APPENDIX A. NUMERIC ENDPOINT DEVELOPMENT FOR POTABLE WATER USE

The development of a TMDL requires a scientifically defensible numeric endpoint which will ensure that the best uses of the water body are met. For the purpose of TMDL development in this watershed, a link between phosphorus concentrations and protection of the best use of the water body as a source of drinking water must be established. New York State's current guidance value for phosphorus is 20 μg/L (NYSDEC 1993) but was derived to protect primary and secondary contact recreational uses from impairment due to aesthetic effects. The current guidance value was not specifically derived to protect the drinking water use of water bodies such as Bear Lake. The link is best made through a site-specific interpretation of New York State's existing narrative ambient water quality standard for phosphorus (6NYCRR 703.2): "none in amounts that will result in growths of algae, weeds and slimes that will impair the waters for their best usages" (New York State, 2008), because an appropriate numeric translator for drinking water use has not been adopted.

In 2000, NYSDEC incorporated such a site-specific interpretation of the narrative criterion protective of drinking water use into TMDLs for the New York City Reservoirs (NYSDEC 2000). The USEPA, NYSDEC and the New York City Department of Environmental Protection (NYCDEP) worked toward the development of water supply-based phosphorus criteria for the New York City Reservoir Watershed, as part of the Phase II TMDL process. A weight-of-evidence approach utilized all available NYC reservoir-specific data to develop a relationship between phosphorus and chlorophyll-*a* levels, and a selected set of water quality variables which have been demonstrated to negatively affect the water quality of the drinking water supplied by the reservoirs in the watershed. Five water quality variables that are important concerns to water supply and are associated with excessive nutrient loading and reservoir water quality were selected, including trihalomethane (THM) precursor concentrations for certain reservoirs (Stepczyk 1998, NYCDEP 1999). Using the weight-of-evidence approach, the EPA-approved TMDL used a site-specific phosphorus guidance value of 15 μg/L as the ambient phosphorus level to protect NYC source water reservoirs used directly for public water supply.

Eutrophication-related water quality impairments adversely affect a broad spectrum of water uses, including water supply and recreation, and also adversely affect aquatic life. Concerns about cultural eutrophication (human induced enhancement of primary productivity) are not unique to New York, and the issue is widely recognized as a significant water quality concern at the national and international levels. These concerns lead the USEPA to initiate a National Nutrient Strategy in 1998 with the goal of assisting all states in the development of numeric nutrient criteria (USEPA, 1998).

To further the process of developing numeric nutrient criteria protective of potable water use, the NYSDEC, in collaboration with investigators from the New York State Department of Health (NYSDOH), Upstate Freshwater Institute (UFI), State University of New York College of Environmental Science and Forestry (SUNY-ESF), and Morgan State University, conducted a study to investigate the relationship between nutrient-related indices and certain human health related indices. The study was funded by the USEPA as part of that agency's National Nutrient Criteria Strategy (USEPA, 1998). The study involved the monthly collection of paired water column samples from 21 lakes and reservoirs during the growing season (May to October, 2004 and/or 2007). The study systems were distributed throughout New York State, and spanned a relatively broad range of trophic conditions ranging from oligotrophic systems (low primary productivity) to eutrophic systems (high primary productivity).

NYSDEC has developed a draft Fact Sheet based on that research for Ambient Water Quality Values for ponded sources of potable waters in New York State, (NYSDEC, 2010) which has undergone EPA and peer-review, but is still in development. That research, which is further described in a peer-review journal (Callinan 2013), is used as the basis for the site-specific phosphorus targets in this Bear Lake TMDL, given the limited data available for those sources.

USEPA recently issued guiding principles "to offer clarity to states about an optional approach for developing a numeric nutrient criterion that integrates causal (nitrogen and phosphorus) and response parameters into one water quality standard (WQS). …These guiding principles apply when states wish to rely on response parameters to indicate that a designated use is protected. …A criterion must protect the designated use of the water, and states should clearly identify the use(s) they are seeking to protect. Where a criterion is intended to protect multiple designated uses, states must ensure that it protects the most sensitive one (40 CFR 131.11(a))…. Documentation supporting the criterion should identify all applicable nutrient pathways, addressing all potential direct and indirect effects (e.g., as identified in a conceptual model that outlines the effects of nutrient pollution)" (USEPA, 2013).

A.1 Conceptual Model

Nutrient enrichment of lakes and reservoirs used for potable water supply (PWS) can cause adverse effects, ranging from operational problems to increases in health related risks such as disinfection by-products (DBPs), cyanotoxins, and arsenic.

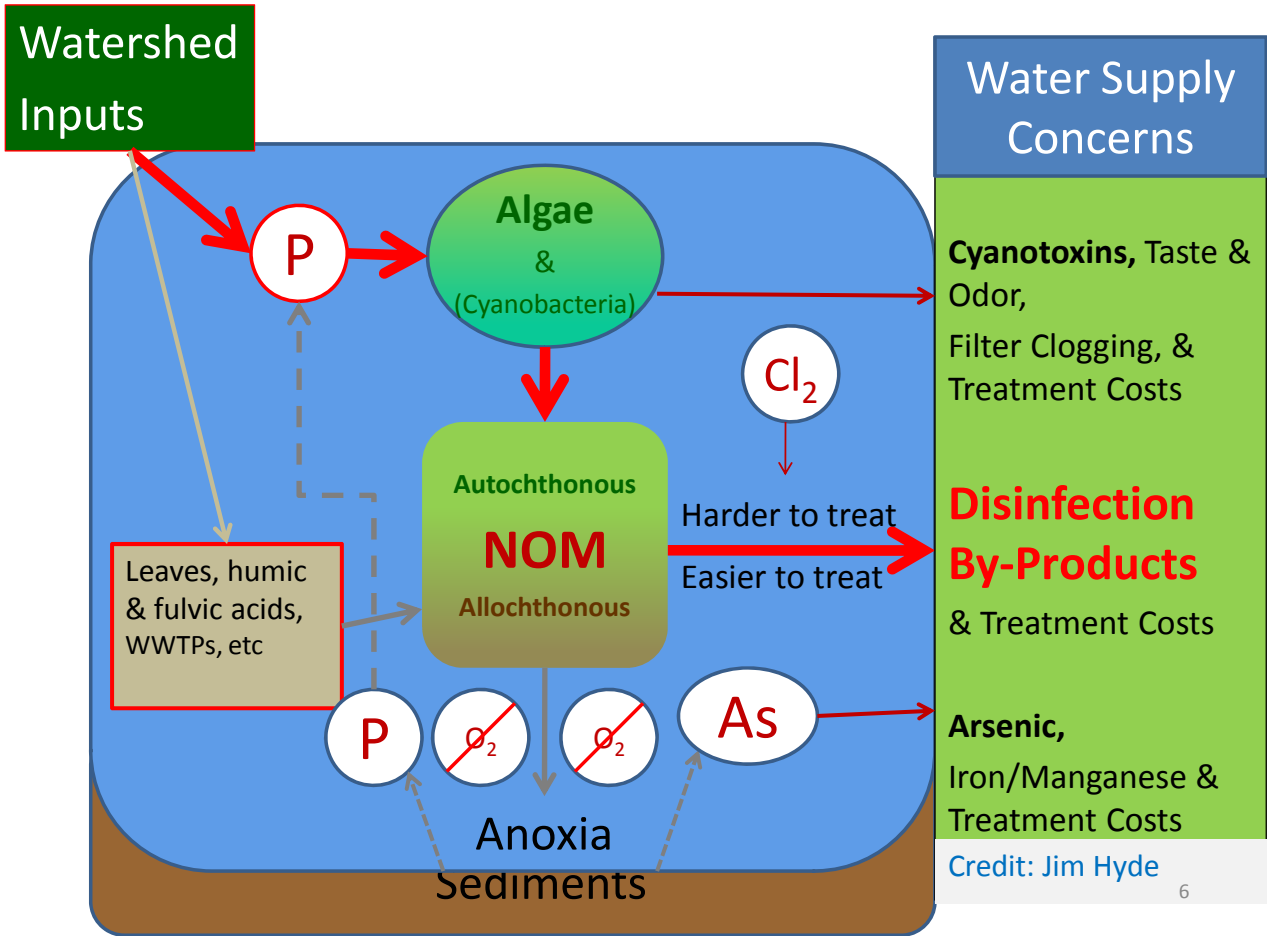
The linkages between eutrophication and PWS concerns are shown in Figure 11. As illustrated by the red arrows in the figure, the primary route of concern is: (1) nutrient (P) enrichment leads to (2) increases in algae (measured as chlorophyll-*a*), which results in (3) increases in natural organic matter (NOM), which (4) combines with chlorination (Cl_2) to form disinfection by-products.

Additional phosphorus inputs may further accelerate eutrophication, which may lead to oxygen depletion, which may cause reductive release of sediment-bound arsenic and phosphorus, which can provide a positive feedback to further nutrient enrichment, and production of cyanotoxins. Although an increase in arsenic levels and production of cyanotoxins are health concerns for PWS, the NYSDEC study found that formation of DBPs was likely to be the most sensitive endpoint for developing a phosphorus criterion for PWS, and it is the relationship to formation of DBPs that is the focus of the site-specific phosphorus target in this TMDL.

Disinfection By-Products

Disinfection by-products (DBPs) are a group of compounds formed as a result of chemical reactions between natural organic matter (NOM) and certain disinfection agents (e.g., chlorine). The two major classes of DBPs are trihalomethanes (THMs) and haloacetic acids (HAAs). Several of these compounds (e.g., bromodichloromethane, trichloroacetic acid) are considered to be carcinogenic (ATSDR 1997, USEPA 2006). There is also some evidence linking DBPs to adverse reproductive effects (USEPA, 2006).

Figure 11. Conceptual model of the linkages between eutrophication and potable water supply concerns



The link between nutrient enrichment and increased production of DBPs occurs because in many temperate freshwater systems, phosphorus acts as the limiting growth factor for primary production. This increase in primary production leads to: (a) an increase in the level of NOM, and (b) a change in the nature of NOM within the system, which heightens the risk for DBP production when the water is subjected to disinfection. The NYSDEC study discussed below was limited to total THMs (TTHMs).The USEPA (2006) defines TTHMS as the sum of four chlorinated compounds: chloroform, bromodichloromethane, dibromochloromethane, and bromoform.

