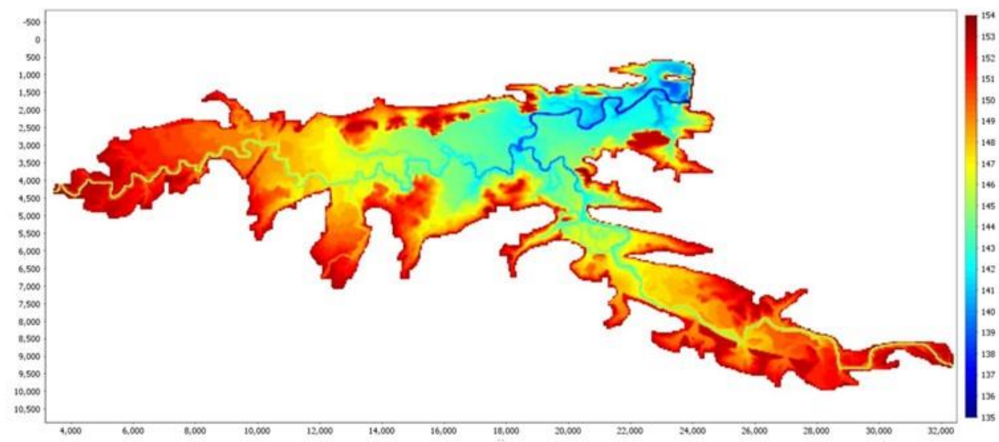
Lake Wister Water Quality Modeling in Support of Nutrient and Sediment TMDL Development



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Bio x Design

July 5, 2022

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A copy of this report can be read or downloaded from pvia.org/lake-modeling-report-final

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Executive Summary

Lake Wister is listed on the Oklahoma Department of Environmental Quality's 2020 303(d) list of Impaired Waters as impaired for beneficial uses of *Public and Private Water Supply*, *Fish Consumption*, *Warm Water Aquatic Community*, and *Aesthetics* (ODEQ 2020).

ODEQ uses the following identifying information for Lake Wister.

Waterbody ID	Waterbody Name	Waterbody Size	WB Category
OK220100020020_00	Wister Lake	7333[1]	5a

Table ES-1. Lake Wister TMDL Identifying Information (ODEQ 2020).

Cause Category	Impaired Use	Cause of Impairment	TMDL Priority	Unconfirmed Potential Sources
5a	Public and Private Water Supply	Chorophyll-a	2	140 – Source Unknown
5c	Fish Consumption	Mercury	2	140 – Source Unknown
5a	Warm Water Aquatic Community	pH	2	140 – Source Unknown
5a	Aesthetics	Phosphorus, Total	2	140 – Source Unknown
5a	Warm Water Aquatic Community	Turbidity	2	140 – Source Unknown

Table ES-2. Lake Wister Impairments (ODEQ 2020).

[1] The most recent bathymetric survey of Lake Wister found the lake size to be 6,259 acres (AquaStrategies 2018).

A Waterbody Category of 5 indicates that a water quality standard is not attained. The waterbody is impaired or threatened for one or more designated uses by a pollutant(s) and requires a TMDL. Causes of listed water quality impairments are listed in Table ES-2. Cause Category 5a indicates that development of a TMDL is underway; for Category 5c additional data and information will be collected before a TMDL is scheduled. The modeling analysis and TMDL recommendations reported here are designed to address nutrient and sediment-based impairment to Lake Wister.

Water quality modeling simulations developed for Lake Wister and reported here indicate that a *78% reduction* in the average *Total Phosphorus* (TP) load delivered to the lake will be required for the lake to meet the Oklahoma Water Quality Standard of *10 µg/L of chlorophyll-a (chl-a)*. Model simulations indicate that a *71% reduction* in the average *Total Suspended Solids* (TSS) load delivered to the lake will be required for the lake to meet the Oklahoma Water Quality Standard of *no more than 10% of samples exceeding 25 NTU turbidity*.

Table ES-3. Target load recommendations to meet water quality standards in Lake Wister.

	Average Load 2011-2015 (kg/yr)	TMDL (Annual Basis) @ a78% Reduction (kg/yr)	10% Margin of Safety (kg/yr)	Target Annual Load (kg/yr)	Target Daily Load (kg/yr)
Total Phosphorus	221,787	48,793	4,879	43,914	120
Total Suspended Solids	142,560,053	41,342,415	4,134,242	37,208,174	101,940

The Total Maximum Daily Load (TMDL) of Total Phosphorus (TP) to Lake Wister may be subdivided into a Waste Load allocation (for the load derived from point sources) and a Load Allocation (for the load derived from nonpoint sources) as shown in Table ES-2. (The current

Table ES-4. Load allocations for recommended TMDLs for Lake Wister.

	Total Phosphorus TMDL (kg/day)	% Total Phosphorus TMDL	Total Suspended Solids TMDL (kg/day)	% TSS Total Load
Waste Load Allocation	13.4	11.2	321.8	0.3
Load Allocation	94.6	78.8	91,339.5	89.7
MOS	12.0	10.0	10,184.6	10.0
Total	120.0	100.0	101,845.9	100.0

Total Suspended Solids (TSS) Waste Load allocation (WLA) from point sources comprises 0.3% of the l TSS TMDL to Lake Wister; the TSS WLA has a very small contribution to TSS TMDL). Model simulations further show that an in-lake application of alum that reduced internal phosphorus loading by 90% would reduce the required watershed load reduction to 58%.

ES.1 Background

Lake Wister is a 25.4 km^2 (6,259-acre) flood control, water supply, and recreation reservoir located in LeFlore County in eastern Oklahoma. (While ODEQ lists Wister Lake as 7,333 acres the most recent bathymetric survey of Lake Wister found the lake size to be 6,259 acres (AquaStrategies 2018). Water quality in Lake Wister does not currently meet State of Oklahoma Water Quality Standards. This report describes the estimated reduction in nutrients and sediment entering the lake that are required in order for the lake to meet those standards. The report also examines the potential for in-lake restoration actions to mitigate some of the effects of excess nutrients and sediments once they have entered the lake. The information reported here was developed through the construction and application of a detailed numerical computer model tailored to the specific conditions of Lake Wister. The model makes use of five years of both stream inflow and in-lake data collected from 2011 to 2015.

The Poteau Valley Improvement Authority (PVIA) sponsored this water quality modeling project in order to better assess and address Lake Wister's water quality impairments and what is required to improve them. The computer modeling effort provides the scientific information necessary for the establishment of one or more Total Maximum Daily Loads (TMDLs) for the lake.

A TMDL is the total amount of a pollutant that a given waterbody can receive and still meet state water quality standards). A *load* is the quantity of a given constituent delivered to a lake or stream. Loads are defined as quantities per time, for example, pounds per year or milligrams per day. The legal basis for TMDLs is found in the Federal Clean Water Act and implemented in Oklahoma by the Oklahoma Department of Environmental Quality.

Wister Lake is on the Oklahoma 303(d) list of impaired water bodies, identified as impaired for its beneficial uses of *Public and Private Water Supply*, *Fish and Wildlife Propagation*, *Fish Consumption*, and *Aesthetics*. Causes of impairment include excess chl-a, turbidity, mercury, and phosphorus (Tables ES-3).

This report describes the current water quality conditions in Lake Wister and a modeling application to derive load reduction estimates and inform the development of TMDLs for Total Phosphorus (TP) and Total Suspended Solids (TSS). Our goal was to identify specific load reduction estimates that would result in water quality in Lake Wister meeting the targets of a long-term average of 10 µg/L chl-a and less than 10% of observations exceeding 25 NTU turbidity, as specified in Oklahoma Water Quality Standards. This report does not address lake impairments due to excessive mercury.

ES.2 Lake Model

ELCOM-CAEDYM is a three-dimensional hydrodynamic and water quality model capable of simulating thermal stratification and mixing, as well as horizontal and lateral hydraulic variation, and water quality dynamics in lakes and reservoirs. The required boundary conditions for the model include lake morphometry, meteorology, surface hydrology, and water quality initial conditions. Given these inputs, the model can fully simulate water quality conditions of interest, including growth and biomass of many phytoplankton groups as well as sediment transport and resuspension. The ELCOM-CAEDYM model for Lake Wister was developed for the 2011 – 2015 calendar years, with odd number years used for model calibration activities and even number years reserved for independent verification of calibration. With very few exceptions, the model calibration met all of the required industry benchmarks and performed very well at predicting water quality conditions in Lake Wister, including the weak thermal stratification that occurs during summer months resulting in brief periods of hypoxia or anoxia in the lower water column.

ES.3 Load Reduction Estimates

Inputs to the calibrated model were modified to simulate phosphorus, nitrogen, and sediment load reductions to the lake to estimate their relative influence on in-lake water quality. Both external (i.e. watershed load reductions) and internal (i.e. in-lake sources such as sediment phosphorus and resuspended sediments) loading were evaluated as potential management options. Reduction of internal phosphorus loads alone showed a small, but significant effect on the average chl-a concentration in Lake Wister over the five-year (2011-2015) modeling period (hereafter long-term chl-a). On the other hand, reducing the external phosphorus load had a pronounced effect on long-term chl-a. Reducing nitrogen loads alone, or in combination with phosphorus had no effect on the long-term chl-a other than what was observed in the phosphorus alone reduction scenarios. A 78% reduction in TP loads from external sources will be required for the lake to meet the long-term average of 10 µg/L chl-a concentration. When combined with a 90% internal phosphorus load reduction, only a 58% reduction in external phosphorus load is required for the lake to be in compliance with the chl-a standard.

The simulated reductions of wave energy in the lake reduced the turbidity of Lake Wister, but reductions in watershed suspended sediment inputs had an even greater effect on turbidity. While the model calibration for suspended sediments met state-recommended targets, the model was better at simulating long-term averages than specific high turbidity events. Since the state standard for turbidity is (in contrast to that for chlorophyll-a) based on monitoring results from specific events, we combined an analysis of 287 specific turbidity monitoring results with modeled TSS reductions to estimate that a 71% reduction in TSS from the watershed to the lake will be required to reduce in-lake turbidity measurements to less than 10% of samples exceeding 25 NTU turbidity.

While the load reduction goals for both phosphorus and sediments are large, model results also show that incremental improvements as we progress toward them will benefit the lake. The long-term average chlorophyll-a concentration in the lake decreased by 0.12 µg/L for every 1% reduction in external phosphorus load. Long-term average turbidity in the lake decreased by 0.2 NTU for every 1% reduction in external sediment load.

Permitted point source dischargers in the Lake Wister watershed contributed an average 5,831 kg TP per year. This is approximately 2.6% (with a range from 1.3 – 5.5%) of the average 221,787 kg/yr TP load to Lake Wister. If Oklahoma major dischargers adopted and achieved a 1 mg/L TP discharge limit, the TP load to Lake Wister would decrease by an average of 1,706 kg/yr, about 1% of the current total phosphorus load. As noted above, a 1% reduction in the total phosphorus load will result in a decrease in the long-term average chlorophyll-a concentrations in the lake of 0.12 µg/L. The adoption of the 1 mg/L standard by Oklahoma major dischargers (the Wilburton discharge already achieves a TP discharge of less than 1 mg/L with a five year average of 0.73 mg/L) would decrease the TP load Lake Wister by 4.7 kg/day, or 3.9% of the of the 120 kg/day TP value.

Implementing 1 mg/L TP concentration discharge limit for Lake Wister watershed major dischargers results in a Waste Load Allocation (WLA) of 13.4 kg TP/day, 11.2% of the total load (Table ES-3).

TMDLs established for Lake Wister set target levels of key constituents required to improve water quality in the lake. The next step will be to establish how those loads will be reduced. This will be approached through the development of a watershed based plan that examines pollutant sources, locations within the watershed, available load-reducing techniques and technologies, and costs relative to effectiveness.

Section 1 – Introduction

Water quality in Lake Wister does not currently meet State of Oklahoma Water Quality Standards. This report describes the estimated reduction in nutrients and sediment entering the lake that are required in order for the lake to meet those standards. The report also examines the potential for in-lake restoration actions to mitigate some of the effects of excess nutrients and sediments once they have entered the lake. The information reported here was developed through the construction and application of a detailed numerical computer model tailored to the specific conditions of Lake Wister. The model makes use of five years of both stream inflow and in-lake data collected from 2011 to 2015.

The Poteau Valley Improvement Authority (PVIA) sponsored this water quality modeling project in order to better assess and address Lake Wister's water quality impairments and what is required to improve them. The computer modeling effort provides the scientific information necessary for the establishment of one or more Total Maximum Daily Loads (TMDLs) for the lake.

A TMDL is the total amount of a pollutant that a given waterbody can receive and still meet state water quality standards (ODEQ n.d.). A *load* is the quantity of a given constituent delivered to a lake or stream. Loads are defined as quantities per time, for example, pounds per year or milligrams per day. The legal basis for TMDLs is found in the Federal Clean Water Act (US EPA 2017) and implemented in Oklahoma by the Oklahoma Department of Environmental Quality (ODEQ n.d.).

TMDLs established for Lake Wister will set target levels of key constituents required to improve water quality in the lake. The next step will be to establish how loads of those constituents will be reduced. This will be developed through a watershed restoration planning effort. What are the sources? Where within the watershed are they located? What load-reducing techniques and technologies are available? What are their costs in relation to their effectiveness? These questions will be analyzed and addressed in a *watershed based plan* (US EPA 2008) that will be developed as the next step in the Lake Wister water quality restoration process.

The current water quality modeling project does not address all water quality concerns at Lake Wister. This project specifically focused on excessive algae and cyanobacteria in the lake and on lake turbidity; it does not address fish consumption concerns due to elevated mercury levels.

1.1 Lake Wister & Its Watershed

Figure 1-1. Lake Wister (Google Earth, imagery 3-20-2016).

Lake Wister (Figure 1-1) is a 25.4 km^2 (6,259-acre) flood control, water supply, and recreation reservoir located in LeFlore County in eastern Oklahoma (OWRB 2011a). (While ODEQ lists Wister Lake as 7,333 acres the most recent bathymetric survey of Lake Wister found the lake size to be 6,259 acres (AquaStrategies 2018). Wister Dam, constructed by the US Army Corps of Engineers in 1949, impounds the Poteau River and its tributary, Fourche Maline Creek. At its conservation pool elevation of 145.7 m (478 ft.), Lake Wister has an average depth of 2.4 m (8 ft.) resulting in the storage of approximately 62.3 x 10^6 m^3 water (50,529 acre-feet) (OWRB 2011). The surface area of the lake can increase by almost four times at maximum flood pool (Figure 1-2) resulting in a potential storage of 473 x 10^6 m^3 (383,302 acre-ft.) of water (USACE 2017) for flood control purposes.

Table 1-1. Lake Wister beneficial uses and their status (ODEQ 2020)

Beneficial Use	Status	Cause
Public and Private Water Supply	Not Supporting	Chlorophyll-a
Warm Water Aquatic Community (Fish & Wildlife Propagation)	Not Supporting	Turbidity pH
Aesthetic	Not Supporting	Total Phosphorus
Primary Body Contact Recreation	Supporting	
Fish Consumption	Not Supporting	Mercury
Agriculture	Supporting	

Lake Wister is the source of drinking water and water for commercial and industrial uses for most of LeFlore County and portions of adjacent counties. PVIA treats from Lake Wister and distributes it to sixteen cities and rural water districts in the region. The quality of the water in Lake Wister directly affects the cost and difficulty of water treatment and therefore the ability of PVIA to affordably supply safe drinking water to its customers.

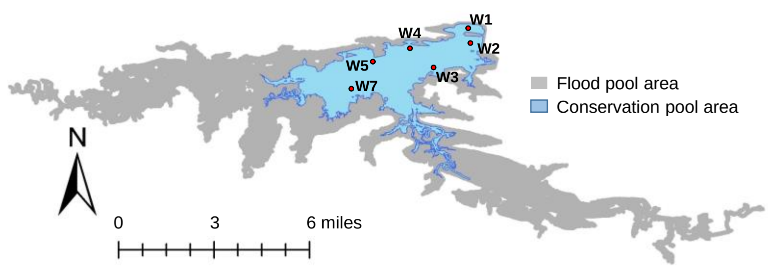


Figure 1-2. Map of the Lake Wister. Gray area represents the flood pool area and blue represents the conservation pool. The PVIA monitoring locations are also shown.

Water quality in the reservoir has deteriorated since construction and especially over the last several decades. Lake Wister is currently listed on the Oklahoma 303(d) list of impaired water bodies, identified as impaired for beneficial uses of *Public and Private Water Supply*, *Fish and Wildlife Propagation*, *Fish Consumption*, and *Aesthetics* (Table 1-1). Causes of these impairments include excessive chlorophyll-a (chl-a), turbidity, mercury, and total phosphorus (TP) (ODEQ 2014).

Certain characteristics of the lake itself can also contribute to reduced water quality. For example, the lake's primary function as flood control means that it is often flooded for extended periods of time in the spring. This kills most shoreline and shallow water vegetation, leading to bank erosion and contributing to turbidity in the lake. The shallowness of Lake Wister means that sediment from the bottom of the lake can be resuspended relatively easily by wind-driven wave action. Resuspended sediment can cause turbidity levels to increase. Some of the phosphorus that comes into the lake from the watershed settles to the lake bottom and becomes incorporated into lake sediments. Under the right conditions, this phosphorus is released from the sediments back into the water in the lake. At any given time, therefore, algae and cyanobacteria in the lake may be fertilized by both new nutrients from the watershed as well as recycled nutrients from the lake bottom.

Lake Wister is located in the Arkansas Valley Level III Ecoregion (US EPA 2013). The Poteau River rises on the slopes of the Ouachita Mountains in western Arkansas, east of the city of Waldron. It runs west to Lake Wister where it turns north and flows to its confluence with the Arkansas River at Fort Smith, Arkansas (Figure 1-3).

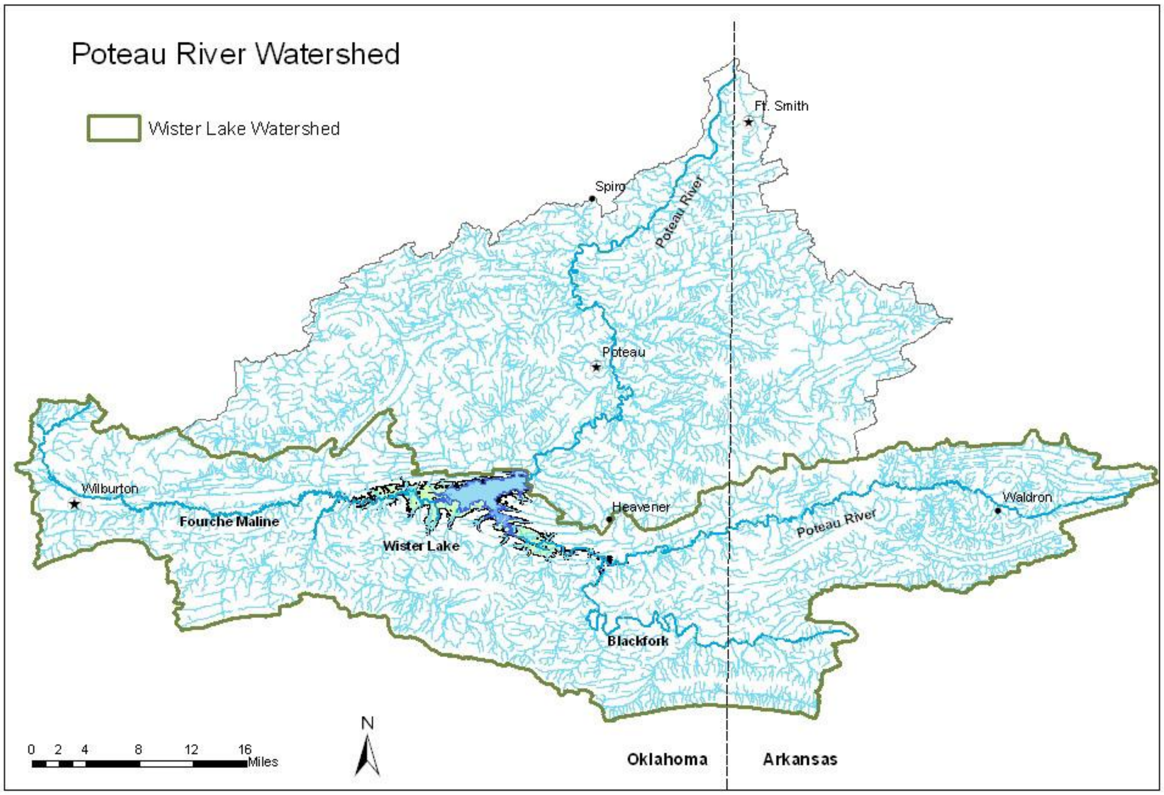


Figure 1-3. Poteau River Watershed

The second major tributary of the lake, Fourche Maline Creek, rises on the slopes of the San Bois Mountains north of Wilburton, Oklahoma. The Fourche Maline likewise descends quickly to a lower gradient and flows east to Lake Wister. Wister Dam was constructed on the Poteau River just downstream from where the Fourche and Poteau River converged.

The Poteau River watershed drains approximately 4890 sq. km (1,888 sq. miles) (Lindsay et al. 1974). Wister Lake is located approximately mid-way in the Poteau River watershed and receives water from a watershed of approximately 2572 km^2 (993 sq. miles) (USACE 2017). The watershed area to lake area ratio for Lake Wister and its watershed is 100:1. This means for every surface acre of the lake, runoff from 100 acres enters the lake. Put simply, Lake Wister receives and must process a high quantity of runoff relative to its size.

All exposed rocks in the Lake Wister watershed are sedimentary in origin—shales and sandstones predominate (Lindsay et al. 1974). Soils derived from shale weather to clays and silts. Transport of those clays by streams contributes to generally turbid conditions in Wister watershed streams and in the lake. The land cover in the Wister watershed is approximately 72% forest, 19% grassland/pasture/hay, 1.6% open water, 0.6% developed, and 6.8% other (e.g., cultivated crops, wetlands, and barren rock) (Homer et al. 2015).

1.2 Water Quality Targets

The State of Oklahoma has designated that Lake Wister has the same chlorophyll-a standard as do those listed as *Sensitive Public and Private Water Supplies* (OWRB 2016: 785:45-5-10(7)), though Wister Lake is not formally listed as a Sensitive Water Supply. In accordance with this designation, the long-term average cholorophyll-a concentration at a depth of 0.5 meters below the surface in Lake Wister shall not exceed 10µg per liter (OWRB 2016). Chlorophyll, the molecule that makes plant leaves appear green, is used as a measure of the quantity of algae and cyanobacteria in the water.

