Summer (segment c–d in Figure 27): The summer target water level in Lake Winnebago is 3.0 feet. When water levels in the lake exceed 3.0 feet, the resulting high water levels in the Upper Pool Lakes can lead to substantial environmental damage. Water levels at times drop below 3.0 feet due to low precipitation and high evaporation losses.
- Between navigation season and freeze-up (segment d–e in Figure 27): The water level is lowered gradually to a *freeze-up* water level that is determined annually in the fall. Large changes in water level, which can impact lake habitat, are avoided from approximately October 1 through freeze-up. Ice formation typically begins between mid-November to late December.

The Great Lakes Hydraulics and Hydrology Office of the USACE Detroit District oversees water level regulation of the Winnebago Pool Lakes. Water levels at the USACE's four Lake Winnebago gages are averaged and compared to seasonal water level targets. Daily regulation decisions are made in conjunction with local personnel at the USACE field office in Kaukauna, Wisconsin. Adjustments of the private dam at Neenah are made by private contractors of Neenah Paper in coordination with USACE staff.

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[2] A nautical chart of Lake Winnebago and the Lower Fox River was published by NOAA (2017). WDNR also published a navigation chart with lake depths of the Winnebago Pool Lakes (*Navigation Chart of Lake Winnebago and Surrounding Navigable Waters*, WDNR n.d.); however, the document states that it is not for navigation purposes.

[3] The Oshkosh Datum is a local datum from approximately 100 years ago. The original survey points no longer exist and it is not possible to tie back to the datum (C. Shaw, USACE, personal communication). In managing water levels on Lake Winnebago, the USACE compares water levels to the current crest of the Menasha Dam (1.68 feet).

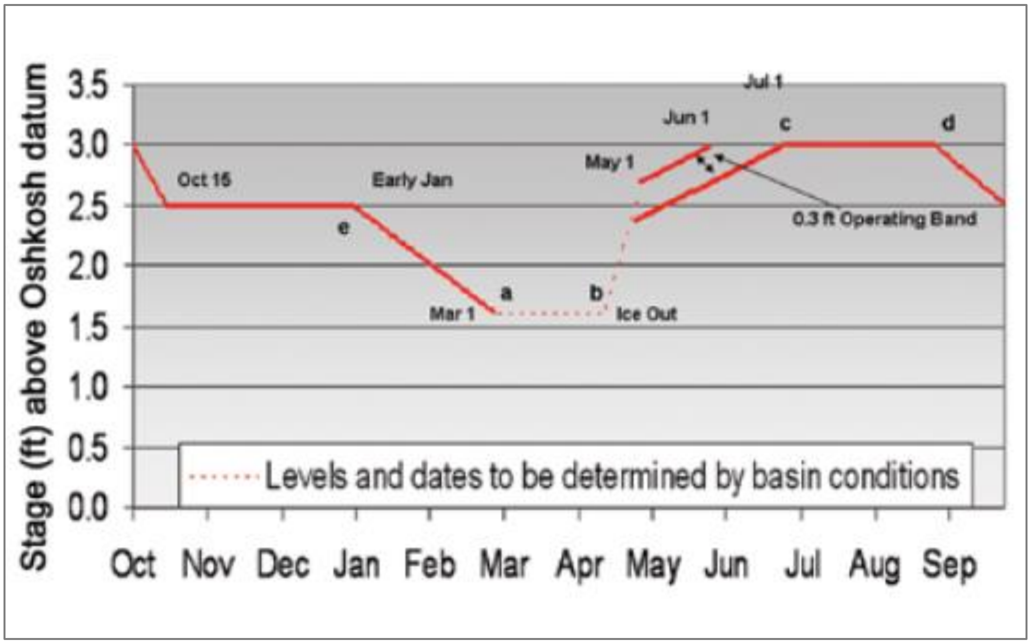

Figure 27. Seasonal targets. Figure from "Building Strong, Lake Winnebago: Fox-Wolf River Basin" (USACE n.d.).

Suggested Water Level Control for the Winnebago Pool (Linde 1976b) is locally referred to as the "Linde Plan" and recommends water level targets within the confines of the Marshall Order. The recommendations aim to balance the needs of the various lake users with the ecological health of the lake (Figure 28):

- The March water level low should be less than 1.0 feet (noted as "Oshkosh gage" in the *Linde Plan*), and subsequent increases in water level should be slow enough to allow growth of submergent vegetation before peak water levels are reached later in the growing season.
- To minimize negative ecological impacts of ice action during the ice break-up period, water levels shouldn't reach the spillway level until April 10.
- Increases in water level in the spring should be timed to allow sufficient light to reach growing submergent macrophytes. Gradual increases in water level will allow growing macrophytes to keep pace with the increases in water levels.
- Water levels should peak in June, with no further increases throughout the summer. A slow decrease in water levels from June through August would be preferable to stable water levels, because declining water levels can help plants set seeds.
- Water levels in September and October can rise in response to fall storms; these higher water levels would provide better conditions for waterfowl hunting and boating.
- Sudden increases in water levels should be avoided year-round because of the damaging effect on vegetation.

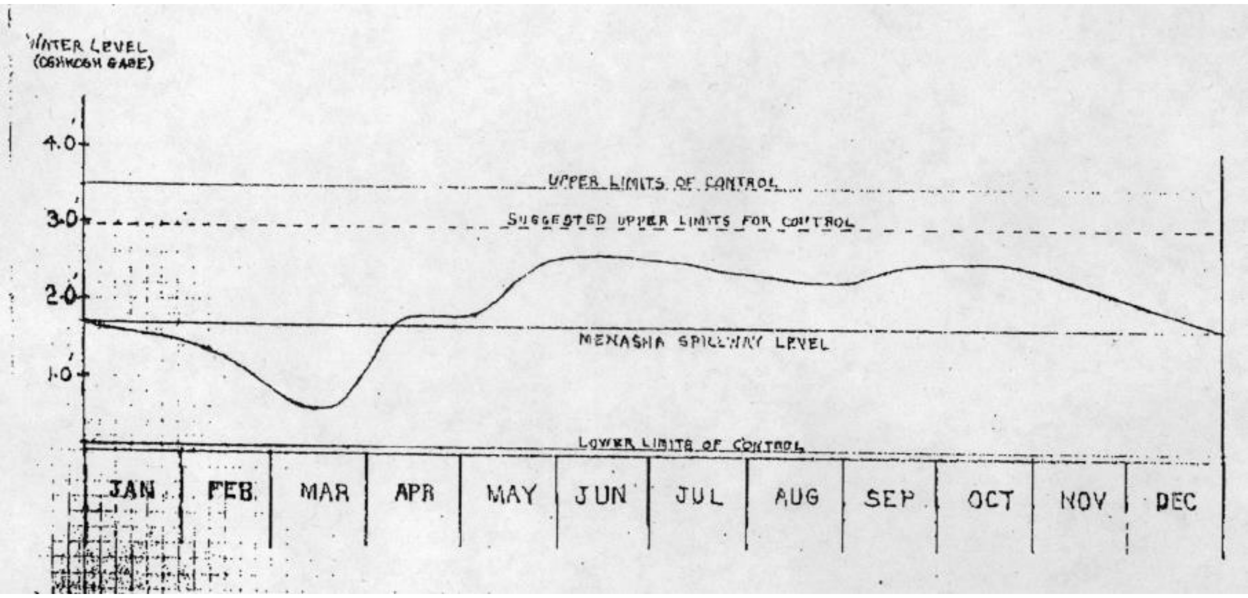

Figure 28. Suggested water level regime for Winnebago Pool (Figure from Linde 1976b).

Lake level management has led to deeper waters in all of the Winnebago Pool Lakes. The water depths in a substantial area of the Upper Pool Lakes are still shallow enough to support aquatic macrophytes (Figure 25). Other stressors, such as the timing and rate of water level change, wind and wave action, and nutrient loading, prevent the lake from sustaining a healthy aquatic macrophyte assemblage. Lake Winnebago, on the other hand, is on average deeper than the Upper Pool Lakes and a smaller proportion of the lake can support aquatic macrophyte growth (Figure 26). Altered hydrology in the watershed has changed the flow rate, volume, and timing of watershed inputs to the Winnebago Pool Lakes, which has also altered residence time in the lakes.

Anecdotal observations suggest that the aquatic macrophyte assemblage is typically higher quality when the *Linde Plan* (Figure 28) is followed more closely, and plant survival is often lower in years when the *Linde Plan* is not followed (Ted Johnson, WDNR, personal communication).

The State of the Upper Fox River Basin (WDNR 2001) includes discussion about water level manipulation in the watershed:

- Lake level manipulation in the shallow lakes in the Upper Fox River Basin, which includes the Winnebago Pool Lakes, has “disrupted the natural high and low water cycles that the shallow lakes depend on to maintain natural habitat” (Figure 29). The resulting stable water levels in the summer have decreased aquatic plant generation. The report states that, “The best management option to restore habitat and fish and wildlife populations on these very valuable shallow lakes is to restore natural fluctuations in water elevations, or at a minimum, manage water levels in a manner that mimics natural fluctuations.” One of the priorities laid out in the report is to “pursue ecologically sound water level management on shallow lakes.”
- Public pressure to raise water elevations in the shallow lakes in the Upper Fox River Basin is due in part to the perception that lakes should be deep and free of “weeds.”
- Water levels in Lake Winnebago can increase due to high water levels in the Wolf and Upper Fox Rivers, resulting in gradual increases in lake water levels that can last for a week or more. Heavy localized precipitation on the surface of Lake Winnebago can lead to more abrupt increases in lake water level. Additionally, strong winds across the lake surface can raise the water level on one side of the lake, which increases the potential for flooding and shoreline erosion. North-south winds have led to water level differences of up to 1.25 feet on opposite sides of the lake.

- Seasonal fluctuations in lake water levels are influenced by spring snowmelt and precipitation, which lead to the greatest rise in lake water levels. Evaporation from the lake's surface can represent substantial water losses and decreases in lake water levels.
- The USACE keeps water levels as low as possible when the layer of ice on the lake is thin. Strong winds can push thin ice around the lake, leading to "ice shoves," where ice piles up against the shoreline. Ice expansion, as a result of periods of varying temperatures, can crack and form pressure ridges where ice will pile up. Ice shoves and ice expansion both have the potential to damage shoreline structures such as docks.

There is potentially room for improvement in macrophyte establishment within the constraints of the *Linde Plan* (Ted Johnson, WDNR, personal communication). One option could be to lower water levels earlier in August, which would better mimic the natural system and provide better habitat for macrophytes. Another option is to manage water levels to be slightly lower in the spring and early summer.

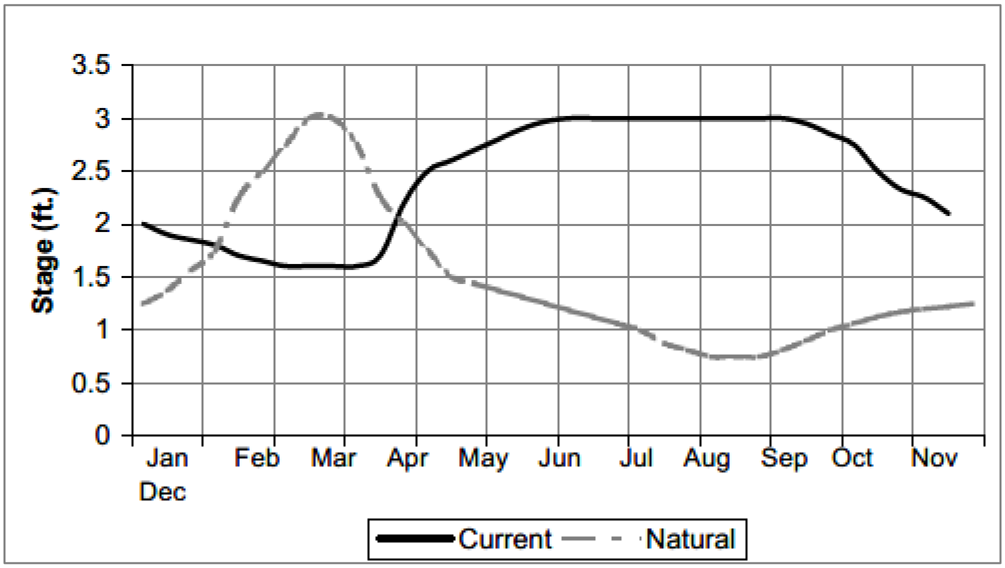

Figure 29. Lake Winnebago current and natural fluctuations in water level.

Figure from Water Quality in the Lake Winnebago Pool (WDNR 2004).

Gabriel (2004) surveyed residential property owners along the shorelines of the Winnebago Pool Lakes to determine their hazard experiences, adjustments, and management preferences in regards to fluctuating water levels. The majority of respondents had been impacted by shoreline hazards (e.g., flooding or high water levels) and support water level regulation to reduce shoreline hazards. A majority of respondents also believe that managing lake water levels with dams can stop most flooding and erosion. However, the majority of respondents are not aware of the technological limitations of using lake level regulation to reduce shoreline hazards, and the majority of respondents prefer water levels to stay the same or increase, which would actually increase the likelihood of shoreline hazards. There is a need to educate the public on the technical limitations of water level management to reduce shoreline hazards, the types of strategies used to mitigate the effects of hazards, and the financial and technical assistance available for hazard management from government agencies.

4.1.1 Water Level Data Inventory

Water level data on Lake Winnebago were compiled from the following sources:

- Water level data of Lake Winnebago at four USACE gages: Menasha, Oshkosh, Stockbridge, and Fond du Lac. Daily average water levels provided by USACE.
- Water level data of Lake Winnebago at Oshkosh and near Stockbridge. Downloaded from USGS NWIS.
- Water level data of Lake Poygan. Provided by USGS to Winnebago Waterways.

4.1.2 Water Level Data Assessment

Water levels vary annually in Lake Winnebago (Figure 30) due to water level management and the variability and unpredictability of weather and hydrologic response. Annual maximum water levels in Lake Winnebago typically occur in May, June, or July, and range from 2.96 feet in 2006 to 3.79 feet in 2008. Annual minimum water levels occur in February or March and range from 1.12 feet in 2014 to 2.00 feet in 2012.

Figure 30 compares water level data from Lake Winnebago to established water level limits and targets:

- Daily average water levels in Lake Winnebago (2002–2017) are provided to show the general pattern of measured water levels.
- USACE seasonal targets (Figure 27) are shown in black.
- The thick yellow curve is the suggested water level regime in the Linde Plan, the thinner yellow line at 3.0 feet is the Linde Plan's suggested upper limit for control, and the yellow dashed line at 3.45 feet is the Marshall Order's and Linde Plan's upper limit of control (Figure 28).
- The green curve represents the natural fluctuations expected in Lake Winnebago (Figure 29).

The measured water levels generally follow the USACE's seasonal targets, however, the USACE summer target of 3.0 feet is the suggested *upper* limit for control in the Linde Plan, which is higher than the Linde Plan's suggested water level regime (thick yellow line in Figure 30).

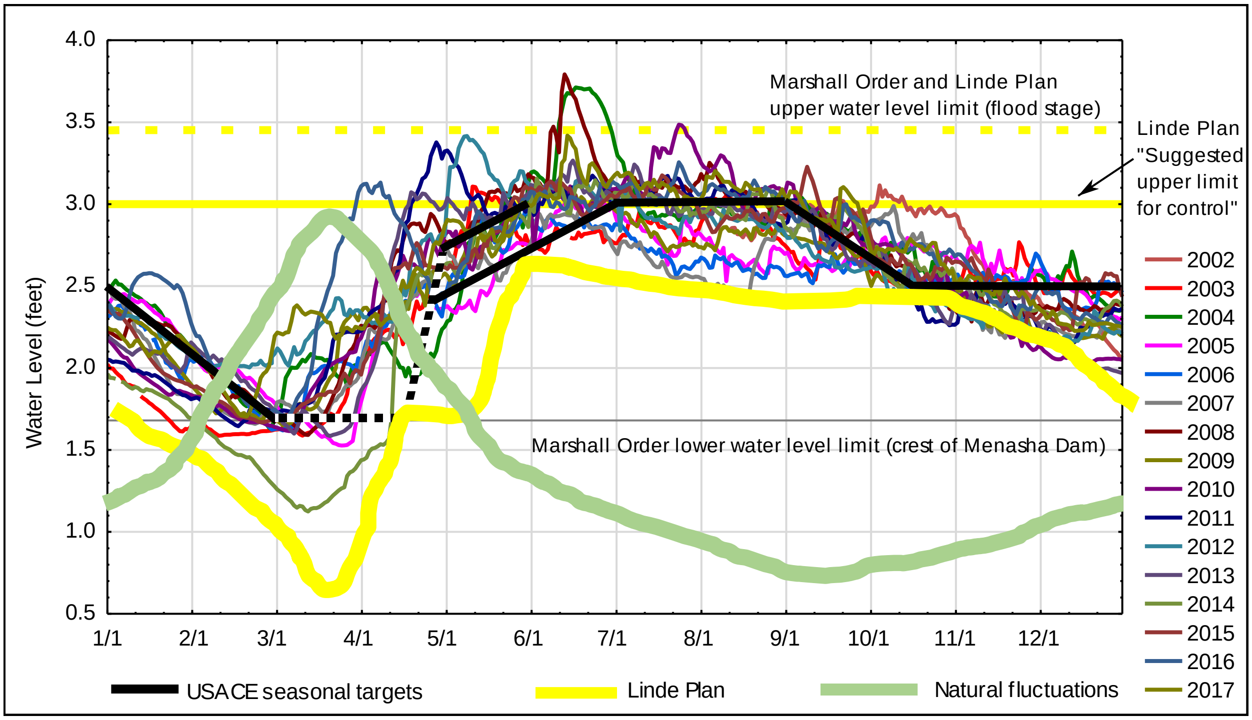

Figure 30. Daily average Lake Winnebago water levels, averaged over four USACE gages (Menasha, Oshkosh, Stockbridge, and Fond du Lac) 2002–2017.

The relationship in 2009–2011 between water level and water quality, as measured by chlorophyll concentration, was examined. These three consecutive years have the greatest number of samples out of the most recent 10 years. Chlorophyll concentration increased over the course of the growing season, with the highest chlorophyll concentrations observed in 2011 (Figure 31). The water level in 2011 peaked earlier than in the other two years, with a steeper rate of water level increase in the spring (Figure 30, Figure 31). While this one observation is not enough to conclude that earlier and more rapid spring increases in water level lead to increased algal growth, the observation does fit the expected relationship of water level to water quality in shallow lakes. Rapid increases in water level in the spring can damage sensitive stages of aquatic macrophytes and limit light availability to macrophytes. The loss of macrophytes can destabilize lake sediments, increase wind and wave action on the lake sediments, and increase nutrient availability in the water column, all of which can lead to higher algal growth.

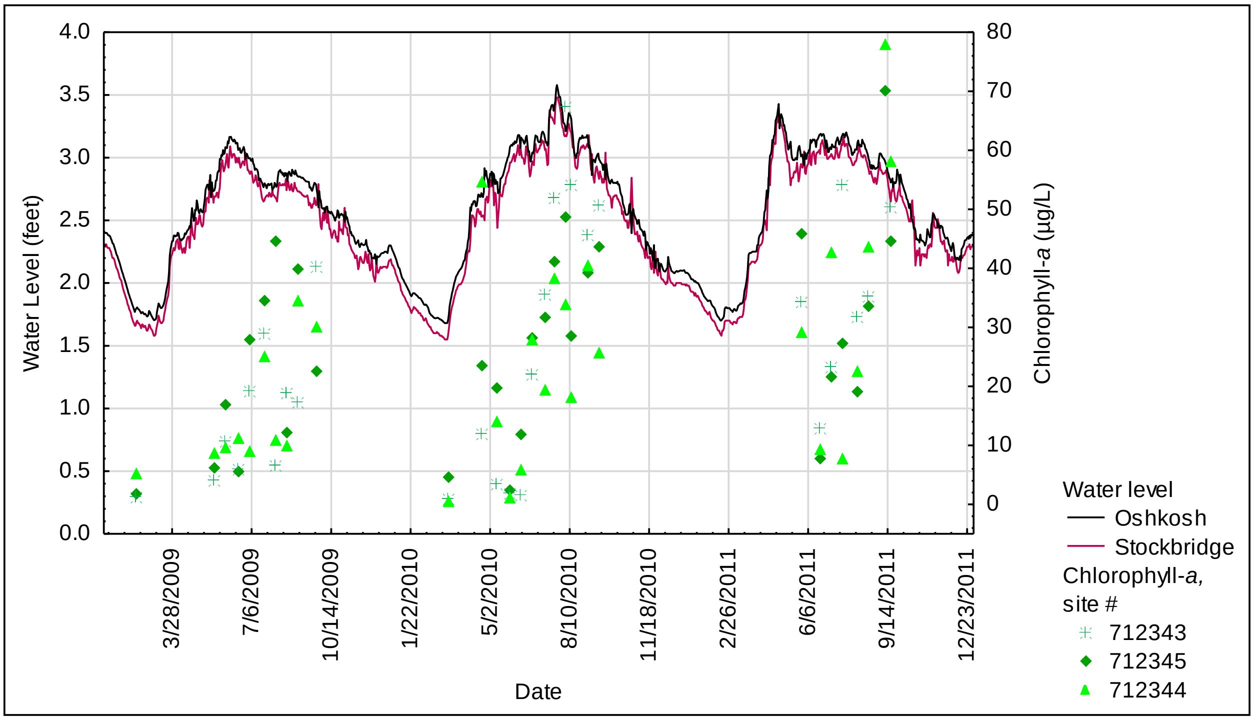

Figure 31. Lake Winnebago water levels and chlorophyll concentrations, 2009–2011.

4.2 NUTRIENT LOADS

Many studies have investigated the Lake Winnebago and Upper Pool Lakes watersheds and have estimated watershed sediment and phosphorus loading in the region. Watershed phosphorus loading is being addressed in the effort to develop phosphorus TMDLs for the Winnebago Pool Lakes by WDNR and others. Several prior studies describing watershed loading are described briefly below:

- *Sources and Transport of Phosphorus in the Western Lake Michigan Drainages* (Robertson 1996) presents information on an investigation of the Western Lake Michigan Drainages, which includes Lake Winnebago. Export of total phosphorus was quantified for each subunit in the Green Bay watershed.
- *Water Quality Assessment of the Western Lake Michigan Drainages—Analysis of Available Information on Nutrients and Suspended Sediment, Water Years 1971–1990* (Robertson and Saad 1996) addresses the USGS's Western Lake Michigan Drainage study unit, which includes Lake Winnebago. The report summarizes nutrient and suspended sediment data collected in the study unit, evaluates long term trends, and describes the factors that affect patterns of concentrations and loads.

- *Simulated TSS and Phosphorus Export to Lake Winnebago and Green Bay from the Fox–Wolf River Basin* (Baumgart 2002) describes a watershed model that was developed with the Soil and Water Assessment Tool (SWAT) and Geographic Information Systems (GIS) to estimate phosphorus and total suspended solids (TSS) loads to Lake Winnebago and to Green Bay. An updated SWAT model is being developed as part of the phosphorus TMDLs for the Winnebago Pool Lakes.
- *Nutrient Inputs to the Laurentian Great Lakes by Source and Watershed Estimated Using SPARROW Watershed Models* (Robertson and Saad 2011) describes nitrogen and phosphorus watershed loads estimated with the model Spatially Referenced Regressions On Watershed Attributes (SPARROW). The model covers the Great Lakes Basin and the Upper Midwest. The Fox River had the highest total phosphorus loading in the Lake Michigan drainage area.

High internal loading of phosphorus in the Winnebago Pool Lakes is ultimately due to phosphorus loads that have accumulated in the sediment from external sources. Mechanisms that lead to the release of the phosphorus from the sediments to the lake water column include re-release of phosphorus from oxidized precipitates to the sediment that become soluble under hypoxic conditions, disturbance of the sediments by wind and wave action, and other biological and physical disturbances such as foraging by benthivorous fish and motorboat activity. Phosphorus information in the sediments was collected as part of the TMDL study, and preliminary results from lake modeling efforts (including Bathtub and Jensen models) indicate that internal loading represents approximately 40 percent of the annual load to the lake and 53 percent of the growing season load to Lake Winnebago (Dale Robertson, USGS, personal communication). Information on sediment phosphorus and internal loading is not yet available.

Lake water quality data suggest that algal growth in Lake Winnebago is at times limited by nitrogen (section 3.1.2). Nitrogen limitation can favor certain species of cyanobacteria that fix their own nitrogen and therefore can use atmospheric nitrogen. However, the effect of nitrate concentrations on cyanobacteria densities in Lake Winnebago is unknown. Nitrogen data in the Winnebago Pool Lakes are limited (Figure 5), and the conclusions drawn from the data are preliminary. Nitrate concentrations in the Fox River inflow to and outflow from Lake Winnebago have increased in recent years (Figure 22). Whereas phosphorus must be reduced to lower algal and cyanobacteria growth and shift the lake back to the clear water phase, nitrogen loading can influence algal biomass and community composition on shorter, ecologically-meaningful time scales.

4.3 WIND AND WAVE ACTION

The Winnebago Pool Lakes is a shallow system with a relatively large surface area and long fetch. Wind energy has the potential to disturb and prevent re-establishment of aquatic macrophytes. Wind energy also disturbs lake sediments, leading to sediment resuspension and release of phosphorus. In the 1977–1982 study on the Winnebago Pool Lakes, wind induced wave action accounted for over 25 percent of the variation in the April–May photic zone depth (Kahl 1993). The USACE is developing wind fetch and wave model applications and management scenarios for Terrell’s Island (a portion of Lake Butte des Morts) and the northern part of Lake Poygan for the WDNR and the USACE Detroit District.

4.4 BENTHIVOROUS FISH

Benthivorous fish such as common carp forage in lake sediments. This activity can directly damage rooted macrophytes and can lead to the release of nutrients from the sediments to the water column. Based on observations by WDNR, carp are considered a stressor, primarily in the Upper Pool Lakes.

4.5 OTHER PHYSICAL DISTURBANCES

Anthropogenic disturbances such as motorboat activity, channel dredging, and nuisance macrophyte removal can also directly disturb bottom sediments and vegetation. Snowmobile activity in the winter months can be a stressor to aquatic vegetation.

5 CONCEPTUAL MODEL

Algal and aquatic macrophyte dynamics in the Winnebago Pool Lakes are driven by a combination of altered water level regime, wind and wave action, nutrient loading, benthivorous fish, and other physical disturbances (Figure 32):

- Altered water level regime: The USACE manages water levels in the Winnebago Pool Lakes to balance the needs of multiple users of the watershed's water resources. However, this management strategy has disrupted the natural water levels of the lakes, both in terms of lake depth and the timing and rate of water level change (Figure 30), affecting macrophyte establishment and survival (Figure 33):
    - Water levels are currently drawn down over the winter to provide capacity to hold spring flood waters. When water levels are low, the lakes are prone to wave action, ice scour, and freezing of sediments, which damage plant roots and rhizomes. Natural water levels would have been higher in the late winter.
    - Water levels are currently raised gradually through the early spring, and Lake Winnebago is refilled after ice out, which typically occurs at the beginning of April. During wet seasons, there is a greater potential for flooding and damage to aquatic vegetation due to deeper water and rapid increases in water level. The high water levels reduce light availability for rooted macrophytes. Natural water levels would have peaked from spring snowmelt and then gradually decreased.
    - Water levels currently peak in June and are maintained relatively constant throughout the summer to support recreation. Natural water levels would have gradually declined over the summer, with minimum water levels typically occurring in August–September.
    - Water levels are lowered gradually in the fall to a winter target level. If large changes in water level occur during this period, vegetation can be damaged. Natural water levels would have gradually increased over the fall after the late summer water level minimum.

  While the water levels across the Winnebago Pool Lakes are on average deeper than historical water levels, the timing and rate of change of water level fluctuations appears to have more of an effect on plant establishment and survival than just water level depth. This is evidenced by the lack of aquatic macrophytes in the Upper Pool Lakes, where water depths are shallow enough to support aquatic vegetation, yet macrophytes do not exist where one would expect from water depth alone (Figure 25).
- Nutrient loads. High phosphorus loading is from both external and internal sources; internal loading represents approximately 50 percent of the growing season load to Lake Winnebago (Dale Robertson, USGS, personal communication). In-depth analysis of in-lake nutrients is being completed as part of TMDL development; this analysis was not available for this project. Because of the poor quality of the aquatic macrophyte habitat in the lake, primary production tends to be in the form of algal growth in the water column (i.e., phytoplankton) as opposed to rooted plants and associated periphyton. Watershed nitrate concentrations have increased in recent years (Figure 22), although the link between nitrogen concentrations and cyanobacteria in the Winnebago Pool Lakes is not known. Whereas phosphorus must be reduced to lower algal and cyanobacteria growth and shift the lake back to the clear water phase, nitrogen loading can influence algal biomass and community composition on shorter, ecologically-meaningful time scales.
- Wind and wave action: Lake Winnebago and the Upper Pool Lakes are shallow lakes with a relatively large surface area. Wind energy has the potential to disturb and prevent re-establishment of aquatic macrophytes (Figure 34). Wind energy also disturbs lake sediments, leading to sediment resuspension and release of phosphorus.
- Benthivorous fish can directly damage rooted macrophytes and disturb bottom sediments, leading to nutrient release from the sediments.
- Other physical disturbances such as motorboats and snowmobiles can also directly damage rooted macrophytes and disturb bottom sediments, leading to nutrient release from the sediments. The extent to

which these physical disturbances drive algal and aquatic macrophyte dynamics in the Winnebago Pool Lakes is unknown, although it is assumed that these disturbances play a relatively minor role relative to the other stressors.

The above stressors maintain the lake in a stable, turbid, phytoplankton-dominated state (Figure 32). The relative balance of nitrogen and phosphorus availability affects the algal species assemblage. When phosphorus concentrations are low relative to nitrogen, which typically occurs in the spring and early summer in Lake Winnebago, primary production in the system is likely limited by phosphorus. As phosphorus increases relative to nitrogen (lowered N:P), co-limitation by both nitrogen and phosphorus is likely. However, phosphorus is typically the limiting nutrient for cyanobacteria growth and production of cyanotoxins in lakes. Ultimately, phosphorus must be reduced to control algal growth (Schindler 2012) and shift the lake back to the clear water phase. Increases in light availability (e.g., through water level management) without reductions in phosphorus availability could lead to increases in cyanobacteria biomass and cyanotoxin production.

The low abundance of aquatic macrophytes decreases sediment stability, making the sediments more susceptible to disturbance by wind and wave action, benthivorous fish such as carp, and other physical disturbances. Nutrients are released from the sediments to the water column, further fueling algal growth. The high concentrations of algae in the water column increase the turbidity of the water and decrease light availability. Lower levels of light limit growth of rooted macrophytes; high turbidity in the lake is a result of sediment inputs from the watershed, wind and wave resuspension of bottom sediments, and high concentrations of phytoplankton. The timing and rate of change of water level fluctuations also limit plant establishment and survival. The multiple stressors reinforce the turbid state, making it difficult to shift the system to the clear-water, macrophyte-dominated state.

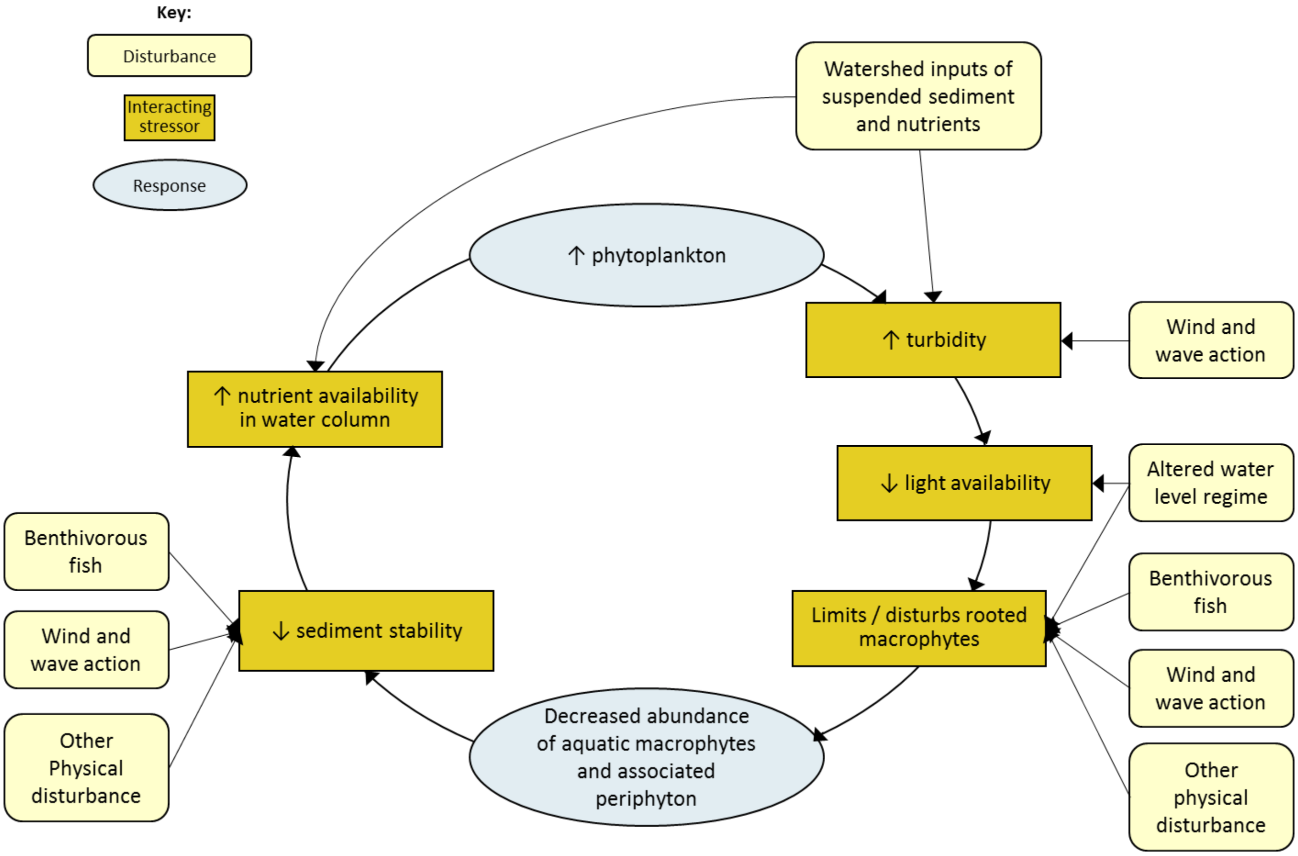

Figure 32. Conceptual model of algae and aquatic macrophytes in the Winnebago Pool Lakes.

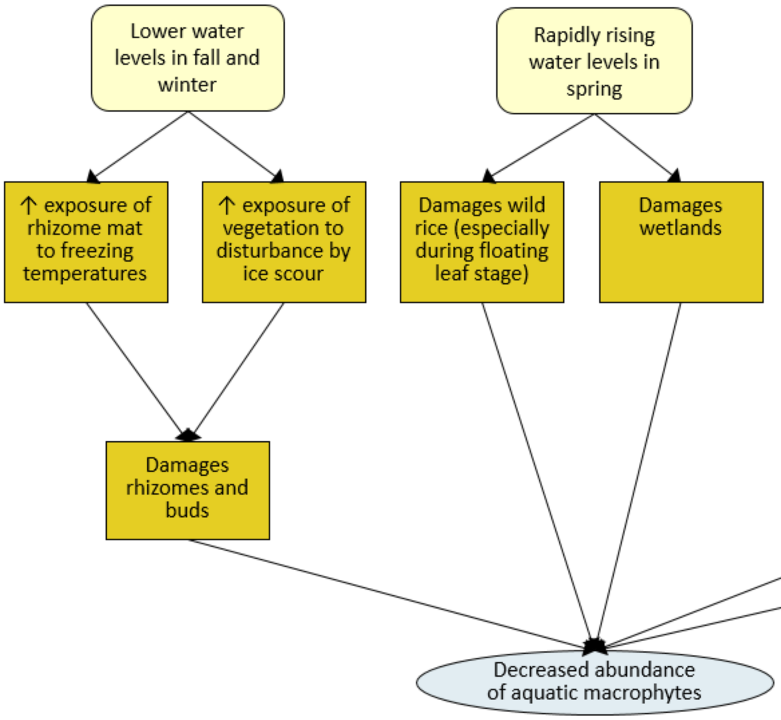

Figure 33. Altered water level regime component of conceptual model of algae and aquatic macrophytes in the Winnebago Pool Lakes.

See key in Figure 32

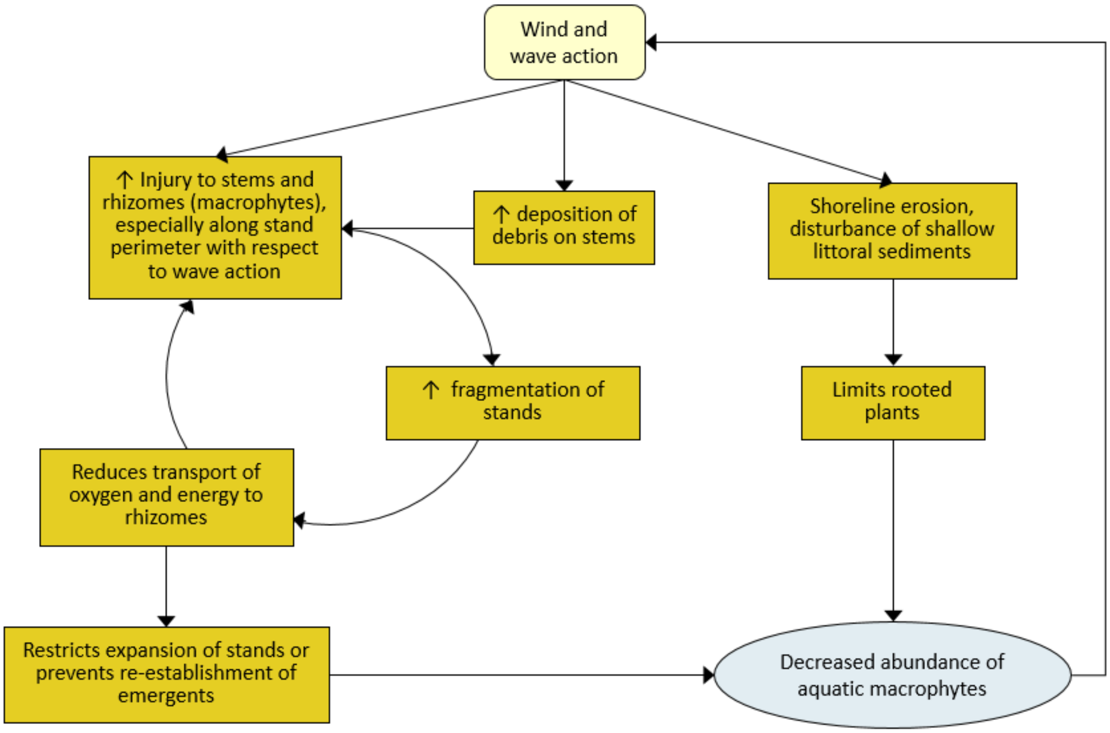

Figure 34. Wind and wave action component of conceptual model of algae and aquatic macrophytes in the Winnebago Pool Lakes.

See key in Figure 32.

6 COMPUTER MODEL SELECTION

Computer models can be used to simulate natural processes and the effect of implementation activities on in-lake dynamics. A discovery process was conducted to identify models and tools capable of addressing the following:

1) Simulate historic and baseline ecological conditions in Lake Winnebago and the Upper Pool Lakes (e.g., macrophytes, algae, etc.).
2) Simulate the predictive relationships between in-lake management activities such as manipulation of nutrients and hydromorphology (e.g., break walls, islands) with aquatic ecology (primarily macrophytes) and water quality to develop a range of management options.
3) Predict the extent of implementation needed to shift the lakes from a turbid to clear lake state (alternative stable states). Simulate the transition between stable states.

Based on the conceptual model, the following should be represented as external disturbances in the chosen model(s):

- Nutrient loading—the model(s) should have the ability to alter external loads over time.
- Water level changes over time—the model(s) should have the ability to simulate daily water level fluctuations. Based on review of the available water level data, a monthly or annual timestep will be inadequate.
- Wind and wave action—the model(s) should have the ability to simulate the effect of wind and wave action on in-lake sediment and macrophyte disturbance. This could include the ability to link to external wind-wave modeling applications.
- Benthivorous fish—the model(s) should account for the ecological interactions of benthivorous fish and macrophytes.
- Physical disturbances—the model(s) should have the ability to account for physical disturbances on sediment and macrophyte such as motorboat activity.

The model(s) will need to simulate the ecological response (e.g., algae growth, macrophyte establishment) to the above external disturbances. The chosen model should include multiple planktonic and benthic algal components, preferably with explicit representation of different macrophyte types (e.g., floating-leaf), to simulate alternating stable states (macrophyte dominated and algae dominated).

The model(s) should be continuous simulation with a minimum daily timestep. In addition to providing the needed resolution to simulate changing water levels, a daily timestep will also provide the ability to determine if water quality standards and thresholds are being met, specifically the chlorophyll-a threshold that requires less than 30 percent of days in the sampling season with nuisance algal blooms as indicated by chlorophyll-*a* concentrations greater than 20 µg/L (WDNR 2017).

6.1 EXISTING MODELING EFFORTS

Existing in-lake modeling efforts include Bathtub, which is being used to estimate the load reductions in Lake Winnebago that are needed to meet TMDL requirements (Dale Robertson, USGS) and a wind fetch and wave model that is being developed by the Army Corps of Engineers St. Paul District for Terrell’s Island and the northern part of Lake Poygan. The wind-wave model being used was developed by the USGS Upper Midwest Environmental Sciences Center and is based on an ArcGIS platform. Five years of wind data were used, and the wind model integrates the highest sustained winds and direction per day. The wave model predicts the probability of sediment resuspension in the lakes based on wind fetch, bathymetry, and the minimum orbital wave velocity at a 10-meter grid cell scale. Existing condition models have been completed and scenarios are underway as of 11/2017 (D. Ingvalson, USACE).

6.2 AVAILABLE MODELS

Selection of an appropriate model or tool depends on several factors:

1) Ability of the model to address the study questions (complexity, biological/ecological processes)
2) Availability of data to support model development and calibration/validation and the ability to collect these data
3) Ability of the model to integrate with other modeling efforts (e.g., wind-wave modeling)
4) Flexibility of the model to allow for source code changes if needed

In addition, the availability of resources (technical and financial) to support the modeling effort should be considered. Several receiving water models were investigated for use in the Winnebago Pool Lakes including PCLake, CAEDYM/AEM3D, CE-QUAL-W2, and WASP, which are summarized below. These models focus primarily on in-lake nutrient and ecological interactions.

6.2.1 PCLake

PCLake is an open source and public domain ecosystem model maintained jointly by the Netherlands Environmental Assessment Agency and the Netherlands Institute of Ecology. PCLake was originally developed by Janse and van Liere (1995) as a zero-dimensional ecological model for non-stratifying, shallow lakes. It has been widely applied to shallow lakes, mainly in the Netherlands, Denmark, and northern Europe. PCLake has been calibrated with data from more than 40 temperate shallow lakes located in the Netherlands, Belgium, and Ireland and thus has a fairly wide geographic applicability (Janse et al. 2010, Kuiper et al. 2017). There have been successful case studies using PCLake in Mediterranean Greece (Mellios et al. 2015) and Subtropical China (Kong et al. 2016) as well.

The PCLake model operates with a fully mixed water column and subsurface sediment layer. PCLake is highly parameterized and complex and therefore requires a very rich input dataset. The model describes the dynamics of phytoplankton and macrophytes; includes a simplified food web including zooplankton, zoobenthos, zooplanktivorous fish, benthivorous fish, and piscivorous fish; and accounts for mass balances of nitrogen, phosphorus, and silica (represented by dry weight) (Hu et al. 2016). The original model also includes a marsh module describing (helophytic) marsh vegetation in a zone around the lake. The PCLake model was developed to simulate eutrophication of shallow lakes and to estimate critical loadings for transitions between the clear, vegetation dominated state and the turbid, phytoplankton dominated state in shallow lakes (Janse 1997, Janse et al. 2008, Nielsen et al. 2014, and Hu et al. 2016).

Limitations of the PCLake model include, but are not limited to, the following:

- Inability to account for possible near-surface accumulations of phytoplankton, especially surface blooms of cyanobacteria that could develop during calm and warm periods, due to the zero-dimensional model setup (Nielsen et al. 2014).
- No vertical gradients within the model for various water quality parameters (i.e., chl-*a*, TP, TN, and DO). Some shallow lakes can experience gradients of water quality parameters on a short-term basis, in particular DO gradients with occasional low levels that would facilitate release of phosphorus from the sediments, increasing internal loading.
- Assumption of a uniform water depth across a waterbody, leading to an “all or nothing” response in lakes (Mooij et al. 2010).
- Ice-cover formation is not included in PCLake (Nielsen et al. 2014).
- Grass carp and similar species, which have a strong trophic interaction with aquatic plants, are not included in the PCLake model (Kuiper et al. 2017).
- The model macrophyte species in PCLake are waterweeds in general (*Elodea spp.*) (Kuiper et al. 2017).
- Inorganic carbon (CO_2) is not explicitly modeled and, therefore, the elevation of pH due to algal growth cannot be evaluated using PCLake (Utah DEQ 2016).

There are studies currently implementing PCLake into FABM (Framework for Aquatic Biogeochemical Models), which allows PCLake to be run in zero-dimensional, one-dimensional, and three-dimensional physical environments (Trolle et al. 2012, Hu et al. 2016). This allows the detailed ecological processes provided by PCLake to be used to study deeper and spatially complex aquatic ecosystems. For example, macrophytes in PCLake are represented by a single value in g/m^2, but are able to colonize different depths when linked to a one-dimensional hypsographic hydrodynamic model (Hu et al. 2016). It also becomes possible to study the concept of critical nutrient loading for spatially heterogeneous aquatic systems, such as very large shallow lakes.

A customized PCLake model, whereby the model is adapted, calibrated, and validated for a specific lake/case, will provide the most accurate predictions for use in the management of that system. Examples include Nielsen et al. (2014), Trolle et al. (2014), and Kong et al. (2016).

6.2.2 CAEDYM (AEM3D)

CAEDYM was a proprietary aquatic ecology model previously maintained by the Centre for Water Research at the University of Western Australia. The Centre was discontinued by the government, and CAEDYM is now contained within the AEM3D package, maintained by HydroNumerics (http://www.hydronumerics.com.au). Most of the available literature refers to the older version; that name is maintained in this discussion. CAEDYM (computational aquatic ecosystem dynamics model) is a process-based library of water quality, biological, and geochemical sub-models that may be run independently or driven by either the Dynamic REServoir simulation Model (DYRESM, one-dimensional hydrodynamics model) or the Estuary and Lake COMputer model (ELCOM, three-dimensional hydrodynamics model) (Mooij et al. 2010). AEM3D is an updated and recoded version of the CAEDYM-ELCOM modeling pair.

The CAEDYM portion of the model simulates a range of biological and chemical variables, such as inorganic and organic forms of phosphorus and nitrogen, up to seven phytoplankton groups, and up to five zooplankton groups (Trolle et al. 2008, Trolle et al. 2014). If the lake/waterbody being modeled is characterized by complex bathymetry and a higher spatial resolution is required due to complex horizontal circulation and transport processes, CAEDYM should be run with the three-dimensional ELCOM hydrodynamics model.

The more recent versions of CAEDYM (v3.3, Hipsey and Hamilton 2008) and AEM3D (Hodges and Dallimore 2016) can model stratification in lakes, inflow/outflow dynamics of waterbodies, suspended solids, oxygen and organic/inorganic nutrients (C, N, P, and Si), multiple phytoplankton functional groups, zooplankton, fish, benthic communities (including macrophytes), pathogens, geochemistry and sediment oxygen, and nutrient and metal fluxes. In either one-dimensional or three-dimensional applications, CAEDYM has been used widely throughout the world (over 70 peer reviewed studies [Trolle et al. 2012]) for studying nutrient cycling, the effects of increased nutrient loads on algal blooms, and changes to phytoplankton succession, as well as for identifying conditions that favor cyanobacteria (Mooij et al. 2010). United States applications of the model include Lake Mead in Arizona and Nevada, Lake Elsinore and Canyon Lake in California, and Coeur d'Alene Lake in Idaho (Hipsey et al. 2007). It is unclear at this time if the model is being maintained or distributed. The current AEM3D model code is proprietary; the executable model is available for a small fee ($100 Australian).

6.2.3 CE-QUAL-W2

CE-QUAL-W2 is an open source and freeware program supported by the US Army Corps of Engineers and maintained by the Portland State University Water Quality Research Group. CE-QUAL-W2 is a two-dimensional laterally averaged hydrodynamic and water quality model that simulates vertical stratification and longitudinal variability. The model can be run for rivers, estuaries, lakes, reservoirs, and river basin systems. The model simulates basic eutrophication processes such as temperature, nutrient, algae, DO, organic matter, and sediment relationships (Cole and Wells 2015). The model also has the ability to simulate optional biotic groups including multiple periphyton, phytoplankton, zooplankton, benthic algae, and macrophytes groups, with interacting hydrodynamics (Mooij et al. 2010). CE-QUAL-W2 does not simulate macroinvertebrates, fish, or bird response.

CE-QUAL-W2 Version 3.7.2 predicts sediment oxygen demand and nutrient fluxes from the underlying benthos through either zero-order or first-order decay. A fully predictive sediment diagenesis routine can be implemented in the newest version (4.0) of the model (Utah DEQ 2016).

CE-QUAL-W2 has been widely used in the United States as a management and research tool, particularly for studying nutrient and sediment dynamics of reservoirs and river impoundments (Mooij et al. 2010). It can be run in a quasi-3-D mode by using additional model branches (Portland State University 2017). The model's main limitation is that it assumes the waterbody being modeled is well-mixed in the lateral direction, hence its popularity and success with long, narrow reservoirs and river impoundments. Simulating sediment oxygen demand and nutrient release dynamics have been identified as model weaknesses. We were not able to identify examples of application of CE-QUAL-W2 to stable state analysis of shallow lakes.

6.2.4 WASP

The Water Quality Analysis Simulation Program (WASP) is a public domain and freeware water quality model maintained by the EPA. Current versions of the model code are not readily available but could likely be obtained by request via EPA. WASP simulates nutrients and water quality dynamically in both the water column and the underlying sediments. The time varying processes of advection, dispersion, point and diffuse mass loading, and boundary exchange are represented in the model (USEPA 2017). WASP can also be linked with hydrodynamic and sediment transport models that provide flows, depths, velocities, temperature, salinity, and sediment fluxes. WASP can simulate five different phytoplankton groups, three benthic algae groups, and one zooplankton group. The most recent version (WASP 8) includes a full representation of various types of macroalgae, including rooted macrophytes. The current model also contains a sediment diagenesis model linked to the Advanced Eutrophication sub model, that predicts sediment oxygen demand and nutrient fluxes from the underlying sediments (USEPA 2017).

WASP itself does not contain a hydraulic model, although it can perform level pool or kinematic wave routing. Where consideration of complex hydrodynamics is required, WASP is linked to a hydrodynamic model, such as Environmental Fluid Dynamics Code (EFDC).

WASP can simulate the settling, accumulation, burial, and resuspension of three different inorganic sediment size classes. Resuspension is predicted on specification of scour velocity, a formulation that is less suitable for a system where erosion is wind and wave induced. Coupling WASP with a 3-D hydrodynamic model such as EFDC would adequately simulate the resuspension of sediment resulting from wind forces and wave action (Utah DEQ 2016). WASP does not simulate benthic macroinvertebrates, fish, or bird response.

WASP is one of the most widely used water quality models in the United States and throughout the world and has frequently been applied to the development of TMDLs (USEPA 2017). WASP has capabilities of linking with both hydrodynamic and watershed models, which allows for multi-year analysis under varying conditions. WASP models have been applied to all of the major estuaries in Florida to support the development of numeric nutrient criteria. A WASP Version 5 model was developed for shallow Lake Okeechobee, and the source code was modified to compute sediment resuspension based on wind-wave action and bottom shear stress (James et al. 1997, James et al. 2005). Additional enhancements have since been made to the Lake Okeechobee Water Quality Model (James 2013). While WASP simulates both planktonic algae and macrophytes, it does not explicitly simulate transition between alternative stable states.

6.3 MODEL SELECTION

Table 2 provides a comparison of several model components for each of the potential models, and Table 3 summarizes the model inputs, applicability, pros, and cons associated with each of the models. The level of effort is estimated to be similar among all of the proposed models; each model requires detailed inputs that will require comprehensive data collection efforts.