Research on DBPs initially focused on the allochthonous (watershed; e.g., leaves and wastewater) precursor pool; however, subsequent studies also identified the autochthonous (in-lake; e.g., algae) precursor pool as important (Figure 11). There are important distinctions between allochthonous and autochthonous precursors that are relevant to PWS management. For example, autochthonous precursors are both more amenable to mitigation through nutrient management and more difficult to remove through water treatment. Furthermore, autochthonous precursors may produce greater quantities of unregulated DBPs.

A.2 Derivation of Site-specific Ambient Water Quality Values (Criteria)

The approach taken in the NYSDEC study to derive appropriate site-specific ambient water quality values (AWQVs) is based upon findings from a NYSDEC's Disinfection By-Product/Algal Toxins Project (DBP-AT Project), as well as pertinent material from other independent investigations (both peer review literature and technical reports).

The toxicological basis for the criteria in the DBP-AT Project was based upon previous drinking-water related toxicological findings for disinfection by-products (specifically total trihalomethanes) derived to meet the current maximum contaminant levels (MCLs) as summarized and presented in the Code of Federal Regulations (40 CFR January 4, 2006).

Several assumptions were made in the derivation of nutrient thresholds THMs.

1. The target nutrient thresholds are designed to attain the current maximum contaminant level (MCL) for TTHMs, presently set at 80 μg/l per the USEPA Stage 2 Disinfectants and Disinfection Byproducts Rule (USEPA 2006).
2. The applicable toxicological evidence as presented in the USEPA Stage 2 Rule in support of the current MCL is adequate for the protection of human health. The current MCL for TTHMs is deemed the appropriate target value given that the criteria are directed toward protection of public water supply use which, in all instances for ponded surface waters, involves disinfection.
3. The nutrient thresholds defined for THMs are sufficient to protect for HAAs. Some studies suggest that algae are equally important in the generation of HAAs and TTHMs (Nguyen, et al., 2005), thus, it is assumed that limiting algae production will have comparable effects of both major classes of DBPs.

The NYSDEC DBP-AT Study involved the collection of paired ambient water samples that were analyzed for THM Formation Potential (THMFP) and nutrient-related indices. THMFP is commonly used in research investigations to normalize results for the purpose of system comparisons.

The study developed relationships for each step in the conceptual model. For the first step, the regression relationship between mean chlorophyll-*a* and TP indicates that approximately 78% of the variability in phytoplankton biomass (based on chlorophyll-*a*) is accounted for by changes in TP, which supports the idea that phytoplankton biomass is controlled by phosphorus during the growing season. Study findings also offer several lines of evidence in support of the hypothesis that increased primary productivity (or cultural eutrophication) leads to an increase in the generation of THMFP:

- The relationship between mean DOC (a measure of NOM) and chlorophyll-*a* indicates a trend of increasing DOC concentrations with increasing chlorophyll-*a*.
- THMFP levels are substantially influenced by algal biomass. (The importance of the autochthonous precursor pool is supported by observed increases in THMFP concentrations with increases in trophic state, observed correlations between mean concentrations of THMFP and trophic indexes, and observed increases in THMFP concentrations during the growing season in most study systems).

- The relationship between mean THMFP and DOC, shows that approximately 80% of the variation in mean THMFP is attributable to mean DOC.

The observed relationships between THMFP and trophic indexes in the NYSDEC DBP-AT Project provide a sound basis for the derivation of nutrient-related thresholds protective of PWS. These findings are also consistent with a significant body of literature demonstrating a qualitative relationship between nutrient enrichment and the risk of increased THMFP production (Palmstrom, et al 1988, Wardlaw, et al. 1991, Cooke and Kennedy 2001) and showed similar quantitative relationships to research by Arruda and Fromm (1989) and the Colorado Department of Public Health and Environment (2011).

Building upon the relationships discussed above, the next step in the criteria development process is to identify potential AWQVs for the nutrient indices that are protective of potable waters with respect to DBPs. This required associating the measured THMFP to the TTHM drinking water standard. THMFP represents something of a "worst case" scenario in that the analytical protocol is designed to fully exploit the reaction between the available natural organic matter (NOM) and the disinfectant agent. In contrast, water treatment plant (WTP) operators attempt to minimize the generation of TTHMs, and other DBPs, while providing adequate disinfection.

This THMFP to TTHM translation, involved fitting observed THMFP data to a TTHM simulation model, and running the model using representative treatment/distribution system conditions coupled with the TTHM maximum contaminant level (MCL) of 80 µg/L. Using the relationships among chlorophyll-*a*, DOC and THMs established in the NYSDEC DBP-AT Project, a threshold of chlorophyll-*a* = 4.0 µg/L was derived, where values apply as growing season (May-October) means within the photic zone of the lake or reservoir.

AWQVs (Criteria)

The draft fact sheet applied the NYSDEC DBP-AT Project derived threshold of chlorophyll-*a* = 4.0 µg/L as an AWQV to water use Class *AA* waters, given that these systems are required to meet applicable drinking water standards following only disinfection.[1]

The draft fact sheet for ponded waters considered it appropriate to derive distinct AWQVs for different water use classes of ponded surface waters carrying best usage of source of potable water supply because of the differing level of expected treatment inherent in the specific use classes. Class *AA* will be subject to the more stringent AWQVs given that these waters are expected to meet applicable drinking water standards after only disinfection, whereas, ponded water supply source waters carrying water use Class *A* will be subject to a somewhat less stringent AWQV given that they are expected to meet applicable drinking water standards following "conventional" water treatment.[2]

---

[1] *Class AA: "This classification may be given to those waters that, if subjected to approved disinfection treatment, with additional treatment if necessary to remove naturally present impurities, meet or will meet New York State Department of Health drinking water standards and are or will be considered safe and satisfactory for drinking water purposes"*. (6 NYCRR Part 701).

[2] *Class A: "This classification may be given to those waters that, if subjected to approved treatment equal to coagulation, sedimentation, filtration and disinfection, with additional treatment if necessary to reduce naturally present impurities, meet or will meet New York State Department of Health drinking water standards and are or will be considered safe and satisfactory for drinking water purposes.*" (6 NYCRR Part 701)

Conventional water treatment processes (coagulation, sedimentation and, filtration) can reduce levels of DOC in raw source water, however, removal efficiency diminishes as trophic level increases. Thus, the draft fact sheet assumed a somewhat conservative DOC removal efficiency of 10% - note, this is a reduction in DOC, not in phosphorus or chlorophyll-*a*. Thus, using the relationships among chlorophyll-*a*, DOC and THMs established in the NYSDEC DBP-AT Project, the draft fact sheet proposed a chlorophyll-*a* concentration of 6.0 μg/L for Class *A* waters.

Although water use classes listed above include a caveat relating to "naturally present impurities", this was not deemed applicable for situations of cultural eutrophication, which, by definition are driven by anthropogenic-driven processes.

The NYSDEC findings compare well with other independent investigations. Arruda and Fromm (1989) investigated the relationship between trophic indexes and THMs in 180 Kansas lakes and arrived at a recommended chlorophyll-*a* threshold of 5 μg/L to attain a TTHM limit of 100 μg/L (MCL in place at that time). Colorado (Colorado DPHE, 2011) conducted a study patterned on New York's study, although with enhancements including use of the Uniform Formation Conditions method (Summers 1996) that also targeted HAA formation and alternative methods of interpretation, and determined that a mean chlorophyll-*a* concentration of 5 μg/L would be an appropriate threshold for direct use public water supply reservoirs.

Based upon the above, for the purpose of this TMDL, a chlorophyll-*a* concentration of 6.0 μg/L has been identified as an appropriate endpoint protective of the water supply use of the Class *A* Bear Lake.

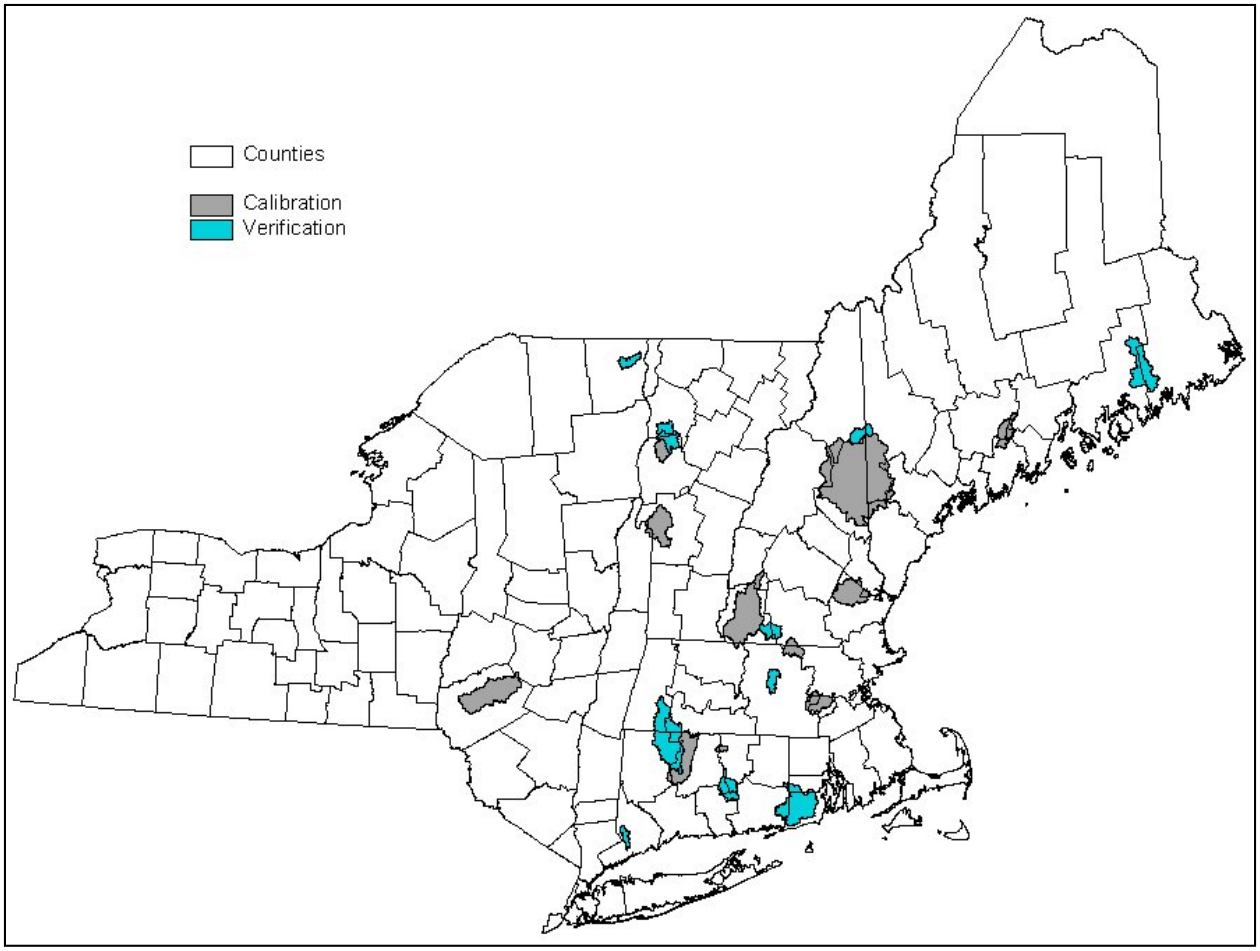
APPENDIX B. MAPSHED MODELING ANALYSIS

Northeast MAPSHED Model

As described in the following, the watershed model was originally calibrated for the northeast for the ArcView Generalized Watershed Loading Function (AVGWLF) model (Evans et al., 2007). AVGWLF utilized proprietary geographic information system (GIS) software. The model was later ported to an open source GIS software system and re-released as MAPSHED (Evans and Corradini, 2012). As the theoretical basis for the model remained unchanged the following accurately reflects the MAPSHED model development process.