The State of Oklahoma maintains an anti-degradation policy for water quality. As found in Section 785:45-3-1 of the Oklahoma Water Quality Standards: (a) Waters of the state constitute a valuable resource and shall be protected, maintained and improved for the benefit of all the citizens. (b) It is the policy of the State of Oklahoma to protect all waters of the state from degradation of water quality, as provided in OAC 785:45-3-2 and Subchapter 13 of OAC 785:46 (OWRB 2016).

Excessive algae and cyanobacteria in the lake, driven by excessive phosphorus levels, is also the primary basis for the lake not meeting the *Aesthetics* beneficial use. There is no quantitative standard for excessive algae for aesthetic purposes. Rather, this is a narrative standard which states: “To be aesthetically enjoyable, the surface waters of the state must be free from floating materials and suspended substances that produce objectionable color and turbidity” (OWRB 2016).

In accordance with its designation as a *Warm Water Aquatic Community*, no more than 10% of samples collected from Lake Wister over a long-term sampling period (a minimum of 20 samples) should exceed 25 Nephelometric Turbidity Units (NTUs; OWRB 2016). NTUs are a measure of the cloudiness or murkiness of the water. Excessive turbidity may be caused by algal growth, suspended sediment, or a combination.

The Lake Wister watershed is also listed as a *nutrient limited watershed* in the Oklahoma Water Quality Standards (OWRB 2016). Poultry farmers in watersheds designated as nutrient limited must conduct annual soil testing to determine nutrient levels and apply litter in quantities consistent with soil nutrient levels and USDA Waste Utilization Standards (OSU 2013).

1.3 Current Water Quality Conditions

PVIA began a regular source water quality monitoring program on Lake Wister in March 2011 that has remained continuous to the present. In this program, five stations in Lake Wister (Figure 1-2) are monitored monthly for a variety of physicochemical and water chemistry variables[2]. At each station, a profile is collected from lake surface to bottom measuring water temperature, dissolved oxygen, specific conductivity, and pH. Additionally, a water sample is collected immediately below the surface and analyzed for alkalinity, ammonia-nitrogen, nitrate plus nitrite-nitrogen, total nitrogen (TN), soluble reactive phosphorus, TP, turbidity, total suspended solids (TSS), volatile suspended solids, and total organic carbon. In two locations (Quarry Island Cove and near the dam) water samples are also collected from 0.5 m above the lake bottom, and analyzed for the same suite of constituents.

The results of much of this monitoring activity are not reviewed in this report. However, most of these data were used in the calibration and validation of the model discussed in Section 2, below.

[2] In March 2015, monitoring site W4 was replaced with a new site, W7, in the routine monthly monitoring program.

Water quality conditions recorded at Lake Wister over the modeling period demonstrate that excessive chl-a and turbidity are frequent conditions at the lake, consistent with its listing in the State's water quality reporting (ODEQ 2014). Figure 1-4A shows the average chl-a concentrations across the five monitoring locations in Lake Wister from 2011-2015. The whole-lake average chl-a concentration was less than the 10 µg/L standard on only four sampling dates across the five-year monitoring period and the long-term average chl-a for this period was approximately double the state standard at 20.8 ± 11.1 µg/L.

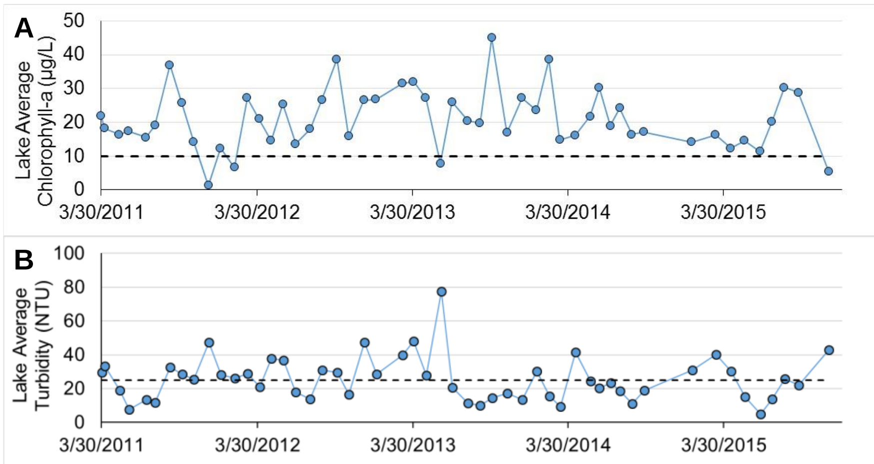


Figure 1-4. Whole lake average A) chl-a concentrations and B) turbidity from the five PVIA monitoring locations on Lake Wister from 2011-2015.

In 1974, as part of a nationwide lake assessment program, Lake Wister was sampled four times (once per quarter) in two locations (US EPA Region VI 1977). The highest chl-a measurement for any of those eight samples was 8.4 µg/L. The average for the eight samples was 4.8 µg/L. The sampling method used was different than current methods so the chl-a results are not directly comparable; nevertheless, it appears likely that in 1974 Lake Wister would have easily achieved the current "less than 10 µg/L" standard and that in the four decades-plus since this assessment, the average chl-a level in the lake has more than quadrupled.

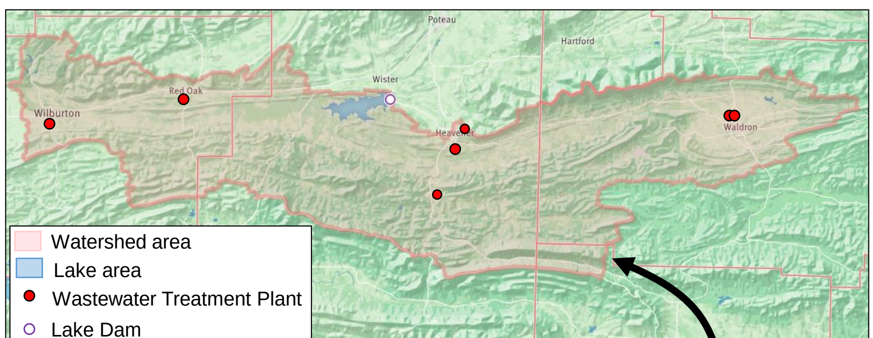
Figure 1-4B shows the average turbidity across the five monitoring locations in Lake Wister. The whole lake average turbidity frequently exceeded the 25 NTU turbidity standard. More importantly, when data were analyzed individually, 43% of individual samples (123 of 287) were in violation of the 25 NTU turbidity standard. The average turbidity of samples that were in violation of the standard was 39.2 NTU.

1.4 Pollutant Sources

Water quality in streams in the watershed and in the lake are affected by two types of sources--point and non-point. These source types are treated differently under current state and federal law.

Point sources discharges have a specific, discrete location. They typically "come out of the end of a pipe." The discharge of treated water from a city's wastewater treatment plant or from an industrial manufacturing plant are typical point sources. Point source discharges must have a permit. They have discharge limits established for constituents of concern and approved treatment technologies designed to achieve those limits. The Oklahoma Department of Environmental Quality (ODEQ) develops and administers point source discharge permits in Oklahoma; the Arkansas Department of Environmental Quality (ADEQ) does so in Arkansas.

There are seven currently permitted point sources in the Lake Wister watershed. Five wastewater discharges are located in the Poteau River arm of the Lake Wister drainage (two in Arkansas and three in Oklahoma) and two are located in the Fourche-Maline Creek watershed area (Figure 1-5).



NPDES	Short ID	Full Name
OK0038407	Heavener	City of Heavener, OK
OK0022951	Hamilton	Hamilton Correctional Center
OK0031828	Cedar	USDA Forest Service Cedar Lake Rec. Area
OK0021881	Wilburton	Wilburton Public Works Authority
OK0031631	Red Oak	Red Oak Public Works Authority
AR0038482	Tyson	Tyson Foods INC., Waldron, AR
AR0035769	Waldron	City of Waldron, AR

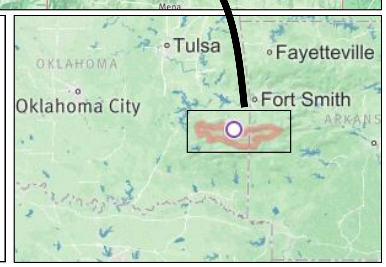


Figure 1-5. Locations and identifications of permitted wastewater treatment facilities in the Lake Wister Watershed.

Nonpoint sources are diffuse, they can't be tracked to a single pipe or point. Nonpoint source pollution is caused by rainfall or snowmelt moving over and through the ground. As it moves, the surface runoff or groundwater flow picks up and carries with it materials and chemicals it encounters, and deposits them into lakes, rivers, wetlands, or other waters (OCC 2014). In Oklahoma, the Oklahoma Conservation Commission oversees efforts to reduce nonpoint source pollution (OCC 2014). Because they are diffuse and widespread, nonpoint sources of pollution are often challenging to identify and reduce.

Nonpoint sources are the primary sources of nutrients and sediments that enter Lake Wister. Among the important nonpoint sources are animal manures and soil erosion. In 2009, there were 451 poultry houses in the Lake Wister watershed—220 in Oklahoma and 231 in Arkansas (PVIA 2009; based on a Google Earth aerial photo analysis). The number of houses may have declined since that time, as some producers have gone out of business (J. Britton, pers. com.). There were also some 33,000 head of cattle in LeFlore County during the same period (NASS & ODA 2016). Most of the chicken litter created by broiler production is applied to pasture for cattle production, although in recent years an increasing amount is being sold and exported from the watershed. A portion of applied litter runs off the land and into streams and the lake. Phosphorus that does not run off remains in the soil, bound to soil particles and then moves with the soil when it is eroded.

Table 1-2. Poultry litter applied, LeFlore County, OK (OCC 2002-2015 and ODAFF 2016-2017).

Year	Poultry Litter Produced (Tons)	Poultry Litter Applied (Tons)	Phosphorus Applied* (Pounds)	Phosphorus Applied* (kg)
2001	58,469	57,278	1,718,340	781,064
2002	37,592	36,990	1,109,700	504,409
2003	31,998	32,207	966,210	439,186
2004	38,295	36,314	1,089,420	495,191
2005	46,714	42,419	1,272,570	578,441
2006	49,552	36,575	1,097,250	498,750
2007	46,512	53,824	1,614,720	733,964
2008	40,001	30,382	911,460	414,300
2009	45,270	32,025	960,750	436,705
2010	36,492	26,780	803,400	365,182
2011	34,030	23,714	711,420	323,373
2012	36,696	21,589	647,670	294,395
2013	78,767	9,596	287,880	130,855
2014	28,459	13,928	417,840	189,927
2015	45,244	16,150	484,500	220,227
2016	50,738	9,766	292,980	133,173
Total			14,386,110	6,539,141

As noted earlier, the Lake Wister watershed is listed in Oklahoma as a *nutrient limited watershed* (meaning that nutrients are in excess and limit water quality). This means that poultry farmers in the watershed must conduct annual soil testing to determine nutrient levels and apply litter in quantities consistent with soil nutrient levels and USDA Waste Utilization Standards, which are generally lower levels than what may be applied outside the watershed (OSU 2013).

Neither Oklahoma nor Arkansas tracks chicken houses or litter production by watershed. Since 2001 Oklahoma has tracked poultry litter production by county. In 2015-2016 LeFlore County is reported to have had 367 broiler houses with a licensed capacity 9,355,531 birds (ODAFF 2017). The 367 houses in the county is 84 less than the 451 estimated for the Wister watershed (including Arkansas). Therefore, the LeFlore County numbers can be used to give an approximation (but only a rough approximation) of the phosphorus application rate in the Wister watershed. Between 2001 and 2016, 14.3 million pounds of phosphorus in the form of chicken litter was applied to pastures in LeFlore County (Table 1-2).

Soil erosion is a second significant source of phosphorus as well as sediment to the lake. Soil erosion comes from overgrazed pastures, from unpaved roads, and from eroding streambanks. Many of the roads in the Lake Wister watershed are unpaved. This includes county roads, private drives, forest roads, and the roads cut to provide access to gas wells. The number of such roads and the quantity of soil erosion they produce has not been quantified. However, soil erosion and phosphorus supply from unpaved roads can be significant. For example, in one summary Turton (2011) estimated that sediment from unpaved rural roads may be responsible for about 25% of the sediment entering streams in north-central Oklahoma. Measured erosion rates ranged from 7 tons per mile to over 400. Eroded soil entering watershed streams and eventually the lake contributes to the lake turbidity, to loss of water supply capacity, and carries phosphorus along with it as it moves.

Nor have Wister watershed streams been studied to quantify streambank erosion rates or associated phosphorus load production. However, soil erosion and phosphorus supply from eroding stream banks can be significant. For example, one study in the erosion-prone soils of the

Barren Creek watershed in northeastern Oklahoma found that 55km (34 miles) of stream produced 93,000 kg (102 tons) of TP per year (Miller et al. 2014).

1.5 Existing TMDLs

There are several existing TMDLs in effect in the Lake Wister watershed. There are TMDLs for phosphorus, copper, and zinc for a segment of the Poteau River in Arkansas (Reach 11110105-031L) upstream from the state line and downstream from the City of Waldron (FTN 2006). The phosphorus TMDL established a phosphorus discharge limit of 1.0 mg/L TP for the City of Waldron and 1.5 mg/L for the Tyson Waldron production plant. There is also a TMDL for bacteria for a segment of the Fourche Maline upstream from the lake (Parsons 2008).

Section 2 – Lake Model Development, Calibration, and Validation

The current project began in 2013 when PVIA contracted with the University of Arkansas to develop a lake modeling plan for Lake Wister. The goal was to identify a modeling platform that could be used to address water quality concerns and support the development of one or more TMDLs for the lake (Scott and Patterson 2014). A number of multi-dimensional models were evaluated for their potential utility in modeling water quality in Lake Wister. The final report recommended the use of the Estuarine, Lake, and Coastal Ocean Model (ELCOM) in conjunction with the Computational Aquatic Ecosystem Dynamic Model (CAEDYM). Coupling of these models allows for three-dimensional simulation by linking ELCOM's 3D hydrodynamics with the water quality and biological components of CAEDYM. Among the reasons for choosing ELCOM-CAEDYM was the way that the model handled nutrient cycling from lake sediments, considered to be particularly important to understanding Lake Wister's water quality processes.

PVIA subsequently contracted with the University of Arkansas in 2014 to develop, calibrate, validate, and simulate water quality in Lake Wister under a variety of management conditions. Modeling work was largely completed at the University of Arkansas by June 2016 with the exception of final calibration routines. The model and associated data were transferred at that time to Baylor University and PVIA subsequently contracted with Baylor to complete modeling activities.

2.1 ELCOM-CAEDYM Model Description

ELCOM and CAEDYM were developed by the Centre for Water Research at the University of Western Australia (Hodges and Dallimore 2013, Hipsey et al. 2013). ELCOM is a three-dimensional hydrodynamic model intended to simulate thermal stratification and mixing, as well as, horizontal and lateral hydraulic variation. CAEDYM is a biogeochemical and food-web model capable of simulating water quality conditions in lakes and estuaries. ELCOM-CAEDYM has been widely used for estimating load reductions necessary to support water quality goals (Burger et al. 2008; Trolle et al. 2010). State variables utilized in ELCOM-CAEDYM include water temperature, dissolved oxygen, suspended solids, nitrogen (NH_4-N, NO_2+NO_3-N, particulate organic N, dissolved organic N, and TN), phosphorus (soluble reactive P, particulate organic P, dissolved organic P, and TP), organic carbon, and multiple phytoplankton groups. The model predicts advective and diffusive factors influencing the biogeochemical cycling of elements and the biological responses to these hydrodynamic and chemical variations (Figure 2-1). The ELCOM-CAEDYM model requires substantial input data that includes lake morphometry, time varying meteorology, and time varying inflow and outflows water volumes and water constituent concentrations.

2.2 Data Sources

Data sources used for model boundary conditions included: 1) lake morphometry data provided by the Oklahoma Water Resources Board (OWRB), 2) meteorological data generated by the Oklahoma MESONET network, 3) hydrologic inputs from US Geological Survey (USGS) stream gauges on the Poteau River arm and the Fourche-Maline Creek arm, 4) Hydrologic outputs and withdrawals from the US Army Corps of Engineers (USACE), 5) water quality across baseflow and stormflow conditions at river gauges measured by USGS, and 6) estimated initial conditions for Lake Wister from water quality monitoring data collected from 2011-2015 by PVIA. Additionally, the PVIA water quality monitoring dataset was used for model calibration and independent verification.

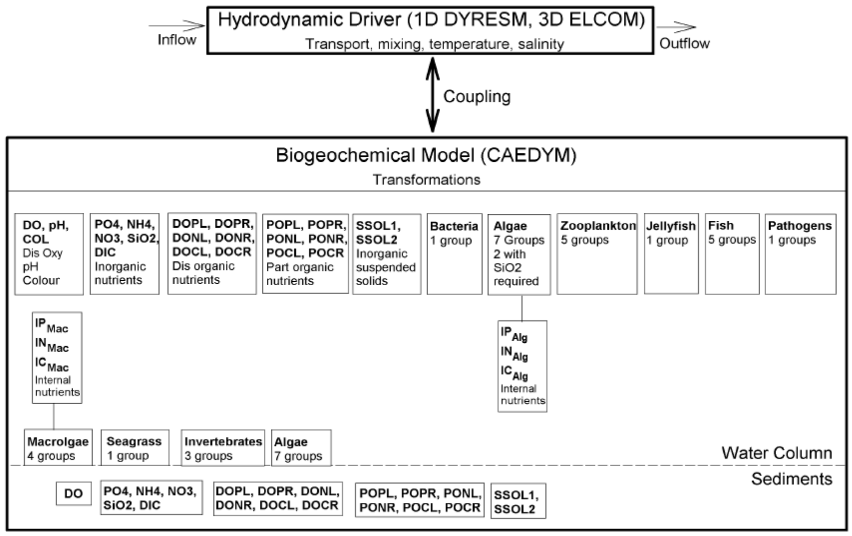


Figure 2-1. Schematic representation of the ELCOM-CAEDYM model and its state variables (Hipsey et al. 2013)

2.3 Boundary Conditions

2.3.1 Lake Morphometry

2.3.1 Lake Morphometry – One of the most significant challenges to modeling Lake Wister is that water storage can increase by 7x and surface area by 4x when the reservoir changes from conservation pool storage to maximum flood storage. Thus, the bathymetry of the lake generated from data produced by the OWRB had to be merged with a high-resolution digital elevation model generated from lidar imagery. The resulting floodpool boundary (Figure 1-2) allowed us to derive a seamless bathymetric model for the complete pool conditions on Lake Wister (Figure 2-2). CAEDYM requires the use of equal grid sizes in the *x* and *y* directions, but permits variable thicknesses in the *z* direction. We chose a 50 m x 50 m x/y grid and used 0.5 m resolution to delineate vertical thickness. Each grid cell covers 2,500 square meters, or a little over 0.6 of an acre.

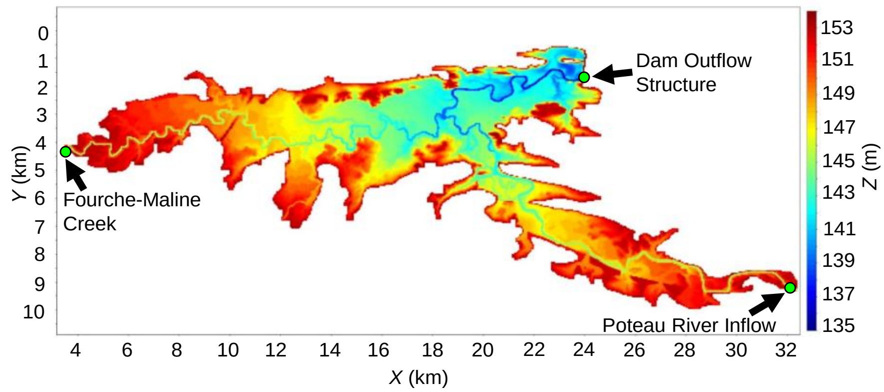


Figure 2-2. Lake Wister input bathymetry. Grid size was 50 m by 50 m in the X and Y directions. Vertical layers were 1 m thick.

Lake model cells receiving water inflows were located at the two major river inflows, Fourche-Maline Creek and the Poteau River. The location of monitoring sites on these two rivers accounts for approximately 84% of the drainage area contributing to Lake Wister (USGS 2014).Quantities of these inflows were adjusted to account for the drainage from the remainder of the watershed draining directly to the lake. The lake outlet was located at the USACE dam gate structures.

2.3.2 Meteorological Data – Meteorological data including air temperature, irradiance, relative humidity, and wind speed and direction are required boundary condition inputs for the ELCOM-CAEDYM model. We utilized data from the Oklahoma Mesonet network to derive these inputs for Lake Wister from 2011-2015. The vast majority of meteorological input were derived from the Oklahoma Mesonet site in Wister, Oklahoma. However, some minor gaps existed in this data set for the modeling period and were filled with data from nearby Mesonet stations in Salisaw and Talihina, Oklahoma.

Figure 2-3 shows the meteorological input data used for the Lake Wister ELCOM-CAEDYM model. The daily and seasonal periodicity in both air temperature and solar irradiance are obvious across the period of record for both variables. Similarly, data variability in wind speed over the lake is also quite dramatic, regularly exceeding 10 m/s (> 20 mph).

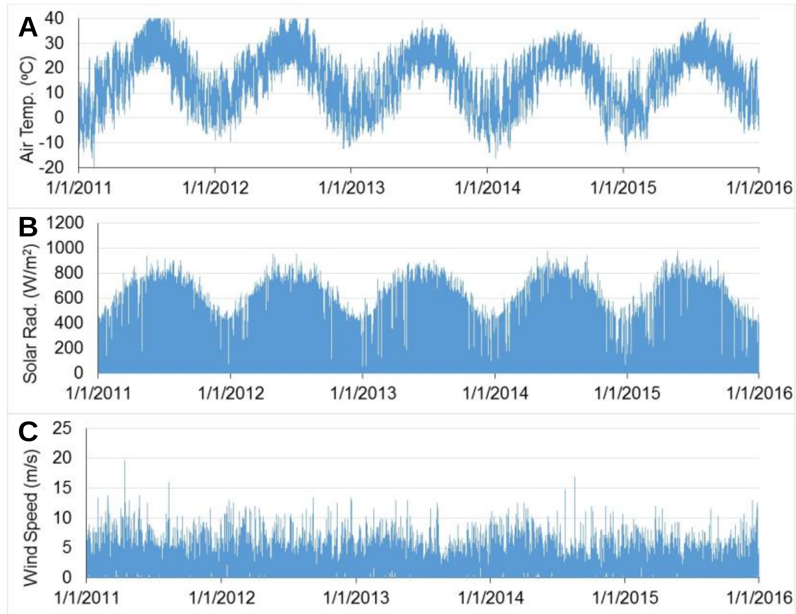


Figure 2-3. Meteorological conditions at Lake Wister measured by Oklahoma Mesonet 2011-2015: A) Air temperature, B) Total Solar Irradiance, and C) Wind Speed.

