Final Report for TMDL Monitoring of Arcadia Lake and the Arcadia Lake Watershed

FY 20/21 Section §106 I-006400-19 Project 04

OKLAHOMA
Water Resources Board

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Table of Contents

Table of Figures

Table of Tables

EXECUTIVE SUMMARY

Arcadia Lake, Oklahoma Waterbody ID (WBID) OK520710020020_00, is a 1,676-acre, eutrophic, urban lake located in the City of Edmond, Oklahoma. Arcadia Lake currently appears on the 303(d) list for several impairments including turbidity and chlorophyll *a*, therefore a Total Maximum Daily Load (TMDL) must be developed to further investigate these impairments. A three-year study was conducted to gain an understanding of water quality for both the lake and watershed for use in the TMDL development process.

The intent of this project was to collect water quality data to verify waterbody impairment and provide data in support of TMDL development for Arcadia Lake. Data collection was designed to characterize the watershed and loading into the lake to assist in identifying the source(s) and extent of impairment to the lake. The bullets below highlight key findings from the study.

- Spring Creek and Deep Fork, the two major inputs into Arcadia Lake, are both considered nutrient threatened and will be included on the 2022 Integrated Report
- 63% of chlorophyll *a* concentrations exceeded the OWQS 10 mg/m^3, indicating the lake is still not meeting the Public and Private Water Supply beneficial use
- 33% of in-lake turbidity concentrations exceeded the water quality standard of 25 NTU, indicating the lake is still not meeting the Fish and Wildlife Propagation beneficial use

The Oklahoma Water Resources Board (OWRB), in service to all Oklahomans, has a long-standing commitment to monitor lake water quality and guide actions to better manage Oklahoma lake ecosystems. This monitoring initiative provides a perspective on the condition of Arcadia Lake and its watershed allowing scientists, lake managers, and other decision makers to work together to protect this valuable lake ecosystem.

INTRODUCTION

Protecting and improving the water quality of Oklahoma's lakes is vital for healthy lake ecosystems, enhancing the quality of life for Oklahomans through access to safe drinking water and recreational opportunities, and providing economic benefits for the state. Quality of life and economic benefits are both directly connected to maintaining healthy lake ecosystems.

Arcadia Lake was constructed as a cooperative effort between the City of Edmond and the U.S. Army Corps of Engineers and opened in 1987. It has a surface area of approximately 7.4 km^2 (2.8 mi^2) with about 41.8 km (26 mi) of shoreline at a normal elevation of 1006 ft (NGVD88). Arcadia Lake is a 1,676-acre, hypereutrophic urban reservoir located in central Oklahoma that serves as the primary drinking water source for the City of Edmond's estimated population of 94,054 (US Census, 2019), along with serving as flood control for the Deep Fork of the North Canadian River (Deep Fork), providing fisheries and wildlife habitat, and offering recreation in the lake and along its adjacent landscape. Although a relatively young reservoir, Arcadia Lake has been identified as a eutrophic system in previous studies mainly due to excess nutrients and other pollutants being delivered from the urban land use in the watershed (OWRB, 2000).

The lake currently appears on Oklahoma's Impaired Waters List, pursuant to requirements of the Federal Clean Water Act (CWA), Section 303(d). The 2020 Oklahoma Integrated Report lists Arcadia Lake as not meeting designated beneficial uses for Public and Private Water Supply (PPWS) and Fish and Wildlife Propagation (FWP). Based on water quality criteria, chlorophyll *a* is the cause of Arcadia Lake's PPWS impairment and turbidity is causing the impairment for FWP. Consistent with the state's listing prioritization and CWA requirements, a TMDL must be developed to further investigate these impairments. A TMDL analysis and document identifies the amount of pollutant which a waterbody can assimilate without exceeding the established water quality standard for that particular pollutant. TMDLs determine pollutant numeric targets and allocate load reductions necessary to achieve and maintain a waterbody's designated beneficial uses.

The Oklahoma Water Resources Board (OWRB), in collaboration with the Oklahoma Department of Environmental Quality (ODEQ), conducted monitoring on Arcadia Lake to verify water quality impairments in the lake and collect data for use in the TMDL development process. In order to accomplish this task, in-lake and watershed data collection efforts were undertaken to represent both Arcadia Lake and its watershed. Characterization of the watershed is a key to understanding pollutant inputs to the lake and is reflected in the parametric coverage that was chosen. Data needs, such as parameters of interest, were identified for inclusion through partner discussions prior to the project getting underway. All data were collected in accordance with the approved Quality Assurance Project Plan (QAPP) (OWRB 2018) for this project with sample analysis performed at the Oklahoma Department of Environmental Quality State Environmental Laboratory (ODEQ-SEL). Oklahoma's Use Support Assessment Protocols (USAP)(OAC, 2020a; OAC, 2020b) were followed for the assessment of impairment status in both lake and streams. Where USAPs do not exist, acceptable scientific methods, such as those outlined in Oklahoma's Continuing Planning Process (ODEQ, 2012), were followed to assess water quality.

METHODS

Study Design

This project was designed to collect the necessary data to verify water quality impairments and support TMDL development for Arcadia Lake. The project included 5 in-lake sites and 3 watershed sites: Deep Fork, Wynn Creek, and Spring Creek (Figure 1). The study was conducted from February 2018 to December 2020. Field sampling occurred monthly for both the watershed and in-lake monitoring locations. For the lake, additional biweekly sampling trips occurred annually during the growing season (May-September). To characterize watershed inputs, additional samples were collected during targeted rainfall events (TREs). A TRE was identified as any event where the water level was above seasonal baseflow due to rainfall in the watershed.

Planning activities, including site selection and installation of equipment, were conducted from mid-2017 through early 2018. During the planning process for this project, meetings were held to determine both data needs and availability for verifying impairment status and for future modeling efforts. Data collection was designed to assist in identifying sources and extent of impairments to Arcadia Lake as well as provide data to be used for load allocation and modeling approaches for use by ODEQ staff. Monitoring activities were carried out from February 2018 through the end of December 2020. These activities included regular and targeted site visits to collect water samples, discharge measurements and maintenance of deployed equipment.

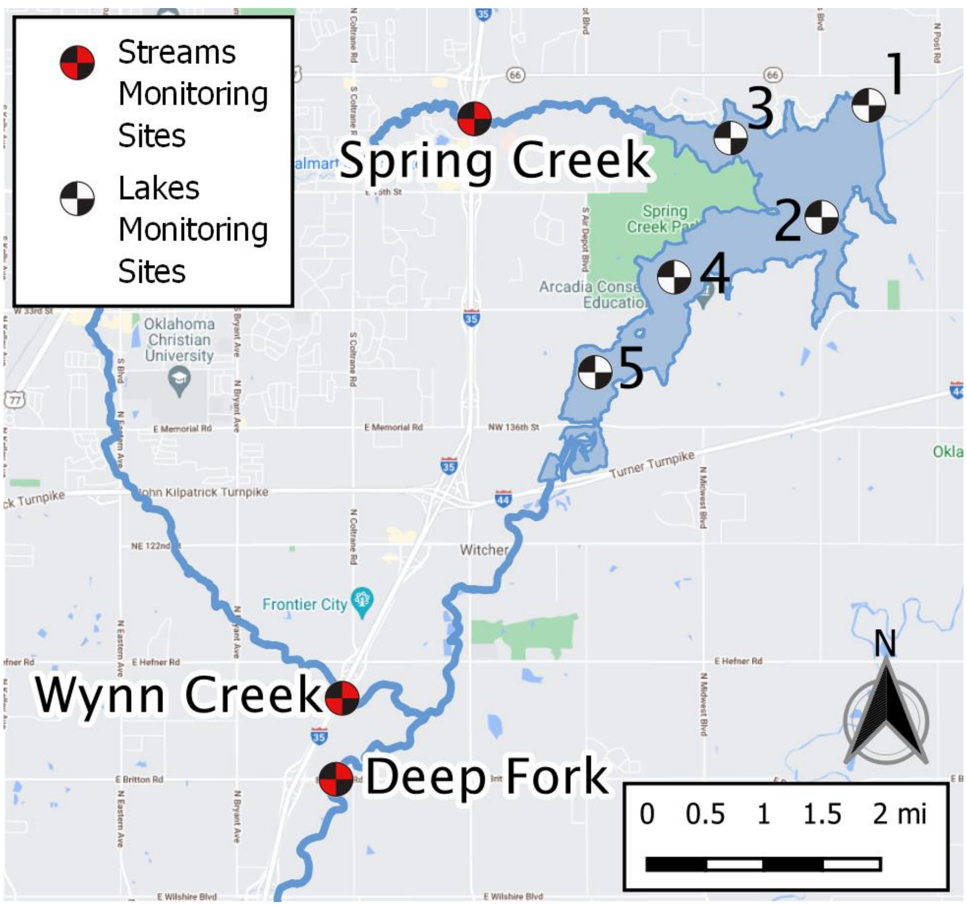


Figure 1. Arcadia watershed and in-lake monitoring locations.

Lake Site Descriptions

Five locations in the reservoir were sampled throughout the study period (Figure 1). These sites represent the lacustrine zone (Site 1), transitional zones (Site 2 and Site 4), and the riverine zones (Site 3 and Site 5) of the reservoir. Site 5 represents the Deep Fork arm and Site 3 represents the Spring Creek arm. Briefly, these zones are defined by Cooke et al. (2005) as:

- Riverine zone is a narrow portion of the reservoir nearest the lotic input typified by potentially high flow with greater suspended solids, nutrients, and turbidity. Photosynthesis is generally light limited in this zone.
- Transition zone is broader and deeper than the riverine zone where reduced flow causes sediments to settle out and water becomes clearer. Photosynthesis is predicted to be highest in this zone.
- Lacustrine zone is a broader, deeper, and more lake-like section of the reservoir nearest the dam where the water tends to be clearest, nutrient concentrations are lowest, and photosynthesis is nutrient limited.

The in-lake sites generally followed these definitions with Site 1 being the deepest site (usually ~12 to 15 m), located in the former river channel nearest the dam. Site 2 was the second deepest site (~8-11 m), located in the approximate middle of lake. Site 4 was a hybrid transitional-riverine site with a moderate depth (5-8 m) located between the transitional Site 2 and the riverine Site 5. Lastly, Sites 3 and 5 were considered riverine sites. Site 3 had a similar depth to Site 4. Site 5 was the shallowest site (<1 m to ~4 m) with possibly the most influence from an inflow stream. These sites were established at the outset of the Beneficial Use Monitoring Program (BUMP) and have been maintained in the program since Fall 1998 and during a previous study completed by the OWRB (OWRB, 2000).

Watershed Site Descriptions

The watershed monitoring effort of this project focused on the three streams, including two direct inputs into Arcadia Lake—Spring Creek (segment WBID OK520710020030_00) and Deep Fork (segment WBID OK520710020060_00)—and Wynn Creek (segment WBID OK520710020050_00), a tributary of the Deep Fork. These waterbodies are representative of the three Arcadia Lake watershed subbasins, Lake Arcadia-Deep Fork subbasin, White Turkey Creek-Deep Fork subbasin, and Headwaters Deep Fork subbasin (Figure 2). All are upstream of the lotic-lacustrine transition of Arcadia Lake. Watershed and subbasin area, as well as stream length, were collected by analyzing shapefiles available from the United States Geological Survey's (USGS) National Hydrography Dataset (USGS, 2021).

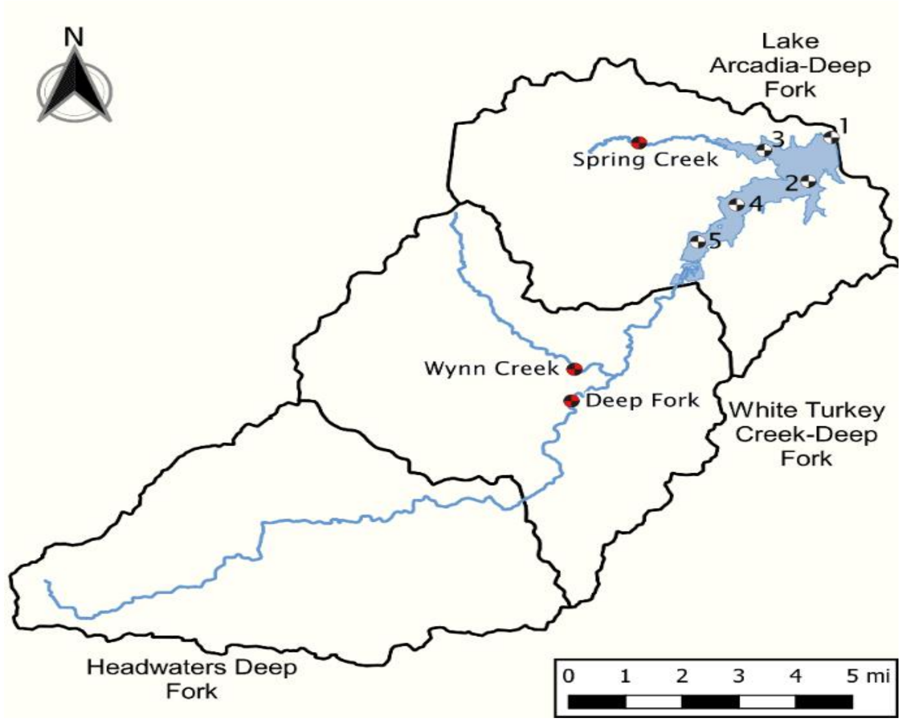


Figure 2. Watershed Subbasins

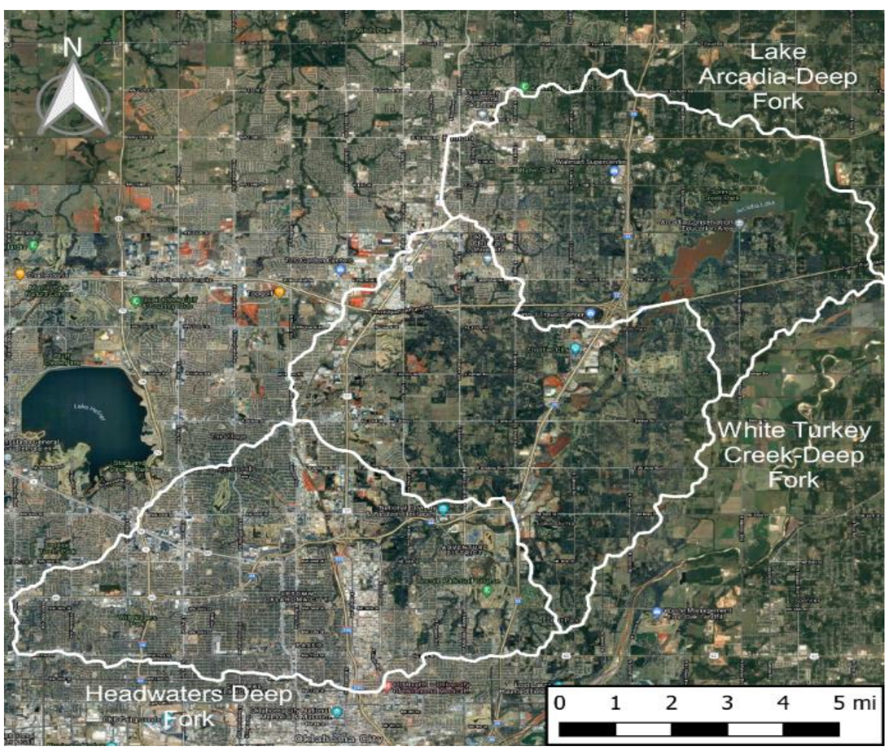


Figure 3. Satellite view of Subbasins

The Spring Creek monitoring station and all in-lake monitoring locations were in the Lake Arcadia-Deep Fork subbasin, HUC 111003030103 (Figure 2). The Lake Arcadia-Deep Fork subbasin is an 81.1 km^2 (31.3 mi^2) drainage basin consisting of Arcadia Lake and its tributaries. The Lake Arcadia-Deep Fork subbasin encompasses the Deep Fork downstream of Interstate-44 to the Arcadia Lake dam (Figure 3). The lake was formed by impounding the Deep Fork near its confluence with Spring Creek and is the main hydrological feature of the subbasin. There are approximately 122 km (75.6 mi) of streams within the subbasin, however several kilometers of streams, specifically the Deep Fork and Spring Creek, are encompassed within Arcadia Lake (Figure A3). Spring Creek is located, in its entirety, within this subbasin and runs for approximately 9.87 km (6.13 mi) from its source in Edmond, OK, to its mouth at the Deep Fork near the Arcadia Lake dam. It should be noted, however, that about half of the Spring Creek stream length is encompassed within Arcadia Lake. The Deep Fork runs approximately 7.78 km (4.84 mi) in the subbasin, all encompassed within Arcadia Lake, with the upstream portion of the Deep Fork in the subbasin constituting a riverine zone of the lake (USGS, 2021).

The Wynn Creek and Deep Fork monitoring locations are in the White Turkey Creek-Deep Fork subbasin, HUC 111003030102 (Figure 2). The White Turkey Creek-Deep Fork subbasin is a 90.0 km^2 (34.7 mi^2) drainage basin consisting of the Deep Fork and its tributaries from Interstate-35 downstream to Interstate-44 (Figure 3). There are approximately 127 km (79 mi) of streams within the subbasin. The Deep Fork runs for approximately 12.5 km (7.78 mi) through the subbasin and is the main hydrological feature. Wynn Creek is located, in its entirety, in the White Turkey Creek-Deep Fork subbasin and runs for approximately 9.89 km (6.15 mi) from its source in Edmond, OK, to its confluence with the Deep Fork near Sooner Road between Hefner Road and Britton Road in Oklahoma City (USGS, 2021).

The Headwaters Deep Fork subbasin, HUC 111003030101, is an 89 km^2 (34 mi^2) drainage basin consisting of the Deep Fork and its tributaries from its source in western Oklahoma City downstream to Interstate-35 (Figure 3). There are approximately 72 km (45 mi) of streams within the subbasin, with the Deep Fork accounting for approximately 20 km (12 mi). The Deep Fork runs for approximately 19.7 km (12.3 mi) through the subbasin and is the main hydrological feature in the subbasin. Due to the highly urbanized nature of the subbasin, many stretches of streams within the subbasin, including the Deep Fork, are channelized for stormwater management (USGS, 2021).

Stream order was determined using the Strahler (1952) method with waterbody data available from the National Hydrography Dataset (USGS, 2021). The sampling station location for both Wynn Creek and Deep Fork are both located in fourth-order segments of their respective waterbodies. Downstream of its confluence with Wynn Creek, however, the Deep Fork is a fifth-order stream, and remains as such throughout the rest of its course up to the dam at Arcadia Lake. Spring Creek is a third-order stream at the location of its sampling station and remains a third-order stream until its confluence with the Deep Fork within Arcadia Lake. Stream slope for all three streams monitoring locations was calculated at each individual station using the USGS StreamStats tool, version 4.6.2 (USGS, 2016). The slope of the Deep Fork, Wynn Creek, and Spring Creek at their study monitoring locations were 13.0 feet per mile, 17.1 feet per mile, and 23.2 feet per mile, respectively.

The individual monitoring locations were strategically chosen along each stream to be representative of the load inputs from the three subbasins that comprise the Arcadia Lake watershed. Additionally, each monitoring location includes examples of all land-use types in the study which is an important consideration in the development of watershed models (Figure 2 and Figure 5). The location of each site was established to be close enough to Arcadia Lake to capture all major input loads to the lake, but far enough upstream as to avoid backwater effects from a rise in lake water level. Accessibility of each monitoring location was also essential for field staff to safely access the monitoring locations even in times of excessive rainfall. Each monitoring location was established at a bridge crossing to allow for staff to mount equipment and work from the bridge during sampling events.

Water Quality Indicators

To characterize the physical, chemical, and biological condition of Arcadia Lake and its watershed, a group of water quality indicators were selected and sampled at each in-lake site and watershed monitoring location (Table 1). During each in-lake sampling event, all water quality parameters listed in Table 1 were collected, asterisks indicate which parameters are in-lake specific. For the watershed monitoring locations, in-situ parameters varied by the type of collection needed, but all other field, general chemistry and biological parameters remained consistent. The only exception was Ammonia-Nitrogen (NH3) which was added to the watershed monitoring locations after reevaluating parameters midway throughout the project period. This section describes each water quality indicator and their associated impact to overall water quality.

Table 1. Water Quality Parameters

In-situ Parameters	Field Parameters	General Chemistry Parameters	Biological Parameters
Dissolved Oxygen (DO)	Turbidity	Total Kjeldahl Nitrogen (TKN)	Chlorophyll a
% DO Saturation	Total Alkalinity	Total Phosphorus (TP)	Zooplankton*
pH	Total Hardness	Total Suspended Solids (TSS)	Phytoplankton*
Water Temperature	Secchi Disk Depth*	Nitrate-Nitrogen (NO3)	
Specific Conductance		Nitrite-Nitrogen (NO2)	
Salinity		Ammonia-Nitrogen (NH3)	
Total Dissolved Solids		Dissolved Orthophosphate (DOP)*	
Oxidation-Reduction Potential (ORP)*		Total Organic Carbon (TOC)*	

*Indicates analytes measured only in-lake water samples

Dissolved Oxygen

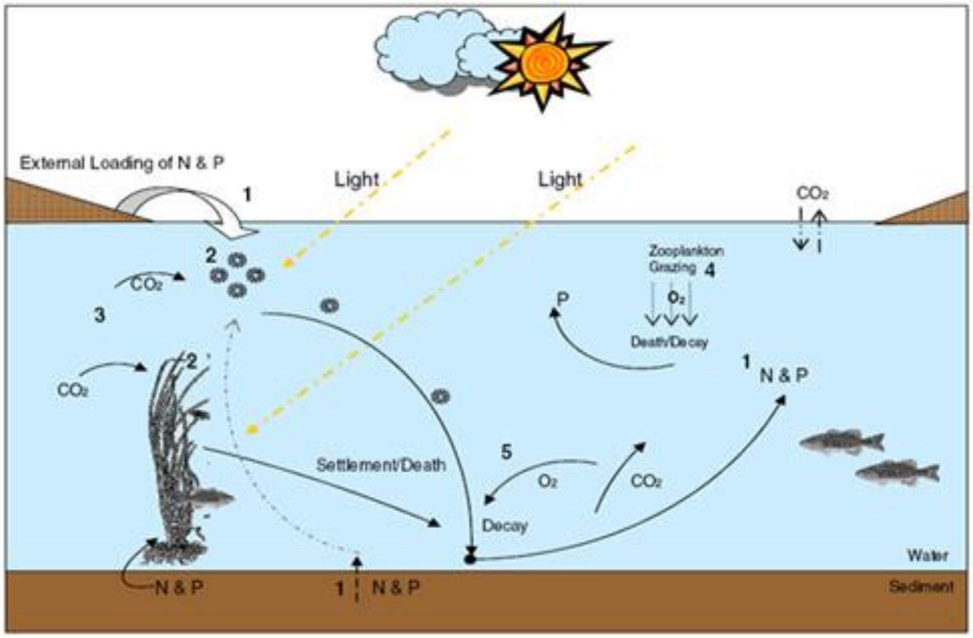
Dissolved oxygen (DO) is fundamental to the lake ecosystem and is essential to all aquatic organisms. Thus, the dynamics and distribution of oxygen within lakes is extremely important. Oxygen is supplied to the lake from the atmosphere and photosynthesis and distributed throughout the lake via diffusion and physical mixing. Respiration and decomposition processes are the key drivers of oxygen consumption within the lake. Photosynthesis is a light reaction and therefore only occurs during the daylight hours whereas, respiration and decomposition occur at all times. This difference produces daily (diurnal) variations in DO concentrations. At night dissolved oxygen concentrations steadily decline due to ongoing respiration and decomposition without photosynthesis to replenish the oxygen.

Eutrophication aggravates typical lake DO dynamics; for example, abundant algal biomass can increase oxygen concentrations via photosynthesis such that the oxygen concentration becomes saturated. Yet, conversely the additional biomass accelerates the rate of oxygen depletion due to decomposition in the deeper areas of the lake, especially when the lake is stratified in the summer season (Water on Web, 2004). This feedback loop increases the opportunity for summer fish kills in eutrophic and hypereutrophic lakes. Fish kills are most apt to occur at times when daytime photosynthesis is diminished due to clouds and calm winds minimize the entrainment of oxygen from the atmosphere thus, oxygen production at the lake surface is reduced. At the same time, the generous amounts of organic material hasten respiration and decomposition processes, which deplete the lake's oxygen. Through the combination of these events oxygen in the lake can be consumed causing the lake to become hypoxic (2 mg/L DO) and/or anoxic (0 mg/L DO) causing a direct impact of the lake's fish community (Water on Web, 2004).

Nutrients, Phosphorus and Nitrogen

Phosphorus and nitrogen are two essential nutrients necessary for all aquatic life. A fundamental ecological process in lakes is nutrients supporting algal growth and algae providing the foundation for the overall lake food web. Phosphorus and nitrogen are present within waterbodies, in various organic and inorganic forms as well as dissolved and particulate forms. Phosphorus and nitrogen can come from natural sources through physical, chemical, and biological processes; but they also come from anthropogenic sources including agricultural activities (synthetic fertilizer and animal manure application), wastewater discharges (municipal wastewater treatment plants and septic systems), industrial discharges (nitrogen fertilizer production, paper mills, and petroleum refining), and stormwater runoff.

There are many biological responses to nutrients in lakes and Figure 4 (EPA 1999) outlines the basics of nutrient cycling in lakes. The biologically available nutrients and light stimulate phytoplankton (algae) and/or macrophyte growth. As these plants grow, they provide food and habitat for other organisms such as, zooplankton and fish. When these aquatic plants die, they will release nutrients back into the water through decomposition. The decomposition of plant material consumes oxygen from the water column and recycled nutrients are available to stimulate additional plant growth. Physical properties including light, temperature, residence time, and wind mixing also play integral roles throughout the pathways described.



1. Nutrients (N and P) enter the lake through external loading from the surrounding watershed and internal recycling processes
2. Nutrients and light stimulate the growth of phytoplankton and macrophytes (aquatic plants)
3. Aquatic plants consume carbon dioxide and increase the pH of the lake
4. Zooplankton (aquatic invertebrates) graze on the phytoplankton population
5. Aquatic plants break down and/or die, consume oxygen as part of decomposition, and recycle ammonia, phosphorus, and carbon dioxide into the water and the sediments

Figure 4. Nutrient Cycling in Lakes (Adapted from EPA 1999)

Chlorophyll *a*

Chlorophyll is a green pigment found in plants and phytoplankton responsible for photosynthesis, the biological process of converting light energy into chemical energy. Chlorophyll *a* is the dominant chlorophyll pigment in plants and is often the dominant chlorophyll pigment in phytoplankton (Hauer and Lamberti, 1996). The concentration of chlorophyll *a* is used to estimate the amount of phytoplankton biomass present in a lake or stream. Phytoplankton serve a foundational role in the lake food web as primary producers. Primary producer is a term for organisms that can utilize light to convert inorganic chemicals such as, nitrogen, phosphorus, carbon dioxide, and other minerals into living biomass (Water on the Web, 2004). Therefore, measurements of chlorophyll *a* are a useful way to estimate waterbody productivity. The biologic productivity of a lake or stream, measured as chlorophyll *a*, also influences the trophic state and dissolved oxygen.

Increased lake algal productivity, measured as elevated chlorophyll *a* concentrations, can have a myriad of impacts on public water supplies including operational problems (e.g., clogged filters), taste and odor complaints, and increased disinfection by-product formation (Jüttner & Watson, 2007, Rashash et al., 1997, Young et al., 1996, Cooke & Kennedy, 2001, Wardlaw et al., 1991). Particular algal species are known to produce musty/earthy odors that lead to taste and odor problems at drinking water treatment facilities. Additionally, drinking water facilities that use a chlorination process are at risk of forming disinfection by-products such as carcinogenic trihalomethane (THM) when chlorophyll concentrations are high (Callinan et al.,2013, Cooke & Kennedy, 2001, Wardlaw et al., 1991). Excessive algal growth in a lake increases the levels of organic matter which is a precursor to THM formation. Many of Oklahoma's public water supply lakes are subject to nutrient pollution and elevated chlorophyll *a* concentrations; consequently, it is valuable to use chlorophyll *a* as an indicator to evaluate the condition of these lakes in the context of water supply. As a lake with the Sensitive Water Supply (SWS) designation, the water quality criterion of 10 mg/m^3 is in place.

As chlorophyll concentration increases there is greater and greater likelihood that the phytoplankton biomass in the lake is dominated by cyanobacteria (Tetra Tech, 2018, Havens, 2014). When cyanobacteria are dominating the phytoplankton community there is a greater prospect for a harmful algal bloom (HAB) event if/when the advantageous conditions occur within the lake. Thus, chlorophyll concentrations can be used as a proxy for the potential presence of HAB toxins.

Water Clarity

Turbidity is a measure of water clarity and relates to erosion and sedimentation. The greater the amount of total suspended solids in the water, the less clear the water will be, and the higher measured turbidity. Suspended solids that contribute to turbidity include silt, clay, algae, plankton, and organic matter. Increased turbidity affects lakes and streams in a myriad of ways. For example, the suspended particles absorb more heat, which can raise water temperature and reduce the dissolved oxygen concentration. This happens as a result of the water's oxygen saturation threshold being lower when water is warmer (Water on Web, 2004). Turbidity also influences algal growth by limiting the amount of light penetration into the water column. Aquatic organisms impacted by increased turbidity, as particles of silt, clay, and/or organic material settle to the lake or stream bottom they can suffocate larvae and fill in areas around rocks that serve as benthic habitat (Water on Web, 2004). Moreover, as the suspended solids settle to the lake bottom, the lake becomes shallower, and its capacity is reduced limiting water supply availability. Lastly, high turbidity can also negatively impact the aesthetic and recreational qualities of lakes.

Total Organic Carbon

Total organic carbon (TOC) is a measure of all the carbon containing compounds present in a water sample, allowing insight to the amount of organic material present. Sources of these organic compounds include soil and plant detritus and to a lesser degree, even carbon present in living material such as bacteria and plankton (Wetzel, 2001). Wetzel (2001) presents median organic carbon content for eutrophic lakes as 12.0 mg/L, oligotrophic lakes as 2.2 mg/L, and rivers as 7.0 mg/L.

TOC is an especially important measure for water treatment plants to inform on potential creation of Disinfection By-Products (DBPs). Chlorine compounds used in disinfection can react with organic matter to create by-products that could be carcinogenic (TCEQ, 2002). Reducing TOC in the source water could lead to reducing the drinking water treatment cost.

Phytoplankton

Phytoplankton are free-floating, microscopic algae that live in open water, taking up nutrients from the water and energy from sunlight (Water on the Web, 2004). They inhabit the sunlit uppermost layers of the water column in order to photosynthesize (USEPA, 2019a). The ability of phytoplankton to photosynthesize makes them a primary producer of food and energy within the lake ecosystem; for example, phytoplankton are the food source for other organisms such as zooplankton. Phytoplankton are uniquely adapted to specific depths, habitats, and nutrients conditions. The composition and diversity of the phytoplankton community are affected by a myriad of environmental conditions and can be used as an indicator of the biological condition of a waterbody (USEPA, 2016).

Cyanobacteria are a particular group of phytoplankton that under certain conditions (e.g., excessive nutrients, warm water temperatures, and slow-moving/calm water) can rapidly multiply and produce a HAB with toxins (USEPA, 2019). The toxins they produce can harm humans, pets, wildlife, and livestock. Children and dogs are the most frequently affected by cyanotoxins because they are smaller, more likely to ingest water, and tend to stay in the water for longer periods of time (CDC, 2017). It is important to note that not all species of cyanobacteria produce toxins and those that do often require certain environmental conditions.

Zooplankton

Zooplankton are small, free-floating aquatic microorganisms in lakes. They live near the surface and can migrate vertically within the water column to be near food sources. The zooplankton community is composed of both primary consumers, which eat free-floating algae, and secondary consumers, which feed on other zooplankton (USEPA, 2016). Zooplankton are an important part of the food web in lakes, transferring energy between primary producers and other levels in the food chain. As a result of their central position in lake food webs, zooplankton can strongly affect water quality, algal densities, fish production, and nutrient cycling (International Institute for Sustainable Development, 2016). Through grazing zooplankton help maintain the balance of algae population within the lake. Zooplankton have close links with the surrounding environment throughout their life cycles and they demonstrate rapid changes in their populations when disturbances such as eutrophication occurs. As such, changes in zooplankton community structure and composition can indicate water quality changes in lakes making them a good indicator of biological condition in lake systems.

ODEQ-SEL analyzed samples for parameters listed in Table 2 in accordance with the ODEQ's Quality Assurance Plan (ODEQ, 2019) and Data Quality Manual (ODEQ, 2018). The Compliance Plan requires the analytical laboratory to use Environmental Protection Agency (EPA) certified methods and report to certain detection limits, also referred to as method reporting limits (MRLs), as shown in Table 2.

Table 2. Test Methods and Reporting Limits

Parameter	Method	Units	Method Reporting Limit
TKN	EPA 351.2	mg/L	0.10
Nitrate/Nitrite	EPA 353.2	mg/L	0.05
TP	EPA 365.3	mg/L	0.01
TSS	SM 2540D	mg/L	5.00
Total Organic Carbon	SM 5310C	mg/L	0.50
Dissolved Orthophosphate	EPA 365.1	mg/L	0.005
Ammonia	EPA 350.1	mg/L	0.10
Chlorophyll *a*	OWRB-Chlorophyll*	mg/m^3	0.50
Pheophytin a	OWRB-Chlorophyll*	mg/m^3	0.50

* The chlorophyll-*a* method follows EPA SM10200H with some slight modifications in sample handling to include the use of magnesium carbonate solution during filtering and modifications in sample extraction to adhere to the maximum steep time of 24 hours as prescribed in EPA 446.0

Data Collection

Sampling of both in-lake and watershed sites began in February of 2018 and was completed in December of 2020. Data collected included water quality sampling, biological sampling (in-lake only), and continuous data collections. All data reported were collected as stated in the Quality Assurance Project Plan (QAPP) and meets data quality objectives and quality assurance protocols. Each site was sampled monthly throughout the study period, with bi-weekly sampling of the lake during the critical period (May-Sept.). Arcadia Lake sites were sampled a total of 49 times with the mid-May 2019 collection missed due to flooding and restricted lake access. Watershed baseflow events were sampled a total of 25 times with targeted rainfall events (TREs) sampled 18, 16, and 13 times at Deep Fork, Wynn Creek, and Spring Creek, respectively.

During each sampling event, staff noted field observations, collected water samples, and recorded information specific to each site. Data for water quality indicators were collected following OWRB standard operating procedures (SOPs) for the water quality samples (OWRB, 2018b). Data for water quality parameters (Table 1) were collected following OWRB SOP for water quality sampling (OWRB, 2018a). Several parameters were monitored *in-situ* utilizing a YSI® EXO2 multi-parameter sonde. In accordance with manufacturer's specifications and/or published SOPs, all parameters (excluding water temperature which is factory calibrated) were calibrated weekly and verified daily with appropriate standards. Multi-parameter sonde measurements at the watershed monitoring locations were taken in the stream's thalweg near the surface, at a depth of approximately 0.5 m. OWRB personnel measured hardness and alkalinity using Hach® titration protocols, and nephelometric turbidity using a Hach® Portable turbidimeter.

Continuous Data Collection

Capturing and documenting hydrologic conditions is critical to understanding in-lake and watershed dynamics. Automated data loggers were used to capture high resolution measurements of temperature and dissolved oxygen in Arcadia Lake, while water level telemeters and autosamplers were used at the three watershed monitoring locations.

Temperature String Buoy: In January 2018, a Nexsens CB-50 Data Buoy and X2-SDL Submersible Data Logger was installed at the lacustrine site on Arcadia Lake equipped with a TS2100 Thermistor String (Nexsens Technology, Fairborn, OH) that recorded temperature every 15 minutes. The strings consisted of twelve sensors spaced 1-meter apart with measurements starting at 0.5 m below the surface to a depth of 11.5 m. The temperature string was deployed to aid in determining lake stratification, mixing events, and its mictic nature in addition to evaluation of beneficial uses. During monthly sampling visits regular operation and maintenance occurred that included the retrieval of buoy and data logger for battery replacement and retrieval of temperature string thermistors for cleaning of any biofouling. Maintenance was attempted within the 15-minute time window between logging events to ensure uninterrupted data collection when possible. Data were transmitted from the buoy to the WQdatalive website (https:///www.wqdatalive.com) every 4 hours and downloaded weekly to OWRB network. The full temperature dataset was uploaded to AQUARIUS® Time-Series database (Aquatic Informatics, 2020).

Dataloggers: In summer 2019 and spring 2020, three HOBO® dissolved oxygen and temperature data loggers (U26-001) were installed at 0.8 m (equipped with a U26-GUARD-2 anti-fouling guard), 3.2 m, and 7.5 m depths on the temperature string. The middle depth of 3.2 m represented the threshold of 50% of the volume of Arcadia at conservation pool elevation. Each of the loggers recorded data for 5-6 months after installation. Data were downloaded via a HOBO® U-DTW-1 Waterproof Shuttle during each sampling trip, and loggers were set to continuous recording. Data logger deployment, data retrieval, and data export were accomplished using HOBOware® Pro software for Windows. Data were validated or calibrated for biofouling and/or cleaning events using the HOBOware® Pro Dissolved Oxygen Assistant or AQUARIUS® Time-Series and data is currently stored in the AQUARIUS® Time-Series database. Three watershed monitoring locations were equipped with telemetered river gauges that use a Campbell CR300 datalogger, which allowed for real-time monitoring of stream stage through web-based software. Real-time data were transmitted through cellular telemetry at 15-minute intervals and stored on an OWRB server. Elevation surveys were conducted at each bridge location to establish at least two reference points (RPs) and benchmarks according to the OWRB SOPs (OWRB, 2004). These elevation surveys created a consistent vertical reference datum at each site, allowing staff to establish a measurable stream stage, recorded by river gauging equipment mentioned above, and by physical measurements by field staff using weighted steel tape measures. Telemetered stage data were adjusted and validated for drift in AQUARIUS® Time-Series.

Autosamplers: Teledyne ISCO® Glacier autosamplers were installed at all three of the watershed monitoring locations in March 2018. These autosamplers were essential to collecting water quality parameters during TREs, especially since stage and discharge are elevated during periods of precipitation. The autosamplers also had the ability to collect samples during unattainable targeted rainfall events, such as during an active thunderstorm or overnight hours. The samples collected by the autosamplers were via a flow weighted sampling method. Flow weighted samples are collected using proportionate volumes of water over the course of a TRE. This sample collection method allows for a more representative sample from the stream during the entire course of the TRE, which in most instances lasted several hours to a day. The autosamplers were programmed to begin collections for a TRE at a site-specific trigger stage that was established by analyzing the frequency and duration of elevated stream stages during TREs. The intakes for the autosamplers were installed at or below any designated trigger stage to ensure water was at an adequate level. TREs were not collected by the autosamplers unless at least 3 liters of water was able to be pumped to fill the collection bottles. Once an event was triggered, the autosamplers were programmed to continue collecting water every 15 minutes during the rise, peak and fall of the hydrograph for that TRE. The volume of water collected at each 15-minute interval was determined by the relative, calculated volume of water in the stream at the measured stage value. Collections continue every 15 minutes until the hydrograph falls below the trigger stage value or until the sample container in the autosampler was full, whichever came first. All water collected by an autosampler during a TRE were composited into a single churn splitter and subsequently homogenized into four sample bottles, in the same manner as the monthly baseflow collections.

In-Lake Collections

Measurements were recorded at each in-lake sampling site in the form of a vertical profile. Vertical profiles were recorded in 1-meter increments from the lake surface to the lake bottom, with additional readings at 0.5 m below the surface and 0.2 m above the lake bottom. During periods with anoxic conditions (dissolved oxygen < 2.0 mg/L) an additional reading was taken 0.5 m above the first depth with measured anoxia to narrow down the point of transition.

Data for all other indicators were amassed from water quality samples collected. Water quality samples were collected in-lake at all sites utilizing a depth-integrated sampler (DIS) and churn splitter. A DIS is designed to collect a representative sample of the water column to a targeted depth, which is calculated by first measuring the Secchi disk depth (cm) at each site. That depth is doubled to represent the photic zone and is the targeted DIS depth. For instance, if a Secchi disk depth is 80 cm, the targeted depth for collecting a DIS is 160 cm. If the photic zone is less than 0.5 m, a surface water grab is collected 0.5 m below surface. Each sampling event included four bottles—one clear 1-liter bottle for mineral analysis, one clear 1-liter bottle preserved with sulfuric acid for nutrient analysis , one amber 1-liter bottle for field chemistry analyses, and one amber one liter bottle for chlorophyll *a* filtering and analysis, with all samples preserved on ice until delivery to ODEQ-SEL. Dissolved orthophosphate, was collected and filtered using a FlipMate filtration system (Environmental Express FlipMate 100 SC0308 Filtration Assemblies, Certified, PES/PTFE, 0.45µm), and total organic carbon (TOC) samples, collected in a 500 m amber glass bottle, were stored on ice and preserved upon deliver to ODEQ-SEL. Additionally, a Van Dorn sampler was used to collect at depth samples near the lake bottom, just above the sediment-water interface. Bottom samples were collected at both sites 1 and 2 to capture internal nutrient dynamics within the lake.

Watershed Collections

The watershed monitoring collections were designated in to two separate types of collections: monthly baseflow and targeted rainfall events (TREs). Both types of collections were essential to representing the dynamics of the watershed. Monthly baseflow samples were collected when streamflow was stable between precipitation events. The additional samples were collected during targeted rainfall events, when the water level was above seasonal baseflow due to rain in the watershed. Water was collected at each of the watershed monitoring locations during baseflow collections via the composite grab method as outlined in the OWRB SOP for water quality sampling in streams (OWRB, 2013). The baseflow water collection consisted of six 1-liter grab samples taken from a cross-section of the stream and composited into an 8-liter churn splitter and subsequently homogenized into four sample bottles—minerals bottle, nutrients bottle, field chemistry bottle, and chlorophyll *a* bottle for filtering. Dissolved orthophosphate and TOC were not analyzed at the three watershed monitoring locations.

Surface runoff following rainfall events are the most significant sources to nutrient and sediment loadings, therefore additional samples were collected during targeted rainfall events (TREs) for the three watershed monitoring locations. These targeted rainfall events give a better understanding of loads that are being input into Lake Arcadia via runoff from a rainfall event. To aid in the collection of TRE samples, all watershed sites had a refrigerated autosamplers installed April 2018 in conjunction with the continuous stage recorder. Due to the use of refrigerated autosampler equipment, some in-situ parameters (temperature and dissolved oxygen) are not valid since the variables are temperature dependent and samples collected are immediately refrigerated. Like the baseflow collections, each TRE sample was analyzed for the same suite of parameters. Each autosampler was programmed to collect water at site-specific stream stages, or trigger values. This ensured that an adequate number of representative TREs were collected from a variety of points along the hydrograph and allow for better characterization of loadings. Discharge measurements were also taken according to the OWRB SOPs (OWRB, 2016), using acoustic doppler current profilers (ADCPs) for non-wadeable conditions, and acoustic doppler velocimeters (ADVs) for wadeable conditions. A rating curve for each watershed location was developed to represent each hydrograph.

Biological Collections

Samples for chlorophyll *a*, as a measure of algal biomass, were collected at all sample sites and processed in accordance with standard procedures outlined (OWRB, 2018). All chlorophyll *a* samples were analyzed by the ODEQ-SEL. Additionally, phytoplankton and zooplankton samples were collected at the dam site on Arcadia Lake during each sampling visit for taxonomic identification. Taxonomic identification was performed by Scott Lab at Baylor (phytoplankton) and Dr. Dzialowski's lab at Oklahoma State University (zooplankton). Phytoplankton samples were collected as a surface grab sample, while zooplankton were collected as a tow using a Wisconsin-style plankton net (130 mm diameter, 243 µm mesh). The length of the tow was the entire depth of the water column. All samples were collected and processed in accordance with standard procedures (OWRB, 2020c).

PHYSICAL CHARATERISTICS OF THE WATERSHED

Land Use Characteristics

Land use characteristics were analyzed for each of the three HUC-12 subbasins using data from Multi-Resolution Land Characteristics Consortium's (MRLC) National Land Cover Database (NLCD) for the years 2001 and 2019 (MRLC, 2021). Data from 2019, the most recent NLCD product during report writing, was used to assess current conditions within the Arcadia Lake watershed. Land use characteristics were analyzed at the subbasin and watershed levels using BASINS 4.5 and QGIS. There are 16 land uses categorized in the Continental United States, 15 of which are present in the Arcadia Lake watershed (Figure 5). Furthermore, these categories were grouped into eight categories—Open Water, Developed, Barren Land, Forest, Shrubland, Herbaceous, Planted/Cultivated, and Wetlands—to aid in subbasin and watershed comparisons. Land uses were categorized using the Anderson Land Cover Classification System (Anderson et al., 1976). More information on NLCD Land Cover classes and values, as well as classification descriptions can be found via the MRLC NLCD website.

The Developed category is comprised of four land uses—Developed, Open Space; Developed, Low Intensity; Developed, Medium Intensity; and Developed, High Intensity—representing different levels of urbanized development and impervious cover. Impervious cover in this land use category will be further explained in the following section. Four of the eight land use categories—Open Water, Shrubland, Herbaceous, and Barren Land—are comprised of only one land use, each of which share the same or similar name with the category. The Forest category is comprised of three types of forested land use—Deciduous Forest, Evergreen Forest, and Mixed Forest—which differ in percentage typical tree species found in each 30 m^2 pixel. The Planted/Cultivated category consists of Pasture/Hay and Cultivated Crops, and are areas of agriculture. The Wetlands category, land uses Woody Wetlands and Emergent Herbaceous Wetlands, are areas of wooded or perennially vegetated wetlands (MRLC, 2021).

The White Turkey Creek-Deep Fork and Lake Arcadia-Deep Fork subbasins are dominated by a mix of Developed, Forest, and Herbaceous land use categories, while the Headwaters Deep Fork subbasin is highly urbanized, with Developed land uses accounting for over 90% of the total subbasin (Table A5, Table A6, Table A7). Developed land uses accounted for 82.1 km^2 (31.7 mi^2) of the Headwaters Deep Fork subbasin, 36.3 km^2 (14.0 mi^2) of the White Turkey Creek-Deep Fork subbasin, and 33.6 km^2 (13.0 mi^2) of the Lake Arcadia-Deep Fork subbasin, or 92.7%, 40.3%, and 41.4%, respectively. The Arcadia Lake Watershed as a whole is comprised of 58.5% Developed land use categories as of 2019 (Table A8). Forest land use categories make up 3.73 km^2 (1.44 mi^2) of the Headwaters Deep Fork subbasin, 26.4 km^2 (10.2 mi^2) of the White Turkey Creek-Deep Fork subbasin, and 25.1 km^2 (9.70 mi^2) of the Lake Arcadia-Deep Fork subbasin, or 4.21%, 29.4%, and 31.0%, respectively (Table A5, Table A6, Table A7, Figure A2). The Arcadia Lake Watershed is comprised of 21.3% Forest land use categories as of 2019 (Table A8). Herbaceous land use categories accounted for 2.20 km^2 (0.85 mi^2) of the Headwaters Deep Fork subbasin, 22.6 km^2 (8.74 mi^2) of the White Turkey Creek-Deep Fork subbasin, and 13.7 km^2 (5.31 mi^2) of the Lake Arcadia-Deep Fork subbasin, or 2.49%, 25.2%, and 16.9%, respectively (Table A5, Table A6, Table A7, Figure A1). The Arcadia Lake Watershed is comprised of 14.9% Herbaceous land use category as of 2019 (Table A8). Wetlands land uses only accounted for 0.15 km² (0.058 mi²), or less than 0.1% of the total area of the Arcadia Lake Watershed. Planted/Cultivated land use categories were also uncommon within the three subbasins, totaling 2.68 km² (1.03 mi²), or approximately 1.03% of the Arcadia Lake watershed. Barren Land is the least common land use category in the Arcadia Lake watershed at less than 0.1 km² (0.040 mi²) (Table A5, Table A6, Table A7, Table A8).

All three subbasins had an increase in the High and Medium Developed land uses from 2001 to 2019, with the White Turkey Creek-Deep Fork and Lake Arcadia-Deep Fork subbasins increasing over 30% for each category, while the Headwaters Deep Fork subbasin had an increase of over 5% in both categories (Table A1, Table A2, Table A3). Overall, the Arcadia Lake watershed has seen a 7.44% increase in Developed land use categories from 2001 to 2019, up a total of 10.5 km² (4.05 mi²) (Table A4). The Open Developed land use was the only developed use that decreased from 2001 to 2019 in the watershed, from 31.7 km² (12.2 mi²) in 2001 to 30.3 km² (11.7 mi²) in 2019, a net loss of 1.43 km² (0.552 mi²) (Table A4). The loss in Open Developed land use from 2001 to 2019 is almost exclusively due to a conversion to other Developed land uses. This land use category has accounted for over 50% of the total land use in the Arcadia Lake watershed for all NLCD years from 2001 to 2019 and increased from 54.5% to 58.5% in those two years, respectively (Table A4).

The Forest land use category, as compared to Developed, decreased significantly in the Arcadia Lake Watershed from 2001 to 2019, and accounted for the largest single category loss in the watershed over that time span (Table A4). Deciduous Forest land use accounted for the most forest area in the watershed at 53.3 km² (20.6 mi²) in 2019. Evergreen Forest and Mixed Forest land uses combined for an additional area of 2.02 km² (0.780 mi²) for a total combined Forested land use in the watershed of 55.3 km² (21.4 mi²) (Table , Table A2, Table A3, Table A4). The Forest land use categories have accounted for over 20% of the Arcadia Lake watershed for all years from 2001 and 2019 but have decreased 11.0%, down from 23.9% of the total area in the watershed in 2001 to 21.3% in 2019. This loss in area totaled 6.85 km² (2.64 mi²), down from 62.1 km² (24.0 mi²) in 2001 (Table A8). Most of the loss in area from 2001 to 2019 was from decreases in the Deciduous Forest land use, accounting for decrease of 6.68 km² (2.58 mi²), or 11.2%; 6.20 km² (2.39 mi²) of the Deciduous Forest loss contribution was in the White Turkey Creek-Deep Fork and Lake Arcadia-Deep Fork subbasins (Table A2, Table A3). Percent canopy, also available from the NLCD, was also assessed from percentages in all three subbasins, and showed the same urban trends in the Headwaters Deep Fork subbasin with higher percent canopy rates in the downstream White Turkey Creek-Deep Fork and Lake Arcadia-Deep Fork subbasins (Table A13).

The Herbaceous land use category, consisting solely of the Grassland land use, declined the second most, behind the Forest land use category, from 2001 to 2019. There was an overall loss of 10.5%, or 4.53 km² (1.75 mi²) in the watershed, with 4.27 km² (1.65 mi²) of the loss from the White Turkey Creek-Deep Fork and Lake Arcadia-Deep Fork subbasins (Table A2, Table A3, Table A4). Conversely, Shrubland land use categories saw a modest increase from 2001 to 2019, increasing 0.9 km² (0.347 mi²) in the Arcadia Lake watershed, and the highest non-Developed land use growth each of the three subbasins. The Planted/Cultivated land use category decreased 0.10 km² (0.040 mi²) between 2001 and 2019, while the remaining land use categories—Water, Barren, and Wetlands—changed less than 0.10 km² (0.040 mi²) in that time period (Table A8).

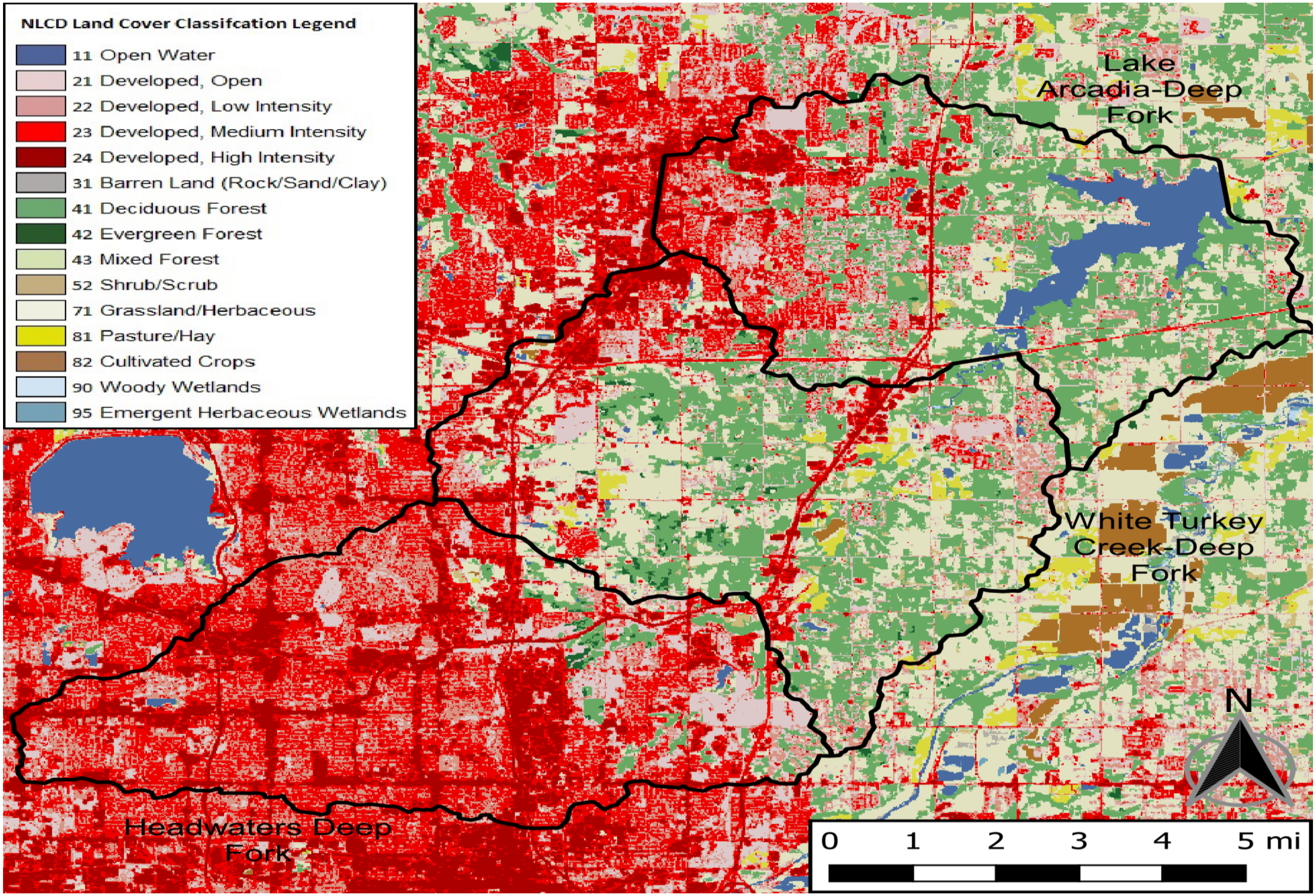


Figure 5. Land use map of the Arcadia Lake watershed. Reference appendix for land use tables.