Table 2. Overview of lake ecosystem model components (adapted from Mooij et al. 2010, Utah DEQ 2016)

Model Component	Model			
	PCLake	**CAEDYM (AEM3D)**	**CE-QUAL-W2**	**WASP**
Clear-Turbid Stable State Transition	Y	partial	partial	partial
Nutrient Loading	Y	Y	Y	Y
Water Level Changes over Time	Y	Y	Y	Y
Wind and Wave Action	Y	Y	Y	N^
Benthivorous Fish	partial	partial	N	N
Physical Disturbances	partial	N	N	N
Spatial Dimension	0-D 1 to 3-D*	3-D	2-DV	1-D 3-D, if linked
Stratification	Y, if linked	Y	Y	N
Sediment	Y	Y	Y	Y
Littoral Zone	Y	Y	N	N
# Phytoplankton Groups	1-3	7	3+	5
# Zooplankton Groups	1	5	3+	1
# Benthic Groups	1	6	3+	3
# Fish Groups	3	3	0	0
# Macrophyte Groups	1-6	1	3+	1
# Bird Groups	0-1	0	0	0
Hydrodynamics	Y, if linked	Y	Y	Y, if linked
Temperature Dynamics	Y	Y	Y	Y
Oxygen Dynamics	Y	Y	Y	Y
Internal Phosphorus Dynamics	Y	Y	Y	Y
Internal Nitrogen Dynamics	Y	Y	Y	Y
Internal Silica Dynamics	partial	Y	Y	Y
Sedimentation/Resuspension	Y	Y	Y	partial
Diagenesis	partial	Y	partial	Y
Dredging	Y	N	N	N
Mowing	Y	N	N	N
Ice Regime	N	N	Y	Y

*PCLake can be 1 to 3-D when used with compartments or linked with various hydrodynamic models, i.e., DUFLOW (1-DH).
^ The public release of WASP does not address wind and wave resuspension, but this is incorporated in the Lake Okeechobee version of the model.

Table 3. Model Summary

Model Name	Primary Data Input Requirements	Applicability to Large Shallow Lakes	Pros	Cons
PCLake	Water inflow and outflow, infiltration or seepage rate (if any), nutrient (N, P) loading, particulate loading, temperature and light, dimensions (lake depth and size), size of the marsh zone (if any), sediment features, and loading history (initial conditions). Initial conditions include plankton, submerged vegetation, and fish inputs. Additional detail is provided in section 8.2. Bathymetry (if linking/coupling with higher dimension physical model)	Yes; applicability increases with lake specific adapted model and/or coupling with 1-D or 3-D hydrodynamic model to better represent spatially heterogeneity in a large system Note—Lake Winnebago is substantially larger than the lakes used to develop the PCLake model, see Appendix A.	• Only model described in the existing literature that is able to account for interactions between macrophyte recovery and decay and fish, zooplankton, and phytoplankton in shallow lakes (Nielsen et al. 2014) • Can be used to estimate critical nutrient loadings • Can be linked/embedded in models focusing on spatial dynamics	• Requires linking to create a version to account for variability in lake morphometry • Highly complex model with a large number of model parameters
CAEDYM (AEM3D)	Lake bathymetry, inflows and outflows (including flow, nutrients, temperature, and oxygen), lake water quality data, air temperature, relative humidity, atmospheric pressure, cloud cover, wind speed/wind direction, solar radiation, precipitation	Somewhat; applicable to large waterbodies with complex morphology. Most US applications have been on deeper large lakes with stratification.	• Ecosystem representation is configurable and can be varied by the user depending on purpose of the model and availability of data • Flexibility to be run in either 1-D or 3-D • Detailed aquatic ecology, strong ecological modeling capability	• Complexity of the model requires the use of a large number of model parameters • Code not public domain

Model Name	Primary Data Input Requirements	Applicability to Large Shallow Lakes	Pros	Cons
CE-QUAL-W2	Lake bathymetry and topographic information, lake volume, surface area, water surface elevation data, inflows and outflows (including flow, nutrients, temperature, and oxygen), lake water quality data, air temperature, dew point temperature, cloud cover, wind speed/direction, solar radiation, precipitation	Somewhat; developed for modeling long, narrow rivers and reservoirs; can be used in a quasi-3-D model with multiple branches; maybe better suited if waterbody is laterally well mixed	• Widely used in the US • Model program is very accessible • Can simulate several different groups of phytoplankton, zooplankton, and macrophytes	• Complexity of the model requires the use of a large number of model parameters • Requires high quality input data • Better suited for laterally well mixed waterbodies such as reservoirs and rivers • No demonstrated ability to predict transition between stable states
WASP	Lake volume, depth and surface area, inflows and outflows (including flow, nutrients, temperature, and oxygen), lake water quality data, air temperature, dew point temperature, cloud cover, wind speed/direction, solar radiation, precipitation	Yes; has been used successfully in large shallow lakes (i.e., Lake Okeechobee) although with some modification or linking with hydrodynamic models	• Widely used in the US • Model program is very accessible • Current version incorporates ice cover • Can simulate several different groups of phytoplankton and macrophytes	• Would most likely need to be adapted to unique characteristics of the lake • Model code is not readily available • Ability to predict transition between stable states not demonstrated

6.4 MODEL RECOMMENDATION

A single model does not have to contain all of the required elements; multiple models can often be linked to provide the needed inputs and responses. PCLake is the only model that has some proven track record in addressing stable state transitions in shallow lakes, although application in the U.S. has been limited. Application of PCLake is recommended following a comprehensive data collection effort addressing the model inputs identified in section 8.2. The primary inputs to PCLake include lake hydrology, nutrient loading, temperature and light, dimensions (depth and size), size of marsh zone (if applicable), sediment properties, and loading history (initial conditions). However, PCLake is a highly parameterized model with over 200 parameters. These parameters, listed in Janse (2005), would need to be reviewed to ensure that default information is applicable to the modeled lake(s).

Based on the available literature, there remains uncertainty of PCLake's ability to explicitly represent physical disturbances or benthivorous fish and their effect on sediment suspension or macrophytes. It would be possible to alter the model code to better account for these activities, thus developing a customized version of PCLake. The effect of wind and wave action in the Upper Pool Lakes, as modeled in PCLake, should be scrutinized to ensure adequate representation. If needed, PCLake can be linked to a 3-D hydrodynamic model that can integrate additional detailed wind-wave effects.

The anticipated level of effort to develop a customized version of PCLake for Lake Winnebago and the three Upper Pool Lakes along with scenario development is 1,200–1,500 hours, not including additional wind-wave modeling, linking to a 3-D model, or monitoring and data collection. Suggested expertise to use PCLake includes the following:

- Detailed knowledge of ecosystem processes in shallow lakes including ecological, chemical, and physical interactions
- Extensive experience using numeric lake response models, including familiarity with differential equations
- Ability to read and modify code contained within the model

An alternate option would be to apply the WASP model, likely using the Lake Okeechobee version that incorporates wind/wave erosion. The application would be linked to a hydrodynamic model, such as EFDC, to meet the requirements of evaluating daily water level fluctuations and addressing wind and wave action. WASP would not provide direct predictions of stable state transitions but could be used to evaluate ecosystem conditions under both clear and turbid states that could form the basis for an external analysis of the likelihood of shift back to a clear state.

7 LAKE RESTORATION ACTIVITIES

The goal of the management options explored in this section is to restore aquatic macrophytes in the Winnebago Pool Lakes to shift the lakes from a turbid state to a clear water state. Whereas this report focuses on in-lake restoration alternatives, reduction of external nutrient loads is often a prerequisite for a successful shift back to a clear state. External nutrient loading is often a “forward switch” in shallow lakes, shifting the lake from a clear water state to a turbid state, or maintaining the lake in a turbid state (see Figure 3). The literature suggests that to reverse the switch to a clear state, a much lower phosphorus concentration is needed, and therefore external nutrient reductions should be considered the highest priority. However, activities that address internal loading of nutrients can be conducted concurrently to address internal phosphorus reserved in the sediments.

Both short- and longer-term management options are provided below. The management options described below address the “disturbance” components of the conceptual models in Figure 32 through Figure 34. Many of the proposed restoration alternatives will need to be implemented in combination with other alternatives. For example, reductions in water level due to Altered Water Level Regime activities could increase light availability, which could in turn increase phytoplankton (including cyanobacteria) if nutrient levels remain high in the water column.

The effect that the management options would have on water quality can be simulated with computer models, which are explored in section 6: Computer Model Selection. Case studies are also included that illustrate lake restoration efforts in large, shallow lakes. In addition to the management options provided below, programmatic and regulatory actions could also be explored.

7.1 RESTORATION ALTERNATIVES

7.1.1 Altered Water Level Regime

Work with the USACE to alter the water level regime in the Winnebago Pool Lakes to more closely resemble the suggested water level regime in the Linde Plan (Figure 30). Consider the water level management recommendations in *Aquatic Macrophyte Ecology in the Upper Winnebago Pool Lakes, Wisconsin* (Kahl 1993), which include establishing a spring–summer target water level that is under 2.5 feet and allowing seasonal and annual fluctuations around this water level to mimic a natural hydrologic cycle. Even seemingly small reductions in water levels from spring to fall could benefit aquatic plants.

Adherence to the spring refill targets in the Linde Plan are especially important to prevent damage to submerged aquatic vegetation and to allow growing macrophytes to keep pace with the increasing water levels. Because one year of high water can damage aquatic plant populations and undermine previous gains, high waters should be avoided to the extent possible.

7.1.2 Nutrient Loads

Reduction in nutrient loads is needed to shift a shallow lake from a turbid to a clear state. The draft lake TMDLs are identifying and addressing phosphorus reductions needed to lower in-lake phosphorus concentrations to below state water quality standards. As part of TMDL development, it is anticipated that watershed and point source load phosphorus reductions will be identified. At this time, watershed nitrogen sources are not being addressed as part of TMDL development. It is possible that as additional information is collected to further describe the conceptual model of the lakes, watershed nitrogen load reduction may be identified as an important implementation activity.

While external nutrient sources (watershed and point source) are being reduced, reductions in internal phosphorus loading can be concurrently occurring. A cost-benefit analysis could be used to determine the cost optimal combination of external and internal load reductions. The following are options to reduce internal phosphorus loading in Lake Winnebago and the Upper Pool Lakes.

- Chemical treatment such as in-lake phosphorus inactivation and tributary interception with alum or iron treatment. As a phosphorus inactivation agent, aluminum sulfate (alum) chemically binds with phosphorus and precipitates out of the water column as part of the alum floc, which is an aluminum hydroxide complex. Dosing is regulated or buffered as to not have an impact on water pH. The alum treatment would likely result in a short-term decrease in turbidity. Another potential chemical treatment is the addition of iron. Although iron-bound phosphorus is often released back into the water column under anoxic conditions, recent evidence from mesocosm experiments suggests that gradual, long-term reductions in phosphorus concentrations can be achieved even under anoxic conditions (Orihel et al. 2016). This technology requires a large iron to phosphorus ratio.

Chemical treatment could occur within the lake or it could be used to treat tributary inflows. Tributary treatment typically occurs through the use of an off-line detention treatment system with alum injection.

- Installation of hydroponic islands to increase aquatic habitat, reduce wind mixing of sediment, and protect shoreline erosion. Consists of floating platforms that support aquatic vegetation, both submergent and emergent. This activity would serve as a physical wind barrier, nutrient sink, and fisheries habitat enhancement.
- Dredging removes sediment and contaminants from the lake. Scale and associated cost is an issue to consider, the Upper Pool Lakes are likely better candidates for this management activity. Phosphorus inactivation of newly exposed soils would be needed following a dredging project to prevent the release of phosphorus.

7.1.3 Wind and Wave Action

Breakwalls and constructed islands can be used to protect sensitive areas in lakes from damage caused by wind and wave action. Management recommendations in *Aquatic Macrophyte Ecology in the Upper Winnebago Pool Lakes, Wisconsin* (Kahl 1993) include implementing large-scale breakwater projects to reduce turbidity. This type of activity will reduce the lake fetch leading to less resuspension of sediment in the water column. On a smaller scale, hydroponic islands described in section 7.1.2 could also be used as a physical wind barrier.

Where there is wind abatement (e.g., in and around natural islands, reefs, points, harbors, and bays) in Lake Winnebago and the Upper Pool lakes, there is higher macrophyte coverage. Implementation of breakwalls and islands will need to be balanced with the recreational needs of the lakes and private ownership.

Terrell’s Island in Lake Butte des Morts is the result of a breakwall that was constructed by the WDNR to protect habitat and associated fish and wildlife. A two-mile long rock breakwall encircles a portion of the lake, referred to as Terrell’s Island, to allow reestablishment of native plants. After construction of the breakwall, there was initially a positive macrophyte response that resulted in a shallow clear lake state. However, after about 10 years the lake flipped back to a turbid state.

A series of break walls are planned in Lake Poygan near the Wolf River inlet. Approximately 1,200 feet were constructed in 2016, and two additional phases are planned over the next three to five years. It is likely that at least 2,500 to 3,500 feet will be added. Macrophyte response is not yet known because phase 1 alone does not provide sufficient energy dissipation.

7.1.4 Benthivorous Fish

The role of benthivorous fish such as carp requires additional study to understand the potential effect on macrophyte establishment and water clarity. Results from a WDNR carp exclusion study in the Terrell’s Island area suggest that carp are not a major stressor leading to the turbid state, but rather a lack of water circulation is. Observations from WDNR staff indicate that carp might be a stressor, particularly in the Upper Pool Lakes.

Carp control can target weaknesses in carp’s life cycle. Because adult carp aggregate under the ice in winter, commercial winter harvesting can target areas of high carp densities. Additionally, carp migrate from lakes to wetlands to spawn; these routes can be blocked and/or targeted for removal. Carp eggs and larvae can be controlled by abundant panfish populations.

An investigation should be conducted over multiple years to quantify the density of carp and other benthivorous fish and to track their movement within the Winnebago Pool Lakes. If it is determined that benthivorous fish such as carp substantially contribute to the lack of rooted macrophytes, regular commercial harvesting should be considered. Lake level drawdowns could also be considered in select locations as well as carp exclosures.

In addition to a system-wide carp investigation, when planning restoration practices in the Winnebago Pool Lakes, the effects of carp should be studied in order to mitigate any negative effect that carp might have on the restoration activity.

7.1.5 Other Physical Disturbances

Other physical disturbances such as motorboat activity, snowmobile activity, and private removal of aquatic macrophytes should be addressed through education. In addition to education, an inventory and review of existing regulations and guidance related to these activities may identify areas where strengthened regulation could be used to further address these disturbances.

7.2 RESTORATION CASE STUDIES

7.2.1 Grand Lake St. Marys

Grand Lake St. Marys (GLSM) is a large, shallow, hypereutrophic lake in Ohio spanning 20 square miles (5,200 ha) with a fetch of ten miles. GLSM is a man-made reservoir that is also Ohio’s largest inland lake. GLSM is an important recreation lake; however, harmful algal blooms and excess algae have been common over the past decade.

Many plans have been developed over the years to provide guidance on water quality improvements and restoration. As part of the plans, computer modeling has been conducted to understand the existing conditions and evaluate proposed scenarios. A Bathtub model was developed as part of TMDL development in 2007 (Ohio EPA 2007), and mass balance modeling of phosphorus on a two-week timestep was also developed (Tetra Tech 2013). More recently, a Conceptual Ecosystem Revitalization Model (CERM) was developed (Figure 35). The CERM provides performance measures to monitor over time to determine effectiveness of the various activities.

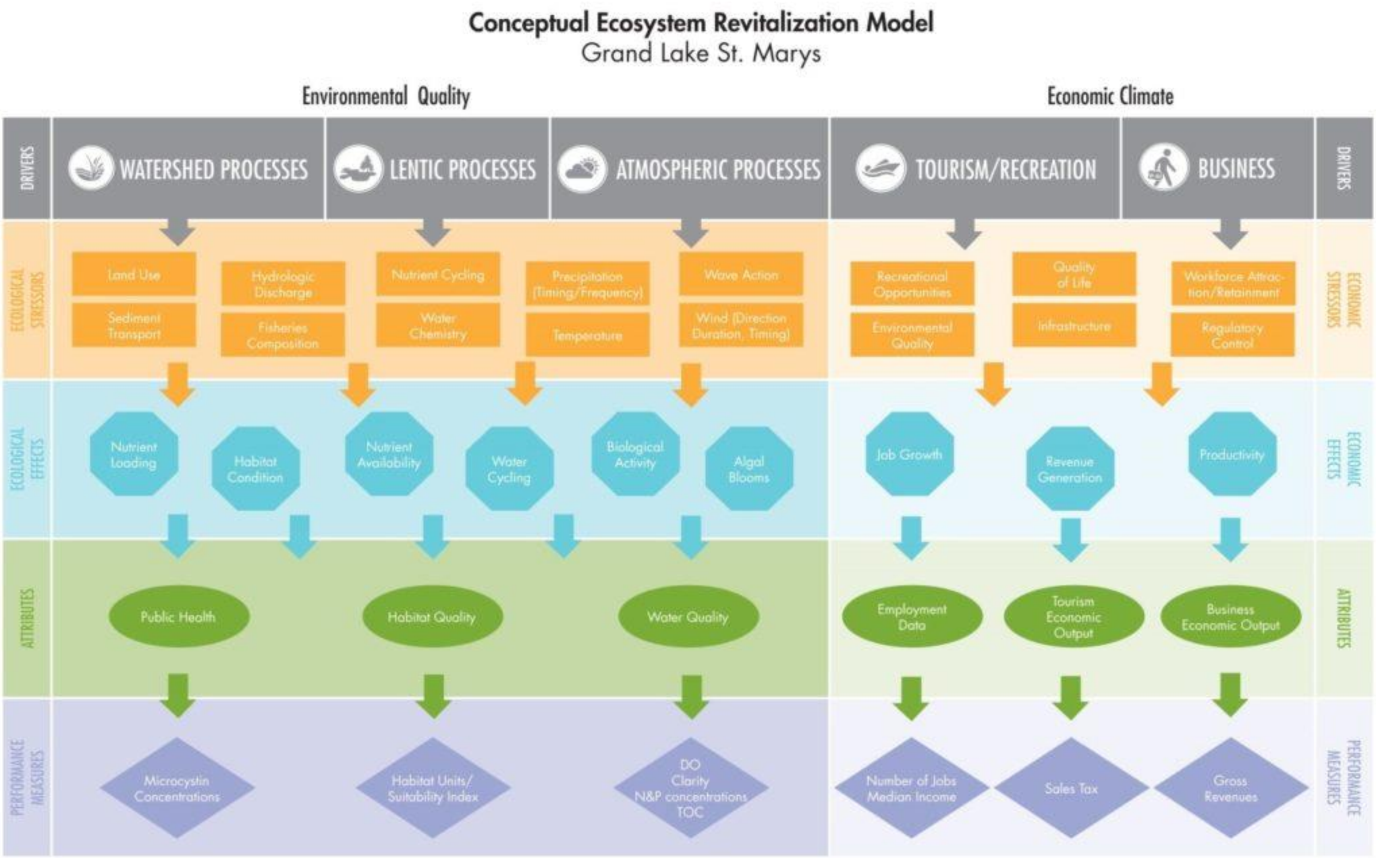

Figure 35. Conceptual Ecosystem Revitalization Model (image from KCI 2017).

A strategic plan was developed in 2011 following algae blooms during the summer of 2010 which resulted in closure of the lake to all recreational activities. The strategic plan integrated efforts by various agencies and organizations and provided a framework and timeline for restoration. The primary objectives of the strategic plan actions were to improve water quality and increase habitat. Implementation through 2016 included various activities:

- In-lake alum treatment was applied to the central portion of GLSM (40 percent of the total lake area) during the spring of 2011 and spring of 2012. These partial lake treatments resulted in a 55 percent reduction in internal loading for the whole lake (Welch et al. 2017).
- BMP implementation in the watershed including agricultural BMPs and septic system improvements.
- Three treatment train systems on tributary systems, as described in KCI (2017):

  > Prairie Creek is an engineered system that includes a Mobile Alum Injection Device as well as extensive constructed and restored wetlands. Coldwater Creek is also an engineered system, similar to Prairie Creek TT, but is larger. Both of these projects are operational and have successful preliminary results. Monitored removal efficiencies on Prairie Creek TT were 31% and 71% for nitrogen (NO2-N, NO3-N, NH3-N) and phosphorus (P04) respectively. Scheduled to be completed in the fall of 2017, Beaver Creek is a Biofilter Complex treating water in three vegetated cells. Big Chickasaw Creek TT is in design and scheduled for implementation in 2018. Beaver Creek/Prairie Creek TTs was funded by the 319 Program. Coldwater Creek/Big-Little Chickasaw Creek TTs are funded through state appropriations.

- Rough fish removal including commercial harvesting.
- Dredging and removal of over 1,000,000 cubic yards of sediment from the lake.
- Linear aeration/circulation.

In addition, implementation activities have been programmatic in nature:

- Establishment and filling of a dedicated lake manager position
- Establishment and filling of a full-time Agricultural Solutions position at the county level to focus on researching agricultural BMPs
- Establishment of a Lake Facility Authority to provide a funding source for implementation
- Development and implementation of a communications plan and fund raising program
- Ongoing water quality monitoring
- Various regulatory and state-wide initiatives

An adaptive management plan was developed in 2017 to further guide restoration activities. The following water quality improvement activities are included:

- Treatment trains and littoral wetland systems
- Rough fish removal
- Re-establishment of public connection with the lake as a recreational destination for swimming and water contact sports
- Watershed BMPs
- In-lake features development (e.g., islands and near shore bars)
- Management of channel water
- Natural/man-made infrastructure management
- Monitoring, documentation and modeling of scientific data
- Beneficial use of organic waste

In addition to these activities, programmatic and regulatory activities are also included.

7.2.2 Lake Okeechobee

Lake Okeechobee is a large, shallow lake in southern Florida with a surface area of 730 square miles and an average depth of nine feet. The lake is part of the Greater Everglades watershed, which begins with the Kissimmee River in the Lake Okeechobee watershed and extends through the Everglades and into Florida Bay. The lake provides a multitude of uses, including drinking water supply, agricultural irrigation water and frost protection, aquifer recharge, fish and wildlife habitat, flood control, navigation, and recreational activities including boating and fishing. The South Florida Water Management District, Florida Department of Environmental Protection, and the Florida Department of Agriculture and Consumer Services work with the United States Army Corps of Engineers, other federal agencies, the Florida Fish and Wildlife Conservation Commission, local governments, and other stakeholders to address the challenges that the lake faces and to protect and restore the lake's many uses.

Watershed practices are the focus of the Lake Okeechobee Basin

Figure 36. Lake Okeechobee watershed (Florida Department of Environmental Protection 2014).

Water quality in the lake declined in recent decades due to excessive nutrients from agricultural and urban activities, extreme water levels (both low and high), and aquatic invasive vegetation. In 1998, the lake was listed as impaired due to nutrients (particularly phosphorus), dissolved oxygen, un-ionized ammonia, chlorides, coliforms, and iron. A phosphorus TMDL was developed in 2001 (Florida Department of Environmental Protection 2001). The TMDL report presents an average annual phosphorus load estimate and goal. Approximately half of the annual phosphorus loads to the lake are from internal sources.

The *Basin Management Action Plan (BMAP) for the Implementation of Total Maximum Daily Loads for Total Phosphorus by the Florida Department of Environmental Protection in Lake Okeechobee* "represents the joint efforts of multiple stakeholders to identify and implement projects that ultimately achieve the TP TMDL for Lake Okeechobee" (Florida Department of Environmental Protection 2014). The BMAP addresses watershed loads to the lake (Figure 36); it does not include management practices that would directly reduce internal phosphorus loading. Watershed management strategies include practices that address urban stormwater, dispersed water management, hydrologic restoration, and agricultural BMPs (e.g., managing nutrient application, irrigation, livestock, and operations).

The BMAP uses a watershed assessment model (WAM), which is a load estimation tool that calculates daily nutrient and sediment loads and can simulate agricultural and urban stormwater best management practices. The WAM partitioned loads among the six subwatersheds of the northern Lake Okeechobee Watershed, which are the focus area for the first BMAP phase. The BMAP compiles projects that were completed, planned, or ongoing and includes the nutrient reductions estimated for each project.

Extensive monitoring and analysis are completed and reported annually in the *Lake Okeechobee Watershed Research and Water Quality Monitoring Results and Activities* (Zhang and Welch 2018), which also provides an update on the planning and project status for the Lake Okeechobee watershed. This report includes updates on conditions of hydrology, water quality, and aquatic habitat in the lake and watershed. Monitoring includes surface water flow; phosphorus, nitrogen, chlorophyll, and microcystin concentrations; water level and water releases; emergent and submergent aquatic vegetation (Figure 37); fish; and wading birds. An algorithm that relates spectral imagery to chlorophyll-*a* concentrations in the lake is under development.

Other models have been developed that evaluate in-lake dynamics. The Lake Okeechobee Environmental Model (LOEM) simulates sediment resuspension and transport in the lake based primarily on wind waves, surface wind stresses, and inflows and outflows (Jin and Ji 2004). A submerged aquatic vegetation (SAV) model was integrated into the LOEM to allow estimates of water quality, SAV, and other variables of interest under multiple management scenarios (Jin and Ji 2013).

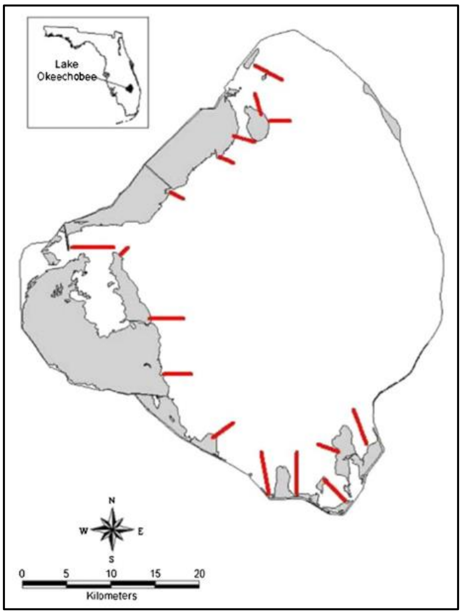

Figure 37. Lake Okeechobee nearshore transects for quarterly sampling (2000–2009) of submerged aquatic vegetation biomass (Jin and Ji 2013).

The various studies, modeling efforts, and restoration activities work towards improving the water quality in Lake Okeechobee and support the Northern Everglades and Estuaries Protection Act, the Lake Okeechobee Protection Program, and the Comprehensive Everglades Restoration Plan.

8 DATA NEEDS

Additional information is needed to further refine the conceptual model and to inform future computer modeling.

8.1 CONCEPTUAL MODEL ENHANCEMENT

The available data on the Winnebago Pool Lakes, in conjunction with general knowledge about shallow lake ecology, are sufficient to provide a basic understanding of the ecological interactions among aquatic macrophytes, water levels, phytoplankton, nutrient loading, wind and wave action, and other external stressors. However, the data are limited and prevent a more nuanced understanding of the ecological interactions in the lake. Data for all of the primary ecological components (e.g., macrophytes, wind and wave action, algal growth, nutrient availability, water levels, etc.) are needed from the same time period to enhance the conceptual model.

The existing monitoring program on the Winnebago Pool Lakes should be expanded to include the following components:

- Water chemistry and water clarity data for all four Winnebago Pool Lakes: chlorophyll-*a*, microcystin, total phosphorus, soluble reactive phosphorus, total Kjeldahl nitrogen, nitrate plus nitrite nitrogen, Secchi depth transparency, total suspended solids, volatile suspended solids, and algal community composition.
    - Data collected twice monthly, year-round; weekly data collection would provide additional benefit.
    - Three existing long-term trend sites in Lake Winnebago.
    - A minimum of one site from each of the Upper Pool lakes.
- Flow and water chemistry data for major tributary inputs and outlet: total phosphorus, soluble reactive phosphorus, total Kjeldahl nitrogen, nitrate plus nitrite nitrogen, total suspended solids, volatile suspended solids.
- Temperature and dissolved oxygen profiles for all four Winnebago Pool Lakes, on same data collection schedule as water chemistry data.
- Lake water levels: Continued data collection at the four USACE gages on Lake Winnebago and the USGS site on Lake Poygan. Tie water levels to elevation so that data sets can be compared to one another. Consider collecting water level data on Lake Winneconne and Lake Butte des Morts as well.
- Macrophyte surveys: Annual site index survey on all four Winnebago Pool Lakes, as outlined by Johnson (2017).
- Fish surveys: Investigation into carp population densities and movement within the Winnebago Pool Lakes (see section 7.1.4).
- Lake depth information for all four Winnebago Pool Lakes, tied to an elevation that can be tied to the USACE's water level data and seasonal targets (Figure 30).

Pre- and post-implementation monitoring should be conducted to determine the effectiveness of the management practice(s).

8.2 MODEL PARAMETERS

Additional data are needed to develop a predictive model of the relationship between aquatic macrophytes and the biotic and abiotic factors that control their abundance and distribution as recommended in section 6. The following section assumes the use of PCLake; if a different computer model is chosen, these model parameters would change to some extent. This section provides an overview of the model parameters needed to run PCLake, as described by Janse (2005).

Primary inputs include the following (modified from Janse 2005); those in *italics* can be input to the model as timeseries. Multiple years of data are recommended in order to provide for both model calibration and validation.

Lake characteristics:

- Mean water depth [m]
- Lake size, expressed as fetch [m]
- Sediment:
    - dry-weight content (d.m.) [%]
    - organic content (or loss on ignition) (OM) [% of d.m.]
    - lutum [%] and/or Fe and Al [mg/g]
    - or (if not available): estimate of sediment type, e.g. clay, sand, peat, mud
- Marsh area [-] (if any)

Water and nutrients:

- *Water inflow and outflow [mm/d]* or retention time [d]. Note that water level changes are simulated by varying the inflow and outflow rate in the model.
- Infiltration / seepage (if any)
- *External P, N and Si loading [g m^{-2} d^{-1}] or concentrations in inflowing water [mg/l]*: sum of point sources, diffuse and sources, surface inflow. Estimate of % dissolved / particulate loading.
- *Input or inflow concentrations of (inorganic) suspended matter*

Ecology (dry weight, N and P variables):

- Phytoplankton—diatoms, small edible algae, cyanobacteria
- Submerged vegetation—multiple groups can be simulated
- Zooplankton
- Zoobenthos
- Fish (optional)—multiple groups can be simulated, however the effect of benthivorous fish is not explicitly modeled
- Marsh vegetation (optional)—shoot and rhizomes

Other:

- *Water temperature*
- *Day light (light irradiation)*
- *Wind*

Lake history and management:

- P and N concentrations in the sediment top layer (give depth), or estimate of historical nutrient loading
- *Intensity of fishery [d^{-1}]*
- Any management measures (being) conducted (e.g., biomanipulation, dredging, mowing)

9 NEXT STEPS

The conceptual model presented in section 5 summarizes the current understanding of algal and aquatic macrophyte dynamics in the Winnebago Pool Lakes. In general, additional data, as recommended in section 8, are needed to increase the level of confidence in the conceptual model and provide the level of detail needed to propose many of the lake restoration activities. In addition, the computer model PCLake is recommended in section 6 to simulate existing conditions and the effect of potential implementation activities. This model is very data intensive and requires extensive monitoring and data collection.

In the short-term, suggested next steps include:

1. Initiate communication with the USACE to evaluate the potential for changes to the water level regime in the Winnebago Pool Lakes. An adaptive management approach is recommended over the period of several years to determine the potential effect on the macrophyte assemblage in the lakes.
2. Develop partnerships to collect data and begin initial modeling activities, including final model selection, considering pros and cons of both PCLake and a linked WASP/EFDC model. It is likely that coordination will be needed with the Netherlands Institute of Ecology to more thoroughly understand the requirements of PCLake if that model is chosen. Identify funding mechanisms for computer model development.
3. Continue existing data collection and implement additional recommendations as provided in section 8. Develop and implement monitoring plan to collect data needed for model simulation and calibration, as needed.
4. Continue to compile other known datasets that were unavailable for this project (e.g., TMDL-related information, continuous monitoring data on algal pigments and other water quality parameters collected by the *Aquatic Environmental Microbiology and Chemistry* group at the University of Wisconsin Milwaukee, and data collected by Bart De Stasio at Lawrence University). These data, along with new information being collected, should be used to update the conceptual model.
5. Complete feasibility studies and implement pilot studies to test the various lake restoration alternatives.

A programmatic approach is also recommended to integrate the work being conducted in the Winnebago Pool Lakes and their watershed. This approach could include a designated lake manager and a central clearing house for information and data related to the Winnebago Pool Lakes.

Additional study questions and research needs identified by Wisconsin DNR for follow-up include:

1. What are the drivers of phytoplankton succession in the Winnebago Pool Lakes?
    a. Are there historical data on seasonal succession of diatoms, green algae, diazotrophic (nitrogen-fixing) cyanobacteria, and non-diazotrophic cyanobacteria, both in terms of abundances and relative proportions? If historical data are made available, the data should be evaluated with respect to nitrogen and phosphorus concentrations in the tributary inputs and in the lakes themselves.
    b. Future data collection should include monitoring of algal species abundance and community composition throughout the year. These data can then be evaluated with respect to seasonal nutrient concentrations, cyanotoxin production, water clarity, and macrophyte densities and species composition.

2. Do we have increasing toxin production or increasing *Microcystis* abundance, as has been observed in similar lake systems?
3. Additional data could be collected to provide more detail for the conceptual model on nutrient cycling in the lake. Characteristics of interest include phytoplankton nutrient assimilation, phytoplankton abundance, taxa competitive advantage, and relationship to nutrient availability and nutrient limitation. Implications for toxin production could also be addressed.
4. Determine if algal growth, and more specifically, cyanotoxin production, is affected by nitrogen inputs. If it is, should stakeholders examine lake nitrogen budgets and explore if there are any readily implementable, grant eligible, or reasonable options for reducing nitrogen loading to the lake?

5. Is there a strong theoretical or observed empirical basis to consider dual nutrient reductions (i.e., nitrogen and phosphorus) as a voluntary, community-driven strategy to more quickly reduce frequency and intensity of cyanotoxin occurrence?
6. While phosphorus can limit cyanobacterial occurrence if reduced below a target threshold, due to the time lag for the target threshold to be achieved (even under fully successful load reduction implementation scenarios), should dual nutrient management be considered as an interim measure to reduce cyanotoxins in the near to medium term?
7. Literature review to identify examples of modeling phytoplankton community and composition changes.

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APPENDIX A. ADDITIONAL INFORMATION ON PCLAKE

Applicability

PCLake was built upon empirical data for 43 lakes. The model was tested to ensure the response reflected the data available for these calibration lakes. Before applying PCLake to other lakes outside of the calibration set, it is good practice to ensure that the model is representative of the applicable conditions. These calibration lakes had the following characteristics, as described by Janse (2005):

"both 'clear' and 'turbid', all sediment types, with P inflow concentration ranging from 0.03 - 2 mgP l_{-1}, depth from 0.8 - 6.8 m, area from 1 – 4500 ha and retention time from 7 to over 500 days. Total P concentrations measure between 0.001 and 1.5 mgP/l, total N between 0.2 and 6.6 mgN/l, chlorophyll-*a* between 2 and > 200 mg m_{-3}, vegetation cover between 0 and 90 % and Secchi depth between 0.2 and 2.0 m."

The Upper Pool Lakes generally fit within this set of parameters, and therefore PCLake is likely applicable in those cases. Lake Winnebago is considerably larger than the biggest lake in the calibration set (4,570 ha versus 53,400 ha), therefore care must be taken to ensure that the model equations and calculations are relevant.

Management Options

The following management options can be evaluated using PCLake (as described in Janse 2005):

Catchment management:

- Increase or reduction in nutrient (P, N) loading
- Hydrological measures, flushing
- Wetland restoration
- Water level, water level fluctuations

Local factors or management:

- Biomanipulation
- Fishery
- Birds
- Mowing
- Dredging
- Reducing fetch
- Sediment traps (local pits)

Autonomous factors:

- Type of sediment
- Water temperature

As described in Janse (2005), the following are examples of options could be modeled:

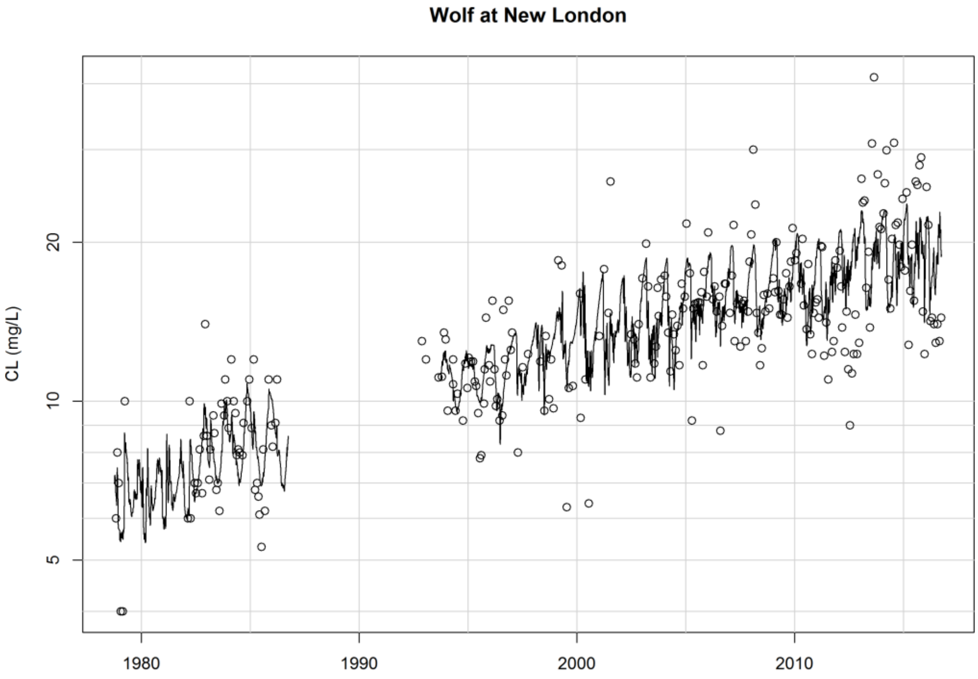
- Increasing the hydraulic loading rate and thus decreasing the water retention time (provided that the extra inflowing water has a lower P concentration than the lake).
- Decreasing the wind effects on settling and resuspension, for instance by making sediment traps (deep areas) or by compartmentalization.
- Decreasing the water depth, by water level manipulation.
- Improving the conditions for reestablishment of a marsh zone surrounding or connected to the lake. This may be enhanced by natural water level fluctuations.
- Fishery management specifically removing planktivorous fish.
- Dredging to remove nutrient–rich sediment layers.

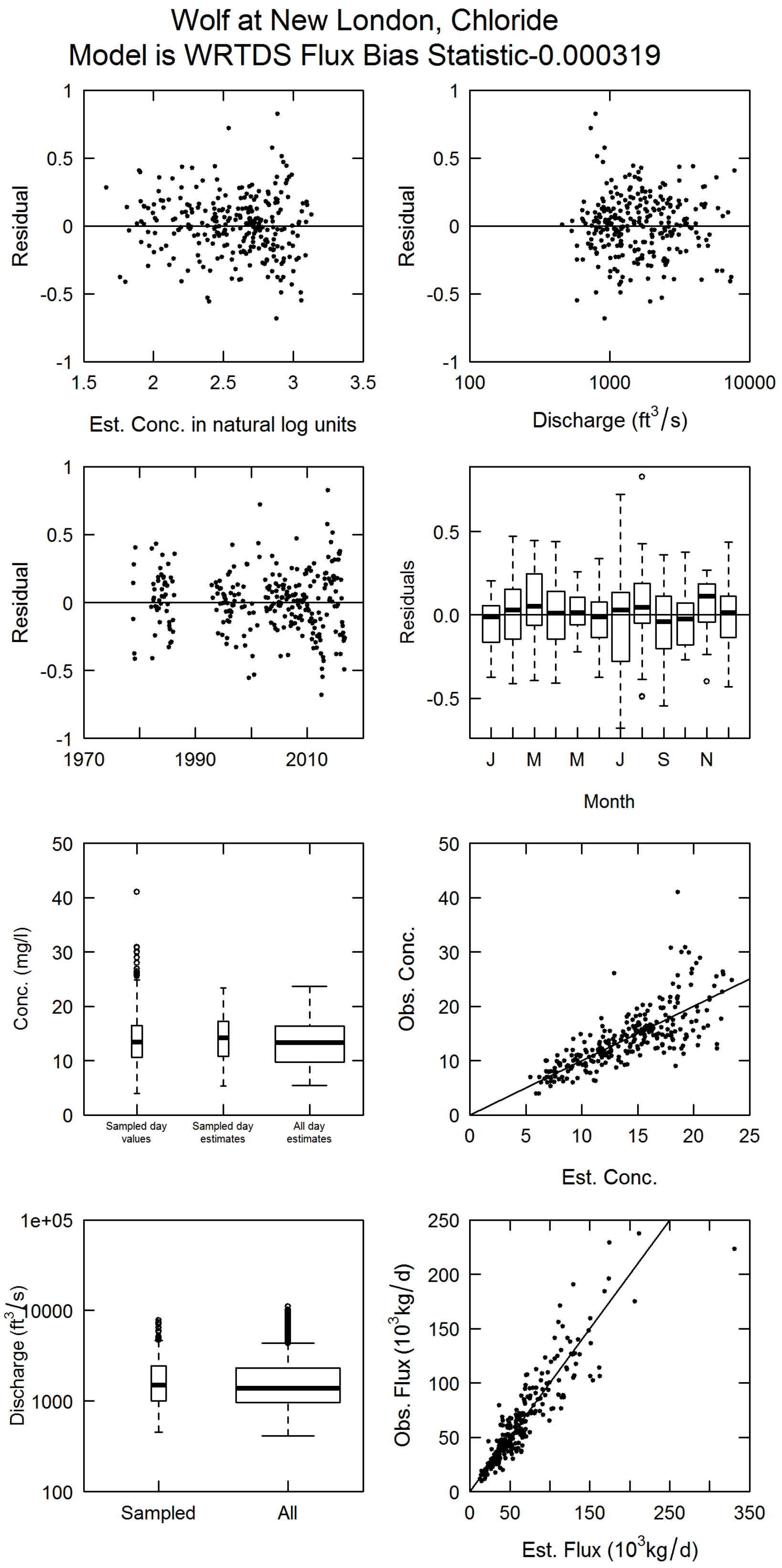
APPENDIX B. TRIBUTARY LONG TERM TREND WATER QUALITY DATA SUMMARY

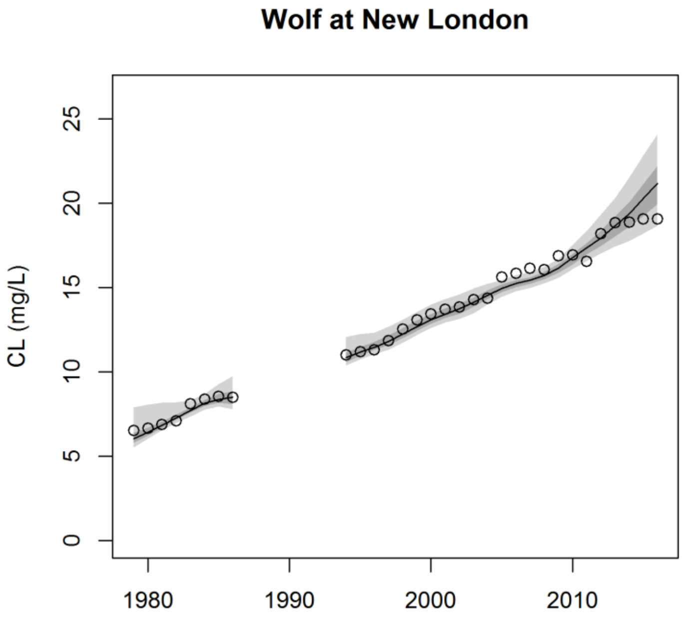
WDNR evaluated water quality data from four river stations with long term data—Wolf River at New London, Fox River at Berlin, Fox River at Oshkosh, and Fox River at Neenah. The trend analysis used the R package *EGRET* (Exploration and Graphics for RivEr Trends) and its *Weighted Regressions on Time, Discharge and Season* (WRTDS) model to describe long-term trends in both concentration and flux.

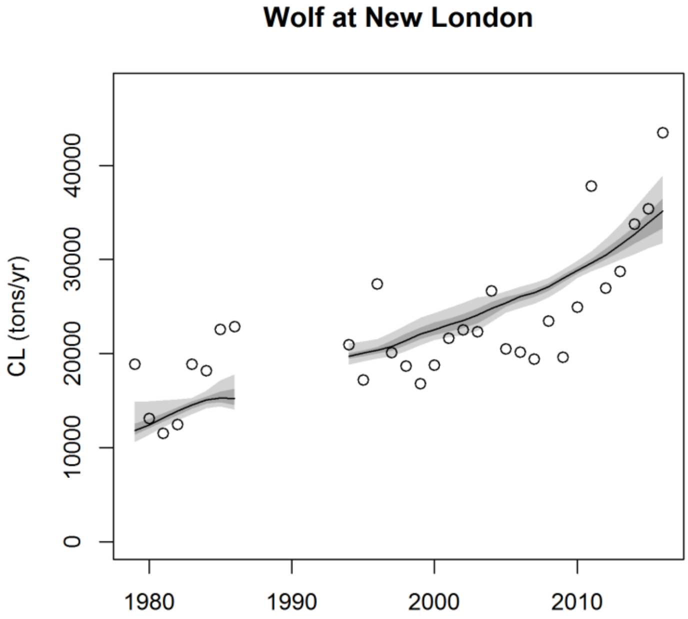
WOLF RIVER AT NEW LONDON (693035)

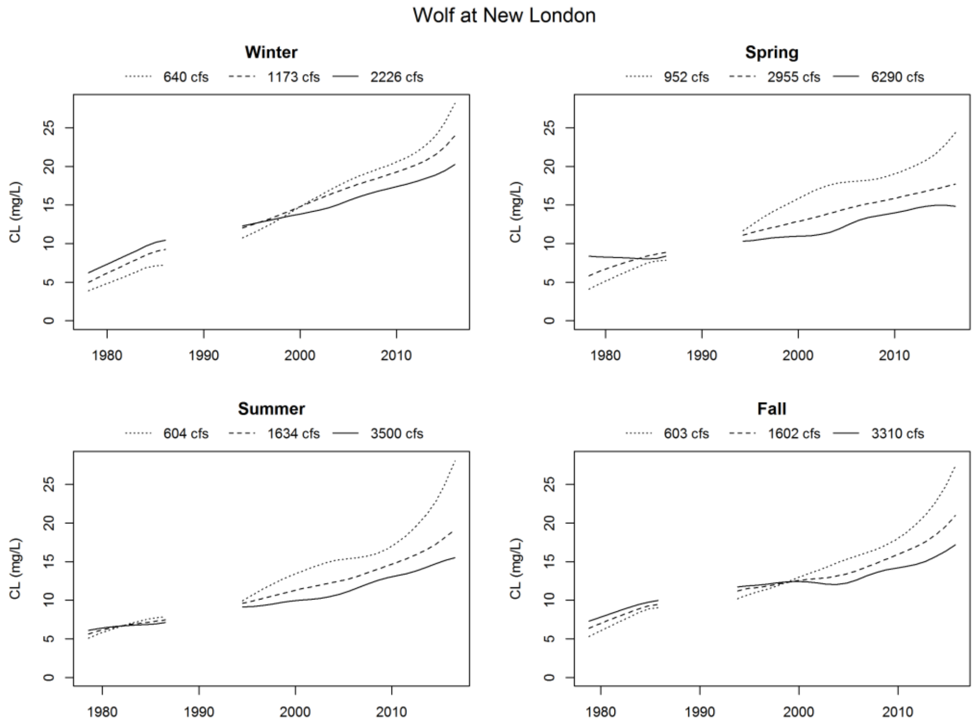
Chloride



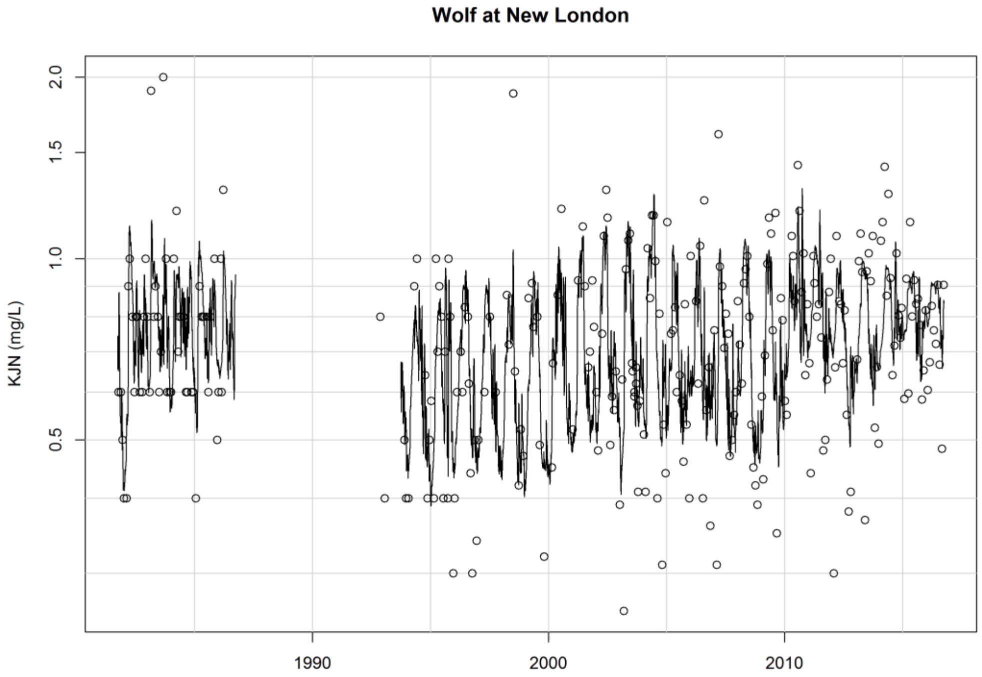

Wolf at New London, Chloride
Model is WRTDS Flux Bias Statistic-0.000319
Residual
1
0.5
0
-0.5
-1
1.5
2
2.5
3
3.5
Est. Conc. in natural log units
Residual
100
1000
10000
Discharge (ft3/s)
Residual
1970
1990
2010
Residuals
0.5
0.0
-0.5
J
M
M
J
S
N
Month
Conc. (mg/l)
50
40
30
20
10
0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
0
5
10
15
20
25
Est. Conc.
Discharge (ft3/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (10³kg/d)
250
200
150
100
50
0
0
50
150
250
350
Est. Flux (10³kg/d)