2.3.3 Hydrologic Inflows – PVIA began funding the USGS to collect baseflow and stormflow samples at the two primary river inflows, Fourche-Maline Creek near Leflore, Oklahoma (07247650) and Poteau River near Heavener, Oklahoma (07247350) in the fall of 2010. These sampling locations were chosen to capture as much of watershed flows as possible, while minimizing chances of lake backwater effects. Although neither of these sampling locations have a continuous streamflow monitoring gauge, USGS stations upstream (07247500, 07247015, and 07247250) have continuous streamflow monitoring that permits flow estimation at the two sample locations (USGS 2014). Thus, flow was reported by USGS by estimating flow using the drainage area ratio method. In this method, flow at ungauged stations on rivers with upstream or downstream gauges is estimated by:

$$Q_s = \left[DA_s/DA_g\right] * Q_g$$

where Q_s is the daily mean streamflow at the ungauged station, Q_g is the daily mean streamflow at the gauged station, DA_s is the drainage area contributing to flow at the ungauged location, and DA_g is the drainage area contributing to flow at the gauged location. Estimated inflows were available in 15-minute intervals for the Fourche-Maline

Creek location and in 30-minute intervals for the Poteau River location over the 2011-2015 period of record. Average flow in the Fourche-Maline Creek was 12.5 ± 35.4 m^3/s (441 ± 1,250 cfs) from 2011-2015 with typical peakflow reaching 200 m^3/s (7,062 cfs) (Figure 2-4A). Average flow in the Poteau River was 35.4 ± 124 m^3/s (1,250 ± 4,378 cfs)

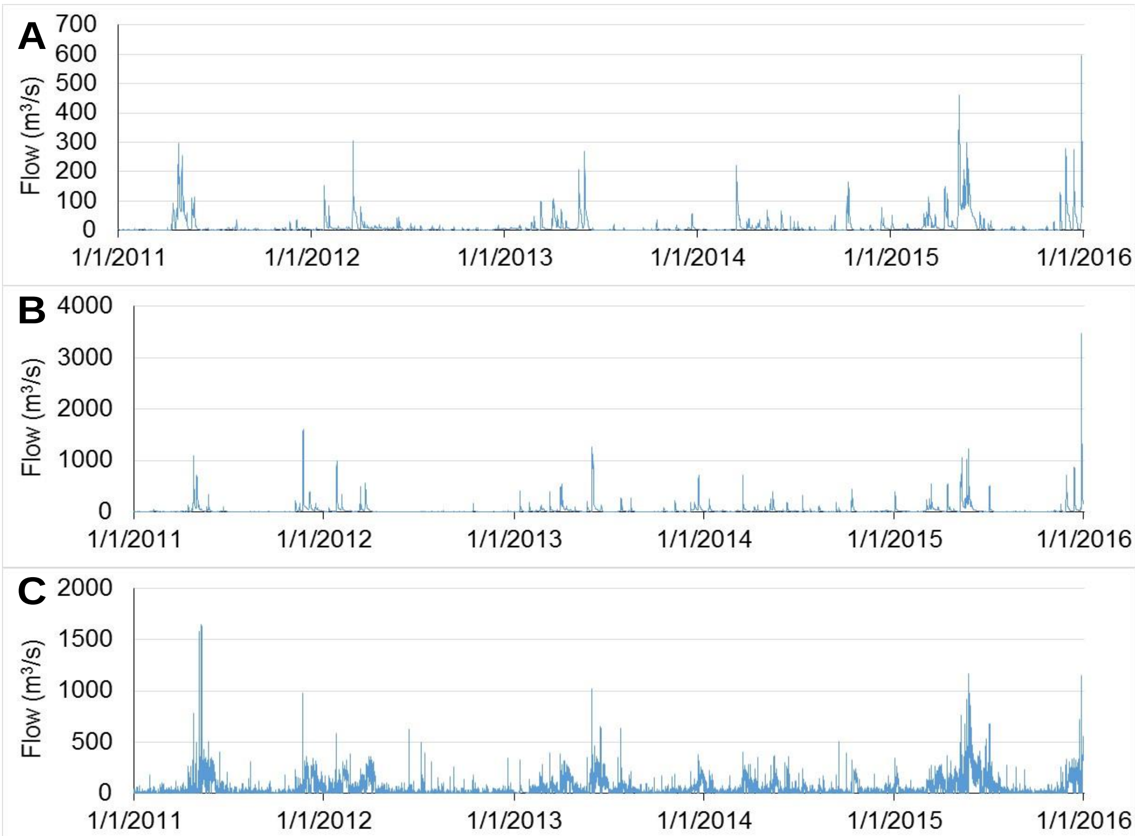


Figure 2-4. Inflows and outflow to Lake Wister: A) Inflow from Fourche-Maline Creek, B) Inflow from Poteau River, and C) Outflow from dam.

from 2011-2015, but typical peakflows reached 1,000 m^3/s (35,310 cfs) (Figure 2-4B).

2.3.4 Lake Withdrawals and Outflows – The USACE generates continuous pool elevation and outflow discharge data for Lake Wister at the dam. Average outflow at the dam was 43.4 ± 77.7 m^3/s (1,532 ± 2,744 cfs) from 2011-2015, and typical peak discharge rarely exceeded 500 m^3/s (17,655 cfs) (Figure 2-4C). A few lake withdrawals for water supply exist. The average PVIA withdrawal for water supply purposes is approximately 0.24

m^3/s (8.5 cfs)[3], which we considered insignificant to the water balance structure for the model. Therefore, the PVIA withdrawal was not simulated. The City of Heavener and the Oklahoma Department of Tourism also hold minor water rights in the lake. The quantities of these potential withdrawals are much smaller than the PVIA usage; they too were not simulated.

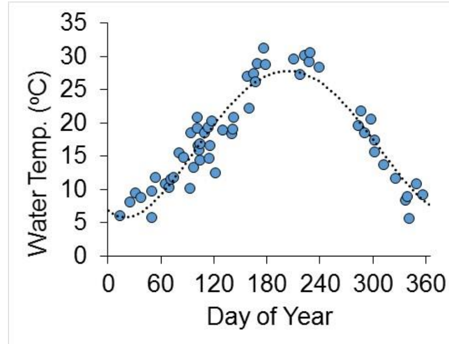


Figure 2-5. Inflow temperature modeled by day of year

2.3.5 Water Quality of River Inflows – The USGS collected water quality samples during baseflow or stormflow conditions 12 times per year from 2011-2015 (USGS 2014). During sampling, field data were generated for water temperature, dissolved oxygen, pH, and conductivity. Additionally, samples were collected for NH_4-N, NO_2+NO_3-N, TN, SRP, TP, and TSS.

A continuous data set on river inflow temperatures was needed for boundary conditions in the ELCOM-CAEDYM model. Initially, we utilized water temperature data collected by USGS to derive a seasonal model of water temperature using a polynomial regression with both the Fourche-Maline Creek and Poteau River. The regression model is shown in Figure 2-5. However, preliminary calibration activities indicated that this seasonally-based model was not sensitive to the cold water conditions that exist during winter and spring inflow events. Although the regression model of the seasonal model was strong (R^2 = 0.83) the initial ELCOM-CAEDYM simulations conducted with these inflow temperatures never achieved thermal stratification and never had hypolimnetic anoxia, both of which are common every year in Lake Wister. We evaluated several potential options for estimating river inflow temperatures including flow-dependent corrections from continuous air temperature data. In the end, the most useful dataset for calibrating water temperature in the lake was the 4-day average air temperature. Therefore, the final

[3] PVIA's annual water usage is approximately two billion gallons or 6,138 acre-feet (7,571,100 m^3). Average daily usage is approximately 5.5 million gallons/day (20,820 m^3/day).

calibrated model used 4-day average air temperature to compute the water temperature of the Fourche-Maline Creek and Poteau River for the inflow boundary conditions (Figure 2-6).

We used estimated flow data along with measured TN, TP, and TSS concentrations to develop models for flow-dependent concentrations for these variables in both the Fourche-Maline Creek and Poteau River. However, we also expected seasonal variation to drive variability in these water quality constituents. Thus, we evaluated regression models for each water quality variable that included flow only as the independent variable or a combination of flow and sample day of year (DOY) as a proxy for seasonal influence on water quality using a multiple sin structure (Hirsch et al. 1993). All flow and

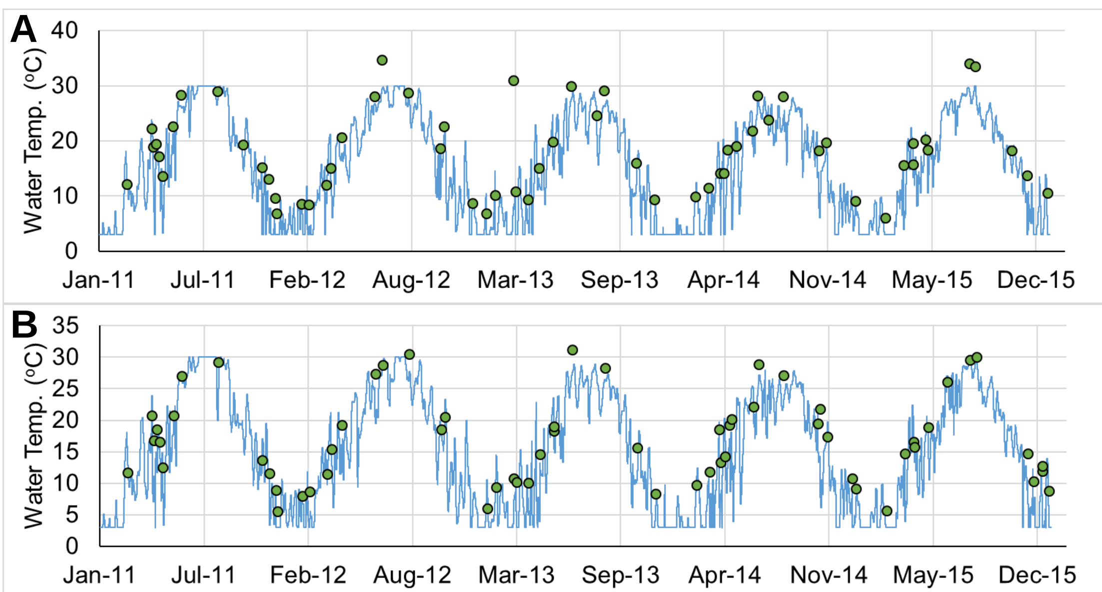


Figure 2-6. Inflow temperature boundary conditions for A) Poteau River, and B) Fourche-Maline Creek. Solid lines represent continuous estimates based on air temperature and day of year. Green circles represent measured data collected by USGS.

water quality data were natural log transformed prior to regression modeling. The two possible model structures for use in TN, TP, and TSS input models were:

$$\ln C = a + b1 \ln Q$$

or

$$\ln C = a + b1 \ln Q + b2 \ \sin 2\pi \ DOY + \ b3 \ \cos 2\pi \ DOY$$

where *C* is the concentration of TN, TP, or TSS, *a*, *b1*, *b2*, and *b3* are regression coefficients, *Q* is flow in m^3/s, and *DOY* is the day of the year. We evaluated both models for each water quality variable and chose the model with the largest coefficient of determination (R^2) for use in prediction. These regression models allowed us to compute daily TN, TP, and TSS load estimates from the Fourche-Maline Creek and the Poteau River and are common in applications of river loading estimations into lakes (Grantz et al. 2014). The regression models allowed us to have a continuous input of these variables available for use as the ELCOM-CAEDYM boundary conditions.

Table 2-1. Regression model statistics for nutrient and sediment rating curves derived from USGS data for the Fourche-Maline Creek and Poteau River.

Variable		Model Structure	a ± SE	b1 ± SE	b2 ± SE	b3 ± SE	*F* (*p*)	R^2
Fourche-Maline Creek								
	TN	Flow	-0.39 ± 0.06	0.13 ± 0.02	--	--	44.7 (<0.001)	0.48
	TP	Flow	-2.58 ± 0.09	0.21 ± 0.03	--	--	56.6 (<0.001)	0.53
	TSS	Flow	3.36 ± 0.13	0.36 ± 0.04	--	--	69.1 (<0.001)	0.62
Poteau River								
	TN	Flow	-0.53 ± 0.08	0.10 ± 0.02	--	--	30.2 (<0.001)	0.37
	TP	Flow	-2.92 ± 0.11	0.23 ± 0.03	--	--	66.5 (<0.001)	0.57
	TSS	Flow + DOY	-3.01 ± 2.78	0.41 ± 0.05	-3.37 ± 1.69	4.70 ± 2.41	23.4 (<0.001)	0.62

The final regression models for TN, TP, and TSS, including the individual parameter estimates and whole-model statistics are shown in Table 2-1. A flow only model was the strongest predictor of water quality conditions for all water quality variables except TSS in the Poteau River, for which the strongest model included flow and DOY. All models were highly statistically significant (i.e. $p < 0.001$) and the coefficient of determination indicated that flow (or flow and DOY in combination for TSS in Poteau River) explained between 37 to 62 % of the variation in water quality in both rivers. From these regression models, we computed the TN, TP and TSS concentrations for all 15- or 30-minute intervals to match the flow data for the Fourche-Maline Creek and Poteau River, respectively. From these estimates, we integrated the flow and constituent concentration for each interval to compute a mass of TN, TP, or TSS per interval. These intervals were then summed to derive an annual mass loading (kg/year) of TN, TP, or TSS (Table 2-2).

Table 2-2. Annual nitrogen, phosphorus and sediment loads delivered to Lake Wister from the Fourche-Maline Creek and Poteau River.

	TN (kg)	TP (kg)	TSS (kg)	TN (kg)	TP (kg)	TSS (kg)
	Fourche-Maline Creek			Poteau River		
2011	257,795	41,841	31,110,621	863,156	157,207	100,634,398
2012	182,201	28,356	19,757,036	415,340	68,923	31,198,168
2013	270,669	41,587	28,160,315	993,123	167,800	100,667,981
2014	188,233	28,354	18,655,800	533,508	82,056	44,057,691
2015	938,920	158,049	124,827,717	1,773,789	334,764	213,730,539
TOTAL	**1,837,818**	**298,187**	**222,511,488**	**4,578,916**	**810,750**	**490,288,777**

The USGS also calculated loading rates for TSS, TP, and TN to Lake Wister, but data were only available from 2011 – 2013. We compared our mass loading estimates to the available USGS loading estimates. Although some specific within-year variation did exist among our regression model loading estimates and USGS estimates, the models were comparable in ranges as exemplified by the scatterplot (Figure 2-7) between our estimates and USGS estimates for the 2011, 2012, and 2013 model years. Because the axes of this graph are both log-transformed, a power function provided the best linear fit to the data where the slope of the line is represented in the exponent of the equation. The exponent of the best fit line was 1.03 with an R^2 of 0.97, indicating a strong correlation between our loading estimates and those from the USGS.

Although water quality loading models are useful for computing loads of total nutrients and sediment to Lake Wister, these variables are not specifically useful as inputs for ELCOM-CAEDYM. Instead, the lake model requires inputs that are more specific to the physical and chemical form of the material entering the lake. Thus, instead of TN, the model requires flow-corrected continuous concentration data for NH_4-N, NO_2+NO_3-N, particulate organic N (PON), and dissolved organic N (DON). Instead of TP, the model requires flow-corrected continuous concentrations for soluble reactive P (SRP), particulate organic P (POP), dissolved organic P (DOP), and particulate inorganic P (PIP).

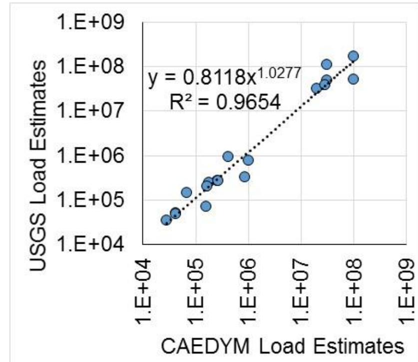


Figure 2-7. Relationship between nutrient and sediment loading estimates derived in this study (CAEDYM Load Estimates) and nutrient and sediment loading estimates derived by USGS (2014) on the same data sets.

NH_4-N, NO_2+NO_3-N, and SRP were measured directly at USGS gauging stations so that a direct flow x concentration regression analysis could be used to derive their continuous input concentrations. Variables for which measured data were not directly available were

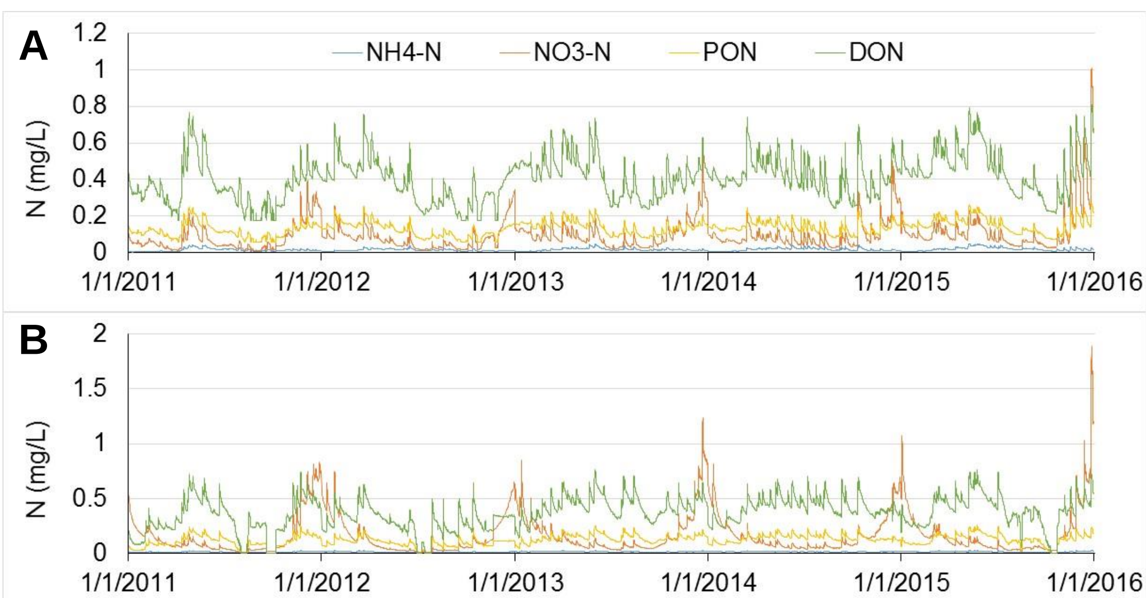


Figure 2-8. Nitrogen concentrations in inflows to Lake Wister from A) Fourche-Maline Creek and B) Poteau River.

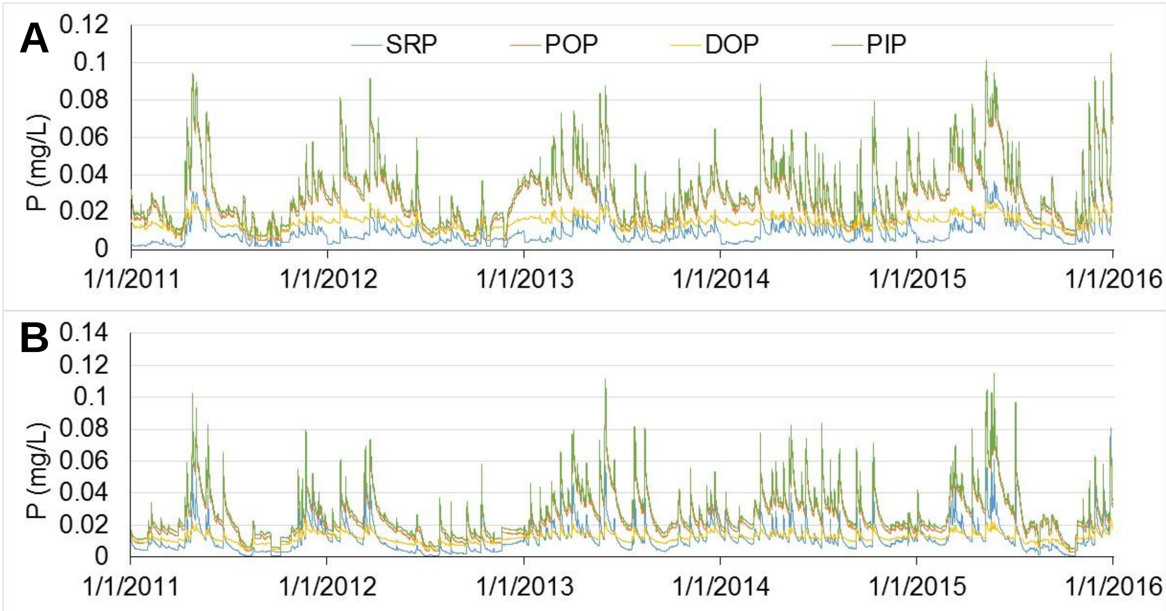


Figure 2-9. Phosphorus concentrations in inflows to Lake Wister from A) Fourche-Maline Creek and B) Poteau River.

derived using a set of common assumptions about the partitioning of N, P, and sediment among physical and chemical forms. Total organic N (TON) was computed as measured total Kjeldahl N (TKN) minus NH_4-N. Particulate P (PP) concentration was derived by subtracting the measured total dissolved P (TDP) from measured TP. DOP was estimated by subtracting measured SRP from measured TDP.