The AVGWLF model was calibrated and validated for the northeast (Evans et al., 2007). AVGWLF requires that calibration watersheds have long-term flow and water quality data. For the northeast model, watershed simulations were performed for twenty-two (22) watersheds throughout New York and New England for the period 1997-2004 (Figure 12). Flow data were obtained directly from the water resource database maintained by the U.S. Geological Survey (USGS). Water quality data

Figure 12. Location of Calibration and Verification Watersheds for the Northeast AVGWLF Model



were obtained from the New York and New England State agencies. These data sets included in-stream concentrations of nitrogen, phosphorus, and sediment based on periodic sampling.

Initial model calibration was performed on half of the 22 watersheds for the period 1997-2004. During this step, adjustments were iteratively made in various model parameters until a "best fit" was achieved between simulated and observed stream flow, and sediment and nutrient loads. Based on the calibration results, revisions were made in various AVGWLF routines to alter the manner in which model input parameters were estimated. To check the reliability of these revised routines, follow-up verification runs were made on the remaining eleven watersheds for the same time period. Finally, statistical evaluations of the accuracy of flow and load predictions were made.

To derive historical nutrient loads, standard mass balance techniques were used. First, the in-stream nutrient concentration data and corresponding flow rate data were used to develop load (mass) versus flow relationships for each watershed for the period in which historical water quality data were obtained. Using the daily stream flow data obtained from USGS, daily nutrient loads for the 1997-2004 time period were subsequently computed for each watershed using the appropriate load versus flow relationship (i.e., "rating curves"). Loads computed in this fashion were used as the "observed" loads against which model-simulated loads were compared.

During this process, adjustments were made to various model input parameters for the purpose of obtaining a "best fit" between the observed and simulated data. With respect to stream flow, adjustments were made that increased or decreased the amount of the calculated evapotranspiration and/or "lag time" (i.e., groundwater recession rate) for sub-surface flow. With respect to nutrient loads, changes were made to the estimates for sub-surface nitrogen and phosphorus concentrations. In regard to both sediment and nutrients, adjustments were made to the estimate for the "C" factor for cropland in the USLE equation, as well as to the sediment "a" factor used to calculate sediment loss due to stream bank erosion. Finally, revisions were also made to the default retention coefficients used by AVGWLF for estimating sediment and nutrient retention in lakes and wetlands.

Based upon an evaluation of the changes made to the input files for each of the calibration watersheds, revisions were made to routines within AVGWLF to modify the way in which selected model parameters were automatically estimated. The AVGWLF software application was originally developed for use in Pennsylvania, and based on the calibration results, it appeared that certain routines were calculating values for some model parameters that were either too high or too low. Consequently, it was necessary to make modifications to various algorithms in AVGWLF to better reflect conditions in the Northeast. A summary of the algorithm changes made to AVGWLF is provided below.

- **ET**: A revision was made to increase the amount of evapotranspiration calculated automatically by AVGWLF by a factor of 1.54 (in the "Pennsylvania" version of AVGWLF, the adjustment factor used is 1.16). This has the effect of decreasing simulated stream flow.
- **GWR:** The default value for the groundwater recession rate was changed from 0.1 (as used in Pennsylvania) to 0.03. This has the effect of "flattening" the hydrograph within a given area.
- **GWN:** The algorithm used to estimate "groundwater" (sub-surface) nitrogen concentration was changed to calculate a lower value than provided by the "Pennsylvania" version.
- **Sediment "a" Factor:** The current algorithm was changed to reduce estimated stream bank-derived sediment by a factor of 90%. The streambank routine in AVGWLF was originally

developed using Pennsylvania data and was consistently producing sediment estimates that were too high based on the in-stream sample data for the calibration sites in the Northeast. While the exact reason for this is not known, it's likely that the glaciated terrain in the Northeast is less erodible than the highly erodible soils in Pennsylvania. Also, it is likely that the relative abundance of lakes, ponds and wetlands in the Northeast have an effect on flow velocities and sediment transport.

- **Lake/Wetland Retention Coefficients:** The default retention coefficients for sediment, nitrogen and phosphorus are set to 0.90, 0.12 and 0.25, respectively, and changed at the user's discretion.

To assess the correlation between observed and predicted values, two different statistical measures were utilized: 1) the Pearson product-moment correlation (R^2) coefficient and 2) the Nash-Sutcliffe coefficient. The R^2 value is a measure of the degree of linear association between two variables, and represents the amount of variability that is explained by another variable (in this case, the model-simulated values). Depending on the strength of the linear relationship, the R^2 can vary from 0 to 1, with 1 indicating a perfect fit between observed and predicted values. Like the R^2 measure, the Nash-Sutcliffe coefficient is an indicator of "goodness of fit," and has been recommended by the American Society of Civil Engineers for use in hydrological studies (ASCE, 1993). With this coefficient, values equal to 1 indicate a perfect fit between observed and predicted data, and values equal to 0 indicate that the model is predicting no better than using the average of the observed data. Therefore, any positive value above 0 suggests that the model has some utility, with higher values indicating better model performance. In practice, this coefficient tends to be lower than R^2 for the same data being evaluated.

Adjustments were made to the various input parameters for the purpose of obtaining a "best fit" between the observed and simulated data. One of the challenges in calibrating a model is to optimize the results across all model outputs (in the case of AVGWLF, stream flows, as well as sediment, nitrogen, and phosphorus loads). As with any watershed model like GWLF, it is possible to focus on a single output measure (e.g., sediment or nitrogen) in order to improve the fit between observed and simulated loads. Isolating on one model output, however, can sometimes lead to less acceptable results for other measures. Consequently, it is sometimes difficult to achieve very high correlations (e.g., R^2 above 0.90) across all model outputs. Given this limitation, it was felt that very good results were obtained for the calibration sites. In model calibration, initial emphasis is usually placed on getting the hydrology correct. Therefore, adjustments to flow-related model parameters are usually finalized prior to making adjustments to parameters specific to sediment and nutrient production. This typically results in better statistical fits between stream flows than the other model outputs.

For the monthly comparisons, mean R^2 values of 0.80, 0.48, 0.74, and 0.60 were obtained for the calibration watersheds for flow, sediment, nitrogen and phosphorus, respectively. When considering the inherent difficulty in achieving optimal results across all measures as discussed above (along with the potential sources of error), these results are quite good. The sediment load predictions were less satisfactory than those for the other outputs, and this is not entirely unexpected given that this constituent is usually more difficult to simulate than nitrogen or phosphorus. An improvement in sediment prediction could have been achieved by isolating on this particular output during the calibration process; but this would have resulted in poorer performance in estimating the nutrient loads for some of the watersheds. Phosphorus predictions were less accurate than those for nitrogen. This is not unusual given that a significant portion of the phosphorus load for a watershed is highly related to sediment transport processes. Nitrogen, on the other hand, is often linearly correlated to flow, which typically results in accurate predictions of nitrogen loads if stream flows are being accurately simulated.

As expected, the monthly Nash-Sutcliffe coefficients were somewhat lower due to the nature of this particular statistic. As described earlier, this statistic is used to iteratively compare simulated values against the mean of the observed values, and values above zero indicate that the model predictions are better than just using the mean of the observed data. In other words, any value above zero would indicate that the model has some utility beyond using the mean of historical data in estimating the flows or loads for any particular time period. As with R^2 values, higher Nash-Sutcliffe values reflect higher degrees of correlation than lower ones.

Improvements in model accuracy for the calibration sites were typically obtained when comparisons were made on a seasonal basis. This was expected since short-term variations in model output can oftentimes be reduced by accumulating the results over longer time periods. In particular, month-to-month discrepancies due to precipitation events that occur at the end of a month are often resolved by aggregating output in this manner (the same is usually true when going from daily output to weekly or monthly output). Similarly, further improvements were noted when comparisons were made on a mean annual basis. What these particular results imply is that AVGWLF, when calibrated, can provide very good estimates of mean annual sediment and nutrient loads.

Following the completion of the northeast AVGWLF model, there were a number of ideas on ways to improve model accuracy. One of the ideas relates to the basic assumption upon which the work undertaken in that project was based. This assumption is that a "regionalized" model can be developed that works equally well (without the need for resource-intensive calibration) across all watersheds within a large region in terms of producing reasonable estimates of sediment and nutrient loads for different time periods. Similar regional model calibrations were previously accomplished in earlier efforts undertaken in Pennsylvania (Evans et al., 2002) and later in southern Ontario (Watts et al., 2005). In both cases this task was fairly daunting given the size of the areas involved. In the northeast effort, this task was even more challenging given the fact that the geographic area covered by the northeast is about three times the size of Pennsylvania, and arguably is more diverse in terms of its physiographic and ecological composition.

As discussed, AVGWLF performed very well when calibrated for numerous watersheds throughout the region. The regionalized version of AVGWLF, however, performed less well for the verification watersheds for which additional adjustments were not made subsequent to the initial model runs. This decline in model performance may be a result of the regionally-adapted model algorithms not being rigorous enough to simulate spatially-varying landscape processes across such a vast geographic region at a consistently high degree of accuracy. It is likely that un-calibrated model performance can be enhanced by adapting the algorithms to reflect processes in smaller geographic regions such as those depicted in the physiographic province map in Figure 13.