Impervious Cover

Urban imperviousness data from the NLCD for years 2001 and 2019 were analyzed in the Arcadia Lake watershed using QGIS (Figure 6). Imperviousness is reported as a percentage of developed surface over a 30 m^2 pixel. The percentages were grouped for ease of analysis, using the four categories—0-10 percent, 11-25 percent, 26-60 percent, and 61-100 percent—developed from the Impervious Cover Model (ICM) and based on the role of impervious cover in hydrological, biological, and water quality stream health indicators (Schueler et al, 2009). The 0-10 percent imperviousness category was the most dominant in the watershed, making up the greatest total area in the watershed, over 50% of total area, as well as the White Turkey Creek-Deep Fork and Lake Arcadia-Deep Fork subbasins (Table A12). The highly urbanized Headwaters Deep Fork subbasin is comprised of a relatively even mix of the 26-60 percent and 61-100 percent categories, correlating with the high levels of urbanization in Oklahoma City (Table A9). The largest increases from 2001 to 2019 in all three subbasins, and, intuitively, the watershed, were in the 26-60 percent and 61-100 percent categories, reinforcing the increased Developed land use category changes over the same time period. All subbasins had decreases in the 0-10 percent impervious category, with a loss of 11.5%, or 8.27 km^2 (3.19 mi^2) in the Arcadia Lake Watershed (Table A9, Table A10, Table A11). Conversely, the 26-60 percent and 61-100 percent categories increased 7.81% and 16.5% in the Arcadia Lake watershed, or 4.42 km^2 (1.71 mi^2) and 7.66 km^2 (2.96 mi^2), respectively (Table A12).

Impervious cover coefficients represent the ratio of runoff to precipitation (Oregon Department of Transportation, 2014). While these coefficients are dependent on many factors, such as soil moisture, surface slope, infiltration rate, ground cover, and potential water storage, they are nonetheless important in representing the rate and magnitude of runoff events, particularly in watershed with high amounts of urbanization and impervious cover. Fully impervious surfaces, such as streets, sidewalks, and developed buildings have a runoff coefficient of 0.85 or greater, meaning that almost all of precipitation not lost to evaporation becomes surface runoff (Oregon Department of Transportation, 2014). These high runoff coefficients are expected in the purple areas in Figure 6. Residential areas, mostly denoted as land use 23 Developed, Medium Intensity in Figure 5 and as medium to dark reds in Figure 6 have runoff coefficients ranging from 0.50 to 0.70, indicating high runoff despite lawns and trees with typically low runoff coefficients. Forests and unimproved areas, the Forest and Herbaceous categories found in Figure 5 and Table 1, in contrast to urbanized land uses, have a runoff coefficient of 0.10, suggesting high infiltration and retention of precipitation.

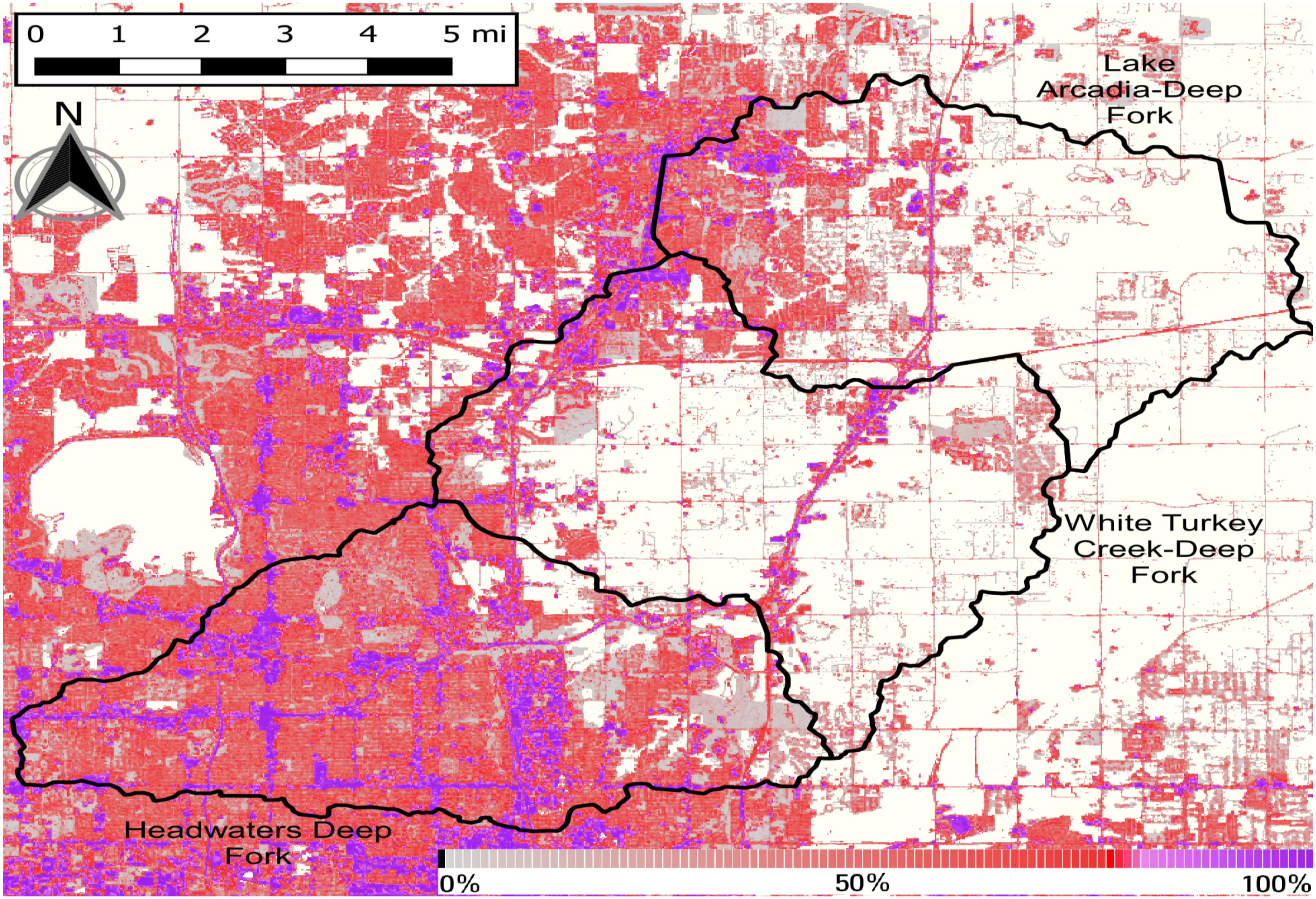


Figure 6. Impervious surface throughout the watershed. Purple representing impervious.

Soil Type Characteristics

Soil type area and percentages within the Arcadia Lake Watershed was compiled using BASINS 4.5 software and utilized soil data from the State Soil Geographic (STATSGO) database held by the National Resources Conservation Service (NRCS) of the United States Department of Agriculture (USDA). Soil type characteristics are available from Soil Survey of Oklahoma County (USDA, 2003), and can be accessed, along with soil surveys of other counties in Oklahoma, via the NRCS website.

There are three soil complexes within the Arcadia Lake watershed, OK151, OK094, and OK146. OK151 is the dominant soil complex in the Arcadia Lake watershed, accounting for over 65% of the total area, and is the dominant soil complex in both the White Turkey Creek-Deep Fork (80.5%) and Lake Arcadia-Deep Fork subbasins (92.0%) (Table A14). OK151, also known as Stephenville-Darnell-Newalla, is a complex of well drained and excessively drained loamy and sandy soils on forested uplands, ranging from very deep to shallow soil depths. The soil complex is comprised of Stephenville, Harrah, and Darsil soil types. Stephenville and Harrah soil types are deep to very deep, well drained brown fine sandy loam surface soils, with sandy clay loam subsoils. Darsil is a shallow, excessively drained brown loamy fine sand over sandstone bedrock. Stephenville and Harrah soil types are suitable for cropland, rangeland, pastureland, and urban development, while Darsil soil is only suited for rangeland (USDA, 2003).

OK094, also known as Kirkland-Renfrow-Zaneis, is most common in the Headwaters Deep Fork subbasin, comprising of almost 70% of the total area (Table A14). The soil complex is comprised of Kirkland and Renthin soil types, as well as urban lands. The Kirkland and Renthin soil types are well-drained deep to very deep silt loam surface soil types, with silty clay or clay loam subsoils over shale and/or sandstone bedrock, suitable for cropland, rangeland, and pastureland (USDA, 2003).

OK146, also known as Konawa-Eufaula-Dougherty, is the least dominant soil type in the Arcadia Lake watershed, comprising of less than 4% of the total area, and is completely absent from the Lake Arcadia-Deep Fork subbasin (Table A14). OK146 is a soil complex of very deep, somewhat excessively drained loamy and sandy soils on upland terraces and dunes commonly found in the North Canadian River basin. The soil complex is comprised of Konawa and Derby soil types, as well as urban lands. The Konawa and Derby soil types are very deep, well drained to somewhat excessively drained soils with a loamy fine sand surface layer. Konawa subsoil is a red sandy clay loam, while Derby subsoil is a loamy fine sand to a sandy fine loam. The soil complex is suitable for cropland, rangeland, and pastureland, as well as urban development (USDA, 2003).

Watershed Precipitation Characteristics

Precipitation data was sourced from the Army Corps of Engineers (USACE) rainfall gauge that is located at the Arcadia Lake dam. The overall watershed precipitation data was sourced from the National Weather Service River Forecast Center (NWS-RFC) for Arcadia Lake Basin Rainfall (Figure 7). Both were analyzed over the period of record, November 1994 – December 2020, as well as the monitoring period from, January 2018 to December 2020. The annual average dam precipitation for the period of record was 31.48 in (799.6 mm), and 34.14 in (867.2 mm) during the monitoring period (Table A15). The annual average basin precipitation for the period of record was 34.12 in (866.8 mm), and 39.90 in (1013 mm) during the monitoring period. Based on these averages, the project period had more precipitation than normal, especially in 2019. Precipitation throughout the basin during the monitoring included 35.16 inches in 2018, 46.75 inches in 2019 and 37.78 inches in 2020. Seasonality is shown in the precipitation data with June accounting for 15.01% of the basin rainfall throughout the period of record. Following along with seasonal trends, May is the second wettest month attributing for 12.98% and July ranking third with 10.55% (Table A15). During these spring months, the increased precipitation was represented through increased TRE collections, while baseflow conditions were observed more often during the winter months such as January and February (Table A5). This is also due to January and February accounting for less than 4% of basin rainfall throughout the period of record.

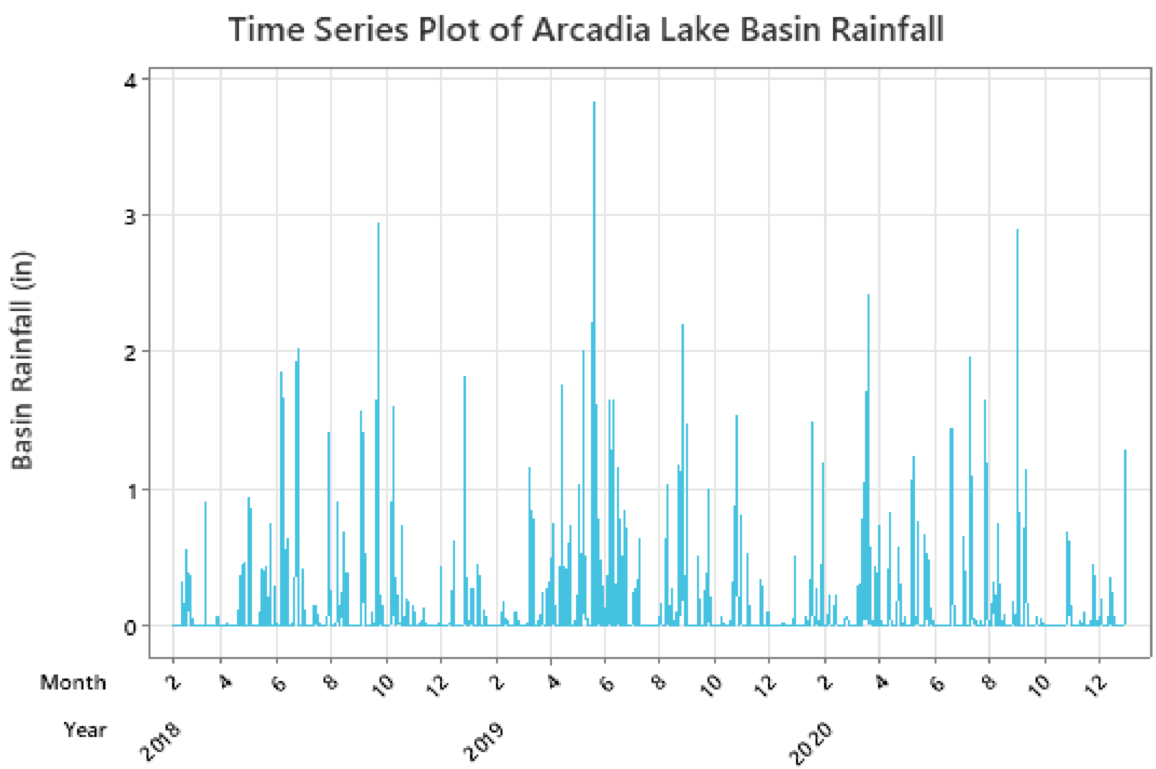


Figure 7. Arcadia Lake Basin Precipitation in inches, 2018-2020.

Stormwater Permitting

ODEQ issues four types of Oklahoma Pollutant Discharge Elimination System (OPDES) stormwater permits to point-source dischargers: Municipal Separate Storm Sewer Systems (MS4s) Phase I and Phase II, OKR05 Industrial Stormwater Permits, and OKR10 Construction Stormwater Permits (ODEQ, 2021).

Both Phase I and Phase II MS4 permits are public, municipal conveyance systems for urban stormwater and overland flow for discharge directly into local water bodies, that are not part of a combined sewer system and are not part of a publicly owned treatment works system. Phase I MS4s are required for medium and large cities or counties with populations over 100,000 persons, such as Oklahoma City and Tulsa. Phase II MS4s are covered by the general OKR04 Permit and issued by ODEQ for municipalities under 100,000 persons. The City of Oklahoma City is the only Phase I MS4 permittee in the Arcadia Lake watershed, operating under permit number OKS000101. Additionally, four Phase II MS4 permits operate within the Arcadia Lake watershed for smaller municipalities such as Edmond, Nichols Hills, The Village, and Warr Acres.

Industrial facilities operating under the OKR05 Permit are required to obtain an OKR05 Industrial Stormwater Permit from ODEQ. Industrial facilities performing material storage and handling, vehicle fueling and maintenance, and other operations outdoors can lead to the discharge of pollutants into local waterbodies and storm sewer systems via surface runoff. Accidental spills or leaks, improper waste disposal, and illicit discharges via storm sewer systems may also lead to increased pollutant loads in waterbodies, which can result in waterbody impairment, degradation of aquatic habitat, and pollution of drinking water. As of June 7, 2021, there are 22 active OKR05 Permits in the Arcadia Lake watershed and there an additional 11 permits that are near, but outside, of the three HUC-12 subbasins that comprise the Arcadia Lake watershed. It is unclear without further investigation of these sites could possibly discharge into the Deep Fork or its tributaries either directly or via an MS4.

Construction sites that disturb more than one acre of land require an OKR10 Construction Stormwater Permit, which is required before construction activity can start. There are currently 149 active OKR10 Permits in the Arcadia Lake watershed as of June 7, 2021. There are also 37 permits that are near, but outside, of the three HUC-12 subbasins that comprise the Arcadia Lake watershed. The effective dates of these permits range from October 2017 to May 2021, so the status of these construction projects is unknown. The main pollutant of concern from construction sites is sediment, as bare soil is often exposed during construction activities and is easily transported as surface runoff to a nearby waterbody or MS4. Chemicals, trash and debris, and other pollutants may be of concern at construction sites as well.

There are multiple stormwater canal conveyances in Northwest and Northeast Oklahoma City that discharge directly into the Deep Fork or are channelized sections of the Deep Fork itself. High amounts of impervious surfaces from urbanization directs large quantities of stormwater through the watershed. This leads to quickly rising water levels that could be described as “flashy” because the hydrograph has a steep rising limb and a high peak discharge. Pollutants such as sediments, heavy metals, oil and grease from vehicles, pesticides, organics, nutrients, trash and debris are conveyed at much higher levels during rain events. These pollutants and debris are transported into the Deep Fork via direct discharge, tributaries, or stormwater conveyances, and eventually discharged downstream into Arcadia Lake. Urban runoff from impervious surfaces in the subbasins can also affect other tributaries of Arcadia Lake, such as Spring Creek.

For the most current data on active permits: Active OKR10 and OKR05 Stormwater Permits

RESULTS

Data collected for this project were used to evaluate current waterbody condition, verify magnitude and extent of impairment, identify numeric targets for attaining beneficial uses, and overall support the development of a TMDL.

Temperature String Buoy

The temperature isopleth for Arcadia Lake at Site 1 exhibited relatively consistent inter-annual temperature patterns throughout the study period with the lake mixed during colder months and lake stratification setting up around mid-May each year (Figure 8). The buoy recorded several surface temperatures >30°C during the summer of each sample year with a thermocline that resided between 6 and 10 m deep through most of the summer until breaking down in late August or early September each year. A previous study had reported that Arcadia Lake exhibited a polymictic pattern for stratification, where the stratification was established and broke down several times over the summer (OWRB, 2000). However, polymictic stratification was not observed during this recent study period. Instead, there was a gradual breakdown of stratification as warmer temperatures likely encroached deeper into the water column during the summer.

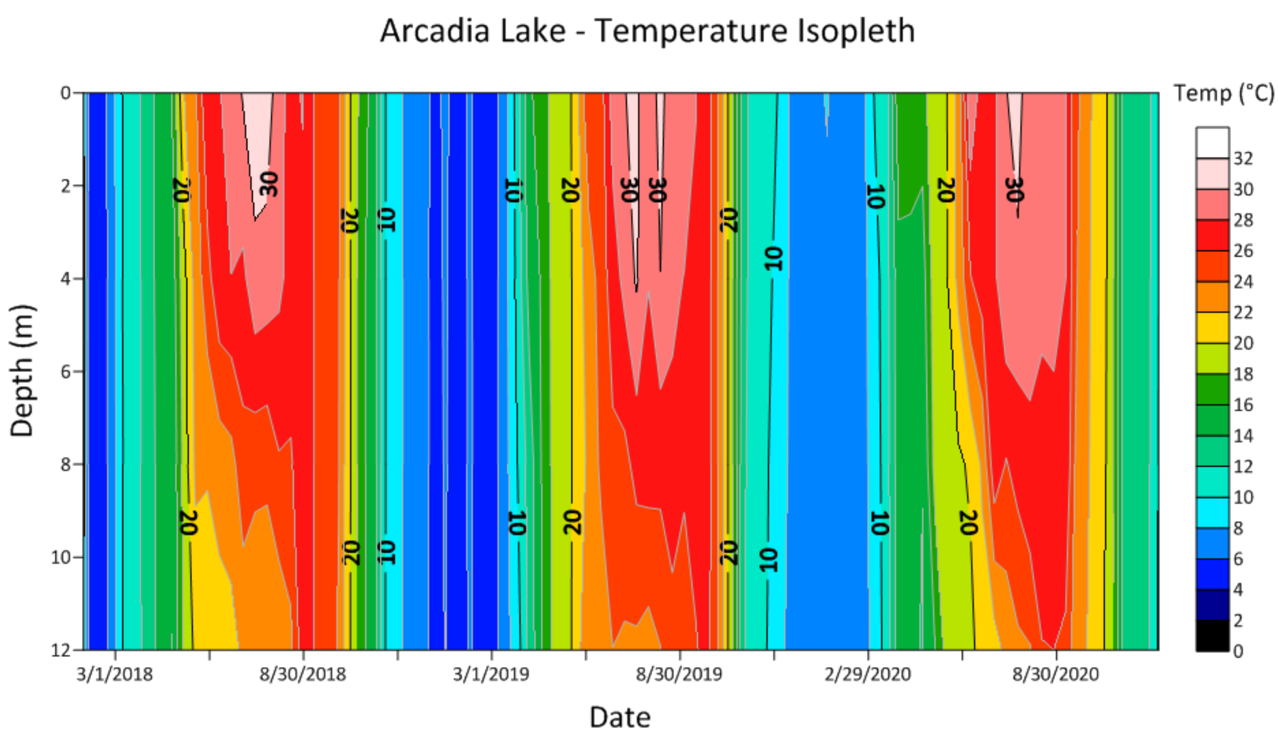


Figure 8. Temperature isopleth for Arcadia Lake at Site 1 from January 29, 2018 to December 7, 2020.

Dissolved Oxygen and Profile Data

Dissolved oxygen (DO) criteria are designed to protect the diverse aquatic communities found throughout Oklahoma waterbodies. For warm water aquatic communities, such as Arcadia Lake, two assessment methodologies apply to protect the Fish and Wildlife Propagation beneficial use: surface and water-column/volumetric (OAC 785:46-15-5). Surface water DO criteria for not supporting is a seasonal threshold of 4.0 mg/L during the summer months and 5.0 mg/L in spring and fall. The volumetric criteria threshold for fully supporting the Fish and Wildlife Propagation beneficial is less than 50% of the cumulative lake volume measuring anoxic (< 2 mg/L DO).

Water below the thermocline was almost always hypoxic as confirmed from discrete DO-depth profiles measured during each sampling event at Site 1 (Figure 9) and relatively high hypolimnion temperatures coupled with hypoxia resulted in percent DO saturations approaching 0% (Figure 10). When the lake was mixed with similar temperature throughout the profile, higher DO concentrations were measured along the entire profile and cooler temperatures resulted in DO saturations between 69% and 115%. Profiles at Site 2, Site 3, and Site 4 showed similar DO-temperature profiles with summer anoxia occurring at similar depths to Site 1. Site 5 was never deep enough to observe hypoxia. Based on the discrete profile data, Arcadia Lake is supporting the FWP beneficial use based on volumetric dissolved oxygen. Anoxic (DO <2.0 mg/L) volumes rose to 46.06% in 2018, 40.23% in 2019, and 32.13% in 2020. Lowest DO levels were seen in July for all years.

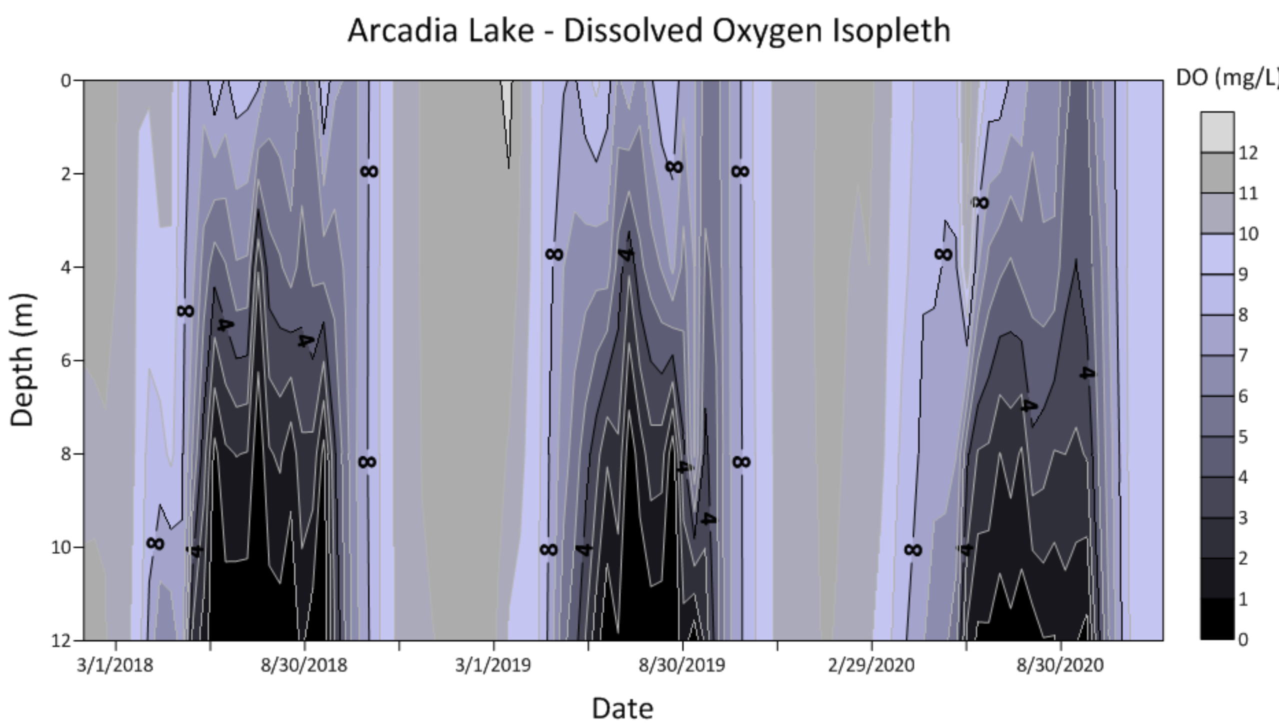


Figure 9. Dissolved oxygen concentration isopleth for Arcadia Lake taken from discrete profiles at Site 1.

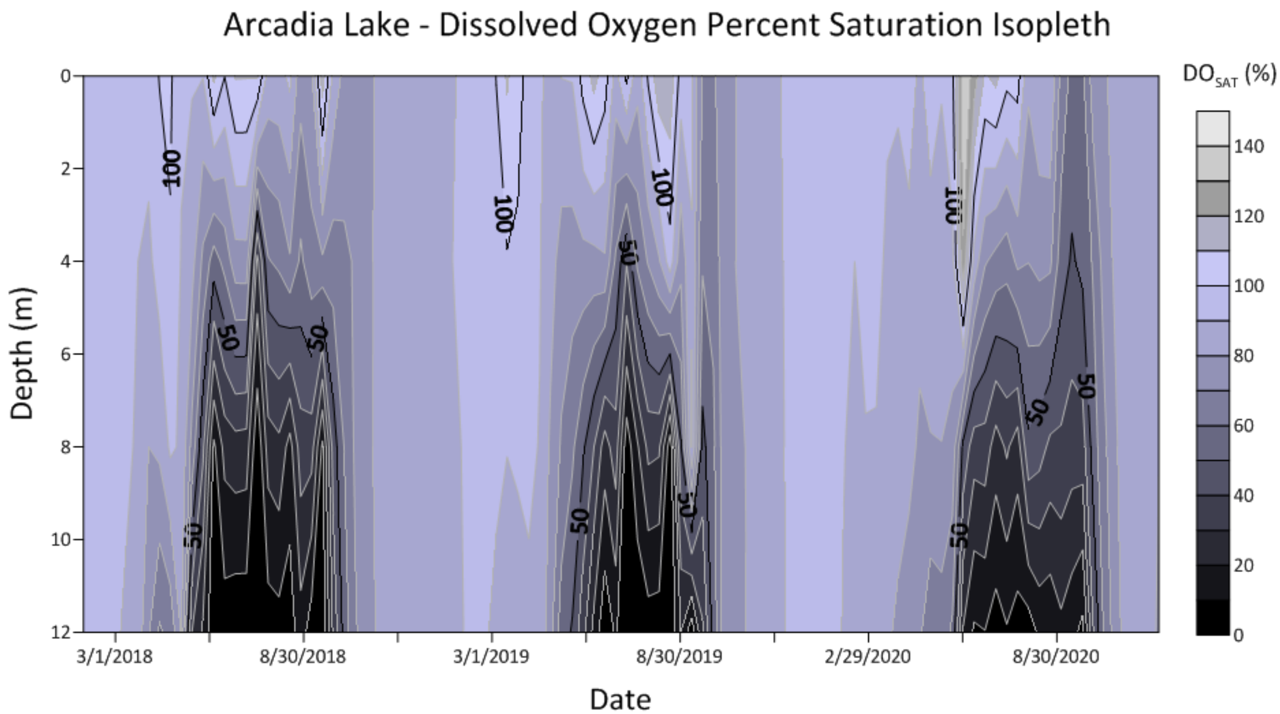


Figure 10. Dissolved oxygen percent saturation isopleth for Arcadia Lake taken from discrete profiles at Site 1.

HOBO® DO dataloggers measured DO data at a finer temporal scale for approximately six months in 2019 (July 6 – December 2, 2019) and 2020 (May 18 – Oct 9, 2020), recording data in 15-minute intervals. In 2019, DO dropped below 2.0 mg/L at 0.8 meters at night (0.92 mg/L at 3:00 am on 9/23/2019) and several times at 3.2 meters for as long as 4 hours at a time at night (Figure 11). The bottom sensor (7.5 m) recorded hypoxia in the late summer and early fall during lake stratification. In 2020, the surface data logger at 0.8 m and the 3.2 m data logger recorded several events where DO dropped below 2.0 mg/L at night (Figure 12). Deployment of dataloggers occurred prior to lake stratification in 2020 where DO was relatively homogeneous throughout the water column to stratified in late May 2020. Similar to 2019, the bottom sensor recorded hypoxia though most of the summer and into early fall. According to this data and given the depth of 3.2 m represents 50% of the conservation pool elevation, there were several periods at night when the lake is not meeting the criteria for DO.

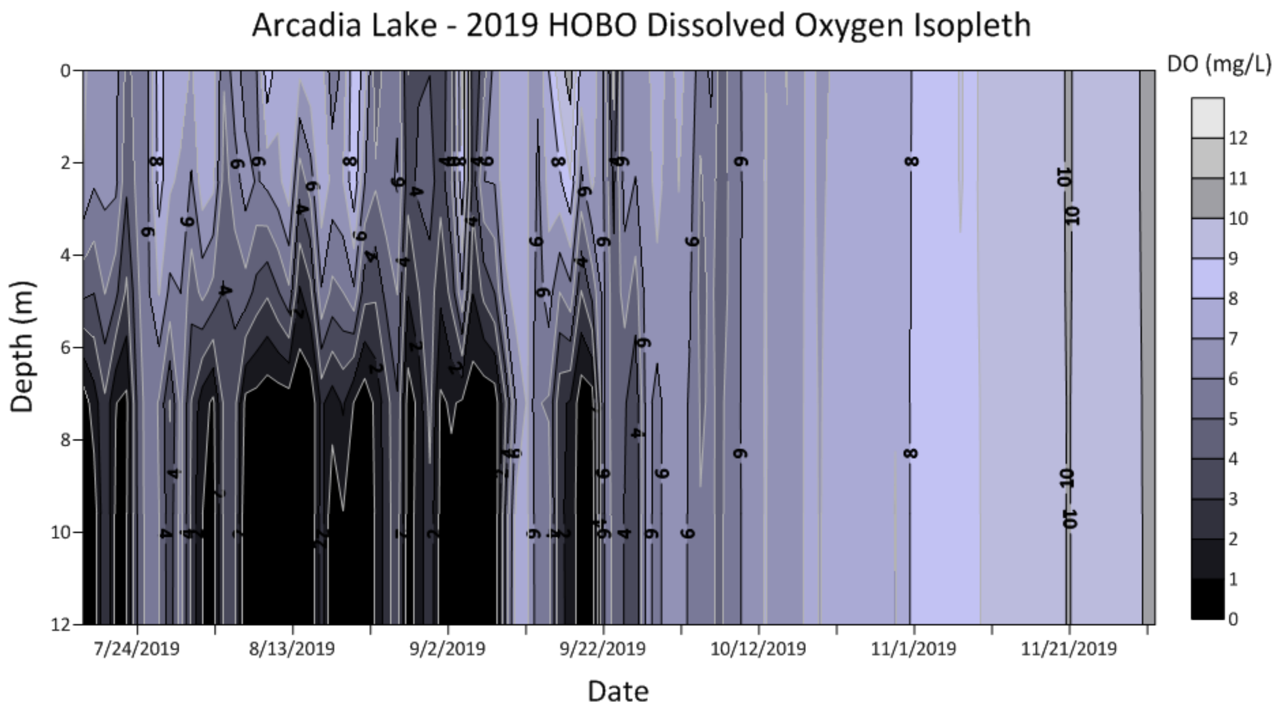


Figure 11. Dissolved oxygen concentration isopleth for 2019 taken from three HOBO® data loggers (depths 0.8 m, 3.2 m, 7.5 m) from 7/16/2019 to 12/2/2019.

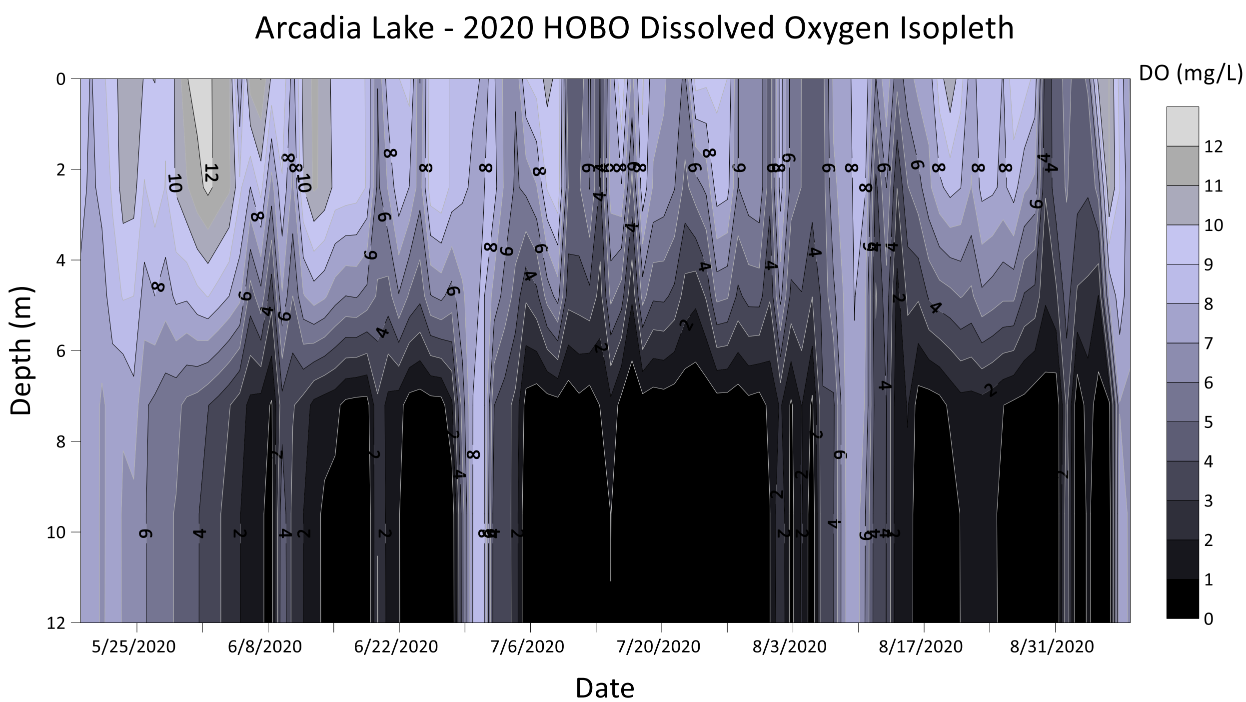


Figure 12. Dissolved oxygen concentration isopleth for 2020 taken from three HOBO® data loggers (depths 0.8 m, 3.2 m, 7.5 m) from 5/18/2020 to 10/9/2020.

Oxidation-reduction potential shows similar results to DO profiles where reduced conditions (<100 mV) were present in the hypoxic hypolimnion (Figure 13) and is consistent with predicted conditions in eutrophic reservoirs (Cooke et al., 2005; Thornton et al., 1990) and with the previous Phase 1 study (OWRB, 2000). In a reduced environment, oxygen is no longer available as an electron acceptor in some nutrient cycles and the valence of iron changes resulting in a release of orthophosphate from the sediment. There was very little difference within a single profile for specific conductance, however specific conductance does change throughout the year with increased values in the spring likely due to increased input of various ions from spring rains (Figure 14).

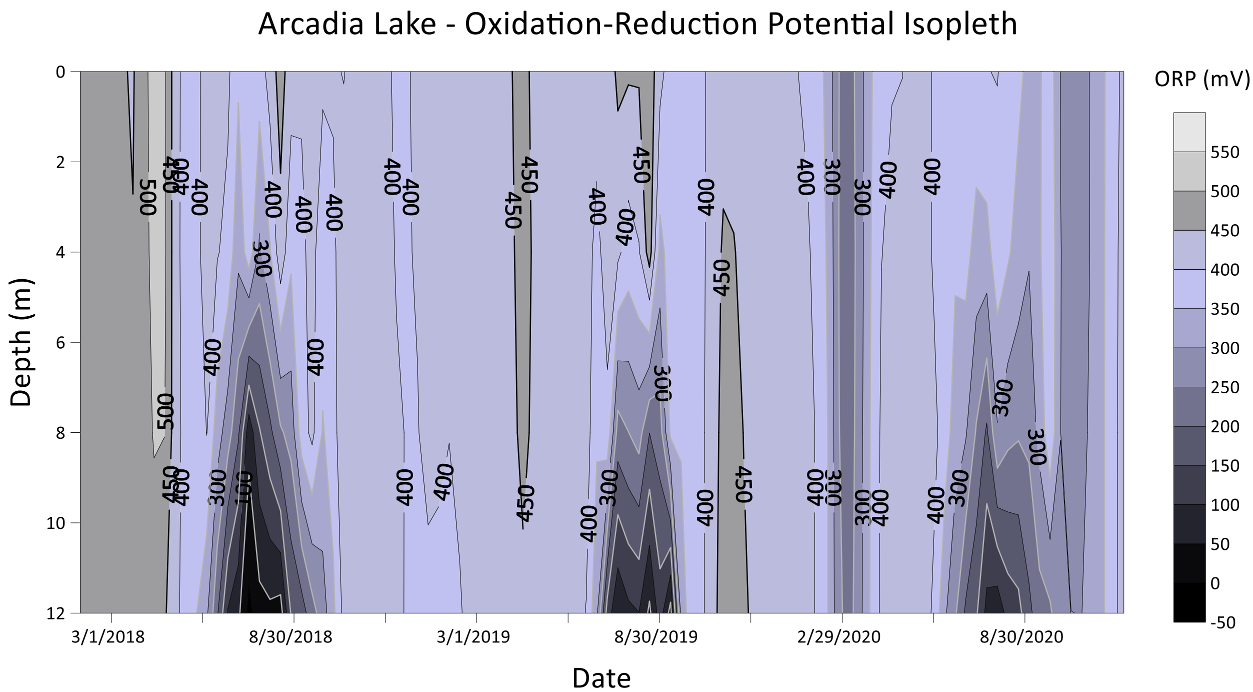


Figure 13. Oxidation-reduction potential isopleth for Arcadia Lake taken from discrete profiles at Site 1.

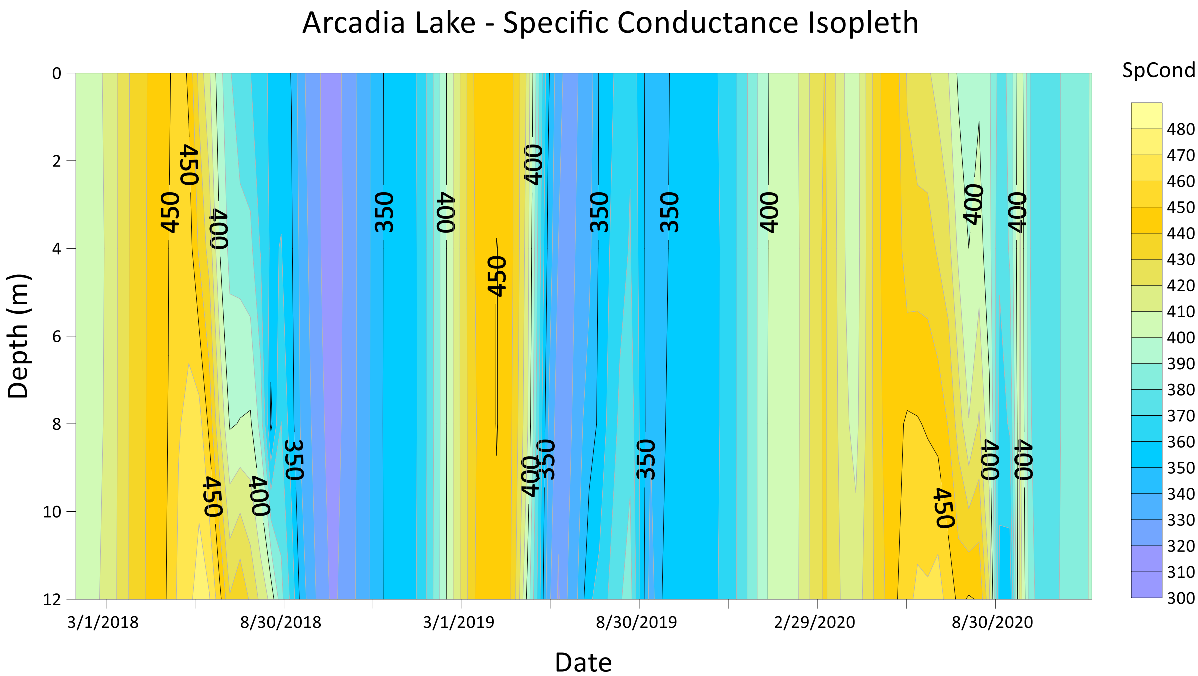


Figure 14. Specific conductance isopleth for Arcadia Lake taken from discrete profiles at Site 1.

Nutrients

There was considerable variation in nutrient concentrations observed throughout the study period with apparent patterns likely due to rainfall events and associated inputs of nutrients at the riverine sections of the lake, or due to oxygen- or hypoxia-mediated conditions within the lake. Many of the nutrients of concern have steps in their cycles that can be promoted or stopped in the presence or absence of oxygen, with the latter observed in the hypolimnion of the lake during stratification. Observations for nutrients (and other water quality parameters) were categorically divided into seasonal data where Winter included January – March, Spring included April – June, Summer included July – September, and Fall included October – December.

To understand the following box and whisker plots, each box is divided into quartiles, with the bottom of the box representing the lower 25% quartile and the top of the box representing the upper 75% quartile. The line through the middle of the box is represents the median of each dataset while the 'X' represents the mean. The whiskers coming from the boxes represent the minimum and maximum of each dataset. There were some outliers observed and those are represented as data points outside of the box and whiskers. The red line denotes the Method Reporting Limit (MRL) for each parameter based on their individual reporting limits (see Table 2). Nutrient and mineral concentrations at watershed monitoring locations are considerably variable by nature due to flowing water, box and whisker plots are a preferred way of showing the watershed results. For example, the taller the box, the more variability at the monitoring location.

Phosphorus

Lake surface concentrations ranged from non-detected (<0.005 mg/L) to 0.108 mg/L with a mean of 0.025 mg/L for dissolved orthophosphate (DOP) and 0.035 to 0.207 mg/L with a mean of 0.078 mg/L for total phosphorus (TP). Seasonal boxplots of each site for DOP (Figure 15) and TP (Figure 16) show that there was little difference in surface concentrations among Site 1, Site 2, and Site 3 seasonally. Site 5, however, tended to have higher concentrations of both analytes, especially TP, while Site 4 tended to have slightly higher concentrations during spring. The higher concentrations in spring for both DOP and TP at these sites were likely attributed to input from Deep Fork, with Site 4 being downstream of Site 5 but more of a transitional site. The shallow depth in the area at Site 5 with frequent resuspension of sediment by wind or wave activity throughout the year may have been responsible for higher concentrations observed in other seasons. These findings at Site 5 also agreed with a previous study at Arcadia (OWRB, 2000). Other than the general higher concentrations at Site 4 and Site 5, DOP (Figure 17a) and TP (Figure 18a) showed little predictability or obvious patterns at the other sites during the study period when looking at surface concentrations from each sampling trip. However, bottom samples showed peak concentrations in both TP and DOP at Site 1 and Site 2 during late spring and summer months corresponding to DOP release from sediments in the reduced conditions in the hypolimnion (Figure 17b and Figure 18b). In some cases, hypolimnion concentrations of DOP were several orders of magnitude higher than in surface samples, which were close to the detection limit during the same sampling event at Site 1 and Site 2. It is typical to see high concentrations of DOP in the hypolimnion of eutrophic lakes due to the reduced conditions in the hypoxic environment (e.g., the subsequent solubility as ferrous phosphate is reduced to ferric phosphate; Wetzel, 2001; Cole and Weihe, 2016). Previous studies at Arcadia Lake and a nearby reservoir have found similar results of increased phosphorus in the hypolimnion (OWRB, 2000; OWRB, 2020b).

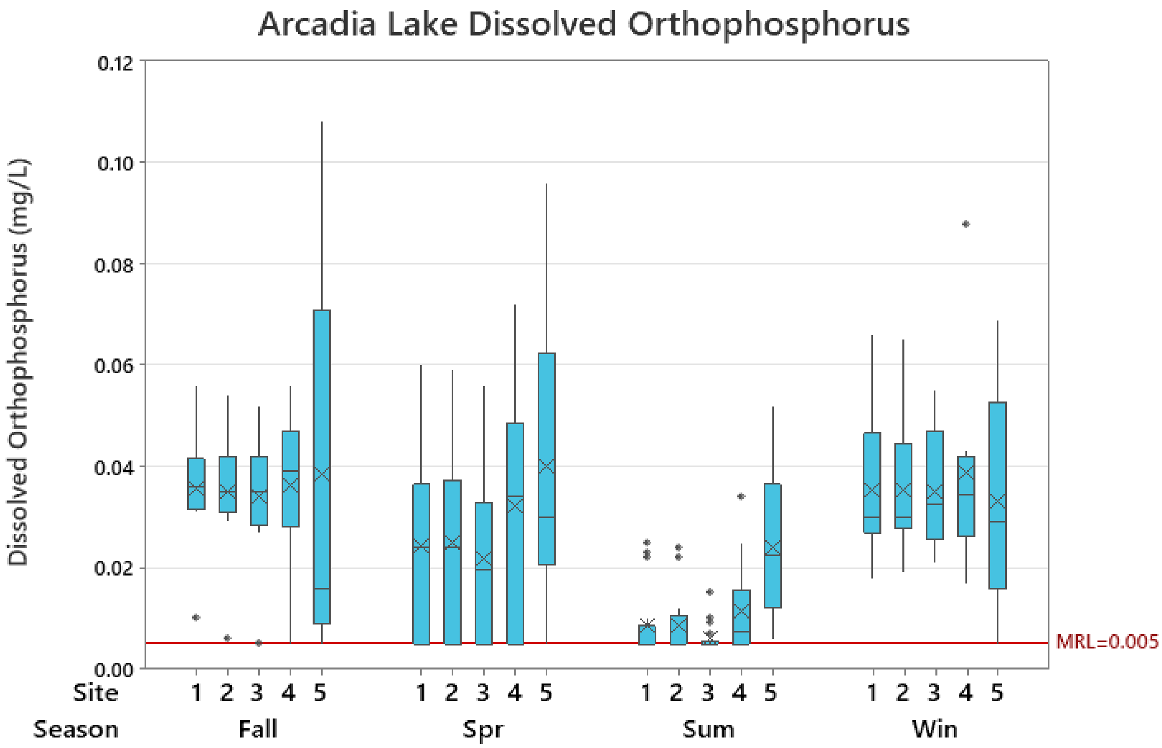


Figure 15. Arcadia Lake boxplots for dissolved orthophosphate surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8).

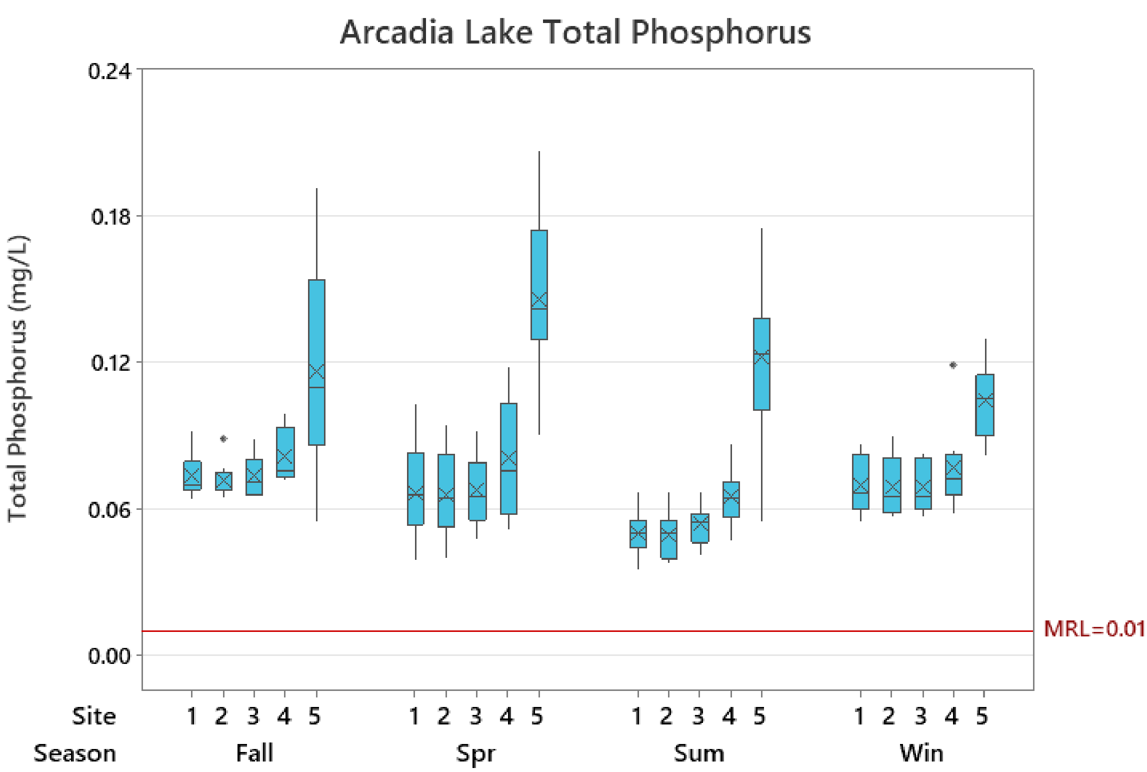


Figure 16. Arcadia Lake boxplots for total phosphorus surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8).

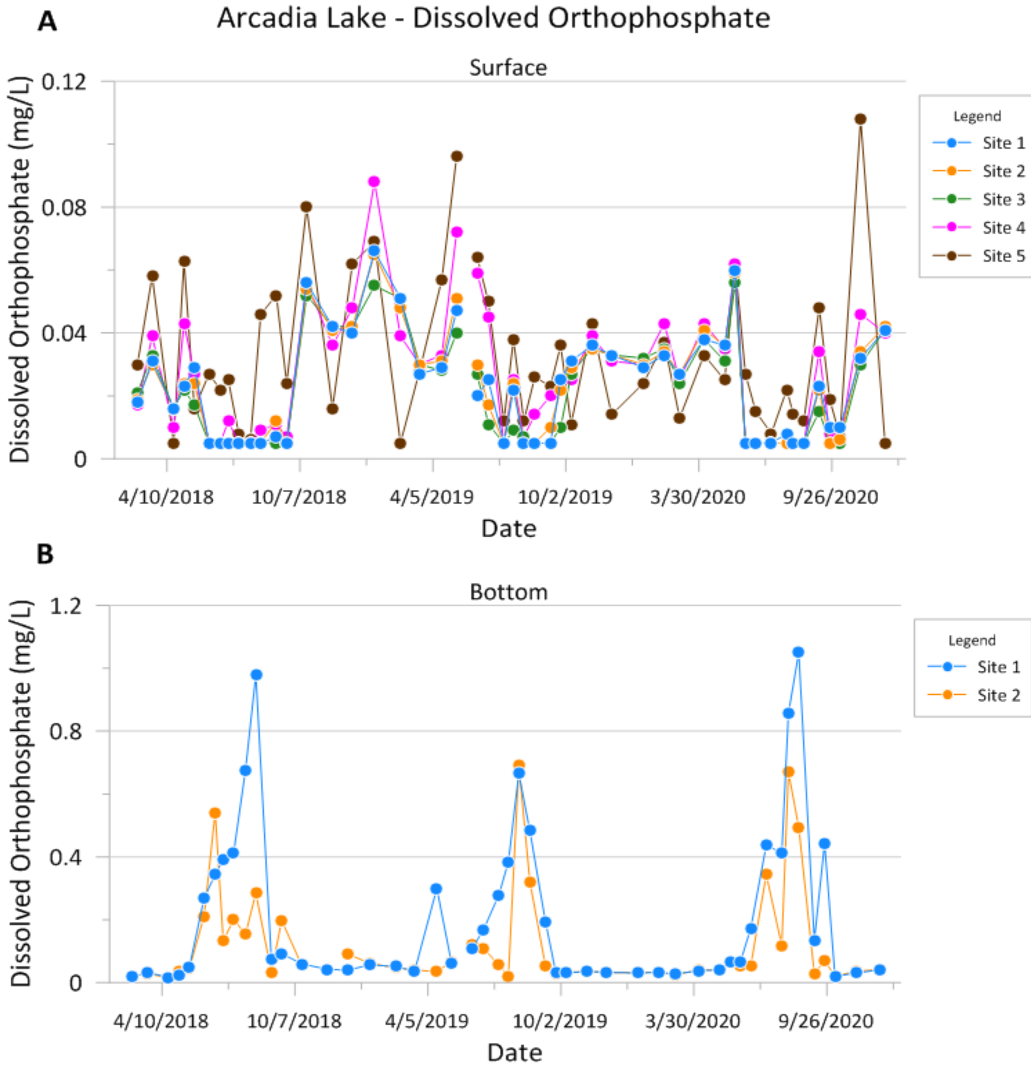


Figure 17. Arcadia Lake (a) surface and (b) bottom dissolved orthophosphate plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

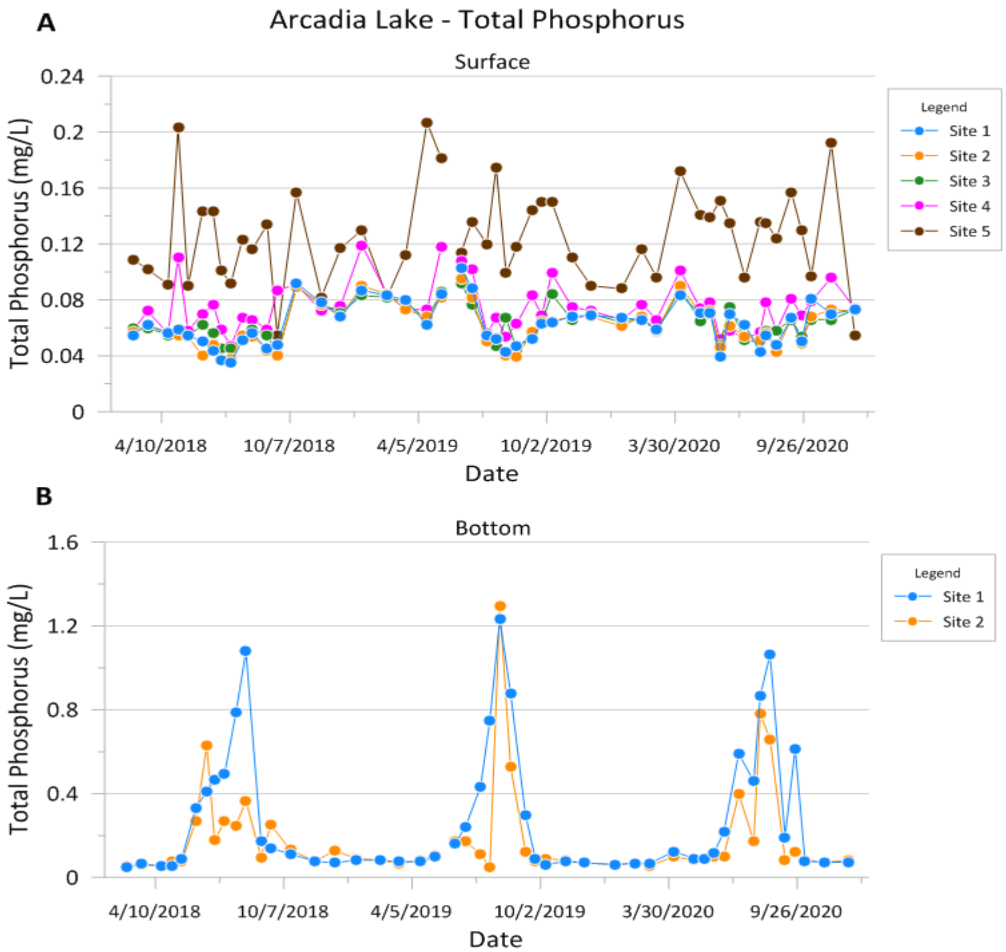


Figure 18. Arcadia Lake (a) surface and (b) bottom total phosphorus plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

The watershed monitoring locations showed TP concentrations consistent with the findings from the in-lake results (Figure 16, Figure 19, and Figure 20), particularly in-lake Site 5. Baseflow concentrations of TP showed minor fluctuations throughout the study period with seasonal variation. The mean TP concentration for Spring Creek at baseflow conditions is 0.06 mg/L with a median of 0.04 mg/L (Figure 19). There were two elevated baseflow TP concentrations that were observed during February of 2018 and May of 2019 showing concentrations of 0.22 mg/L and 0.24 mg/L, respectively (Figure 23). These heightened concentrations are also shown in collections from Wynn Creek (Figure 22) and Deep Fork (Figure 21) during the same time period. Deep Fork had a mean TP concentration 0.09 mg/L during baseflow conditions and is the predominant source of TP input into Arcadia Lake.