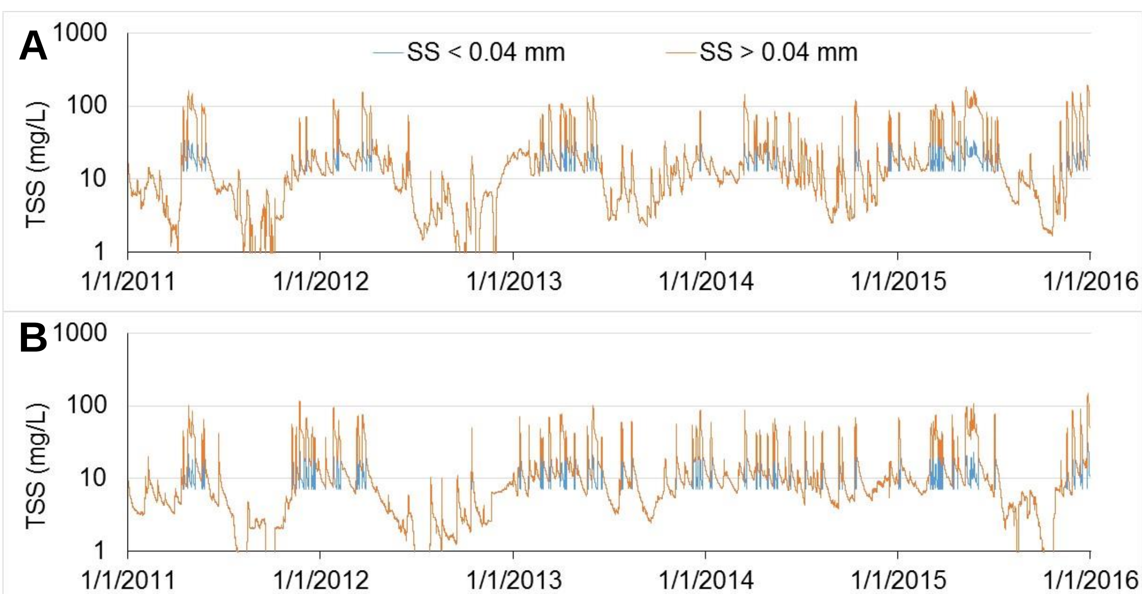


Figure 2-10. Total suspended solids concentrations in inflows to Lake Wister from A) Fourche-Maline Creek and B) Poteau River.

We used estimated flow data along with each of the measured (NH_4-N, NO_2+NO_3-N, SRP, and total organic carbon [TOC]) and derived (TON, PP, DOP) variables required for ELCOM-CAEDYM to develop regression models for flow-dependent concentrations for these variables in both the Fourche-Maline Creek and Poteau River. We expected seasonal variation to drive variability in these water quality constituents. Thus, we evaluated regression models for each water quality variable that included flow only as the independent variable or a combination of flow and sample day of year (DOY) as a proxy for seasonal influence on water quality using a multiple sin structure (Hirsch et al. 1993) as described for TN, TP, and TSS above.

Even the derived variables that could be computed from measured variables did not always match the required ELCOM-CAEDYM inputs. Thus, a variety of other variables were derived based on some basic assumptions on partitioning between chemical and physical forms, that were based on our best scientific judgement using data from nearby watersheds (Grantz et al. 2014). PON was assumed to be 25% of TON and DON be 75% of TON regardless of river flow conditions. POP and PIP were both assumed to be 50% of PP. Particulate organic carbon (POC) concentrations were derived as 25% of measured

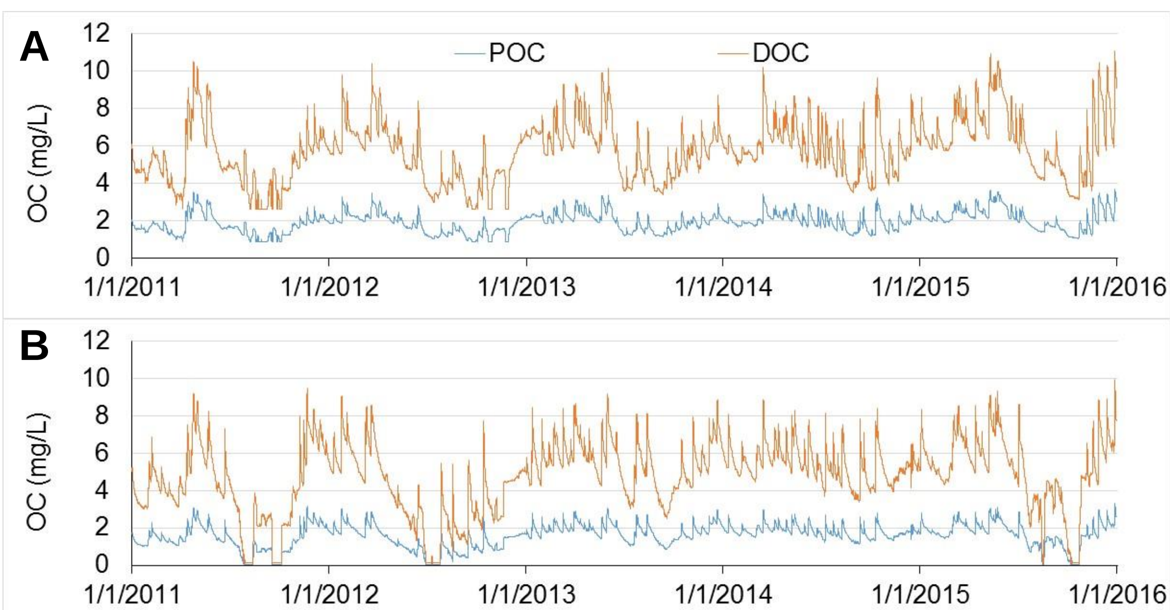


Figure 2-11. Organic carbon concentrations in inflows to Lake Wister from A) Fourche-Maline Creek and B) Poteau River.

total organic carbon (TOC) and dissolved organic carbon (DOC) concentrations were derived as 75% of TOC for samples in both rivers across all flow conditions.

Rather than TSS, ELCOM-CAEDYM requires inputs of the concentrations of suspended sediments of a particular size. For the Lake Wister model, we used the two suspended sediment size classes reported by the USGS--one group representing clay-size particles (< 0.004 mm diameter = SS1) and a second group for larger size sediment particles (> 0.004 mm diameter = SS2).

We simulated these two groups, however, we had no specific information with which to calibrate these divisions except the TSS data for Lake Wister. Suspended sediment groups SS1 (clays) and SS2 (silts and larger) were evenly distributed (SS1 and SS2 each 50% of TSS) during storm flow conditions, which was defined as greater than 20 m^3/s in the Fourche-Maline Creek and 50 m^3/s in the Poteau River. However, we used initial simulation/calibration activities to adjust the proportions of SS1 and SS2 during baseflow conditions. We experimented with 90% SS1 and 10% SS2, as well as 75% SS1 and 25% SS2, which made a drastic difference in model output. We settled on SS1 comprised of 82.5% TSS and SS2 comprised of 17.5% TSS at baseflow as the input which provided the best calibration statistics. Thus, SS1 and SS2 were evenly distributed (SS1 and SS2 each 50% of TSS) during storm flow conditions, which was defined as greater than 20 m^3/s in the Fourche-Maline Creek and 50 m^3/s in the Poteau River. At baseflow conditions, SS1 comprised 82.5% of TSS and SS2 comprised 17.5% of TSS.

From these regression models and variable derivations, we computed all boundary conditions for 15- or 30- intervals to match the flow data for the Fourche-Maline Creek and Poteau River, respectively, which are as continuous model inputs for N (Figure 2-8), P (Figure 2-9), TSS (Figure 2-10), and OC (Figure 2-11).

Although the PVIA water quality database included measured TSS and the ELCOM-CAEDYM model simulates sediments through multiple groups (SS1 and SS2 in the Lake Wister model), the State of Oklahoma assesses water quality in Lake Wister based on a turbidity standard (Section 1.2). PVIA collected turbidity data during all sampling between 2011-2015 at each of their water quality monitoring sites. Measured turbidity was strongly correlated with measured TSS at all sites and in all data combined for the lake (Figure 2-12). We used a conversion of suspended sediment model output (SS1 + SS2 = TSS) to turbidity with these measured relationships to evaluate model output relative to water quality standards.

The final regression models for all measured and derived variables required for ELCOM-CAEDYM, including the individual parameter estimates and whole-model statistics, are shown in Table 2-3. All models were highly statistically significant (i.e. $p \leq 0.001$), however, some of the models had better strength than others. For example, the strongest model for NH_4-N in the Poteau River had a coefficient of variation of 7%, while many other models had coefficients of determination between 30-50%.

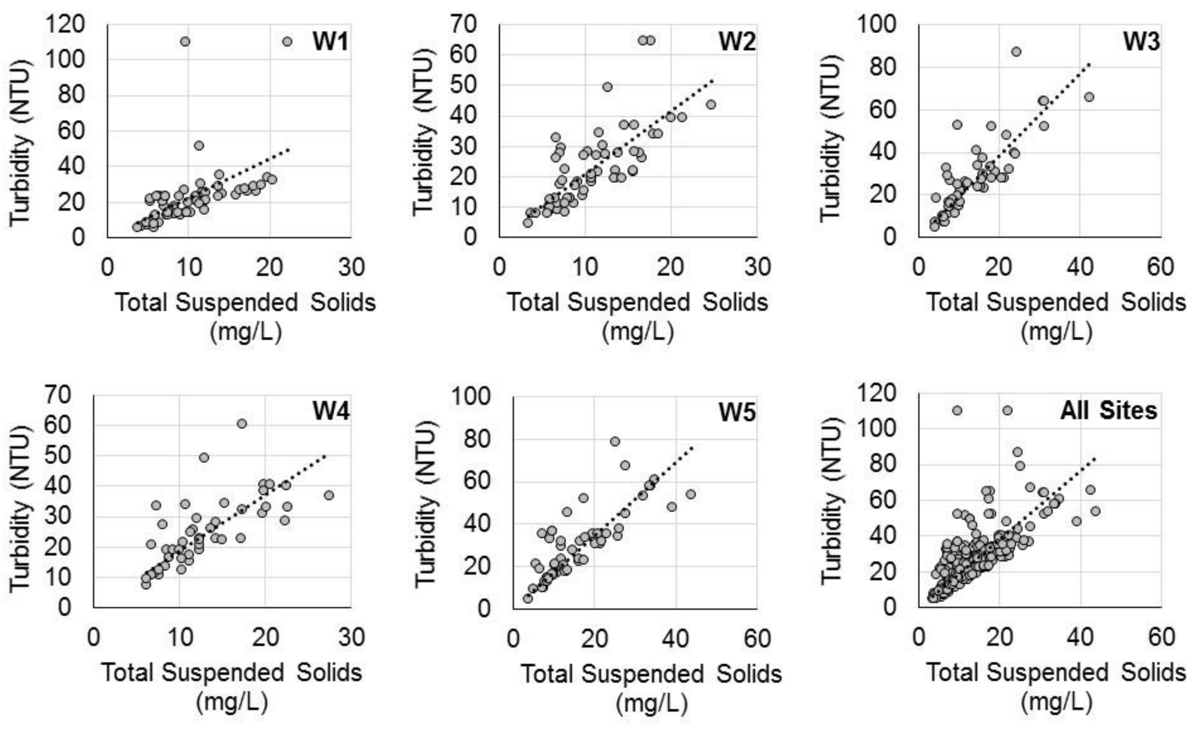


Figure 2-12. Relationship between measured TSS and measured turbidity across the different monitoring locations in Lake Wister.

2.3.6 Water Quality Initial Conditions – Model simulations were run in one-year segments starting on January 1 of each year and ending on December 31. We used measured data from January of each year to set the initial conditions for that year. For example, the

Table 2-3. Regression models for all ELCOM-CAEDYM required boundary conditions for the Lake Wister model.

Variable	Model Structure	a ± SE	b1 ± SE	b2 ± SE	b3 ± SE	*F* (*p*)	R^2
Fourche-Maline Creek							
TON	Flow	-0.64 ± 0.06	0.12 ± 0.02	--	--	35.0 (<0.001)	0.41
NH4-N	Flow + DOY	-7.75 ± 1.49	0.12 ± 0.03	-2.52 ± 1.07	3.08 ± 1.19	6.24 (0.0012)	0.28
NOx-N	Flow + DOY	2.81 ± 2.93	0.25 ± 0.05	2.43 ± 1.82	-5.20 ± 2.54	9.51 (<0.001)	0.38
PP	Flow	-3.01 ± 0.12	0.23 ± 0.04	--	--	40.0 (<0.001)	0.44
DOP	Flow	-4.25 ± 0.08	0.10 ± 0.03	--	--	16.5 (<0.001)	0.25
SRP	Flow + DOY	-8.98 ± 1.94	0.23 ± 0.03	-3.04 ± 1.21	3.28 ± 1.67	16.1 (<0.001)	0.51
TOC	Flow	2.01 ± 0.08	0.11 ± 0.02	--	--	21.3 (<0.001)	0.34
Poteau River							
TON	Flow + DOY	-1.13 ± 0.87	0.08 ± 0.01	-1.05 ± 0.54	1.25 ± 0.75	10.2 (<0.001)	0.38
NH4-N	Flow	-4.08 ± 0.13	0.06 ± 0.03	--	--	3.73 (<0.001)	0.07
NOx-N	Flow + DOY	11.03 ± 2.23	0.17 ± 0.04	7.68 ± 1.37	-12.0 ± 1.91	22.3 (<0.001)	0.58
PP	Flow + DOY	-8.03 ± 2.18	0.24 ± 0.04	-2.72 ± 1.35	4.02 ± 1.87	16.4 (<0.001)	0.50
DOP	Flow	-4.60 ± 0.08	0.11 ± 0.02	--	--	28.4 (<0.001)	0.36
SRP	Flow	-4.88 ± 0.14	0.27 ± 0.03	--	--	62.2 (<0.001)	0.55
TOC	Flow	5.13 ± 0.76	1.02 ± 0.19	--	--	29.4 (<0.001)	0.41

January 4, 2012 monitoring data were used to set the initial conditions for January 1, 2012. We repeated this for every other January 1 across the five-year simulation period so that the actual conditions measured closest to that date were used as the initial conditions. However, monitoring data were not available for January 2011. Thus, in that year, the long-term average conditions (average of January monitoring across 2012, 2013, 2014, and 2015) were used to estimate the 2011 initial conditions. One-year modeling runs were used to shorten computer run times and to facilitate comparison of multiple reduction scenarios for each year.

2.4 CAEDYM Model Parameters

CAEDYM equations represent the physical-chemical-biological water quality variables of interest for load reduction estimates for Lake Wister. A number of parameters exist within the model that control the kinetics of interactions. We identified 17 phytoplankton-related and 6 sediment-water interaction model parameters which have been shown to strongly influence model results from the perspective of sensitivity and uncertainty (Table 2-4; Burger et al. 2008, Missaghi et al. 2014).

The Lake Wister ELCOM-CAEDYM model was set up with three phytoplankton groups: cyanobacteria, green algae, and diatoms, which are numbered in the CAEDYM control file as groups 2, 4, and 7, respectively. In most instances, we used the default rate coefficients for the vast majority of these parameters (Hipsey et al. 2013). However, during calibration some of these parameters were adjusted within acceptable range based on the scientific literature to improve model performance. Missaghi et al. (2014) conducted a thorough evaluation of the sensitivity of the ELCOM-CAEDYM model to a number of phytoplankton growth parameters. They found that the phytoplankton half-saturation constant for P (K_p), the phytoplankton minimum internal P (IP_{min}), and the respiratory and mortality constant (k_r) represented three of most influential parameters within the model. We iteratively adjusted these parameters in simulations for the three calibration years (2011, 2013, and 2015) to identify the most accurate model output. We also experimented with carbon and nutrient recycling parameters (POX_{max} and DOX_{max}, where X was C (carbon), P, or N), to calibrate the model. Additionally, we experimented with a number of sediment-water interactions. Specifically, we set the maximum P release rate from sediment to 0.003 g per day, which represents the mean value derived empirically for sediments in Lake Wister (Haggard et al. 2012). The parameter values used in the final calibrated model are provided in Table 2-4.

2.5 ELCOM-CAEDYM Calibration to Existing Conditions and Independent Verification (Validation)

The five model years were divided into three calibration years (2011, 2013, and 2015) and two validation years (2012 and 2014). All model calibration activities were only conducted by comparing model output from the calibration years to measured data from those years. The comparisons between model out and measured data in 2012 and 2014 were reserved to independently verify the model calibration after the final calibration activities were completed.

The only exception to this was the calibration for water level, as discussed in Section 2.5.2, below.

A detailed description of the calibration benchmarks that were used in the study and how each variable performed in both calibration and validation is provided in the subsections below.

Table 2-4. Model kinetic parameters used in ELCOM-CAEDYM model. Values are specific to phytoplankton group: 2=cyanobacteria, 4=green algae, 7=diatoms.

Parameter	Units	Value	Reference
Phytoplankton kinetics		**2, 4, 7**	
Maximum potential growth	day^{-1}	1.2, 1.8, 1.8	Robson & Hamilton 2004
Carbon to chlorophyll-a ratio	mg C/mg chl-a	80, 27, 32	Missaghi et al. 2014
PS-irradiance slope	µmol m^{-2} s^{-1}	60, 20, 20	Missaghi et al. 2014
Light saturation	µmol m^{-2} s^{-1}	500, 400, 400	Robson & Hamilton 2004
Phosphorus half-saturation	mg/L	0.006, 0.01, 0.01	Holm & Armstrong 1981
Nitrogen half-saturation	mg/L	0.0001, 0.045, 0.045	Hamilton & Schladow 1997
Minimum internal P	mg P/mg chl-a	0.5, 0.25, 0.25	Robson & Hamilton 2004
Maximum internal P	mg P/mg chl-a	2.0, 1.3, 1.3	Hamilton & Schladow 1997
Maximum P uptake	mg P mg C^{-1} day^{-1}	0.3, 2.0, 1.0	Hamilton & Schladow 1997
Minimum internal N	mg N/mg chl-a	2.5, 2.0, 2.0	Robson & Hamilton 2004
Maximum internal N	mg N/mg chl-a	4.0, 4.5, 4.5	Robson & Hamilton 2004
Maximum N uptake	mg N mg chl-a^{-1} day^{-1}	1.5, 3.5, 3.5	Robson & Hamilton 2004
Standard growth temp.	ºC	20, 12, 12	Robson & Hamilton 2004
Optimum growth temp.	ºC	28, 23, 23	Robson & Hamilton 2004
Maximum growth temp.	ºC	40, 30, 30	Robson & Hamilton 2004
Respiration rate coefficient	day^{-1}	0.05, 0.12, 0.12	Robson & Hamilton 2004
Settling velocity	m/day	4.4, 0.5, 0.5	Romero et al. 2004
Sediment Water Interactions			
Maximum potential SOD	g m^{-2} day^{-1}	0.62	Haggard et al. 2012
SOD half-saturation	mg/L	0.5	Burger et al. 2008
Maximum pot. P release	g m^{-2} day^{-1}	0.003	Haggard et al. 2012
Maximum pot. NH4 release	g m^{-2} day^{-1}	0.05	Missaghi et al. 2014
Maximum pot. NO3 release	g m^{-2} day^{-1}	-0.001	Missaghi et al. 2014
Nutrient release DO half-sat.	mg/L	0.5	Burger et al. 2008

2.5.1 Calibration Benchmarks

2.5.1 Calibration Benchmarks – Model performance was measured by comparing model output to measured data using the relative root mean square error (Relative RMSE) and correlation coefficient. The RMSE was:

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}\left(y_p - y_m\right)^2}{n}}$$

where y_p was a predicted value from the model, y_m was the corresponding measured value, and n was the number of paired modeled-measured values.

The relative RMSE was:

$$Relative\ RMSE = \frac{RMSE}{MSE_{max} - MSE_{min}}\ x\ 100$$

where MSE_{max} and MSE_{min} represent the maximum and minimum mean square error (MSE) which was:

$$MSE = \sum_{i=1}^{n} \left(y_p - y_m\right)^2$$

where y_p was the predicted value from the model, y_m was the corresponding measured value.

The correlation coefficient was computed as:

$$R = \frac{\sum_{i=1}^{n} y_{p_i}\, y_{m_i} - n\bar{y}_p\bar{y}_m}{\sqrt{\left(\sum_{i=1}^{n} y_{p_i}^2 - n\bar{y}_p^2\right)}\ \sqrt{\left(\sum_{i=1}^{n} y_{m_i}^2 - n\bar{y}_m^2\right)}}$$

where y_{p_i} and y_{m_i} are individual pairs of model-predicted and measured values and $\bar{y}_p$ and $\bar{y}_m$ are the mean predicted and measured values, respectively.

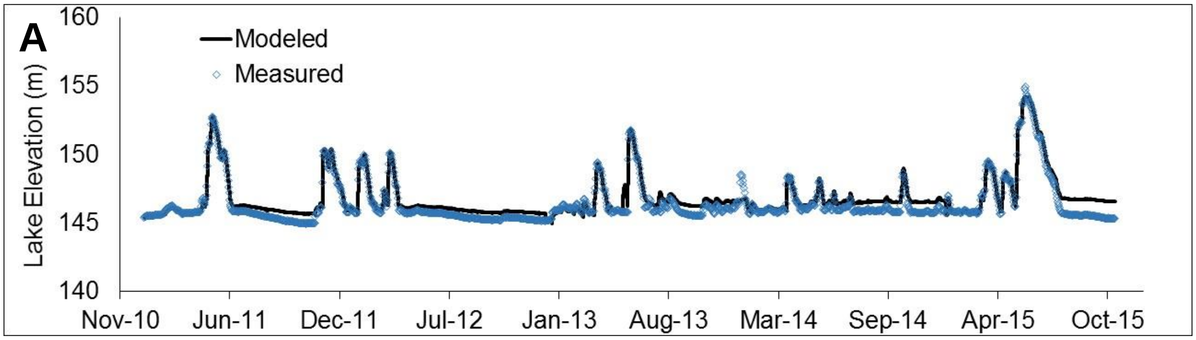
Model performance statistics for both the calibration and validation years were evaluated based on criteria recommended by the ODEQ (Table 2-5). We also used R values to evaluate model performance in addition to ODEQ recommendations. The R value is particularly useful in demonstrating model performance against measurements that are influenced by analytical detection limits. For example, measured quantities of dissolved nutrients such and NH_4-N, NO_2+NO_3-N, and SRP are commonly less than the analytical method detection limit and were assigned the detection limit value in order to have a quantity for comparison with model output. Thus, the RMSE and the relative RMSE can be strongly affected by these measurements. In those cases, the R statistic is useful for evaluating the correlation between predicted and measured values.

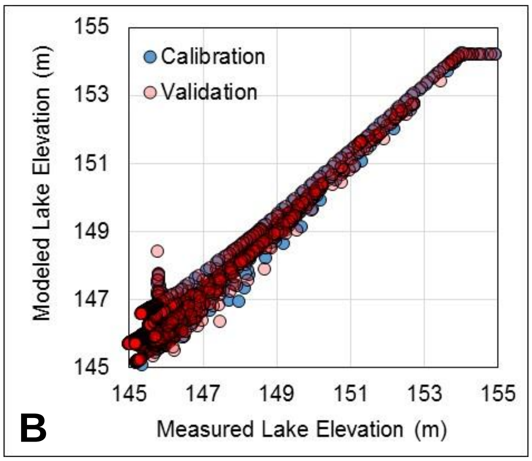
Table 2-5. Target and actual model calibration and validation statistics for the Lake Wister ELCOM-CAEDYM model.

