Modeling Documentation for the Lake Memphremagog TMDL

DRAFT

6-24-16

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VERMONT DEPARTMENT OF ENVIRONMENTAL CONSERVATION
WATERSHED MANAGEMENT DIVISION

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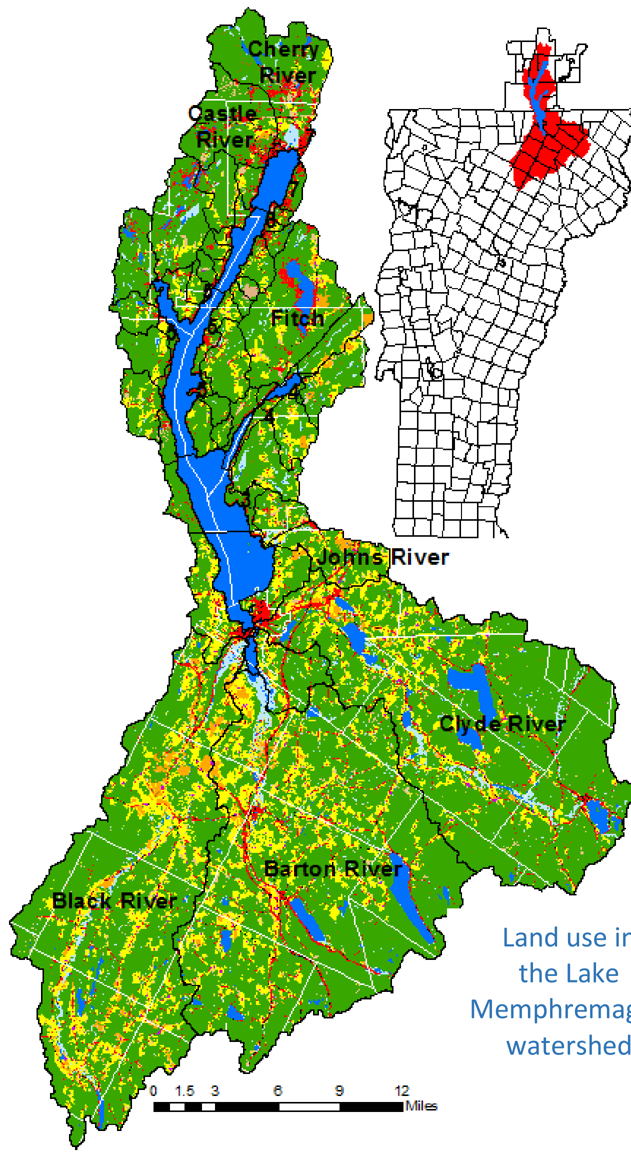
1 Introduction

Lake Memphremagog Cyanobacterial bloom

Phosphorus levels in the Vermont portion of Lake Memphremagog are on average 21% higher than the water quality standard set for the lake of 14 ug/l. Elevated levels of phosphorus contribute to infrequent Cyanobacteria blooms but also support excessive plant and algae growth that limits the quality of the lake for recreational use. A Total Maximum Daily Load (TMDL) is required by the Clean Water Act to set a limit of phosphorus that can enter the lake from its watershed and still meet this Water quality standard. Lake Memphremagog is an international waterbody with over 73% of its surface area in Quebec, while 27% is in Vermont. While most of the lake surface area is in Quebec, the majority of the Lake Memphremagog watershed lies in Vermont (71%). Although the Lake Memphremagog TMDL is only focused on the Vermont Lake Segment it is necessary to model the entire lake and watershed to understand the load reductions required to meet the lakes water quality standard. The Vermont portion of the watershed covers most of Orleans County including the three major lake tributary rivers: The Black, Barton, Clyde in addition to the smaller Johns River. Smaller shoreline areas drain directly to the lake including Newport City and town and the town of Derby.

The purpose of this report is to document the modeling steps that were taken in support of developing the Lake Memphremagog TMDL. This monitoring and modeling was done by Vermont Watershed Management Division staff in collaboration with a number of organizations in both Quebec and Vermont including joint water sampling efforts and an extensive collaborative phosphorus land use modeling effort funded through the MRC du Memphremagog which was adapted for use in this TMDL modeling effort. Many aspects of this modeling effort closely follow what was done for the Lake Champlain TMDL, however the shorter length of the water quality sampling record and lower sampling intensity and the larger extent of the Lake and watershed in Quebec increase modeling uncertainties. In addition to this, the approach to modeling watershed loading used in this effort was a Land use based phosphorus export model and not the Soil and Water Assessment tool model used for Lake Champlain. This report is broken into sections describing how information was collected, analyzed and how modeling was done including:

- Estimating flow into the lake
- Bathymetric features of the lake
- Chloride loading estimates from tributaries
- Phosphorus loading from tributaries
- Land use phosphorus export model
- In-Lake phosphorus measurements
- Lake Modeling approach
- Scenario tool development.



Land use in the Lake Memphremagog watershed

2 Estimating Flow for Tributaries and direct drainages to Lake Memphremagog.

Goal: Estimate mean daily discharge from all portions of the Lake Memphremagog watershed as needed for a model of chloride and phosphorus in Lake Memphremagog.

Timeframe for developing Phosphorus / flow relationships for tributaries in:
Vermont from 2005-2013
Quebec from 1998 – 2013
Timeframe for Correlations between Gages and Hydrotel model output: 1/1/1990 - 12/14/2010
Flows estimated from: 1990-2013

2.1 Sources for measured tributary inflows to Lake Memphremagog

USGS Gages – Barton, Black, Clyde **(64.8% watershed);** Missisquoi (used for regression)
Barton estimated prior to 7/16/2010 Log Barton= 0.70565(Log Black +.5 Log Clyde) - 0.00931 R^2=.94 (at outlet)

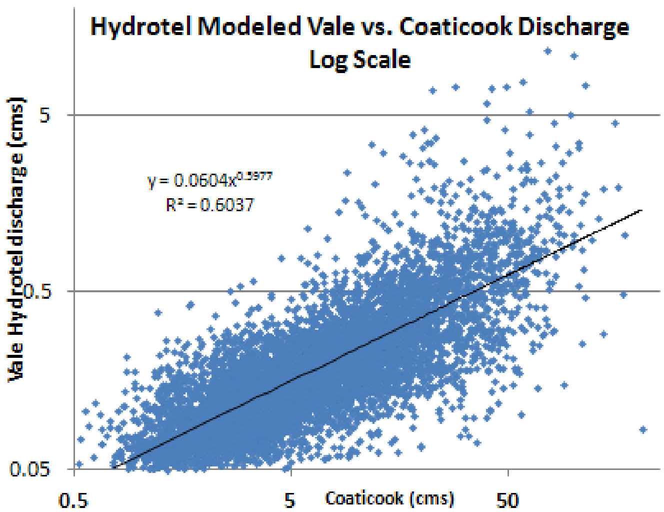
VT DEC Gage – Johns River 2008-2013 **(1.4% watershed)** or Johns discharge=0.005231*Black $(cfs)^{0.803245}$ R^2=.78

