Total Maximum Daily Load (TMDL) for Phosphorus in Engleville Pond

Schoharie County, New York

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1.0 INTRODUCTION

1.1. Background

In April of 1991, the United States Environmental Protection Agency (EPA) Office of Water's Assessment and Protection Division published "Guidance for Water Quality-based Decisions: The Total Maximum Daily Load (TMDL) Process" (USEPA 1991b). In July 1992, EPA published the final "Water Quality Planning and Management Regulation" (40 CFR Part 130). Together, these documents describe the roles and responsibilities of EPA and the states in meeting the requirements of Section 303(d) of the Federal Clean Water Act (CWA) as amended by the Water Quality Act of 1987, Public Law 100-4. Section 303(d) of the CWA requires each state to identify those waters within its boundaries not meeting water quality standards for any given pollutant applicable to the water's designated uses.

Further, Section 303(d) requires EPA and states to develop TMDLs for all pollutants violating or causing violation of applicable water quality standards for each impaired waterbody. A TMDL determines the maximum amount of pollutant, or load, that a waterbody is capable of assimilating while continuing to meet the existing water quality standards. Such loads are established for all the point and nonpoint sources of pollution that cause the impairment, and levels necessary to meet the applicable standards are specified. TMDLs provide the framework that allows states to establish and implement pollution control and management plans with the ultimate goal indicated in Section 101(a) (2) of the CWA: "water quality which provides for the protection and propagation of fish, shellfish, and wildlife, and recreation in and on the water, wherever attainable" (USEPA, 1991a).

Due consideration is given to seasonal variations and margin of safety. In the case of an algal impairment, or protection of Class A waters from too much algal growth, this may include assessment of the seasonal nutrient loadings responsible for the vegetative growth. A seasonal chlorophyll-a (chl-a) and corresponding Phosphorous seasonal target have been determined. In properly assessing these targets, this document goes beyond the usual annual Mapshed default assessment parameters and choses a watershed weather rolling average parameters to match the algal growing season, and uses monitoring data to consider the seasonal changes that occur due to changes in internal loading.

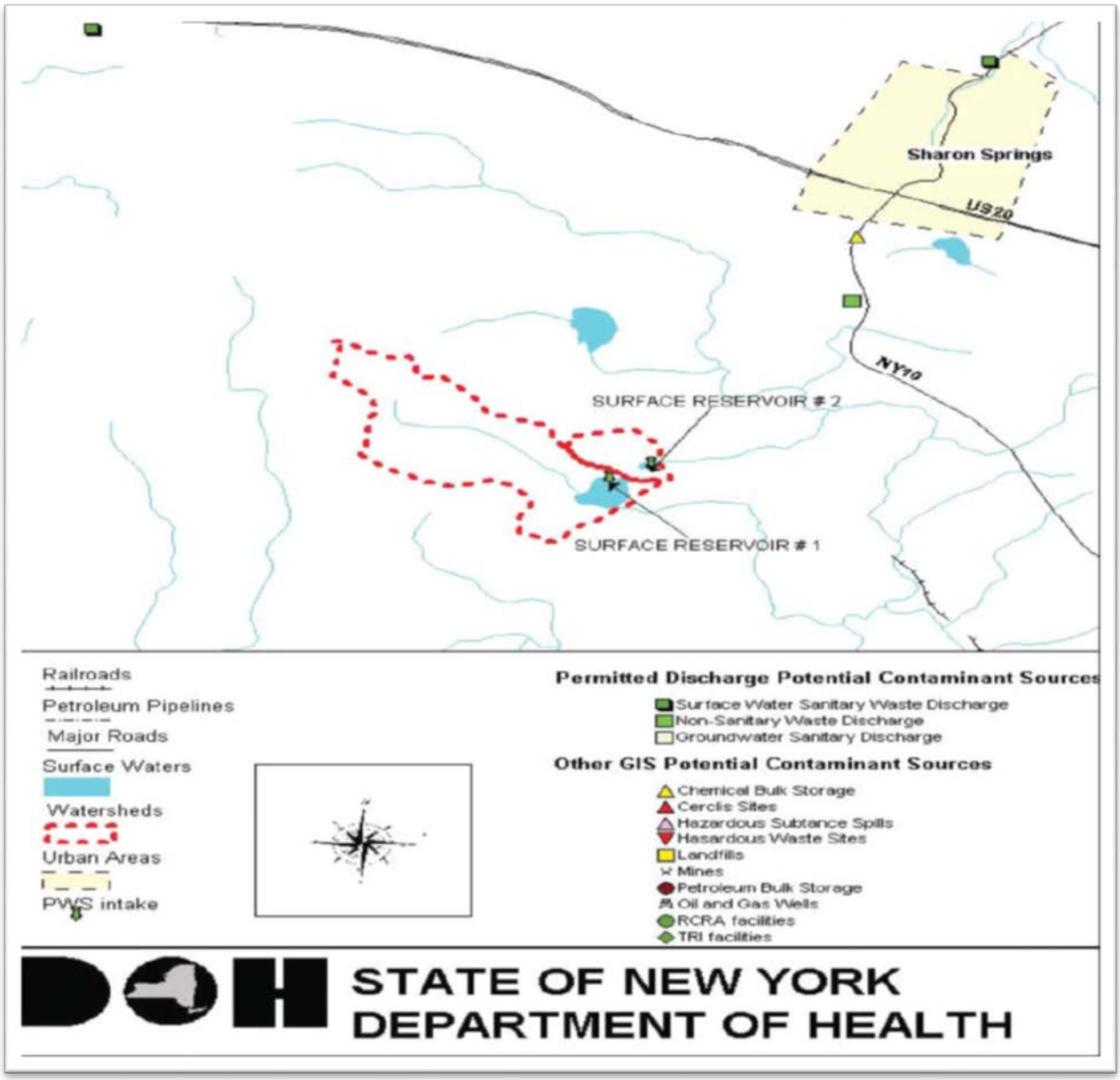
In addition to the responsibility to correct impaired waters uses as stated above, New York State Department of Conservation (DEC) has the duty to protect potentially threatened uses. In addition to being impaired for recreational use, Engleville Pond is considered stressed as a drinking water supply. Engleville Pond and the tributary leading to it are New York State Class A Waters. 6 NYCRR Part 701.6 states that; "The best usages of Class A waters are: a source of water supply for drinking, culinary or food processing purposes; primary and secondary contact recreation; and fishing. … ". This is the primary drinking water source for the Village of Sharon Springs, New York. Class A water standards are generally more stringent than those of recreational waters.

1.2. Problem Statement

1.2.1. Scope of Waterbody Impairment

Engleville Pond (WI/PWL ID 1202-0009) refers to Pond 1 which is situated in the Town of Sharon, within Schoharie County, New York. In 1997 this waterbody and its watershed were assessed pursuant to the Safe Water Drinking Act Amendment (SWDA) of 1996 requirements. The assessment found it to be sensitive to nutrient contamination from agriculture. Over the previous couple of decades, the lake had experienced degraded water quality that has reduced the lake's recreational and aesthetic value. Engleville Pond is presently among the lakes listed on New York State's Priority Waterbody List (PWL) Section 303(d) list of impaired waterbodies needing a TMDL for phosphorus

Figure 1. DOH Drinking Water Assessment Map Delineating Engleville Pond Basin



Pond 1 is the larger of the two surface water reservoirs that comprise the Sharon Springs drinking water supply with a groundwater well accessible as an auxiliary or emergency supply. Engleville Pond , referred to in Figure 1 as Surface Reservoir 1 (which shall be known as Pond 1), has a surface area of 29.5 acres, about 43 Million Gallon capacity, and receives water from a watershed projected to be 576 acres. (Software differences between DOH and the National Land Cover Database appear to delineate slightly different shapes for the western portion of the watershed, and there are very slight variations in the NLCD from 1997 to 2014). The un-named pond that leads to a sub-tributary of West Creek, which is referred to in Figure 1 as Surface Reservoir 2 (which shall be known as Pond 2), according to a 2012 DOH report, has a watershed area of about 43 acres and a capacity of about 3 Million Gallons.

Both ponds were tested by DEC in 1997 and in 2014. In 1997 both Ponds had high levels of Total Phosphorous (TP) and Chlorophyll-a (chl-a). In 2014 DEC testing showed Pond 1 to still have high TP and chl-a levels, though both were lower than in 1997. In 2014 Pond 2 both parameters levels were lower and were at levels DEC considers acceptable for Class A waterbodies. The data for the two ponds is summarized in Table 3, and further illustrated in Figures 8 through 10.

Although Pond 1 is listed on the Mohawk River Basin PWL, Pond 2 is not individually listed. This may have been an oversight since Pond 2 also had elevated phosphorous and chl-a levels during the 1997 testing. Both Pond 1 and Pond 2 were discussed in the Safe Drinking Water Act (SDWA) assessment as parts of the drinking water system. The 1997 lake monitoring data was initially considered a dataset as if for one waterbody for purpose of watershed modeling. The two ponds were hydraulically connected in 1997, and both were impacted by the watershed being modeled. Since that time the water flow to Pond 2 has changed in origin. Separate impairment assessments are now justified, and Pond 1 data for 2014 was modeled separately.

Both ponds were tested again in 2014 to assess water quality and suitability for best uses. Based on the 2014 test results, a Phosphorous TMDL is deemed necessary for Pond 1. Pond 2 water quality has improved and it is currently meeting water quality targets for phosphorus and chl-a.

In 2014 both ponds were sampled more rigorously than in 1997, with sampling occurring 10 times in two week intervals during the period from May to September. 2014 data indicates Pond 1 has improved but is still experiencing eutrophic conditions. Pond 2 improved and did not exceed either the TP or the chl-a guidance intended to prevent it from being a 'threatened' drinking water source.

It is possible that the 2014 water quality results for Pond 2 were impacted by a change in the piping of water. In 1997 both ponds were hydraulically impacted by the watershed. By 2014 the conduit to Pond 2 was disconnected, and water appears to be only from the 43 acres of watershed. This may account for the water being of acceptable quality without additional action. Either Pond, however, can physically still be tapped for drinking water. Since both ponds are still part of the water supply system, and piping could change again in the future, it is still appropriate

to discuss both in this TMDL. Future monitoring of Engleville Pond will also continue to evaluate both Ponds 1 & 2.

Pond 1 water quality has also improved since 1997, but still needs further phosphorous reduction. The present New York PWL Section 303(d) list, places this pond in Category 1 as an impaired waterbody due to algal and weed growth resulting from excess nutrients. Phosphorus is cited as causing an impairment for use as a recreational water. The PWL further notes that its best use as a drinking water is considered to be 'threatened'**.** It is further placed it in Category 4c has having a pollutant based impairment for which a TMDL is not appropriate. This 4c listing is a reference to the fact that the impairment is algal based. Algae is itself not a 'pollutant'.

The sources of phosphorus impacting the water quality of Engleville Pond are primarily agricultural and internal loading. The water quality is influenced by runoff events from the drainage basin and seasonal impacts of lake stratification induced anoxia that results in sediment releasing phosphorous. In response to precipitation nutrients such as phosphorus, naturally found in New York soils, drain into the lake from the surrounding drainage basin by way of; stream, overland flow, and subsurface flow. Nutrients are then deposited and stored in the lake bottom sediments. This accumulated bottom sediment phosphorus is then released during summer stratification and the resulting anoxic conditions. Phosphorus is often the limiting nutrient in temperate lakes and ponds and can be thought of as a fertilizer; a primary food for plants, including algae. When lakes receive excess phosphorus, it "fertilizes" the lake by feeding the algae. Too much phosphorus can result in algae blooms, which can damage the ecology/aesthetics of a lake, as well as the economic well-being of the surrounding drainage basin community.

There is a small Class A unnamed tributary leading to Engleville Pond that may add a contribution of soil carried phosphorous when flows are large enough to cause stream erosion.

1.2.2. Additional Consideration for Class A water

In addition to the impairment listing, DEC believes it has the responsibility in this TMDL to address the Class A waterbody best usage as a drinking water source. Although not listed as 'impaired' for this use, the waterbody is still 'threatened' as a drinking water source due to the potential for Disinfection By-Products in the water treatment system resulting from excess algal growth. It is appropriate to protect this best usage, even though the water treatment plant should mitigate some of the potential health impacts on the drinking water.

DEC believes that Class A potable water chl-a concentrations for Ponds 1 and 2 should not exceed a seasonal mean target value of 6 µg/l. Furthermore, a site specific level of TP that achieves this chl-a target should be determined and be viewed as a goal for each specific Class A waterbody. The statistical method correlating TP to chl-a is presented in Appendix A, and results in a phosphorous target very close to the BATHTUB Model phosphorous target prediction of 12 µg/l required in Pond 1 to protect it as a drinking water source. This target is intended to minimize the possibility of harmful Disinfection- By-Products (DBPs) in the drinking water, by limiting the

mean seasonal chl-a to 6 µg/l (see Appendix A: Numeric Endpoint Development for Potable Water Use). The Target level of 20 µg/l TP associated with the recreational impairment would have been selected if it were more conservative than the Appendix A result.

The Pond 1 watershed is the subject of this TMDL since the 2014 sampling data shows that Pond 1 remains impaired and above the desired concentrations for a drinking water source. Pond 2 is not individually listed on the PWL, and has improved to acceptable concentrations from the high phosphorous readings in 1997. Both Ponds are discussed, however, since they both are part of the drinking water system and it is desirable to protect their drinking water best use quality accordingly.

1.2.3. Dam Safety Issue Coordination

There are NYS Dam Safety files for Pond 1 Dam ID #158-4283 and Pond 2 Dam ID# 158-5801. Dams are periodically inspected by DEC, and if any TMDL Implementation Section measures impact the Pond 1 dam, these measures would need to consider dam safety and may need to be coordinated with DEC's Dam Safety Section for review, and may require permits from DEC. If a mitigation measure such as hypolimnetic extraction is selected, for example, then care will have to be taken that any new piping not interfere with dam integrity or exacerbate seepage problems.

2.0 WATERSHED AND RESERVOIR CHARACTERIZATION

2.1. History of the Reservoir and Watershed

Figure 2 Pond 1 - vantage point between outlet and Mill Pond Road

Construction: Pond 1 is on a tributary of West Creek in Schoharie County, New York and is used for both recreational fishing and as a drinking water supply. Construction of the Dam that impounds the water was completed in 1910. Its normal surface area is 30 acres, and it is owned by the Village of Sharon Springs. Pond 1 receives flow from the almost 1 square mile watershed including a Class A unnamed tributary.

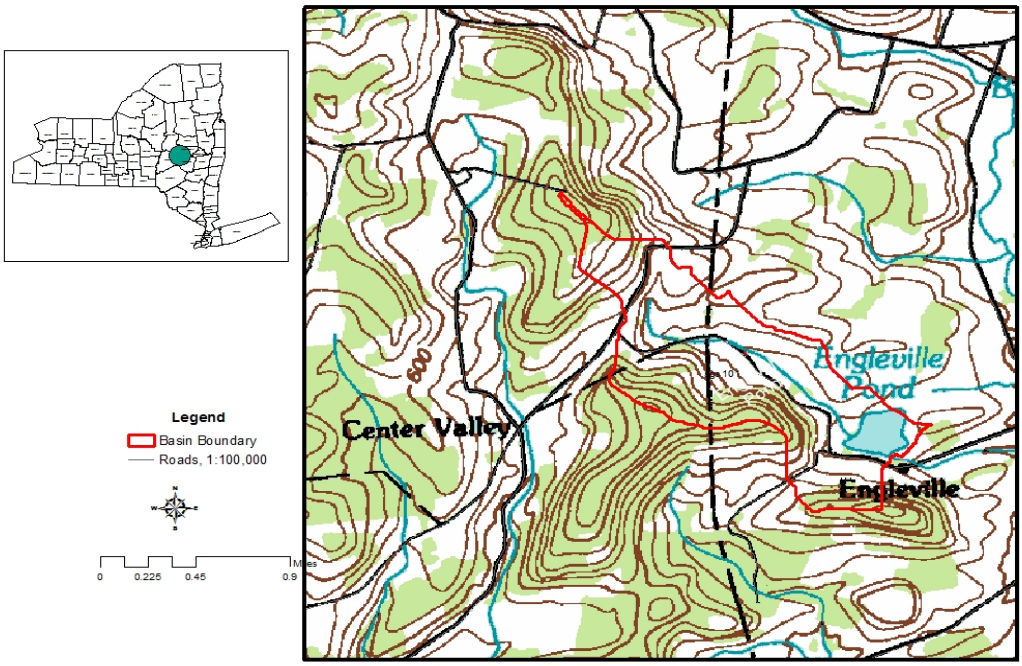
Pond 1 has two outlets. The natural outlet, over dam ID # 158-4283 to a West Creek tributary, is presently controlled by use of stoplogs to modify the volume of this reservoir. The second outlet is piped to the northeast, past Pond 2, to the Water Treatment Plant (WTP). At the WTP water is processed to become drinking water for the Village of Sharon Springs.

Fishing is allowed in Pond 1. DEC regional Fisheries staff have indicated that Pond 1 has been known to have Chain Pickerel, Smallmouth Bass, Pumpkinseed, Brown Bullhead, Black Crappie, and Yellow Perch. There are also some fish in the restricted Pond 2, but less information was available on the species. Public Fishing is allowed at Pond 1 but Pond 2 is secured from public access by a locked chain gate.

2.2. Watershed Characterization

Calculations indicate that Pond 1 has a direct drainage basin area of 576 acres excluding the surface area of the lake (Figure 3). Elevations in the lake's basin range from approximately 2,169 feet above mean sea level (AMSL) to as low as 1,400 feet AMSL at the surface of Pond 1.

Figure 3 Map Delineating Pond 1 Direct Drainage Basin



Minor variations in watershed delineations may have contributed to slight differences in the watershed characterizations in different years. Comparisons of the differences will be stated as information to provide context for the corresponding years of monitoring data and modeling. Differences include;

- In a 2012 Report, the DOH assessed Pond 1 has having a Capacity of 43 Million Gallons and watershed area of 590 acres, and Pond 2 has having a capacity of 3 Million Gallons and a watershed area of 43 acres, and
- DEC compared, using the 2011 USGS Stream Stats delineation shapefile downloaded to the DEC GIS system, the 2001 with the 2011 National Land Cover Database (NLCD). The main difference in land usage appeared to be that several percent of the area defined as forest in the 2001 NLCD was redefined as wetland and/or 'open land' in the 2011 NLCD.

The smaller Pond 2 watershed and morphometric characteristics were not assessed by CADMUS. In 1997 Pond 2 received most of its water from the same watershed as Pond 1. By 2014 they were no longer hydraulically connected and Pond 2 was shown to be within Class A water quality goals.

2.2.1 Pond Inlet Flow: There is a Class A tributary that flows into Pond 1's west shore. The spring fed stream originates near Palmer Road and runs about 2.5 kilometers ending at a dirt drive and wetland area that abuts Pond 1 to the west. Usually the stream flow is minimal, but during significant rain events there is flow from the spring down the dirt/gravel road and washes into the larger wetland area to the west of the pond. This area may present a phosphorus reduction opportunity.

Figure 4. Primary Tributary outlet through adjacent wetland area and gravel drive.

