Hydros Consulting

DRAFT TECHNICAL MEMORANDUM

TO: The Cherry Creek Basin Water Quality Authority
FROM: Christine Hawley, Jean Marie Boyer, PhD, PE, and Taylor Adams; Hydros Consulting Inc.
SUBJECT: Cherry Creek Reservoir Water-Quality Modeling Project: Model Calibration and Sensitivity Analyses
DATE: July 31, 2015

This technical memorandum provides key findings of Tasks 3 and 3A of the Cherry Creek Reservoir Water-Quality Modeling Project for the Cherry Creek Basin Water Quality Authority (Authority). Task 3 covers development and calibration of the numerical model. Task 3A is the sensitivity analysis. This memorandum is not a complete documentation of the model (model documentation is Task 6), but it purposefully includes greater detail than a basic summary. This additional level of detail is provided at this time is to support the peer review and provide context for development of management scenarios.

This technical memorandum is organized in seven sections, starting with a summary of key findings and recommendations followed by more in-depth discussions:

1. **Executive Summary**
2. **Introduction**
3. **Conceptual System Understanding Highlights**
4. **Model Development**
5. **Model Calibration**
6. **Sensitivity Analysis**
7. **Recommendations**
8. **References**

There are also four attachments to this technical memorandum:

- **Attachment A – Response to Comments.** This Attachment provides a comment-by-comment response to input received from Dr. Wells (Wells, 2015), the peer reviewer, following preliminary review of the draft calibration on June 22, 2015.
- **Attachment B – Temperature Profiles and Thermistors – Observed versus Simulated**
- **Attachment C – Dissolved Oxygen Profiles – Observed versus Simulated**
- **Attachment D – Detailed Sensitivity Analysis Results**

1 Executive Summary

Tasks 3 and 3A of the Cherry Creek Reservoir Modeling Project have been completed. This up-front summary is a high-level overview of work completed, key findings, and monitoring recommendations.

1.1 Tasks 3 and 3A

For Task 3, a two-dimensional hydrodynamic and water-quality model of Cherry Creek Reservoir was developed using CE-QUAL-W2 (W2), v 3.71. The model simulated the years 2003 through 2013 and was calibrated to observed data. A key calibration target was summertime chlorophyll *a* concentrations, recognizing the site-specific chlorophyll *a* standard which has not been met in recent years. The calibration also focused on the simulation of surface water elevation, water temperature, nutrients, light attenuation, and dissolved oxygen.

Simulated summertime (July through September) average chlorophyll *a* concentrations are compared to observed concentrations in Figure 1. These show a good pattern and value match for all simulated years. Absolute mean error, average error, and root mean squared error are all less than 3 ug/L for the summer average prediction. The model was also successful at simulating other important constituents and the predicted relationships between constituents make sense.

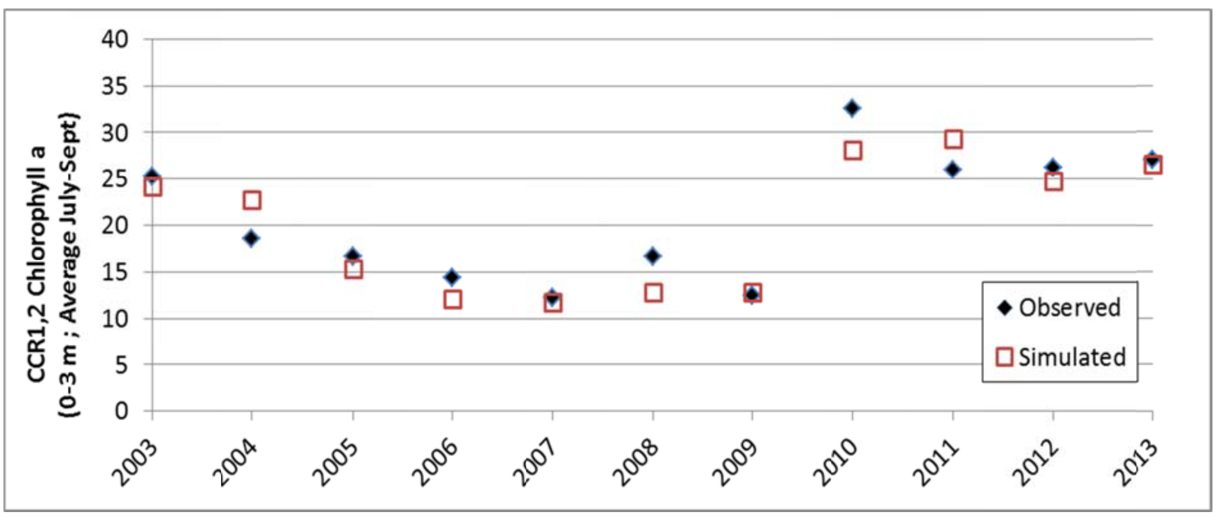

Figure 1. Simulated and Observed Summertime Average Chlorophyll a Concentrations in the Photic Zone, 2003-2013, July – September, CCR1 and CCR2

For Task 3A, a sensitivity analysis was conducted using the calibrated model. The sensitivity analysis focused on assessing modeled water-quality response to major forcing functions[1] for this system. Specifically, effects of the following factors were evaluated:

- Destratification System Mixing;
- Wind;
- Nutrient Loading from Major Tributaries; and
- Internal Loading.

Some of these forcing functions can be managed, and some cannot (e.g., wind). As such, these do not represent realistic management scenarios, but they were defined to provide an understanding of the system and modeling tool to ultimately support selection of management scenarios.

1.2 Key Findings

The model calibration and sensitivity analysis work served to advance the conceptual understanding of the system. Several major findings were generated that provide refined and, in some cases, new perspectives on the system and what can and cannot be achieved through certain types of management. The following subsections highlight some key findings that may be of greatest interest to the Authority in planning for management scenarios.

Overall Water-Quality Dynamics

Briefly, the observed data and modeling show Cherry Creek Reservoir to be very dynamic. This shallow, eutrophic reservoir is subject to significant mixing from wind throughout the year, with the exception of periods of ice cover. The mixing effects of the wind are significant and exceed those of the destratification system. The mixing serves to bring nutrients released from sediments to the upper layers of the reservoir, resulting in algal growth. Through this mixing, nutrients cycle from algal uptake, to algal settling and decay, to release back into the water column to support more algal growth. Nutrients introduced by tributary inflows are also significant contributors to nutrient concentrations throughout the water column. Thus, both internal and external loading of nutrients are important with respect to algal growth.

Nitrogen Limited Conditions are an Important Factor Driving Algal Growth

Both observed data and modeling confirm that the reservoir is strongly nitrogen limited through summer months due to high phosphate (bioavailable phosphorus - PO4) loading. Modeling

[1] Forcing functions are external forces that can cause a water-quality response (e.g., wind, inflow concentrations, changes in operations, etc.)

indicates that, because of this nitrogen limitation, nitrogen-fixing cyanobacteria are active in the reservoir, which is consistent with observations. These cyanobacteria can fix atmospheric nitrogen, thereby introducing additional nitrogen into the reservoir. It is hypothesized that the total inorganic nitrogen (TIN) to PO4 ratio in the reservoir affects algal species assemblages, resulting zooplankton species, and total chlorophyll *a* concentrations in the reservoir. Lower TIN:PO4 ratios further increase growth of nitrogen-fixing cyanobacteria in the model. This is relevant to setting appropriate goals for management of inflowing nutrients.

<u>Tributary Nutrient Concentrations</u>

The simulated algal response to reducing inflow tributary concentrations of nutrients by one-half was tested as part of the sensitivity analysis. Nutrient concentrations were reduced first for Cherry Creek (with no change to Cottonwood Creek), then vice versa. The results for these simulations offer some initial perspective on appropriate and reasonable goals for management of the two watersheds. The simulated 50% reduction in inflow nutrients from Cherry Creek did lower in-reservoir chlorophyll *a* concentrations significantly; however, the results indicate that the current site-specific chlorophyll *a* standard would still not be met for all years (Figure 2).

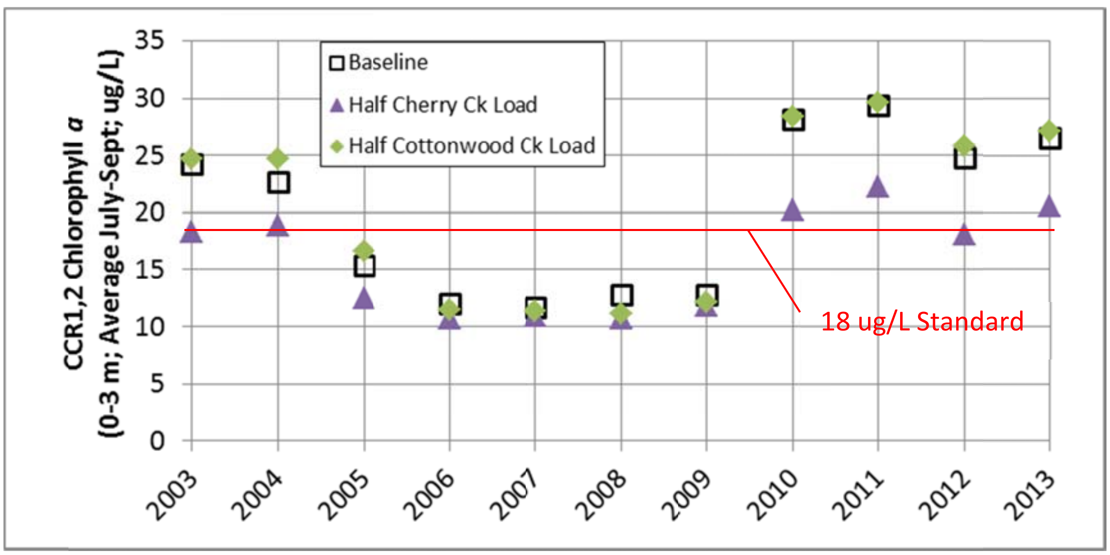

Figure 2. Simulated Average Summertime (July-September) Chlorophyll *a* (CCR2), Half Tributary Nutrient Loading and Baseline Simulations, 2003-2013 (Baseline = Results from Calibrated Model)

Interestingly, halving nutrient concentrations in Cottonwood Creek resulted in small simulated increases in chlorophyll *a* in roughly half of the simulation years. Cottonwood Creek is characterized by lower PO4 and higher inorganic nitrogen; therefore, halving nutrient concentrations increased the relative fraction of PO4 present in the reservoir. As a result, the model simulated increasing growth of nitrogen-fixing cyanobacteria.

The differences in the direction and magnitude of the response for halving nutrient concentrations in each of tributaries are due to 1) the relative differences in flow and 2) the significantly different nutrient concentrations and resulting nitrogen-to-phosphorus ratios.

Based on these findings, it is important for the Authority to view the effectiveness of potential nutrient reduction strategies differently for the two tributaries. Specifically, further reductions in nitrogen loading without corresponding reductions in phosphorus loading could have unintended consequences, encouraging additional nitrogen-fixing cyanobacteria.

Internal Nutrient Loading

Internal loading of nutrients from the sediments can occur by aerobic and anaerobic processes. Anaerobic processes dominate in many reservoirs because, through warmer months, larger reservoirs may stratify, resulting in consistent anaerobic conditions at the sediment-water interface. In Cherry Creek Reservoir, however, because of mixing by wind (and aerators), dynamic oxygen conditions occur at the bottom of the reservoir. In addition, Cherry Creek Reservoir is highly productive and has high concentrations of algae and non-algal organic matter that settles and decays under aerobic conditions. As a result, aerobic processes at the sediment-water interface are also relevant.

Sensitivity analysis simulations were conducted to demonstrate the relative roles of aerobic and anaerobic sediment processes on algal growth. These model runs indicate that aerobic processes generate more internal loading of nutrients than anaerobic processes, though both were significant. This finding is important because the destratification system is designed to address anaerobic loading by bringing additional oxygen from the surface to the bottom of the reservoir. As such, the potential benefits of a destratification system on reducing internal load in this shallow, polymictic, productive reservoir are limited.

Destratification System

The modeling indicates that the current destratification system increases vertical mixing in the reservoir by about 60% and results in slightly warmer water at the bottom of the reservoir. The data and the model also show that this results in an increase in DO at the bottom of the reservoir at CCR3. There may also be a slight decrease in DO at the bottom of the reservoir at CCR2 based on observed data, which would indicate an induced increase in the rates of oxygen consumptions by sediment processes in response to aerator operation (termed “induced sediment oxygen demand [SOD]”). Similar responses to increased mixing or oxygen addition in systems with high SOD have been observed elsewhere (e.g., Ashley, 1983, Soltero et al., 1994, Moore et al., 1996 and Gantzer et al., 2009).

Sensitivity analysis simulations indicate that the existing destratification system has a small beneficial effect on summertime chlorophyll *a* concentrations, reducing summertime averages by less than 1 ug/L up to 4 ug/L from 2008-2013 (Figure 3). Even this relatively small effect may be somewhat overstated since the model is not currently simulating all of the mechanisms behind the induced SOD effect.

The model was run assuming a 10-fold increase in destratification system induced missing, relative to current destratification system mixing (2008-2013 only). Results indicate this greater mixing would produce consistently aerobic conditions at the bottom of the reservoir and serve as an estimate of the maximum potential effectiveness of a destratification system in Cherry Creek Reservoir. This simulation resulted in an average decrease in simulated average summertime chlorophyll *a* of 5 ug/L, ranging from 2 to 9 ug/L (Figure 3). The simulation indicates that the 18 ug/L chlorophyll *a* standard would still not have been met in recent years.

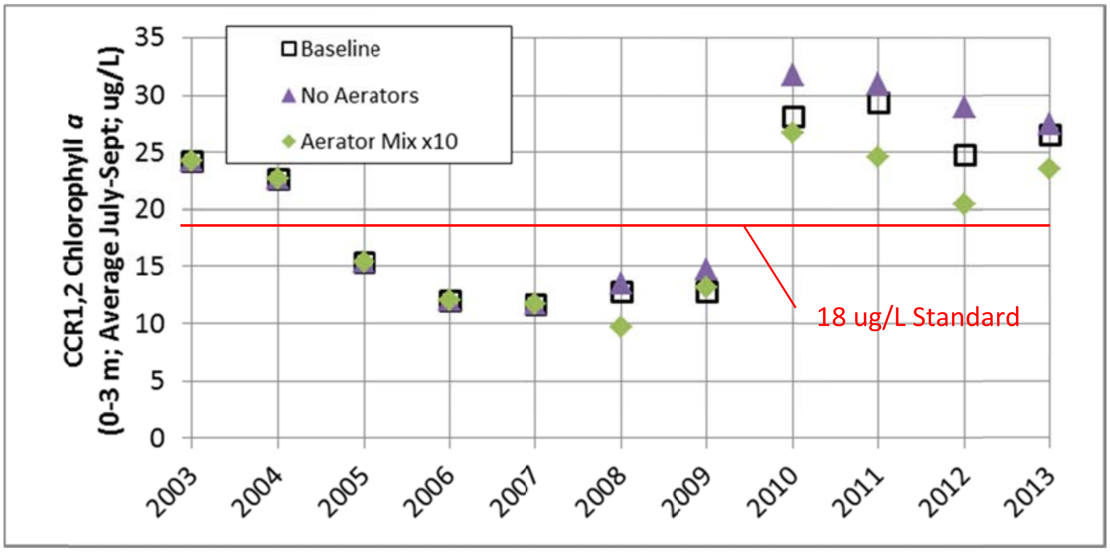

Figure 3. Simulated Average Summertime (July-September) Chlorophyll *a* (CCR2), Aerator Mixing Simulations Compared to Baseline, 2003-2013 (Note that the Aerator Mix x10 model run only applies to 2008-2013, when the destratification system existed.)

In summary, the destratification system does bring oxygen from the surface to the bottom of the reservoir to reduce anaerobic internal loading. The design target of at least 5 mg/L DO at the bottom is not being achieved, however (Boyer, et al., 2014a). The system is likely under-designed, given the apparent induced SOD and the greater potential for reduction in algal concentrations simulated by the model for greater mixing. If the system did achieve the target 5 mg/L at the bottom, model simulations indicate that chlorophyll *a* concentrations would be reduced, but the site-specific chlorophyll *a* standard would not be met. This is due to significant aerobic internal loading and high inflow tributary loading to this shallow and productive reservoir.

1.3 Monitoring Recommendations

Through the model development and calibration process, recommendations were generated regarding monitoring. Overall, the existing water-quality dataset is an excellent record of conditions in the reservoir and was critical to successful model development. The following recommendations are offered to support future modeling and ongoing development of the conceptual understanding.

- **Continuous In-Reservoir DO Measurement** – It is strongly recommended that the Authority install continuous DO probes at 1 m below the surface and at 0.5 m above the bottom of the reservoir at CCR2. Similar monitoring at CCR3 could also be helpful; however, continuous DO data at CCR3 is considered less critical.

- **Review of DO Data Collection Procedures** – In response to some uncertainty and questions about the observed DO dataset, it is recommended that calibration and measurement protocols be reassessed and documented in greater detail. Additionally, periodic duplicate observations with a second probe may be appropriate.

- **Meteorological Data** – Given the major effects of weather conditions on the water-quality response of Cherry Creek Reservoir, is it is recommended that the Authority coordinate with Colorado Parks and Wildlife (CPW) to check and improve data collection and data management protocols for the CPW met station located next to the reservoir. Alternatively, a new meteorological station could be installed at the dam, monitoring at a minimum, air temperature, solar radiation, wind speed, and wind direction.

- **Inflow Algae** – Cherry Creek and Cottonwood Creek include areas of briefly impounded water that exhibit a green color in some aerial photographs at times. Therefore, as discussed in Boyer et al. (2014a), sampling of algae at CC-10 and CT-2 is recommended to improve the understanding of algal loading from the tributaries. This should include both storm and non-storm-event sampling. Results could be reviewed each year to determine whether or not to continue this sampling.

- **Critical Sampling to Continue** – Continued collection of thermistor data and algal species and biovolume data is critical. Additionally, more recently added data collection of organic carbon and zooplankton are also expected to be valuable to help refine areas of uncertainty in the model.

- **Possible Sampling to Discontinue** – The D-series profile transect data was not critical for modeling. Given the dynamic nature of the system, the thermistor data are much more useful. Additionally, for the purposes of modeling, location CCR1 data collection could be discontinued.

2 Introduction

Cherry Creek Reservoir (Figure 1) is a 13,000 acre-ft (AF; ~16 million m^3) reservoir near Denver, Colorado. It was constructed in 1950 by the U.S. Army Corps of Engineers (USACE) for flood control purposes. Cherry Creek Park is comprised of the reservoir and surrounding land. The park is a popular recreation area, receiving approximately 1.5 million visitors per year to enjoy boating, fishing, camping, swimming, hiking, and more (Colorado Parks and Wildlife [CPW], 2013). The reservoir is also considered a high-quality walleye fishery and is heavily relied upon for providing eggs to fish hatcheries for subsequent stocking in lakes throughout the state.

Figure 4. Image of Cherry Creek Reservoir Looking South (Photo by Harry Weddington, USACE)

The Cherry Creek Basin Water Quality Authority (Authority) exists to protect and improve water quality in the reservoir to meet applicable water-quality standards. Cherry Creek Reservoir has been the focus of major watershed and in-reservoir monitoring and mitigation projects to improve water quality for its designated beneficial uses. The reservoir exhibits high chlorophyll *a* concentrations and has failed to consistently meet the current site-specific state standard of 18 ug/L, assessed as a July – September average.

In 2008, a destratification system was installed in the reservoir to mix oxygen from the surface to the bottom with the intention of reducing internal loading of nutrients and ultimately chlorophyll *a* concentrations. Mixing from the aerators was also intended to reduce blue-green algae concentrations by disrupting their buoyancy advantage over other algal types. In total, 123 aerator heads were installed covering roughly 350 of the 850 acres (1.4 of the 3.4 million m^2) of the reservoir (Figure 2). The aerators release air roughly 0.75 m above the sediment-water interface and are generally operated from April through November.

Figure 5. Aerator Footprint; Image from GEI Presentation, April 30, 2013

Since the start of aerator operations in 2008, the site-specific standard for chlorophyll *a* has only been met in two out of seven years, with all years exceeding the standard in the recent five-year compliance period (Figure 6).

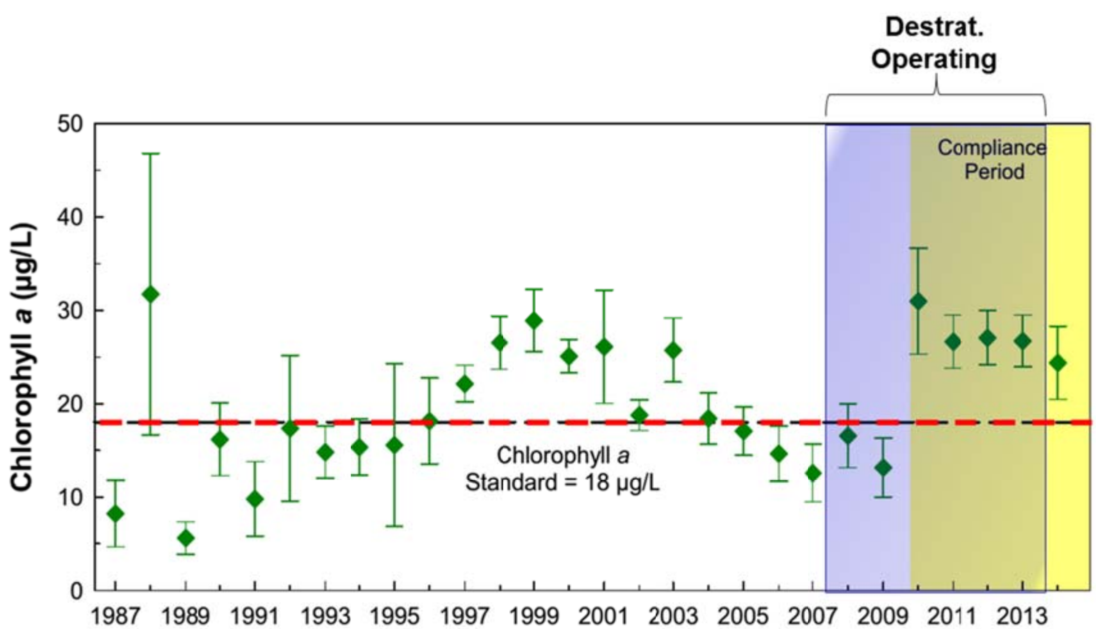

Figure 6. Cherry Creek Average Summer Chlorophyll *a* Compared to Site-Specific Standard, 1987-2013; Graph from GEI, 2015

Based on this, the Authority identified a need to develop a water-quality model of the reservoir to:

- Better understand the mechanisms behind chlorophyll *a* exceedances,
- Determine the impacts of the aeration system, and
- Provide a tool that can help predict effects of future management strategies.

To address this need, the Authority commissioned development of a water-quality model to simulate reservoir response from 2003 through 2013. Development and calibration of the model is the subject of this technical memorandum.

3 Conceptual System Understanding Highlights

The conceptual system understanding developed based on review of existing data is presented in a technical memorandum provided for Task 1 (Boyer et al., 2014a). This section provides highlights of key aspects of that analysis, as well as some updates and additions, to support subsequent discussions of model development and calibration. To the extent observed data are available, the discussion considers observations over the model simulation period (2003 and 2013).

3.1 Bathymetry and Storage

The 13,000 AF reservoir covers approximately 850 acres (3.4 million m^2). The average depth of the reservoir is ~9 ft (~2.7 m), with a maximum depth of ~27 ft (~8.2 m). The bathymetry of the reservoir is presented in 1 ft contours in Figure 7, along with the location of the three main in-reservoir, water-quality sampling locations.

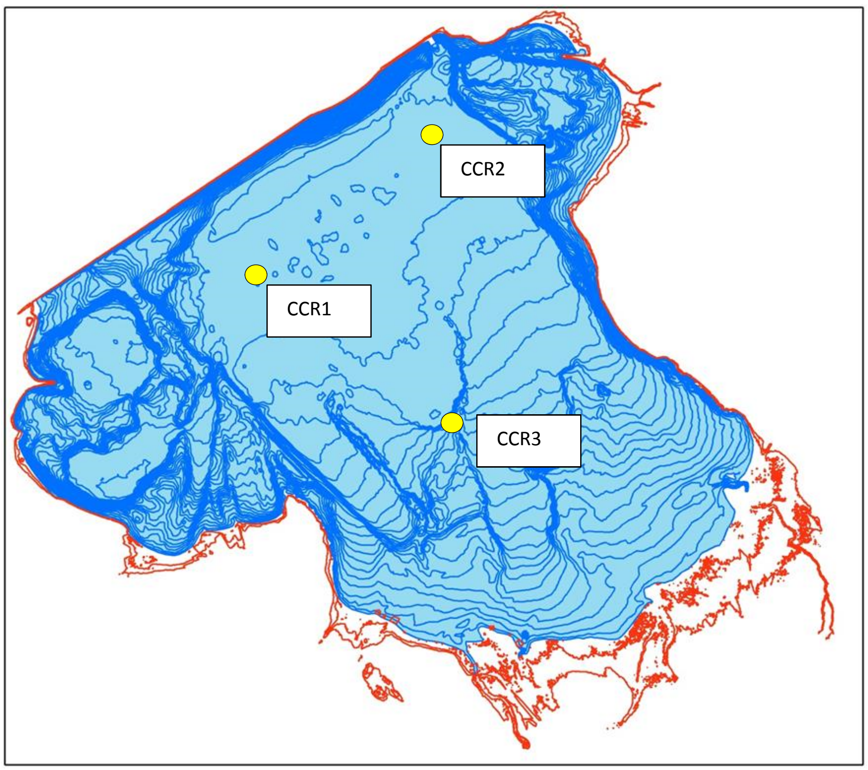

Figure 7. Cherry Creek Reservoir Bathymetry, 1 Ft. Contours, and In-Reservoir Sampling Locations

The elevation contours shown here were measured as part of two separate studies. Contours from elevation 5,512' to 5,550' (Local Project Datum) were measured at one-foot intervals by Absolute Natural Resources (ANR) in 2013, and contours from elevation 5,552.2' to 5,558.2' at two-foot intervals were from a LiDAR survey carried out by Southeast Metro Stormwater Authority (SEMSWA) in 2008. These two sets of contours were merged by Shane Michael of Leonard Rice Engineers and delivered to Hydros by email on 3/28/14 as a shapefile.

The reservoir water surface elevation is recorded daily by the USACE (Figure 8). These data show that the water surface elevation has varied only by ~6.5 ft (2 m) between 2003 and 2013. Due to reservoir operations, there has been a general decrease in the average annual storage over that period of time.

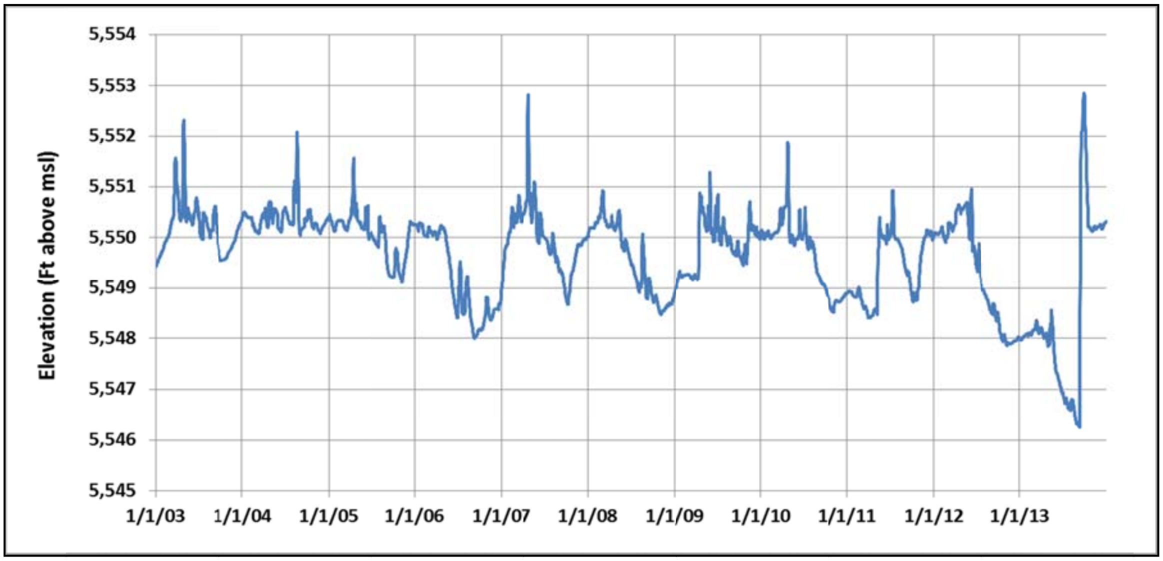

Figure 8. Observed Reservoir Water Level, 2003-2013

3.2 Hydrology

There are two major gaged tributaries flowing into Cherry Creek Reservoir: Cherry Creek and Cottonwood Creek (Figure 9). Other inflows include alluvial inflow, ungaged surface inflow, and precipitation. Water is released from the reservoir on the northwest end of dam from the bottom of the reservoir. Water is also lost through evaporation and outflow seepage.