Quebec CEHQ – Coaticook River (used for regression)

Precipitation to Lake Memphremagog (5.5% watershed) - Estimates of precipitation directly to the lake (and islands) based on size of each lake segment times precipitation based on mean of three rain gages: Magog (in city of Magog on north end, maintained by Environment Canada), Georgeville (eastern shore about midway up lake, maintained by Environment Canada) and Newport (at south end of lake, provided by NOAA National Climatic Data Center). Average monthly pan evaporation at Essex Junction with a pan coefficient of 0.79 is used for months with measurements available, otherwise used Burlington Mean Monthly pan evap estimated with the penman-monteith calculations (and pan coefficient of 0.79). When converted to daily evaporation yields depths of less than 0.005m.

2.2 Modeled or estimated flows

Hydrotel modeled discharge from 1/1/1990-12/14/2010 (16.5% of watershed). Discharge was estimated by CEHQ for many of the subwatersheds in the watershed using the Hydrotel hydrologic model. Modeling used a number of input parameters including soils, land use, precipitation and then calibrated the model to the estimated flows to lake Memphremagog outside the Black and Clyde River watersheds and validated the model using the Black and Clyde Rivers.



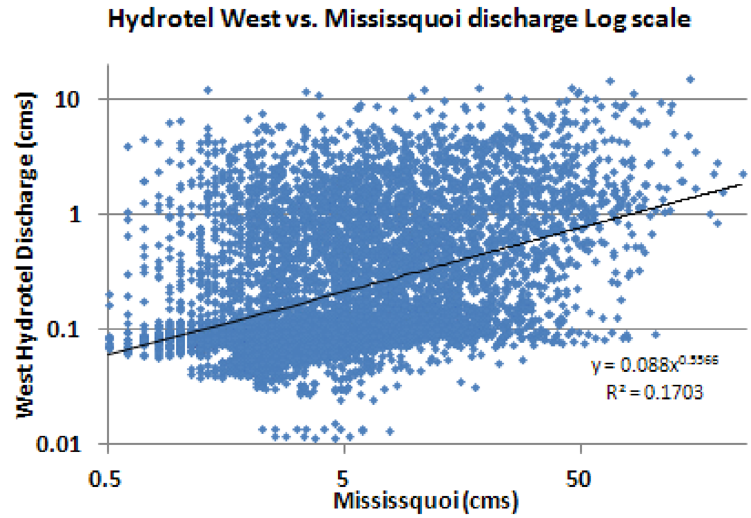


Figure 1. Regressions between hydrotel discharge and Coaticook or Missisquoi Gauged discharge from 1/1/1990 – 12/14/2010. Vale relationship was typical of tributaries except West which had a poor relationship.

Flows projected for hydrotel watersheds 12/15/2010-12/31/2012 (16.5% of watershed). Flows for 13 watersheds that were modeled through hydrotel were filled in beyond the hydrotel record based on the best regression with the hydrotel estimated flows and the Coaticook, Missisquoi, or Black gages. All watersheds had best regression with Coaticook gage (R^2 between .52 and .61) except Halls Creek which had a better relationship with Black and West which had poor regressions with all three gages as shown in Figure 1. Due to this uncertainty a DA Ratio with Coaticook gage was used for estimating the West discharge.

Flows estimated for Québec watersheds not broken out in Hydrotel model (3.5% of watershed). Flows for nine watersheds that were not broken out in the Hydrotel analysis and West drainage were estimated based on drainage area ratio with the Coaticook River. Johns River could not be used due to short overlap with phosphorus sampling data (Johns River monitoring began in 2008 and phosphorus data go back to 1998) and centroid of Coaticook watershed was closer then Black River (25-41km vs. 38-54 km) to the centroid of all nine watersheds. An evaluation of flow on the Johns River gage compared to estimate discharge based on a drainage area ratio with the Black River or through hydrotel model shows that between these two methods of estimating flow the Hydrotel model performs slightly better than the DA Ratio but both are close in all measures. Low bias in DA ratio may be due to rain event in August that hit Johns but not the Black River.