The TP contribution from Deep Fork to Arcadia Lake is shown best through the TRE collections, where TP concentrations reached a maximum of 3.65 mg/L for a single TRE during August of 2018 (Figure 21). TRE collections showed significantly higher concentrations at all three watershed sites, with mean TP concentrations at or near ten-fold higher for TRE collections than at baseflow (Figure 20). TP concentrations showed a direct relationship with respect to discharge at Spring Creek and Deep Fork, indicating that as discharge increases, so will TP (Figure 26 and Figure 24, respectively). This relationship was reflected in much higher TRE concentrations as compared to baseflow concentrations, as stated above. Wynn Creek did not show the same level of correlation as the other sites, but still showed a relationship (Figure 25).

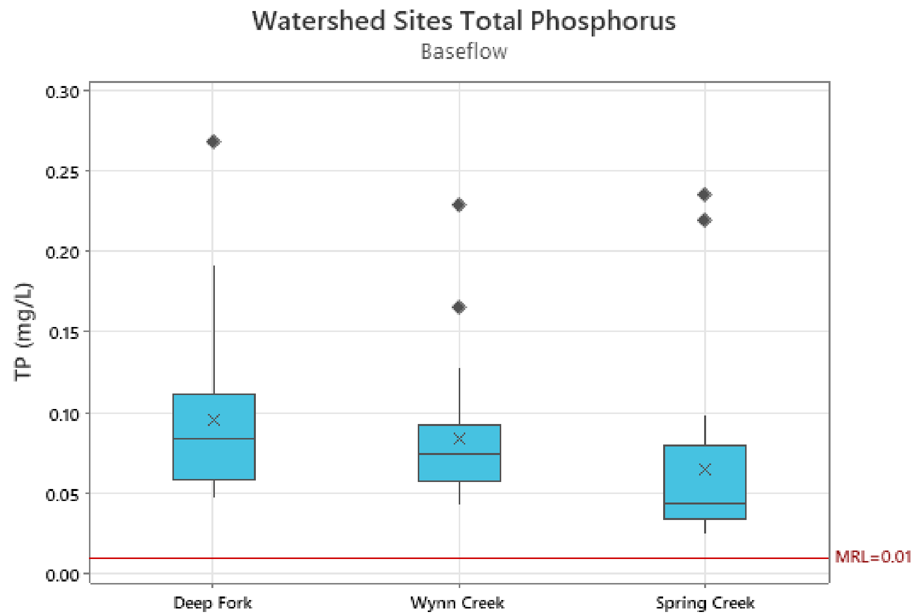


Figure 19. Watershed sites total phosphorus in mg/L, baseflow collections at Deep Fork (n=25), Wynn Creek (n=25), and Spring Creek (n=25). X denotes mean.

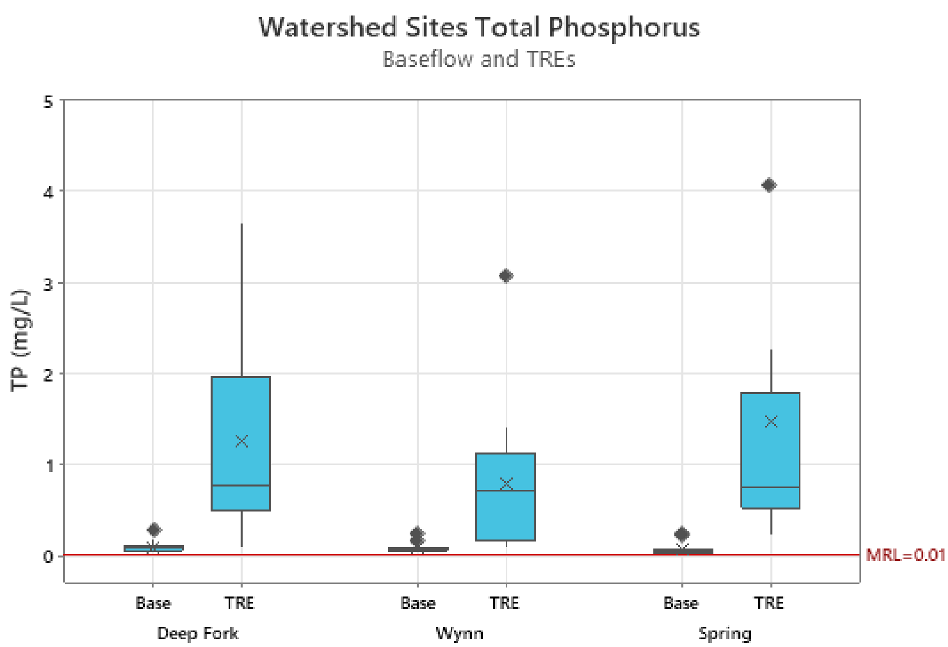


Figure 20. Watershed sites total phosphorus in mg/L, Deep Fork baseflow (n=25) and TRE (n=18), Wynn Creek baseflow (n=25) and TRE (n=16), and Spring Creek baseflow (n=25) and TRE (n=13). X denotes mean. One Spring Creek outlier value (6.22 mg/L) not shown.

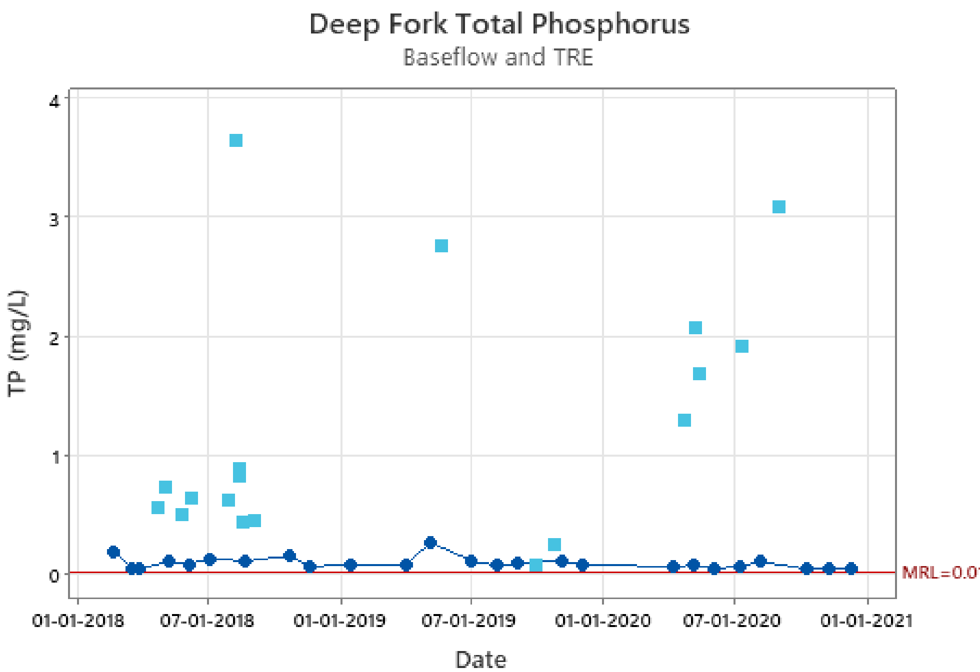


Figure 21. Total phosphorus in mg/L at Deep Fork over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=18) are represented by light blue squares.

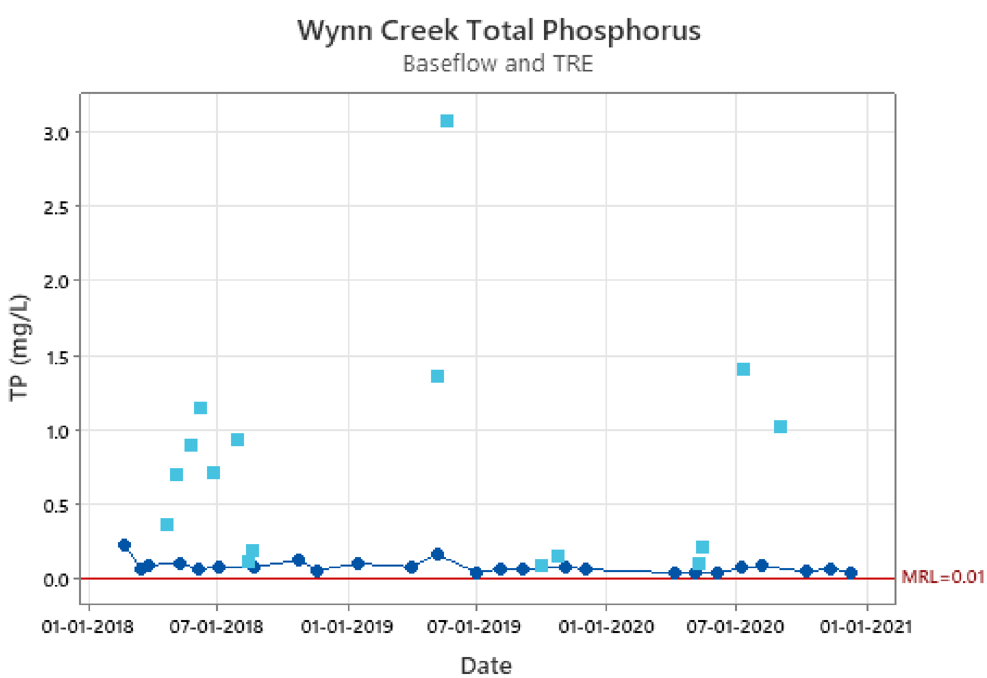


Figure 22. Total phosphorus in mg/L at Wynn Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=16) are represented by light blue squares.

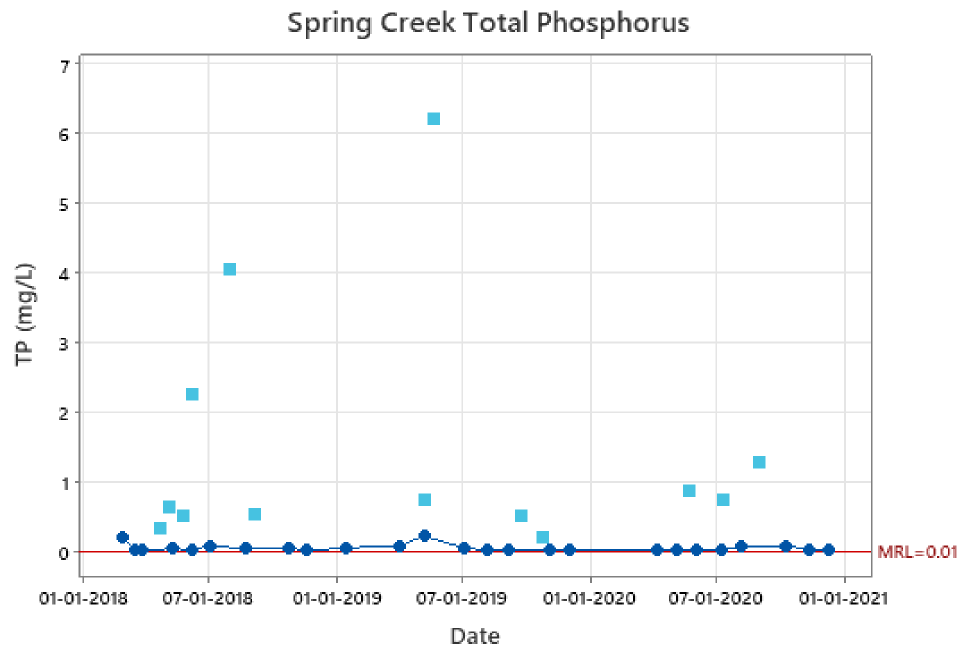


Figure 23. Total phosphorus in mg/L at Spring Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=13) are represented by light blue squares.

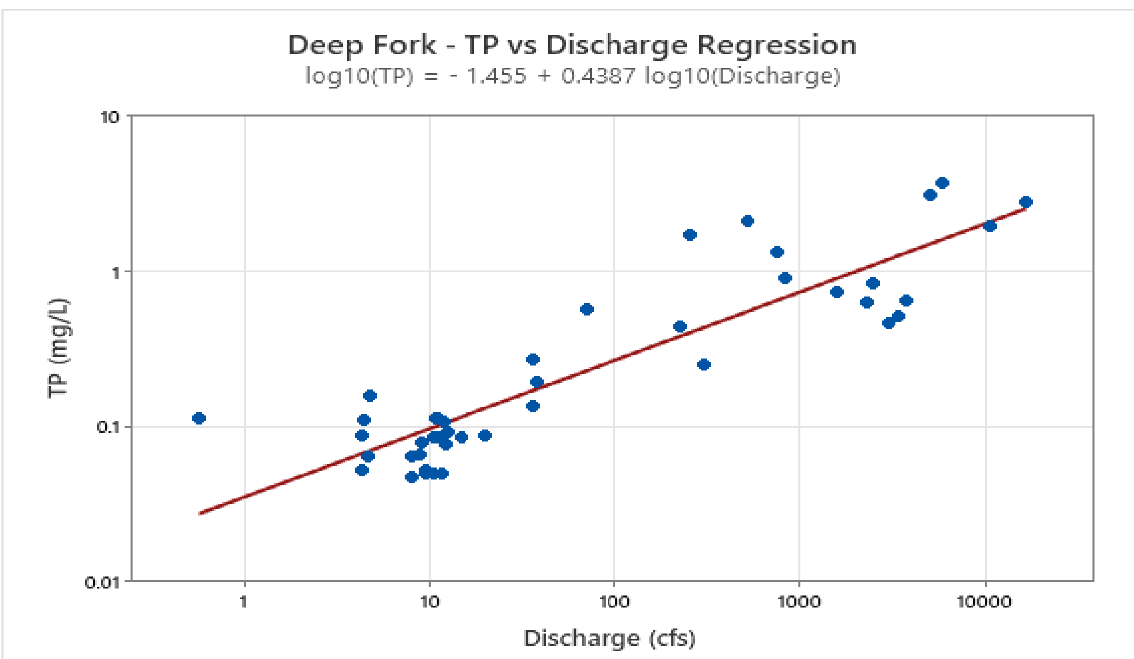


Figure 24. Total phosphorus versus discharge regression at Deep Fork. R^2=79.7%, p<0.001, n=43.

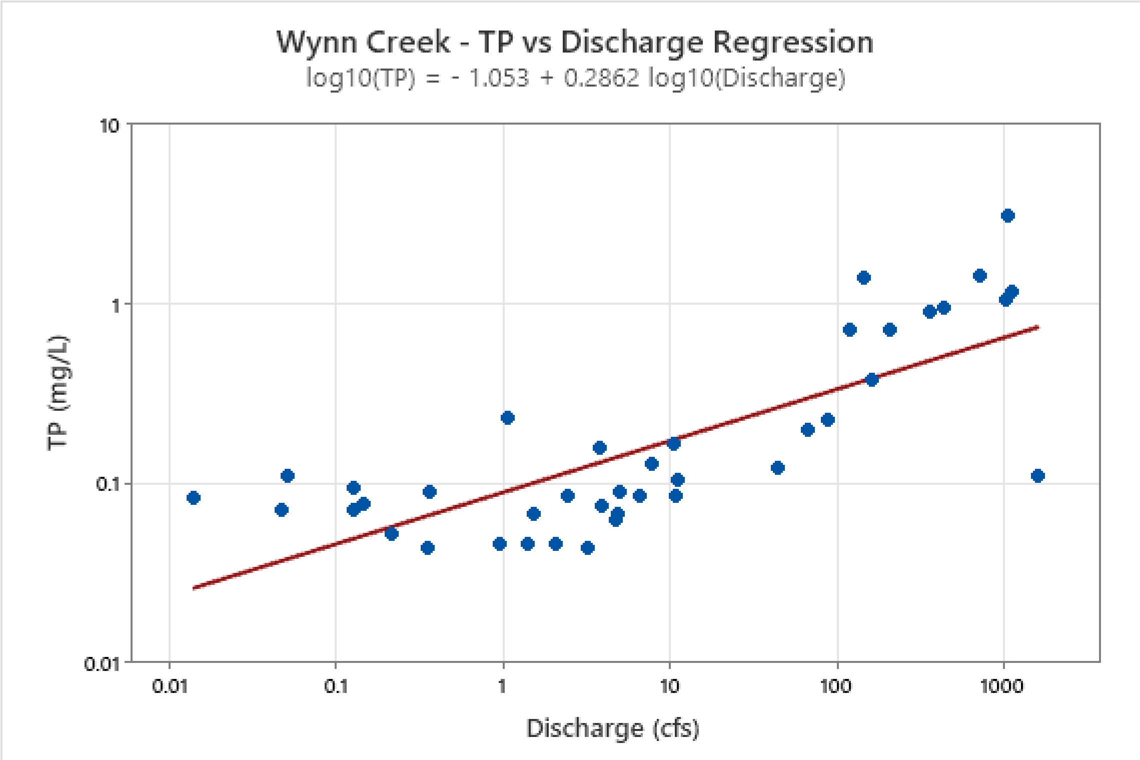


Figure 25. Total phosphorus versus discharge regression at Wynn Creek. R^2=59.3%, p<0.001, n=41.

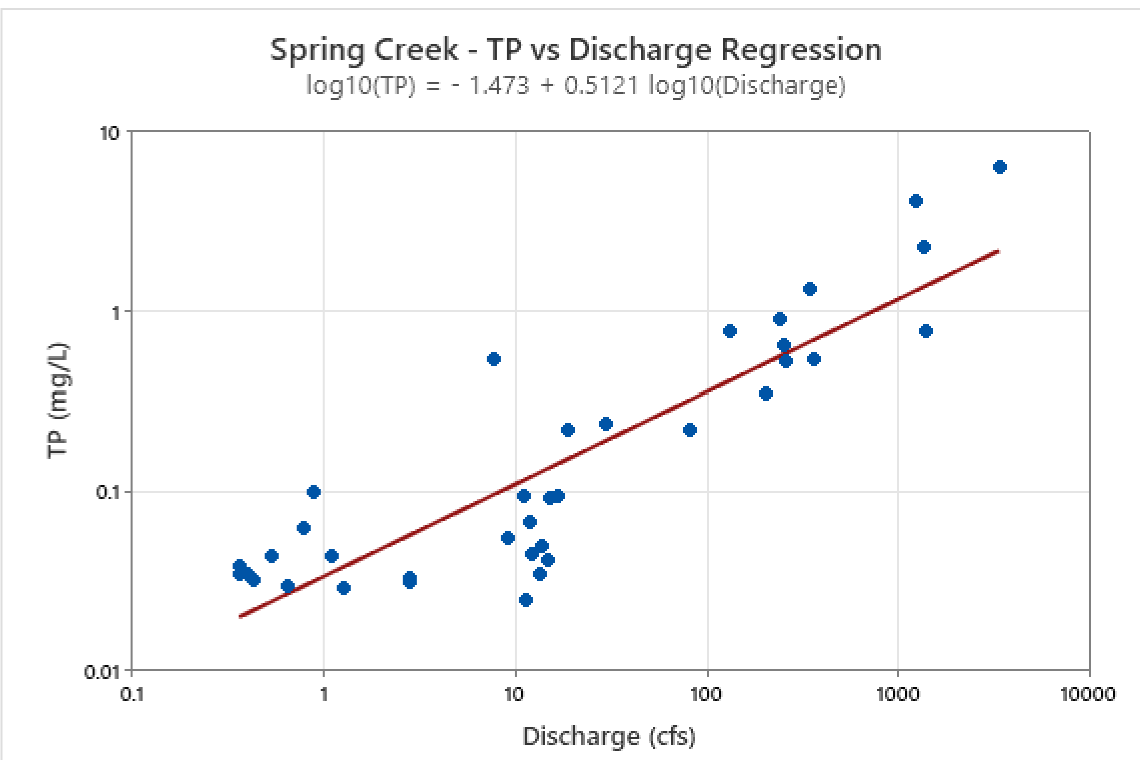


Figure 26. Total phosphorus versus discharge regression at Spring Creek. R^2=76.8%, p<0.001, n=38.

Nitrogen

Nitrate plus nitrite (NO3+NO2) concentrations in Arcadia Lake surface samples ranged from non-detected (0.05 mg/L) to 0.69 mg/L with a mean of 0.28 mg/L. Seasonally, boxplots of each site show surface NO3+NO2 concentrations were variable at each site except during the summer sampling period when concentrations were either at or below reporting limit (Figure 27). These results were confirmed from observing temporal data throughout the study period where NO3+NO2 concentrations in both surface and bottom samples occurred at or near reporting limits during lake stratification in summer and tended to covary, especially at Sites 1-4, when the lake was mixed (Figure 28). During the growing season, any NO3+NO2 at the surface was probably actively taken up by phytoplankton and with a hypoxic environment in the hypolimnion, no nitrification could occur to generate any NO3+NO2 in the lower depths of the lake.

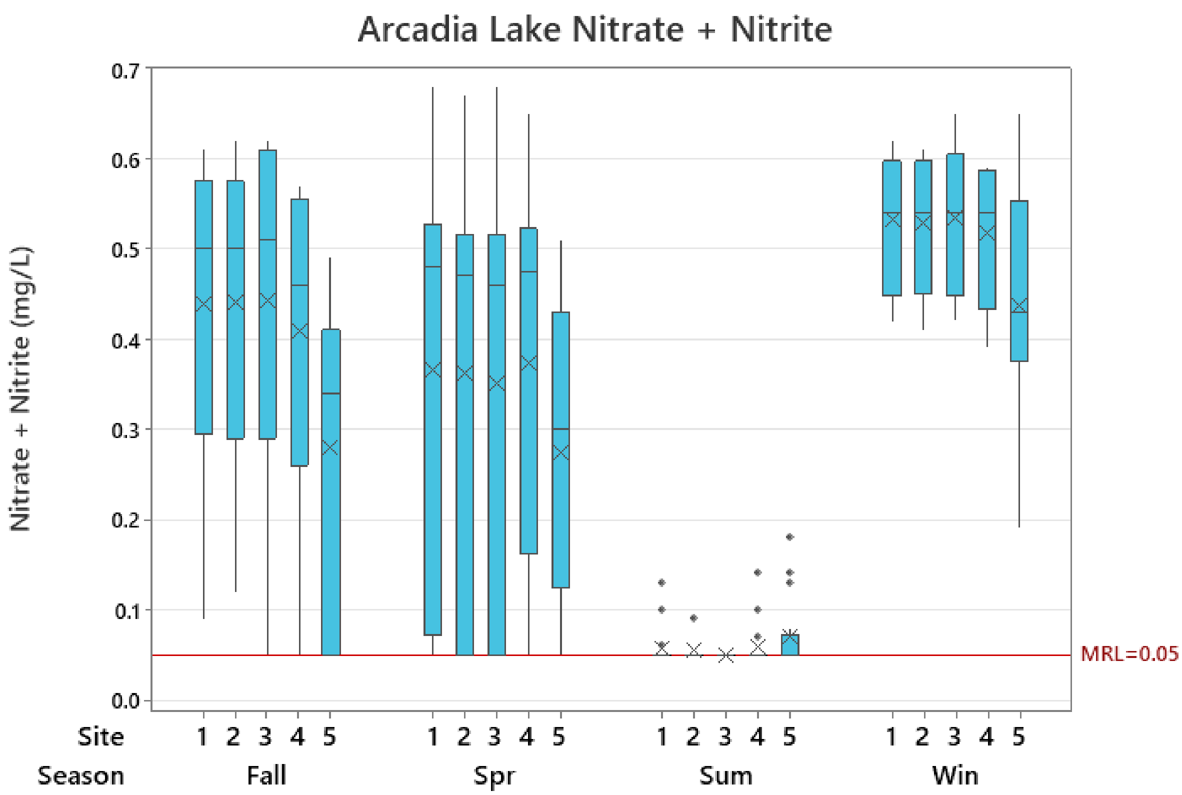


Figure 27. Arcadia Lake boxplots for nitrate plus nitrite (NO3+NO2) surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8). X denotes mean.

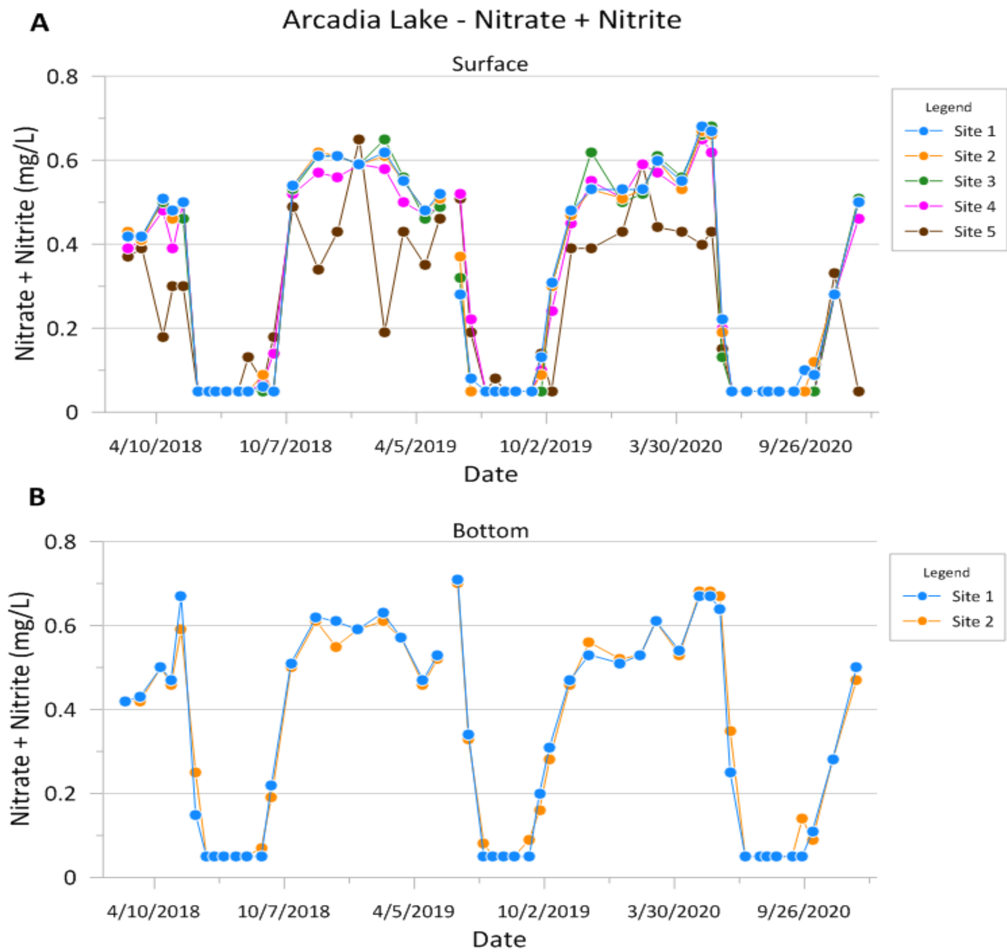


Figure 28. Arcadia Lake (a) surface and (b) bottom nitrate plus nitrite (NO3+NO2) plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

Other than a few peaks in winter and at Site 5 in spring, ammonia (NH3) concentrations were typically below reporting limits in surface samples (0.1 mg/L) and ranged up to 0.27 mg/L, with a mean of 0.108 mg/L (Figure 29 and Figure 30). As NH3 is usually quickly converted to NO3+NO2 by nitrification (Wetzel, 2001), any measurable NH3 may have been a result of input from Deep Fork or due to mineralization from organic matter in the resuspended sediments at the shallower sites. Bottom concentrations for NH3 showed a similar concentration pattern to DOP and TP where ammonification could occur in the reduced conditions in the hypolimnion during lake stratification and any subsequent nitrification was blocked by the lack of oxygen as an electron acceptor in this environment (Figure 30b). Increased concentrations of NH3 have been a common feature in the hypolimnion of eutrophic lakes (Wetzel, 2001; CADDIS 2017) and was observed in the Arcadia Lake Phase 1 report and at nearby Lake Thunderbird (OWRB, 2000; OWRB, 2020b). However, benthic concentrations were sometimes an order of magnitude greater than surface concentrations at this lake (when detected) and those reported from the previous study at Arcadia Lake, possibly due to the increased length of time or extent of hypolimnion hypoxia suggesting greater eutrophication is occurring in this system (OWRB, 2000).

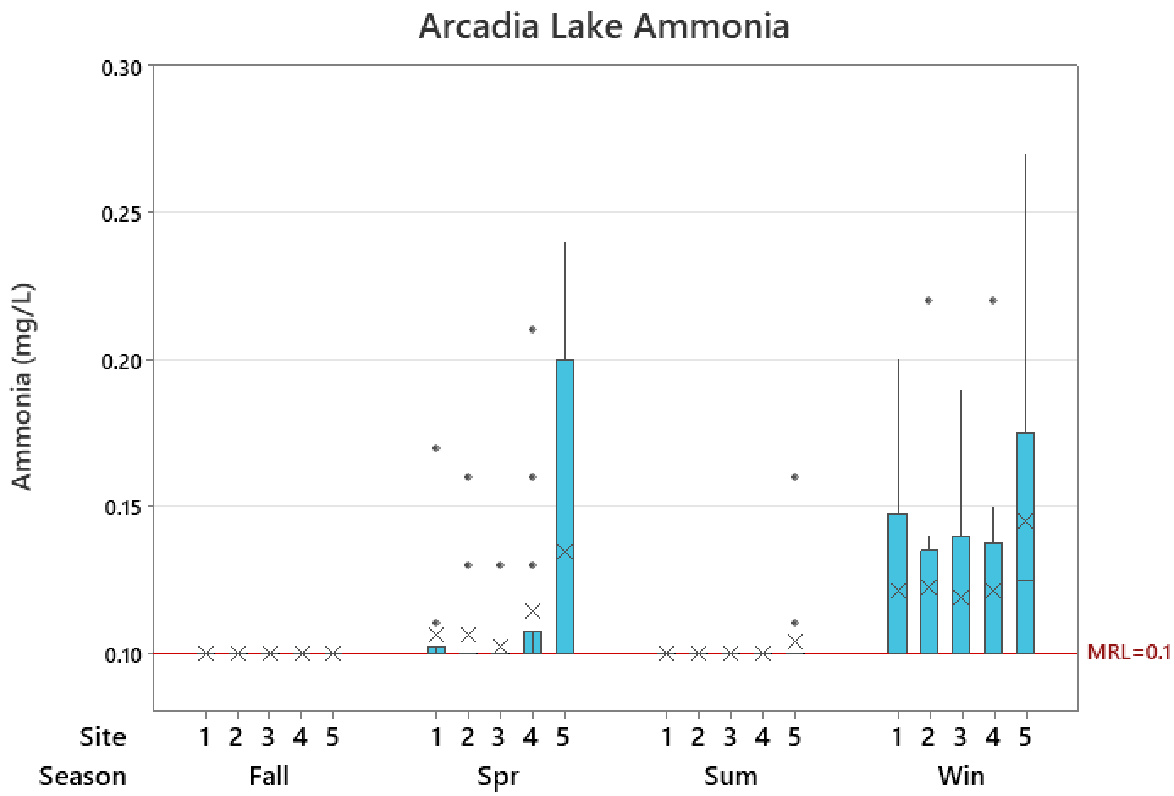


Figure 29. Arcadia Lake boxplots for ammonia surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8). X denotes mean.

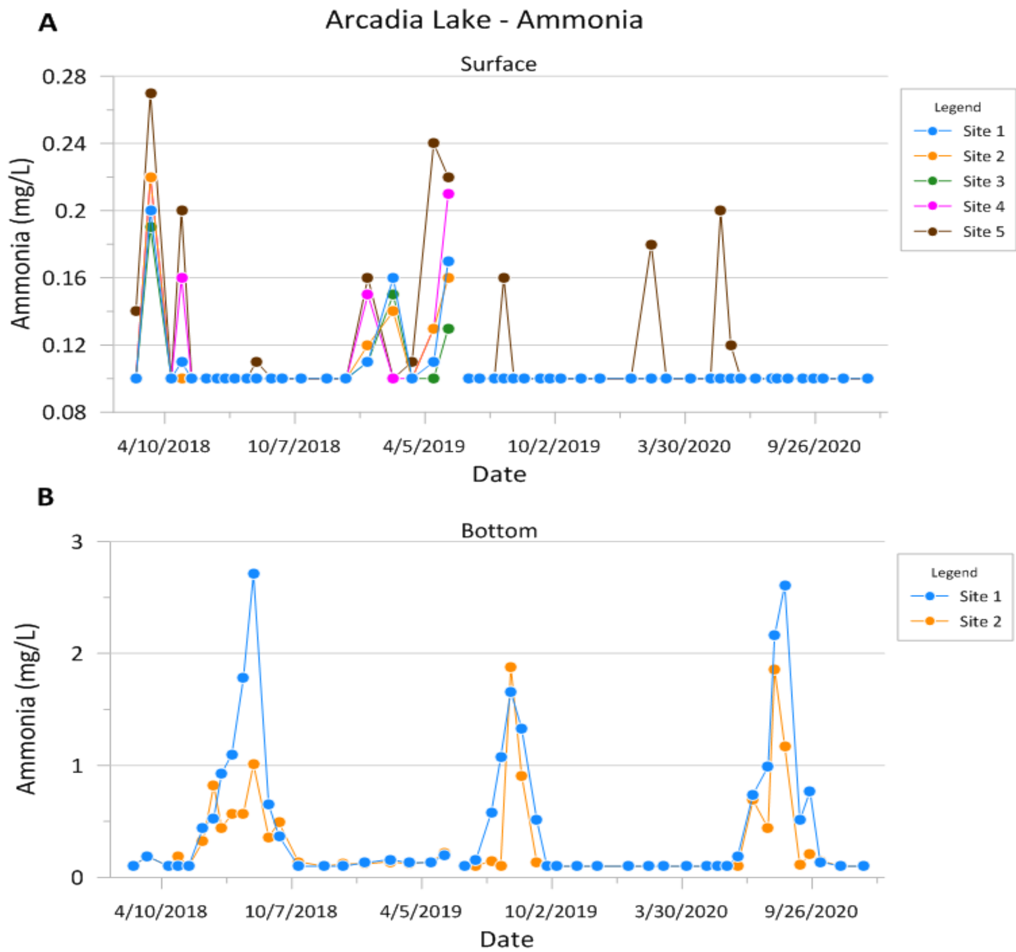


Figure 30. Arcadia Lake (a) surface and (b) bottom ammonia plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

Total Kjeldahl Nitrogen (TKN), ranging from 0.34 - 1.34 mg/L (mean 0.70 mg/L), and total nitrogen (TN), ranging from 0.54 - 1.64 mg/L (mean 0.97), were generally higher in surface samples at Site 5 but showed few other spatial or temporal patterns in the surface waters at the other lake sites (Figure 31 and Figure 32). These increased concentrations at Site 5 were likely due to allocthonous input of organic matter from the Deep Fork as seen with other water quality parameters. Both TKN and TN had increased concentrations in the bottom samples at Site 1 and Site 2 during lake stratification when compared to the surface concentrations (Figure 33 and Figure 34). These results are likely due in part to the increase in NH3 seen in the hypolimnion in this reducing environment as NH3 is a component of both TKN and TN. Other sources of TKN or TN in the hypolimnion were possibly from autocthonous input from the epilimnion (e.g., settling plankton).

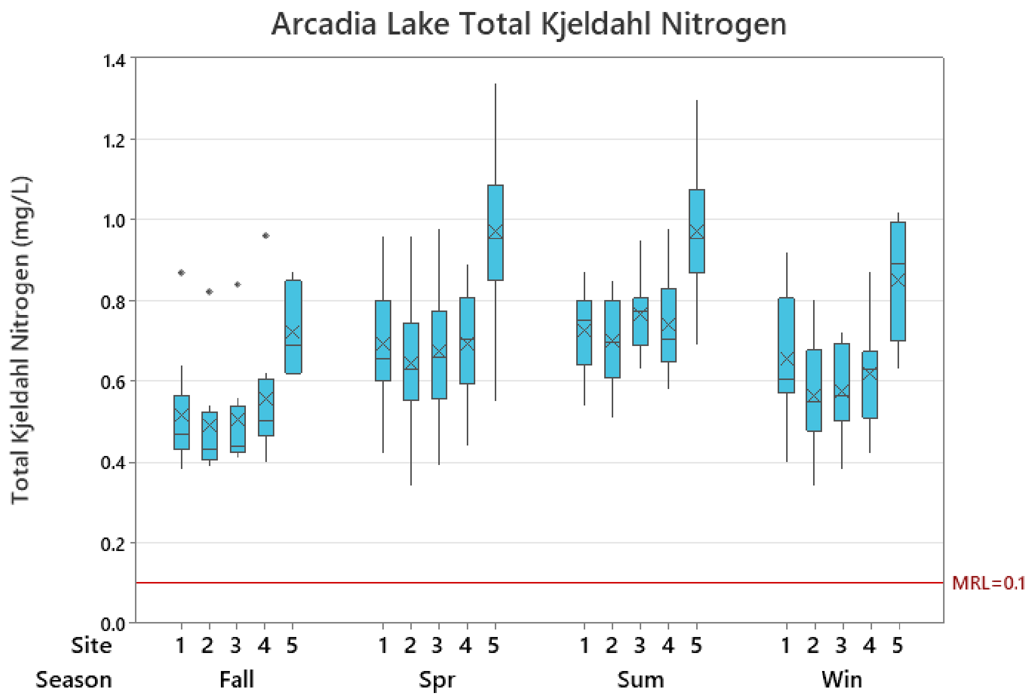


Figure 31. Arcadia Lake boxplots for Total Kjeldahl Nitrogen surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8). X denotes mean.

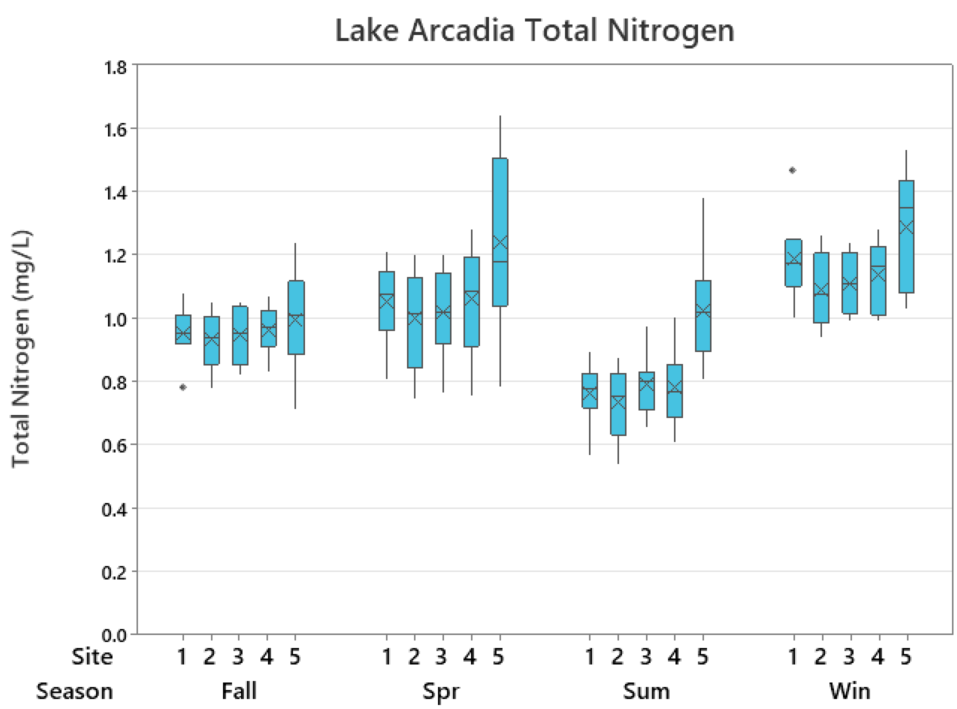


Figure 32. Arcadia Lake boxplots for total nitrogen surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8). X denotes mean.

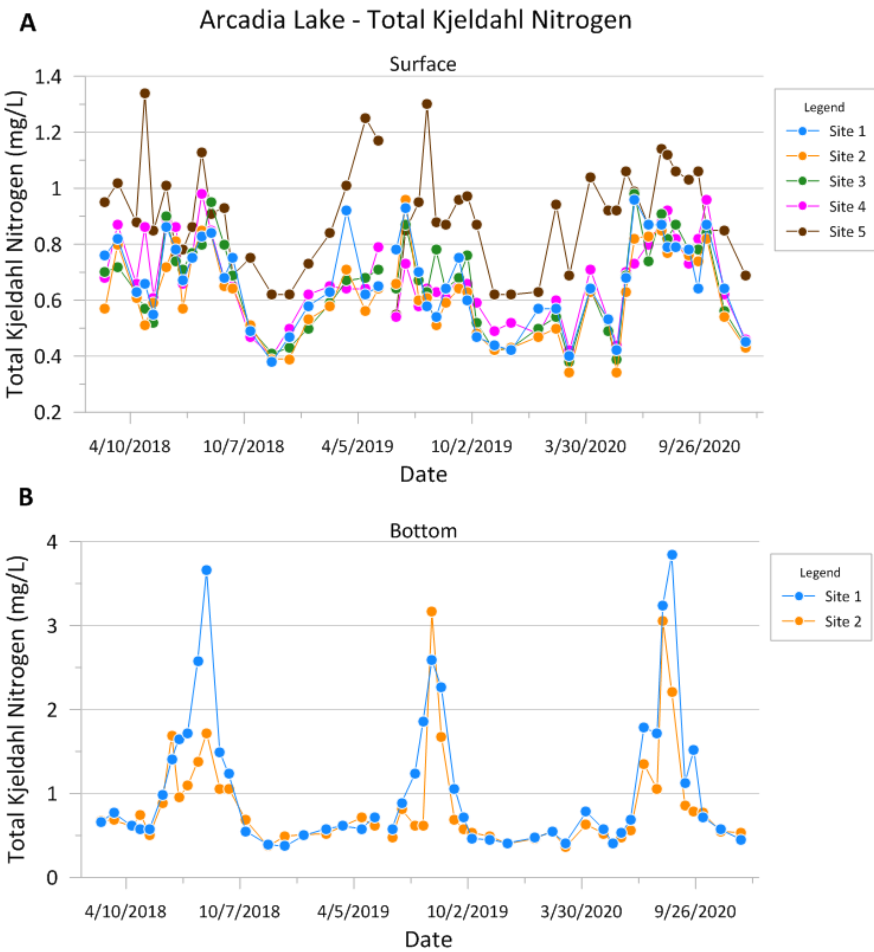


Figure 33. Arcadia Lake (a) surface and (b) bottom Total Kjedahl Nitrogen (TKN) plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

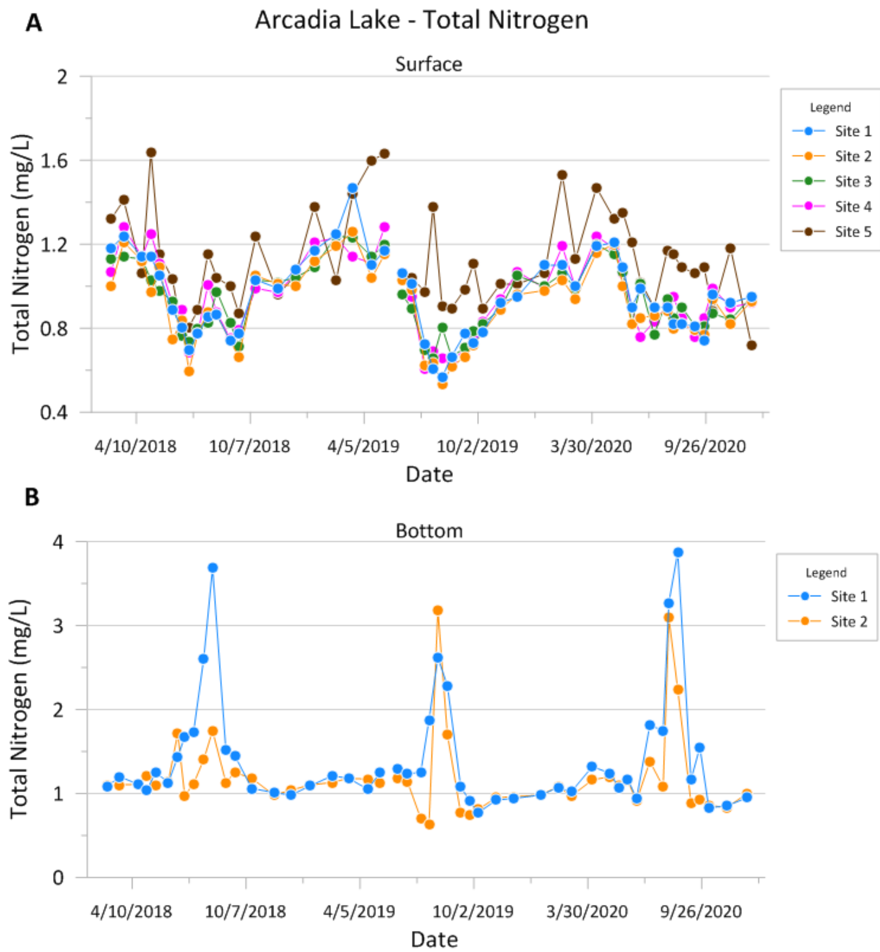


Figure 34. Arcadia Lake (a) surface and (b) bottom total nitrogen plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

Baseflow TN (Figure 35) for the watershed sites, similar to TP, mirrored observations at Site 5 (Figure 32 and Figure 34a). TN had slight variations at baseflow conditions for all three sites, with interquartile ranges of 0.55 mg/L at the Deep Fork site, and 0.39 mg/L for both Spring Creek and Wynn Creek locations (Figure 35, Figure 37, Figure 38, and Figure 39), and only one collection at the Deep Fork site exceeding 2.0 mg/L. TRE collections at all three watershed sites were significantly higher than baseflow collections, with median values of 3.87 mg/L, 3.41 mg/L, and 4.29 mg/L at Deep Fork, Wynn Creek, and Spring Creek, respectively (Figure 36, Figure 37, Figure 38, and Figure 39). These higher concentrations are likely attributed to high levels of urban runoff during precipitation events. All three sites showed strong relationships between TN and discharge (Figure 40, Figure 41, and Figure 42).

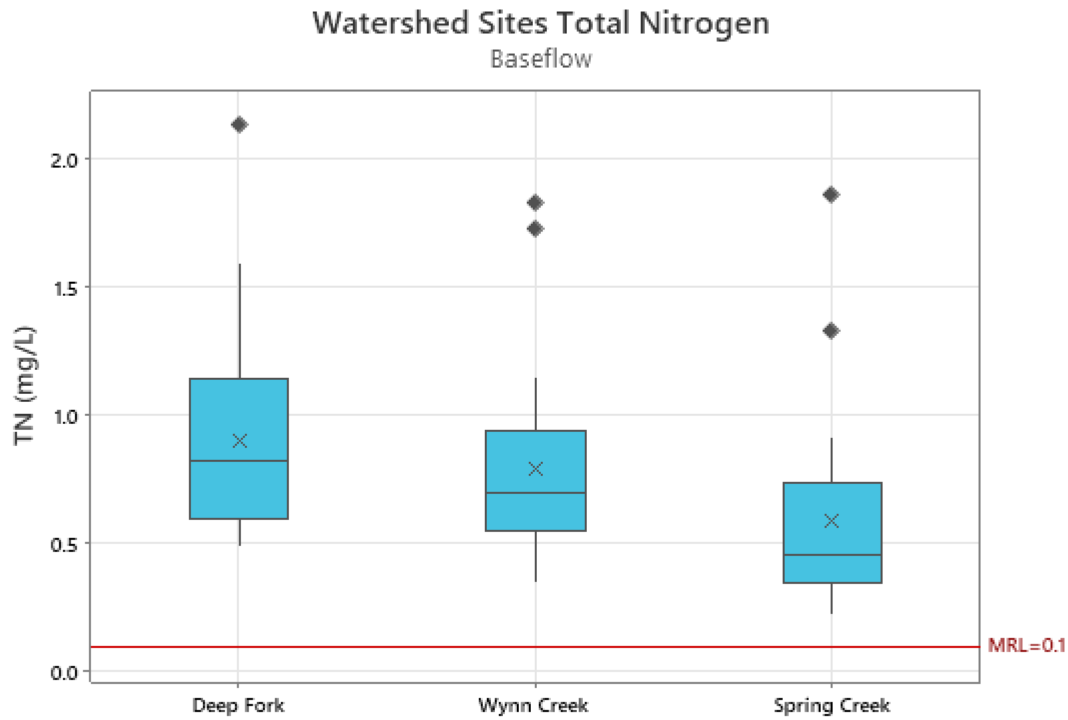


Figure 35. Watershed sites total nitrogen in mg/L, baseflow collections at Deep Fork (n=25), Wynn Creek (n=25), and Spring Creek (n=25). X denotes mean.

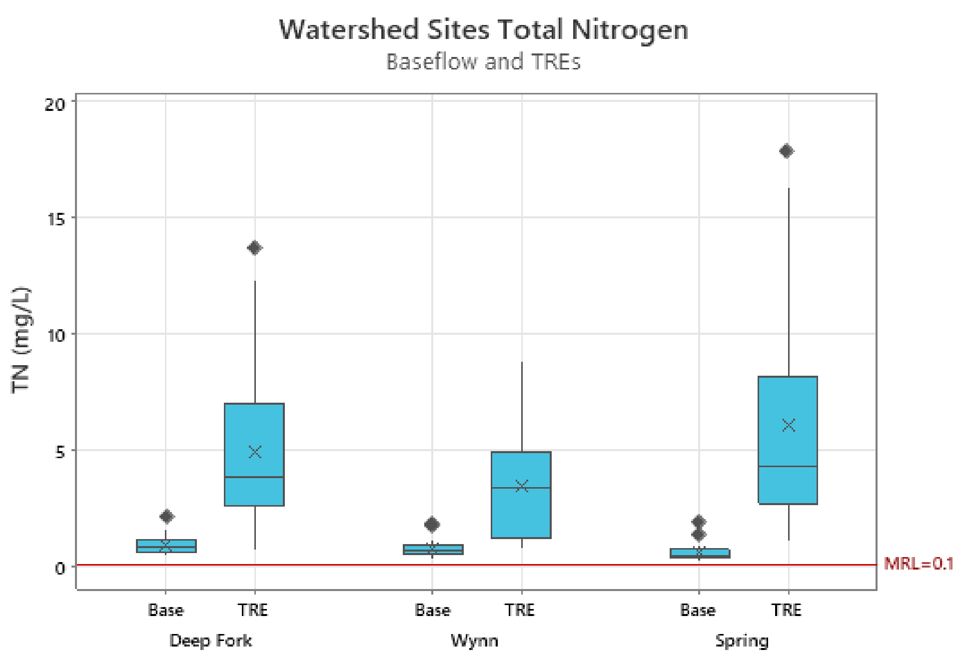


Figure 36. Watershed sites total nitrogen in mg/L, Deep Fork baseflow (n=25) and TRE (n=18), Wynn Creek baseflow (n=25) and TRE (n=16), and Spring Creek baseflow (n=25) and TRE (n=13). X denotes mean.

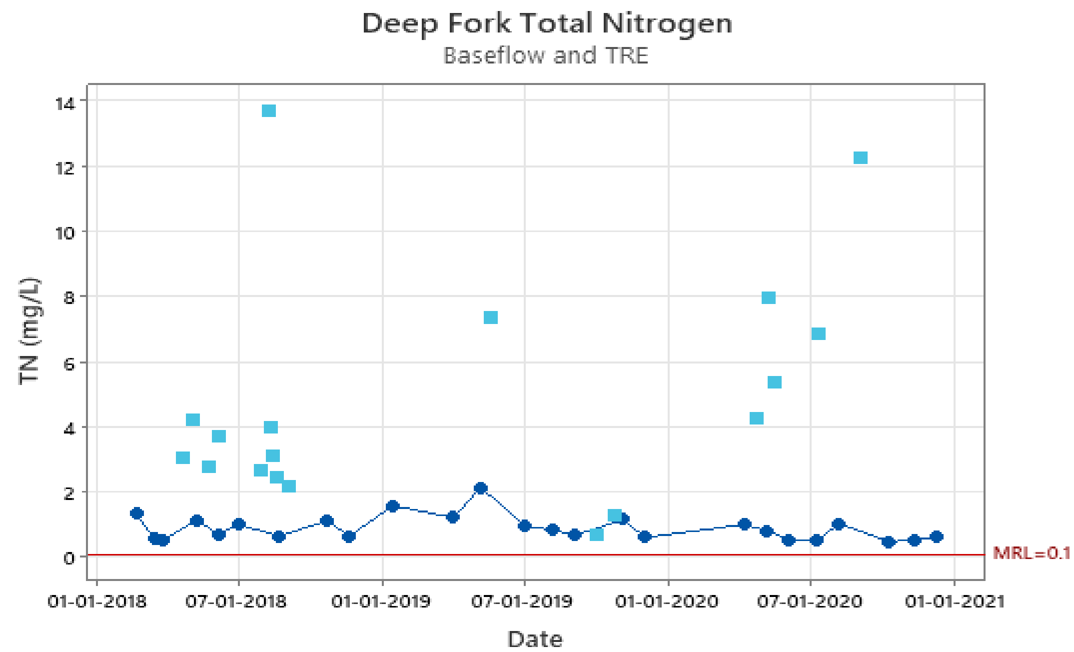


Figure 37. Total nitrogen in mg/L at Deep Fork over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=18) are represented by light blue squares.