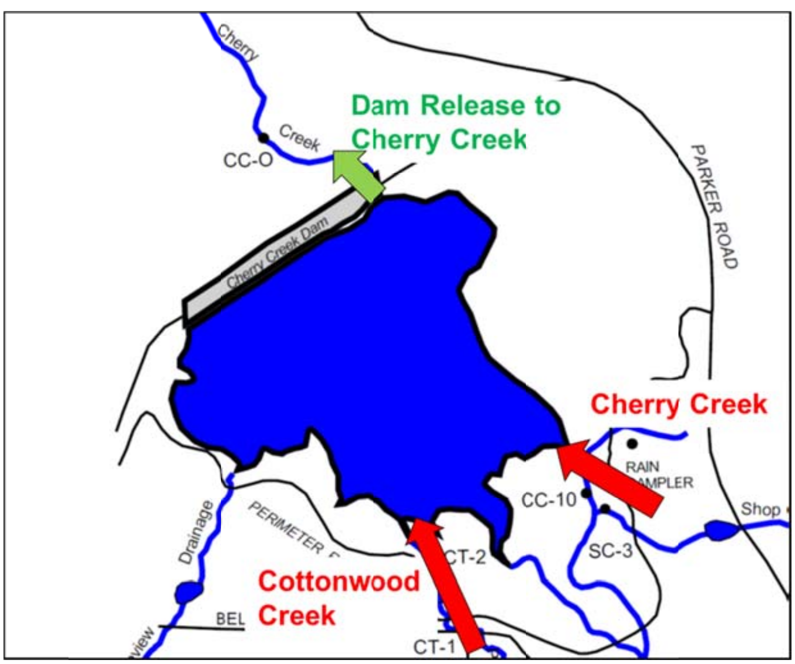

Figure 9. Cherry Creek Reservoir Gaged Inflows and Outflows

To evaluate the system hydrology and develop a water balance for the model, observed precipitation and daily flow data were compiled. For ungaged flows, the following calculations and assumptions were made:

- **Outflow seepage** was estimated from USACE measurements conducted during dam construction[2].
- **Ungaged surface inflows** for the 5.2 square mile direct watershed were estimated from precipitation using the Rational Method, applying averaged parameters developed by Brown and Caldwell for watershed phosphorus modeling (Brown and Caldwell, 2009).
- **Evaporation** was estimated by applying the methodology outlined in the ASCE Standardized Reference Evapotranspiration Equation (Walter et al., 2001) and the FAO Irrigation and Drainage Paper 56 (Allen et al., 2006). The long-term average annual evaporation rate resulting from these calculations is ~ 42 in/yr (107 cm/yr), which is very close to the expected value for this region and close to the value reported by USACE in for the reservoir.
- **Alluvial inflow seepage** was assumed to be a constant equal to the 2,200 AF/ yr (~2.7 million m^3/yr) estimate developed by Lewis et al. (2005). Attempts to solve for alluvial inflow seepage as the closure term of the water balance produced similar seepage estimates in some years. However, there were seasons when increases in the closure term were calculated that did not correlate with times when inflow seepage would be expected to increase significantly. This suggested other sources of error in the water balance and led to assumption of constant inflow seepage.

Compiling these terms on a daily basis, along with daily reservoir storage, a water balance was developed. The closure term from these inputs was assumed to be error in the reported gaged inflows. Errors were smoothed and split proportionately between Cherry Creek and Cottonwood Creek inflows, adjusting these terms on a daily basis to close the water balance. The resulting adjustments to Cherry Creek and Cottonwood Creek flow rates were less than 5% of the observed flow data, which is well below the typical 10 to 15% estimate of accuracy for good to fair streamflow monitoring (Risley and Gannett, 2006).

The water balance inflows for 2003-2013 in Figure 10 show that Cherry Creek is the dominant source of water into the reservoir, averaging more than 60%. Cottonwood Creek is the next

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[2] As part of a supplemental design analysis carried out during preparation for reservoir construction in 1947, USACE calculated the horizontal and vertical hydraulic conductivity of the soils underlying Cherry Creek Reservoir. These conductivity estimates used to develop a linear relationship between reservoir surface elevation and seepage outflow. Application of this elevation-seepage relationship results in a relatively constant estimated outflow seepage.

largest term at 20%, and is nearly twice the next highest term, alluvial inflow. The water balance outflows for 2003-2013 are summarized in Figure 11, and show outflow seepage to be relatively small compared to outflow released at the dam and evaporation.

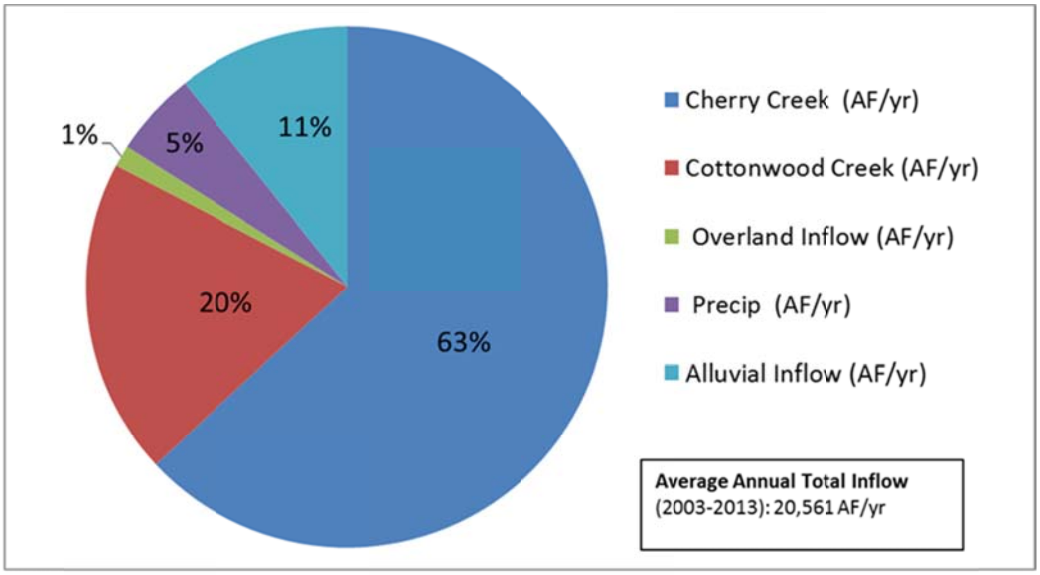

Figure 10. Relative Fractions of Cherry Creek Inflows, 2003-2013

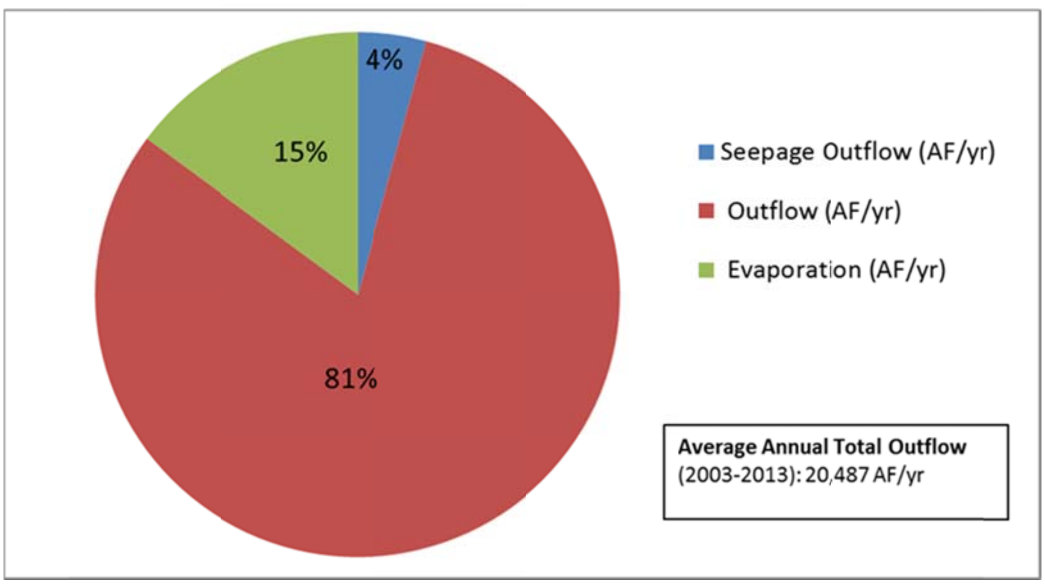

Figure 11. Relative Fractions of Cherry Creek Outflows, 2003-2013

The daily inflow pattern exhibits a wide range of inflow rates, characterized by periodic storm event hydrograph peaks (Figure 12). Over the modeling years (2003-2013), total annual inflows tend to be similar to total annual outflows each year (Figure 13). Variability among the years is apparent in the ratio of Cherry Creek to Cottonwood Creek flows and in the total volume of water moving through the reservoir.

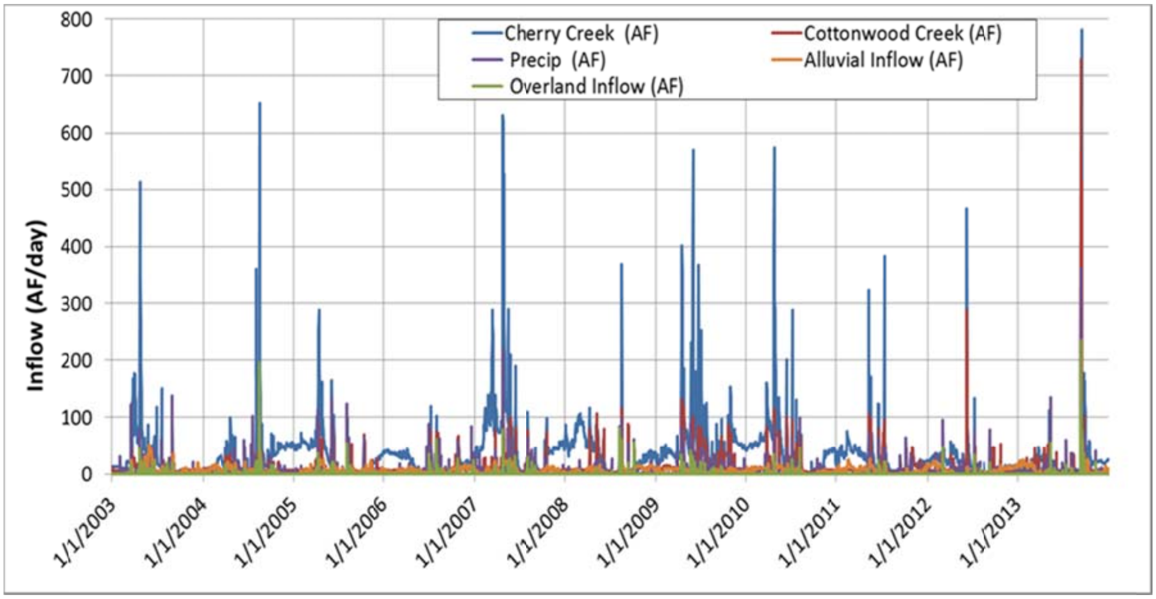

Figure 12. Cherry Creek Reservoir Daily Inflow Rates, 2003-2013

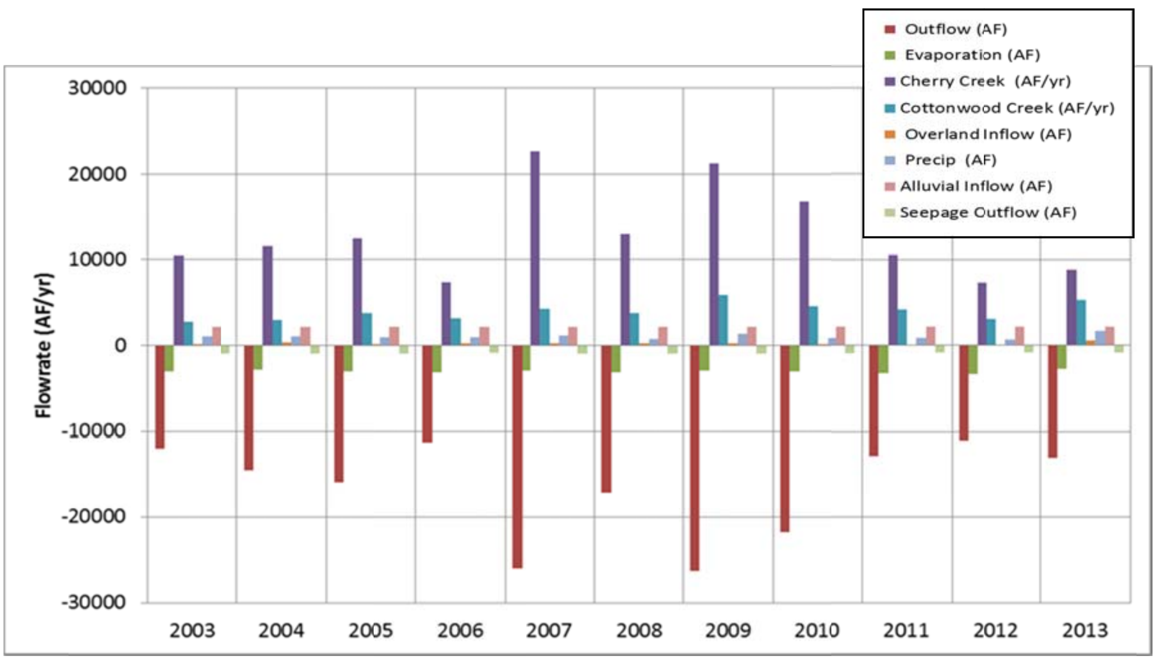

Figure 13. Cherry Creek Reservoir Annual Balance, 2003-2013

Estimated annual hydraulic residence time, based on observed outflows, averaged 10.4 months from 2003-2013 (Figure 14). The average annual residence time ranged from six months in 2009 to 14 months in 2012.

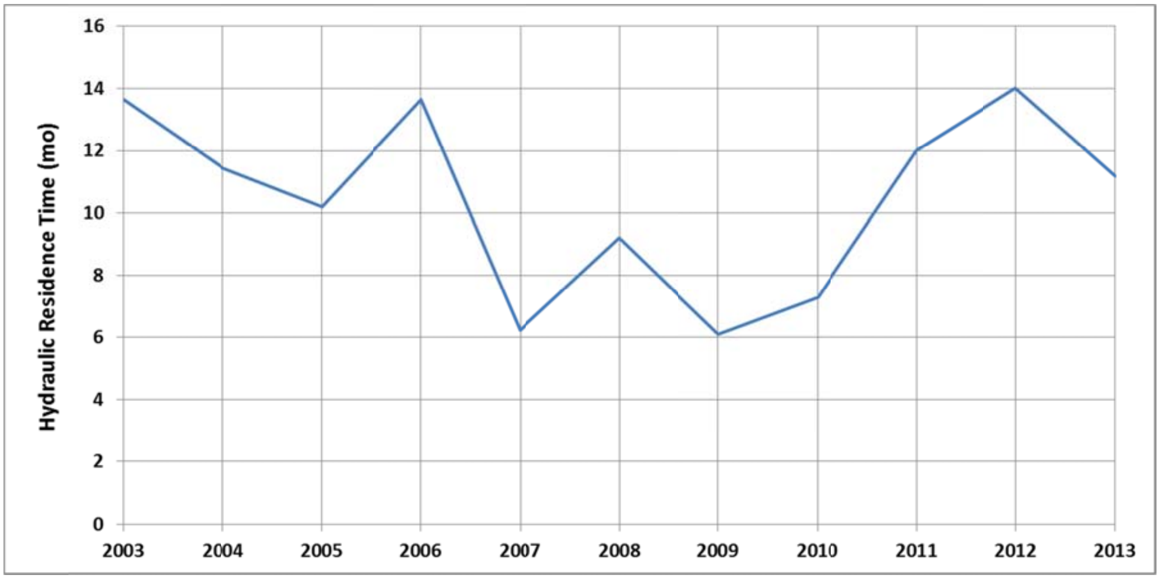

Figure 14. Annual Hydraulic Residence Time (Flow Based on Reservoir Releases)

3.3 Inflow Water Quality

Water-quality data are collected at the inflow locations to Cherry Creek Reservoir on Cherry Creek and Cottonwood Creek at gages CC-10 and CT-2, respectively. Groundwater-quality samples are also collected at MW-9, located approximately 440 ft (~130 m) upgradient of the reservoir next to Cherry Creek. Inflow water quality is characterized by high nutrient and TSS concentrations and has been the subject of numerous water-quality improvement projects implemented by the Authority.

Surface inflow and groundwater soluble reactive phosphorus (SRP) concentrations are presented in Figure 15. Groundwater SRP concentrations are consistently high, averaging 188 ug/L. SRP concentrations in Cherry Creek at CC-10 vary more, ranging from less than 100 ug/L to over 350 ug/L, with a high average concentration of 170 ug/L. In Cherry Creek, there is a seasonal pattern of higher SRP concentrations in the summer months. Cottonwood Creek exhibits much lower SRP concentrations, averaging 18 ug/L. The combined, volume-weighted average concentration for Cherry and Cottonwood Creeks is approximately 120 ug/L SRP for 2003-2013 (~190 ug/L for total phosphorus [TP]).

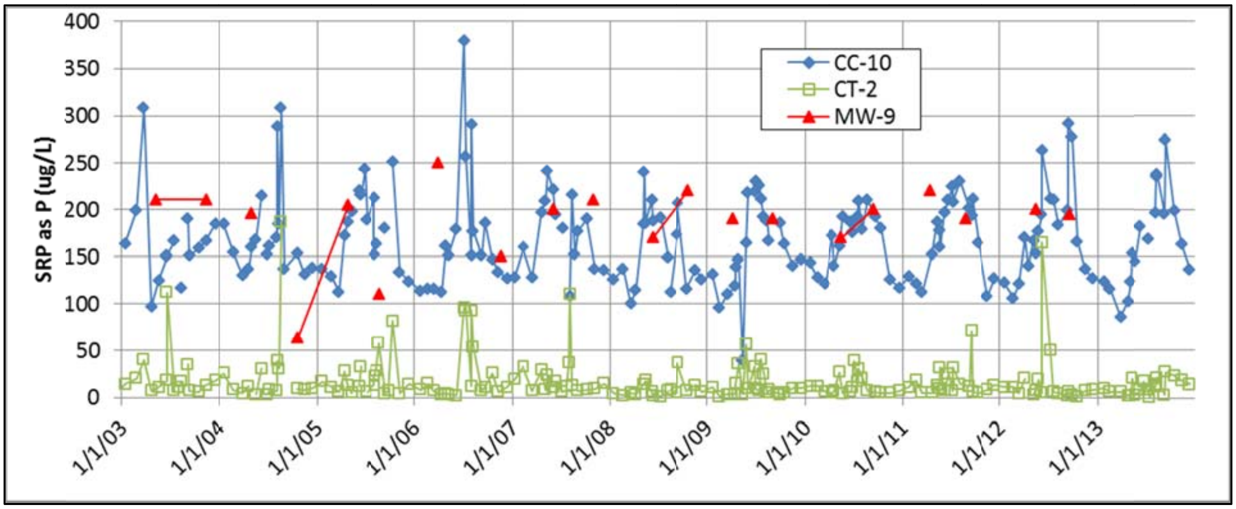

Figure 15. Inflow Soluble Reactive Phosphorus Concentrations, 1999-2013

Inflow nitrate and ammonia concentrations exhibit different patterns compared to SRP. For ammonia and nitrate, Cottonwood Creek concentrations are higher than those in Cherry Creek, particularly before 2009. Peak nitrate concentrations tend to occur in winter months in both Cherry Creek and Cottonwood Creek. The average combined Cherry and Cottonwood Creek volume-weighted inflow concentrations from 2003-2013 are estimated to be 840 ug/L and 83 ug/L for nitrate and ammonia, respectively. Note that MW-9 ammonia data from 2004-2009 (shown on Figure 17) were not included in calculations of loading. Values provided are noted as being below the detection limit and the detection limit was high.

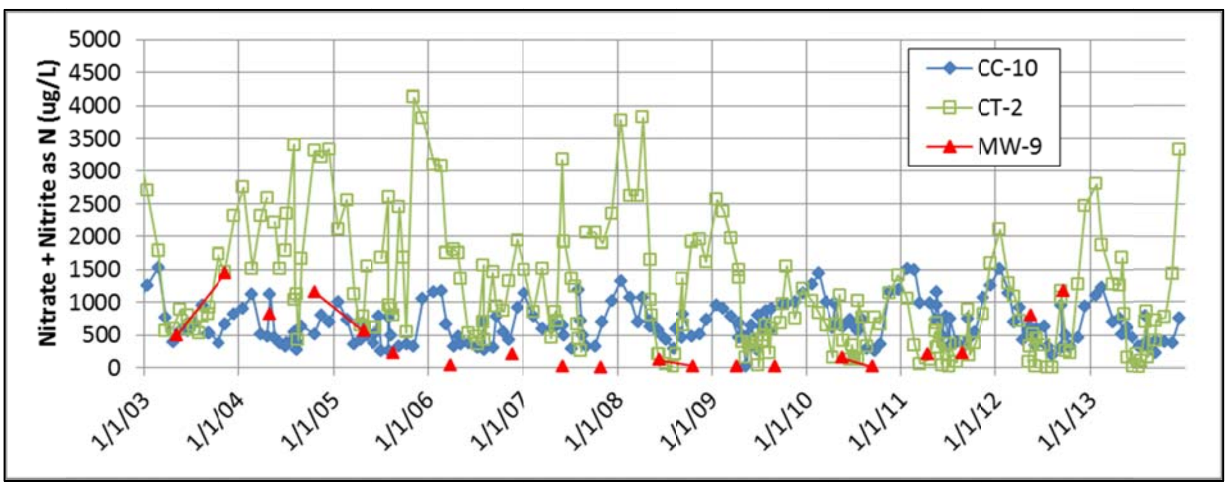

Figure 16. Inflow Nitrate + Nitrite Concentrations, 2003-2013

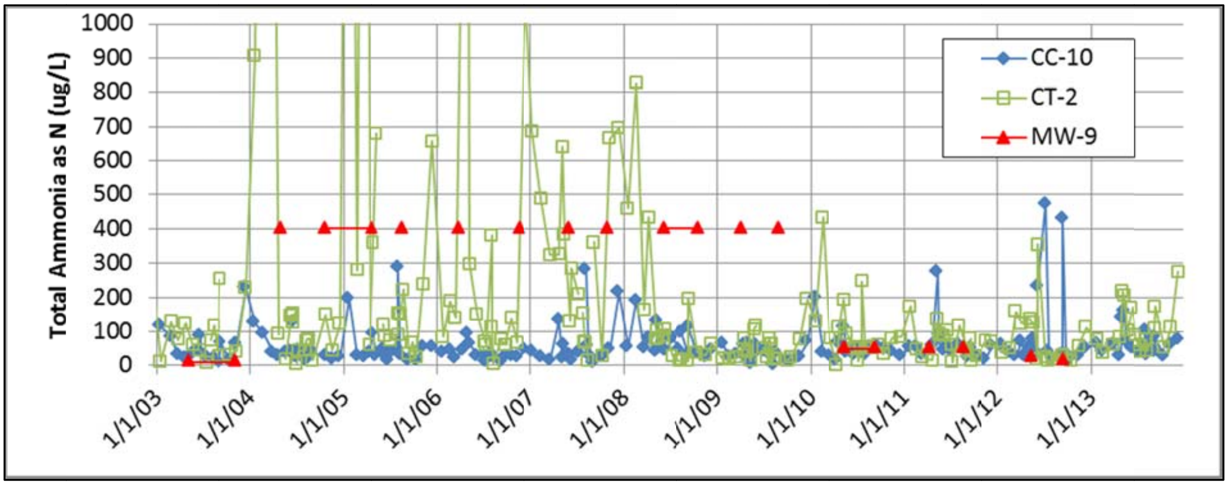

Figure 17. Inflow Ammonia Concentrations, 2003-2013

Looking more closely at the change in nitrate and ammonia concentrations in 2009, Figure 18 presents the ratios of TIN (total inorganic nitrogen = nitrate plus ammonia) to SRP for Cottonwood Creek and Cherry Creek from 2000 through 2013. A period of higher relative TIN:SRP ratios for Cottonwood Creek is apparent between roughly 2004 and 2009. The volume-weighted average concentrations for SRP were fairly consistent comparing periods before and after 2009, but there was a shift in nitrate and ammonia after 2009, as shown in Table 1. A shift in TIN:SRP ratios is relevant because it can affect the in-reservoir algal response and assemblage in eutrophic lakes (e.g., Barica et al., 1980, Smith, 1983, Schindler et al., 2008).

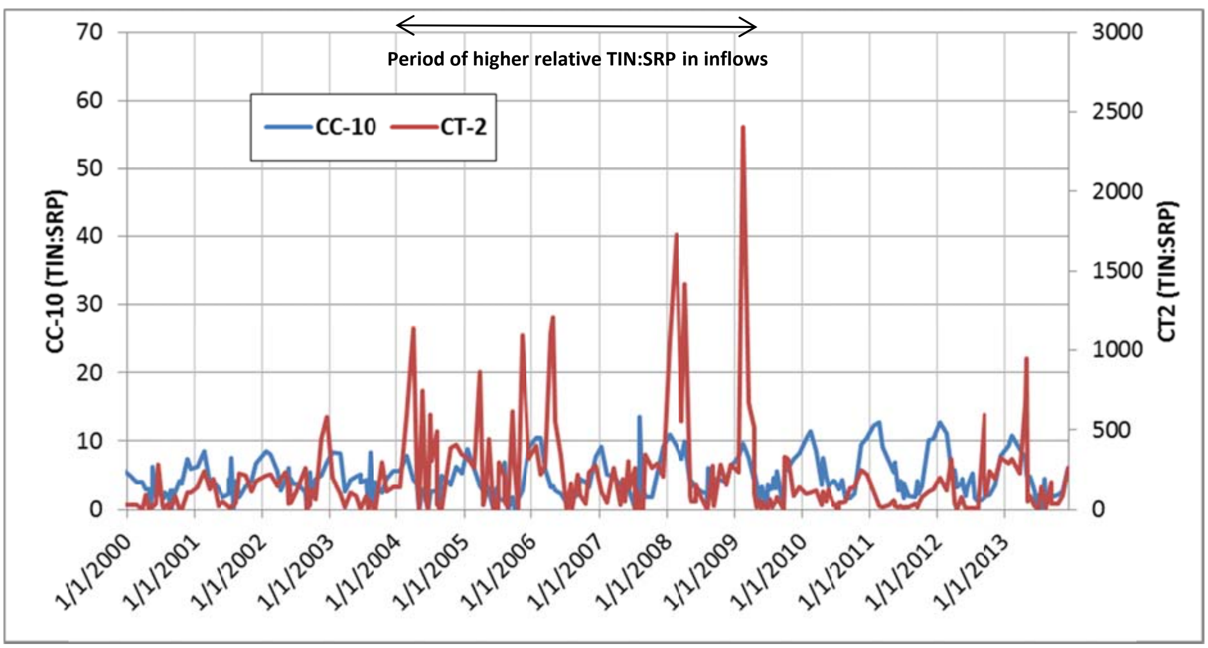

Figure 18. Observed Ratios of TIN to SRP for Cherry Creek (CC-10) and Cottonwood Creek (CT-2); Note Different Scales, 2000-2013

Table 1. Comparison of Volume-Weighted Average Nutrient Concentrations for Inflows, 2003-2009 and 2010-2013

Averaging Period	Volume-Weighted Average Inflow Concentration (ug/L) (Cottonwood Creek and Cherry Creek)		
	SRP	Nitrate	Ammonia
2003-2009	119	880	97
2010-2013	114	782	59

Total suspended solids (TSS) concentrations are also relatively high on the inflows (Figure 19). Median TSS concentrations for Cherry Creek and Cottonwood Creek are similar for 2003-2013, at 28 and 29 mg/L, respectively. Much higher TSS concentrations, however, can be observed during storm events on each of the tributaries. Both tributaries have shown an apparent reduction in non-storm event TSS concentrations since 2008, likely reflecting water-quality improvement projects in the watersheds.

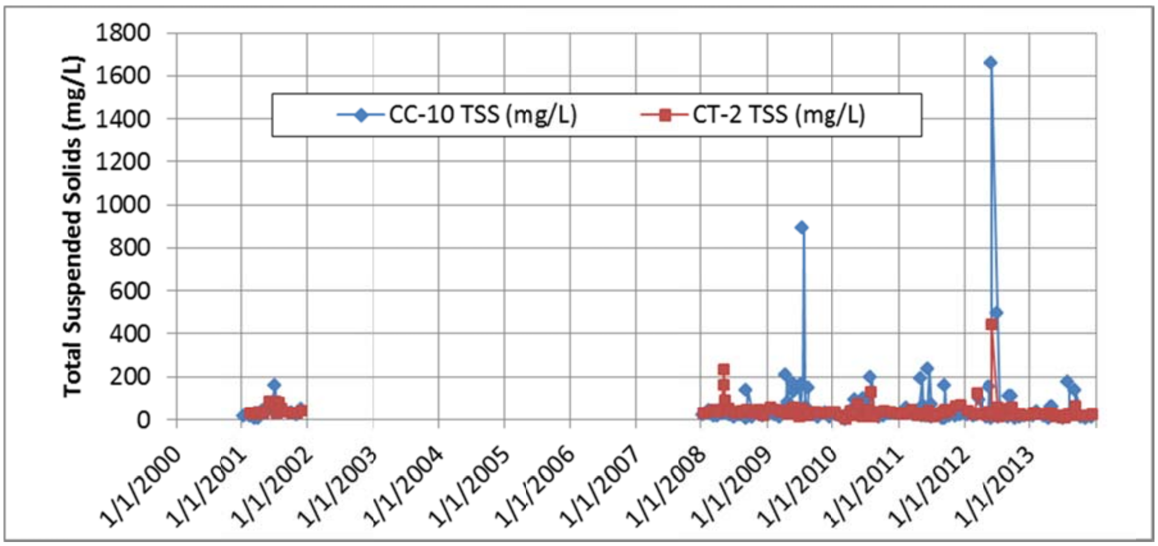

Figure 19. TSS Concentrations in Cottonwood Creek (CT-2) and Cherry Creek (CC-10). 2000-2013

3.4 Meteorology

Meteorological data were compiled from four locations (Figure 20) to support the development of the conceptual understanding and compile inputs for the model. These included KAPA (Centennial Airport), CPW Met (Colorado Parks and Wildlife Met Station at Cherry Creek), KBKF (Buckley Air Force Base), and LRSS (Lowry Range Solar Station). KAPA data were used for all but solar radiation and air temperature. CPW Met data were used for solar radiation and air temperature. Unfortunately, wind data from CPW Met could not be used due to extensive data formatting and date-stamp issues. Missing data for air temperature were filled in from KAPA, and missing data for solar radiation were filled in from LRSS. Data from KBKF were used as a secondary backfill source for missing data which could not be filled from other sources.