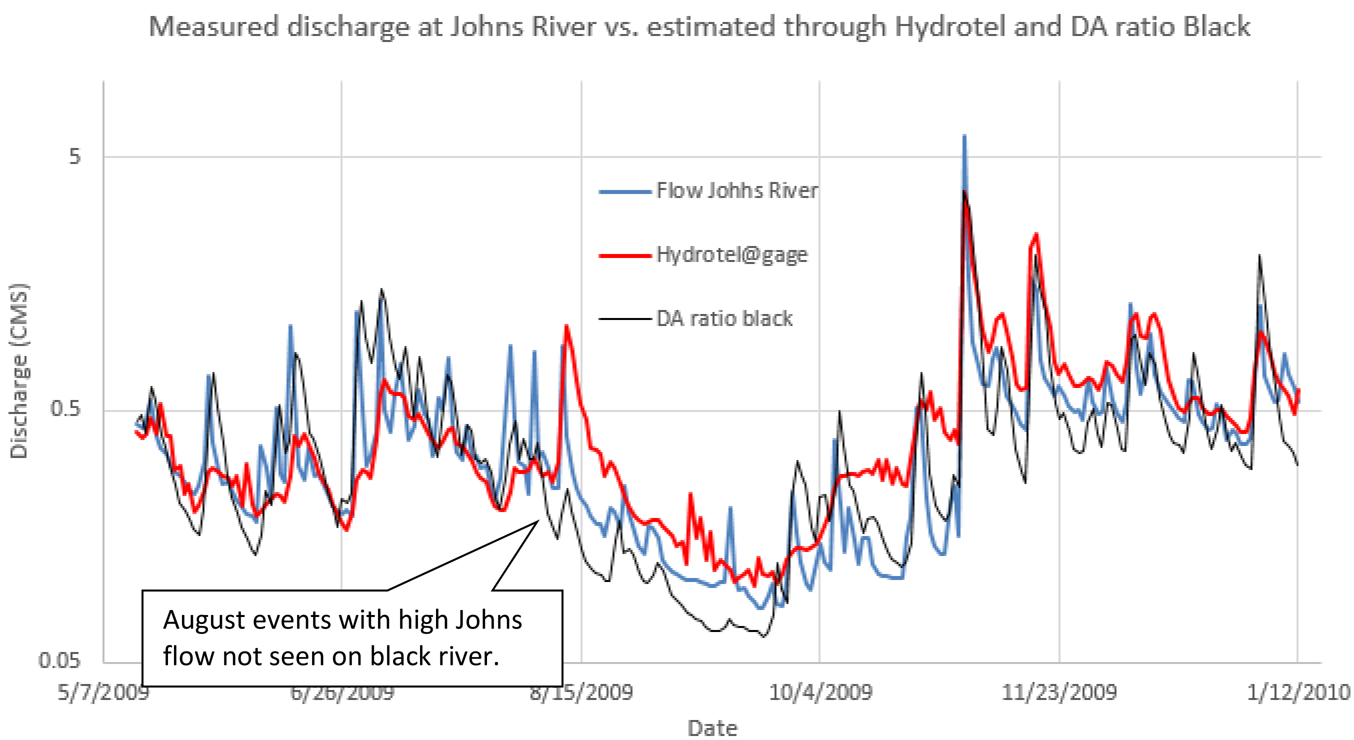


Figure 2. Comparison of measured Johns Discharge vs discharge modeled through hydrotel model or based on drainage area ratio with the Black River.

Table 1. Statistical comparison of measured discharge on Johns with discharge modeled through Hydrotel and based on drainage area ratio with the Black River

	Hydrotel	DA w/ Black
R^2	**0.64**	0.62
Nash-Sutcliff Coefficient of Efficiency	**0.62**	0.60
Root means squared error	**0.29**	0.30
Bias	**1.07**	0.90
Peak-weighted root mean sq error	**0.52**	0.55

Flow estimates for direct lake watersheds 1990-2012 (8.1% or watershed) - Flows for direct watersheds were estimated based on drainage area ratio with the Johns River. Johns River was chosen as a proxy for these watershed based on watershed centroid being closest to all segments and the fact that the watershed size of all direct watersheds was also closer in size to the Johns River watershed then either the Black or Coaticook watersheds. Flows from these watersheds are only used in the lake model from 2010-2012. The Johns River measured discharge and estimated discharge based on drainage area ratio with the Coaticook are quite close.

Table 2. Methodology for estimating discharge for each tributary to Lake Memphremagog

Tributary name	area (km)	Method for estimating Discharge
Coaticook	514.0	Quebec Gage (used for estimating flows)
Barton	429.5	USGS gage adjusted by 107.0% (after 7/16/2010) - correlation between Clyde and Black prior R^2=.94
Black	350.1	USGS gage adjusted by 110.79%
Clyde River	374.6	USGS gage adjusted by 101.78%
Johns	25.2	Vermont Gage 2008-2013 or regressed against Black River mean daily flow for missing data- R^2=.78
Bunker	12.4	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.60
Castle River	37.8	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.60
Chateau	11.5	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.58
Cherry River	54.8	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 R^2=.61
de l'Anse	3.2	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.60
Fitch Trib	60.4	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.56
Halls Creek	11.4	Hydrotel - Projected by regressing against Black mean daily flow 12/15/2010-12/31/2013 - R^2=.56
McCutcheon	12.9	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.52
McIntosh	13.7	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.60
Powell	29.8	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.60
Taylor	16.4	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.61
Tomkin	20.3	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.60
Vale	13.3	Hydrotel - Projected by regressing against Coaticook mean daily flow 12/15/2010-12/31/2013 - R^2=.61
West	17.7	Not modeled well in hydrotel - Flows estimated using DA Ratio with Coaticook – 3.43%
Boynton	3.4	Not monitored or modeled - Flows estimated using DA Ratio with Coaticook - 0.65%
Chemin Taylor	3.0	Not monitored or modeled - Flows estimated using DA Ratio with Coaticook - 0.58%
Gale	17.3	Not monitored or modeled - Flows estimated using DA Ratio with Coaticook - 3.36%
Glen	8.0	Not monitored or modeled - Flows estimated using DA Ratio with Coaticook - 1.55%
Hermitage	1.5	Not monitored or modeled - Flows estimated using DA Ratio with Coaticook - 0.30%
Limeklin	2.6	Not monitored or modeled - Flows estimated using DA Ratio with Coaticook - 0.51%
Patterson	2.8	Not monitored or modeled - Flows estimated using DA Ratio with Coaticook -0.55%
Price	2.2	Not monitored or modeled - Flows estimated using DA Ratio with Coaticook - 0.42%
Saint Benoit	3.4	Not monitored or modeled - Flows estimated using DA Ratio with Coaticook - 0.67%
Segment 1 unmodeled	24.0	Not monitored or modeled - Flows estimated using DA Ratio with Johns - 95.29%
Segment 2 unmodeled	36.8	Not monitored or modeled - Flows estimated using DA Ratio with Johns – 146.43%
Segment 3 unmodeled	10.3	Not monitored or modeled - Flows estimated using DA Ratio with Johns – 41.01%

Segment 4 unmodeled	5.6	Not monitored or modeled - Flows estimated using DA Ratio with Johns – 22.24%
Segment 5 unmodeled	30.6	Not monitored or modeled - Flows estimated using DA Ratio with Johns – 121.71%
Segment 6 unmodeled	16.2	Not monitored or modeled - Flows estimated using DA Ratio with Johns – 64.50%
Segment 7 unmodeled	10.7	Not monitored or modeled - Flows estimated using DA Ratio with Johns – 42.70%
Segment 8 unmodeled	6.2	Not monitored or modeled - Flows estimated using DA Ratio with Johns – 24.57%
North Lake QC	9.6	Precipitation
Center Lake QC	26.3	Precipitation
Inner Fitch Bay	2.2	Precipitation
Magog Lake segment	9.8	Precipitation
South Bay	2.3	Precipitation
South Lake QC	21.3	Precipitation
South Lake VT	24.1	Precipitation
Outer Fitch Bay	2.8	Precipitation

