Watson Lake TMDL Receiving Water Modeling

Prepared for

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August 2012

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1 Introduction

1.1 PROBLEM STATEMENT

EPA Region 9 has listed Watson Lake as water quality impaired for high nitrogen, low dissolved oxygen (DO), and high pH. The lake was first listed as impaired on the Arizona's 2004 303(d) list based on samples collected between 1996 and 2003. The Arizona Surface Water Quality Standards designate the uses of the lake as warm-water Aquatic and Wildlife Protection (A&Ww), Full Body Contact (FBC), Fish Consumption (FC), Agriculture Irrigation (AgI), and Agriculture Livestock (AgL). Watson Lake is categorized as an igneous lake. Table 1 lists the relevant numeric surface water standards for the lake according to the unofficial copy[1] of the 2009 surface water quality standards.

Table 1. Applicable Numeric Standards for Watson Lake (18 A.A.C., Chap 11, Art. 1; ADEQ, 2009)

Designated Use	Lake Category	Dissolved Oxygen	pH
Full Body Contact	Igneous	NA	6.5 to 9
Warm-Water Aquatic and Wildlife Protection	All except Urban	6 mg/L within top 1 meter depth	6.5 to 9

In addition to the numeric standards, the Arizona narrative surface water quality criteria state that:

> A surface water shall not contain pollutants in amounts or combinations that cause the growth of algae or aquatic plants that inhibit or prohibit the habitation, growth, or propagation of other aquatic life or that impair recreational uses.

Table 2 lists endpoint thresholds supporting narrative criteria applicable to Watson Lake (warm water lakes peak season, April – October). The narrative nutrient criteria refer to these ranges, stating that the narrative standard is met if the mean reservoir chlorophyll *a* concentration is less than the minimum threshold value, or the reservoir is within the target range for chlorophyll *a* and meets a list of additional criteria relating to the thresholds in Table 2 and qualitative observations of fish kills, nuisance algal blooms, and submerged aquatic vegetation (SAV). The Verde River nutrient criteria, listed in Table 3, apply to Granite Creek and its tributaries, upstream and downstream of Watson Lake.

Table 2. Thresholds Applied to Narrative Criteria, Applicable to Watson Lake (18 A.A.C., Chap 11, Art. 1; ADEQ, 2009)

Watson Lake Endpoints	Chlorophyll *a*	Secchi Depth	Blue Green Algae	Total Phosphorus	Total Nitrogen	Total Kjeldahl Nitrogen
Full Body Contact, Igneous	20 to 30 µg/L	0.5 to 1 meters	20,000 per mL	0.100 to 0.125 mg/L	1.5 to 1.7 mg/L	1.2 to 1.4 mg/L
Warm Water Aquatic and Wildlife, All Except Urban	24 to 40 µg/L	0.8 to 1 meters	NA	0.115 to 0.140 mg/L	1.6 to 1.8 mg/L	1.3 to 1.6 mg/L

[1]The Notice of Final Rulemaking was published in the Arizona Administrative Register on December 26, 2008. The Secretary of State has not yet posted the official rule.

Table 3. Nutrient Criteria for the Verde River and its Tributaries (18 A.A.C., Chap 11, Art. 1; ADEQ, 2009)

Nutrient	Annual Mean	90th Percentile	Single Sample Mean
Total Phosphorus (mg/L)	0.10	0.30	1.00
Total Nitrogen (mg/L)	1.00	1.50	3.00

A major source of impairment is thought to be historic wastewater effluent leaching from impoundments associated with the wastewater treatment plant adjacent to Granite Creek, occurring for decades prior to the mid-1980s. The impoundments were reconstructed and lined, and a pipeline was built to route effluent around the lake to recharge basins near the Prescott Airport. While this source no longer exists, it is expected that nutrients from this historic loading have accumulated in lake sediments. Additional sources of impairment include urban stormwater runoff influenced by fertilizer use, atmospheric deposition, domestic animal waste, streambank erosion, and other non-point nutrient sources. Existing wastewater-related sources include septic leach fields, sanitary sewer system leaks, and package treatment plants. Historic dumping and landfills are also a concern. Background sources of nutrient loading include forest fires and wildlife.

A significant presence of algae has been observed within the lake, and dominant algal genera observed by Arizona Department of Environmental Quality (ADEQ) in the last five years include the cyanobacteria (cyanophytes) *Anabaena, Cylindrospemopsis, Cylindrospermum, Gloeocapsa, Gloeotrichia,* and *Microcyctis*. Most recently, *Gloeotrichia* species have been observed to dominate the algal community during the growing season. This genus is a type of cyanobacteria, is nitrogen-fixing, and forms dense fibrous clusters within the water column.

Submerged aquatic vegetation (SAV) dominates the upper, shallower portion of the lake, with peak coverage occurring during July and August. Common names of plants observed include Coontail, Sago Pondweed, Leafy Pondweed, Chara, and Water Milfoil.

The lake provides a number of recreational uses, including boating, fishing, hiking, and bird watching; however, the City of Prescott does not allow swimming in the lake. The presence of algae and SAV directly impact these recreational uses, and low dissolved oxygen could be impacting game fish populations within the lake. Nutrient loading from the lake impacts the downstream concentrations within the Verde River, with both water supply and aquatic life uses.

The goal of the receiving water modeling was to estimate the lake nutrient balance and simulate algal response to support the determination of TMDLs for nitrogen, dissolved oxygen, and pH. The model approach sought to link the achievement of numeric and narrative standards both external and internal nutrient loading to the lake. Management scenarios considered how the multiple uses of the lake could be addressed while achieving the recommended reductions in nutrients loading.

1.2 SETTING

Granite Creek and its tributaries comprise the majority of the Watson Lake watershed (40 square mile drainage area). The lake and its entire watershed are located within Yavapai County, Arizona in the Upper Verde River Watershed (Figure 1). Watson lake drains a portion of the City of Prescott, and the lake itself is located about six miles northeast of the city center. The major land uses in the watershed are high density urban, low density urban, residential and commercial urban, and natural areas (forest and scrub-shrub). Land within the watershed is owned by the City of Prescott, Yavapai County, Prescott National Forest, the State of Arizona (State Trust lands), Yavapai-Prescott Tribe, and private landowners.

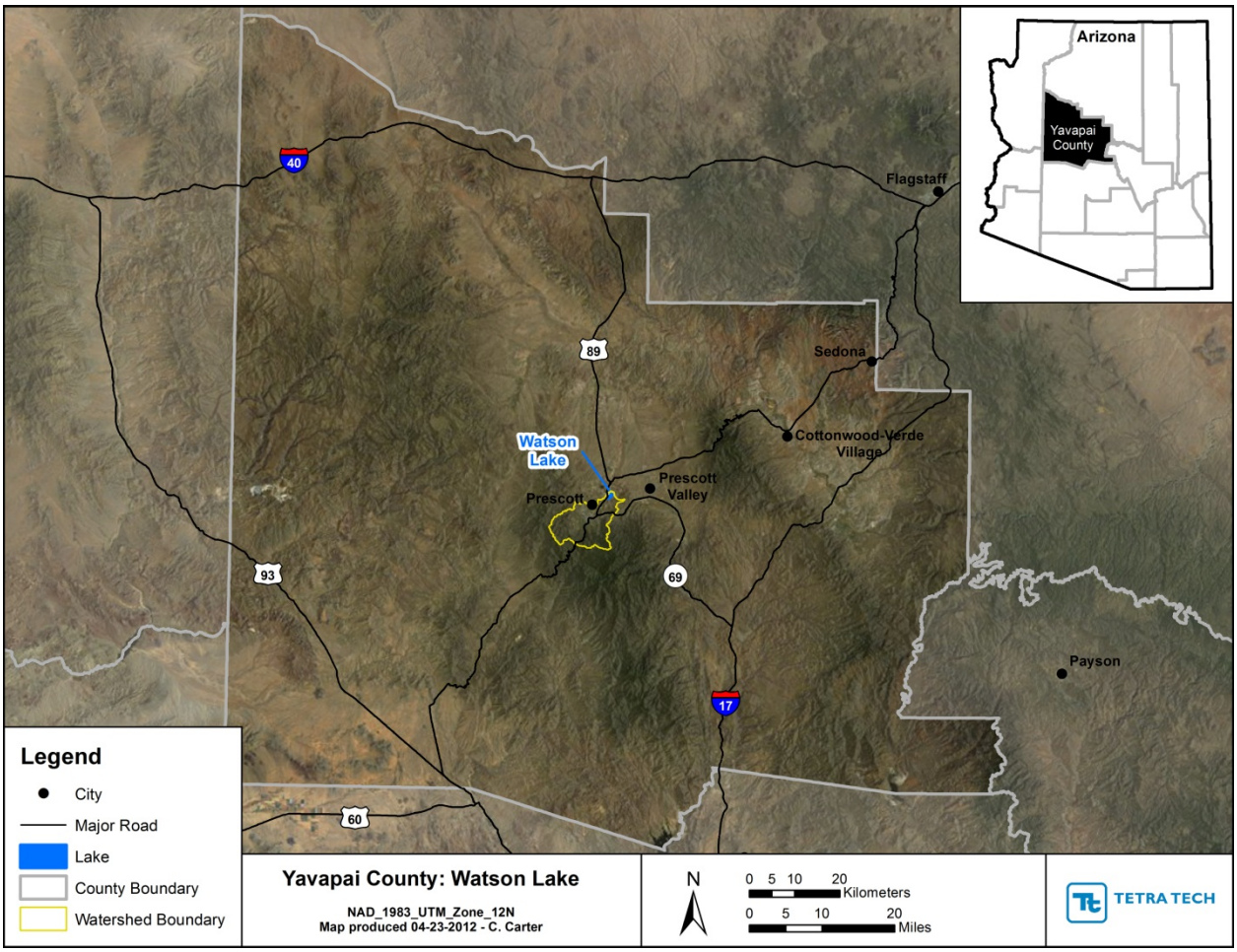

Figure 1. Watson Lake Vicinity

The surface area of Watson Lake is 192 acres at full pool. The upper portion of the lake where Granite Creek enters is shallow (average 1.7 meters) and supports the majority of SAV present in the lake. An overflow channel exists midway along the northern side of the shallow portion, which is used to passively route water to Willow Creek Reservoir during very wet years. The lower, deeper portion of the lake is fragmented by granite formations that form islands and peninsulas within the lake. The depth in this portion averages 5.7 meters with a maximum depth of 14.9 meters at full pool. The surface elevation of the lake is about 5,100 feet.

The City of Prescott manages lake levels, and water is periodically released from the dam for irrigation and recharge downstream. The city has a goal of maintaining a recreational pool of no less than 7 feet below the spillway, but this may not be possible in dry years. The lake is stocked for recreational fishing, and boating is allowed (water skiing and boat wakes are prohibited).

Watson Lake experiences significant fluctuation in water levels during each year as well as variation across years. The most precipitation and inflow occurs during winter and spring. Winter inflows to the lake consist of snowmelt and rain-on-snow. Precipitation declines in early summer until the lake experiences large storm events that occur during the summer monsoon season. Summer inflows to the lake are extremely flashy and highly variable with regard to the amount of runoff that reaches the lake. The lake levels respond to the seasonal variation in water inputs combined with periodic dam releases, overflow to Willow Creek reservoir, and evaporation due to the arid climate.

1.3 TMDL ENDPOINTS

The nutrient loading capacity for Watson Lake was determined based on achieving nutrient endpoints selected to represent low risk of impairment to the lake. A weight of evidence approach was used to select nutrient targets, involving review of current ADEQ standards and narrative endpoints, literature references, TMDL endpoints used for similar lakes, and other information sources. Table 4 summarizes the potential endpoints that were reviewed. While many endpoints listed in Table 4 are relevant to the nutrient ecoregion in which Watson Lake resides (Ecoregion II), some of the endpoints presented here were developed for temperate lakes and may not be relevant to lakes influenced by arid climates. These references were selected based on best available data.

Table 4. Relevant Endpoints for Watson Lake Nutrient Concentrations

Descriptions	TN (mg/L)	TP (mg/L)	Notes	Source
Full Body Contact, Current Narrative Threshold Numeric Translator	1.5 to 1.7 (TKN: 1.2 to 1.4)	0.100-0.125	Igneous lake category	Malcolm Pirnie (2005)
Warm water Aquatic and Wildlife Protection, Current Narrative Threshold Numeric Translator	1.6 to 1.8	0.115 to 0.140	All except urban category	Malcolm Pirnie (2005)
EPA Ecoregion Guidance	0.40 to 0.88	0.0125	Ambient Water Quality Criteria Recommendations, Lakes and Reservoirs Specific to sub-ER 23 (TN range reflects estimated versus calculated); estimate of natural.	USEPA (2000)
Literature on risk of cyanophyte dominance	NA	0.03	At TP above this value, Cyanobacteria are likely to dominate (>40% risk according to Downing et al.).	Watson et al. (1992), Downing et al. (2001)
CO Proposed Interim Standards	0.41 to 0.80	0.020 to 0.080	Cold to warm, lakes>25 acres	CO WQCC (2012)
Consensus on Mesotrophic Threshold	0.65	0.025	International	Welch and Jacoby (2004)
Equations relating TP or TN with Chlorophyll ***a***	NA	0.043 to 0.048	Based on Watson lower Chlorophyll *a* endpoint (20 ug/L); TP range reflects 3 sources. Dillon and Rigler: log chl *a*=1.449 log TP -1.136 (r^2=0.90) Jones and Bachmann: log chl *a*=1.46 log TP -1.09 (r^2=0.90)	Dillon and Rigler (1974), Jones and Bachmann (1976)

Descriptions	TN (mg/L)	TP (mg/L)	Notes	Source
Equation relating both TP and TN with Chlorophyll ***a***	0.32 to 0.87	0.058 to 0.089	Calculated separately for TN and TP keeping other constituent constant at observed annual average; calculated separately for lake segments 1 and 2;range reflects variability across both segments. Smith: log chl *a*=0.6531 log TP + 0.548 log TN -1.517 (r^2=0.76)	Smith (1982)
Rainbow Lake Endpoints	0.49 to 0.61	0.054 to 0.069	Similar elevation, and ecoregion; range of scenario results; Watson segment 1 similar depth.	Tetra Tech (1999)
Big Bear Lake Endpoints	NA	0.035	Similar elevation, ecoregion, and average/maximum depth	CARWCB (2006)
Verde River Numeric Endpoints	1	0.1	Annual mean	ADEQ (2009)
Lowest Measured Watershed Concentrations	0.3	0.04	Estimate by ADEQ based on measured data in upper watershed	NA

Algae uptake inorganic nutrients; however, various researchers have found that total nitrogen (TN) and total phosphorus (TP) are better predictors of eutrophication response because organic forms are cycled back to inorganic forms by bacteria and can then be rapidly taken up by algae (e.g., Dodds et al., 1997). Across all references, TN concentration recommendations ranged from 0.3 mg/L to 1.8 mg/L, and TP concentration recommendations from 0.0125 mg/L to 0.140 mg/L. Considering these broad ranges, the references were further reviewed to determine the most relevant values for Watson Lake and provide the greatest evidence that nutrient concentrations at the stated level would minimize risk of impairment, particularly nuisance algal blooms.