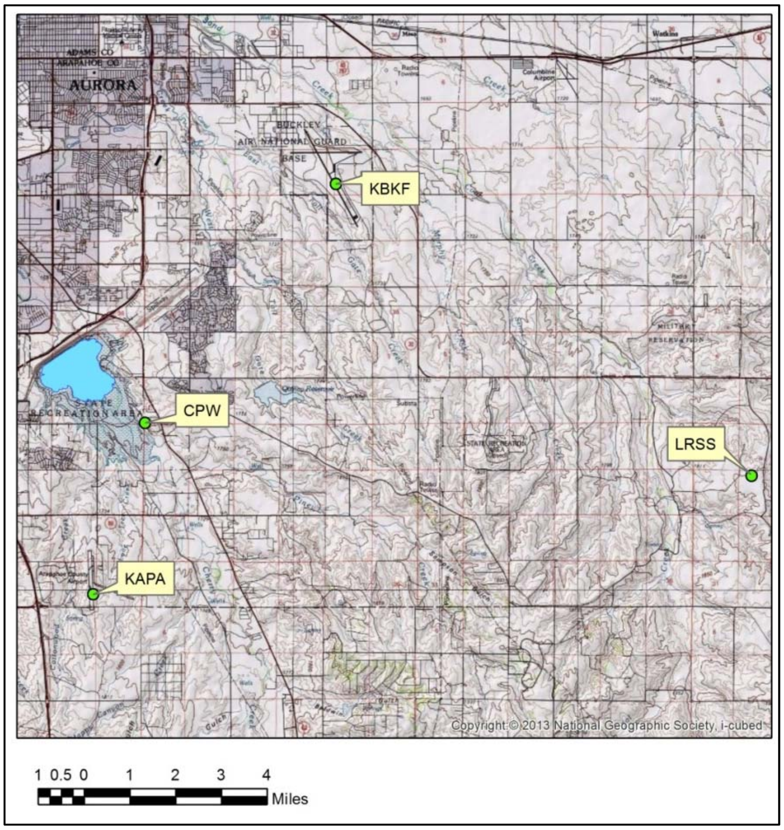

Figure 20. Locations of Met Stations Used to Compile Meteorological Inputs

Meteorological conditions at the reservoir are characterized by sunny and warm summers, relatively cool winters, low precipitation, and high winds. Peak summer air temperatures range from 96 °F (36 °C) to 107 °F (41.7 °C) for the simulation period of 2003-2013. Within that period, the warmest summers were 2003, 2005, and 2012. The coolest summers were 2004 and 2009. Annual winter low temperatures range from 0.2 °F (-18 °C) to 12 °F (-11 °C). The coolest months in the simulation period were February of 2007 and February of 2012. The average precipitation of ~17 inches per year (43 cm/yr) is typical for the semi-arid, high plains. Finally, average wind speeds (at the 10 m elevation of the wind data from KAPA) were approximately 9 mph (4 m/s). The wind comes most frequently from the south or north (Figure 21) and is characterized by occasional strong gusts, with calm conditions occurring during 10% of the record. The reservoir has little natural sources of physical sheltering from the wind.

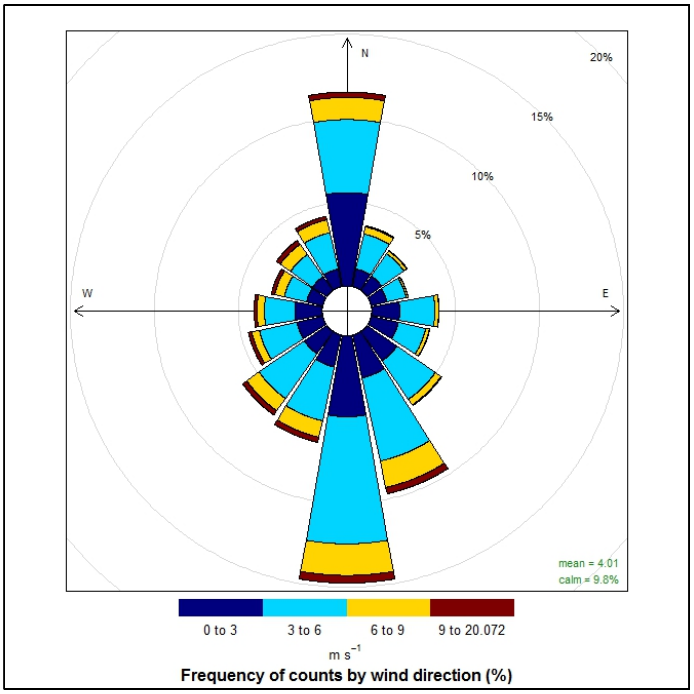

Figure 21. Wind Rose, 2003-2013, KAPA Data from 10 m

3.5 In-Reservoir Water Quality

Water-quality data from stations CCR1, CCR2 and CCR3 (Figure 7) were reviewed in detail in development of the conceptual understanding. This section presenting highlights focuses primarily on CCR2, the deepest sampling location. Data reflect a warm, relatively shallow, eutrophic, polymictic reservoir.

Temperature profile data indicate the reservoir is frequently isothermal or nearly isothermal through the summer months (Figure 22). Peak surface temperature each summer ranges from ~23 °C (73 °F) to ~25 °C (77 °F). Thermistor data indicate that top to bottom temperatures in the reservoir vary from 0 to 4 °C from spring through fall (Figure 23). As a result, temperatures at the deepest part of the reservoir reach relatively high peak temperatures, ranging from ~21 °C (70 °F) to ~23.5 °C (74 °F).

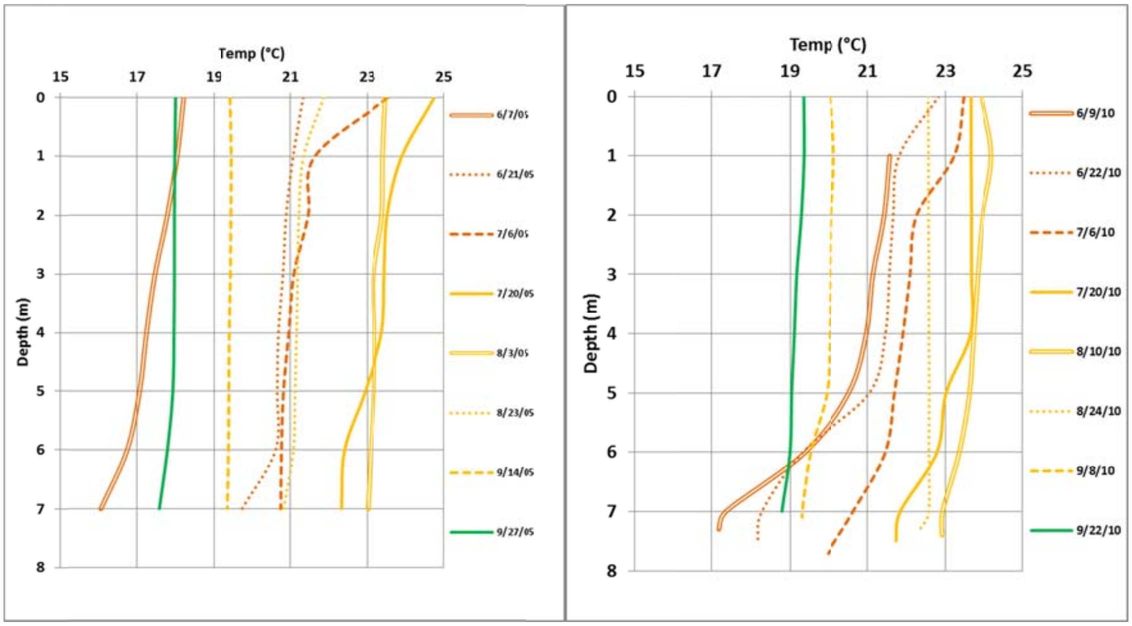

Figure 22. June through September Temperature Profiles at CCR2, 2005 and 2010

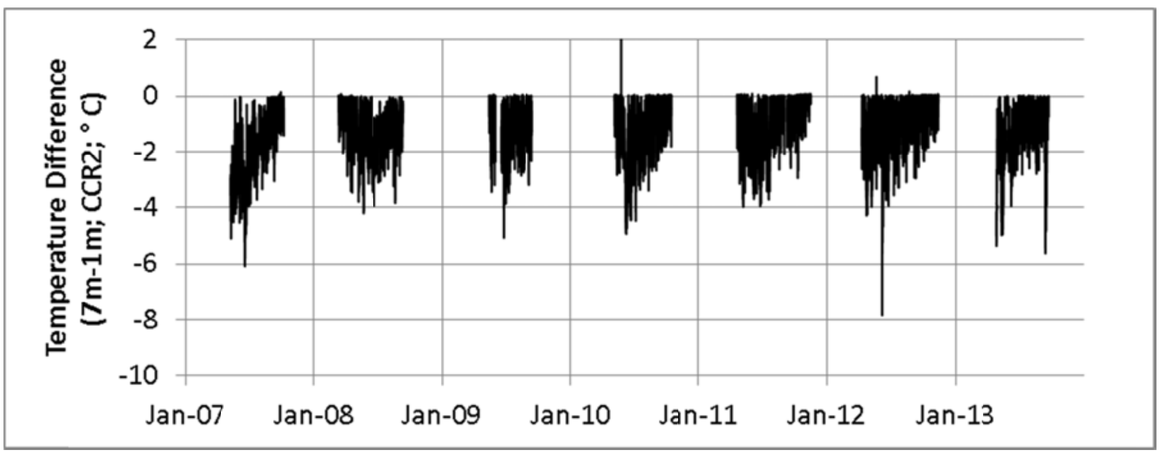

Figure 23. Thermistor-Measured Temperature Difference between 1 m and 7 m at CCR2, 2007-2013

While the reservoir exhibits isothermal conditions periodically throughout the summer, there are typically still vertical concentration gradients apparent in dissolved oxygen (DO) and nutrient data through the summer months. Dissolved oxygen data from 1 m and the bottom-most observation (usually within 1 m of the bottom[3]) are plotted from CCR2 and CCR3 in Figure 24. These data show anoxia or hypoxia at the bottom the reservoir at CCR2 each summer, and higher concentrations at the top. This gradient is driven by high sediment oxygen demand (SOD)

[3] Note that "Bottom" data at CCR2 and CCR3 prior to 2006 are often more than one meter above the bottom, resulting in higher DO values.

at the bottom and reaeration and photosynthesis at the top. Since initiation of the aerators, bottom DO concentrations have decreased slightly at CCR2 and increased at CCR3. The decrease at CCR2 likely reflects induced sediment oxygen demand. This is a phenomenon observed in other systems with oxygenation / circulation systems (e.g., Ashley, 1983, Soltero et al., 1994, Moore et al., 1996 and Gantzer et al., 2009), reflecting increased gradients and reductions in the diffusive boundary layer at the sediment water interface. Induced sediment demand occurs when existing potential for DO consumption exceeds available DO. It is expected, as supported by sediment core data, that the SOD at CCR2 is higher than that at CCR3, possibly explaining the difference in response to the destratification system.

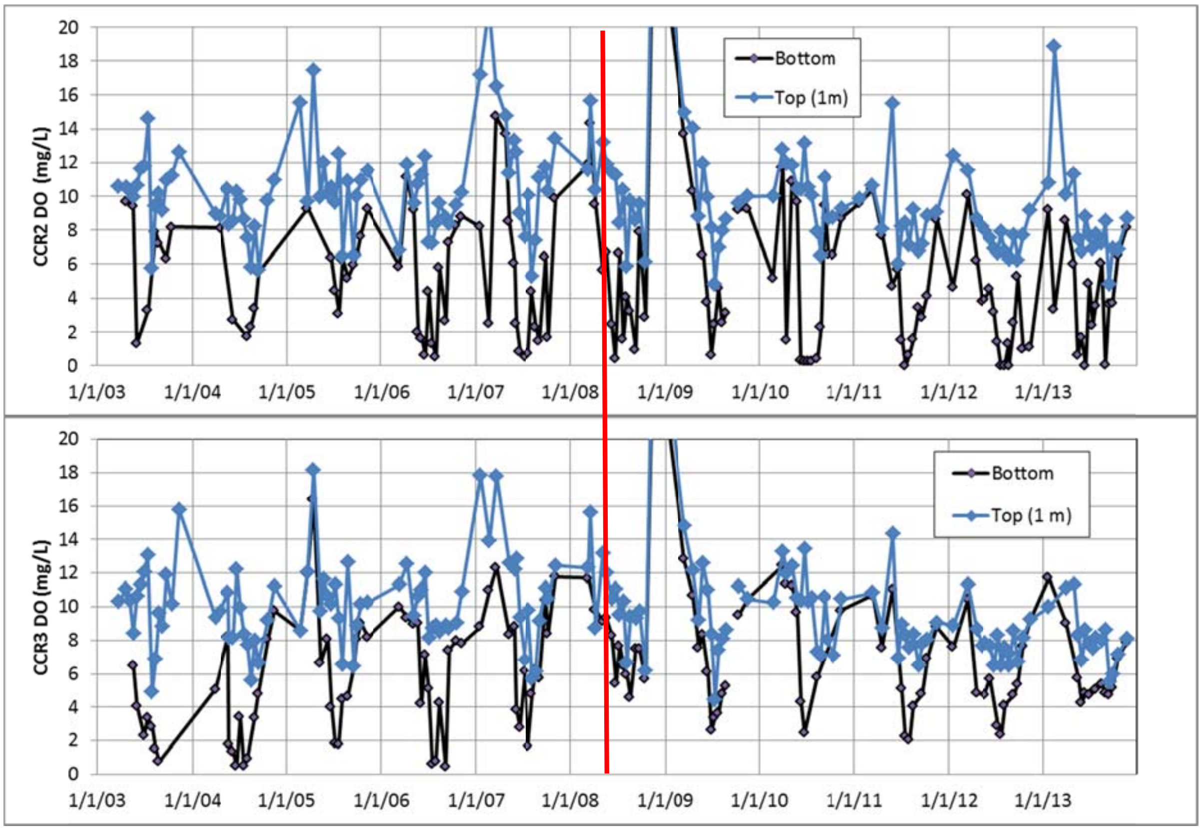

Figure 24. Dissolved Oxygen Concentration at 1 m and Near the Bottom of the Reservoir at CCR2 and CCR3; Red Line Indicates Start of Aeration System

Top and bottom concentrations for nitrate and ammonia are shown in Figure 25. Concentrations of inorganic nitrogen species tend to be relatively low at the top of the reservoir, with an average of 12 ug/L for nitrate and 23 ug/L for ammonia. Recalling that the volume-weighted average inflow concentrations for nitrate and ammonia are 840 ug/L and 83 ug/L, respectively, it is apparent that concentrations at the top of the reservoir show a pattern of algal uptake. At the bottom of the reservoir, nitrate concentrations tend to also be low (averaging 22 ug/L). Ammonia at the bottom, however, exhibits elevated concentrations (averaging 85 ug/L), indicative of decay of organic matter and ammonia release from sediments. Ammonia concentrations at the bottom can show multiple peaks in the summer, reflecting periodic mixing or partial vertical mixing.

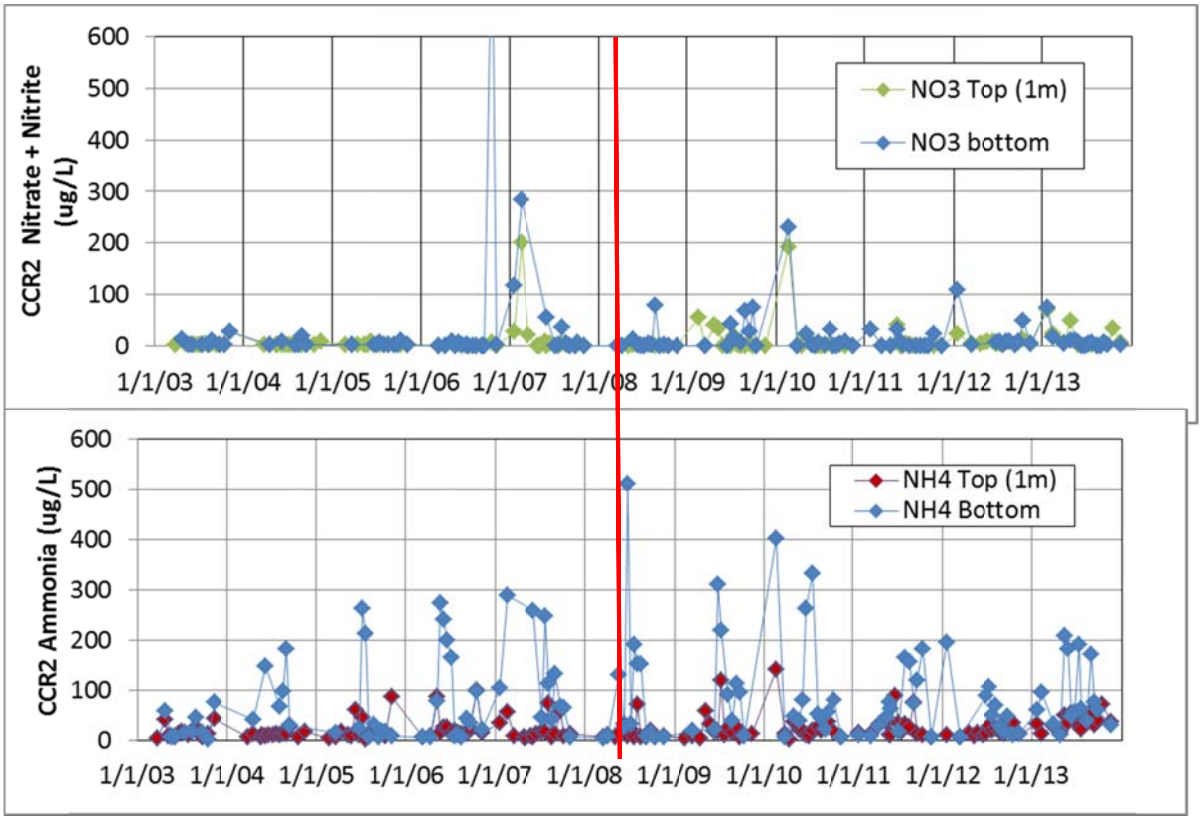

Figure 25. Nitrate and Ammonia Observations from the Top and Bottom of the Reservoir, 2003-2013; Red Line Indicates Start of Aeration System

Observed concentrations of SRP at the top and the bottom of the reservoir indicate a seasonal pattern of low winter concentrations and summertime peaks (Figure 26). There is a concentration gradient from the bottom to the top, indicative of nutrient release from sediments at the bottom and uptake from algae closer to the surface. Surface concentrations average 22 ug/L and bottom concentrations average 50 ug/L. These in-reservoir concentrations are low relative to the average inflow volume-weighted average concentration of 120 ug/L. Unlike, nitrate and ammonia however, concentrations near the surface of the reservoir exhibit a clear pattern of peak values in the summer. This is directly indicative of two things. First, there is some mixing from the bottom to the top of the reservoir, providing nutrients from sediment release to the surface throughout the summer months. Second there is excess phosphate at the surface relative to inorganic nitrogen, indicating nitrogen limitation for algal growth.

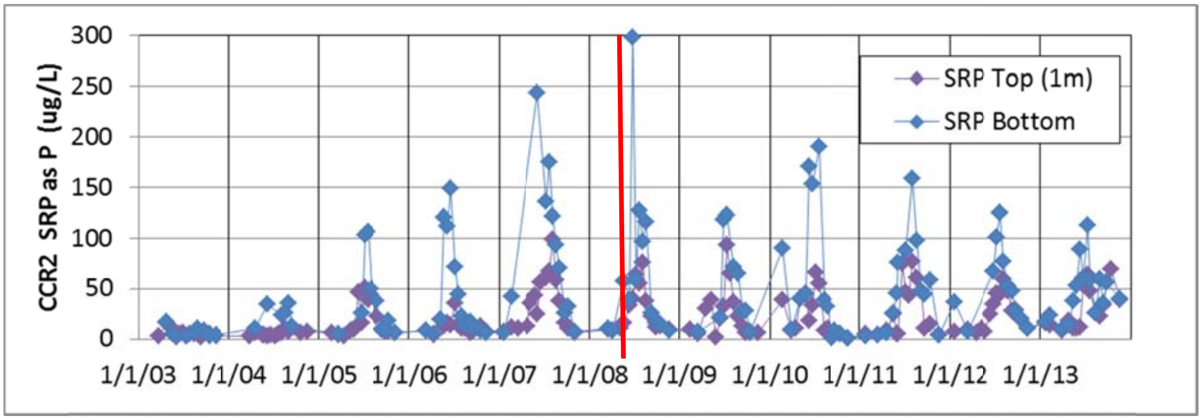

Figure 26. SRP Observations from the Top and Bottom of the Reservoir, 2003-2013; Red Line Indicates Start of Aeration System

The algal response in the reservoir, as indicated by chlorophyll *a*, is shown in samples from the photic zone at CCR2 in Figure 27. Chlorophyll *a* concentrations range from 1 ug/L to 57 ug/L, with an average for the 2003-2013 period of 20 ug/L. The summer chlorophyll *a* concentrations show a period from 2004 through 2009 of relatively lower values. Interestingly, this corresponds to the period of time with increased inflow ratios of TIN: SRP (see Figure 18 and associated discussion). It is possible that higher inflow TIN:SRP ratios are beneficial, reducing the chlorophyll *a* response in the reservoir. Further, a change in algal assemblage to more edible algal types in response to higher TIN:SRP ratios (as observed by Barica et al., 1980, Smith, 1983, Schindler et al., 2008) could result in a change in zooplankton species and concentrations, which could have a net effect of reducing algal concentrations. Unfortunately, there are no zooplankton data available from 2002-2010. Algal species data do not show clear changes in species dominance through this period; however, due to the lack of biovolume data prior to 2009 and the lab change for algal assessment in 2009 (discussed in greater detail Boyer et al., 2014a), the hypothesis is difficult to definitively verify or refute with the data.

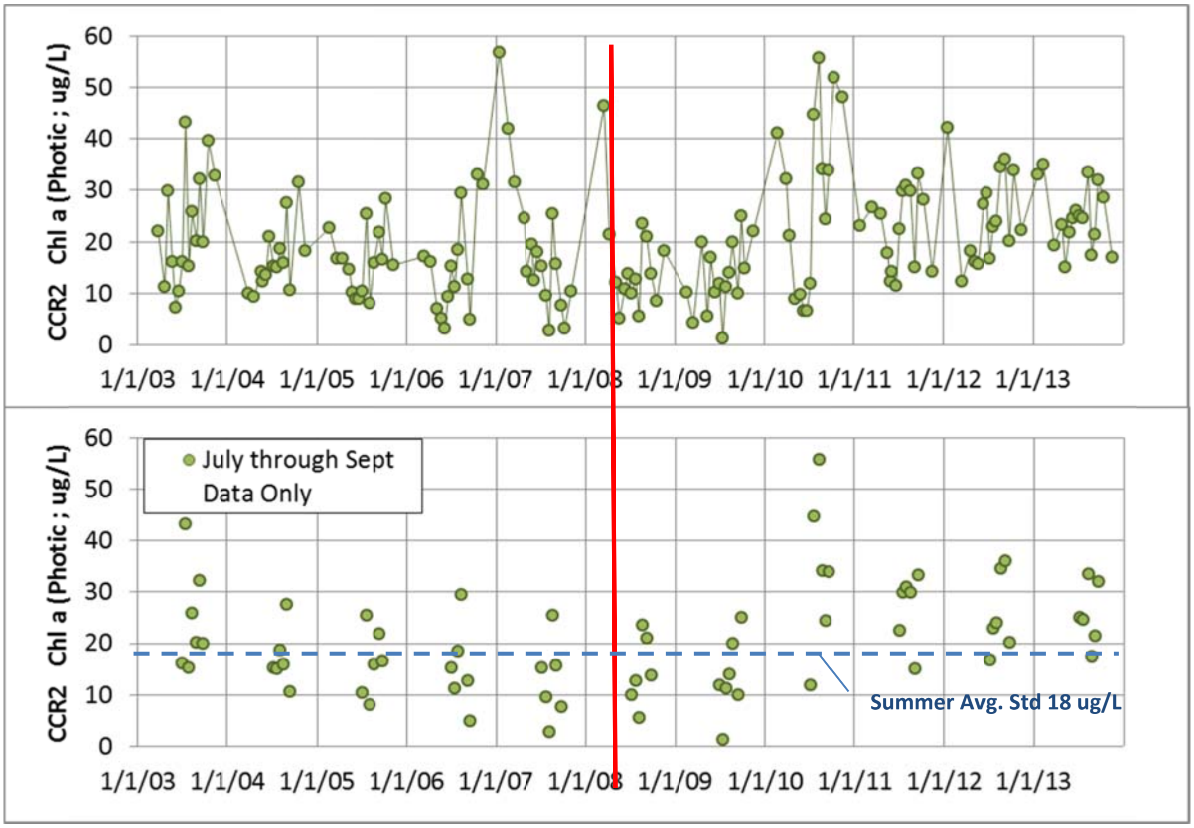

Figure 27. Observed Chlorophyll a Concentrations from CCR2 (2003-2013) and Comparison to Standard; Red Line Indicates Start of Aeration System

Due to the high algal concentrations, as well as high TSS (averaging 15 mg/L in the photic zone, per data available from 2011-2013), the reservoir exhibits low clarity (Figure 28). This can affect algal growth patterns, indicating low light transmission to deeper parts of the reservoir. The average clarity is 0.9 m. Clarity ranged from 0.4 m to 2.9 m; both measured in 2009.

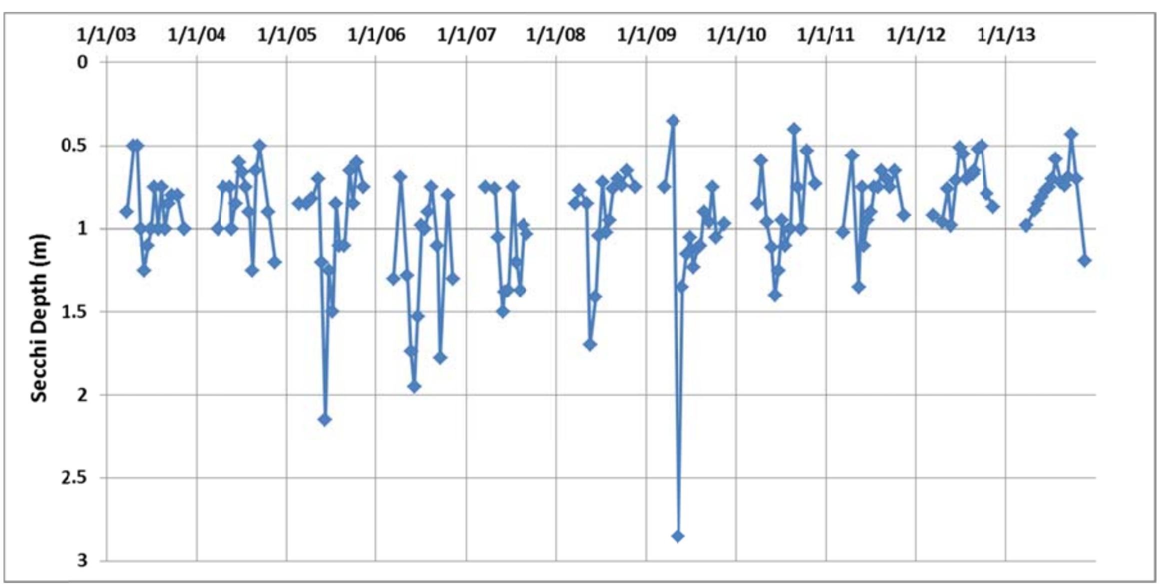

Figure 28. Secchi Depth at CCR2

Lastly, in 2012, there was a fish kill in the reservoir between August 14 and August 23. The cause of the fish kill is expected to be a combination of high water temperatures and low dissolved oxygen creating poor conditions, also known as a “temperature-oxygen squeeze”. Of the 548 fish collected by Colorado Parks and Wildlife, 97% were gizzard shad and 3% were walleye. There were no other such fish kills observed in the reservoir in there period between 2003 and 2013 (Wolf, personal communication, 2015).

4 Model Development

For reasons detailed in the Task 2 Technical Memo (Boyer et al., 2014b), the modeling software CE-QUAL-W2 v3.71 (Cole and Wells, 2011) was selected for development of the hydrodynamic and water-quality model of the reservoir. CE-QUAL-W2 (W2) is a two-dimensional hydrodynamic and water-quality model. The model assumes lateral homogeneity, but simulates variation longitudinally and vertically to the resolution specified. The U.S. Army Corps of Engineers originally developed the model, and it is currently supported and updated by Portland State University. The model is coded in FORTRAN, available in the public domain, and can be operated from a Windows environment. Detailed documentation of the software and associated technical manuals are available at http://www.ce.pdx.edu/w2/.

4.1 Bathymetry

Developing the bathymetric representation of the reservoir in W2 requires designation of segments and layers. Segments were oriented perpendicular to the direction of flow of the original thalweg of Cherry Creek. This resulted in segments laid out parallel to the dam. Spacing of segments considered capturing the variations in the bathymetric structure and the aeration system, as well as the location of in-reservoir sampling data. Ultimately, an even segment spacing of 200 m was set, for a total of 12 active segments, as shown in Figure 27.

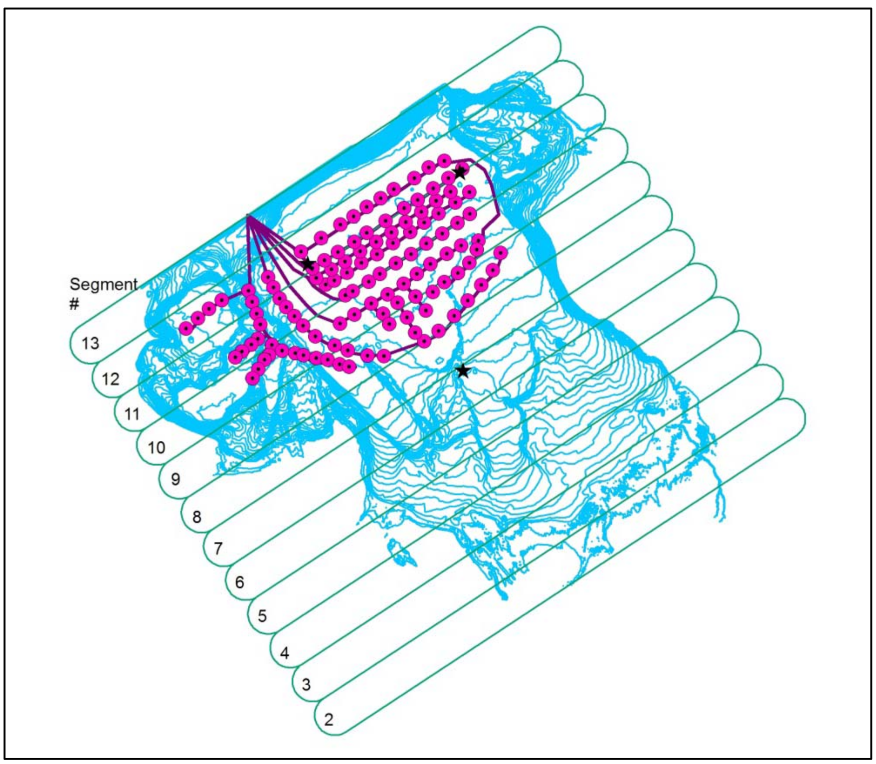

Figure 29. Model Segment Layout Shown with Bathymetric Contours, Aeration System (Pink Circles), and Sampling Locations (Three Black Stars)

Model layer spacing was defined considering observed vertical variability in water-quality response and the need to simulate aerator placement. Based on profile data such as that shown in Figure 30, a vertical discretization of 0.5 m was selected. This resulted in 19 active layers in the model.