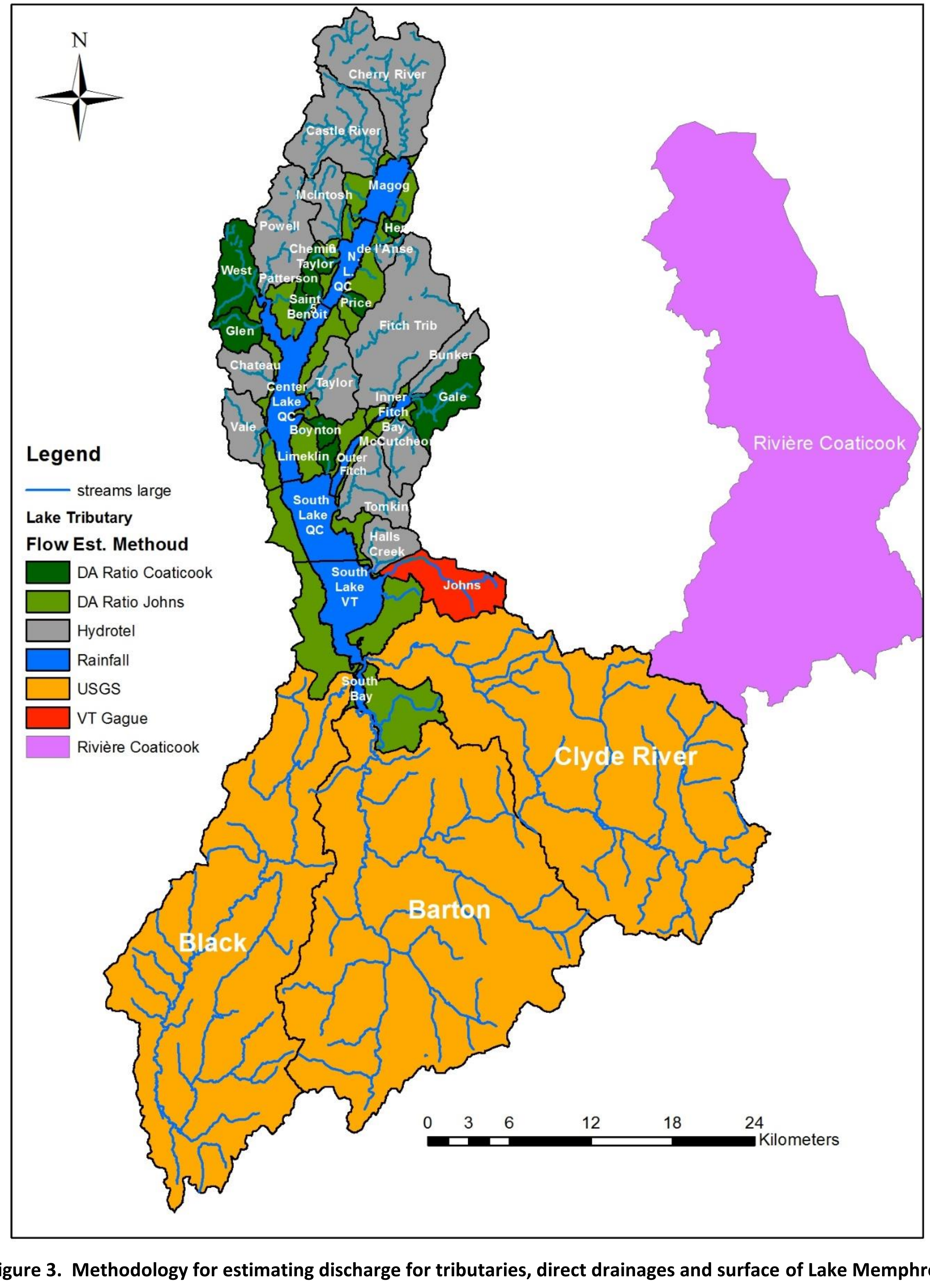


Figure 3. Methodology for estimating discharge for tributaries, direct drainages and surface of Lake Memphremagog

3 Bathymetry

Lake Bathymetry was based on a digital elevation model from data collected by MDDEFP, EPA, VTDEC, and CHS. The map was produced with a bench mark depth of 207.29 based on the Canadian datum for the gage in Quebec and equivalent to a depth of the USGS gage of 680.25. The datum was adjusted for estimating lake segment volumes and cross sectional areas adjusted to mean water levels at the USGS gague in Newport Vermont from 2000-2013 of 681.99 or an increase in depth of .42m or 1.38ft.

Table 3 Calculations to establish difference between bathometric chart datum and mean depth of .42M

Data	Elevation	Units
chart datum	207.29	Meters
Vt mean depth 2000-2013	207.87	Meters
conversion to Quebec datum	-0.160	Meters
Quebec mean depth	207.71	Meters
chart conversion to mean depth	0.41998	Meters
rounded conversion	0.42	Meters

Data collected in the Quebec portions of the lake were done at high resolution by CHS, areas of the Vermont portions of the lake were done by EPA and MDDEFP at high resolution and South Bay was surveyed by EPA and VT DEC at lower resolution but still sufficient for approximation of segment volumes and cross sectional areas. Digital depths were not available for Fitch bay but volume was estimated using 6 foot contours that were published on a hard cover map that was scanned in and converting to depth in meters and used to estimate Fitch Bay volume.

Lake areas were based on GIS shapefile of the lake split into segments at narrow locations or at the Vermont Quebec boarder as shown in Figure 1. Lake volumes for each segment were calculated in GIS and plus 0.42 meters times the segment surface area. In addition to this, cross sectional areas at the segment boundaries were calculated though a similar method in GIS with an addition of .42 meters but also the addition of width based on the slope of the lake near the shore at these cross sections. The resulting cross sectional areas are listed in Table 4 and segment area and volumes are listed in Table 5.

Table 4 Cross Sectional areas between lake segments.

Segment boundary	Area hm^2
South Bay – South Lake VT	0.0437
South Lake VT – South Lake QC	2.090
03 South Lake QC – Center Lake	0.927
05 Center Lake – North Lake	0.488
06 North Lake – Magog Bay	1.625
04 Inner Fitch Bay – Outer Fitch Bay	0.002
08 Outer Fitch Bay – South Lake	0.328