Figure 5: Approximate Water Outlets leading to the Water Treatment Facility

Class A Tributary **Pond 1** **Pond 2** **Water Treatment Plant**

The stream and the dirt road during rainfall events, are probable sources of phosphorous contributing sediment going to the Pond 1. As such erosion control efforts on these two areas are likely to reduce the sediment phosphorous loading to the pond (further discussed in the Implementation Section). Since this is an intermittent stream and an even more sporadically impacted dirt road, the contribution is difficult to quantify and is considered included in the general overland flow estimate.

2.2.2 Pond Outlet Flow: Either Pond may be used as a drinking water source:

a. Pond 2, the pond to the north, has an outlet conduit that flows to the treatment plant, or may be turned off. Otherwise water may depart via a vegetated spillway at the northern end of the dam/berm at the eastern edge of the pond.

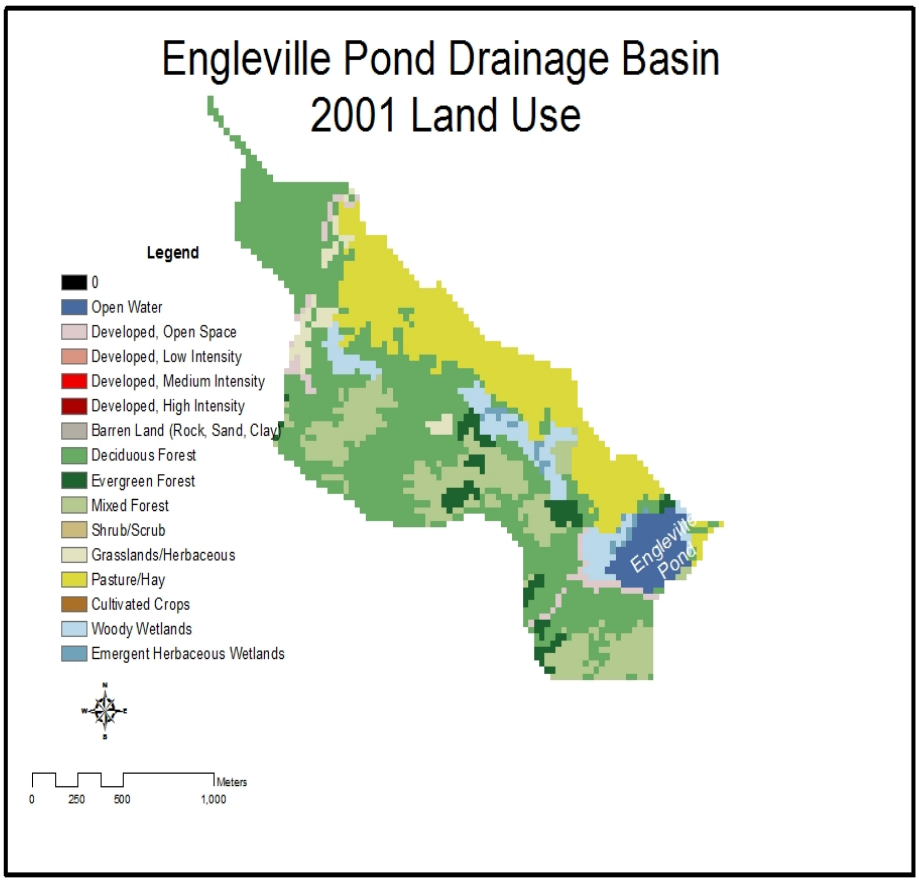
b. Pond 1, the pond to the south, may flow out either of two outlets:

    1) The northeast flow is through a conduit past Pond 2 to the treatment plant.
    2) The natural surface water flow outlet is southeast to a Class C tributary of West Creek next to Mill Pond Road. The Water Treatment Plant Operator may modify this flow by adding and removing dam stop logs. This provides some control of the Pond 1 reservoir water quality and quantity.

2.3. 2015 Watershed Land Use Verification

Existing land use and land cover in the Engleville Pond drainage basin was determined from digital aerial photography and geographic information system (GIS) datasets. Digital land use/land cover data were obtained from the 2001 National Land Cover Dataset (NLCD) (Homer, 2004).

Figure 6. 2001 Land Use in the Engleville Pond Drainage Basin



The NLCD is a consistent representation of land cover for the conterminous United States generated from classified 30-meter resolution Landsat thematic mapper satellite imagery data. High-resolution color orthophotos were used to manually update and refine land use categories for portions of the drainage basin to reflect current conditions in the drainage basin (Figure 6). Figure 6 shows the relevant NLCD initially used for modeling when only the 1997 lake data was available. Appendix B provides additional detail about the refinement of land use for the drainage basin. Land use categories (including individual category acres and percent of total) in Engleville Pond's drainage basin are listed in Tables 1 and 2 presented in Figure 7. It should be noted that both the 2014 and 1997 data were modeled by DEC using the 2011 National Land Cover Database whereas CADMUS modeled only the 1997 concentration data using the 2001 NLCD version seen in Figure 6. Land delineations are quite similar with a little change from 2001 to 2011 mostly from the forest and wetland depictions.

Table 1. Land Use Acres and Percent Pond 1 Drainage Basin

Land Use Category	Acres 2001 NLCD	% of Drainage Basin via 2001 NLCD	Acres 2011 NLCD	% of Drainage Basin via 2011 NLCD
Agriculture	132.5	23%	140.8	23.6%
Hay & Pasture	(131.9)	(22.9%)		(23.6%)
Cropland	(0.6)	(0.1%)		(0.0%)
Open Land			12.4	2.1%
Developed Land	16.7	3%	12.4	2.1%
Low Intensity	(16.7)	(3.0%)	(12.4)	(2.1%)
High Intensity	(0)	(0.0%)		
Forest	418.6	73%	388.0	65.1%
Wetlands	8.5	1%	42.0	7.1%
TOTAL	576.3	100%	595.6	100%

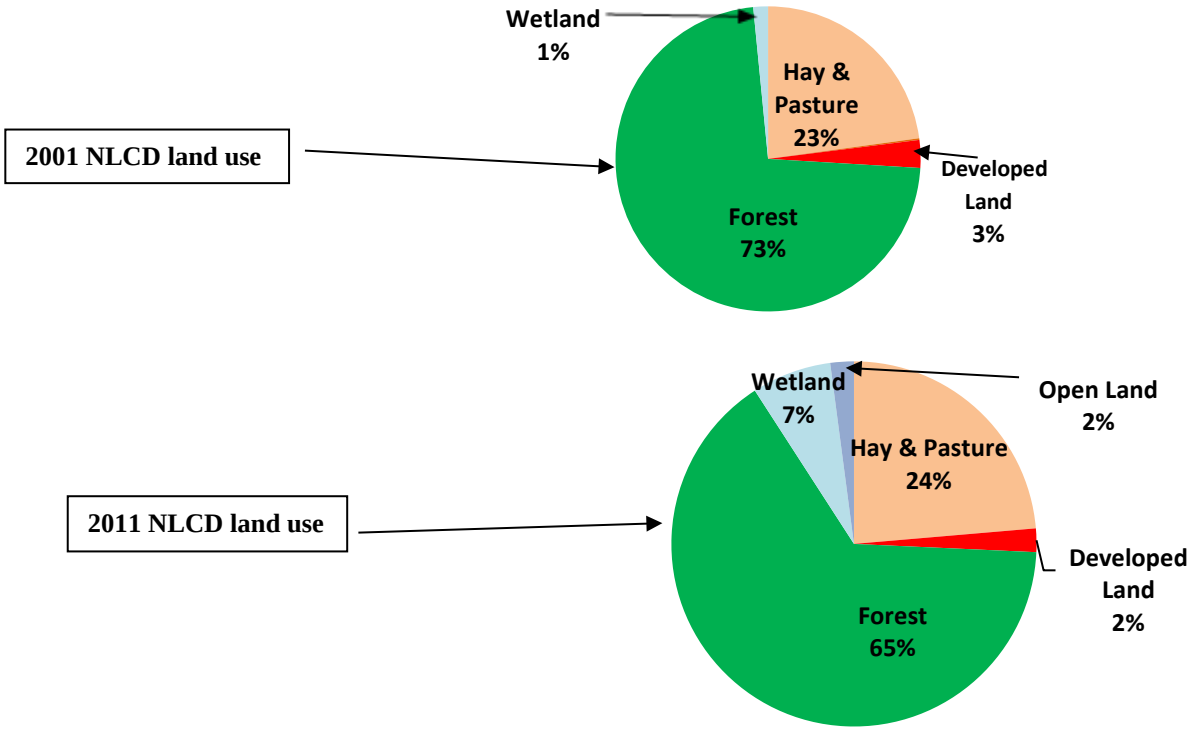
Table 1 shows a comparison of the 2001 NLCD vs the more current 2011 NLCD presently populating the Mapshed database currently used.

USGS Streamstats shapefile now defines the watershed slightly differently than in 2001, resulting in a slightly larger watershed area. Other slight variations in land used resulted from using the newer 2011 NLCD rather than the 2001 NLCD. A small percentage of land was redefined from forest to wetland, and the category of 'Open Land' was introduced. The Mapshed Manual discusses Open Land as follows:

> **Open Land -** this category is intended to depict such land types similar to "open range" or "grassland", such as are often found in the western United States. These essentially "natural" areas are typically not cultivated or heavily pastured.

If USGS StreamStats is used to identify the Pond 1 watershed, using the natural outlet to West Creek as the delineation point, the resulting watershed was nearly identical in shape except for a slight variation in the western portion**.** The resulting shapefile was downloaded to the DEC Geo Information System, and interfaced with the National Land Cover Database (NLCD) for 2011. Aggregate watershed changes were minimal, and the basic conclusion that reduction is needed from agriculture and internal loading clearly still held true.

Figure 7. Percent Land Use in Pond 1 Drainage Basin



2.4. Lake Morphometry

The primary body of water, Pond 1, is a 30 acre waterbody at an elevation of about 1,400 feet AMSL. Pond 2 is a smaller body of water and a part of the water supply system, that has a minimal actual watershed of its' own. Drinking water may be taken from either, or both, of these Ponds. Table 2 summarizes key morphometric Pond 1 characteristics.

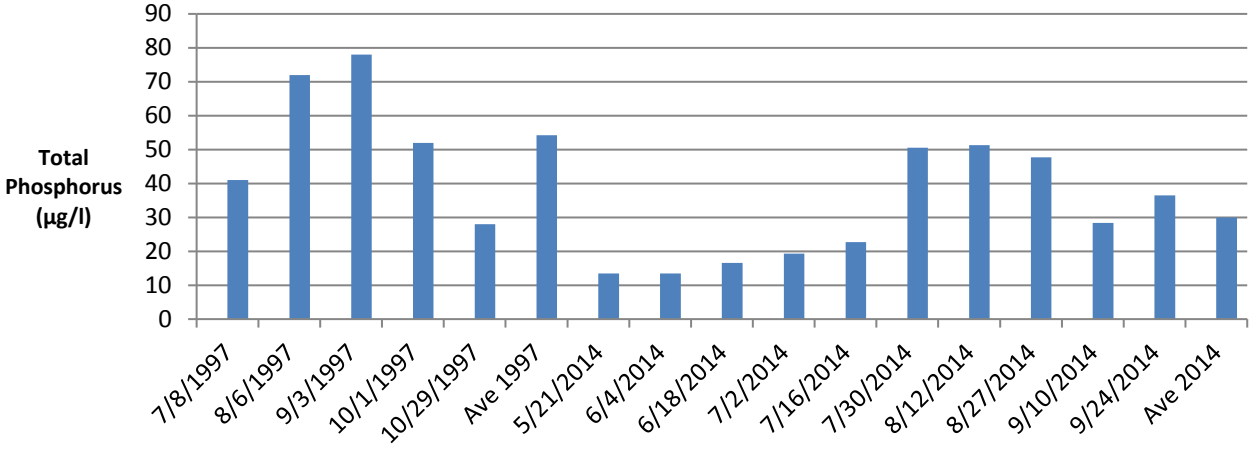
Table 2. Pond 1 Characteristics

Surface Area (acres)	**30**
Elevation (ft AMSL)	1400
Mean Depth (ft)	6.5
Length (ft)	1,716
Width at widest point (ft)	1,216
Shoreline perimeter (ft)	4,705
Direct Drainage Area (acres)	576
Watershed : Lake Ratio	19:1
Mass Residence Time (years)	0.2
Hydraulic Residence Time (years)	0.2

2.5. Water Quality

In 2014, water quality samples were collected at Ponds 1 and 2 for a 20 week period, 10 samples at each lake were collected every alternate week. For 5 of the sample weeks, the DEC Lake Classification Index (LCI) set of parameters were evaluated, including; Dissolved Oxygen (DO), Total Phosphorous (TP) and Chlorophyll–a (Chl-a). The other 5 weeks of testing was added to the usual LCI sampling schedule in order to better assess the parameters of interest for this TMDL including DO, TP and Chl-a. LCI and TMDL focused testing was done on alternating trips to Pond 1. This data was evaluated and compared to the 1997 TP and Chl-a data. Some progress in water quality was observable in the improved Phosphorous and Chl-a concentrations.

Figure 8. Measured Concentrations of Total Phosphorus in Pond 1

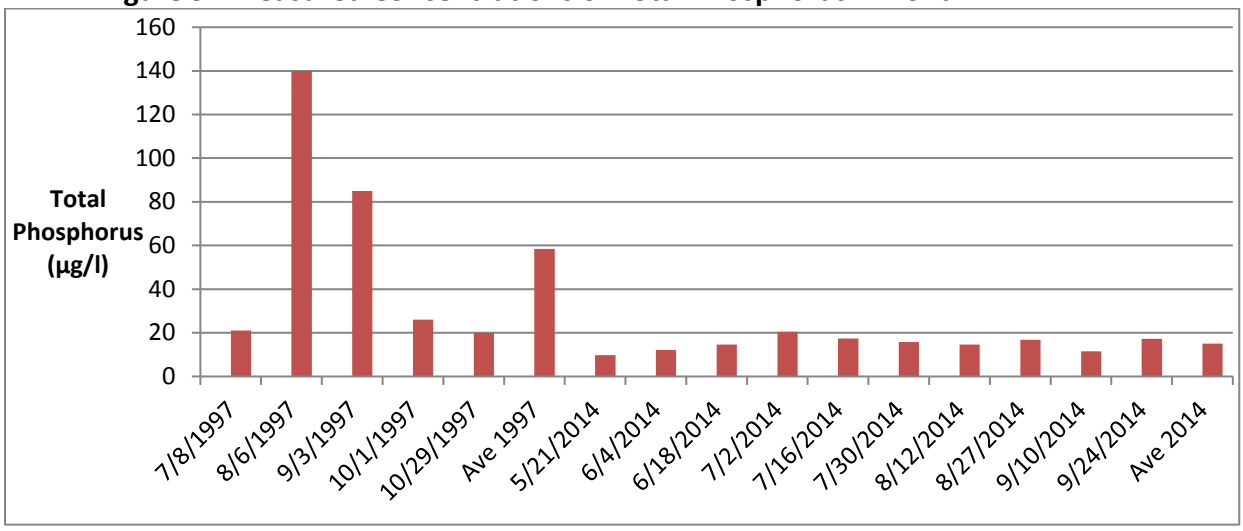


Meeting Drinking Water Target Values:
Ultimate compliance with water quality standards for the TMDL will be determined by measuring the lake's water quality to determine when the phosphorus guidance value and Chl- a target is attained. The 2014 monitoring data shows Pond 2 presently in attainment of the 6 µg/l drinking water goal for chl-a, as well as the 20 µg/l phosphorous goal for recreation. Pond 1 has improved, but is not yet in in attainment of the goals of this TMDL. These TMDL targets and goals are discussed in more detail in Appendix A.

In 1997 the mean Pond 1 Phosphorous concentration was 54 µg/l. This improved to a summer mean 30 µg/l in 2014. This is still above both the 20 µg/l recreational target, and the level that is *expected* to correlate to the protective Class A Water target of 6 µg/l chl-a discussed below. The Pond 1 TP target value expected to result in meeting 6 µg/l chl-a is 12 µg/l TP.

When the Appendix A method is applied to the 2014 data, it appears that a TP concentration of 11.4 µg/l correlates best to a level of 6 µg/l chl-a. When applied to the 1997 data alone this method suggested 15 µg/l would correlate to 6 µg/l chl-a. If a mean value of these two 'snapshots' were used, the TP target implied would be 13.2 µg/l. In Appendix C, the Bathtub Models runs done in 2015 predict 12 µg/l TP to correlate to 6 µg/l chl-a in Pond 1. Given that the 1997 data is less representative of the present waterbody due to data age, sample size, and flow configuration changes, the Bathtub prediction and/or 2014 sampling data based predictions seem more reliable. Therefore, a Phosphorous Target of 12 µg/l TP is selected as the concentration that is expected to protect the drinking water best use.

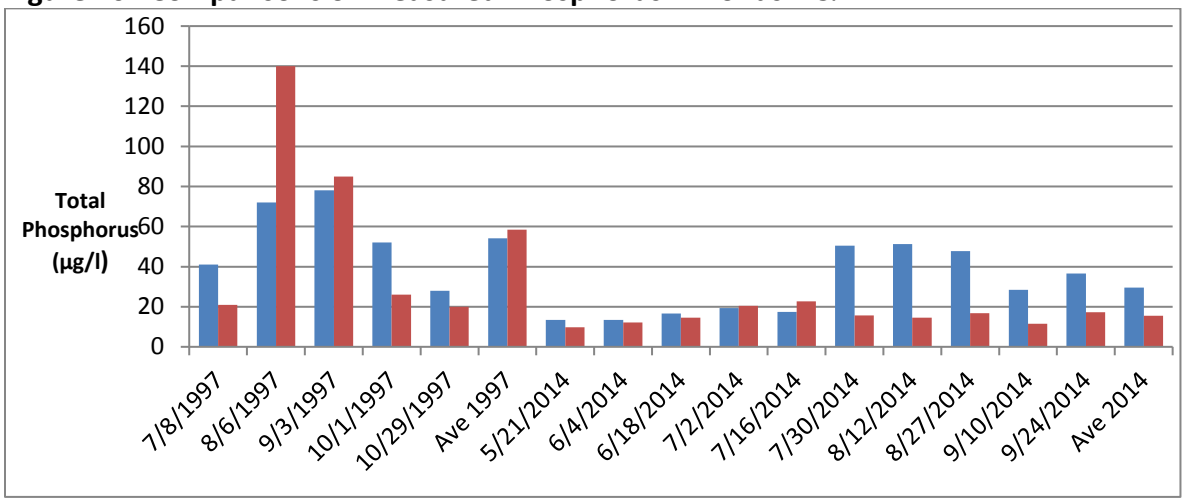
Figure 9. Measured Concentrations of Total Phosphorus in Pond 2



The 1997 Pond 2 mean Phosphorous concentration was 58 µg/l. This improved to a summer mean of 16 µg/l in 2014. This is less than both the 20 µg/l recreational use target and the lake specific concentration expected to achieve an acceptable level of algal growth for a Class A potable waterbody. There are multiple factors that may have contributed to this reduction and TP target changes for these ponds, including:

1) A change in plumbing resulted in Pond 2 no longer receiving water directly from the main watershed by 2014 as it did in 1997, and
2) Farming practices may have changed by 2014 with the fields being use for hay but not actively use for raising livestock.