Fine-tuning & Re-Calibrating the Northeast AVGWLF for New York State

For the TMDL development work undertaken in New York, the original northeast AVGWLF model was further refined by The Cadmus Group, Inc. and Dr. Barry Evans to reflect the physiographic regions that exist in New York. Using data from some of the original northeast model calibration and verification sites, as well as data for additional calibration sites in New York, three new versions of AVGWLF were created for use in developing TMDLs in New York State. Information on the fourteen (14) sites is summarized in Table 7. Two models were developed based on the following two physiographic regions: Eastern Great Lakes/Hudson Lowlands area and the Northeastern

Figure 13. Location of Physiographic Provinces in New York and New England

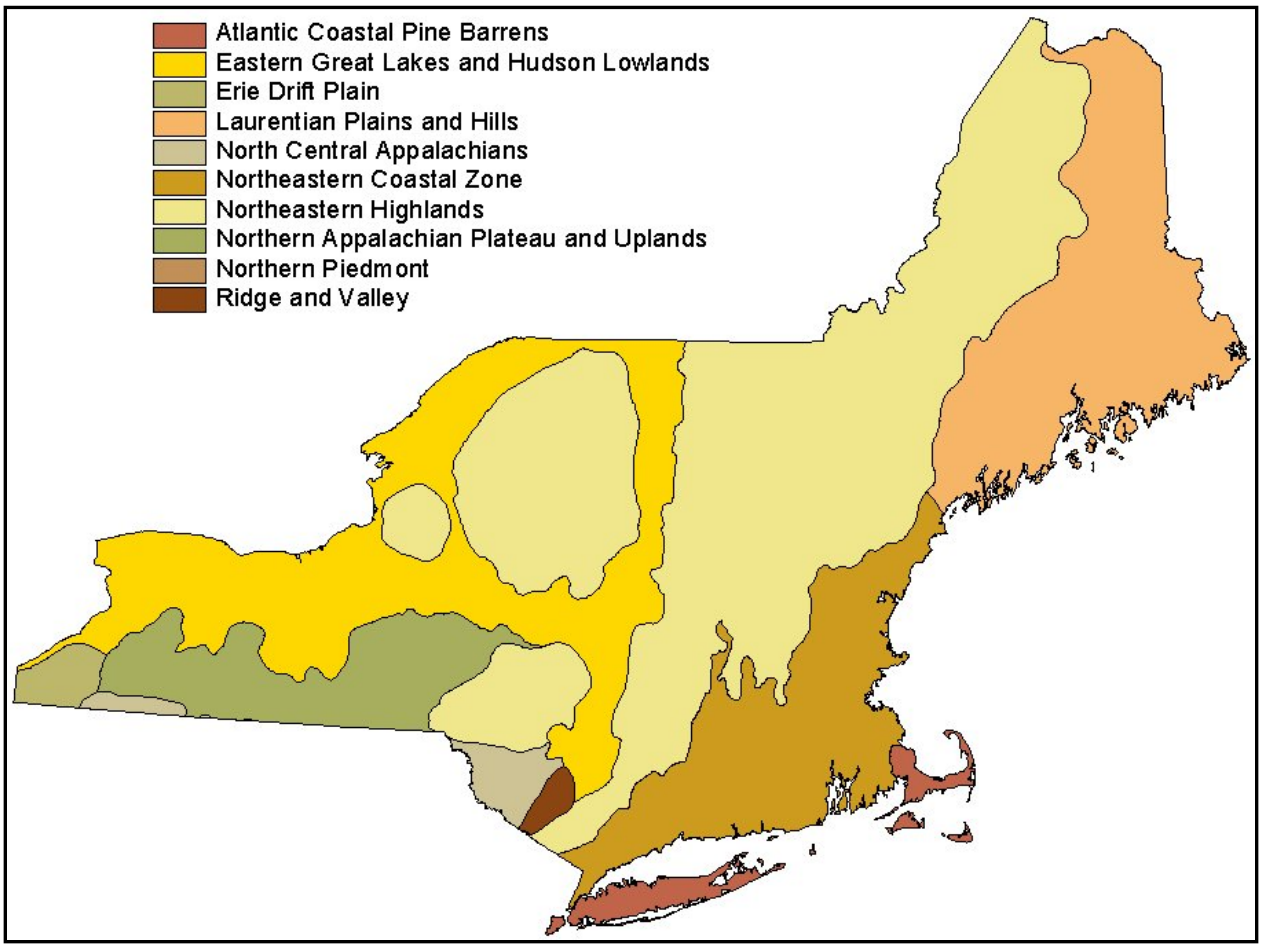


Table 7. AVGWLF Calibration Sites for use in the New York TMDL Assessments

Site	Location	Physiographic Region
Owasco Lake	NY	Eastern Great Lakes/Hudson Lowlands
West Branch	NY	Northeastern Highlands
Little Chazy River	NY	Eastern Great Lakes/Hudson Lowlands
Little Otter Creek	VT	Eastern Great Lakes/Hudson Lowlands
Poultney River	VT/NY	Eastern Great Lakes/Hudson Lowlands & Northeastern Highlands
Farmington River	CT	Northeastern Highlands
Saco River	ME/NH	Northeastern Highlands
Squannacook River	MA	Northeastern Highlands
Ashuelot River	NH	Northeastern Highlands
Laplatte River	VT	Eastern Great Lakes/Hudson Lowlands
Wild River	ME	Northeastern Highlands
Salmon River	CT	Northeastern Coastal Zone
Norwalk River	CT	Northeastern Coastal Zone
Lewis Creek	VT	Eastern Great Lakes/Hudson Lowlands

Highlands area. The model was calibrated for each of these regions to better reflect local conditions, as well as ecological and hydrologic processes. In addition to developing the above mentioned physiographic-based model calibrations, a third model calibration was also developed. This model calibration represents a composite of the two physiographic regions and is suitable for use in other areas of upstate New York.

Set-up of the "New York State" MAPSHED Model

Using data for the time period 1982-2012, the calibrated MAPSHED model was used to estimate phosphorus loading to the lake. Table 8 provides the sources of data used for the MAPSHED modeling analysis. The various data preparation steps taken prior to running the final calibrated MAPSHED Model for New York are discussed below the table.

Land Use

The 2011 NLCD land use coverage was obtained, recoded, and formatted specifically for use in MAPSHED. Following the public meeting a map was provided to NYSDEC upon which the land cover for a number of individual agricultural fields had been changed based upon local knowledge of the watershed. A number of fields operated by Amish farmers were also identified. These notes were used to manually change the land use codes for those fields in the GIS dataset.

Phosphorus retention in wetlands and open waters in the basin can be accounted for in MAPSHED. MAPSHED recommends the following coefficients for wetlands and pond retention in the northeast: nitrogen (0.12), phosphorus (0.25), and sediment (0.90). Wetland retention coefficients for large, naturally occurring wetlands vary greatly in the available literature. Depending on the type, size and quantity of wetland observed, the overall impact of the wetland retention routine on the original watershed loading estimates, and local information regarding the impact of wetlands on watershed loads, wetland retention coefficients defaults were adjusted accordingly. The percentage of the drainage basin area that drains through a wetland area was calculated and used in conjunction with nutrient retention coefficients in MAPSHED. To determine the percent wetland area, the total basin land use area was derived using ArcView. Of this total basin area, the area that drains through emergent and woody wetlands were delineated to yield an estimate of total watershed area draining through wetland areas. If a basin displays large areas of surface water (ponds) aside from the water body being modeled, then this open water area is calculated by subtracting the water body area from the total surface water area.

On-site Wastewater Treatment Systems ("septic tanks")

GWLF simulates nutrient loads from septic systems as a function of the percentage of the unsewered population served by normally functioning vs. three types of malfunctioning systems: ponded, short-circuited, and direct discharge (Haith et al., 1992).

- **Normal Systems** are septic systems whose construction and operation conforms to recommended procedures, such as those suggested by the EPA design manual for on-site wastewater disposal systems. Effluent from normal systems infiltrates into the soil and enters the shallow saturated zone. Phosphates in the effluent are adsorbed and retained by the soil and hence normal systems provide no phosphorus loads to nearby waters.

Table 8. Information Sources for AVGWLF Model Parameterization

WEATHER.DAT file	
Data	**Source or Value**
Temperature, precipitation	Historical weather data from Fredonia, NY, Dunkirk, NY and Sinclairville, NY from the National Climate Data Center
TRANSPORT.DAT file	
Data	**Source or Value**
Basin size	GIS/derived from basin boundaries
Land use/cover distribution	GIS/derived from land use/cover map
Curve numbers by source area	GIS/derived from land cover and soil maps
USLE (KLSCP) factors by source area	GIS/derived from soil, DEM, & land cover
ET cover coefficients	GIS/derived from land cover
Erosivity coefficients	GIS/ derived from physiographic map
Daylight hrs. by month	Computed automatically for state
Growing season months	Input by user
Initial saturated storage	Default value of 10 cm
Initial unsaturated storage	Default value of 0 cm
Recession coefficient	Default value of 0.06
Seepage coefficient	Default value of 0
Initial snow amount (cm water)	Default value of 0
Sediment delivery ratio	GIS/based on basin size
Soil water (available water capacity)	GIS/derived from soil map
NUTRIENT.DAT file	
Data	**Source or Value**
Dissolved N in runoff by land cover type	Default values/adjusted using GWLF Manual
Dissolved P in runoff by land cover type	Default values/adjusted using GWLF Manual
N/P concentrations in manure runoff	Default values/adjusted using AEU density
N/P buildup in urban areas	Default values (from GWLF Manual)
N and P point source loads	Derived from SPDES point coverage
Background N/P concentrations in GW	Derived from new background N map
Background P concentrations in soil	Derived from soil P loading map/adjusted using GWLF Manual
Background N concentrations in soil	Based on map in GWLF Manual
Months of manure spreading	Input by user
Population on septic systems	Estimated based upon data provided during public meeting
Per capita septic system loads (N/P)	Default values

- **Short-Circuited Systems** are located close enough to surface water (~15 meters) so that negligible adsorption of phosphorus takes place. The only nutrient removal mechanism is plant uptake. Therefore, these systems are always contributing to nearby waters.
- **Ponded Systems** exhibit hydraulic malfunctioning of the tank's absorption field and resulting surfacing of the effluent. Unless the surfaced effluent freezes, ponding systems deliver their nutrient loads to surface waters in the same month that they are generated through overland flow. If the temperature is below freezing, the surfacing is assumed to freeze in a thin layer at the ground surface. The accumulated frozen effluent melts when the snowpack disappears and the temperature is above freezing.
- **Direct Discharge Systems** illegally discharge septic tank effluent directly into surface waters.