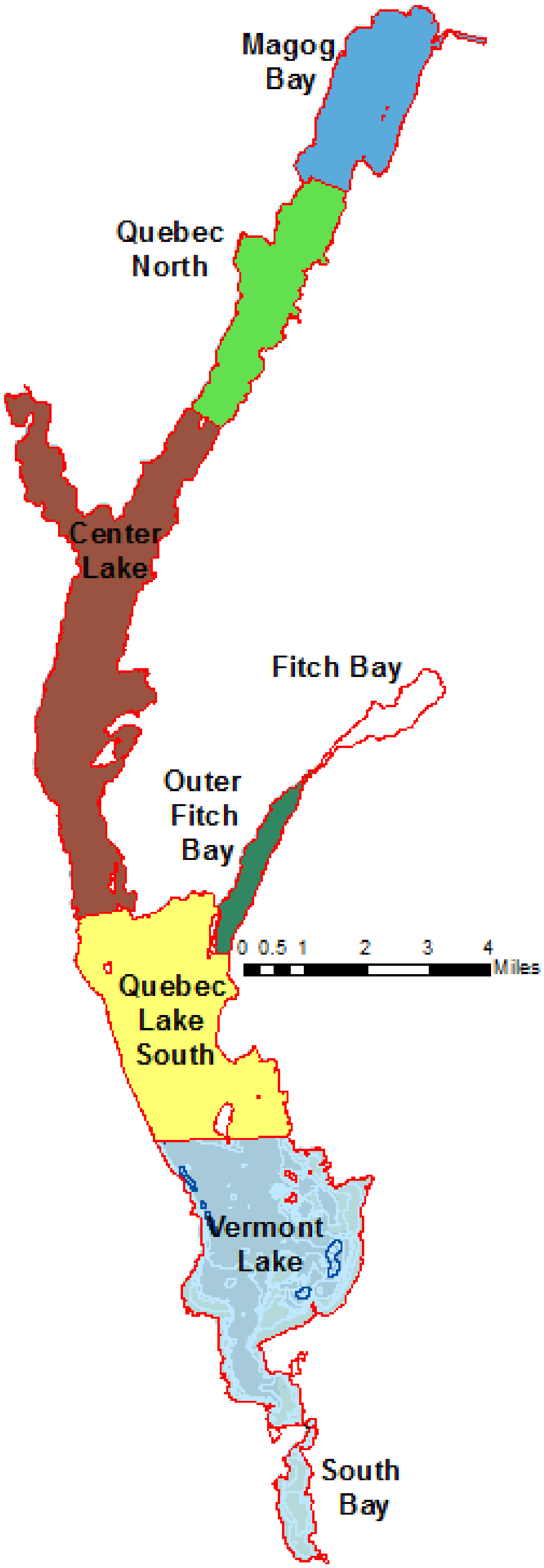


Figure 4. Lake Memphremagog lake segmentation used in modeling

Table 5. Morphologic features used in the modeling

Segment	Area (km)	Volume (hm)	Mean depth	2009-2012 annual Flow (hm/yr)	Residence time (yr)
South Bay	2.27	7.42	3.26	543.3	0.014
South Lake VT	23.80	160.65	6.75	878.7	0.183
South Lake QC	20.92	169.10	8.08	978.3	0.173
Inner Fitch Bay	2.21	6.34	2.87	60.6	0.105
Outer Fitch Bay	2.71	29.79	10.99	66.4	0.449
Center Lake QC	26.09	1307.09	50.11	1084.0	1.206
North Lake QC	9.45	140.13	14.83	1112.9	0.126
Magog	9.91	126.05	12.72	1181.2	0.107

4 Estimating Chloride loading to Lake Memphremagog.

Timeframe for in-lake modeling: 1/1/2010 through 12/31/2012
Timeframe for developing Chloride / flow relationships for tributaries in: Vermont from 2010-2013
Quebec from 2010 – 2012

4.1 Sources for chloride loading to Lake Memphremagog

Vermont water quality sampling – Barton, Black, Clyde, and Johns **(66.2% watershed)** chloride loading was estimated using the relationship between chloride and discharge which was strong for all four tributaries. Flux was used to estimate daily chloride loading which was done with a low coefficient of variation for each tributary. Sampling was done from 2010 through 2013 with a total number of chloride samples of 71 for the Black River, 72 for the Barton River, 73 for the Clyde River, and 62 for the Johns River all processed by the VT DEC laboratory. Sampling frequency varied from monthly with high flow samples taken when flows were above the Q90% as measured at the Black River as has been done since 2013, Biweekly sampling plus samples taken when flows were above 90% discharge from 2010-2012. Sampling on the Black, Barton, and Clyde Rivers was done using the DH 59 "bomb" sampler at the centroid of flow to capture a single depth integrated sample. Samples on the smaller Johns River were taken as grab samples at the centroid of flow. Quality assurance measures were met for phosphorus for all years with an average RPD of 1.68% and there were no instances of blank samples above detection.

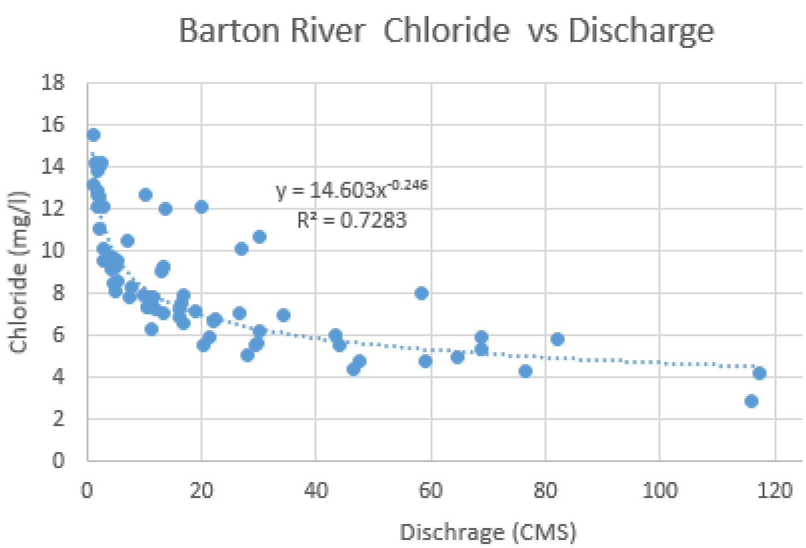


Figure 5 relationship between chloride concentration and discharge on the Barton River from 2010-2013.

Table 6. Chloride loading estimates using flux for the Black, Barton, Clyde from 2010- 2013								
River	Number of samples	*Log/Log slope*	*R^2*	*p > C/Q*	*Load An.*	*Var.*	*Chloride mg/L*	*C.V.*
Barton River	**72**	-0.247	0.730	0	2349573	19286091	7.99	0.024
Black River	**71**	-0.3057	0.726	0	1584344	12522053	6.29	0.029
Clyde River	**73**	-0.1937	0.693	0	2386620	27768150	8.18	0.028
Johns River	**62**	-0.292	0.842	0	297354.4	154363	18.5	0.017

However, based on an evaluation of residuals it appears that there was a seasonal variation in the loading relationship with higher chloride levels for given flows with higher levels for a few weeks around showmelt and in the fall possibly related to chloride application to suppress dust on dirt roads. These seasonal changes appear to be different each year depending on the timing of snowmelt or major rainfall events such as in 2011 which appear to have reduced chloride levels for nearly a year possibly related to storage of low chloride water in lakes as well as ground water supplies. To address this seasonal variation an adjustment factor to the estimated concentrations was made based on the residuals when these were consistent over a period of time. The first graph below shows the residual of the chloride concentration and the applied adjustment factor. The second graph shows the measured chloride levels in the Barton River from 2011-2012 overlaid with the estimated chloride levels with and without the seasonal adjustment factors.