For TP, the 0.03 mg/L concentration cited by Watson et al. (1992) and Downing et al. (2001) provides a strong case as Downing et al. (2001) indicates that phosphorus concentrations above this value present greater than 40 percent risk of cyanobacteria dominance. Some cyanobacteria species can produce toxins, and since many cyanobacteria can fix atmospheric nitrogen and are thus not limited by watershed nitrogen loads, high phosphorus loading can lead to their dominance and cause excessive productivity, leading to nuisance algal blooms, low DO, increased turbidity, and other impacts to the designated uses of Watson Lake. Of the references reviewed in Table 4, Rainbow Lake and Big Bear Lake TMDL endpoints (0.49 to 0.61 mg/L TN and 0.035 to 0.069 mg/L TP) are most promising for application to Watson Lake as these lakes have the most similar characteristics to Watson Lake. Measured background concentrations within the watershed provide a lower bound estimate for TMDL endpoints (0.3 mg/L TN and 0.04 mg/L TP). Selecting endpoints below the Verde River Numeric Endpoints was also a relevant consideration to ensure that downstream nutrient loading is protective of the Verde River.

With greater weight given to the above mentioned lakes (Rainbow Lake and Big Bear Lake), the following endpoint ranges were selected for use in determining the loading capacity:

- Total Nitrogen Growing Season Endpoint Range: 0.3 to 0.8 mg/L
- Total Phosphorus Growing Season Endpoint Range: 0.03 to 0.06 mg/L

It was assumed that the loading capacity should produce growing season (May through October) concentrations within this range. Loading capacity calculations, explained in more detail in Section 6.1, were performed based on the management scenario that resulted in percent load reductions in both TN and TP loading to the lake that were found to result in TN and TP concentrations within and toward the high end of the TN and TP ranges selected above. To be protective of downstream conditions, annual average nutrient endpoints of 1 mg/L TN and 0.1 mg/L TP were selected based on the Verde River numeric endpoints.

2 Conceptual Model

Watson Lake is a monomictic lake, indicating that the lake stratifies into two layers (the shallower epilimnion and the deeper hypolimnion) during one season, and the entire lake mixes during the rest of the year. Stratification generally occurs between June and September and has a strong influence on nutrient balances, algal productivity, and dissolved oxygen concentrations within the growing season. Characteristics of the lake that affect the degree of impairment include lake shape, shoreline complexity, depth, and water level fluctuations, thermal stratification, historic and current nutrient loading, SAV growth and decay, algal growth and decay, and composition of bed sediment.

The conceptual model for Watson Lake, shown in Figure 2, illustrates the major physical, chemical, and biological processes that affect the lake impairments. Nutrient loading, light, and warm temperatures stimulate the growth of primary producers (algae, plankton, SAV, etc.). Excess nutrient loading can increase productivity to the point of causing lake impairments, as explained further below. Sources of external nutrient loading include urban stormwater runoff, point source as well as nonpoint source wastewater sources, atmospheric deposition directly to the lake, and other nonpoint sources, including runoff from heavily managed forested areas and undisturbed natural land cover. Lake levels affect the surface area of substrate available for SAV growth as well as the volume of water conducive for algal and planktic growth.

Die-off of algae, plankton, SAV, and other organisms results in either the release of dissolved nutrients from decomposition or the storage of organic matter in bed sediments, which can later be returned to the water column as bioavailable dissolved nutrients through longer-term decomposition and chemical equilibrium processes, such as the dissolution of phosphorus-iron complexes that occurs under reducing conditions when oxygen is depleted in the sediment . The return flux of dissolved nutrients from the sediment to the water column represents the internal nutrient loading in the lake that can further stimulate productivity. Excessive productivity can lead to algal blooms and low dissolved oxygen.

Low dissolved oxygen is of particular concern in the deeper hypolimnion layer that is not exposed to the atmosphere during stratification. Low or near-zero concentrations of dissolved oxygen can lead to the death of fish and other aquatic life if these conditions occur within the majority of a species' water habitat (i.e., depth range of appropriate water temperature and food sources).

Increased toxicity is also a potential concern with excessive productivity. As primary producers consume dissolved carbon dioxide during photosynthesis, the pH of the water column increases during the day and decreases at night (as is confirmed by monitoring diel cycles of pH and DO in Watson Lake). Elevated pH can lead to increased concentrations of dissolved, un-ionized ammonia (NH_3), which is potentially toxic to aquatic life. Basic conditions (high pH) above habitable ranges also can directly lead to toxic conditions within the lake. In addition to these concerns, some algal species can directly produce substances toxic to both humans and aquatic life. No evidence exists of toxic conditions within Watson Lake although potentially toxin-producing cyanophytes dominate the planktonic algal community. Eutrophic conditions and high pH are indicators that toxicity should be evaluated as a potential concern, either under current conditions or in the future if excessive productivity is not addressed.

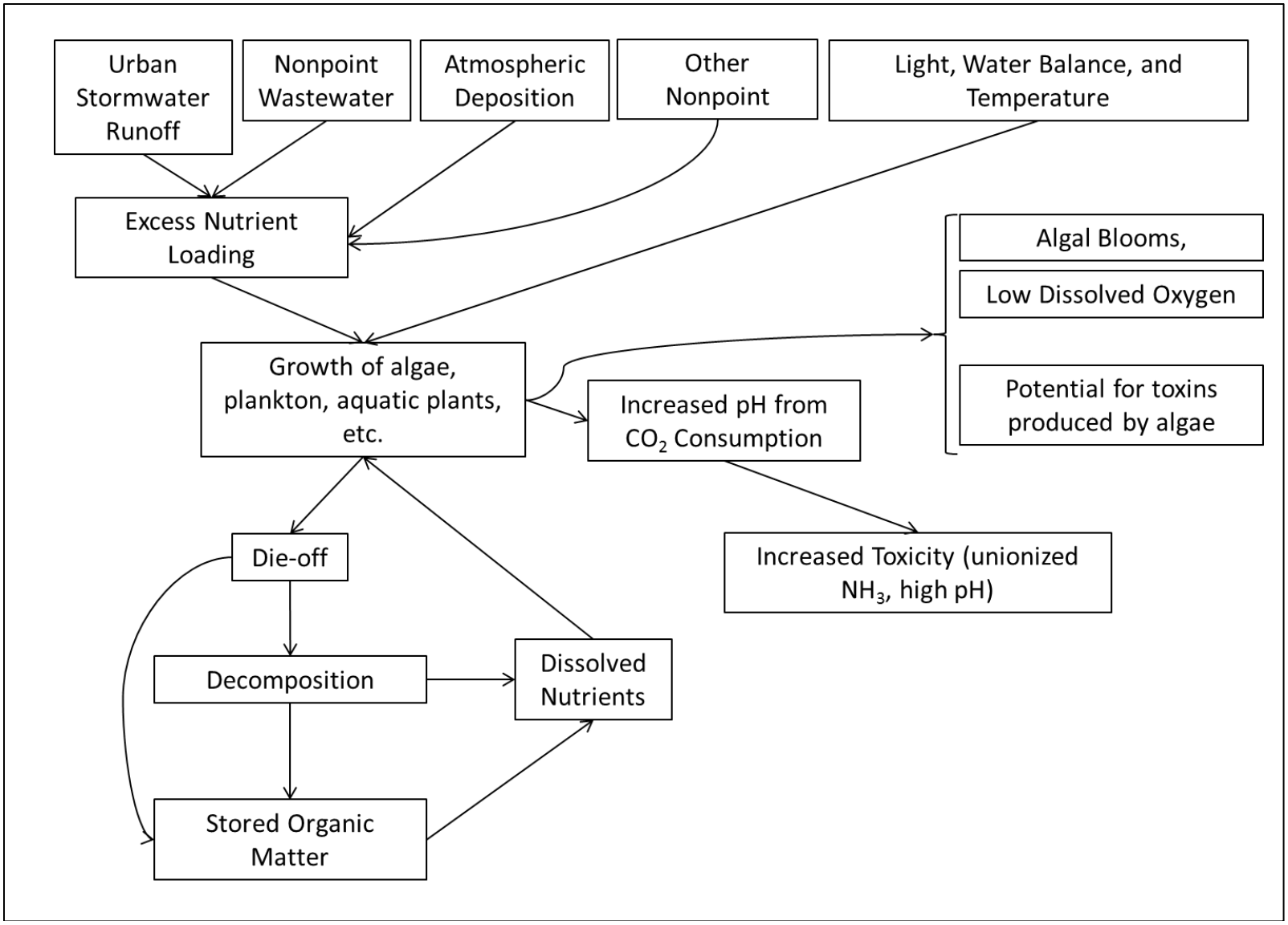

Figure 2. Conceptual Model for Watson Lake

3 Model Framework

Development of the Watson Lake receiving water model began with a review of available water quality, biological, hydrologic, climate, and morphological data. The BATHTUB model (Walker 2004) was selected based on its relevance to the study questions and applicability to the data sample sizes available. A number of model assumptions were required to develop inputs to BATHUTUB, including setting the growing season as May through October and dividing the lake into two segments with separate model input. The data and model assumptions are described in more detail in the following sections.

3.1 DATA USED

Data used by Tetra Tech throughout model development was received from the following entities: ADEQ (2000-2011), the City of Prescott (2003-2011), Dr. Paul Gremillion at Northern Arizona University (NAU; 2009-2011), and Dr. David Walker at the University of Arizona (UA; 2010-2011). Data was also downloaded from USGS (flow data) and NCDC (precipitation and evaporation). Data reviewed but not used directly for the model development are listed in Appendix A. The monitoring data provided by ADEQ that was used for model development included nutrients (nitrogen and phosphorus species), secchi depth, dissolved oxygen, chlorophyll *a*, and water temperature measured at various depths from the surface to 12.5 meters at four locations throughout Watson Lake from 2000 to 2011. The ADEQ and City of Prescott sample locations are displayed in Figure 3. For model development, all samples collected at VRWAT-C were combined with samples collected at VRWAT-B because very little data was collected at VRWAT-C. Together, data collected at these two stations represented mid-lake conditions. Data collected at VRWAT-A represented near-dam conditions, and data collected at VRWAT-SO represented conditions in the upstream portion of the lake under greatest influence from growth of submerged aquatic vegetation (SAV).

Calculated from USGS gaging data, ADEQ also provided monthly mean flow rates from Granite Creek upstream of the lake inlet for January through December for 2007 through 2011. Gage data was retrieved by ADEQ from the lower USGS along Granite Creek at Sundog Ranch Road. This gage is located about a half-mile upstream of the Sundog Wastewater Treatment Plant, which is about a quarter-mile upstream of Watson Lake. ADEQ estimated monthly loading rates to Watson Lake for total phosphorus (TP), total nitrogen (TN), inorganic nitrogen (TIN), and orthophosphorus (OrthoP) from grab sample data collected at the Watson Woods site located across from the Sundog Wastewater Treatment Plant upstream from Watson Lake. These estimations were applied to the average of daily mean flows for each month. Data for 2007, 2010, and 2011 were used in model development.

The City of Prescott provided two sets of data. One set included dam and lake level elevation data for Watson Lake from 2003 to 2011. The second set of data included nitrogen data from 2005-2008 that Tetra Tech used to calculate TN at three sample locations (Figure 3). Two of these sample locations were coincident with stations sampled by ADEQ, VRWAT-A and VRWAT-B, and the third was located off the end of the boat ramp (Boat Ramp Station).