GWLF requires an estimation of population served by septic systems to generate septic system phosphorus loadings. In reviewing the orthoimagery for the lake, it became apparent that septic system estimates from the 1990 census were not reflective of actual population in close proximity to the shore. Shoreline dwellings immediately surrounding the lake account for a substantial portion of the nutrient loading to the lake. Therefore, the estimated number of septic systems in the drainage basin was refined using a combination of 1990 and 2000 census data and GIS analysis of orthoimagery to account for the proximity of septic systems immediately surrounding the lake. If available, local information about the number of houses within 250 feet of the lakes was obtained and applied. Great attention was given to estimating septic systems within 250 feet of the lake (those most likely to have an impact on the lake). To convert the estimated number of septic systems to population served, an average household size of 2.57 people per dwelling was used based on the circa 2010 USCB census estimate for number of persons per household in New York State.

GWLF also requires an estimate of the number of normal and malfunctioning septic systems. This information was not readily available for the lake. Therefore, several assumptions were made to categorize the systems according to their performance. These assumptions are based on data from local and national studies (Day, 2001; USEPA, 2002) in combination with best professional judgment. To account for seasonal variations in population, data from the 2000 census were used to estimate the percentage of seasonal homes for the town(s) surrounding the lake. The failure rate for septic systems closer to the lake (i.e., within 250 feet) were adjusted to account for increased loads due to greater occupancy during the summer months. If available, local information about seasonal occupancy was obtained and applied. For the purposes of this analysis, seasonal homes are considered those occupied only during the month of June, July, and August.

Local information provided during the public meeting was used to refine the initial analysis. This information indicated 11 homes within 50 feet of the Bear Lake shoreline, only one of which is occupied for the entire year. An additional 58 homes are located 50 to 250 feet from the lake shoreline, of which 12 are occupied for the entire year. Finally, there are two seasonal campgrounds within 250 feet of the shoreline which have a combined 45 campsites, all of which are served by septic systems. For these campgrounds an additional correction factor of 0.29 was applied to the populations based upon estimated occupancy for 2 out of 7 days each week. Population estimates were based upon an assume occupancy rate of 2.57 people per dwelling as before.

Groundwater Phosphorus

Phosphorus concentrations in groundwater discharge are derived by MAPSHED. Watersheds with a high percentage of forested land will have low groundwater phosphorus concentrations while watersheds with a high percentage of agricultural land will have high concentrations. The GWLF manual provides estimated groundwater phosphorus concentrations according to land use for the eastern United States. Completely forested watersheds have values of 0.006 mg/L. Primarily agricultural watersheds have values of 0.104 mg/L. Intermediate values are also reported. The MAPSHED-generated groundwater phosphorus concentration was evaluated to ensure groundwater phosphorus values reasonably reflect the actual land use composition of the drainage basin and modifications were made if deemed necessary.

Point Sources

If permitted point sources exist in the drainage basin, their location was identified and verified by NYS DEC and an estimated monthly total phosphorus load and flow was determined using either actual reported data (e.g., from discharge monitoring reports) or estimated based on expected discharge/flow for the facility type. No point sources were identified within the Bear Lake basin.

Concentrated Animal Feeding Operations (CAFOs)

A state-wide Concentrated Animal Feeding Operation (CAFO) shapefile was provided by NYS DEC. CAFOs are categorized as either large or medium. The CAFO point can represent either the centroid of the farm or the entrance of the farm, therefore the CAFO point is more of a general gauge as to where further information should be obtained regarding permitted information for the CAFO. If a CAFO point is located in or around a basin, orthos and permit data were evaluated to determine the part of the farm with the highest potential contribution of nutrient load. In ArcView, the CAFO shapefile was positioned over the basin and clipped with a 2.5 mile buffer to preserve those CAFOS that may have associated cropland in the basin. If a CAFO point is found to be located within the boundaries of the drainage basin, every effort was made to obtain permit information regarding nutrient management or other best management practices (BMPs) that may be in place within the property boundary of a given CAFO. These data can be used to update the nutrient file in AVGWLF and ultimately account for agricultural BMPs that may currently be in place in the drainage basin. No CAFOs were identified within the Bear Lake basin.

Municipal Separate Storm Sewer Systems (MS4s)

Stormwater runoff within Phase II permitted Municipal Separate Storm Sewer Systems (MS4s) is considered a point source of pollutants. Stormwater runoff outside of the MS4 is non-permitted stormwater runoff and, therefore, considered nonpoint sources of pollutants. Permitted stormwater runoff is accounted for in the wasteload allocation of a TMDL, while non-permitted runoff is accounted for in the load allocation of a TMDL. DEC determined that there are no MS4s in the Bear Lake drainage basin.

Figure 14: Input Transport File

Urban Land	Area (ha)	%Imp	CNI	CNP
LD Mixed	54	0.15	92	74
MD Mixed	1	0.52	98	79
HD Mixed	0	0.0	0	0
LD Residential	0	0.0	0	0
MD Residential	0	0.0	0	0
HD Residential	0	0.0	0	0

Rural Land	Area (ha)	CN	K	LS	C	P
Hay/Pasture	306	75	0.24	0.511	0.03	0.45
Cropland	156	82	0.24	0.32	0.42	0.45
Forest	1524	73	0.24	0.629	0.002	0.45
Wetland	355	87	0.24	0.09	0.01	0.1
Disturbed	0	0	0.0	0.0	0.0	0.0
Turf/Golf	0	0	0.0	0.0	0.0	0.0
Open Land	218	87	0.24	0.494	0.04	0.45
Bare Rock	0	0	0.0	0.0	0.0	0.0
Sandy Areas	0	0	0.0	0.0	0.0	0.0
Unpaved Road	0	0	0.0	0.0	0.0	0.0

Month	Ket	Adjust %ET	Day Hours	Grow Seas	Eros Coef	Stream Extract	Ground Extract
Jan	0.85	1.0	9.0	0	0.08	0.0	0.0
Feb	1.04	1.0	10.0	0	0.08	0.0	0.0
Mar	1.18	1.0	12.0	0	0.08	0.0	0.0
Apr	1.29	1.0	13.0	0	0.26	0.0	0.0
May	1.54	1.0	15.0	1	0.26	0.0	0.0
Jun	1.72	1.0	15.0	1	0.26	0.0	0.0
Jul	1.85	1.0	15.0	1	0.26	0.0	0.0
Aug	1.95	1.0	14.0	1	0.26	0.0	0.0
Sep	2.02	1.0	12.0	1	0.26	0.0	0.0
Oct	2.07	1.0	11.0	1	0.08	0.0	0.0
Nov	1.93	1.0	9.0	0	0.08	0.0	0.0
Dec	1.84	1.0	9.0	0	0.08	0.0	0.0

Sediment A Factor	2.4511E-04
Sed A Adjustment	1.0
Avail Water Cap (cm)	0.63
Sed Delivery Ratio	0.164

Values 0 - 1

GW Recess Coeff	0.06
GW Seepage Coeff	0.0
% Tile Drained (Ag)	0.0

Save File | Export to JPEG | Close

Figure 15: Input Nutrient File

Dissolved Runoff Coefficients (mg/L)

Rural Runoff	Dissolved N	Dissolved P
Hay/Pasture	0.75	0.18667
Cropland	2.9	0.18667
Forest	0.19	0.01
Wetland	0.19	0.01
Disturbed	0.0	0.0
Turf/Golf	0.0	0.0
Open Land	0.5	0.01
Bare Rock	0.0	0
Sandy Areas	0	0
Unpaved Rd	0	0

	N	P	Sed
Groundwater (mg/L)	0.82	0.01	
Tile Drain (mg/L)	15.0	0.1	50.0
Soil Conc (mg/Kg)	2000.0	571.0	
% Bank Frac (0-1)	0.25	0.25	

Nitrogen and Phosphorus Loads from Point Sources and Septic Systems

Month	Point Source Loads/Discharge Kg N	Kg P	MGD	Septic System Populations Normal	Pond	Short Cir	Direct
Jan	0.0	0.0	0.0	20	3	10	0
Feb	0.0	0.0	0.0	20	3	10	0
Mar	0.0	0.0	0.0	20	3	10	0
Apr	0.0	0.0	0.0	20	3	10	0
May	0.0	0.0	0.0	20	3	10	0
Jun	0.0	0.0	0.0	118	18	74	0
Jul	0.0	0.0	0.0	118	18	74	0
Aug	0.0	0.0	0.0	118	18	74	0
Sep	0.0	0.0	0.0	20	3	10	0
Oct	0.0	0.0	0.0	20	3	10	0
Nov	0.0	0.0	0.0	20	3	10	0
Dec	0.0	0.0	0.0	20	3	10	0

Growing season uptake (g/d): N 1.6 P 0.4

Per Capita Tank Load (g/d): N 12 P 2.5

Urban Buildup (kg/Ha/day)

	Area (Ha)	Nitrogen Acc Imp	Acc Perv	Dis Fract	Phosphorus Acc Imp	Acc Perv	Dis Fract	TSS Acc Imp	Acc Perv
LD Mixed	54	0.095	0.015	0.33	0.0095	0.0021	0.4	2.8	0.8
MD Mixed	1	0.105	0.015	0.33	0.0105	0.0021	0.4	6.2	0.8
HD Mixed	0	0	0	0	0	0	0	0	0
LD Residential	0	0	0	0	0	0	0	0	0
MD Residential	0	0	0	0	0	0	0	0	0
HD Residential	0	0	0	0	0	0	0	0	0

Save File | Export to JPEG | Close

APPENDIX C. BATHTUB MODELING ANALYSIS

Model Overview

BATHTUB is a steady-state (Windows-based) water quality model developed by the U. S. Army Corps of Engineers (USACOE) Waterways Experimental Station. BATHTUB performs steady-state water and nutrient balance calculations for spatially segmented hydraulic networks in order to simulate eutrophication-related water quality conditions in lakes and reservoirs. BATHTUB's nutrient balance procedure assumes that the net accumulation of nutrients in a lake is the difference between nutrient loadings into the lake (from various sources) and the nutrients carried out through outflow and the losses of nutrients through whatever decay process occurs inside the lake. The net accumulation (of phosphorus) in the lake is calculated using the following equation:

Net accumulation = Inflow – Outflow – Decay

The pollutant dynamics in the lake are assumed to be at a steady state, therefore, the net accumulation of phosphorus in the lake equals zero. BATHTUB accounts for advective and diffusive transport, as well as nutrient sedimentation. BATHTUB predicts eutrophication-related water quality conditions (total phosphorus, total nitrogen, chlorophyll-*a*, transparency, and hypolimnetic oxygen depletion) using empirical relationships derived from assessments of reservoir data. Applications of BATHTUB are limited to steady-state evaluations of relations between nutrient loading, transparency and hydrology, and eutrophication responses. Short-term responses and effects related to structural modifications or responses to variables other than nutrients cannot be explicitly evaluated.