Figure 10. Comparisons of Measured Phosphorus in Ponds 1 & 2



Ponds 1 – Blue Bars Ponds 2 – Red Bars

To calibrate the Bathtub Model in 2015 DEC staff used both the 2014 data and the 1997 data. However, in using the 1997 data DEC used four (of five) Pond 1 data points, keeping the 10/1/97 data point and discarding the late October reading. This compared Pond 1 to Pond 1 seasonal data more directly since present LCI sampling (as in 2014) ends in September.

2014 data indicated further progress in reducing phosphorous loading is still needed for Pond 1, but that Pond 2 water quality is presently acceptable. The data from both sampling efforts are referenced in Figures 11 to 13 and are the basis for the options and recommendations presented.

DEC water quality monitoring program description:

DEC's Lake Classification and Inventory (LCI) program was initiated in 1982 and is conducted by DEC staff. Each year, approximately 10-25 water bodies are sampled in a specific geographic region of the state. The waters selected for sampling are considered to be the most significant in that particular region, both in terms of water quality and level of public access. Samples are collected for pH, ANC, specific conductance, temperature, oxygen, chl-a, nutrients and plankton at the surface and with depth at the deepest point of the lake, 4-7 times during the sampling year (with stratified lakes sampled more frequently than shallow lakes). Sampling generally begins during May and ends by October.

The LCI effort had been suspended after 1992, due to resource (mostly staff time) limitations, but was resumed again in 1996 on a smaller set of lakes. Since 1998, this program has been geographically linked with the Rotating Integrated Basin Sampling (RIBS) stream monitoring program conducted by the DEC Bureau of Watershed Assessment. LCI sites are chosen within the RIBS monitoring basins. RIBS and LCI monitoring is conducted in 2 to 4 of the state's 17 major

drainage basins each year, resulting in data available statewide over a 5-year cycle. These data include water column, sediment, and organism tissue chemistry and biological assessment of water quality using macroinvertebrate community analysis and toxicity testing. Pond 1 is in the Mohawk River Basin and will be in the set of lakes that may be tested every 5 years when Mohawk Basin waterbodies are selected for testing. Waterbodies tested are generally from among the waterbodies listed on the NYS Priority Waterbodies List (PWL) for which water quality data are incomplete or absent, from the largest lakes in the respective basin in which no water quality data exists within the DEC database, or to assess the progress being made by a TMDL Implementation Plan as will be the case for Pond 1.

There were LCI water samples collected in both Ponds 1 & 2 during the summer of 1997. The results from these sampling efforts show eutrophic conditions with the 54 µg/l mean concentration of phosphorus in the lake exceeding the state guidance value for phosphorus (20 µg/l (or 0.020 mg/l), applied as the mean summer, epilimnetic total phosphorus concentration), which increases the potential for nuisance summertime algae blooms. Pond 2 also showed eutrophic conditions and an average concentration of 58 µg/l of phosphorus.

Although not in the original 2014 LCI basin sampling schedule, Ponds 1 and 2 were added to the 2014 monitoring to facilitate the development of this TMDL with more current data. The resulting 10 sampling weeks in 2014 showed improvement in both Ponds, with Pond 1 still having unacceptably high phosphorous and chl-a, and Pond 2 achieving acceptable levels of both.

3.0 NUMERIC WATER QUALITY TARGET

The TMDL target is a numeric endpoint specified to represent the level of acceptable water quality that is to be achieved by implementing the TMDL. The water quality classification for both Ponds 1 and 2 is A, which means that the best uses of the lake are as a source of water supply for drinking, culinary or food processing purposes; primary and secondary contact recreation; and fishing. The lakes must also be suitable for fish propagation and survival. New York State has a narrative standard for nutrients: "none in amounts that will result in growths of algae, weeds and slimes that will impair the waters for their best usages" (6 NYSCRR Part 703.2). As part of its Technical and Operational Guidance Series (TOGS 1.1.1 and accompanying fact sheet, NYS, 1993), DEC has suggested that for waters classified as ponded (i.e., lakes, reservoirs and ponds, excluding Lakes Erie, Ontario, and Champlain), the epilimnetic summer mean total phosphorus level shall not exceed 20 µg/l (0.02 mg/l) for recreational uses, based on biweekly sampling, conducted from June 1 to September 30.

The Priority Waterbody List (PWL) lists Pond 1 as Impaired for Recreational Use and Threatened as a Drinking Water Supply. TMDLs are intended to address water impairments and protect best uses. Accordingly, this guidance value of 20 µg/l would be the only TMDL target for Ponds 1 and 2 were they not an active water supply. However because it is actively used potable water, a second target was calculated to protect the use as a drinking water source.

The guidance value of 20 µg/l total phosphorus has been developed for ponded waters and is protective of aesthetics. This guidance value was not specifically derived to protect the drinking water use of waterbodies such as Ponds 1 and 2. Site specific numeric translations of the state's narrative standard for the protection of the drinking water use was, therefore, developed for both Ponds. The TMDL establishes a Total Phosphorous target of 12 µg/l for Pond 1, using both the BATHTUB correlation to the chl-a target of 6 µg/L and considering the Appendix A statistical method, in order to provide a site-specific numeric translation of the state's narrative standard.

Using the Appendix A correlation of NYS Class A and Class AA lakes, the 1997 monitoring data indicates a target of 15 µg/l TP should result in 6 µg/l chl-a, and the 2014 data indicates that a concentration of 11.4 µg/l TP should meet this 6 µg/l chl-a target. The BATHTUB predictive model correlates 12 µg/l TP to 6 µg/l chl-a**.**

A target of 12ug/l is chosen: The mean of these two values would be 13.2 µg/l, however since the 2014 data was more thorough and recent it was given greater weight, and the BATHTUB derived value of 12 µg/l was chosen to better correspond to the 2014 data results. The 2014 data derived target value was also the more restrictive of the two monitoring data sets, providing more reasonable assurance that the 6 µg/l chl-a target desired for Class A water will be achieved.

Table 3 includes a comparison of the mean summer eplimnetic monitoring data for TP and the statistically implied chl–a targets for Ponds 1 and 2.

Table 3: Monitoring Data vs Target Values

Site	**1997 data TP(µg/l)**	**1997 Target TP(µg/l)**	**2014 data TP(µg/l)**	**2014 Target TP (µg/l)**	**Mean Target TP (µg/l)**	**BATHTUB TP (µg/l)**
Pond 1	54	15	30	11.4	13.2	12
Pond 2	58	20 (39 calc*)	16	Complies *	na	na
Site	**1997 chl-a(µg/L)**	**Reduction to 6µg chl-a**	**2014 chl-a (µg/L)**	**Reduction to 6µg chl-a**	**Mean Reduction**	
Pond 1	31	25 decrease	17.75	11.75	18	na
Pond 2	18	12 decrease	2.77	Complies *	na	na

**The target statistically correlating TP to chl-a reductions, per Appendix A, is outside of the prediction bands for the regression model and higher than the 20 µg/l TP recreational guidance value. In this case the Target TP would revert to the recreation 20 µg/l TP over the 39 µg/l TP correlating to 6 µg/l chl-a.*

The amount of reduction required is based on monitoring data of the specific body of water as well as the TP vs chl-a ratio. The site monitoring data is applied to the statistical correlation of TP vs chl-a chart shown in Appendix A, and the resulting TP target is intended to protect the drinking water usage. The TP targets are statistically derived concentrations expected to achieve 6 µg/l chl-a based on the NYS DEC Lake Classification Index (LCI) monitoring data. (Appendix A.)

The Appendix A response model provides a total phosphorus target endpoint for each pond discussed in this TMDL. The decision of how and when the endpoints are to be applied is, however, still informed by the science behind the development of the response model. As noted above, application of the response model includes specific limitations over the range of observations to specify how this endpoint is to be applied, such as was done for Pond 2 where the recreational standard would have been chosen by the 1997 data but is shown in compliance the 2014 data. The chl-a ratio based TP target for Pond 2 would have been greater than 20 µg/l.

The 2014 data was more extensive than the 1997 data, and appeared to be a narrower range of data with no obvious data outliers. Further, there were flow changes after 1997 that may be better represented by the 2014 data, as it is simply a more current picture of the watershed. Therefore, the TMDL's primary conclusions will be based on modeling and consideration that the 2014 data probably better represents the watershed current condition. Therefore, the 2014 data was considered more reliable and was given greater weight in forming conclusions.

4.0 SOURCE ASSESSMENT

4.1 Analysis of Phosphorus Contributions

The MapShed watershed runoff model was used in combination with the BATHTUB lake response model to develop the Engleville Pond TMDL. This approach usually consists of using MapShed to determine mean annual phosphorus loading to the lake, and BATHTUB to define the extent to which this load must be reduced to meet water quality targets. MapShed incorporates an enhanced version of the Generalized Watershed Loading Function (GWLF) model developed by Haith and Shoemaker (1987) and the RUNQUAL model also developed by Haith (1993). GWLF and RUNQUAL simulate runoff and stream flow by a water-balance method based on measurements of daily precipitation and average temperature. The complexity of the two models falls between that of detailed, process-based simulation models and simple export coefficient models that do not represent temporal variability. The GWLF and RUNQUAL models were determined to be appropriate for this TMDL analysis because they simulate the important processes of concern, but do not have onerous data requirements for calibration. MapShed was developed to facilitate the use of the GWLF and RUNQUAL models via a Map Window interface (Evans, 2009). Appendix B discusses the setup, calibration, and use of the MapShed model for lake TMDL assessments in the Northeast including New York State.

The AVGWLF calibration and development is discussed in Appendix B. There were numerous changes in model parameters from the original Pennsylvania based version. Because of the work that CADMUS had done to recalibrate the original AVGWLF model parameters, to better address New York lakes than the original AVGWLF developed for Pennsylvania, it was decided to model with the CADMUS parameters. DEC used both the 1997 and 2014 data in a MapShed model, as well as updating land use data from the 2001 and the 2011 NLCDs.

The product derived by use of this default annual mean loading and feeding it to the BATHTUB model for a similar annual calculation is a table depicting annual loadings that can be divided by 365 and provide the daily loading values found in most TMDLs. However, due to the seasonal nature of the impairment, in combination with the very short hydraulic residence time for the lake in question, this annual calculation was deemed inadequate. The tremendous seasonal differences is some important watershed parameters is demonstrated by the MapShed charts below showing the average Evapotranspiration and Total Phosphorous overland Runoff Trends for the 25 year modeled period.

Figure 11: 25 Year Average Pond 1 Evapotranspiration Annual Pattern

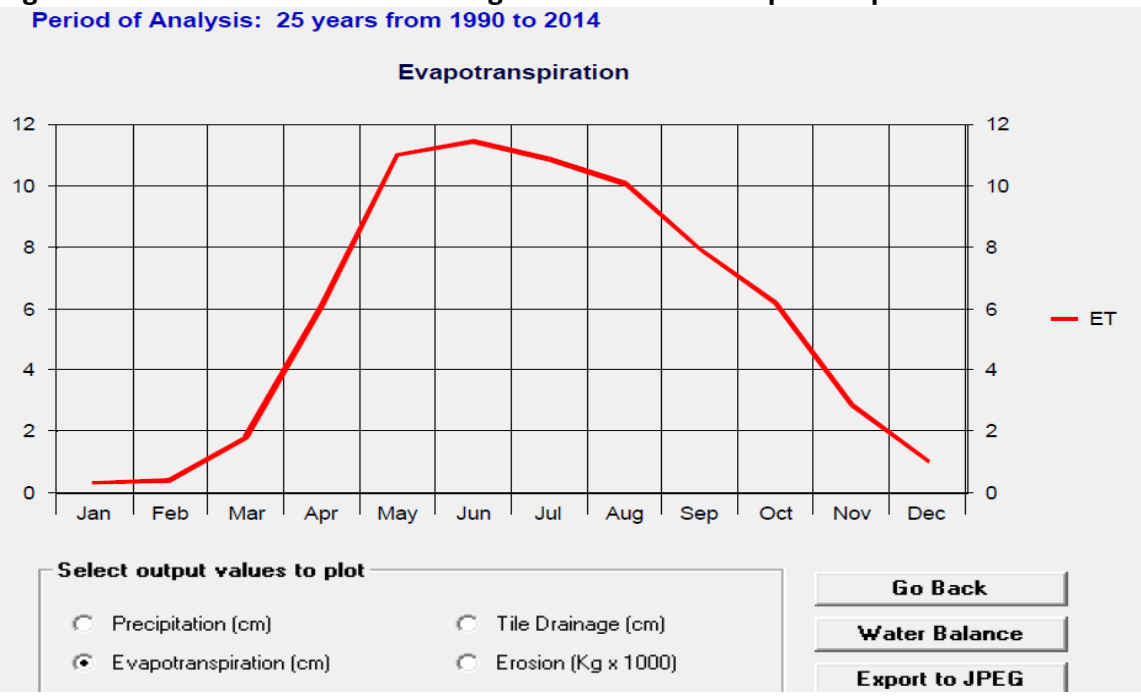
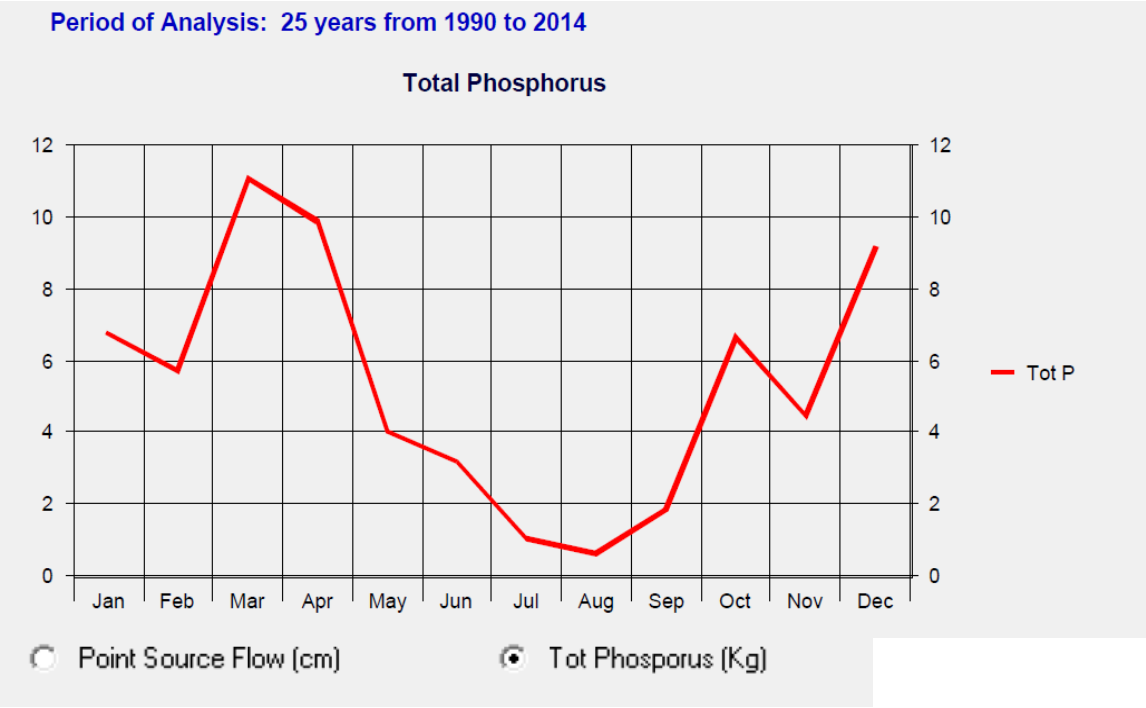


Figure 12: 25 Year Average Pond 1 Phosphorous Overland Runoff Pattern



There is clearly much greater evapotranspiration during the warmest months which results in a much lower overland flow of Total Phosphorous during these same months. For a lake with such

a small hydraulic residence time as Engleville, 0.2 year residence time, this causes overland flow to be proportionally much less important during than if the annual mean overland flow were considered for a lake with a longer residence time.

When the weather data and the resulting watershed nutrient and flow parameters that impact lake loading are fed on a 5 month rolling average basis; the results then become more attuned to the actual lake conditions during summer algal growing season. In addition, the watershed parameters indicate a decreased overland flow during this seasonal growing period. The anoxia in this stratified lake then results in a seasonally higher amount of internal loading as will be discussed further in the Internal Loading discussion later in this chapter. The result is a seasonally greater contribution of internal loading and less overland flow than would be the case if annual mean values were used.

4.2. Sources of Phosphorus Loading

Table 4. Estimated Sources of Phosphorus Loading to Pond 1

Source	Total Phosphorous (lb/yr) 1997&2014 data (DEC-2016)	% Phosphorous Contribution
Agriculture	27.9	12.6
Hay/Pasture	(27.9)	(12.6)
Cropland	(0)	(0)
Forest	5.8	2.6
Open Land	2.0	0.9
Wetland	0.6	0.3
Developed Land	0.5	0.2
Low Density Mixed	(0.5)	(0.2)
High Density Mixed	(0)	(0)
Groundwater	145.4	65.7
Internal Loading	39.2	17.7
TOTAL	**221.4**	**100.0**

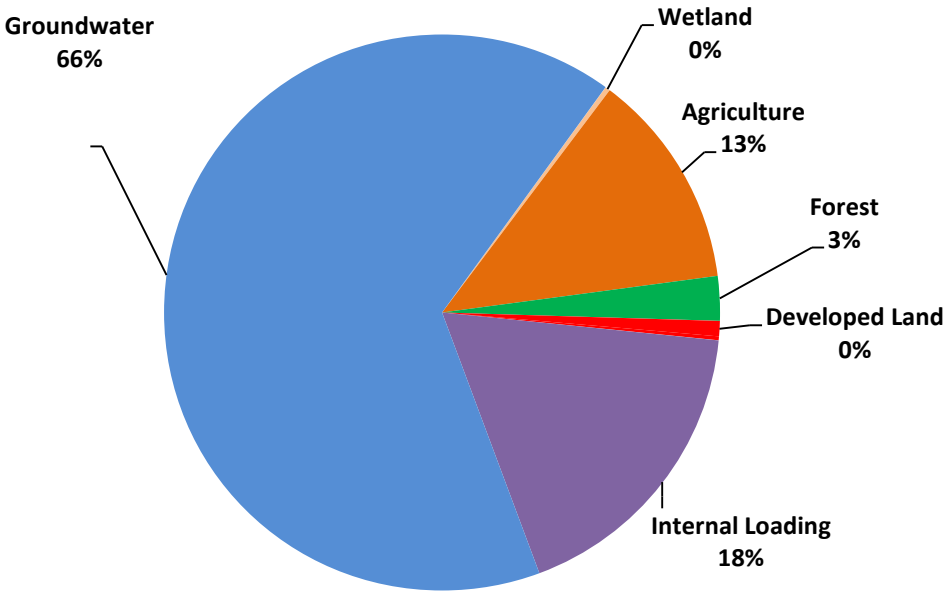
MapShed was used to estimate long-term (1990-2007) mean annual phosphorus loading to Pond 1 to be 224 lb/yr while considering only the 1997 monitoring data for model calibration. The more recent modeling, considering both the 1997 data and the 2014 data, yielded an estimate of 221.4 lb/yr of phosphorous loading. BATHTUB was then used to evaluate the effects of the phosphorous load on the Pond.

The NLCD in 2011 defined the land use as having a slightly more wetland and less forest, but that changed the overall phosphorous loading very little. There was a slight difference in watershed

area delineation using USGS Streamstats, and the land usage acreage changed a few percent according to the National Land Cover Database. There were no new point sources adding to the sectors of phosphorous contributors that would need to be evaluated.