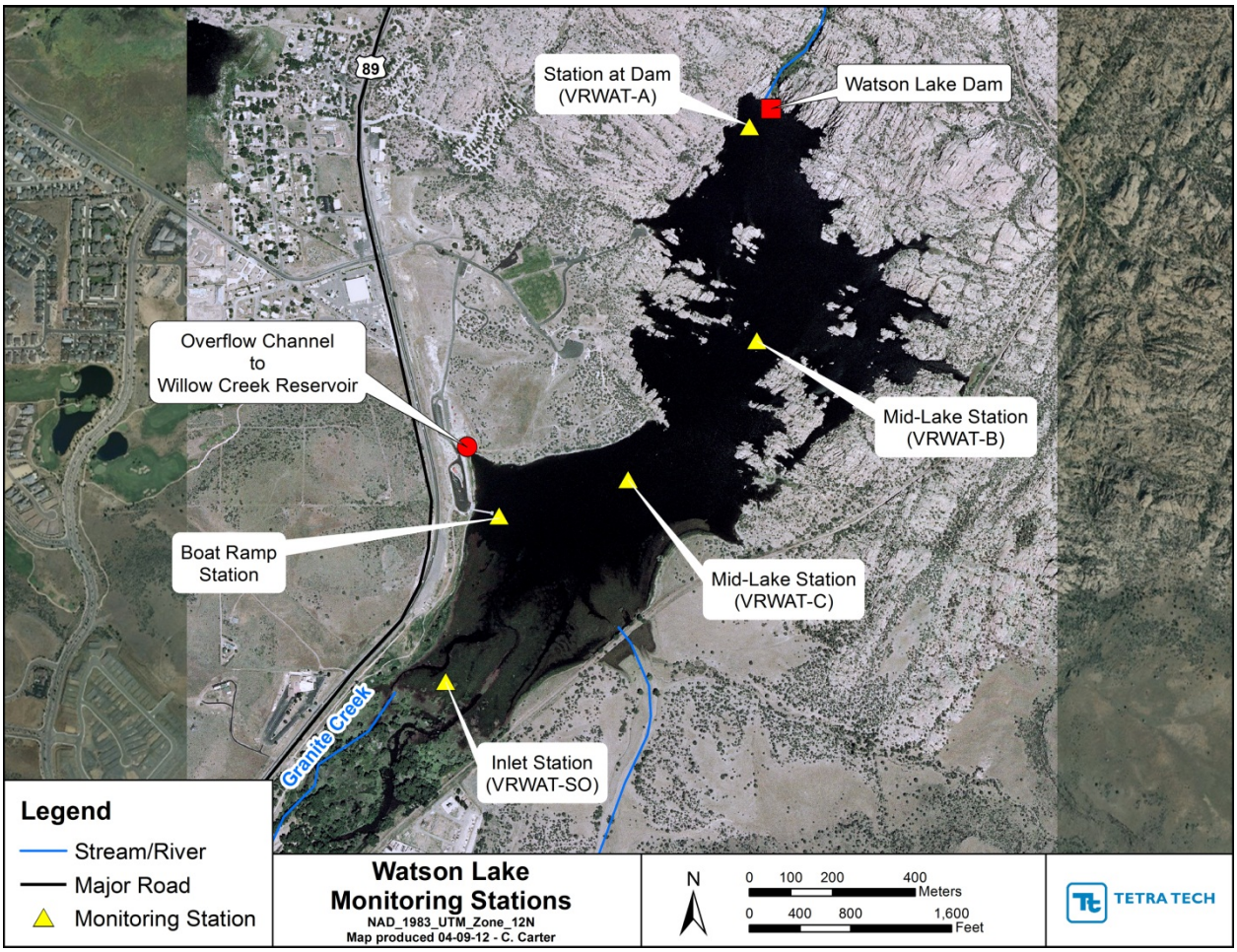

Figure 3. In-lake Monitoring Station Locations

Dr. Paul Gremillion (NAU) provided lake depth data resulting from a bathymetric survey of Watson Lake performed on May 14, 2009. Survey results showed that Watson Lake depth ranged from 0 to 14.9 meters below full pool level. Dr. Gremillion also provided Tetra Tech with a 2007 USGS high resolution orthoimagery for Arizona. The final dataset included results from the Watson Lake Sediment Core Analysis (Gremillion, 2012). The analysis included an assessment of two sediment cores, one collected near the Watson Lake dam and the other collected near the upstream portion of the lake near the inflow of Granite Creek.

Dr. David Walker (UA) provided algal biovolume results collected at ADEQ sample locations (VRWAT-A, B, and SO) for 2010 to 2011 for depths ranging 0 to 12 meters. Data recorded for biovolume also included identification of major divisions present (Chlorophyta, Chrysophyta, Pyrrophyta, Euglenophyta, and Cyanobacteria) and speciation of the cyanobacteria dominant species. Dr. Walker performed a limno-corral study within Watson Lake from August to October 2011 (Walker, 2012). The results from this study were used to interpret model results relating to nutrient dynamics and algal response.

Water quality data from all years was used to inform model development. In particular, a comparison of inlake stations indicated that nutrient concentrations in the upper, southern portion of the lake differed from concentrations in the lower, northern portion of the lake. This evidence, as well as knowledge of morphological differences and presence versus absence of SAV led to the designation of two model segments. The full water quality data set was also used to select model years for calibration and validation based, in part, on data availability.

The depth limit for data used in model calibration was approximately equal to the average secchi depth recorded for the mid-lake and near dam station locations for the years of interest. Half-substitution was used for all data values reported below the detection limit. Significant numbers of samples were below the method detection limit in the nutrient datasets used to calculate TN and organic nitrogen, and in the ortho-phosphorus dataset. Due to the level of reported non-detects, the half-substitution method used in model input data processing introduces uncertainty into the analysis and may impact the quality of model performance.

From the CASTNET website, Tetra Tech retrieved data for atmospheric deposition for TN and TIN from two air monitoring stations located closest to Watson Lake (USEPA, 2012): the Grand Canyon NP site (GRC474) and the Petrified Forest site (PET427). CASTNET stands for Clean Air Status and Trends Network and is a national air quality monitoring network developed by the U.S. Environmental Protection Agency (USEPA). This website provides both dry deposition data collected directly by CASTNET as well as wet deposition data collected by the National Atmospheric Depositional Program.

Tetra Tech also retrieved data for precipitation and computed potential evapotranspiration for model annual inputs for 2007, 2010, and 2011 (see explanation for model years under Section 3.2. Precipitation data was retrieved from the Global Historical Climatology Network-Daily (GHCN-D) database (http://www.ncdc.noaa.gov/oa/climate/ghcn-daily/). Tetra Tech summed Summary of the Day precipitation (Prescott station 026796 located approximately 1 mile southwest of Watson Lake) for each year of interest to equal total precipitation for that year. Data to estimate potential evaporation was retrieved from the National Climatic Data Center (NCDC) database (http://www.ncdc.noaa.gov/oa/ncdc.html), from Surface Airways station 23184 located at the Prescott Municipal Airport. Tetra Tech used the Penman Pan method to calculate potential evaporation from solar radiation, air temperature, wind, relative humidity or dew point, latitude, date, and time data.

3.2 MODEL ASSUMPTIONS

BATHTUB version 6.14 was selected to assist in developing management scenarios and TMDL calculations for the Watson Lake TMDL. The model simulates steady-state water and nutrient mass balances in a spatially segmented hydraulic network that accounts for advective transport, diffusive transport, and nutrient sedimentation. Empirical relationships previously developed and tested for reservoir applications (Walker, 1999) form the basis for model simulation of eutrophication-related water quality conditions (expressed in terms of growing season average total phosphorus, total nitrogen, chlorophyll *a*, transparency, organic nitrogen, non-ortho-phosphorus, and hypolimnetic oxygen depletion rate). The most recent version available (6.14) was used for the Watson Lake modeling. This model was chosen for Watson Lake because it does not require extensive watershed or lake input data and it provides a simulation of lake sedimentation rates, which are important for considering the effect of internal loading on lake nutrient concentrations.

Model input files are provided in Appendix B. Model development focused on the years of 2010 and 2011 because these years provided the most nutrient and chlorophyll *a* data for calibration. The year of 2007 was also modeled and served as a model validation year. The year 2010 represents a relatively wet year, and the year 2007 represents a relatively dry year. The year 2011 was determined to be dryer than 2010 but a fairly typical year for Watson Lake's climate. A BATHTUB model was developed for each year to account for hydrologic variation.

To capture the variation in water quality and morphology between the upper and lower portions of the lake, Tetra Tech divided Watson Lake into two model segments. The first segment, identified in the model as Segment 1, was the upstream portion of the lake that is greatly influenced by the growth of SAV (Figure 4). The second segment, identified in the model as Segment 2, represents the remaining downstream portion of the lake, from nearby the boat ramp to the dam (Figure 4). Segment areas were

selected by Tetra Tech based on what appeared to be an obvious and natural division within the lake. Segment 1 includes very shallow areas with depths ranging from 0 to 4.5 meters and is predominantly covered by SAV throughout the growing season. Segment 2 was predominantly open water throughout the year with depths ranging from 0 to 14.9 meters with greatest depths recorded near the dam.

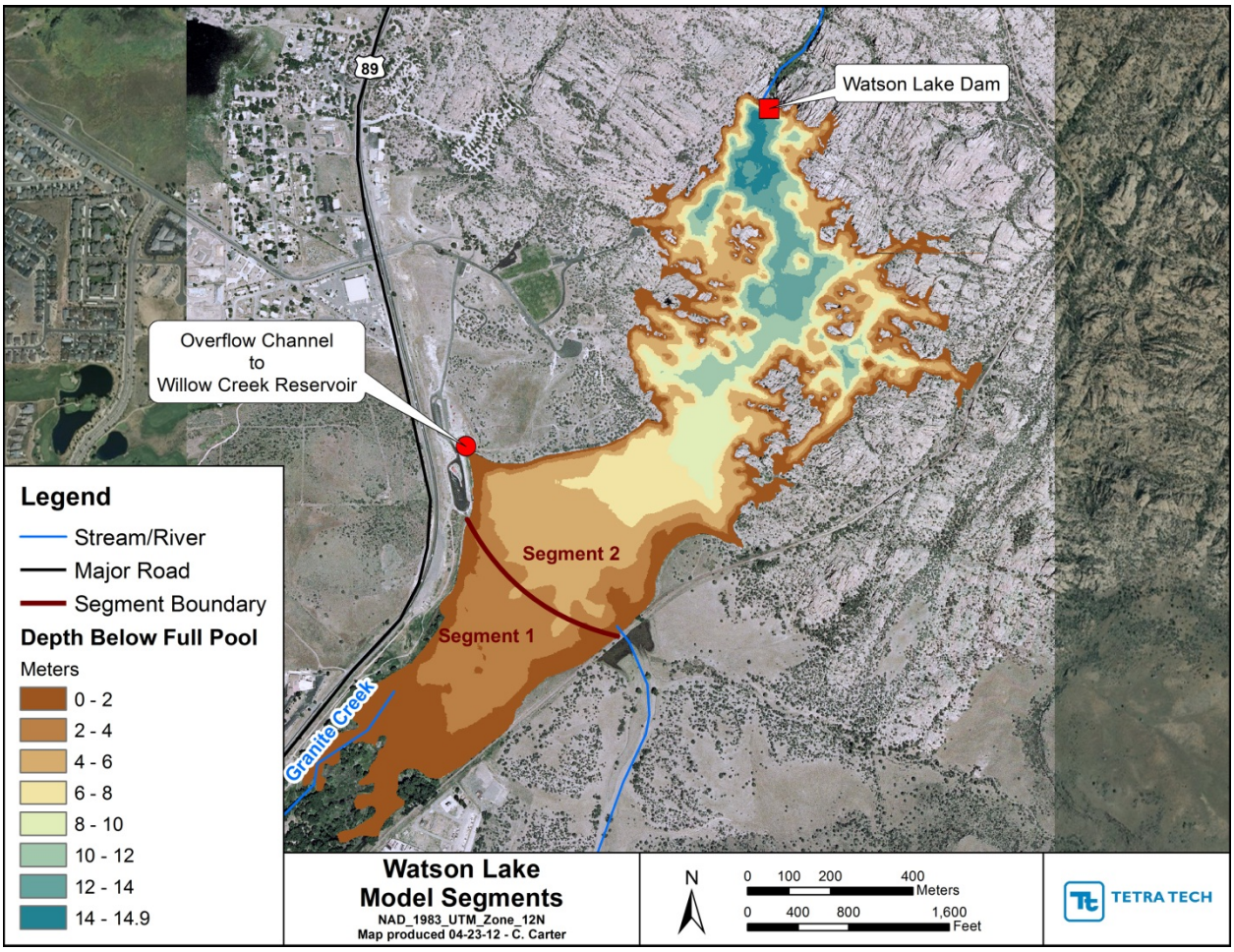

Figure 4. Model Segmentation for Watson Lake

To account for loading to the lake from tributaries and direct drainage, Tetra Tech divided the Watson Lake watershed into three sections. The first and largest section, covering approximately 26,400 acres, includes the Granite Creek drainage area upstream from Watson Lake and is identified in the model as Tributary 1 (Figure 5). The second section, covering approximately 247 acres, includes land area directly draining to Segment 1 and is identified in the model as Tributary 2 (Figure 5). The third and final section, covering approximately 1,980 acres, includes land area directly draining to Segment 2 and land drained by an unnamed tributary and is identified in the model as Tributary 3 (Figure 5). Model inputs for flow rate for Tributaries 2 and 3 were estimated as a proportion of the flow rate calculated for Tributary 1 based on differences in drainage areas between Tributaries 2 and 3 and Tributary 1. Model inputs for nutrient concentrations from the tributaries were calculated as flow-weighted concentrations of observed data from Tributary 1. Because these concentrations were calculated as flow-weighted, the same concentrations were used as model inputs for the other tributaries. BATHTUB performs a calculation of watershed nutrient loading from each tributary using the flow-weighted nutrient concentrations and the estimated area-weighted flow rates.