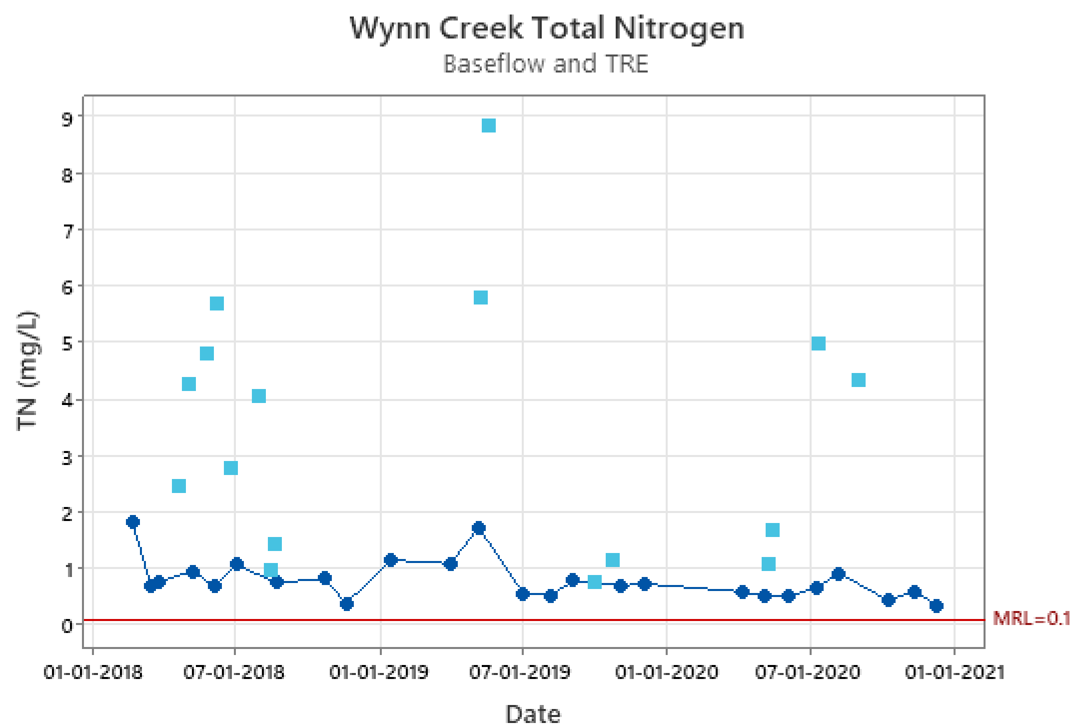


Figure 38. Total nitrogen in mg/L at Wynn Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=16) are represented by light blue squares.

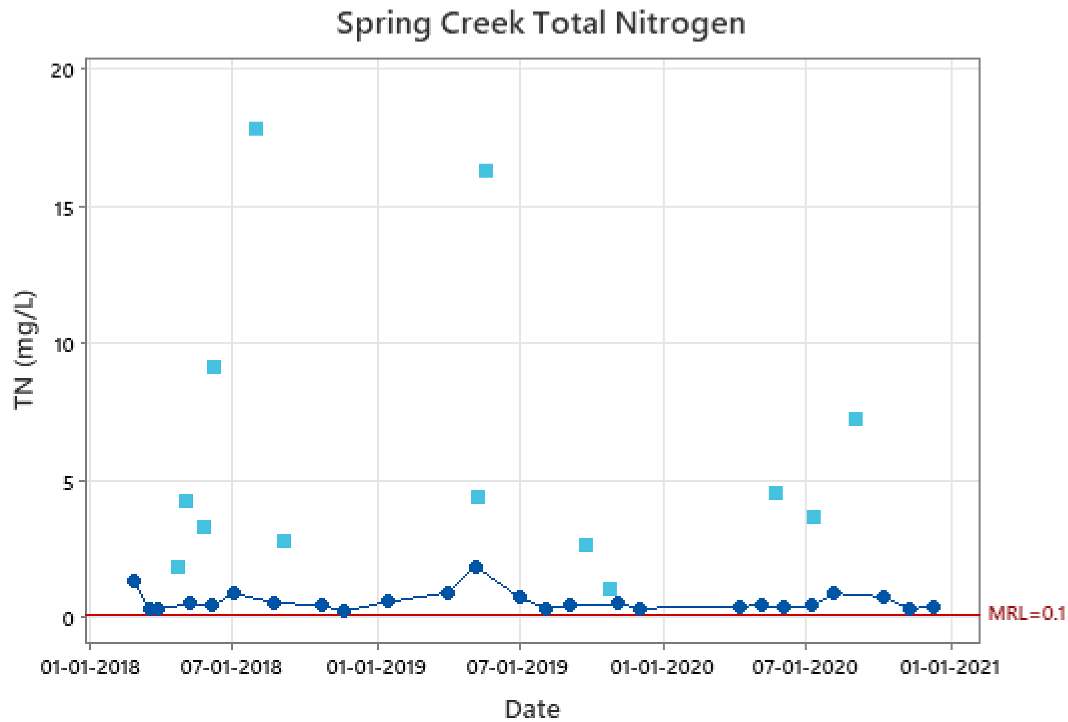


Figure 39. Total nitrogen in mg/L at Spring Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=13) are represented by light blue squares.

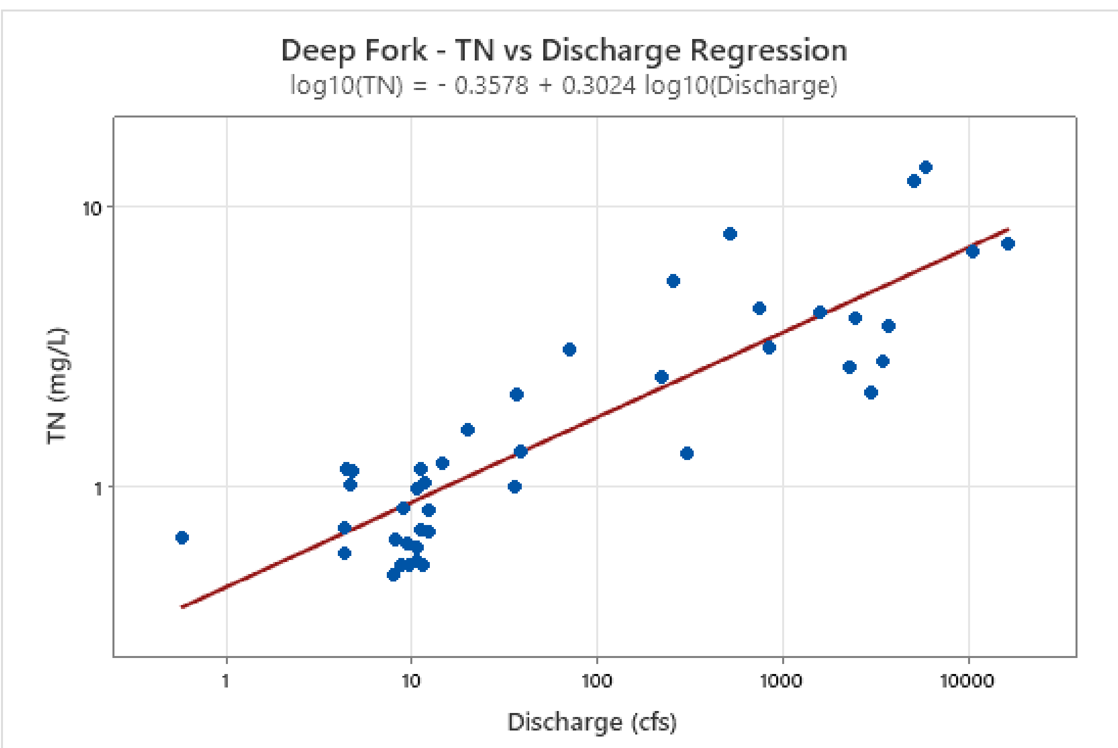


Figure 40. Total nitrogen versus discharge regression at Deep Fork. R^2=76.8%, p<0.001, n=43.

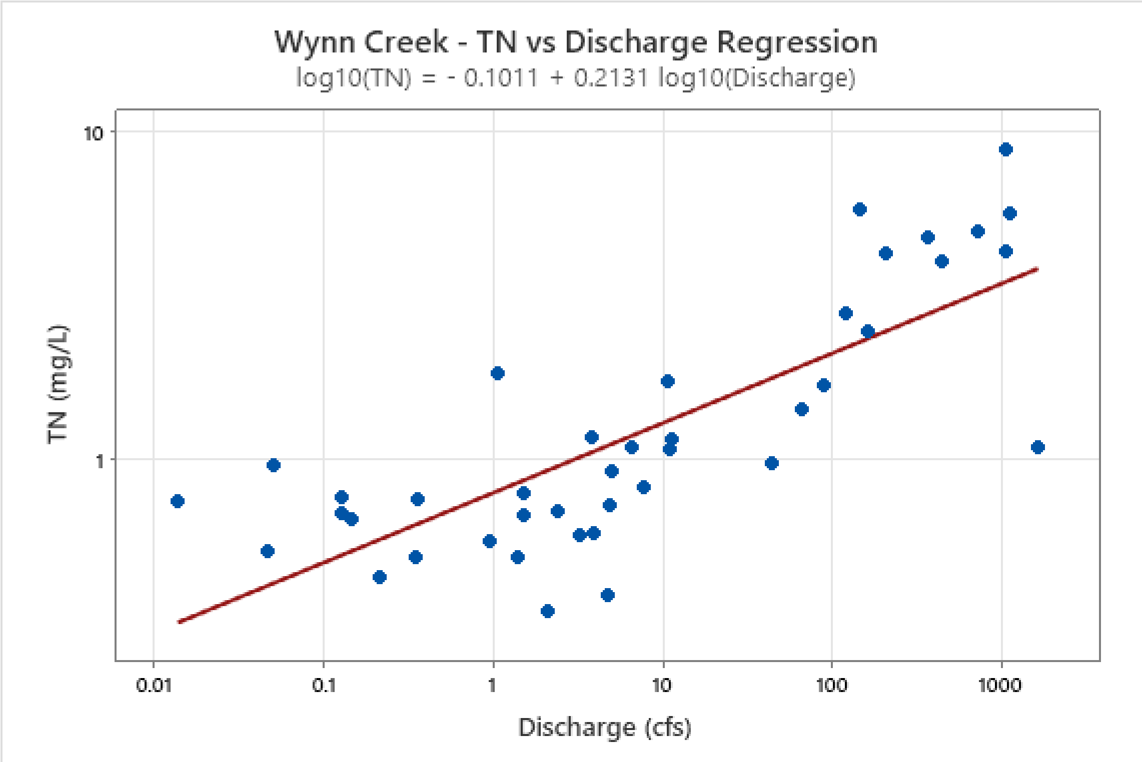


Figure 41. Total nitrogen versus discharge regression at Wynn Creek. R^2=61.3%, $p<0.001$, n=41.

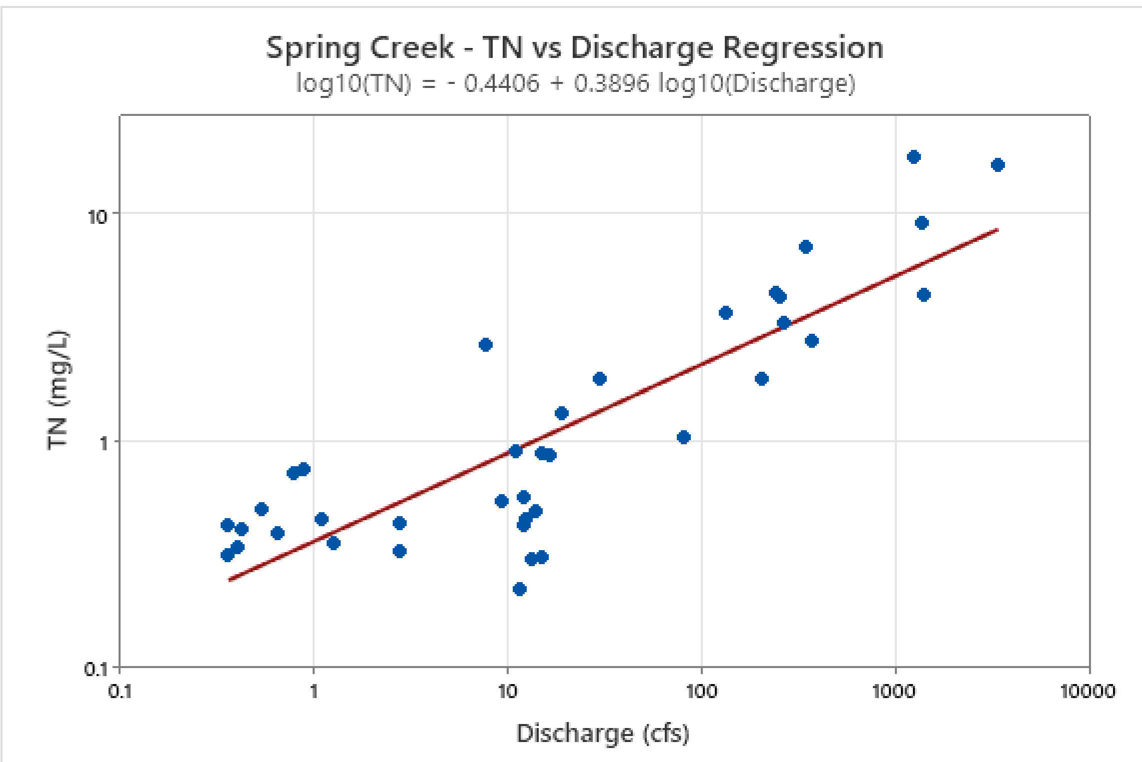


Figure 42. Total nitrogen versus discharge regression at Spring Creek. R^2=72.5%, $p<0.001$, n=38.

TKN at the watershed sites was the bulk component of measured TN values, comprising of over 50% of calculated TN values for 118 of the 122 samples collected across all three watershed sites (Figure 37, Figure 38, Figure 39, Figure 45, Figure 46, and Figure 47). Baseflow collection mean and median TKN values at the three sites were 0.67 mg/L and 0.63 mg/L respectively at Deep Fork, 0.65 mg/L and 0.60 mg/L respectively at Wynn Creek, and 0.46 mg/L and 0.40 mg/L respectively at Spring Creek (Figure 43). Mean and median TRE collection TKN values at the three sites were 4.53 mg/L and 3.55 mg/L, respectively at Deep Fork, 3.15 mg/L and 3.08 mg/L, respectively at Wynn Creek, and 5.75 mg/L and 3.86 mg/L, respectively at Spring Creek (Figure 44). Regression analysis showed relationships between discharge and TKN at all three sites, with Wynn Creek once again showing the weakest relationship of the three (Figure 48, Figure 49, and Figure 50).

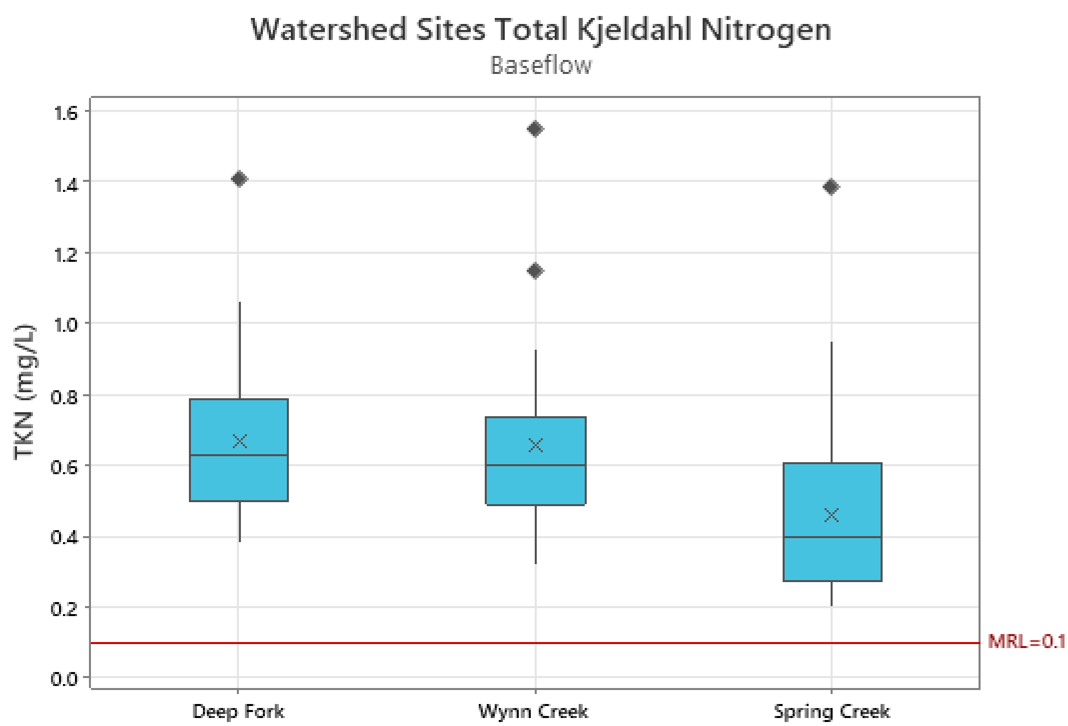


Figure 43. Watershed sites Total Kjeldahl Nitrogen in mg/L, baseflow collections at Deep Fork (n=25), Wynn Creek (n=25), and Spring Creek (n=25). X denotes mean.

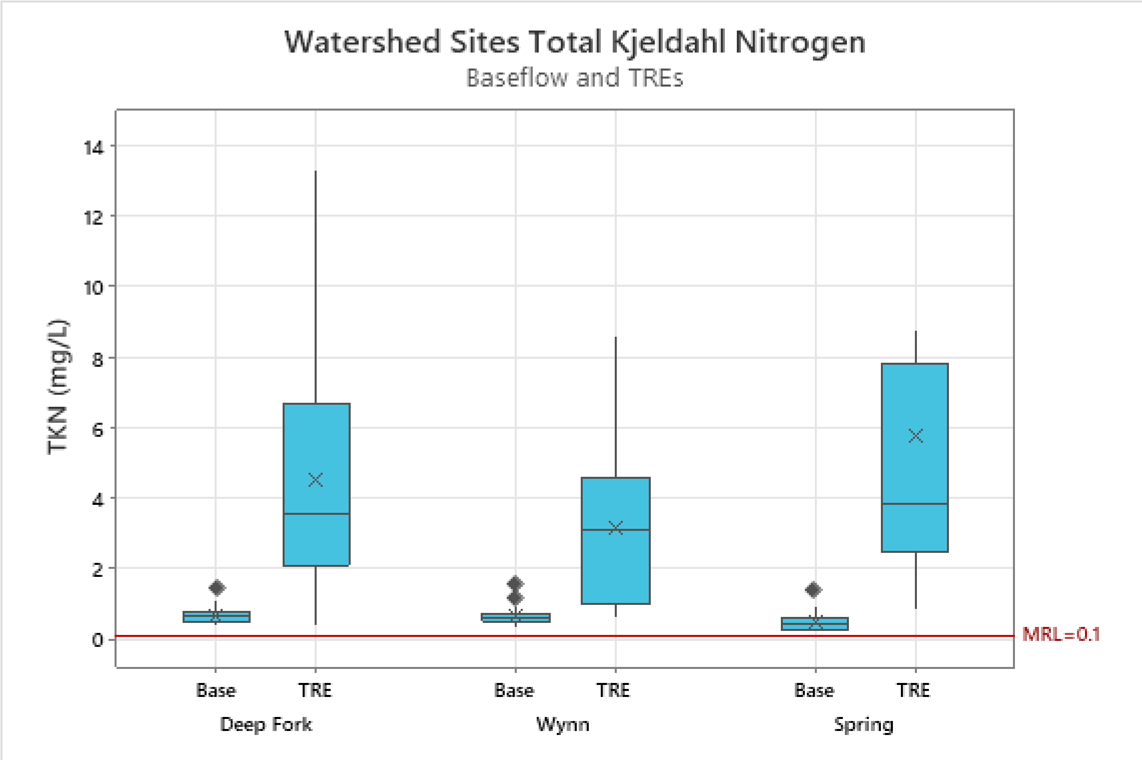


Figure 44. Watershed sites Total Kjeldahl Nitrogen in mg/L, Deep Fork baseflow (n=25) and TRE (n=18), Wynn Creek baseflow (n=25) and TRE (n=16), and Spring Creek baseflow (n=25) and TRE (n=13). X denotes mean. Two Spring Creek TRE outliers (16.0 mg/L and 17.6 mg/L) not shown.

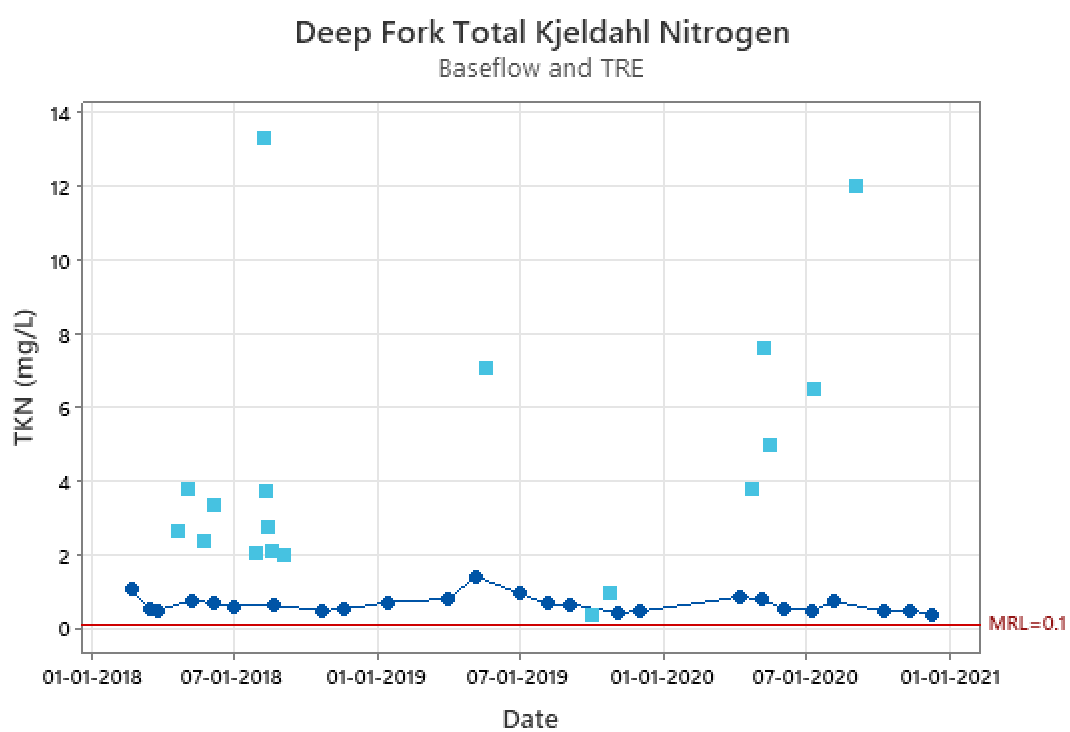


Figure 45. Total Kjeldahl Nitrogen in mg/L at Deep Fork over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=18) are represented by light blue squares.

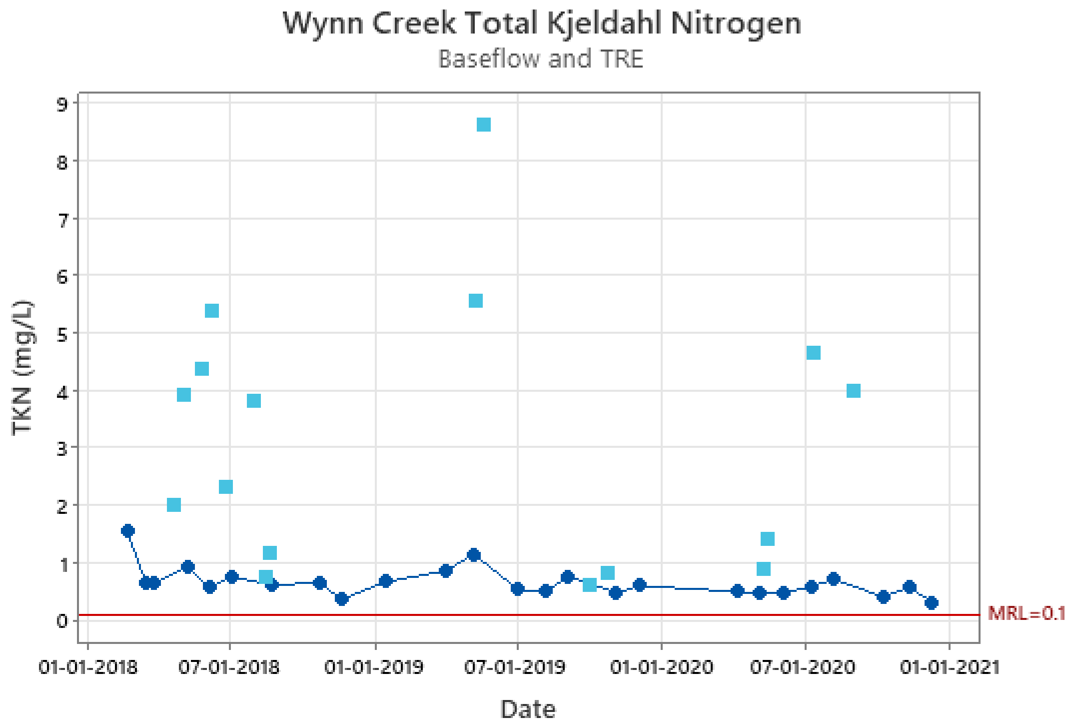


Figure 46. Total Kjeldahl Nitrogen in mg/L at Wynn Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=16) are represented by light blue squares.

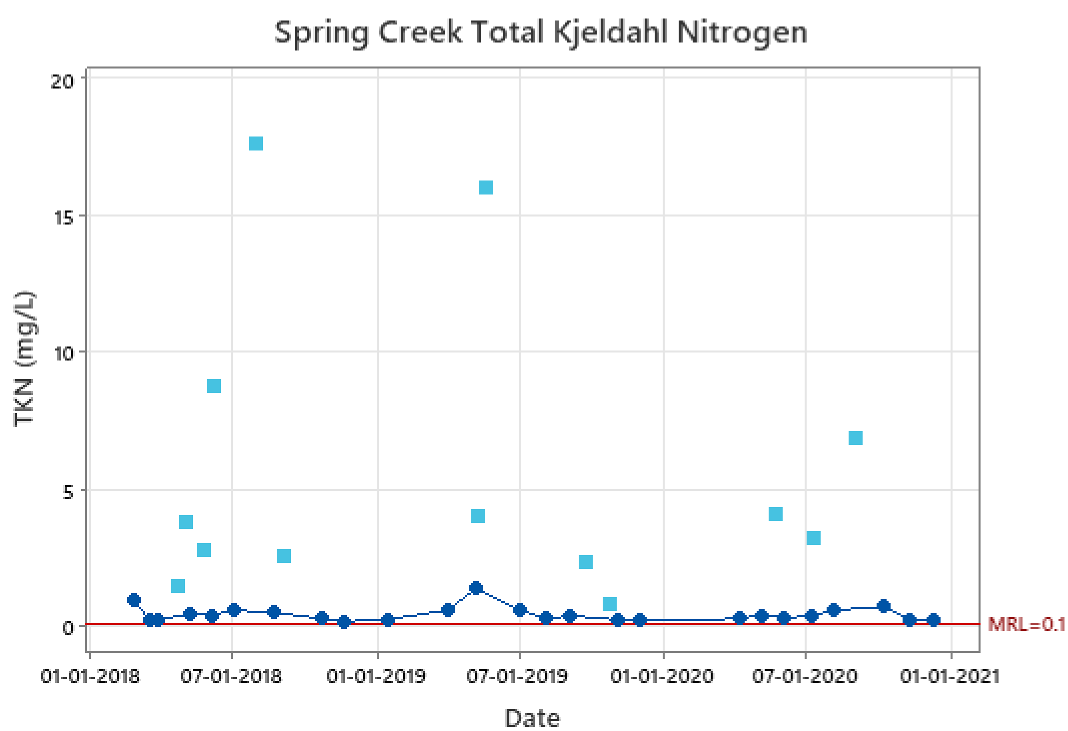


Figure 47. Total Kjeldahl Nitrogen in mg/L at Spring Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=13) are represented by light blue squares.

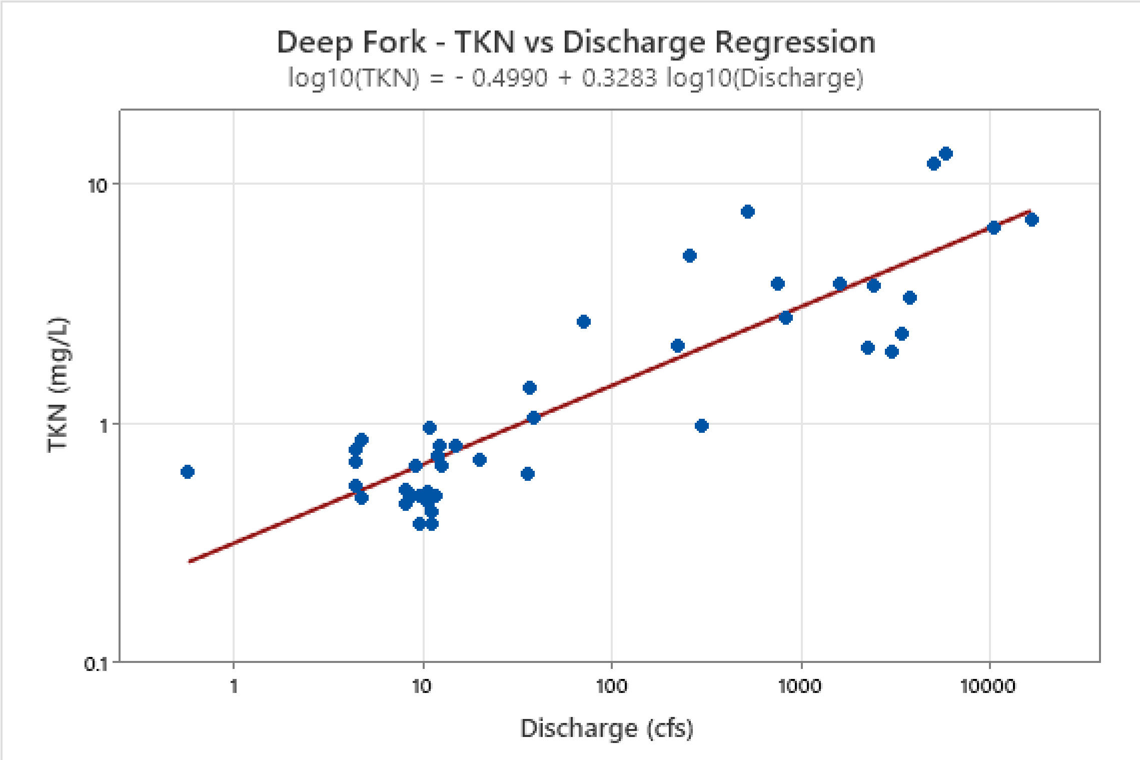


Figure 48. Total Kjeldahl Nitrogen versus discharge regression at Deep Fork. R^2=77.7%, p<0.001, n=43.

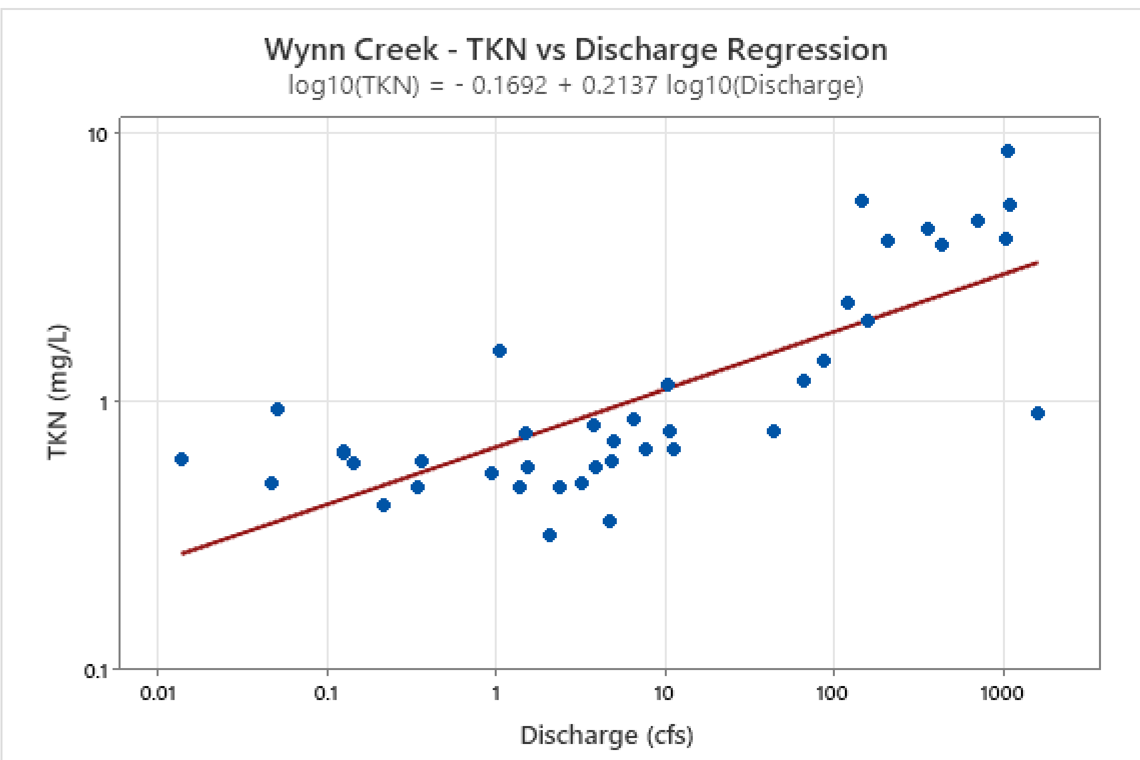


Figure 49. Total Kjeldahl Nitrogen versus discharge regression at Wynn Creek. R^2=58.7%, p<0.001, n=41.

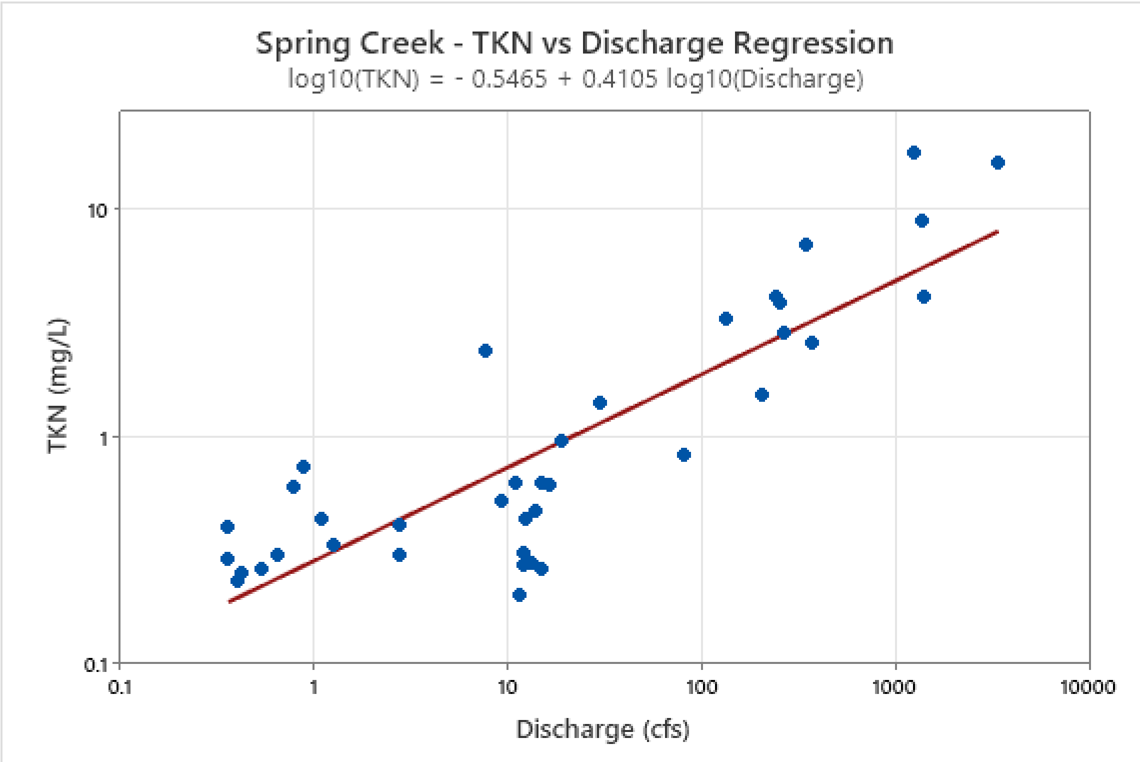


Figure 50. Total Kjeldahl Nitrogen versus discharge regression at Spring Creek. R^2=72.7%, p<0.001, n=38.

Analysis of ammonia was added to the watershed monitoring sites during the second half of the monitoring period, and subsequently has fewer samples for each site. During baseflow conditions, ammonia was not measured above the 0.1 mg/L method reporting limit, therefore a graphical representation of ammonia at baseflow conditions is not included by itself but is included alongside TRE collections in Figure 51. The lack of measurable ammonia above detection is likely due to high in-stream nitrification during aerobic baseflow conditions (Allen, 1995).

Higher levels of ammonia above the MRL were only observed during TREs, likely due to a lower water column nitrification rate during these events, as well as ammonia and other nutrient release from disturbed sediments. The highest ammonia sample observed was 0.38 mg/L at the Deep Fork during a TRE collection on 5/8/2020 (Figure 52, Figure 53, and Figure 54).. TRE collection mean and median ammonia values at the three sites were 0.20 mg/L and 0.17 mg/L respectively at Deep Fork, 0.12 mg/L and 0.1 (MRL) mg/L respectively at Wynn Creek, and 0.19 mg/L and 0.2 mg/L respectively at Spring Creek (Figure 51). Due to baseflow measurements and a low number of overall samples, regression analysis was not performed for ammonia.

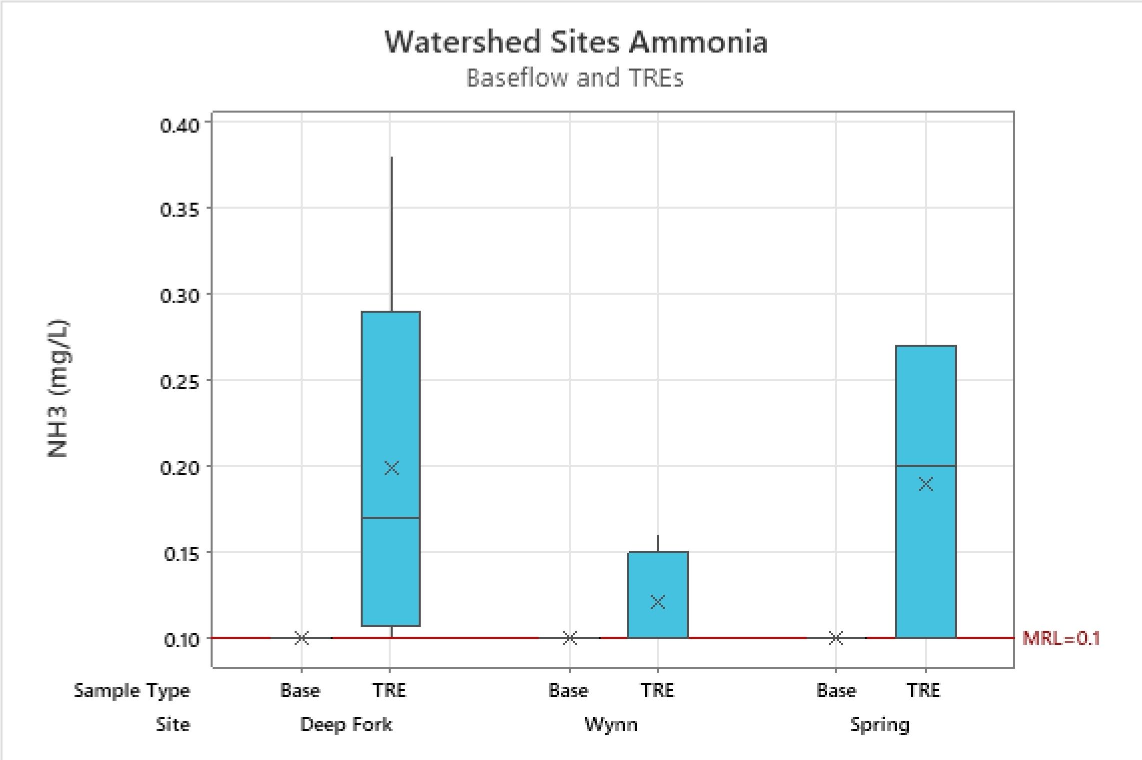


Figure 51. Watershed sites ammonia in mg/L, Deep Fork baseflow (n=13) and TRE (n=8), Wynn Creek baseflow (n=13) and TRE (n=7), and Spring Creek baseflow (n=13) and TRE (n=6). X denotes mean. 0% of baseflow collections were measured above the MRL of 0.1 mg/L.

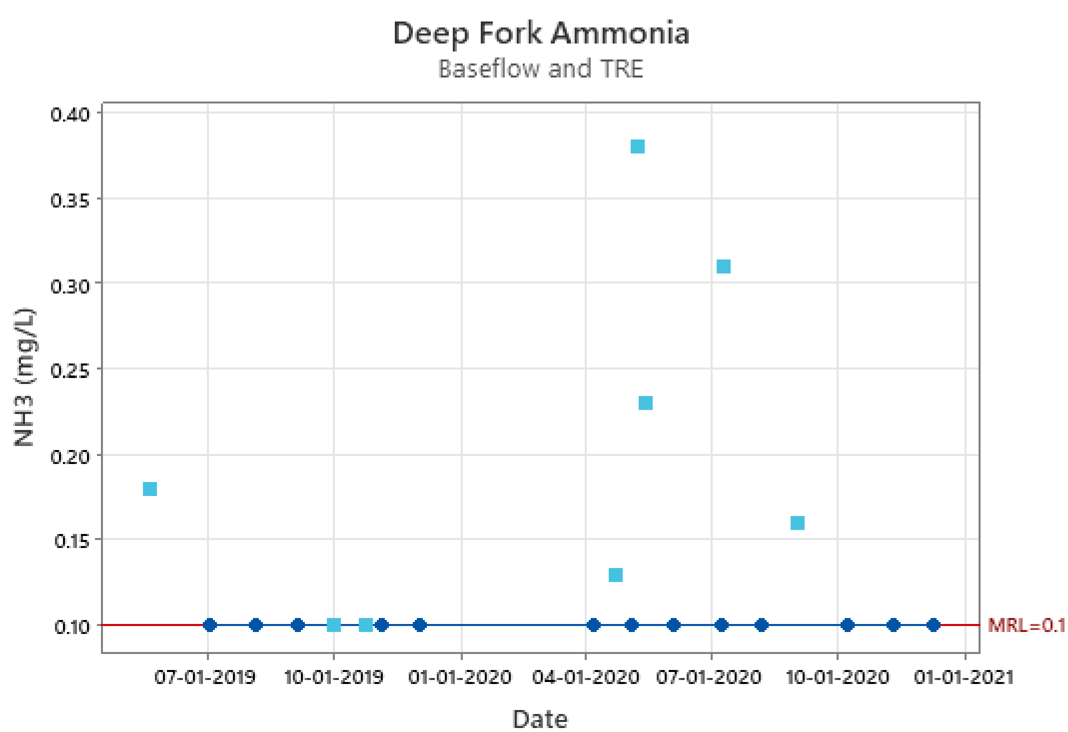


Figure 52. Ammonia in mg/L at Deep Fork over the sample period, baseflow collections (n=13) are represented by dark blue dots with connect line, and TRE collections (n=8) are represented by light blue squares.

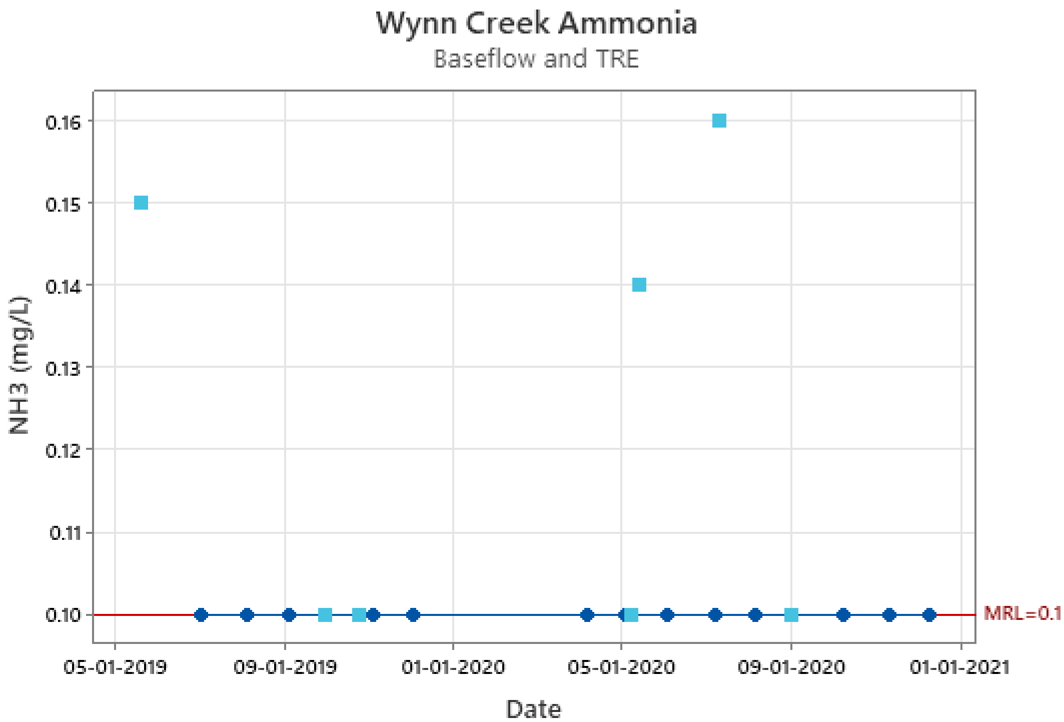


Figure 53. Ammonia in mg/L at Wynn Creek over the sample period, baseflow collections (n=13) are represented by dark blue dots with connect line, and TRE collections (n=7) are represented by light blue squares.

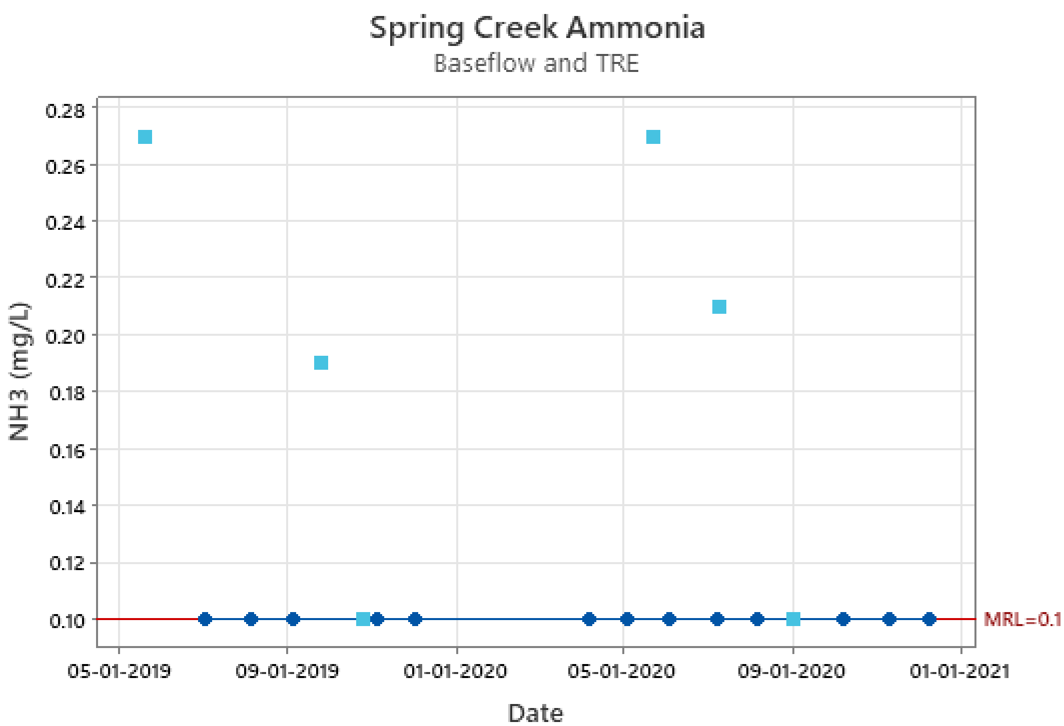


Figure 54. Ammonia in mg/L at Spring Creek over the sample period, baseflow collections (n=13) are represented by dark blue dots with connect line, and TRE collections (n=6) are represented by light blue squares.

Nitrate and nitrite, unlike other in-stream nutrients, did not show strikingly higher measurements at TRE collections as opposed to baseflow collections (Figure 56). As noted above, ammonia is readily available for nitrification under aerobic conditions, oxidizing to the intermediate nitrite before oxidizing further to nitrate (Allen, 1995). This likely led to higher nitrate and nitrite values at baseflow, with diminished values at higher discharges during TREs as compared to other nutrients. There was also less variability in TRE values as compared to baseflow values. The four highest measured concentrations at Deep Fork were baseflow collections (Figure 57).

Baseflow collection mean and median nitrate and nitrite values at the three sites were 0.24 mg/L and 0.14 mg/L respectively at Deep Fork, 0.15 mg/L and 0.09 mg/L respectively at Wynn Creek, and 0.14 mg/L and 0.05 mg/L respectively at Spring Creek (Figure 55, Figure 57, Figure 58, and Figure 59). TRE collection mean and median nitrate and nitrite values at the three sites were 0.37 mg/L for both mean and median at Deep Fork, 0.29 mg/L for both at Wynn Creek, and 0.34 mg/L and 0.35 mg/L respectively at Spring Creek (Figure 56, Figure 57, Figure 58, and Figure 59). Regression analysis showed no relationships at any of the three sites, likely due to in-stream nitrification at baseflow conditions as mentioned above.

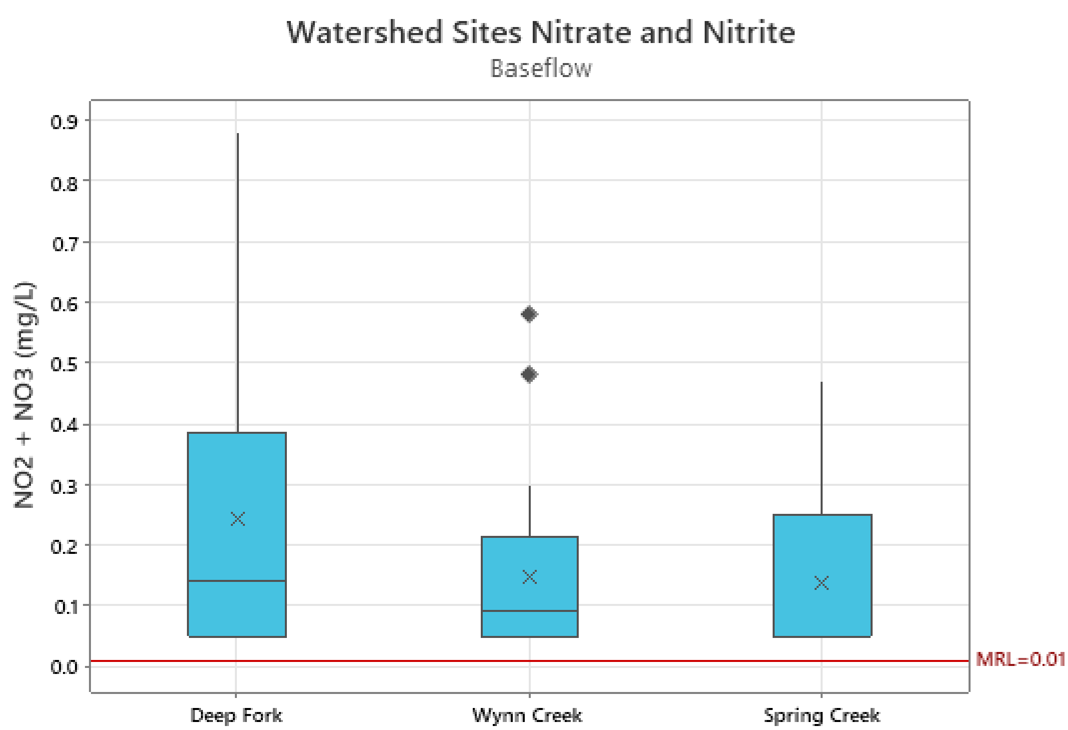


Figure 55. Nitrate and nitrite in mg/L, baseflow collections at Deep Fork (n=25), Wynn Creek (n=25), and Spring Creek (n=25). X denotes mean.

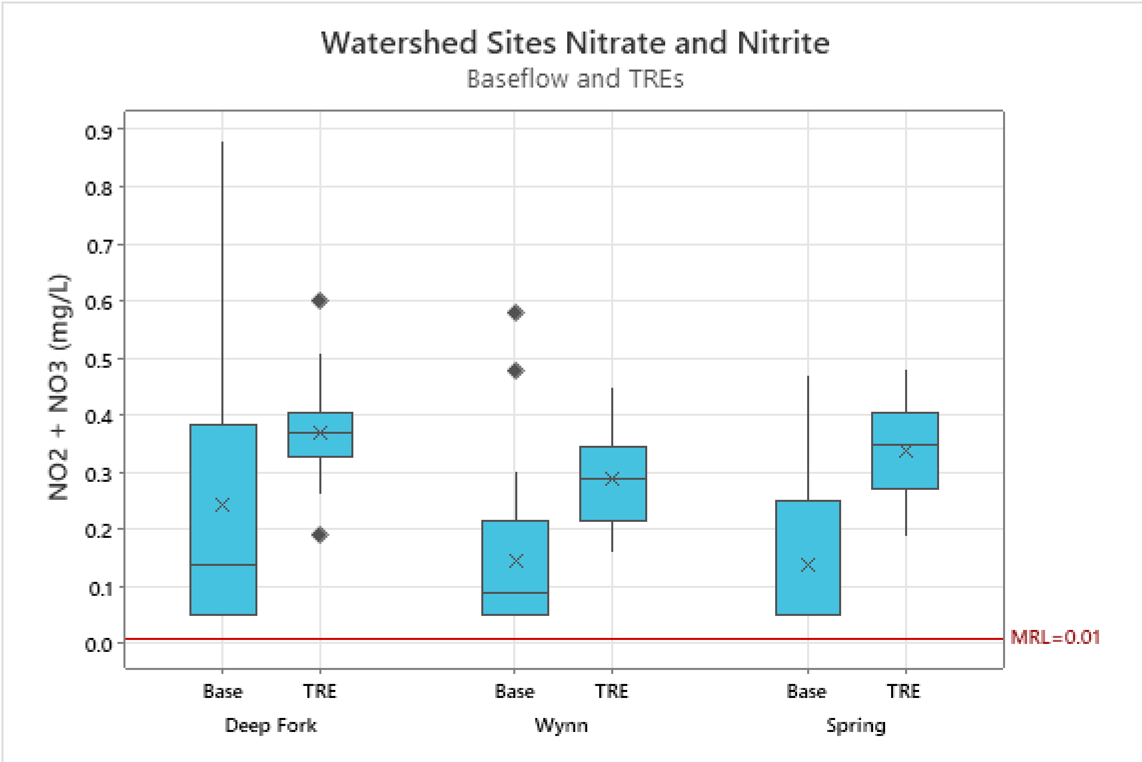


Figure 56. Watershed sites nitrate and nitrite in mg/L, Deep Fork baseflow (n=25) and TRE (n=18), Wynn Creek baseflow (n=25) and TRE (n=16), and Spring Creek baseflow (n=25) and TRE (n=13). X denotes mean.