Input data requirements for BATHTUB include: physical characteristics of the watershed lake morphology (e.g., surface area, mean depth, length, mixed layer depth), flow and nutrient loading from various pollutant sources, precipitation (from nearby weather station) and phosphorus concentrations in precipitation (measured or estimated), and measured lake water quality data (e.g., total phosphorus concentrations).

The empirical models implemented in BATHTUB are mathematical generalizations about lake behavior. When applied to data from a particular lake, actual observed lake water quality data may differ from BATHTUB predictions by a factor of two or more. Such differences reflect data limitations (measurement or estimation errors in the average inflow and outflow concentrations) or the unique features of a particular lake (no two lakes are the same). BATHTUB's "calibration factor" provides model users with a method to calibrate the magnitude of predicted lake response. The model calibrated to current conditions (against measured data from the lakes) can be applied to predict changes in lake conditions likely to result from specific management scenarios, under the condition that the calibration factor remains constant for all prediction scenarios.

Model Set-up

Using descriptive information about Bear Lake and its surrounding drainage area, as well as output from MAPSHED, a BATHTUB model was set up for Bear Lake. Seasonal phosphorus loading to the lake was simulated using MAPSHED for the period 1982-2012. After initial model development, NYSDEC sampling data were used to assess the model's predictive capabilities and, if necessary, "fine tune" various input parameters and sub-model selections within BATHTUB during

a calibration process. Once calibrated, BATHTUB was used to derive the total phosphorus load reduction needed in order to achieve the TMDL target.

Sources of input data for BATHTUB include:

- Physical characteristics of the watershed and lake morphology (e.g., surface area, mean depth, length, mixed layer depth) - Obtained from CSLAP and bathymetric maps provided by NYS DEC or created by the Cadmus Group, Inc.
- Flow and nutrient loading from various pollutant sources - Obtained from MAPSHED output.
- Precipitation – Obtained from MAPSHED output (derived from NCDC data).
- Phosphorus concentrations in precipitation (measured or estimated), and measured lake water quality data (e.g., total phosphorus concentrations, chlorophyll-*a* and secchi disk depth) – Obtained from NYS DEC or USGS.

Tables 9 – 13 summarize the primary model inputs for Bear Lake, including the coefficient of variation (CV), which reflects uncertainty in the input value. Default model choices are utilized unless otherwise noted. Spatial variations (i.e., longitudinal dispersion) in phosphorus concentrations are not a factor in the development of the TMDL for Bear Lake. Therefore, division of the lake into multiple segments was not necessary for this modeling effort. Modeling the entire lake with one segment provides predictions of area-weighted mean concentrations, which are adequate to support management decisions. Water inflow and nutrient loads from the lake's drainage basin were treated as though they originated from one "tributary" (i.e., source) in BATHTUB and derived from MAPSHED.

BATHTUB is a steady state model, whose predictions represent concentrations averaged over a period of time. A key decision in the application of BATHTUB is the selection of the length of time over which water and mass balance calculations are modeled (the "averaging period"). The length of the appropriate averaging period for BATHTUB application depends upon what is called the nutrient residence time, which is the average length of time that phosphorus spends in the water column before settling or flushing out of the lake. Guidance for BATHTUB recommends that the averaging period used for the analysis be at least twice as large as nutrient residence time for the lake. The appropriate averaging period for water and mass balance calculations would be 1 year for lakes with relatively long nutrient residence times or seasonal (6 months) for lakes with relatively short nutrient residence times (e.g., on the order of 1 to 3 months). The turnover ratio can be used as a guide for selecting the appropriate averaging period. A seasonal averaging period (April/May through September) is usually appropriate if it results in a turnover ratio exceeding 2.0. An annual averaging period may be used otherwise. Other considerations (such as comparisons of observed and predicted nutrient levels) can also be used as a basis for selecting an appropriate averaging period, particularly if the turnover ratio is near 2.0.

Precipitation inputs were the observed total precipitation values from Fredonia, NY, Dunkirk, NY and Sinclairville, NY from the NCDC for 1982-2012 as output by MAPSHED. Evapotranspiration was derived from MAPSHED using daily weather data (1982-2012) and a cover factor dependent upon land use/cover type. The values selected for precipitation and change in lake storage have very little influence on model predictions. Atmospheric phosphorus loads were specified using data collected by USGS from a collection site at Mendon Ponds County Park, in New York (Sherwood,

2005). Atmospheric deposition is not a major source of phosphorus loading to Bear Lake and has little impact on simulations.

Lake surface area, mean depth, and length were derived using GIS analysis of bathymetric data. Depth of the mixed layer was estimated using a multivariate regression equation developed by Walker (1999). Existing water quality conditions in Bear Lake were represented using an average of the observed summer mean phosphorus concentrations for years 1985, 2006 and 2012. These data were collected through the NYSDEC LCI. The phosphorus loading to the lake was calculated using the monthly average flow and phosphorus loads simulated by MAPSHED. To obtain flow in units of volume per time, the depth of flow was multiplied by the drainage area for each month. To obtain phosphorus concentrations, the nutrient mass was divided by the volume of flow.

Internal loading rates reflect nutrient recycling from bottom sediments. Internal loading rates are normally set to zero in BATHTUB since the pre-calibrated nutrient retention models already account for nutrient recycling that would normally occur (Walker, 1999). Walker warns that nonzero values should be specified with caution and only if independent estimates or measurements are available. In some studies, internal loading rates have been estimated from measured phosphorus accumulation in the hypolimnion during the stratified period. Results from this procedure should not be used for estimation of internal loading in BATHTUB unless there is evidence the accumulated phosphorus is transported to the mixed layer during the growing season. Specification of a fixed internal loading rate may be unrealistic for evaluating response to changes in external load. Because they reflect recycling of phosphorus that originally entered the reservoir from the watershed, internal loading rates would be expected to vary with external load. In situations where monitoring data indicate relatively high internal recycling rates to the mixed layer during the growing season, a preferred approach would generally be to calibrate the phosphorus sedimentation rate (i.e., specify calibration factors < 1). However, there still remains some risk that apparent internal loads actually reflect under-estimation of external loads.

Eutrophication Model

The Bathtub model contains numerous equations which may be chosen to model the different components of eutrophication. Table 9 identifies the different equations used to model Bear Lake. Additional information on these equations and the other equations which may be used can be found in the Bathtub User's Manual (Walker, 1999). Information on the selected phosphorus, chlorophyll-*a* and non-algal turbidity equations is provided below. Values for the calibration factors may be set by the user, while the other inputs are derived from the watershed model and the lake characteristics.

Phosphorus in Bear Lake was modeled using the default model in Bathtub, which uses a 2nd order available phosphorus equation:

$$P = \frac{-1 + \sqrt{1 + 4 \times CP \times A1 \times P_i \times T}}{2 \times CP \times A1 \times T}$$

where P is the phosphorus concentration (µg/L), CP is the calibration factor for the phosphorus sedimentation rate (default value of 1), P_i is the inflow total phosphorus concentration (µg/L), and T is the hydraulic residence time (yr). $A1$ is the intercept of the phosphorus sedimentation term and is given by:

$$A1 = \frac{0.17 \times Q_s}{Q_s + 13.3}$$

$$Q_s = max\left(\frac{Z}{T}, 4\right)$$

in which Q_s is the surface overflow rate (m) and Z is the mean total depth (m).

Chlorophyll-*a* (B, in µg/L) is modeled using the default Bathtub equation which incorporates phosphorus, light and temperature using the following equations:

$$B_P = \frac{P^{1.37}}{4.88}$$

$$G = Z_{mix} \times (0.19 + 0.0042 \times F_s)$$

$$B = \frac{CB \times B_P}{(1 + 0.025 \times B_P \times G)(1 + G \times a)}$$

where B_P is the phosphorus potential chlorophyll-*a* concentration (µg/), G is the kinetic factor used in the chlorophyll-*a* model, Z_{mix} is the mean depth of the mixed layer (m), F_s is the summer flushing rate (yr^{-1}), and CB is the calibration factor for chlorophyll-*a*.

The non-algal turbidity (*a*) is estimated using a multivariate turbidity model as:

$$log(a) = 0.23 - 0.28 \times log(Z) + 0.20 \times log(F_s) + 0.36 \times log(P) - 0.027 \times LAT$$

in which LAT is the latitude of the lake in degrees North and other variables were defined above.

Table 9. BATHTUB Model Input Variables: Model Selections

Water Quality Indicator	Option	Description
Conservative Substance	00	Not Computed*
Total Phosphorus	01	2nd Order Available Phosphorus*
Total Nitrogen	00	Not Computed*
Chlorophyll-*a*	02	P, Light, T*
Transparency	01	VS. Chl-*a* & Turbidity*
Longitudinal Dispersion	01	Fischer-Numeric*
Phosphorus Calibration	01	Decay Rates*
Nitrogen Calibration	01	Decay Rates*
Error Analysis	01	Model and Data*
Availability Factors	00	Ignore*
Mass Balance Tables	01	Use Estimated Concentrations*
Output Destination	02	Excel Worksheet*

* Default model choice

Table 10. BATHTUB Model Input: Global Variables

Model Input	Mean	CV
Averaging Period (years)	0.42	NA
Precipitation (meters)	**	0.2*
Evaporation (meters)	**	0.3*
Atmospheric Load (mg/m^2-yr)- Total P	75.48	0.5*
Atmospheric Load (mg/m^2-yr)- Ortho P	43.52	0.5*
Atmospheric Load (mg/m^2-yr)- Total N	1000	0.5
Atmospheric Load (mg/m^2-yr)- Inorganic N	5000	0.5

* Default model choice

**Values for each model year were derived from the MapShed output. See Table 13

Table 11. BATHTUB Model Input: Lake Variables

Morphometry	Mean	CV
Surface Area (km^2)	0.47	NA
Mean Depth (m)	3.7	NA
Length (km)	1.48	NA
Estimated Mixed Depth (m)	3.7	0.12
Observed Water Quality	**Mean**	**CV**
Non-Algal Turb. (1/m)	†	0.2
Total Phosphorus (ppb)	*	*
Chlorophyll-*a* (ppb)	*	*
Secchi Depth (m)	*	*
Total P - Ortho P (ppb)	*	*
Calibration Factors	**Mean**	**CV**
Chlorophyll-*a*	1.6	0
Internal Load	**Mean**	**CV**
Total Phosphorus (mg/m2-day)	0.6	0

* Values derived from measurements were used for those years when measurements were made.