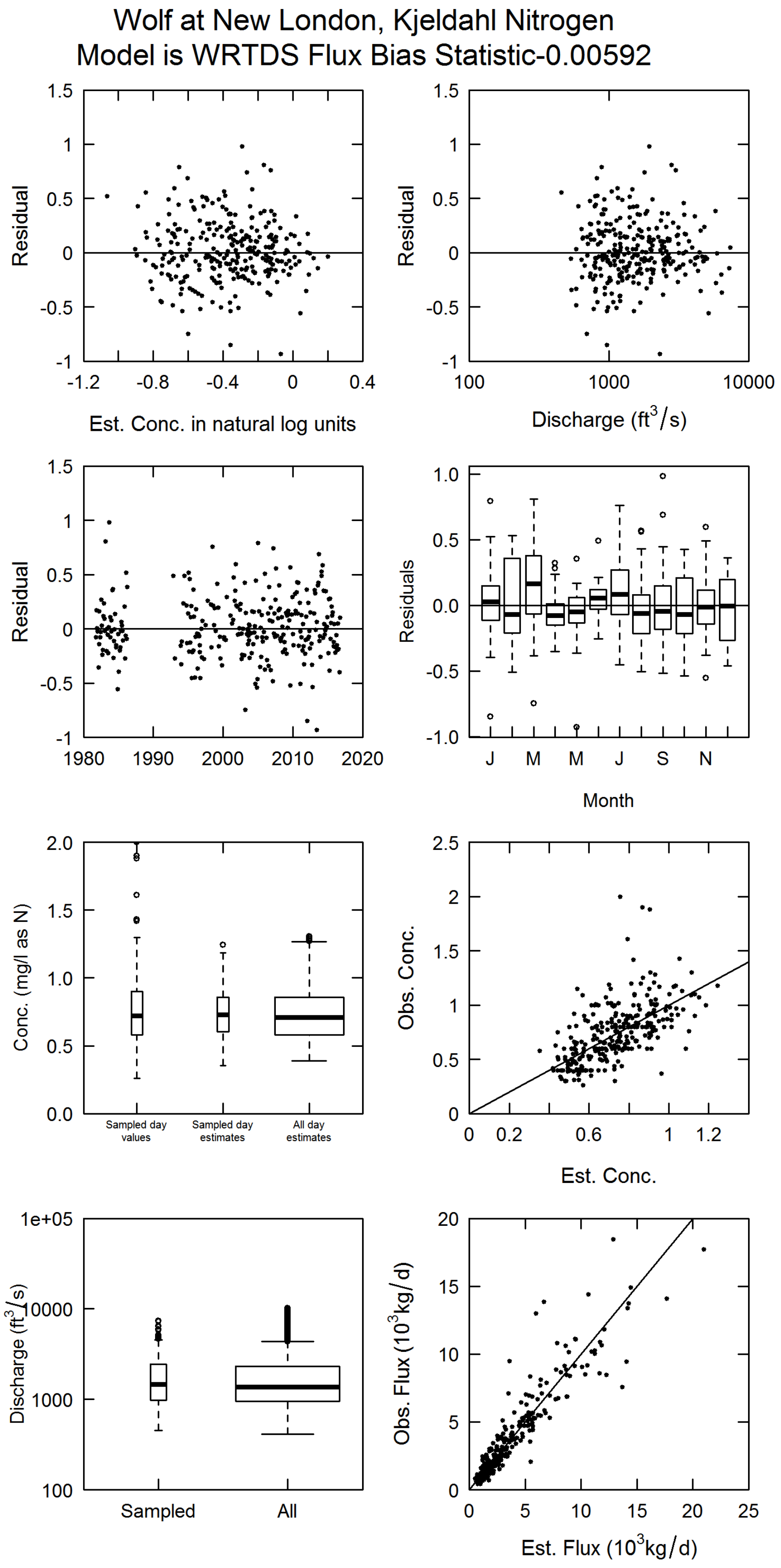

Wolf at New London
CL (mg/L)
0
5
10
15
20
25
1980
1990
2000
2010

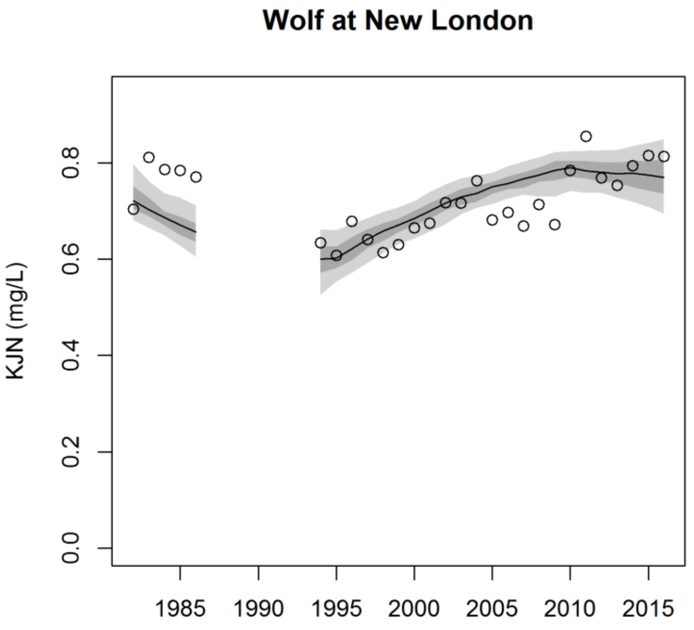

Wolf at New London
CL (tons/yr)
0
10000
20000
30000
40000
1980
1990
2000
2010

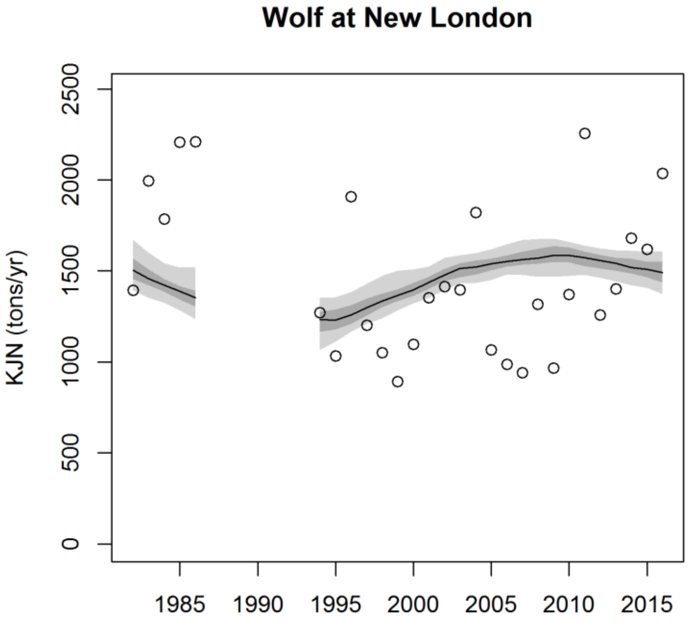


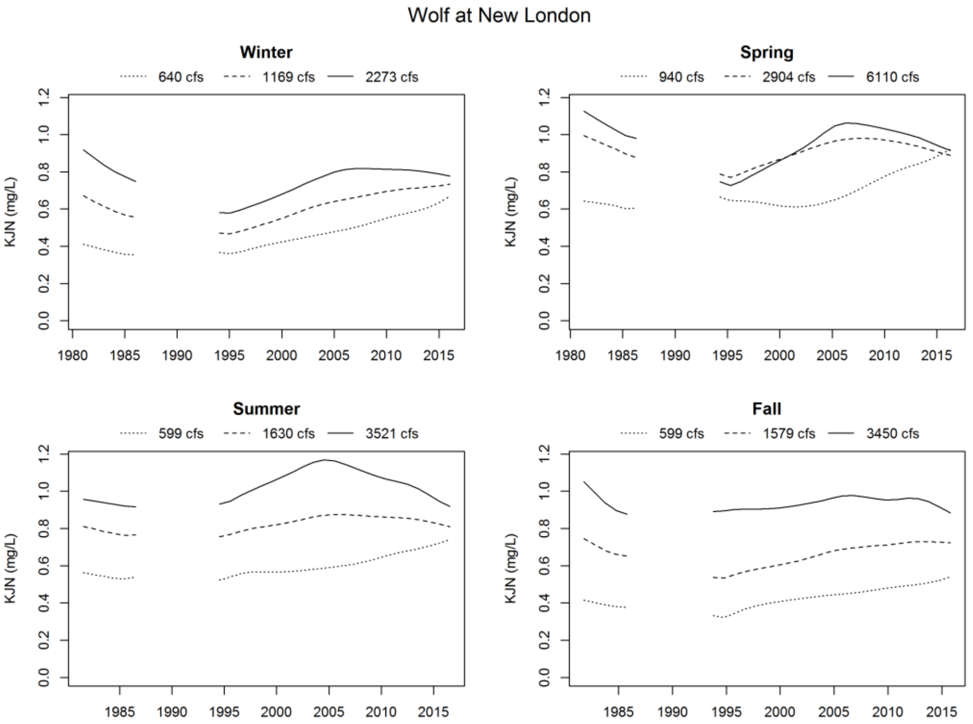
Total Kjeldahl Nitrogen



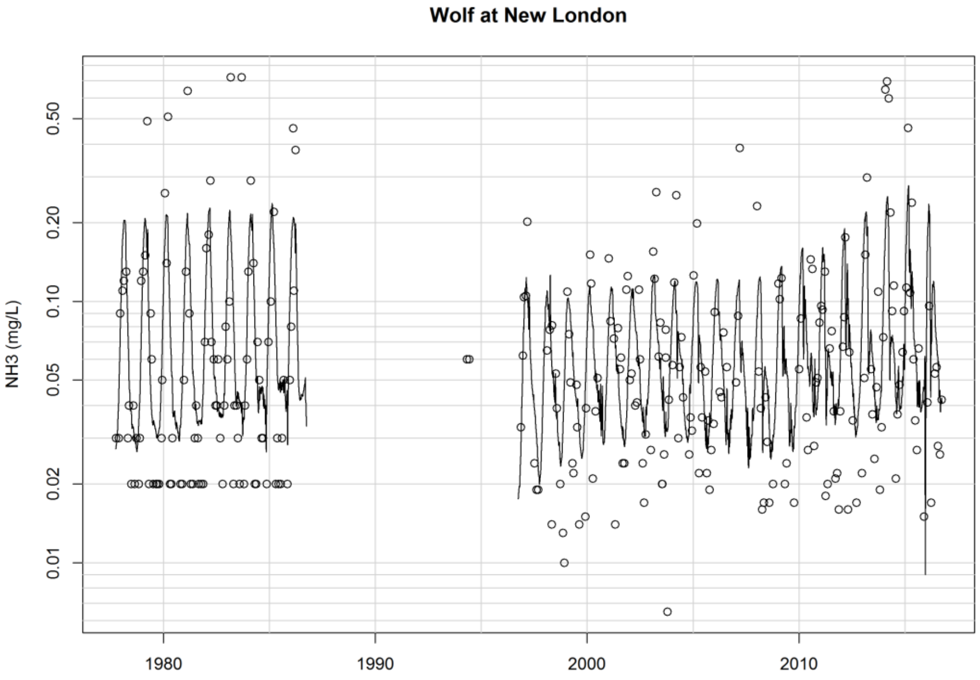

Wolf at New London, Kjeldahl Nitrogen
Model is WRTDS Flux Bias Statistic-0.00592
Residual
1.5
1
0.5
0
-0.5
-1
-1.2
-0.8
-0.4
0
0.4
Est. Conc. in natural log units
Residual
100
1000
10000
Discharge (ft3/s)
Residual
1980
1990
2000
2010
2020
Residuals
1.0
0.5
0.0
-0.5
-1.0
J
M
M
J
S
N
Month
Conc. (mg/l as N)
2.0
1.5
1.0
0.5
0.0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
2.5
2
1.5
1
0.5
0
0
0.2
0.6
1
1.2
Est. Conc.
Discharge (ft3/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (103kg/d)
20
15
10
5
0
0
5
10
15
20
25
Est. Flux (103kg/d)

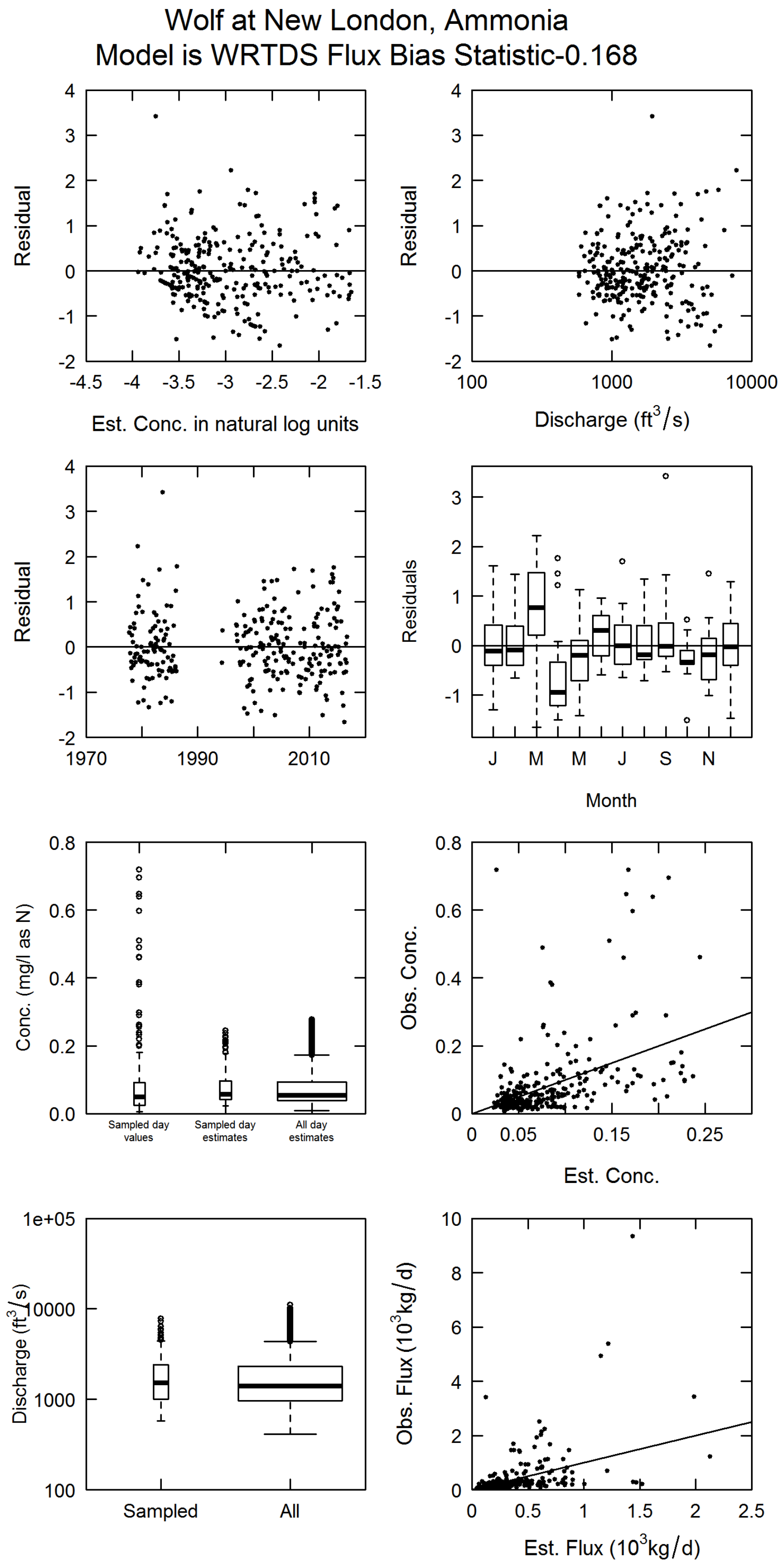

Wolf at New London
KJN (mg/L)
0.0
0.2
0.4
0.6
0.8
1985
1990
1995
2000
2005
2010
2015

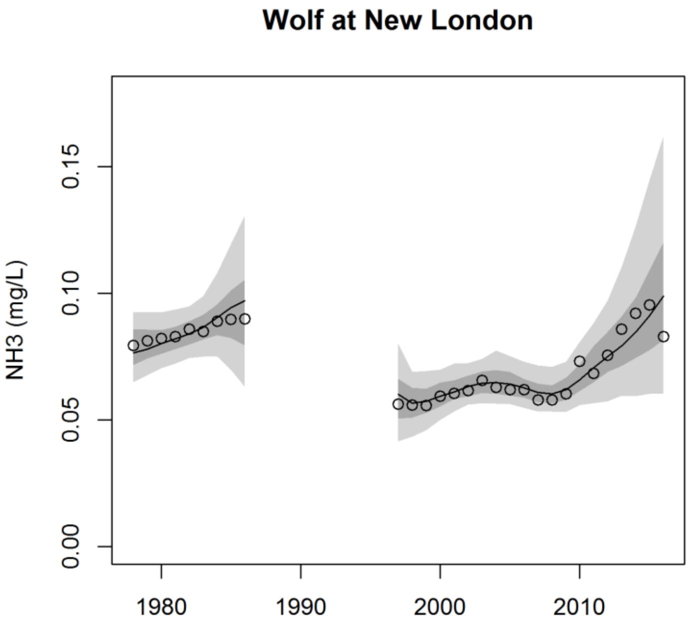

Wolf at New London
KJN (tons/yr)
0
500
1000
1500
2000
2500
1985
1990
1995
2000
2005
2010
2015

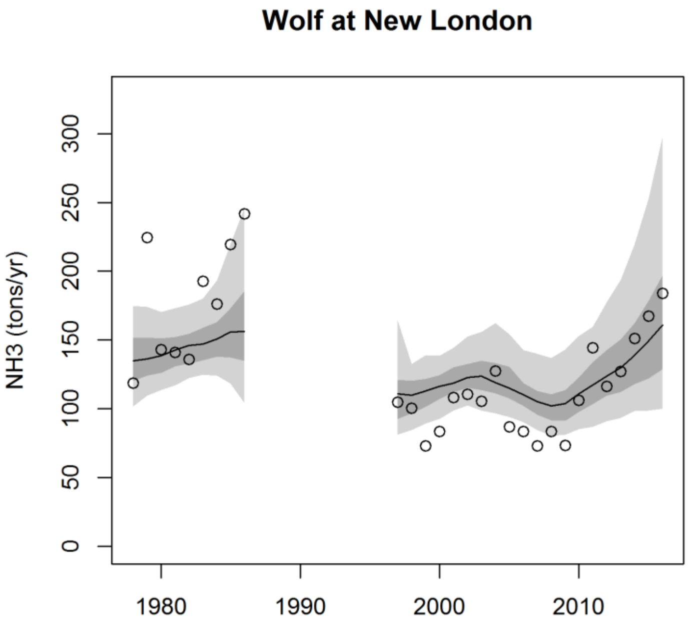

Wolf at New London
Winter
640 cfs
1169 cfs
2273 cfs
Spring
940 cfs
2904 cfs
6110 cfs
Summer
599 cfs
1630 cfs
3521 cfs
Fall
599 cfs
1579 cfs
3450 cfs
KJN (mg/L)
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1980
1985
1990
1995
2000
2005
2010
2015

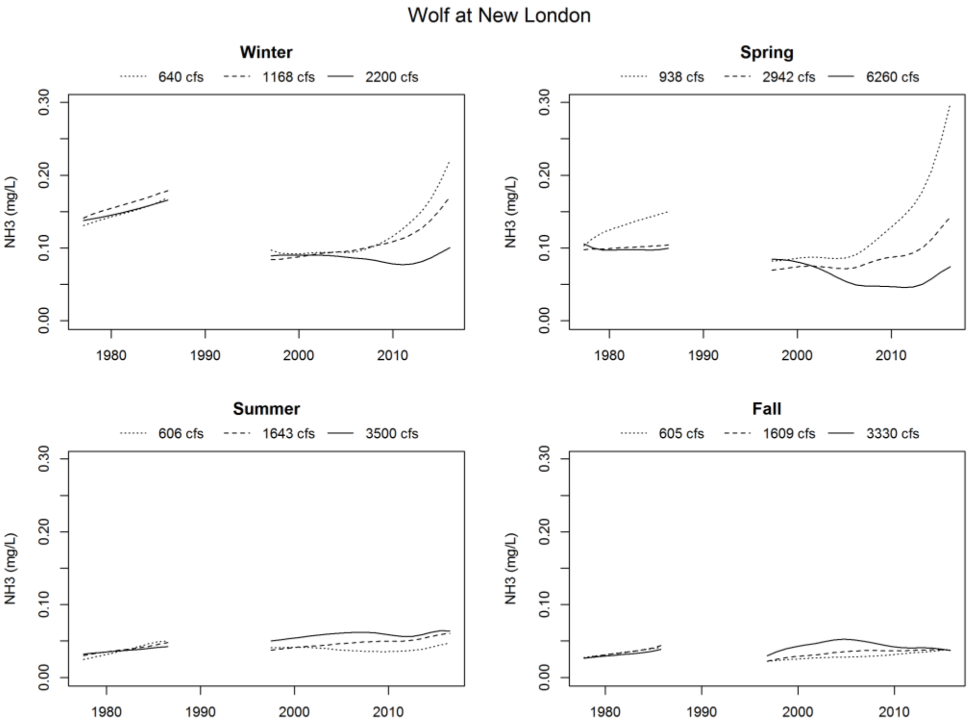
Ammonia


Wolf at New London
NH3 (mg/L)
0.01
0.02
0.05
0.10
0.20
0.50
1980
1990
2000
2010

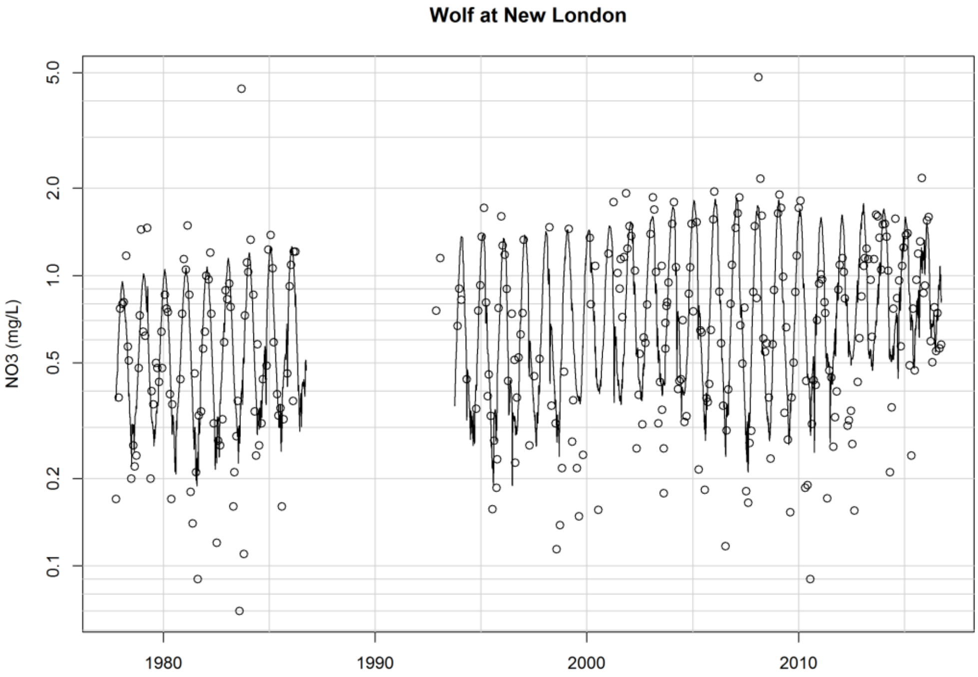

Wolf at New London, Ammonia
Model is WRTDS Flux Bias Statistic-0.168
Residual
Est. Conc. in natural log units
-4.5
-4
-3.5
-3
-2.5
-2
-1.5
Residual
Discharge (ft^3/s)
100
1000
10000
Residual
1970
1990
2010
Residuals
Month
J
M
M
J
S
N
Conc. (mg/l as N)
0.0
0.2
0.4
0.6
0.8
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
Est. Conc.
0
0.05
0.15
0.25
Discharge (ft^3/s)
100
1000
10000
1e+05
Sampled
All
Obs. Flux (10^{3}kg/d)
Est. Flux (10^{3}kg/d)
0
0.5
1
1.5
2
2.5
0
2
4
6
8
10

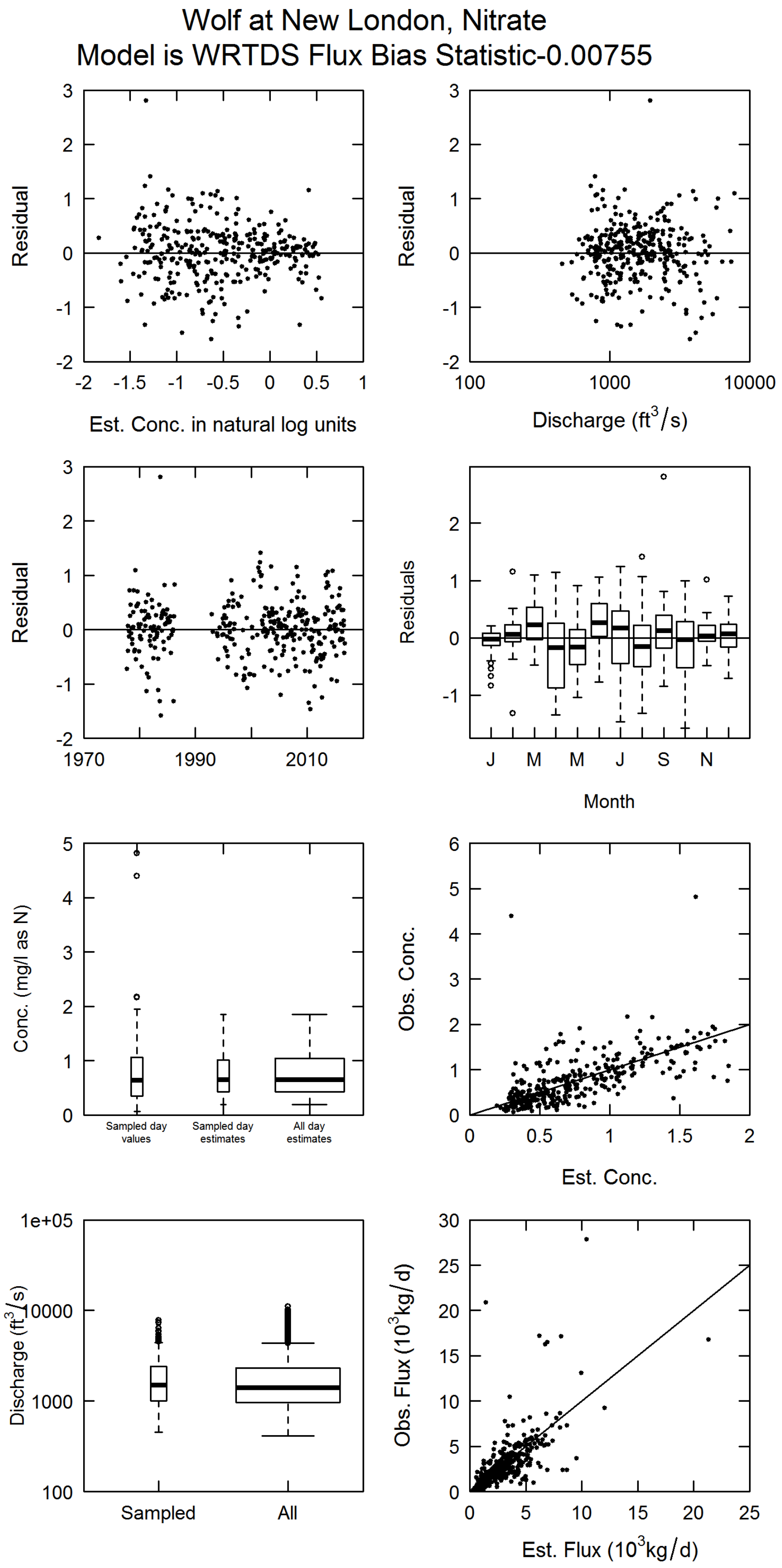

Wolf at New London
NH3 (mg/L)
0.00
0.05
0.10
0.15
1980
1990
2000
2010

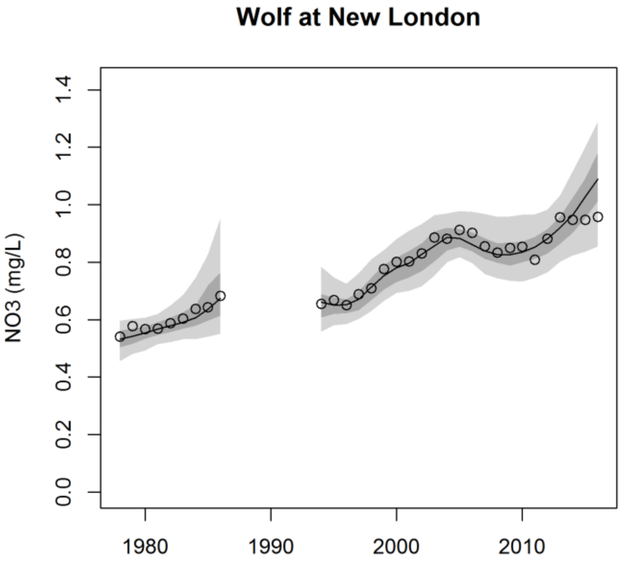

Wolf at New London
NH3 (tons/yr)
0
50
100
150
200
250
300
1980
1990
2000
2010

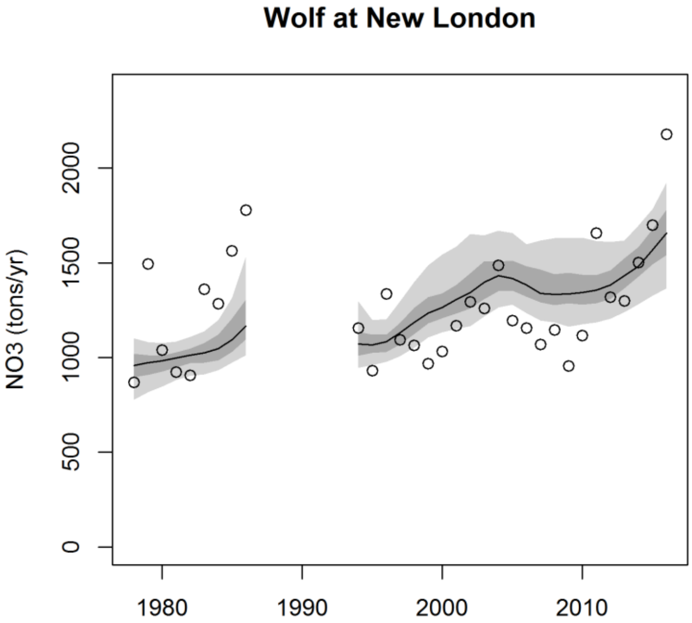

Wolf at New London
Winter
640 cfs
1168 cfs
2200 cfs
Spring
938 cfs
2942 cfs
6260 cfs
Summer
606 cfs
1643 cfs
3500 cfs
Fall
605 cfs
1609 cfs
3330 cfs
NH3 (mg/L)
0.00
0.10
0.20
0.30
1980
1990
2000
2010

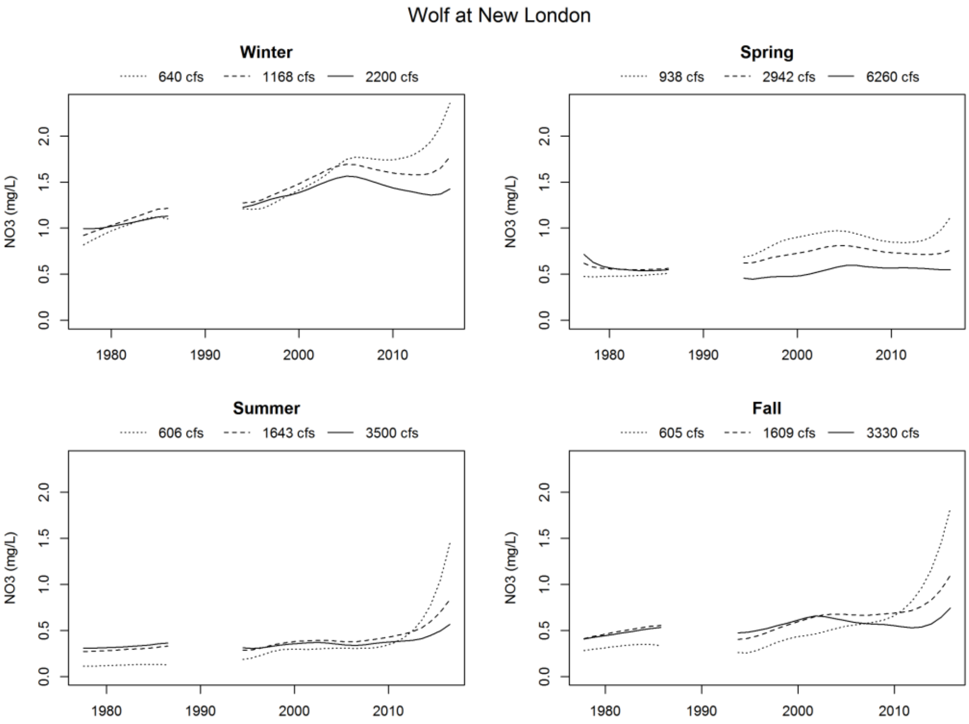
Nitrate


Wolf at New London
NO3 (mg/L)
0.1
0.2
0.5
1.0
2.0
5.0
1980
1990
2000
2010

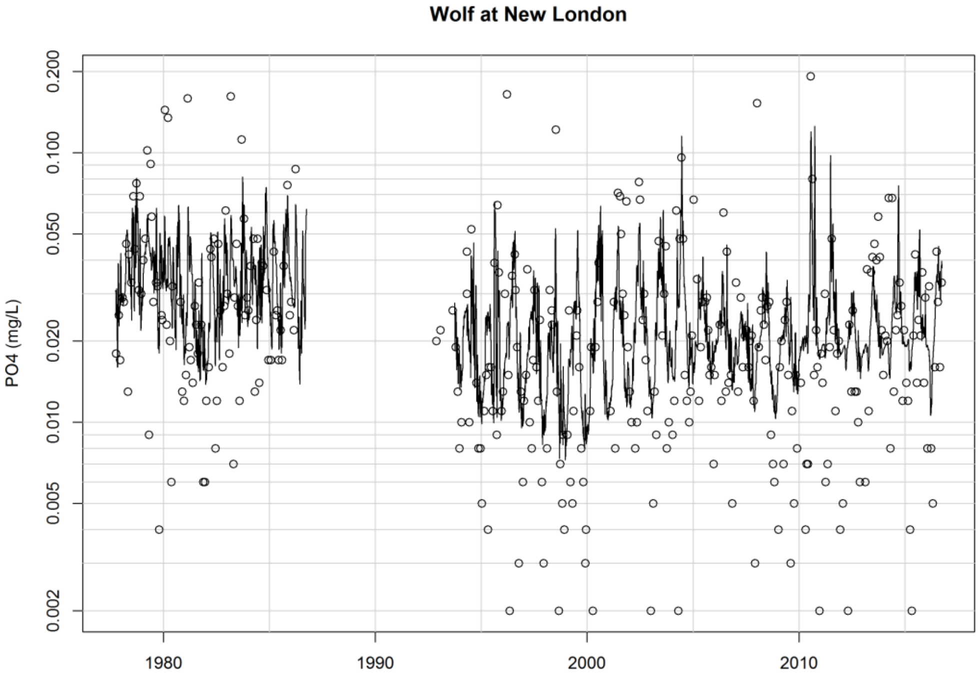

Wolf at New London, Nitrate
Model is WRTDS Flux Bias Statistic-0.00755
Residual
Est. Conc. in natural log units
-2
-1.5
-1
-0.5
0
0.5
1
3
2
1
0
-1
-2
Residual
Discharge (ft³/s)
100
1000
10000
Residual
1970
1990
2010
Residuals
Month
J
M
M
J
S
N
Conc. (mg/l as N)
Sampled day values
Sampled day estimates
All day estimates
5
4
3
2
1
0
Obs. Conc.
Est. Conc.
0
0.5
1
1.5
2
6
Discharge (ft³/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (10³kg/d)
Est. Flux (10³kg/d)
0
5
10
15
20
25
30

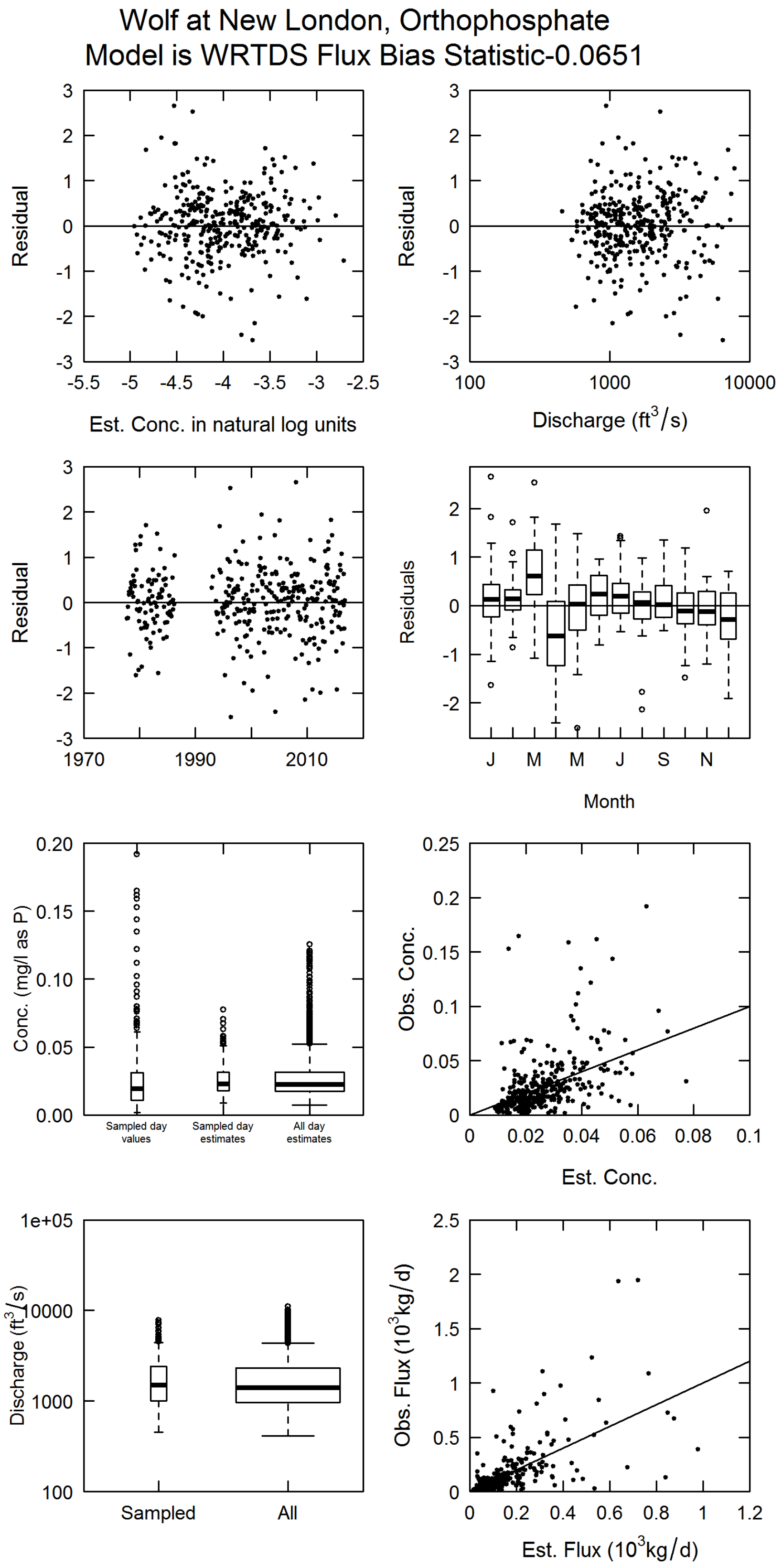

Wolf at New London
NO3 (mg/L)
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1980
1990
2000
2010

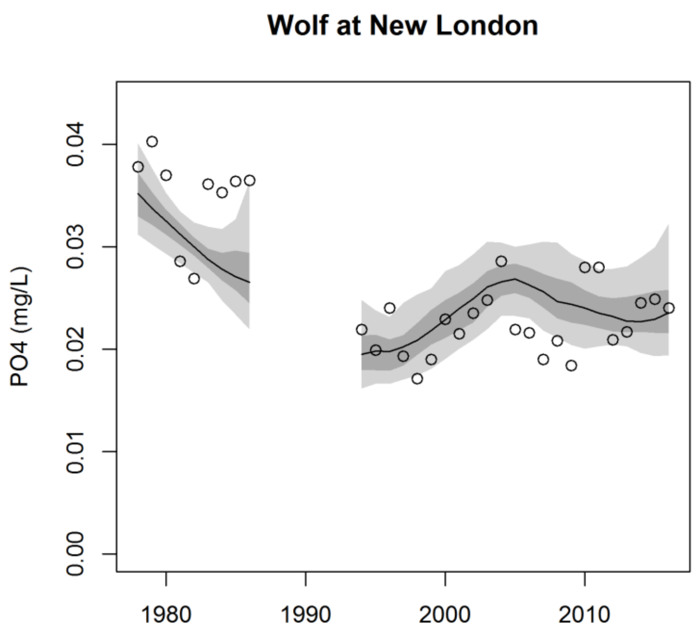

Wolf at New London
NO3 (tons/yr)
0
500
1000
1500
2000
1980
1990
2000
2010

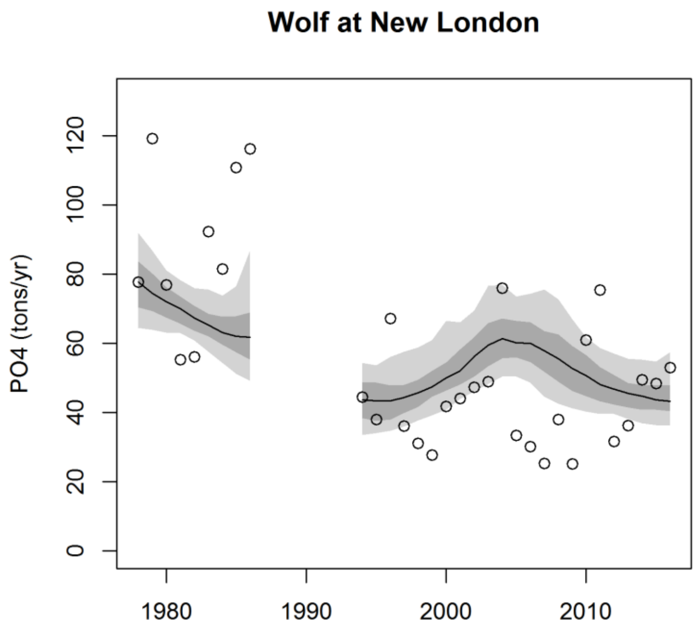

Wolf at New London
Winter
640 cfs
1168 cfs
2200 cfs
Spring
938 cfs
2942 cfs
6260 cfs
Summer
606 cfs
1643 cfs
3500 cfs
Fall
605 cfs
1609 cfs
3330 cfs
NO3 (mg/L)
0.0
0.5
1.0
1.5
2.0
1980
1990
2000
2010

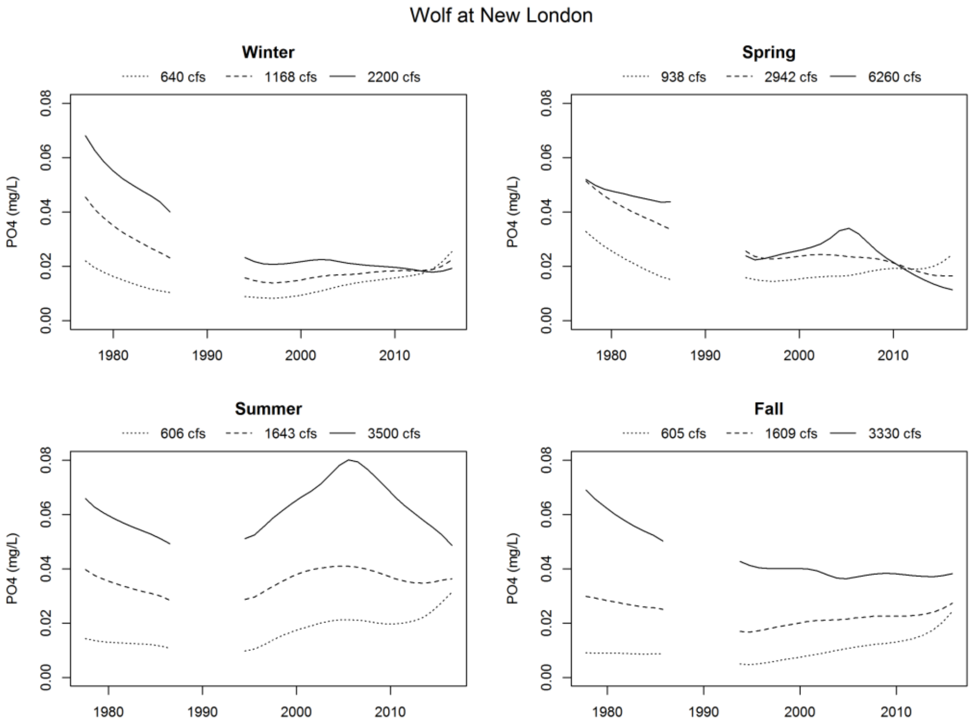
Phosphate


Wolf at New London
PO4 (mg/L)
0.002
0.005
0.010
0.020
0.050
0.100
0.200
1980
1990
2000
2010

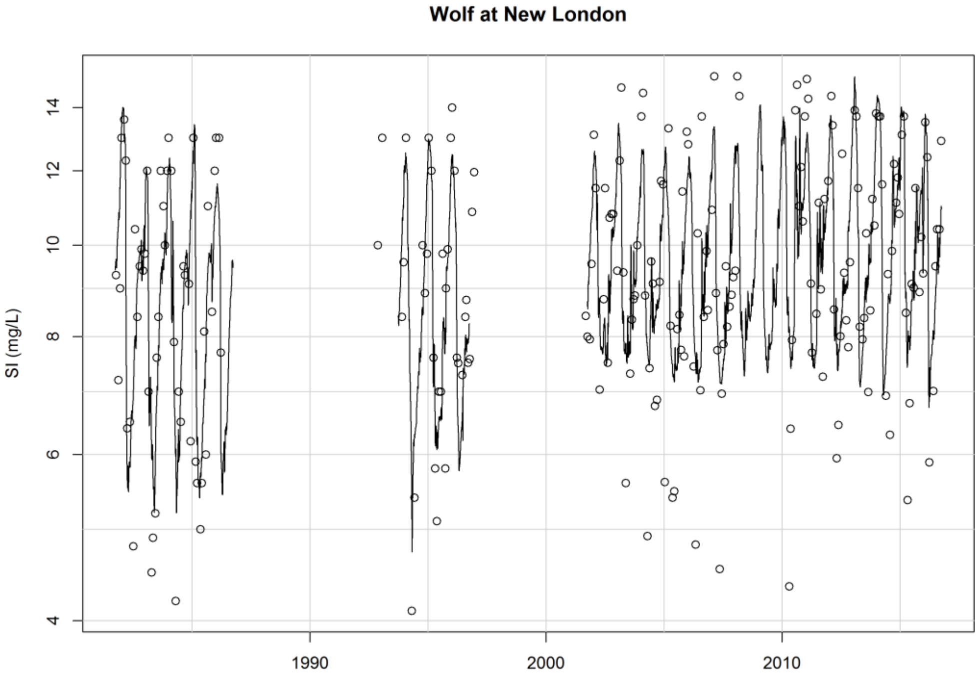

Wolf at New London, Orthophosphate
Model is WRTDS Flux Bias Statistic-0.0651
Residual
3
2
1
0
-1
-2
-3
-5.5
-5
-4.5
-4
-3.5
-3
-2.5
Est. Conc. in natural log units
Residual
100
1000
10000
Discharge (ft^3/s)
Residual
1970
1990
2010
Residuals
J
M
M
J
S
N
Month
Conc. (mg/l as P)
0.20
0.15
0.10
0.05
0.00
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
0.25
0.2
0.15
0.1
0.05
0
0.02
0.04
0.06
0.08
0.1
Est. Conc.
Discharge (ft^3/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (10^{3}kg/d)
2.5
2
1.5
1
0.5
0
0.2
0.4
0.6
0.8
1
1.2
Est. Flux (10^{3}kg/d)

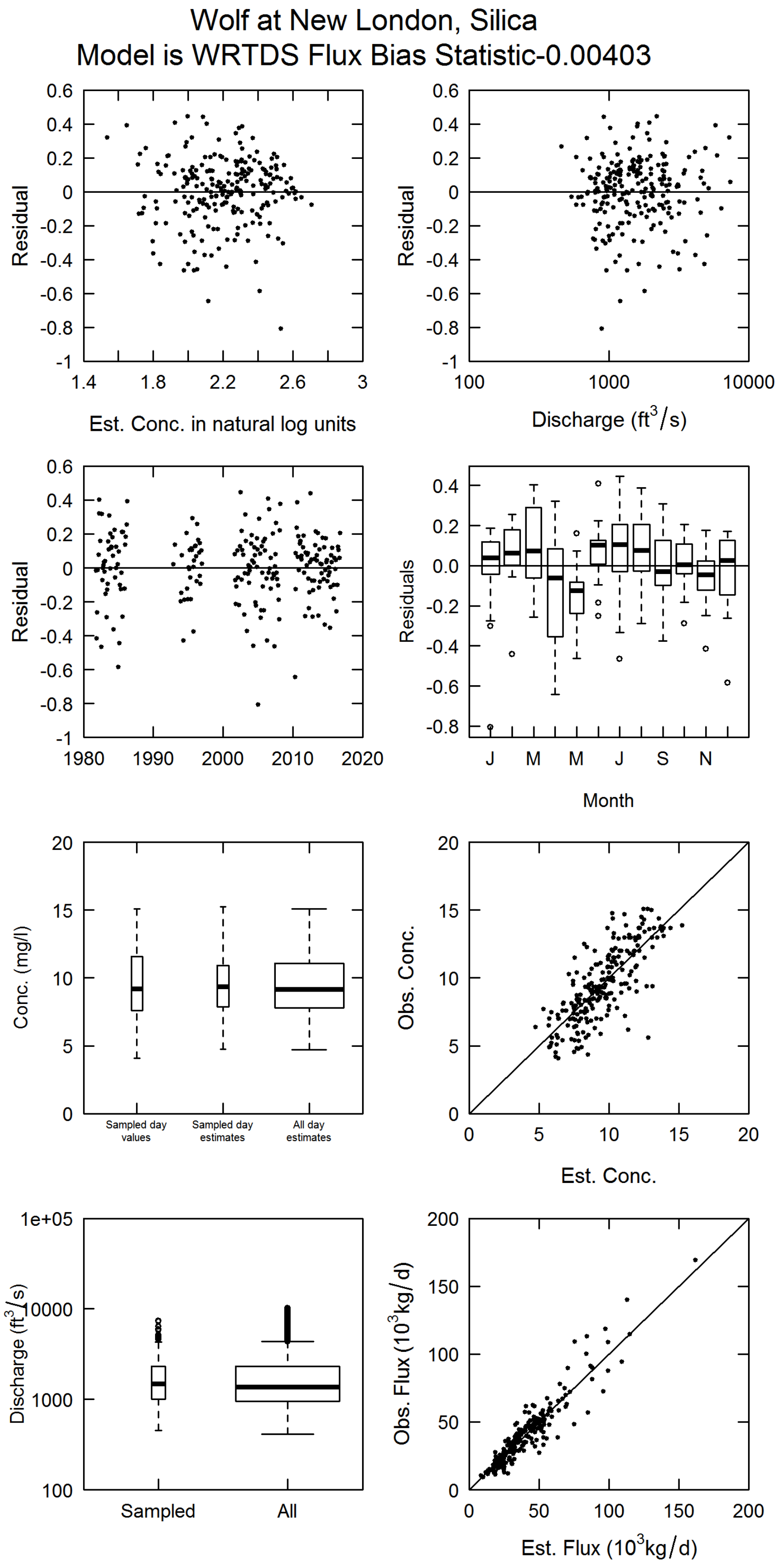

Wolf at New London
PO4 (mg/L)
0.00
0.01
0.02
0.03
0.04
1980
1990
2000
2010

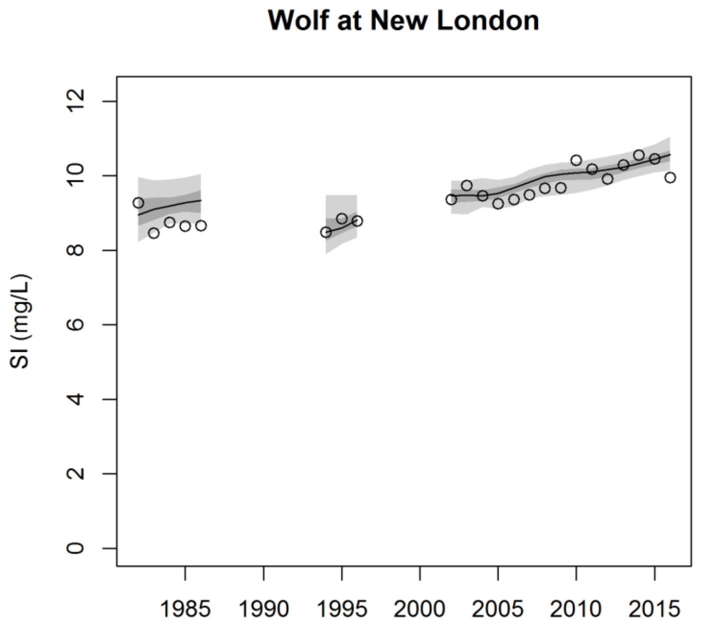

Wolf at New London
PO4 (tons/yr)
0
20
40
60
80
100
120
1980
1990
2000
2010

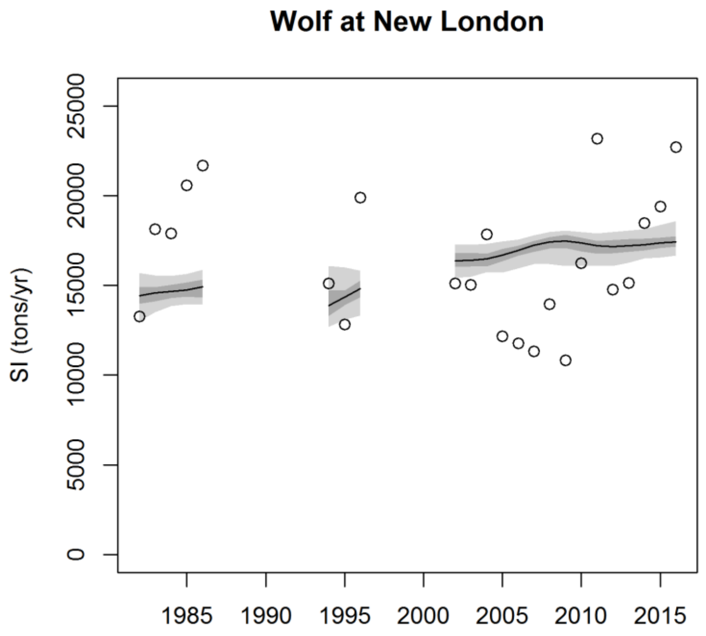

Wolf at New London
Winter
640 cfs
1168 cfs
2200 cfs
Spring
938 cfs
2942 cfs
6260 cfs
Summer
606 cfs
1643 cfs
3500 cfs
Fall
605 cfs
1609 cfs
3330 cfs
PO4 (mg/L)
0.00
0.02
0.04
0.06
0.08
1980
1990
2000
2010