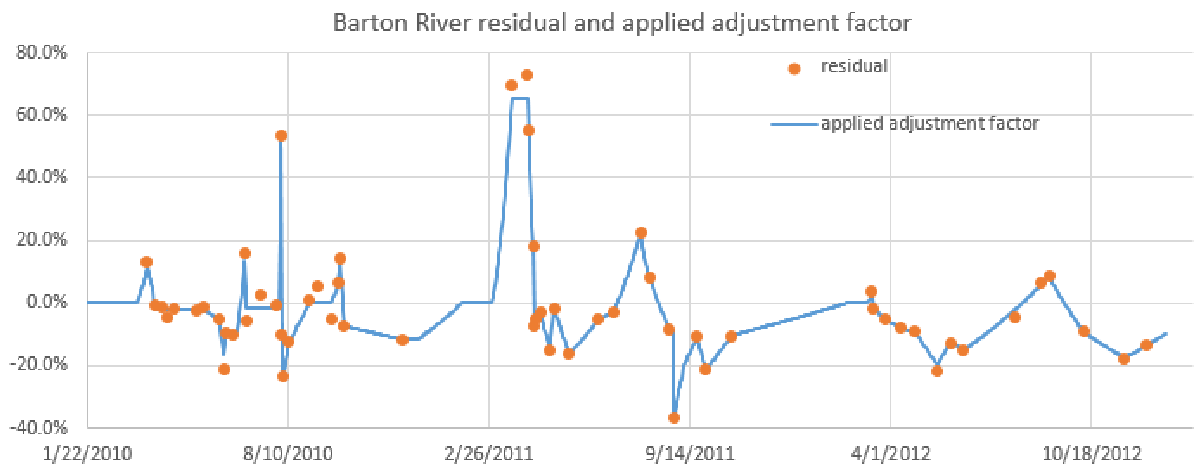


Figure 6 Chloride residual on the Barton River based on relationship Cl = 14.603x $Q^{-0.246}$ and the applied seasonal adjustment factor.

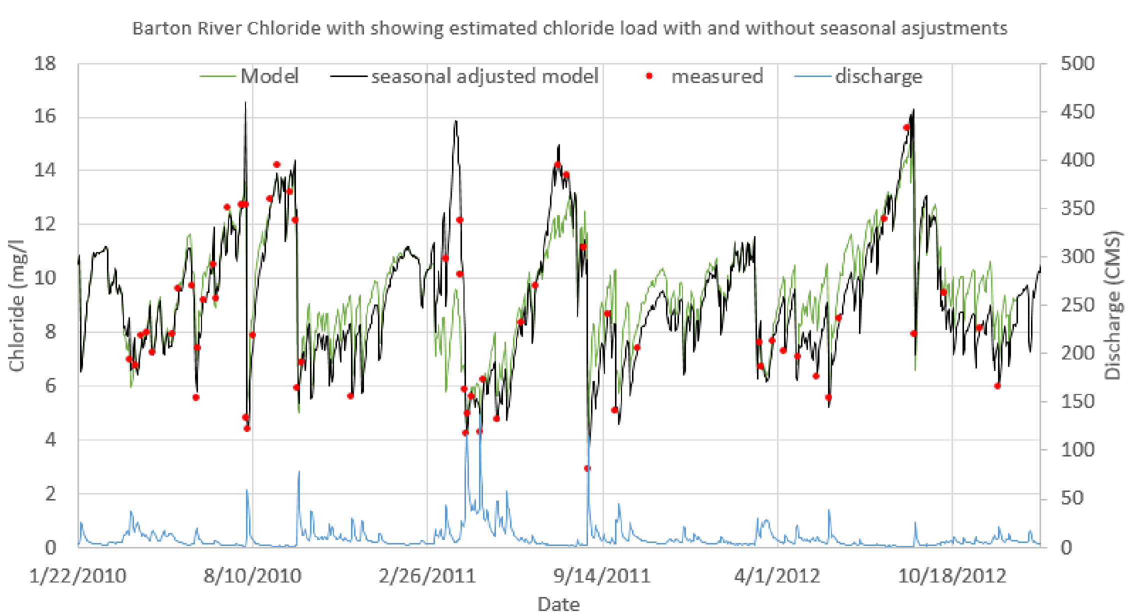


Figure 7 Estimated chloride levels on the Barton River based on relationship Cl = 14.603x $Q^{-0.246}$ along with the estimated concentrations with the applied seasonal adjustment factor and measured concentrations.

As you can see in the graph below these adjustment factors are similar for many of the tributaries. The general trend is higher chloride levels in the spring and fall although in the fall of 2011 the rain from tropical storm Irene after the major spring runoff event appears to have driven chloride levels down into the late fall and winter of 2011.