† Value calculated according to multivariate turbidity model given above and found in Table 4.6 of the BATHTUB Manual

Table 12. BATHTUB Model Input: Watershed "Tributary" Loading

Monitored Inputs	Mean	CV
Total Watershed Area (km^2)	26.4	NA
Flow Rate (hm^3/yr)	**	0
Total P (ppb)	**	0
Organic P (ppb)	**	0

**Values for each model year were derived from the MapShed output. See Table 13

Table 13. Annual Input Data Derived from MapShed

Year	Precipitation	Evaporation	Annual Flow	TP Conc.	Ortho P Conc.
	m/avg. per.	m/avg. per.	hm^3/yr	ppb	ppb
1982	0.52	0.31	12.2	17.2	15.0
1983	0.61	0.36	14.0	29.8	17.1
1984	0.67	0.38	18.9	25.5	17.1
1985	0.44	0.34	7.0	19.7	18.1
1986	0.77	0.43	19.5	28.5	20.2
1987	0.49	0.35	7.9	21.1	16.8
1988	0.38	0.30	5.0	25.0	21.0
1989	0.50	0.34	10.0	20.0	16.1
1990	0.58	0.38	11.8	15.7	14.2
1991	0.33	0.27	5.1	21.8	19.6
1992	0.61	0.39	13.9	29.4	18.6
1993	0.45	0.35	4.7	24.4	20.5
1994	0.59	0.32	15.9	25.5	16.1
1995	0.40	0.33	5.4	29.1	22.2
1996	0.69	0.39	17.7	20.7	15.4
1997	0.64	0.41	12.5	18.4	15.3
1998	0.43	0.33	8.1	19.1	16.1
1999	0.40	0.28	6.3	18.9	17.4
2000	0.57	0.38	11.3	19.1	15.6
2001	0.37	0.29	4.1	22.9	21.2
2002	0.51	0.35	10.6	21.8	17.7
2003	0.56	0.38	8.0	19.5	16.4
2004	0.64	0.35	17.9	27.1	19.2
2005	0.46	0.24	12.6	34.2	17.38
2006	0.60	0.33	14.3	30.0	16.8
2007	0.30	0.21	5.2	29.8	21.6
2008	0.50	0.31	13.0	23.8	17.6
2009	056	0.29	12.8	37.8	20.5
2010	0.48	0.25	14.7	26.8	15.4
2011	0.49	0.30	13.9	22.5	14.1
2012	0.26	0.21	3.2	30.3	24.7

Model Calibration

BATHTUB model calibration consists of:

1. Applying the model with all inputs specified as above
2. Comparing model results to observed phosphorus data
3. Adjusting model coefficients to provide the best comparison between model predictions and observed phosphorus data (only if absolutely required and with extreme caution.)

Model calibration focused first upon matching the modeled average total phosphorus concentration in Bear Lake to the measured summer mean concentrations from 1985, 2006 and 2012. As Bear Lake exhibits significant rooted aquatic life and field measurements have indicated hypoxic conditions in the bottom waters of the lake, internal loading was found to be appropriate. An internal load of 1.1 $mg/m^2/day$ yielded an acceptable phosphorus calibration (Figure 16).

Chlorophyll-*a* measurements were compared against the modeled chlorophyll-*a* values. The model was calibrated to match the average modeled and measured values (Figure 17). A calibration factor of 1.5 was needed.

Secchi disk depths were collected during the field measurements. Measured and modeled values are are shown in Figure 18. Although no calibration of the Secchi disk depth was undertaken the measured and modeled values are in good agreement.

Figure 16. Measured and modeled total phosphorus concentrations

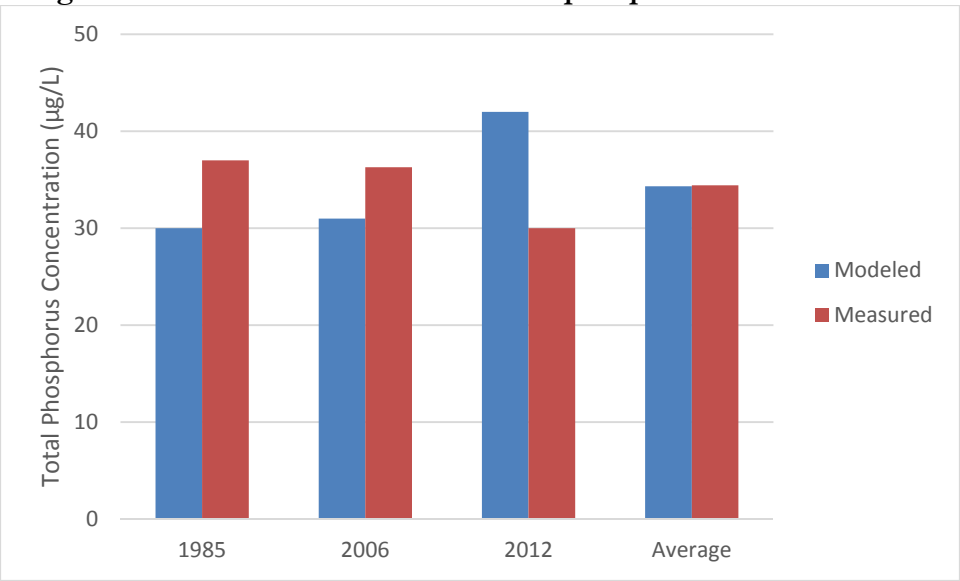


Figure 17. Measured and modeled chlorophyll-*a* concentrations

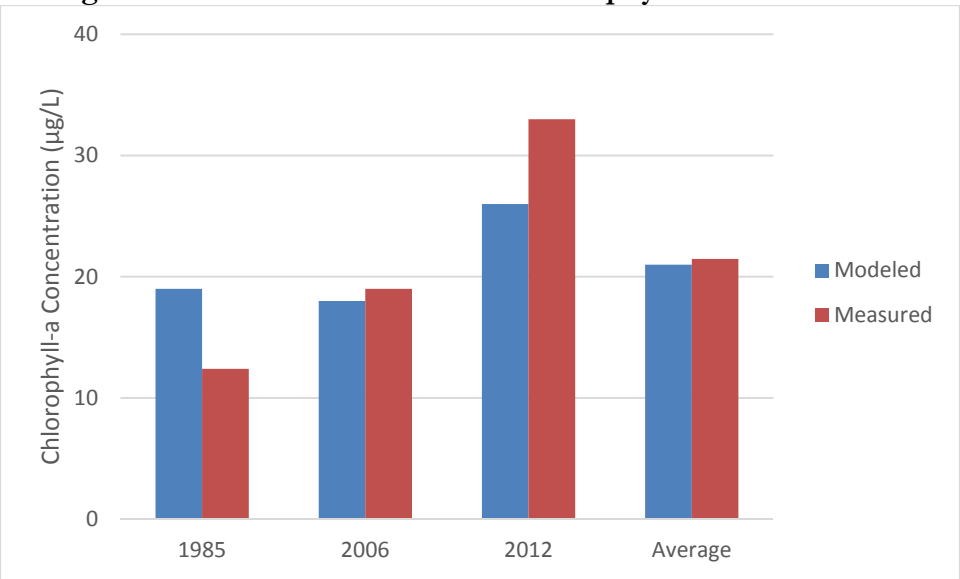
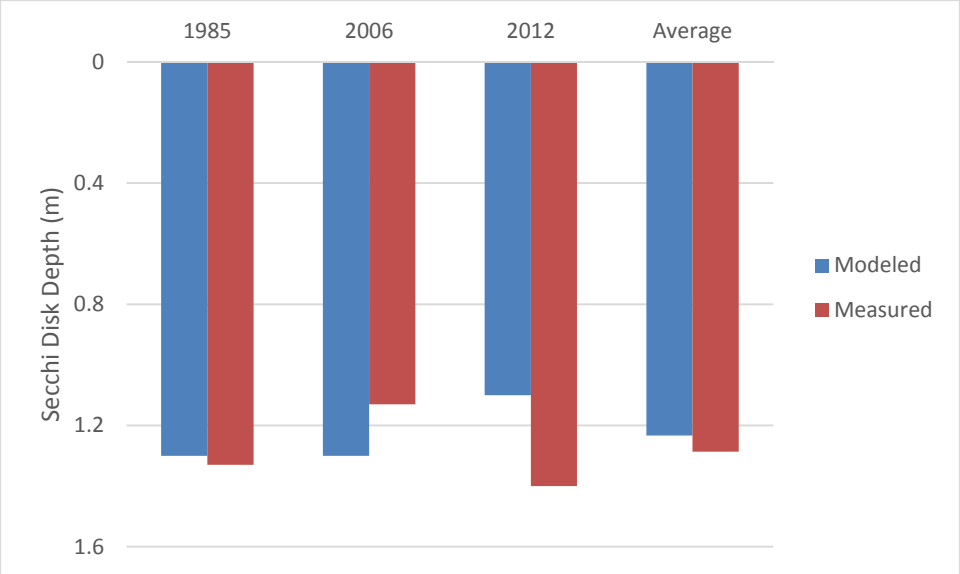


Figure 18. Measured and modeled Secchi disk depths



APPENDIX D. TMDL ALLOCATIONS FOR CLASS B WATER

Bear Lake is currently a Class *A* water, suitable as a source of water supply for drinking, and serves as an emergency water supply for the Village of Brocton. An effort to utilize Lake Erie as a water supply source is currently being considered as part of a regional effort. Should this occur, Bear Lake could be abandoned as a water supply. This would create the possibility that Bear Lake could be reclassified to a Class *B* water, suitable for primary and secondary contact recreation.