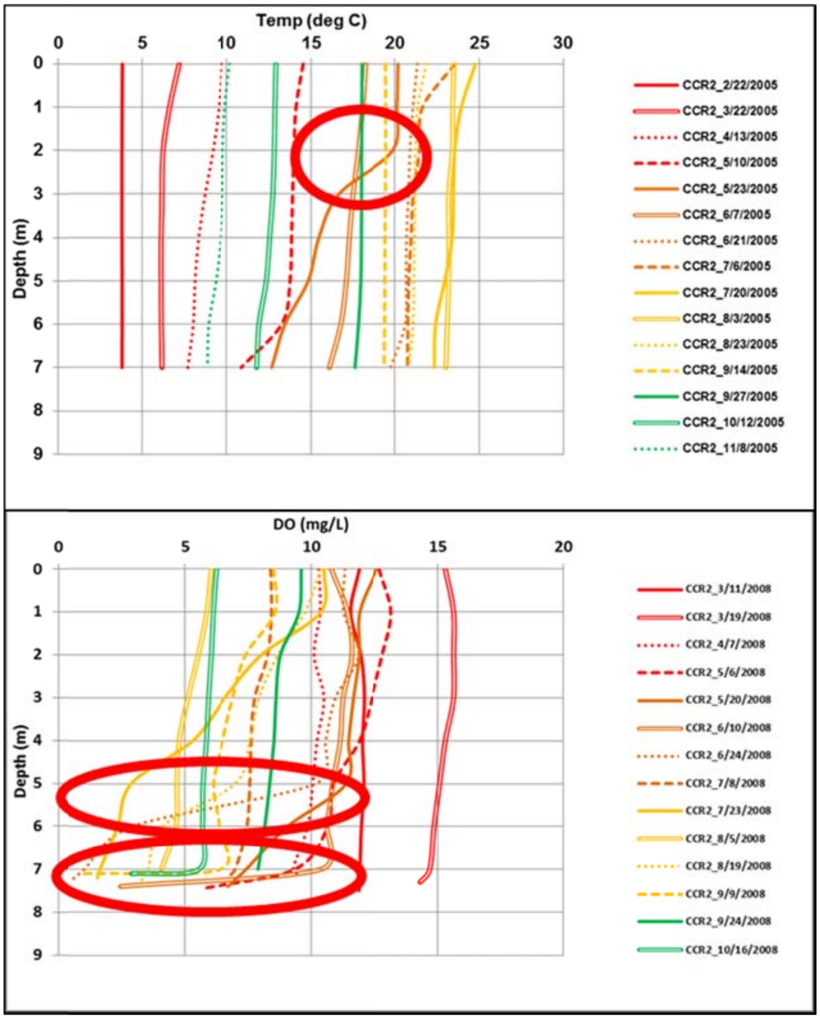

Figure 30. Example of Profile Data Considerations in Selecting Vertical Discretization of Model Layers

A view of the model profile without vertical exaggeration is presented in

Figure 32 to support visualization of the reservoir, and a plan view of the model is shown in Figure 33. Cross-sections developed from the 1-ft bathymetric contours were used to develop the grid and widths for each cell[4]. An example section-view of a segment (segment 12) is shown in Figure 33.

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[4] Model bathymetry was calculated from these contours by measuring widths perpendicular to the principal direction of flow at each contour depth along 24 transects equally-distributed at 100 meter intervals over the reservoir. The average width at each depth in each of the 12 active model segments was calculated, and the vertical discretization of the model bathymetry was set at 0.5 meter resolution

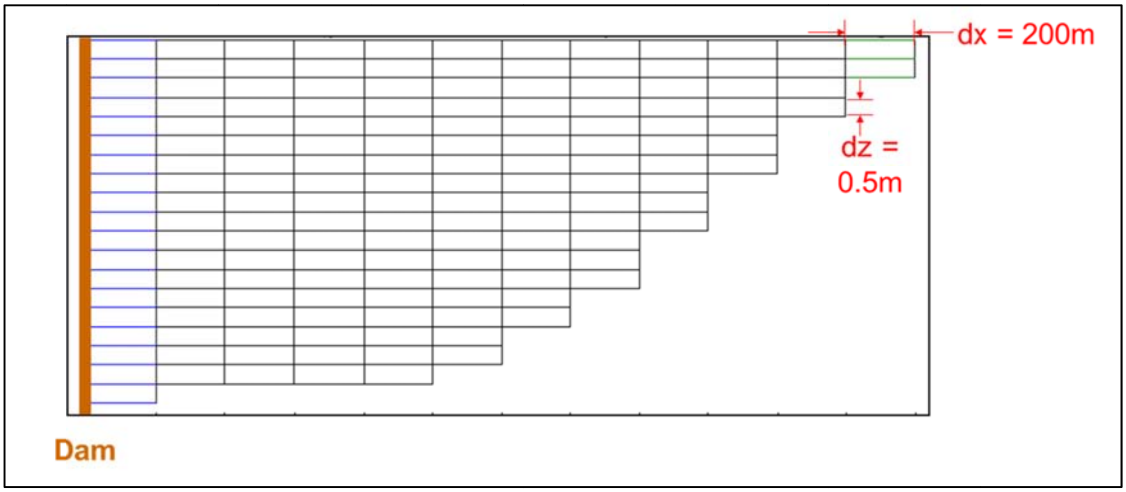

Figure 31. Profile View of Model Grid

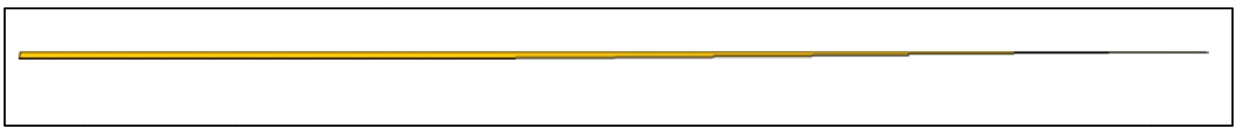

Figure 32. Model Profile without Exaggerated Vertical Scale

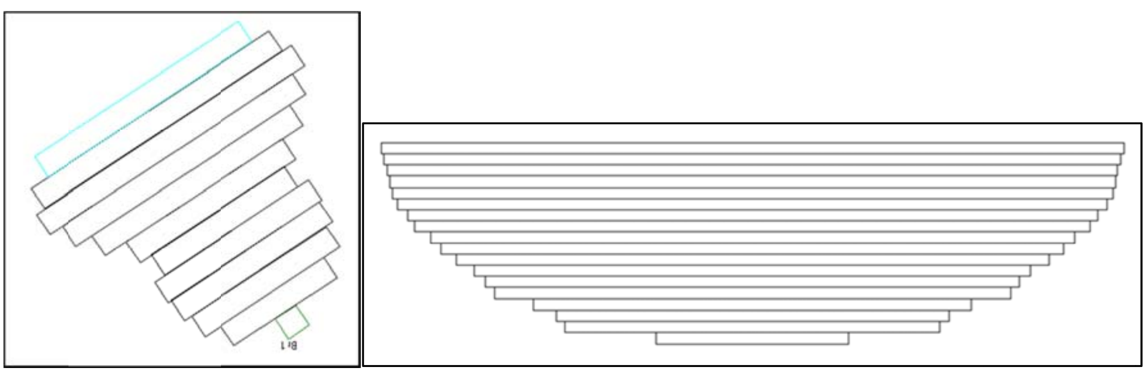

Figure 33. Plan View of Model Grid (Left) and Section View of Segment 12 (Right)

The resulting elevation-to-volume and are elevation-to-surface area relationships were compared to those based on the 1-ft contour data. The bathymetry in the model matches the observed bathymetry well. A comparison of model and contour-based elevation-to-volume

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using linear interpolation between measured contours to calculate center-line widths. Where necessary, adjustments were made to ensure bottom widths were greater than or equal to 10 meters, and so that the width of each cell was at least 10% of the width of the cell directly above it, as recommended for optimal model performance.

relationship is presented in Figure 34, with an indication of the operating range of the 2003-2013 period. The absolute mean error (AME) is 0.28% (49 AF; 60,400 m^3), and the root mean squared error (RMSE) is 0.37% (66 AF; 81,400 m^3).

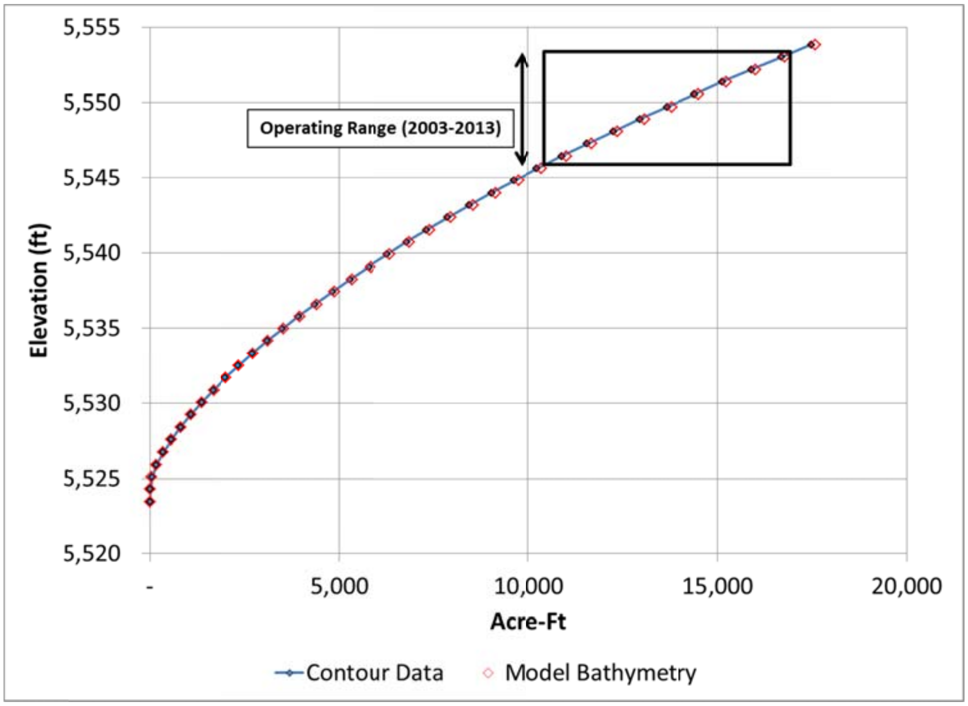

Figure 34. Comparison of Model Bathymetry and Contour Data Elevation - Volume Relationship

A comparison of model and contour-based surface area-to-volume relationship is presented in Figure 35, with an indication of the operating range of the 2003-2013 period. The AME is 0.49% of the total area (4.8 acres; 19,400 m^2), and the RMSE is 0.60% of the total area (6.0 acres; 24,300 m^2).

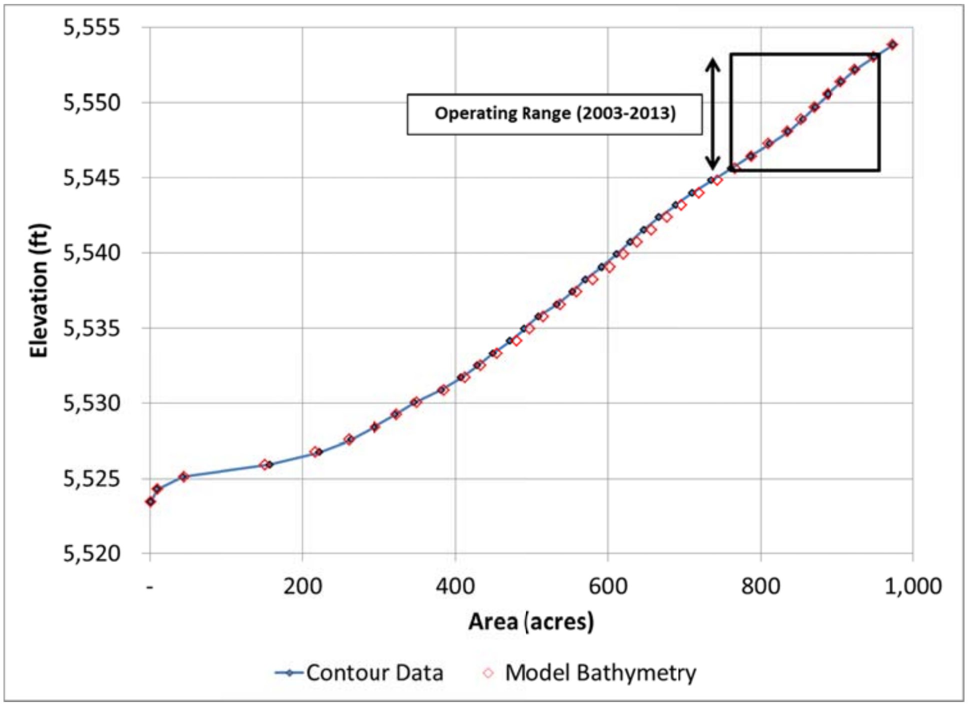

Figure 35. Comparison of Model Bathymetry and Contour Data Elevation - Volume Relationship

4.2 Application of the Water Balance

Daily flow rates developed for the water balance were applied in the model to approximate actual flow patterns as described below and as shown on Figure 36:

Inflows:

- **Cherry Creek** inflows were assigned as the branch inflow, entering the model at the first active segment. Cherry Creek inflows are set to flow in according to relative thermal density.
- **Cottonwood Creek** inflows were assigned as a tributary inflow, entering the model at the second active segment and set to flow in according to relative thermal density.
- **Groundwater** inflows were set to enter the model in the second segment also. This is based on the location a greatest seepage identified by Lewis et al. (2005) through seepage meter measurements and sediment conductivity and water content observations. The groundwater inflows are set to flow in according to relative density.
- **Ungaged Surface Water** inflows were assigned as tributary inflows into the eighth active segment, recognizing that some of this inflow enters the reservoir via ungaged channelized flows adjacent to this segment. These flows are also set to flow in according to relative density.
- **Precipitation** enters the model distributed across the reservoir surface in proportion to the surface area of the segments.

Outflows:

- **Dam Releases** are simulated as releases from a point-type structure in the last active segment, defined in the model by elevation of the outlet within the reservoir. The outlet is set up to use the selective withdrawal algorithm in the W2 code. This algorithm calculates a time-varying withdrawal zone for each outlet based on outflow, outlet geometry, and upstream density gradients**.**
- **Evaporation** is set to be removed from the top layer of the reservoir, distributed in proportion to the surface area of the segments.
- **Outflow Seepage** is designated in the model as an outflow from the final active segment and seepage is specified to come from the bottom layer of that segment.

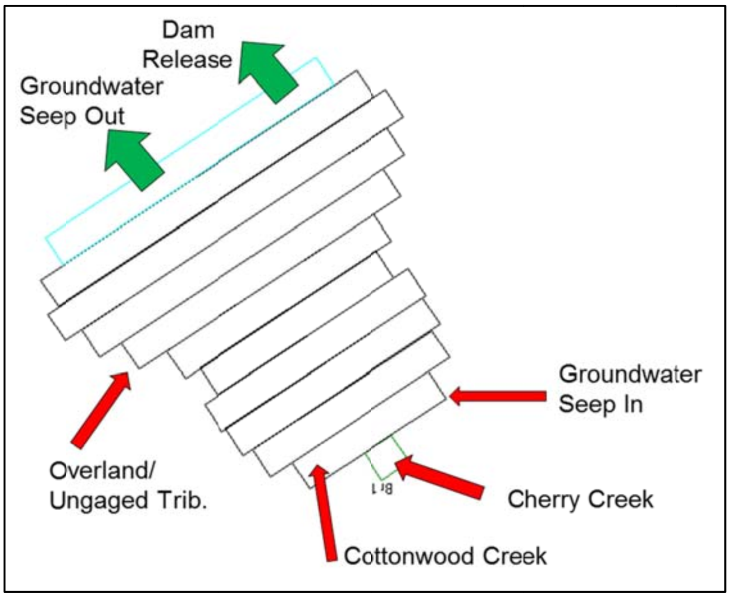

Figure 36. Location of Inflows and Outflows Applied in Model

4.3 Algae and Zooplankton

Algal species biovolume and density data were reviewed for seasonal progression patterns to develop inputs for algal groups. There are some patterns in the data, but there is a lot of variability from year to year. From the data, four algal groups were assigned in the model:

1. **Group 1: Spring and Fall.** The data show spring and fall dominance of cryptophytes, green algae, diatoms, euglenoids, chrysophytes, and dinoflagellates in some years.
2. **Group 2: Winter.** Typically, winter algae are dominated by cryptomonads, green algae, and diatoms. In some years, there are also significant percentages of golden algae, euglenoids, chrysophytes, and cryptophytes through this time.
3. **Group 3: Summer, Nitrogen Fixing Cyanobacteria.** This algal group was added to represent nitrogen-fixing cyanobacteria behavior in summer months. This algal group was also assigned zero settling rates to reflect buoyancy characteristics.

4. **Group 4: Summer.** The warmest months exhibit a wide range of algal types comprising significant proportions of the algal density and biomass over the years. These include chlorophytes, diatoms, cyanobacteria, green algae, and dinoflagellates.

Maximum growth rates specified in the model for these four algal groups are presented as a function of water temperature in Figure 37. This figure also presents the maximum settings for zooplankton growth, which will be discussed in more detail in Section 5. The maximum growth rates of nitrogen-fixing cyanobacteria are lower than other algal groups and largely limited to warmer water conditions, reflecting the energetically taxing nature of fixing atmospheric nitrogen.

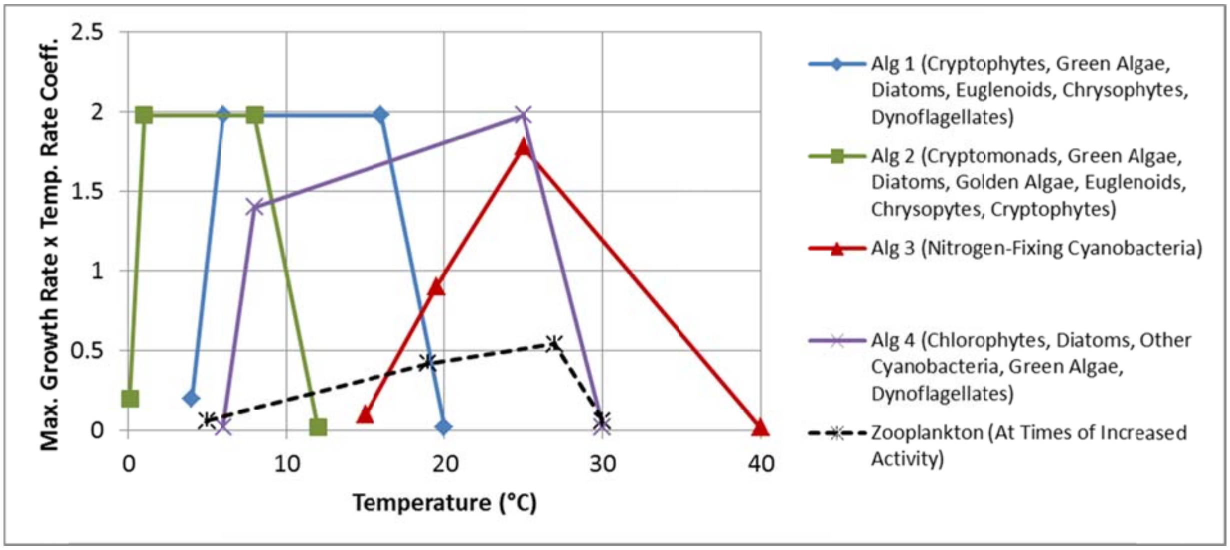

Figure 37. Maximum Growth Rate and Water Temperature for the Four Algal Groups and Zooplankton in the Model

4.4 Sediment

Sediment oxygen demand and internal loading are simulated applying both the zero-order and first-order sediment compartments in W2. The zero-order sediment compartment simulates anaerobic oxygen demand and internal loading. The first-order sediment compartment simulates aerobic decay of detritus at the sediment-water interface, consuming oxygen and providing nutrients to the water column. These are explained in detail in the W2 documentation (Cole and Wells, 2011). It was decided to apply both sediment compartments, recognizing the highly productive nature of the reservoir and the variable dissolved oxygen concentrations at the bottom, even during warm summer months.

4.5 Destratification System

The destratification system is represented in the model using the W2 Aeratec controls. This module allows the user to apply a multiplier to the vertical mixing coefficient on a segment-by-segment basis, over the layers of interest (Cole and Wells, 2011). CE-QUAL-W2 also allows for the direct addition of oxygen to the water. In the model, aerators were designated in the second active layer above the bottom (corresponding to 0.5 to 1 m above the bottom, reflecting the ~0.6 to 0.75 m height of the aerator heads). Mixing was designated from the aerator heads to the surface of the reservoir, recognizing that bubbles from this system are routinely observed at the surface. The aerators were designated in four segments, patterning their actual location in the reservoir (Figure 38).

Figure 38. Actual Location of Aerators on Reservoir Footprint (Left) and Model Segments Designated to Contain Aerator Heads (Right)

Because the aerator bubbles have a relatively short contact time with the water before reaching the surface (due to the relatively shallow nature of the reservoir), and because the aerators supply air and not oxygen, limited direct oxygen loading is expected. Addition of oxygen by aerator bubbles was assigned by first estimating the maximum possible oxygen load from the aerators. The aerators provide 200 to 250 cubic feet of air per minute (or 8,155 cubic meters per day). Assuming a density of air of 0.93 kg/m^3 at an altitude of 5,500 ft at 90°C, this corresponds to 7,584 kg of air per day. Assuming 23% oxygen, this equates to 1,744 kg oxygen per day. Since the aerators are distributed over four reservoir segments, this is the equivalent of a maximum potential of 436 kg of oxygen per day per segment, assuming an even distribution. A total of 200 kg of oxygen per day per segment was used in the modeling. This is a little less than half the maximum potential. The effect of this directly-added oxygen is minimal in the modeled response, as was simulation of the maximum potential load in a test run.

4.6 Model Code Adjustments

To better simulate the observed algal response, two adjustments were made to the W2 code to allow for simple time-variation of select parameters:

1. **Fish Kill Effects**

   The fish kill in August of 2012 would be expected to have resulted in an increased organic matter load to the bottom of the reservoir (in the form of dead fish). This organic matter would also be expected to not be buried as quickly as algal detritus.

Based on this, the model was coded to allow for an increase in organic matter concentration in the 1st-order sediment compartment following the fish kill. Further, sediment burial rates were also decreased somewhat, starting after the fish kill in 2012.

To estimate the increase in organic matter delivered to the bottom the reservoir, it was assumed that 90% of the fish settled to the bottom of the reservoir (Johnson, personal communication, 2015). Based on the number of fish collected at the surface, this was assumed to be roughly 5,000 fish. It was assumed that the average weight of each fish was 0.2 kg/fish, resulting in ~1,000 kg of settled dead fish. Assuming 850 acres of bottom surface (3.4E6 m^2), this corresponds to 0.29 g of organic matter per square meter. Assuming that half of this organic matter was carbon and applying the Redfield ratio (C:N:P = 106:16:1), the following increases in sediment compartment concentrations were applied: Carbon +0.14 g/m^2, Nitrogen +0.02 g/m^2, Phosphorus +0.001 g/m^2.

2. Zooplankton Effects

In general, the role of zooplankton on algal growth is expected to be minimal in Cherry Creek Reservoir due to predation by gizzard shad and young walleye (Boyer et al, 2014a and Lewis et al., 2004). Based on this the small amount of pressure on the phytoplankton was simulated with slightly increased mortality rates in lieu of simulating the highly suppressed zooplankton. Note: W2 does not simulate higher trophic response of fish in the reservoir.

The observed data indicated a change in algal response in the reservoir, particularly between 2004 and 2009 (Figure 27). This pattern does not correspond to a time of observed reductions in in-reservoir nutrient concentrations, differences in DO at the bottom, or different water temperatures at the surface. This period of lower summertime algal concentrations does, however, correspond to the period of increased TIN:SRP ratios on the inflow (Figure 18). As discussed in Section 3.5, it was hypothesized that this change in inflow nutrient ratios could affect algal species response and, in turn zooplankton response. Unfortunately, there are no zooplankton data from 2003 through 2010 to verify speculation about an increased influence of zooplankton on phytoplankton over part of the record. There may also be a pattern of increased numbers of smaller gizzard shad outside of this period, as reflected in the fish sampling data (**Error! Reference source not found.**). However, CPW sampling protocols did not always include collection of smaller gizzard shad, so it is difficult to draw clear conclusions from these data (Johnson, personal communication, 2015).

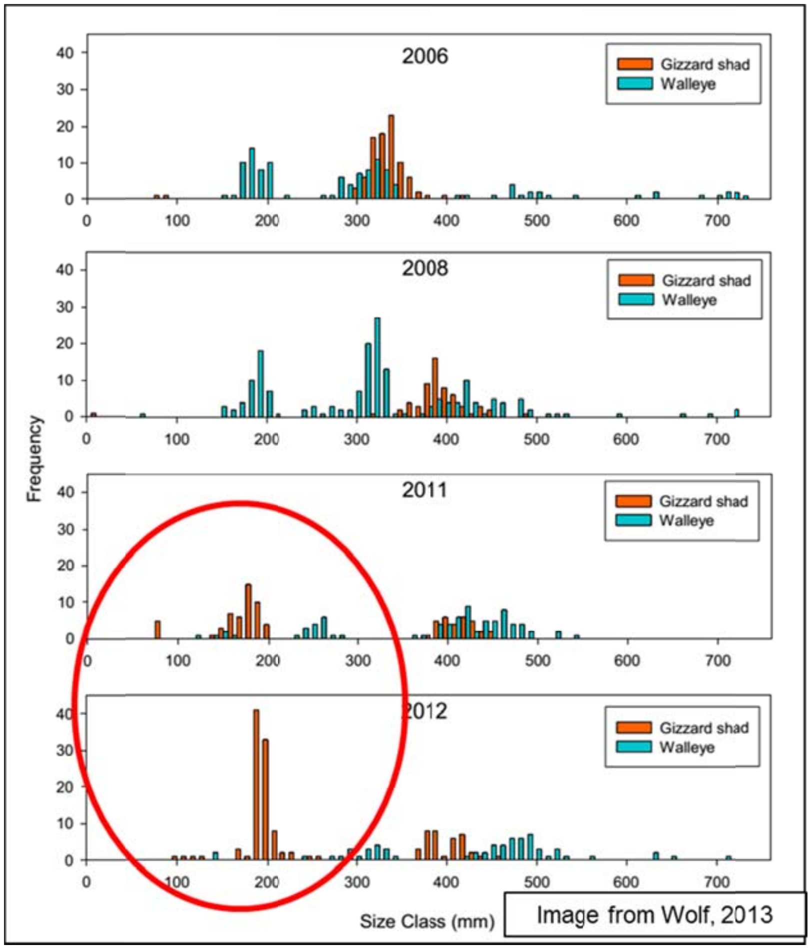

Figure 39. CPW Fish Sampling Data Showing Possible Period of Increased Smaller Gizzard Shad

In short, changes in zooplankton pressure on algae (possibly in response to changing algal and zooplankton species patterns, ultimately due to increased TIN:PO4 ratios) represents the best available hypothesis at this time for this observed decrease in algal concentrations from 2005 through 2009. Therefore, the model code was modified to simulate higher zooplankton growth rates between 2005 and 2009. The resulting zooplankton growth rate was still relatively low with high mortality and predation[5]. The adjustment produced good results in the simulation.

[5] Resulting simulated zooplankton (2005-2009) were compared to observed zooplankton concentrations (2011-2013). The average simulated zooplankton concentration for 2005-2009 was ~1.1 mg/L. Average observed zooplankton concentrations for 2011-2013 were 0.7 mg/L. The comparison, however, is considered imperfect, since Lewis et al. (2004) indicated that the dominant zooplankton for their study in Cherry Creek was a cyclopoid, *Diacyclops thomasi,* which is more likely to feed on protozoan than phytoplankton, citing Dobberfuhl et al. (1997). Therefore, even fewer than 0.7 mg/L of zooplankton that fed on phytoplankton were likely present from 2011-2013.

Moving forward, this is recognized as an uncertainty in the model. Zooplankton data, including species information, are currently being collected. Continued collection of these data and review relative to the TIN:SRP ratios of inflows and the in-reservoir algal response are recommended. Review of this information over time should help evaluate the hypothesis presented here. Additionally, as a possible future refinement to the model, zooplankton could be simulated for the entire period with a greater number of algal groups in the summer months to better reflect the growth of more and less-edible species. Such a refinement would require some recalibration and recoding for the obligate nitrogen-fixation effect (discussed in Section 5.3).

5 Model Calibration

The calibration of the Cherry Creek Reservoir Model is described in this section. Calibration was performed by adjusting conceptually-relevant coefficients within reasonable ranges. Reasonable ranges of coefficient settings were defined by model guidance documents and literature. The process was iterative and not prescriptive. More than 1,000 model runs were conducted in the calibration process for this model. The calibration focused on the observed record from January 2003 through December 2013.

The ultimate calibration target for this model, based on the site-specific standard driver for this work, was the summertime chlorophyll *a* concentrations. Simulating chlorophyll *a,* however, required reasonable simulation of water temperature, nutrient concentrations, sediment dynamics and oxygen. Therefore, these are also considered calibration targets for this effort. The focus in review of simulation results through calibrations was on simulating concentration ranges, seasonal patterns, spatial patterns, and year-to-year differences by visual review of graphical comparisons of observed and simulated results. The goal of this approach was to reflect water-quality mechanisms at work in the reservoir instead of simply focusing on minimizing residual error statistics. In addition to visual review of patterns, calibration summary statistics were calculated and are presented.