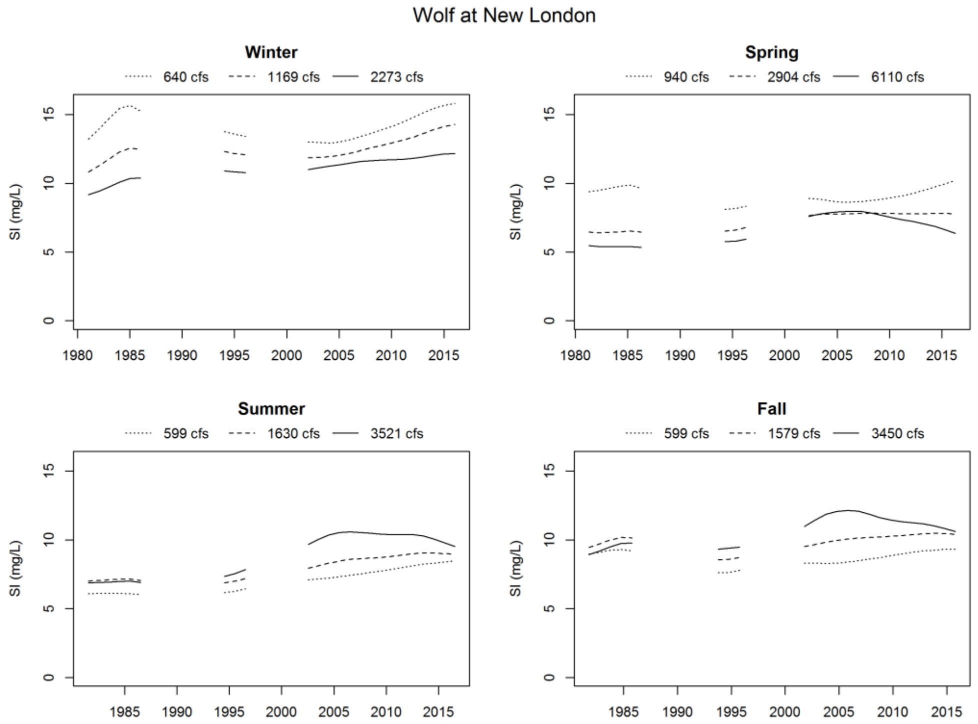
Silica


Wolf at New London
SI (mg/L)
4
6
8
10
12
14
1990
2000
2010

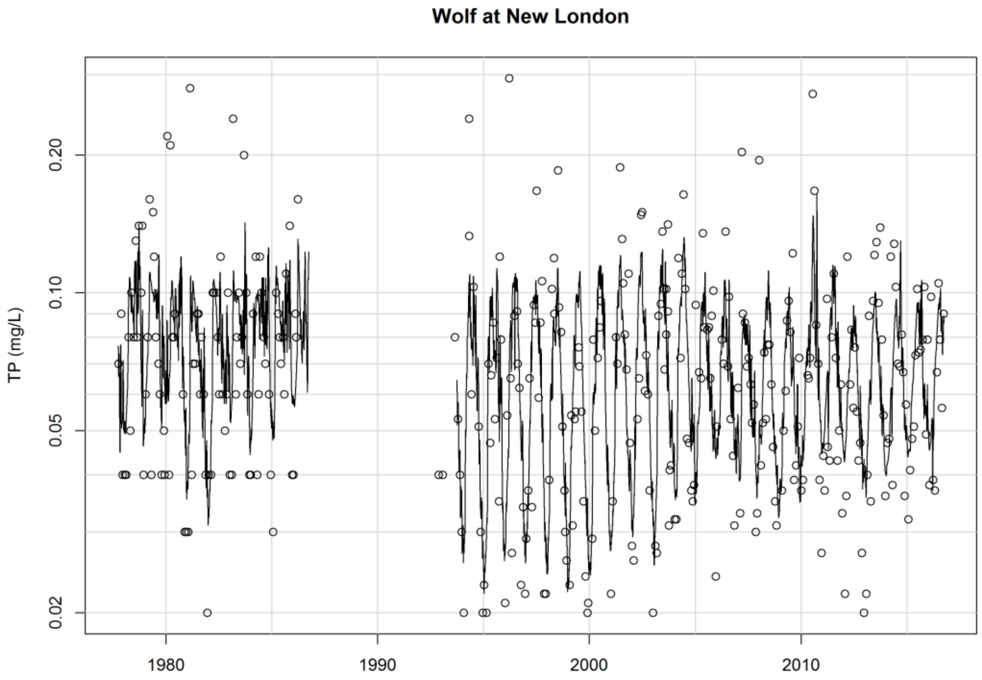

Wolf at New London, Silica
Model is WRTDS Flux Bias Statistic-0.00403
Residual
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
-0.8
-1
1.4
1.8
2.2
2.6
3
Est. Conc. in natural log units
100
1000
10000
Discharge (ft³/s)
1980
1990
2000
2010
2020
Residuals
0.0
J
M
M
J
S
N
Month
Conc. (mg/l)
20
15
10
5
0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
Est. Conc.
Discharge (ft³/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (10³kg/d)
200
150
100
50
Est. Flux (10³kg/d)


Wolf at New London

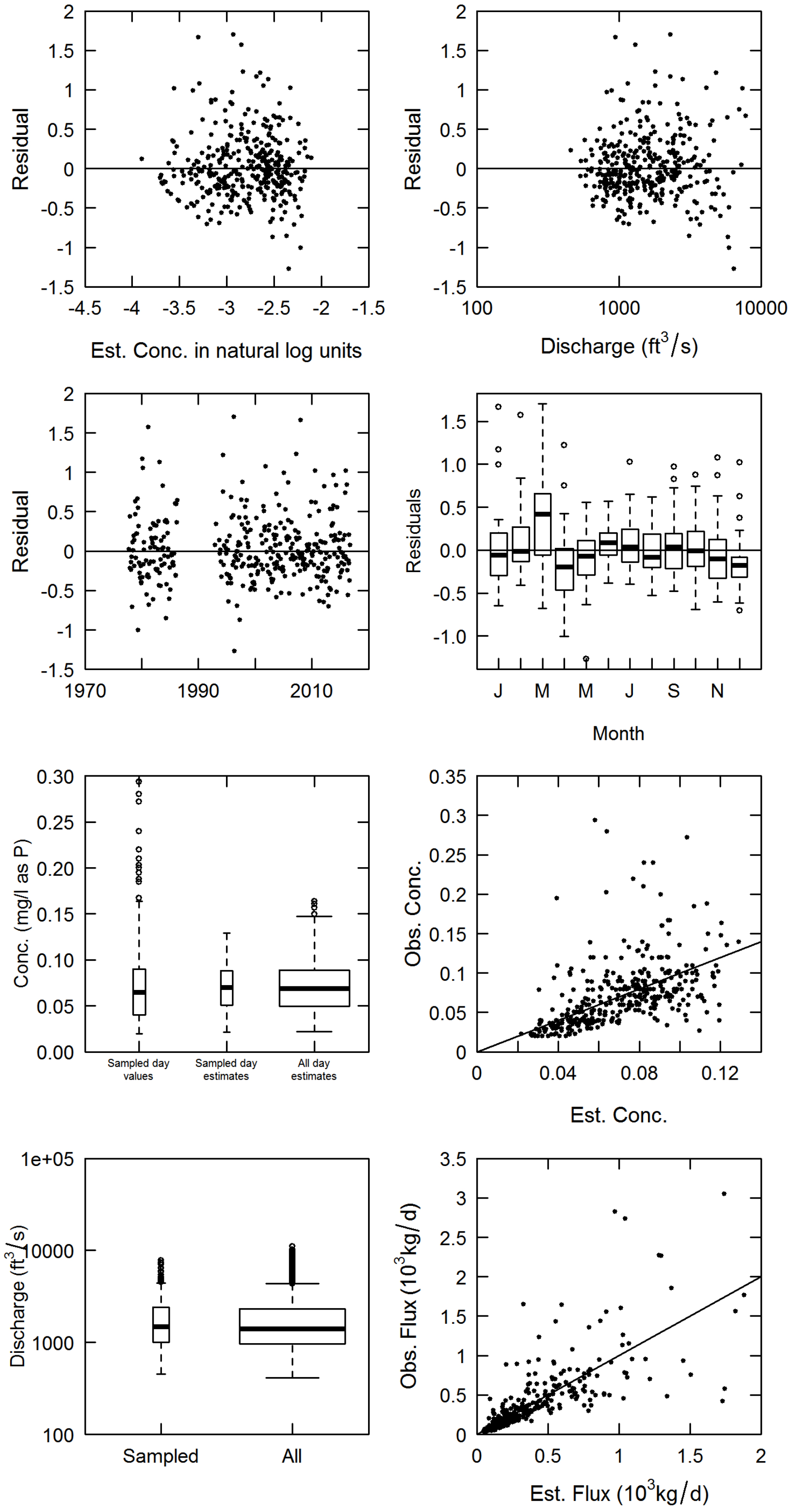

Wolf at New London

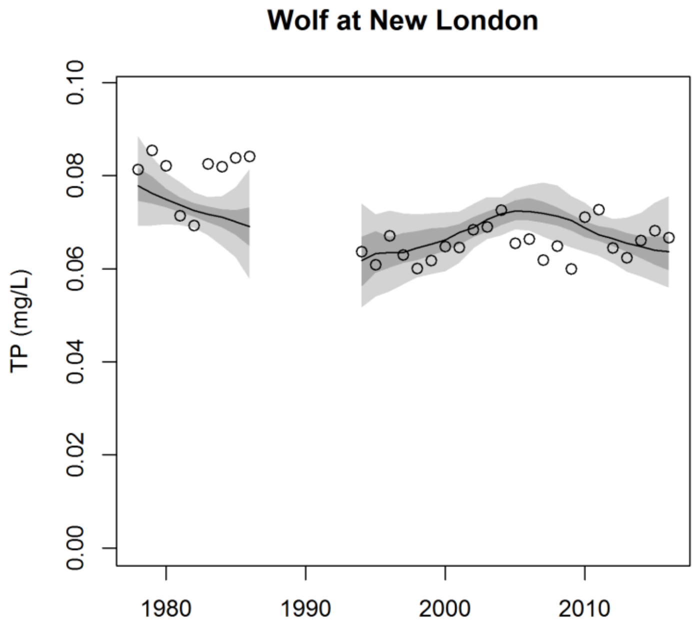


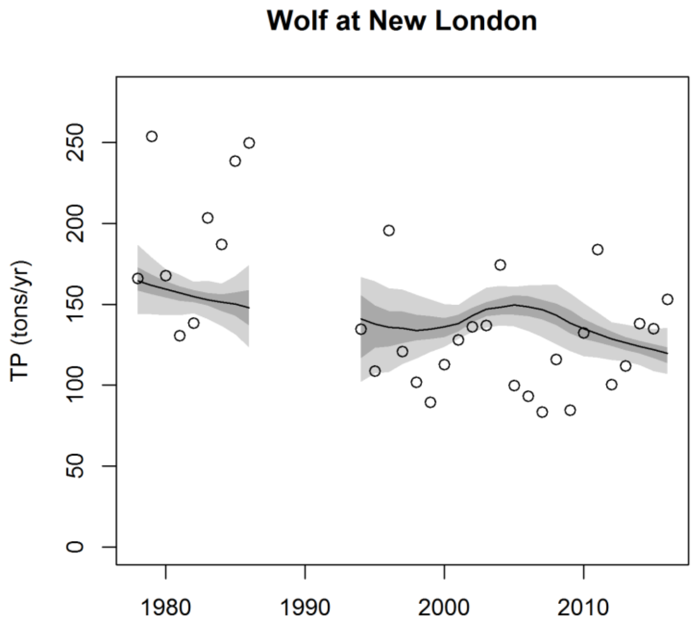
Total Phosphorus

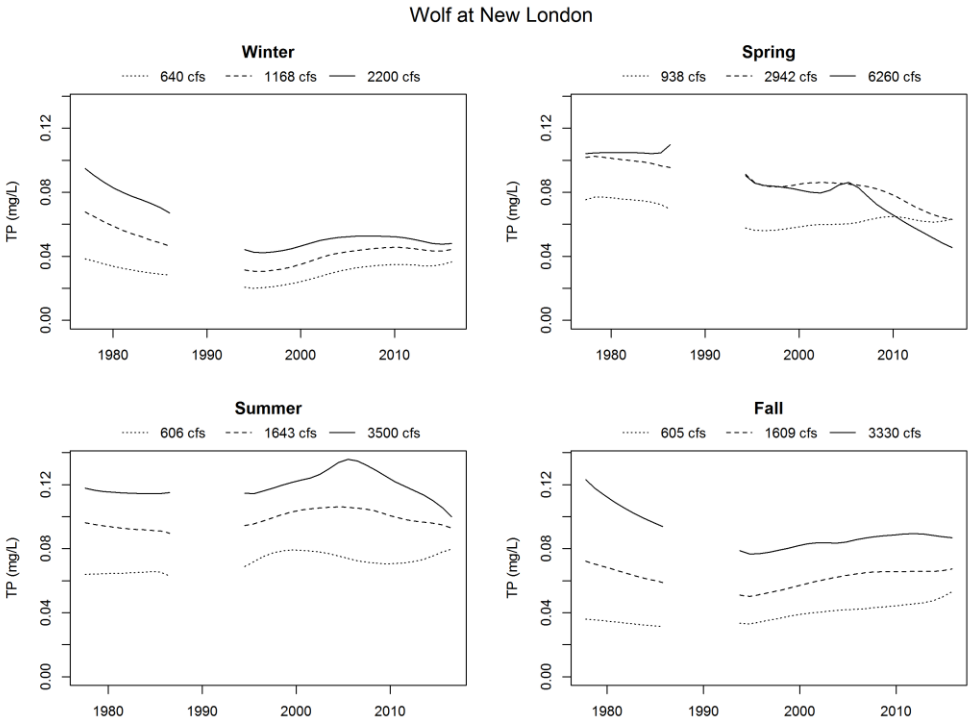


Wolf at New London, Total Phosphorus
Model is WRTDS Flux Bias Statistic-0.0405

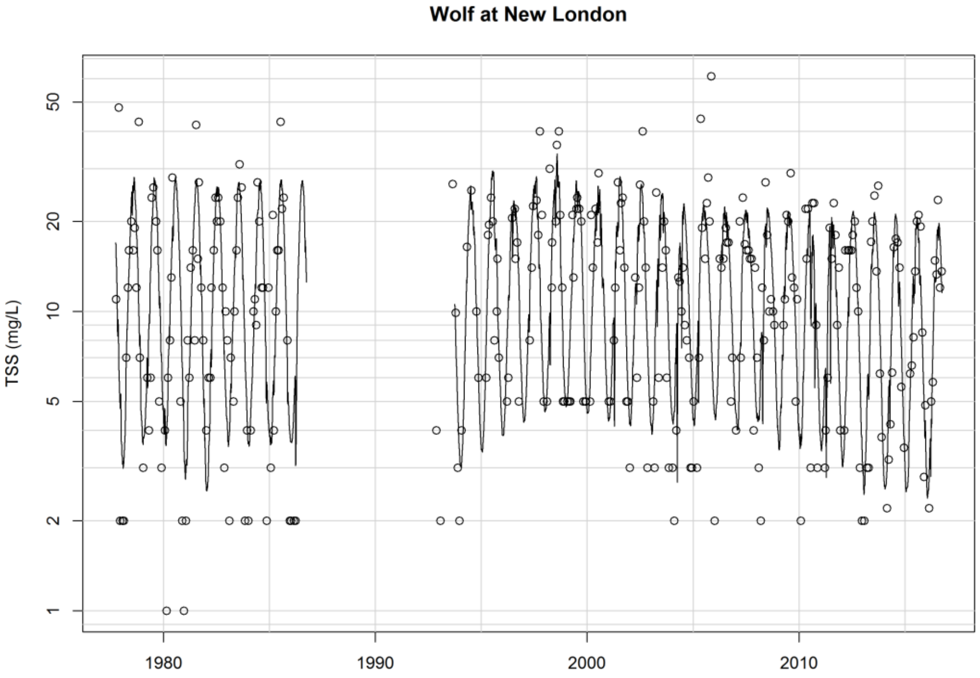


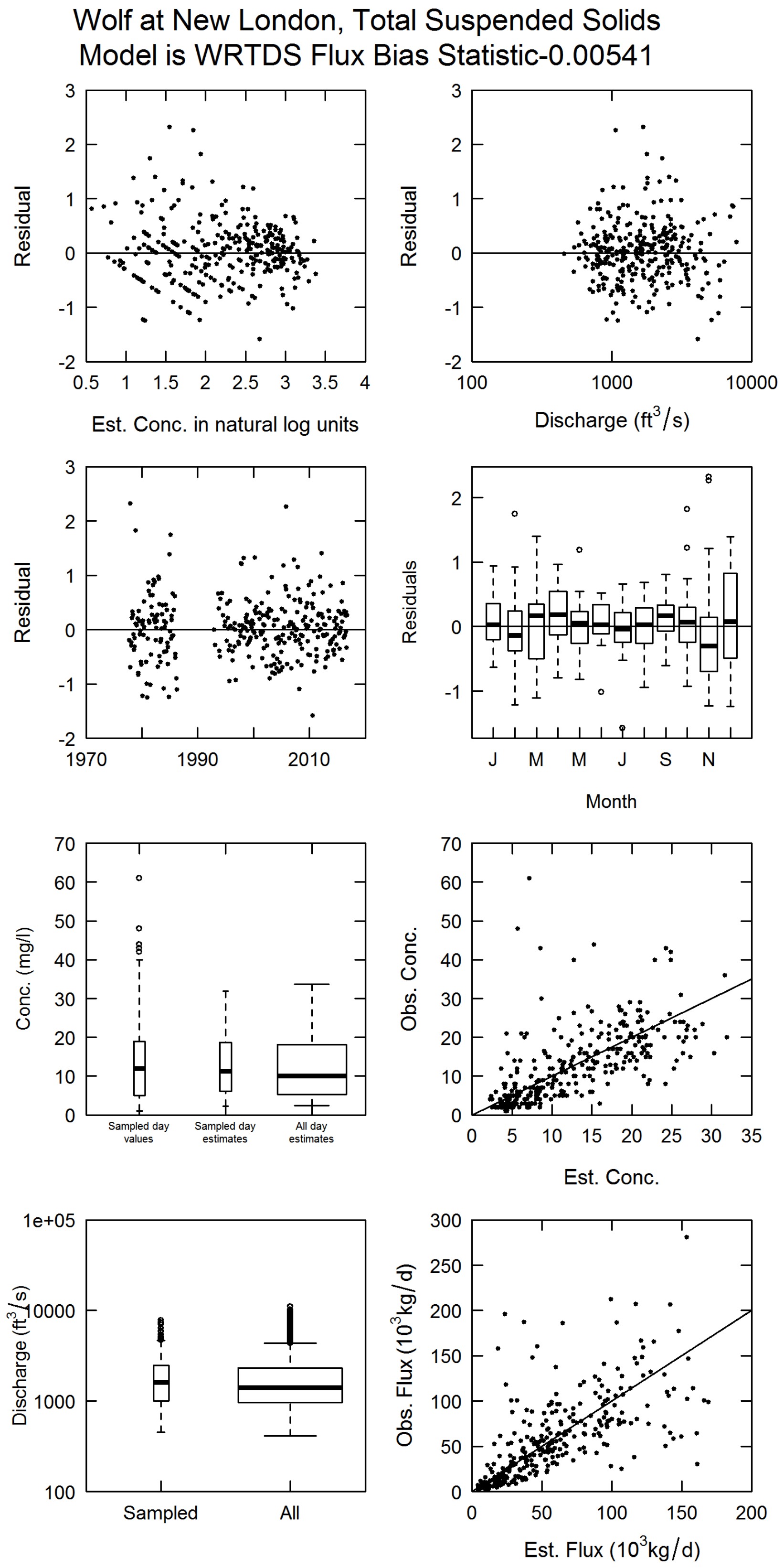

Wolf at New London
TP (mg/L)
0.00
0.02
0.04
0.06
0.08
0.10
1980
1990
2000
2010


Wolf at New London
TP (tons/yr)
0
50
100
150
200
250
1980
1990
2000
2010

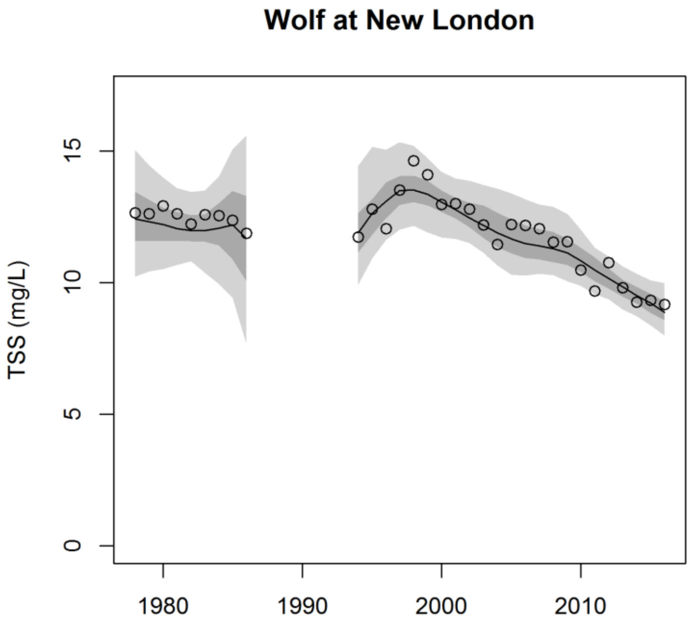


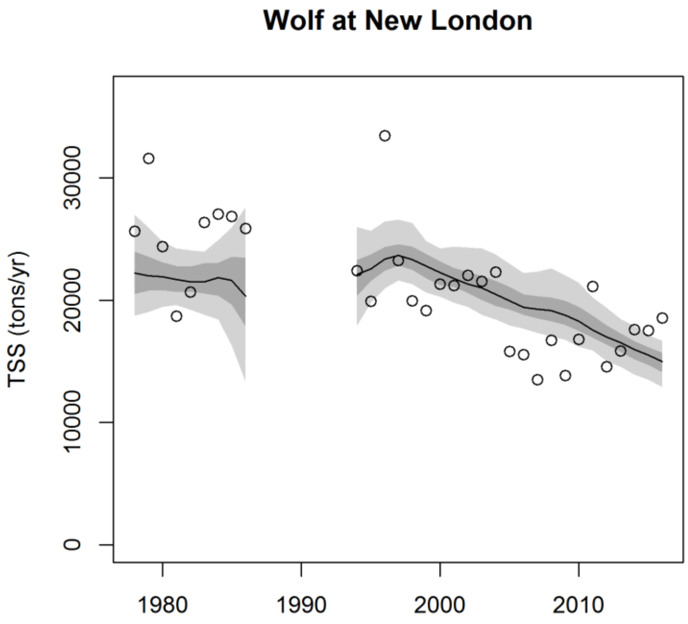
Total Suspended Solids

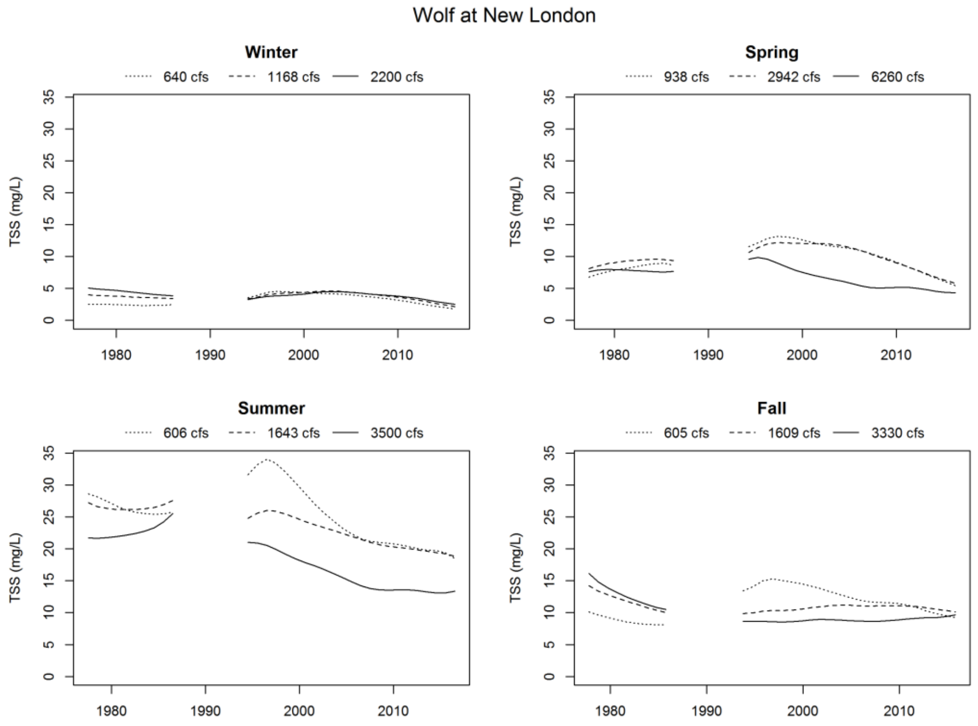




Wolf at New London, Total Suspended Solids
Model is WRTDS Flux Bias Statistic-0.00541

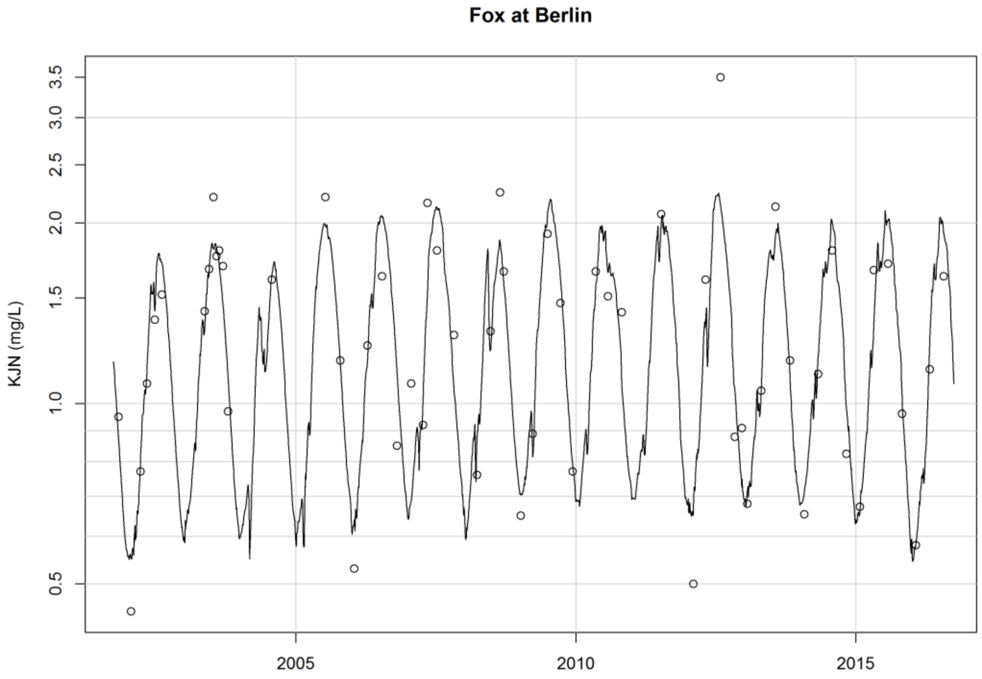

Wolf at New London
TSS (mg/L)
0
5
10
15
1980
1990
2000
2010

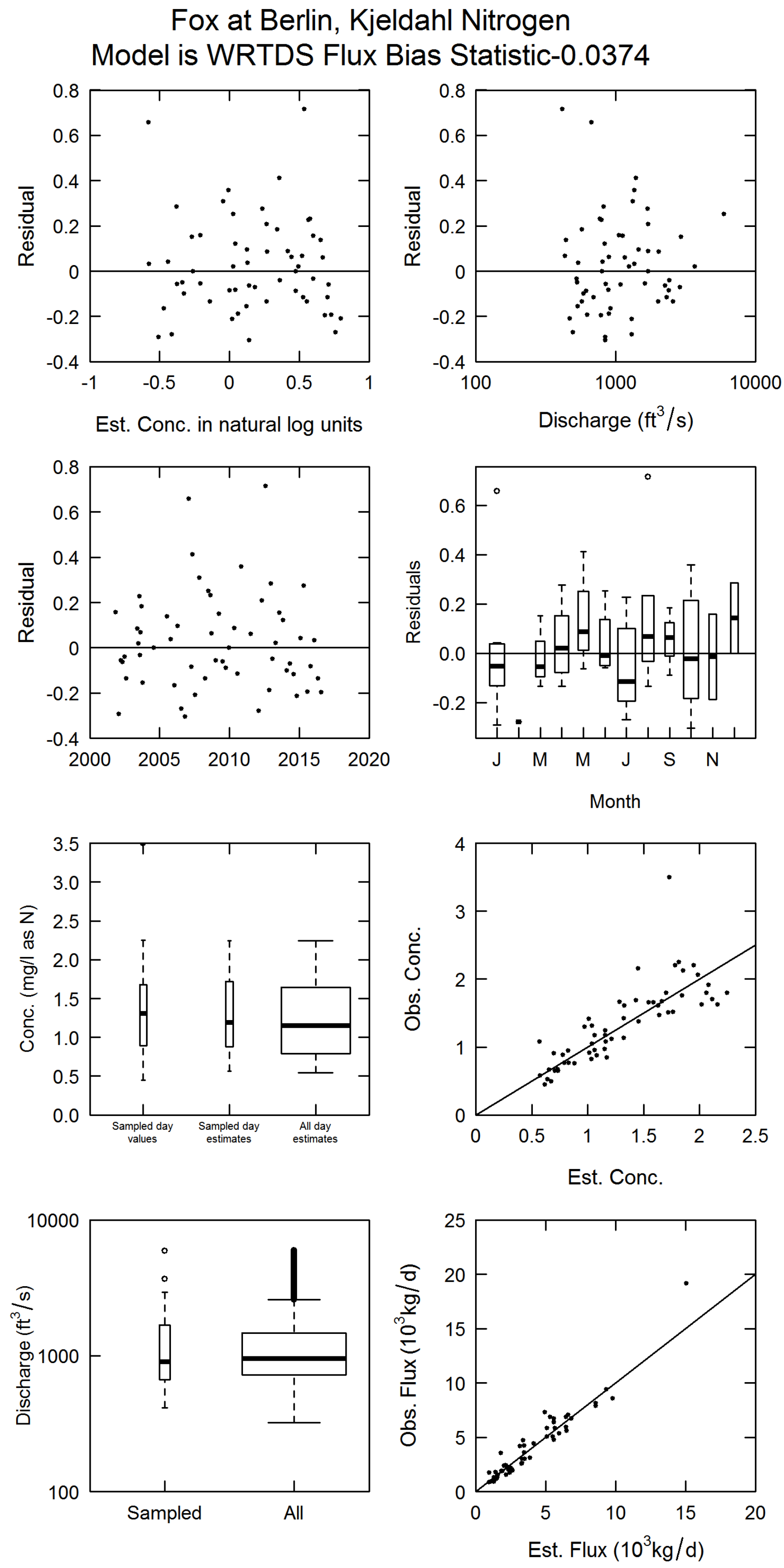

Wolf at New London
TSS (tons/yr)
0
10000
20000
30000
1980
1990
2000
2010

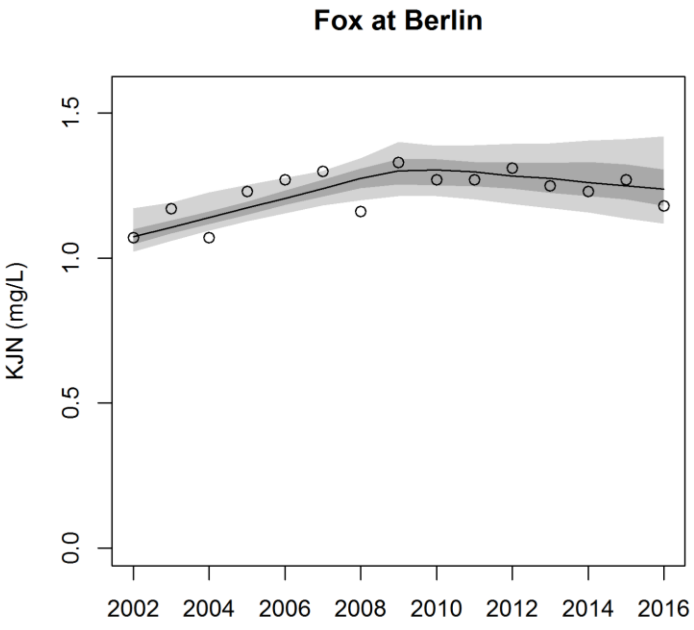


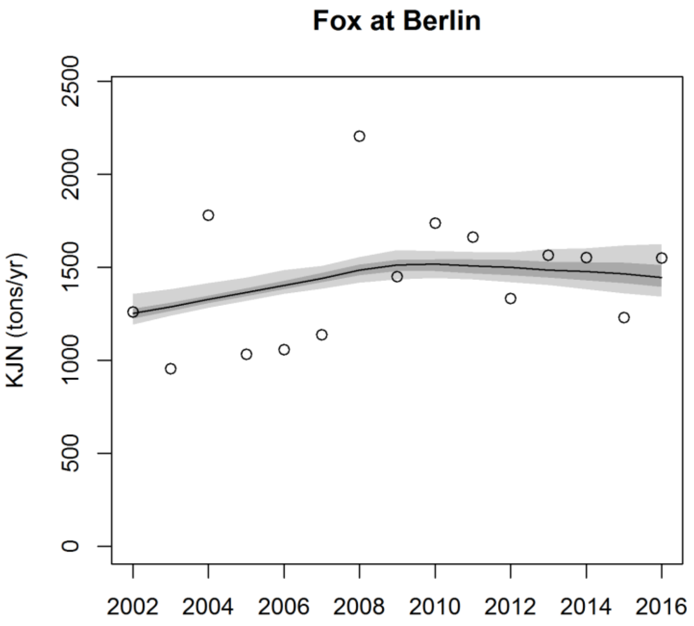
FOX RIVER AT BERLIN (243020)

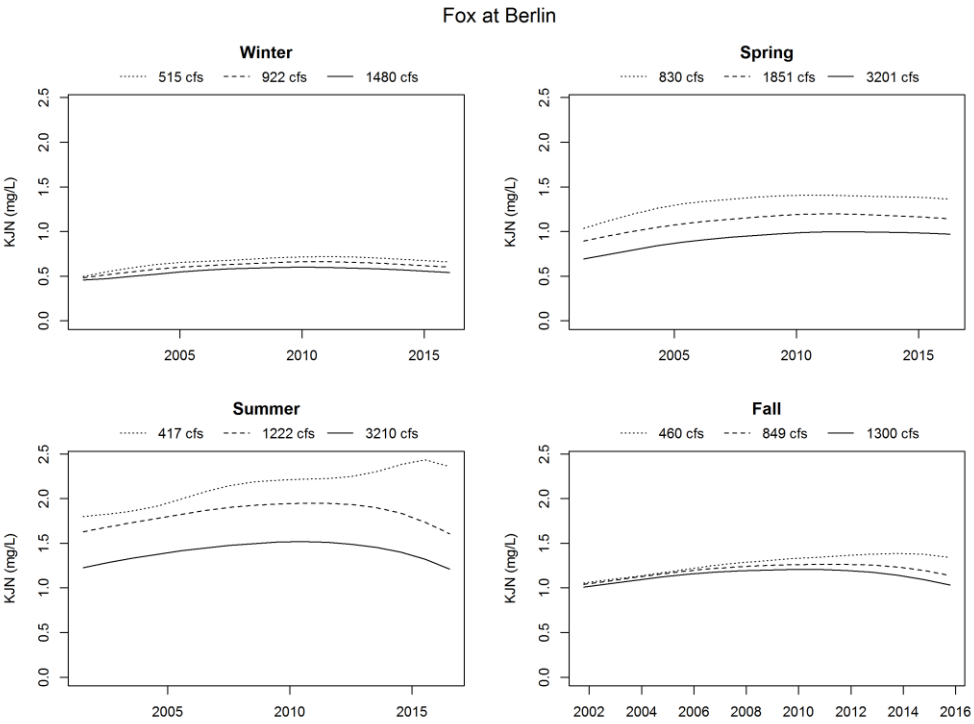
Total Kjeldahl Nitrogen



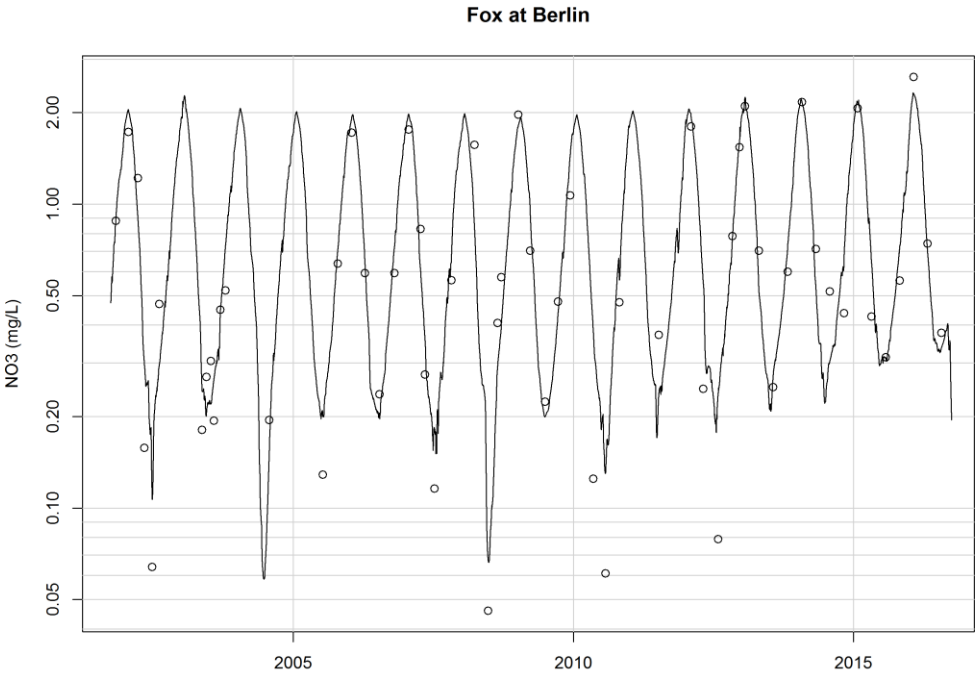

Fox at Berlin, Kjeldahl Nitrogen
Model is WRTDS Flux Bias Statistic-0.0374
Residual
0.8
0.6
0.4
0.2
0
-0.2
-0.4
-1
-0.5
0
0.5
1
Est. Conc. in natural log units
100
1000
10000
Discharge (ft3/s)
2000
2005
2010
2015
2020
Residuals
0.6
0.4
0.2
0.0
-0.2
J
M
M
J
S
N
Month
Conc. (mg/l as N)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
4
3
2
1
0
0.5
1
1.5
2
2.5
Est. Conc.
Discharge (ft3/s)
10000
1000
100
Sampled
All
Obs. Flux (103kg/d)
25
20
15
10
5
0
5
10
15
20
Est. Flux (103kg/d)

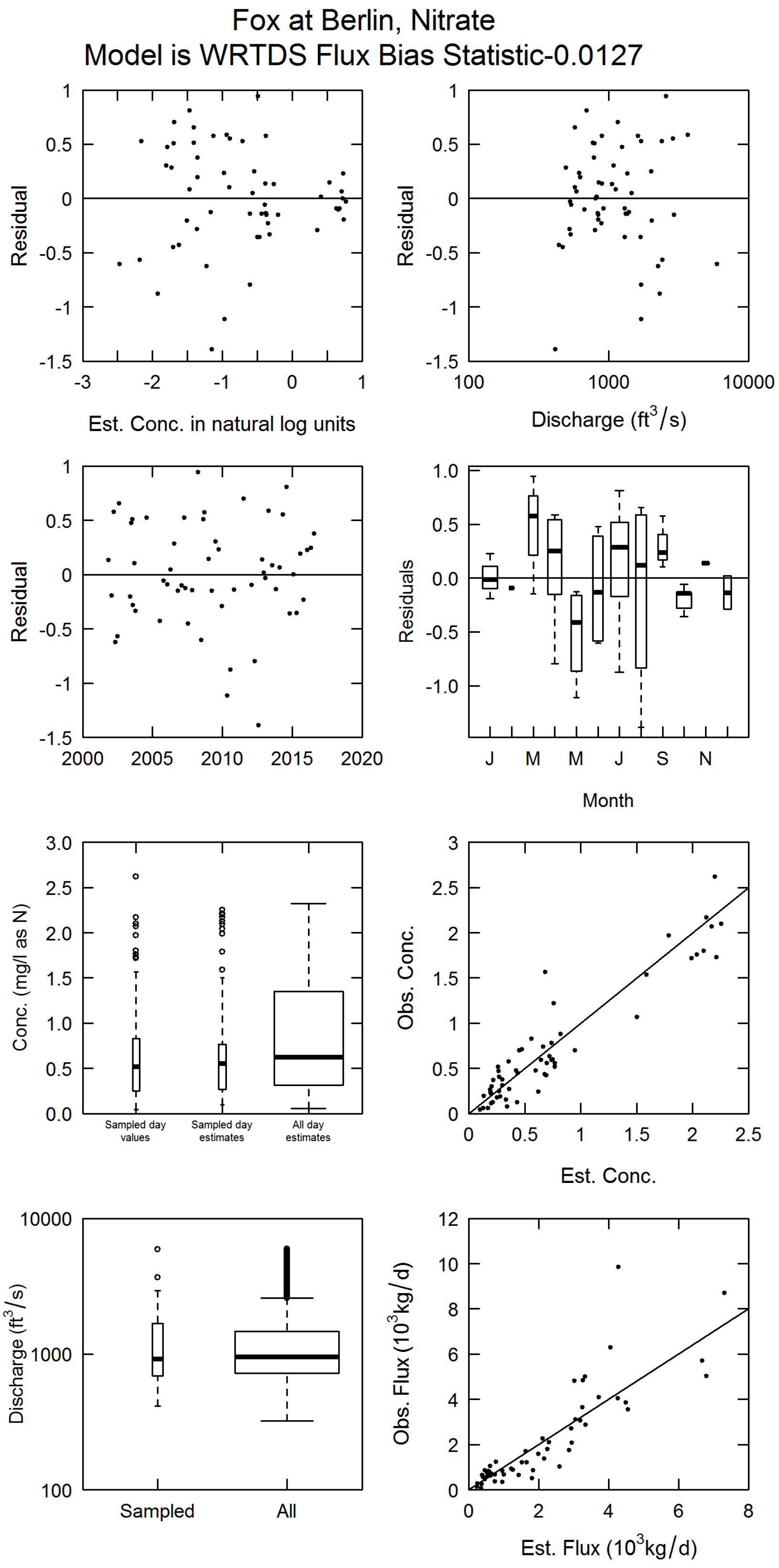

Fox at Berlin
KJN (mg/L)
0.0
0.5
1.0
1.5
2002
2004
2006
2008
2010
2012
2014
2016

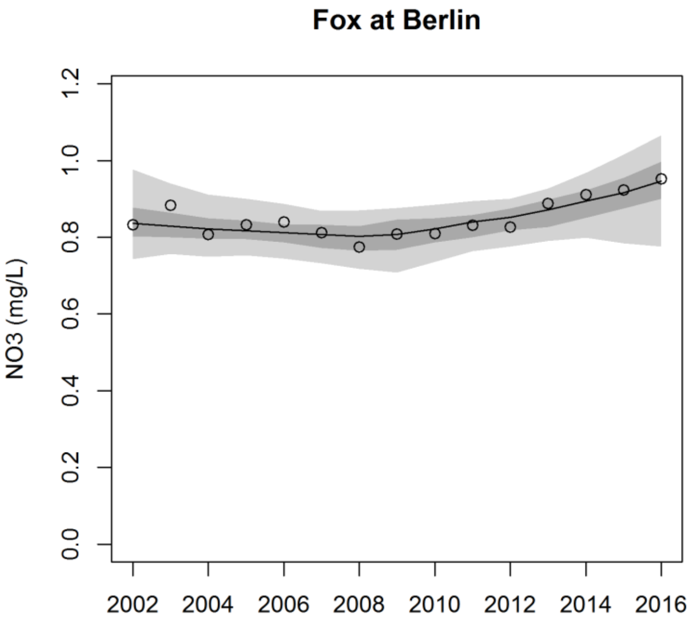

Fox at Berlin
KJN (tons/yr)
0
500
1000
1500
2000
2500
2002
2004
2006
2008
2010
2012
2014
2016

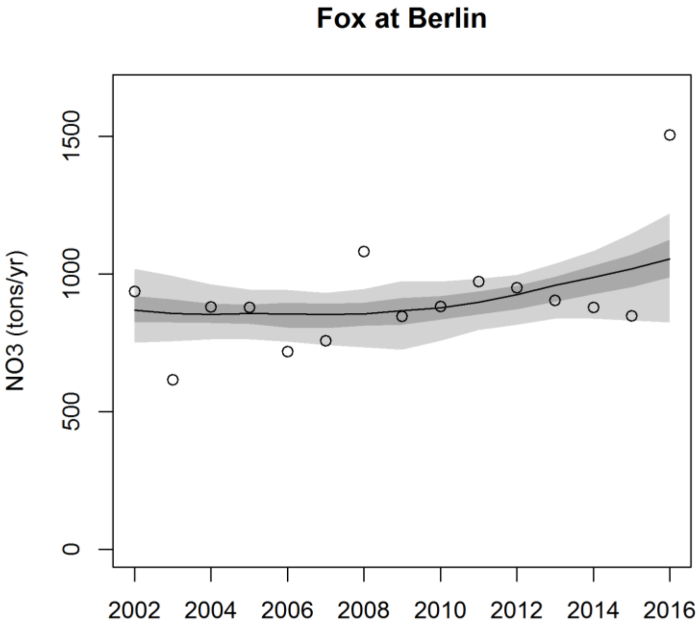

Fox at Berlin
Winter
515 cfs
922 cfs
1480 cfs
Spring
830 cfs
1851 cfs
3201 cfs
Summer
417 cfs
1222 cfs
3210 cfs
Fall
460 cfs
849 cfs
1300 cfs
KJN (mg/L)
0.0
0.5
1.0
1.5
2.0
2.5
2002
2004
2005
2006
2008
2010
2012
2014
2015
2016

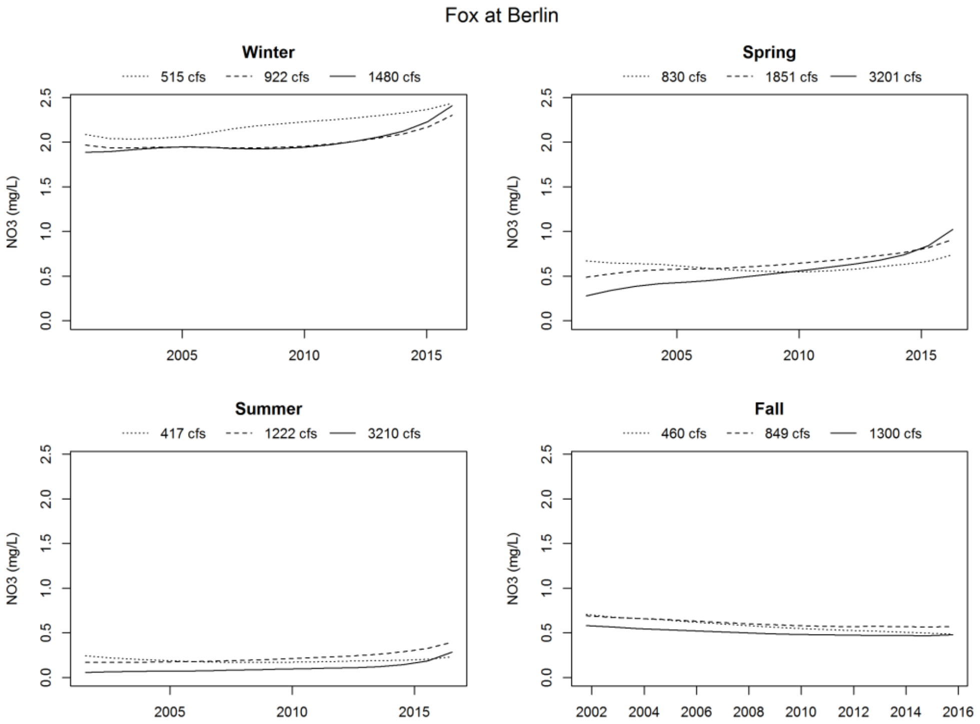
Nitrate


Fox at Berlin
NO3 (mg/L)
0.05
0.10
0.20
0.50
1.00
2.00
2005
2010
2015

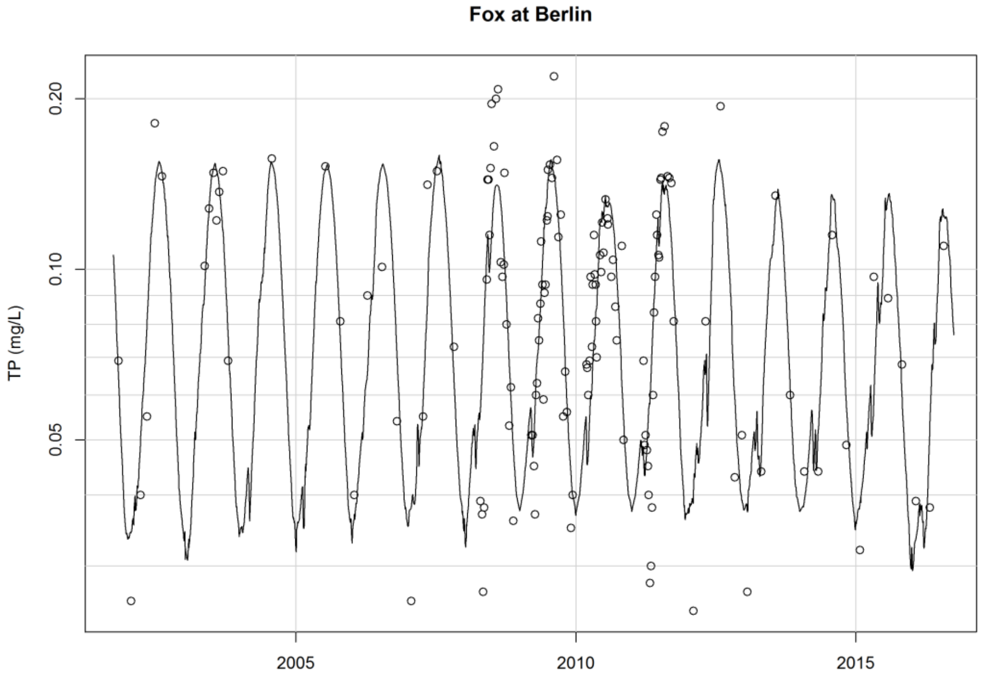

Fox at Berlin, Nitrate
Model is WRTDS Flux Bias Statistic-0.0127
Residual
1
0.5
0
-0.5
-1
-1.5
-3
-2
-1
0
1
Est. Conc. in natural log units
100
1000
10000
Discharge (ft^3/s)
2000
2005
2010
2015
2020
Residuals
1.0
0.5
0.0
-0.5
-1.0
J
M
M
J
S
N
Month
Conc. (mg/l as N)
3.0
2.5
2.0
1.5
1.0
0.5
0.0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
3
2.5
2
1.5
1
0.5
0
Est. Conc.
Discharge (ft^3/s)
10000
1000
100
Sampled
All
Obs. Flux (10^3kg/d)
12
10
8
6
4
2
0
Est. Flux (10^3kg/d)