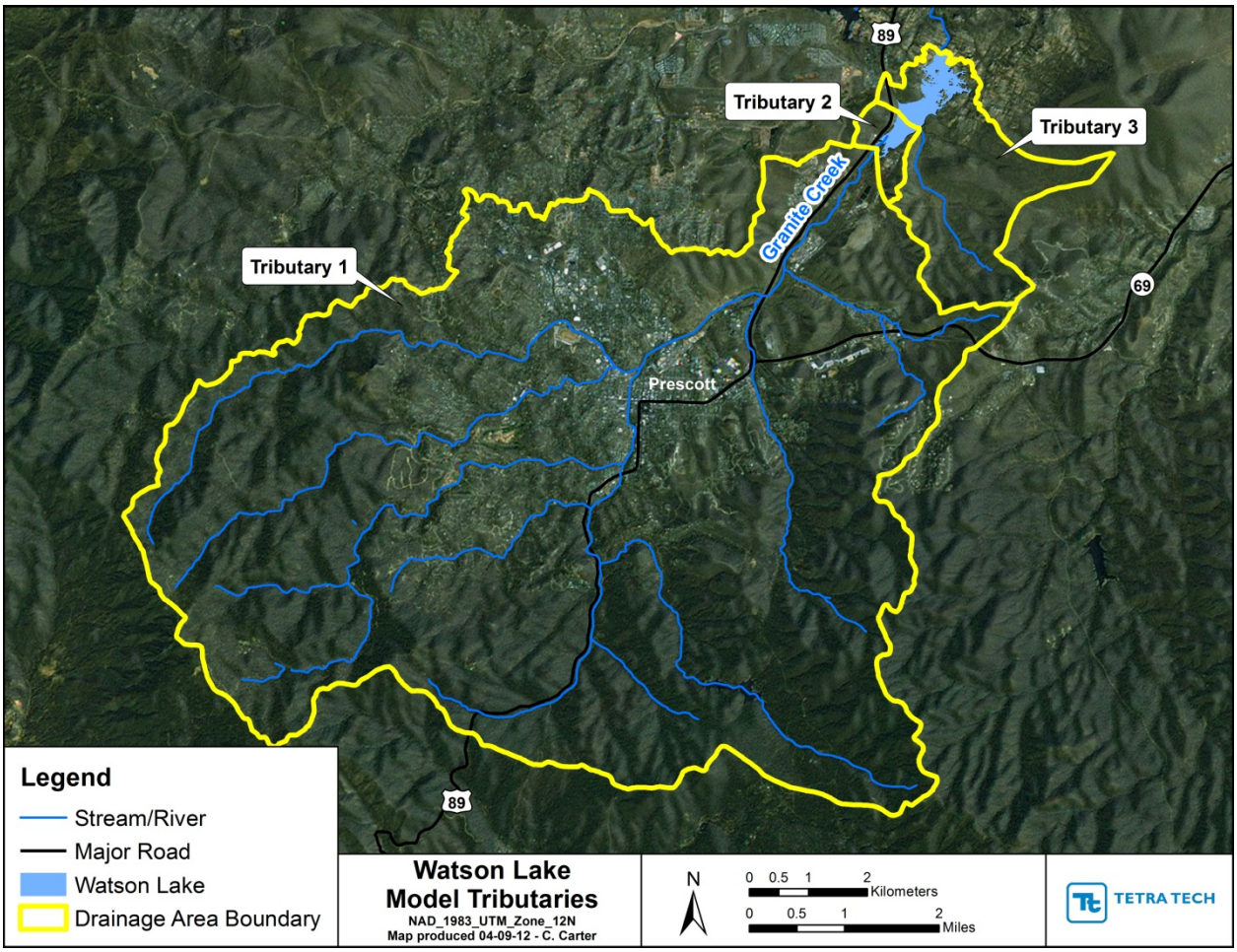

Figure 5. Model Tributaries for Watson Lake

Tetra Tech calculated the average wet plus dry nitrogen deposition across the available years at both CASTNET stations (2000 to 2009 for GRC474 and 2003, 2006, 2007, and 2008 for PET427) and then averaged the results between the two sites to equal 2.27 kg-N/ha (227mg/m^2). This value was used for model input for both TN and TIN loading from atmospheric deposition. Tetra Tech determined that a generalization in atmospheric N deposition at this extent was sufficient for modeling purposes because the Watson Lake surface area is small compared to the watershed area.

Tetra Tech assumed loading of TP and OrthoP from atmospheric deposition to be zero throughout model development. Data for atmospheric deposition of TP and OrthoP were not readily available to estimate accurate values for the study area, and Tetra Tech found no evidence to suggest that TP and OrthoP contributions from atmospheric deposition would be considerable proportions of total load to the lake.

Tetra Tech considered several options for model averaging period, including full year, growing season (May through October), and February through September. Based on BATHTUB model guidance, the appropriate averaging period for each model year was the annual averaging period (1 year). Since Watson Lake was not found to be phosphorus limited, the calculations to estimate the appropriate averaging period were based on an evaluation of the turnover ratio for nitrogen under growing season and annual loading conditions.

Tetra Tech estimated normal pool elevation for 2007 (5,152ft), 2010 (5,160ft), and 2011 (5,155ft) from an assessment of frequency distribution plots of daily lake level elevation for each year. For 2007, there was

a clear unimodal frequency distribution peaking at 5,152ft above sea level. Both 2010 and 2011 frequency distributions for lake level elevation were bimodal. Best professional judgment and knowledge of the lake level seasonal patterns were used to estimate normal pool elevation for both 2010 and 2011. Normal pool elevation was used to calculate surface area, mean depth, hypolimnetic thickness, and volume for model inputs and diagnostic variables.

4 Model Calibration

4.1 Model Limitations for Algal Response

Initial model testing indicated that BATHTUB provided a reasonable representation of growing season average nutrient concentrations; however, it was not possible to calibrate to BATHTUB to reliably predict observed chlorophyll *a* concentrations. In addition, observed nutrient levels are in a range suggesting a eutrophic system that supports significant algal growth. It is suspected that observations may not accurately reflect chlorophyll *a* concentrations when *Gloeotrichia* species dominate the algal community because these species form dense clusters that are difficult to sample. Also, many cyanophytes rely on other pigments in addition to chlorophyll *a*, and chlorophyll *a* may not be an accurate indicator of algal biomass in Watson Lake.

Concurrent to the BATHTUB model development, data from limnocorrals were analyzed by the University of Arizona (Walker, 2012). The limnocorrals performed as expected for nutrients and physical-chemical parameters but did not reveal a clear relationship between nutrients and either chlorophyll *a* or algal biovolume. The reasons for this are not yet well understood but could include temporal changes (temperature decline over the simulation period), influence of periphyton growing on the sides of the limnocorrals that were not accounted for in chlorophyll *a* or algal biovolume measurements, differences in ratios of chlorophyll *a* and biovolume to actual biomass among different algal species, and other influences.

The University of Arizona study started late in the growing season for a lake at the elevation of Watson and time given for the algal response to each treatment was likely far too short to adequately capture highly dynamic algal responses and temporal variability (Walker, 2012). In addition, the limnocorrals used throughout the study have a light transparency of approximately 85 percent ambient light. The amount of periphytic biomass growing on the inside of the limnocorrals that was not accounted for in this study likely resulted in under-estimating the amount of algal biomass inside the limnocorrals and affected the results (Walker, 2012). Also, within the lake, the dominant phytoplankton is the cyanobacteria *Gloeotrichia*. This cyanobacteria is difficult to sample representatively as it forms macroscopic balls suspended at varying depths throughout the water column. It was observed in situ in each limnocorral during the baseline condition of the study. Colonies of *Gloeotrichia* can be quite large and can dominate biovolume in the lake. Despite its apparent dominance, *Gloeotrichia* was found in relatively low levels in grab or composite samples collected from the lake during this study. Researchers suggested that this observation was likely due to a failure to adequately and representatively capture such large phytoplankton using the gear selected for the study (Walker, 2012).

The mesocosm tests involved testing response to nitrogen addition (as phosphorus was apparently not limiting) followed by alum addition that primarily reduced phosphorus with some reduction in TKN as well. Massive changes in algal communities occurred as a result of the nutrient addition, so the effects of the alum addition on the native *Gloeotrichia*-dominated community are unclear. The limnocorral data did not provide clear evidence on the effects of nutrient reductions, particularly nitrogen reductions, on the existing dominant summer algal community in Watson Lake, nor did the experiment provide a clear correlation between nutrient levels and either chlorophyll *a* or biovolume.

Based on both the model and limnocorral findings, it was determined that additional data are required before a reliable simulation of algal response to nutrient loading can be developed. The TMDL was developed using the BATHTUB nutrient simulation and selected TMDL nutrient endpoints, as described in Section 1.3.

4.2 CALIBRATION TARGETS

BATHTUB offers a series of model options for simulating both TN and TP. Prior to calibration, Tetra Tech tested the available nutrient sedimentation models to determine which model best simulated the observed annual average concentrations for lake TN and TP while providing a reasonable level of complexity for simulating in-lake processes. Models selected were 2nd Order, Available Phosphorus for TP, and 2nd Order, Decay for TN. These models performed reasonably well for simulating observed concentrations. Other model options provided a closer simulation to observed concentrations, but the model equations considered fewer input variables and did not provide a consistent fit between different years. In particular, the chosen models provide a simulation of the variability in lake sedimentation rates derived from real reservoir data whereas the other BATHTUB model selections apply either fixed sedimentation rates or are not based on the reservoir dataset.

The model selected for TP sedimentation, 2nd Order Available Phosphorus, performs mass balance calculations on the available phosphorus which is calculated as the weighted sum of ortho-phosphorus and non-ortho-phosphorus placing a heavier emphasis on the ortho-phosphorus component that is more biologically available (Walker, 1999). This model accounts for inflow nutrient partitioning by adjusting the inflow concentrations of phosphorus and using a fixed sedimentation coefficient. The effects of inflow partitioning are incorporated prior to the mass balance calculation (Walker, 1999). In general, nitrogen balances are much less sensitive to inflow nutrient partitioning than are phosphorus balances, potentially because inflow nitrogen tends to be less strongly associated with suspended sediments. The sedimentation model selected for TN, 2nd Order Decay, accounts for inflow nutrient partitioning by adjusting the effective sedimentation rate coefficient as opposed to the inflow concentrations (Walker, 1999). Because the sedimentation models selected for both TP and TN have been empirically calibrated using the reservoir dataset, effects of internal loading from bottom sediments are inherently reflected in the model output parameter values and error statistics (Walker, 1999).

The 2011 model was calibrated to observed growing season median concentrations for in-lake TN and TP. The growing season for Watson Lake is defined as May through October, which represents the typical time period during which productivity increases are observed. Stratification typically begins in June and early July and has been observed to continue into October in some years. Lake Segments 1 and 2 were calibrated separately for both TN and TP for the 2011 model year. Calibration of TN and TP models for the Watson Lake 2011 model year were performed by adjusting sedimentation coefficients within the ranges recommended for application of the model (roughly a factor of 2 for TP and a factor of 3 for TN) to improve the agreement between observed and predicted nutrient concentrations (Walker, 1999).

A data set of 41 US Army Corps of Engineers reservoirs was used to develop and test the BATHTUB model, and the expressions for net sedimentation rates were fit to this reservoir dataset. Since sedimentation rates in individual lakes vary, calibration factors are used to adjust the net influence of sedimentation rates that determine in-lake concentrations. The BATHTUB documentation recommends using calibration factors within a range of 0.5 to 2 for TP and 0.33 to 3 for TN. Using calibration factors within this range helps ensure that the model results fall within expected ranges based on the reservoir dataset used to derive the model equations. For the 2011 model, calibration factors were set to achieve in-lake concentrations closest to the observed growing season medians while staying within the recommended ranges (Table 5).

This process provided a reasonable model fit for 2011 considering the range in observed data, as shown in Figure 6 and Figure 7. The model performs well for TP and TN within Segment 2 and for TN within Segment 1. For TP in Segment 1, the model is overestimating TP compared to the observed growing season median.

Tetra Tech attempted to adjust the calibration factors to provide a reasonable fit across both year 2010 and 2011 using the same model selections and calibration factors. Resulting calibration factors for the 2011 model were used to fit the 2010 model because considerably more data were available to calibrate the 2011 model. In contrast to 2011 data which cover the entire growing season, 2010 data were only available for late June and August. Sample sizes are much smaller, especially for Segment 1 (one sample date in late June for TN and TP). Calibration factors for each lake segment are displayed in Table 5, and observed water quality data with simulated concentrations for 2010 are displayed in Figure 8 and Figure 9.

Table 5. Calibration Factors for 2010 and 2011 Nutrient Simulation

Calibration Factors (applied to decay rates)			
TP		TN	
Seg1	Seg2	Seg1	Seg2
2	2	0.33	0.38

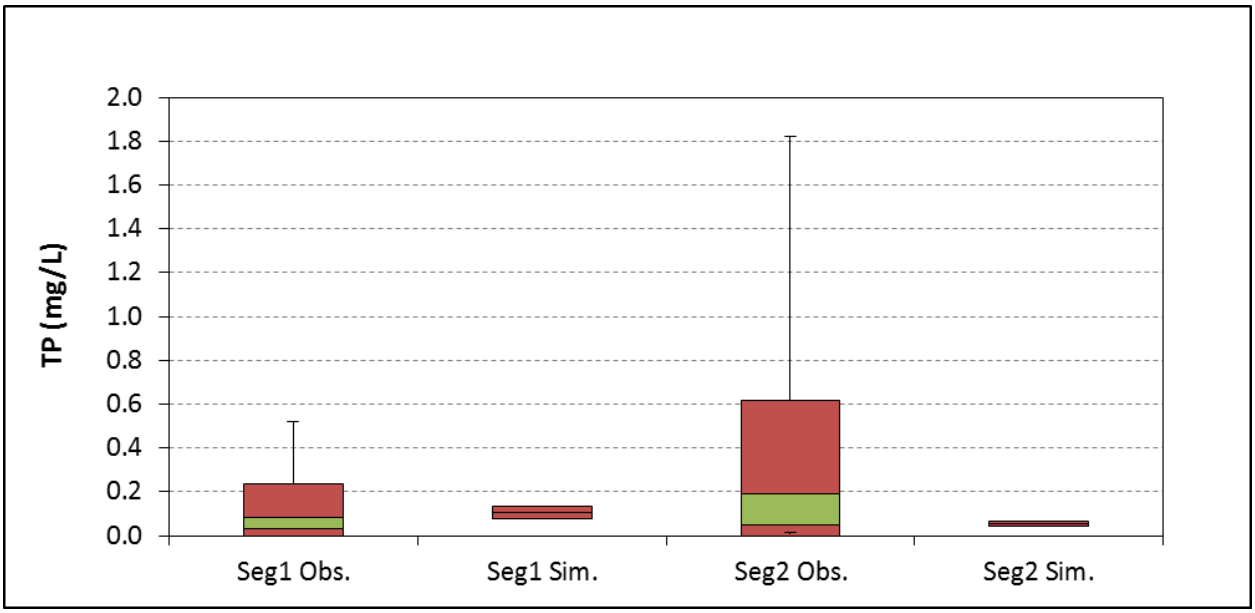

Figure 6. Observed TP Data and Simulated Concentrations for the 2011 Growing Season