Lake Model Variable	Acceptable Relative RMSE	Calibration Relative RMSE	Validation Relative RMSE	Calibration R	Validation R
Elevation	± 20%	5.5%	5.6%	0.98	0.97
Surf. Temp.	± 20%	11%	10%	0.96	0.97
6m Temp.	± 20%	12%	24%	0.95	0.73
Surf. D.O.	± 20%	17%	14%	0.89	0.89
6 m D.O.	± 20%	14%	9.2%	0.93	0.96
TSS	± 50%	57%	36%	0.21	0.07
Total N	± 50%	17%	20%	0.39	0.01
Nitrate	± 50%	19%	14%	0.78	0.85
Ammon.	± 50%	37%	22%	0.13	0.27
Total P	± 50%	94%	21%	0.10	0.01
SRP	± 50%	22%	25%	0.74	0.36
Chl-a	± 100%	22%	21%	0.03	0.40

2.5.2 Hydrology Calibration – We evaluated the hydrologic mass balance of the Lake Wister ELCOM model by comparing model output to USACE-measure water elevation data. The USACE reports water elevation data at Lake Wister twice daily, at midnight and at 0800. We computed calibration statistics by pairing model output with the data recorded at midnight over the 5-year period of record for the modeling study, and computed validation statistics by pairing model output with the data recorded at 0800 each day.

The Lake Wister ELCOM model predicted lake elevation well (Figure 2-13). The RMSE was 0.5 m and the relative RMSE for lake elevation was 5.5% and 5.6% for the calibration and validation data, respectively, which was less than the 20% benchmark requirement (Figure 2-13C). The correlation coefficient between modeled and measured



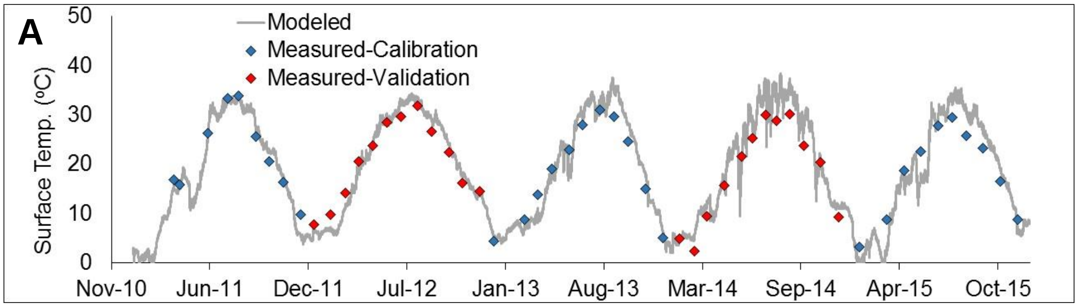


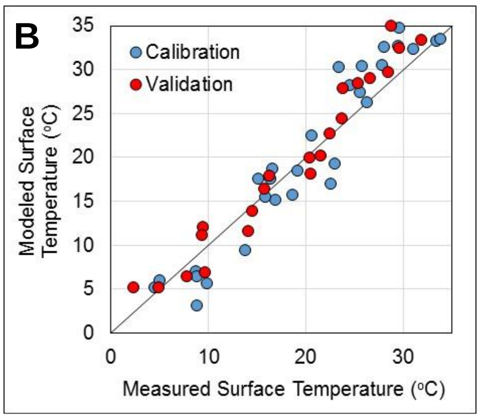
C	Calibration	Validation
RMSE	0.50	0.51
Relative RMSE	5.5% (*20%*)	5.6% (*20%*)
R	0.98	0.97
p	< 0.0001	< 0.0001
n	1750	1750

Figure 2-13. Model calibration information for lake elevation: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

data was 0.98 and 0.97 for the calibration and validation periods, respectively. Interestingly, the model tended to overestimate the lake elevation during extended periods of low inflows, and this pattern was worse following any large flood events as exemplified by the periods between August-December 2011 and July-December 2015 (Figure 2-13A). However, this small difference in elevation was insignificant in terms of volume when compared with the peak elevations that occurred during storm events.

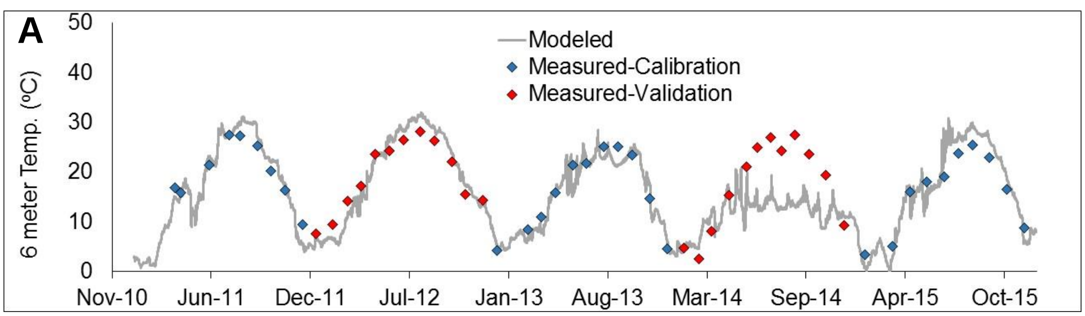
2.5.3 Temperature Calibration – The Lake Wister ELCOM-CAEDYM model was calibrated using water temperature data from the lake surface and a depth of 6 m at site

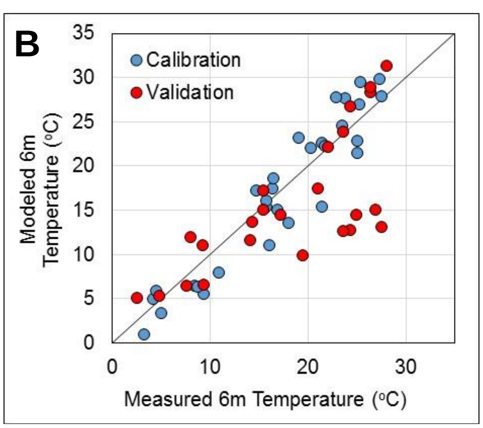




C	Calibration	Validation
RMSE	3.2	2.9
Relative RMSE	11% (*20%*)	10% (*20%*)
R	0.96	0.97
p	< 0.0001	< 0.0001
n	32	24

Figure 2-14. Model calibration information for surface water temperature at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.





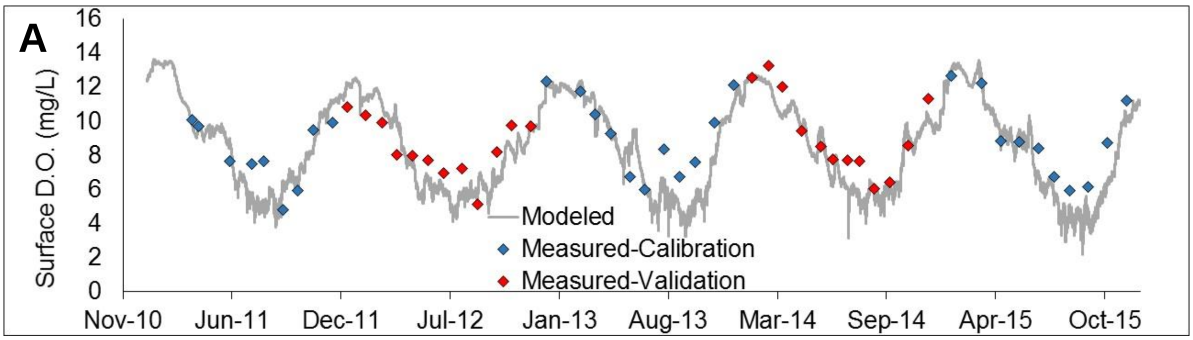
C	Calibration	Validation
RMSE	2.9	6.1
Relative RMSE	12% (*20%*)	24% (*20%*)
R	0.95	0.73
p	< 0.0001	< 0.0001
n	32	24

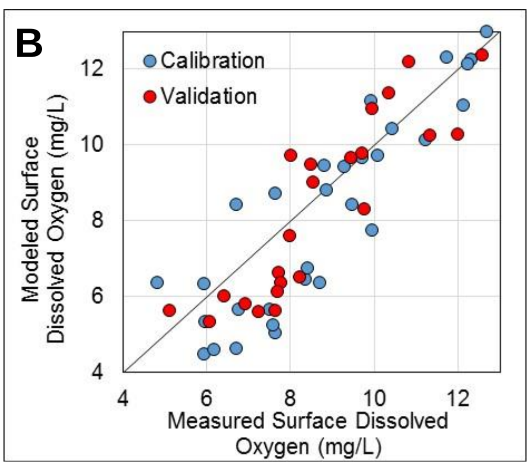
Figure 2-15. Model calibration information for water temperature at 6 meter depth at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

W2 immediately adjacent to the dam. Of the five lake monitoring data collection sites available (Figure 1-2), site W2 in our judgement represented the best single location for a water quality assessment of the lake. W2 is a deep water site (for Lake Wister), located near the dam. Calibrating the model to a single site is significantly less complex than calibrating to multiple sites. Calibrating to multiple sites would have required tradeoff decisions about which sites were more important for which parameters to achieve the best calibration. Through inspection, we noted there were not major differences in water quality due to spatial variability in the lake in either the modeled output or measured data. The RMSE of surface temperatures was 3.2ºC for the calibration data and 2.9ºC for the validation data (Figure 2-14). The resulting relative RMSE was 11% and 10% for the calibration and validation periods, respectively, well below the 20% relative RMSE target. The correlation coefficient for surface temperature was 0.96 for the calibration period and 0.97 for the validation period, suggesting a strong correlation between modeled and measured data.

The RMSE of water temperatures at 6-m depth was 2.9 ºC and 6.1 ºC for the calibration and validation periods, respectively (Figure 2-15). The resulting relative RMSE was 12% for the calibration period and 24% for the validation period. Thus, the relative RMSE for the calibration period was met, but the relative RMSE during validation did not meet the 20% target. However, the correlation coefficient for water temperature at depth 6 m was 0.95 for the calibration period and 0.73 for the validation period. The somewhat lower correlation coefficient in the validation period was caused by an underestimation of 6 m water temperature by the model in 2014 (Figure 2-15A). This pattern was discovered at validation after all calibration activities were complete, and no model inputs or calibration parameters were adjusted.

2.5.4 Dissolved Oxygen Calibration – The Lake Wister ELCOM-CAEDYM model was calibrated using water temperature data from the lake surface and a depth of 6 m at site W2 immediately adjacent to the dam. The RMSE of surface dissolved oxygen was 1.4 mg/L for the calibration data and 1.2 mg/L for the validation data (Figure 2-16). The resulting relative RMSE was 17% and 14% for the calibration and validation periods, respectively, which was less than the 20% relative RMSE target. The correlation coefficient for surface dissolved oxygen was 0.89 for the calibration period and 0.89 for the validation period, suggesting a strong correlation between modeled and measured data.

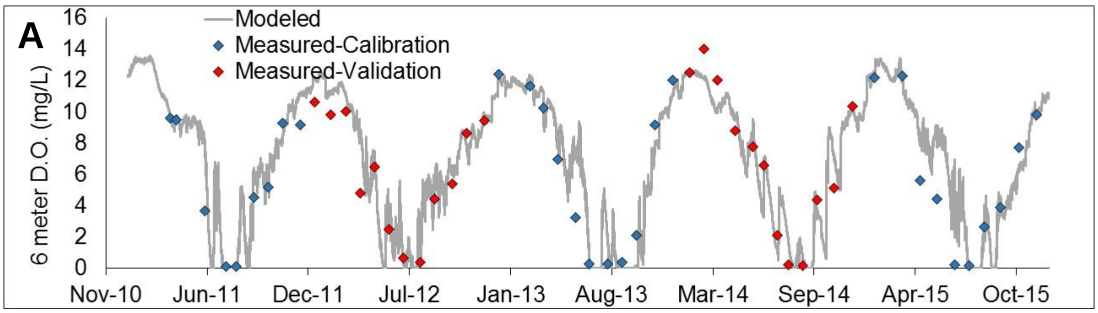


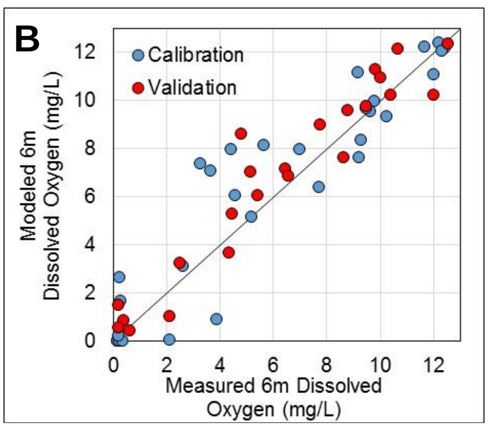


C	Calibration	Validation
RMSE	1.4	1.2
Relative RMSE	17% (*20%*)	14% (*20%*)
R	0.89	0.89
p	< 0.0001	< 0.0001
n	32	24

Figure 2-16. Model calibration information for surface water dissolved oxygen at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

The RMSE of dissolved oxygen at 6 m depth was 1.7 mg/L and 1.3 mg/L for the calibration and validation periods, respectively (Figure 2-17). The resulting relative RMSE was 14% for the calibration period and 9.2% for the validation period, which was less than the 20% relative RMSE target. The correlation coefficient for 6 m water temperature was 0.93 for the calibration period and 0.96 for the validation period,



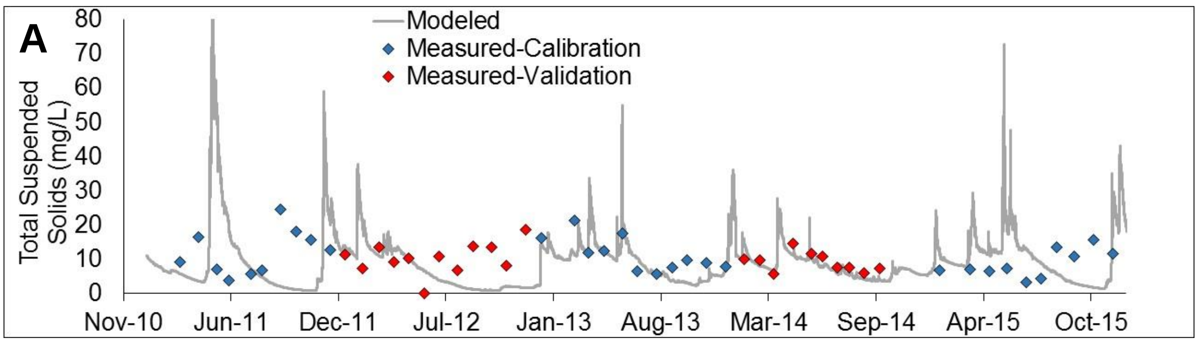


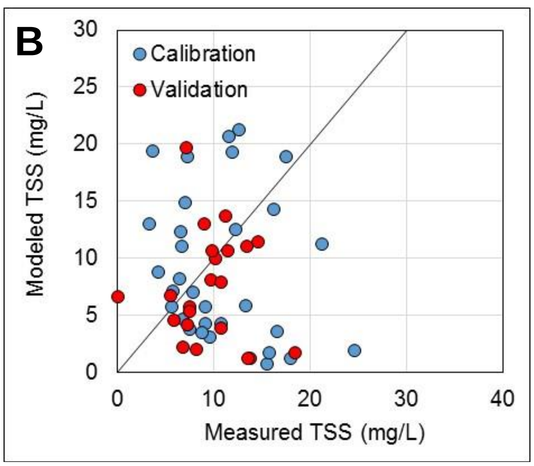
C	Calibration	Validation
RMSE	1.7	1.3
Relative RMSE	14% (*20%*)	9.2% (*20%*)
R	0.93	0.96
p	< 0.0001	< 0.0001
n	32	24

Figure 2-17. Model calibration information for dissolved oxygen at 6 meter depth at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

suggesting a strong correlation between measured and modeled data.

2.5.5 Suspended Solids Calibration – The Lake Wister ELCOM-CAEDYM model was calibrated using TSS data from the lake surface at site W2, immediately adjacent to the dam. The RMSE of TSS was 12.1 mg/L for the calibration data and 6.7 mg/L for the validation data (Figure 2-18). The resulting relative RMSE was 57% and 36% for the calibration and validation periods, respectively, which was greater than the 20% relative RMSE target. However, the correlation coefficient for TSS was 0.21 for the calibration period and 0.07 for the validation period. The correlation between model data and measured data for TSS was not statistically significant for either the calibration (p = 0.7336) or validation (p = 0.7629) periods (Figure 2-18C). This pattern in statistical output suggests that the model performed poorly at predicting the average TSS



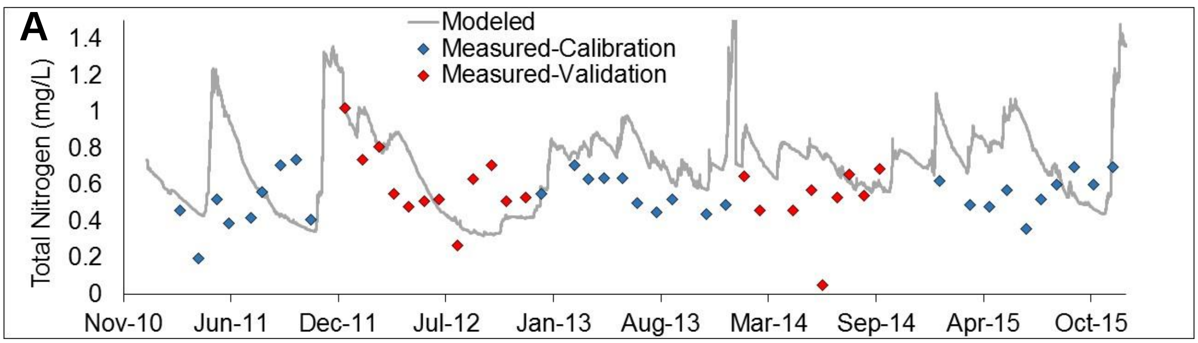


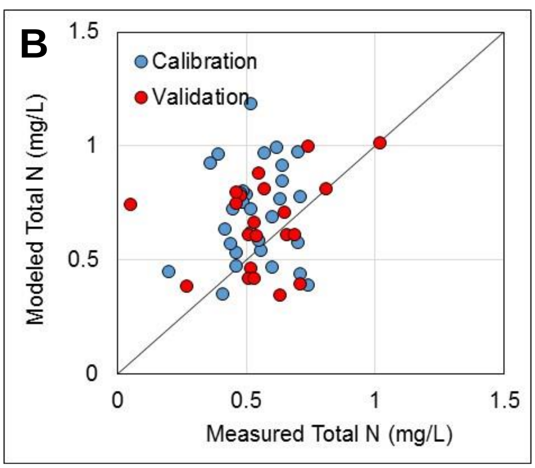
C	Calibration	Validation
RMSE	12.1	6.7
Relative RMSE	57% (*20%*)	36% (*20%*)
R	0.21	0.07
p	0.7336	0.7629
n	31	22

Figure 2-18. Model calibration information for surface TSS at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

concentrations in Lake Wister (i.e. high relative RMSE) and poorly predicted the TSS concentrations that occurred during specific stormflow events. The model typically overestimated TSS at high flow conditions (Figure 2-18A). Further, the model tended to underestimate TSS concentrations during extended periods of low river flow into the lake, as seen in the summer-fall periods of 2011, 2012, and 2015. Interestingly, the model performed best at predicting TSS concentrations in years where there were relatively frequent rainfall events that were not particularly large relative to other years (e.g. calendar years 2013 and 2014; Figure 2-18A).

2.5.6 Nutrient Calibration – The Lake Wister ELCOM-CAEDYM model was calibrated using TN data from the lake surface at site W2, immediately adjacent to the dam. The RMSE of TN was 0.81 mg/L for the calibration data and 2.7 mg/L for the validation data (Figure 2-19). The resulting relative RMSE was 17% and 20% for the calibration and



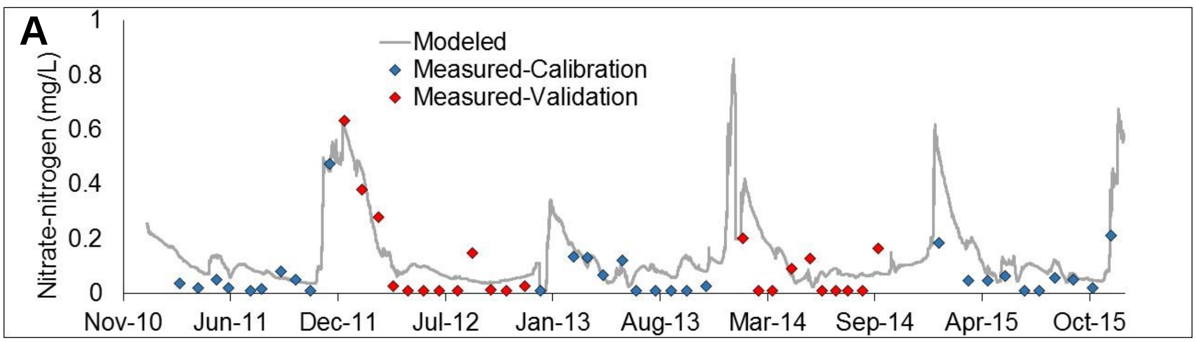


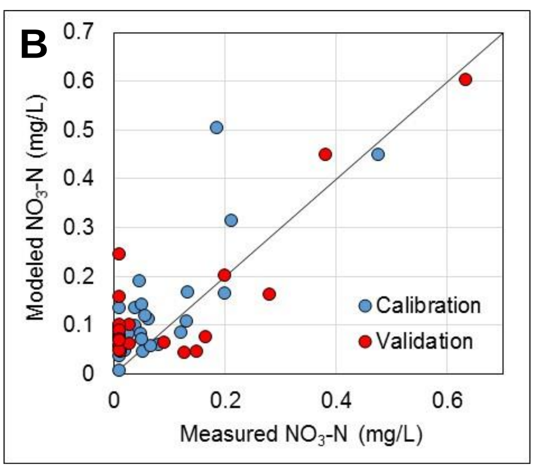
C	Calibration	Validation
RMSE	0.81	2.7
Relative RMSE	17% (*50%*)	20% (*50%*)
R	0.39	0.01
p	0.0273	0.9796
n	32	22

Figure 2-19. Model calibration information for surface TN at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

validation periods, respectively, which was less than the 50% relative RMSE target. The correlation coefficient for TN was 0.39 for the calibration period, but only 0.01 for the validation period. The correlation between model data and measured data for TSS was statistically significant for the calibration period (p = 0.0273) but not the validation period (p = 0.9796). This pattern in statistical output suggests that the model performed well at predicting the average TN concentrations in Lake Wister (i.e. low relative RMSE). However, the model performance was weaker in regard to predicting the extreme events, particularly in the validation periods (Figure 2-19A).