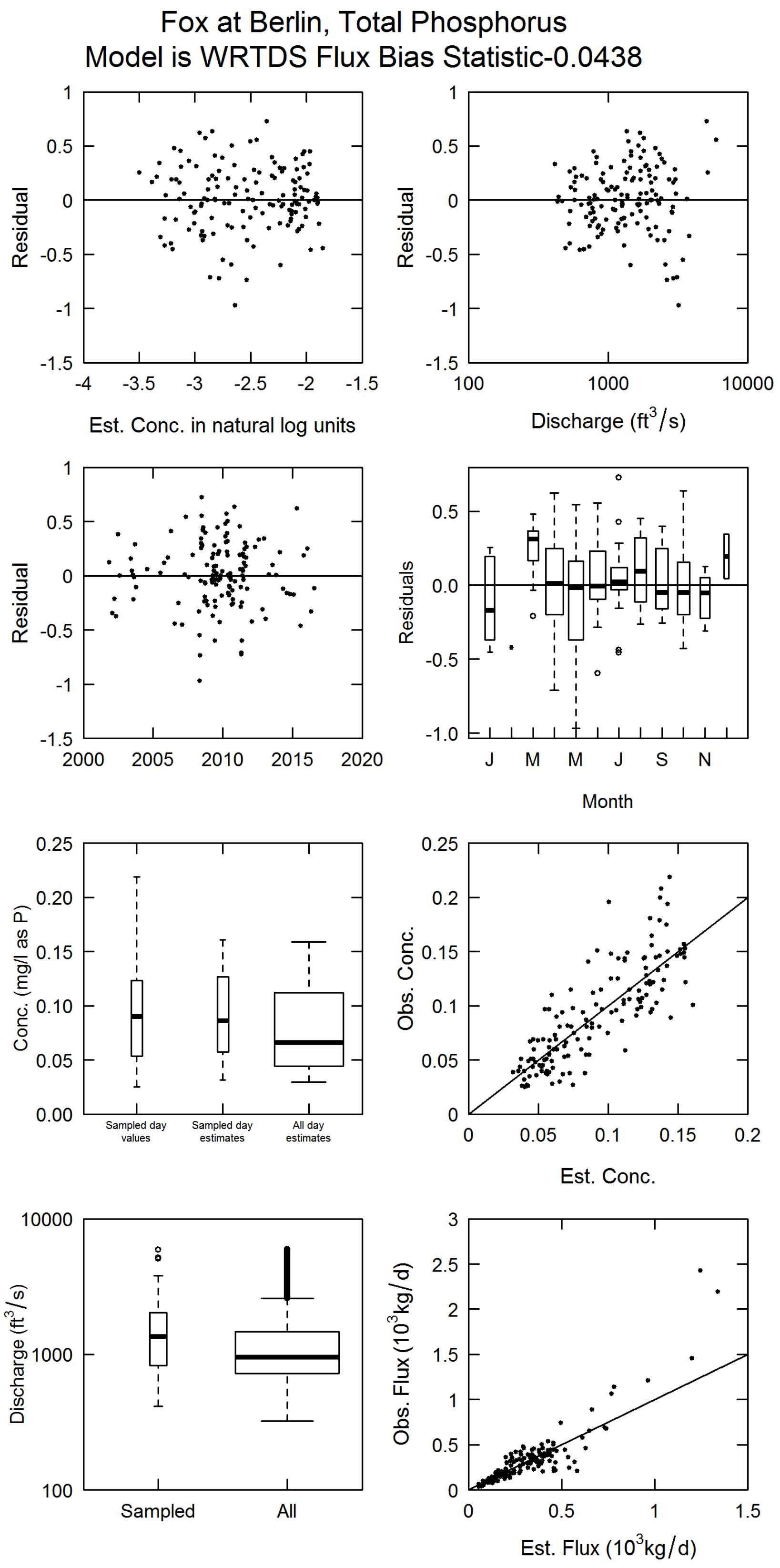

Fox at Berlin
NO3 (mg/L)
0.0
0.2
0.4
0.6
0.8
1.0
1.2
2002
2004
2006
2008
2010
2012
2014
2016

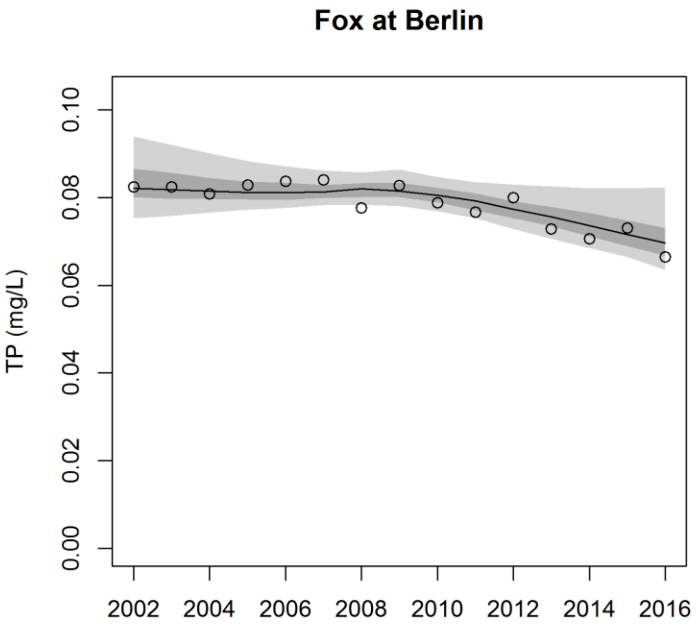

Fox at Berlin
NO3 (tons/yr)
0
500
1000
1500
2002
2004
2006
2008
2010
2012
2014
2016

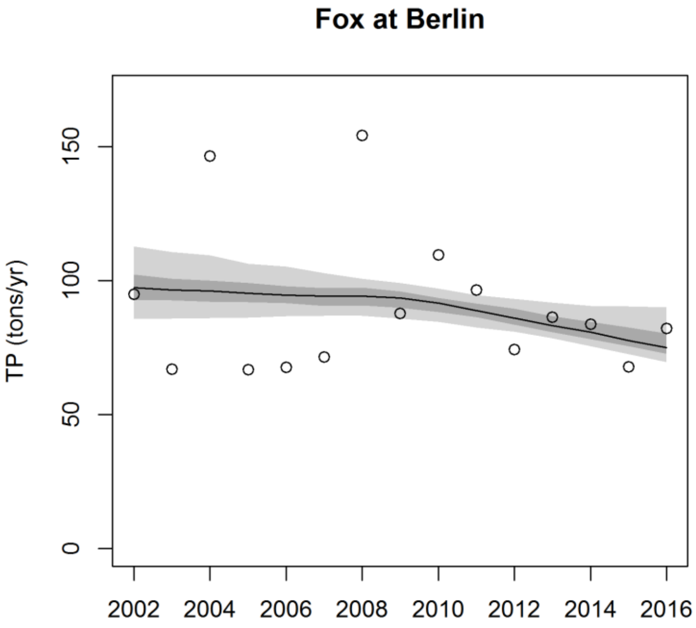


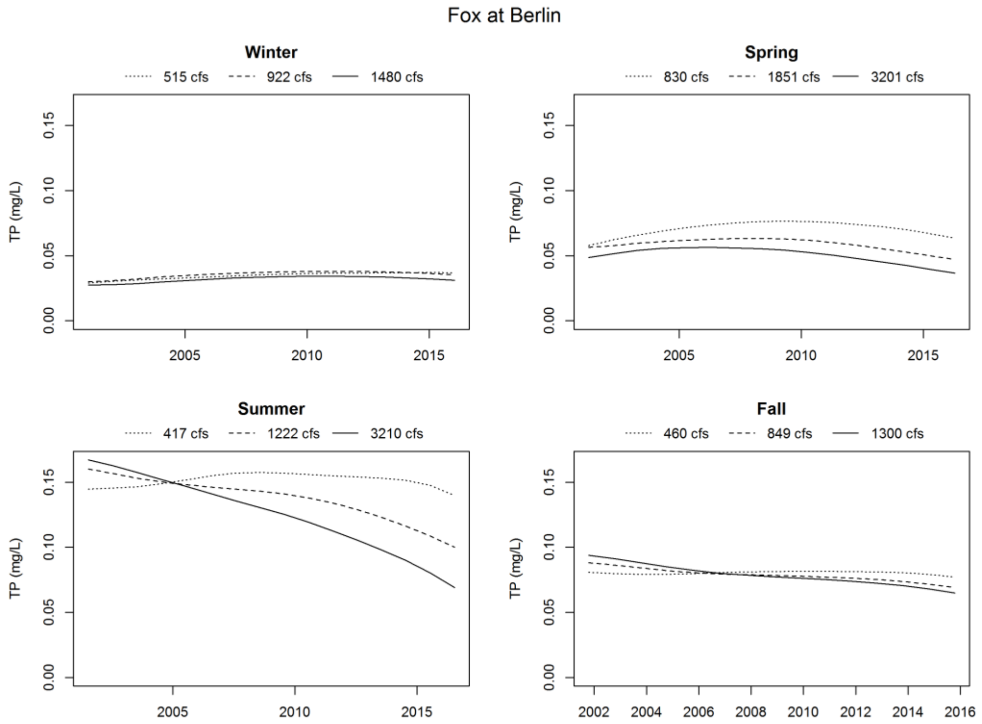
Total Phosphorus



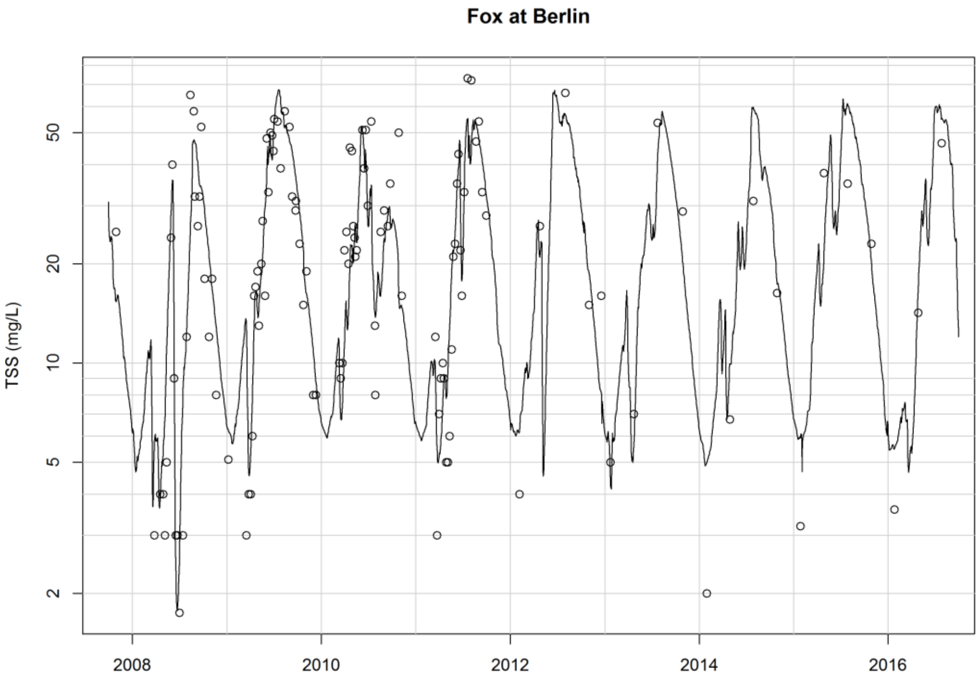

Fox at Berlin, Total Phosphorus
Model is WRTDS Flux Bias Statistic-0.0438
Residual
1
0.5
0
-0.5
-1
-1.5
-4
-3.5
-3
-2.5
-2
-1.5
Est. Conc. in natural log units
Residual
100
1000
10000
Discharge (ft^3/s)
Residual
2000
2005
2010
2015
2020
Residuals
0.5
0.0
-0.5
-1.0
J
M
M
J
S
N
Month
Conc. (mg/l as P)
0.25
0.20
0.15
0.10
0.05
0.00
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
0.25
0.2
0.15
0.1
0.05
0
0
0.05
0.1
0.15
0.2
Est. Conc.
Discharge (ft^3/s)
10000
1000
100
Sampled
All
Obs. Flux (10^{3}kg/d)
3
2.5
2
1.5
1
0.5
0
0
0.5
1
1.5
Est. Flux (10^{3}kg/d)

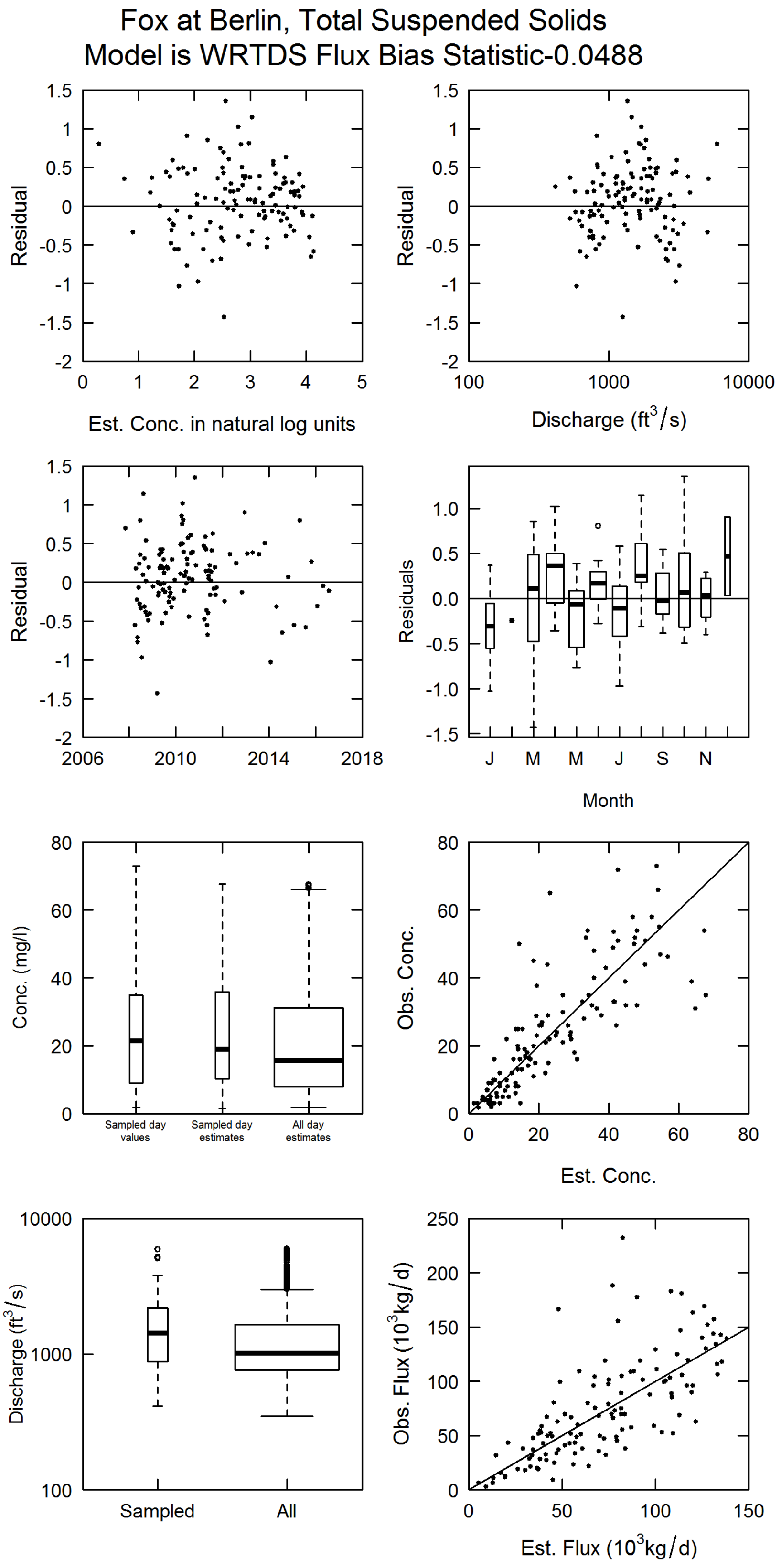

Fox at Berlin
TP (mg/L)
0.00
0.02
0.04
0.06
0.08
0.10
2002
2004
2006
2008
2010
2012
2014
2016

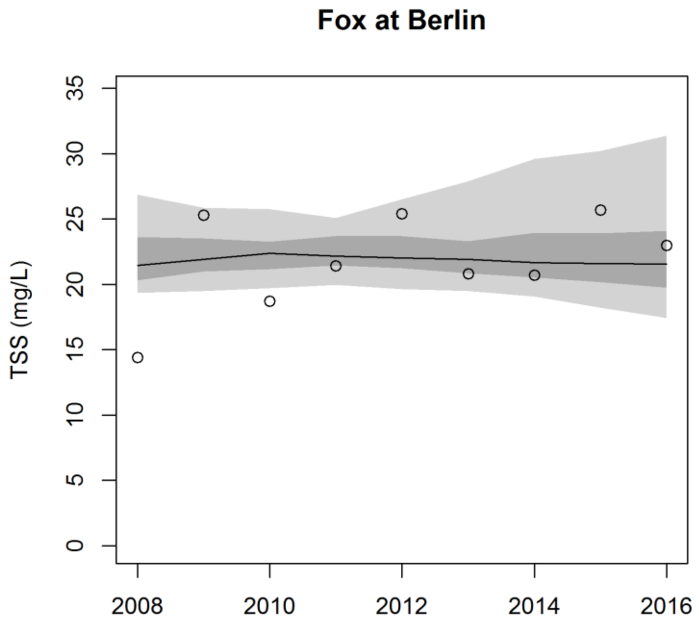

Fox at Berlin
TP (tons/yr)
0
50
100
150
2002
2004
2006
2008
2010
2012
2014
2016

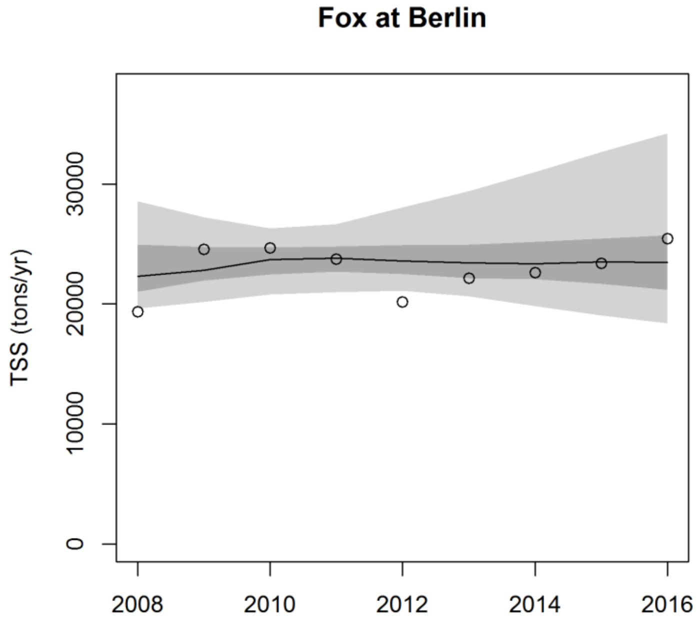


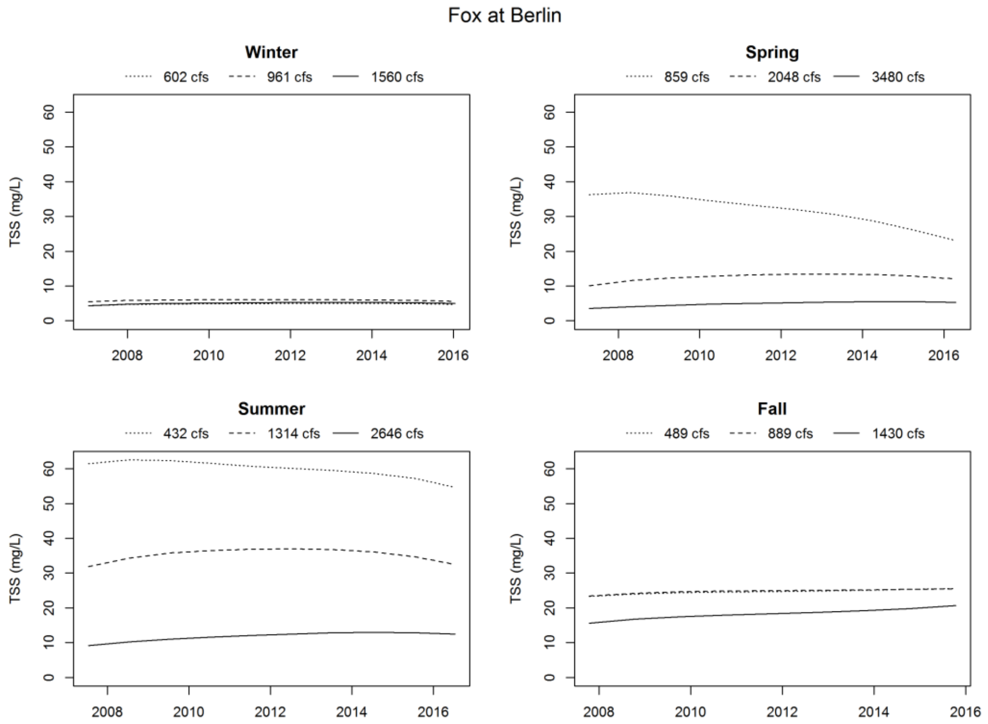
Total Suspended Solids




Fox at Berlin, Total Suspended Solids
Model is WRTDS Flux Bias Statistic-0.0488
Residual
1.5
1
0.5
0
-0.5
-1
-1.5
-2
0
1
2
3
4
5
Est. Conc. in natural log units
100
1000
10000
Discharge (ft3/s)
2006
2010
2014
2018
Residuals
1.0
0.5
0.0
-0.5
-1.0
-1.5
J
M
M
J
S
N
Month
Conc. (mg/l)
80
60
40
20
0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
Est. Conc.
Discharge (ft3/s)
10000
1000
100
Sampled
All
Obs. Flux (103kg/d)
250
200
150
100
50
0
Est. Flux (103kg/d)

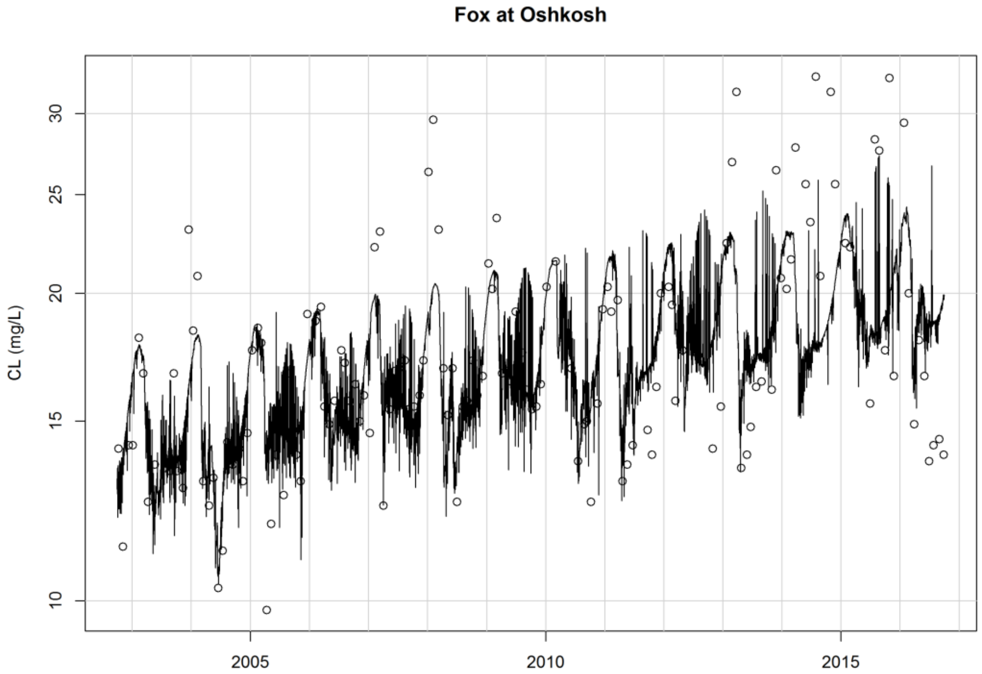

Fox at Berlin
TSS (mg/L)
0
5
10
15
20
25
30
35
2008
2010
2012
2014
2016

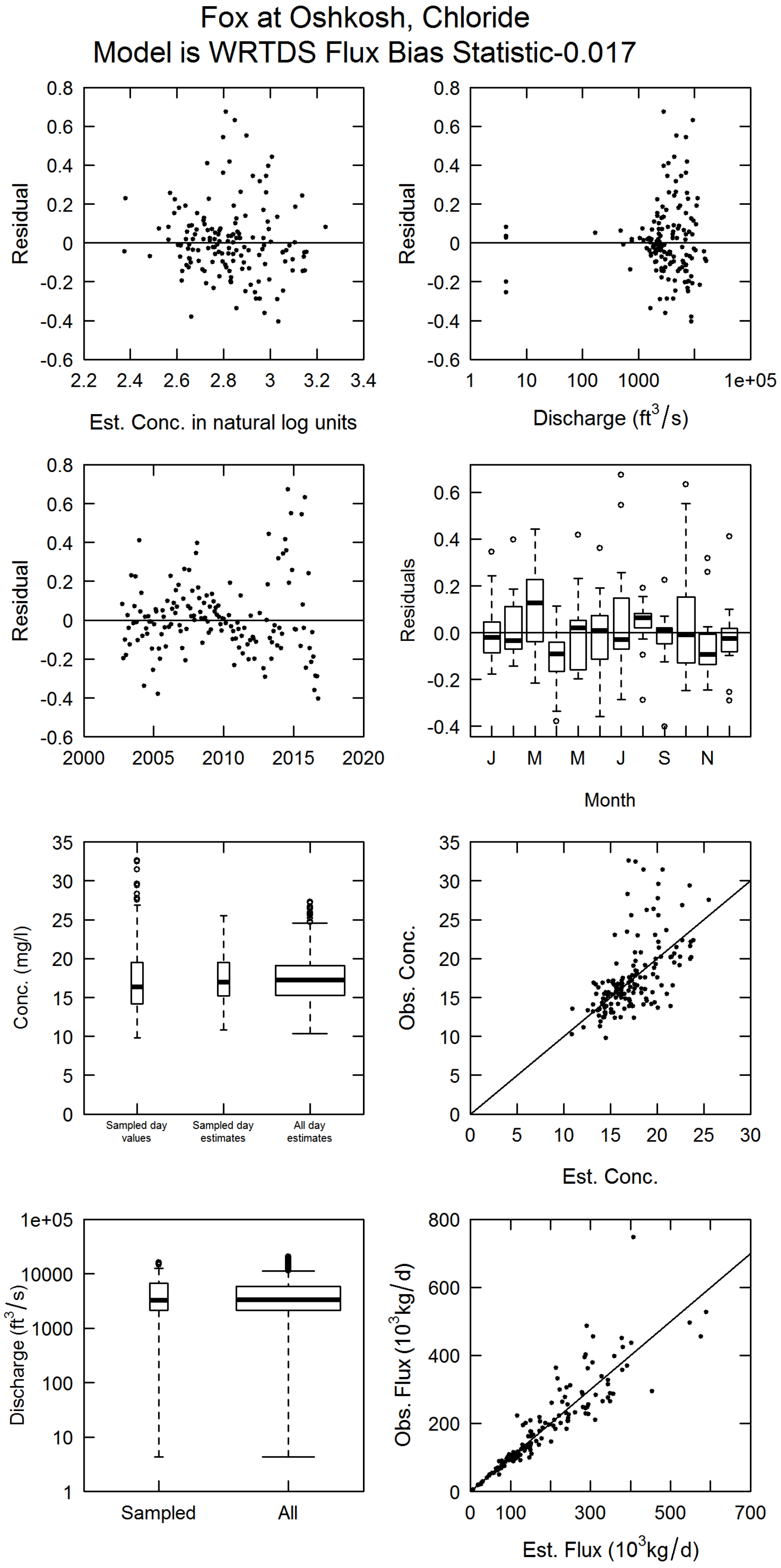

Fox at Berlin
TSS (tons/yr)
0
10000
20000
30000
2008
2010
2012
2014
2016

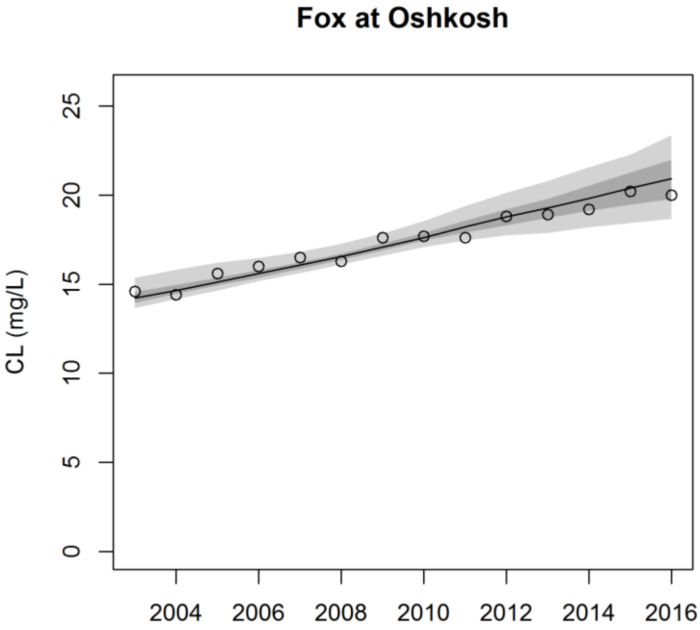


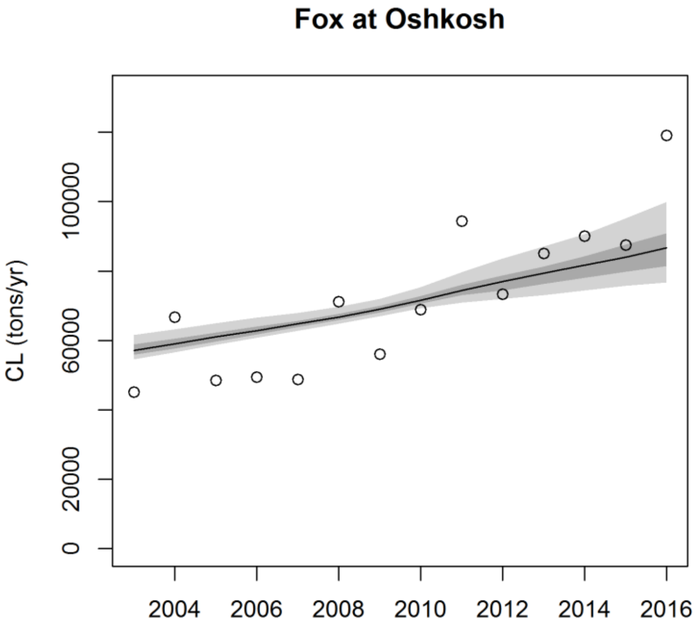
FOX RIVER AT OSHKOSH (713056)

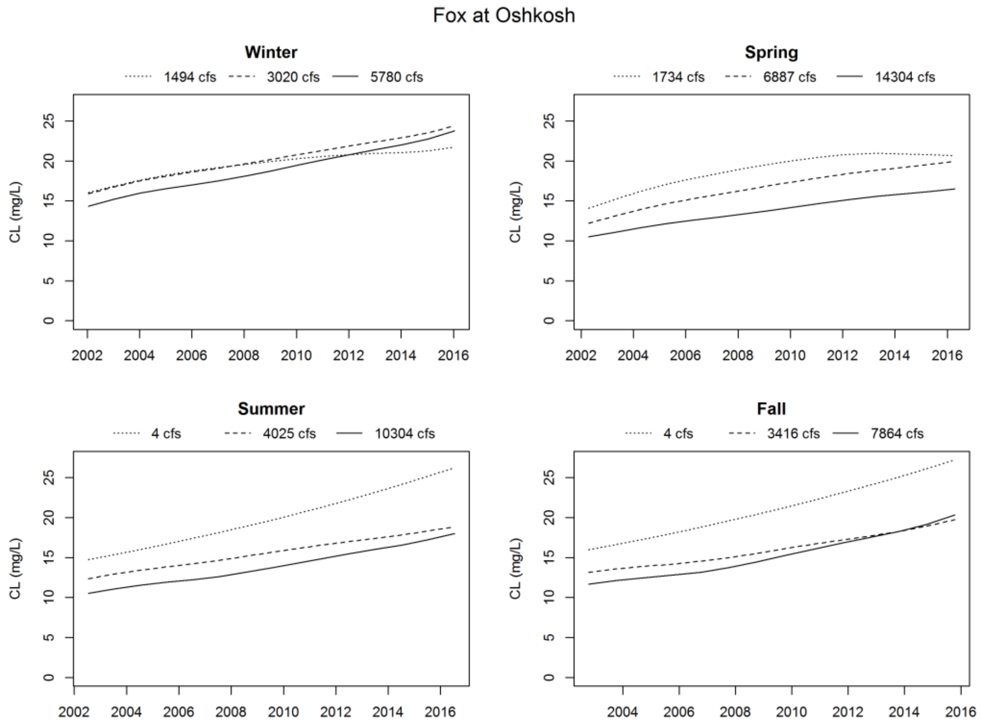
Chloride



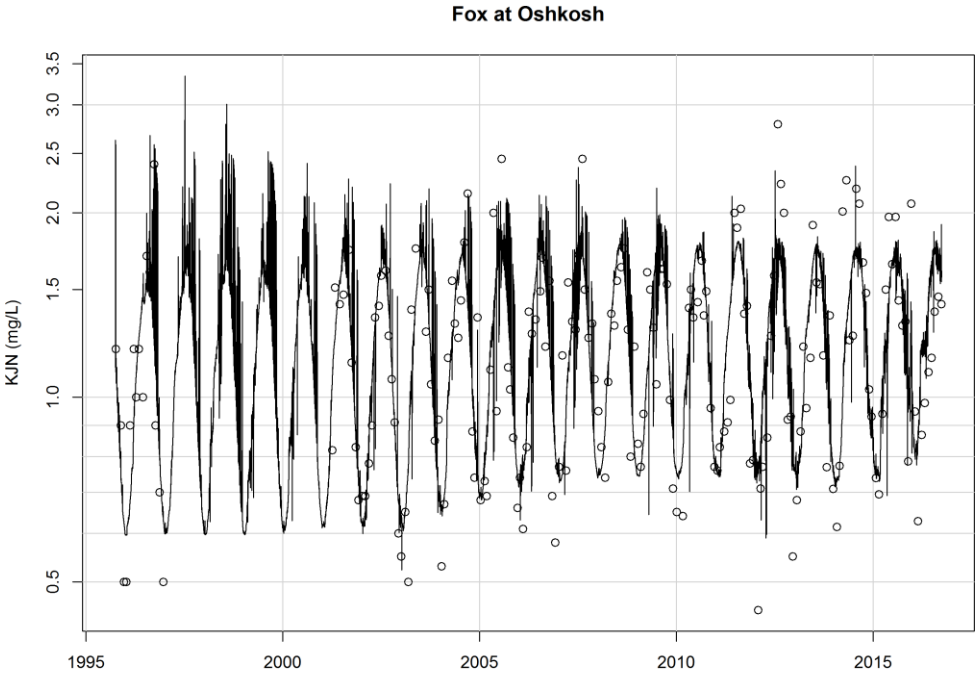

Fox at Oshkosh, Chloride
Model is WRTDS Flux Bias Statistic-0.017
Residual
0.8
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
2.2
2.4
2.6
2.8
3
3.2
3.4
Est. Conc. in natural log units
Residual
1
10
100
1000
1e+05
Discharge (ft3/s)
Residual
2000
2005
2010
2015
2020
Residuals
0.6
0.4
0.2
0.0
-0.2
-0.4
J
M
M
J
S
N
Month
Conc. (mg/l)
35
30
25
20
15
10
5
0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
0
5
10
15
20
25
30
Est. Conc.
Discharge (ft3/s)
1e+05
10000
1000
100
10
1
Sampled
All
Obs. Flux (103kg/d)
800
600
400
200
0
0
100
300
500
700
Est. Flux (103kg/d)

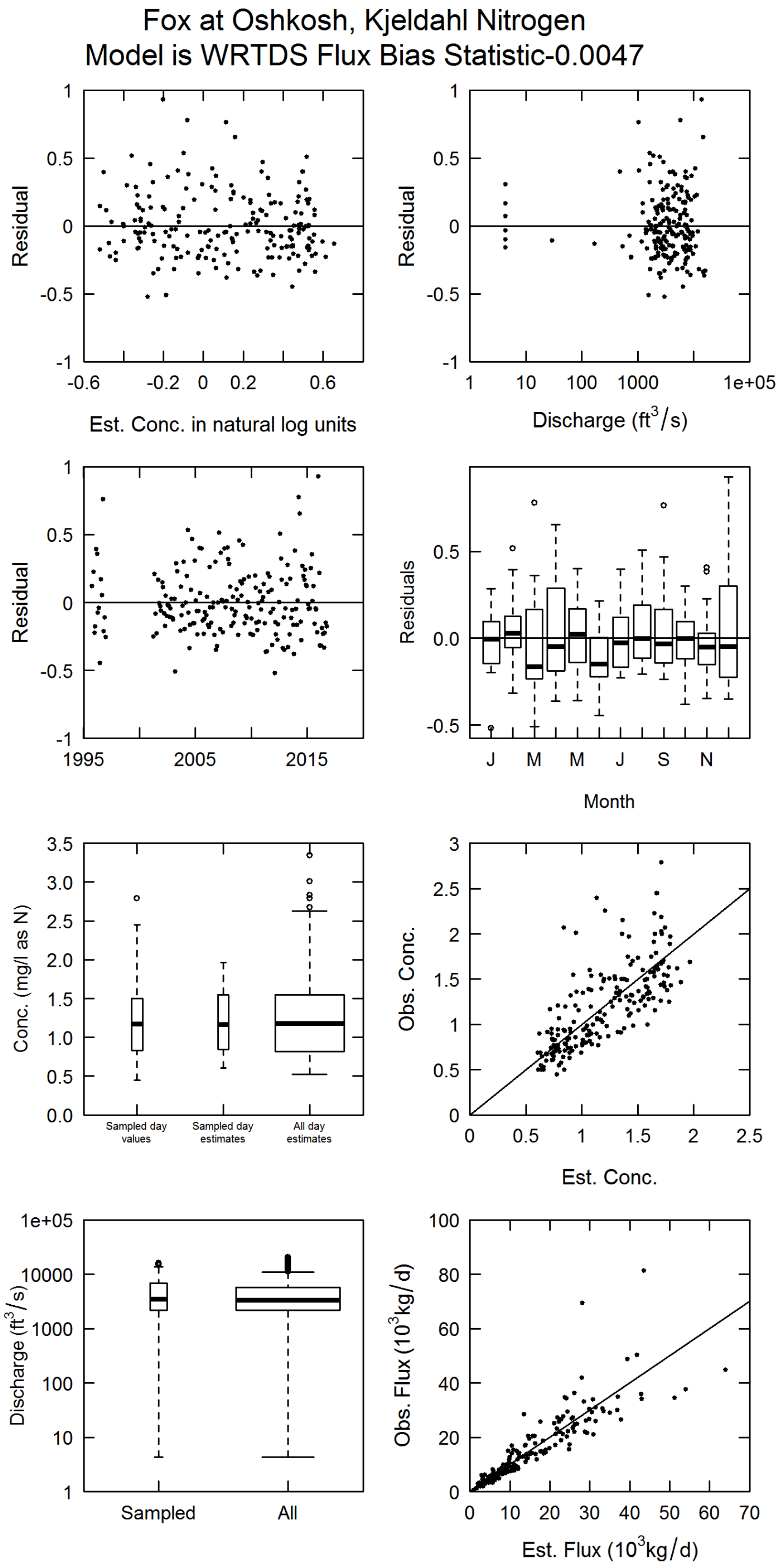

Fox at Oshkosh
CL (mg/L)
0
5
10
15
20
25
2004
2006
2008
2010
2012
2014
2016

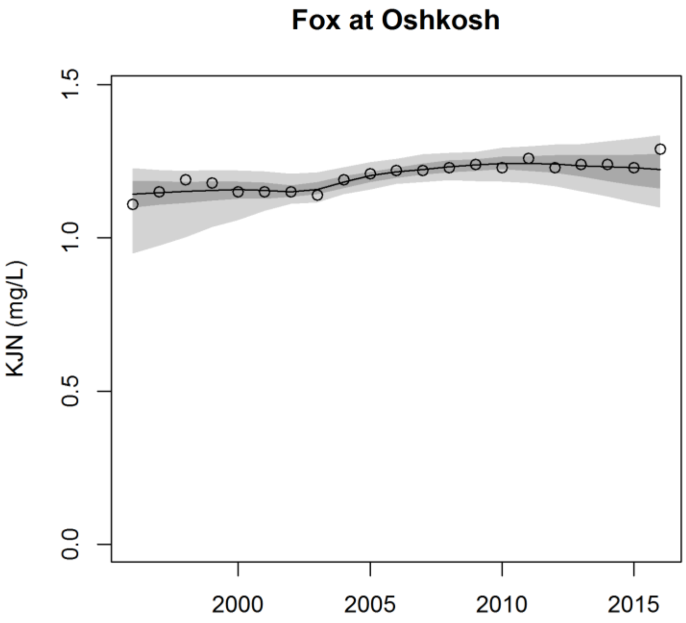

Fox at Oshkosh
CL (tons/yr)
0
20000
60000
100000
2004
2006
2008
2010
2012
2014
2016

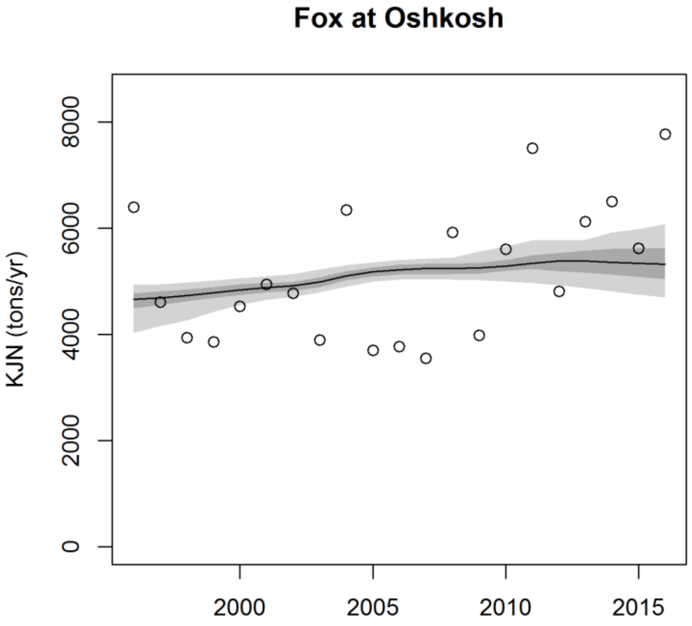


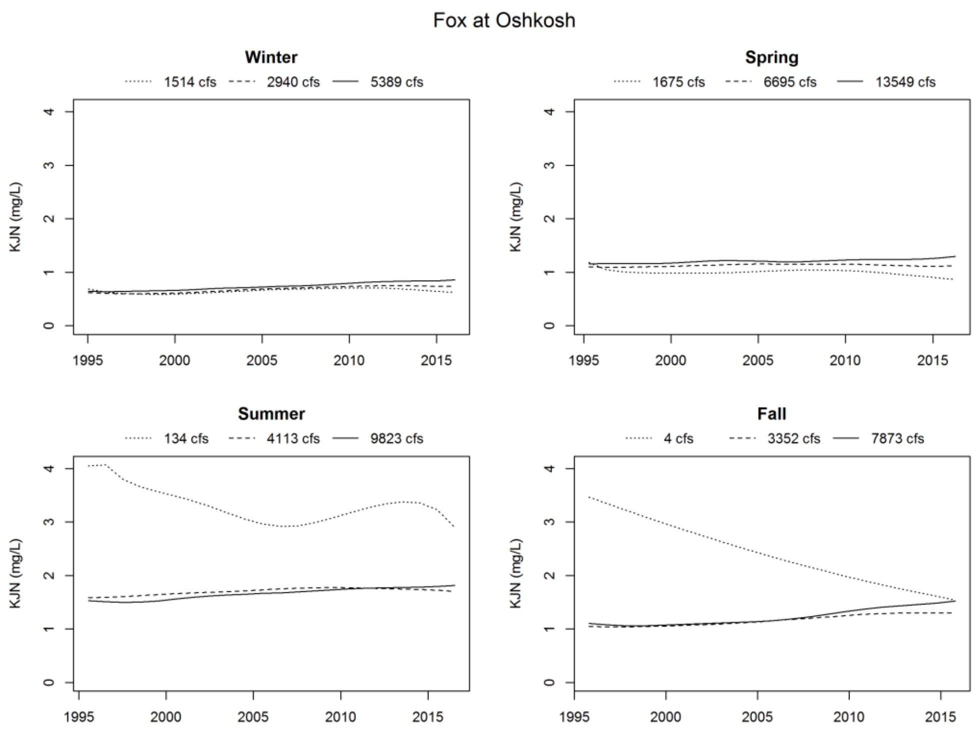
Total Kjeldahl Nitrogen



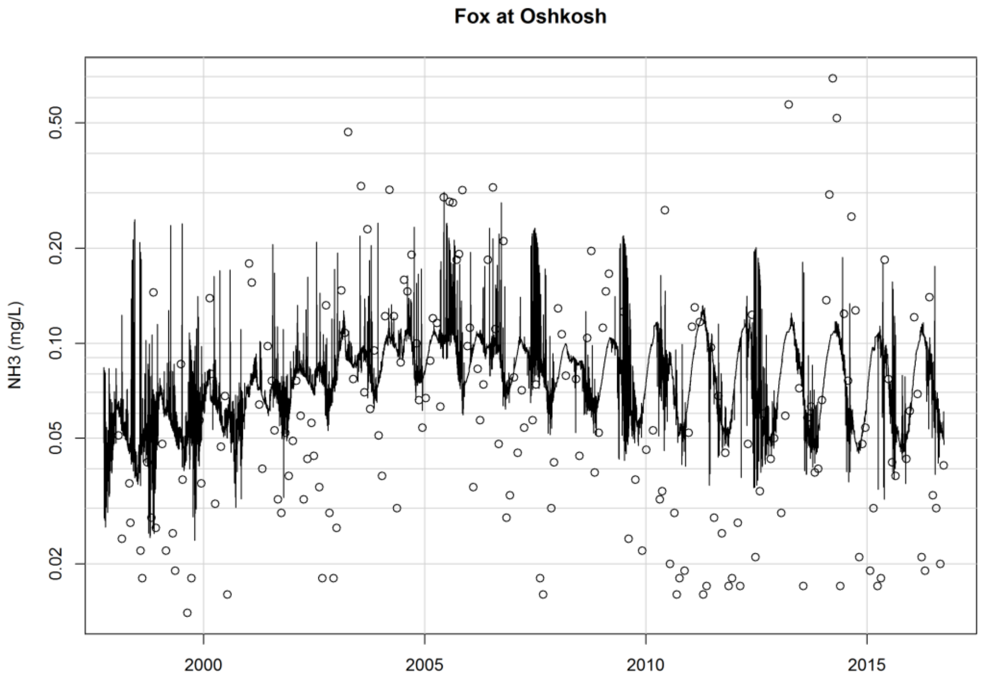

Fox at Oshkosh, Kjeldahl Nitrogen
Model is WRTDS Flux Bias Statistic-0.0047
Residual
1
0.5
0
-0.5
-1
-0.6
-0.2
0
0.2
0.6
Est. Conc. in natural log units
Residual
1
0.5
0
-0.5
-1
1
10
100
1000
1e+05
Discharge (ft^3/s)
Residual
1
0.5
0
-0.5
-1
1995
2005
2015
Residuals
0.5
0.0
-0.5
J
M
M
J
S
N
Month
Conc. (mg/l as N)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
3
2.5
2
1.5
1
0.5
0
0
0.5
1
1.5
2
2.5
Est. Conc.
Discharge (ft^3/s)
1e+05
10000
1000
100
10
1
Sampled
All
Obs. Flux (10^3kg/d)
100
80
60
40
20
0
0
10
20
30
40
50
60
70
Est. Flux (10^3kg/d)

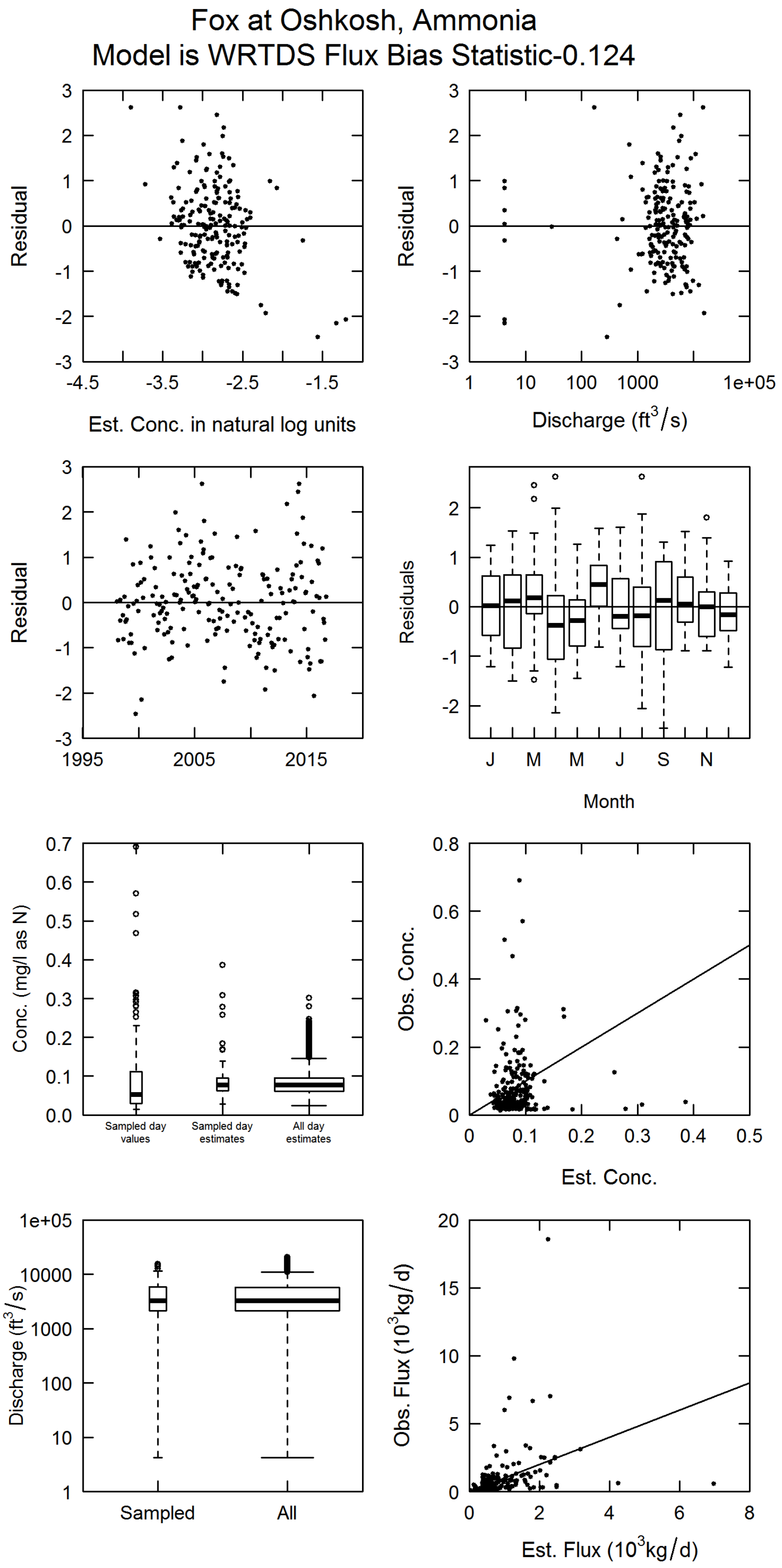

Fox at Oshkosh
KJN (mg/L)
0.0
0.5
1.0
1.5
2000
2005
2010
2015

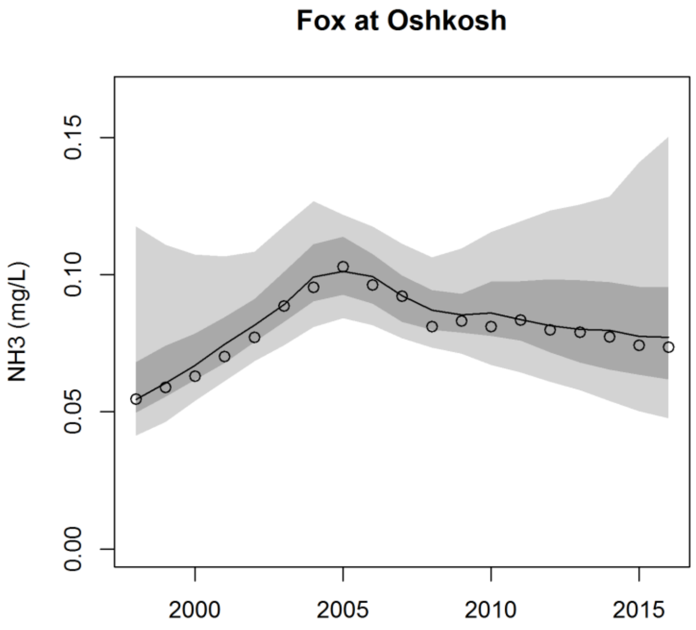

Fox at Oshkosh
KJN (tons/yr)
0
2000
4000
6000
8000
2000
2005
2010
2015

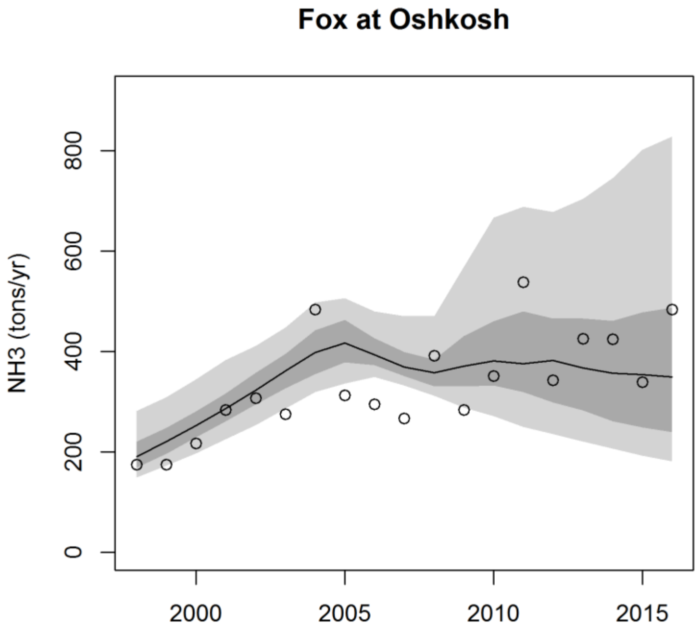

Fox at Oshkosh
Winter
1514 cfs
2940 cfs
5389 cfs
Spring
1675 cfs
6695 cfs
13549 cfs
Summer
134 cfs
4113 cfs
9823 cfs
Fall
4 cfs
3352 cfs
7873 cfs
KJN (mg/L)
0
1
2
3
4
1995
2000
2005
2010
2015