The following subsections present the calibration assumptions and results for water levels, temperature, nutrients, DO, and algae / chlorophyll a. This section is supported by Attachments A, B, and C, which include responses to peer reviewer comments on the draft calibration, simulated and observed temperature results (profile and thermistor), and simulated and observed dissolved oxygen results (profile).

5.1 Surface Water Elevation

Applying the water balance described in Section 3.2, simulated surface water elevations were compared to the observed record. The model matched observations well, with and AME of 0.1 ft (0.04 m), an RMSE of 0.1 ft (0.04 m), and an average error of -0.1 ft (0.04 m). Simulated and observed daily water levels are shown in Figure 40.

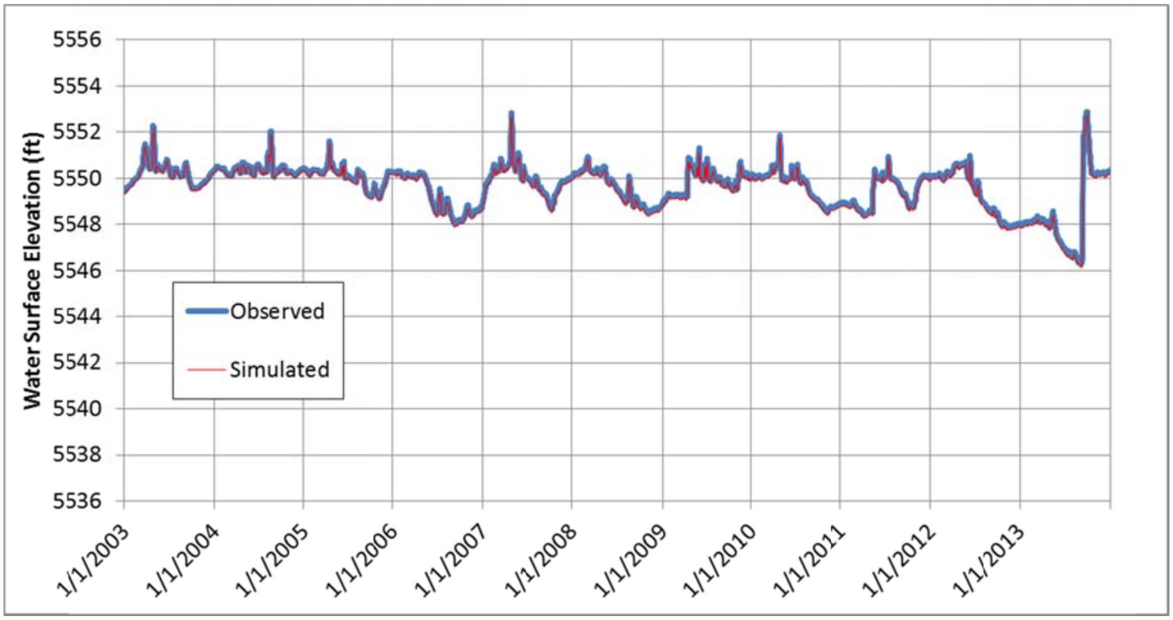

Figure 40. Comparison of Simulated and Observed Daily Water Levels, 2003-2013

The model outputs "water age", indicating how long water in a given cell has been in the reservoir. This output is generally comparable to residence time. Simulated water age in Segment 12 (the segment corresponding to CCR1 and CCR2) at 1 m and at the bottom are shown in Figure 41. Note that water age starts at zero across the reservoir at the start of each simulation, though a steady pattern is reached in the second year. Water age at the top of the reservoir is not very different from that at the bottom, indicating that short-circuiting underflows or overflows are very limited in this part of the reservoir. The exception is some simulated overflow in some winter months and some brief underflow during non-winter storm events. The average water age simulated by the model at CCR2 from 2004-2013 is 247 days.

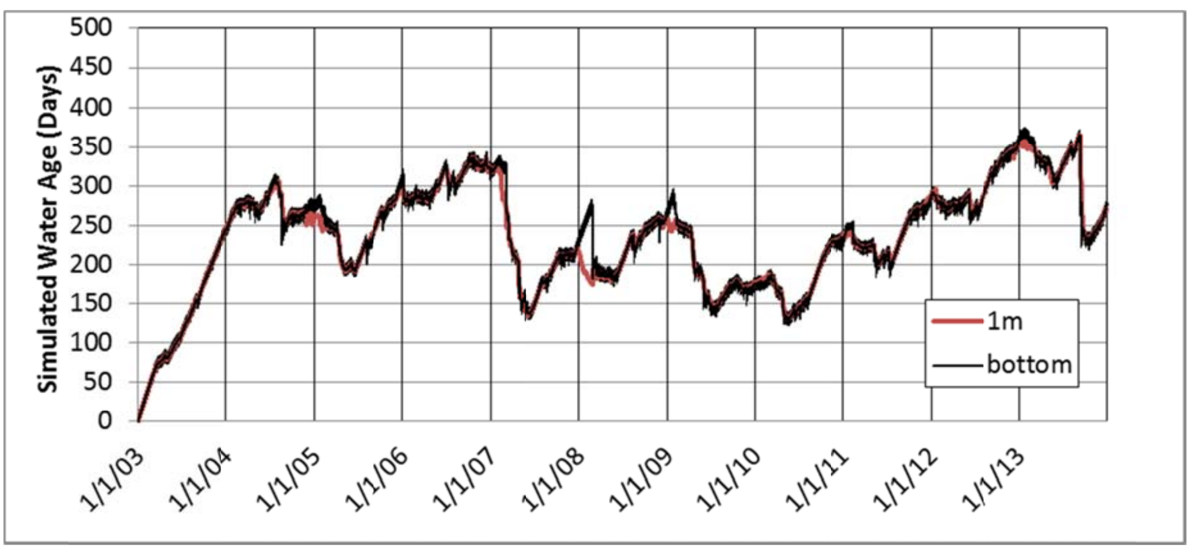

Figure 41. Simulated Water Age at 1 m and at the Bottom at CCR2, 2003-2013

5.2 Temperature

Accurate simulation of water temperature in the reservoir is critical to water-quality modeling. Temperature is one indication of the simulation of mixing within the reservoir. Temperature is also an integral part of simulating various reaction, growth, and decay rates within the model. Calibration of water temperature is also dependent on the water-quality simulation, particularly in a eutrophic system like Cherry Creek Reservoir. The water-quality simulation, which includes simulation of total suspended solids, affects the transmission of light through the reservoir, which in turn, affects the temperature response with depth. Therefore, the temperature calibration was completed iteratively with the water-quality simulation.

In this relatively shallow system with residence time on the order of 250 days, accurate simulation of water temperature required good inputs for meteorological drivers. Air temperature, wind, and solar radiation were the most critical meteorological inputs. Data from the local meteorological stations (Figure 20) were tested in the model, and the compilation described in Section 3.4 was identified as the best combination of information. Air temperature and solar radiation data from the CPW Met station were particularly valuable in improving calibration response. Based on this, it was recommended during a check-in meeting with the Authority Reservoir Modeling Committee that the Authority coordinate with CPW to check and improve data collection and data management protocols for the CPW meteorological station. Alternatively, a meteorological station could be installed at the dam, monitoring, at a minimum, air temperature, solar radiation, wind speed, and wind direction (Hydros presentation on the March 23, 2015 conference call). These data would be valuable for future modeling and model refinements.

The thermal simulation was less sensitive to inflow water temperatures from the tributaries, as compared meteorological inputs, though not completely insensitive, particularly during larger storm events. The 2003-2013 dataset does not have continuous temperature data on the inflows, so inputs were developed by applying seasonal multiple regressions of air temperature, solar radiation, and flow rate to the available monthly temperature observations. Recently-installed thermistors in the inflow tributaries should help refine thermal inputs for future modeling, particularly for storm events.

The 328 observed reservoir temperature profiles from CCR2 and CCR3, collected between 2003 and 2013, were reviewed relative to simulation results to guide the calibration. Additionally, the thermistor data from CCR2 were compared to simulation results. The thermistors have collected continuous temperature data at 1 m intervals since 2007. Thermistors are deployed in March, April, or May of each year and retrieved in September, October, or early November. The thermistor dataset was particularly useful for refining calibration in this very dynamic system. Comparison of the simulated and observed temperature difference from the top to bottom, as measured at CCR2 by the 1 m and 7 m thermistors, was also a useful review for calibration.

The in-reservoir thermistor data were also helpful in calibration of increased vertical mixing due to the destratification system. Following calibration focusing on the pre-destratification system period (2003-2007), simulated aerator mixing was increased progressively from zero until it no longer improved (and started to deteriorate) the calibration of thermistor response at CCR2 for 2008-2013. Specifically, this optimization of calibration focused on matching the thermistor-measured temperature difference between 1 m and 7 m at CCR2. Based on this approach, a 60% increase in vertical mixing was set for destratification system when operating.

Overall, the thermal calibration is very good. The model successfully simulates the overall seasonal patterns and magnitudes of observed water temperatures. This includes simulation of periods of brief stratification and isothermal conditions. Three example CCR2 calibration profiles over a range of temperatures are presented in Figure 42. The full set of calibrated and observed profiles from 2003-2013 for CCR2 and CCR3 are presented in Attachment B.

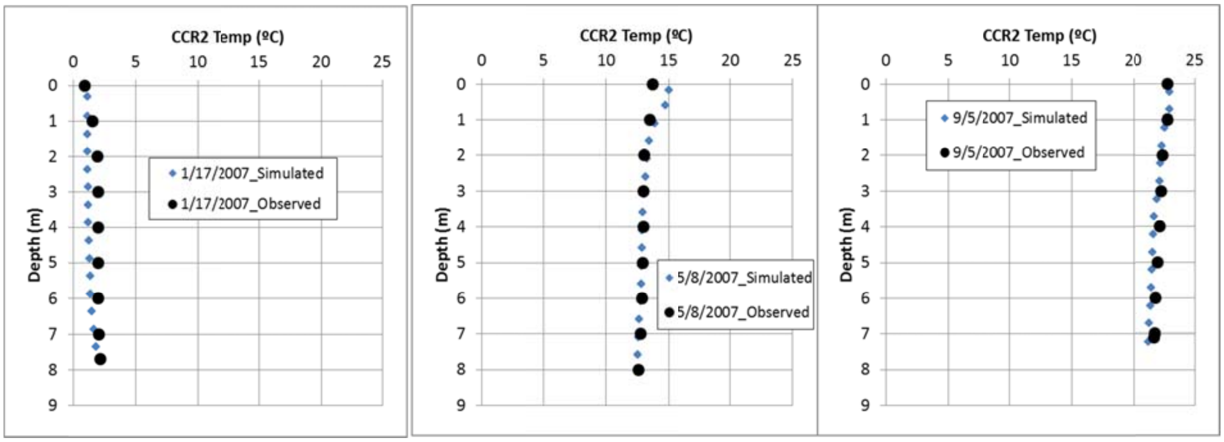

Figure 42. Example Calibration Profiles at CCR2 from 2007

Residuals for each profile observation were calculated and compiled to evaluate simulation error. In general, residuals tended to be slightly greater in May or June, when water temperature were increasing most rapidly. The AME, RMSE, and average error for the temperature profiles are presented in Table 2. All of these metrics were less than 1 °C. Results indicate similar statistics at CCR2 and CCR3. Further, results were the same (to the tenth of a degree) for the period before and after the start of the destratification system (2008).

Table 2. Summary Calibration Statistics for Temperature Profiles, 2003-2013

Metric	CCR2	CCR3
Profiles - Average AME	0.6 °C	0.6 °C
Profiles - Average RMSE	0.7 °C	0.7 °C
Profiles - Average Error	-0.3 °C	-0.1 °C

Observed thermistor data plotted against simulated temperatures also shows good match of seasonal patterns, magnitude, and diurnal range from the top to the bottom in all years. As an example, the hourly simulated and observed temperatures at 1 m and 7m at CCR2 in 2008 are shown in Figure 43 and Figure 44, respectively.

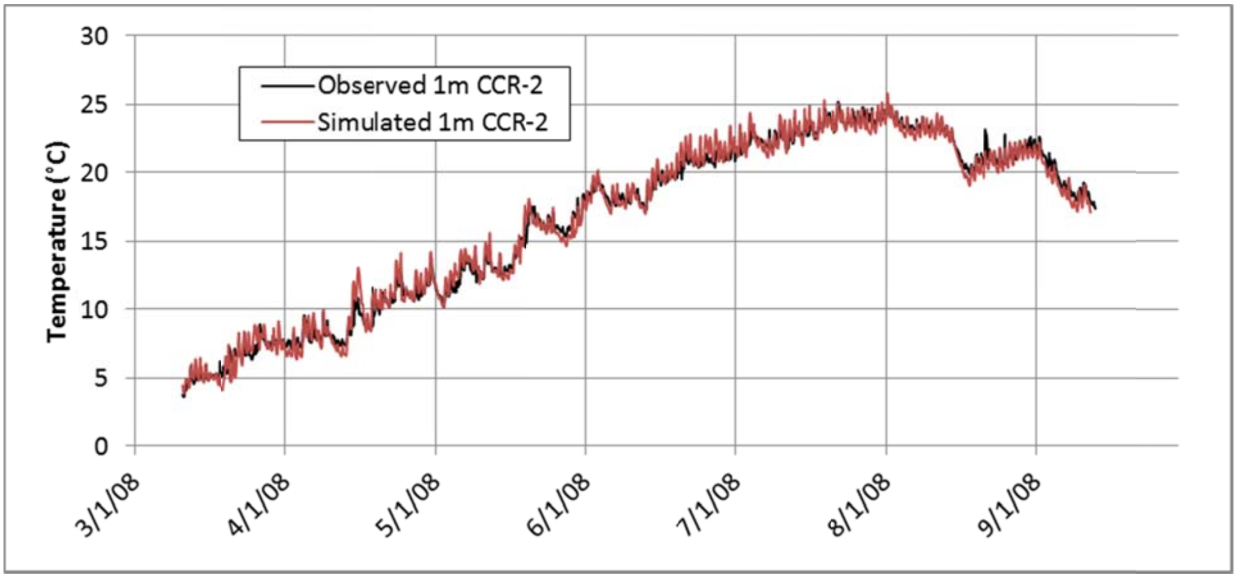

Figure 43. Observed and Simulated Hourly Temperatures at 1 m at CCR2, 2008

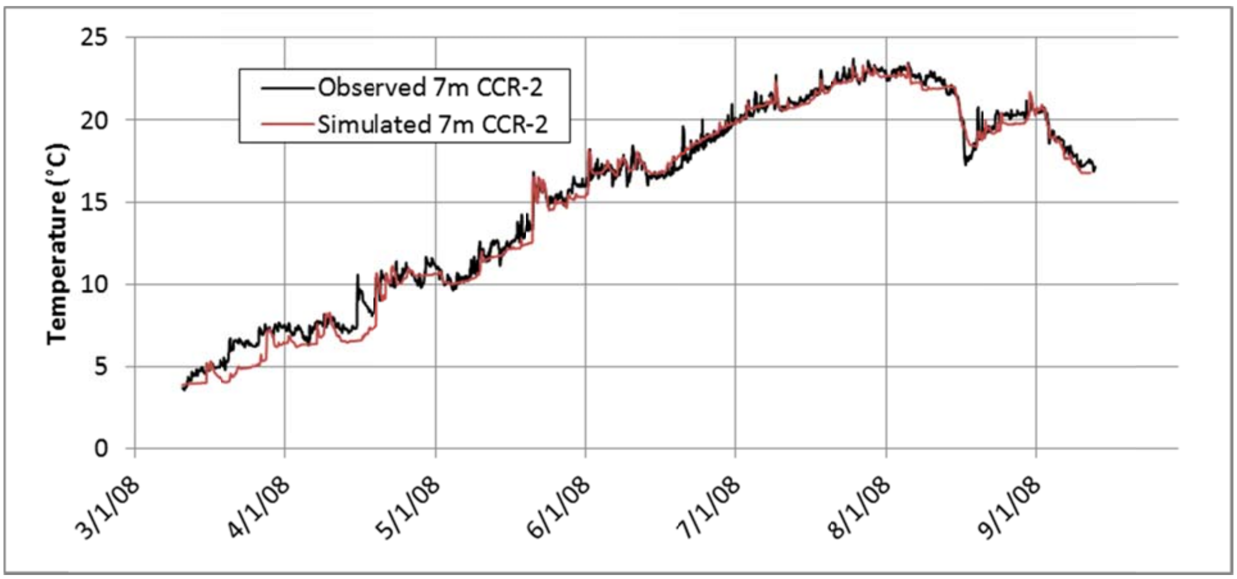

Figure 44. Observed and Simulated Hourly Temperatures at 7 m at CCR2, 2008

Top-to-bottom temperature differences are also well simulated by the model. An example of the observed and simulated 1m to 7m temperature difference is provided in Figure 45. The complete set of observed and simulated data for the CCR2 thermistors is provided in Attachment B.

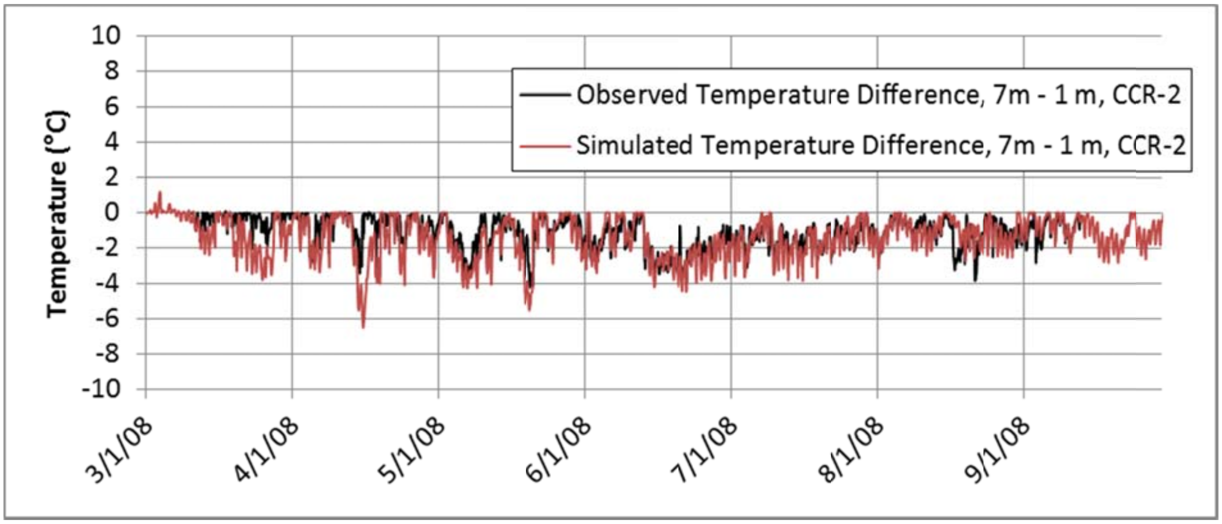

Figure 45. Observed and Simulated Hourly Temperature Differences between 7 m and 1 m at CCR2, 2008

Calibration summary statistics for continuous thermistor data are summarized in Table 3. All of these measures of residuals are less than 1° C, providing good confidence in the thermal simulation.

Table 3. Summary Calibration Statistics for Thermistors at CCR2, 2007-2013

Metric	CCR2		
	AME	RMSE	Average Error
Thermistors - Average AME/ RMSE / Average Error	0.7°C	0.7°C	-0.3°C
Temp. Diff. (1m to 7m) – AME/RMSE/ Average Error	0.6°C	0.9°C	+0.3°C

5.3 Nutrients

The model simulates ammonia, nitrate, and phosphate as state variables. Simulation of these nutrient concentrations in the reservoir was most sensitive to algal settings, anaerobic (0-order) sediment release rates of ammonia and phosphate[6] (PO4), and sediment burial rates affecting aerobic sediment release rates of ammonia and PO4.

The reservoir exhibits significant cycling of nutrients between the water column and the sediments. Aerobic and anaerobic decay of organic matter in the sediments releases ammonia and PO4. Algae take up nutrients and eventually settle to the sediment surface for subsequent decay. Due to mixing from the bottom to the top at times through summer months, this cycling occurs in Cherry Creek Reservoir throughout the growing season.

The calibrated Cherry Creek model simulates these nutrient cycling mechanisms; however, though the calibration process, several assumptions inherent to the existing version of CE-QUAL-W2 were identified for possible future refinements to further improve the simulation. Note that identifying these potential future refinements is not intended to undermine the perceived strength of the current calibration. The calibrated model is a powerful and useful tool for the Authority as is. Instead, these are documented to support informed review of simulation results, consideration of application results, and to keep as placeholders for possible areas of future model development.

- **Sediment Burial Rates:** The sediment burial rate in the model is a constant value. This setting defines a loss rate of settled detritus (primarily algae in this system) from the sediment compartment that would be available of aerobic decay. It is likely that this burial rate varies over time, responding to effects such as storm loading of higher concentrations of solids. Sediment diagenesis simulation may (or may not) be beneficial in improving time-varying response related to sediment burial rates. A draft of W2 v. 4.0 with a beta version of sediment diagenesis capabilities was provided to Hydros by Dr. Wells on June 22, 2015. However, based on the schedule and budget, use of this updated tool has been deferred for future refinements.

- **Stoichiometry of Algae:** The stoichiometry of algae is a constant in the model, but in reality can vary over time. Excess PO4, in particular, can be taken up and stored by algae when it is available in abundance. This is termed “luxury uptake”. The model does not currently simulate luxury uptake; however, recoding this effect may be helpful for future Cherry Creek modeling refinements. Given high concentrations of excess PO4 (relative to nitrate and ammonia) present at the top of the reservoir, simulation of luxury uptake is expected to improve the simulation of PO4 response in Cherry Creek.

[6] For the purposes of modeling, observations of SRP were considered to represent phosphate for inputs and review of results. Throughout the calibration section observations of SRP are referred to as phosphate or PO4.

- **Nitrogen Fixation:** Nitrogen fixation is expected to be occurring in the reservoir, based on simulation and review of observed data patterns for ammonia, nitrate, and PO4. Nitrogen fixation is the process by which some cyanobacteria fix atmospheric nitrogen (N_2) for use in cellular development when nitrate / ammonia availability is limited. The model simulates nitrogen fixation, and a nitrogen fixing algal group (Group 3) is designated in the Cherry Creek Reservoir model. The model, however, currently only simulates nitrogen fixers as obligate nitrogen fixers. In other words, when this algal group grows, it can only get its nitrogen from atmospheric nitrogen. In reality, nitrogen fixation is an energetically expensive process for the cells, and cells revert to direct ammonia / nitrate uptake when these forms of nitrogen become available in the water column again (Bergman et al., 1999, pg. 4). Recoding nitrogen fixation as a future refinement to the Cherry Creek model, to allow nitrogen fixing cyanobacteria to revert to direct ammonia / nitrate uptake in the presence of adequate concentrations of dissolved forms, could further improve the simulation of nitrogen subspecies and algae.

In spite of these items that may merit future refinements, the model currently performs well overall simulating the nutrient response in this dynamic system. The model simulates the major processes of nutrient uptake, nutrient release from sediments, and the resulting nutrient gradient from bottom to top. Calibration summary statistics for nitrate, ammonia, and PO4 are presented in Table 4. These include average error, AME, and RMSE for CCR1, CCR2 and CCR3. The following discussions compare the time-series simulation results to observations.

Table 4. Summary Calibration Statistics for Nutrients, 2003-2013

	CCR1			CCR2						CCR3		
	Photic (0-3m)			Photic (0-3m)			Bottom			Photic (0-3m)		
	Avg. Error	AME	RMSE	Avg. Error	AME	RMSE	Avg. Error	AME	RMSE	Avg. Error	AME	RMSE
Nitrate and Nitrite as N (mg/L)	0.04	0.04	0.08	0.03	0.04	0.09	0.04	0.05	0.09	0.03	0.05	0.09
Phosphate as P (mg/L)	0.03	0.03	0.05	0.03	0.04	0.05	0.03	0.05	0.06	0.03	0.04	0.05
Total Ammonia as N (mg/L)	0.03	0.04	0.06	0.03	0.04	0.06	0.07	0.09	0.12	0.03	0.04	0.06

5.3.1 Ammonia

Observed and simulated ammonia concentrations at the top and bottom of the reservoir at CCR2 are presented in Figure 46. The model effectively simulates uptake of ammonia by algae in the photic zone as well as ammonia release from the sediments. At times, the model over-simulates ammonia concentrations, likely reflecting small differences in actual and simulated algal uptake. However, the general magnitudes and seasonal patterns, as wells as the relative concentration gradient from the bottom to the top of the reservoir are reflected in the simulation results. As a reminder for reference, the inflow ammonia volume weighted average concentration was 83 ug/L through this period.

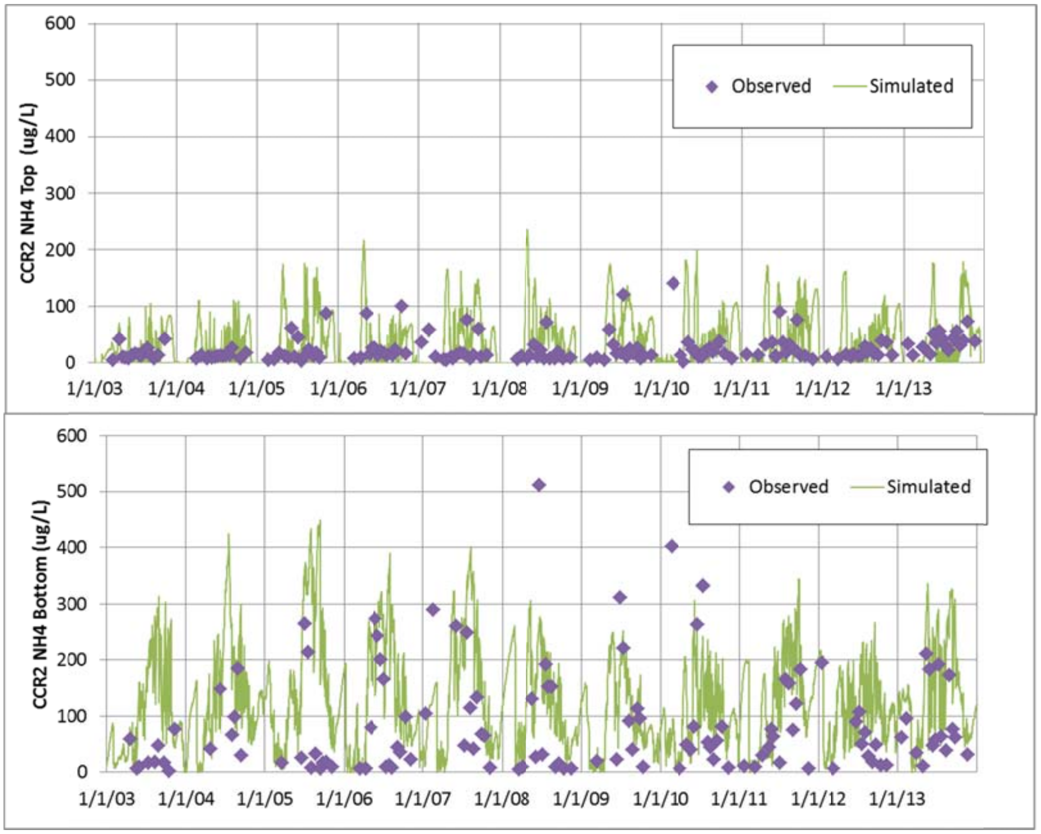

Figure 46. Simulated and Observed Ammonia Concentrations at CCR2, 2003-2013

5.3.2 Nitrate

Observed and simulated nitrate concentrations at the top and bottom of the reservoir at CCR2 are presented in Figure 47. The model effectively simulates uptake of nitrate by algae in the photic zone. Nitrate is not released from the sediments, and nitrification produces very little nitrate in the simulation. As a reminder for reference, the inflow nitrate volume-weighted average concentration was 840 ug/L through this period. As with ammonia, at times, the model over-simulates nitrate concentrations, likely reflecting small differences in actual and simulated algal uptake. However, the model does simulate significant nitrate uptake relative to inflow concentrations and the lack of a significant concentration gradient from top to bottom.

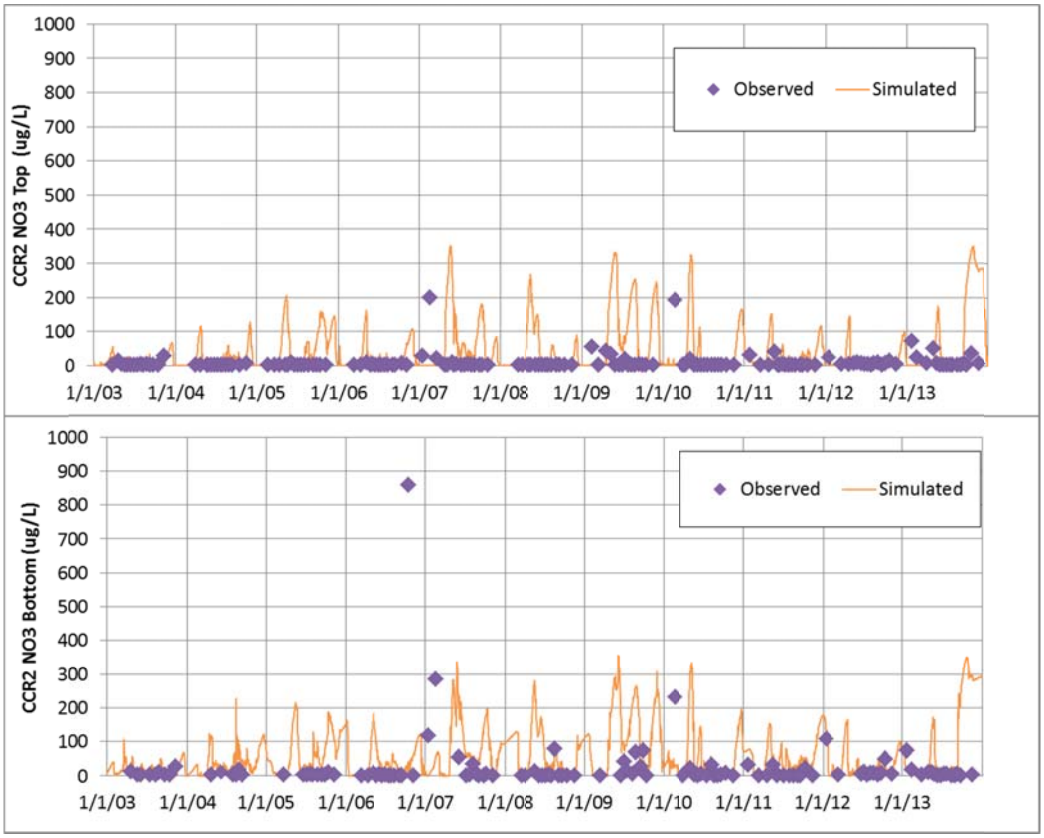

Figure 47. Simulated and Observed Nitrate Concentrations at CCR2, 2003-2013

5.3.3 Phosphate

Observed and simulated PO4 concentrations at the top and bottom of the reservoir at CCR2 are presented in Figure 48. The model effectively simulates the seasonal excess PO4 at the top of the reservoir, sediment releases of PO4, and the concentration gradient of decreasing PO4 concentrations with decreasing depth. At times, particularly in late summer and early fall, the model over-simulates PO4 concentrations in the reservoir. The lack of luxury uptake behavior in the model may help to explain this. As a reminder for reference, the inflow PO4 volume weighted average concentration was 117 ug/L through this period.