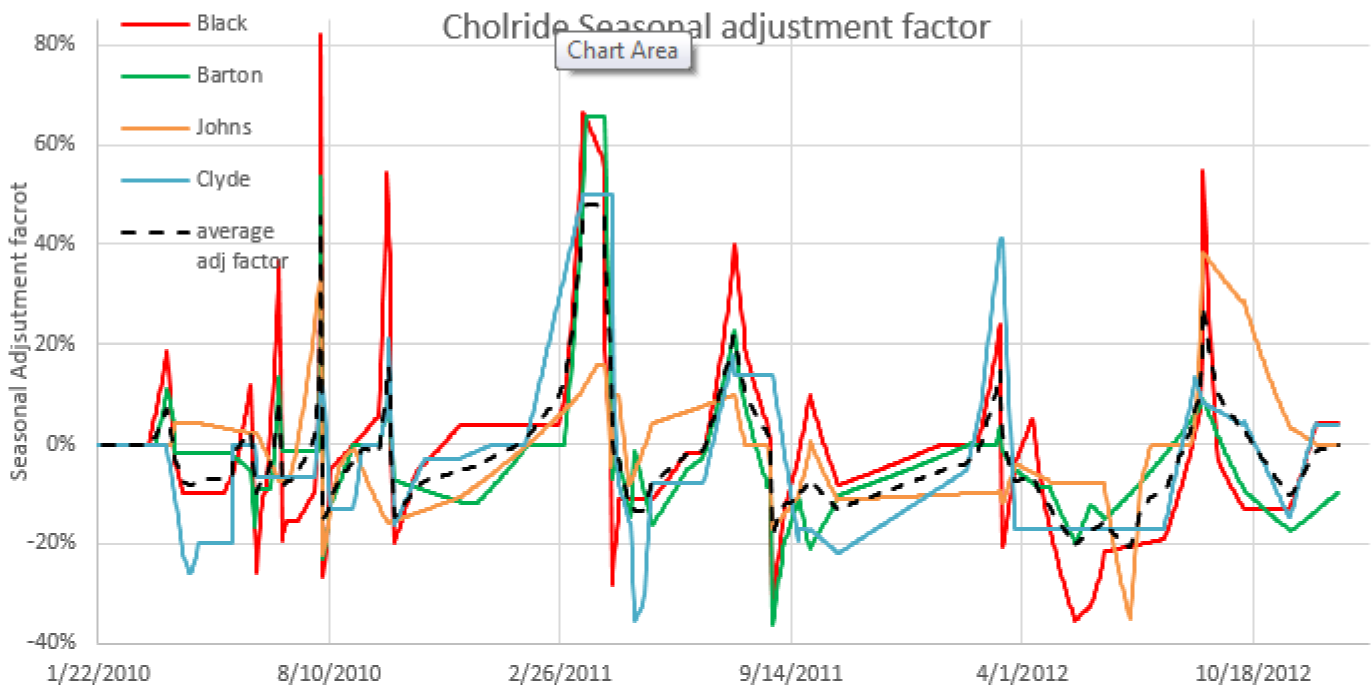


Figure 8 seasonal adjustment factors applied for the Black, Barton, Clyde and Johns Rivers along with an average which was applied to other tributaries in the watershed.

4.2 Castle and Cherry Rivers (5.2 % of watershed)

There is less sample data to use for estimating chloride loading from Quebec Tributaries. All the chloride samples were processed in the Vermont lab so no laboratory comparison was necessary. The Cherry and Castle rivers have 22 and 26 chloride samples respectively with relatively good relationships with discharge estimated through the hydrotel model (R2 of .62 and .68 respectively) also showing inverse relationship with flow. We had a frequency of data to add an adjustment for the castle and cherry Rivers for 2011 and 2012 and so and seasonal adjustment factor was applied to both of these rivers and an average of this was used for other tributaries in lake segments 5 and 6.

4.3 Chloride from Small Quebec Tributaries (8.6% of watershed)

We have Chloride data from 11 other smaller tributaries which range in number of samples of between 5 to 12, and with relationships with discharge that have an R2 ranging from a low of .12 to a high of .83 with most in the .5 - .7 range as shown in table 2. Power relationships were used for each of these sites as this seemed to best describe the relationship between chloride and flow with the exception of the Fitch trib which seemed to be best described with a linear relationship. A number of the sites had non-detects at high to moderate flow. A ratio of the chloride from a nearby tributary at levels just above non-detect was made and this was used to estimate chloride levels below detection limits assuming that this ratio was consistent which was the generally the case above the detection limit.

This method was used for Bunker, Gale, and Powell tributaries. This method assumes the relationship with discharge is similar between these tributaries and the tributary used to estimate the chloride at lower flows but seems better than picking an arbitrary number for non-detect values. In addition to this, because the seasonal adjustments appear likely to apply across watersheds the average of the seasonal adjustment was applied to all the watersheds based on location. For segments 1-2 an average of the Johns, Black, Barton, and Clyde was applied, for segments 3, 4, 8 and average of all adjustment factors was applied and for Segments 5 and 6 an average of the adjustment factor for the Cherry and Castle rivers was applied.

Table 7 Chloride relationship with flow for all monitored watersheds in Vermont and Quebec.

watershed	equation	N	R^2	Segment	Country	Ski Area	Watershed area (km^2)	asphalt km/km^2	Mean Cl- mg/l
Castle	$y = 22.2321x^{-0.8800}$	26	.68	**7**	Canada	YES	37.8	1.19	29.31
aux Cerises	$y = 17.672x^{-0.87}$	22	.63	**7**	Canada	No	54.8	1.06	21.05
Johns	$y = 16.052x^{-0.292}$	62	.84	2	Both	No	25.2	1.23	17.71
Vale	$y = 4.9632x^{-0.695}$	5	.73	**3**	Canada	YES	13.3	0.34	12.95
Fitch	$y = -1.518x + 12.210$	12	.44	**4**	Canada	No	13.6	0.51	9.49
Clyde River	$y = 13.196x^{-0.229}$	73	.69	2	USA	No	374.6	0.44	7.58
Barton	$y = 14.603x^{-0.246}$	71	.73	1	USA	No	429.5	0.50	7.43
Château	$y = 2.4705x^{-0.709}$	5	.48	**5**	Canada	No	11.5	0.32	6.28
Black	$y = 12.524x^{-0.303}$	71	.72	1	USA	No	350.1	0.29	5.97
McIntosh	$y = 1.5769x^{-0.821}$	9	.12	**6**	Canada	No	13.7	0.20	4.70
West	$y = 3.7115x^{-0.266}$	5	.26	**5**	Canada	No	17.8	0.20	4.38
Taylor	$y = 2.1432x^{-0.418}$	6	.71	**5**	Canada	No	16.3	0.21	3.17
McCutcheon	$y = 2.1708x^{-0.329}$	11	.83	**4**	Canada	No	12.9	0.28	3.14
Tomkin	$y = 2.042x^{-0.17}$	10	.48	**3**	Canada	No	20.3	0.12	2.29
Powell	$y = 1.276x^{-0.921}$	8	.58	**5**	Canada	No	29.8	0.03	1.74
Bunker	$y = 0.7243x^{-0.397}$	11	.58	**4**	Canada	No	12.4	0.00	1.30
Gale	$y = 1.8758x^{-0.116}$	11	.79	**4**	Canada	No	17.3	0.00	1.20

4.4 Unmonitored watersheds (14.5% watershed) **–** Finally to fill in for unmonitored watersheds a nearby watershed which has a similar length of paved road per watershed area was used. The watersheds used to estimate loading for each subwatershed is shown in Table 2 below. Watersheds chosen to estimated loads for other watersheds were chosen based on length of road per watershed area which correlated very closely with mean chloride concentration from 2010 - 2012. Figure 6 below shows this relationship separately for Vermont and Quebec watersheds with the Vermont watershed having a lower mean chloride concentration per linear km of road and also separates out the Vale and Castle Rivers which both have ski area development and significantly higher mean chloride concentrations per Km of paved road than other sites. Chloride use tends to be high around ski areas including large ski area parking lots not included in the road length and there may also be increase application rates due to heavy use in winter. Because of this these two watersheds with ski areas were excluded from the correlation estimating loading in unmonitored watersheds without ski areas.