The resulting changes caused MapShed to estimate a different Phosphorous Loading when remodeled with weather data for the years 1990 to 2014, considering the 2011 NLCD, and calibrating to DEC's 2014 monitoring data. (See Table 4 above) An average of 221.4 lb/yr total phosphorous is now estimated as entering Pond 1 in the last column of Table 4, and as shown in Figure 13. Appendices B and C provide additional models results from MapShed and Bathtub respectively.

Figure 13. Estimated Sources of Total Phosphorus Loading to Pond 1



The pond concentration decrease may be reflective of watershed changes that have already occurred in the intervening 17 years in terms of a decrease in phosphorous application to the agricultural land. An elution of phosphorous from the agricultural portion may have been caused in part by the last decade being significantly wetter than the 1990-2007 average precipitation, and this may have also contributed to a different portion of the phosphorous being contributed by groundwater. This may have the effect of producing both more external loading by causing greater surface and subsurface flow in the watershed, as well and decreasing the internal loading

by greatly increasing the flush our rate of this small residence time lake. It is hoped that this also correlates to an improving overall watershed in terms of phosphorus balances.

4.2.1. *Agricultural Runoff*

Agricultural land encompasses 140.8 acres (23.6%) of the lake drainage basin and is mostly hay and pasture land. Overland runoff from agricultural land is estimated to contribute 27.9 lb/yr of external phosphorus loading to Pond 1, which is 12.6% of the total phosphorus loading.

In addition to the contribution of phosphorus to the lake from overland agriculture runoff, additional phosphorus originating from agricultural lands is leached in dissolved form from the surface and transported to the lake through subsurface movement via groundwater. The process for estimating subsurface delivery of phosphorus originating from agricultural land is discussed in the Groundwater Seepage section (below). Phosphorus loading from agricultural land originates primarily from soil erosion and the application of manure and fertilizers. Implementation plans for agricultural sources will require voluntary controls applied on an incremental basis.

In addition to the contribution of phosphorus to the reservoir from overland agriculture runoff, additional phosphorus originating from agricultural lands is leached in dissolved form from the surface and transported to the reservoir through subsurface movement via groundwater. The process for estimating subsurface delivery of phosphorus originating from agricultural land is discussed in the Groundwater Seepage section (below). Phosphorus loading from agricultural land originates primarily from soil erosion and the application of manure and fertilizers. Implementation plans for agricultural sources will require voluntary controls applied on an incremental basis.

4.2.2. *Urban and Residential Development Runoff*

Developed land comprises 12.4 acres (2.1%) of the lake drainage basin. Stormwater runoff from developed land contributes about 0.53 lb/yr of phosphorus to Pond 1, which is less than 0.23% of the total phosphorus loading to the lake.

In addition to the contribution of phosphorus to the lake from overland urban runoff, additional phosphorus originating from developed lands is leached in dissolved form from the surface and transported to the lake through subsurface movement via groundwater. The process for estimating subsurface delivery of phosphorus originating from developed land is discussed in the Groundwater Seepage Section (below).

Phosphorus runoff from developed areas originates primarily from human activities, such as fertilizer applications to lawns. Shoreline development, in particular, can have a large phosphorus loading impact to nearby waterbodies in comparison to its relatively small percentage of the total land area in the drainage basin.

4.2.3. *Forest Land Runoff*

Forested land comprises 388 acres (65.1%) of the lake drainage basin. Runoff from forested land is estimated to contribute about 5.8 lb/yr of phosphorus loading to Pond 1, which is about 2.6% of the total phosphorus loading to the lake. Phosphorus contribution from forested land is considered a component of background loading. Additional phosphorus originating from forest land is leached in dissolved form from the surface and transported to the lake though subsurface movement via groundwater. The process for estimating subsurface delivery of phosphorus originating from forest land is discussed in the Groundwater Seepage Section (below).

4.2.4. *Open Land*

Although not identified as a category for this watershed in the 2001 NLCD**,** the 2011 NLCD identified 12.4 acres as "Open Land". This category is intended to depict such land types similar to "open range" or "grassland". These essentially "natural" areas are typically not cultivated or heavily pastured. Apparently the 2011 NLCD placed some acreage in this category that had previously been considered some other categories of land use in the 2001 NLCD.

The "Open Land" category comprises 12.4 acres (2.1%) of the lake drainage basin. Runoff from this open land contributes about 2.0 lb/yr of phosphorus loading to Pond 1, which is about 0.9% of the total phosphorus loading to the lake.

4.2.5. *Groundwater Seepage*

In addition to nonpoint sources of phosphorus delivered to the lake by surface runoff, a portion of the phosphorus loading from nonpoint sources seeps into the ground and is transported to the lake via groundwater. Groundwater is estimated to transport 145.4 lb/yr (65.7%) of the total phosphorus loading to Pond 1. With respect to groundwater, there is typically a small "background" concentration owing to various natural sources.

The GWLF manual provides estimated background groundwater phosphorus concentrations for ≥90% forested land in the eastern United States, which is 0.006 mg/l. Primarily agricultural watersheds have values of 0.104 mg/l. Intermediate values are also reported. The AVGWLG model calibrated for the Northeast estimates a typical groundwater phosphorous concentration of 0.010 mg/l for lakes in this region. This version of AVGWLF calibrated for lakes in the Northeast is discussed in Appendix B.

Given the 25 years of weather data modeled, the resulting average phosphorous loading from groundwater is estimated to be 145.4 lb/yr. It is estimated that this 145.4 lb/yr of phosphorus transported to the lake through groundwater originates from; agriculture (110.4 lb/yr), natural resources of 25.2 lb/yr [combining forest (22.7 lb/yr) and wetland (2.5 lb/yr)], open land (7.8 lb/yr) and light development residential (2.1 lb/yr). Table 5, land usage data based on the 2011 National Land Cover Database, summarizes this information.

Table 5. Sources of Phosphorus Transported in the Subsurface via Groundwater

	Total Phosphorus (lb/yr)	% of Total Groundwater Load
Natural Sources	**25.2**	**17.3%**
Forest	*22.7*	*15.6%*
Wetland	*2.5*	*1.7%*
Developed Land	**2.1**	**1.4%**
Agricultural Land	**110.3**	**75.9%**
Open Land	**7.8**	**5.4%**
TOTAL	**145.4**	**100%**

4.2.6. *Internal Loading*

Pond 1 has been exposed to nutrient loading that is much higher than its assimilative capacity. Over time, much of this excess phosphorus has been deposited into the bottom sediments. Internal phosphorus loading from lake sediments can be an important component of the phosphorus budget for lakes, especially shallow lakes. Excess phosphorus in a lake's bottom sediments is available for release back into the water column when conditions are favorable for nutrient release; such conditions can include re-suspension of sediments by wind mixing or rough fish activity (e.g., feeding off bottom of lake), sediment anoxia (i.e., low dissolved oxygen levels near the sediment water interface), high pH levels, die-offs of heavy growths of curly-leaf pond weeds, and other mechanisms that result in the release of poorly bound phosphorus.

In order to estimate internal loading in Pond 1, the data from 1997 and 2014 were used to calibrate the BATHTUB Model discussed in Section 5 and Appendix C. The monitored results for the lakes change in the intervening 17 years, as did the weather patterns. Measured lake concentrations were less, and weather patterns were much wetter in the last decade on average.

The calibration of the two years, in combination with the model results, converges to yield an estimate of lake internal loading of 0.17 mg/sq meter/day release of phosphorous and a lake bottom area of 29.54 acres. As per the MapShed modeling directions, this value of 0.17 was then multiplied by a factor of 2.4 to account for the 5 month averaging period. Using this model calibrated prediction, on average, internal loading is estimated to contribute about 39.2 lb/yr of phosphorus to Pond 1, which is about 17.7% of the total phosphorous loading to the lake.

4.2.7. *Other Sources*

Atmospheric deposition, wildlife, waterfowl, and domestic pets are also potential sources of phosphorus loading to the reservoir. All of these small sources of phosphorus are incorporated into the land use loadings as identified in the TMDL analysis (and therefore accounted for). Further, the deposition of phosphorus from the atmosphere over the surface of the reservoir is accounted for in the reservoir model, though it is small in comparison to the external loading to the reservoir.

5.0 DETERMINATION OF LOAD CAPACITY

5.1 Lake Modeling Using the BATHTUB Model

BATHTUB was used to define the relationship between phosphorus loading to the lake and the resulting concentrations of total phosphorus in the lake. The U.S. Army Corps of Engineers' BATHTUB model predicts eutrophication-related water quality conditions (e.g., phosphorus, nitrogen, chlorophyll-a, and transparency) using empirical relationships previously developed and tested for reservoir applications (Walker, 1987). BATHTUB performs steady-state water and nutrient balance calculations in a spatially segmented hydraulic network. Appendix C discusses the setup, calibration, and use of the BATHTUB model.

5.2 Linking Total Phosphorus Loading to the Numeric Water Quality Target

In order to estimate the loading capacity of the lake, simulated external phosphorus loads from MapShed and calculated internal loads were used to drive the BATHTUB model to simulate water quality in Pond 1. MapShed was used to derive a mean annual external phosphorus loading to the lake for the period 1990-2014. Using this external load and the calculated internal load as inputs, BATHTUB was used to simulate water quality in the lake. The results of the BATHTUB simulation were compared against the average of the lake's observed summer mean phosphorus concentrations for the years 1997 and 2014. Year-specific loading was also simulated with MapShed for external loading and calculated for internal loading. The loading was run through BATHTUB and compared against the observed summer mean phosphorus concentration for years with observed in-lake data. Because observed lake concentration data were only available for years 1997 and 2014, the ratio of internal loading concentration to external loading concentration in 1997 and 2014 was used to predict the internal loading concentration in other years (including the long term average year) assuming the ratio remained constant over time.

The combined use of MapShed and BATHTUB provides a decent fit to the observed Phosphorus and chl-a data available for Pond 1 in 1997 and 2014 (See Figures 14 and 15).

Figure 14. Observed vs. Simulated Summer Mean Epilimnetic Total Phosphorus Concentrations (µg/l) in Pond 1 based on 25 years of weather data.

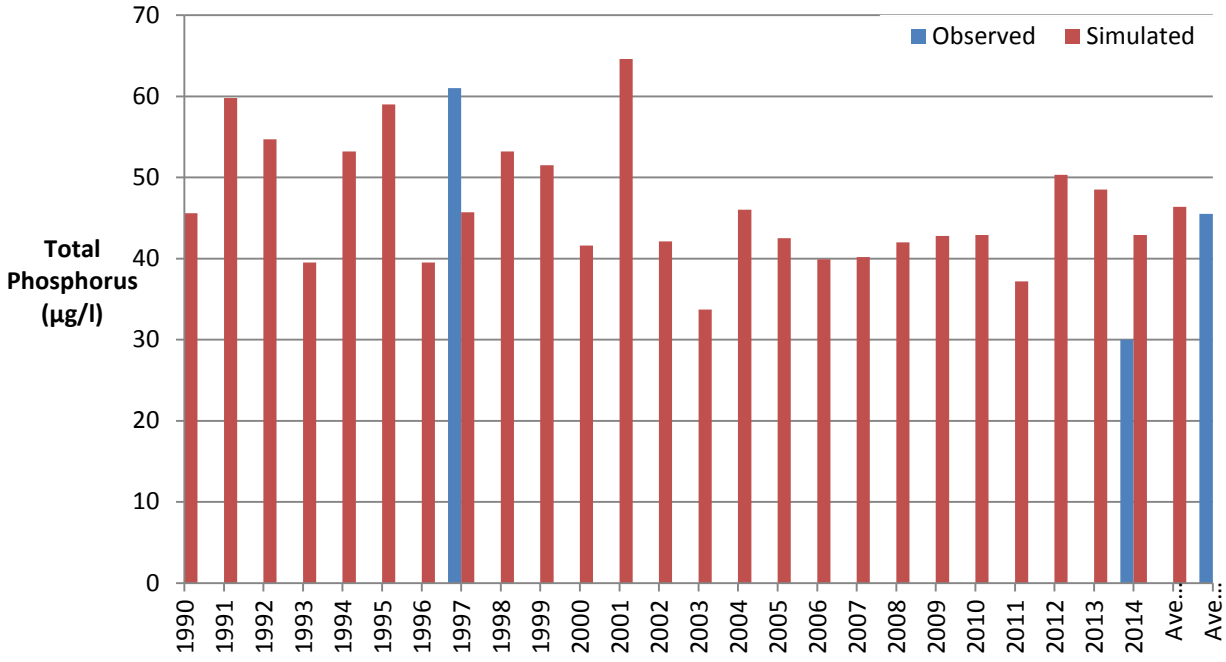
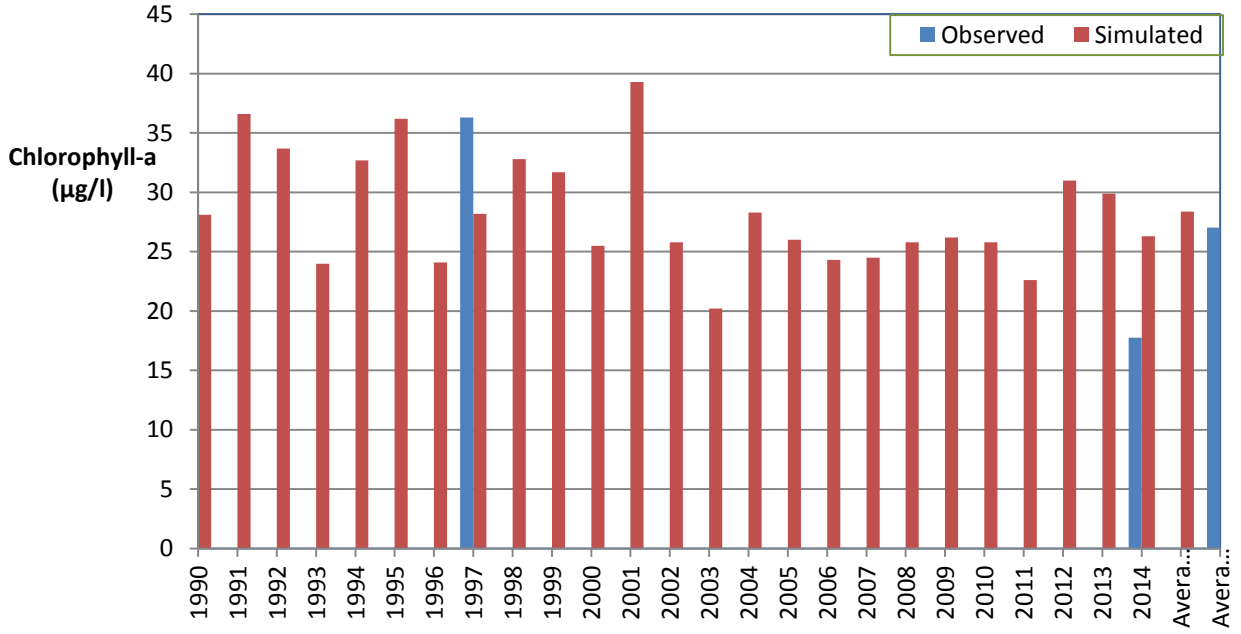


Figure 15. Observed vs. Simulated Summer Mean Epilimnetic Chlorophyll-a Concentrations (µg/l) in Pond 1 based on 25 years of weather data.



In Figure 15, the combined use of MapShed and BATHTUB also provides a statistically acceptable fit to the observed data available for Pond 1 in 1997 and 2014 chl-a concentrations.

These tables do show projected lower concentrations during the latter decade than in the first decade projections. This may possibly be due to the higher precipitation in the last ten years causing a quicker flushing out of the Pond. The average TP from 1990 to 2002 was projected to be 50.0 µg/l, and the average TP from 2003 to 2014 was projected to be 42.4 µg /l. Actual measured concentrations decreased from 61 µg/l in 1997 to 30 µg/l in 2014. The additional decrease may have been influenced by a decrease in activity on the watershed farmland.

Similarly, the projected average chl-a from 1990 to 2002 was 30.7 µg /l and from 2003 to 2014 was projected to be 25.9 µg /l. This also roughly corresponded to a decrease of real measured chl-a from 31 to 17.75 µg/l in Pond 1.

Meeting Recreational Guidance values: The BATHTUB model was used as a "diagnostic" tool to derive the total phosphorus load reduction required to achieve the phosphorus target 12 µg/l. The loading capacity of Pond 1 was determined by running BATHTUB iteratively, reducing the concentration of the drainage basin phosphorus load (which in turn reduced the internal load) until model results demonstrated attainment of the water quality target. As external loading is reduced, internal loading is also reduced; thus the percent reduction in internal loading is estimated to be proportional to the percent reduction in external loading. The maximum concentration that results in compliance with the TMDL target for phosphorus is used as the basis for determining the lake's loading capacity. This concentration is converted into a loading rate using simulated flow from MapShed.

The maximum annual phosphorus load (i.e., the annual TMDL) that will maintain compliance with the water quality goal of 12 µg/l in Pond 1 is a mean annual load of 51.8 lb/yr. The daily TMDL of 0.142 lb/d was calculated by dividing the annual load by the number of days in a year. Lakes and reservoirs store phosphorus in the water column and sediment, therefore water quality responses are generally related to the total nutrient loading occurring over a year or season. For this reason, phosphorus TMDLs for lakes and reservoirs are generally calculated on an annual or seasonal basis. The use of annual loads, versus daily loads, is an accepted method for expressing nutrient loads in lakes and reservoirs. This is supported by EPA guidance such as *The Lake Restoration Guidance Manual* (USEPA 1990) and *Technical Guidance Manual for Performing Waste Load Allocations, Book IV, lakes and Impoundments, Chapter 2 Eutrophication* (USEPA 1986). While a daily load has been calculated, it is recommended that the annual loading target be used to guide implementation efforts since the annual load of total phosphorus as a TMDL target is more easily aligned with the design of best management practices (BMPs) used to implement nonpoint source and stormwater controls for lakes than daily loads.

6 POLLUTION LOAD ALLOCATION

The objective of a TMDL is to provide a basis for allocating acceptable loads among all of the known pollutant sources so that appropriate control measures can be implemented and water quality standards achieved. Individual waste load allocations (WLAs) are assigned to discharges regulated by State Pollutant Discharge Elimination System (SPDES) permits (commonly called point sources) and unregulated loads (commonly called nonpoint sources) are contained in load allocations (LAs). A TMDL is expressed as the sum of all individual WLAs for point source loads, LAs for nonpoint source loads, and an appropriate margin of safety (MOS), which takes into account uncertainty.

Equation 1. Calculation of the TMDL

(Equation 1): $$TMDL = \sum WLA + \sum LA + MOS$$

6.1 Wasteload Allocation (WLA)

There are no permitted wastewater treatment plant dischargers or Municipal Separate Storm Sewer Systems (MS4s) in the Pond 1 basin, therefore the WLA is set at zero.

6.2 Load Allocation (LA)

The LA is set at 46.6 lb/yr. Nonpoint sources that contribute total phosphorus to Pond 1 on an annual basis include loads from developed land and agricultural land. Table 6 lists the current loading for each source and the load allocation needed to meet the TMDL; Figure 16 provides a graphical representation of this information. Phosphorus originating from natural sources (including forested land, wetlands, and stream banks) is assumed to be a minor source of loading that is unlikely to be reduced further and therefore the load allocation is set at current loading. Internal loads were allocated under the assumption that the internal load will decrease proportionally to decreases in external loads. The bulk of the reductions need to come from agricultural land, which accounts for most of the estimated external load in the watershed.