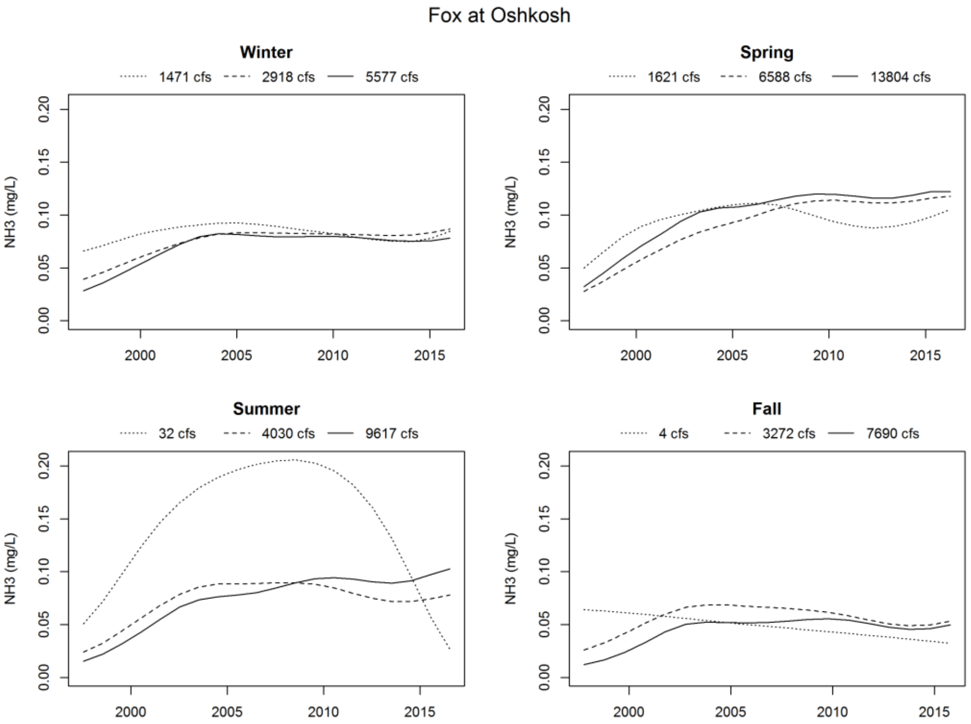
Ammonia


Fox at Oshkosh
NH3 (mg/L)
0.02
0.05
0.10
0.20
0.50
2000
2005
2010
2015

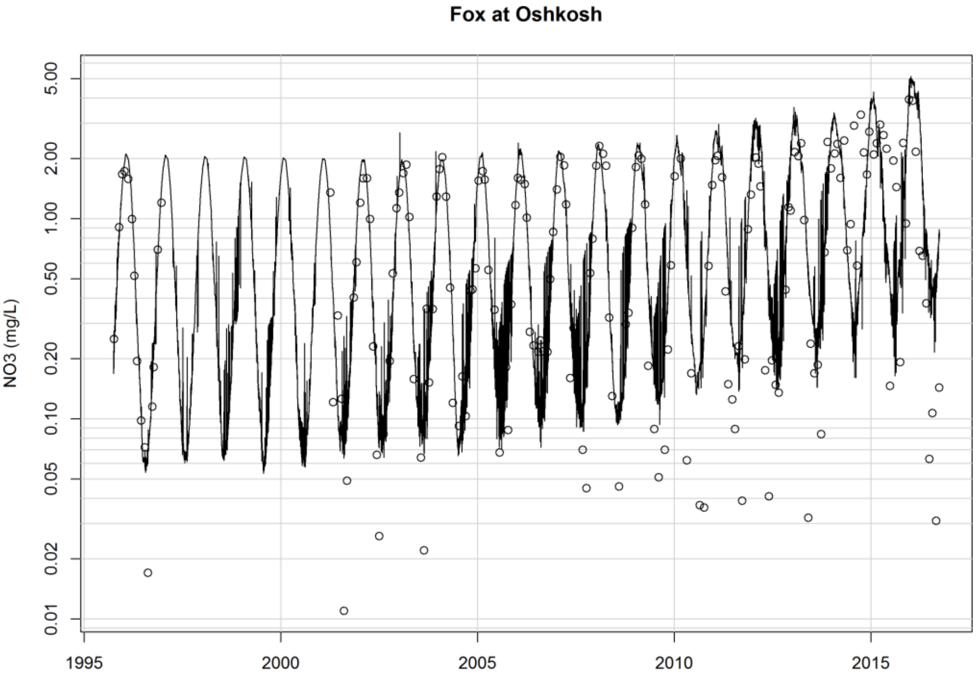

Fox at Oshkosh, Ammonia
Model is WRTDS Flux Bias Statistic-0.124

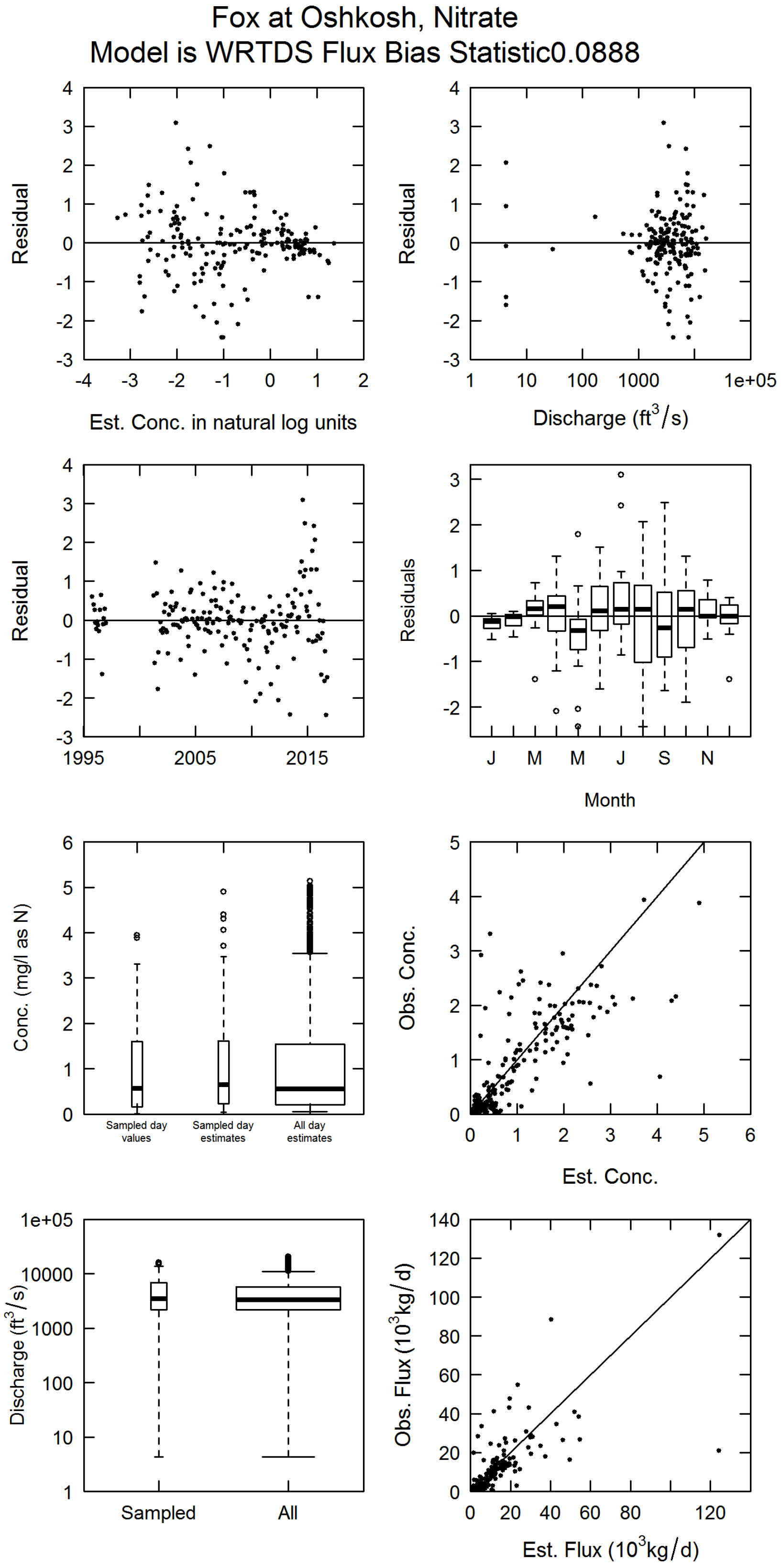

Fox at Oshkosh
NH3 (mg/L)
0.00
0.05
0.10
0.15
2000
2005
2010
2015

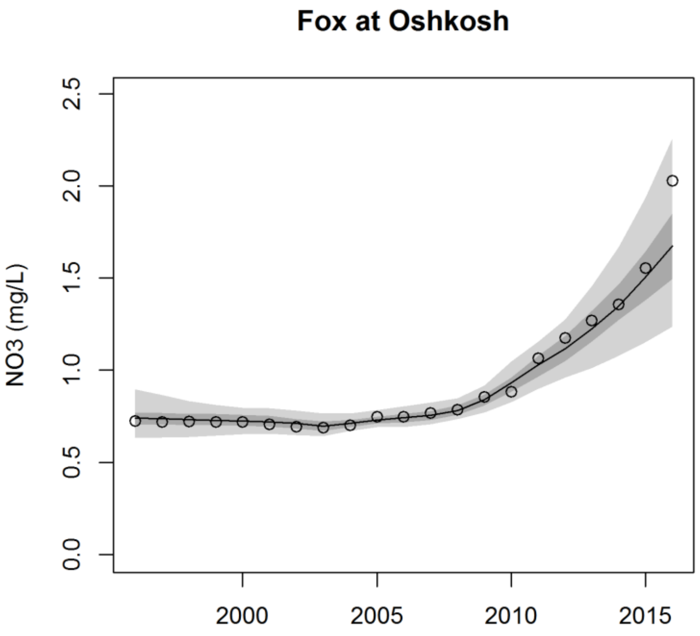

Fox at Oshkosh
NH3 (tons/yr)
0
200
400
600
800
2000
2005
2010
2015

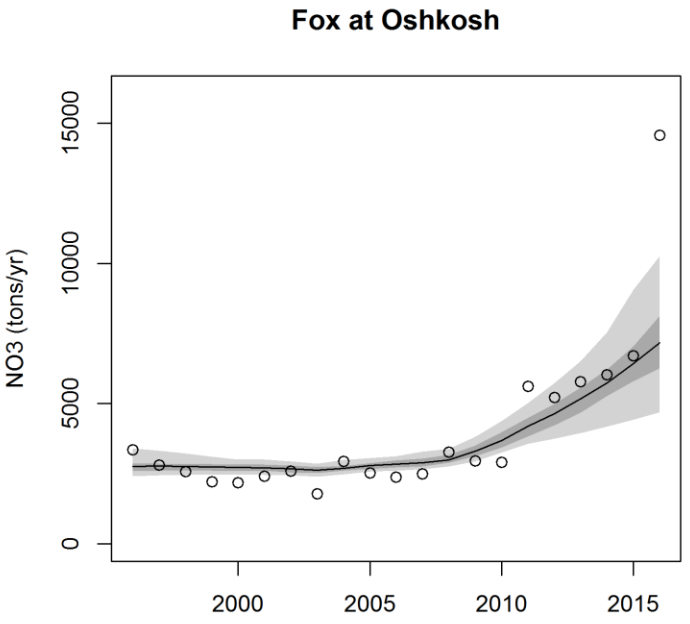

Fox at Oshkosh
Winter
1471 cfs
2918 cfs
5577 cfs
Spring
1621 cfs
6588 cfs
13804 cfs
Summer
32 cfs
4030 cfs
9617 cfs
Fall
4 cfs
3272 cfs
7690 cfs
NH3 (mg/L)
0.00
0.05
0.10
0.15
0.20
2000
2005
2010
2015

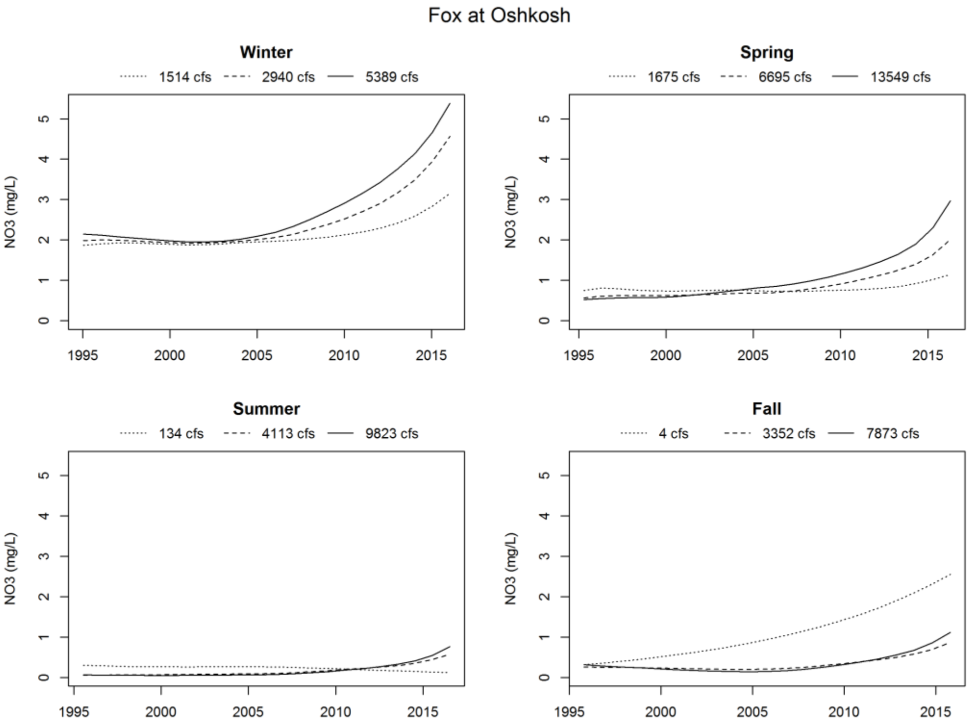
Nitrate


Fox at Oshkosh
NO3 (mg/L)
0.01
0.02
0.05
0.10
0.20
0.50
1.00
2.00
5.00
1995
2000
2005
2010
2015

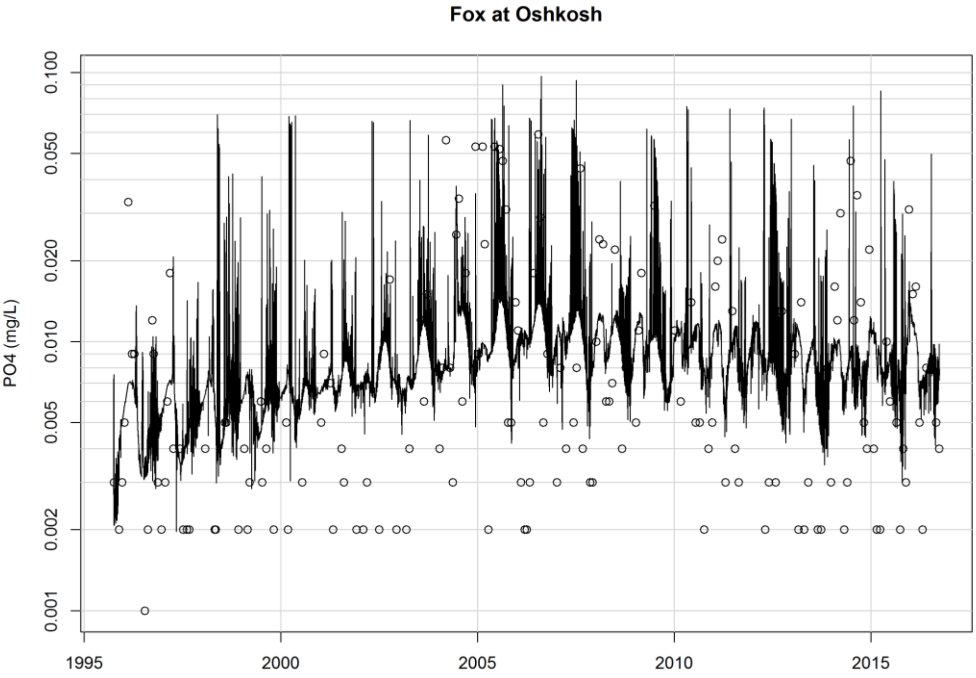

Fox at Oshkosh, Nitrate
Model is WRTDS Flux Bias Statistic0.0888
Residual
4
3
2
1
0
-1
-2
-3
-4
-3
-2
-1
0
1
2
Est. Conc. in natural log units
Residual
1
10
100
1000
1e+05
Discharge (ft3/s)
Residual
1995
2005
2015
Residuals
J
M
M
J
S
N
Month
Conc. (mg/l as N)
6
5
4
3
2
1
0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
Est. Conc.
Discharge (ft3/s)
1e+05
10000
1000
100
10
1
Sampled
All
Obs. Flux (103kg/d)
140
120
100
80
60
40
20
0
Est. Flux (103kg/d)

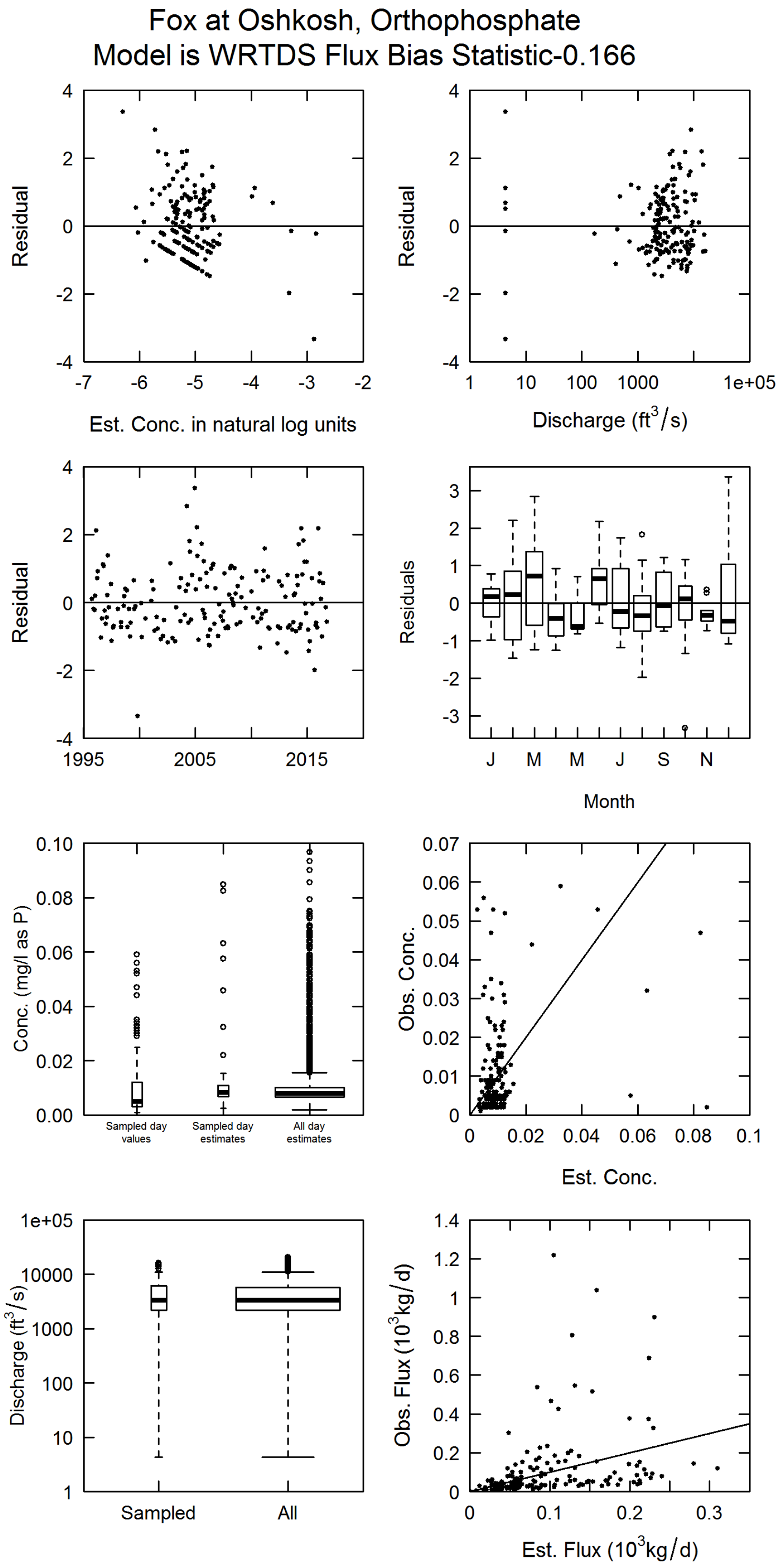

Fox at Oshkosh
NO3 (mg/L)
0.0
0.5
1.0
1.5
2.0
2.5
2000
2005
2010
2015


Fox at Oshkosh
NO3 (tons/yr)
0
5000
10000
15000
2000
2005
2010
2015

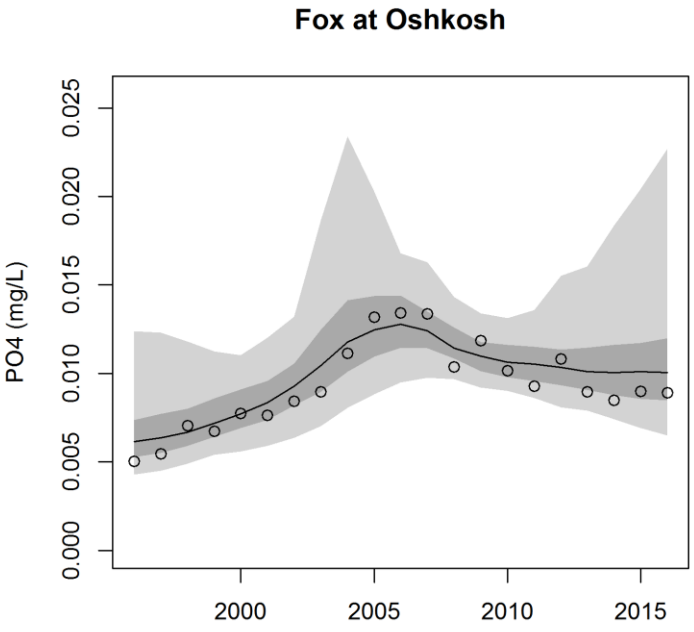

Fox at Oshkosh
Winter
1514 cfs
2940 cfs
5389 cfs
Spring
1675 cfs
6695 cfs
13549 cfs
Summer
134 cfs
4113 cfs
9823 cfs
Fall
4 cfs
3352 cfs
7873 cfs
NO3 (mg/L)
0
1
2
3
4
5
1995
2000
2005
2010
2015

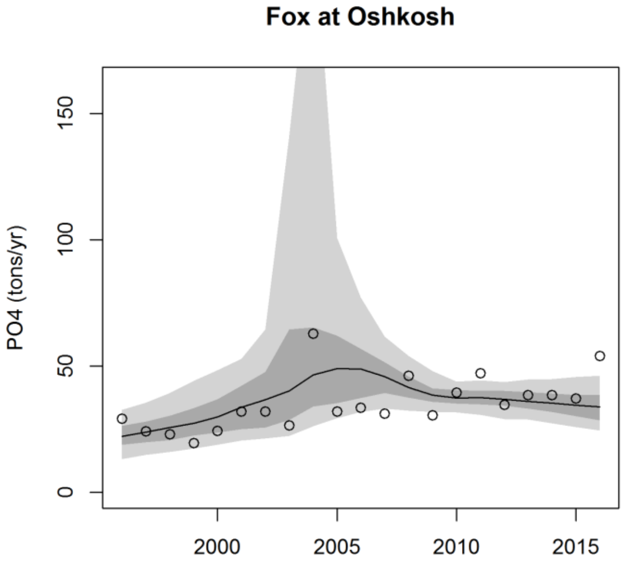
Phosphate

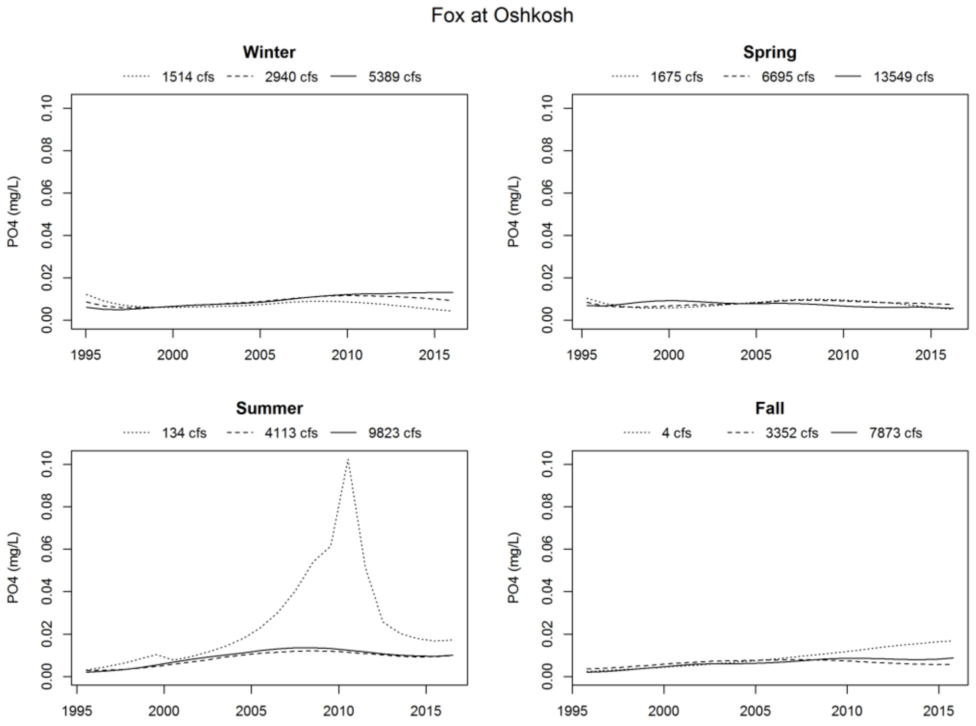

Fox at Oshkosh
PO4 (mg/L)
0.001
0.002
0.005
0.010
0.020
0.050
0.100
1995
2000
2005
2010
2015



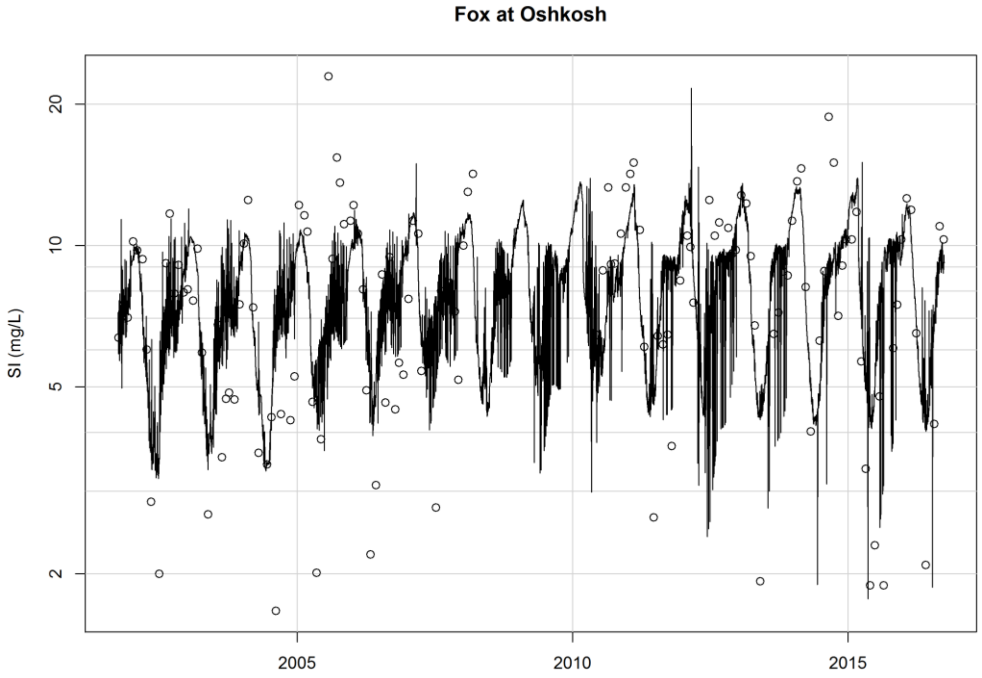
Fox at Oshkosh, Orthophosphate
Model is WRTDS Flux Bias Statistic-0.166

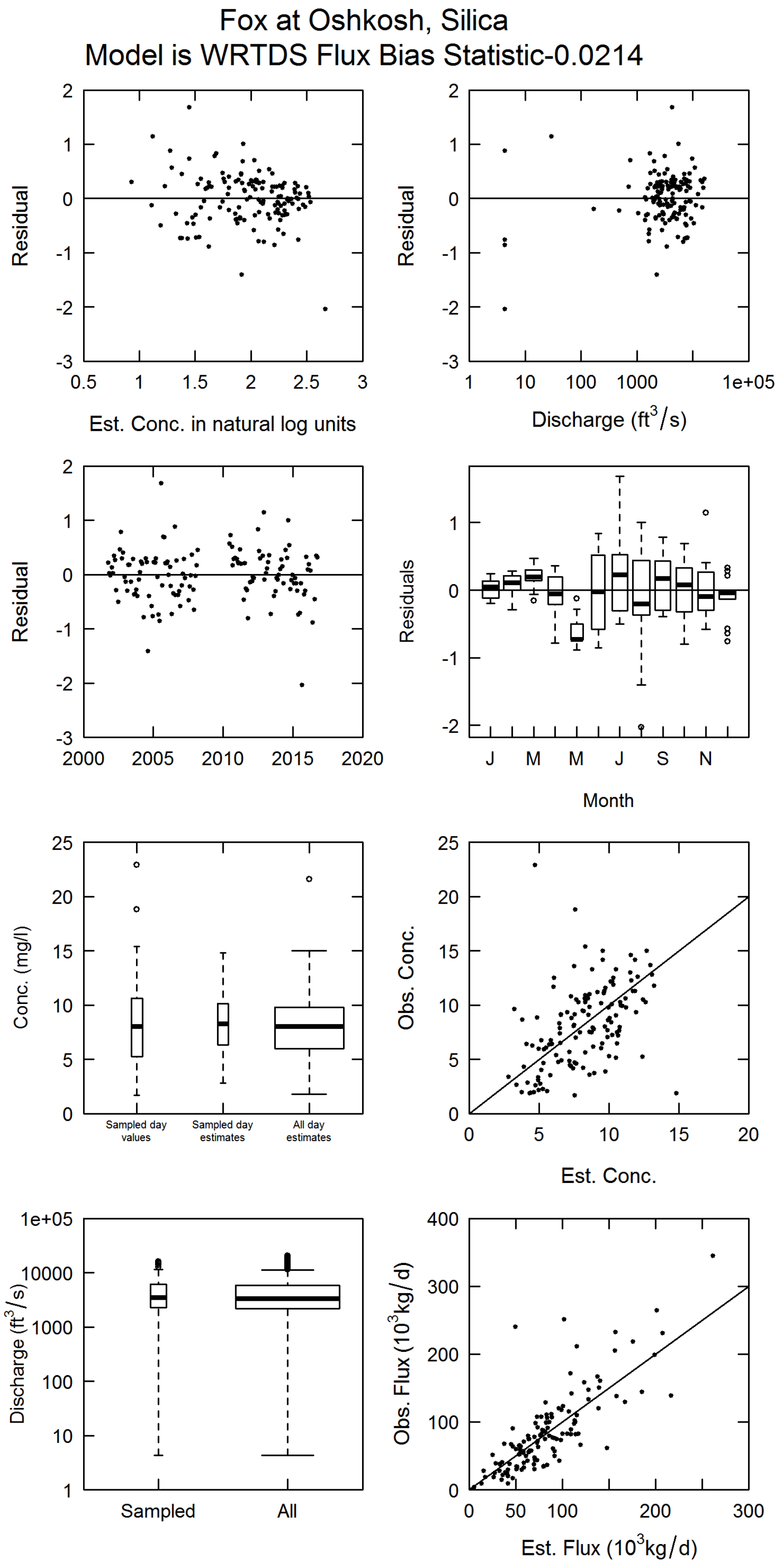

Fox at Oshkosh
PO4 (mg/L)
0.000
0.005
0.010
0.015
0.020
0.025
2000
2005
2010
2015


Fox at Oshkosh
PO4 (tons/yr)
0
50
100
150
2000
2005
2010
2015

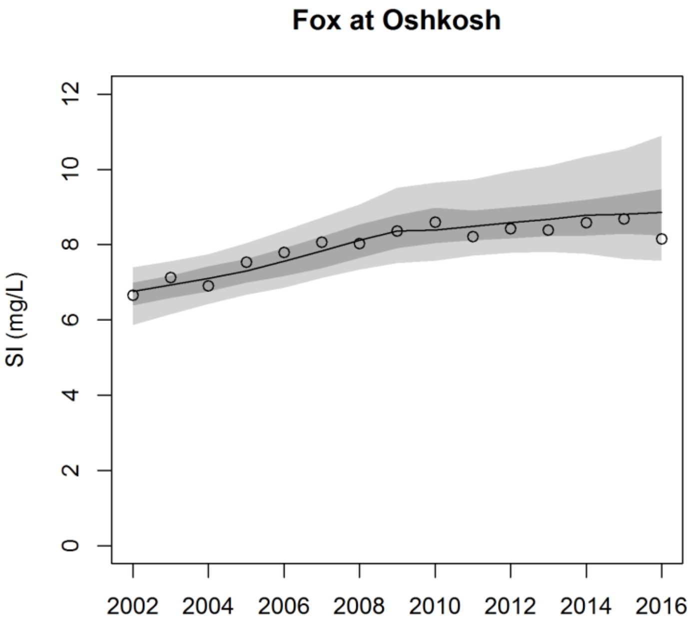

Fox at Oshkosh
Winter
1514 cfs
2940 cfs
5389 cfs
Spring
1675 cfs
6695 cfs
13549 cfs
Summer
134 cfs
4113 cfs
9823 cfs
Fall
4 cfs
3352 cfs
7873 cfs
PO4 (mg/L)
0.00
0.02
0.04
0.06
0.08
0.10
1995
2000
2005
2010
2015

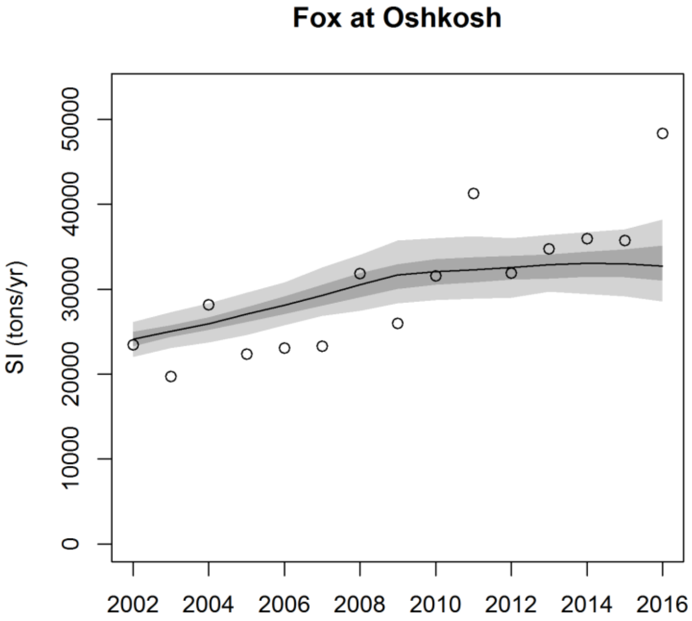
Silica

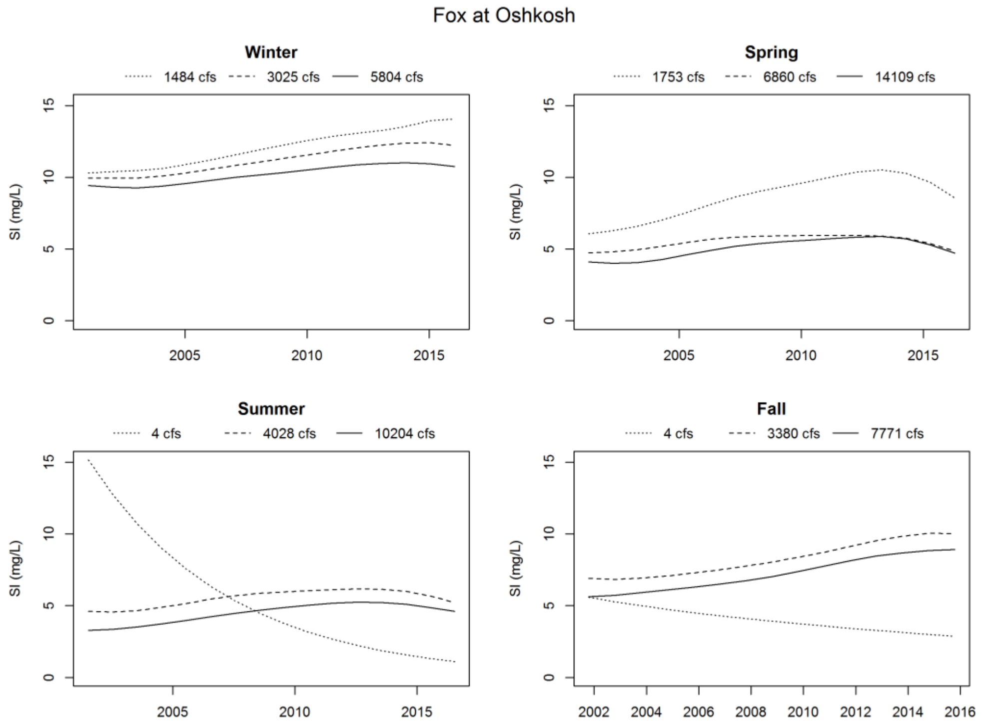

Fox at Oshkosh
SI (mg/L)
2
5
10
20
2005
2010
2015



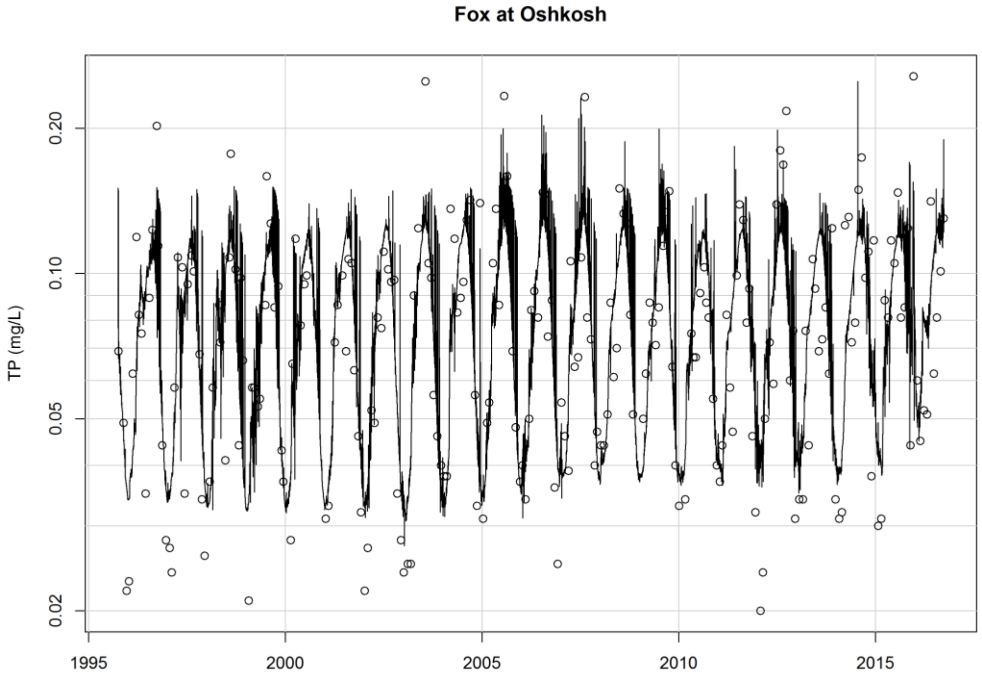
Fox at Oshkosh, Silica
Model is WRTDS Flux Bias Statistic-0.0214

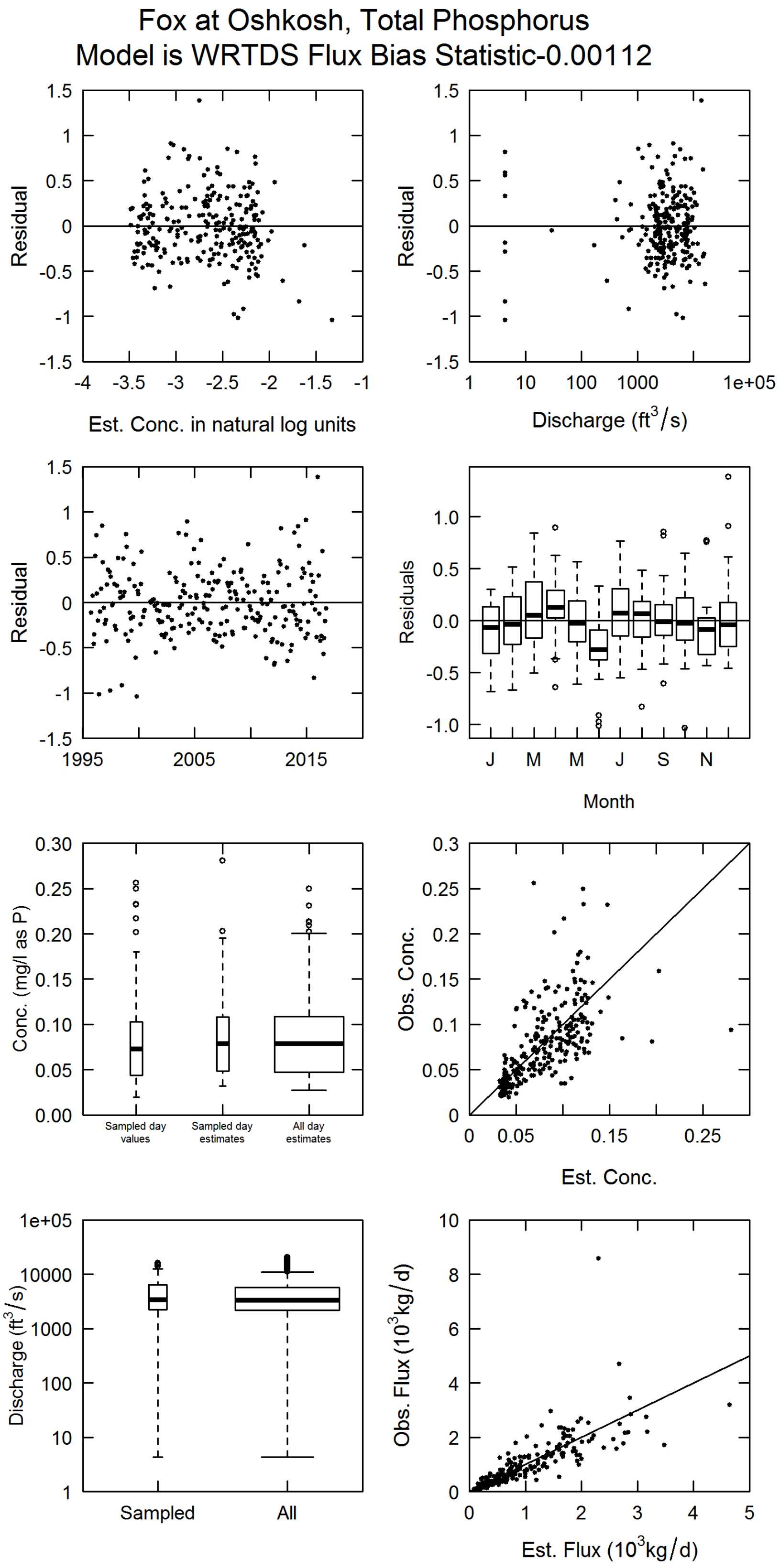

Fox at Oshkosh
SI (mg/L)
0
2
4
6
8
10
12
2002
2004
2006
2008
2010
2012
2014
2016


Fox at Oshkosh
SI (tons/yr)
0
10000
20000
30000
40000
50000
2002
2004
2006
2008
2010
2012
2014
2016

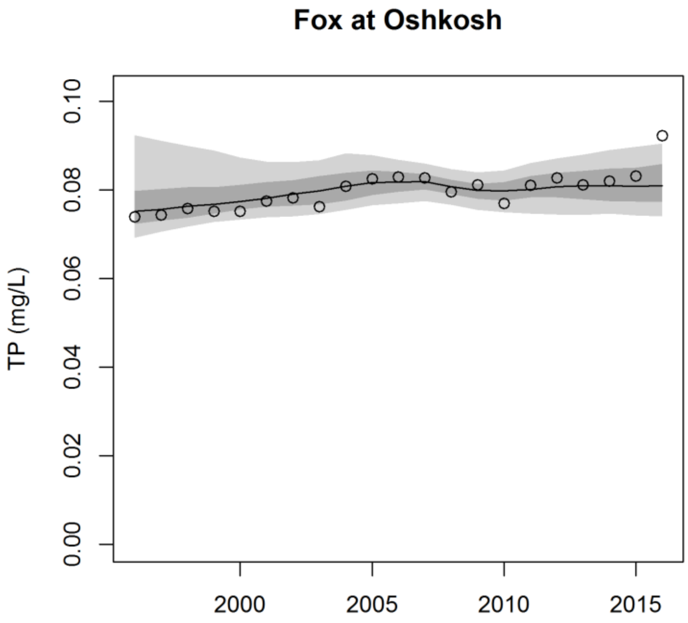


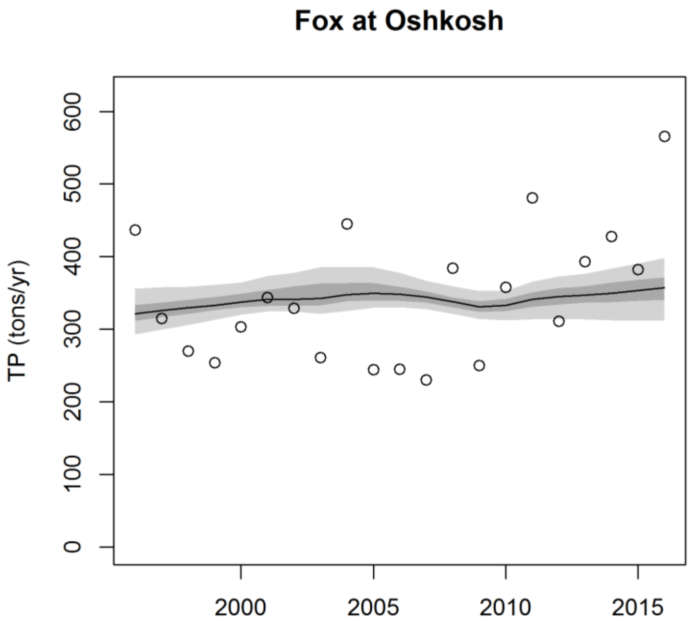
Total Phosphorus

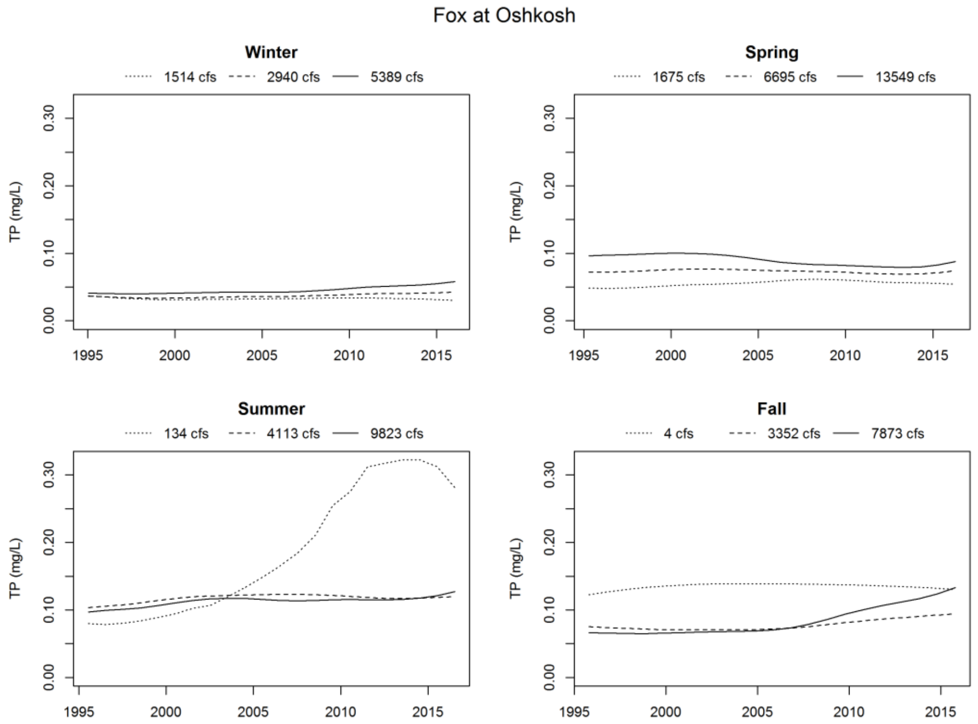




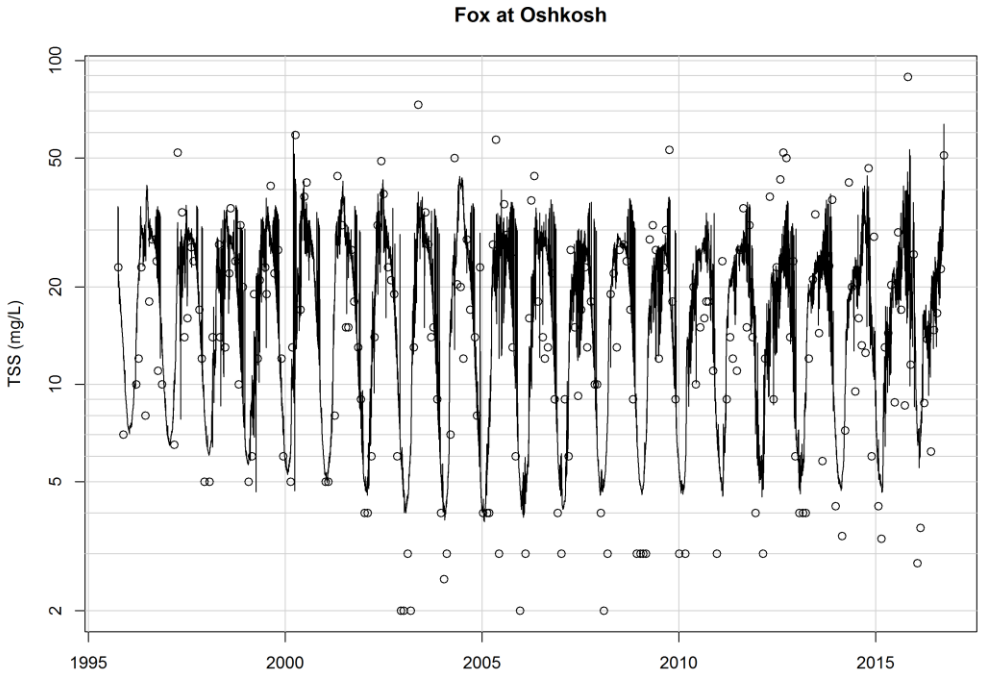
Fox at Oshkosh, Total Phosphorus
Model is WRTDS Flux Bias Statistic-0.00112