The Lake Wister ELCOM-CAEDYM model was calibrated using NO_2+NO_3-N data (referred to as NO_3-N in graphics) from the lake surface at site W2, immediately adjacent to the dam. The RMSE of NO_2+NO_3-N was 0.09 mg/L for the calibration data and 0.09 mg/L for the validation data (Figure 2-20). The resulting relative RMSE was 19% and 14% for the calibration and validation periods, respectively, which was less than the 50% relative RMSE target. The correlation coefficient for NO_2+NO_3-N was 0.78 for the calibration period and 0.85 for the validation period. This pattern in statistical output suggests that the model performed well in predicting the average NO_2+NO_3-N



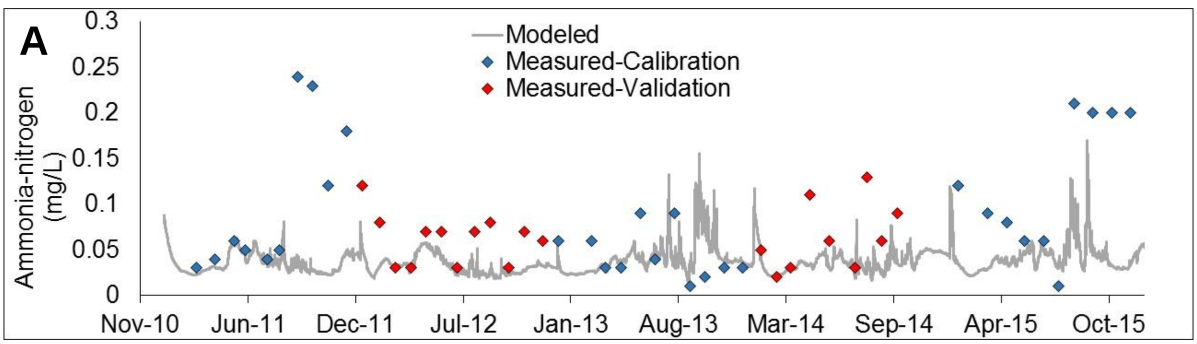


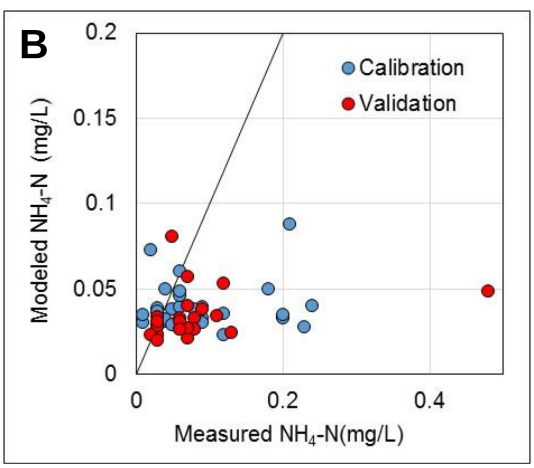
C	Calibration	Validation
RMSE	0.09	0.09
Relative RMSE	19% (*50%*)	14% (*50%*)
R	0.78	0.85
p	< 0.0001	< 0.0001
n	32	22

Figure 2-20. Model calibration information for surface nitrate at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

concentrations (i.e. low relative RMSE), and did predict the extreme concentrations in NO_2+NO_3-N well (i.e. high R; Figure 2-20A).

The Lake Wister ELCOM-CAEDYM model was calibrated using NH_4-N data from the lake surface at site W2, immediately adjacent to the dam. The RMSE of NH_4-N was 0.09 mg/L for the calibration data and 0.10 mg/L for the validation data (Figure 2-21). The resulting relative RMSE was 37% and22% for the calibration and validation periods, respectively, which was less than the 50% relative RMSE target. The correlation coefficient for NH_4-N was 0.13 for the calibration period and 0.27 for the validation period. However, the correlation between model data and measured data for NH_4-N was not statistically significant for either the calibration (p = 0.4660) or validation (p = 0.2266) periods (Figure 2-21C). This pattern in statistical output suggests that the model performed well in predictions for the average NH_4-N concentrations (i.e. low relative

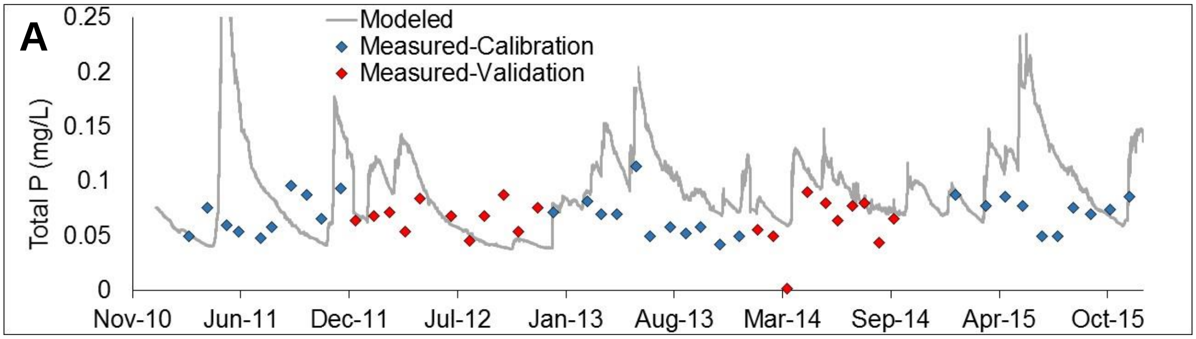


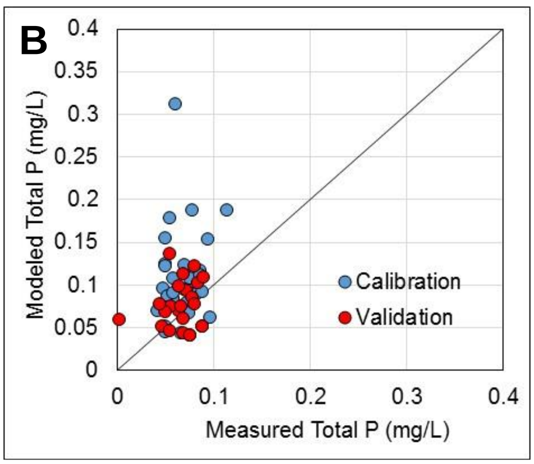


C	Calibration	Validation
RMSE	0.09	0.10
Relative RMSE	37% (*50%*)	22% (*50%*)
R	0.13	0.27
p	0.4660	0.2266
n	32	22

Figure 2-21. Model calibration information for surface ammonium at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation

RMSE), but the model performance was weak in regard to predicting the extreme events in both the calibration and validation periods. (Figure 2-21A).

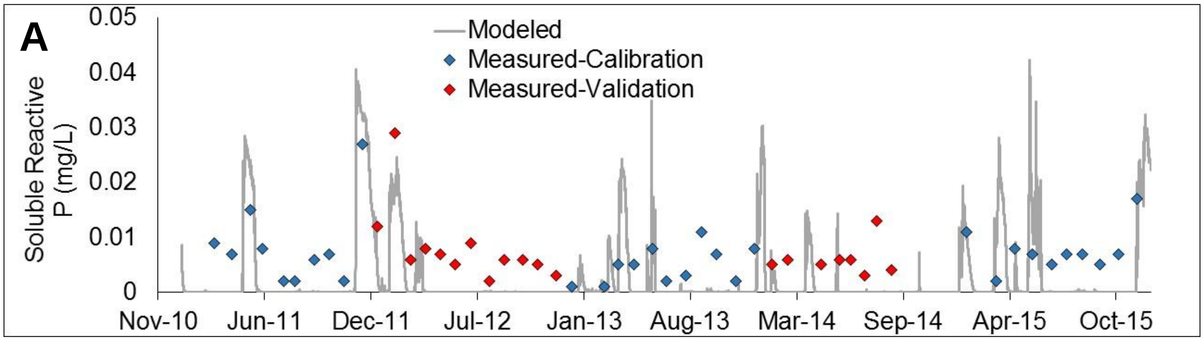


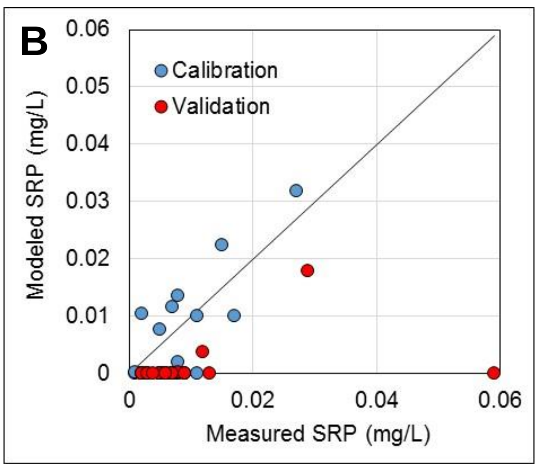


C	Calibration	Validation
RMSE	0.07	0.69
Relative RMSE	94% (*50%*)	21% (*50%*)
R	0.10	0.01
p	0.6019	0.9490
n	32	22

Figure 2-22. Model calibration information for surface TP at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

The Lake Wister ELCOM-CAEDYM model was calibrated using TP data from the lake surface at site W2, immediately adjacent to the dam. The RMSE of TP was 0.07 mg/L for the calibration data and 0.69 mg/L for the validation data (Figure 2-22). The resulting relative RMSE was 94% and 21% for the calibration and validation periods, respectively. Thus, the calibration exceeded the 50% target relative RMSE, but the validation relative RMSE was within acceptable limits. Alternatively, the correlation coefficient for TP was 0.10 for the calibration period and 0.01 for the validation period. However, the correlation between model data and measured data for TP was not statistically significant for either the calibration ($p = 0.6019$) or validation ($p = 0.9490$) periods (Figure 2-22C). This pattern in statistical output suggests that the model predictions for TP concentrations were sometimes weak (Figure 2-22A). Prediction of average conditions during the validation period was strong (i.e. low relative RMSE), but the prediction of average conditions during the calibration period and the prediction of extreme conditions in both the calibration and validation periods was less strong.



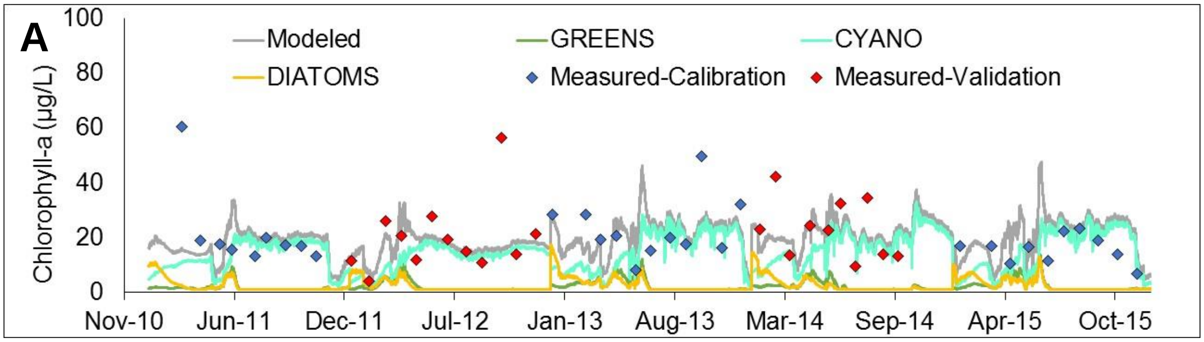


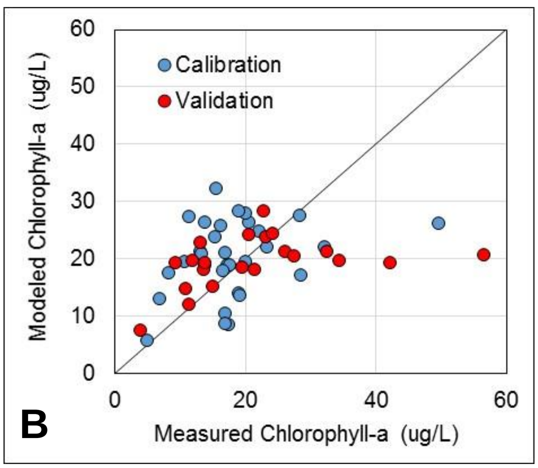
C	Calibration	Validation
RMSE	0.006	0.014
Rel. RMSE	22% (*50%*)	25% (*50%*)
R	0.74	0.36
p	< 0.0001	0.1031
n	32	22

Figure 2-23. Model calibration information for surface soluble reactive phosphorus at station W2: A) Time-series plot of modeled and measured data, B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

The Lake Wister ELCOM-CAEDYM model was calibrated using SRP data from the lake surface at site W2, immediately adjacent to the dam. The RMSE of SRP was 0.006 mg/L for the calibration data and 0.014 mg/L for the validation data (Figure 2-23). The resulting relative RMSE was 22% and 25% for the calibration and validation periods, respectively, which was less than the 50% target relative RMSE. The correlation coefficient for SRP was 0.74 for the calibration period and 0.36 for the validation period. This pattern in statistical output suggests that the model did predict the average SRP concentrations well (i.e. low relative RMSE) anddid predict the extreme concentrations in SRP well (i.e. high R; Figure 2.-23A).

2.5.7 Chlorophyll-a Calibration – The Lake Wister ELCOM-CAEDYM model was calibrated using chl-a data from the lake surface at site W2, immediately adjacent to the dam. The RMSE of chl-a was 11.9 µg/L for the calibration data and 11.0 µg/L for the validation data (Figure 2-24). The resulting relative RMSE was 22% and 21% for the calibration and validation periods, respectively, which was well below the 100% relative RMSE target. Interestingly, the correlation coefficient for the calibration period was only 0.03 (p = 0.8633), but the correlation coefficient for the validation periods was 0.40 (p = 0. 0621). This pattern in statistical output suggests that the model performed well at predicting the average chl-a concentrations in Lake Wister (i.e. low relative RMSE). The model also performed reasonably well at predicting extreme conditions in the validation years, but not in the calibration years (Figure 2-24A).





C	Calibration	Validation
RMSE	11.9	11.0
Rel. RMSE	22% (*100%*)	21% (*100%*)
R	0.03	0.40
p	0.8633	0.0621
n	32	22

Figure 2-24. Model calibration information for surface chl-a at station W2: A) Time-series plot of modeled and measured data (including model estimates of taxa-specific chl-a concentrations), B) scatterplot of modeled versus measured values, and C) calibration and validation statistics.

Section 3 – Simulation Scenarios and Results

The calibrated and validated Lake Wister ELCOM-CAEDYM model was manipulated to simulate watershed and in-lake water quality management scenarios that could potentially improve water quality. Specifically, we were interested in evaluating management options that would 1) reduce the long-term chl-a concentration in Lake Wister to less than 10 µg/L, and 2) reduce the number of turbidity violations (>25 NTU) to less than 10% of observations.

3.1 Nutrients

We examined the potential for both internal and external reductions in phosphorus to reduce the amount of phytoplankton biomass expressed as chl-a in Lake Wister. Management options such as in-lake alum application have the potential to reduce the internal load of P moving from lake sediments to the water column. To simulate this effect, we manipulated the rate coefficient for maximum potential P release from sediment (Table 2-4) in the CAEDYM control file so that sediment P release was varied at decreased percentages of current conditions. We also simulated increased internal P loading to the lake by manipulating the CAEDYM control file so that sediment P release was varied at increased percentages of current conditions. Each of these simulated conditions were modeled across the entire five year modeling timeframe and the details of the parameter and boundary conditions adjustments for each of the simulation frameworks are provided in Sections 3.1.1 – 3.1.5.

To evaluate the effect of variable nutrient concentrations entering the lake from the watershed, we manipulated the concentrations of all nutrient forms for both N and P to be decreased percentages of the current conditions over the five year modeling timeframe. We conducted simulations for P reductions alone, N reductions alone, and combined P and N reductions. We also conducted simulations of increased nutrient loading to the lake by manipulating the concentrations of all nutrient forms for both N and P to be increased percentages of the current conditions over the five year modeling timeframe.

3.1.1 Internal Phosphorus Loading – In order to simulate the potential for in-lake management to reduce chl-a concentrations, the rate coefficient for the maximum potential P release from sediments was adjusted in the CAEDYM control file. Sediment P release in these files was reduced to 1% (0.00003 g P m^{-2} day^{-1}) of the value from the calibrated model (0.003 g P m^{-2} day^{-1}) and the model was run with all other conditions being the same as the final calibrated model in order to simulate a 99% reduction in internal P loading to the lake. In another simulation, we reduced maximum potential P release from sediments to 10% (0.0003 g P m^{-2} day^{-1}) of the calibrated value current values to simulate a 90% reduction in internal P loading. The process was repeated with 25% (0.00075 g P m^{-2} day^{-1}), 50% (0.0015 g P m^{-2} day^{-1}), and 75% (0.00225 g P m^{-2} day^{-1}) reduction of current maximum potential P release to simulate 75%, 50% and 25% internal P reduction scenarios, respectively. We also evaluated the effect of an increase in internal P loading by increasing the maximum potential P release by 10x (0.03 g P m^{-2} day^{-1}) and 100x (0.3 g P m^{-2} day^{-1}).

The simulated five-year average chl-a concentrations decreased with increasing internal load reductions to the model (Figure 3-1). **The five-year average chl-a concentration decreased by 0.027 µg/L for every 1% decrease in internal P load** (Figure 3-1A). The simulated decreases in internal P loading (25% - 99% internal P load reduction) resulted in relatively small decreases in the annual average TP concentration observed at site W2. However, the 10x and 100x simulated increase in internal P loading resulted in significant increases in the annual average TP concentrations. The change in annual TP concentrations at site W2 across all these simulated conditions resulted in major changes in the annual chl-a concentrations (Figure 3-1B). Most of the variability in both TP and

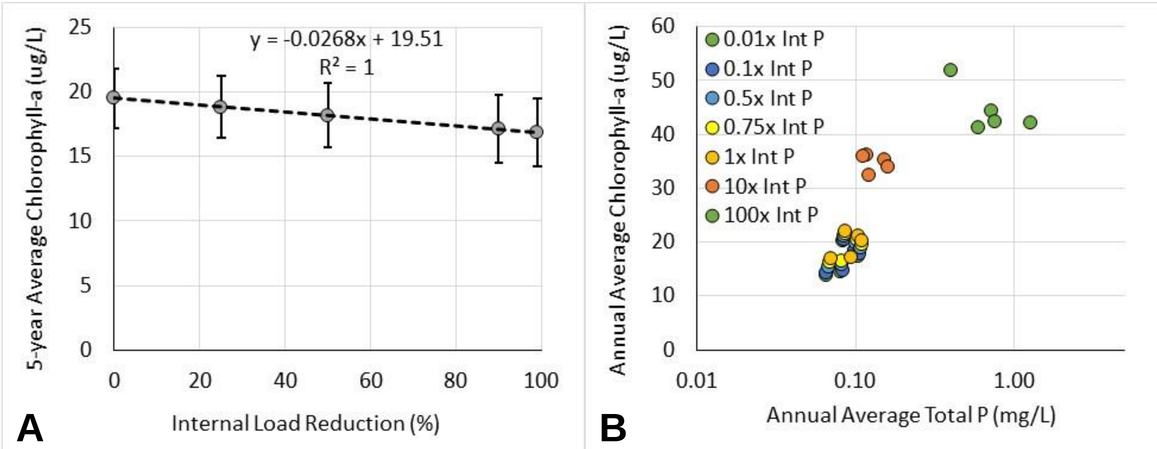


Figure 3-1. Model results from internal P load reduction simulations. A) Effect of various internal percent P reductions on average chl-a concentrations at site W2 in Lake Wister, B) Relationship between TP concentration and chl-a concentrations at site W2 in Lake Wister across each of the five year model years in response to internal P load reductions and 10x and 100x simulated P load increases.

chl-a concentrations resulted from the 10x and 100x simulated increases in internal P loading. This suggests that while current internal P loading contributes to the current trophic state of Lake Wister, an *increase* in internal loading resulting from continued accumulation of P from external sources in lake sediments has the potential to significantly worsen lake trophic state.

3.1.2 Watershed Phosphorus Loading – In order to simulate the potential for watershed management to reduce P concentrations and ultimately chl-a concentrations in Lake Wister, the water quality input files for the Fourche-Maline Creek and Poteau River were directly manipulated. The SRP, DOP, POP, and PIP concentrations in these files were reduced to 10% of current values and the model was run with all other conditions being the same as the final calibrated model in order to simulate a 90% reduction in P loading to the lake. In another simulation, we reduced watershed inputs to 25% of current values to simulate a 75% reduction in P input from the watershed. The process was repeated with SRP, DOP, POP, and PIP concentrations from both rivers to simulate 50% and 25%

reductions in watershed P loading, respectively. We also evaluated the effect of an increase in P loading by increasing the P concentrations of all forms by 150% and 200%.

The simulated five-year average chl-a concentrations decreased with increasing external load reductions to the model (Figure 3-2). **The five-year average chl-a concentration decreased by 0.12 µg/L for every 1% decrease in external P load** (Figure 3-2A). Based on this relationship, we found that a 78% P load reduction from external sources is necessary to achieve an average annual chl-a concentration of 10 µg/L (Figure 3-2B).

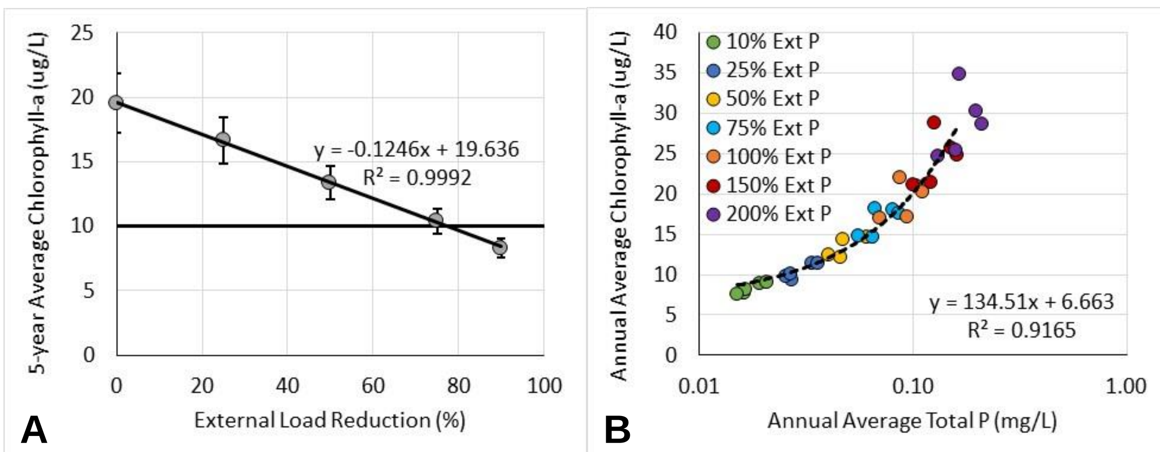


Figure 3-2. Model results from external P load reduction simulations. A) Effect of various percent P reductions from external sources on average chl-a concentrations at site W2 in Lake Wister, B) Relationship between TP concentration and chl-a concentrations at site W2 in Lake Wister across each of the five year model years in response to external P load reductions and 1.5x and 2x simulated P load increases.