NYSDEC currently has a total phosphorus guidance value for ponded waters based upon the protection of primary and secondary contact recreation and aesthetics. As part of its Technical and Operations Guidance Series (TOGS 1.1.1 and accompanying fact sheet, NYSDEC 1993), NYSDEC has suggested that the summer mean total phosphorus concentration shall not exceed 20 µg/L (or 0.02 mg/L), based upon biweekly sampling conducted from June 1 to September 1. Should Bear Lake be reclassified to a Class *B* water, this guidance value of 20 µg/L would be an appropriate TMDL target.

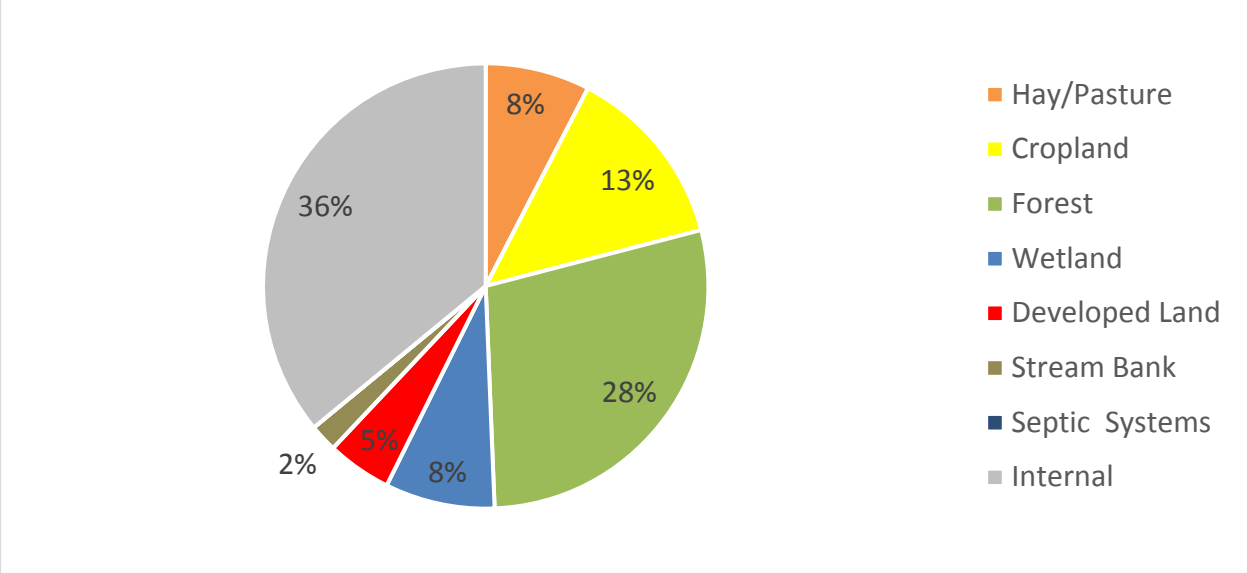
Based upon the potential for reclassification, allocation tables were developed which will allow Bear Lake to meet the 20 µg/L total phosphorus endpoint (Table 14, Figure 19). These allocations would only become effective if Bear Lake is reclassified to a Class *B* water. Many of the management actions needed to achieve the load reductions specified in Table 14 are the preliminary steps needed to achieve the load reductions needed to support the Class *A* best uses (Table 5). Thus, implementation of best management practices should not be delayed even if reclassification will be pursued. Sewering is still recommended to address the phosphorus load attributed to septic systems.

Table 14. Seasonal Total Phosphorus Load Allocations for Bear Lake to Meet a Total Phosphorus Concentration of 20 µg/L

Source	Total Phosphorus Load (lbs) During the Growing Season			% Reduction
	Current	Allocated	Reduction	
Agriculture	101.4	44.7	56.7	56%
Developed Land	22.7	4.3	12.7	56%
Septic Systems	50.2	0	50.2	100%
Forest, Wetland, and Natural Background	77.6	77.6	0	0%
Stream Bank Erosion	9.8	4.3	5.5	56%
Internal Load	174.2	76.7	97.5	56%
LOAD ALLOCATION	**435.9**	**213.3**	**222.6**	**51%**
Point Sources	0	0	0	0%
WASTELOAD ALLOCATION	**0**	**0**	**0**	**0%**
LA + WLA	**435.9**	**213.3**	**222.6**	**51%**
Margin of Safety	---	23.7	---	---
TOTAL	**435.9**	**237.0**	**---**	**---**

*Daily equivalent values are provided in Appendix E

Figure 19. Total Phosphorus Load Allocations for a 20 µg/L Total Phosphorus Endpoint



APPENDIX E. TOTAL EQUIVALENT DAILY PHOSPHORUS LOAD ALLOCATIONS

Table 15. Load allocations for attainment of Class *A* best uses

Source	Daily Total Phosphorus Load During the Growing Season (lb)			% Reduction
	Current	Allocated	Reduction	
Agriculture	0.66	0.03	0.63	94%
Developed Land	0.15	0.01	0.14	94%
Septic Systems	0.33	0	0.36	100%
Forest, Wetland, and Natural Background	0.51	0.51	0	0%
Stream Bank Erosion	0.06	0	0.06	94%
Internal Load	1.14	0	1.14	100%
LOAD ALLOCATION	**2.85**	**0.56**	**2.29**	**94%**
Point Sources	0	0	0	0%
WASTELOAD ALLOCATION	**0**	**0**	**0**	**0%**
LA + WLA	**2.85**	**0.56**	**2.29**	**80%**
Margin of Safety	---	0.06	---	---
TOTAL	**2.85**	**0.62**	**---**	**---**

Table 16. Load allocations for attainment of Class *B* best uses

Source	Daily Total Phosphorus Load During the Growing Season (lb)			% Reduction
	Current	Allocated	Reduction	
Agriculture	0.66	0.30	0.37	56%
Developed Land	0.15	0.07	0.08	56%
Septic Systems	0.33	0	0.33	100%
Forest, Wetland, and Natural Background	0.51	0.51	0	0%
Stream Bank Erosion	0.06	0.03	0.04	56%
Internal Load	1.14	0.50	0.64	56%
LOAD ALLOCATION	**2.85**	**1.39**	**1.45**	**51%**
Point Sources	0	0	0	0%
WASTELOAD ALLOCATION	**0**	**0**	**0**	**0%**
LA + WLA	**2.85**	**1.39**	**1.45**	**51%**
Margin of Safety	---	0.15	---	---
TOTAL	**2.85**	**1.55**	**---**	**---**

APPENDIX F. MONITORING DATA

Table 17. Lake surface water chemistry data

Date	Total Phosphorus (µg/L)	Chlorophyll-a (µg/L)	Secchi Disk (m)	Sampling Program
6/5/1985	No Data	11.1	1.5	LCI
7/17/1985	26	12.5	1.5	LCI
8/28/1985	48	13.6	1.0	LCI
7/1/1997	12.1	No Data	No Data	SUNY
7/1/1997	13.0	No Data	No Data	SUNY
7/21/1997	14.4	No Data	No Data	SUNY
7/21/1997	14.4	No Data	No Data	SUNY
6/12/2006	42.8	2.02	1.1	LCI
7/6/2006	45.3	No Data	1.4	LCI
7/6/2006	32.5	18.64	No Data	LCI
8/21/2006	32.6	25.10	0.9	LCI
9/12/2006	37.3	26.84	1.1	LCI
6/25/2012	20	15.9	1.5	LCI
7/31/2012	25.6	16.3	1.6	LCI
8/28/2012	33.4	46.9	1.1	LCI
9/25/2012	40.6	52.1	1.4	LCI

Table 18. Lake bottom water chemistry

Date	Total Phosphorus (µg/L)	Chlorophyll-a (µg/L)	Secchi Disk (m)	Sampling Program
7/11/2006	45.3	No Data	No Data	LCI
8/25/2012	28.1	No Data	No Data	LCI
7/31/2012	132	No Data	No Data	LCI
8/28/2012	88	No Data	No Data	LCI

Table 19. Temperature and dissolved oxygen profiles

6/12/2006

Depth (m)	Dissolved Oxygen (mg/L)	Temperature (°C)
0.0	9.2	18.5
1.0	9.0	18.7
2.0	8.8	18.7
3.0	8.8	18.6
4.0	4.2	16.7
5.0	1.6	14.5
6.0	0.2	13.7

7/6/2006

Depth (m)	Dissolved Oxygen (mg/L)	Temperature (°C)
0.0	7.6	21.3
1.0	7.6	21.8
2.0	7.5	22.0
3.0	7.6	22.2
4.0	3.5	21.5
5.0	2.8	16.3
6.0	2.6	14.5

9/12/2006

Depth (m)	Dissolved Oxygen (mg/L)	Temperature (°C)
0.0	7.8	16.2
1.0	7.4	16.5
2.0	7.2	16.7
3.0	7.1	16.9
4.0	6.9	17.0
5.0	6.7	17.2
6.0	0.2	17.1

6/25/2012

Depth (m)	Dissolved Oxygen (mg/L)	Temperature (°C)
0.0	9.3	24.5
1.0	9.4	24.5
2.0	9.4	24.5
3.0	9.8	23.7
4.0	1.7	18.9
5.0	0.1	14.5
6.0	0.0	12.9

7/31/2012

Depth (m)	Dissolved Oxygen (mg/L)	Temperature (°C)
0.1	9.27	26.28
0.99	9.47	26.13
2.01	8.61	25.43
2.97	4.20	23.87
3.01	4.24	23.84
4.03	1.61	22.82
5.0	0.01	18.60

8/28/2012

Depth (m)	Dissolved Oxygen (mg/L)	Temperature (°C)
0.27	8.85	23.97
0.96	8.85	23.84
2.03	8.47	23.77
3.04	4.00	22.76
4.00	0.28	21.92
5.04	0.00	21.00
5.96	0.00	18.27
7.03	0.00	15.54

9/25/2012

Depth (m)	Dissolved Oxygen (mg/L)	Temperature (°C)
0.25	8.29	16.27
0.98	8.30	16.25
2.00	8.24	16.26
3.00	8.27	16.26
3.99	8.28	16.26
5.00	8.26	16.24
6.02	6.23	15.37