6.3 Margin of Safety (MOS)

The margin of safety (MOS) can be implicit (incorporated into the TMDL analysis through conservative assumptions) or explicit (expressed in the TMDL as a portion of the loadings) or a combination of both. For the Engleville Pond TMDL, the MOS is explicitly accounted for during the allocation of loadings. An implicit MOS could have been provided by making conservative assumptions at various steps in the TMDL development process (e.g., by selecting conservative model input parameters or a conservative TMDL target). However, making conservative assumptions in the modeling analysis can lead to errors in projecting the benefits of BMPs and lake responses. Therefore, the recommended method is to formulate the mass balance using the best scientific estimates of the model input values and keep the margin of safety in the "MOS"

term. The TMDL contains an explicit margin of safety corresponding to 10% of the loading capacity, or 5.2 lb/yr. The MOS can be reviewed in the future as new data become available.

Table 6. Total Annual Phosphorus Load Allocations for Pond 1*

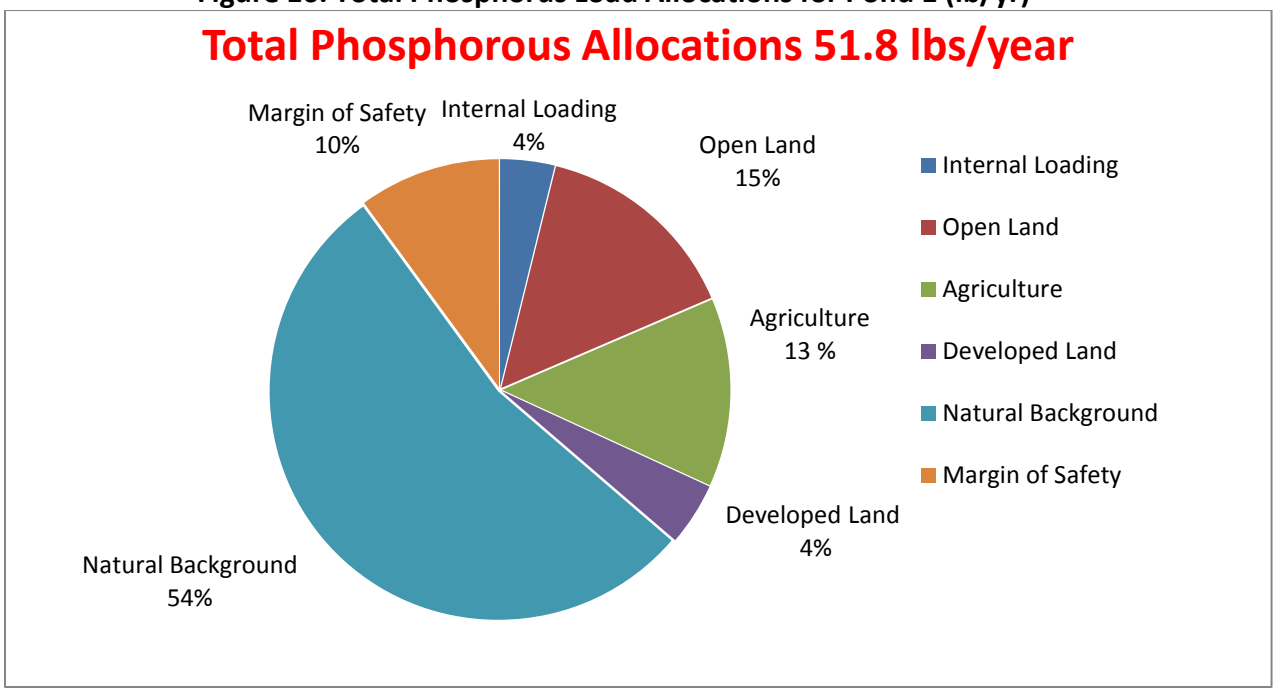
Source	Total Phosphorus Load (lbs/year)			
	Current	Allocated	Reduction	% Reduction
Agriculture**	138.3	6.9	131.4	95%
Developed Land**	2.6	2.3	0.3	12%
Forest, Wetland, Stream Bank, and Natural Background**	31.6	27.8	3.8	12%
Open Land	9.8	7.6	2.2	22%
Internal Loading	39.2	2.0	37.2	95%
LOAD ALLOCATION	**221.5**	**46.6**	**174.9**	**79%**
Point Sources	0	0	0	0%
WASTELOAD ALLOCATION	0	0	0	0%
LA + WLA	**221.5**	**46.6**	**174.9**	**79%**
Margin of Safety	-	5.2	-	-
TOTAL	**221.5**	**51.8**	**-**	**-**

* Values reported in Table 6 are annually integrated. Daily equivalent values are provided in Appendix-D.
** *Includes phosphorus transported through surface runoff and subsurface (groundwater)*

The values in Table 6 are based on calculations using weather data from 1990 to 2014 and two year of monitoring data used for model calibration (1997 and 2014). The 1997 and 2014 data years are not of completely comparable value due to factors varying such as;

- the more recent monitoring shows lower concentrations in the Reservoir,
- recent years have had more precipitation and higher temperatures (evapotranspiration),
- and the actual waterflow/hydraulic connections is different between the ponds,
- The MapShed model does not allot a way to distinguish the difference in the farm TP in 1997 from the farm TP in 2014 due to a lack of data pertaining to farm animal usage and manure application. Agriculture differences in particular are difficult to quantify, therefore, but future BMP initiatives and farm practices are still thought to be a potential source of future TP reductions.

Figure 16. Total Phosphorus Load Allocations for Pond 1 (lb/yr)



6.4 Critical Conditions

TMDLs must take into account critical environmental conditions to ensure that the water quality is protected during times when it is most vulnerable. Critical conditions were taken into account in the development of this TMDL. In terms of loading, spring runoff periods are considered critical because wet weather events transport significant quantities of nonpoint source loads to reservoirs. However, the water quality ramifications of these nutrient loads are most severe during middle or late summer. Therefore, BATHTUB model simulations were compared against observed data for the summer period only. Furthermore, AVGWLF and Mapshed take into account loadings from all periods throughout the year, including spring loads.

6.5 Seasonal Variations

Seasonal variation in nutrient load and response is captured within the models used for this TMDL. In BATHTUB, seasonality is incorporated in terms of seasonal averages for summer. Seasonal variation is also represented in the TMDL by taking 25 years of daily precipitation data when calculating runoff through MapShed, as well as by estimating septic system loading inputs (when this is applicable) based on residency (i.e., seasonal or year-round). This takes into account the seasonal effects the lake will undergo during a given year.

Examples of Seasonal Trends were shown in Section 4 Figures 11 and 12. In addition to the seasonal trends in these charts, the comparatively more rapid hydraulic turnover rate of Pond 1 and the potential for the development of algal growth in this drinking water source during the summer months are considerations require a conservative assessment averaging factors during the sensitive algal production season.

7 IMPLEMENTATION

One of the critical factors in the successful development and implementation of TMDLs is the identification of potential management alternatives, such as BMPs in collaboration with the involved stakeholders. DEC, in coordination with these local interests, will address the sources of impairment, using regulatory and non-regulatory tools in that watershed, matching management strategies with sources, and aligning available resources to effect implementation.

7.1 Reasonable Assurance for Implementation

Meeting the loading limits specified in this TMDL will require reductions from nonpoint sources and the reduction of internal loading. Implementation will rely upon a blend of existing programs which have proven successful in reducing loads from the targeted source sectors and innovative solutions based on proven science to address internal loading.

- There is a portion of the needed reduction that appears to be occurring naturally due to some decrease in the intensity of the agricultural contribution and a natural attenuation of the flow from this portion of the watershed.

- For internal loading mitigation there are a number of options offered by science and discussed somewhat in the DEC Web Publication 'Diet for a Small Lake'. The methods for phosphorous removal and/or other control algal growth techniques include a variety of physical removal, chemical treatment and biomanipulation options.

- For any increase in the agricultural source sector reductions, implementation relies upon voluntary installation of BMPs if reduction from land usages changes prove insufficient.

- A Village Watershed Implementation Plan (WIP), as referenced in Subsection 7.1.10 is encouraged. It is expected that any such WIP would be flexible enough to modify phosphorous control methods if monitoring results warrant it.

Financial assistance and resource conservation may provide incentives for participation in nonpoint source reduction programs. Additional incentive for reduction from both categories is derived from the need to use this water source for drinking water, and to maintain attain compliance with the applicable drinking water standards.

Reasonable Assurance of attainment of phosphorous targets as well as compliance with the optimum drinking water goal will include a Margin of Safety, and follow up monitoring will occur that will trigger additional TP reduction response if needed. In addition to the nonpoint source incentives and BMP options, the 'Diet for a Small Lake' provides multiple corrective options that may be attempted to correct internal loading from a lake the size of Pond 1.

7.1.1 The Role of Natural Improvement by Attenuation

A comparison of the 1997and the 2014 DEC LCI monitoring indicates a reduction has occurred of total phosphorous (TP) and corresponding chl-a concentrations in both ponds.

- The Pond 1 TP decreased from a mean value of 54 µg/l to 30 µg/l, and the chl-a decreased from 31 to 12 µg/l.
- The Pond 2 TP decreased from a mean value of 58 µg/l to 16 µq/l, and the chl-a decreased from 18 to 2.8 µg/l.

From 1997 to 2014, the most significant obvious change in the watershed that would impact both ponds is the decrease in intensity of agricultural application of nutrients. There is no longer live stock on the pasture for some undetermined number of years (now hay fields), and there are of row crops (which also implies there may be less fertilizer application.)

This strongly implies that the change in the watershed has had some positive impact on the water quality. This attenuation of runoff concentrations may well continue since the migration of nutrients from the fields naturally takes time to achieve equilibrium with the decreased land application. There may still be more improvement.

(In addition, Pond 2 appears to have been impacted by the severing of a tile hydraulic connection to the main watershed. There is less activity in the small natural watershed that it now has. It is not clear if the present piping in this regard is intended to be permanent.)

7.1.2 Recommended Phosphorous Management Strategies for Internal Loading

Internal Loading is considered to be a significant source of the seasonally released Phosphorous, promoting the growth of an excess of algal material. This phenomena is most pronounced in stratified lakes that have accumulated an internal loading of the limiting nutrient. CADMUS found the Osgood Index of Pond 1 to be 5.8, and literature stated that Osgood Numbers above 6 when there is an anoxic zone, indicates that there is hypolimnetic transfer of Phosphorous during the growing season. This assessment by CADMUS appears consistent with the DEC water testing in 2014 that found phosphorous in the hypnolimnia to be close to double the concentration as the concentration in the surface water. It should be noted that hypnolimnia vs surface comparisons were added to the sampling regime after stratification was documented, and thus was measured 3 times in Pond 2 and once in Pond 1 in 2014.

For lakes with excessive algal growth resulting from too great of a phosphorous loading there are three categories of mitigation measures that are referenced in DEC's web publication "Diet for a Small Lake": Physical Removal of Phosphorous, Chemical Treatment, and Biological Manipulation.

7.1.3 Physical Removal of Internal Phosphorous

Lakes in New York State may stratify in summer and winter. When a lake is stratified, colder, heavier water sinks to the bottom and lighter, warmer water rises to the top. This creates distinct layers that do not mix easily. In relatively deep lakes, these layers become less distinct during the spring and fall months and mix together in the process known as destratification or turnover.

During stratification, the bottom water, or hypolimnion, receives little or no exposure to the atmosphere, which can lead to oxygen depletion. This is usually much more severe in the summer stratification, during the four warmest months of the year. The hypolimnion is the location for; reactions with the sediment, degradation of organic materials that have settled out of the water column, and significant biological activity. This combination of oxygen depletion and chemical reactions can lead to deoxygenated, high-nutrient conditions.

During the summer 2014 DEC testing of Pond 1, DO gradients were measured and stratification was documented as occurring in both ponds. There were also measurements of Phosphorous at multiple layers during a few of these weeks, three times in Pond 2 and once in Pond 1. Both the average ratio in Pond 2 and the single week of measurement's ratio in Pond 1, showed that Phosphorous in the bottom layer during this stratified time was about twice as high as the Phosphorous concentration in the surface water at the same time.

Three of the principles that may be used to physically control algal blooms are 1) Artificial Circulation, 2) Hypolimnion Aeration, and 3) Hypolimnetic Withdrawal:

1. Artificial Circulation Principle. This principle involves either the injection of compressed air from a pipe or diffuser into the hypolimnion, or water may be moved with impellors. Either method is intended to eliminate thermal stratification and improve the flow and movement of water within a lake. This may lower algae levels by inhibiting the release of phosphorus from oxygen-depleted bottom sediments, thud decreasing the internal loading contribution to the phosphorous concentration.

2. Hypolimnion Aeration Principle. This principle is intended to increase the oxygen concentration of the hypolimnion such that the hypolimnion has sufficient oxygen that phosphorus release from the oxygen-depleted bottom sediments will be minimized. With less phosphorous, it is then hoped that algal production would be minimized or lessened. This aeration of the hypolimnion is accomplished either by the infusion of oxygen into the hypolimnion either or pumping lower stagnant water to a higher elevation and provide some exposure to the atmosphere and thus increase its oxygen and decrease its methane and carbon dioxide.

3. Hypolimnetic Withdrawal Principle. This method makes use of the higher concentration of phosphorous in the anoxic hypolimnion and directly removed phosphorous laden water from this part of the lake with a pipe or siphon placed along the bottom of the lake.

Unlike the other measures stated in the Diet For A Small Lake, there is no New York state case study to reference since this has not been done in this state for this purpose yet.

Direct aeration of the hypolimnion can also have the benefit of increasing the oxygen.

DEC would need to be consulted in advance about the specifics of any plan using this alternative:

- To determine if there were any additional stream disturbance or discharge concerns to be addressed, and
- To determine if there are any dam safety issues or mitigations pertaining to the water removal method.

7.1.4 Recommended Phosphorus Management Strategies for Developed Lands.

Developed lands represent a minor part of the total load delivered to the lake. However, minor reductions may still be realized through the present Nonpoint Source Management Program. There are several measures, which if implemented in the watershed, could directly or indirectly reduce phosphorus loads.

- Public education regarding:
    - Lawn care, specifically reducing fertilizer use or using phosphorus-free products now commercially available. The NYS Household Detergent and Nutrient Runoff Law restricts the sale and application of fertilizers containing phosphorus.
    - Cleaning up pet waste.
- Management practices to address any significant existing erosion sites.
- Construction site and post construction stormwater runoff control ordinance, inspection and enforcement programs.
- Pollution prevention practices for road and ditch maintenance.
  Management practices for the handling, storage and use of deicing products.

7.1.5 Recommended Phosphorus Management Strategies for Agricultural Runoff

The New York State Agricultural Environmental Management (AEM) Program was codified into law in 2000. Its goal is to support farmers in their efforts to protect water quality and conserve natural resources, while enhancing farm viability. AEM provides a forum to showcase the soil and water conservation stewardship farmers provide. It also provides information to farmers about Concentrated Animal Feeding Operation (CAFO) regulatory requirements to assure compliance. Details of the AEM program can be found at the New York State Soil and Water Conservation Committee (SWCC) website, http://www.nys-soilandwater.org/aem/index.html.

Using a voluntary approach to meet local, state, and national water quality objectives, AEM has become the primary program for agricultural conservation in New York. It also has become the

umbrella program for integrating/coordinating all local, state, and federal agricultural programs. For instance, farm eligibility for cost sharing under the SWCC Agricultural Non-point Source Abatement and Control Grants Program is contingent upon AEM participation.

AEM core concepts include a voluntary and incentive-based approach, attending to specific farm needs and reducing farmer liability by providing approved protocols to follow. AEM provides a locally led, coordinated and confidential planning and assessment method that addresses watershed needs. The assessment process increases farmer awareness of the impact farm activities have on the environment and by design, it encourages farmer participation, which is an important overall goal of this implementation plan.

The AEM Program relies on a five-tiered process:

Tier 1 – Survey current activities, future plans and potential environmental concerns.
Tier 2 – Document current land stewardship; identify and prioritize areas of concern.
Tier 3 – Develop a conservation plan, by certified planners, addressing areas of concern tailored to farm economic and environmental goals.
Tier 4 – Implement the plan using available financial, educational and technical assistance.
Tier 5 – Conduct evaluations to ensure the protection of the environment and farm viability.

Schoharie County Soil and Water Conservation District should continue to implement the AEM program on farms in the watershed, focusing on identification of management practices that reduce phosphorus loads. These practices would be eligible for state or federal funding and because they address a water quality impairment associated with this TMDL, should score well.

Tier 1 could be used to identify farmers that for economic or personal reasons may be changing or scaling back operations, or contemplating selling land. These farms would be candidates for conservation easements, or conversion of cropland to hay, as would farms identified in Tier 2 with highly-erodible soils and/or needing stream management. Ideally, Tier 3 would include a Comprehensive Nutrient Management Plan with phosphorus indexing at the appropriate stage in the planning process. Additional practices could be fully implemented in Tier 4 to reduce phosphorus loads, such as conservation tillage, stream fencing, rotational grazing and cover crops. Also, riparian buffers reduce losses from upland fields and stabilize stream banks in addition to reducing load by taking land out of production.

7.1.6 Reducing internal phosphorus loading with inactivants.

Phosphorus Precipitation and Inactivation Principle. Phosphorus precipitation uses a chemical agent, such as alum, to remove phosphorus from the water column. Phosphorus deactivation works by sealing the bottom sediments to prevent the release of phosphorous to the over lying water with low oxygen concentrations.

7.1.7 Algae control with chemicals.

Chemical control issues in a number of lakes in New York State already in order to either control algal growth with algaecides or to control algal growth by decreasing the availability of the nutrients in the lake. The use of any chemical addition method one always has to assess the potential for certain toxicity concerns and consider that there may be regulatory requirements pertaining to the chemical application.

Algaecides are generally copper-based chemicals used to kill algae cells, and to reduce the use impairments associated with excessive algal growth. Copper Sulfate is the most common algaecide and one of the most popular algae control techniques. There are, however, a variety of copper based algaecides that may be chose for various algal problems, and there also conflicting studies on the potential toxic impacts of copper based algaecides on the benthic organisms in lakes where these have been applied. Algaecides may be beneficial in treating the symptoms of eutrophication, but will not result in the attainment of the required phosphorous targets.

7.1.8 Biological Manipulation.

Biomanipulation is abroad term that describes any biological introduction to an ecosystem for the purpose of shifting ecological conditions to the advantage of a desired species or lake condition, or enhance recreational conditions. Biomanipulation can generally be divided into two categories;

- Stocking specific organisms, usually fish, to enhance zooplankton grazing, which will reduce algae populations; and
- Removal of specific organisms, usually bottom-dwelling fish, to enhance water clarity.

Other concepts mentioned in Diet for a Small Lake (available on the DEC website) include:

- Aeration to physically mix the stratified water and thus break up the anoxic condition causing release of the Phosphorous from the sediment.

- Aeration to increase oxygen in the hypolimnion without destroying stratification.