The change in chl-a in response to P loading is caused by the dependence of algal biomass on water column P concentrations. These simulated decreases in external P loading (25% - 90% external P load reduction) resulted in significant decreases in the annual average TP concentration observed at site W2. Additionally, the simulated increases in external P loading resulted in significant increases in the annual average TP concentrations at site W2. The change in annual TP concentrations at site W2 across all of these simulated conditions resulted in major changes in the annual chl-a concentrations at site W2 as well (Figure 3-2B).

The modeled TP versus chl-a relationship shows that both simulated external P reductions and increases affected chl-a concentrations. This suggests that external P loading contributes significantly to controlling the trophic state of Lake Wister and that external P load reductions should be an effective tool for decreasing the long-term average chl-a concentration in the lake.

3.1.3 Combined Internal and Watershed Phosphorus Loading –We also evaluated the effect of simultaneous reductions in both internal and external P loading. For these simulations, sediment P release was reduced to 10% of current values in order to simulate a 90% reduction in internal P loading to the lake. Pilot studies conducted by PVIA in Quarry Island Cove at Lake Wister suggest this is potentially achievable (PVIA 2016). Along with the 90% reduction in internal P loading, we ran five separate simulations on the simultaneous effect of 25%, 50%, 75%, and 90% reductions in external P loading from watershed sources by reducing the SRP, DOP, POP, and PIP concentrations to represent these load reduction scenarios. Thus, we evaluated a 90% internal P reduction across the same gradient of external P reduction scenarios as presented above.

The simulated five-year average chl-a concentrations decreased with increasing load reductions from external and internal sources (Figure 3-3). When 25% - 90% external load reductions were evaluated along with a 90% reduction in internal loads, the resulting five year average chl-a concentrations decreased. The rate of chl-a decrease remained almost constant with external P reductions, but the **five-year average chl-a concentration was consistently 3 µg/L less when a 90% internal load reduction** was simulated on top of the external load reductions (Figure 3-3A). These simulated decreases in combined external and internal P loading resulted in significant decreases in the annual average TP concentration observed at site W2. The change in annual TP

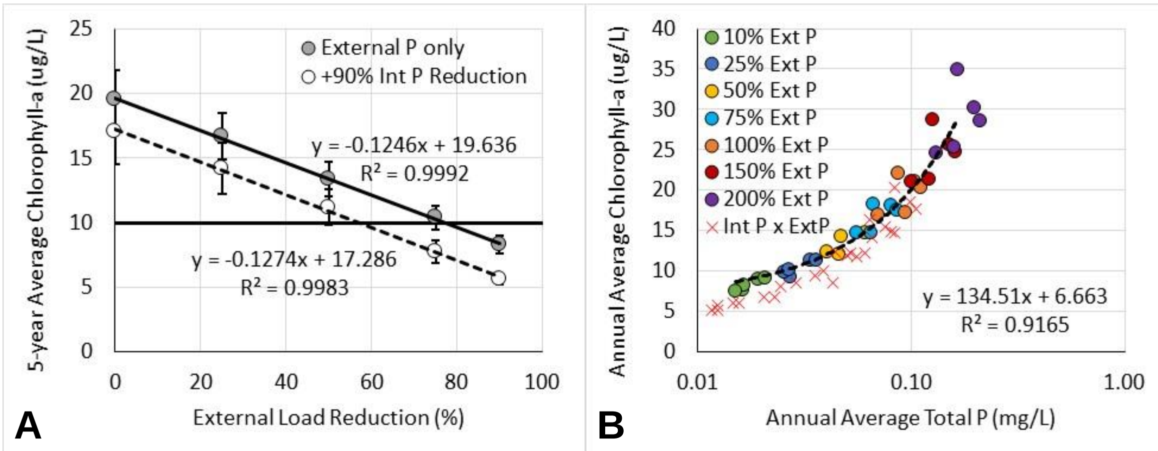


Figure 3-3. Model results from simultaneous external P load reduction and 90% internal P load reductions. A) Effect of various percent P reductions from external and internal sources on average chl-a concentrations at site W2 in Lake Wister, B) Relationship between TP concentration and chl-a concentrations at site W2 in Lake Wister across each of the five year model years in response to external and internal P load reductions and 1.5x and 2x simulated external P load increases.

concentrations at site W2 across all of these simulated conditions resulted in major changes in the annual chl-a concentrations at site W2 as well (Figure 3-3B).

The variability in the TP versus chl-a relationship shows that although external load reductions alone resulted in the most dramatic improvement to trophic state, the internal load reductions provided a significant addition to the external P reductions. Whereas a **78% reduction** in the average external P load is required to achieve a 10µg/L chl-a average (Figure 3-2A), **only a 58% watershed load reduction** would be required in conjunction with the modeled 90% reduction in internal loading (Figure 3-3A).

3.1.4 Watershed Nitrogen Loading – In order to simulate the potential for watershed management to reduce both the P and N concentrations, and ultimately chl-a concentrations in Lake Wister, the water quality input files for the Fourche-Maline Creek and Poteau River were directly manipulated. The NH_4-N, NO_2+NO_3-N, DON, and PON concentrations in these files were reduced to 10% of current values and the model was run with all other conditions being the same as the final calibrated model in order to simulate a 90% reduction in N loading to the lake. In another simulation, we reduced watershed inputs to 25% of current values to simulate a 75% reduction in N input from the watershed. The process was repeated with NH_4-N, NO_2+NO_3-N, DON, and PON concentrations from both rivers to simulate 50% and 25% reductions in watershed N loading, respectively. We also evaluated the effect of an increase in N loading by increasing the N concentrations of all forms by 150% and 200%.

The simulated five year average chl-a concentrations decreased slightly with increasing external N load reductions to the model (Figure 3-4), showing the potential for N to limit phytoplankton growth at extremely low N loads. However, the magnitude of change in the five year average chl-a concentration was much less for the simulated external N load reductions than what was observed from the simulated external P load reductions. In fact, almost no reduction in the five year average chl-a concentration was apparent until at least 75% - 90% of the external N load was eliminated, with only a 2 – 5 µg/L decrease in chl-a observed at those magnitudes, respectively (Figure 3-4A). Also, and importantly, the simulated increases in external N loading increased TN in the water column but did not increase chl-a concentrations proportionally (Figure 3-4B).

ELCOM-CAEDYM, like all other water quality models typically assumes 100% N fixation efficiency. In other words, the model assumes that when phytoplankton become N-limited, certain phytoplankton species become dominant that convert N2 gas into biologically reactive N. Although this does occur in nature, new research has indicated that it may not be as efficient as previously thought (Scott and McCarthy 2010, Paerl et al. 2016). But water quality models have not in general been updated to reflect this

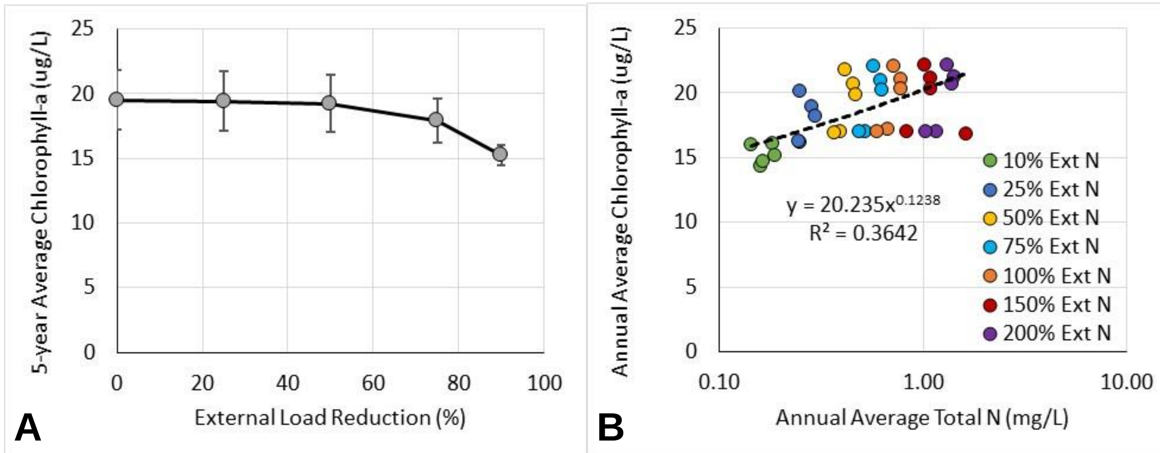


Figure 3-4. Model results from external N load reduction simulations. A) Effect of various percent N reductions from external sources on average chl-a concentrations at site W2 in Lake Wister, B) Relationship between TN concentration and chl-a concentrations at site W2 in Lake Wister across each of the five year model years in response to external N load reductions and 1.5x and 2x simulated N load increases.

changing paradigm. On the other hand, lake mass balance calculations (See Section 3.3, below) indicated that on an annual basis Lake Wister sometimes exports more nitrogen that it receives from its watershed. Further complicating the situation, empirical monitoring data show that Lake Wister, like many regional reservoirs, becomes N limited at times during the summer growing season. Thus, increases in N inputs at that time could increase phytoplankton production.

3.1.5 Combined Watershed Phosphorus and Nitrogen Loading – In order to simulate the potential for watershed management that reduced both P and N concentrations to reduce chl-a concentrations in Lake Wister, the water quality input files for the Fourche-Maline Creek and Poteau River were directly manipulated. The SRP, DOP, POP, PIP, NH_4-N, NO_2+NO_3-N, DON, and PON concentrations in these files were reduced to 10% of current values and the model was run with all other conditions being the same as the final calibrated model in order to simulate a 90% reduction in P and N loading to the lake. In another simulation, we reduced watershed inputs to 25% of current values to simulate a 75% reduction in P and N input from the watershed. The process was repeated with SRP, DOP, POP, PIP, NH_4-N, NO_2+NO_3-N, DON, and PON concentrations from both rivers to simulate 50% and 25% reductions in watershed P and N loading, respectively. We also evaluated the effect of an increase in P and N loading by increasing the N concentrations of all forms by 150% and 200%.

The simulated five-year average chl-a concentrations decreased with decreasing external P and N loads (Figure 3-5). The magnitude of change in the five-year average chl-a concentration was similar in magnitude to the change observed from the P only external load reductions. **The five-year average chl-a concentration decreased by 0.12 µg/L for every 1% reduction in external P and N load** (Figure 3-5). These simulated decreases in external P and N loading (25% - 90% external P and N load reduction) resulted in significant decreases in the annual average TP and TN concentrations observed at site W2. Additionally, the simulated increases in external P and N loading resulted in significant increases in the annual average TP and TN concentrations at site W2. The change in annual TP and TN concentrations at site W2 across all of these simulated conditions resulted in major changes in the annual chl-a concentrations at site W2 as well (Figure 3-6).

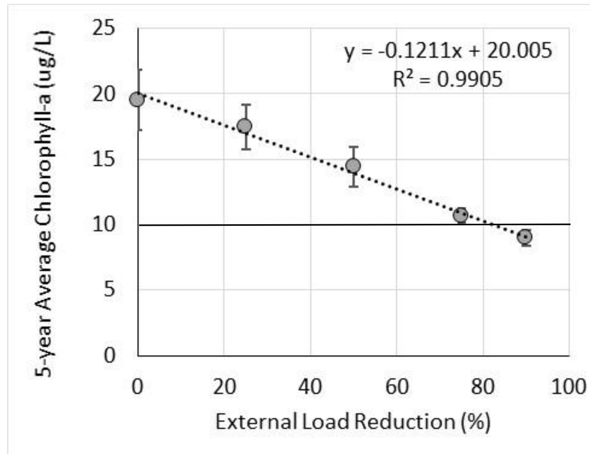


Figure 3-5. Model results showing effect of combined P and N reductions from external sources on average chl-a concentrations at site W2 in Lake Wister.

Although these results imply that both P and N reductions were useful for reducing the five-year average chl-a concentration, the magnitude of response for these combined P and N simulations was virtually identical to the magnitude of response observed for P-only simulations (Figure 3-2). Although these results indicate that the primary management for meeting the chl-a criteria in Lake Wister is reducing TP loading to the lake, they do not necessarily indicate that TN reductions cannot be helpful. Indeed, TN reductions alone made a small but measurable change in five-year average chl-a. Perhaps more importantly, Lake Wister like other regional reservoirs experiences seasonal N limitation of phytoplankton, a nuance that may not have been well captured by the model. Scott et al. (2019) found that many lakes across the US are experiencing summer N deficiencies, and that these lakes are also generally losing reactive N through denitrification. Thus, the N cycle has a natural process to remove N and keep lakes less productive. Warm-water lakes in the south-central US are known to experience annually-perpetuated seasonal N limitation in summer (Scott et al. 2009, Scott and Grantz 2013). Both the modeling the monitoring data from this project support a similar pattern for Lake Wister. Managing for N will further constrain the symptoms of eutrophication and is recommended for sustainable eutrophication management, particularly in summer.

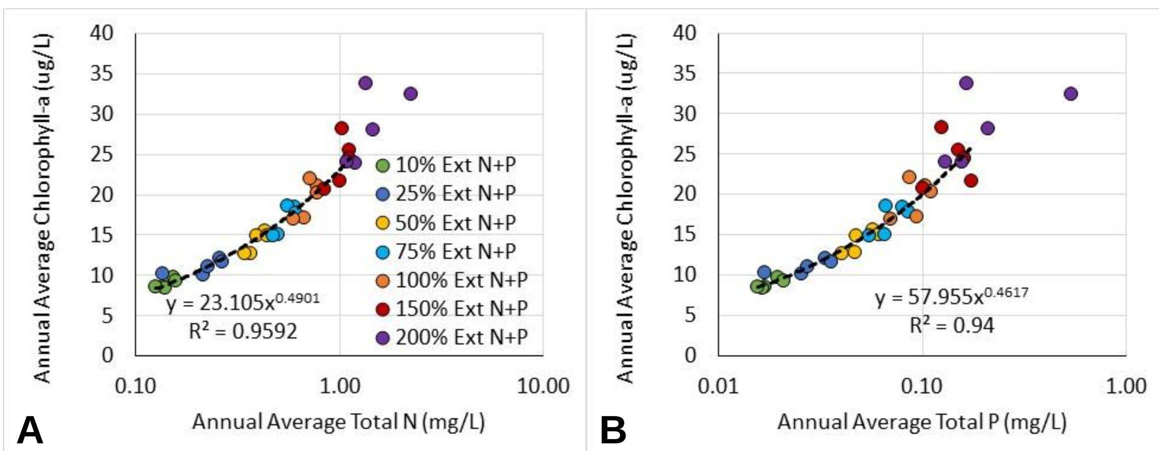


Figure 3-6. Model results from external P and N load reduction simulations. A) Relationship between TN concentration and chl-a concentrations at site W2 in Lake Wister across each of the five year model years in response to external P and N load reductions and 1.5x and 2x simulated P and N load increases, B) Relationship between TP concentration and chl-a concentrations at site W2 in Lake Wister across each of the five year model years in response to external P and N load reductions and 1.5x and 2x simulated P and N load increases.

3.2 Suspended Solids

We examined the potential for both internal and external reductions in TSS to improve turbidity conditions at Lake Wister. To evaluate the potential effects of in-lake management to reduce the amount of sediment resuspension in the lake and therefore reduce turbidity levels, we manipulated model input to simulate decreased sediment resuspension by decreasing the wind velocity inputs. We also conducted simulations of increased sediment resuspension to the lake by increasing wind velocities. Each of these simulated conditions were modeled across the entire five-year modeling timeframe. To evaluate the effect of altered suspended solid quantities entering the lake from the watershed, we manipulated TSS concentrations in lake inputs.

While modeled relationships are based on TSS, the Oklahoma Water Quality Standard is based on turbidity (Section 1.2). To address this, TSS concentrations produced by the model were converted into turbidity values based on the empirical relationship between turbidity and TSS derived from PVIA monitoring data (Figure 2-12). Thus, we were able to evaluate how both in-lake management scenarios and watershed sediment load reductions could influence TSS concentrations in model output and specifically how that may translate into turbidity reductions for comparison to water quality standards.

3.2.1 Wave Energy – In order to simulate the potential for in-lake management to reduce turbidity, the meteorological input files were directly manipulated. The wind speed in these files was reduced to 10% of current values and the model was run with all other conditions being the same as the final calibrated model in order to simulate a 90% reduction in wind energy intercepted by the lake. In another simulation, we reduced wind speed to 25% of current values to simulate a 75% wind energy reduction. The process was repeated with current wind velocities to simulate a 50% and 25% wind energy reduction scenarios, respectively. We also evaluated the effect of an increase in wave energy by increasing the wind speed by 150% and 200%.

Simulated five-year average turbidity levels decreased as wind velocities were reduced in the model (Figure 3-7). **Average turbidity decreased by 0.085 NTU for every 1% reduction in wind velocity**. This is consistent with the observation that wave action within Lake Wister influences the suspension and resuspension of inorganic particles which in turn influences turbidity levels in the lake. Decreased wind velocity allows particle settling and therefore a reduction in lake turbidity levels. Thus, internal management strategies, such as wind break structures, have the potential to reduce turbidity in Lake Wister.

3.2.2 Watershed Sediment Loading – In order to simulate the potential for watershed management to reduce turbidity, the water quality input files for the Fourche-Maline Creek and Poteau River were directly manipulated. The SS1 and SS2 concentrations in these files were reduced to 10% of current values and the model was run with all other conditions being the same as the final calibrated model in order to simulate a 90% reduction in suspended sediment loading to the lake. In another simulation, we reduced watershed inputs to 25% of current values to simulate a 75% reduction in suspended sediment input from the watershed. The process was repeated with 50% reduction and 75% reduction of current SS1 and SS2 concentrations from both rivers to simulate 50% and 25% reductions in watershed sediment loading, respectively. We also evaluated the effect of an increase in sediment loading from watershed by increasing the SS1 and SS2 by 150% and 200%.

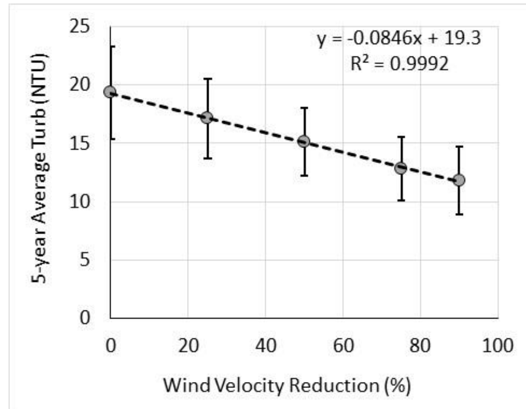


Figure 3-7. Model results showing effect of reductions of internal wave velocities caused by decreased wind velocities on average turbidity at site W2 in Lake Wister.

Simulated five-year average turbidity levels decreased with increasing external sediment load reductions in the model (Figure 3-8). **Long-term average turbidity decreased by 0.2 NTU for every 1% decrease in external sediment load.** Thus, external sediment reductions were more than twice as effective at reducing long-term average turbidity levels in the lake compared to internal reductions. Therefore, watershed management strategies that reduce the supply of suspended sediment to the lake have the potential to reduce turbidity in Lake Wister.

The model simulated average TSS conditions in the lake better than it simulated the magnitude of specific events (Section 2.5.5). However, in contrast to chl-a where assessment of water quality standards is based on an average of monitoring data, the Oklahoma turbidity standard is based on values observed during events (Section 1.2). Specifically, no more than 10% of monitored values should exceed 25 NTU.

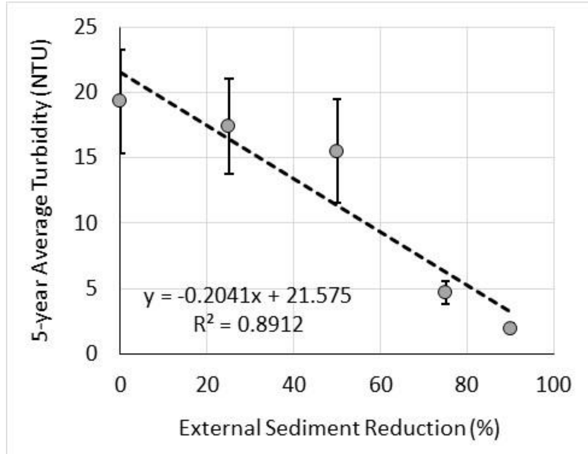


Figure 3-8. Model results showing effect of external sediment load reduction on average turbidity at site W2 in Lake Wister.

The average observed turbidity at Lake Wister over five years of monitoring was 26.1 ± 15.9, only slightly above the 25 NTU threshold. However, 123 of 287 individual samples exceeded 25 NTU (43%; Section 1.3). It is clear that a reduction in the frequency of high turbidity events and of turbidity levels during high turbidity events is desirable. However, relating modeled reductions in average TSS loads to average in-lake turbidity is unproductive because that latter average is only slightly above the 25 NTU standard.

We focused, therefore, on the measured turbidity events that exceeded the standard, rather than on the average of all events. The average value of the 123 in-lake turbidity samples that exceeded 25 NTU was 39.2 NTU. Therefore, the average magnitude of exceedance of the 25 NTU standard was 14.2 NTU. Load reduction simulations show that in-lake average turbidity decreased by 0.2 NTU for every 1% reduction in external sediment load. For example, a 10% reduction in external TSS supplied to the lake would produce a 2 NTU reduction in average in-lake turbidity (Figure 3-8).

Based on the combined analysis from the monitoring data showing that the average magnitude of exceedance was 14.2 NTU and the modeled decrease of 0.2 NTUs per 1% reduction in TSS loading, a 71% reduction in the external TSS load supplied to the lake would reduce average in-lake turbidity such that no more than 10% of samples would exceed 25 NTU (for this data set, and assuming the same patterns in future data collection) (14.2 NTU ÷ 0.2 NTU/% load reduction = 71 % load reduction).