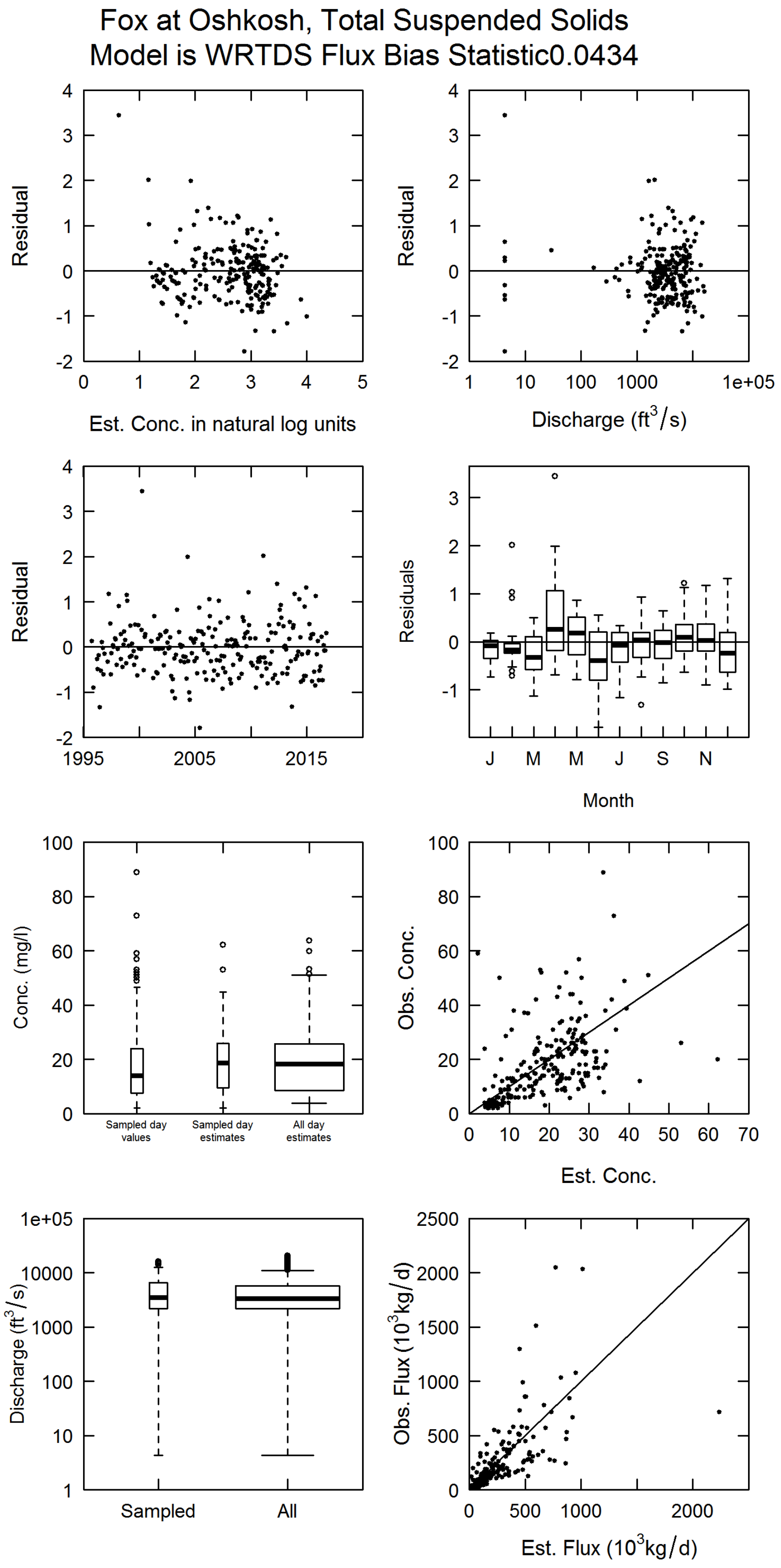

Fox at Oshkosh
TP (mg/L)
0.00
0.02
0.04
0.06
0.08
0.10
2000
2005
2010
2015

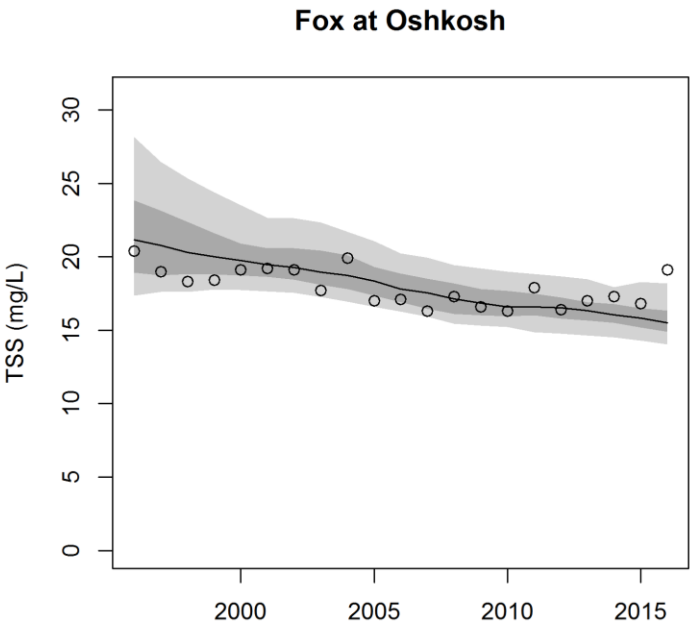

Fox at Oshkosh
TP (tons/yr)
0
100
200
300
400
500
600
2000
2005
2010
2015

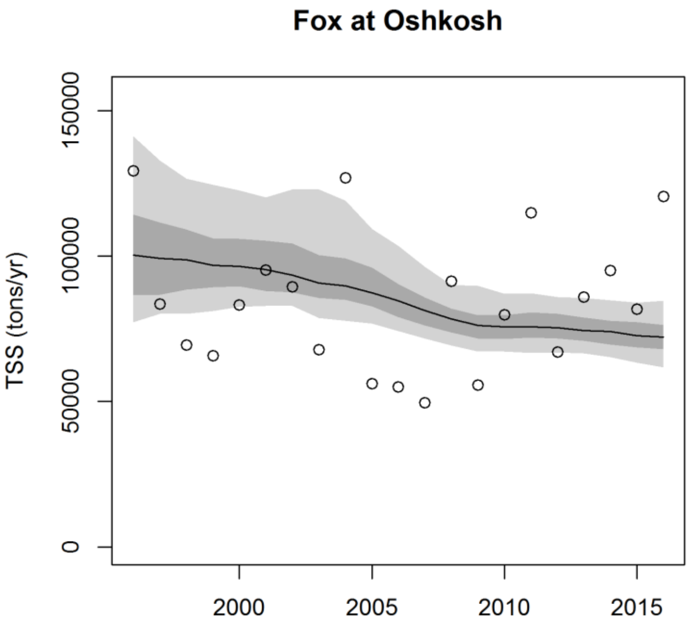


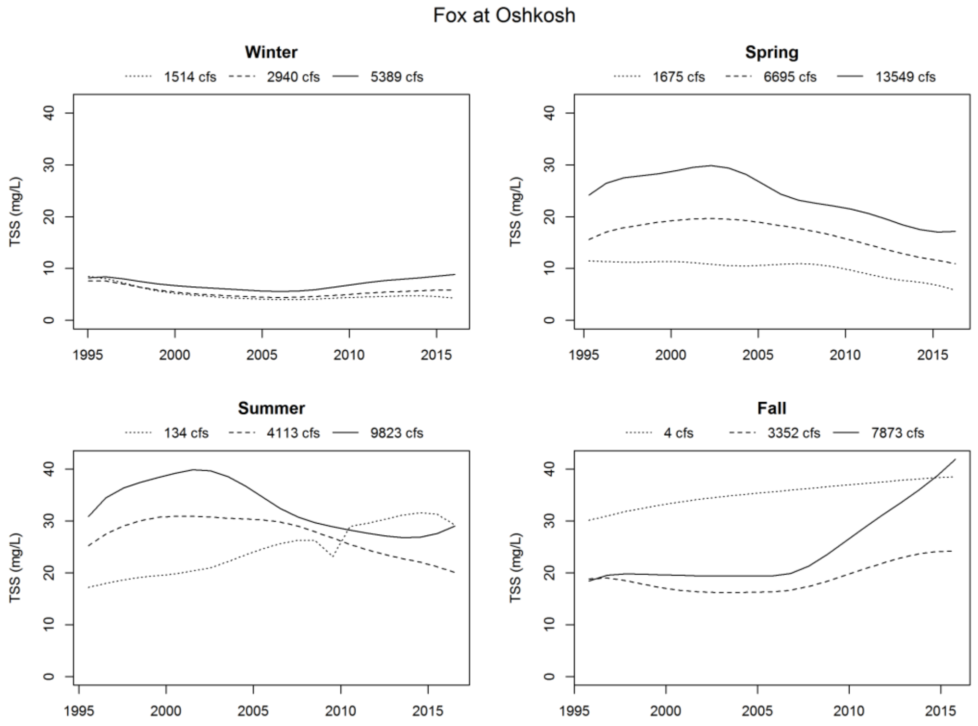
Total Suspended Solids




Fox at Oshkosh, Total Suspended Solids
Model is WRTDS Flux Bias Statistic0.0434

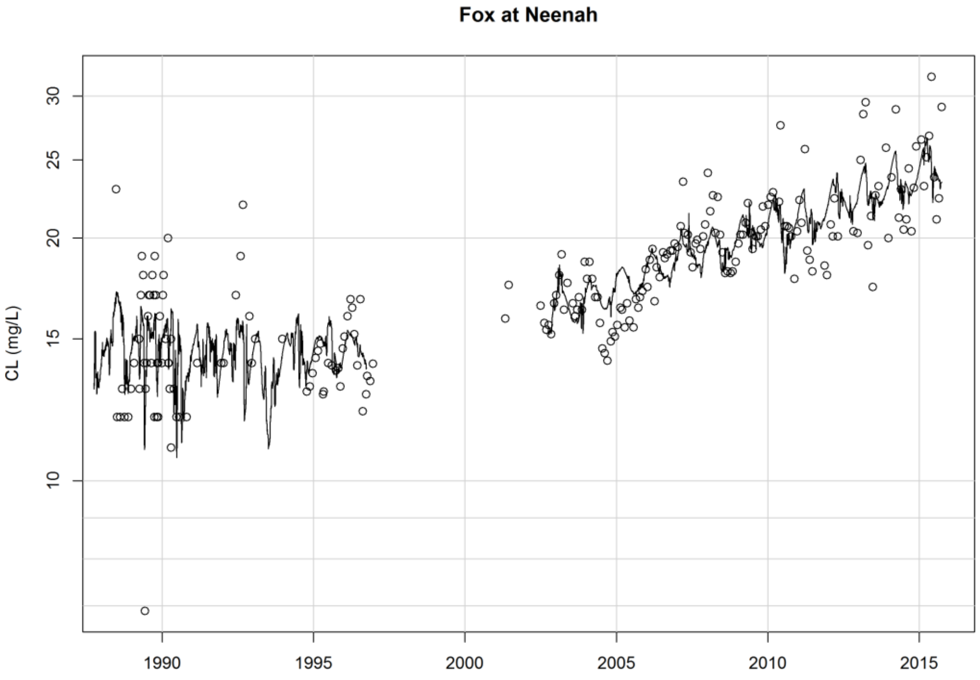

Fox at Oshkosh
TSS (mg/L)
0
5
10
15
20
25
30
2000
2005
2010
2015

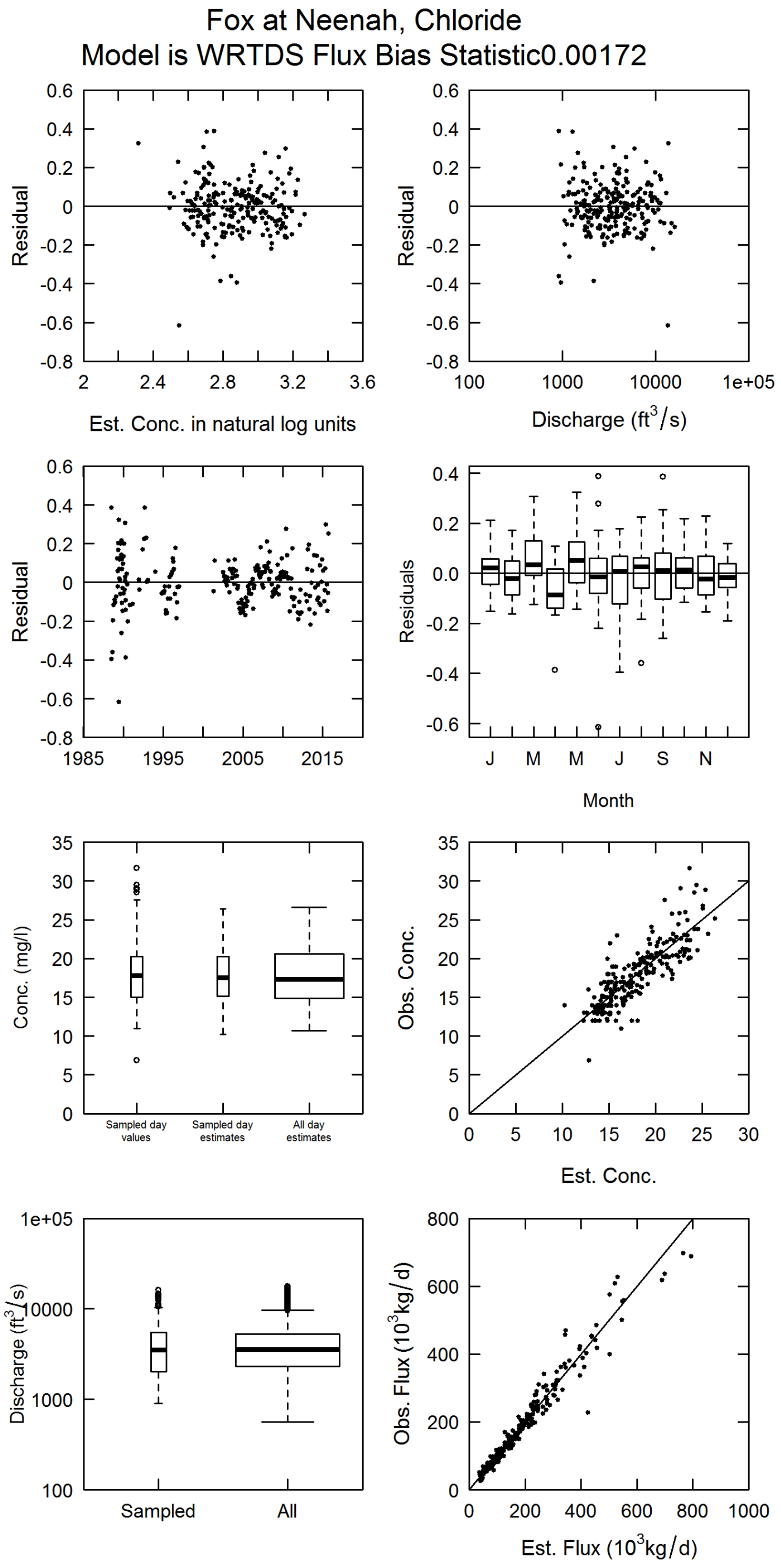

Fox at Oshkosh
TSS (tons/yr)
0
50000
100000
150000
2000
2005
2010
2015

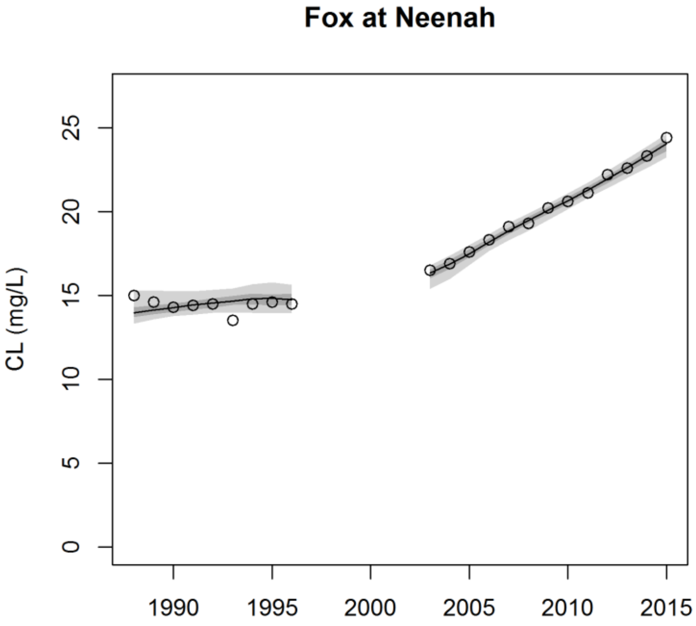


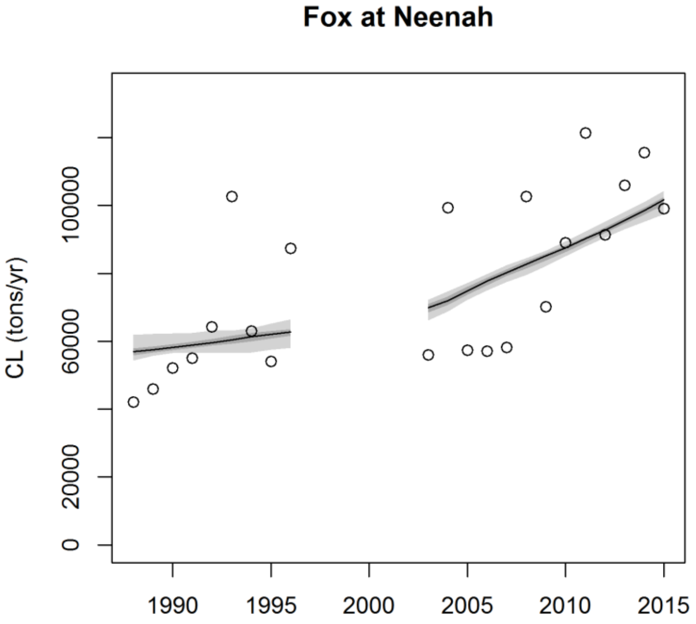
FOX RIVER AT NEENAH (713002)

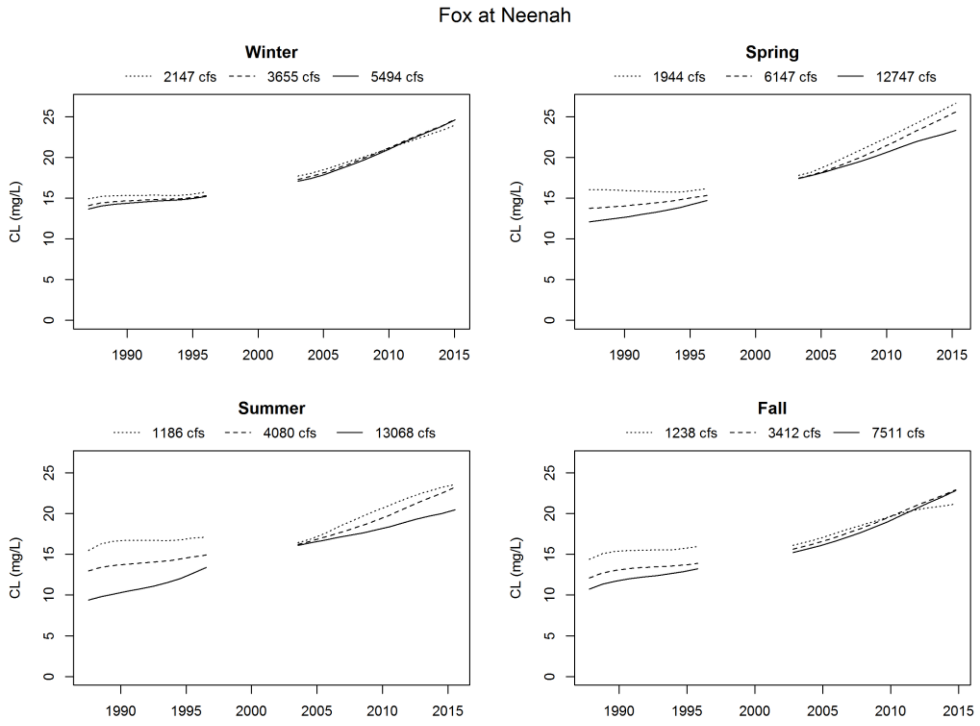
Chloride



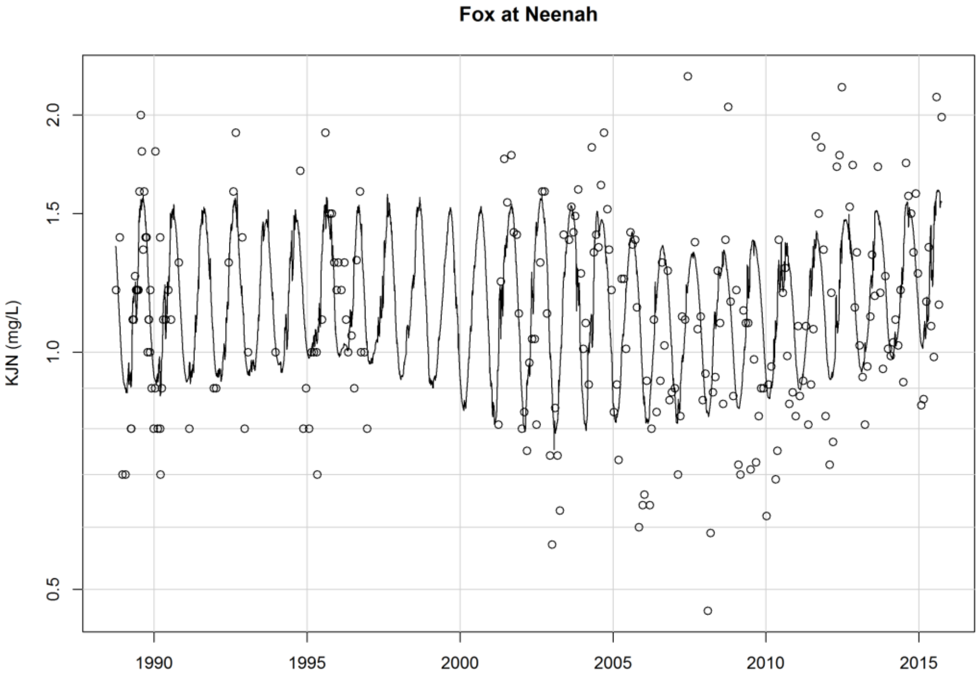

Fox at Neenah, Chloride
Model is WRTDS Flux Bias Statistic0.00172
Residual
Est. Conc. in natural log units
2
2.4
2.8
3.2
3.6
Residual
Discharge (ft3/s)
100
1000
10000
1e+05
Residual
1985
1995
2005
2015
Residuals
Month
J
M
M
J
S
N
Conc. (mg/l)
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
Est. Conc.
Discharge (ft3/s)
Sampled
All
Obs. Flux (10³kg/d)
Est. Flux (10³kg/d)

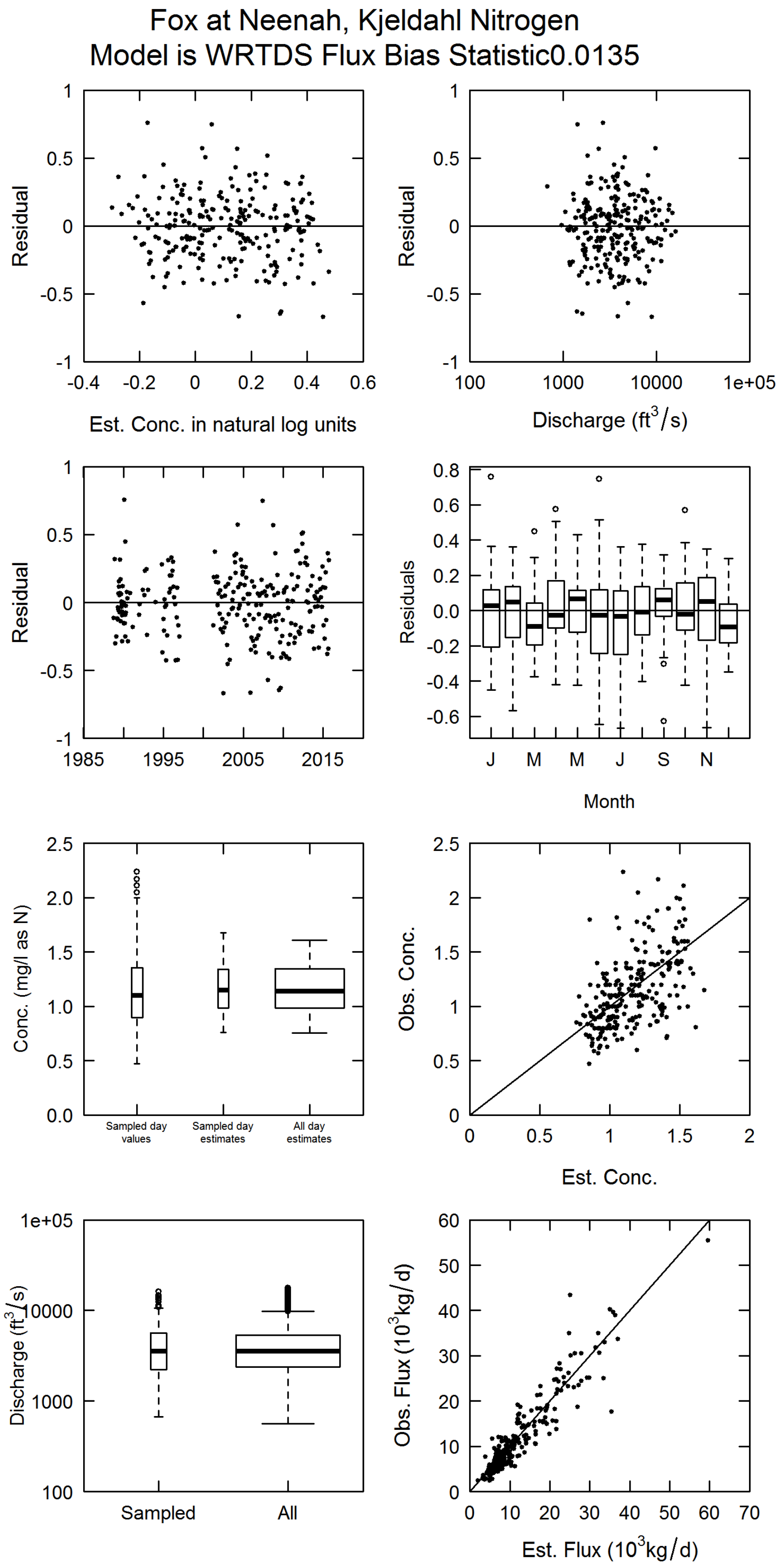

Fox at Neenah
CL (mg/L)
0
5
10
15
20
25
1990
1995
2000
2005
2010
2015

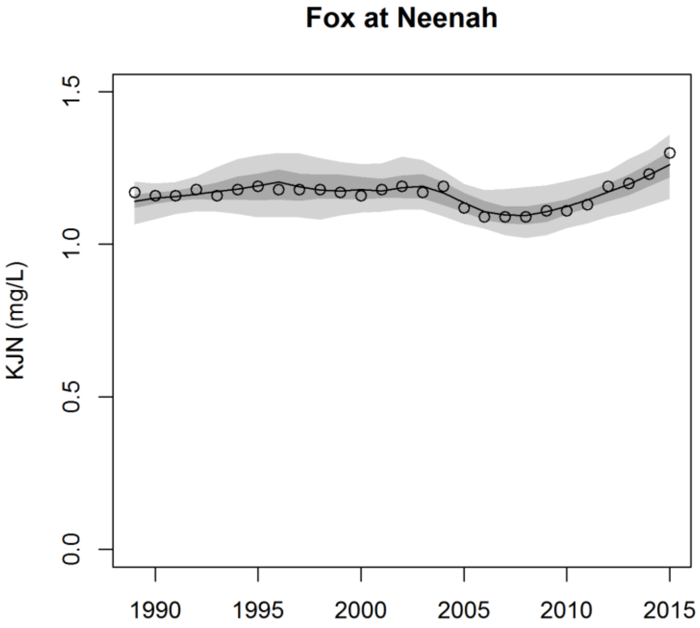

Fox at Neenah
CL (tons/yr)
0
20000
60000
100000
1990
1995
2000
2005
2010
2015

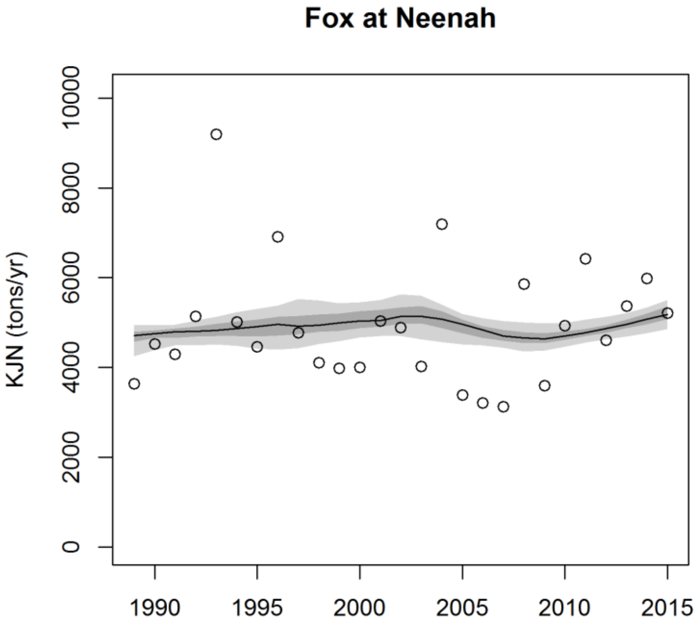


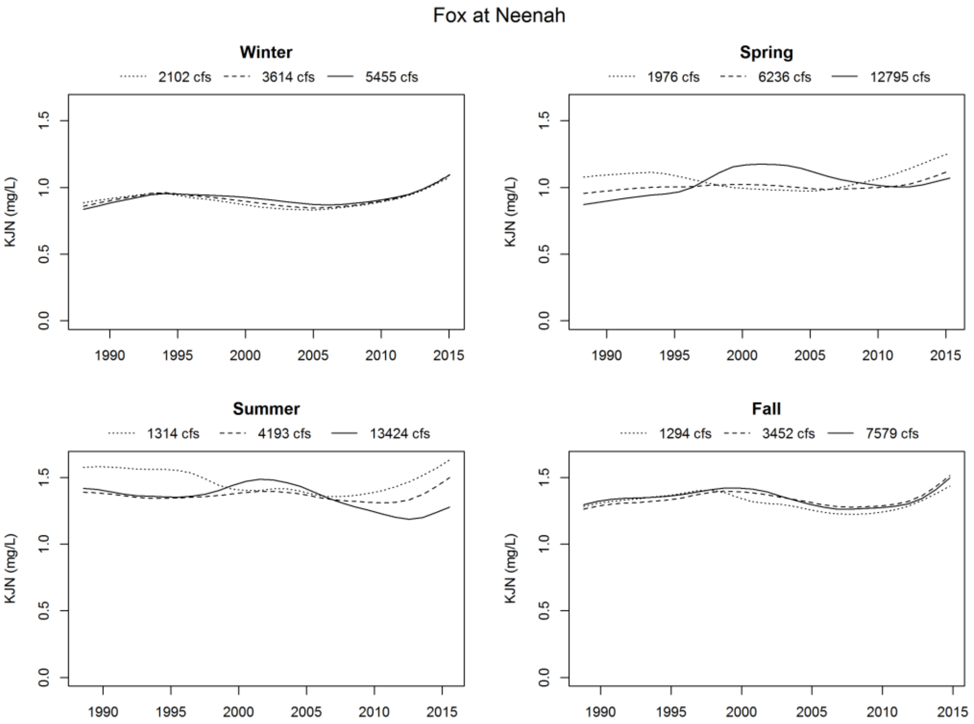
Total Kjeldahl Nitrogen



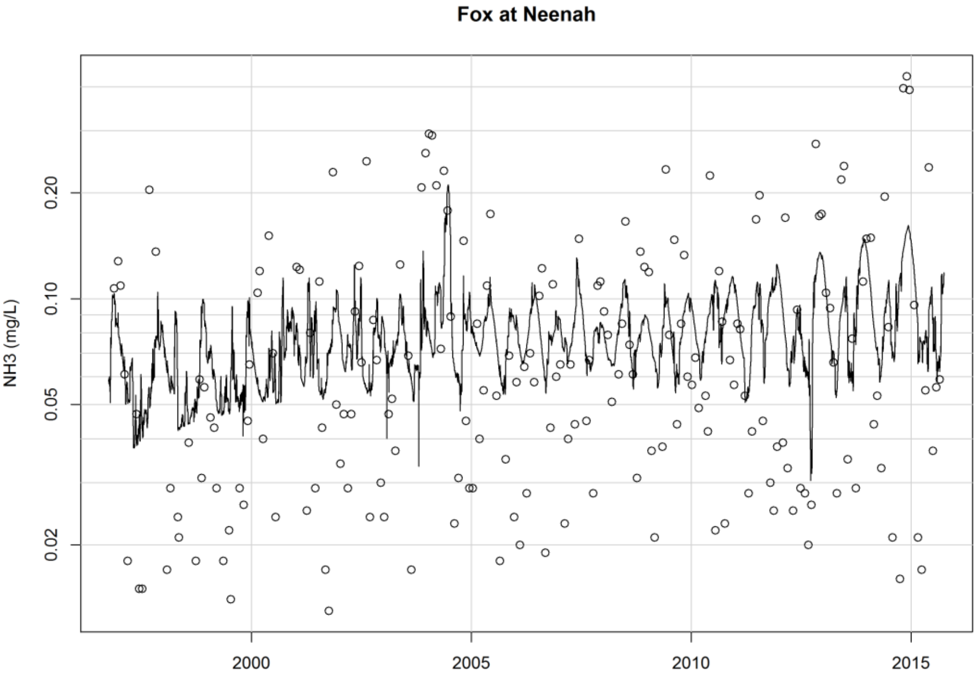

Fox at Neenah, Kjeldahl Nitrogen
Model is WRTDS Flux Bias Statistic0.0135

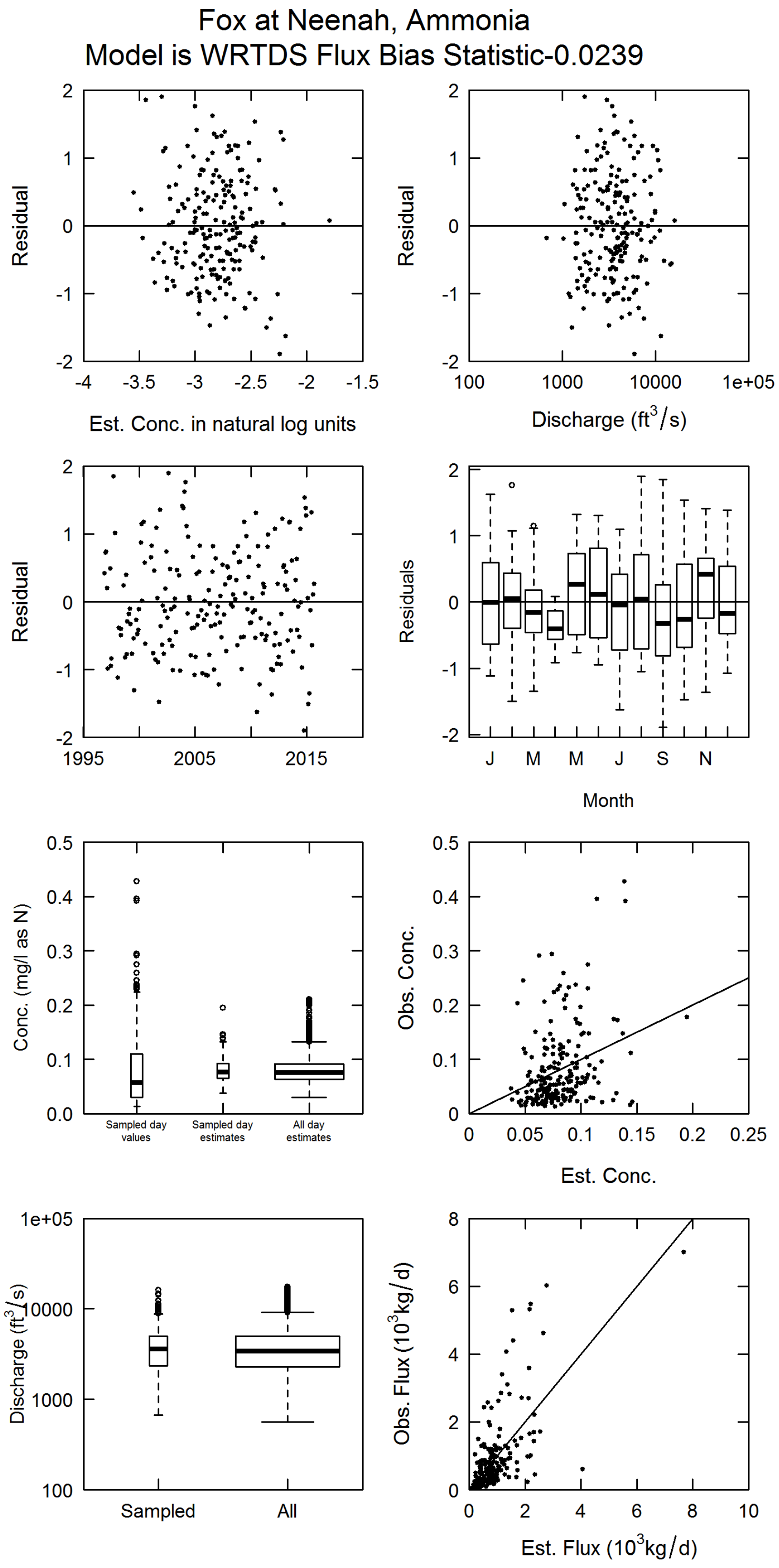

Fox at Neenah
KJN (mg/L)
0.0
0.5
1.0
1.5
1990
1995
2000
2005
2010
2015

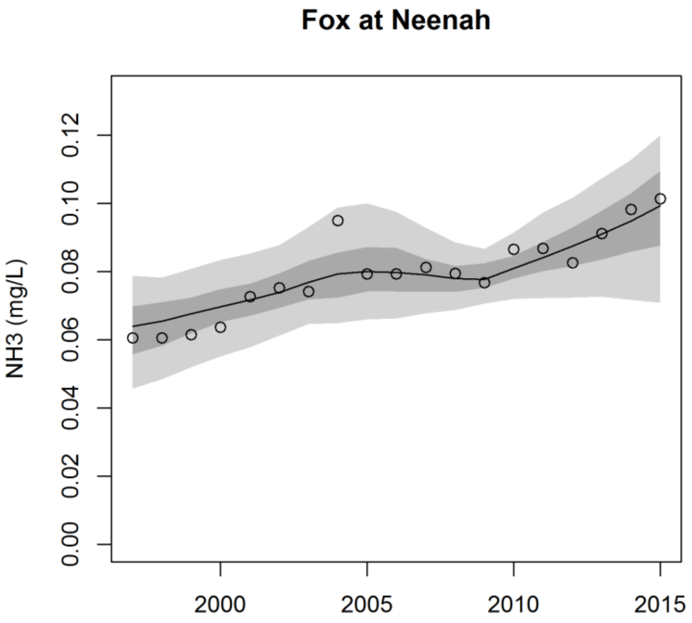

Fox at Neenah
KJN (tons/yr)
0
2000
4000
6000
8000
10000
1990
1995
2000
2005
2010
2015

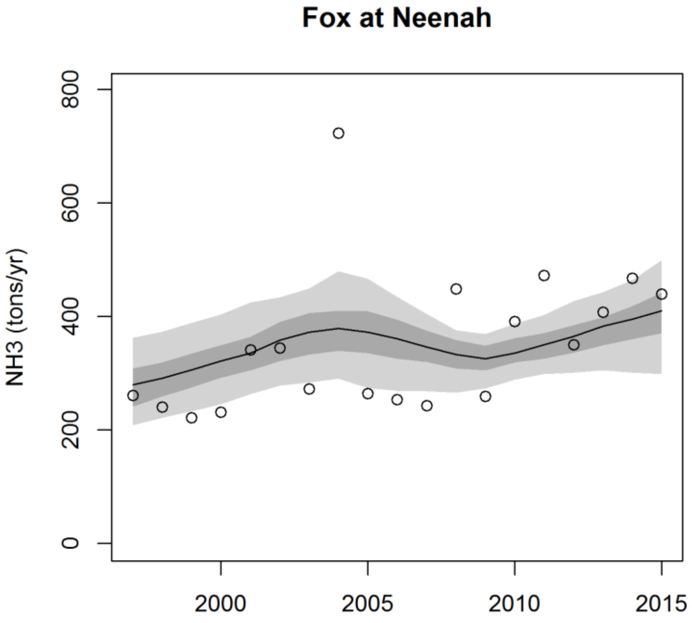

Fox at Neenah
Winter
2102 cfs
3614 cfs
5455 cfs
Spring
1976 cfs
6236 cfs
12795 cfs
Summer
1314 cfs
4193 cfs
13424 cfs
Fall
1294 cfs
3452 cfs
7579 cfs
KJN (mg/L)
0.0
0.5
1.0
1.5
1990
1995
2000
2005
2010
2015

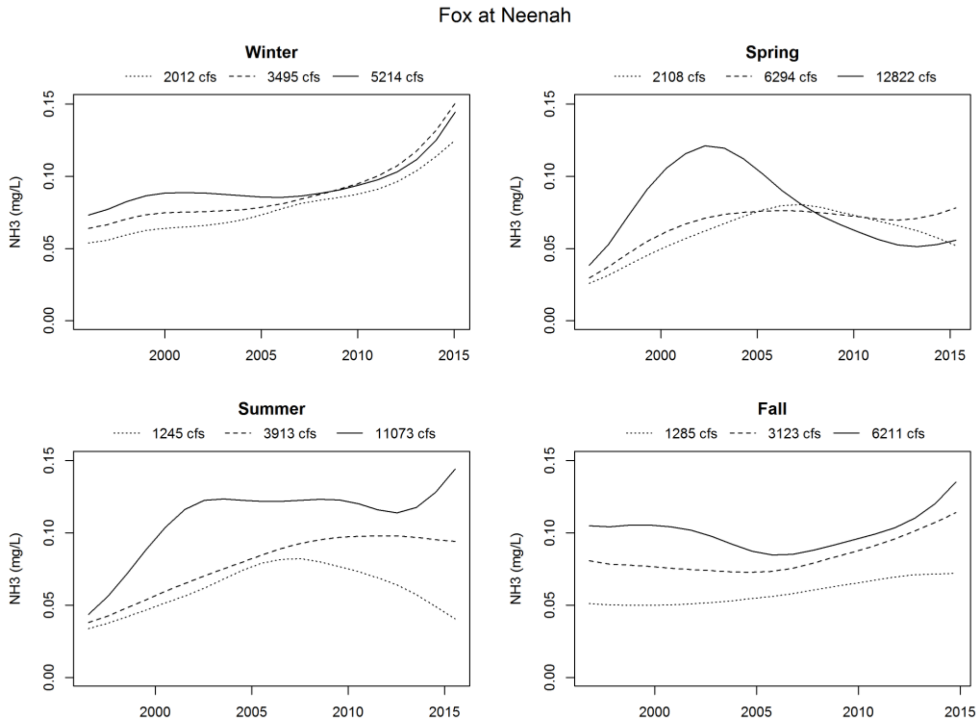
Ammonia


Fox at Neenah
NH3 (mg/L)
0.02
0.05
0.10
0.20
2000
2005
2010
2015

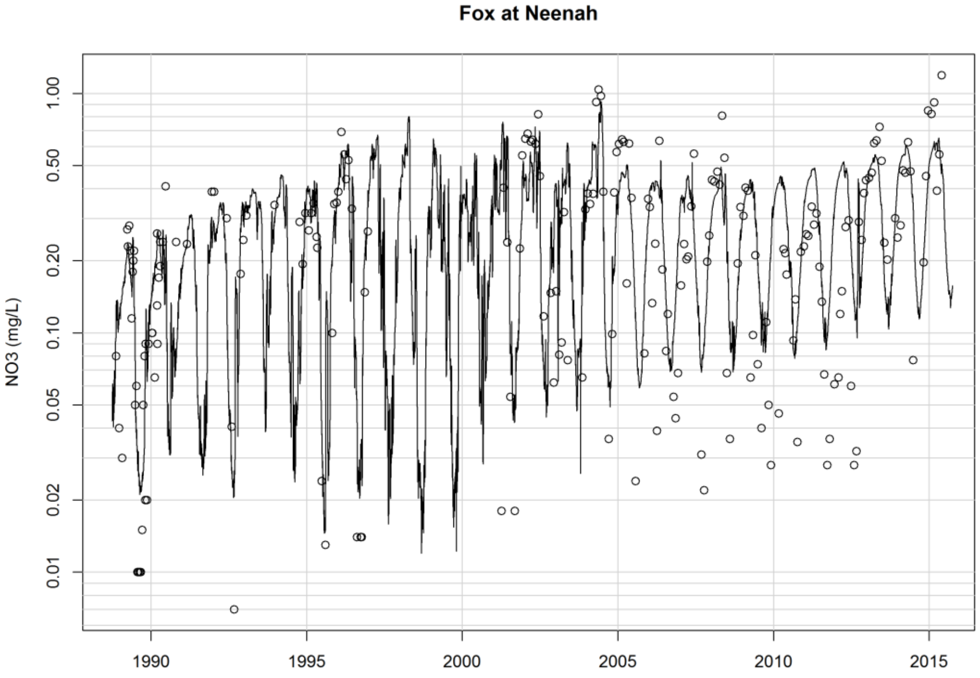

Fox at Neenah, Ammonia
Model is WRTDS Flux Bias Statistic-0.0239
Residual
2
1
0
-1
-2
-4
-3.5
-3
-2.5
-2
-1.5
Est. Conc. in natural log units
Residual
100
1000
10000
1e+05
Discharge (ft³/s)
Residual
1995
2005
2015
Residuals
J
M
M
J
S
N
Month
Conc. (mg/l as N)
0.5
0.4
0.3
0.2
0.1
0.0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
0
0.05
0.1
0.15
0.2
0.25
Est. Conc.
Discharge (ft³/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (10³kg/d)
8
6
4
2
0
0
2
4
6
8
10
Est. Flux (10³kg/d)

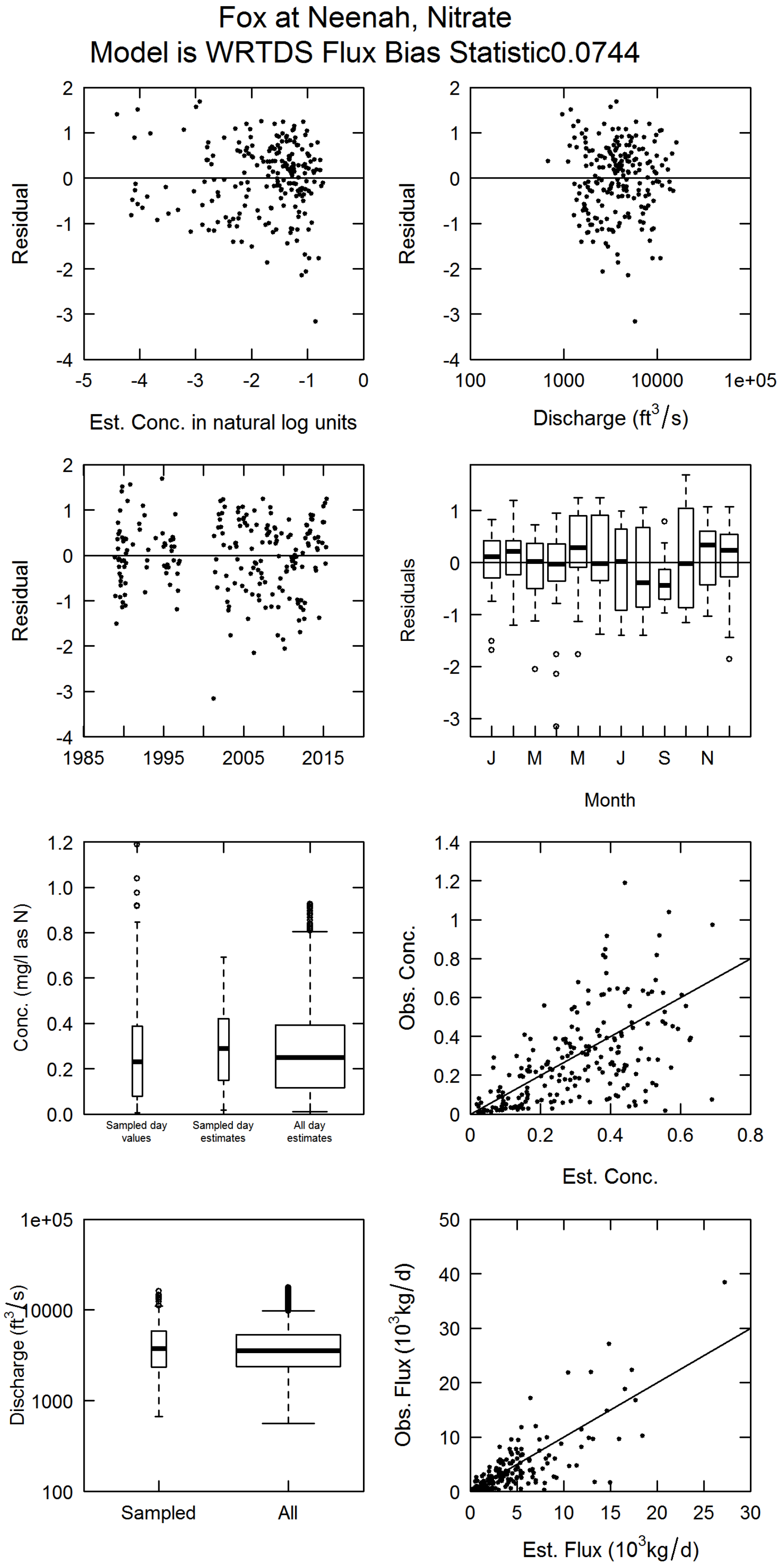

Fox at Neenah

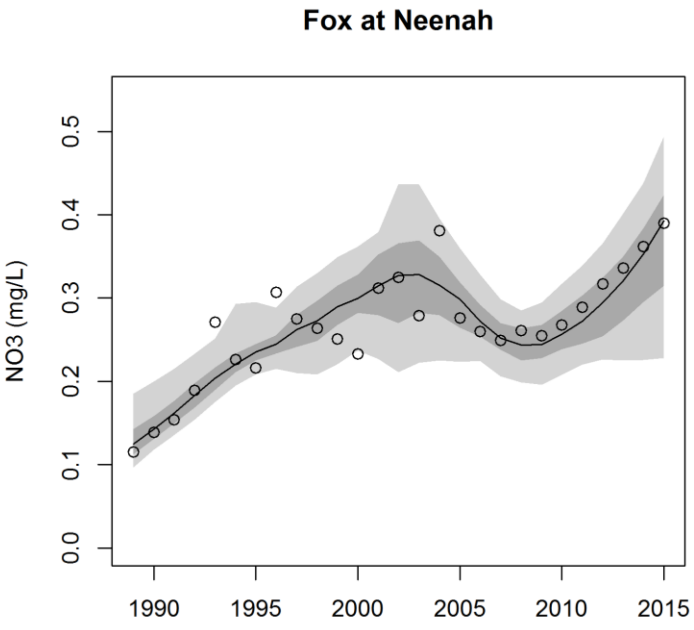

Fox at Neenah

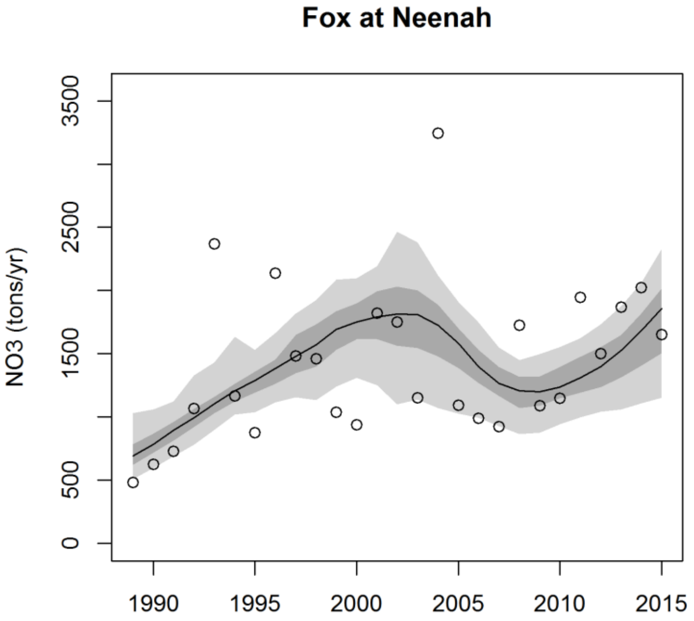

Fox at Neenah
Winter
2012 cfs
3495 cfs
5214 cfs
Spring
2108 cfs
6294 cfs
12822 cfs
Summer
1245 cfs
3913 cfs
11073 cfs
Fall
1285 cfs
3123 cfs
6211 cfs
NH3 (mg/L)
0.00
0.05
0.10
0.15
2000
2005
2010
2015