(Whiskers indicate data range; red box indicates 1 standard deviation; green box indicates range between mean and median; middle black line indicates simulated mean)

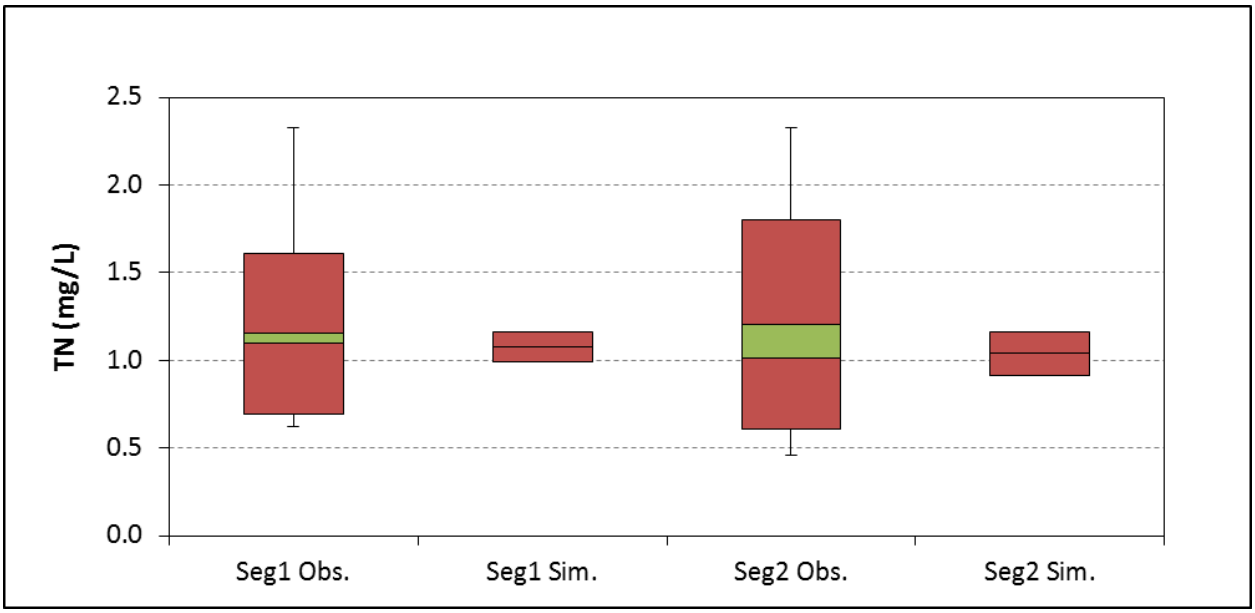

Figure 7. Observed TN Data and Simulated Concentrations for the 2011 Growing Season

(Whiskers indicate data range; red box indicates 1 standard deviation; green box indicates range between mean and median; middle black line indicates simulated mean.)

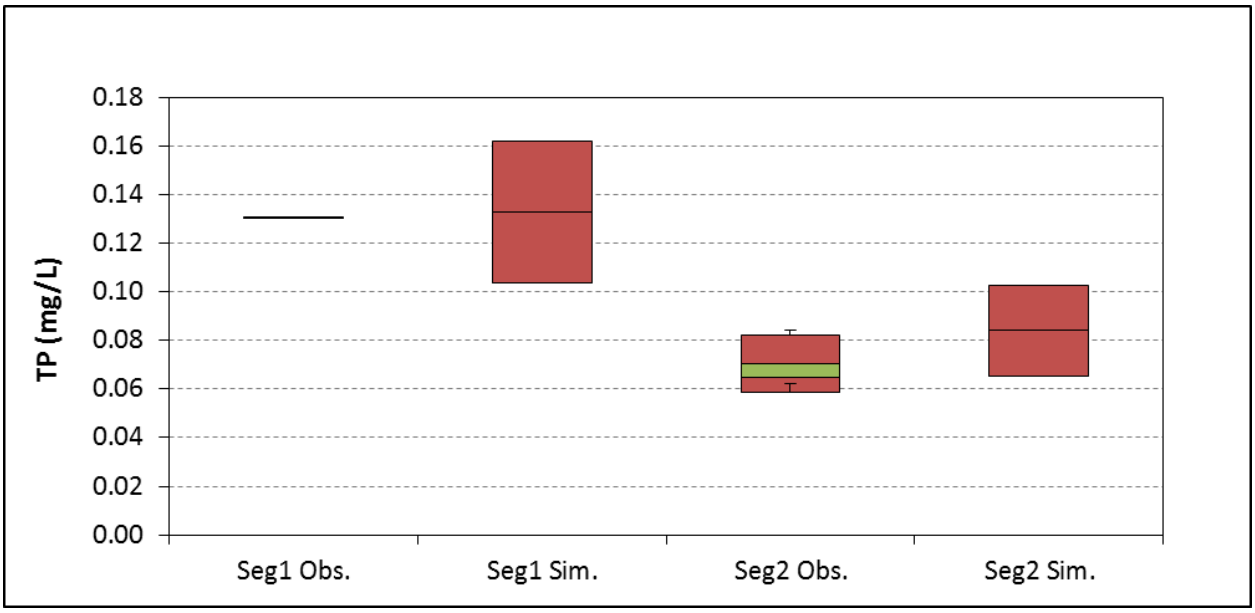

Figure 8. Observed TP Data and Simulated Concentrations for the 2010 Growing Season

(Whiskers indicate data range; red box indicates 1 standard deviation; green box indicates range between mean and median; middle black line indicates simulated mean); Single black line indicates one observed data value (n=1) for the monitoring period.)

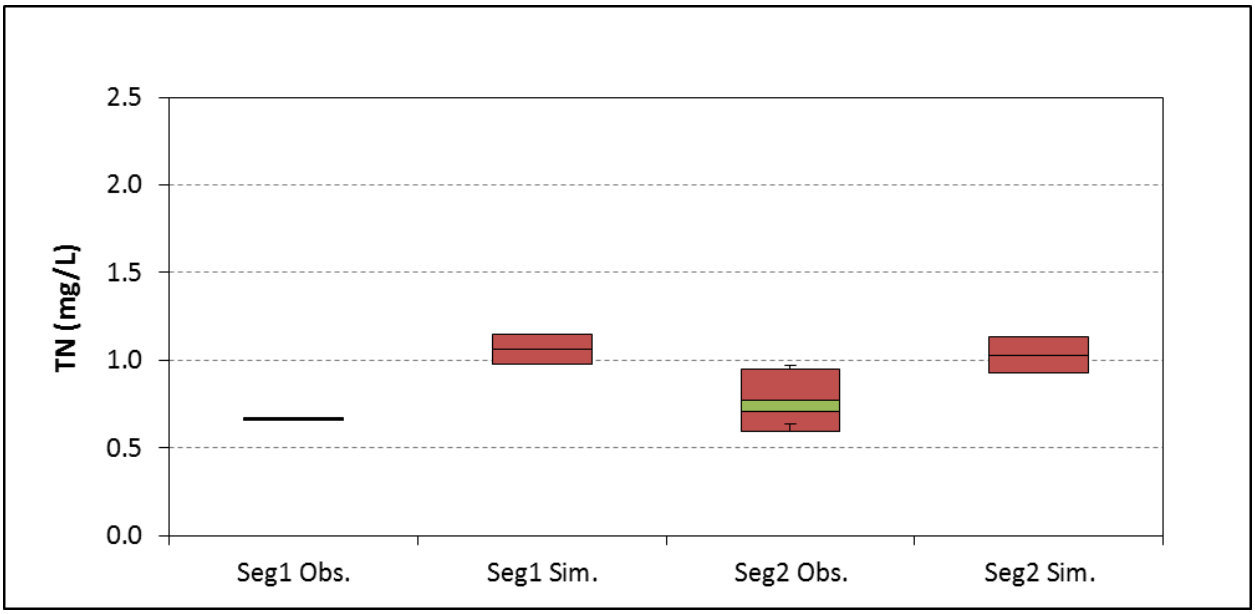

Figure 9. Observed TN Data and Simulated Concentrations for the 2010 Growing Season

(Whiskers indicate data range; red box indicates 1 standard deviation; green box indicates range between mean and median; middle black line indicates simulated mean; a single black line indicates n=1 for observed)

Model performance for 2011 is acceptable given the range of data values for both TN and TP. Even though there was one outlier (greater than two standard deviations above the mean) measured at Segment 2 for TP in August, 2011, the simulated concentration for 2011 TP for Segment 2 appears to be in agreement with the other observed data values at this location during this time. Model performance for 2010 is more acceptable for TP than for TN; simulated concentrations for TN in 2010 are outside of the range of observed data values. Apparent imprecise fit in 2010 could be due to the limitation of data that were available during this model year. As previously stated, 2010 data were only available for late June and August and Segment 1 only had one sample available in late June for both TN and TP.

Model selection and calibration were performed for chlorophyll *a*; however, resulting calibration factors were outside of the acceptable range for this parameter. Both 2010 and 2011 models calibrated for nutrients were grossly over-predicting chlorophyll *a* when compared to observed growing season median chlorophyll *a* data presented in Table 6.

BATHTUB uses growing-season average chlorophyll *a* concentration and the average thickness of the hypolimnion to calculate the metalimnetic and hypolimnetic oxygen depletion rates (MODv and HODv), which are the rates of decrease of the volume-weighted average dissolved oxygen concentration in the metalimnion and hypolimnion. The hypolimnetic oxygen depletion rate (HODv) implicitly accounts for the long-term contribution of sinking algae to sediment oxygen demand. In Table 6, these rates are calculated using the observed and predicted chlorophyll *a*. Since Segment 1 is not deep enough to contain the metalimnion or hypolimnion, these rates are only calculated for Segment 2.

Based on the reservoir dataset used to develop BATHTUB, an HODv greater than 0.10 mg/L-day will typically result in the depletion of the hypolimnetic oxygen supply in the hypolimnion within 120 days after stratification begins (Walker, 1999). According to the June 16, 2011 DO profile at the dam, the average starting DO profile in the hypolimnion was likely about 6 mg/L. If stratification occurred at the beginning of July 2011, the HODv based on observed chlorophyll *a* suggests that anoxic conditions

would occur by the end of July (approximately 24 days). This is confirmed by the DO profile at the dam for July 20, 2011, which shows near anoxic conditions within the hypolimnion on this date. Anoxic conditions would occur much sooner according to the 2011 predicted chlorophyll *a* concentrations. This is an additional indication that BATHTUB is overestimating average chlorophyll *a* concentrations.

Table 6. Chlorophyll *a* Observed Growing Season Medians for Watson Lake

Year	Chlorophyll *a* (µg/L)				HODv (mg/L-day)	MODv (mg/L-day)
	Seg1	cv	Seg2	cv	Seg2	Seg2
2010, Observed	1.9	-	2.0	1.14	0.14	0.05
2010, Predicted	31.9	0.28	22.7	0.31	0.47	0.17
2011, Observed	2.8	1.37	2.2	1.72	0.25	0.05
2011, Predicted	41.5	0.29	21.1	0.31	0.76	0.17

4.3 SENSITIVITY ANALYSIS

Tetra Tech performed a sensitivity analysis to test model sensitivity to the model inputs that were based on limited data or required gross assumptions. This analysis tested whether variation in these inputs would or would not have a significant effect on model results. The following model inputs were tested:

1. Atmospheric phosphorus deposition.
2. Nutrient loading from the watershed due to direct drainage to the lake and the small, unnamed lake tributary.

Model sensitivity to atmospheric phosphorus deposition was selected for analysis because it was assumed throughout model development that there was no input of phosphorus to the lake from atmospheric deposition. To test the model's sensitivity, atmospheric phosphorus deposition rates (mg/m^2-yr) were varied from 0 to 200 mg/m^2-yr at various increments. No significant effect was observed on predicted TP concentrations in response to an increase in loading of phosphorus from atmospheric deposition. Results of the sensitivity analysis indicated that there was less than a 0.1 percent increase in predicted TP concentrations for both Segment 1 and Segment 2 from an increase of 200 mg/m^2-yr in atmospheric phosphorus deposition loading.

Model sensitivity to nutrient loading from direct drainage to the lake and the small tributary was selected for analysis because it was assumed that nutrient loadings from Tributaries 2 and 3 were equally proportional by area to loading calculated for Tributary 1 (Granite Creek) even though land use composition varies between tributary drainage areas. Sensitivity runs were performed to test model sensitivity to either a 10 percent increase or a 10 percent decrease in nutrient loading from the drainage areas of Tributaries 2 and 3.

A 10 percent increase in nutrient loading from Tributary 2 and 3 drainage areas was found to increase predicted TP concentrations by 0.2 and 0.4 percent and decrease predicted TN concentrations by 0.4 and 0.6 percent in model Segments 1 and 2, respectively. A 10 percent decrease in nutrient loading from the watershed was found to decrease predicted TP by 0.2 and 0.6 percent and increase predicted TN concentrations by 0.4 and 0.7 percent in model Segments 1 and 2, respectively.

In general, the greatest sensitivity to increased nutrient loading from Tributary 2 and 3 drainage areas was observed in Segment 2 with a lesser degree of sensitivity observed in Segment 1. The inverse response to

changes in TN loading was attributed to the relationship between nitrogen sedimentation rates and the ratio of tributary inorganic nitrogen to tributary total nitrogen in Model 2, the chosen BATHTUB model for nitrogen. Since the response to TN change was minor, the sensitivity of the TN and TP inorganic fractions was not analyzed. Results of sensitivity runs are provided in Appendix C.