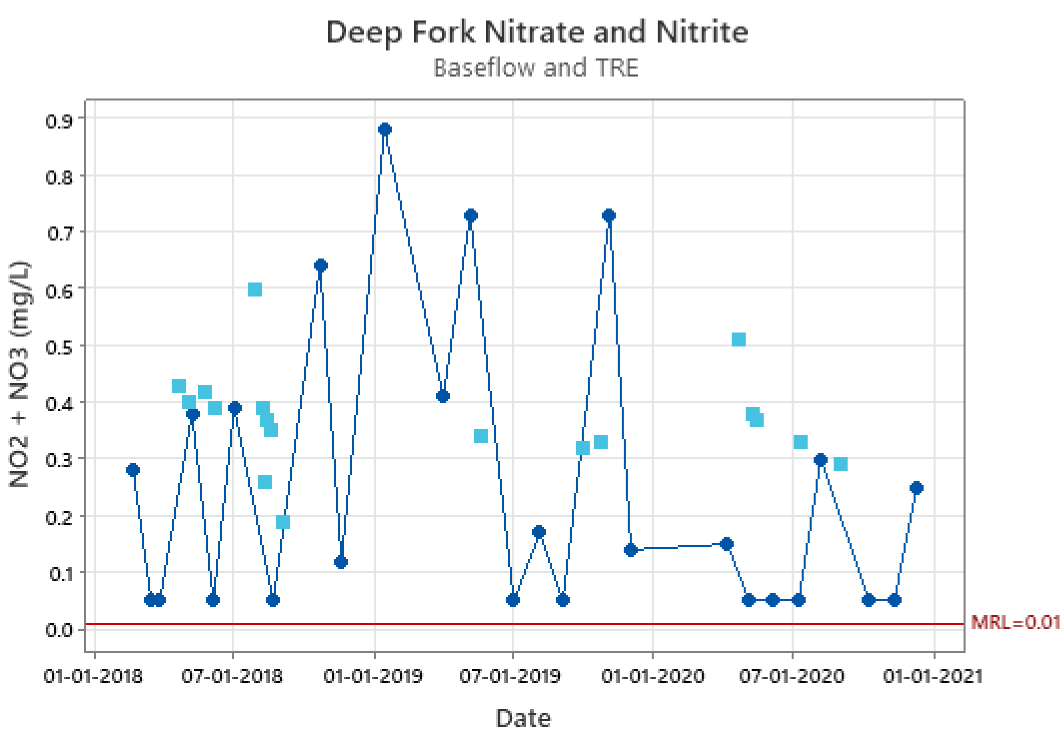


Figure 57. Nitrate and nitrite in mg/L at Deep Fork over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=18) are represented by light blue squares.

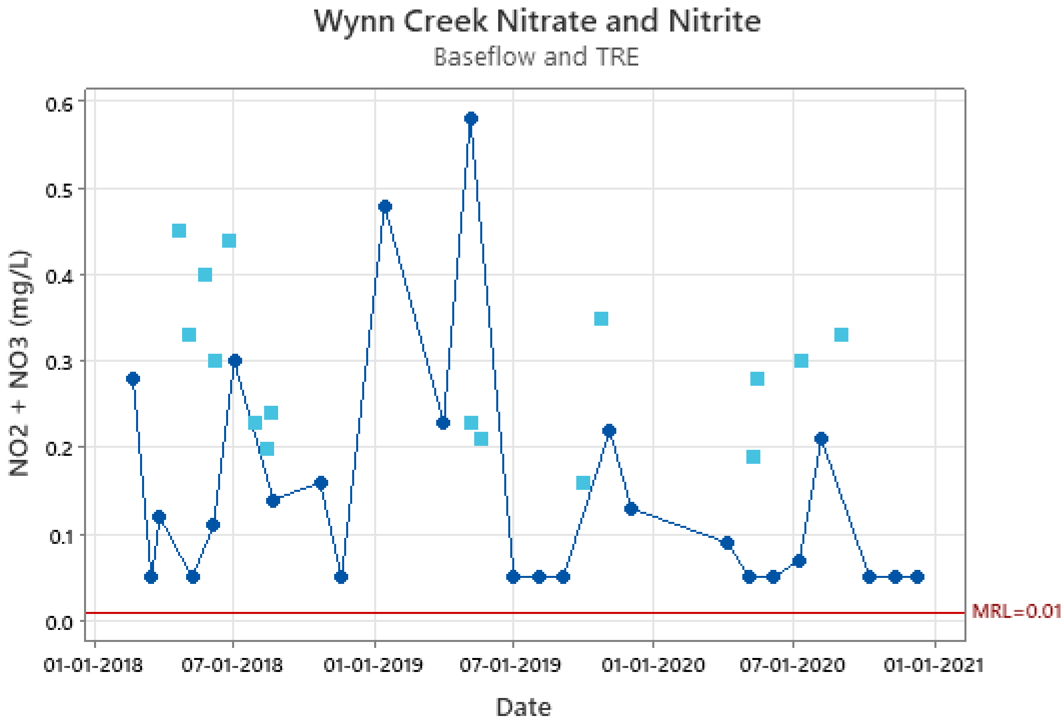


Figure 58. Nitrate and nitrite in mg/L at Wynn Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=16) are represented by light blue squares.

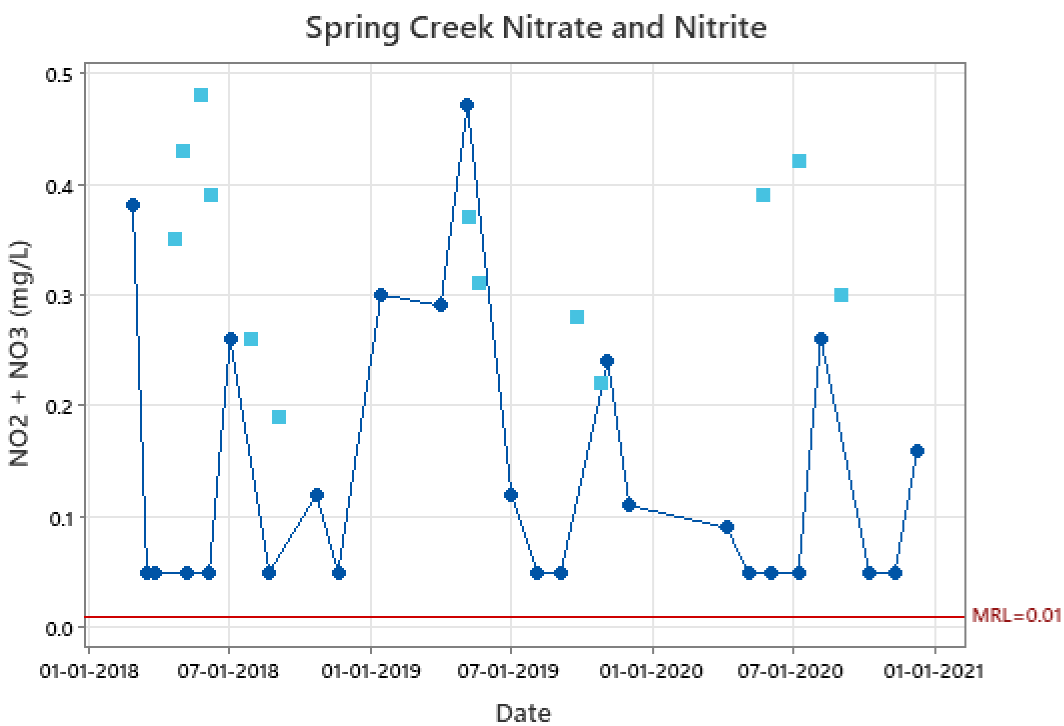


Figure 59. Nitrate and nitrite in mg/L at Spring Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=13) are represented by light blue squares.

Chlorophyll *a*

Arcadia Lake chlorophyll *a* concentrations ranged from 0.65 to 64.4 mg/m^3 with a mean of 18.9 mg/m^3 and tended to have its highest values in the growing season (Spring and Summer) and at Site 5 in the Winter (Figure 60). These observations are confirmed when looking at the results from each sampling trip where chlorophyll *a* tended to have greater concentrations at Sites 1-4 during the growing season (May-September) although much temporal variation is apparent (Figure 60). Site 5, however, tended to have the highest concentrations relative to the other sites throughout the study period and these observations are similar to the results seen in a previous Arcadia Lake study (OWRB, 2000) and at other Oklahoma BUMP lakes throughout the state (OWRB Lakes, unpublished data). Interestingly, a few of the highest observed concentrations during the study period were observed outside of the growing season at Site 5 (52.9 mg/m^3 on 4/19/2018, 64.4 mg/m^3 on 2/20/2019, and 52.9 mg/m^3 on 12/7/2020). Those results, along with the observation that some of the lowest concentrations occurred in April of each year at Sites 1-4, suggest that a priori categorization into contemporary seasons may not be as effective at predicting patterns for chlorophyll *a* as the growing, non-growing season, despite those 3 out of season maxima (The next highest recorded chlorophyll *a* at the other sites was 51.0 mg/m^3 on 6/4/2019 at site 1). Lastly, 63% of the observed concentrations for chlorophyll *a* were above the water quality standard for the PPWS beneficial use of 10 mg/m^3 for chlorophyll *a.* Based on this information, Arcadia Lake is not meeting its beneficial uses related to chlorophyll *a*.

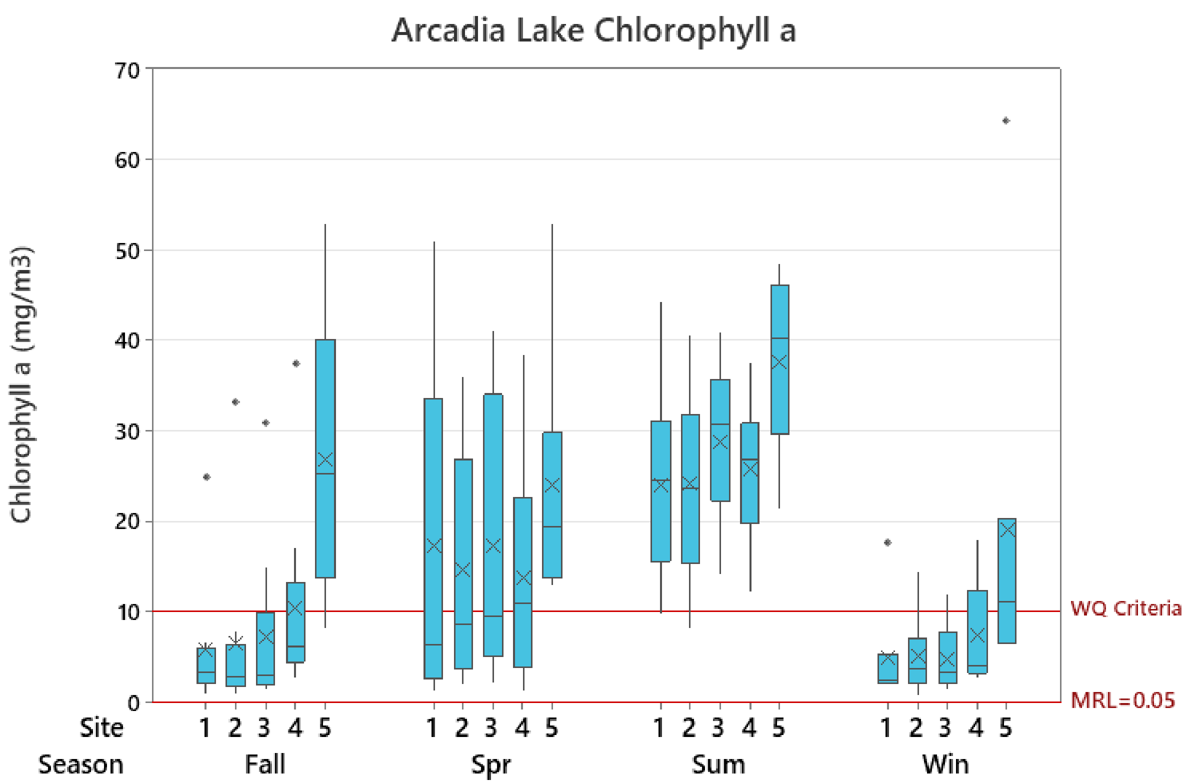


Figure 60. Arcadia Lake boxplots for chlorophyll a surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=7).

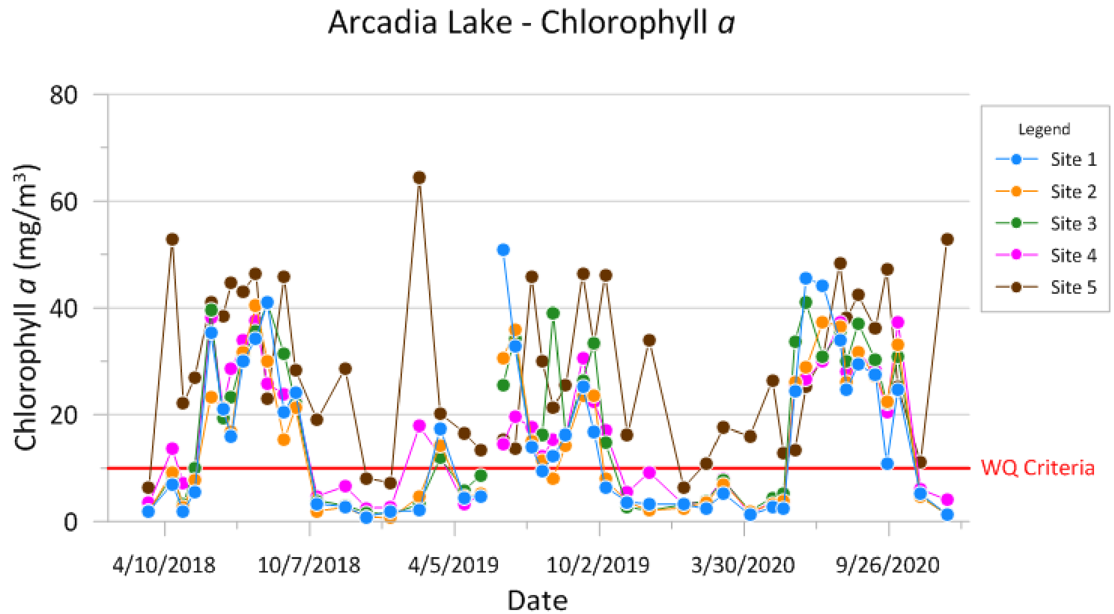


Figure 61. Arcadia Lake surface chlorophyll a plotted for each sampling trip (n=48) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

Watershed chlorophyll *a* showed a notable increase from baseflow collections to TRE collections, with TRE mean concentrations at or near double baseflow mean concentrations at all three sites (Figure 62 and Figure 63). Baseflow collection mean and median chlorophyll *a* values at the three sites were 20.2 mg/m³ and 14.4 mg/m³ respectively at Deep Fork, 7.90 mg/m³ and 6.75 mg/m³ respectively at Wynn Creek, and 5.80 mg/m³ and 3.73 mg/m³ respectively at Spring Creek (Figure 62, Figure 64, Figure 65, and Figure 66). TRE collection mean and median chlorophyll *a* values at the three sites were 45.8 mg/m³ and 31.8 mg/m³ respectively at Deep Fork, 17.1 mg/m³ and 17.3 mg/m³ respectively at Wynn Creek, and 14.5 mg/m³ and 8.33 mg/m³ respectively at Spring Creek (Figure 63, Figure 64, Figure 65, and Figure 66). Regression analysis showed no relationships at any of the three sites, likely due to in-stream flushing during TREs and other physical and chemical factors inhibiting further sestonic algal propagation. No algal blooms were observed at any of the watershed sites during the sampling period.

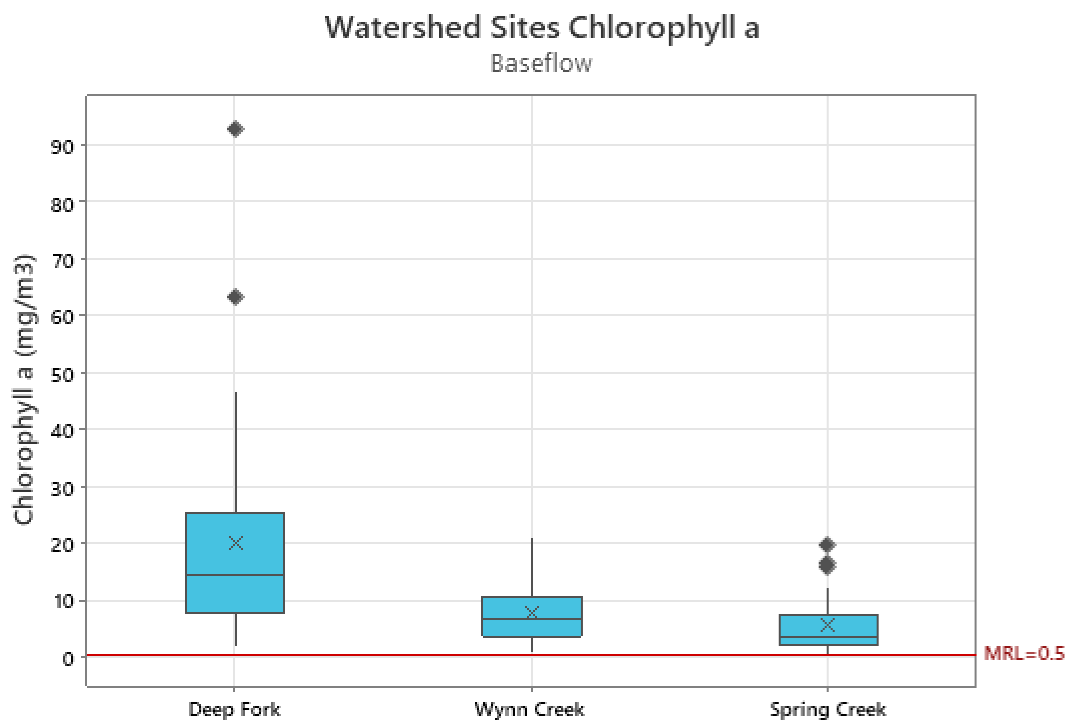


Figure 62. Watershed sites chlorophyll a in mg/m³, baseflow collections at Deep Fork (n=25), Wynn Creek (n=25), and Spring Creek (n=24). X denotes mean.

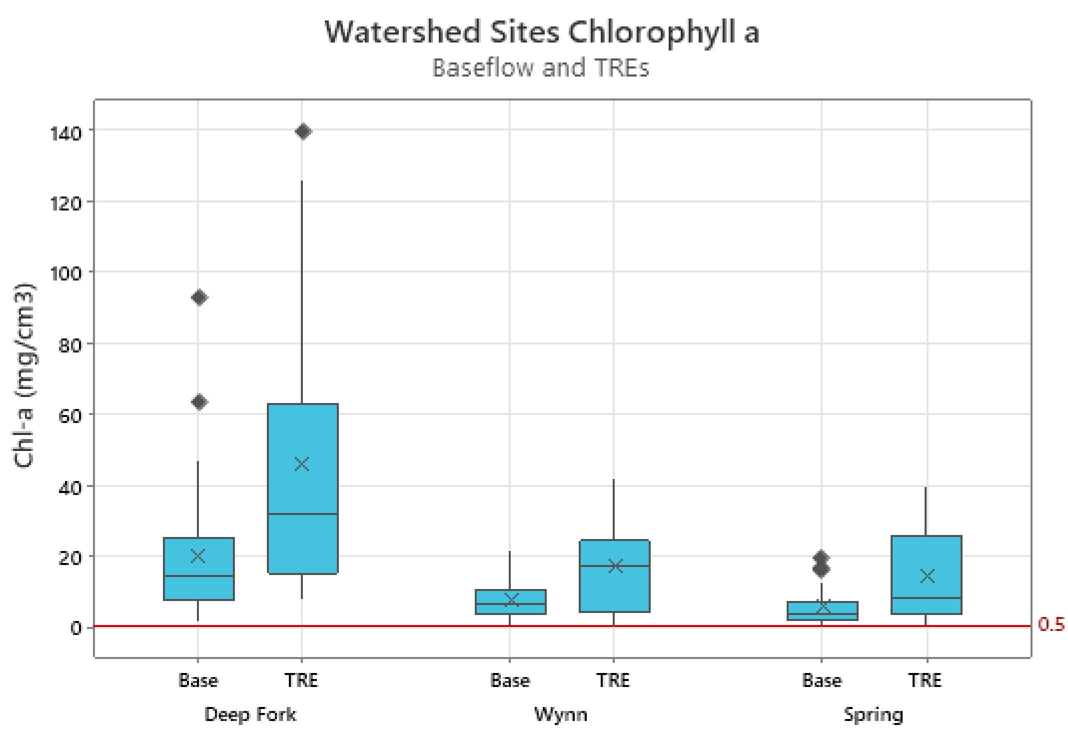


Figure 63. Watershed sites chlorophyll a in mg/m³, Deep Fork Baseflow (n=25) and TRE (n=18), Wynn Creek Baseflow (n=24) and TRE (n=15), and Spring Creek Baseflow (n=24) and TRE (n=12). X denotes mean.

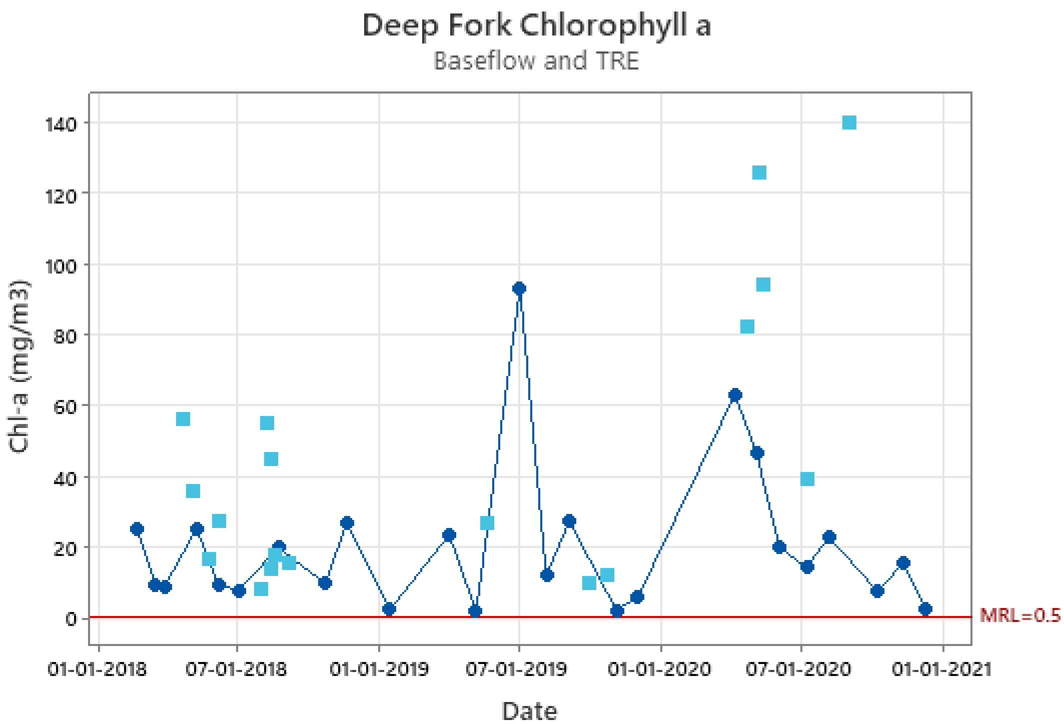


Figure 64. Chlorophyll a in mg/m³ at Deep Fork over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=18) are represented by light blue squares.

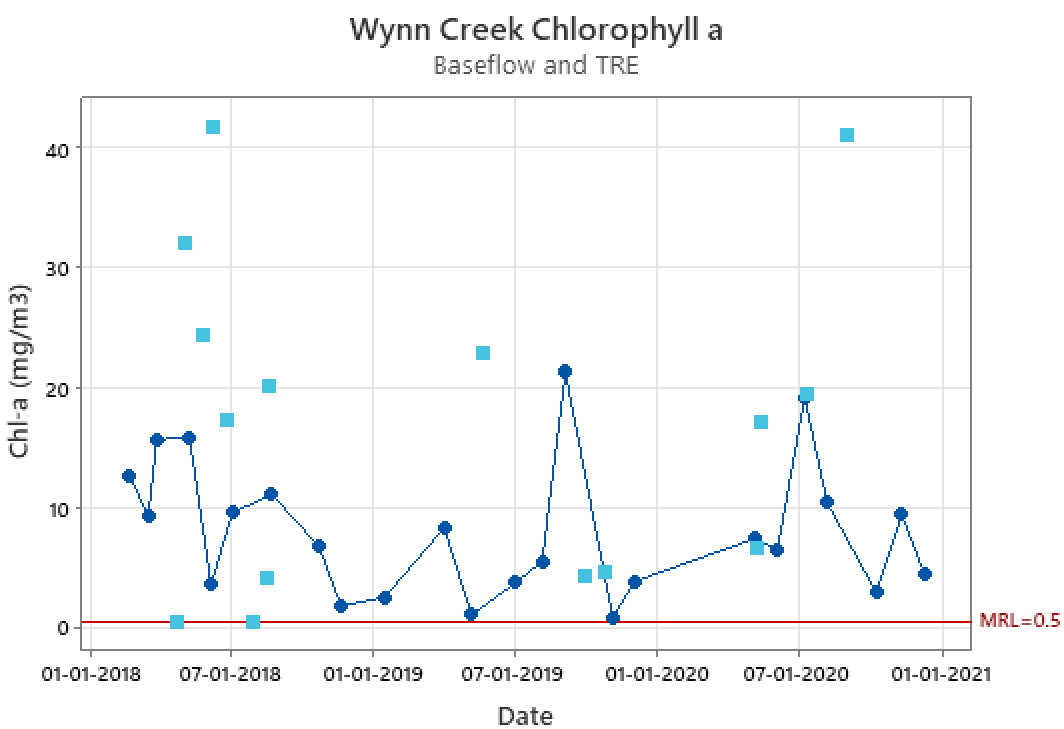


Figure 65. Chlorophyll a in mg/m³ at Wynn Creek over the sample period, baseflow collections (n=24) are represented by dark blue dots with connect line, and TRE collections (n=15) are represented by light blue squares.

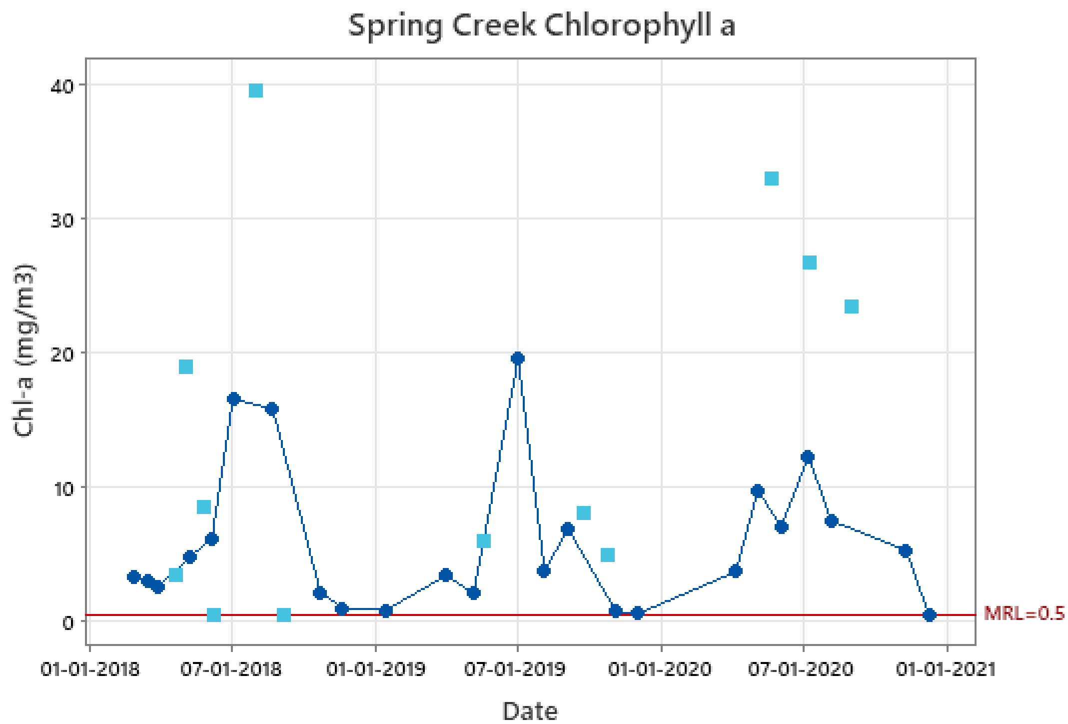


Figure 66. Chlorophyll a in mg/m³ at Spring Creek over the sample period, baseflow collections (n=24) are represented by dark blue dots with connect line, and TRE collections (n=12) are represented by light blue squares.

Water Clarity

Water clarity was analyzed through turbidity, Secchi depth (in-lake only) and TSS. Arcadia Lake turbidity ranged from 5 to 197 NTU with a mean of 28.6 NTU with Site 5 having consistently higher turbidity than Sites 1 – 4 likely due to the increased input of sediment-rich water from the Deep Fork, especially in Spring and Summer (Figure 67). Indeed, the highest recorded turbidity occurred at Site 5 on 5/3/2018, a few hours after an early morning thunderstorm where the water was noticeably more turbid at Sites 4 and 5 than in the rest of the lake (Figure 68). Site 4 tended to have slightly higher turbidity than Sites 1-3, as it is a transitional site between the riverine and lacustrine zones of the lake. Thirty-three percent of the observed concentrations for turbidity were above the water quality standard for beneficial use of 25 NTU for turbidity. Based on this information, Arcadia Lake is not meeting its beneficial uses related to turbidity.

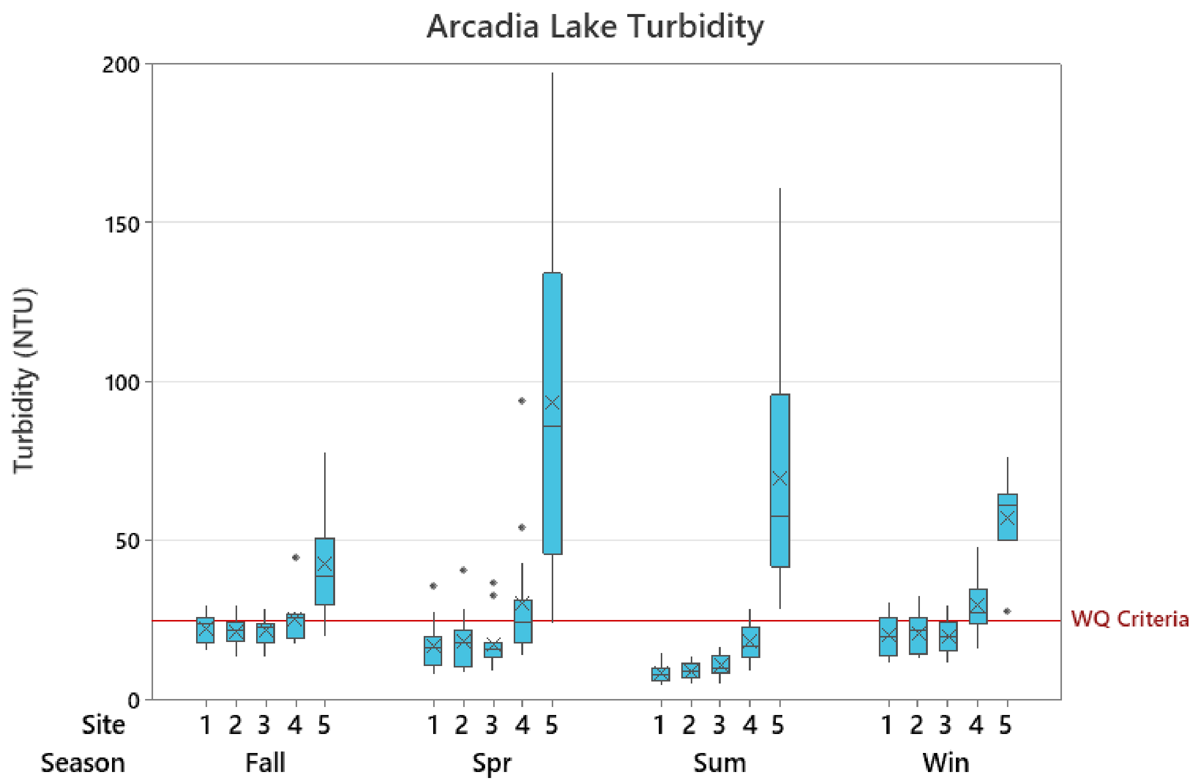


Figure 67. Arcadia Lake boxplots for turbidity surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8).

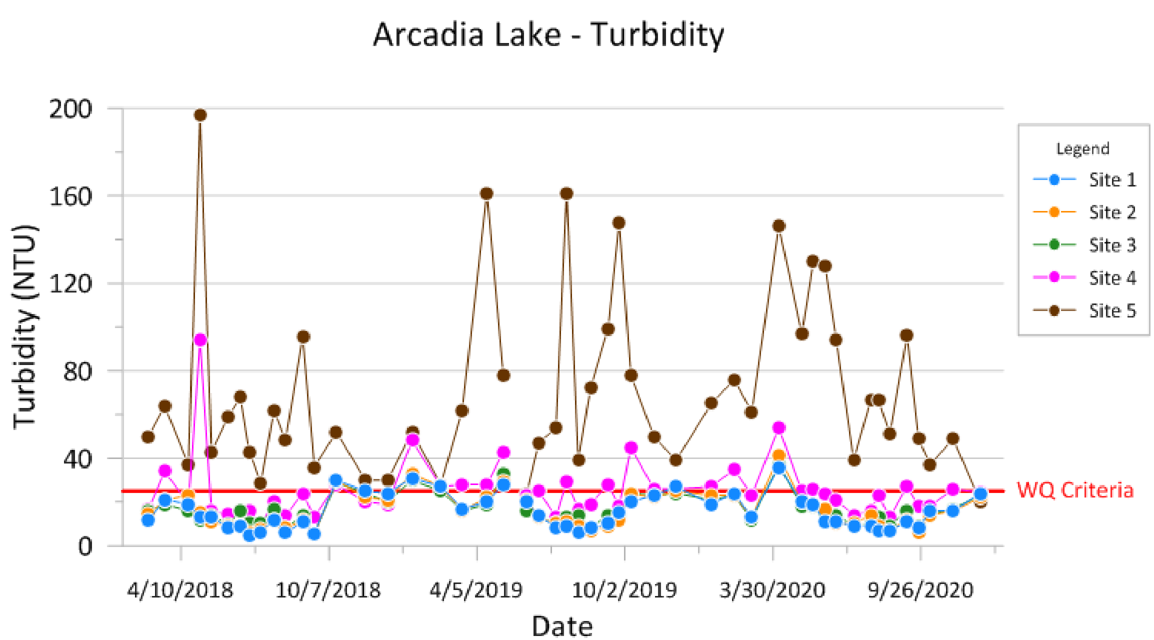


Figure 68. Arcadia Lake surface turbidity plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

Turbidity was the only in-situ measurement for water clarity at the watershed sites. Secchi depth is typically not measured in flowing waters. Watershed sites were typically clear but too shallow at baseflow conditions, and generally opaque or highly cloudy during TRE collections. Arcadia Lake Site 5 turbidity and Secchi disk depth show sediment transport from Wynn Creek and Deep Fork into Arcadia Lake (Figure 67 and Figure 75), while also demonstrating prolonged suspension as compared to the related watershed sites. The watershed sites had much higher turbidity variability than Site 5 during TREs, and higher mean and median concentrations for both watershed sites versus the in-lake counterpart suggest downstream sedimentation as runoff laden waters approach and subsequently pass the riverine Site 5 (Figure 67 and Figure 70).

Baseflow collection mean and median turbidity values at the three sites were 27 NTU and 14 NTU respectively at Deep Fork, 45 NTU and 25 NTU respectively at Wynn Creek, and 27 NTU and 12 NTU respectively at Spring Creek (Figure 69**).** TRE collections at all three sites hit the instrument reporting limit (IRL) of 1000 NTU, the maximum detection value for the in-situ turbidimeters used (Figure 71, Figure 72, and Figure 73). TRE collection mean and median turbidity values at the three sites were 702 NTU and 781 NTU respectively at Deep Fork, 665 NTU and 1000 NTU respectively at Wynn Creek, and 760 NTU and 843 NTU respectively at Spring Creek (Figure 70). Regression analysis was not performed on turbidity due to the IRL.

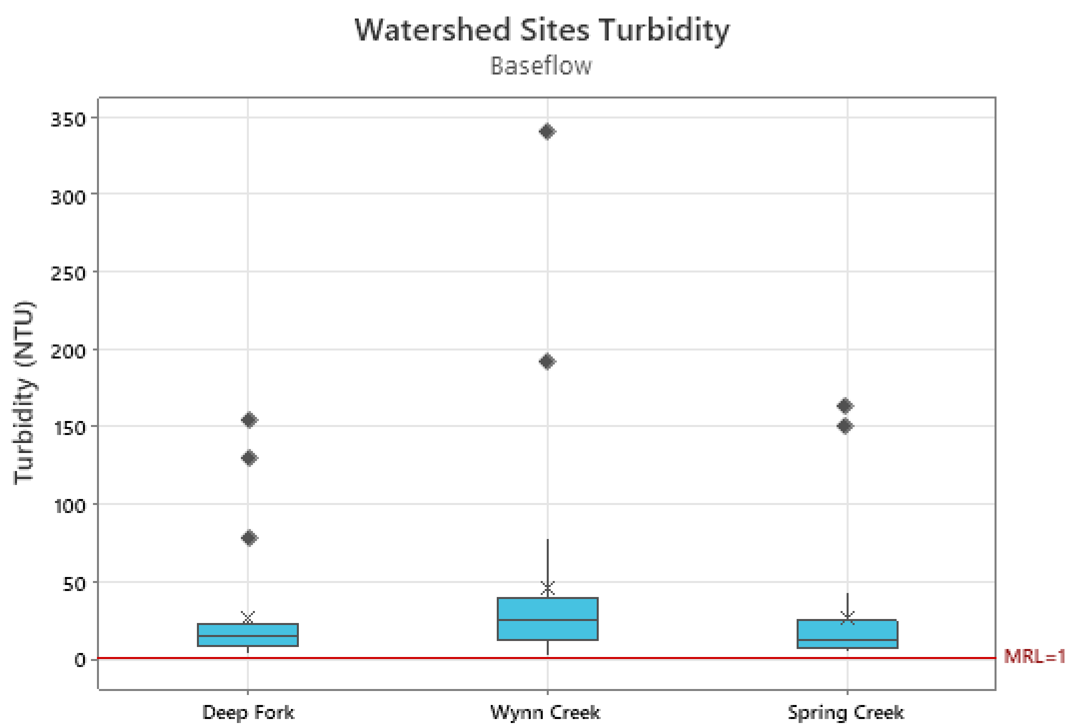


Figure 69. Turbidity in NTU, baseflow collections at Deep Fork (n=24), Wynn Creek (n=24), and Spring Creek (n=23). X denotes mean.

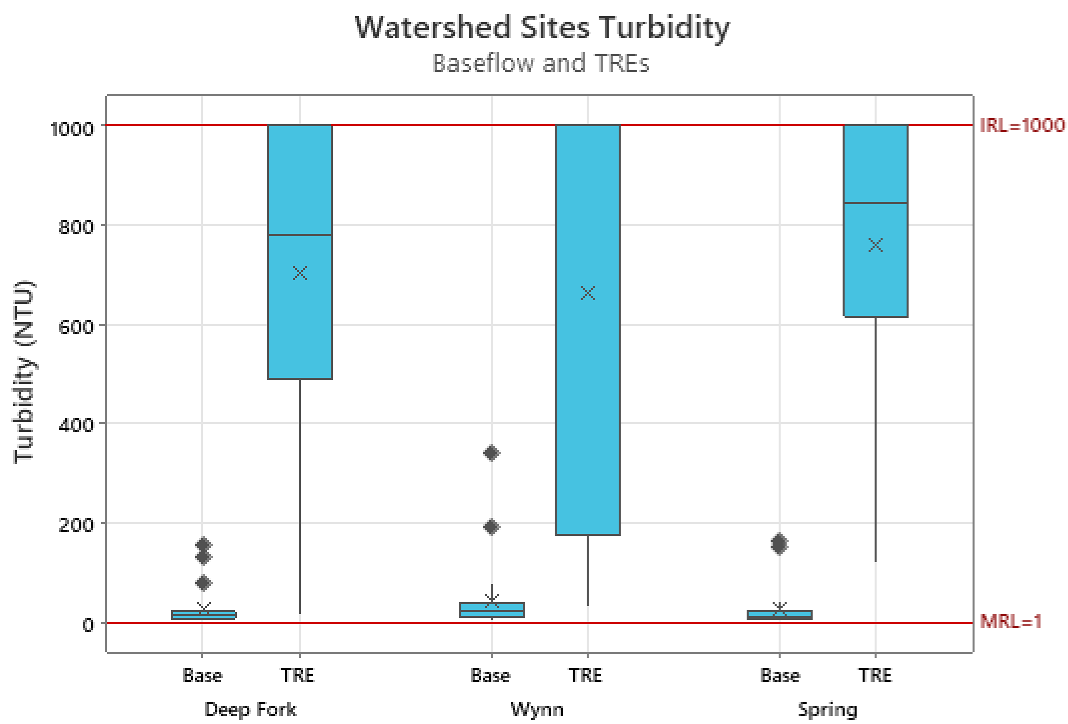


Figure 70. Watershed sites turbidity in NTUs, Deep Fork baseflow (n=24) and TRE (n=18), Wynn Creek baseflow (n=24) and TRE (n=15), and Spring Creek baseflow (n=23) and TRE (n=12). X denotes mean.

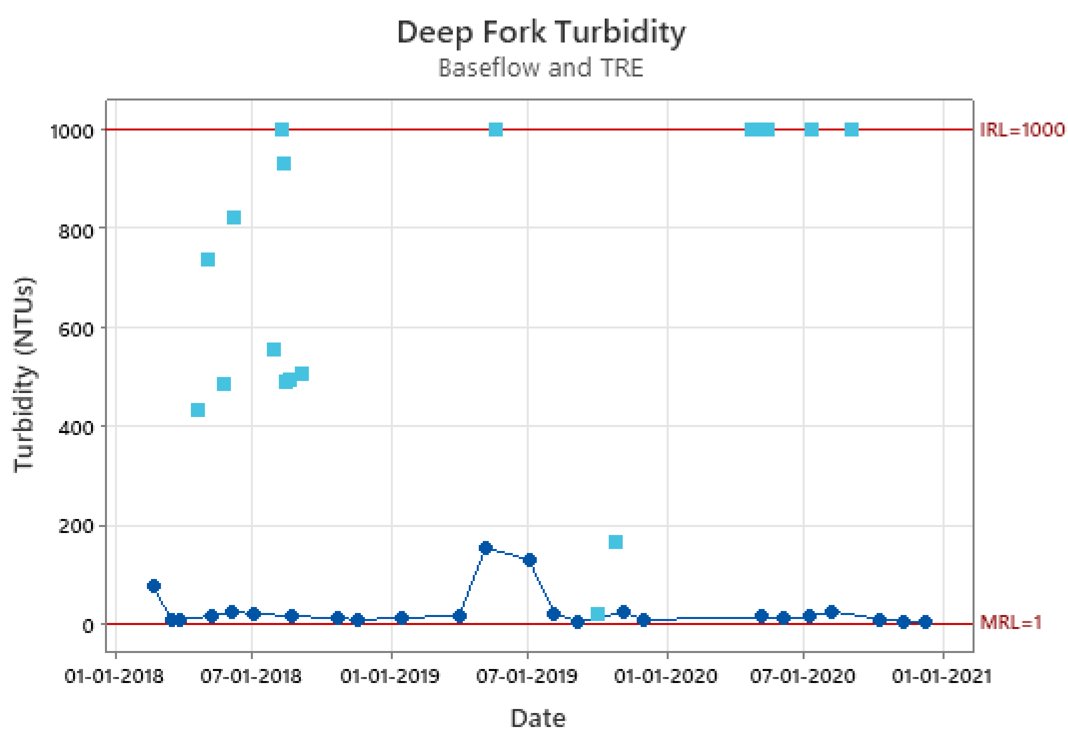


Figure 71. Turbidity in NTU at Deep Fork over the sample period, baseflow collections (n=24) are represented by dark blue dots with connect line, and TRE collections (n=18) are represented by light blue squares.

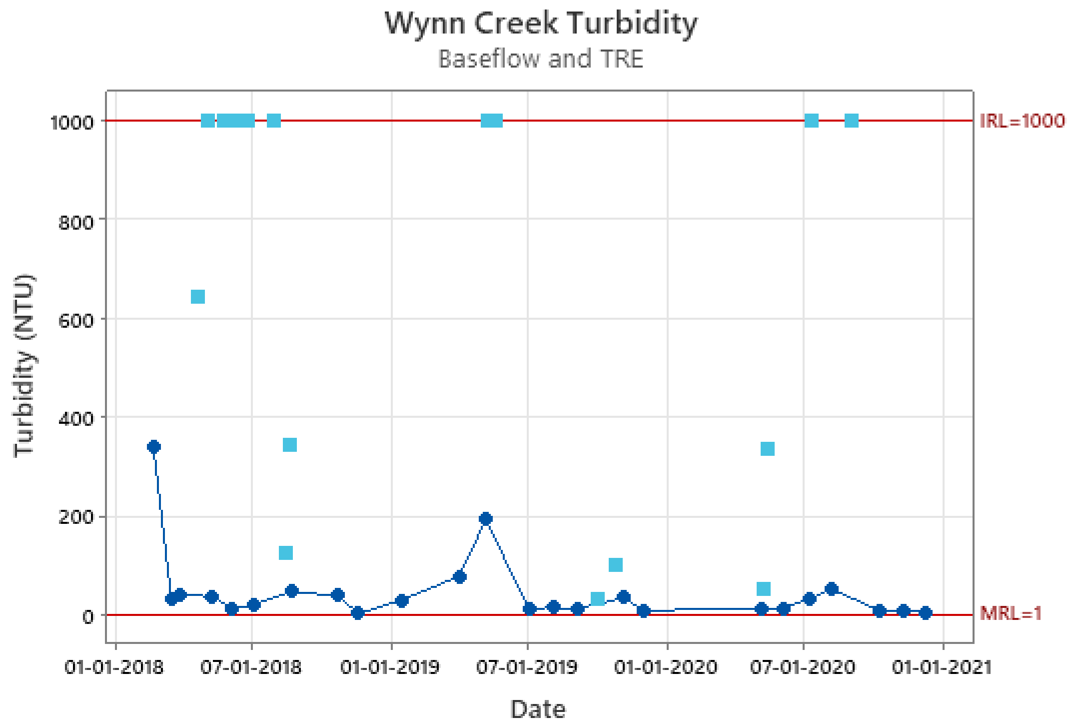


Figure 72. Turbidity in NTU at Wynn Creek over the sample period, baseflow collections (n=24) are represented by dark blue dots with connect line, and TRE collections (n=15) are represented by light blue squares.

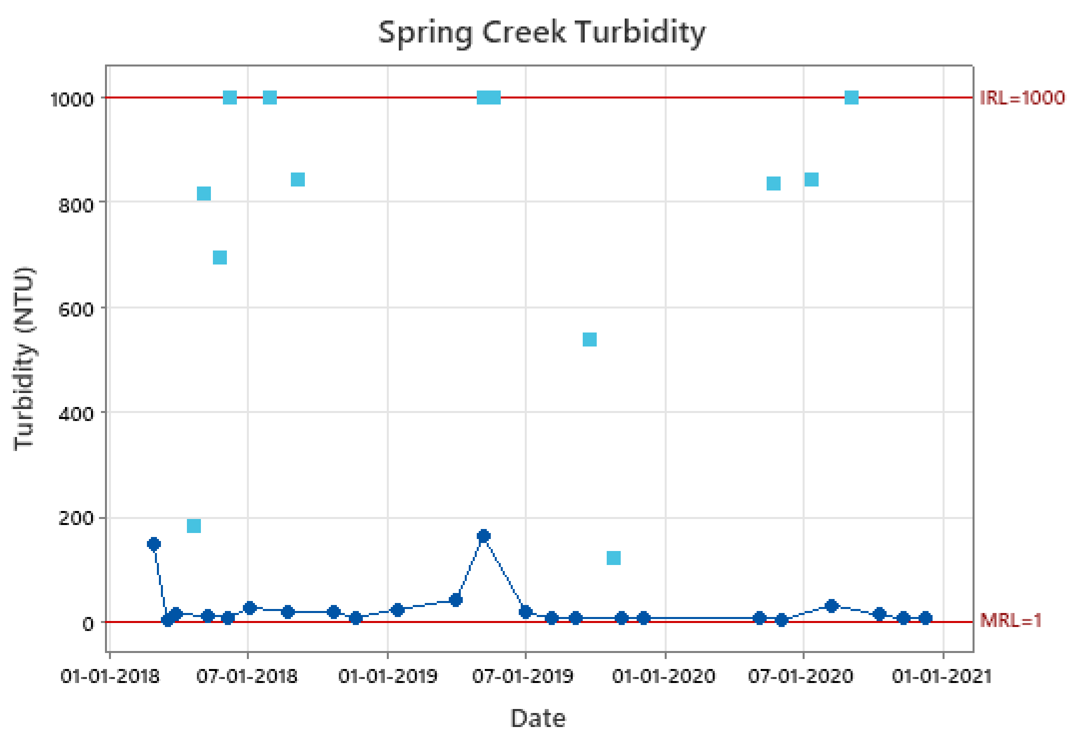


Figure 73. Turbidity in NTU at Spring Creek over the sample period, baseflow collections (n=24) are represented by dark blue dots with connect line, and TRE collections (n=13) are represented by light blue squares.

Secchi disk depth provides an indication of the depth that light can penetrate into the water column so greater depths represent increased water clarity and should in general have an inverse relationship to turbidity. Arcadia Lake Secchi disk depths ranged from 8 – 122 cm with a mean of 52.8 cm with a general trend of deepest at Site 1 to shallowest at Site 5 with fall having the least difference among the sites (Figure 74). As with turbidity, the decreased water clarity at Site 5 is likely attributed to input from Deep Fork as two of the shallowest observations occurred after the storm of May 2018 at Sites 4 and 5 (Figure 75). There appeared to be a gradient in Secchi disk depth moving from Site 5 to Site 1 allowing greater penetration of light at the lacustrine site and very little penetration in the riverine site associated with Deep Fork.

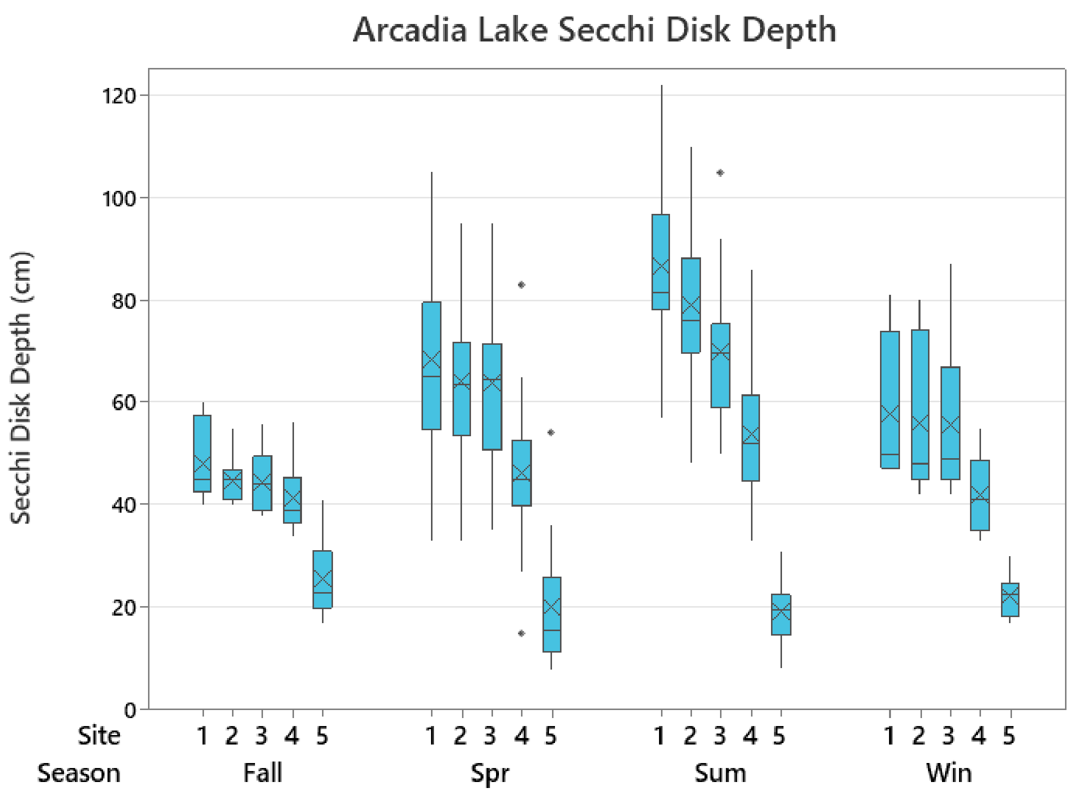


Figure 74. Arcadia Lake boxplots for Secchi disk depth categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8).

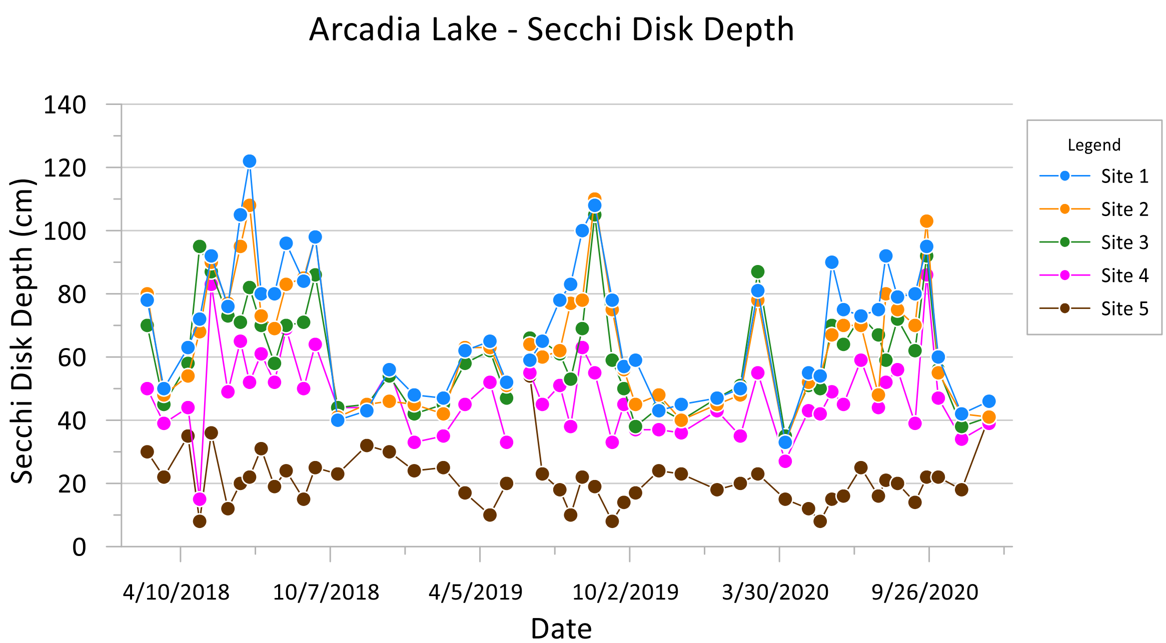


Figure 75. Arcadia Lake Secchi depth plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

Total suspended solids (TSS) showed similar patterns to turbidity and Secchi depth where Site 5 had consistently higher concentrations than the other sites (Figure 76). Riverine sites have been reported as generally higher in both turbidity and TSS with a shallower Secchi depth in a previous study at Arcadia Lake (OWRB, 2000) as well as at other Oklahoma BUMP lakes (OWRB Lakes, unpublished data). TSS concentrations throughout the study period ranged from below reporting limit (5 mg/L) to 118 mg/L with a mean of 17.6 in surface samples (Figure 77a) and below reporting limit to 102 mg/L with a mean 17.1 mg/L in the bottom samples (Figure 77b). TSS showed some variability in bottom samples with a maximum of 102 mg/L on Aussgust 6, 2019 at Site 2 (Figure 77b). It is unclear what caused this peak in TSS at this time, but it did correspond with the highest concentrations at Site 2 for TP, DOP, NH3, TKN, and TN and given its location in the lake may be a result of turbulence reaching all the way to the bottom.