- Placement of barley straw, a material found to counteract the additional Phosphorous that may occur from this internal loading release (*CAUTION should be exercised if this option is chosen due to the possibility that barley straw might legally be viewed as an algaecide and need permits for algaecides.*). When it works, apparently the byproducts that result from the decomposition of the barley straw inhibit algal growth.

- Chemical treatments are mentioned, but must be evaluated for potential unwanted toxic impacts to wildlife and people. DEC and DOH should be consulted prior to use of either of these options.

- Dredging may remove this sediment to address internal loading, but this can be expensive and may have to be repeated and would require permitting.

7.1.9 Erosion Control Measures.

As stated in the Watershed Characterization Section there is a Class A tributary that flows Pond 1's west shore. The stream runs about 2.5 km, originating near Palmer Road and ends at an unnamed dirt drive and wetland area to the west of Pond 1. At times the stream is dry or nearly dry, and during significant rain events there is flow from the stream, and down the dirt road that washes into the main wetland area to the west of the pond. (See Figure 7)

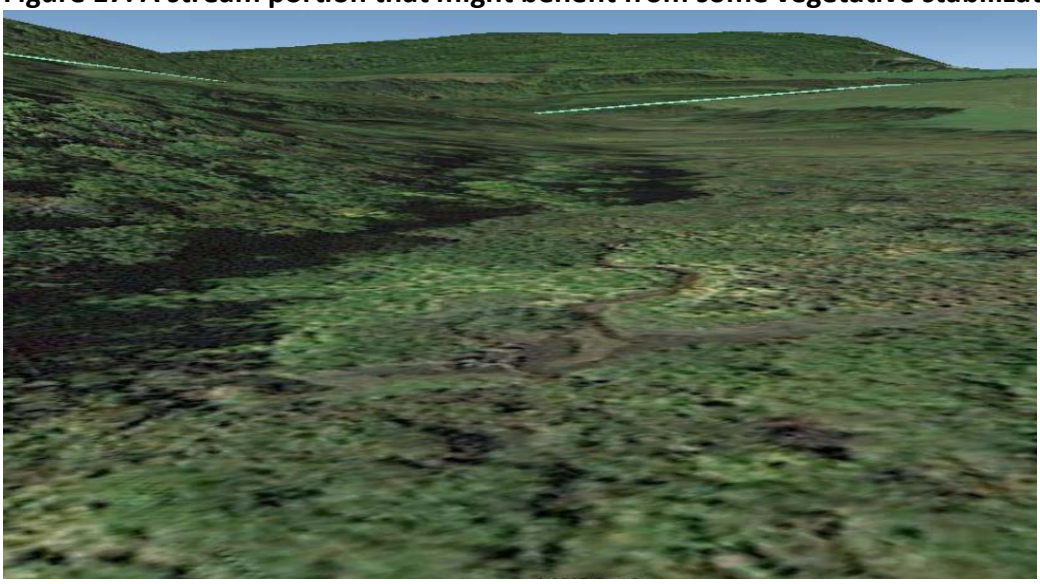
In addition to the agricultural areas previously mentioned, the small stream and the dirt road may be sources of phosphorous contributing sediment during rainfall events. As such it is likely that erosion control efforts on these two areas would likely reduce the actual phosphorous loading to the pond. This contribution is difficult to quantify, and is included in the overland flow estimate

7.1.10 Stream Stabilization

It is still possible that stream stabilization efforts may assist in reducing this phosphorous contributor, and is one of the options that the Village may select from in determining the menu of mitigation measures. Although some erosion is a natural occurrence and expected of any stream, unstable streambeds result in higher amounts of erosion sediment material be transported by the stream to the receiving pond or other waterbody downstream. Any excess of this sediment material that could have been prevented if the stream banks were stabilized will result in a similar excess of Phosphorous being transported along with the sediment to the receiving waterbody. Since there appear to be some areas that are have minimal vegetative cover and appear susceptible to erosion, there is likely to be benefit from some stream stabilization efforts.

Dramatic increases in stormwater runoff through the stream channel may cause accelerated streambank erosion (the process of a stream seeking to reestablish a stable size and pattern due to an external change). An increase in runoff within a stream channel will result in the stream channel adjusting to the additional flow, which will increase streambank erosion. Any land use changes in a watershed, such as clearing land or development, can increase stream bank erosion. The damage or removal of streamside vegetation reduces bank stability and can cause an increase in stream bank erosion. A degraded streambed results in higher and often unstable, eroding banks.

Figure 17. A stream portion that might benefit from some vegetative stabilization.



This aerial view of a portion of the Engleville stream shows a portion that might benefit from some vegetative or other stream stabilization.

Stream stability is an active process, and while streambank erosion is a natural part of this process, human development activities often exacerbates erosion rates. Streambank erosion increases the amount of sediment transported by the stream, resulting in the loss of fertile stream bed causing a decline in the quality of riparian and stream habitat, in addition, depositing excess sediment and phosphorus to Pond 1, where some of the sediment eventually settles.

Many of the methods for dealing with streambank erosion, stabilization and restoration are expensive to install and maintain. Solutions such as rock riprap or gabions (wire baskets filled with rock) may solve some erosion problems (typically for larger streams than this) but may not improve stream habitat or its aesthetic value.

Natural channel design principles look to nature for the blueprint to restore a stream to an appropriate dimension, pattern and profile. Soil bioengineering practices, native revetments and in-stream structures help to stabilize eroding banks. The following techniques may be used to move a stream toward a healthy, naturally stable and self-maintaining system:

<u>Soil Bioengineering Practices:</u>
Bioengineering uses plant materials in a structural way to reinforce and stabilize eroding streambanks. This technique relies on the use of dormant cuttings of willows, shrub dogwoods and other plants that root easily. Bioengineering practices range from simple live stakes to complex structures such as fabricated lifts incorporating erosion control blankets, plants and compacted soil.

<u>Native Material Revetments:</u>

These practices use native materials, wood and stone, to armor streambanks and deflect flow away from them. Low rock walls and log crib walls can be used to armor the bank. Root wads armor the bank and provide protection downstream by deflecting the flow away from the banks.

In-Stream Structures:
Rocks and logs can be used to construct a variety of structures that stabilize the streambank and banks. Cross vanes are rock structures that stabilize the streambed while aiding in streambank stabilization. Rock or log vanes redirect stream flow away from the toe of the streambank and help to stabilize the bank upstream and downstream from the structure. Where these practices are used, the protection should last long enough to allow appropriate vegetation to become established and provide for long term bank stability. The streamside vegetation improves habitat on the land and in the stream by providing hade, cover and flood. Several of the streambank stabilization structures, such as root wads, are also excellent fish habitat improvement structures.

Selection of the stabilization vegetation method or bank protective structure would be based on specifics of the size and characteristics of the stream types and area land uses. Selection of options in this case may be impacted by the due to the small size of this stream and potential for much of the watershed to be used agriculturally.

7.1.11 Additional Protection Measures

Measures to further protect water quality and limit the growth of phosphorus load that would otherwise offset load reduction efforts should be considered. The basic protections afforded by local zoning ordinances could be enhanced to promote smart growth, limit non-compatible development and preserve natural vegetation along shorelines and tributaries. Identification of wildlife habitats, sensitive environmental areas, and key open spaces within the watershed could lead to their preservation or protection by way of conservation easements or other voluntary controls. Engleville may also select to explore treatment options that may take advantage that much of the watershed flow is from the shoreline and tributary on the west side of the pond.

7.1.12 Watershed Implementation Plan

A watershed implementation plan that selects measures listed here and other steps deemed useful in reducing Pond phosphorous levels would be beneficial to protect this waterbody. Success of this effort will be evaluated based on monitoring, and will be modified if necessary based on any measured need for further reductions.

7.2 Follow-up Monitoring

Pond 1 will be a part of a targeted post-assessment monitoring effort initiated to determine the effectiveness of this TMDL implementation. Sampling will be coordinated with the existing Lake Classification and Inventory (LCI) program. Samples will be analyzed for standard lake water quality indicators, with a focus on evaluating eutrophication status: total phosphorus, nitrogen (nitrate, ammonia, and total), chl-a, pH, conductivity, color, and calcium. Field measurements include water depth, water temperature, and Secchi disk transparency. The program is next scheduled to conduct sampling in the basin every 5 years. Although not all lakes are monitored during each of these cycles, the TMDL status should result in Pond 1 being monitored during the majority of these cycles until compliance has been attained.

The 2014 lake sampling was initiated in specifically in support of the TMDL. Otherwise LCI sampling for the Mohawk Basin would have typically been scheduled for 2015. Therefore the next LCI monitoring would not typically occur until at least the next scheduled basin sampling in 2020, and potentially may or may not be included in the incremental sampling every 5 years thereafter. This schedule still may be modified if warranted to support local needs and based on the loading reduction strategy selected.

In 1997 the Pond 1 mean phosphorous concentration was 54 µg/l. This improved to a mean phosphorous concentration of 30 µg/l in 2014. This is still above both the 20 µg/l recreational target, and the 12 µg/l target that is expected to result in the chl-a concentration of 6 µg/l which is desired for a Class A drinking water source when the statistical method in Appendix A is applied. Phosphorous levels should be reduced and kept below both targets, and continued monitoring will track and evaluate the desired results as the actual phosphorous and chl-a concentrations diminish.

8 PUBLIC PARTICIPATION

The municipality worked with DEC to coordinate the 10 weeks of 2014 sampling that supplied data referenced in this that provided the basis for modeling and the report conclusions. In 2015 the data modeling procedure and Implementation options were discussed with the responsible municipality responsible parties and local DOH while the TMDL was in development.

Notice of availability of the Draft TMDL was made to local government representatives and interested parties. The Draft TMDL was public noticed in the Environmental Notice Bulletin on 8/3/2016. A 30-day public review period was established for soliciting written comments from stakeholders prior to the finalization and submission of the TMDL for USEPA approval.

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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 (DEC 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 the Ponds 1 & 2. 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"(DEC 2008), because an appropriate numeric translator for drinking water use has not been adopted.

In 2000, DEC incorporated such a site-specific interpretation of the narrative criterion protective of drinking water use into TMDLs for the New York City Reservoirs (DEC 2000). The USEPA, DEC 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 chl-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 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 (USEPA, 1998) to initiate a National Nutrient Strategy in 1998 with the goal of assisting all states in the development of numeric nutrient criteria.

To further the process of developing numeric nutrient criteria protective of potable water use, the DEC, in collaboration with investigators from the New York State Department of Health (DOH), 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).

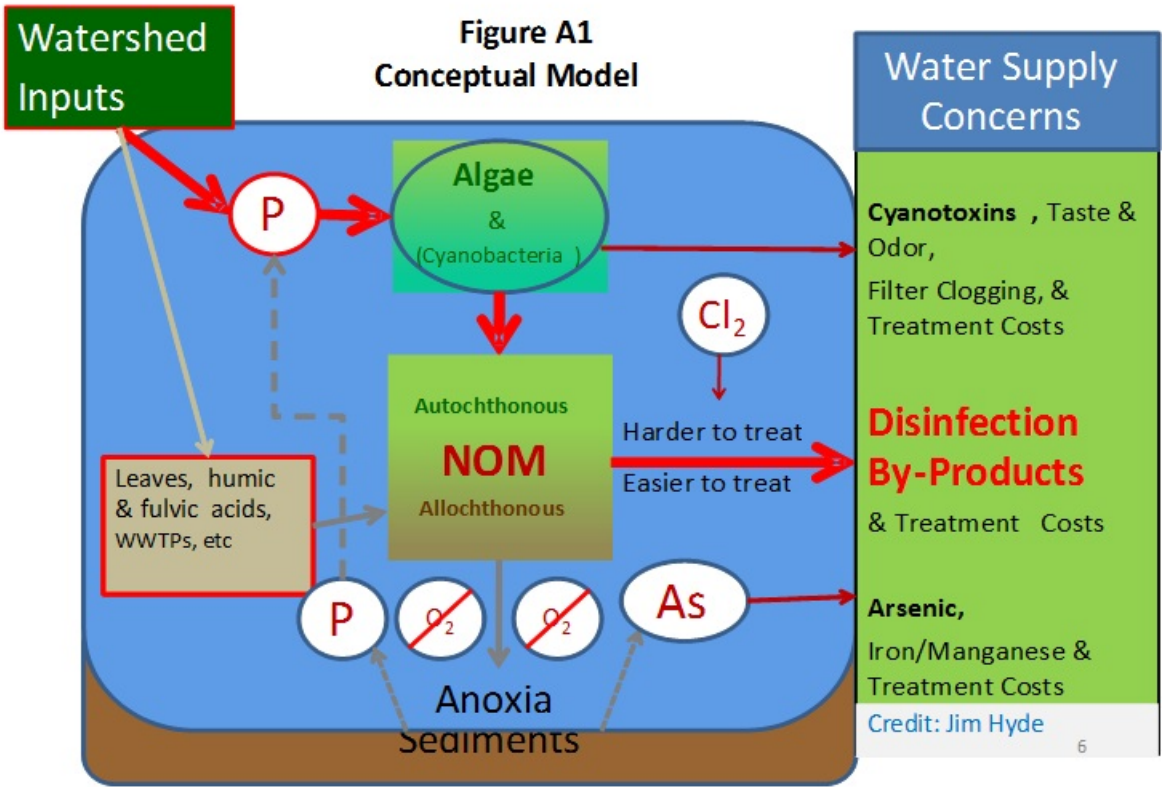
From that study, DEC has developed a process for determining Ambient Water Quality Values for *ponded sources of potable waters* in New York State, (DEC, 2010) which has undergone EPA and peer-review. That research for that process, as described in a peer-review journal (Callinan 2013), is used as the basis to evaluate the degree to which the TMDL target is adequately protective for the Engleville Pond TMDL, and to provide a second TP goal that is suggested as being a more optimal protective value for drinking water given the site-specific data available for those sources. This methodology results in a target concentration of 12 µg/l Total Phosphorous for Pond 1.

USEPA recently issued guiding principles "to offer clarity to states about an optional approach for developing a numeric nutrient criterion {Editor's Note: Herein referred to as target concentration} 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 A1. 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 chl-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 DEC 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).

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 DEC 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 1). 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 DEC study to derive appropriate site-specific ambient water quality values (AWQVs) is based upon findings from DEC's Disinfection By-Product/Algal Toxins Project (DBP-AT Project), as well as pertinent material from other independent investigations (both peer reviewed 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 DEC's DBP-AT Study involved the collection of paired ambient water samples that were analyzed for Trihalomethane 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 chl-a and TP indicates that approximately 78% of the variability in phytoplankton biomass (based on chl-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 Dissolved Organic Carbon (DOC) (a measure of NOM) and chl-a indicates a trend of increasing DOC concentrations with increasing chl-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 DEC's 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 chl-a, DOC and THMs established in the DEC's DBP-AT Project, a threshold of chl-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.

Target Concentrations (Endpoint)

DEC's DBP-AT Project derived threshold for chl-a is 4.0 µg/l as an AWQV to protect Class AA waters, given that these systems are required to meet applicable drinking water standards following only disinfection[1] (without coagulation, sedimentation and/or filtration treatments).

For ponded waters it is appropriate to derive distinct target concentrations 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. Classes AA will be subject to the more stringent target concentrations given that these waters are expected to meet applicable drinking water standards after only disinfection, whereas, ponded water supply source waters carrying water use Classes A will be subject to a somewhat less stringent target concentrations given that they are expected to meet applicable drinking water standards following "conventional" water treatment.[2]

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 chl-a. Thus, using the relationships among chl-a, DOC and THMs established in the DEC DBP-AT Project, the draft fact sheet proposed a chl-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 DEC 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 chl-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 chl-a concentration of 5 µg/l would be an appropriate threshold for direct use public water supply reservoirs.

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[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)

An endpoint for phosphorus is premised on an extensive body of literature indicating that phosphorus is the limiting nutrient (or causal variable) for primary productivity in most temperate, freshwater, ponded waters. The rationale behind setting criteria for chl-a is that it provides the most widely accepted measure of primary productivity (response variable) within freshwater ponded systems.

DEC has focused on the response variable, chl-a as the more appropriate ambient target because of its closer relationship to NOM and DBPs which directly affect the drinking water use. Thus, demonstration of the achievement of the water quality standard for Total Phosphorus, including for the purpose of a TMDL, would be informed by site-specific biomass response. This approach is consistent with the EPA guiding principles about an optional approach for developing a numeric nutrient criterion that apply when states wish to rely on response parameters to indicate that a designated use is protected (USEPA 2013). The EPA recognized that developing numeric values for phosphorus may present challenges associated with the temporal and spatial variability, as well as the ability to tie them directly to environmental outcome. Therefore, the USEPA guiding principles allow a State approach that integrates causal (nitrogen and phosphorus) and response parameters into one water quality standard.

DEC's subsequent study**,** *River Disinfection By-Product/Algal Toxin Study***,** prepared for the USEPA recommended that the primary metric for the establishment of numerical nutrient criteria be chl-a (response variable) because it is the parameter most closely linked to autochthonous DBP precursors (DEC 2010). While consideration was given to establishing a single numerical stressor (total phosphorus) criteria for flowing potable waters, the study concluded that the available dataset could not support the establishment of a single criteria value due to the variability in the relationships between both total phosphorus and chl-a as well as between total phosphorus and THMFP. Such variability is to be expected in natural systems including ponded water as the relationship between stressor and response variables has inherent variability.

Given findings from the DEC ponded and flowing water studies, as well as findings from other comparable studies, the more appropriate approach for establishing the stressor target (total phosphorus) is to establish a criteria "band" delineated by the prediction bands for the regression relationships. USEPA has proposed such an approach for the derivation of nutrient criteria in the state of Florida (USEPA, 2010). Ideally such an approach would use site-specific information regarding the response variable to fine-tune the stressor target, but would also be informed by general relationships demonstrated in robust datasets of multiple water bodies. Site-specific information, even where collected over several years with a variety of hydrological conditions is limited to the empirical range of the measurements. In the case of impaired waters, observations generally would not include chl-a levels that meet the target threshold, so the relationship would need to be extrapolated. Therefore a broader database of lakes, covering a broad band of trophic conditions including those which meet the target threshold chl-a level, provides additional context to a stressor-response model.

A.3 Model Development

The general approach for establishing the stressor target (total phosphorus) for Ponds 1 & 2 was to:

- Select a criterion for the response variable (chl-a = 6 µg/l) appropriate for protection of a drinking water use in a Class A water based on the DEC's DBP-AT Project;
- Use the (slope of the regression) relationship between mean chl-a and mean total phosphorus in combination with the 50% prediction interval to establish possible stressor criteria based on best-fit; and,
- Define the upper and lower prediction bands in which the criteria relationship would be used.

The process to establish a best fit and prediction bands for the total phosphorus to chl-a relationship considered available DEC and other quality assured data for lakes in New York State. Figure 2 shows the total phosphorus to chl-a relationship for lakes in NYS (PWS or otherwise denoted) with at least three year of extensive seasonal data. The prediction bands are denoted by the dashed lines around the regression line of best fit. This broader database was chosen over the DEC's DBP-AT Study results because the latter only covered 21 lakes/reservoirs with a single year of data, but had a similar TP to chl-a relationship. *(Figure will use µg/l units, note symbols for ponds and prediction lines (dashed)).*

A.4 Model Application

Application of the stressor-response model developed in the previous section requires specification of how and when the model will be applied. The rationale used to make decisions on how to account for assessed conditions within the model framework and how the target values will be expressed are described in the following sections.