4.4 MODEL CORROBORATION

Tetra Tech performed model corroboration using data from 2007. This year was selected due to the amount of data available compared with data available for other monitoring years aside from 2010 and 2011. While there was little data available for 2007 compared to 2010 and 2011, 2007 offered more consistent data availability throughout the year to provide annual averaging period inputs compared to other monitoring years. Simulated concentrations resulting from the corroboration run are displayed in Figure 10 and Figure 11.

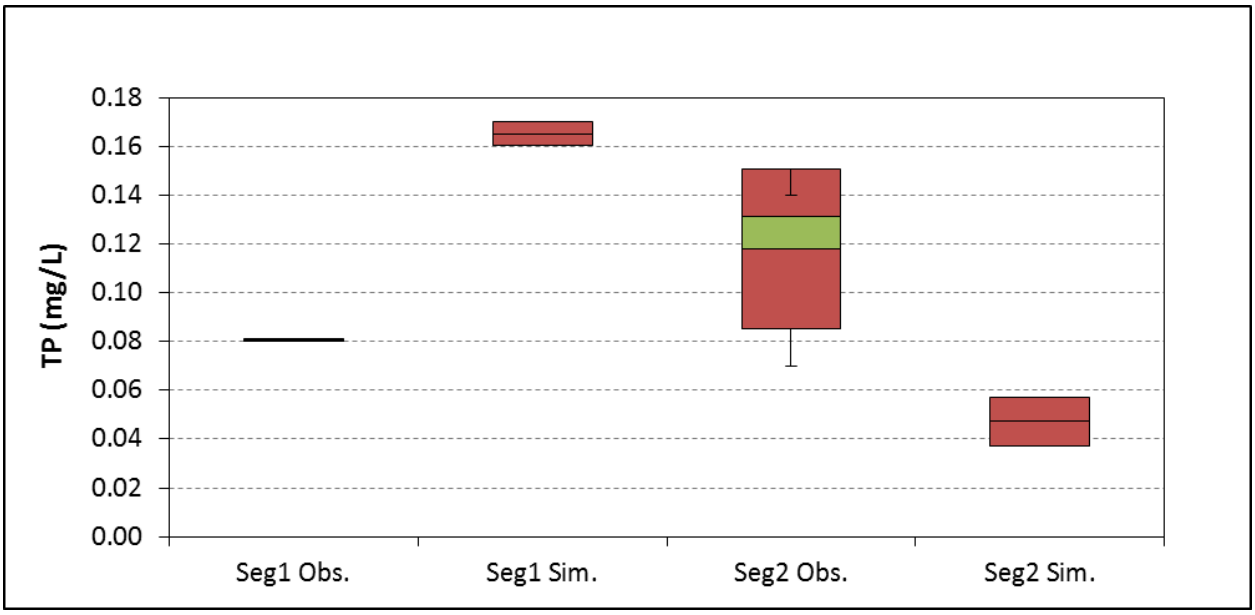

Figure 10. 2007 Growing Season Observed TP Data and Simulated Concentrations

(Whiskers indicate data range; red box indicates 1 standard deviation; green box indicates range between mean and median; middle black line indicates simulated mean; a single black line indicates n=1 for observed.)

Model performance appears to be weak for predicting TP for the 2007 growing season; simulated concentrations are outside of the range of observed data values. Model performance appears to slightly improve when predicting TN for Segment 2, but still has a weak performance for predicting TN for Segment 1. Weak model performance could be explained by the lack of data available for 2007. Since it was difficult to achieve a reasonable fit across multiple years, uncertainties associated with watershed-wide representation of inflow loading estimates may need to be further explored to assess whether they could be contributing to weak model performance.

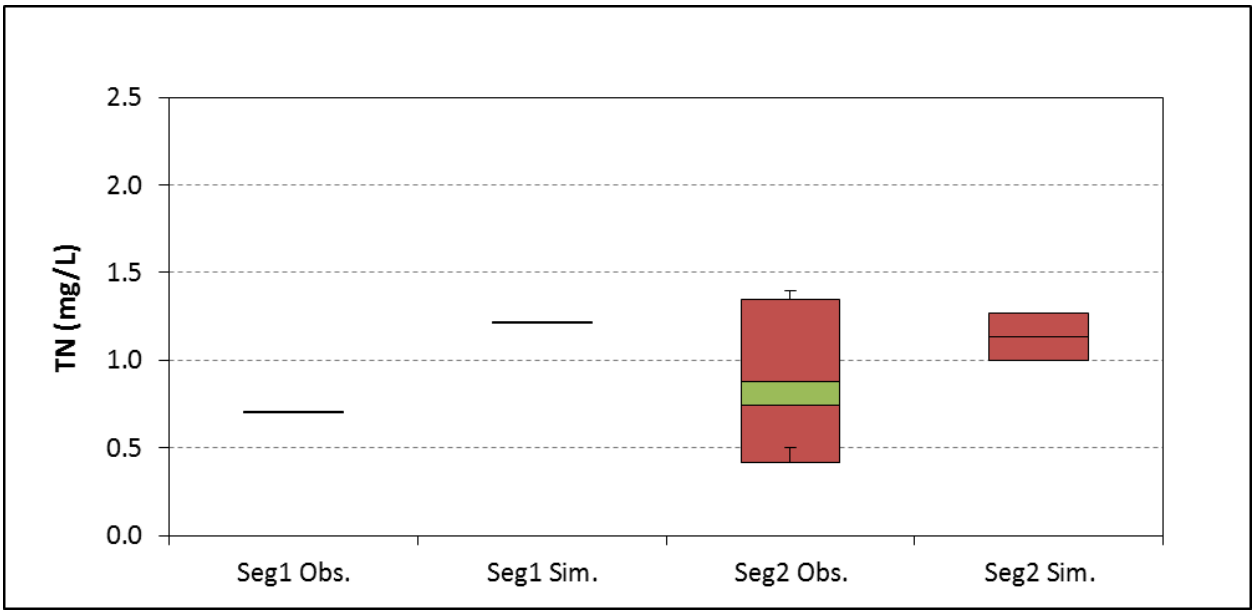

Figure 11. 2007 Growing Season Observed TN Data and Simulated Concentrations

(Whiskers indicate data range; red box indicates 1 standard deviation; green box indicates range between mean and median; middle black line indicates simulated mean; a single black line indicates n=1 for observed and cv=0 for simulated.)

4.5 INTERNAL LOAD

The sedimentation equations used in BATHTUB are semi-empirical representations of net settling (gross settling minus resuspension) derived from a 41-reservoir data set. Thus, some internal loading is inherently accounted for in model simulations. (Note that this internal loading is ultimately derived from storm loads delivered to the lake and retained in the bottom sediment.) The BATHTUB model calibrations for years 2007, 2010, and 2011 suggest positive retention for both nitrogen and phosphorus, indicating that loss of nutrients to the sediment exceeds internal regeneration. Recycling of nutrients from lake sediments is expected to occur within Watson Lake, but the model suggests that the net effect watershed loading, internal loading, nutrient uptake, settling, denitrification, and other processes leads to a net retention of nutrients in the lake, and a reduced outflow load compared to the lake inflow load. These results are expected because the estimated flow weighted concentrations entering the lake (0.23 mg-L TP and 1.13 mg/L TN in 2011) are greater than the in-lake area-weighted concentrations (0.06 mg-L TP and 1.04 mg/L TN in 2011). Watershed loading appears to overwhelm the load contribution from sediment release, and in-lake settling results in a net retention of nutrient load.

These results were confirmed with independent calculations of internal load using empirical equations, one from Welch and Jacoby (2004) and one from Nurnberg (1984). Internal load was calculated separately with two different equations using the following variables: outflow rate, volume, flushing rate, mean depth, lake or segment TP concentration, and inflow TP concentration. Net retention estimated by these equations was about 1,060 lbs/year TP using either equation, which is a similar order of magnitude to the retention predicted by BATHTUB (about 1,600 lbs/year TP).

5 Modeled Scenarios

5.1 Scenario Descriptions and Methods

The model with sedimentation rates calibrated to 2011 data provides an uncertain fit when applied to other years, possibly due to the shortage of available calibration data to estimate growing season concentrations with precision in years other than 2011 as well as uncertainty in estimates of inflow loading. Despite the imprecise fit, the BATHTUB application provides a basis for evaluating the relative impact of management scenarios on nutrient balance. This approach assumes that nutrient reduction will improve algal conditions and that adaptive management will be needed to ensure restoration of designated uses.

Several model scenarios were developed to test the load reductions that can be achieved by watershed and lake management options. The scenarios provide an indication of which management options would provide significant load reductions towards addressing lake impairments and also provide an estimate of the maximum technically achievable reductions in nutrient loads. The following sections provide a description of each management option and methods for simulating the load reductions from each option. The scenario results are summarized at the end of this section.

The BATHTUB model for the year 2011 was chosen to be applied to the model scenarios and resulting TMDL calculations. Based on the calibration results, this model provides a more reliable estimate compared to the 2010 model and is likely more representative of the typical water and nutrient balance of the lake.

5.1.1 Watershed Load Reduction

Management of stormwater and wastewater loading, collectively, is likely to address the majority of the anthropogenic nutrient loading from the watershed. The ideal stormwater treatment facilities would provide large reductions in both nitrogen and phosphorus. More dense urban areas within the City of Prescott may be constrained by space and steep slopes. While wet detention ponds (commonly-used stormwater facilities) provide moderate nitrogen and phosphorus reduction, these facilities may be difficult to site in the more dense urban areas. Even in low density urban areas, steep slopes may constrain the ability to site wet detention ponds or other large, centralized stormwater facilities. Smaller, more distributed stormwater facilities would likely provide more promising options for stormwater treatment throughout the developed watershed areas. Bioretention areas were chosen as the representative distributed stormwater treatment facility for the purpose of this scenario.

Bioretention areas are relatively small depressions filled with sandy soil and planted with vegetation that receive stormwater runoff and slowly infiltrate the runoff into the underlying soil. Where native soils do not provide sufficient infiltration rates, a gravel underdrain can be constructed underneath the sandy soil layer. These facilities can be incorporated into existing landscaping, parking medians, and other small areas available for retrofits. Filter strips or other pretreatment devices should be used to remove sediment from runoff before it enters a bioretention area, as these the sandy soil layer can become clogged with sediment. The maximum recommended drainage area is 5 acres. Bioretention areas can be expensive to implement, but provide multiple advantages in addition to nutrient reduction, including landscaping amenities, control of downstream flow, and potential for groundwater recharge.

Hirschman et al. (2008) suggests that load reductions of 64 and 55 percent can be achieved by treating stormwater runoff with bioretention. Greater reductions have been measured from bioretention, but these

values provide a conservative estimate of load reductions that can be achieved, given that bioretention has not been studied directly within the watershed.

To apply the estimated reductions to loading from urban runoff, it was necessary to estimate the portion of watershed inflow load attributed to urban land uses. Loading rates from a recently developed SWAT model for the Verde River watershed (submitted by Tetra Tech to the EPA Office of Research and Development Global Change Research Program) were applied to land cover areas from the 2006 USGS National Land Cover Dataset. Then the urban proportion of areal loading was calculated from these loads. This proportion was applied to the estimated inflow loads to the lake to approximate the loading from urban land uses. This approximate load proportion does not account for travel time to the reservoir, but provides a rough approximation for the purposes of simulating the effect of reducing urban loading.

In the 2011 BATHTUB model, the load reductions achieved through bioretention were applied to the urban proportion of the inflow loads to the lake. This assumes that 100 percent of the urban drainage area would be treated. In application, it will not be feasible to treat all surfaces, due to site constraints and other factors, but some of the load reduction could be accomplished through sewer and septic management and other methods. Since readily available reduction estimates are not available for wastewater management options, the 100 percent treatment scenario was used to provide an upper boundary for what can be achieved through all available management techniques.

Urban land uses accounted for 14 percent of watershed area and approximately 50 percent of TP or TN load. Treating 100 percent of urban nutrient loads achieves 32 and 34 percent reductions in total inflow load for TP and TN, respectively.

5.1.2 Lake Dredging

Watson Lake has received significant nutrient loading across its lifetime. A sediment core analysis was recently performed for Watson Lake to assess patterns of sedimentation and nutrient content from historic loading (Gremillion, 2012). From this analysis, sedimentation rates were estimated to be 1.5 cm/year after 1964 and ranged from 6 to 9 cm/year prior to 1964. Nutrient (TN and TP) and total carbon content in the sediment were lowest at sediment depths dated to 1954 and earlier. Historic loading has resulted in a layer of accumulated sediment, which is especially apparent in the upper portion of the lake (Segment 1). The sediment layer and historic nutrient loading has provided an ideal substrate for SAV. Removal of this accumulated sediment layer through dredging could provide control of SAV as well as reduction in internal load. The effect of dredging of Segment 1 was tested under this scenario (Gremillion, 2012).

Lake dredging can be expensive and can significantly change lake processes, both in the short and long term. In the short term, dredging causes disturbance of the lake sediment, increased turbidity, and potential release of contaminants stored in the sediment. Changing the depth of the lake can dramatically alter lake processes and biological communities in the long term, and more detailed analysis would be needed to determine if this option is feasible and appropriate for Watson Lake.