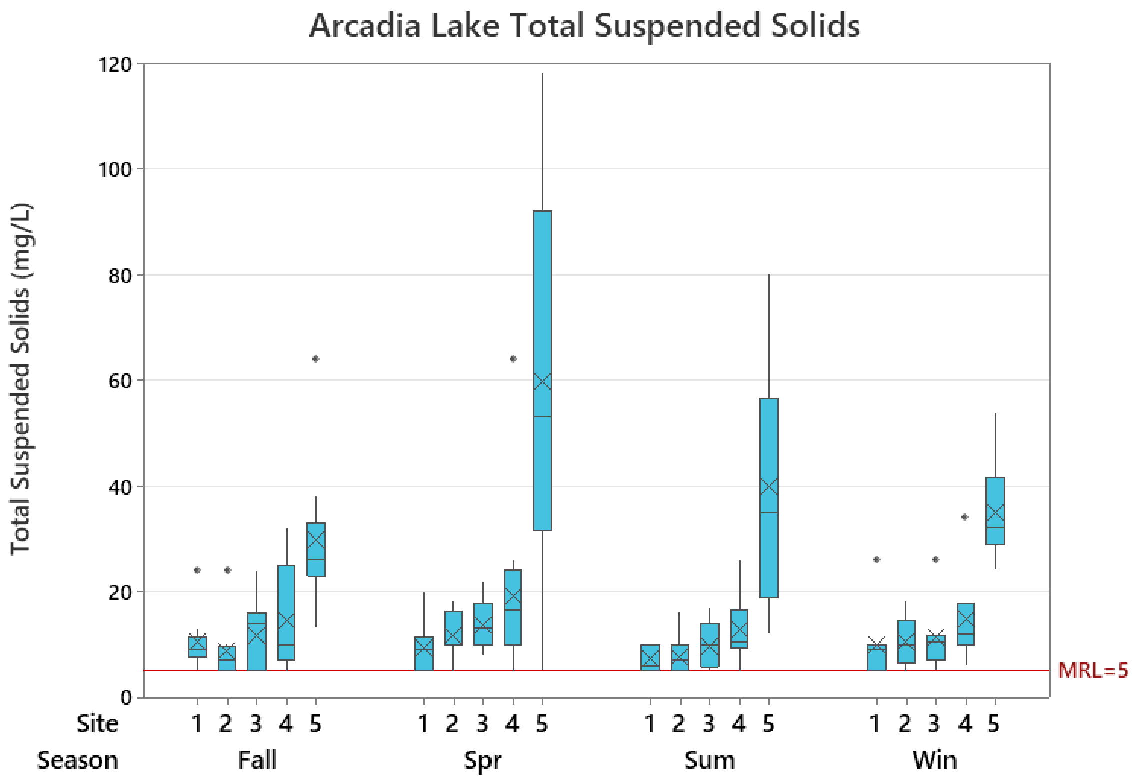


Figure 76. Arcadia Lake boxplots for total suspended solids surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8).

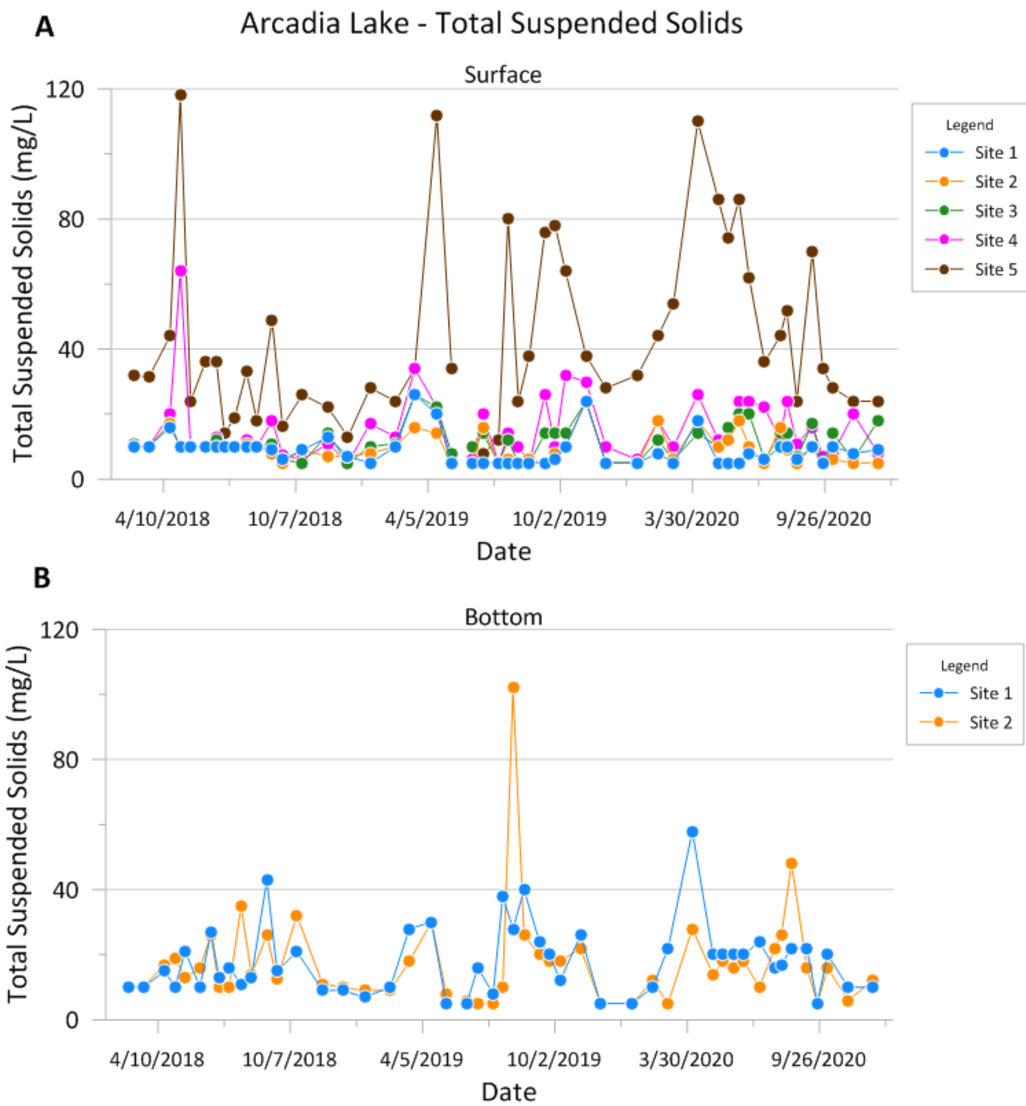


Figure 77. Arcadia Lake (a) surface and (b) bottom total suspended solids plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

All measurements of water clarity showed a general trend of lower water clarity moving from the watershed sites, attributed to input from Deep Fork (Site 5) and less so from Spring Creek (Site 3), to greater water clarity at the lacustrine site. Both Cooke et al. (2001) and Thornton et al. (1990) predicted this increase in water clarity moving towards the dam as sediment and other material settles from the water column as it flows its course through the reservoir. As each of these parameters provide an estimate of water clarity, we would expect a strong relationship among them. Turbidity and TSS had a strong relationship during the study period (Figure 78**;** $p<0.01$; $R^2=0.84$). A regression of Secchi disk depth with turbidity (Figure 79; $p<0.01$; $R^2=0.89$) and TSS (Figure 80; $p<0.01$; $R^2=0.57$) showed similar relationships; however, accuracy is lost with increased depth as the optical estimate of depth in water can be affected by several factors including surface reflectance or light refraction between the two media. But in general, all of these parameters observed in Arcadia Lake agree with the previous predictions of how water clarity increases during its flow through reservoirs.

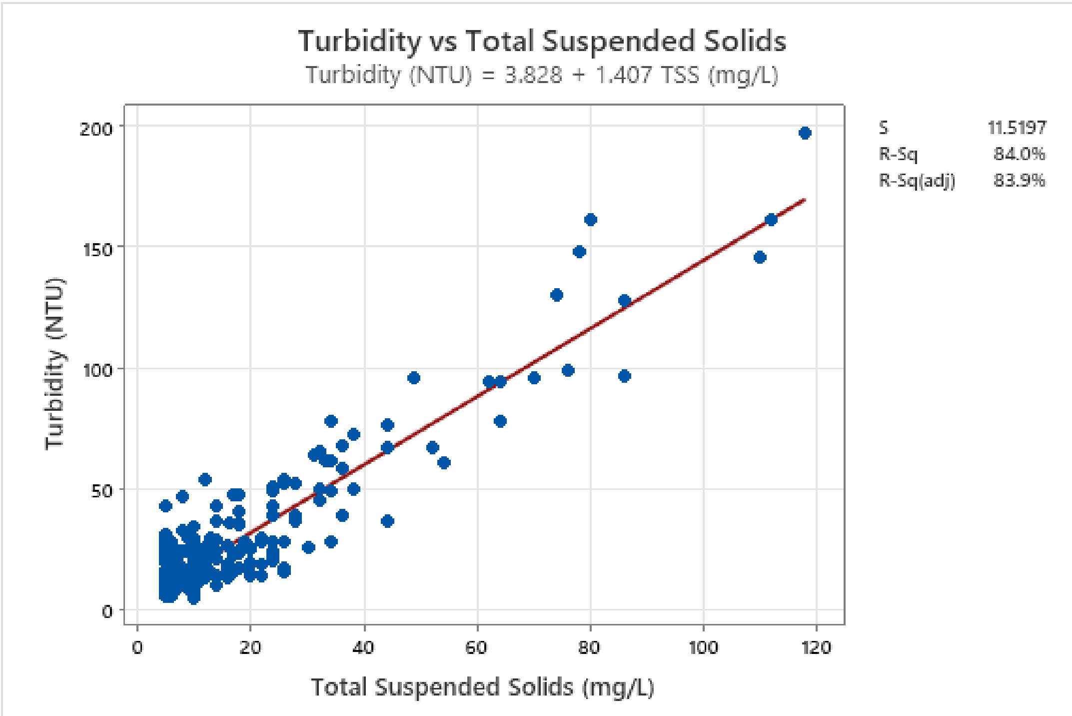


Figure 78. Linear relationship of Arcadia Lake turbidity versus total suspended solids (n=244).

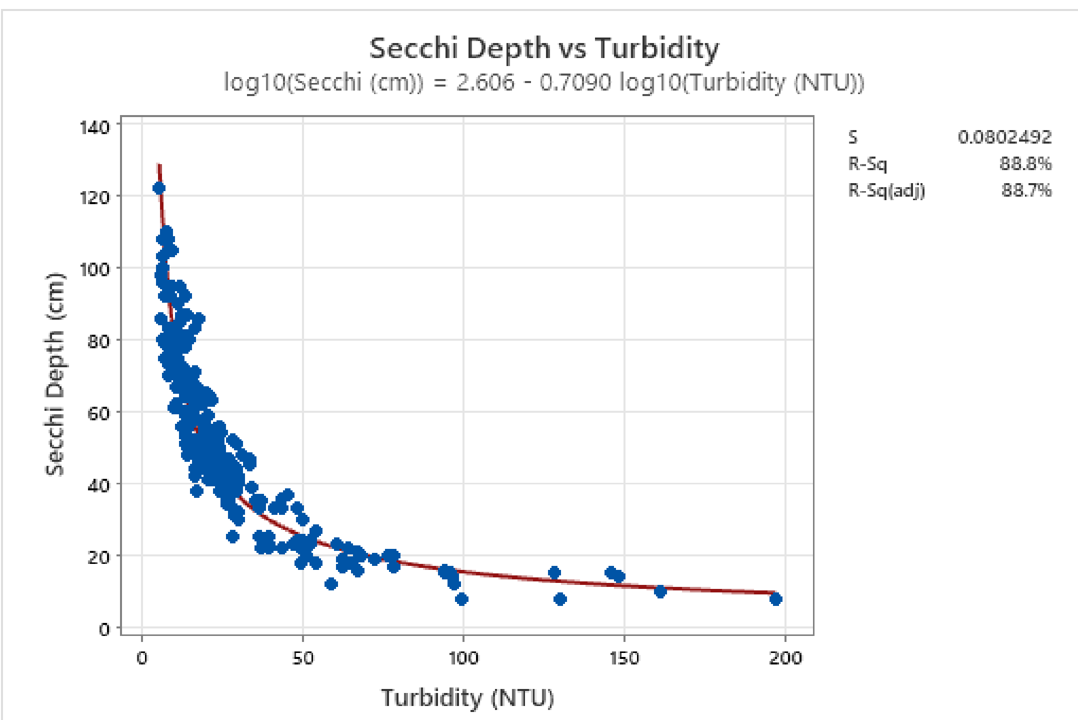


Figure 79. Linear relationship of Arcadia Lake Secchi depth versus turbidity (n=244).

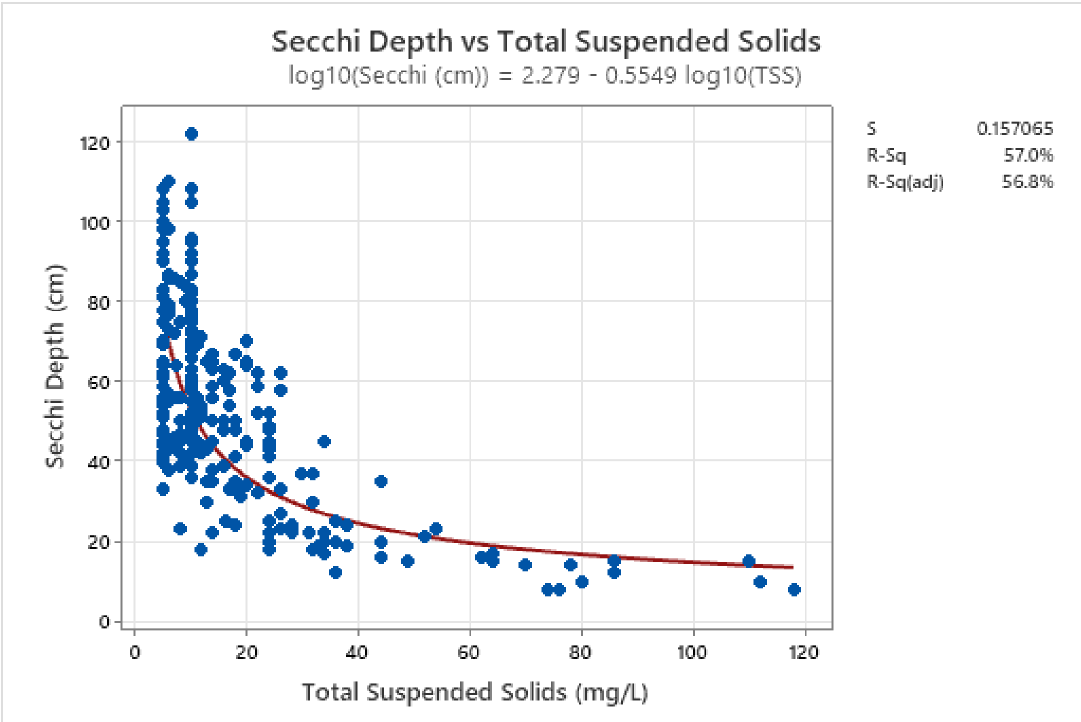


Figure 80. Linear relationship of Arcadia Lake Secchi depth versus total suspended solids (n=244).

Similar to turbidity, TSS showed high variability and high concentrations during TRE collections. High sediment was visible during TREs due to increased precipitation runoff, particularly the highly urbanized Deep Fork (Figure 81). Baseflow collection mean and median TSS values at the three sites were 24 mg/L and 11 mg/L respectively at Deep Fork, 31 mg/L and 16 mg/L respectively at Wynn Creek, and 22 mg/L and 10 mg/L respectively at Spring Creek (Figure 81, Figure 83, Figure 84, and Figure 85). TRE collection mean and median TSS values at the three sites were 2521 mg/L and 765 mg/L respectively at Deep Fork, 1499 mg/L and 969 mg/L respectively at Wynn Creek, and 3624 mg/L and 945 mg/L respectively at Spring Creek (Figure 82, Figure 83, Figure 84, and Figure 85). Regression analysis showed relationships between TSS and discharge at all three sites (Figure 86, Figure 87, and Figure 88).

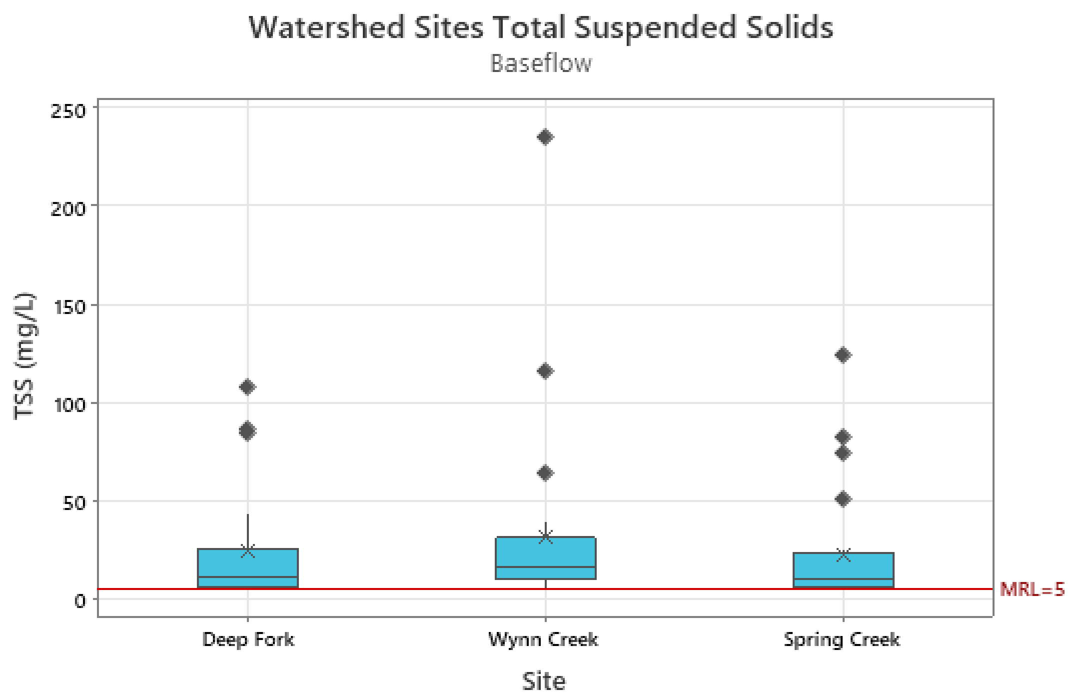


Figure 81. Watershed sites total suspended solids in mg/L, baseflow collections at Deep Fork (n=25), Wynn Creek (n=25), and Spring Creek (n=25). X denotes mean.

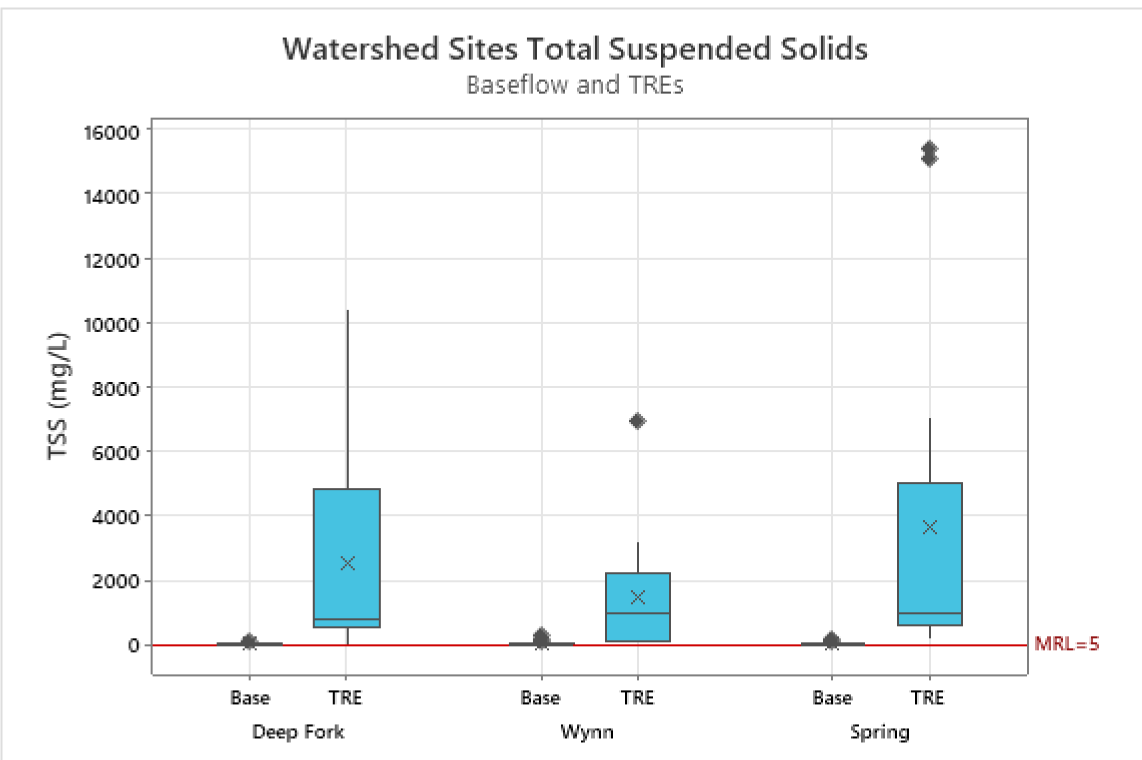


Figure 82. Watershed sites total suspended solids in mg/L, Deep Fork baseflow (n=25) and TRE (n=18), Wynn Creek baseflow (n=25) and TRE (n=16), and Spring Creek baseflow (n=25) and TRE (n=13). X denotes mean.

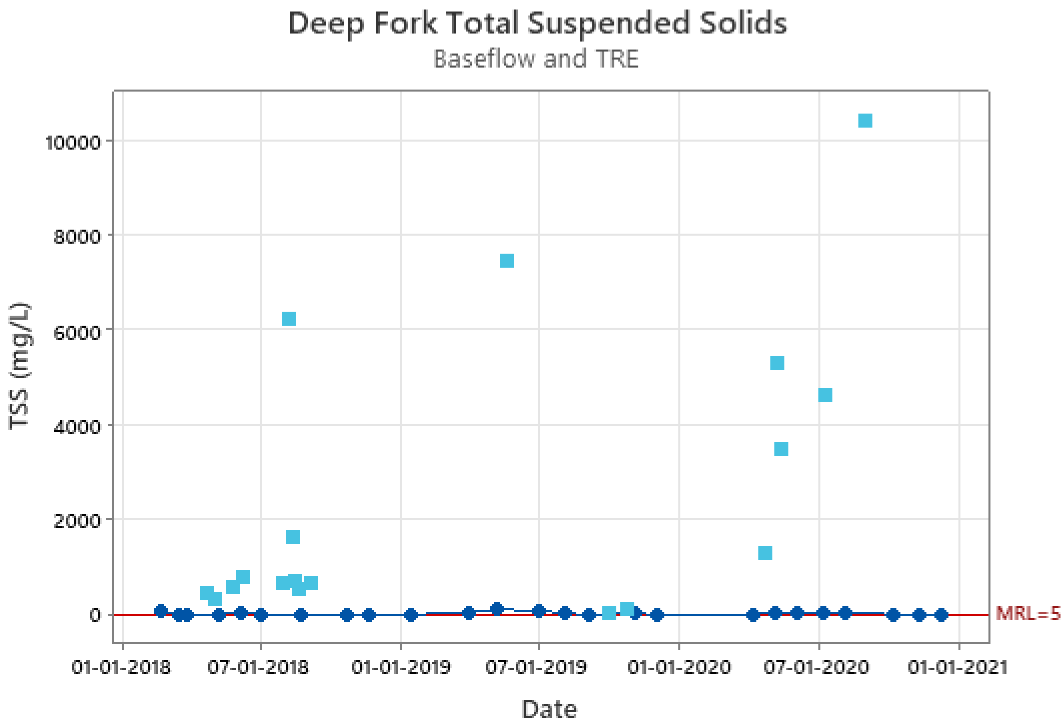


Figure 83. Total suspended solids in mg/L at Deep Fork over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=18) are represented by light blue squares.

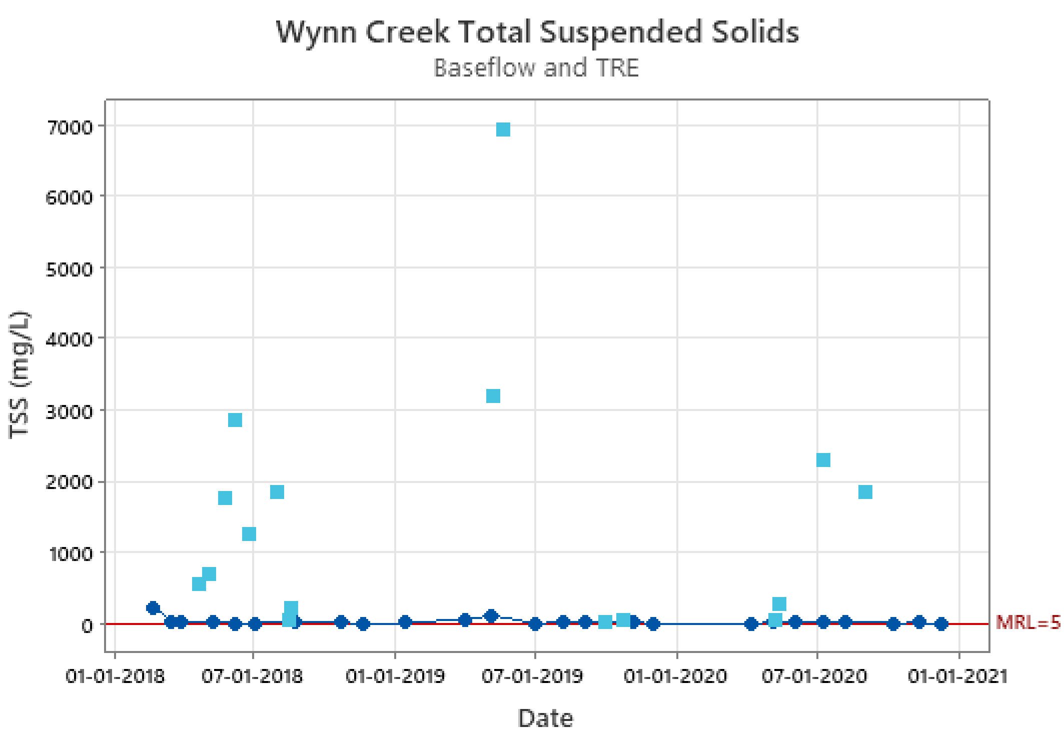


Figure 84. Total suspended solids in mg/L at Wynn Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=16) are represented by light blue squares.

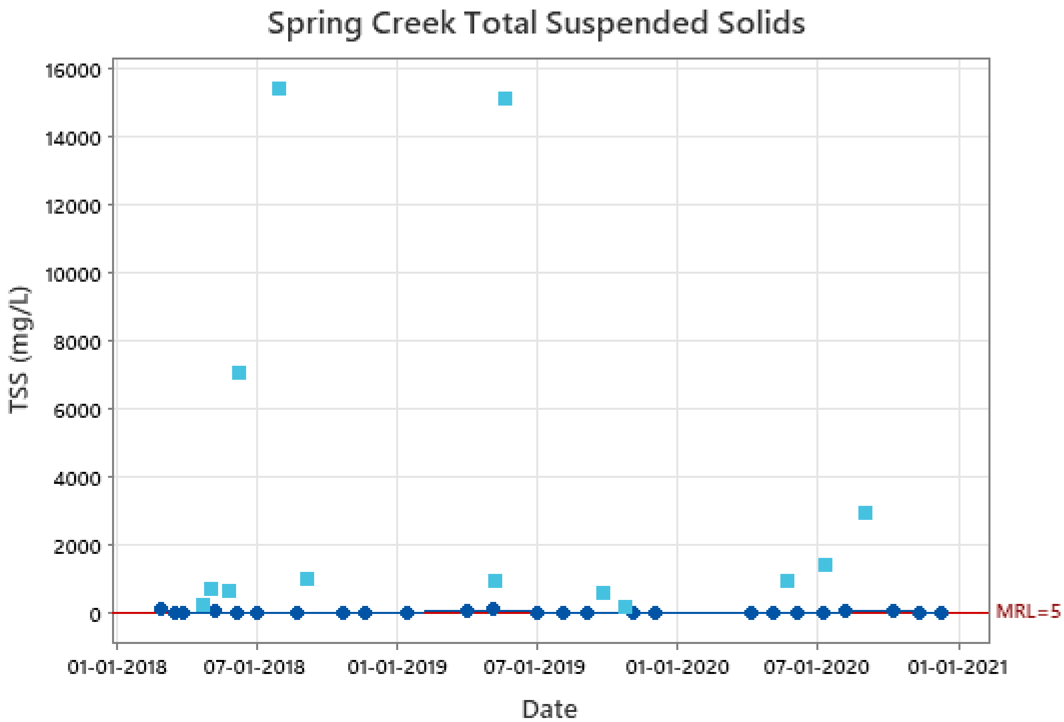


Figure 85. Total suspended solids in mg/L at Spring Creek over the sample period, baseflow collections (n=25) are represented by dark blue dots with connect line, and TRE collections (n=13) are represented by light blue squares.

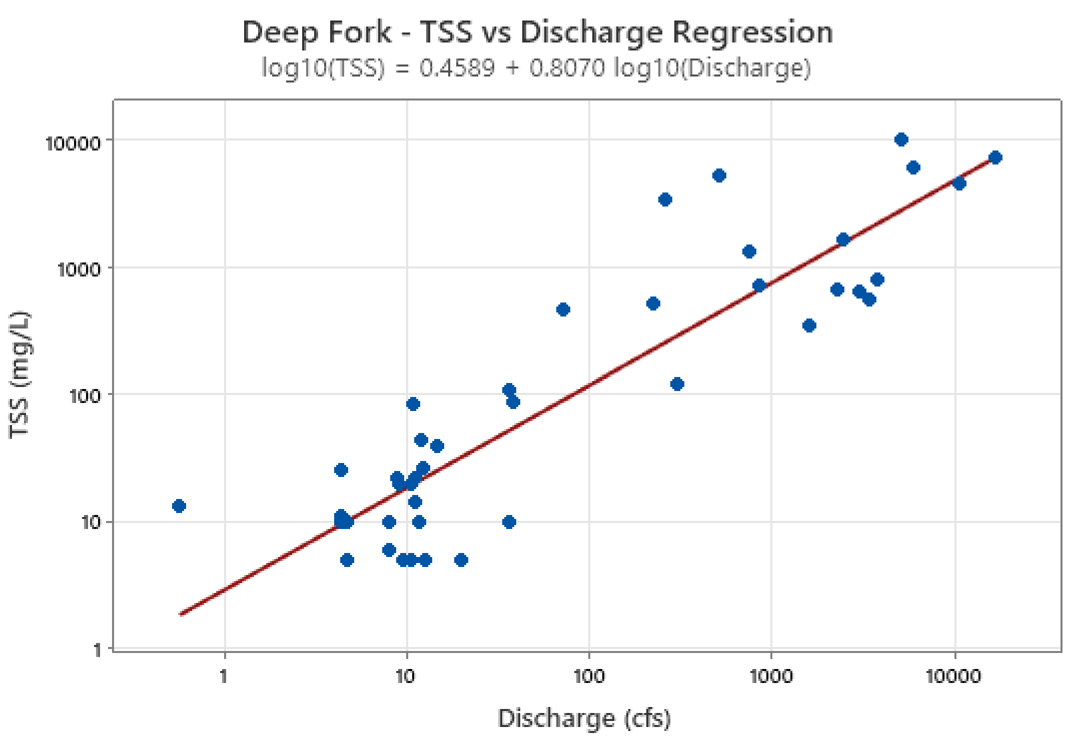


Figure 86. Total suspended solids versus discharge regression at Deep Fork. R^2=80.4%, p<0.001, n=43.

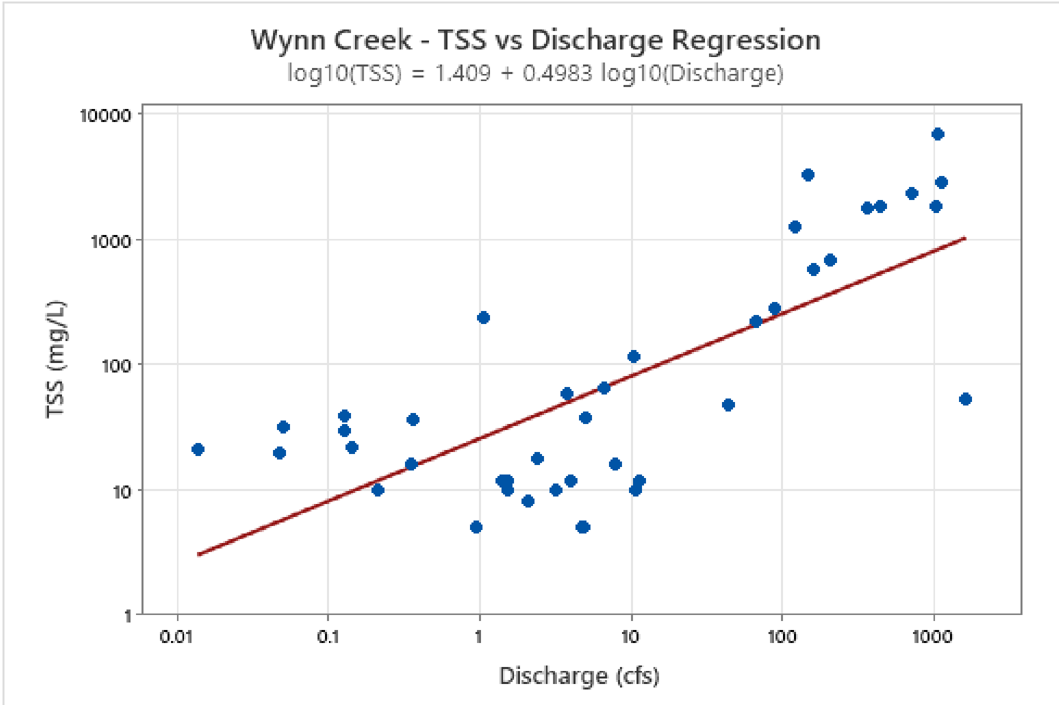


Figure 87. Total suspended solids versus discharge regression at Wynn Creek. R^2=54.1%, p<0.001, n=41.

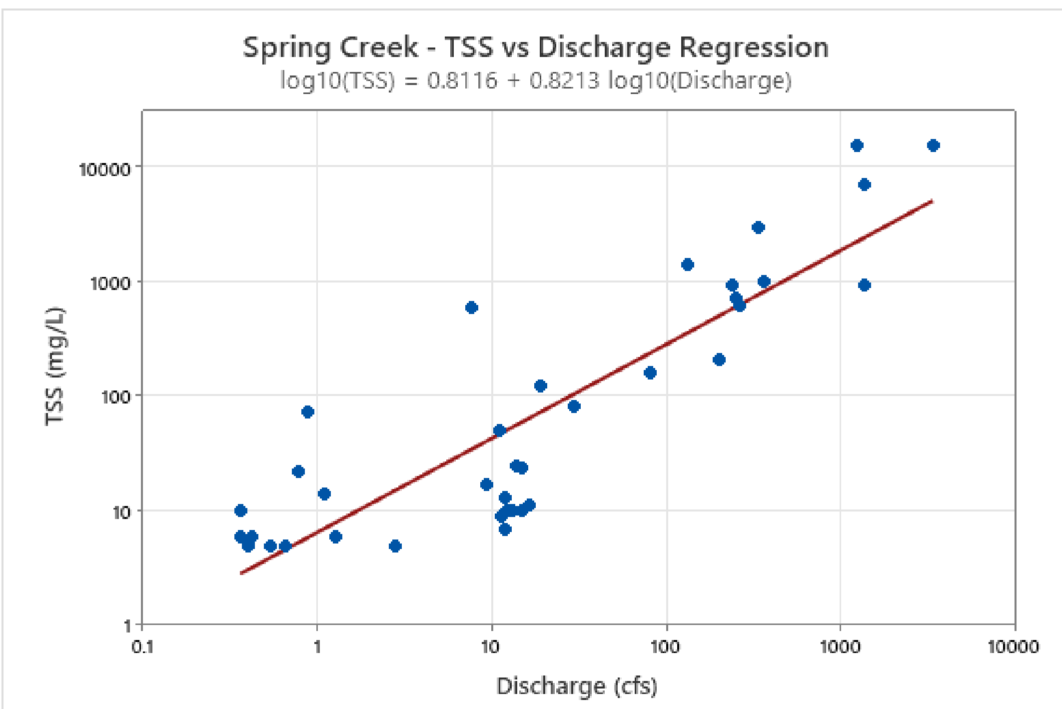


Figure 88. Total suspended solids versus discharge regression at Spring Creek. R^2=75.3%, p<0.001, n=38.

Total Organic Carbon

TOC surface samples had a range of 3.4 to 5.7 mg/L and a mean of 4.1 mg/L and showed increased concentrations during the growing season and at Site 5 in spring and fall (Figure 89). Observations from each sampling trip showed that sites tended to covary with an occasional peak (e.g., 5.7 mg/L on May 7, 2019 at Site 5 or 5.2 mg/L on 7/6/2020 at Site 2; Figure 90a). Likewise, bottom samples showed a similar trend of higher concentrations in the spring and summer months (Figure 90b). Although it is unclear the origin of the organic matter, it would be safe to assume that it was comprised of autochthonous at both the surface and bottom locations at Sites 1 and 2 and allochthonous at Site 5 with a combination of sources between those two sites (cf. Cooke et al. 2005). Increased concentrations in the growing season at Sites 1 and 2 were most likely due to the increase in biomass for planktonic organisms along with suspended bacteria and organic matter in the surface samples and sinking detritus or bacterial degradation of sediment organic matter in the bottom samples (Wetzel, 2001). TOC observed in the riverine section of the lake was probably derived from increased lotic input of organic matter from the watershed (e.g., leaf litter) in the spring and fall (Wetzel, 2001).

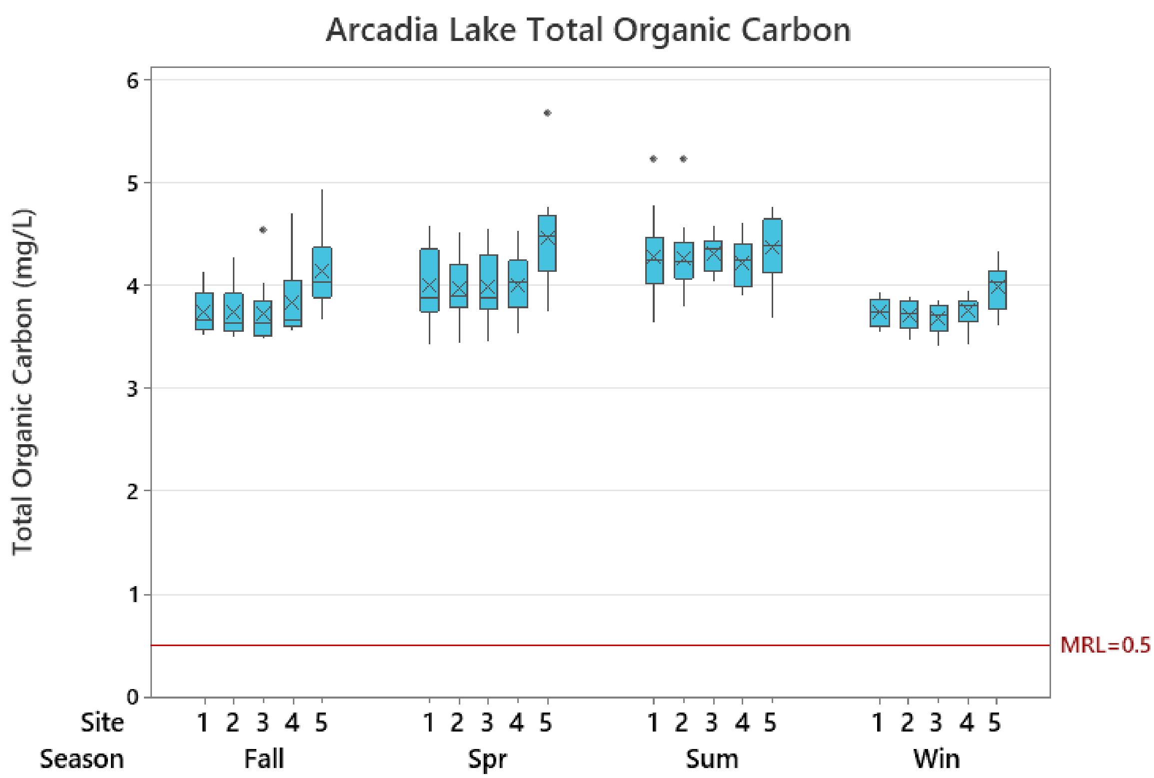


Figure 89. Lake Arcadia boxplots for total organic carbon surface collections categorized by season and site (Fall: n=9, Spring: n=14, Summer n=18, Winter n=8).

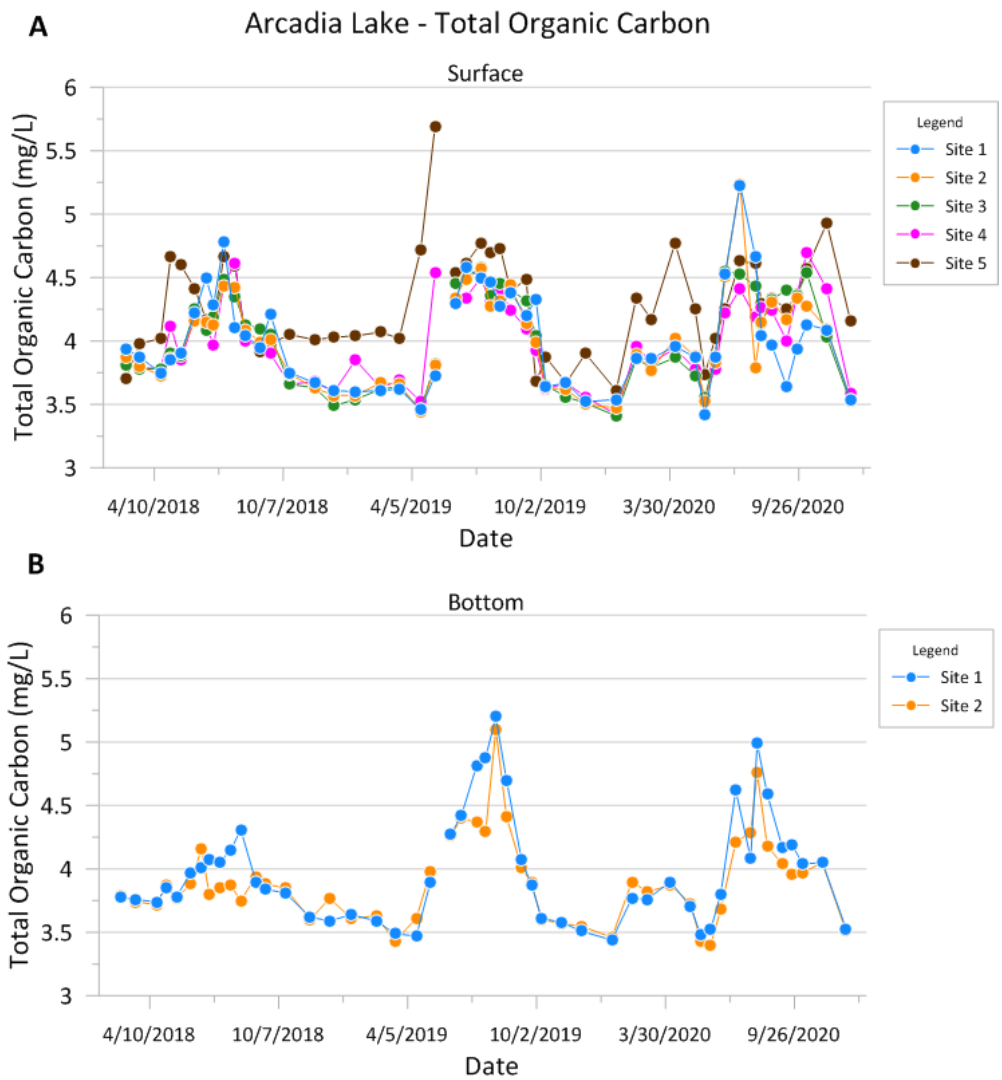


Figure 90. Lake Arcadia (a) surface and (b) bottom total organic carbon plotted for each sampling trip (n=49) by site. Note that sampling occurred bi-monthly during the growing season (May-October).

Phytoplankton

Phytoplankton taxonomy was completed by Baylor University. The data indicate Arcadia Lake is rich in nutrients as shown in the diverse algal community present. The taxa present indicate a eutrophic to hypereutrophic system. Predominant taxa were blue-greens that are mostly small and highly buoyant. Arcadia Lake's hypereutrophic waters allow it to host a myriad of different phytoplankton species, including over a dozen genera of cyanobacteria (Figure 91).

Cyanobacteria made up a significant portion of total cell biovolume and are present throughout all seasons (Figure 92). Diatoms become dominant (i.e., contribute 50% or more to overall biovolume) in the late spring to early summer in 2018 and 2019. In 2020, there was an increase in the presence of diatoms, however their abundance is less than the previous two years, and they are not the dominant group. These peaks in diatom abundance were followed the presence of green algae and cryptophytes in the mid to late summer, which is consistent with findings from previous studies on Arcadia Lake (OWRB, 2000). However, cyanobacteria became dominant in the early fall and remained throughout the winter. There were 6 sampling dates in which 100% of algal biovolume consisted of cyanobacteria, all of which occurred during the fall, winter, and spring. Although total biovolume stayed relatively consistent throughout the seasons—apart from summer 2020—the community shifted during the summer when cyanobacteria contributed less to the overall algal assemblage than it did in the winter—69% on average in the summer as opposed to 99% on average in the winter.

Twelve different genera of cyanobacteria were identified throughout the study period. *Synechococcus* was the predominant genera found during all seasons (Figure 93). *Synechococcus*, which is found in both freshwater and marine environments, is a small unicellular non-bloom-forming cyanobacterium ranging in size from about 1–22 µm and thrives in well-lit surface waters (Phyco Key). Throughout the study period, *Synechococcus* outcompeted all other genera of cyanobacteria when turbidity was high, indicating they are better equipped to regulate buoyancy and remain in the well-lit photic zone than the other 11 genera present (Figure 93). This genus made up a significant portion of the overall algal assemblage, ranging from only about 3% of the total biovolume during times of low turbidity to approximately 98% during peak turbidity.

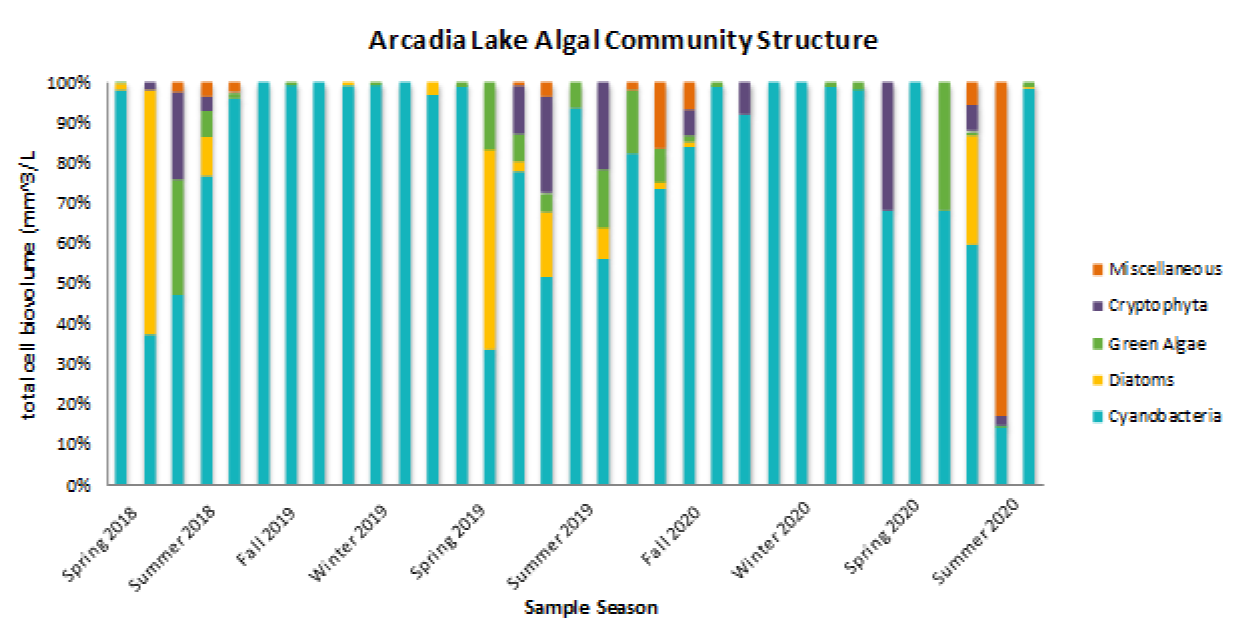


Figure 91. Arcadia Lake Algal Community Structure by Percent of Biovolume. Distribution of five major phytoplankton groups that make up the algal community in Arcadia Lake and what percentage each group contributes to overall algal cell biovolume (mm3/L) at each sampling date. Note that all phytoplankton samples were only taken at Site 1.

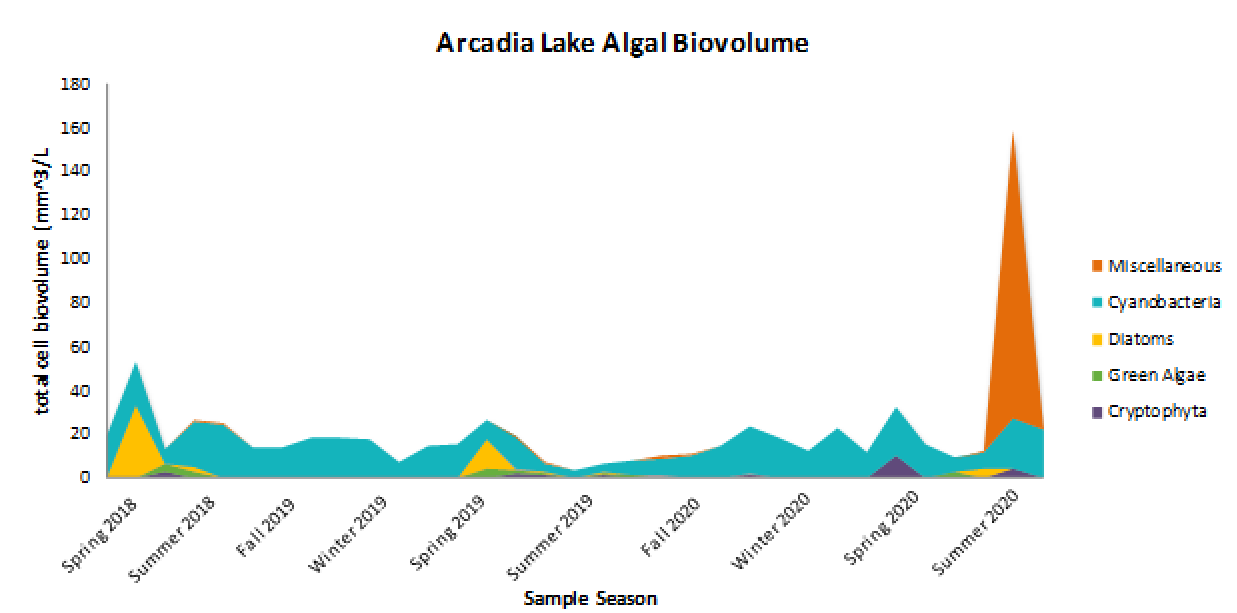


Figure 92. Arcadia Lake Algal Community Structure by Total Biovolume. Total algal cell biovolume (mm3/L) of Arcadia Lake broken into five major groups and plotted over the entire study period, denoted by sampling season. Note that all phytoplankton samples were only taken at Site 1.

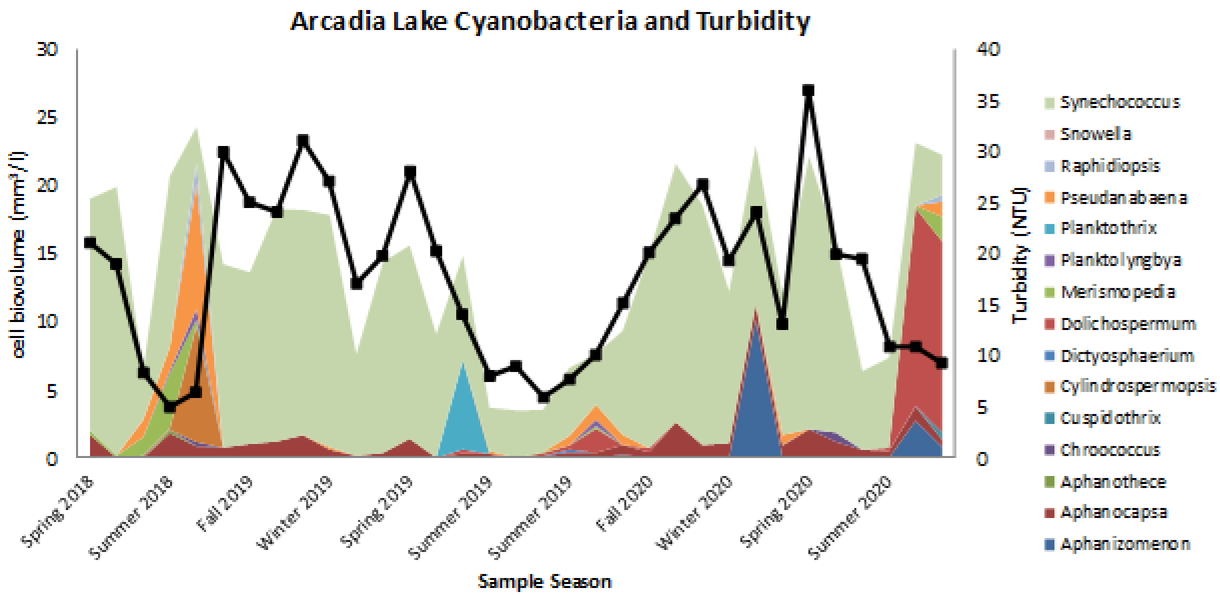


Figure 93. Arcadia Lake Cyanobacteria and Turbidity. Total cell biovolume (mm3/L) of cyanobacteria in Arcadia Lake grouped by genus and plotted over the entire study period with turbidity (NTU) plotted on a secondary axis. Note that all phytoplankton samples were only taken at Site 1.