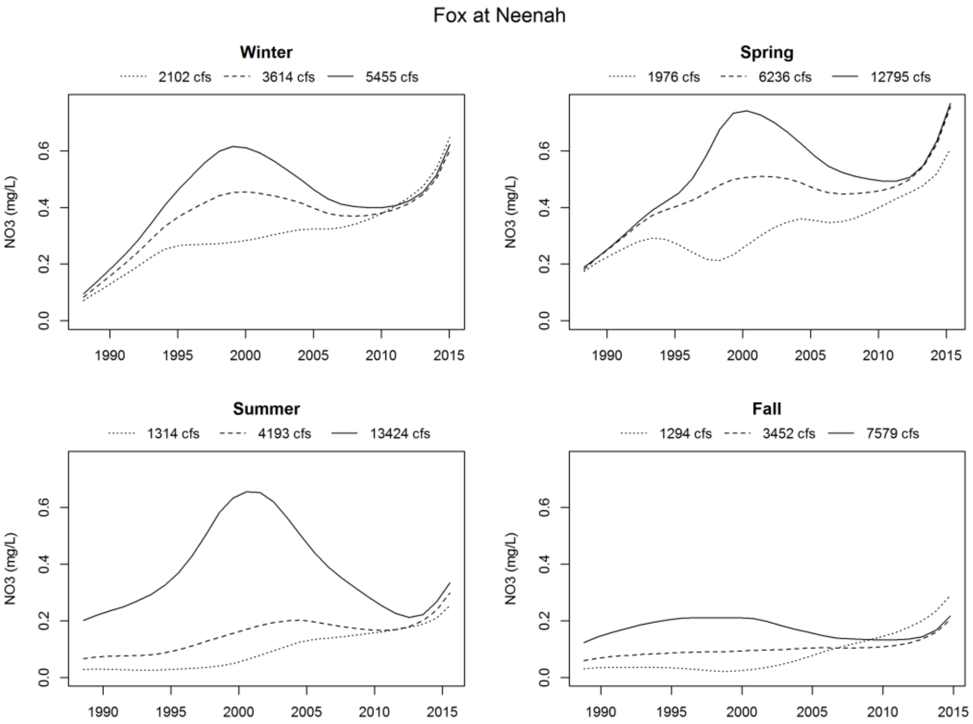
Nitrate


Fox at Neenah
NO3 (mg/L)
0.01
0.02
0.05
0.10
0.20
0.50
1.00
1990
1995
2000
2005
2010
2015

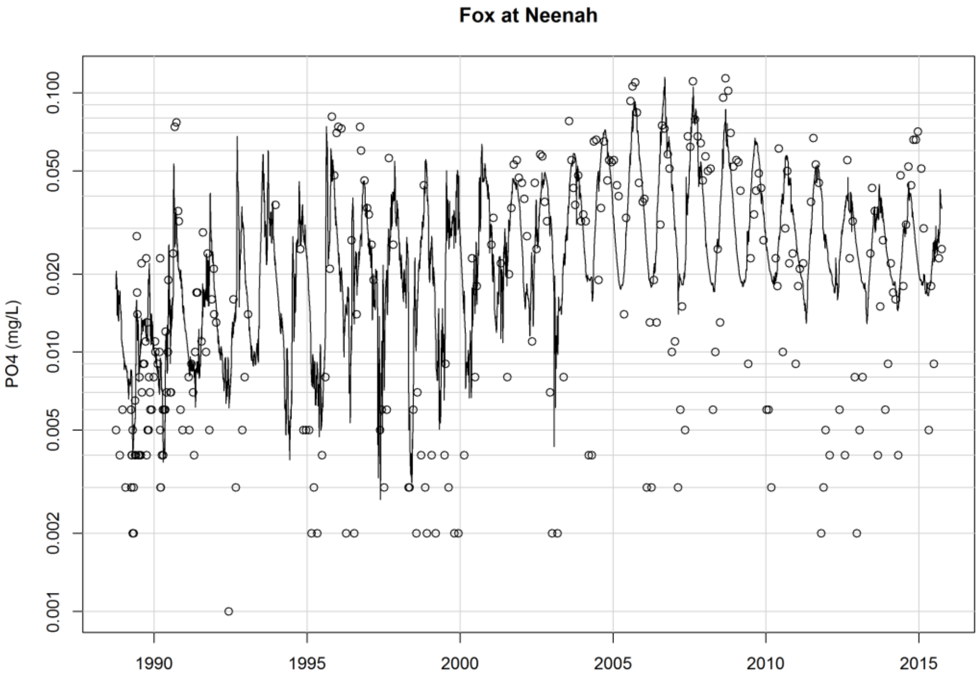

Fox at Neenah, Nitrate
Model is WRTDS Flux Bias Statistic0.0744
Residual
2
1
0
-1
-2
-3
-4
-5
-4
-3
-2
-1
0
Est. Conc. in natural log units
Residual
100
1000
10000
1e+05
Discharge (ft3/s)
Residual
1985
1995
2005
2015
Residuals
J
M
M
J
S
N
Month
Conc. (mg/l as N)
1.2
1.0
0.8
0.6
0.4
0.2
0.0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
1.4
0.8
Est. Conc.
Discharge (ft3/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (10^3kg/d)
50
40
30
20
10
0
5
15
25
Est. Flux (10^3kg/d)

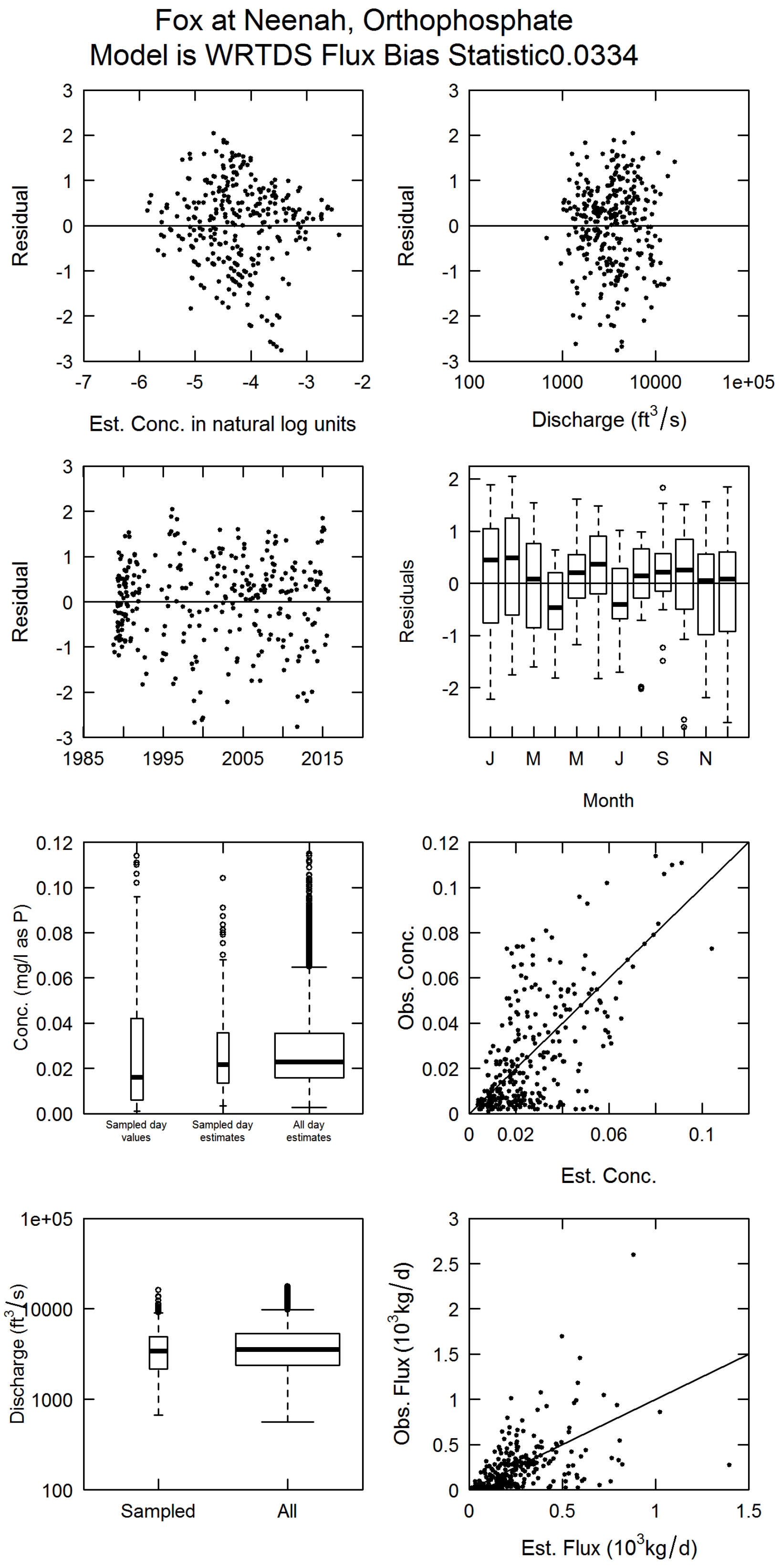

Fox at Neenah
NO3 (mg/L)
0.0
0.1
0.2
0.3
0.4
0.5
1990
1995
2000
2005
2010
2015

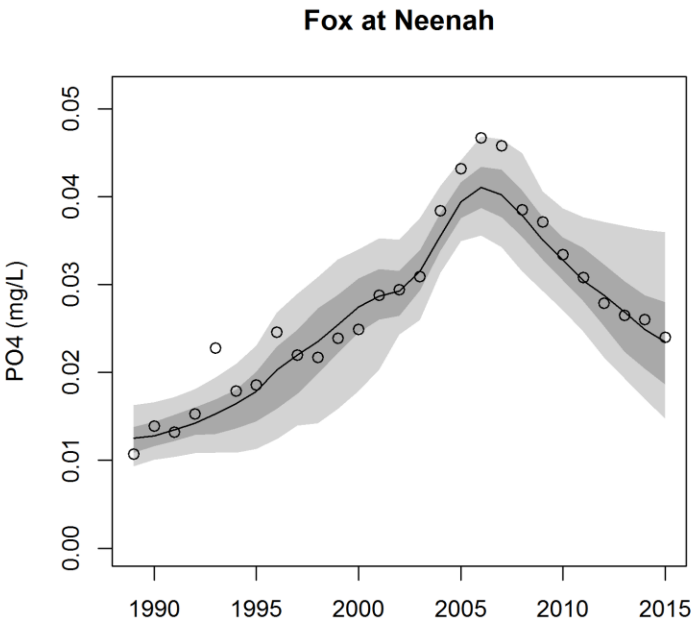

Fox at Neenah
NO3 (tons/yr)
0
500
1500
2500
3500
1990
1995
2000
2005
2010
2015

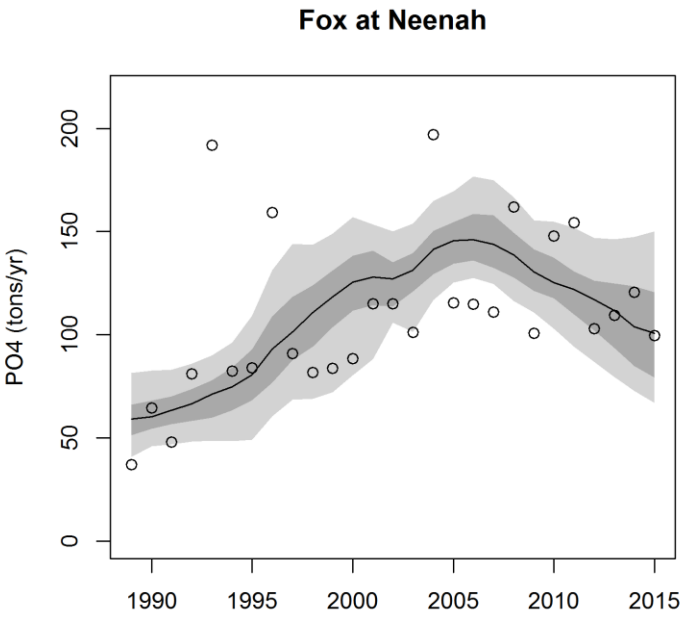

Fox at Neenah
Winter
2102 cfs
3614 cfs
5455 cfs
Spring
1976 cfs
6236 cfs
12795 cfs
Summer
1314 cfs
4193 cfs
13424 cfs
Fall
1294 cfs
3452 cfs
7579 cfs
NO3 (mg/L)
0.0
0.2
0.4
0.6
1990
1995
2000
2005
2010
2015

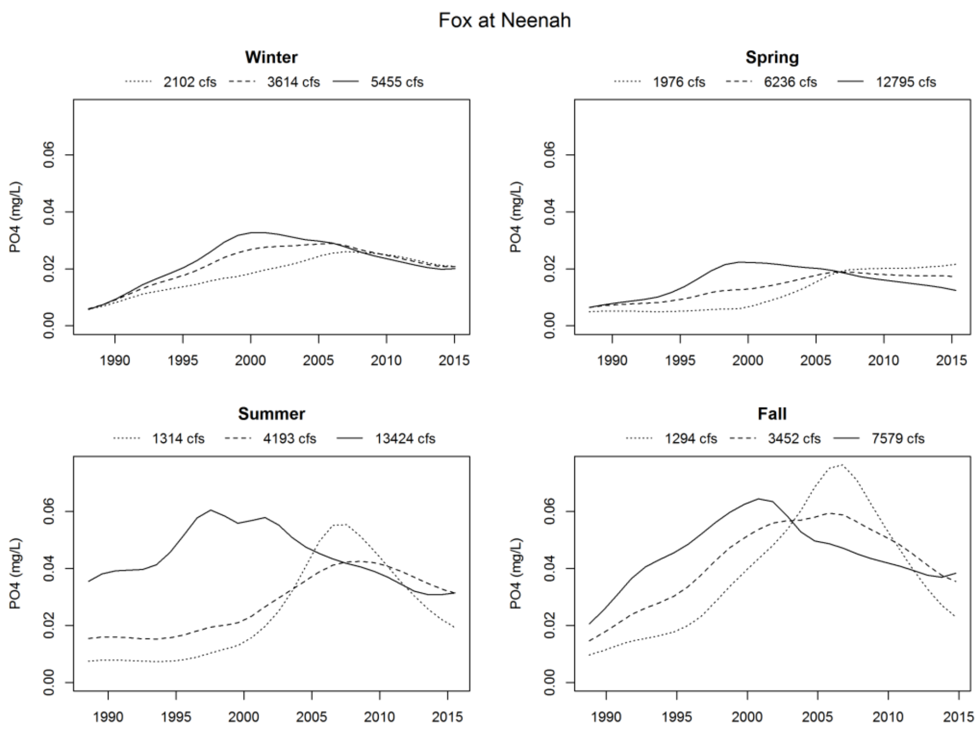
Phosphate


Fox at Neenah
PO4 (mg/L)
0.001
0.002
0.005
0.010
0.020
0.050
0.100
1990
1995
2000
2005
2010
2015

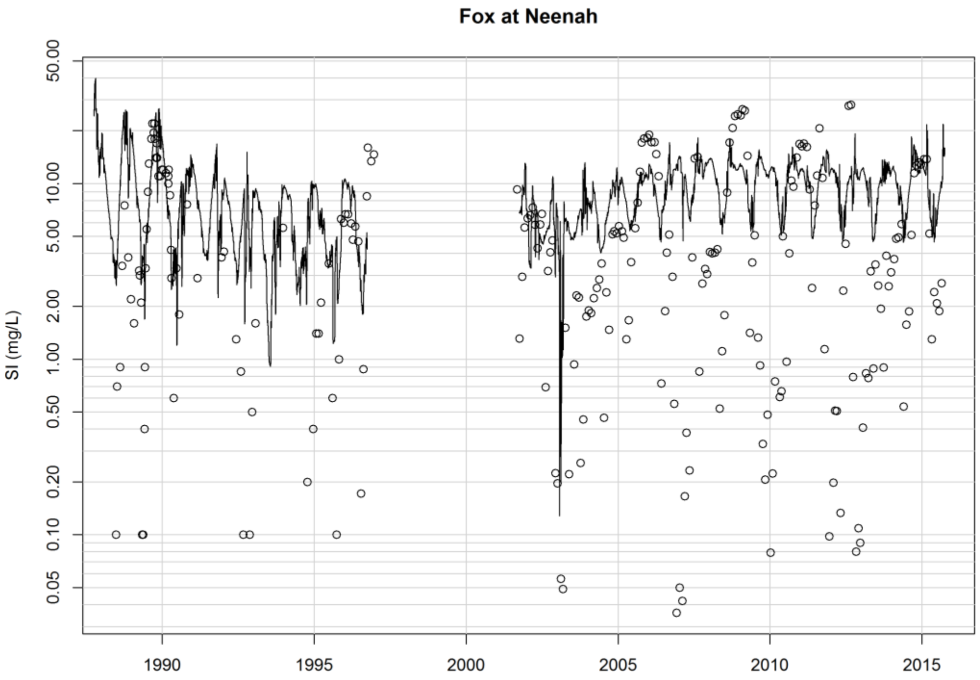

Fox at Neenah, Orthophosphate
Model is WRTDS Flux Bias Statistic0.0334

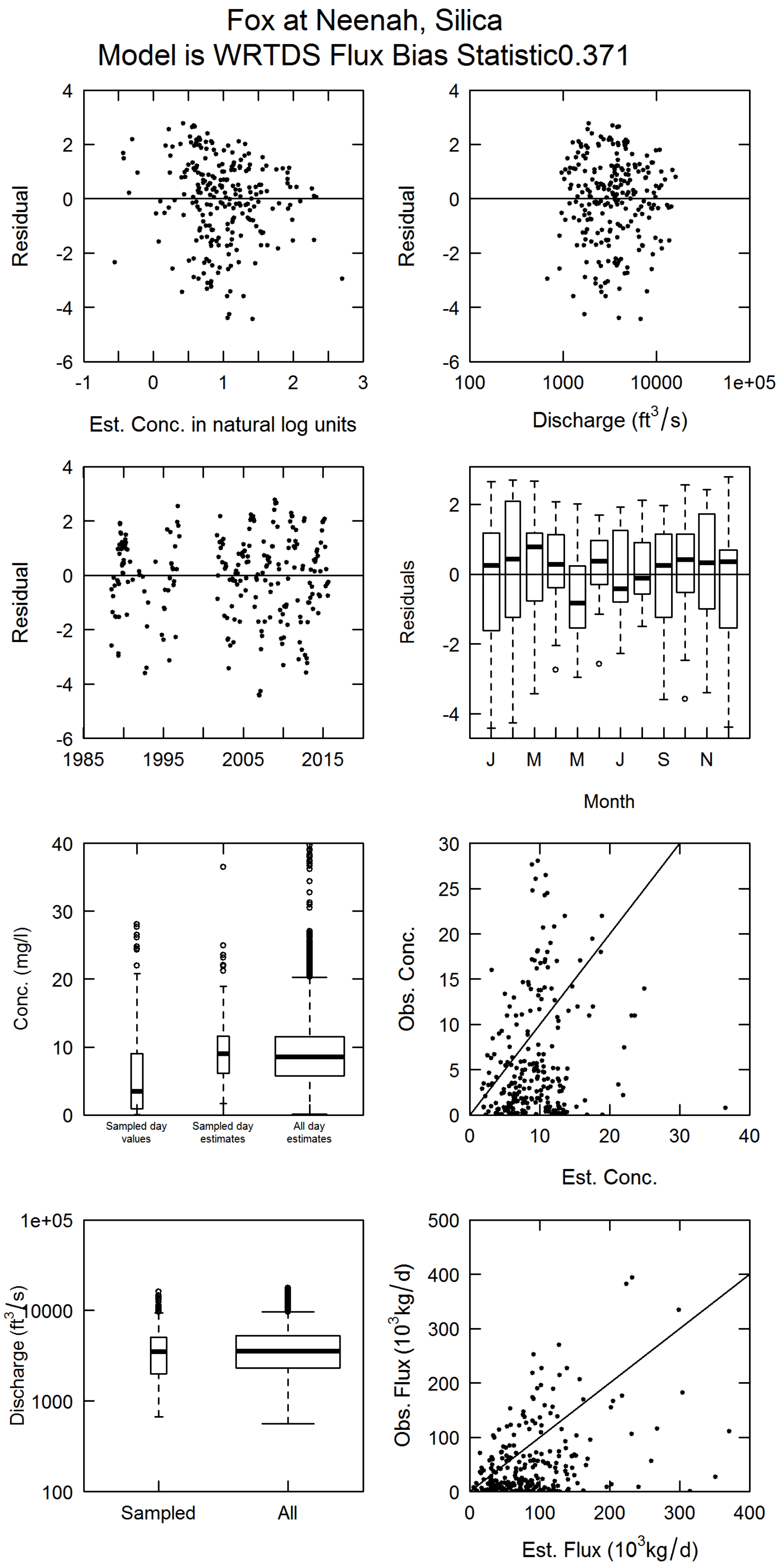

Fox at Neenah
PO4 (mg/L)
0.00
0.01
0.02
0.03
0.04
0.05
1990
1995
2000
2005
2010
2015

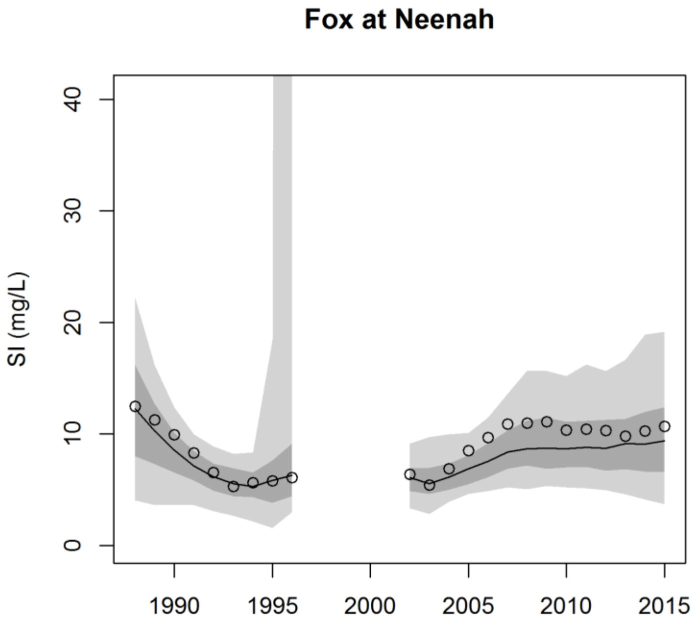

Fox at Neenah
PO4 (tons/yr)
0
50
100
150
200
1990
1995
2000
2005
2010
2015

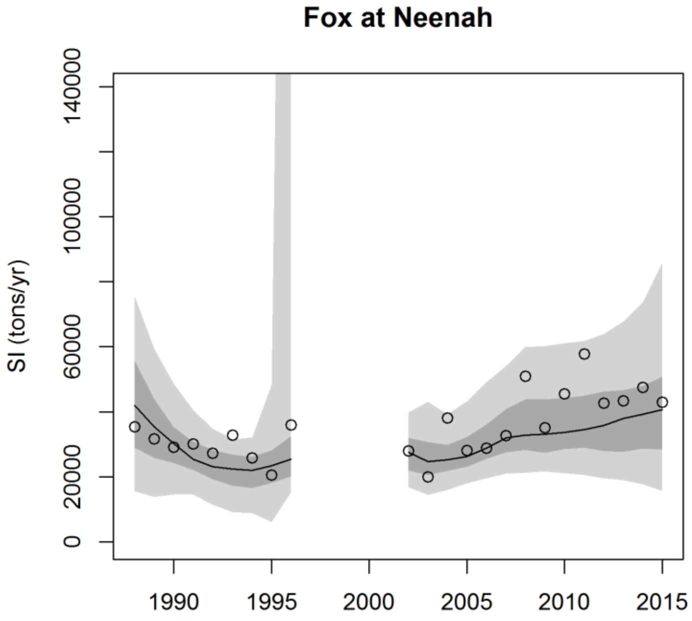

Fox at Neenah
Winter
2102 cfs
3614 cfs
5455 cfs
Spring
1976 cfs
6236 cfs
12795 cfs
Summer
1314 cfs
4193 cfs
13424 cfs
Fall
1294 cfs
3452 cfs
7579 cfs
PO4 (mg/L)
0.00
0.02
0.04
0.06
1990
1995
2000
2005
2010
2015

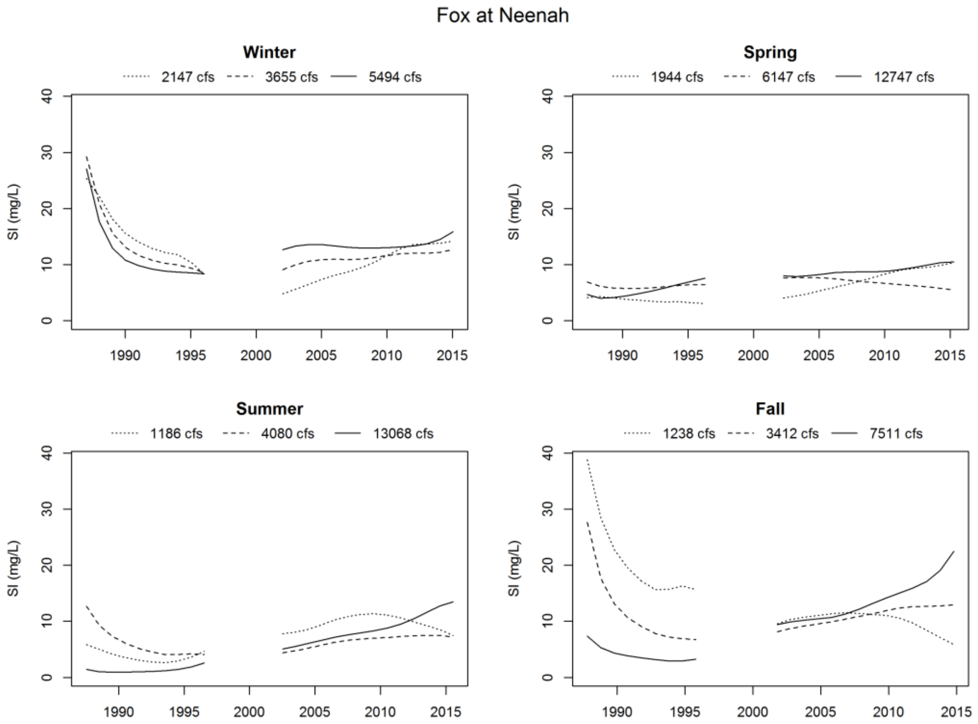
Silica


Fox at Neenah
SI (mg/L)
0.05
0.10
0.20
0.50
1.00
2.00
5.00
10.00
50.00
1990
1995
2000
2005
2010
2015

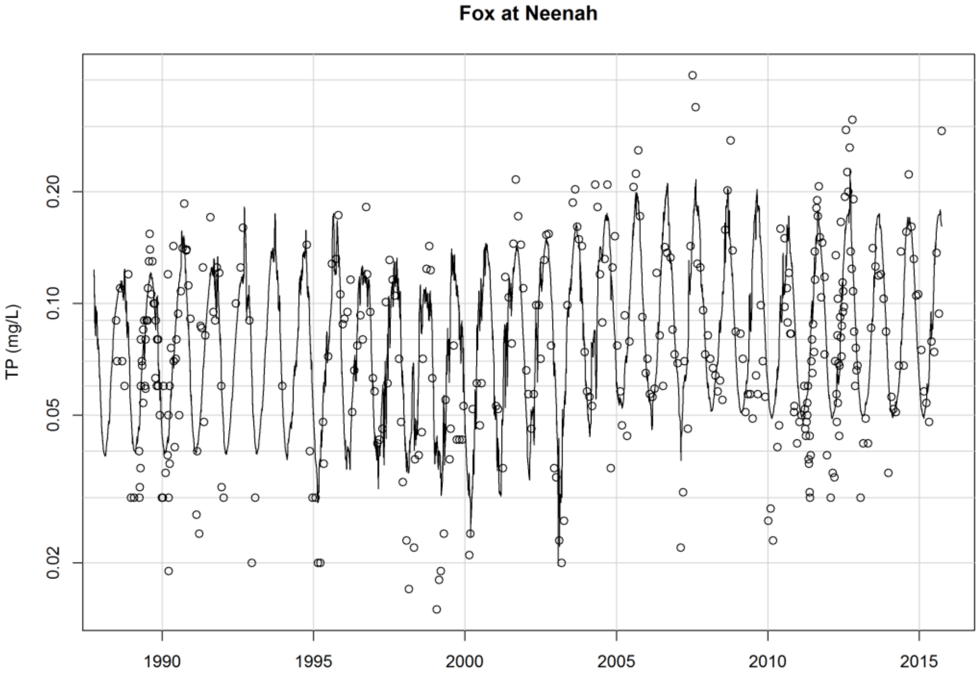

Fox at Neenah, Silica
Model is WRTDS Flux Bias Statistic0.371
Residual
4
2
0
-2
-4
-6
-1
0
1
2
3
Est. Conc. in natural log units
Residual
100
1000
10000
1e+05
Discharge (ft^3/s)
Residual
1985
1995
2005
2015
Residuals
J
M
M
J
S
N
Month
Conc. (mg/l)
40
30
20
10
0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
30
25
20
15
10
5
0
0
10
20
30
40
Est. Conc.
Discharge (ft^3/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (10^{3}kg/d)
500
400
300
200
100
0
0
100
200
300
400
Est. Flux (10^{3}kg/d)

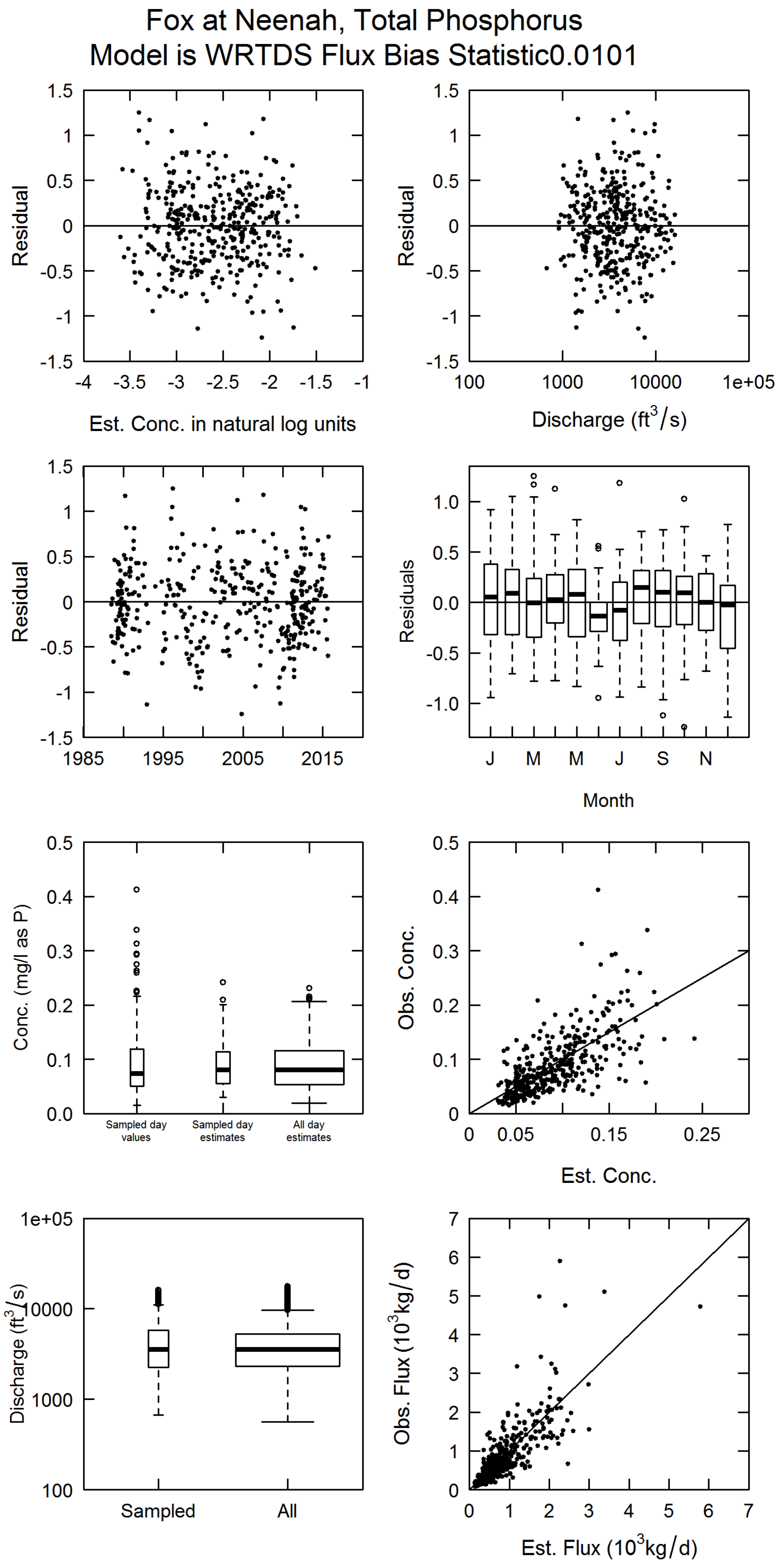

Fox at Neenah
SI (mg/L)
0
10
20
30
40
1990
1995
2000
2005
2010
2015

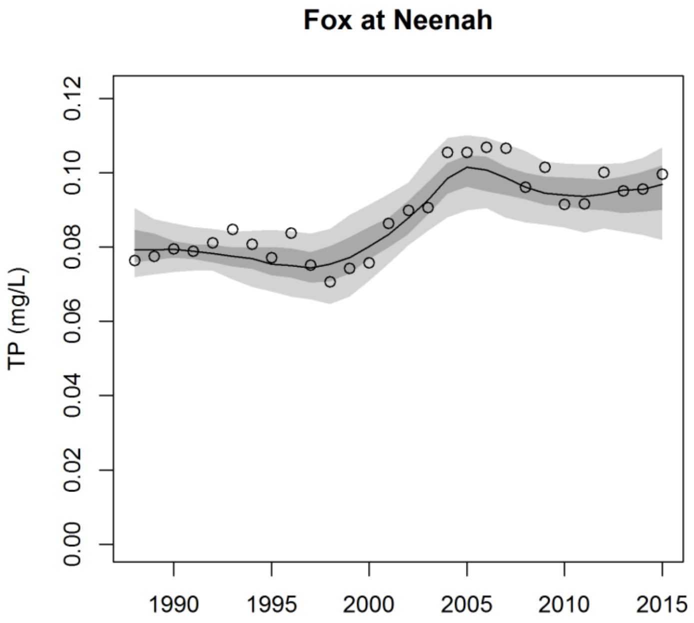

Fox at Neenah
SI (tons/yr)
0
20000
60000
100000
140000
1990
1995
2000
2005
2010
2015

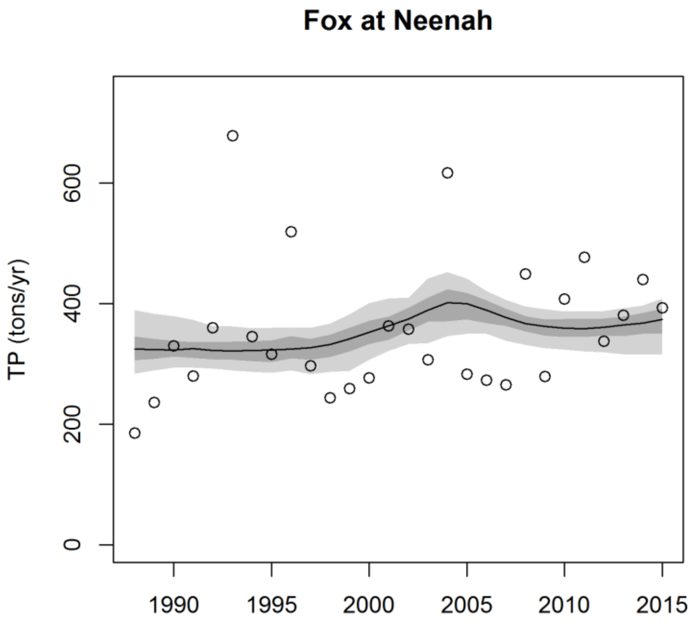


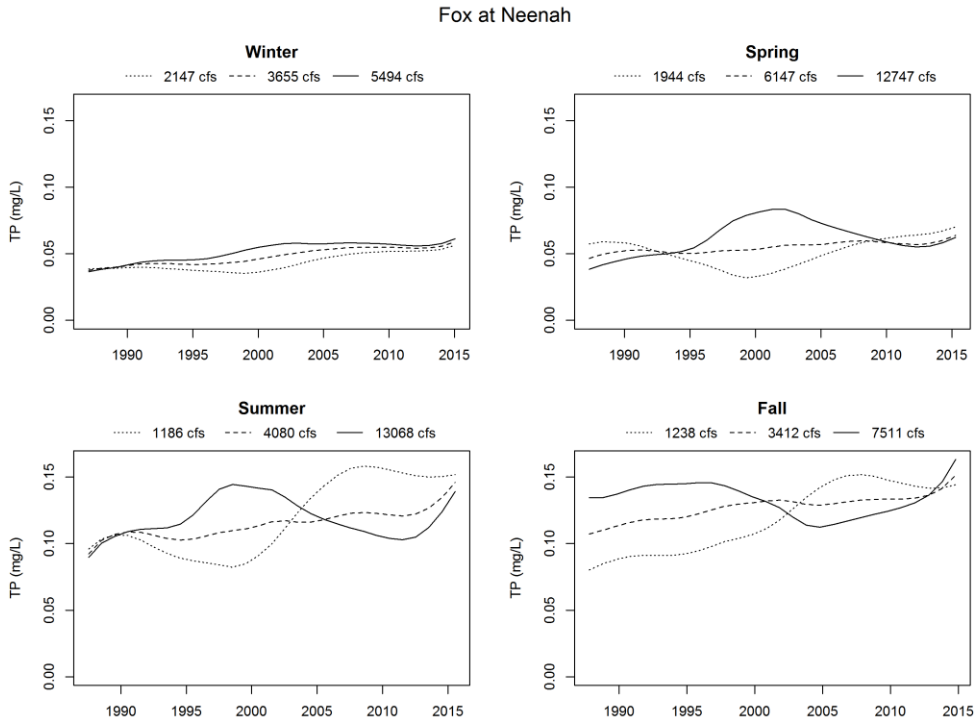
Total Phosphorus



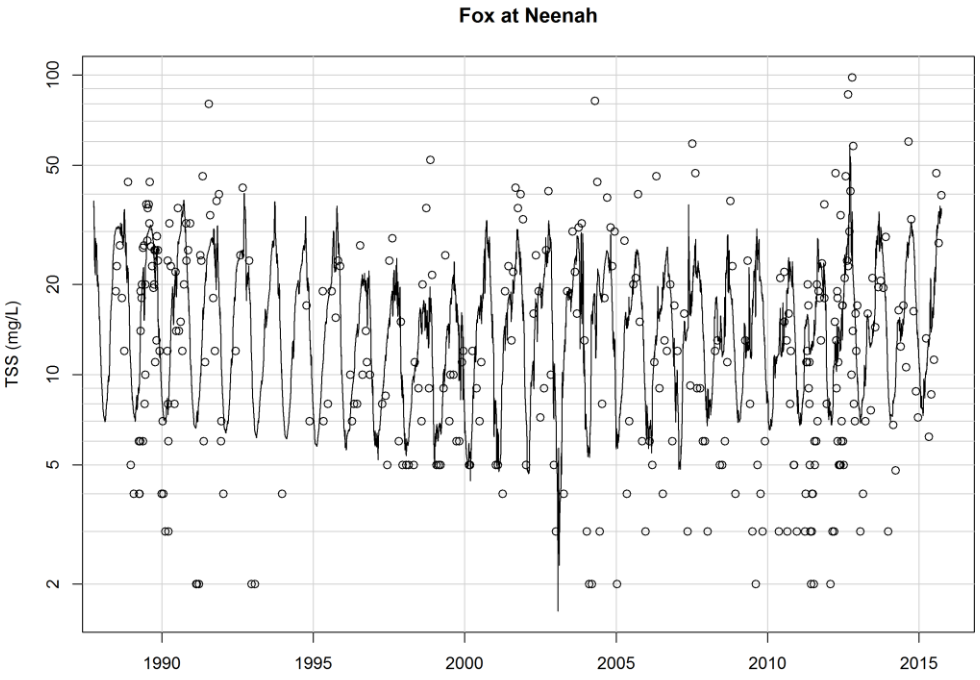

Fox at Neenah, Total Phosphorus
Model is WRTDS Flux Bias Statistic0.0101
Residual
Est. Conc. in natural log units
-4
-3.5
-3
-2.5
-2
-1.5
-1
1.5
1
0.5
0
-0.5
-1
-1.5
Residual
Discharge (ft3/s)
100
1000
10000
1e+05
Residual
1985
1995
2005
2015
Residuals
1.0
0.5
0.0
-0.5
-1.0
J
M
M
J
S
N
Month
Conc. (mg/l as P)
0.5
0.4
0.3
0.2
0.1
0.0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
Est. Conc.
0
0.05
0.15
0.25
Discharge (ft3/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (103kg/d)
Est. Flux (103kg/d)
0
1
2
3
4
5
6
7

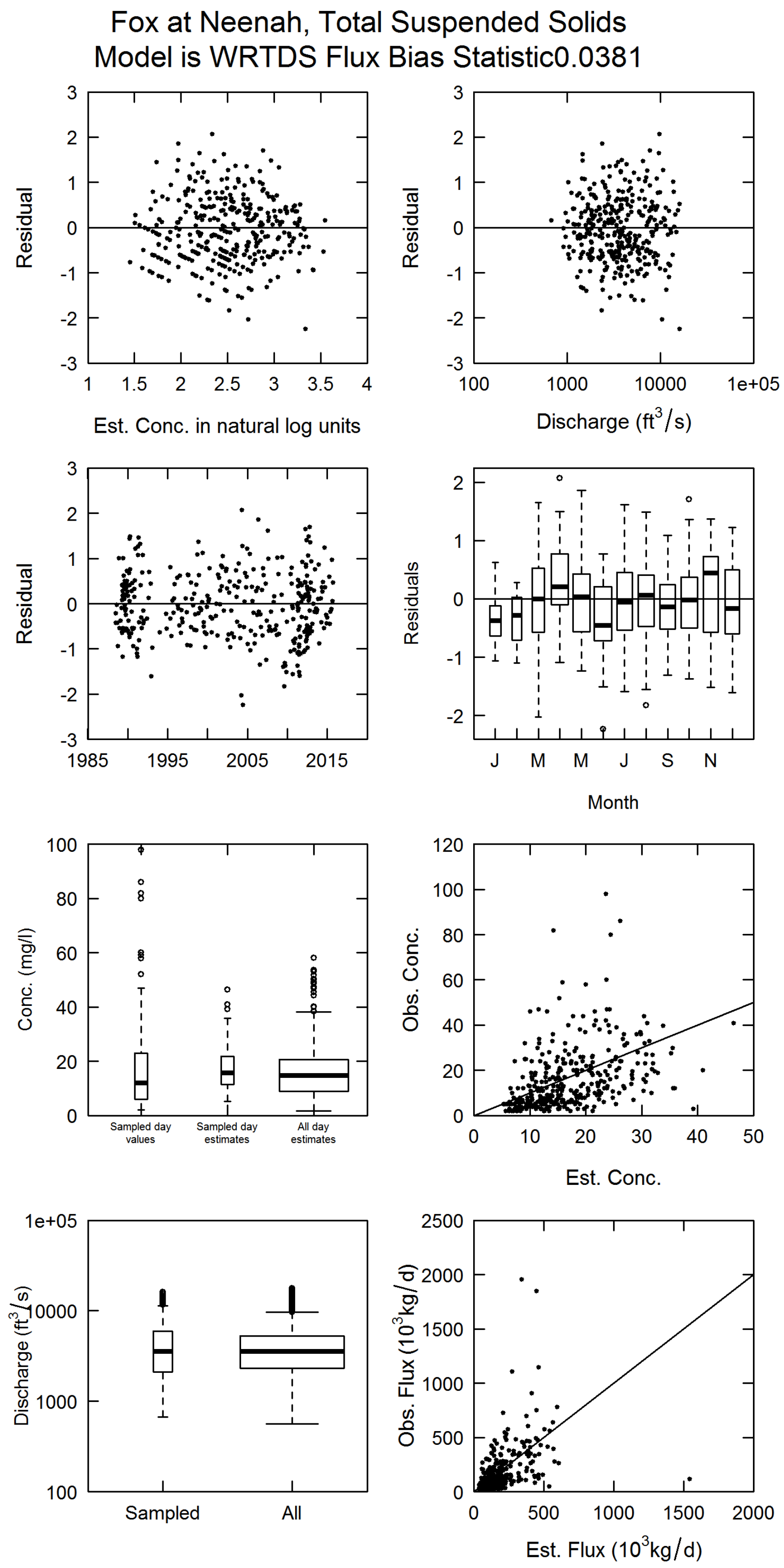

Fox at Neenah
TP (mg/L)
0.00
0.02
0.04
0.06
0.08
0.10
0.12
1990
1995
2000
2005
2010
2015

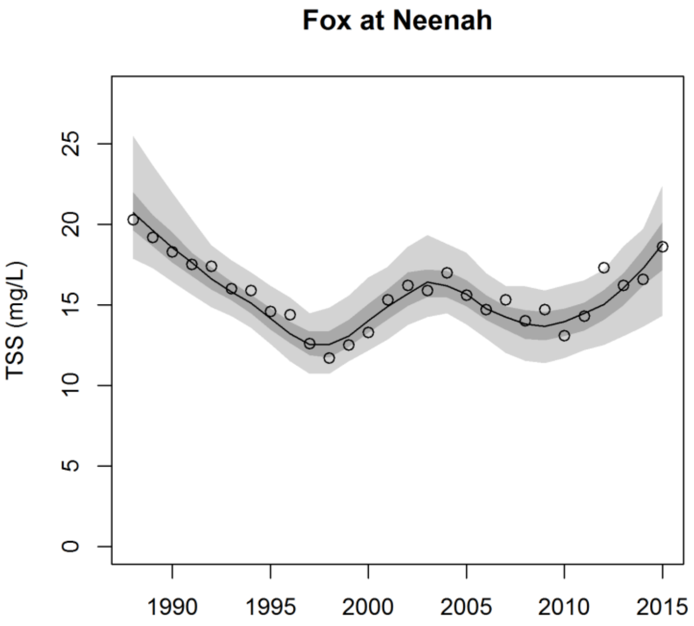

Fox at Neenah
TP (tons/yr)
0
200
400
600
1990
1995
2000
2005
2010
2015

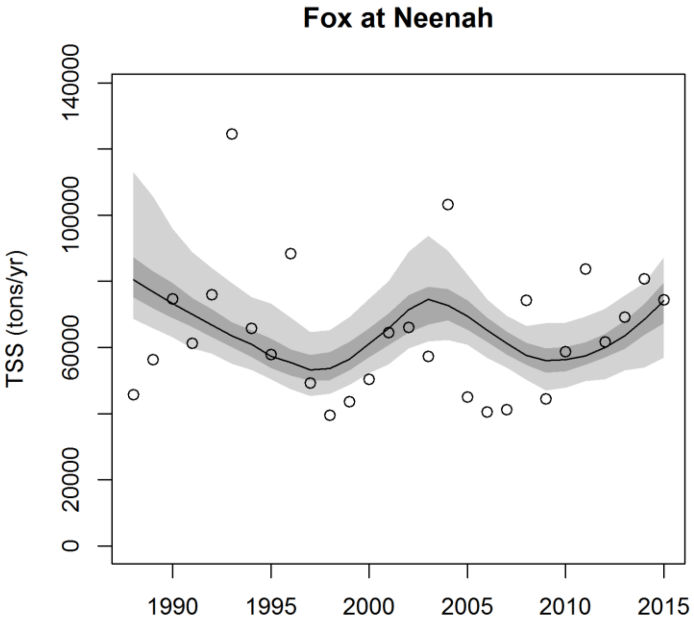


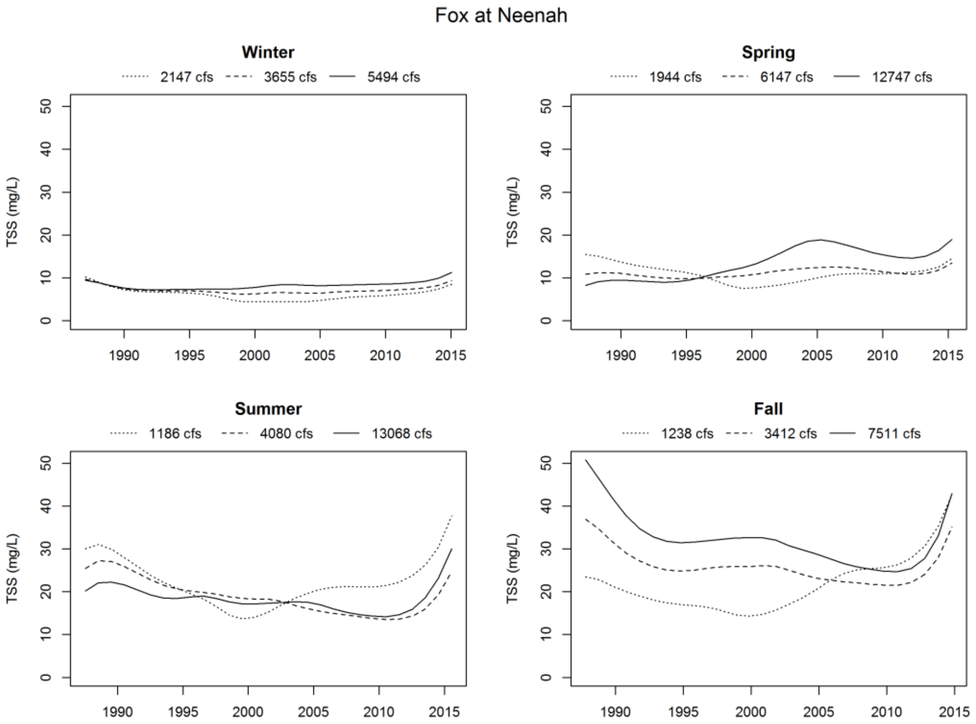
Total Suspended Solids




Fox at Neenah, Total Suspended Solids
Model is WRTDS Flux Bias Statistic0.0381
Residual
3
2
1
0
-1
-2
-3
1
1.5
2
2.5
3
3.5
4
Est. Conc. in natural log units
Residual
100
1000
10000
1e+05
Discharge (ft^3/s)
Residual
1985
1995
2005
2015
Residuals
J
M
M
J
S
N
Month
Conc. (mg/l)
100
80
60
40
20
0
Sampled day values
Sampled day estimates
All day estimates
Obs. Conc.
120
0
10
20
30
40
50
Est. Conc.
Discharge (ft^3/s)
1e+05
10000
1000
100
Sampled
All
Obs. Flux (10^3kg/d)
2500
2000
1500
1000
500
0
Est. Flux (10^3kg/d)


Fox at Neenah
TSS (mg/L)
0
5
10
15
20
25
1990
1995
2000
2005
2010
2015


Fox at Neenah
TSS (tons/yr)
0
20000
60000
100000
140000
1990
1995
2000
2005
2010
2015


Fox at Neenah
Winter
2147 cfs
3655 cfs
5494 cfs
Spring
1944 cfs
6147 cfs
12747 cfs
Summer
1186 cfs
4080 cfs
13068 cfs
Fall
1238 cfs
3412 cfs
7511 cfs
TSS (mg/L)
0
10
20
30
40
50
1990
1995
2000
2005
2010
2015