The effect of dredging of Segment 1 was tested at two dredge depths – 0.8m and 4.16m. A dredge depth of 0.8m was selected from sediment core results analyzed from the southwest upstream portion of the lake (Core B) that indicated TN, TP, and total carbon content of the sediment greatly declined at this depth from the sediment-water interface, a depth that was dated to 1954 (Dr. Paul Gremillion, Northern Arizona University, personal communication, with Heather Fisher, Tetra Tech, Inc., April 13, 2012). Sediment content of TP appeared to continue to decline as the sediment depth increased to 1.6m from the sediment-water interface, but a depth of 0.8m was selected to test this scenario because it was the depth where TN and carbon content of the sediment were at their lowest and TP was still fairly low. A depth of 4.16m was selected to test the scenario of increasing Segment 1 depth to match the average depth calculated for Segment 2. When testing each dredge depth for Segment 1, 0.8m and 4.16m were added to the mean

depth calculated for Segment 1 and mean depth for Segment 2 was adjusted to maintain a constant total lake volume as there was not an expected increase of water volume as a result of dredging. This resulted in a mean depth of 1.14m and 4.5m for Segment 1 under the two dredging scenarios.

Sedimentation rates were not adjusted for this scenario. The change in lake morphology may increase sedimentation rates and provide an additional decrease in water column nutrient concentrations. The conservative assumption not to adjust these rates provides a margin of safety for the reduction estimates under this scenario.

5.1.3 Lake Level Change

Watson Lake levels are periodically adjusted to maintain the recreational pool and to provide downstream agricultural water supply. The adjustment of lake levels to affect water quality provides a relatively straightforward lake management option, assuming that this adjustment would continue to support both the recreational and water supply uses of the lake. This scenario tests the effect on water quality of increasing and decreasing of lake levels. The lake levels simulated may represent more extreme scenarios than lake levels that would be chosen for actual management, but this analysis provides an indication for whether lake level changes should be further considered as nutrient management options. As noted under the lake dredging scenario, changing the depth of the lake can dramatically alter lake processes and biological communities in the long term and should be approached with caution and more detailed analysis.

Tetra Tech tested an increase in lake level by using the normal pool elevation of Watson Lake during 2005, a year known to more frequently have higher lake levels when compared to 2011. The 2011 normal pool elevation was estimated to be 5,155ft. To model the 2005 lake level, the normal pool elevation for this year was selected as the full pool level (5,161.5ft). To model a decrease in lake level, Tetra Tech used the normal pool elevation of Watson Lake during 2007 (5,152ft). Morphometric parameters for the lake, such as surface area, mean depth, hypolimnetic thickness, and storage gain were adjusted to reflect either the increase or decrease in lake level. Calculations performed for hypolimnetic thickness used the 2005 or 2007 lake level as a starting point and applied the hypolimnetic depth from 2011 (estimated as 10m). Calculations performed for storage gain were based on a proportion of lake level increase or decrease relative to the 2011 lake level, which was then applied to the 2011 storage gain. In general, storage gain was inversely related to lake level change.

5.1.4 Alum Treatment

Alum treatment is widely used to reduce phosphorus concentrations in lakes and is a potential management option for Watson Lake. Treatment with alum can reduce both internal load and inflow load of P by at least 85 percent (Cook et al., 2005; Gibbons and Welch, 2011). This depends on method of treatment, whether alum is added to the water column, and whether inflow is treated. The Watson Lake limnocorral results indicated that alum treatment was successful at reducing phosphorus concentrations in the lake during a period of high nutrient loading (following fertilizer addition to the limnocorral; Walker, 2012). For the phosphorus load existing in the water column, reductions of 72 and 50 percent were observed within the limnocorral at the near dam and uplake sites, respectively. The near dam site is likely to be a more accurate measurement of phosphorus reduction due to complications with creating a seal with the sediment in the SAV-dominated portion of the lake (D. Walker, University of Arizona, personal communication to H. Fisher, April 2012). Since the limnocorral results pertain to artificially high nutrient concentrations and a relatively small alum input, the 85 percent reduction assumption was used to simulate the extent of reductions that could be achieved.

Duration of treatment is relative to the amount of external phosphorus load from the watershed. If external load is well controlled, some treatments can last up to 25 years. If external load is under a low level of control, the treatment may only last 5 years. Typically, in the arid Southwest United States, alum treatments tend to last about 5 years. Seasonal impacts in the watershed should be considered for alum treatment plans. The treatment plan may include maintenance doses to aid in keeping phosphorus depressed after an external load may have replenished phosphorus concentrations in the water column. Alum can be toxic to fish and other invertebrates, and smaller, more frequent doses can help reduce this detrimental effect. Other disadvantages include potential release of phosphorus during anoxia or extreme pH, fluctuation in pH or other water chemistry during treatment, and resuspension of floc in shallow areas during turbulence.

For this scenario, it was assumed that an initial alum treatment would be applied to the lake which would provide 85 percent reduction in both TP inflow and internal loading. To ensure that the internal load reduction would be maintained over the long term, maintenance treatments would be applied to control the TP inflow load. Since net internal loading was estimated as zero, the 85 percent reduction was applied to the TP inflow load in the BATHTUB model. Additional load reduction would also be expected for any gross TP internal loading, and small reductions in TN internal loading may be possible as well; these quantities could not be explicitly estimated.

5.1.5 Additional Management Options

The scenarios modeled the most promising management options that could be estimated quantitatively. Additional management options may be considered during TMDL implementation and adaptive management. Aquatic vegetation cutting or harvesting could provide removal of SAV and net removal of nutrients in sediment while minimizing sediment disturbance.

Aeration of the water column is a potential option that can reduce phosphorus and ammonia recycling from the sediments while suppressing some of the competitive advantages of problematic cyanobacteria (e.g., ability to adjust position in the water column through buoyancy changes under quiescent conditions, ability to out-compete other algae for CO_2 when water column concentrations are low). Whole lake aeration is expensive in terms of energy costs, but could be employed if nutrient reduction methods do not result in improvement to algal response or DO concentrations. Partial aeration combined with external load reductions could be a useful management scenario to evaluate.

Lake flushing is also an option but could be expensive considering the arid climate and limited water supply within the vicinity of Watson Lake. As noted under the watershed scenario, a number of watershed management options exist that, if employed together, could result in dramatic nutrient reductions to the lake.

5.2 SCENARIO RESULTS

Table 7 displays the concentration and oxygen depletion rate results for each scenario, and Figure 12 compares the scenario loading results for lake inflow, flow between segments, and lake outflow. The watershed load reduction and alum treatment scenarios appear to be the most promising management options for reducing nutrient loading to and from the lake as well as in-lake concentrations and DO depletion rates. The dredging and lake level scenarios provided some promising results for reduction of TP, HODv, and MODv, but provided only a small degree of TN reduction, if any. One additional scenario is presented, combining the watershed load reduction and alum treatment scenario. The results of this additional scenario represent the estimate of maximum technically achievable reduction in nutrient loads to the lake. Note that this is a conservative estimate as the alum treatment would be expected to provide

reduction in TP internal loading as well as the potential for reduction in TN loading, which could not be reliably estimated.

Table 7. Scenario Results: TN and TP Segment Concentration and Hypolimnetic Oxygen Depletion (HODv) and Metalimnetic Oxygen Depletion (MODv)

Scenario	Segment 1		Segment 2			
	TN (mg/L)	TP (mg/L)	TN (mg/L)	TP (mg/L)	HODv (mg/L-day)	MODv (mg/L-day)
Existing Conditions	1.08	0.105	1.04	0.053	0.760	0.168
Watershed Load Reduction	0.80	0.081	0.79	0.045	0.682	0.150
Lake Dredging to 1.14m	1.10	0.121	1.03	0.045	0.721	0.159
Lake Dredging to 4.5m	1.05	0.084	1.00	0.035	0.660	0.145
Lake Level, Lower (2007 levels)	1.13	0.214	1.03	0.056	1.282	0.171
Lake Level, Higher (2005 levels)	1.03	0.068	1.01	0.043	0.495	0.150
Alum Treatment	1.08[1]	0.023	1.04[1]	0.018	0.478	0.105
Watershed and Alum Treatment	0.80[1]	0.018	0.79[1]	0.015	0.421	0.093

[1]Nitrogen reduction due to alum treatment is expected but could not be quantified.

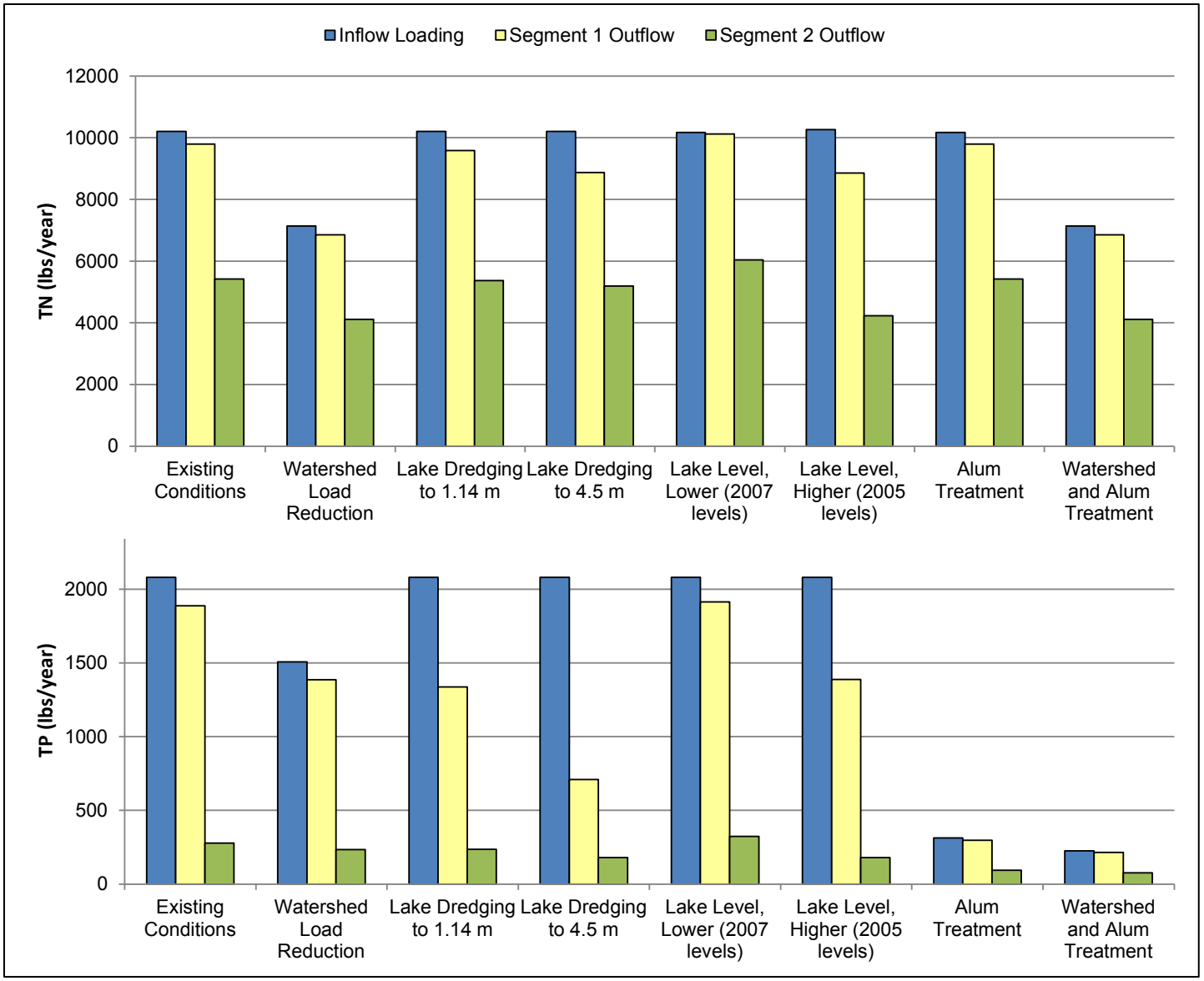

Figure 12. TN and TP Loading Scenario Result

6 Linkage Analysis

6.1 TMDL RECOMMENDATIONS

The modeled scenarios suggest that in-lake concentrations can be reduced to within the nutrient endpoint ranges if loading to the lake is reduced by 34 percent for TN and 32 percent for TP. This analysis was based on the watershed load reduction scenario in which all loads from urban lands were treated; however, the reductions could be achieved in a variety of ways. It is anticipated that opportunities exist to reduce nutrient loading from aging sewer infrastructure, septic leach fields, and other wastewater sources as well as from streambank erosion and other nonpoint sources. An adaptive management approach to TMDL implementation is proposed, in which equal reductions are applied to point and nonpoint sources. As more data become available on the extent of loading due to individual nonpoint sources, the load allocations could be revisited based on the potential to reduce these individual loads. Additional stormwater treatment would provide opportunities for additional load reduction if the load allocations cannot be met through nonpoint source management.

Table 8 presents the existing loads and recommended loading capacity and allocations. The loading capacity represents a reduction in loading to the lake of 34 percent for TN and 32 percent for TP. A 10 percent explicit margin of safety (MOS) is provided to account for uncertainty in the loading estimates that has not already been accounted for by conservative model assumptions. An implicit margin of safety is also provided through conservative assumptions used throughout the model and scenario development. The watershed load reduction scenario from which the 34 percent reduction in TN loading and 32 percent reduction in TP loading was derived produced TN and TP concentration reductions that were within and close to the high end of the target ranges for both TN (0.3 to 0.8 mg/L) and TP (0.03 and 0.06 mg/L) concentrations when compared to analyses performed on other management scenarios. Accounting for the MOS, the waste load allocation (WLA) and load allocation (LA) represent reductions of 37 percent for TN and 35 percent TP.