Zooplankton

Taxonomic identification of zooplankton was completed by Oklahoma State University. The zooplankton community in Arcadia Lake primarily consists of cladocerans and copepods (Figure 94). Percent of biomass levels of the different zooplankton groups remained relatively consistent throughout the seasons with copepods contributing slightly more to overall biomass during the summer. There was, however, a significant increase in overall zooplankton biomass in the fall of 2019 before it stabilized again in early winter 2019 (Figure 95). This spike in copepods occurred simultaneously with a major shift in the cyanobacterial assemblage, when cyanobacteria were contributing over 95% to algal biovolume but *Synechococcus* was at much lower levels and genera including *Psuedanabaena* and *Cylindrospermopsis* were abundant (Figure 91).

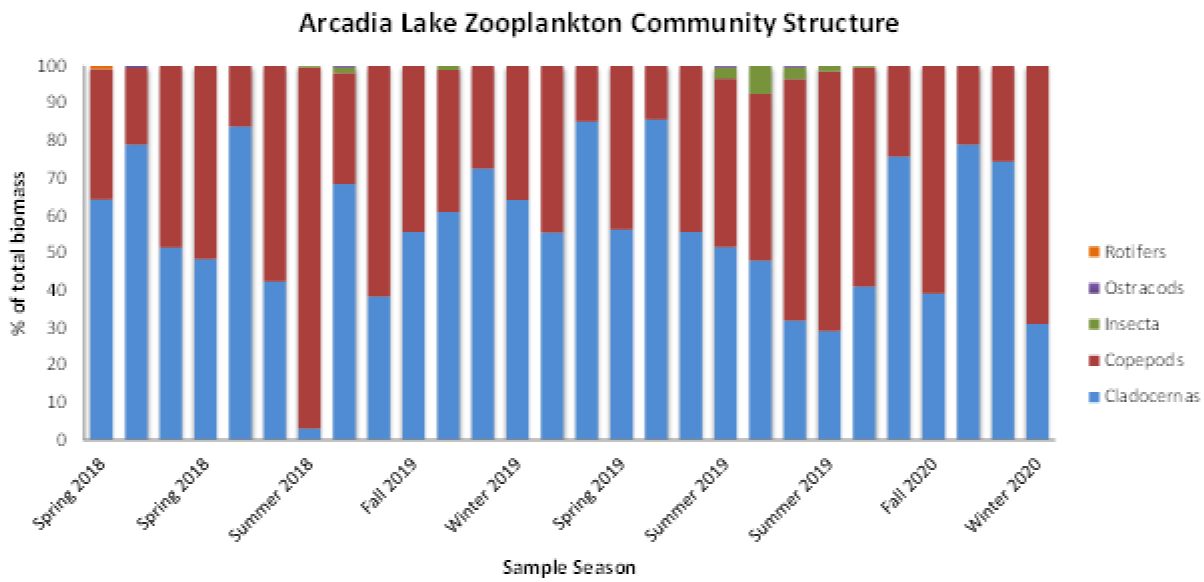


Figure 94. Arcadia Lake Zooplankton Community Structure. Shows which of the five major zooplankton groups makeup the zooplankton community in Arcadia Lake and what percentage each group contributes to overall biomass (individuals/L) at each sampling date. Note that all zooplankton samples were taken only at Site 1.

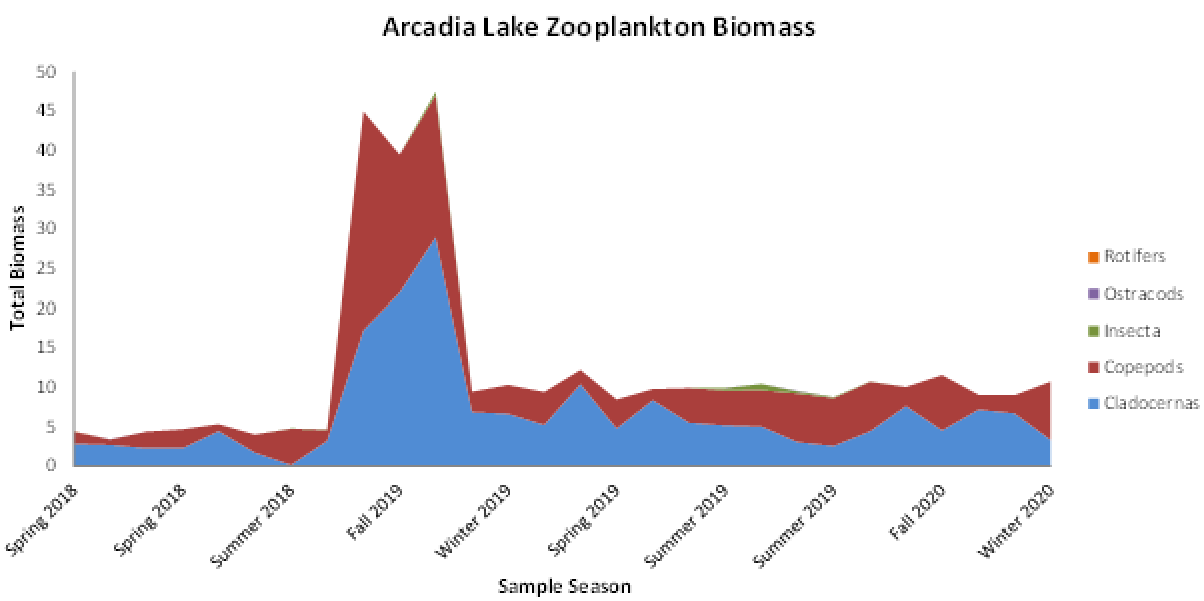


Figure 95. Arcadia Lake Zooplankton Biomass. Shows the total zooplankton biomass (individuals/L) of Arcadia Lake broken into five major groups and plotted over the entire study period, denoted by sampling season. Note that all zooplankton samples were taken only at Site 1.

Discharge

Discharge measurements were taken at each of the watershed monitoring locations throughout the monitoring period. These discharge measurements were taken regularly during baseflow conditions and as often as possible during TREs. Measuring discharge at multiple different stages has led to the development of a rating curve that allows for estimation of discharge at any given point along the hydrograph. The following flow duration curves show the percentage of time each watershed monitoring location presents at that discharge in cubic feet per second. As shown below (Figure 96, Figure 98, and Figure 100), baseflow conditions were observed most often because in-stream high flows were almost completely rainfall dependent.

The Deep Fork is the major flow contributor to Arcadia Lake. Annual mean discharge at the site was 27.6 cfs in 2018, 138 cfs in 2019, and 37.4 cfs in 2020. This is consistent with elevated rainfall totals in 2019 as compared to the other sampling years. Maximum rated instantaneous discharges were 6771 cfs in 2018, 16193 cfs in 2019, and 13594 cfs in 2020. The maximum mean daily flow at the Deep Fork was 1380 cfs in 2018, 3060 cfs in 2019, and 2010 cfs in 2020 (Figure 97). A total of 90 events–21 in 2018, 43 in 2019, and 26 in 2020—totaling one hour or greater exceeded 196.7 cfs, the 5% threshold on the site's flow duration curve (Figure 96). These events ranged from just over the one-hour lower limit to several days, with a median exceedance interval of approximately four hours. The short-lived intensity of these events illustrates the high levels of impervious cover in the Headwaters Deep Fork and White Turkey Creek-Deep Fork subbasins (Figure 6, Table A9, and Table A10), as well as the use of the Deep Fork as a receiving stream for Oklahoma City MS4 stormwater conveyances. Baseflow discharges are the dominant flow measures throughout the year, and range between 3 cfs and 10 cfs (Figure 96 and Figure 97).

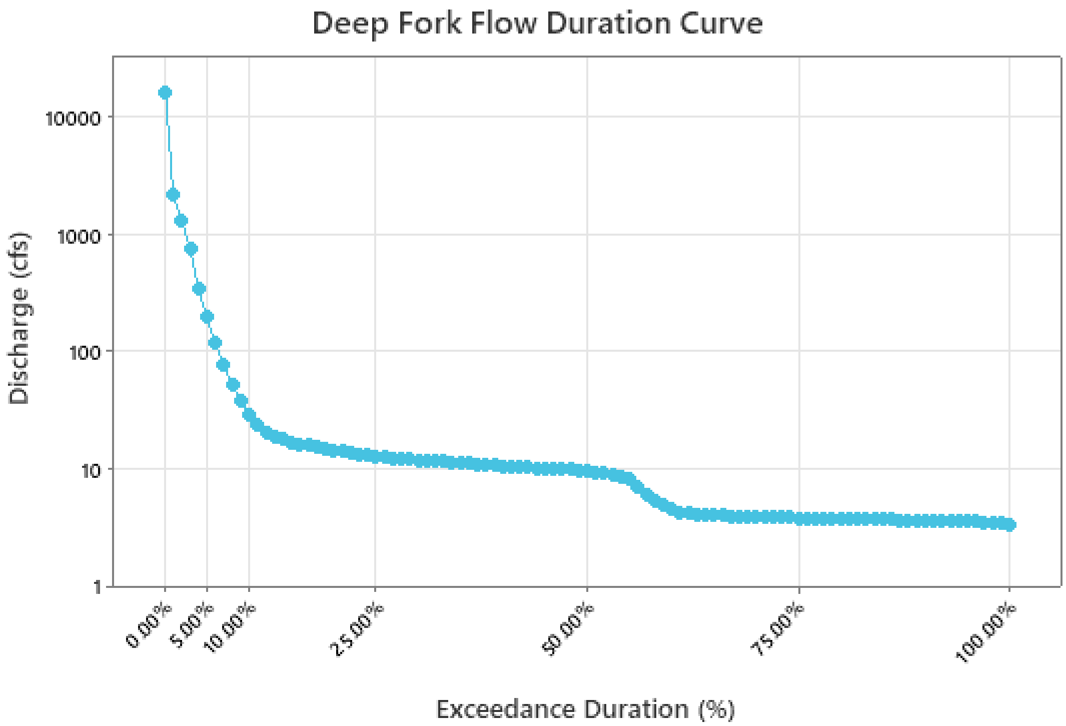


Figure 96. Flow duration curve, Deep Fork.

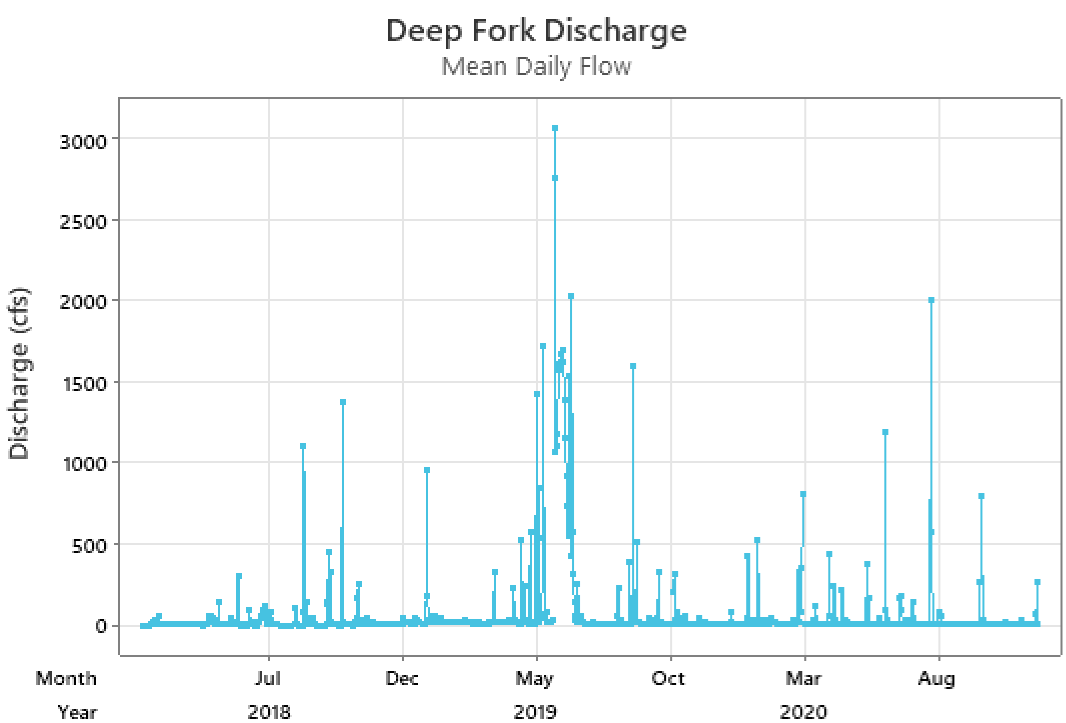


Figure 97. Deep Fork mean daily flow (cfs)

Wynn Creek is the lowest flow contributor of the three watershed sites, discharging into the Deep Fork a few miles upstream of the Deep Fork-Arcadia Lake lotic-lentic transition. Annual mean discharge at the site was 8.03 cfs in 2018, 13.1 cfs in 2019, and 7.77 cfs in 2020. Maximum rated instantaneous discharges at Wynn Creek were 1106 cfs in 2018, 1556 cfs in 2019, and 1297 cfs in 2020. The maximum mean daily flow was 278 cfs in 2018, 301 cfs in 2019, and 195 cfs in 2020 (Figure 99). A total of 97 events—26 in 2018, 43 in 2019, and 28 in 2020—totaling one hour or greater exceeded 35.9 cfs, the 5% threshold on the site's flow duration curve (Figure 98). These events ranged from just over the one hour lower limit to a couple days, with a median exceedance interval of approximately nine and a half hours. The runoff contributing area, developed land uses, and impervious cover is significantly lower than the Deep Fork site, leading to longer events at lower intensities (Figure 6, Figure 97, Figure 99, Table A9, and Table A10). Urbanization was high enough in the Wynn Creek watershed to lead to flashy events typical of an urban stream. Baseflow discharges are the dominant flow values throughout the year, and range between just above 0 cfs to 3 cfs (Figure 98 and Figure 99).

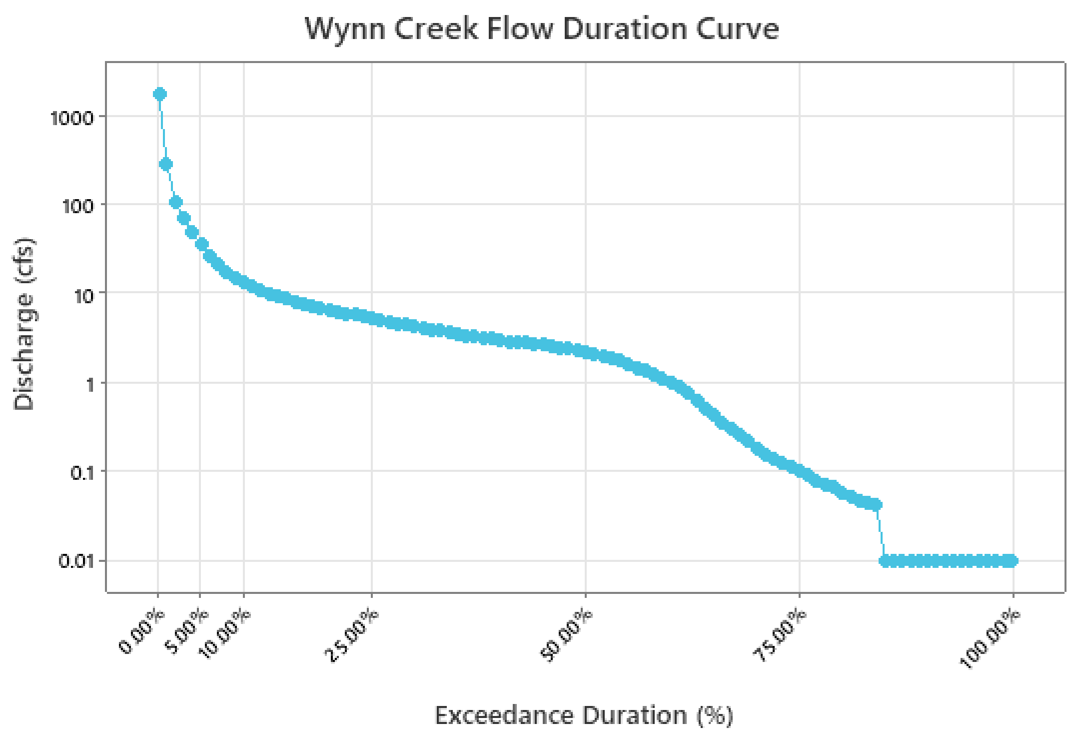


Figure 98. Flow duration curve, Wynn Creek.

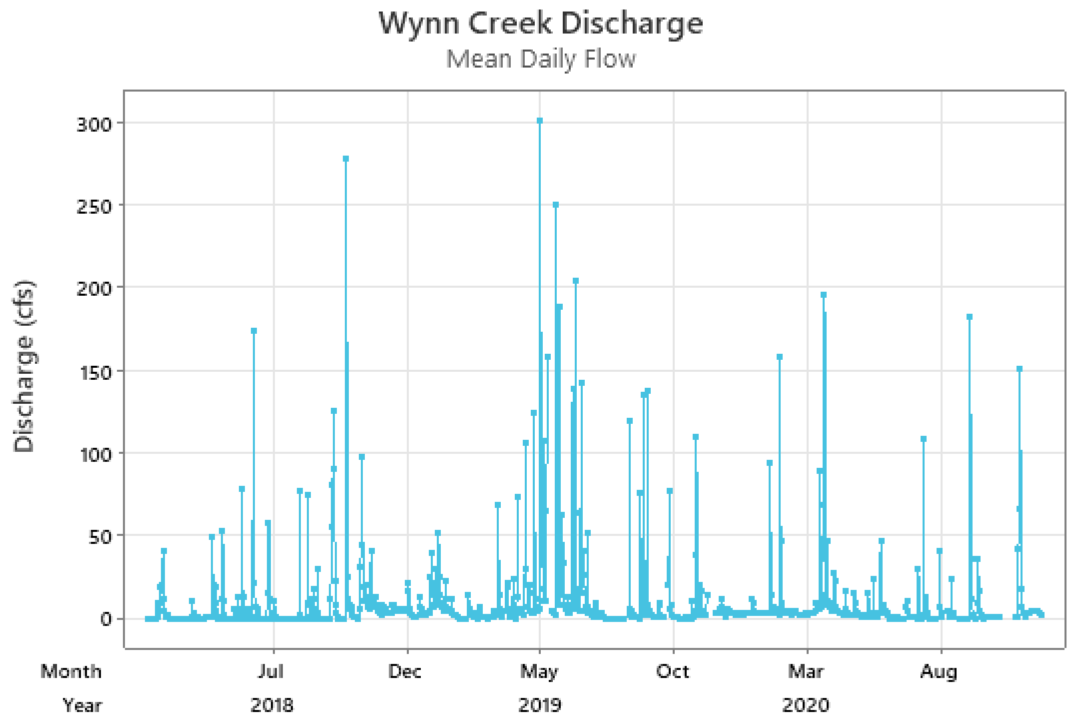


Figure 99. Wynn Creek mean daily flow (cfs).

Spring Creek is the second largest flow contributor to Arcadia Lake, behind the Deep Fork, the lake's primary inflow stream. Annual mean discharge at the site was 15.6 cfs in 2018, 20.3 cfs in 2019, and 4.03 in 2020. Heavy rainfall accounted for higher discharges in Spring Creek during 2019, similar to the Wynn Creek and Deep Fork sites. Maximum rated instantaneous discharges for Spring Creek were 1599 cfs in 2018, 3083 cfs in 2019, and 2183 cfs in 2020. The maximum mean daily flow at Spring Creek for those same years was 211 cfs, 550 cfs, and 201 cfs, respectively (Figure 101). A total of 93 events—29 in 2018, 32 in 2019, and 32 in 2020—totaling one hour or greater exceeded 59.0 cfs, the 5% threshold on the site's flow duration curve (Figure 100). These events ranged from just over one hour to over ten hours, with a median exceedance interval of approximately four hours. The Spring Creek watershed, similar to Wynn Creek and in the same subbasin, is highly urbanized and susceptible to flashy high flow events and large amounts of runoff, but not to the same degree as the Deep Fork (Figure 6, Table A9, and Table A10). Baseflow discharges are the dominant flow values throughout the year, and range between just above 0 cfs to 3 cfs (Figure 100 and Figure 101)

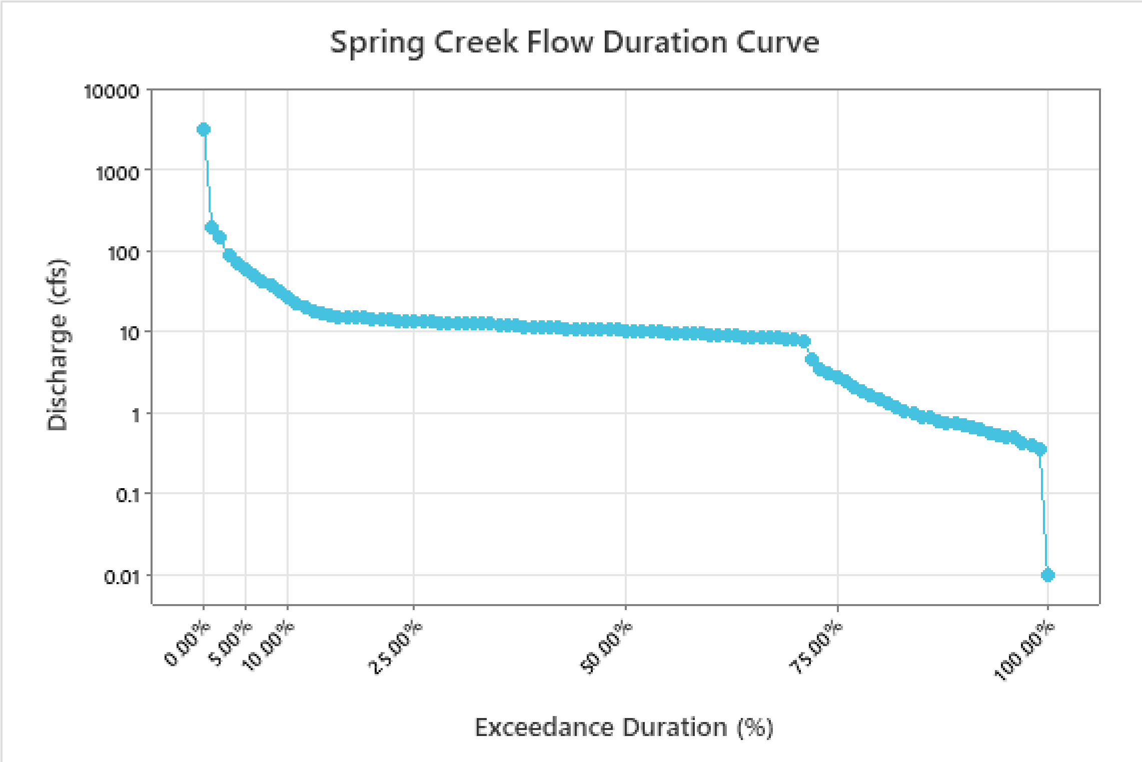


Figure 100. Flow Duration Curve, Spring Creek.

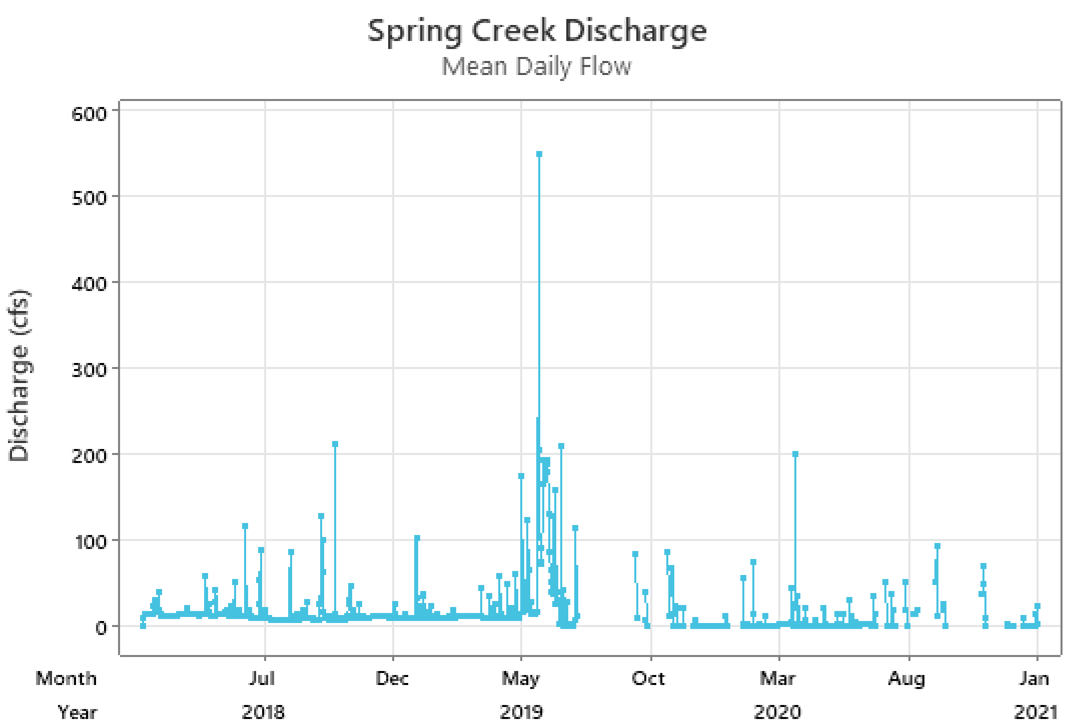


Figure 101. Spring Creek mean daily flow (cfs).

Loadings

Constituent loadings were estimated using LOADEST. LOADEST is a FORTRAN based software developed by the United States Geological Survey (USGS) for estimating constituent loads in rivers and streams (USGS, 2013). Total nitrogen, total phosphorus, and total suspended solids were analyzed at all three watershed sites to determine the estimated daily load in kilograms per day. These three parameters were chosen to best represent nutrient and sediment loading into Arcadia Lake. To estimate loads, the LOADEST model provides a maximum likelihood estimation and adjusted maximum likelihood estimation (AMLE). AMLE was used for all three parameters at all three watershed monitoring locations. Load estimation bias is measured in the software using several bias diagnostics, including Load Bias in Percent (Bp), Partial Load Ratio (PLR), and the Nash Sutcliffe Index. The USGS strongly recommends that load estimations only be used when Bp is less than or equal to +/- 25%. The model guidance also suggests that the observed mean may be a better estimate than the model when the Nash Sutcliffe Index is less than 0. All AMLE models for total phosphorus and total nitrogen had Bp values less than +/- 25%, allowing AMLE models to be used for load estimation. For several of the TP and TN load estimates, the Nash Sutcliffe was less than 0, but for the current analysis, load estimations were used. However, TSS load estimations at all three sites resulted in Bp values outside of the allowable percent bias. For TSS, an empirical estimation of loading should be used. Those empirical estimations are presented later in this section.

Consistent with constituent concentration measurements and discharge, the Deep Fork site was the largest contributor of TN and TP loads. Over the estimation period—February 2018 - December 2020—the estimated mean loads at the Deep Fork were 934 kg/day for TN and 240 kg/day for TP. Estimated daily mean loads for the same constituents ranged from 135 kg/day to 1,206,220 kg/day for TN (Figure 102) and 14 kg/day to 350,791 kg/day for TP (Figure 103). According to bias statistics calculated during the LOADEST model runs, TN and TP are likely slight underestimations of actual loadings.

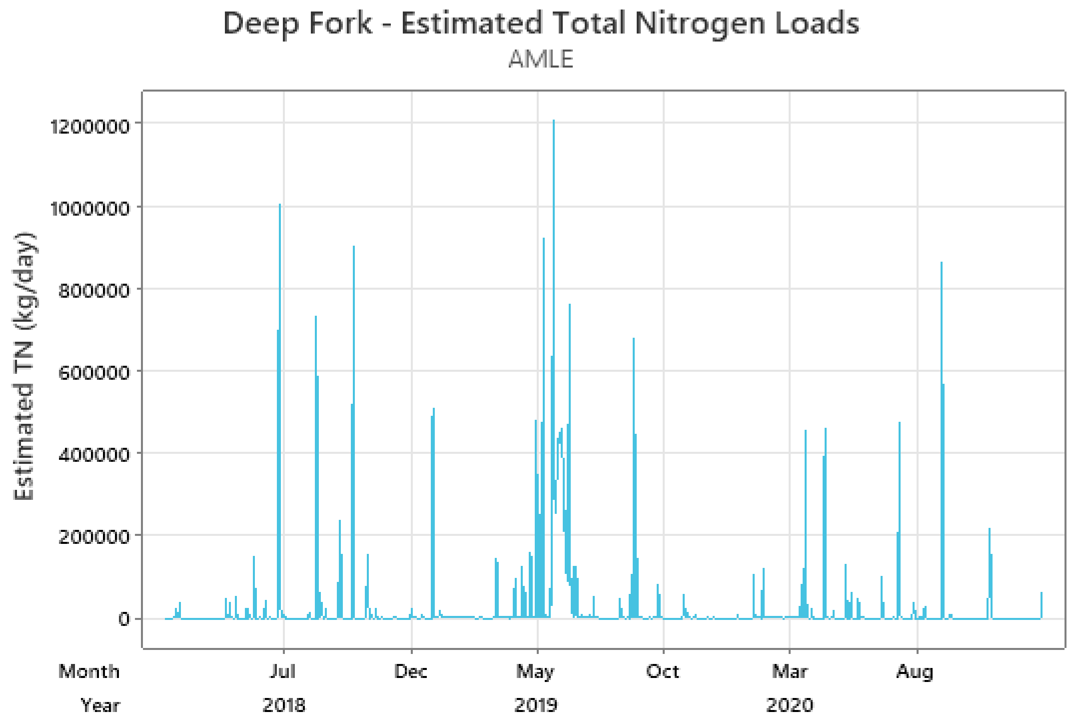


Figure 102. AMLE estimated total nitrogen loads in kg/day at Deep Fork.

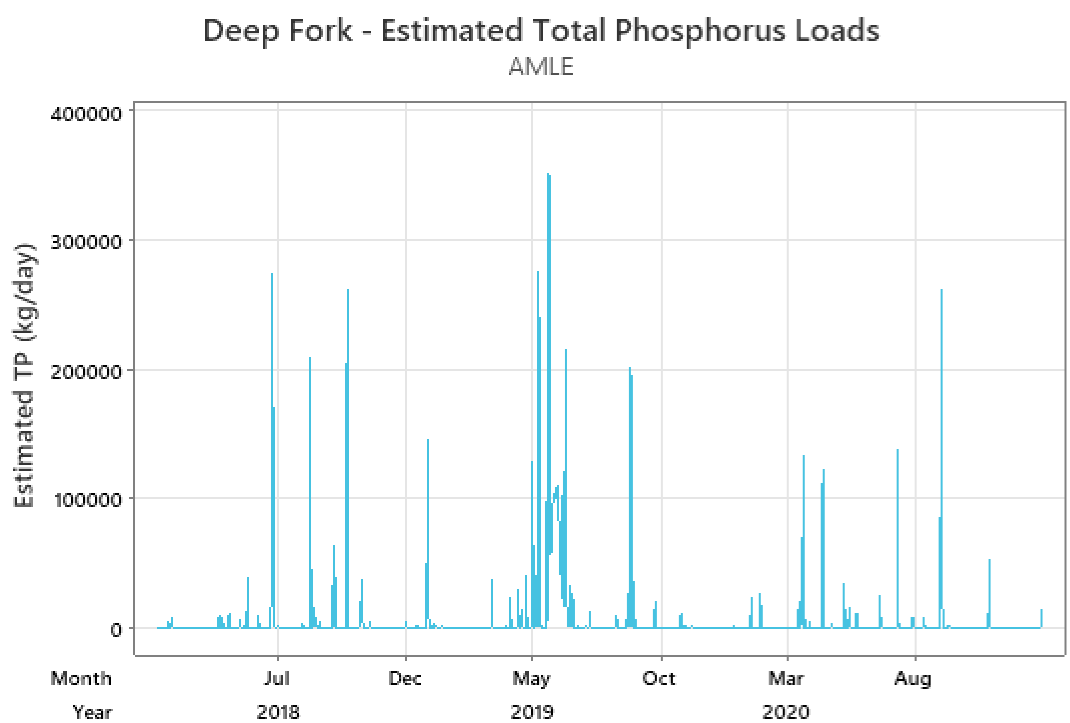


Figure 103. AMLE estimated total phosphorus loads in kg/day at Deep Fork.

Wynn Creek was the lowest contributor of estimated loads of the three watershed sites. Over the estimation period the estimated mean loads at Wynn Creek were 63.9 kg/day for TN and 12.7 kg/day for TP. Estimated daily mean loads for the same constituents ranged from 1.0 kg/day to 111,417 kg/day for TN (Figure 104) and 0.1 kg/day to 27,724 kg/day for TP (Figure 105). According to bias statistics calculated during the LOADEST model runs, TN and TP are likely slight underestimations of actual loadings.

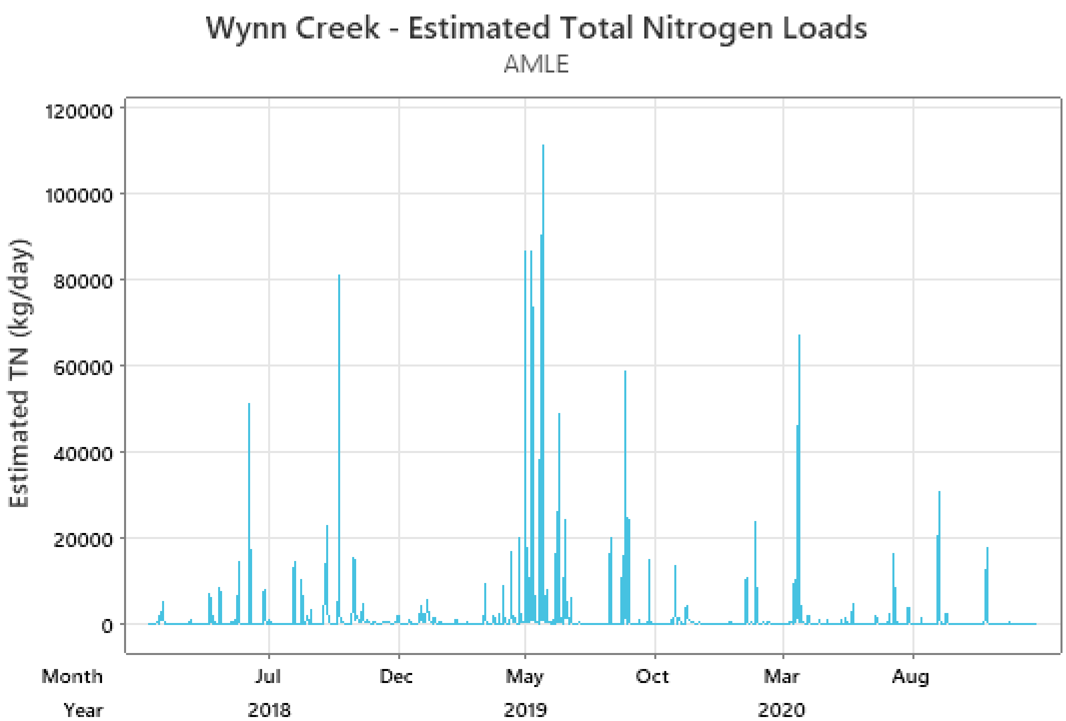


Figure 104. AMLE estimated total nitrogen loads in kg/day at Wynn Creek.

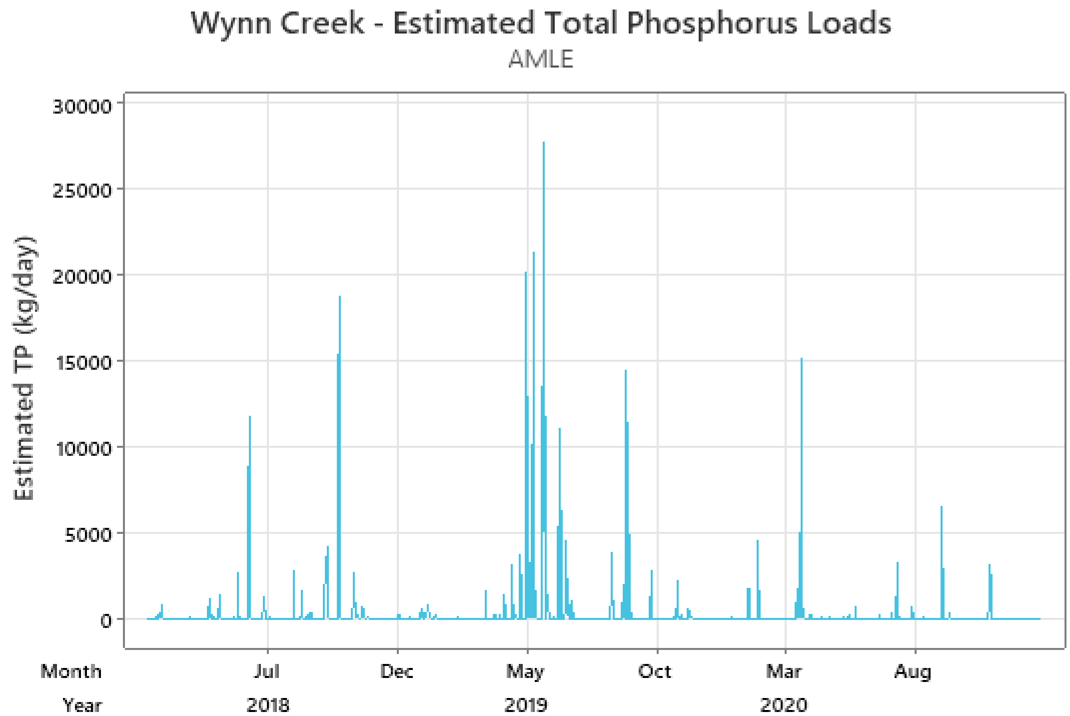


Figure 105. AMLE estimated total phosphorus loads in kg/day at Wynn Creek.

Spring Creek estimated mean loads during the estimation period were 117 kg/day for TN, 27.3 kg/day for TP, and 115,087 kg/day for TSS. Estimated daily mean loads for the same constituents ranged from 8.0 kg/day to 426,982 kg/day for TN (Figure 106) and 1.0 kg/day to 120,332 kg/day for TP (Figure 107). According to bias statistics calculated during the LOADEST model runs, TN and TP are likely slight overestimations of actual loadings.

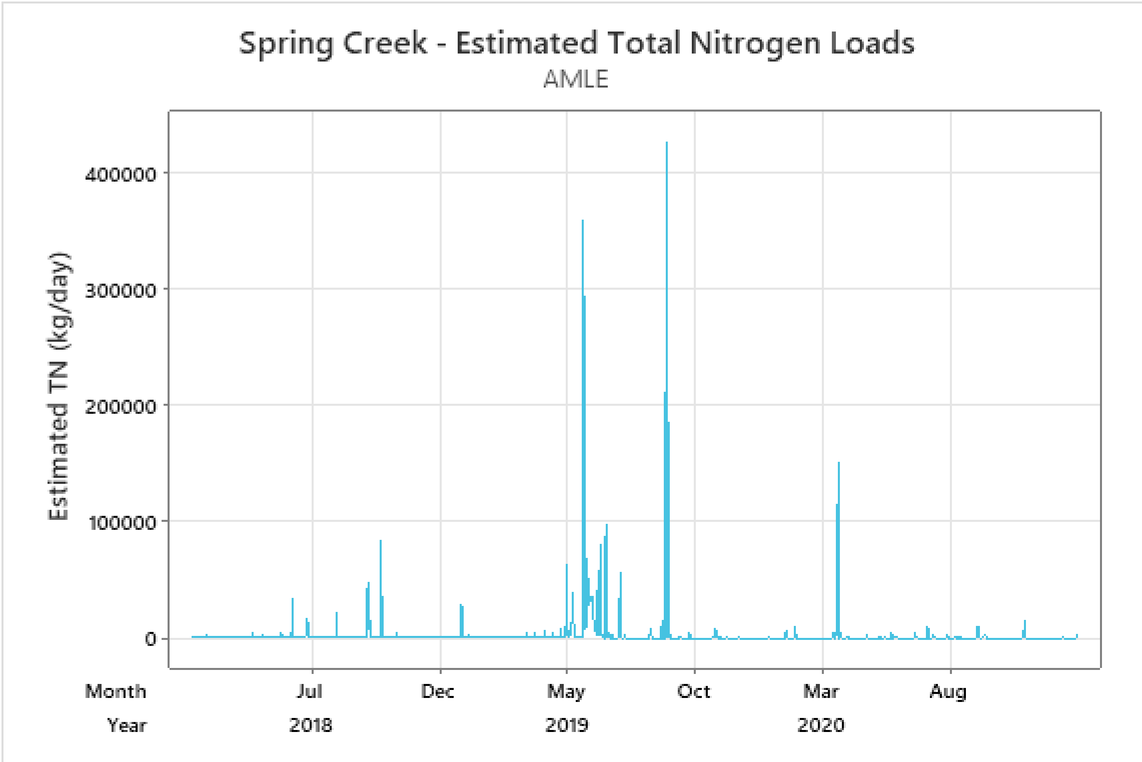


Figure 106. AMLE estimated total nitrogen loads in kg/day at Spring Creek.

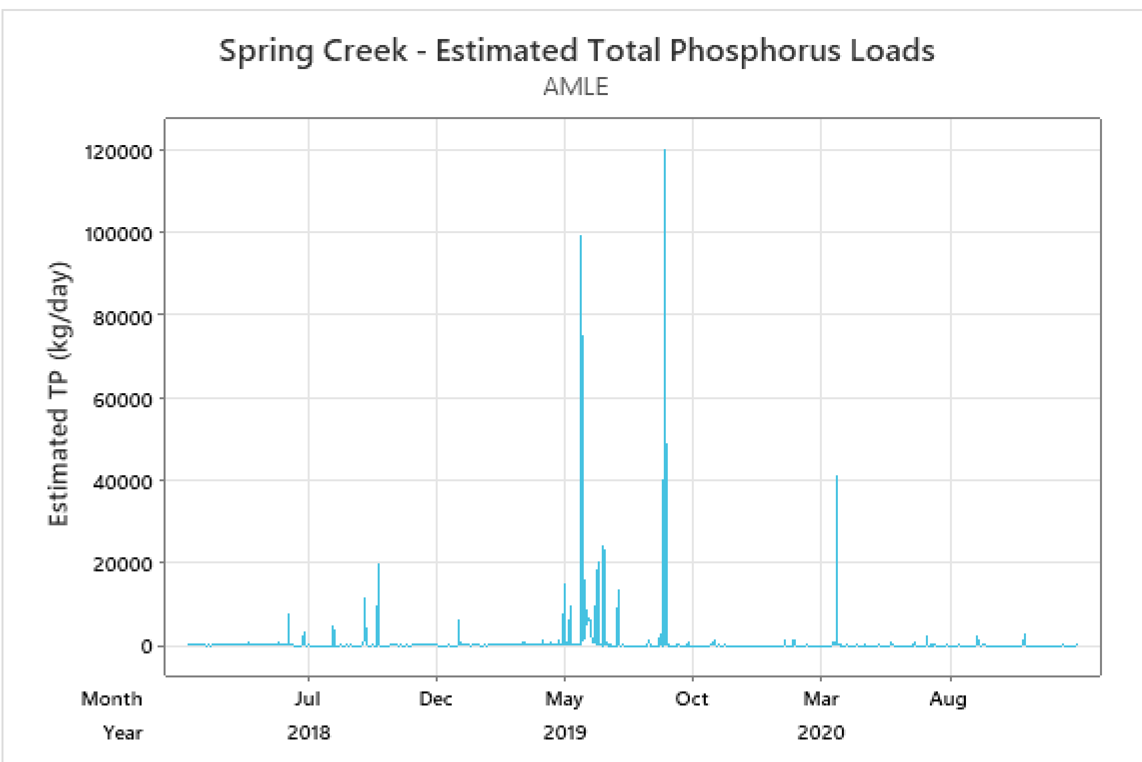


Figure 107. AMLE estimated total phosphorus loads in kg/day at Spring Creek.

As noted in the above sections, TSS loads as estimated by LOADEST were not used due to bias statistics indicating the model estimations were outside of acceptable limits, specifically Bp. Although this may have occurred for several reasons, a short period of record is likely the dominant reason for high level of apparent bias in model estimates. However, empirical loads are useful and are presented to inform on sediment movement from the watershed to the eventual receiving water, Arcadia Lake. To test the efficacy of using empirical loadings, linear regressions of discharge to TSS were made. All three sites demonstrate good correlation at both baseflow and high flow periods (Figure 111, Figure 112, and Figure 113). For each time at each site, relationships are highly significant ($p < 0.01$), and R^2 values are typically near 80% or higher. Only the Deep Fork baseflow period had an R^2 less than 79% but was still relatively high at 64%.

To estimate empirical loads, the median value is used because data are not normally distributed and are highly positively skewed (Figure 108, Figure 109, and Figure 110). The means for all 3 datasets approach or exceed the 75th percentile of the data. Given the flashy characteristics of all three watershed sites, and TSS load dominance from TRE collections despite temporal dominance of baseflow conditions, TSS loadings were also split into baseflow and TRE load mean and median calculations.

The median TSS load, including both baseflow collections and TRE collections, at each watershed site were as follows: 1,270 kg/day at Deep Fork, 304 kg/day at Wynn Creek, and 411 kg/day at Spring Creek (Figure 108). These calculated values are orders of magnitude lower than their counterpart mean values. Mean and median TSS loads for baseflow collections were 1,130 kg/day and 265 kg/day respectively at Deep Fork, 265 kg/day and 44 kg/day respectively at Wynn Creek, and 709 kg/day and 205 kg/day at Spring Creek (Figure 109). Mean and median TSS loads for TRE collections were 38,013,371 kg/day and 4,214,277 kg/day respectively at Deep Fork, 2,528,743 kg/day and 357,838 kg/day respectively at Wynn Creek, and 15,470,863 kg/day and 549,942 kg/day respectively at Spring Creek (Figure 110 and Figure 111)

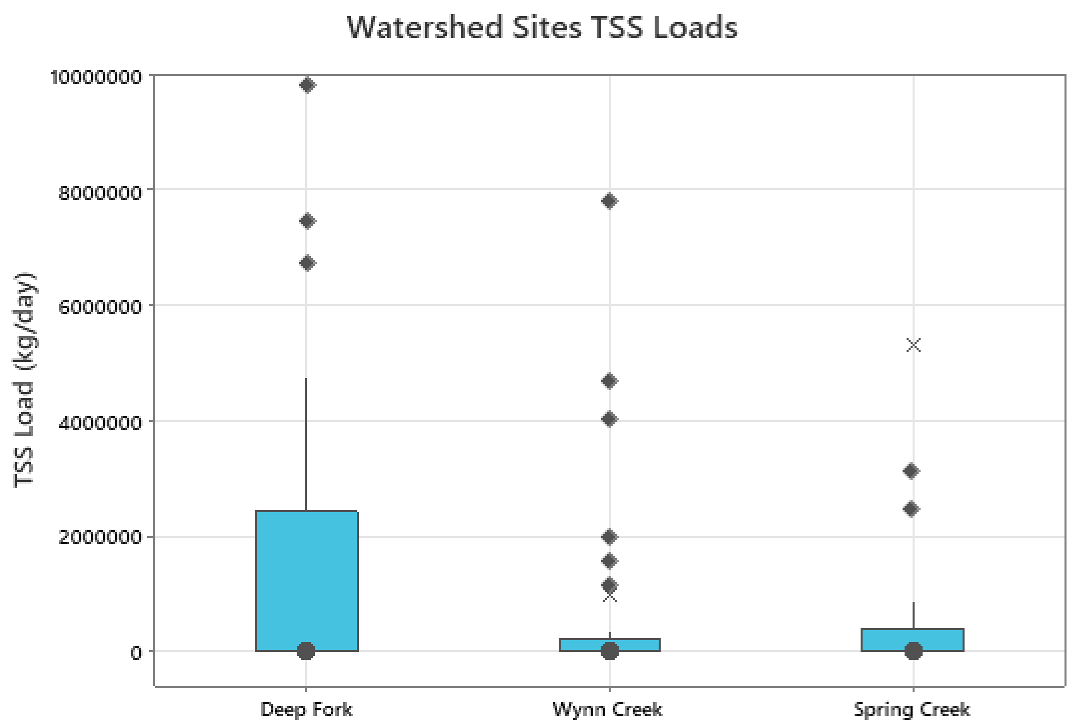


Figure 108. Watershed sites TSS loads in kg/day, Deep Fork (n=43, 4 outliers not shown), Wynn Creek (n=41, 1 outlier not shown), and Spring Creek (n=38, 3 outliers not shown). Circle denotes median. X denotes mean (Deep Fork mean not shown).

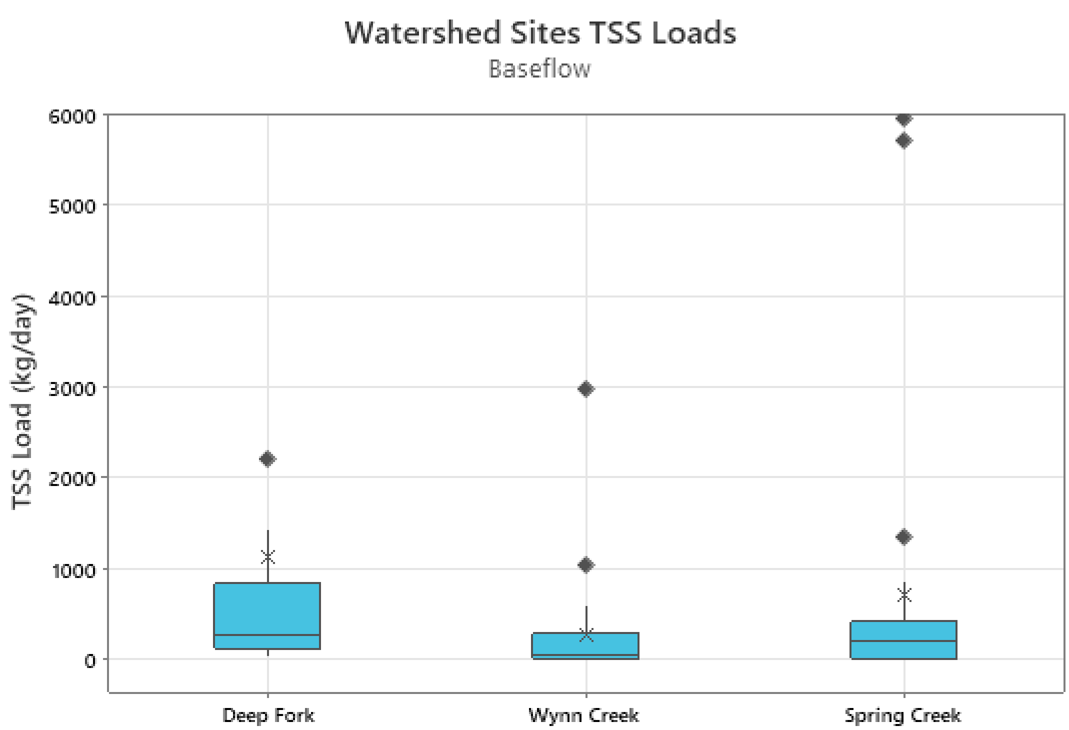


Figure 109. Watershed sites TSS loads in kg/day, Deep Fork (n=25, 2 outliers not shown), Wynn Creek (n=25), and Spring Creek (n=25). X denotes mean.

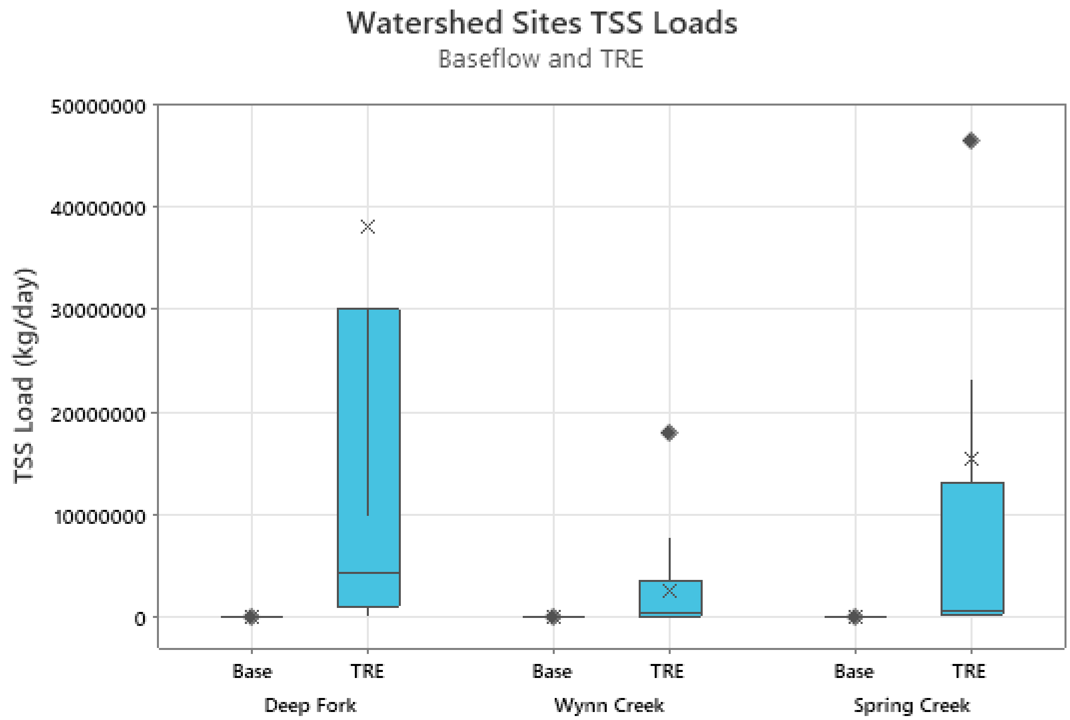


Figure 110. Watershed sites TSS loads in kg/day, Deep Fork baseflow (n=25) and TRE (n=18, 4 outliers not shown), Wynn Creek baseflow (n=25) and TRE (n=16), and Spring Creek baseflow (n=25) and TRE (n=13, 2 outliers not shown). X denotes mean.