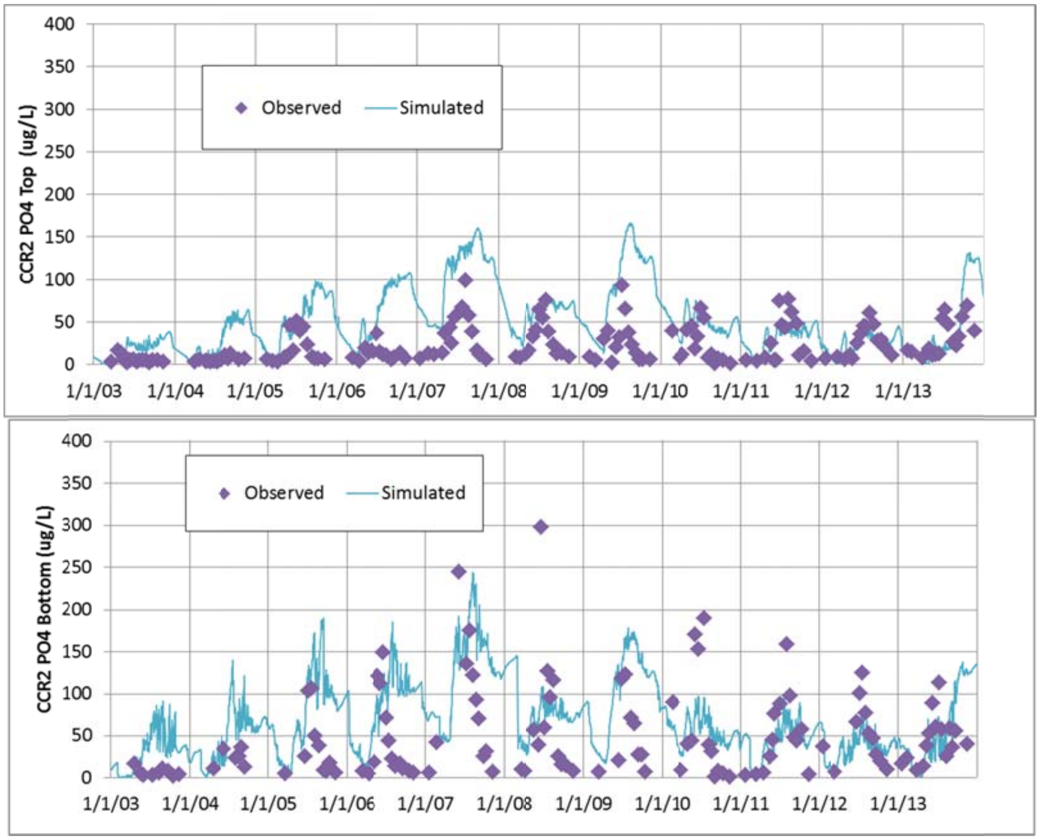

Figure 48. Simulated and Observed Phosphate Concentrations at CCR2, 2003-2013

5.4 Dissolved Oxygen

In Cherry Creek Reservoir, dissolved oxygen is primarily controlled by aerobic and anaerobic SOD, organic matter decay in the water column, and reaeration from the surface. The dissolved oxygen response, therefore, reflects the dynamic nature of mixing and algal activity in the reservoir. Calibration of dissolved oxygen was sensitive to anaerobic (0-order) SOD rates, the 1^{st}-order sediment burial rates, and the amount of algal biomass simulated.

5.4.1 Dissolved Oxygen Dataset Concerns

Review and attempted simulation of DO observations generated some questions about the dataset. First, there is an apparent shift in the observed DO values after 2010. Specifically, observed dissolved oxygen concentrations at the top are consistently lower, by a few mg/L, than the earlier period (2003 -2010) (Figure 49). In contrast, the model simulates fairly consistent, seasonally-varying DO concentrations at the top through the entire period (2003-2013). Other data were reviewed for relevant changes that could correspond to this temporal pattern in the observed DO dataset, but none were found. Specifically, there was no change in wind, relative chlorophyll *a* concentrations, or nutrient loading that matched this pattern.

Recognizing that calibration of DO probes can require input of barometric pressure, a theory was tested. Barometric pressure is typically reported as a sea-level pressure in weather reports and must be converted to a pressure at the altitude of the reservoir. If an altitude adjustment is not made, an approximate error of a 25% increase in observations might be expected. Such an adjustment produces a better fit with modeled data. Because this theory is speculative, however, no adjustments were made to observed data in comparison with model data.

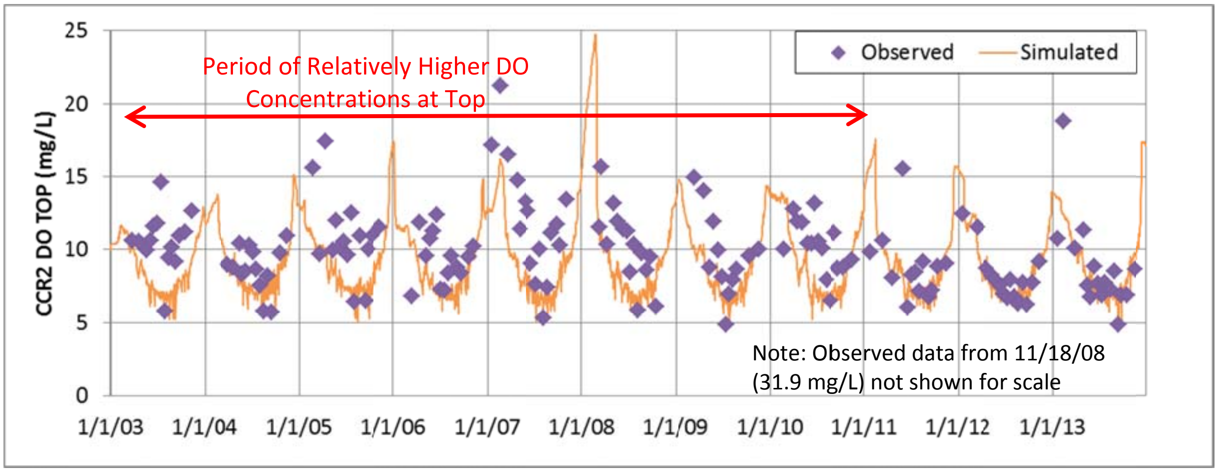

Figure 49. Observed and Simulated Dissolved Oxygen at the Top, CCR2, 2003-2013

Other questions about the observed DO data were also noted. First, there are cases of high super-saturation of DO at the top (and sometimes at the bottom) when chlorophyll *a* concentrations are relatively low. For example, on April 13, 2005, the DO profile at CCR2 indicates 17.1 mg/L DO at the top and 16.8 mg/L at the bottom (Figure 50). For the given water temperature, this corresponds to a super-saturation of 185% and 182%, respectively. While algae can cause super-saturation of DO, such extreme super-saturation would not be expected at this time when winds were not particularly calm and observed chlorophyll *a* was 16.8 ug/L. Examples of this are also apparent in more recent years (e.g., May 31, 2011 data indicate at dissolved oxygen concentration of 15.5 mg/L [153% saturation] with a chlorophyll *a* concentration of 12.4 ug/L).

Second, fish response does not support such high percent DO saturation. Lethal toxicity due to DO super-saturation for walleye was found to occur at 125-130% super-saturation in a study from the State of Washington (Beeman et al., 2003). In that study, half the population of walleye died within 62 hours of exposure to 130% DO super-saturation conditions. Based on this, it is unlikely that super-saturation at or above this value occurred from top to bottom in Cherry Creek Reservoir over the period of record. Yet super-saturation at or much greater than 125% is in the observed dataset to depths of 7+ m on multiple dates between 2003 and 2013.

Third, there were also cases of unexpected large differences in observed super-saturation of DO between CCR1 and CCR2 that are not explained by differences in chlorophyll *a*[7]. For example,

[7] As described in Boyer et al. (2014) chlorophyll *a* concentrations between CCR1 and CCR2 match well throughout the years with very little variation between the locations.

on October 16, 2005, DO at the top of the reservoir at CCR1 was 12.5 mg/L (143% saturation) with a chlorophyll *a* concentration of 6.3 ug/L. The same-day sample at CCR2 was 6.5 mg/L DO (75% saturation) with a chlorophyll *a* concentration of 9 ug/L. These profiles are shown in Figure 50.

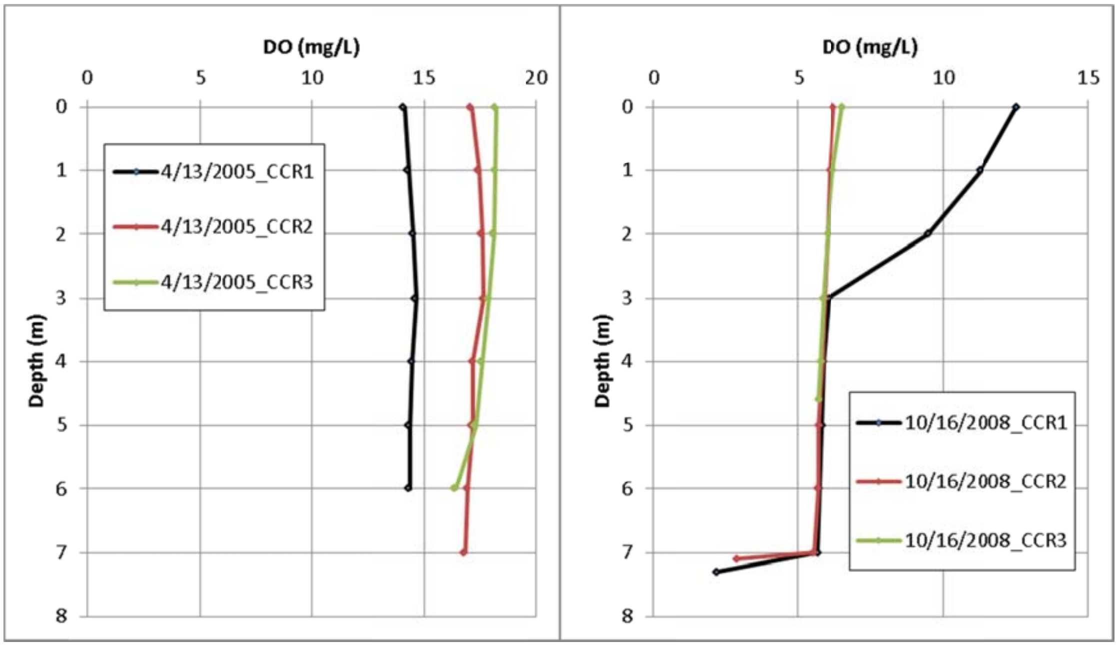

Figure 50. Observed DO Profiles at CCR1, CCR2, and CCR3 on April 13, 2005 and October 16, 2008

This does not represent a complete QAQC review of all observed DO profiles relative to observed chlorophyll *a* data, since that was beyond the scope of this effort. Instead, this is a note of perceived uncertainty in the observed dissolved oxygen dataset.

5.4.2 DO Calibration Results

Despite the perceived uncertainty in the observed dataset, these data were used without adjustment or censoring for calibration and generation of calibration statistics. Calibration statistics were generated from observed data profiles at CCR2 and CCR3. The complete set of observed and simulated DO profiles is presented in Attachment C.

Table 5. Summary Calibration Statistics for Dissolved Oxygen Profiles, 2003-2013

Metric	CCR2	CCR3
Profiles - Average AME	1.4 mg/L DO	1.3 mg/L DO
Profiles - Average RMSE	1.6 mg/L DO	1.5 mg/L DO
Profiles - Average Error	-0.4 mg/L DO	0.1 mg/L DO

Simulated bottom DO concentration data are plotted against the deepest observed DO data at CCR2 in Figure 51. The model simulates highly variable DO concentrations at the bottom the reservoir, reflecting intermittent mixing due to intermittent isothermal or near isothermal conditions and associated wind and aerator-driven mixing. The model also simulates an increase in dissolved oxygen at the bottom at CCR3 starting in 2008 (Figure 52), matching relative observations during destratification system operations. The increased mixing, bringing oxygenated water to the bottom, is responsible for this increase, in the model and in the field. As noted in Section 3.5, the observed data suggest possible apparent induced sediment oxygen demand at CCR2. This effect is not clearly reflected in the simulation results, which would not be expected to simulate reductions to the diffusive boundary layer. A future refinement could consider modifying the code or testing the new sediment diagenesis version to improve simulation of apparent induced increases to SOD. The current calibrated model, however, reasonably simulates patterns and ranges of DO at the bottom of the reservoir.

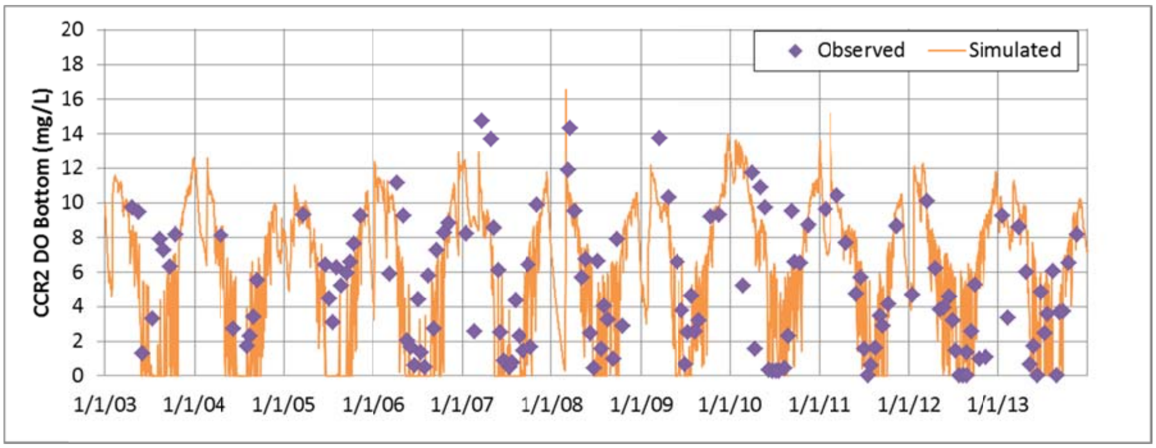

Figure 51. Observed and Simulated Dissolved Oxygen at the Bottom, CCR2, 2003-2013
(Note: Not all observations were within 0.5 m of the bottom. Data from 2003-2006 are typically greater than 0.5 m from the bottom.)

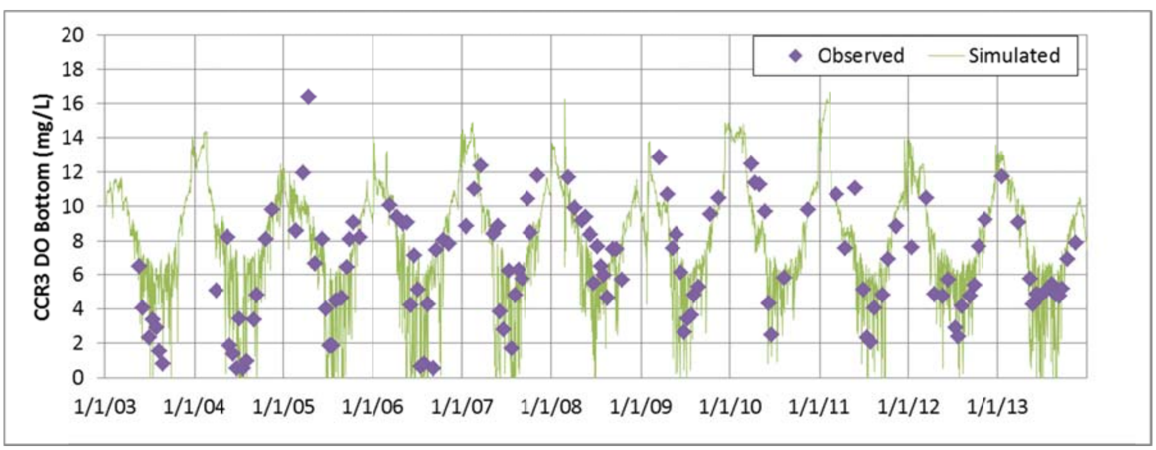

Figure 52. Observed and Simulated Dissolved Oxygen at the Bottom, CCR3, 2003-2013
(Note: Not all observations were within 0.5 m of the bottom. With the exception of several months in 2007, 2008, and 2010, most of these observed data are from roughly 1 m above the bottom.)

The simulated data were also reviewed to evaluate the simulation of the DO gradient from the top of the reservoir to the bottom. As shown in Figure 53, the model simulates a gradient similar to the observed patterns.

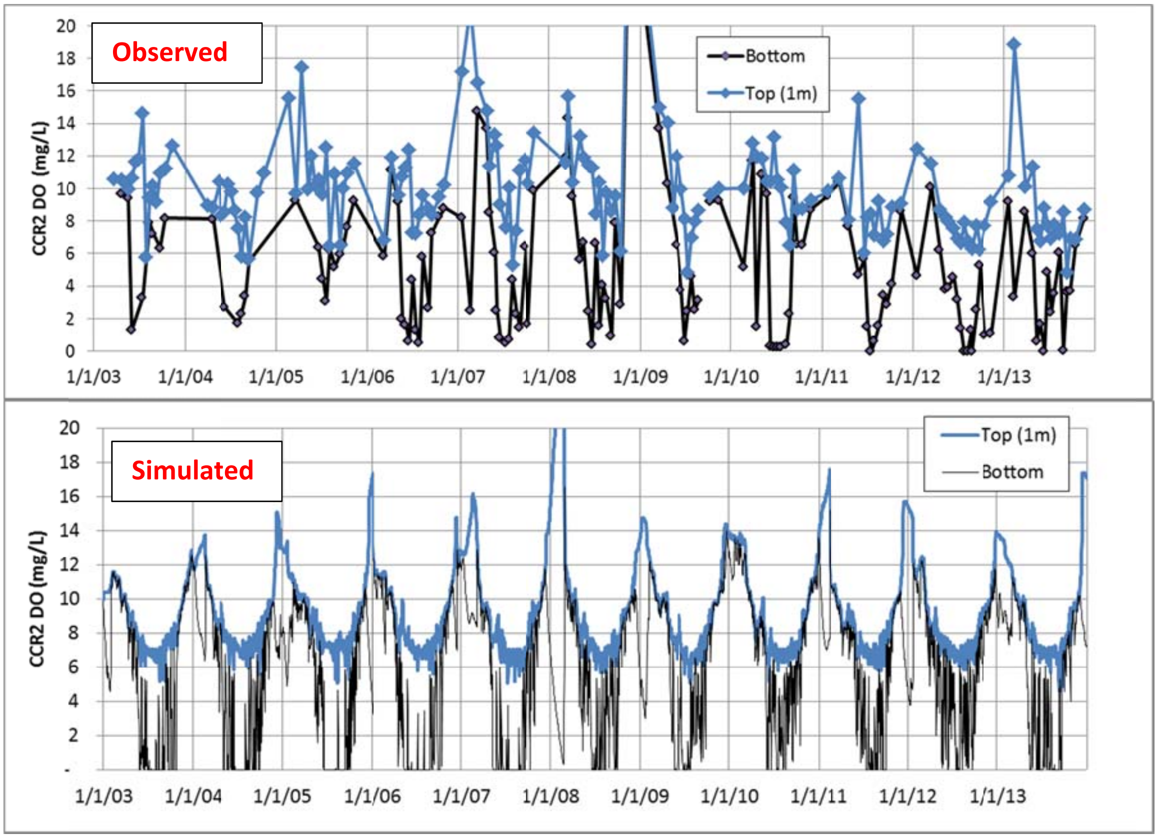

Figure 53. Observed and Simulated Dissolved Oxygen Gradient from Top to Bottom, CCR2, 2003-2013

(Note: Not all "bottom" observations in the upper figure were within 0.5 m of the bottom. Data from 2003-2006 are typically greater than 0.5 m from the bottom.)

Overall, despite the remaining uncertainty the observed DO data at times, the model produces a reasonable simulation of concentration patterns and ranges as well as the general gradient from top to bottom. Further, the observed increase in DO at the bottom the reservoir at CCR3, attributable to mixing by the destratification system, is being simulated by the model. To help resolve some of the remaining uncertainty, it was recommended during a check-in meeting with the Reservoir Modeling Committee that the Authority install continuous DO probes at 1 m below the surface and at 0.5 m above the bottom of the reservoir at CCR2 (Hydros presentation on March 23, 2015 conference call). These data will help determine how dynamic the actual DO response is in the deepest parts of the reservoir and provide more information to help assess potential apparent induced SOD. Similar monitoring at CCR3 could also be helpful to refine SOD settings in the model; however, continuous DO data at this location is considered less critical.

5.5 *Algae*

Simulation of summertime chlorophyll *a* in the reservoir was the primary calibration target for model development due to the site-specific standard. Simulation of algal response in the reservoir was highly sensitive to a wide range of algal settings, including algal growth rates, mortality, settling, temperature rate multipliers, and stoichiometry. The nitrogen-fixing algal group was also sensitive to the half-saturation concentrations for PO4. Algal growth was also sensitive to sediment burial rates and ammonia and PO4 release rates from the 0-order sediment compartment. Reasonable simulation of light transmission in the reservoir was also needed. Adjustments to reflect the 2012 fish kill and possible variability in zooplankton pressure (described in Section 4.6) were also important for the simulation of chlorophyll *a*.

The simulated chlorophyll *a* concentrations are plotted with observed data for CCR2 in Figure 54. The model reasonably simulates general patterns and concentration ranges, though the timing of peaks do not always match, as reflected in point-by-point statistics. Calibration summary statistics for the entire simulation period, including months of the year are presented in Table 6.

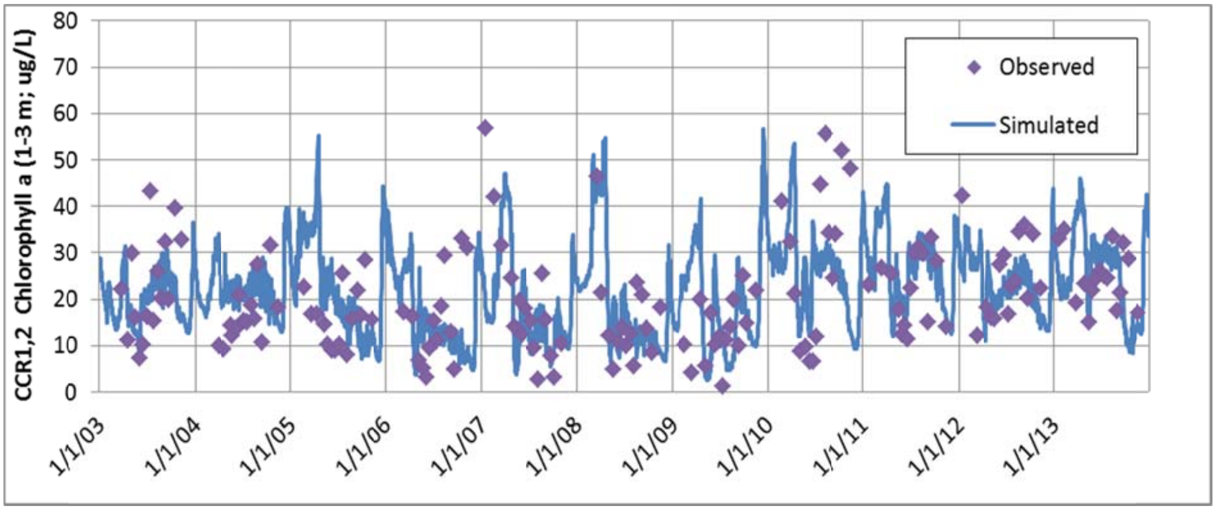

Figure 54. Simulated and Observed Chlorophyll *a* Concentrations in the Photic Zone, 2003-2013

Table 6. Summary Calibration Statistics for Chlorophyll *a* Concentrations in the Photic Zone, 2003-2013, All Months

Metric	CCR1	CCR2	CCR3
Average Error	-0.1 ug/L	0.5 ug/L	0.4 ug/L
AME	9 ug/L	9 ug/L	8.5 ug/L
RMSE	12 ug/L	12 ug/L	11.6 ug/L

Simulated summertime (July through September) average chlorophyll a concentrations are compared to observed concentrations in Figure 55. These show a good pattern and value match for all simulated years. Summary calibrations statistics for the summer average chlorophyll *a* concentrations are presented in Table 7. All error statistics are less than 3 ug/L for the summer average prediction. Further, predictions for all individual years were within 5 ug/L of observations.

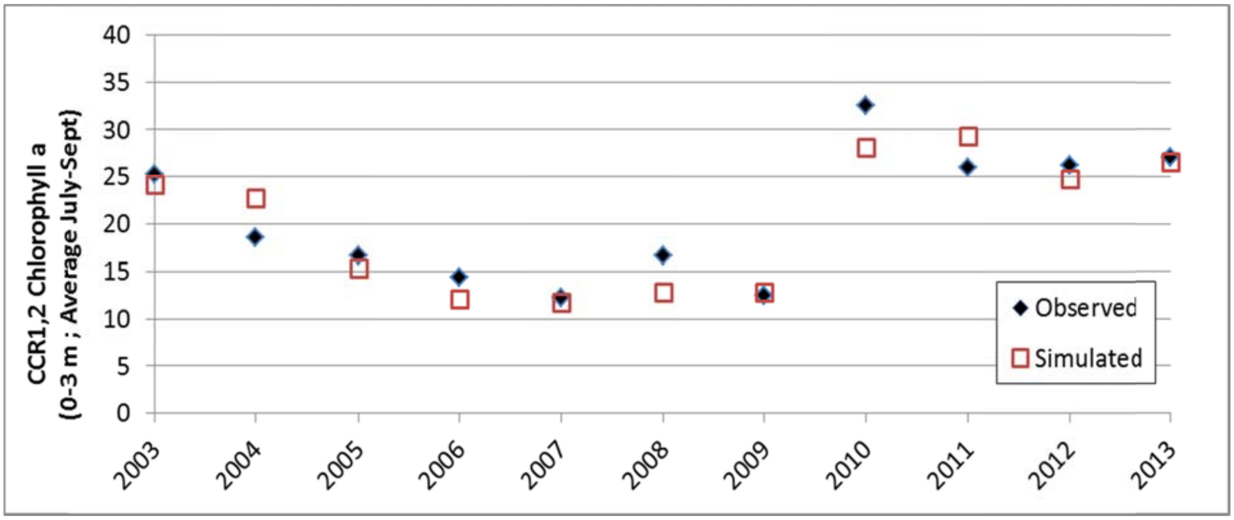

Figure 55. Simulated and Observed Summertime Average Chlorophyll a Concentrations in the Photic Zone, 2003-2013, July – September, CCR1 and CCR2

Table 7. Summary Calibration Statistics for Summertime Average Chlorophyll *a* Concentrations in the Photic Zone, 2003-2013, July – September, CCR1 and CCR2

Metric	Summertime Average Chlorophyll *a*
Average Error	-0.7 ug/L
AME	2 ug/L
RMSE	3 ug/L

6 Sensitivity Analysis

Task 3A of the Cherry Creek Reservoir Modeling Project was to conduct a sensitivity analysis on the calibrated model. The following subsections describe the approach taken and results.

6.1 Approach

A sensitivity analysis tests the effects on simulation results of varying select settings or model inputs. Sensitivity analyses can help identify areas of greatest uncertainty and possibly direct additional data collection to reduce that uncertainty. The sensitivity analysis was generally scoped to include variation of five parameters, to be informed by the process of model development and calibration.

Through development of the calibrated model, numerous model runs were conducted, varying a wide range of settings relevant to the Cherry Creek Reservoir. Through this process, an understanding of model sensitivity to these settings was developed. From that work, a relatively long list of sensitive settings was identified, including:

- Algal Settings for Groups 1,2, and 4 (growth, mortality settling, temperature rate multipliers, stoichiometry)
- Algal Settings for Group 3 (growth, mortality settling, temperature rate multipliers, stoichiometry, half-saturation for PO4)
- Zooplankton (growth, mortality settling, preferences., temperature rate multipliers)
- Sediment Burial Rates
- Ammonia, PO4 Release Rates / SOD

Also, through the calibration effort, areas of greater uncertainty and recommendations for additional data collection and possible future refinements were identified (discussed in Section 5).

Therefore, much of a sensitivity analysis was completed through the calibration process. This was discussed at the June 22, 2015 meeting with the peer reviewer Dr. Wells, and appropriate additional runs were proposed for the sensitivity analysis task. These runs focus on understanding the relative roles of major forcing functions in the model, including wind, aerator mixing, internal loading, and nutrient loading from major tributaries. Some of these forcing functions can be managed, and some cannot (e.g., wind). As such, these do not represent realistic management scenarios, but they were defined to help support selection of management scenarios. A total of seven model runs were performed, as listed in Table 8.

Table 8. Sensitivity Analysis Runs

Group	Run #	Run Description
Mixing	1	No Aerators
	2	Aerator Mixing x10
	3	No Wind
Inflow Nutrient Loading	4	Cherry Creek - Half Concentration NO3, NH4, PO4
	5	Cottonwood Creek - Half Concentration NO3, NH4, PO4
Internal Loading	6	Turn off 0-Order (Anaerobic) Sediment Compartment
	7	Turn off 1st-Order (Aerobic) Sediment Compartment

6.2 Results

Model simulations were conducted and reviewed relative to the calibration simulation (termed *Baseline* in these comparisons). Results were assessed for temperature, nutrient, dissolved oxygen, and algal response. Graphical output of results is presented in Attachment D.