A.4.1 Accounting for Site-specific Information

To incorporate site-specific context into the stressor-response relationship, the actual measured mean chl-a concentration is used as a starting point for the analysis. Next, the slope of the general stressor-response relationship is used to determine an appropriate mean Total Phosphorus concentration target, by solving for the response threshold of 6 µg/l chl-a. The relative improvement in the chl-a at each site is accomplished through changes in the Total Phosphorus concentration, weighted by the pre-factor from the regression equation.

For Pond 1, the calculation is for Targets Values changed as a result of the difference between the 1997 and the 2014 water sampling that was done. To provide proper context, the target calculations for both 1997 and 2014 are discussed below.

A.4.2 Site Specific Sampling Data as Target Basis for Pond 1: For Pond 1 Target Calculations as follows:

- Change in chl-a needed = Measured chl-a –Target chl-a :

1997 data >> 31-6.0 = 25 µg/l 2014 data >>17.75 -6.0 = 11.75 µg/l

- Change in regression TP variable = Measured TP variable –Target TP variable

1997 data> = $[0.634\times TP_m] - [0.634\times TP_t]$
$= [0.634\times 54] - [0.634\times TP_t]$

2014 data >= $[0.634\times TP_m] - [0.634\times TP_t]$
$= [0.634\times 30] - [0.634\times TP_t]$

Where the subscripts m and t are for the measured and target values of the TP variable, respectively. The change in Chlorophyll a is set equal to the change in regression TP variables, -

- thus allowing the following to be solved for target values for TP:

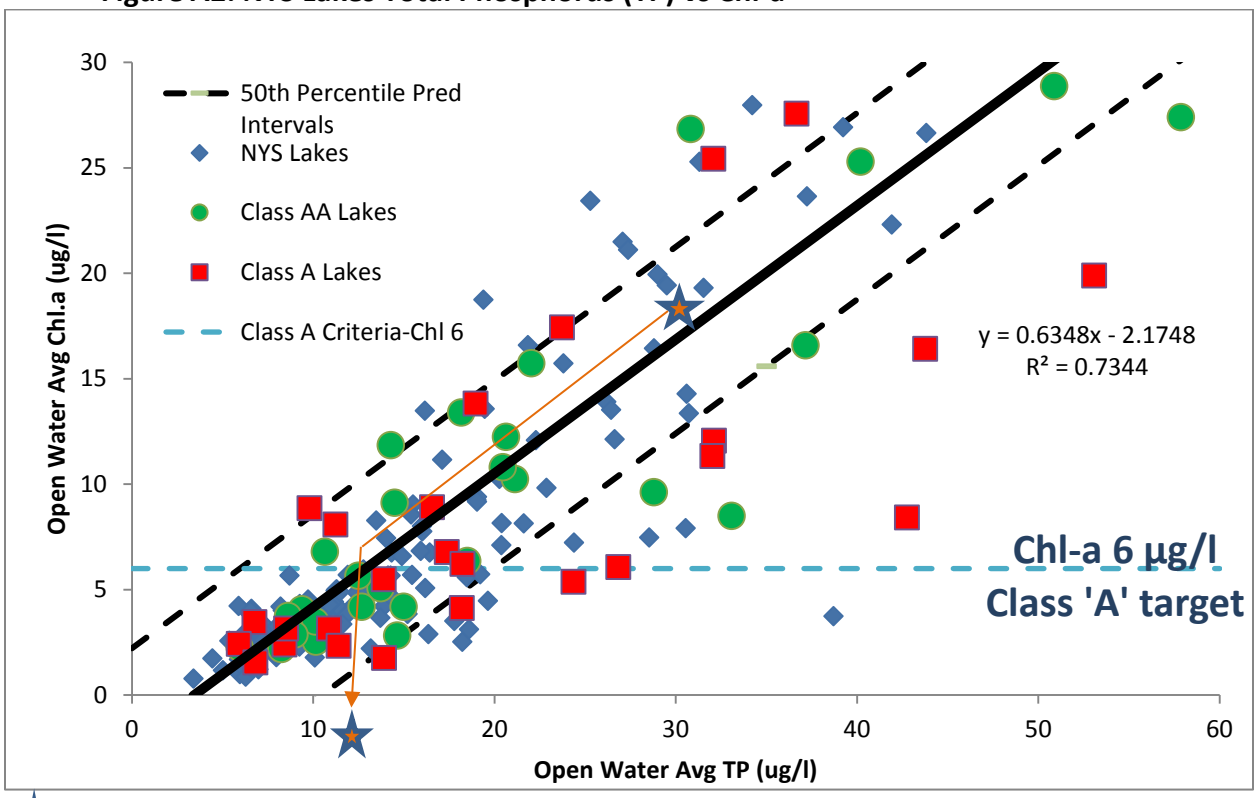
1997 data> 25 = $[0.634\times 54] - [0.634\times TP_t]$
Therefore, $TP_t = 15$ µg/L.

2014 data> 25 = $[0.634\times 54] - [0.634\times TP_t]$
Therefore, $TP_t = 11.4$ µg/L.

A target of 12ug/l is chosen:

The mean of these two values would be 13.2 µg/l, however since the 2014 data was more thorough and recent it was given greater weight, and the BATHTUB derived value of 12 µg/l was chosen to better correspond to the 2014 data results.

Figure A2: NYS Lakes Total Phosphorus (TP) vs Chl-a

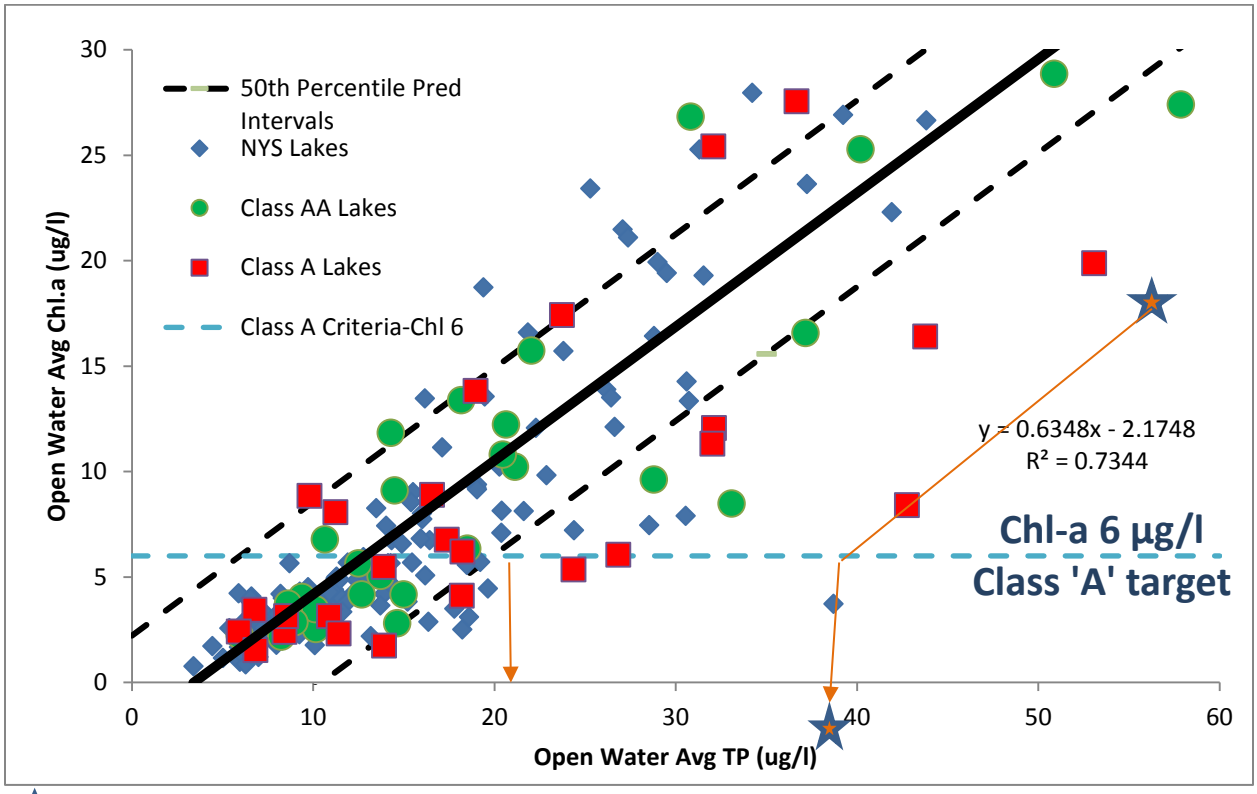


2014 Pond 1 monitoring data correlates to 11.4 µg/l chl-a, near to BATHTUB of 12ug/l

A.4.3 Site Specific Sampling Data as Target Basis for Pond 2:

For Pond 2, if the 1997 monitoring data were used, the similar approach to calculation would result in: $TP_{t=}$ 39 µg/L. However, as shown on Figure A3 by a blue star, the trophic state variables, TP and chl-a, for Pond 2 fall outside of the prediction bands (dashed lines) for the regression model. This apparent muted phytoplankton response to phosphorus could just be the result of temporal variation or monitoring error with a single year of monitoring, but could also represent a situation where there is a light limitation or inadequate retention time to grown out the algae from the available phosphorus. When the regression slope is used starting from the measured trophic state variables, as depicted by the red dashed line/arrow, the resulting TP concentration of 39 µg/L (as calculated above) is above the lower prediction band stressor threshold of 21 µg/L. Because the ambient data is outside of the prediction bands, the slope of the restoration would be less certain, but would likely be a shallower slope that the regression relationship. Therefore, to be conservative, the lower prediction band stressor threshold of 21 µg/L is instead chosen as the TP target for that Pond 2.

Figure A3: Derivation of Total Phosphorus (TP) Target



1997 Pond 2 monitoring data correlates to 39 µg/l chl-a, target would revert to 20ug/l as the NYS recreational protective target value

2014 data was not graphed for a potential TP target concentration in this derivation explanation since the mean value of 2.77 µg/l chl-a was well below the target value of 6 µg/l.

On the other hand the trophic state variables, TP and chl-a, for Pond 1, depicted in Figure A2, fell within the prediction bands and very close to the best-fit line for the regression model. Thus, the resulting translation of TP of 11.4 µg/l for the chl-a target would be a good approximation for an appropriate restoration target. (Corresponding approximately to the BATHTUB target of 12 was selected as explained in the previous section.)

A.4.4 Application of the target concentrations

When evaluating the 1997 data targets of 15 µg/l and 20 mg/l were tentatively developed for Ponds 1 and 2 respectively. Due to the age of the data, potential changes in the watershed, and some modification to one of the pond intakes, sampling occurred in 2014 to either verify or modify these targets as appropriate.

A.4.4.a Initial/Tentative 1997 based target concentrations

Sampling was done in 1997 as DEC's response to the Safe Drinking Water Act of 1996 to adequately characterize this waterbody being classified by DEC as a Class A with the best use as drinking water. It was, as still is the primary drinking water source for Sharon Springs, New York. The data from this testing effort was assessed for the TMDL to derive initial targets, later to be verified or modified by additional testing.

Data assessment indicated that achievement of the best use would be equivalent to setting phosphorus reduction targets based upon achieving a chl-a <= 6 µg/l through phosphorus reductions. The resulting TP targets were:

Table A1: Monitoring Data vs Target Values

Site	**1997 data TP (µg/l)**	**1997 Target TP (µg/l)**	**2014 data TP (µg/l)**	**2014 Target TP (µg/l)**	**BATHTUB**
Pond 1	54	15	30	11.4	**12.0 µg/l TP**
Pond 2	58	20 (39 calc)	16	In compliance	
Site	**1997 data chl-a (µg/l)**	**1997 chl-a reduce to 6µg**	**2014 data chl-a (µg/l)**	**2014 chl-a reduce to 6µg**	
Pond 1	31	25 reduction	17.75	11.75	**6.0 µg/l chl-a**
Pond 2	18	12 reduction	2.77	In compliance	

(1) *The target of 21 was chosen by using the prediction band intercept of the chl-a target, rather than calculating from the measured trophic values which yields a target 39. Then the 20µg/l from recreation was reverted to as being more conservative than this calculated value of 21.*

(2) *The target correlating to the statistical TP to chl-a reductions shown above. In this case the Target TP would revert to the recreation protection value of 20 µg/l TP correlating to 6 µg/l chl-a.*

The response model developed above provides a total phosphorus target endpoint for each pond which has been used for the development of this TMDL. The decision of how and when the endpoint is to be applied is, however, still informed by the science behind the development of the response model. As noted above, application of the response model includes specific limitations over the range of observations to specify how this endpoint is to be applied, as was done for Pond 2.

The response model was developed using average phosphorus concentrations from May through October (growing season). This was done because this was the identified critical period when phosphorus concentrations were measured and sunlight and temperature are favorable, creating the best condition for the production of algae. The associated NOM from production of algae is available for formation of DBPs. The applicability of the response model is therefore the same: an average TP concentration calculated over the May through October growing season.

In 2014 additional sampling of Pond 1 was done from May to September, at approximately two week intervals, in support of this TMDL development. This sampling was done to determine both the overall levels of TP and chl-a in the Ponds, as well as to determine the statistical correlation of these contaminants in order to either verify or modify the initially determined Phosphorus Targets.

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[3] The target of 21 was chosen by using the prediction band intercept of the chl-a target, rather than calculating from the measured trophic values which yields a target 39.

APPENDIX B. GENERAL MAPSHED MODELING ANALYSIS

The MapShed model was developed in response to the need for a version of AVGWLF that would operate in a non-proprietary GIS package. AVGWLF had previously been calibrated for the Northeastern U.S. in general and New York specifically. Conversion of the calibrated AVGWLF to MapShed involved the transfer of updated model coefficients and a series of verification model runs. The calibration and conversion of the models is discussed in detail in this section.

Northeast Arc View Generalized Watershed Loading Functions (AVGWLF) Model

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 B1). Flow data were obtained directly from the water resource database maintained by the U.S. Geological Survey (USGS). Water quality data 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.

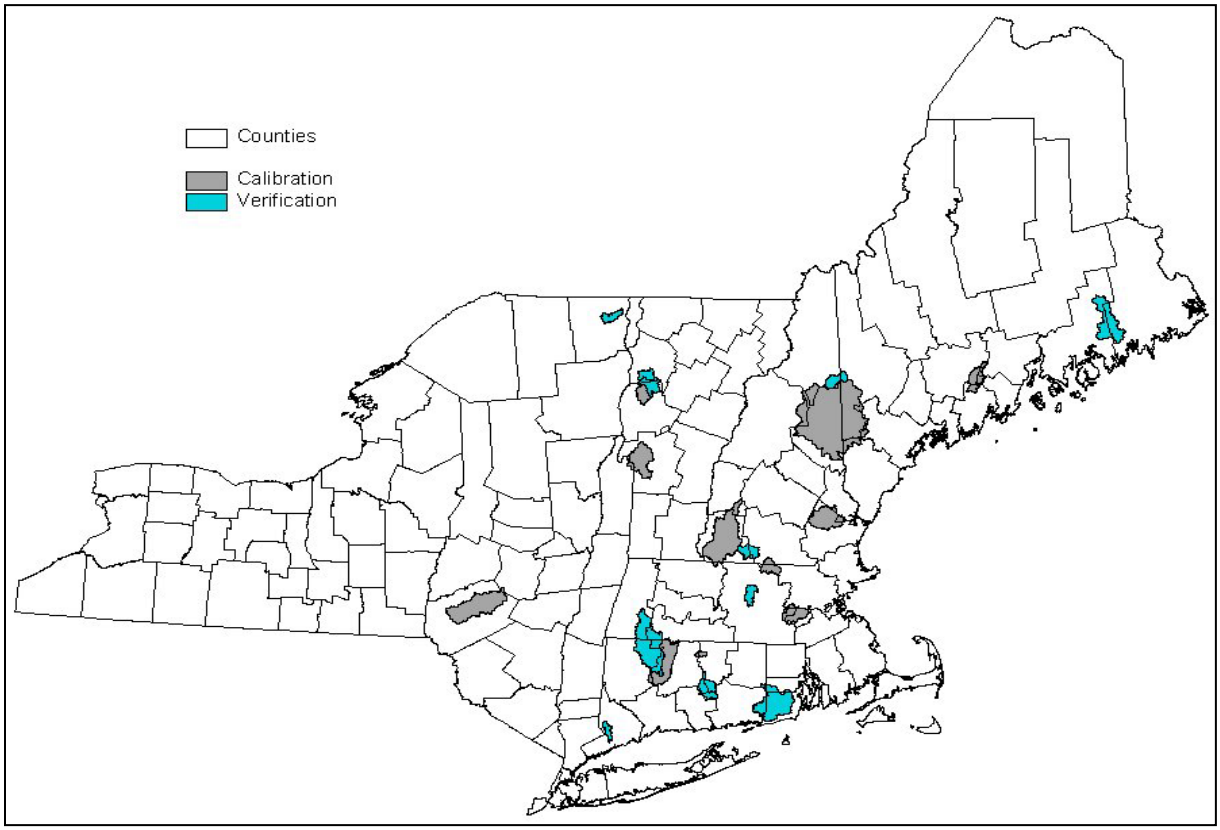

Figure B1. Location of Calibration and Verification Watersheds for the Original Northeast AVGWLF Model

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 B2.

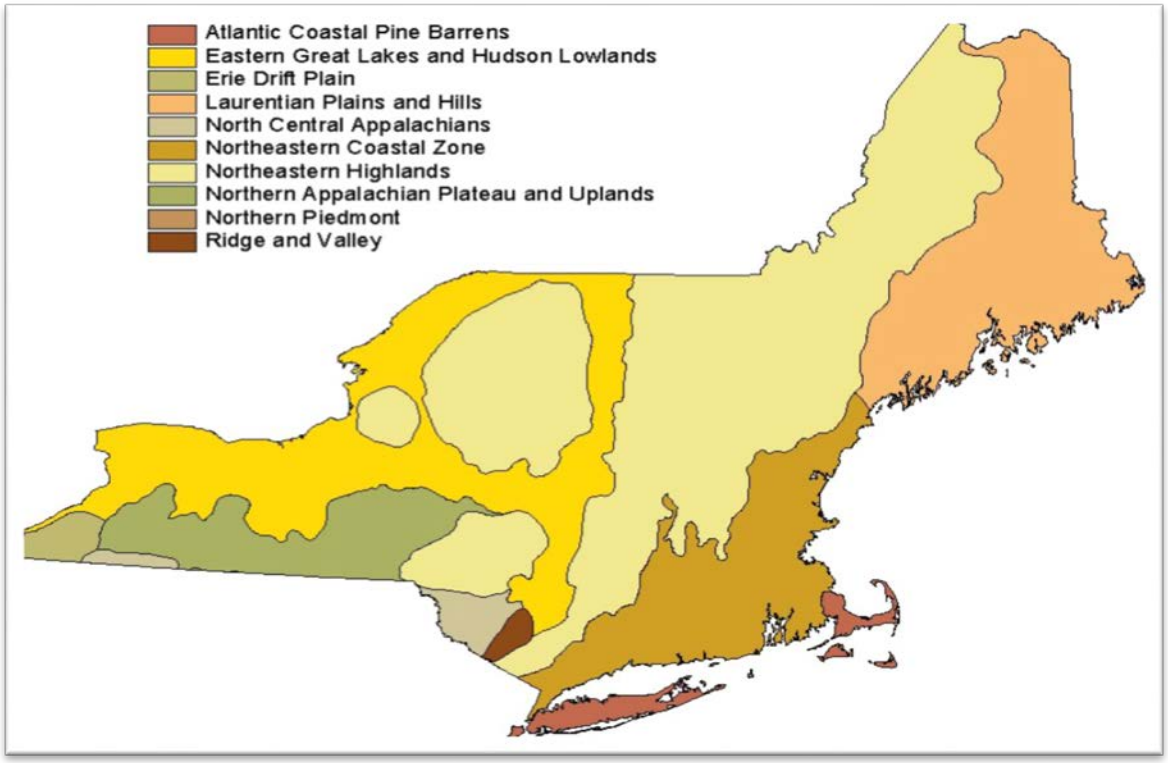

Figure B2. Location of Physiographic Provinces in New York and New England

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 B1. Two models were developed based on the following two physiographic regions: Eastern Great Lakes/Hudson Lowlands area and the Northeastern 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.