Table 8. Existing Loads, Loading Capacity, and Allocations

Conditions/Allocations	Loading to the Lake		Area-weighted Lake Concentration	
	TN (lbs/yr)	TP (lbs/yr)	TN (mg/L)	TP (mg/L)
Existing Conditions	10,888	2,228	1.04	0.060
MS4 Jurisdictions	5,123	1,119		
Nonpoint Sources	5,764	1,109		
Loading Capacity	7,141	1,506	0.79	0.05
Waste Load Allocation	3,011	681		
Load Allocation	3,416	674		
Margin of Safety	714	151		

6.2 CRITICAL CONDITIONS

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. Critical conditions for nutrient impaired lakes typically occur during the warm summer months when water temperatures are elevated and algal growth rates are high. Elevated temperatures not only reduce the saturation levels of DO, but also increase the toxicity of ammonia and other chemicals in the water column. Excessive rates of algal growth may cause large swings in DO, elevated pH, odor, and aesthetic problems. Loading of nutrients to lakes during winter months are often biologically available to fuel algal growth in summer months. The recommended loading capacity accounts for summer season critical conditions by using BATHTUB to calculate possible annual loading rates consistent with meeting the selected growing season nutrient endpoint ranges. The recommended load reductions are expected to alleviate any pH and odor problems associated with excessive nutrient loading and eutrophication. Oxygen depletion rates in the meta- and hypolimnia are also expected to decrease with the recommended reductions. These recommendations therefore protect for critical conditions.

6.3 INFLUENCE OF SAV

Submerged aquatic vegetation (SAV) is expected to have an important influence on nutrient balance and algal response within Watson Lake. The receiving water modeling did not provide a basis for quantifying the effect of SAV. However, SAV can be considered in light of the model results.

It is likely that SAV removes nutrients from the system during the growing season, especially nitrogen. Some of these nutrients are returned to the lake during die down at the end of the growing season. This effect may partially account for difficulties in calibrating nutrients and chlorophyll *a* in Segment 1. In addition, these processes could provide advantages or disadvantages towards addressing impairments. The timing of this nutrient uptake and release may help to decrease nutrient availability for algae during the growing season, but the decomposition of plant biomass may lead to a greater net availability of nutrients in the water column throughout the year.

It is also important to consider how the SAV communities would react to a large reduction in nutrients. For example, if algal biomass decreases as a result of nutrient reduction, light availability may increase, and SAV growth could increase. Based on secchi depth measurements, the lake currently has considerable light availability, so this effect may not be significant.

SAV is also expected to contribute to reduced CO_2 and elevated pH – a possible advantage to cyanobacteria, which can exist more successfully under reduced CO_2 concentrations than many other algal groups. While reduction in algal biomass will also help reduce pH, control of SAV could be considered if nutrient reduction alone does not improve pH.

Overall, the uncertainty in the relationships between SAV and Watson Lake processes supports the need for adaptive management. Additional study of lake processes and incremental changes to lake inputs could provide additional insight into the influence of SAV.

7 Summary and Conclusions

Watson Lake presents a challenge for simulating nutrient balances and algal response due to considerable variability in hydrology across different years. As noted, inconclusive evidence of nutrient effects on algal response prevented a useful prediction of chlorophyll *a* concentrations and related parameters. A reliable nutrient simulation for Watson Lake was developed, with a focus on the year 2011, in which the largest data set was available for calibration and reflected relatively typical hydrologic conditions.

It is important to note that the empirical equations used in BATHTUB were derived from a dataset of reservoirs that may not reflect some of the unique characteristics of Watson Lake. In particular, the lakes within the dataset were likely to experience a dry period during the growing season instead of large storm events as occurs during the summer monsoon season. The effect of the nutrient loading experienced by the lake during the summer monsoon season may not be fully reflected in the BATHTUB simulation, and this may partially account for difficulties encountered during calibration.

Additional data collection is recommended that would provide a stronger foundation for model simulation and greater understanding of lake processes. These recommendations include:

- Limnocorral studies conducted during the portion of the growing season in which peak productivity occurs.
- Continued lake sampling that covers the growing season similar to the sample counts available for 2011.
- Gaging of the lake outflow along the channel to Willow Creek Reservoir to provide more accurate water balance information.

The simulated management scenarios provide a basis for recommending nutrient load reductions allocations for the Watson Lake TMDL. Considering that additional study of algal response is necessary, the nutrient reduction recommendations provide an interim target for lake management that can be re-evaluated once additional data are available.

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9 Appendices

Appendix A: Data Not Used

Tetra Tech limited water quality and biological data usage for model inputs to depths of 1x the average growing season secchi depth for each model year. In general, samples collected at depths greater than 4 meters were used to inform model development and interpretation, but these data were not used to generate model input values.

Tetra Tech limited model development to the years of 2010 and 2011, and validation with data from 2007. The year of 2011 had the most data available throughout the year and across all parameters when compared to all other years with available data. For this purpose, the model from 2011 was selected as priority for calibration and calibration factors used to fit the 2011 model were also used to fit the model built for 2010. These calibration factors were also applied during model validation using data from 2007. Data collected during all other years was not used to generate model input values.

The following list includes additional data provided to Tetra Tech but was not used to inform model development or to generate model inputs:

- Groundwater well monitoring data from Watson Woods Riparian Preserve (2006 through 2011)
- In-lake inorganics, metals, and sediment data collected by ADEQ
- Willow Lake levels provided by City of Prescott
- Sundog Wastewater Treatment Plant effluent data
- Sommerfeld and Ellingson's (1984) Water Quality Analysis of Watson Lake

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Appendix B: Model Input Files

The following data tables include all model input values for 2007, 2010, and 2011.

Global Variables

Parameter	2007	2010	2011
Averaging Period (yrs)	1	1	1
Precipitation (m)/Avr Period	0.391	0.484	0.339
Evaporation (m)/Avr Period	1.788	1.692	1.782
Storage Gain (m)/Avr Period	3	3	2
Atmospheric Loads (mg/m^2-yr)			
Total Phosphorus	0	0	0
Ortho Phosphorus	0	0	0
Total Nitrogen	227	227	227
Inorganic Nitrogen	227	227	227

Segment Data

Parameter	2007				2010				2011			
	Seg 1	cv	Seg 2	cv	Seg 1	cv	Seg 2	cv	Seg 1	cv	Seg 2	cv
Total Phosphorus (ppb)	80.000	-	118.000	0.276	130.000	-	70.300	0.170	74.325	1.914	162.217	2.398
Total Nitrogen (ppb)	700.000	-	785.000	0.542	660.000	-	771.667	0.228	1040.167	0.473	1088.783	0.551
Chlorophyll-a (ppb)	2.500	-	5.779	0.390	1.940	-	5.513	1.137	5.005	1.507	5.422	1.849
Secchi Depth (m)	0	-	3.200	0.236	0	-	3.767	0.271	2.233	0.493	4.705	0.307
Organic Nitrogen (ppb)	680.000	-	685.000	0.072	560.000	-	632.667	0.333	952.792	0.522	983.500	0.626
TP - OrthoP (ppb)	-	-	-	-	-	-	-	-	44.996	-	113.903	-
Hypolimnetic oxygen depletion rate (ppb/d)	-	-	189.034	-	-	-	231.613	-	-	-	-	-
Metalimnetic oxygen depletion rate (ppb/d)	-	-	87.694	-	-	-	85.654	-	-	-	-	-

Tributary Data – cvs were not calculated for inputs

Parameter	2007			2010			2011		
	Trib1	Trib2	Trib3	Trib1	Trib2	Trib3	Trib1	Trib2	Trib3
Watershed Area (km^2)	107.457	0.970	7.661	107.457	0.970	7.661	107.457	0.970	7.661
Annual Flow Rate (hm^3/yr)	3.033	0.027	0.216	16.300	0.147	1.162	4.038	0.036	0.288
Total Phosphorus (ppb)	179.545	179.545	179.545	438.111	438.111	438.111	231.683	231.683	231.683
Ortho Phosphorus (ppb)	179.545	179.545	179.545	438.111	438.111	438.111	231.683	231.683	231.683
Total Nitrogen (ppb)	1214.570	1214.570	1214.570	1228.792	1228.792	1228.792	1132.182	1132.182	1132.182
Inorganic Nitrogen (ppb)	643.380	643.380	643.380	589.608	589.608	589.608	575.220	575.220	575.220

Morphometry

Parameter	2007				2010				2011			
	Seg 1	cv	Seg 2	cv	Seg 1	cv	Seg 2	cv	Seg 1	cv	Seg 2	cv
Surface Area (km^2)	0.006	-	0.475	-	0.174	-	0.581	-	0.070	-	0.510	-
Mean Depth (m)	0.801	-	4.000	-	1.300	-	5.413	-	0.340	-	4.500	-
Length (km)	0.766	-	1.346	-	0.766	-	1.346	-	0.766	-	1.346	-
Mixed Layer Depth (m)	0.800	0.120	3.900	0.120	1.300	0.120	4.800	0.120	0.300	0.120	4.200	0.120
Hypolimnetic Thickness (m)	0	-	3.052	-	0	-	2.433	-	0	-	1.450	-

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Appendix C: Sensitivity Runs

Tetra Tech performed two tests of model sensitivity:

1. Sensitivity to atmospheric phosphorus deposition
2. Sensitivity to nutrient loading from the watershed due to direct drainage to the lake and the small, unnamed lake tributary (drainage areas of Tributaries 2 and 3).

Sensitivity to atmospheric phosphorus (Atm P) deposition

Methods -

1. Increased atmospheric phosphorus load in increments of 0.5mg/m^2-yr from 0 to 10mg/m^2-yr
2. Increased atmospheric phosphorus load in increments of 10mg/m^2-yr from 10 to 100mg/m^2-yr
3. Increased atmospheric phosphorus load in increments of 50mg/m^2-yr from 100 to 200mg/m^2-yr

Results –

Atm P Load (mg/m^2-yr)	Predicted TP Concentration (ppb)		Percent Change from Atm P Load of Zero (%)	
	Seg 1	Seg 2	Seg1	Seg2
0	172.3	125.4	0.000	0.000
0.5	172.4	125.4	0.001	0.000
1	172.4	125.4	0.001	0.000
1.5	172.4	125.4	0.001	0.000
2	172.4	125.5	0.001	0.001
2.5	172.4	125.5	0.001	0.001
3	172.4	125.5	0.001	0.001
3.5	172.4	125.5	0.001	0.001
4	172.4	125.5	0.001	0.001
4.5	172.4	125.5	0.001	0.001
5	172.4	125.5	0.001	0.001
5.5	172.4	125.5	0.001	0.001
6	172.4	125.5	0.001	0.001
6.5	172.4	125.5	0.001	0.001
7	172.4	125.5	0.001	0.001
7.5	172.4	125.5	0.001	0.001
8	172.4	125.5	0.001	0.001
8.5	172.4	125.5	0.001	0.001
9	172.4	125.5	0.001	0.001
9.5	172.4	125.5	0.001	0.001
10	172.4	125.5	0.001	0.001
20	172.5	125.6	0.001	0.002

Atm P Load (mg/m^2-yr)	Predicted TP Concentration (ppb)		Percent Change from Atm P Load of Zero (%)	
	Seg 1	Seg 2	Seg1	Seg2
30	172.6	125.7	0.002	0.002
40	172.6	125.7	0.002	0.002
50	172.7	125.8	0.002	0.003
60	172.8	125.9	0.003	0.004
70	172.9	125.9	0.003	0.004
80	172.9	126.0	0.003	0.005
90	173.0	126.1	0.004	0.006
100	173.1	126.2	0.005	0.006
150	173.4	126.5	0.006	0.009
200	173.8	126.9	0.009	0.012

Summary –

No significant effect was observed on predicted TP concentrations in response to an increase in loading of P from atmospheric deposition. There was less than a 0.1 percent increase in predicted TP concentrations from both Segment 1 and Segment 2 from an increase in atmospheric phosphorus deposition loading from 0mg/m^2-yr to 200 mg/m^2-yr.

Sensitivity to nutrient loading from the watershed

Methods –

1. Decreased both TN and TP concentrations from the watershed for Tributaries 2 and 3 by 10 percent (Sensitivity Runs 1 and 3)
2. Increased both TN and TP concentrations from the watershed for Tributaries 2 and 3 by 10 percent (Sensitivity Runs 2 and 4)

	Model Input for TP (ppb)		
	Trib 1	Trib2	Trib3
Original Model	231.683	231.683	231.683
Sensitivity Run 1	231.683	208.515	208.515
Sensitivity Run 2	231.683	254.851	254.851

	Model Input for TN (ppb)		
	Trib 1	Trib2	Trib3
Original Model	1132.182	1132.182	1132.182
Sensitivity Run 3	1132.182	1018.964	1018.964
Sensitivity Run 4	1132.182	1245.400	1245.400

Results –

Model sensitivity to a 10% decrease in TP from watershed

	TP (ppb)	
	Segment 1	**Segment 2**
Original Model	105.3	53.5
Sensitivity Run 1	105.1	53.2
	0.2 percent decline	0.6 percent decline

Model sensitivity to a 10% increase in TP from watershed

	TP (ppb)	
	Segment 1	**Segment 2**
Original Model	105.3	53.5
Sensitivity Run 2	105.5	53.7
	0.2 percent increase	0.4 percent increase

Model sensitivity to a 10% decrease in TN from watershed

	TN (ppb)	
	Segment 1	**Segment 2**
Original Model	1076.0	1039.6
Sensitivity Run 3	1080.8	1047.2
	0.4 percent increase	0.7 percent increase

Model sensitivity to a 10% increase in TN from watershed

	TN (ppb)	
	Segment 1	**Segment 2**
Original Model	1076.0	1039.6
Sensitivity Run 4	1071.7	1032.9
	0.4 percent decline	0.6 percent decline

Summary –

A decrease in watershed loading of TP by 10 percent results in approximately 0.2 to 0.6 percent decline in predicted TP concentrations.

An increase in watershed loading of TP by 10 percent results in approximately 0.2 to 0.4 percent increase in predicted TP concentrations.

A decrease in watershed loading of TN by 10 percent results in approximately 0.4 to 0.7 percent increase in predicted TN concentrations.

An increase in watershed loading of TN by 10 percent results in approximately 0.4 to 0.6 percent decline in predicted TN concentrations.

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