6.2.1 Aerator Mixing Effects

The first set of sensitivity runs focused on extremes of aerator mixing. As discussed in Sections 4 and 5, data and modeling indicate Cherry Creek aerator operations affect temperature and dissolved oxygen concentrations at the bottom of the reservoir. Two aerator runs were conducted:

- **No Aerators**: The aerator module (AERATEC) was turned off.
- **Aerator Mix x10**: The vertical mixing factor for the aerators was increased to be ten times that simulated in the calibration run for the period 2008-2013. No change was made to the amount of oxygen loaded to the water column.

Review of water temperature results shows that aerator mixing, through the range of no aerator mixing to ten times the current aeration mixing, has a relatively small effect on water temperature at the surface of the reservoir. Increased mixing primarily serves to decrease the diurnal temperature range at the top of the reservoir, as shown in Figure 56 for 2009.

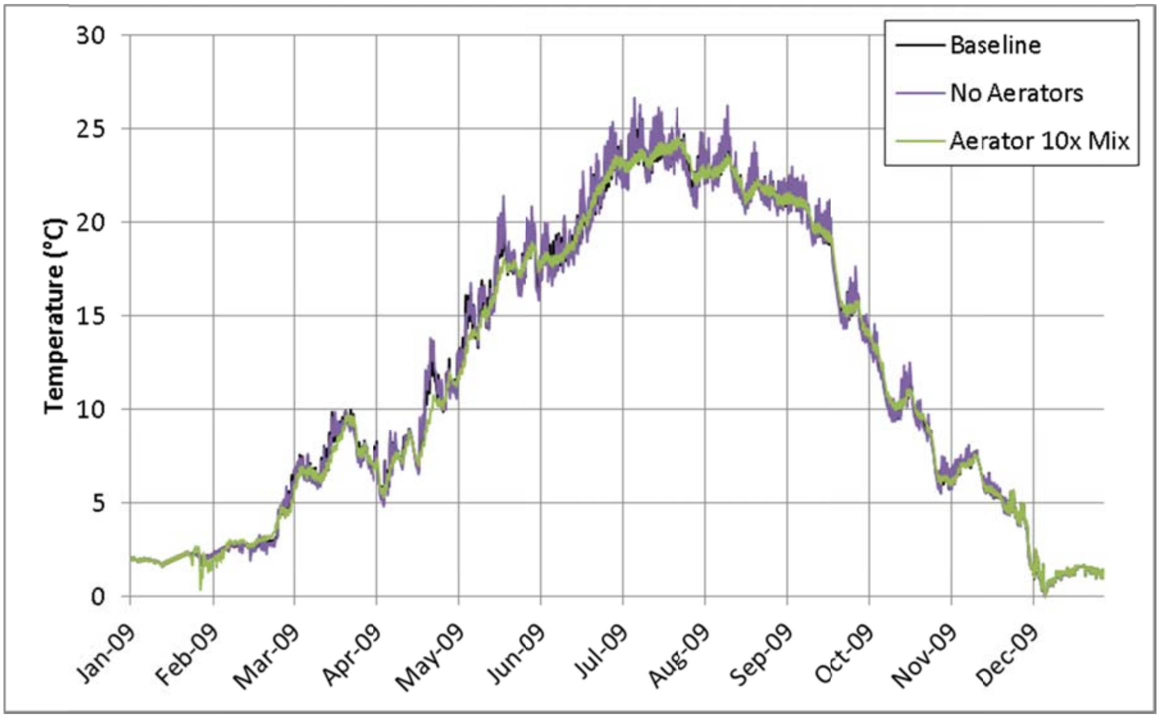

Figure 56. Hourly Simulated Water Temperature at 1 m for Aerator Mixing Simulations, CCR2, 2009

There is more of an effect of aerator mixing on temperature at the bottom of the reservoir. As shown in the example for 2009 (Figure 57), increased aerator mixing results in higher temperatures at the bottom of the reservoir. The effect at ten times the current level of mixing is to create continuous isothermal conditions from top to bottom in the reservoir while the aerators are operating (Figure 58). In the other direction, no mixing results in greater top-to-bottom temperature differences.

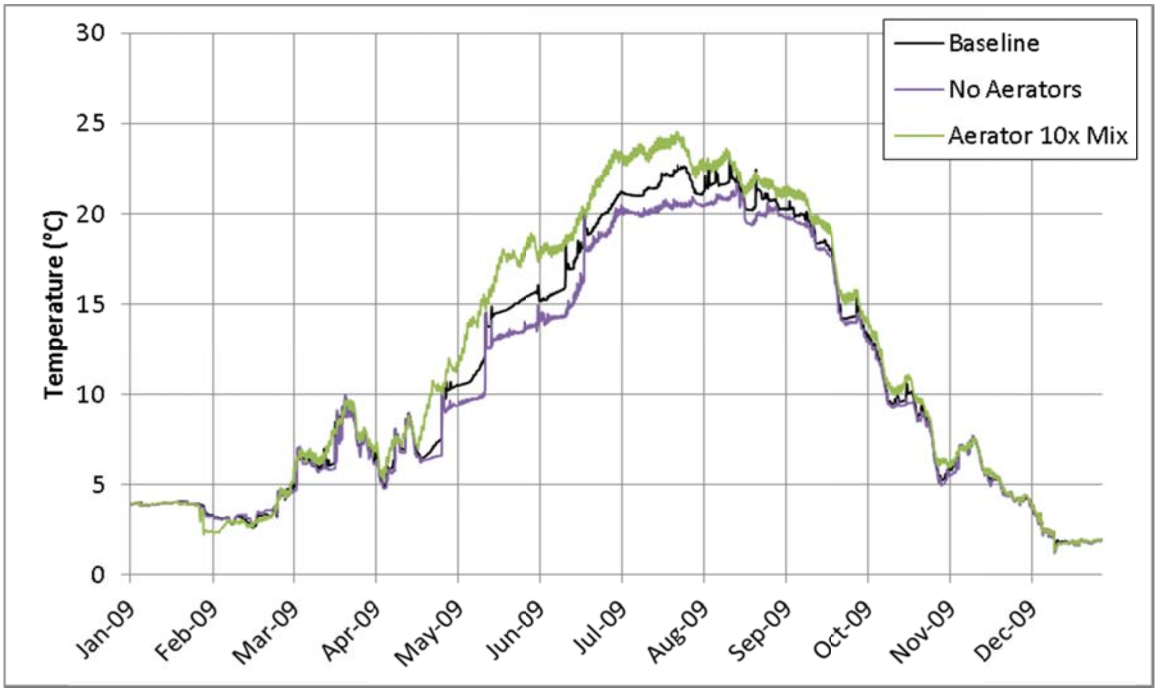

Figure 57. Hourly Simulated Water Temperature at the Bottom for Aerator Mixing Simulations, CCR2, 2009

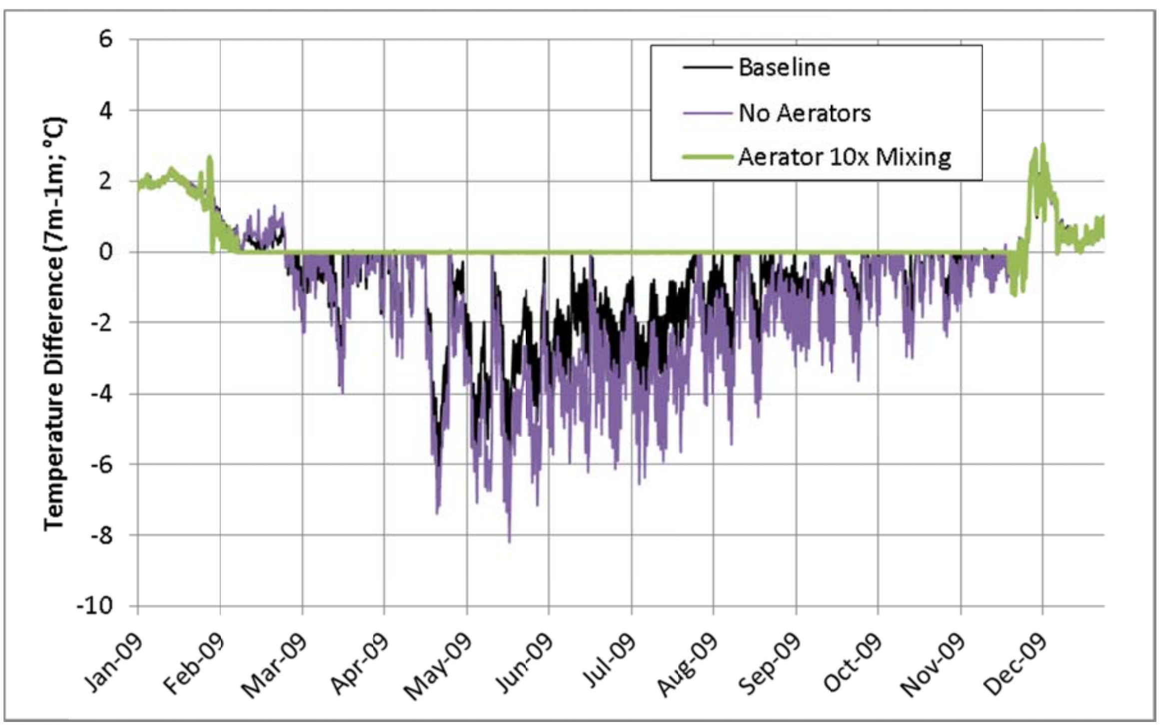

Figure 58. Hourly Simulated Difference in Water Temperature from 1 m to 7 m for Aerator Mixing Simulations, CCR2, 2009

Increased aerator mixing results in mixing of more DO from the surface to the bottom of the reservoir. With the 10x mixing simulation, the model indicates that dissolved oxygen concentrations at the bottom of the reservoir can be maintained above 5 mg/L in summer months (Figure 59). In contrast, the No Aerators simulation shows slightly lower DO concentrations at the bottom of the reservoir, relative to simulation of current conditions. Neither scenario results in large changes in DO at the top of reservoir.

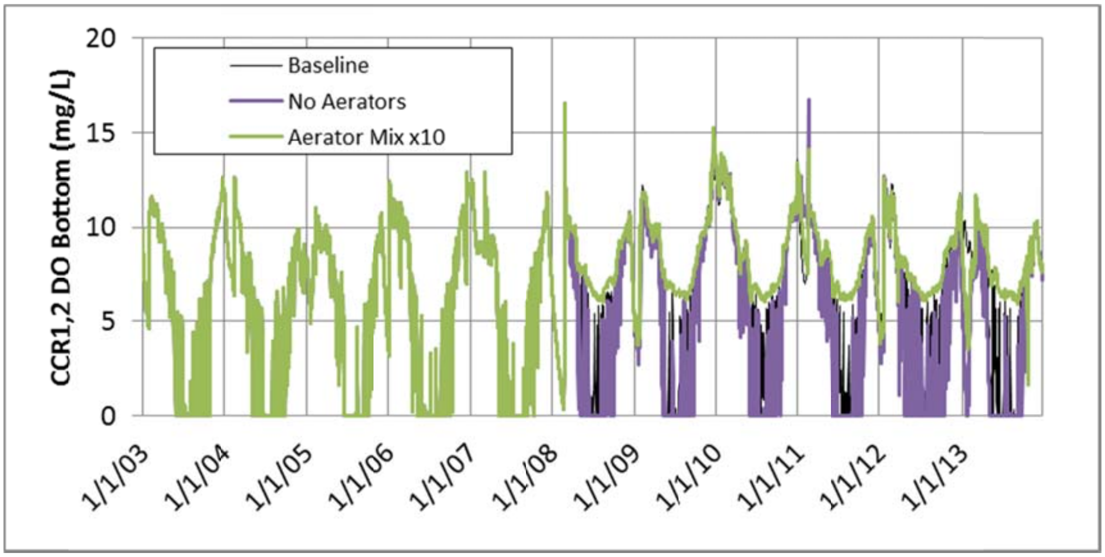

Figure 59. Simulated DO at the Bottom of the Reservoir (CCR2), Aerator Mixing Simulations, 2003-2013

The increased DO at the bottom with greater aerator mixing results in decreased anaerobic loading of ammonia and phosphate. The decreases in loading are apparent in nutrient concentrations at the bottom relative to the No Aerator simulation (e.g., Figure 60). Nutrient concentrations at the top of the reservoir show smaller changes, as compared to effects at the bottom. Remaining ammonia and phosphate concentrations reflect loading from external inflow and aerobic internal loading.

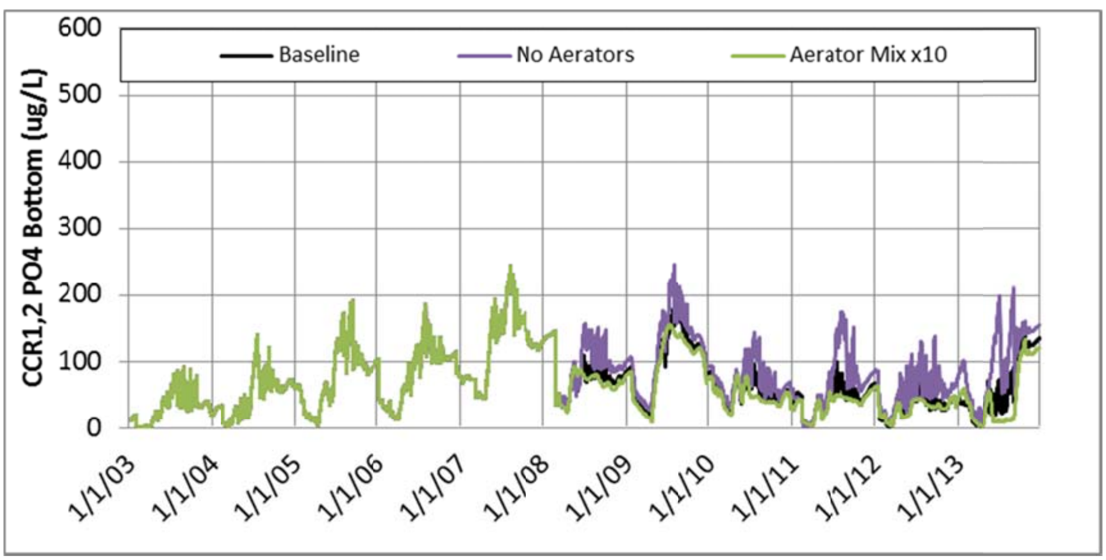

Figure 60. Simulated PO4 at the Bottom of the Reservoir (CCR2), Aerator Mixing Simulations, 2003-2013

The resulting decrease in internal anaerobic nutrient loading associated with aerator mixing results in a simulated decrease in chlorophyll *a* concentrations in the reservoir. Specifically, the simulation indicates that current aerator operation reduced summertime average chlorophyll *a* by an average of 2 ug/L from 2008-2013, ranging from less than 1 to 4 ug/L in individual years[8]. The Aerator Mix x10 simulation, compared to No Aerator, results in an average decrease in simulated summertime chlorophyll *a* of 5 ug/L, ranging from 2 to 9 ug/L. The simulation indicates, however, that the reservoir would not have met the 18 ug/L chlorophyll *a* standard in 2010, 2011, 2012, or 2013 even with ten times the current amount of mixing from the aerators.

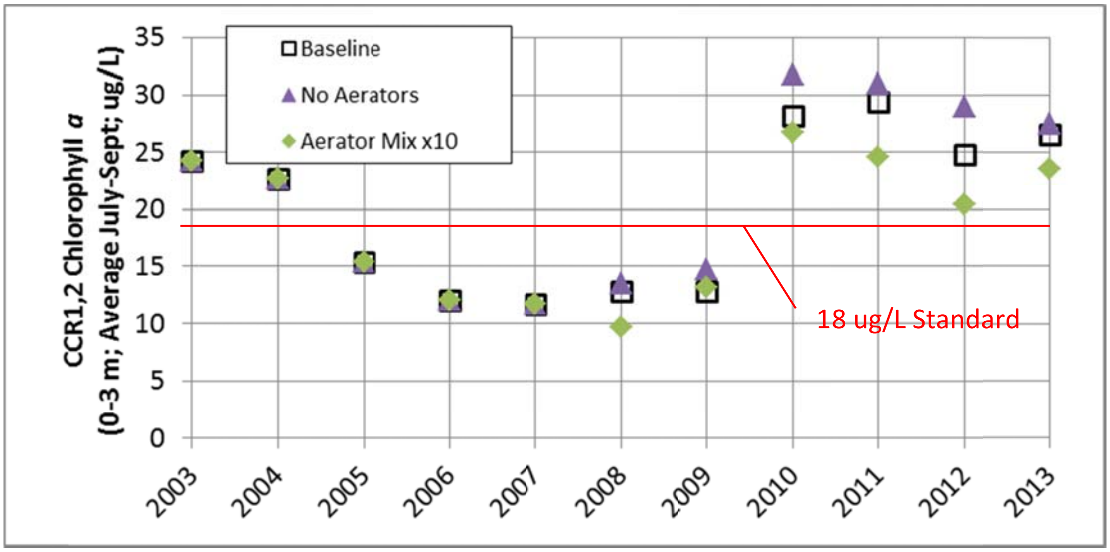

Figure 61. Simulated Average Summertime (July-September) Chlorophyll *a* (CCR2), Aerator Mixing Simulations, 2003-2013

6.2.2 Wind Mixing Effects

The second set of sensitivity analysis runs focused on identifying the role of the wind in the water-quality response of Cherry Creek Reservoir. Through the calibration process, model sensitivity to wind was apparent, particularly on the thermal response, when different meteorological inputs were tested and when wind sheltering coefficients were tested. To evaluate the effects of wind, a model run was conducted setting the wind to zero to evaluate the role of wind on mixing (Run Name: No Wind). There are two key considerations to note about this zero wind simulation before reviewing results:

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[8] The simulated magnitude of reductions in summertime chlorophyll *a* concentrations relative to the No Aerator simulation should be interpreted cautiously. As noted in preceding discussions, the model does not simulate the apparent induced SOD effects of mixing (e.g., reducing the diffusive boundary layer). Depending on the magnitude of any induced SOD at the moderate mixing (Baseline) level, anaerobic loading may not be reduced to the extent simulated. As a result, actual reductions in chlorophyll *a* concentrations may be even less. This effect is not a concern for interpreting the Aerator Mix x10 simulation results, however, because the amount of mixing of DO from the top to the bottom for that simulation is expected to be adequate to compensate for any induced SOD.

- **Effect of No Wind on Aerator Simulation:** Because the aerators must be defined in the model by a multiplier increasing vertical mixing, reducing wind to zero effectively also sharply reduces aerator mixing (2008-2013). Therefore, aerator mixing was also set to zero for this simulation. As such, results for 2008-2013 should be interpreted as the combined effects of no wind and no aerators. Therefore, the Baseline run for comparison to the No Wind simulation is the No Aerators simulation.
- **Evaporative Volume Not Affected:** Setting wind to zero does not reduce evaporation losses in this simulation because of the way evaporation was input into the model (Section 4.2). Therefore, there are not changes to the water surface elevation to consider in the water-quality response. Changes to the heat-exchange effects of wind on evaporative cooling, however, are simulated.

Review of water temperature results shows that wind has a significant effect on water temperature and mixing in Cherry Creek. The No Wind simulation exhibits much higher temperatures at the surface (Figure 62) and much lower temperatures at the bottom (Figure 63), as compared to the Baseline simulation. In fact, in the absence of wind the reservoir consistently remains stratified throughout the summer (Figure 64). The reservoir behaves like a dimictic water body, with turnover occurring in the spring and fall. These effects on water temperature in the reservoir are much greater in magnitude than the range of tested aerator mixing impacts.

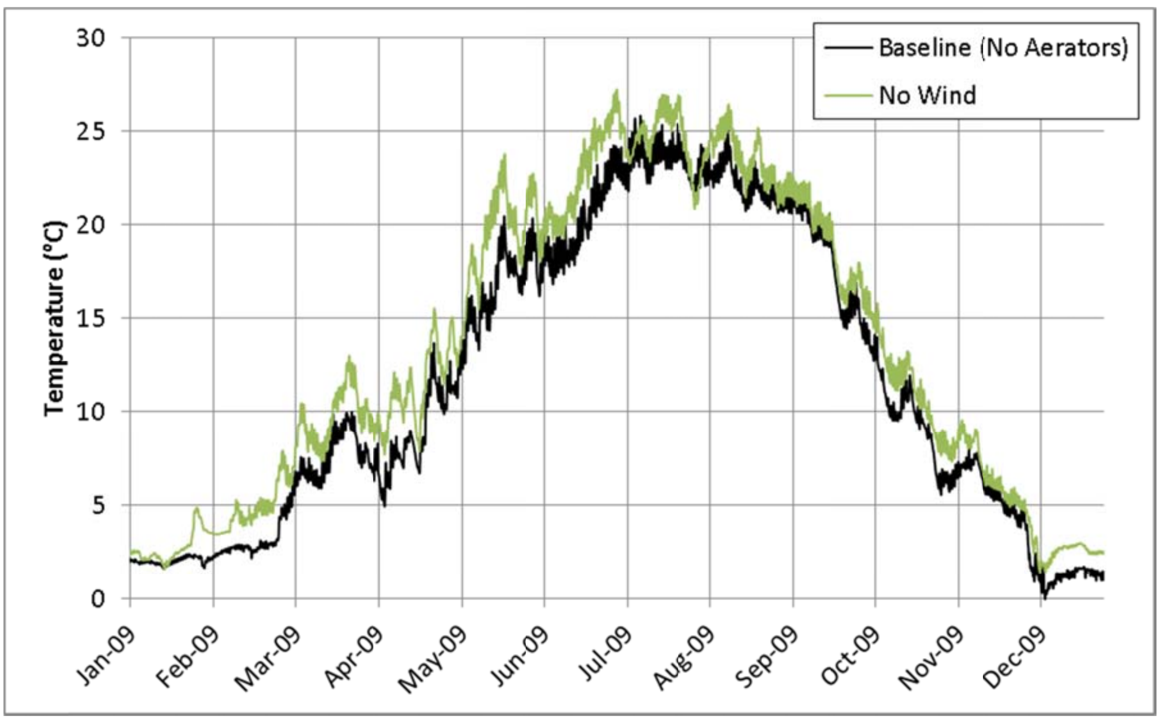

Figure 62. Hourly Simulated Water Temperature at 1 m for the No Wind and Baseline Simulations, CCR2, 2009

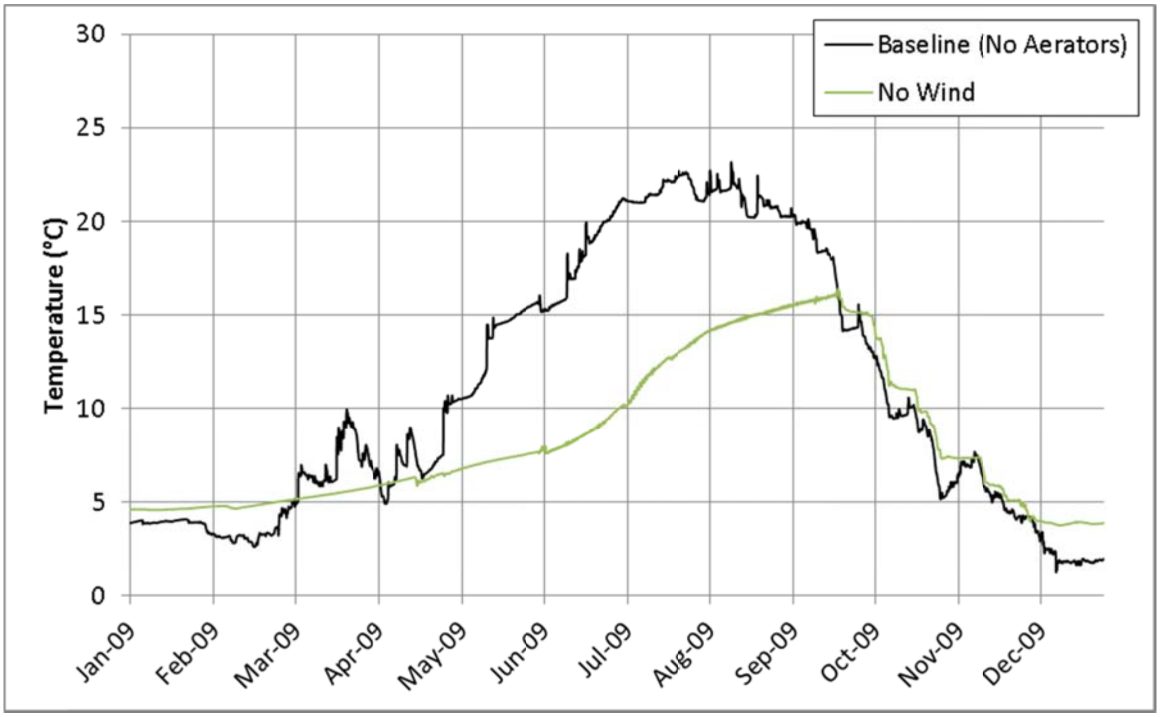

Figure 63. Hourly Simulated Water Temperature at the Bottom for No Wind and Baseline Simulations, CCR2, 2009

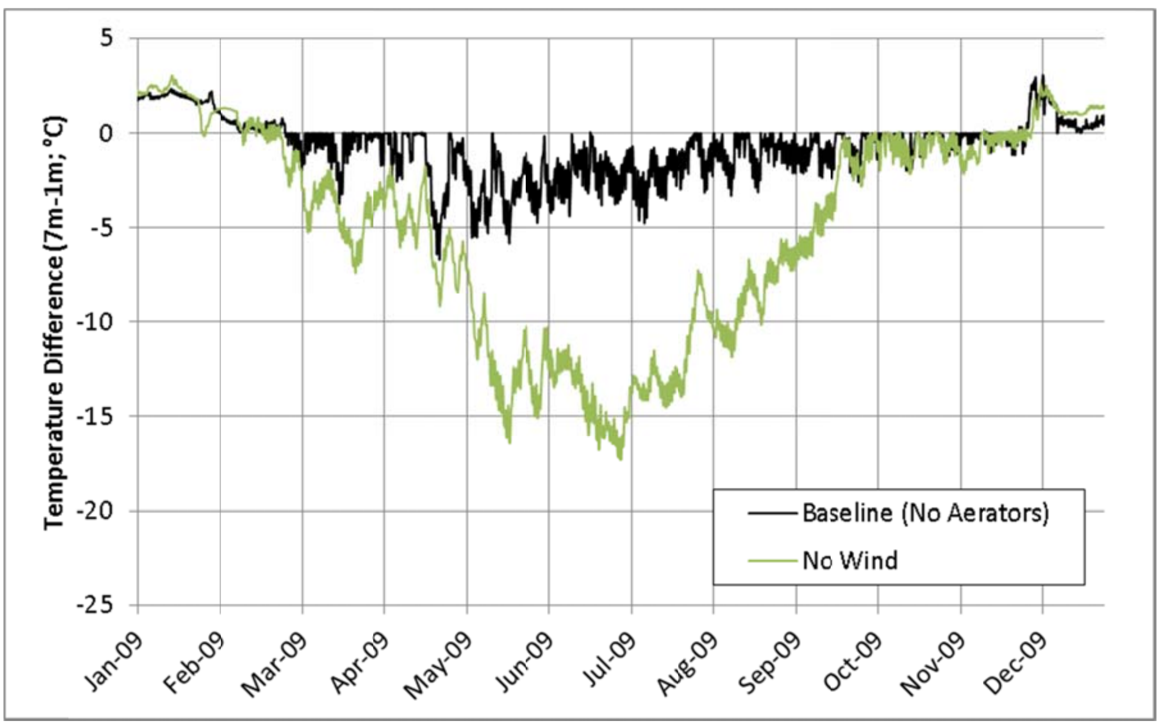

Figure 64. Hourly Simulated Difference in Water Temperature from 1 m to 7 m for No Wind and Baseline Simulations, CCR2, 2009

The stratification occurring because of reduced mixing in the No Wind simulation results in anaerobic conditions at the bottom of the reservoir throughout the summer (Figure 65).

Dissolved oxygen concentrations are simulated to reach levels just below 5 mg/L at the top of the reservoir most years during fall turnover for the No Wind scenario.

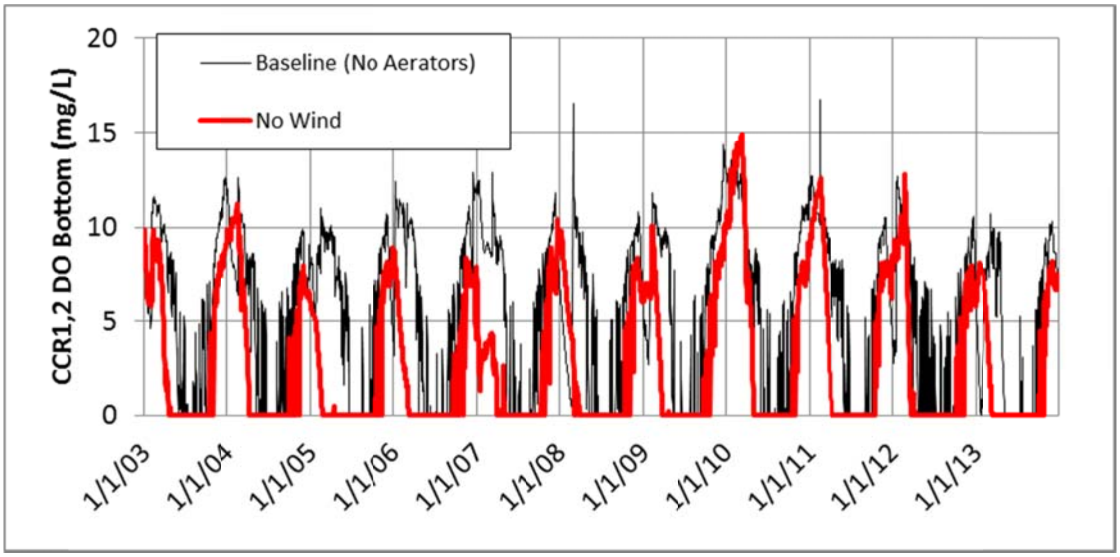

Figure 65. Simulated DO at the Bottom of the Reservoir (CCR2), No Wind and Baseline Simulations, 2003-2013
(No Wind Simulation Results Shown in Red to Make Pattern More Apparent)

As a result of these lower DO concentrations simulated at the bottom, anaerobic nutrient loading of ammonia and phosphate increases. Because there is reduced mixing from top to bottom for this No Wind simulation, nutrient concentrations at the bottom of the reservoir are simulated to increase sharply relative to Baseline (Figure 66). Summer nutrient concentrations at the surface, exhibit smaller decreases with no wind in most years.