Table B1. 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

Conversion of the AVGWLF Model to MapShed and Inclusion of RUNQUAL

The AVGWLF model requires that users obtain ESRI's ArcView 3.x with Spatial Analyst. The Cadmus Group, Inc. and Dr. Barry Evans converted the New York-calibrated AVGWLF model for use in a non-proprietary GIS package called MapWindow. The converted model is called MapShed and the software necessary to use it can be obtained free of charge and operated by any individual or organization who wishes to learn to use it. In addition to incorporating the enhanced GWLF model, MapShed contains a revised version of the RUNQUAL model, allowing for more accurate simulation of nutrient and sediment loading from urban areas.

RUNQUAL was originally developed by Douglas Haith (1993) to refine the urban runoff component of GWLF. Using six urban land use classes, RUNQUAL differentiates between three levels of imperviousness for residential and mixed commercial uses. Runoff is calculated for each of the six urban land uses using a simple water-balance method based on daily precipitation, temperature, and evapotranspiration. Pollutant loading from each land use is calculated with exponential accumulation and washoff relationships that were developed from empirical data. Pollutants, such as phosphorus, accumulate on surfaces at a certain rate (kg/ha/day) during dry periods. When it rains, the accumulated pollutants are washed off of the surface and have been measured to develop the relationship between accumulation and washoff. The pervious and impervious portions of each land use are modeled separately and runoff and contaminant loads are added to provide total daily loads. RUNQUAL is also capable of simulating the effects of various urban best management practices (BMPs) such as street sweeping, detention ponds, infiltration trenches, and vegetated buffer strips.

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

Initially when CADMUS was using data for the time period 1990-2007, the calibrated MapShed model was used to estimate mean annual phosphorus loading to the lake. Table B2 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. In 2015, when Mapshed was run by DEC staff using the more complete and comprehensive sampling data set acquired by the State LCI testing program, the DEC staff included the weather data for the additional years through 2014 but maintained most coefficients derived by CADMUS that appeared to be independent of weather trends and events. In review of the CADMUS work done for many of the contracted draft TMDLs including Engleville, it was found that there were also a few site specifics variables and coefficients that varied from the general defaults values including those previously stated. Some of these are presumably fine tuning of functions specific to the best fit data requirements and the specific variations in the land use and resulting Total Phosphorous loadings.

Table B2. Information Sources for MapShed Model Parameterization

WEATHER.DAT file	
Data	**Source or Value**
	Historical weather data from Cooperstown, NY & Cobleskill, NY of the National Weather Service Stations
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 of 0.1 discussed for Northeast model, 0.05 used for Engleville
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	Derived from census tract maps for 2000 and house counts
Per capita septic system loads (N/P)	Default values/adjusted using AEU density

Initial Mapshed Land Use development by CADMUS:

Initially CADMUS had the 2001 NLCD land use coverage was obtained, recoded, and formatted specifically for use in MapShed. The New York State High Resolution Digital Orthoimagery (for the time period 2003 – 2005) was used to perform updates and corrections to the 2001 NLCD land use coverage to more accurately reflect current conditions. Each basin was reviewed independently for the potential need for land use corrections; however individual raster errors associated with inherent imperfections in the satellite imagery have a far greater impact on overall basin land use percentages when evaluating smaller scale basins. As a result, for large basins, NLCD 2001 is generally considered adequate, while in smaller basins, errors were more closely assessed and corrected. The following were the most common types of corrections applied generally to smaller basins:

1) Areas of low intensity development that were coded in the 2001 NLCD as other land use types were the most commonly corrected land use data in this analysis. Discretion was used when applying corrections, as some overlap of land use pixels on the lake boundary are inevitable due to the inherent variability in the aerial position of the sensor creating the image. If significant new development was apparent (i.e., on the orthoimagery), but was not coded as such in the 2001 NLCD, than these areas were re-coded to low intensity development.

2) Areas of water that were coded as land (and vice-versa) were also corrected. Discretion was used for reservoirs where water level fluctuation could account for errors between orthoimagery and land use.

3) Forested areas that were coded as row crops/pasture areas (and vice-versa) were also corrected. For this correction, 100% error in the pixel must exist (e.g., the supposed forest must be completely pastured to make a change); otherwise, making changes would be too subjective. Conversions between forest types (e.g., conifer to deciduous) are too subjective and therefore not attempted; conversions between row crops and pasture are also too

subjective due to the practice of crop rotation. Correction of row crops to hay and pasture based on orthoimagery were therefore not undertaken in this analysis.

In addition to the corrections described above, low and high intensity development land uses were further refined for some lakes to differentiate between low, medium, and high density residential; and low, medium, and high density mixed urban areas. These distinctions were based primarily upon the impervious surface coverage and residential or mixed commercial land uses. The following types of refinements were the focus of the land use revision efforts:

1) Areas of residential development were identified. Discretion was used in the reclassification of small forested patches embedded within residential areas. Care was taken to maintain the "forest" classification for significant patches of forest within urban areas (e.g. parks, large forested lots within low-density residential areas). Individual trees (or small groups of trees) within residential areas were reclassified to match the surrounding urban classification, in accordance with the land use classifications described in the MapShed manual. Areas identified as lawn grasses surrounding residential structures were reclassified to match the surrounding urban classification, in accordance with the land use classifications in the MapShed manual.

2) Areas of medium-density mixed development were identified. Discretion was used during the interpretation and reclassification of urban areas, based on the land use classification definitions in the MapShed manual. When appropriate, pixels were also reclassified as "low" or "high" density mixed development.

3) Golf courses were identified and classified appropriately.

Total phosphorus concentrations in runoff from the different urban land uses was acquired from the National Stormwater Quality Database (Pitt, *et al.*, 2008). These data were used to adjust the model's default phosphorus accumulation rates. These adjustments were made using best professional judgment based on examination of specific watershed characteristics and conditions.

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.

2015 Mapshed Land Use development by DEC:

The 2015 Mapshed runs were to be used later in conjunction with sampling information from both the 1997 lake testing as well as the 2014 lake testing to assess both Pond 1 and Pond 2. Therefore, the weather station data for an additional 10 years was added to the inputs such that the weather climate data became inclusive of the years from 1990 to 2014. Otherwise the basic format of the Mapshed program was retained, including the same Mapshed Program that CADMUS has used for the data evaluation.

In cases where coefficient uncertainty existed, DEC used the defaults specifically developed for New York State that CADMUS had stated were developed in consultation with Dr. Evans of Pennsylvania State University.

General On-site Wastewater Treatment Systems ("septic tanks") modeling considerations:

MapShed, following the method from 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.
- **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. MapShed 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 2.61 people per dwelling was used based on the circa 2000 USCB census estimate for number of persons per household in New York State.

MapShed 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.

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 unnecessary.

Point Sources

If permitted point sources exist in the drainage basin, their location was identified and verified by 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.

Concentrated Animal Feeding Operations (CAFOs)

A state-wide Concentrated Animal Feeding Operation (CAFO) shapefile was provided by 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, orthoimagery 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 MapShed and ultimately account for agricultural BMPs that may currently be in place in the drainage 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.

MapShed Model Simulation Results:

Input Transport File

Dissolved Runoff Coefficients (mg/L)

Rural Runoff	Dissolved N	Dissolved P
Hay/Pasture	0.75	0.19176
Cropland	0	0
Forest	0.19	0.01
Wetland	0.19	0.01
Disturbed	0	0
Turf/Golf	0	0
Open Land	0.5	0.01
Bare Rock	0	0
Sandy Areas	0	0
Unpaved Rd	0	0

	N	P	Sed
Groundwater (mg/L)	0.97	0.01	
Tile Drain (mg/L)	15	0.1	50
Soil Conc (mg/Kg)	2000	598	
% Bank Frac (0-1)	0.25	0.25	

Nitrogen and Phosphorus Loads from Point Sources and Septic Systems

Point Source Loads/Discharge

Month	Kg N	Kg P	MGD
Jan	0.0	0.0	0.0
Feb	0.0	0.0	0.0
Mar	0.0	0.0	0.0
Apr	0.0	0.0	0.0
May	0.0	0.0	0.0
Jun	0.0	0.0	0.0
Jul	0.0	0.0	0.0
Aug	0.0	0.0	0.0
Sep	0.0	0.0	0.0
Oct	0.0	0.0	0.0
Nov	0.0	0.0	0.0
Dec	0.0	0.0	0.0

Septic System Populations

Month	Normal	Pond	Short Cir	Direct
Jan	0	0	0	0
Feb	0	0	0	0
Mar	0	0	0	0
Apr	0	0	0	0
May	0	0	0	0
Jun	0	0	0	0
Jul	0	0	0	0
Aug	0	0	0	0
Sep	0	0	0	0
Oct	0	0	0	0
Nov	0	0	0	0
Dec	0	0	0	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	Nitrogen Acc Perv	Nitrogen Dis Fract	Phosphorus Acc Imp	Phosphorus Acc Perv	Phosphorus Dis Fract	TSS Acc Imp	TSS Acc Perv
LD Mixed	0	0	0	0	0	0	0	0	0
MD Mixed	0	0	0	0	0	0	0	0	0
HD Mixed	0	0	0	0	0	0	0	0	0
LD Residential	5	0.095	0.015	0.28	0.0095	0.0019	0.37	2.5	1.3
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

Input Nutrient File

Nitrogen and Phosphorus Loads from Point Sources and Septic Systems Runoff Coefficients by Source

Non-Point Source Loads/Discharge

Hay/Pasture	0.75	0.19176
Cropland	0	0
Forest	0.19	0.01
Wetland	0.19	0.01
Disturbed	0	0
Turf/Golf	0	0
Open Land	0.5	0.01
Bare Rock	0	0
Sandy Areas	0	0
Unpaved Rd	0	0

	N	P	Sed
Groundwater (mg/L)	0.97	0.01	
Tile Drain (mg/L)	15	0.1	50
Soil Conc (mg/Kg)	2000	598	
% Bank Frac (0-1)	0.25	0.25	

Septic System Populations

Month	Kg N	Kg P	MGD	Normal	Pond	Short Cir	Direct
Jan	0.0	0.0	0.0	0	0	0	0
Feb	0.0	0.0	0.0	0	0	0	0
Mar	0.0	0.0	0.0	0	0	0	0
Apr	0.0	0.0	0.0	0	0	0	0
May	0.0	0.0	0.0	0	0	0	0
Jun	0.0	0.0	0.0	0	0	0	0
Jul	0.0	0.0	0.0	0	0	0	0
Aug	0.0	0.0	0.0	0	0	0	0
Sep	0.0	0.0	0.0	0	0	0	0
Oct	0.0	0.0	0.0	0	0	0	0
Nov	0.0	0.0	0.0	0	0	0	0
Dec	0.0	0.0	0.0	0	0	0	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	Nitrogen Acc Perv	Nitrogen Dis Fract	Phosphorus Acc Imp	Phosphorus Acc Perv	Phosphorus Dis Fract	TSS Acc Imp	TSS Acc Perv
LD Mixed	0	0	0	0	0	0	0	0	0
MD Mixed	0	0	0	0	0	0	0	0	0
HD Mixed	0	0	0	0	0	0	0	0	0
LD Residential	5	0.095	0.015	0.28	0.0095	0.0019	0.37	2.5	1.3
MD Residential	0	0	0	0	0	0	0	0	0
HD Residential	0	0	0	0	0	0	0	0	0

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, chl-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 Pond 1 and its surrounding drainage area, as well as output from MapShed, a BATHTUB model was set up for Pond 1. DEC staff used the weather database from 1990-2014. DEC sampling data for both 1997 and 2014 were also used to assess the model's predictive capabilities and, to "fine tune" various input parameters and sub-model selections within BATHTUB during the calibration process. Once calibrated, BATHTUB was used to derive the total phosphorus load reduction needed 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 Schoharie County Planning Department and bathymetric maps provided by DEC or created by the Cadmus Group, Inc.
- Flow and nutrient loading from various pollutant sources - Obtained from MapShed output and calculated using Equation 1.
- Precipitation – Obtained from nearby National Weather Service Stations.
- Phosphorus concentrations in precipitation (measured or estimated), and measured lake water quality data (e.g., total phosphorus concentrations) – Obtained from DEC.
- Chl-a concentrations in precipitation (measured or estimated), and measured lake water quality data (e.g., total phosphorus concentrations) – Obtained from DEC

Tables C1 – C4 summarize the primary model inputs for Pond 1, 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 Pond 1. 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.

In the DEC modeling of Engleville a seasonal averaging period of May through September was selected corresponding to the potential algal growing portion of the year. The period of 5 months = 5/12 = 0.42 year, which is about twice the 0.2 yr residence time of this lake as the protocol recommended. This high ratio of growing season to residence time increases the importance of loadings during this time above that for lakes with much smaller ratios due much higher residence times, and for this reason DEC multiplied the projected loadings of the 5 month period by 12/5 to mathematically annualize loadings to better simulate TP and chl-a calculations based otherwise on annual calculations.

Evapotranspiration was derived from MapShed using daily weather data and a cover factor dependent upon land use/cover type. For the DEC modeling effort, which included the 2014 monitoring data, the weather data (1990-2014) was used for stations at Cobleskill, NY and Cooperstown, NY. 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 DEC from the Moss Lake Atmospheric Deposition Station located in Herkimer County. Atmospheric deposition is not a major source of phosphorus loading to Pond 1 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 Pond 1 were represented using an average of the observed summer mean phosphorus concentrations for year 1997 and then again for the year 2014. These data were collected through DEC's LCI program. The concentration of external phosphorus loading to the lake was calculated using the average annual flow and phosphorus loads simulated by MapShed. For years with observed data, the concentration of internal loading was calculated using the concentration of external loading, the hydraulic residence time, and lake phosphorus concentrations. Otherwise, the concentration of internal loading was calculated assuming concentrations were proportional to the average of years with observed data. To obtain flow in units of volume per time, the depth of flow was multiplied by the drainage area and divided by one year. 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). However, for lakes that have been previously exposed to excessive loading, the normal nutrient recycling models may not be sufficient. In these lakes, phosphorus builds up in the sediments, which can then become a significant source of phosphorus loading, especially in shallow lakes such as Pond 1. Walker

warns that nonzero values should be specified with caution. 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 internal loading rates reflect recycling of phosphorus that originally entered the reservoir from the watershed, the rates are 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.

Table C1. BATHTUB Model Input Variables: Model Selections

Water Quality Indicator	Option	Description
Total Phosphorus	01	2nd Order Available Phosphorus*
Phosphorus Calibration	01	Decay Rate*
Error Analysis	01	Model and Data*
Availability Factors	00	Ignore*
Mass Balance Tables	01	Use Estimated Concentrations*

* Default model choice

Table C2. BATHTUB Model Input: Global Variables

Model Input	Mean	CV
Averaging Period (years)	0.42	NA
Precipitation (meters)**	0.4694163	0.2*
Evaporation (meters)**	0.220226	0.3*
Atmospheric Load (mg/m^2-yr)- Total P	4.875	0.5*
Atmospheric Load (mg/m^2-yr)- Ortho P	2.605	0.5*

* Default model choice ** Precipitation and evaporation reflect the averaging period.

Table C3. BATHTUB Model Input: Lake Variables

Morphometry	Mean	CV
Surface Area (km^2)	0.12	NA
Mean Depth (m)	2.000	NA
Length (km)	0.5217	NA
Estimated Mixed Depth (m)	2.0	0.12
Observed Water Quality	**Mean**	**CV**
Total Phosphorus (ppb)- 1997 Mean data with late October omitted	61.0	-
Total Phosphorus (ppb)- 2014 Mean all data between May & Sept	30.0	-
Internal Load	**Mean**	**CV**
Total Phosphorus (mg/m^2 - day) 2015 calibration includes 2014 data	0.1700	-

* Default model choice

Table C4. BATHTUB Model Input: Watershed "Tributary" Loading

Monitored Inputs	(1990-2014) Mean	CV
Total Watershed Area (km^2)	2.32	NA
Flow Rate (hm^3/yr)	0.0564398	0.1
Total P (ppb)	17.42723	0.2
Organic P (ppb)	14.978	0.0

* Table C4 Tributary Loading concentration shows final compliance run assumptions.

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.

Several t-statistics calculated by BATHTUB provide statistical comparison of observed and predicted concentrations and can be used to guide calibration of BATHTUB. Two statistics supplied by the model, T2 and T3, aid in testing model applicability. T2 is based on error typical of model development data set. T3 is based on observed and predicted error, taking into consideration model inputs and inherent model error. These statistics indicate whether the means differ significantly at the 95% confidence level. If their absolute values exceed 2, the model may not be appropriately calibrated. The T1 statistic can be used to determine whether additional calibration is desirable. The t-statistics for the BATHUB simulations for Pond 1 are as follows:

Year	Observed	Simulated	T1	T2	T3
25 year avg. vs data average of 1997 and 2014	45.5*	37.6	0.95	0.71	0.64

In cases where predicted and observed values differ significantly, calibration coefficients can be adjusted to account for the site-specific application of the model. Calibration to account for model error is often appropriate. However, Walker (1999) recommends a conservative approach to calibration since differences can result from factors such as measurement error and random data input errors. Error statistics calculated by BATHTUB indicate that the match between simulated and observed mean annual water quality conditions in Pond 1 is good. Therefore, BATHTUB is sufficiently calibrated for use in estimating load reductions required to achieve the phosphorus TMDL target in the lake.

APPENDIX D. TOTAL EQUIVALENT DAILY PHOSPHORUS LOAD ALLOCATIONS

Source	Total Phosphorus Load (lb/d)			% Reduction
	Current Basis	Allocated	Reduction	
Agriculture*	0.3789	0.0189	0.3600	95%
Developed Land*	0.0071	0.0063	0.0008	12%
Forest, Wetland, Stream Bank, and Natural Background*	0.0866	0.0762	0.0104	12%
Open Land	0.0268	0.0208	0.0060	22%
Internal Loading	0.1074	0.0055	0.1019	95%
LOAD ALLOCATION	**0.6068**	**0.1277**	**0.4792**	**79%**
Point Sources	0	0	0	0%
WASTELOAD ALLOCATION	**0**	**0**	**0**	**0%**
LA + WLA	**0.6068**	**0.1277**	**0.4792**	**79%**
Margin of Safety	---	0.0142	---	---
TOTAL	**0.6068**	**0.1419**	**---**	**---**

* Includes phosphorus transported through surface runoff and subsurface (groundwater)