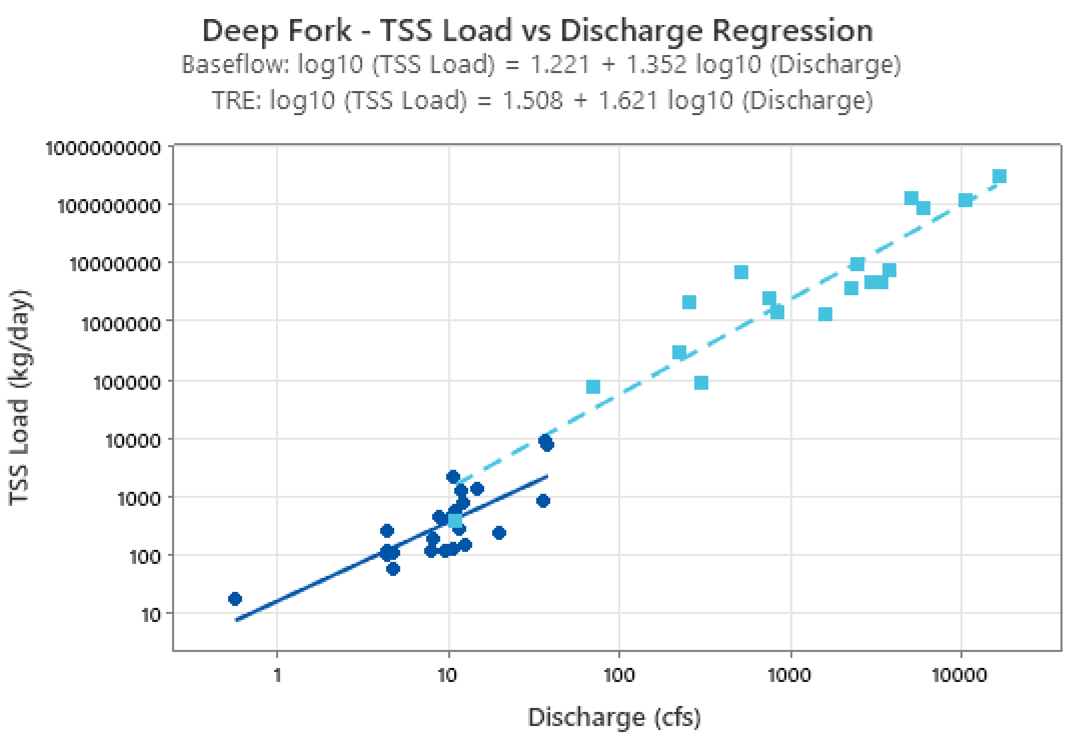


Figure 111. TSS load versus discharge regression at Deep Fork, baseflow calculations (R^2=61.4%, $p<0.001$, n=25) represented by dark blue dots, TRE calculations (R^2=87.0%, $p<0.001$, n=18) represented by light blue squares.

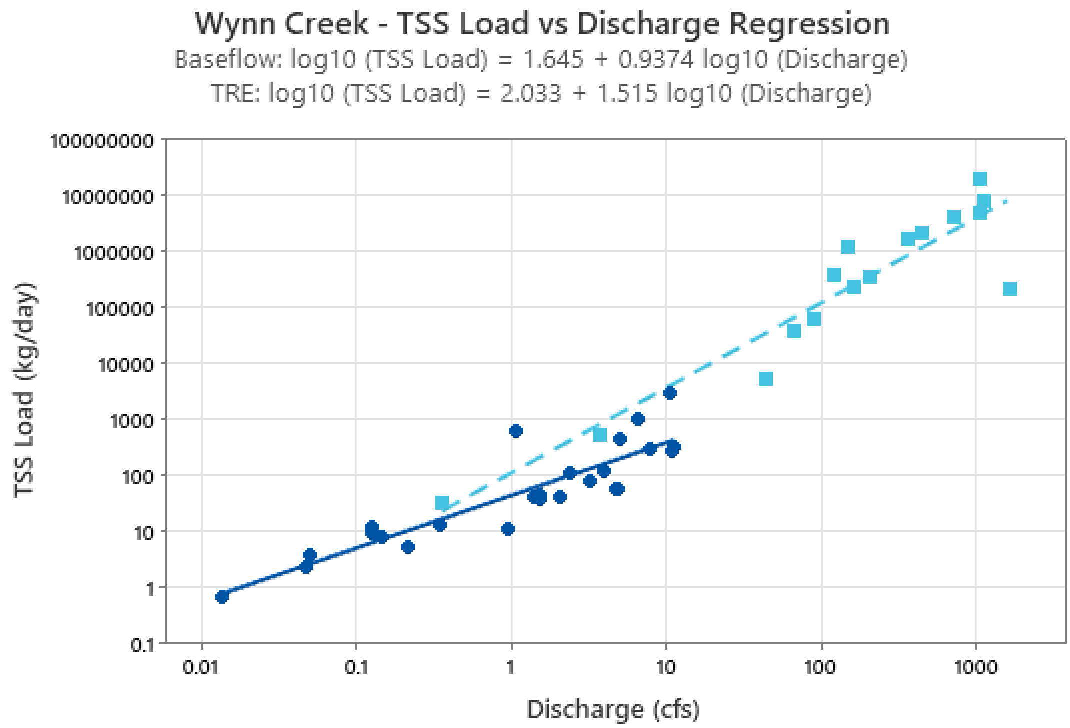


Figure 112. TSS load versus discharge regression at Wynn Creek, baseflow calculations (R^2=79.1%, p<0.001, n=25) represented by dark blue dots, TRE calculations (R^2=87.3%, p<0.001, n=16) represented by light blue squares.

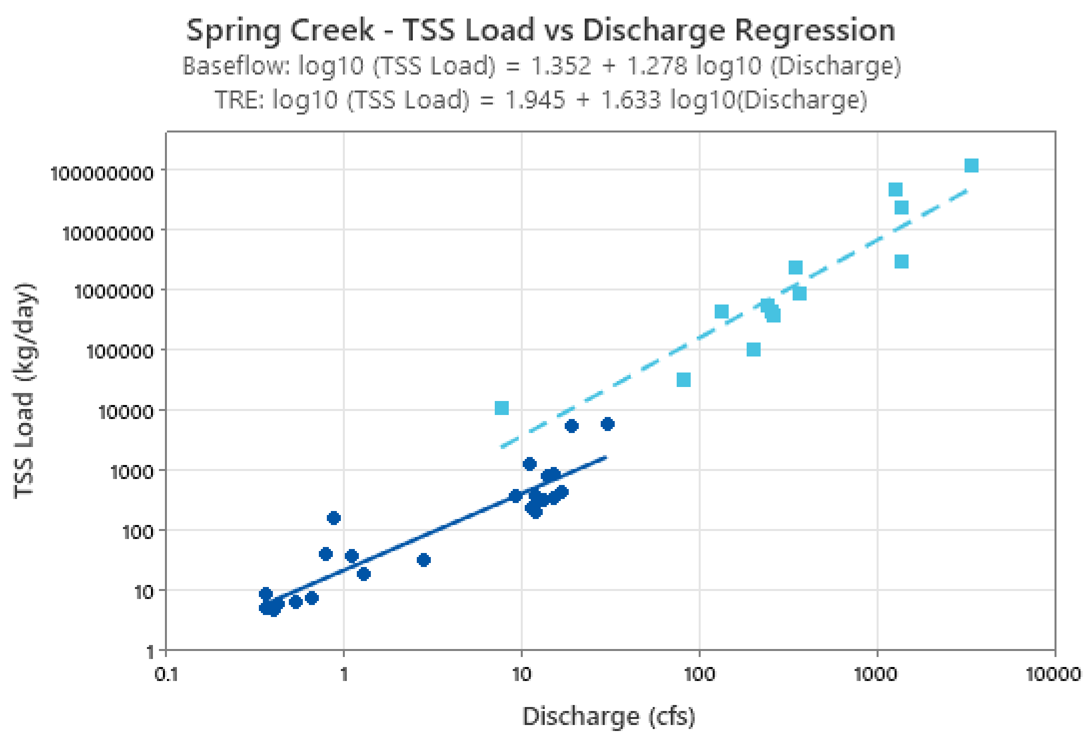


Figure 113. TSS load versus discharge regression at Spring Creek, baseflow calculations (R^2=84.8%, p<0.001, n=25) represented by dark blue dots, TRE calculations (R^2=84.7%, p<0.001, n=13) represented by light blue squares.

DISCUSSION AND CONCLUSIONS

Nutrients are essential for lake ecology; however, there are severe consequences to lake conditions when nutrient concentrations become too high. A statewide lake probabilistic monitoring study report was completed by the OWRB providing a statewide perspective on the condition of Oklahoma lakes (OWRB, 2020a). The key findings from the study identified that 60% of Oklahoma's lake acres are classified as most disturbed for total nitrogen and 43% of Oklahoma's lake acres are classified as most disturbed for total phosphorus. These high nutrient concentrations, combined with the TSI results showing 69% of lake acres are either eutrophic or hypereutrophic, reveal that many of Oklahoma's lakes experience negative water quality conditions on a regular basis. Arcadia Lake, which was part of the statewide study, is classified as eutrophic with high primary productivity and nutrient rich conditions. These results provide strong evidence for the need to reduce and manage nutrient pollution across Oklahoma, particularly nitrogen as it is a prevailing stressor on Oklahoma lakes (OWRB, 2020a).

At the lacustrine locations (Site 1 and Site 2), total and dissolved forms of nutrients, primarily phosphorus and nitrogen, were examined with respect to their spatial and temporal trends, as well as their role in limiting algal growth. Dissolved orthophosphate (DOP), the biologically available form of phosphorus, was not detectable in the epilimnion during the growing season, likely due to uptake by algae. During the growing season hypolimnetic phosphorus values were high, stemming from the effect of thermal stratification and internal release from anoxic sediment. In fall, hypolimnetically stored phosphorus mixed into the water column during destratification and resulted in higher surface concentrations. Nitrogen, another nutrient important for algal growth, followed a similar pattern. Ammonia and nitrate/nitrite are two forms of nitrogen available to algae and remained near zero, below the detection limit, for most of the growing season at the surface indicating a significant amount of algal uptake is occurring in the lake. During stratification, a buildup of ammonia occurred in the reduced conditions of the hypolimnion and upon destratification, this surplus of hypolimnetic ammonia was converted to nitrate/nitrite upon mixing.

Lacustrine surface phosphorus concentrations were generally lower than riverine surface phosphorus, suggesting delivery of a large load of this nutrient to system via runoff from the watershed. Riverine areas allow for the continuous cycling and resuspension of nutrients, due to their shallow depths being susceptible to wind mixing. Similarly, lacustrine nitrogen concentrations were generally lower than riverine nitrogen, again suggesting the tributaries are an important source of both phosphorus and nitrogen inputs. Neither nutrient was likely to be substantially limiting algal growth, as both were present in abundant amounts.

Comparing watershed nutrient values to OWQS, both Spring Creek and Deep Fork should be considered nutrient-threatened. Since the OWQS does not have statewide numeric nutrient criteria for flowing waters, the dichotomous process for determining if a waterbody should be identified nutrient threatened in Chapter 46 (OAC, 2020b 785:46-15) was used for this assessment. This process uses stream order, stream slope, canopy cover, nutrient values, and turbidity to determine if a stream is being degraded due to excessive nutrients. Step one is to determine the stream order for the river segments being assessed. The sampling locations for both Wynn Creek and Deep Fork were in fourth-order stream segments of their respective waterbodies, while the sampling location for Spring Creek was in a third order stream segment. First, second, and third order streams are assessed for nutrient-threatened status using slightly different framework in the dichotomous process than fourth order or larger streams.

After stream order is determined, then stream slope is analyzed. First, second, and third order streams with a slope greater than or equal to 17 feet per mile are first assessed using a threshold concentration of 0.24 mg/L for TP, including both baseflow and TRE collections, or a threshold concentration of 4.95 mg/L for nitrate plus nitrite, where an exceedance of either triggers the next step in the dichotomous framework. Stream slope at the sampling location for Spring Creek is 23.2 ft/mile (USGS, 2016). Spring Creek's mean TP was 0.54 mg/L and its mean nitrate plus nitrite was 0.21 mg/l, including both baseflow and TRE collections. Since Spring Creek's mean TP concentration exceeded the nutrient criterion, percentage of canopy shading was analyzed. This data was not collected as part of this study, but the Oklahoma Conservation Commission did have 6 past habitat assessments that were completed on this stream segment. The average canopy percentage for those 6 visits was 15.8% while the screening criterion for the canopy shading step is 80%. The next decision in the dichotomous process is determination of inorganic or organic turbidity type. Spring Creek's turbidity was determined as inorganic by nature. The final assessment step for first, second, and third order streams is to assess baseflow turbidity values against a threshold of 20 NTUs. If baseflow turbidity is less than 20 NTUs the site will be deemed as nutrient-threatened. It should be noted, however, that baseflow determination as defined in Chapter 46 of the Oklahoma Administrative Code is functionally different than baseflow as defined previously in this report (OAC, 2020b). Spring Creek's baseflow turbidity was 14.5, and therefore should be considered nutrient-threatened for TP.

Fourth order and larger streams adhere to the same 17 feet per mile threshold stream slope framework as first, second, and third order streams. Wynn Creek has a slope of 17.1 feet per mile and Deep Fork has a slope of 13 feet per mile. Since Wynn Creek has a slope of greater than 17 ft per mile it needs to be compared to different nutrient values than the Deep Fork. Decision values for Wynn Creek, a stream with a slope of greater than 17 feet per mile, are mean TP concentration greater than 1.00 mg/L or mean nitrate plus nitrate concentrations greater than 4.65 mg/L. Wynn Creek had a mean concentration of 0.34 mg/L for TP and 0.21 mg/L for nitrate plus nitrite, indicating that Wynn Creek is not nutrient-threatened. Since the Deep Fork's slope is less than 17 feet per mile, it is compared to a TP value of 0.36 mg/L or a nitrate plus nitrite value of 5.0 mg/L, where an exceedance of either triggers the next step in the dichotomous framework. Deep Fork had a mean nitrate plus nitrite value of 0.33 mg/L and a TP value of 0.58 mg/L, including both baseflow and TRE collections.Since Deep Forks TP is greater than the screening value. The last step is if baseflow turbidity is less than 20 NTUs then the site will be deemed as nutrient-threatened. Baseflow values for turbidity are, as noted in the above paragraph, determined differently than in the previous sections of this report. Deep Fork had a baseflow turbidity of 14.3 NTUs, and therefore the Deep Fork should be considered nutrient-threatened for TP.

Chlorophyll *a* is used as a proxy to measure algal biomass and it is important to understand the factors driving growth, due to its potential to cause drinking water and recreation issues. Arcadia Lake's SWS classification requires average chlorophyll *a* to be less than 10 mg/m3; chlorophyll *a* concentrations in the lake were consistently greater than 10 mg/m3. The lake-wide average for chlorophyll *a* of 18.9 mg/m3 greatly exceeds the SWS criterion of 10 mg/m3 for chlorophyll *a*, representing a need to mitigate conditions driving increased algal biomass. Of the 240 samples collected, 63% exceeded the criterion.

Riverine sites, specifically Site 5, experienced higher chlorophyll levels than lacustrine areas, however high turbidity likely limited algal growth and prevented even higher chlorophyll observed values.

Thirty-three percent of in-lake turbidity concentrations exceeded the water quality standard of 25 NTU for lakes. This can be attributed to the direct input into Arcadia Lake via Spring Creek and the Deep Fork. The convergence of Wynn Creek and Deep Fork in the watershed proved to be a leading contributor to turbidity for Arcadia Lake. Wet weather events, characterized by targeted rainfall event (TRE) collections at the watershed sites, were consistent drivers of sediment into the lake. Sedimentation is very apparent when comparing the riverine Site 5 to Site 1, which is located near the dam. As inputs from the tributaries enter Arcadia Lake, particularly where the Deep Fork discharges into the lake at Site 5, most of the sediment remains suspended throughout the water column. The general trend of lower water clarity moving from the riverine sites, attributed to input from Deep Fork and less so from Spring Creek (Site 3), to greater water clarity at the lacustrine site can be seen in Arcadia Lake. Both Cooke et al. (2001) and Thornton et al. (1990) predict this increase in water clarity moving towards the dam as sediment and other material settle from the water column as it flows its course through the reservoir. Increased turbidity can cause an increase in water temperature and reduced dissolved oxygen concentrations further limiting refuge for fish.

The turbidity standard for other surface waters, such as the watershed monitoring locations, is 50 NTU, which can only be assessed during baseflow conditions. During baseflow conditions, the average turbidity of each watershed monitoring location was below the 50 NTU threshold, with Wynn Creek having the highest average at 41 NTU. Wynn Creek also had the most construction observed directly upstream of the collection site, which would be a contributing factor to erosion and therefore turbidity. Based on these baseflow averages, Spring Creek, Wynn Creek and Deep Fork are currently meeting their beneficial use for Fish and Wildlife Propagation.

Two assessment methodologies for DO apply to protect the Fish and Wildlife Propagation beneficial use for warm water aquatic communities. Surface water DO criteria has a minimum seasonal threshold of 4.0 mg/L during the summer months and 5.0 mg/L in spring and fall (OAC 785:46-15-5). The volumetric criteria threshold for fully supporting the Fish and Wildlife Propagation beneficial is less than 50% of the cumulative lake volume measuring anoxic (< 2 mg/L DO). While the discrete profile taken during the day on a monthly sampling trip shows that the lake is meeting the volumetric threshold, temporal DO concentrations suggest that there may be a need to collect finer temporal resolution to determine if standards are met. HOBO® sensors at Site 1 recorded a few moments where DO fell below the criterion for >50% volume.

TREs were significantly more impactful than baseflow conditions at all three watershed monitoring sites to nutrient loading into Arcadia Lake, as alluded to above. Monthly collections at the watershed monitoring location showed significantly less variability and concentrations. Deep Fork is the main contributor to Arcadia Lake; it is the primary inflow, highest discharger, and covers the largest watershed area. The headwaters of Deep Fork subbasin encompasses a vast area of impervious surface from the Oklahoma City metro that leads to White Turkey Creek-Deep Fork subbasin. In the White

Turkey Creek-Deep Fork subbasin, both Wynn Creek and Deep Fork converge and flow directly into Site 5 of Arcadia Lake, an upstream portion of the Lake Arcadia-Deep Fork subbasin. These two tributaries carry nutrient and sediment loads that can be seen directly impacting Arcadia Lake, especially at its initial entry point at Site 5.

The phytoplankton community of a lake can act as an important indicator for water quality and their role as primary producers makes them a vital player in the overall functioning of the lake ecosystem, primarily through the cycling of nutrients. Arcadia Lake is dominated by small blue greens, namely *Synechococcus*, throughout all seasons indicating a eutrophic system unlikely to be limited by nutrients but potentially limited by light. This is indicated by the fact that the dominant taxa are especially suited to regulate buoyancy and remain in the upper layers of the photic zone where they receive adequate light regardless of high turbidity (Reynolds et al., 1987). Although cyanobacteria contribute significantly to the overall algal biovolume of Arcadia Lake, the dominant genera tend to be non-bloom-forming taxa that are unlikely to contribute to taste and odor problems. Therefore, light limitation in the lake likely prevents some nuisance blue-green taxa from forming troublesome surface blooms. However, these nuisance taxa are still present at relatively high levels during certain times. This includes genera such as *Pseudanabaena*, *Cylindrospermopsis*, *Dolichospermum*, and *Aphanizomenon* (Figure 93). *Synechococcus* are also likely an important food source for the zooplankton community, particularly non-selective filter feeders like many cladocerans, as well as phagotropic algae such as many species of cryptophytes (St. Amand, 1990). Therefore, they are a major contributor to nutrient cycling in the upper layers of the lake. Other algal taxa present at different times of year include diatoms, green algae, and cryptophytes and are ideal food sources for filter-feeding zooplankton.

Given that herbivorous zooplankton may exert significant controls over the abundance, biomass, and composition of the phytoplankton community, and as a result determine water clarity and prevalence of nuisance algal species, they may be an important consideration in lake and reservoir management strategies (OWRB, 2000). Cladocerans and copepods are the dominant taxa present in Arcadia Lake during all seasons. Cladocerans, namely *Daphnia*, have been found to have the potential to control HABs because their non-selective filter feeding on large particles can reduce the abundance of large, bloom-forming cyanobacteria (Elser et al., 2000). However, this also makes them more susceptible to the toxins those cyanobacteria may produce. Copepods on the other hand are much more selective and tend to only graze on cyanobacteria when other food sources are scarce and therefore can coexist with toxic cyanobacteria better than non-selective feeders like Daphnia (Engström-Öst et al., 2015). This could explain why copepods are the dominant zooplankton taxa in Arcadia Lake when nuisance blue-green taxa are more abundant.

Arcadia Lake is on Oklahoma's 2020 303(d) list of the Water Quality Integrated Report as impaired due to turbidity and chlorophyll *a*. Monitoring data collected for this project were analyzed for beneficial use impairments in accordance with the USAP (OAC 785:46-15) of the OWQS and Arcadia Lake was found to be not supporting its Fish and Wildlife Propagation beneficial use due to turbidity. Additionally, Arcadia Lake did not meet the 10 mg/m^3 chlorophyll *a* criterion for SWS lakes and is thereby not supporting for its Public and Private Water Supply beneficial use. Further research is needed to assess DO, utilizing data collected at finer temporal resolution for beneficial use attainment. The OWRB will be recommending that Spring Creek and Deep Fork segments be listed as nutrient threatened on the 2022 listing cycle for the States Integrated Report.

Nutrient and sediment reductions are necessary for the lake to meet these water quality standards. Data for this project highlights the need for mitigation to restore impaired beneficial uses, as well as to improve and sustain suitability of a major drinking water source.

Challenges/Issues/Recommendations

Throughout the project, challenges and issues arose alongside the unpredictability of field work. At the Spring Creek monitoring location, an additional stormwater input was observed downstream of where water collections took place. This input came from a stormwater canal conveyance at the right bank of Spring Creek that led into a stormwater drain. On multiple occasions, this stormwater drain was obstructed by trash and debris. When the stormwater drain was obstructed, the stormwater would bypass the drain and input directly into Spring Creek. The presence of this additional stormwater input would only be observed or reflected in that data during a TRE at Spring Creek.

The watershed sites were not always accessible during TREs. If the TRE occurred overnight or accompanied by lightning, discharge measurements were postponed for the safety of field staff. These TRE discharge measurements aid in the development of a rating curve to produce a continuous time-series of discharge at all points along the hydrograph for each monitoring location. The hydrological peaks of the TREs were "flashy" and quick to crest so there was a limited timeframe to catch the peak along the hydrograph. Due to these reasons, field staff was not able to be at the watershed monitoring locations during the hydrological peak of some TREs. Additional discharge measurements would refine the rating curve as more data points along the hydrograph are established. The three-year dataset of the stage and discharge measurements have been used to develop our best fit rating curve within the project timeline and conditions.

Refrigerated autosamplers were essential to collect the TREs, especially when field crews could not be at the monitoring locations during adverse weather. Issues with the autosamplers arose when cloudy conditions impeded the charging of the deep cycle batteries via the solar panels. This led to insufficient power within the autosampler, unable to pump the water to the collection bottles. When water was not collected due to technical difficulties, the equipment was reset to capture the next TRE. Other technical issues were troubleshooted including the dataloggers which were continuously recording stage data for all of the watershed monitoring locations. Occasionally the data from the continuous dataloggers did not record or download correctly, creating minor gaps in the dataset, as evidenced in Figure 97, Figure 99, and Figure 101. These issues were quickly identified, corrected and remedied in post processing. Gaps in the continuous stage dataset led to similar time gaps in mean hourly flow and mean daily flow datasets which inhibited load estimations using LOADEST. These gaps were filled using AQUARIUS® software editing tool and points were provisionally added to the dataset. During dry weather, a linear extrapolation between the bookend points on the gap was drawn. Wet weather events with gaps were filled using surrogate hydrologic responses from the same site at similar precipitation amounts during the period of record. Consequently, these edits may have led to some overestimations and underestimations of constituent loads during the stage data gaps.

Flooding also impacted lake levels and in one instance in-lake sampling was not able to take place. Record rainfall occurred in May/June 2019 and caused lake levels to be greater than 10 feet above normal elevation for more than a month. During this time there was no access to the lake with boat launches and docks being under water. For safety reasons, the City of Edmond closed the lake and barricaded ramps for lake access. Once water levels began to subside, staff contacted the city in order to open these barricades and obtain access to the in-lake monitoring sites.

Construction was observed at the watershed monitoring locations continuously throughout the duration of the project. The presence of construction was impactful on the watershed by leading to decreased riparian buffer and increased sediment into the waterbody channels. With multiple construction sites within the watershed, runoff from these sites can input chemicals, trash, debris, and other pollutants directly into the tributaries which eventually inputs into Arcadia Lake. On more than one occasion, the riparian buffer of the tributaries was altered by cutting trees and overgrown vegetation down. After the trees and vegetation was cut down, the organic debris was left behind to further alter the banks of the watershed monitoring locations. This caused for additional limbs and debris throughout the channel and around the stationary autosamplers making access to the sites difficult.

Recommendations for Arcadia Lake include additional monitoring after the implementation of the TMDL to document any improvement to water quality within the lake or watershed.

APPENDIX

Table A1. Headwaters Deep Fork subbasin land use area and percentages, 2001 and 2019.

2001 and 2019 Land Use - Headwaters Deep Fork Subbasin (111003030101)							
Land Use	Land Use ID	2001 Area (km²)	2019 Area (km²)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km²)	% Change 2001-2019
Open Water	11	0.24	0.22	0.28	0.25	-0.02	-9.19
Developed, Open	21	11.44	10.19	12.91	11.51	-1.25	-10.89
Developed, Low	22	21.79	20.99	24.60	23.69	-0.80	-3.66
Developed, Medium	23	31.52	33.14	35.58	37.41	1.62	5.14
Developed, High	24	16.71	17.75	18.86	20.04	1.04	6.24
Barren Land	31	0.00	0.02	0.00	0.02	0.01	533.33
Deciduous Forest	41	3.77	3.29	4.26	3.71	-0.48	-12.81
Evergreen Forest	42	0.44	0.42	0.50	0.47	-0.02	-5.52
Mixed Forest	43	0.03	0.03	0.04	0.03	-0.01	-17.14
Shrub/Scrub	52	0.11	0.30	0.12	0.33	0.19	167.48
Grassland	71	2.46	2.20	2.77	2.49	-0.26	-10.44
Pasture/Hay	81	0.00	0.02	0.00	0.02	0.02	950.00
Cultivated Crops	82	0.06	0.02	0.07	0.02	-0.05	-74.63
Woody Wetlands	90	0.00	0.00	0.00	0.00	0.00	0.00
Emergent Herbaceous	95	0.00	0.00	0.00	0.00	0.00	50.00

Table A2. White Turkey Creek-Deep Fork subbasin land use area and percentages, 2001 and 2019

2001 and 2019 Land Use - White Turkey Creek-Deep Fork Subbasin (111003030102)							
Land Use	Land Use ID	2001 Area (km^2)	2019 Area (km^2)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km^2)	% Change 2001-2019
Open Water	11	0.53	0.55	0.58	0.61	0.03	4.79
Developed, Open	21	10.52	10.62	11.69	11.79	0.09	0.89
Developed, Low	22	8.52	10.07	9.47	11.19	1.55	18.18
Developed, Medium	23	7.47	10.29	8.30	11.43	2.82	37.76
Developed, High	24	3.89	5.34	4.33	5.93	1.44	37.08
Barren Land	31	0.01	0.02	0.02	0.02	0.01	56.25
Deciduous Forest	41	27.98	25.24	31.08	28.05	-2.73	-9.77
Evergreen Forest	42	0.94	0.86	1.04	0.95	-0.08	-8.27
Mixed Forest	43	0.28	0.33	0.31	0.37	0.05	19.16
Shrub/Scrub	52	1.40	1.56	1.55	1.73	0.16	11.39
Grassland	71	25.88	22.63	28.76	25.15	-3.25	-12.56
Pasture/Hay	81	2.17	2.09	2.41	2.32	-0.08	-3.69
Cultivated Crops	82	0.32	0.32	0.36	0.35	-0.01	-2.50
Woody Wetlands	90	0.03	0.04	0.03	0.04	0.01	30.00
Emergent Herbaceous	95	0.07	0.06	0.07	0.06	-0.01	-16.00

Table A3. Lake Arcadia-Deep Fork subbasin land use area and percentages, 2001 and 2019

2001 and 2019 Land Use - Lake Arcadia-Deep Fork Subbasin (111003030103)							
Land Use	Land Use ID	2001 Area (km²)	2019 Area (km²)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km²)	% Change 2001-2019
Open Water	11	7.13	7.14	8.79	8.80	0.01	0.14
Developed, Open	21	9.76	9.48	12.03	11.69	-0.28	-2.84
Developed, Low	22	9.59	10.54	11.83	12.99	0.95	9.85
Developed, Medium	23	7.65	10.11	9.43	12.47	2.46	32.16
Developed, High	24	2.55	3.42	3.14	4.22	0.87	34.27
Barren Land	31	0.00	0.04	0.00	0.05	0.04	1533.33
Deciduous Forest	41	28.20	24.73	34.76	30.49	-3.47	-12.29
Evergreen Forest	42	0.23	0.13	0.28	0.16	-0.10	-43.43
Mixed Forest	43	0.27	0.26	0.34	0.32	-0.01	-3.95
Shrub/Scrub	52	0.66	1.22	0.82	1.51	0.56	84.38
Grassland	71	14.76	13.74	18.20	16.94	-1.02	-6.93
Pasture/Hay	81	0.23	0.24	0.29	0.30	0.01	5.45
Cultivated Crops	82	0.00	0.00	0.00	0.00	0.00	0.00
Woody Wetlands	90	0.02	0.02	0.02	0.02	0.00	5.26
Emergent Herbaceous	95	0.06	0.03	0.08	0.04	-0.03	-44.93

Table A4. Arcadia Lake watershed land use area and percentages, 2001 and 2019

2001 and 2019 Land Use -Arcadia Lake Watershed							
Land Use	Land Use ID	2001 Area (km²)	2019 Area (km²)	Percent of Watershed 2001	Percent of Watershed 2019	Area Change 2001-2019 (km²)	% Change 2001-2019
Open Water	11	7.90	7.91	3.04	3.05	0.01	0.16
Developed, Open	21	31.72	30.29	12.21	11.66	-1.43	-4.51
Developed, Low	22	39.90	41.60	15.36	16.02	1.70	4.25
Developed, Medium	23	46.64	53.54	17.96	20.61	6.90	14.80
Developed, High	24	23.15	26.51	8.91	10.21	3.36	14.52
Barren Land	31	0.02	0.08	0.01	0.03	0.06	322.73
Deciduous Forest	41	59.94	53.26	23.08	20.51	-6.68	-11.15
Evergreen Forest	42	1.60	1.40	0.62	0.54	-0.20	-12.47
Mixed Forest	43	0.58	0.62	0.22	0.24	0.04	6.34
Shrub/Scrub	52	2.17	3.08	0.84	1.18	0.90	41.61
Grassland	71	43.10	38.57	16.60	14.85	-4.53	-10.51
Pasture/Hay	81	2.40	2.35	0.93	0.91	-0.05	-2.10
Cultivated Crops	82	0.38	0.33	0.15	0.13	-0.05	-13.82
Woody Wetlands	90	0.05	0.06	0.02	0.02	0.01	18.87
Emergent Herbaceous	95	0.13	0.09	0.05	0.04	-0.04	-28.77

Table A5. Headwaters Deep Fork subbasin land use categories area and percentages, 2001 and 2019.

2001 and 2019 Land Use Categories - Headwaters Deep Fork Subbasin						
Land Use	2001 Area (km^2)	2019 Area (km^2)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km^2)	% Change 2001-2019
Open Water	0.24	0.22	0.28	0.25	-0.02	-9.19
Developed	81.45	82.07	91.95	92.65	0.62	0.76
Barren Land	0.00	0.02	0.00	0.02	0.01	533.33
Forest	4.24	3.73	4.79	4.21	-0.51	-12.09
Shrubland	0.11	0.30	0.12	0.33	0.19	167.48
Herbaceous	2.46	2.20	2.77	2.49	-0.26	-10.44
Planted/Cultivated	0.07	0.04	0.08	0.04	-0.03	-44.93
Wetlands	0.01	0.01	0.01	0.01	0.00	16.67

Table A6. White Turkey Creek-Deep Fork subbasin land use categories area and percentages, 2001 and 2019.

2001 and 2019 Land Use Categories - White Turkey Creek-Deep Fork Subbasin						
Land Use	2001 Area (km^2)	2019 Area (km^2)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km^2)	% Change 2001-2019
Open Water	0.53	0.55	0.58	0.61	0.03	4.79
Developed	30.40	36.31	33.78	40.34	5.91	19.42
Barren Land	0.01	0.02	0.02	0.02	0.01	56.25
Forest	29.19	26.43	32.43	29.37	-2.76	-9.45
Shrubland	1.40	1.56	1.55	1.73	0.16	11.39
Herbaceous	25.88	22.63	28.76	25.15	-3.25	-12.56
Planted/Cultivated	2.77	2.67	3.08	2.97	-0.10	-3.54
Wetlands	0.10	0.10	0.12	0.11	0.00	-2.86

Table A7. Lake Arcadia-Deep Fork subbasin land use categories area and percentages, 2001 and 2019.

2001 and 2019 Land Use Categories - Lake Arcadia-Deep Fork Subbasin						
Land Use	2001 Area (km^2)	2019 Area (km^2)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km^2)	% Change 2001-2019
Open Water	7.13	7.14	8.79	8.80	0.01	0.14
Developed	29.56	33.56	36.44	41.37	4.00	13.54
Barren Land	0.00	0.04	0.00	0.05	0.04	1533.33
Forest	28.69	25.12	35.38	30.97	-3.57	-12.46
Shrubland	0.66	1.22	0.74	1.36	0.56	84.38
Herbaceous	14.76	13.74	18.20	16.94	-1.02	-6.93
Planted/Cultivated	0.29	0.30	0.35	0.37	0.02	5.45
Wetlands	0.10	0.06	0.12	0.08	-0.03	-34.09

Table A8. Arcadia Lake watershed land use categories area and percentages, 2001 and 2019.

2001 and 2019 Land Use Categories -Arcadia Lake Watershed						
Land Use	2001 Area (km^2)	2019 Area (km^2)	Percent of Watershed 2001	Percent of Watershed 2019	Area Change 2001-2019 (km^2)	% Change 2001-2019
Open Water	7.90	7.91	3.04	3.05	0.01	0.16
Developed	141.41	151.94	54.45	58.50	10.53	7.44
Barren Land	0.02	0.08	0.01	0.03	0.06	322.73
Forest	62.13	55.28	23.92	21.29	-6.85	-11.02
Shrubland	2.17	3.08	0.84	1.18	0.90	41.61
Herbaceous	43.10	38.57	16.60	14.85	-4.53	-10.51
Planted/Cultivated	3.13	3.01	1.20	1.16	-0.11	-3.65
Wetlands	0.21	0.17	0.08	0.07	-0.04	-16.90

Table A9. Percent impervious cover, Headwaters Deep Fork subbasin, area and percentages, 2001 and 2019.

Percent Impervious Cover - Headwaters Deep Fork Subbasin						
Percent Impervious Category	2001 Area (km²)	2019 Area (km²)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km²)	% Change 2001-2019
0-10	14.59	13.18	16.47	14.88	-1.41	-9.65
11-25	6.78	6.05	7.65	6.83	-0.73	-10.78
26-60	35.24	35.16	39.79	39.69	-0.08	-0.23
61-100	31.97	34.19	36.09	38.60	2.22	6.95

Table A10. Percent impervious cover, White Turkey Creek-Deep Fork subbasin, area and percentages, 2001 and 2019.

Percent Impervious Cover - White Turkey Creek-Deep Fork Subbasin						
Percent Impervious Category	2001 Area (km²)	2019 Area (km²)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km²)	% Change 2001-2019
0-10	66.48	60.59	73.87	67.32	-5.89	-8.86
11-25	5.56	5.80	6.18	6.44	0.24	4.30
26-60	9.98	12.47	11.09	13.86	2.49	24.94
61-100	7.98	11.14	8.86	12.38	3.16	39.64

Table A11. Percent impervious cover, Lake Arcadia-Deep Fork subbasin, area and percentages, 2001 and 2019.

Percent Impervious Cover - Lake Arcadia-Deep Fork Subbasin						
Percent Impervious Category	2001 Area (km²)	2019 Area (km²)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km²)	% Change 2001-2019
0-10	58.09	53.88	71.62	66.42	-4.21	-7.25
11-25	5.26	5.19	6.49	6.40	-0.07	-1.42
26-60	11.38	13.40	14.03	16.52	2.01	17.70
61-100	6.38	8.65	7.86	10.66	2.27	35.65

Table A12. Percent impervious cover, Arcadia Lake watershed, area and percentages, 2001 and 2019.

Percent Impervious Cover - Arcadia Lake Watershed						
Percent Impervious Category	2001 Area (km²)	2019 Area (km²)	Percent of Subbasin 2001	Percent of Subbasin 2019	Area Change 2001-2019 (km²)	% Change 2001-2019
0-10	139.16	127.65	53.59	49.15	-11.51	-8.27
11-25	17.60	17.04	6.78	6.56	-0.57	-3.22
26-60	56.61	61.03	21.80	23.50	4.42	7.81
61-100	46.32	53.98	17.84	20.78	7.66	16.53

Table A13. Tree Canopy, all subbasins and Arcadia Lake watershed. (HDF - Headwaters Deep Fork, WTDF - White Turkey Creek-Deep Fork, LADF - Lake Arcadia-Deep Fork, ALW - Arcadia Lake watershed)

2016 Tree Canopy - All Three Subbasins and Watershed								
Percent Canopy Category	HDF Area (km²)	HDF Percent Area	WTDF Area (km²)	WTDF Percent Area	LADF Area (km²)	LADF Percent Area	ALW Area (km²)	ALW Percent Area
0	46.94	52.99	37.57	41.75	25.03	30.86	109.55	42.18
1-25	24.04	27.15	15.69	17.43	18.10	22.32	57.84	22.27
26-50	9.86	11.13	11.38	12.65	12.90	15.90	34.13	13.14
51-75	4.66	5.26	11.65	12.95	11.41	14.06	27.72	10.67
76-100	3.08	3.48	13.70	15.22	13.67	16.86	30.46	11.73

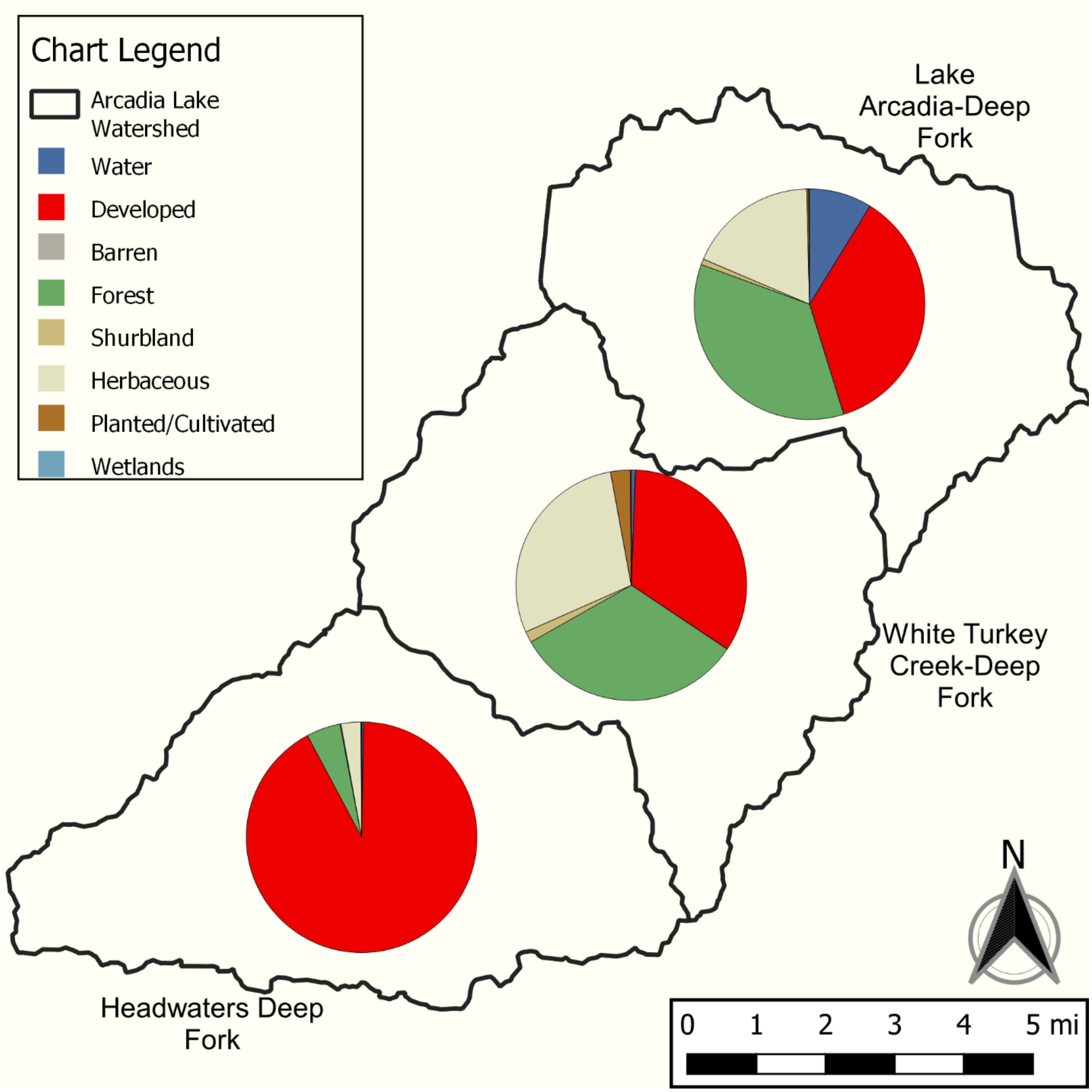


Figure A1. Land use pie charts by subbasin. See Table , Table A2, and Table A3 for more information.

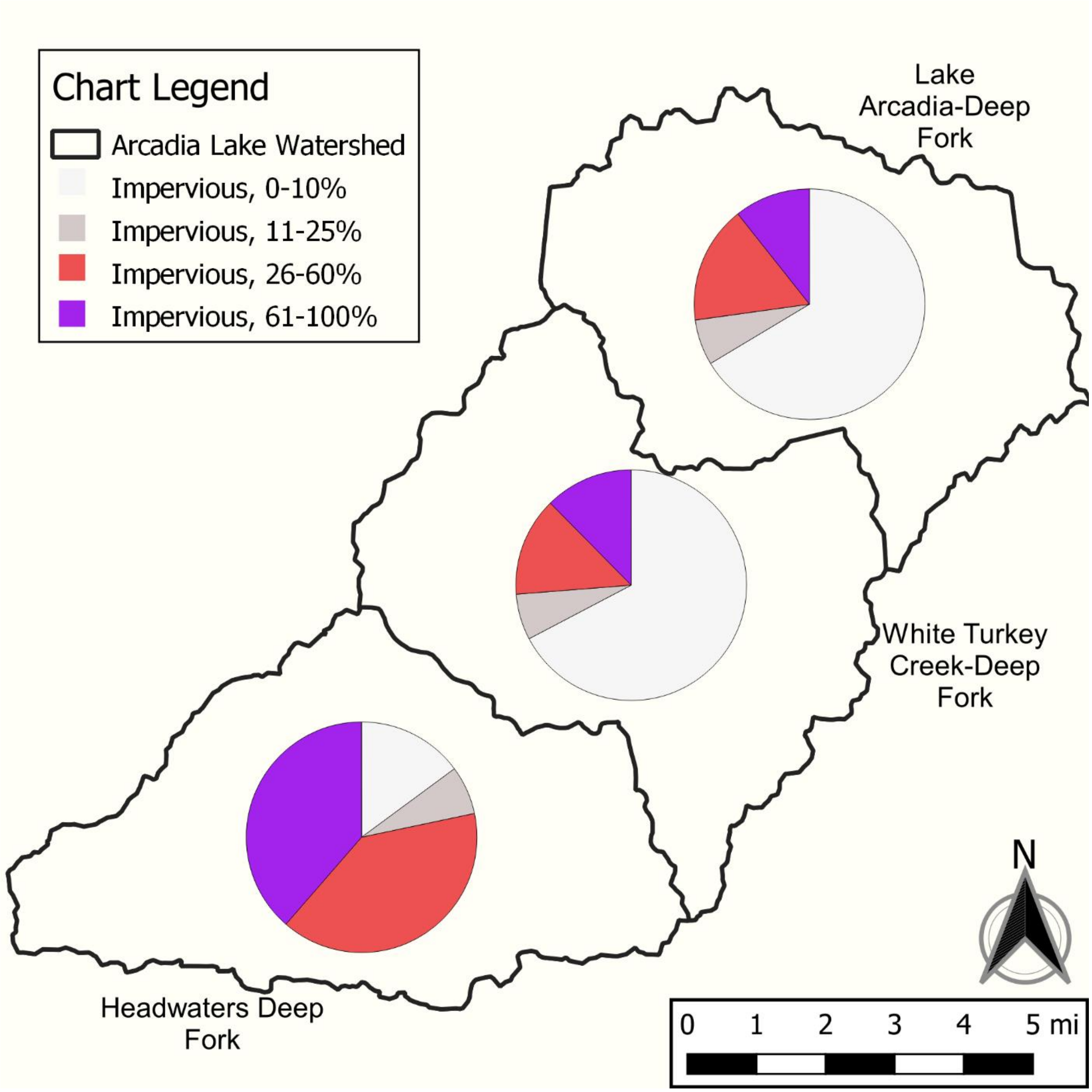


Figure A2. Impervious cover pie charts by subbasin, see Table A9 and Table A10 for more information.

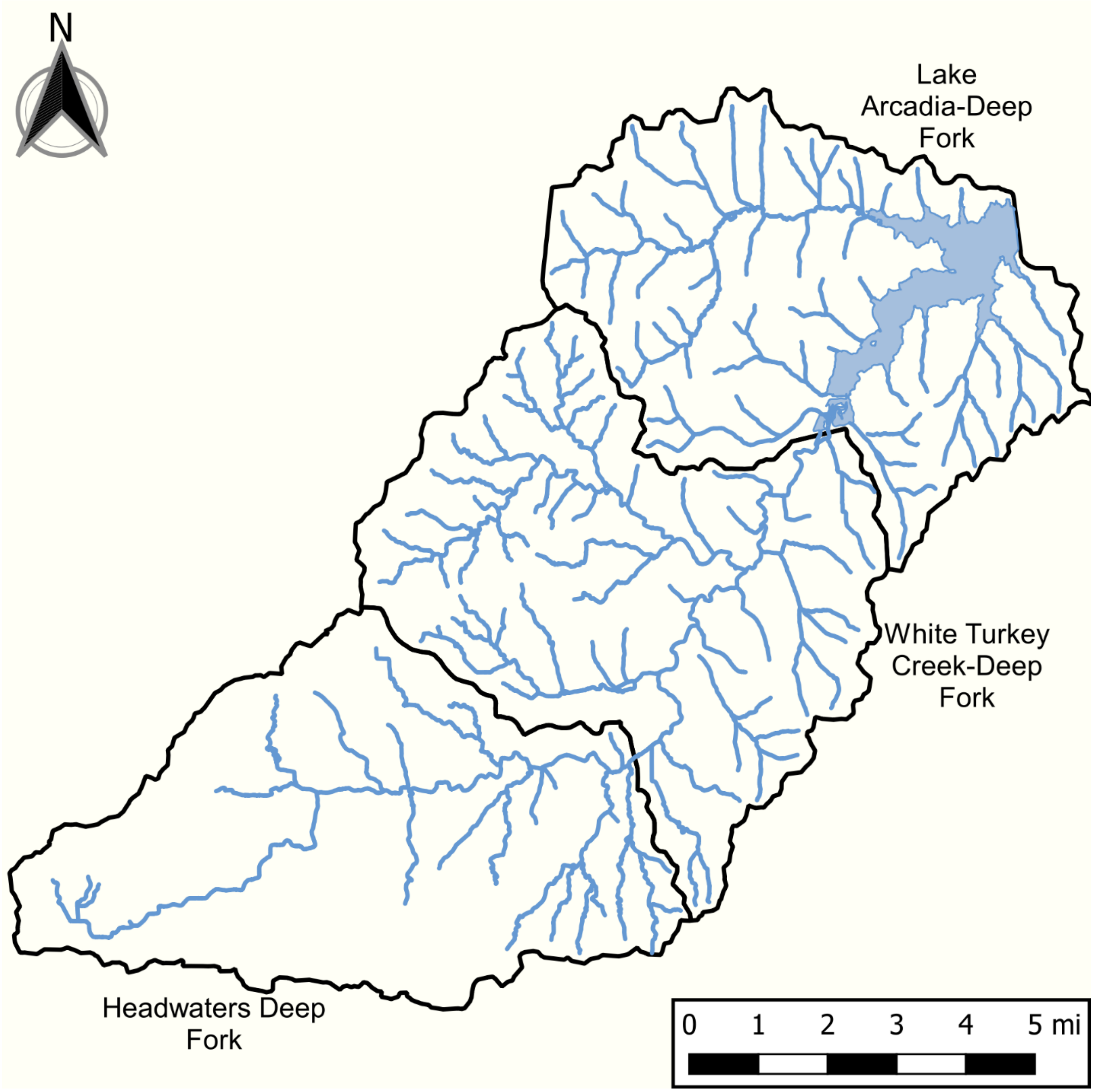


Figure A3. Rivers, streams, and Arcadia Lake in the Arcadia Lake watershed.

Table A14. Soil Complexes in Arcadia Lake watershed

Subbasin	Soil ID	Complex Name	Area (acres)	Area (km²)	Percent of Subbasin
Headwaters Deep Fork	OK094	Kirkland-Renfrow-Zenais	15103.1	61.2	68.9
	OK146	Konawa-Eufaula-Dougherty	1617.7	6.5	7.4
	OK151	Stephenville-Darnell-Newalla	5170.7	20.9	23.6
	Total		21891.5	88.6	100
White Turkey Creek-Deep Fork	OK094	Kirkland-Renfrow-Zenais	3620.3	14.7	16.3
	OK146	Stephenville-Darnell-Newalla	722.6	2.9	3.2
	OK151	Stephenville-Darnell-Newalla	17895.0	72.4	80.5
	Total		22237.9	90.0	100
Lake Arcadia-Deep Fork	OK094	Kirkland-Renfrow-Zenais	506.4	2.1	2.5
	OK151	Stephenville-Darnell-Newalla	18438.2	74.6	92.0
	OKW	Water	1101.8	4.5	5.5
	Total		20046.4	81.2	100

Table A15. USACE Rainfall

Year	Total Dam Rainfall (in)	Total Watershed Rainfall (in)	Dam Daily Maximum Rainfall (in)	Watershed Daily Maximum Daily Rainfall (in)
1995	37.56	40.61	2.86	3.25
1996	27.72	33.52	2.8	4.35
1997	30.57	36.39	3.09	2.01
1998	30.75	34.19	3.15	3.16
1999	28.18	34.02	2.83	2.51
2000	30.33	30.82	2.84	3.52
2001	28.24	27.94	2.75	2.3
2002	31.28	29.26	2.56	1.75
2003	23.72	21.72	4.77	3.26
2004	44.92	38.72	5.25	2.96
2005	27.93	26.81	1.83	1.69
2006	25.29	24.28	2.97	2.47
2007	44.15	48.29	3.29	4.12
2008	35.18	34.41	3.78	2.59
2009	40.23	36.3	3.55	2.69
2010	34.2	35.92	5.53	6.74
2011	29.11	27.37	4.35	2.7
2012	24.62	28.22	2.73	2.68
2013	33.86	45.26	4.13	6.03
2014	29.34	32.31	3.66	3.27
2015	34.35	45.74	2.48	2.82
2016	21.97	24.22	3.12	2.6
2017	22.56	31.23	2.77	2.11
2018	29.22	35.16	3.42	2.94
2019	45.63	46.75	4.98	3.82
2020	27.58	37.78	2.34	2.89
Annual Averages				
1995-2020	31.48	34.12		
2001-2019	31.88	33.68		
2018-2020	34.14	39.90		

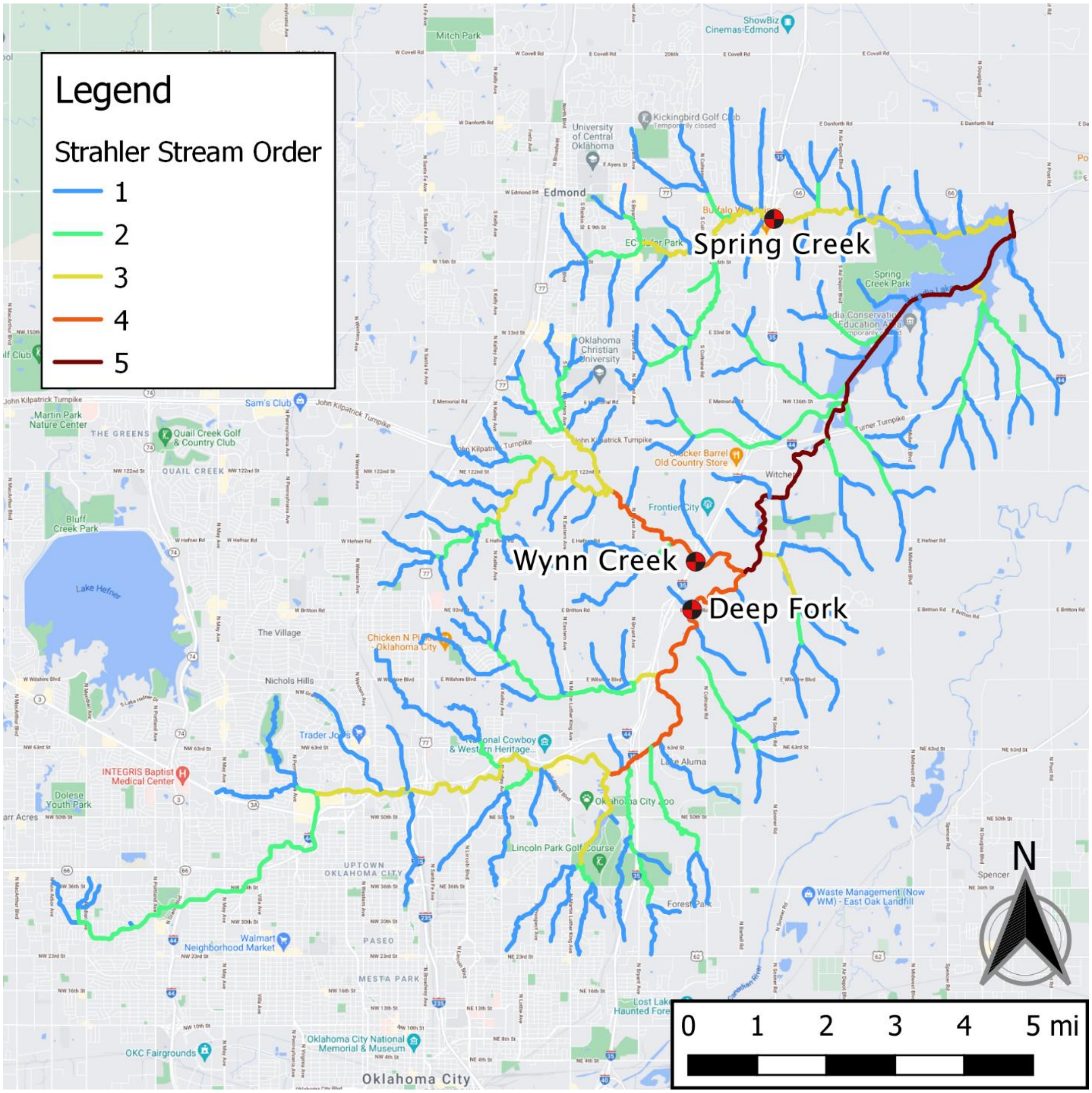


Figure A4. Strahler order for streams in the Arcadia Lake watershed.

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