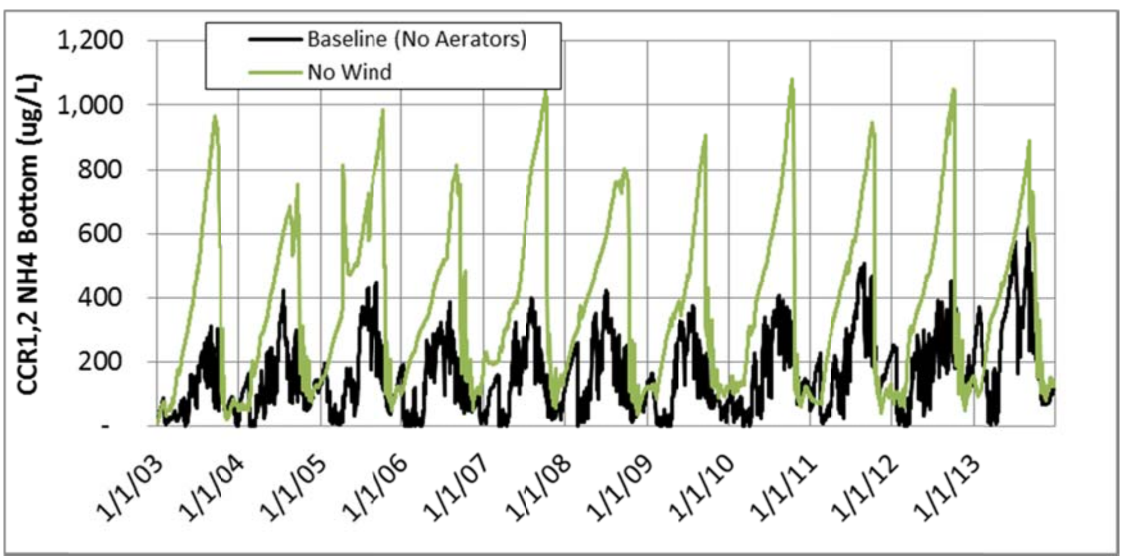

Figure 66. Simulated Ammonia Concentration at the Bottom of the Reservoir (CCR2), No Wind and Baseline Simulations, 2003-2013

Because there is less vertical mixing of internally-loading nutrients in the summer prior to fall turnover in the No Wind simulation, summertime chlorophyll *a* concentrations are simulated to decrease (Figure 67). The average decrease is 4 ug/L, ranging up to 10 ug/L (2013). There are two years with simulated increases in chlorophyll *a* for the No Wind simulation. These are 2007 and 2009. During these years, there was greater flow through the reservoir (see Figure 13 and Figure 14), resulting in more mixing. This allowed expression of some of the increased anaerobic internal load to the surface in the summer months in these years, resulting in increased algal growth.

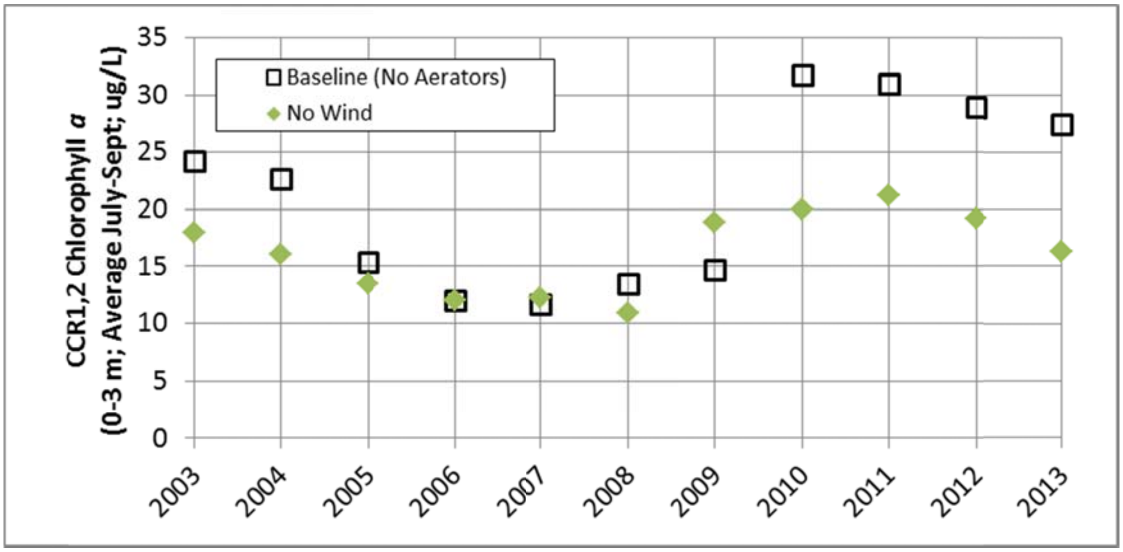

Figure 67. Simulated Average Summertime (July-September) Chlorophyll *a* (CCR2), No Wind and Baseline Simulations, 2003-2013

Based on this, it is conceptually important to understand the relative role of the actual wind over the reservoir. The wind is responsible for preventing stratification and for mixing nutrients from the bottom to the top. The role of the wind is greater than that of the current destratification system in terms of mixing.

6.2.3 Inflow Nutrient Loading

The third set of sensitivity runs was designed to give an indication of the relative sensitivity of water-quality response to nutrient loading from the two major inflow tributaries, Cherry Creek and Cottonwood Creek. Two runs were performed and compared to the Baseline simulation:

- **Half Cherry Creek Load:** Concentrations of ammonia, nitrate, and PO4 were halved for inputs for this run.
- **Half Cottonwood Creek Load:** Concentrations of ammonia, nitrate, and PO4 were halved for inputs for this run.

As would be expected, given the larger relative magnitude of loading from Cherry Creek, particularly for phosphate, the water-quality effects of decreased nutrient loading from Cherry Creek are generally greater than those for Cottonwood Creek. Decreased nutrient loading from both runs resulted in an overall decrease in reservoir productivity over the simulated period. As a result of decreased autochthonous organic matter to decay, simulated DO at the bottom of the reservoir was increased somewhat above Baseline (Figure 68). Though it is difficult to see in the figure, cutting nutrient loading from Cherry Creek in half resulted in a 12% decrease in the number of days with DO below 2 mg/L at the bottom of the reservoir. The effect was smaller for Cottonwood Creek nutrient reductions, with a simulated decrease of 6% for the same metric. There were no significant changes to the temperature or dissolved oxygen at the top of the reservoir in response to these sensitivity runs.

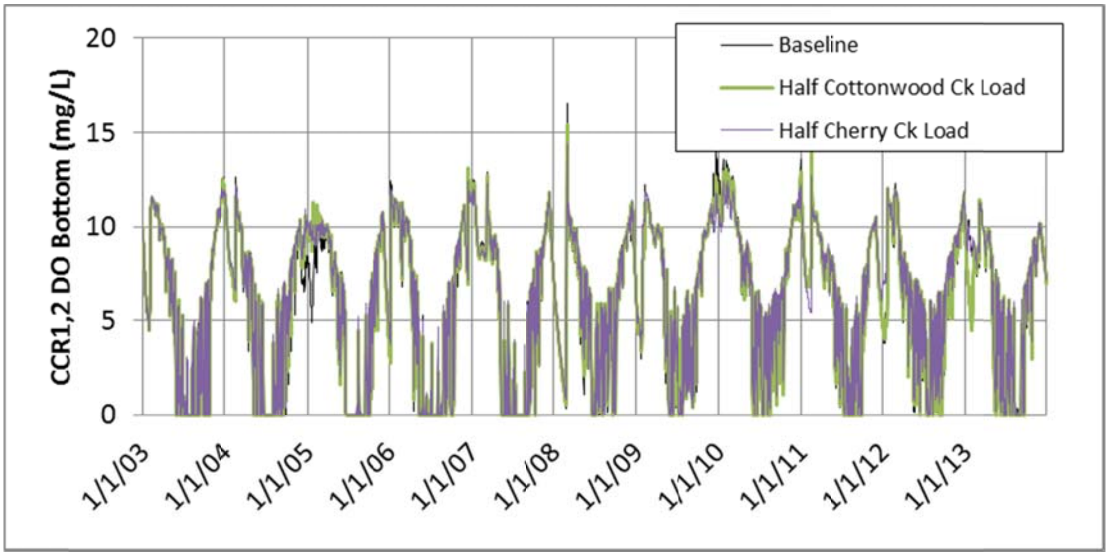

Figure 68. Simulated DO at the Bottom of the Reservoir (CCR2), Half Tributary Nutrient Loading and Baseline Simulations, 2003-2013

Nutrient concentrations simulated at the top of the reservoir decreased generally in proportion with the decreased load, with the greatest effects apparent in phosphate for halving the loading from Cherry Creek (Figure 69). Similar decreases were observed at the bottom of the reservoir. This effect on phosphate concentrations throughout the water column indicates the significance of inflow tributary loading to phosphate concentrations observed in the reservoir.

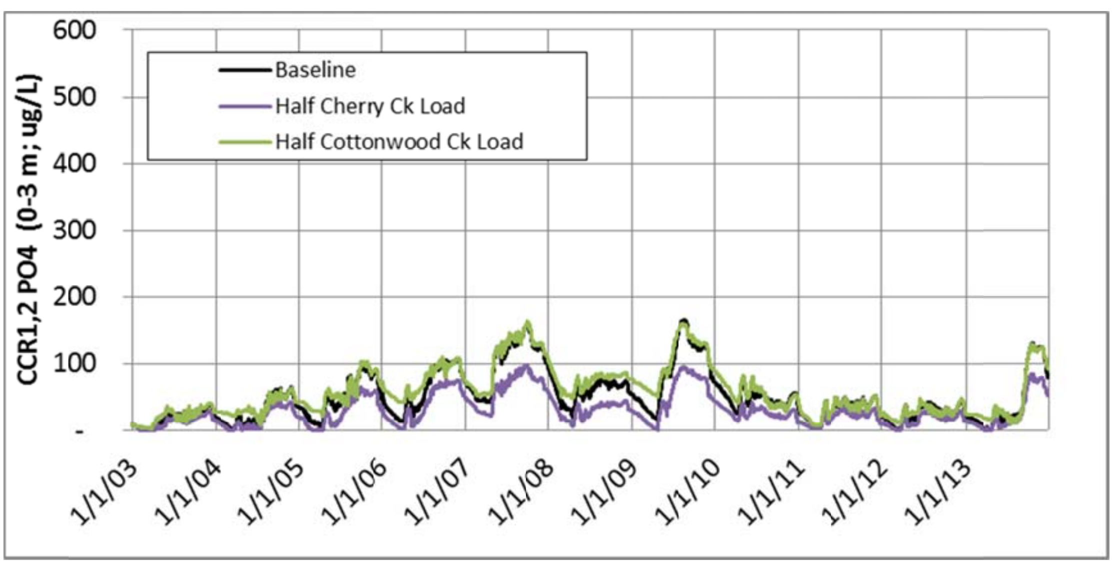

Figure 69. Simulated PO4 at the Top (0-3 m) of the Reservoir (CCR2), Half Tributary Nutrient Loading and Baseline Simulations, 2003-2013

The algal response, as would be expected, shows a greater reduction relative to Baseline for the halving of nutrient loading from Cherry Creek (Figure 70). Even with the phosphorus loading from Cherry Creek cut in half, the model does not simulate meeting the chlorophyll *a* standard in all years.

One thing not reflected in these modeling results is a possible long-term change in the anaerobic phosphate loading rate from the sediments after years of lower inflow phosphate concentrations. If this occurred, the algal concentrations would be expected to be even lower than simulated over the long term, possibly producing algal concentrations below the standard for this 50% inflow loading reduction scenario.

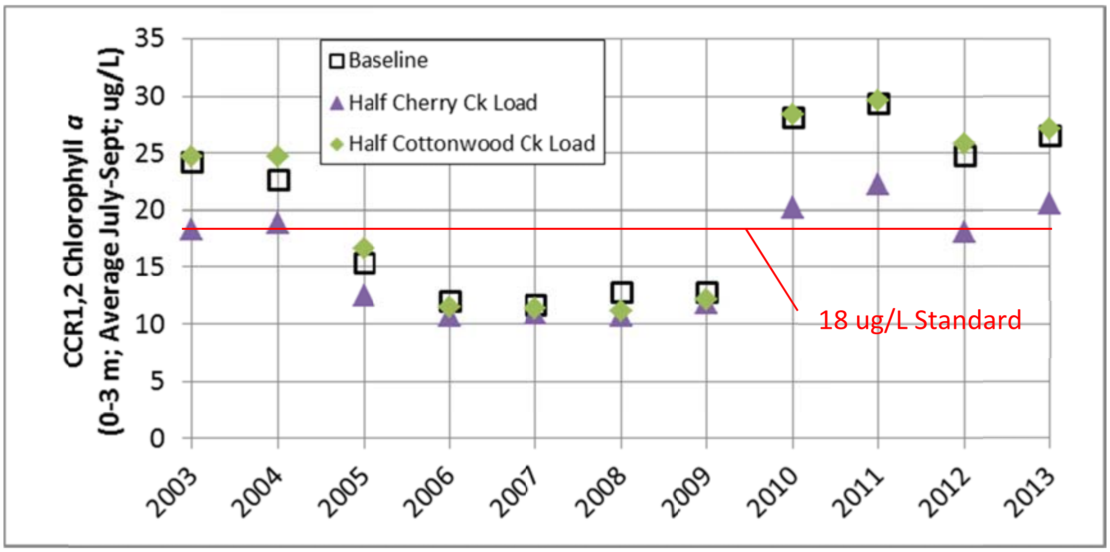

Figure 70. Simulated Average Summertime (July-September) Chlorophyll *a* (CCR2), Half Tributary Nutrient Loading and Baseline Simulations, 2003-2013

Interestingly, halving the nutrient load from Cottonwood Creek resulted in small simulated increases in chlorophyll *a* in roughly half of the simulation years. In the modeling, this is the result of reducing the nitrate and ammonia loading from Cottonwood Creek, which has historically exhibited higher concentrations of these nutrients, relative to Cherry Creek. This decrease in nitrogen nutrient loading, without a larger decrease is phosphorus loading (phosphate concentrations in Cottonwood Creek tend to be much lower than in Cherry Creek), resulted in simulation of a small increase in nitrogen-fixing algae, and a subsequent overall increase in summertime chlorophyll *a*. This response reflects the strongly nitrogen-limited nature of the reservoir. Based on this and the patterns described in Section 3.5 indicating higher TIN:PO4 inflow ratios from 2004-2009, it is recommended to consider the role of the inflow TIN:PO4 ratio as part of future management planning. Specifically, further reductions in nitrogen loading without corresponding reductions in phosphorus loading could have unintended consequences, encouraging additional nitrogen-fixing cyanobacteria.

6.2.4 Internal Loading

As noted in Section 4.4, internal loading from sediments to the water column consists of aerobic loading (1st-order sediment loading) and anaerobic loading (0-order sediment loading). Through calibration and review of model flux output, both are expected to be significant in Cherry Creek Reservoir. This makes conceptual sense because, while there is a high SOD and dissolved oxygen conditions become hypoxic and anoxic at times in the summer months, there are also periods of greater dissolved oxygen concentrations at the bottom due to intermittent mixing events. In more strongly stratified reservoirs, the aerobic processes can be less important through summer months if DO is consistently depleted at the bottom.

Two sensitivity analysis simulations were conducted to indicate relative roles of these processes in water-quality response in the calibrated model:

- **No Aerobic Internal Load:** The 1st-order sediment compartment in W2 was turned off.
- **No Anaerobic Internal Load:** The 0-order sediment compartment was turned off. This was done by setting the SOD value to zero. As a result anaerobic release of ammonia and PO4 was set to zero, and the temperature-based anaerobic oxygen demand was also stopped.

Both aerobic and anaerobic sediment decay processes consume oxygen. Turning of the anaerobic internal loading resulted in a greater increase in DO at the bottom of the reservoir. Figure 71 shows that in the absence of the anaerobic sediment compartment effects, summertime DO concentrations are higher. The simulation indicates that the average number of days per year with DO less than 2 mg/L at the bottom decreases by 90% in the absence of the 0-order compartment. In absence of the aerobic compartment, the decrease is 30% for the same metric. This indicates that oxygen consumption at the bottom of the reservoir is greater due to anaerobic processes, though aerobic processes are not insignificant.

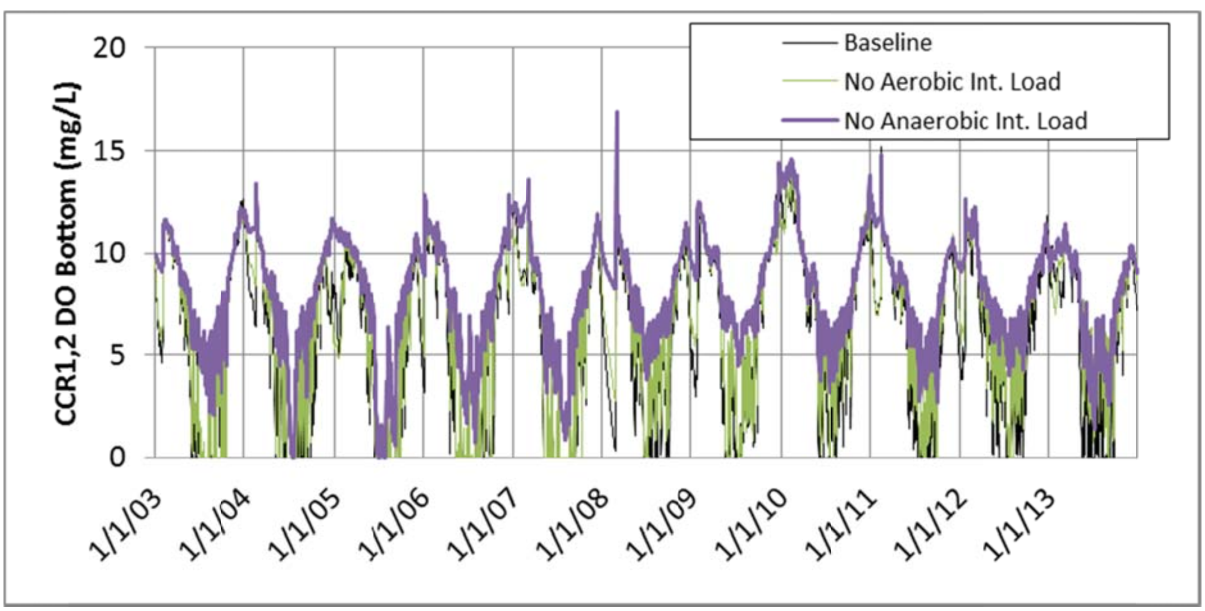

Figure 71. Simulated DO at the Bottom of the Reservoir (CCR2), Aerobic and Anaerobic Internal Loading and Baseline Simulations, 2003-2013

Reduced internal loading of nutrients is apparent for both of these sensitivity runs; however the effect is greater for the No Aerobic Internal Load simulation. Ammonia concentrations in particular are much lower at the top and bottom of the reservoir (Figure 72). Simulated effects on phosphate concentrations through the water column were similar to but slightly less than effects on ammonia.

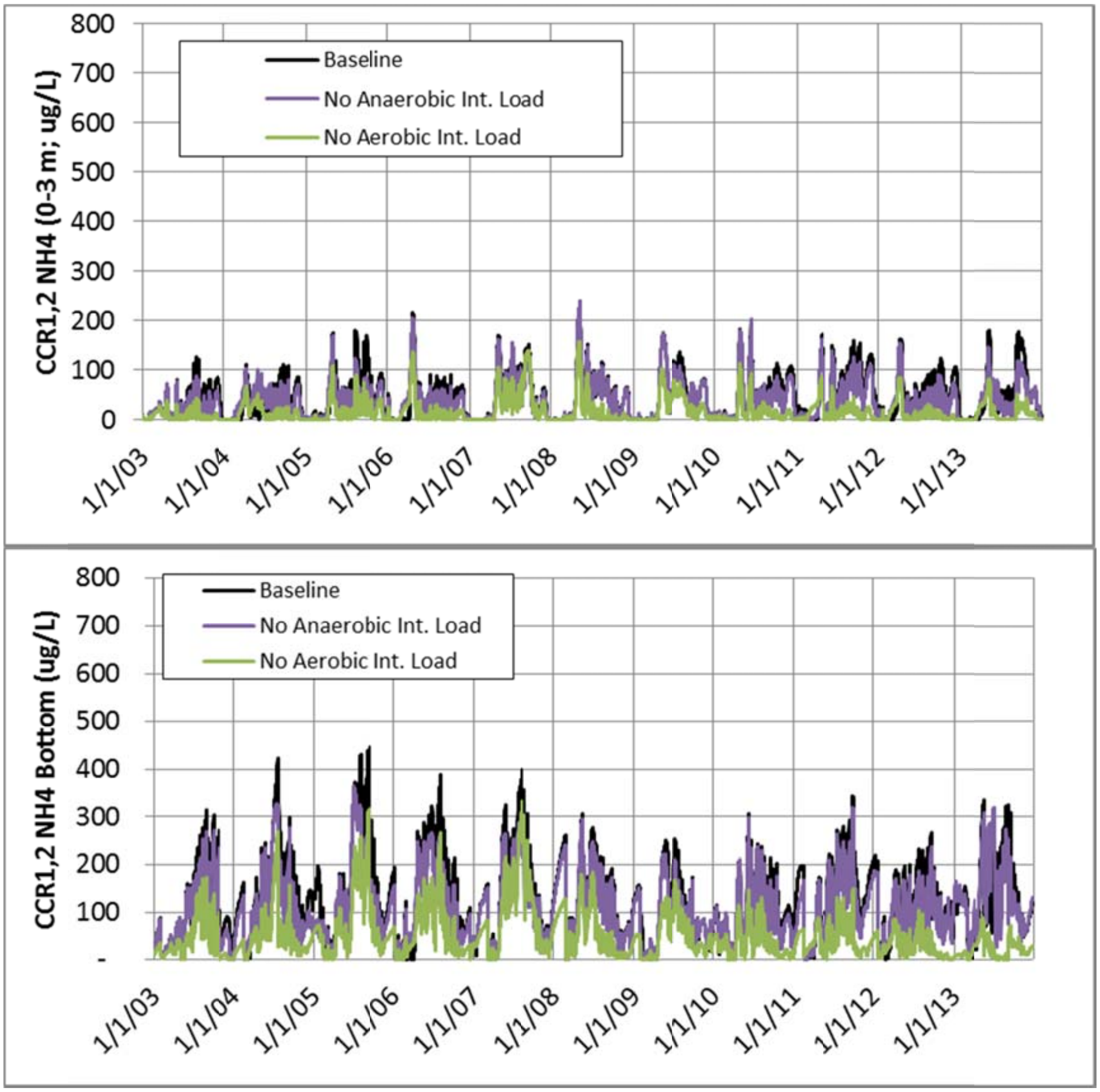

Figure 72. Simulated Ammonia at the Top and Bottom of the Reservoir (CCR2), Aerobic and Anaerobic Internal Loading and Baseline Simulations, 2003-2013

In response to the lower nutrient concentrations at the surface of the reservoir for these internal loading simulations, reduced algal growth was also simulated. Summertime algal concentrations decreased significantly in each run, but the greater effect was simulated for the aerobic portion of the internal loading (Figure 73). This can be understood for this reservoir with significant autochthonous production of organic matter, intermittent and weak stratification, and resulting variable DO concentrations at the bottom through summer months.

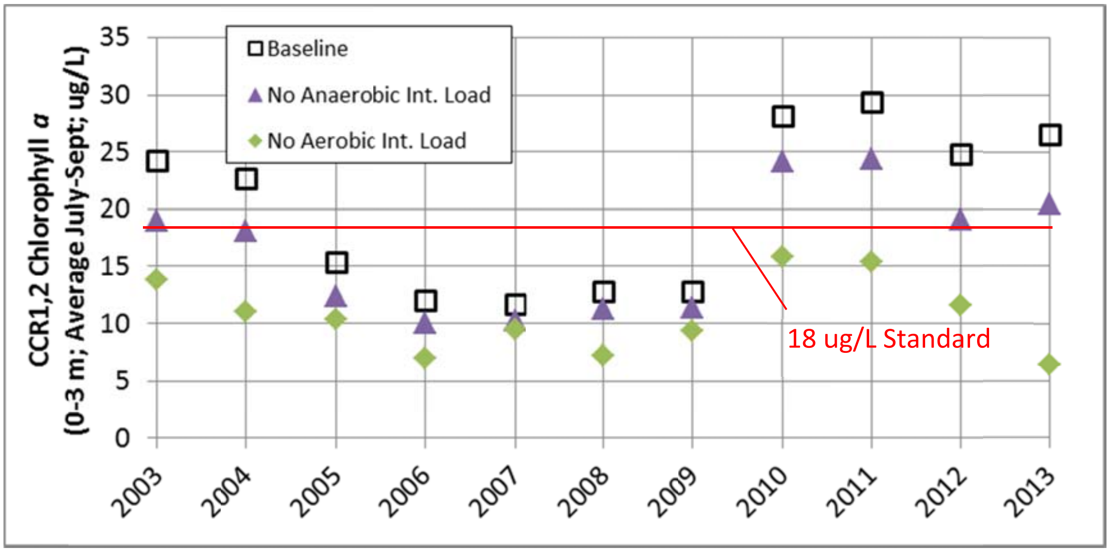

Figure 73. Simulated Average Summertime (July-September) Chlorophyll a (CCR2), Aerobic and Anaerobic Internal Loading and Baseline Simulations, 2003-2013

In the absence of the anaerobic internal loading, the simulation indicates that the reservoir in recent years would still exceed the site-specific standard (Figure 73). This finding is significant because the destratification system is designed to address anaerobic loading. This finding agrees with the results of the 10x Aerator Mixing simulation discussed in Section 6.2.1 and points to limits of potential effectiveness of the aerators relative to the site-specific standard for the given inflow water quality.

7 Recommendations

Through the model development and calibration process, recommendations were generated regarding monitoring and noting possible areas for future refinements of the model.

7.1 Monitoring Recommendations

Overall, the existing water-quality dataset is an excellent record of conditions in the reservoir and was critical to successful model development. The following recommendations are offered to support future modeling and ongoing development of the conceptual understanding.

- **Continuous In-Reservoir DO Measurement** – It is strongly recommended that the Authority install continuous DO probes at 1 m below the surface and at 0.5 m above the bottom of the reservoir at CCR2. Similar monitoring at CCR3 could also be helpful; however, continuous DO data at CCR3 is considered less critical.

- **Review of DO Data Collection Procedures** – In response to some uncertainty and questions about the observed DO dataset, it is recommended that calibration and measurement protocols be reassessed and documented in greater detail. Additionally, periodic duplicate observations with a second probe may be appropriate.

- **Meteorological Data** – Given the major effects of weather conditions on the water-quality response of Cherry Creek Reservoir, is it is recommended that the Authority coordinate with Colorado Parks and Wildlife (CPW) to check and improve data collection and data management protocols for the CPW met station located next to the reservoir. Alternatively, a new meteorological station could be installed at the dam, monitoring at a minimum, air temperature, solar radiation, wind speed, and wind direction.

- **Inflow Algae** – Cherry Creek and Cottonwood Creek include areas of briefly impounded water that exhibit a green color in some aerial photographs at times. Therefore, as discussed in Boyer et al., 2014a, sampling of algae at CC-10 and CT-2 is recommended to improve the understanding of algal loading from the tributaries. This should include both storm and non-storm-event sampling. Results could be reviewed each year to determine whether or not to continue this sampling.

- **Critical Sampling to Continue** – Continued collection of thermistor data and algal species and biovolume data is critical. Additionally, more recently added data collection of organic carbon and zooplankton are also expected to be valuable to help refine areas of uncertainty in the model.

- **Possible Sampling to Discontinue** – The D-series profile transect data was not critical for modeling. Given the dynamic nature of the system, the thermistor data are much more useful. Additionally, for the purposes of modeling, location CCR1 data collection could be discontinued.

7.2 Future Possible Modeling Refinements

The current calibrated Cherry Creek Reservoir Model is a powerful tool for the Authority to help evaluate existing management activities and possible future management work. Numerous insights were gained through the development of the model and it will be very useful for evaluating management scenarios. Like all models, this tool can continue to be improved and refined. Through model development and testing, areas for potential future refinement were identified and are documented here:

- **Luxury Uptake** – Given the high concentrations of PO4 observed through summer months at the top of the reservoir, it is possible that algae uptake PO4 in greater proportions at time (i.e., luxury uptake). The model does not currently simulate luxury uptake; however, recoding for this effect may be helpful for future Cherry Creek modeling refinements to further improve simulation of nutrient concentrations.

- **Nitrogen Fixation -** Nitrogen fixation is expected to be occurring in the reservoir. The model simulates nitrogen fixation; however, the model only allows nitrogen fixers to obtain nitrogen from the atmosphere. In reality, nitrogen-fixing algae revert to direct ammonia / nitrate uptake when these forms of nitrogen become available in the water column again (Bergman et al., 1999, pg. 4). Recoding nitrogen fixation as a future refinement to the Cherry Creek model could further improve the simulation of inorganic nitrogen and algae.

- **Sediment Diagenesis** – A version of CE-QUAL-W2 including sediment diagenesis has been under development. On June 22, 2015, Dr. Wells provided Hydros with a copy of a recently tested and refined version of the software and documentation. Unfortunately, there was not time under this effort to apply the model to that version. However, it may be a useful step for the future. This tool could help further refine the understanding of the sediment dynamics and better simulate time-varying aspects of these processes, such as burial rates.

- **Zooplankton** – Due to lack of zooplankton data from 2003-2010, there is remaining uncertainty in the modeling about the potentially time-varying effects of zooplankton on algal growth. Ongoing collection and review of zooplankton data should be conducted to facilitate critical review and modification of assumptions made in the current calibrated model. Additionally, the model could be modified in the future to simulate more consistent zooplankton settings for the entire period. This might require a greater number of algal groups to better reflect the growth of more and less-edible species. Such a refinement would likely require some recalibration and recoding for the obligate nitrogen fixation effect.

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