In practice, high turbidity runoff events that exceed 25 NTU would be expected, but this load reduction would reduce their frequency and magnitude. As discussed further in Section 5, below, future monitoring and analysis of sediment data may allow an improvement in model sediment and turbidity forecasting and a refinement of this load reduction goal.

Table 3-1. Nutrient and suspended solids mass balance.

Year	TN (kg)	TP (kg)	TSS (Mg)	TN (kg)	TP (kg)	TSS (Mg)	TN (kg)	TP (kg)	TSS (Mg)
	INFLOW			OUTFLOW			RETENTION		
2011	1,120,951	199,048	131,745	1,317,240	221,918	33,486	-196,289	-22,870	98,259
2012	597,541	97,279	50,955	713,243	90,931	11,007	-115,702	6,348	39,948
2013	1,263,792	209,387	128,828	1,266,017	179,548	19,448	-2,225	29,839	109,380
2014	721,741	110,410	62,713	705,652	90,674	9,078	16,089	19,736	53,635
2015	2,712,709	492,813	338,558	2,111,032	327,299	37,062	601,677	165,514	301,495
Total	6,416,734	1,108,937	712,800	6,113,183	910,370	110,082	303,551	198,567	602,719
Avg.	1,283,347	221,787	142,560	1,222,637	182,074	22,016	60,710	39,713	120,544

3.3 Lake Mass Balance

By using Corps of Engineer outflow data (Section 2.3.3) and water quality simulated for the near-dam sampling site (W2, Figure 1-2), we calculated a set of annual mass balances for Lake Wister. Table 3-1 shows the annual outflow of nutrients and sediments from the lake as well as loads entering. This allows a calculation the net retention of nutrients and sediments in the lake. Interestingly, as noted above, the lake sometimes exports more nitrogen than it receives. Although the cause of this phenomenon was unknown and beyond the scope of this study, some long-term forcing factor such as climate cycles or dynamic biogeochemical equilibrium are likely involved. Sediment accumulation is considerable. The phosphorus retained becomes a potential source of ongoing internal P production.

Section 4 – Point Source Loads

Table 4-1. NPDES permit information for Arkansas and Oklahoma WWTPs.

NPDES	ID	Permitted Discharge (MGD)	Permitted Total P (mg/L)	Permitted TSS (mg/L)
OK0038407	Heavener	0.5	2.0	15-30
OK0022951	Hamilton	0.08	---[2]	90
OK0031828	Cedar	0.024	---	30
OK0021881	Wilburton	0.75	2.0	30
OK0031631	Red Oak	0.090[1]	---	90
AR0038482	Tyson	1.25	1.5	15
AR0035769	Waldron	0.85	1.0	15

[1]No permitted discharge June 1 – October 31
[2]Required monitoring of total P

We evaluated the nutrient and suspended solids loading to Lake Wister from permitted dischargers in the Lake Wister watershed by analyzing the National Pollution Discharge Elimination System (NPDES) reports submitted to the Oklahoma and Arkansas Departments of Environmental Quality. There are seven NPDES permitted dischargers in the Lake Wister watershed, with two major dischargers located in Arkansas, and three major and two minor NPDES permitted dischargers located in Oklahoma (Figure 1-5, Table 4-1, and Table 4-2). Major discharger means annual average TP load or permitted TP load greater than 1% of TMDL (438 kg/year TP).

Four of seven dischargers in Oklahoma and Arkansas--the City of Heavener (OK) Utility Authority, the City of Wilburton (OK) Public Works Authority, the City of Waldron (AR), and the Tyson Processing Plant in Waldron, AR--have water quality discharge limits for both TP and TSS (Figure 4-1). Hamilton Correctional Center, the USDA Forest Service Cedar Creek Recreation Area, and the City of Red Oak Public Works Authority have water quality discharge limits for TSS, but not for TP.

Self-reporting data from the dischargers with TP limits included the monthly discharge rate (30 day average million gallons per day), TP concentrations (30 day average mg/L), and TSS concentrations (30 day average mg/L). The self-reporting data from the dischargers with no TP limits included the monthly discharge rate and the TSS concentrations. We compiled these data for all five discharges and computed annual TP and TSS loads for 2011 – 2015 (Table 4-2). For the discharges which had no TP monitoring data, we assumed that the TP concentrations in these discharges was equal to the average TP concentration from the major dischargers across the five-year period of record during the study.

The annual loads of TP and TSS from the NPDES permitted dischargers for 2011 – 2015 are shown in Table 4-2. On average, these WWTPs contributed 5,831 kg TP per year, 16 kg per day. Given that the average annual load of TP to the lake during this time period was 221,787 kg/yr, the WWTPs on average contributed 2.6% (with a range from 1.3 – 5.5%) of the TP to Lake Wister.

WWTPs contributed 34,349 kg TSS per year to Lake Wister from 2011-2015. Given that the average annual load of TSS to the lake between 2011 and 2015 was 142,560,053 kg/yr, the WWTPs on average contributed less than 0.1% of the TSS load to Lake Wister.

Table 4-2. Annual Total P and TSS loads from Arkansas and Oklahoma WWTPs.

Plant/Year	2011	2012	2013	2014	2015	TOTAL	AVERAGE
Total Phosphorus (kg)							
Cedar	41.8	16.7	9.1	14.7	14.8	**97.0**	**19.4**
Hamilton	322.2	422.5	723.3	1033	953.6	**3454**	**691**
Heavener	1221	2575	1591	2840	3946	**12172**	**2434**
Red Oak	213.7	261.7	72.7	0.0	0.0	**548.1**	**109**
Tyson	1257	1380	1556	1468	709.0	**6370**	**1274**
Waldron	1109	1774	962.6	391.3	176.3	**4413**	**882**
Wilburton	459.9	408.7	512.4	333.2	383.1	**2097**	**419**
TOTAL	**4624**	**6839**	**5427**	**6081**	**6183**	**29153**	**5830**
Total Suspended Solids (kg)							
Cedar	97.1	27.1	16.1	19.3	22.5	**182.0**	**36.4**
Hamilton	1630	2257	1688	3041	4470	**13086**	**2617**
Heavener	15263	21492	20245	10578	24300	**91878**	**18375**
Red Oak	903.7	1199	1011	0.0	0.0	**3114**	**622**
Tyson	7455	8312	7225	3661	4484	**31136**	**6227**
Waldron	2524	1269	3607	1969	4453	**13822**	**2764**
Wilburton	3576	4058	4017	3077	3800	**18527**	**3705**
TOTAL	**31449**	**38614**	**37809**	**22344**	**41529**	**171745**	**34349**

Total P discharges from two facilities, Hamilton Correctional and the City of Heavener tripled during the five years analyzed here.

Section 5 – Total Maximum Daily Loads

Model simulations support the establishment of two Total Maximum Daily Loads (TMDLs) for Lake Wister, one for Total Phosphorus (TP) and the second for Total Suspended Solids (TSS).

The annual loads of TP to Lake Wister over the five-year modeling period are shown in Table 5-1. The average annual load for the five-year modeling period was 221,787 kg/yr. Annual loads ranged from 97,279 to 492,813 kg/yr. Model simulations showed that a 78% reduction in the average annual load is required to produce an average chl-a concentration of 10 µg/L in the lake. This results in a target TMDL TP load of 48,793 kg/year. An additional 10% reduction may be calculated as a margin of safety.

Table 5-1. Target load recommendations for TP for Lake Wister.

Year	Total P Load (kg/yr)	TMDL (Annual Basis) @ a 78% Reduction (kg/yr)	10% Margin of Safety (kg/yr)	Target Annual Load (kg/yr)	Target Daily Load (kg)
2011	199,048				
2012	97,279				
2013	209,387				
2014	110,410				
2015	492,813				
Average	**221,787**	**48,793**	**4,879**	**43,914**	**120**

These computed loads are for the external watersheds loads to the lake and do not include internal loads. Although we did model the effect of reduced internal loads on chl-a concentrations, we did not include an internal load reduction recommendation in the TMDL calculation. We chose a conservative approach in which the TMDL is based solely on external load reductions. As described in the main body of the report, *if* internal load reductions are successful, they have the potential to reduce the required external load reductions presented in the TMDL.

The annual loads of TSS to Lake Wister over the five-year modeling period are shown in Table 5-2. The average annual load for the five-year modeling period was 142,560,053 kg/yr. Annual loads ranged from 50,955,204 to 338,558,256 kg/yr. Average outflow TSS loads were 22,016,319 kg, resulting in the lake retaining approximately 120,543,734 kg/yr. Model simulations showed that a 71% reduction in the average annual load is required to produce an average turbidity so that less than 10% of observations would exceed 25 NTU yearly over the five-year modeling period. This results in a target TMDL TSS load of 41,342,415 kg/year. An additional 10% reduction may be calculated as a margin of safety.

Table 5-2. Target load recommendations for TSS for Lake Wister.

Year	TSS Load (kg/yr)	TMDL (Annual Basis) @ a 71% Reduction (kg/yr)	10% Margin of Safety (kg/yr)	Target Annual Load (kg/yr)	Target Daily Load (kg)
2011	131,745,019				
2012	50,955,204				
2013	128,828,296				
2014	62,713,491				
2015	338,558,256				
Average	**142,560,053**	**41,342,415**	**4,134,242**	**37,208,174**	**101,940**

5.1 Waste Load Allocation

Three of the seven permitted NPDES dischargers in the Lake Wister watershed do not have TP limits. The Arkansas dischargers, the City of Waldron and Tyson Foods Inc., have TP discharge concentration limits of 1.0 and 1.5 mg/L, respectively. The Oklahoma discharge limits, where they exist, are 2 mg/L.

Permitted point source dischargers in the Lake Wister watershed contributed an average 5,831 kg TP per year. This is approximately 2.6% (with a range from 1.3 – 5.5%) of the average 221,787 kg/yr TP load to Lake Wister.

While the point source contribution of phosphorus to Lake Wister is small, it is not inconsequential. If Oklahoma major dischargers adopted and achieved a 1 mg/L TP discharge limit at their design flow, the TP load to Lake Wister would decrease by an average of 1,706 kg/yr, which is approximately 1% of the current total phosphorus load. As noted (Section 3.1.2), a 1% reduction in the total phosphorus load to the lake will result in a decrease in the long-term average chlorophyll- concentrations in the lake of 0.12 µg/L.

Further, the contribution of each individual poultry farm in the watershed is also small. Each of them must, however, analyze their soil for its phosphorus concentration, and limit their application of phosphorus-bearing litter accordingly. Therefore,

- All major point source dischargers should have a discharge permit limit of 1 mg/L TP, or less;
- The USDA Forest Service Cedar Creek Recreation Area comprises 0.04% of TP TMDL and the Red Oak Public Works Authority comprises 0.25%. The USDA Forest Service Cedar Creek Recreation Area and the Red Oak Public Works Authority have a very small contribution to the TP load, so they will not be included as part of the WLA.

The adoption of a 1 mg/L standard by Oklahoma dischargers would decrease the TP load to Lake Wister by 4.7 kg/day, or 3.9% of the of the 120 kg/day total phosphorus TMDL value. (The Wilburton discharge, though permitted at 2 mg/L, already achieves a TP discharge of less than 1 mg/L with a five-year average of 0.73 mg/L).

Implementing 1 mg/L TP concentration discharge limit for Lake Wister watershed major dischargers results in a Waste Load Allocation (WLA) of 13.4 kg TP/day, 11.2% of the total load (Table 5-3).

Table 5-3. Load allocations for recommended TMDLs for Lake Wister.

	Total Phosphorus TMDL (kg/day)	% Total Phosphorus TMDL	Total Suspended Solids TMDL (kg/day)	% TSS Total Load
Waste Load Allocation	13.4	11.2	321.8	0.3
Load Allocation	94.6	78.8	91,339.5	89.7
MOS	12.0	10.0	10,184.6	10.0
Total	120.0	100.0	101,845.9	100.0

A similar exercise was not included for TSS because the waste load currently represents less than 0.1% of the total TSS load to Lake Wister. The five-year average discharge for TSS from the five Oklahoma WWTPs was 17.2 mg/L, well below the typical 30 mg/L limit for most NPDES permits. More importantly, the relatively small hydrologic contribution of these dischargers, along with the relatively low TSS concentrations, makes the TSS load from WWTPs exceptionally low relative (<0.1% of total) to non-point source loads (>99.9 % of total).

5.2 Comparison to Previous Estimates

These load estimates and load reduction goals are interesting when compared to past load estimates. In 1996, as part of a Clean Lakes Study, the TP load to the lake was estimated to be 190,000 kg/yr, that is, some 30,000 kg/year less than the average for the last five years (OWRB 1996). At the same time, point source discharges of TP were estimated to be 20,500 kg/yr, or approximately 11% of the total load (OWRB 1996). In 2004, following implementation of phosphorus controls at some point source discharges in the watershed, the point source load was estimated to have been reduced to 7,360 kg TP/yr, a number closer to the current estimate. Nonpoint source loads did not decline during the same period, and may have increased (OWRB 2004).

5.3 State of Arkansas Contributions

As noted in Section 1.1, the Poteau River begins in Arkansas and flows west to Lake Wister. There are approximately equal numbers of poultry houses in the Arkansas portion of the watershed as in the Oklahoma (Section 1-4). Two major point source dischargers are located in Arkansas (Section 4). The load monitoring conducted by the USGS for PVIA (Section 2.3.4) includes sampling on the Poteau River at Loving (OK) (07247015), approximately two miles west of the Oklahoma-Arkansas state border. According to USGS modeling of nutrient and sediment loads in the Poteau River, approximately 55% of the TP load and 35% of the TSS load entering Lake Wister originates upstream of the Loving location (USGS 2013).

Section 6 – Conclusions and Recommendations

The goals of this project were to: 1) develop an accurately calibrated water quality modeling tool for Lake Wister, Oklahoma, 2) establish load reduction goals, where applicable, to bring the lake into compliance with Oklahoma water quality standards, and 3) evaluate the potential effectiveness of watershed and in-lake management options to improve water quality. We developed a three-dimensional lake model using the ELCOM-CAEDYM platform. The model relied on input data derived from the USGS, USACE, Oklahoma Mesonet, and Lake Wister water quality data collected by PVIA. The model was calibrated within the limits prescribed by the ODEQ and based on the best professional judgement of the modeling team.

6.1 Model Performance

Overall, the ELCOM-CAEDYM Lake Wister model performed well with respect to calibration and validation metrics for most of the variables of interest. In particular, we were able to accurately simulate the relatively weak thermal stratification that occurs in Lake Wister (Figure 2-13, 2-14), which results in a general pattern of hypoxia/anoxia in the lower water column (Figure 2-16) that can be interrupted by storm- or wind-driven disturbance. Accurately modeling oxygen dynamics in response to lake mixing is of tremendous importance in Lake Wister, and most lake models, because redox conditions at the sediment water interface are a strong control on internal phosphorus delivery to the water column. We believe this model is a strong improvement over previous modeling efforts for the lake that did not achieve success in modeling oxygen dynamics and used very large P release coefficients which were not representative of realistic conditions in the lake (Haggard et al. 2012).

We specifically targeted accurate predictions of chl-a and dissolved nutrients. Because lake trophic state, as measured by long-term average phytoplankton biomass (i.e., chl-a) is an indirect effect of nutrient loading, we were particularly careful to insure that the chl-a calibration statistics were well within acceptable limits. The relative RMSE for chl-a from both the calibration (0.6) and validation (0.9) periods was more than one order of magnitude less than the target calibration threshold (Figure 2-23). We were also particularly interested in the calibration of dissolved nutrients, where seasonal variation created patterns in dissolved nutrients that were repeated through time. The strongest example of this condition are the nitrate concentrations at site W2 (Figure 2-19), which show a general pattern of maximum concentrations during spring and minimum concentrations in late summer/fall. Although the RMSE statistic for nitrate for the calibration and validation data were not particularly strong, the R statistic shows that the modeled and measured data were strongly correlated, particularly when nitrate concentrations were above the laboratory minimum detection level in the measured data. A similar explanation can be used when evaluating soluble reactive phosphorus (Figure 2-22) and ammonia concentrations (Figure 2-20). However, the model tended to underestimate ammonia concentrations following overturn in some years. Predicting dissolved nutrients with strong accuracy is important because the drawdown of these nutrient forms signifies that phytoplankton growth becomes nutrient-limited in the lake. As such, any management actions that lead to decreases in nutrient concentrations are more likely to decrease the amount of phytoplankton biomass observed in the lake.

Although we were satisfied with model performance in this project, the model did have several shortcomings. Overall, the model performed best at predicting average conditions observed in the lake and less well at predicting extreme conditions and/or the values associated with specific events. This is evident with respect to both chl-a and TSS. First, the model predicted chlorophyll-a concentrations very accurately according to the RMSE statistics, but did not predict the 3 bloom events in which chlorophyll-a at site W2 exceeded 50 µg/L (Figure 2-23). This shortcoming is reflected in the relatively low R value for predicted versus measured chlorophyll-a and in the scatterplot showing that the greatest measured values were did not correspond with equivalent model predictions. Second, the model tended to be overly sensitive with regard to flow-dependent TSS concentrations, as indicated by the large overestimation of TSS following storm events and the less frequent underestimation of TSS following prolonged periods of drought (Summer-Fall 2011, 2012, 2015; Figure 2-17). This effect is most likely a function of our inability to assign appropriate size classes to suspended sediment, which controls the duration in which sediments remain suspended due to specific settling velocity (see Section 6.3 below for additional information on this data gap).

6.2 Wister Lake Processes

While developing TMDL goals for water quality improvement at Lake Wister was the focus of the modeling project, we also learned things about the processes at work in the lake. These improvements in our understanding will help inform future management and restoration actions. For example, the variable frequency and extent of lake mixing at Lake Wister has often been observed. In some years the lake maintains stratification throughout the summer. In other years, the lake may mix several times. Exploring lake temperature and stream inflow temperatures in the model led to the understanding that the strength of stratification in the summer is set up by the temperature of large winter and spring inflow events. Large, cold water inflows set up stronger summer stratification, while warmer large inflows set up weaker.

The importance of watershed influences in creating conditions in the lake was also seen in both nutrient supply and lake turbidity. While internal P loading is important, its effect on chl-a concentrations is dwarfed by the importance of watershed loads. Actions to reduce internal loading will have benefits (Section 3.1.3), but model results emphasize the importance of remediation efforts in the watershed. In terms of turbidity, the lake's relative shallowness has led to the idea that resuspension of lake sediments was a key driver of turbidity in the lake. Model results suggests that the importance of loads of TSS delivered to the lake is greater than internal processes. Loads of sediment delivered to the lake are then *kept* in suspension because the lake is shallow. This process has a larger effect on lake conditions than resuspension of previously deposited sediments. Again, in-lake restoration actions will have benefits, but watershed work to reduce loads to the lake is essential.

6.3 Monitoring and Data Needs

The development of the Lake Wister model has also highlighted several places where routine monitoring parameters and lake model data requirements are not in sync. This gap is not just found in the Lake Wister model, but in most recent and current lake models being developed in the state. (These larger issues were explored in a special session on modeling improvement at the 2017 Oklahoma Clean Lakes and Watersheds Conference (OCLWA 2017)).

One specific need at Lake Wister is to develop a more refined understanding of the particle size distribution of sediment in both the inflowing streams and in the lake. The smaller clay particle size classes are not well-characterized by current analysis, and these small clays are particularly important to the persistence of turbid conditions in the lake because they are most easily kept in suspension.

The importance of water temperature has been discussed several times. We were forced to model inflow water temperatures based on limited data. Because Lake Wister is so sensitive to differences in water temperature, future efforts would benefit from continuous measurements of inflow water temperature and improved monitoring of in-lake water temperatures.

6.4 TMDL Summary

The monitoring data and model results showed that Lake Wister received an annual average of 221,787 kg TP per year through the Fourche-Maline Creek and Poteau River during the years 2011 – 2015. The model results indicated that these external loads were the strongest control on the long-term average chl-a concentration in Lake Wister, which was approximately 20 µg/L over those same years. Load reduction simulations conducted with the calibrated model indicated that the external P load will need to be reduced by 78% for the long-term average chl-a concentrations in Lake Wister to be below the 10 µg/L water quality standard associated with its public water supply designation. If significant internal load reductions were achieved through in-lake management options such as alum treatment, the external loads would only need to be reduced by 58%.

The monitoring data and model results demonstrate that Lake Wister received an annual average of 142.5 million kg (157,146 T) of TSS through the Fourche-Maline and Poteau River during the years 2011-2015. This load will need to be reduced by 71% to reduce the violations of the 25 NTU standard to below 10% of samples. The model results again indicated that external inputs were more responsible for driving the turbidity in Lake Wister than were internal dynamics. The model did indicate that reductions in wave energy within the lake have the potential to contribute to turbidity reductions.

The fact that the model did better with average conditions than specific events also affects our understanding on in-lake turbidity. Future modeling work may include modeling specific potential interventions such as breakwaters in conjunction with watershed analysis of the sources of sediment being supplied to the lake.

These load reduction goals set a high bar. However, lake modeling results also show that incremental improvements will benefit the lake. The average chl-a concentration in the lake decreased by 0.12 µg/L for every 1% increase in external P load reduction and long-term average turbidity decreased by 0.2 NTU in the lake for every 1% decrease in external sediment load. The degradation of water quality at Lake Wister occurred over several decades and improvement will likely take as long. Phosphorus load reductions averaging 2 to 3% per year will result in meeting water quality standards in 20 to 40 years. To improve Lake Wister, every little bit will help.

6.5 Implementation

The establishment of these TP and TSS TMDLs for Lake Wister is an important milestone in efforts to protect and restore water quality in the lake. As important as they are, they only set the stage for future actions that will be required. These TMDLs set the targets for future restoration. The next step will be to establish and plan for how loads may be reduced. What are the sources of nutrients and sediments? Where within the watershed are they located? What load-reducing techniques and technologies are available? What are their costs in relation to their effectiveness? These questions will be analyzed and addressed in a watershed based plan (US EPA 2008) that will be developed as the next step in the Lake Wister water quality restoration process. Some efforts toward this are already underway. PVIA, in cooperation with the Oklahoma Conservation Commission, sponsored a one-year sampling program at the small subwatershed (HUC 12) scale in the Oklahoma portion of the Wister watershed. This will help establish areas to prioritize in future watershed activities. The State of Arkansas is also currently pursuing new sampling efforts in the Arkansas portion of the Poteau River watershed and plans the development of a watershed based plan for that area in the near future. The information reported here in this lake modeling report provides a strong scientific basis on which these future activities may proceed.

Section 7 – Appendix

7.1 Flow Data Inputs – Measured data, models, and input information

Input data and models used to derive the data for river inputs to Lake Wister can be found at: http://www.pvia.org/lake-modeling-report-final/section-7-appendix/

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