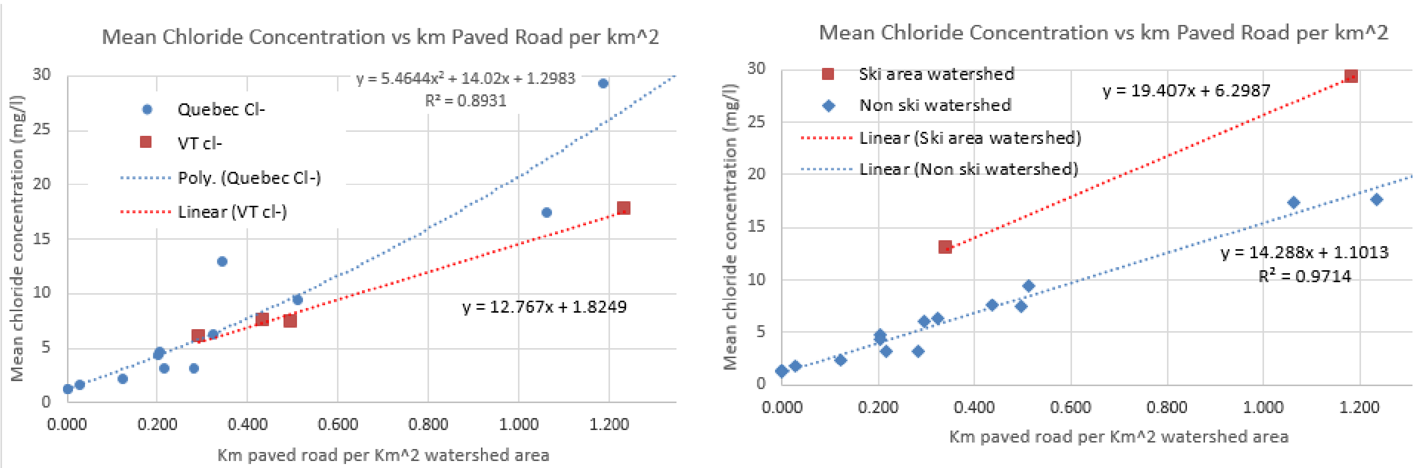


Figure 9. Mean Chloride Concentration vs Km of paved road per Km^2 of watershed showing with Vermont and Quebec watersheds and watersheds seperated between those with ski area development.

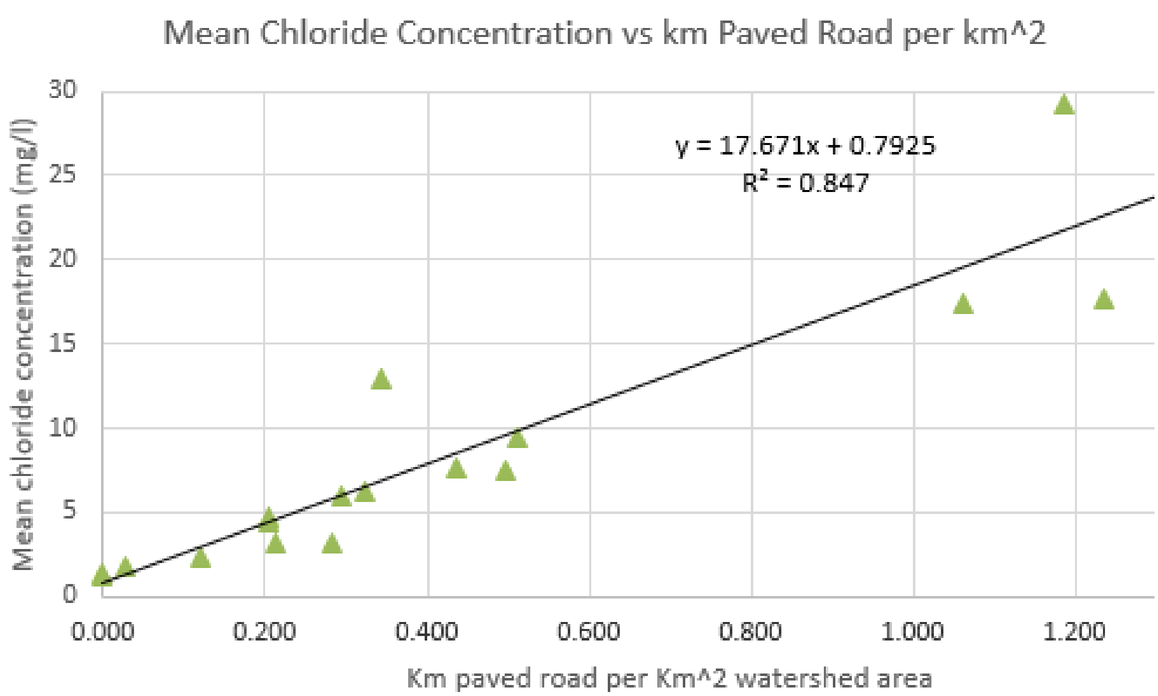


Figure 10. Mean Chloride Concentration vs Km of paved road per Km^2 of watershed with all watersheds together.

Table 8 Proxy watersheds used to estimate chloride for watersheds based on linear km of paved roads per Km^2. Two watersheds unmonitored areas in Segment 7 in Magog, and 1 in VT had to be further adjusted because no nearby watershed had a similar km^2 paved road per km^2

subwatershed name	asphalt km/km^2	Proxy	Proxy Asphalt Km/km^2	Segment	Country	Ski Area	Watershed area (km^2)
unmon seg 7	2.68	Cerises* 2.5[1]	1.06	7	Canada	No	10.4
Halls Creek + Seg 2	1.38	Johns	1.23	2	Both	No	48.2
unmon seg 1	0.98	Johns	1.23	1	Both	No	24.0
de l'Anse	0.87	aux Cerises	1.06	6	Canada	No	0.6
Hermitage	0.72	Fitch	0.51	7	Canada	No	1.5
unmon seg 6	0.71	Fitch	0.51	6	Canada	No	16.6
unmon seg 4	0.71	Fitch	0.51	4	Canada	No	4.8
Patterson	0.48	Fitch	0.51	5	Canada	No	2.8
Price	0.33	Château	0.32	6	Canada	No	2.2
Chemin Taylor	0.32	Château	0.32	6	Canada	No	3.0
Saint-Benoit	0.30	McCutcheon	0.28	5	Canada	No	2.6
unmon seg 5	0.28	McCutcheon	0.28	5	Canada	No	32.3
unmon seg 3	0.14	Tomkin	0.12	3	Canada	No	12.1
unmon seg 8	0.07	Tomkin	0.12	8	Canada	No	7.0
Limekiln	0.00	Gale	0.00	3	Canada	No	2.6
Boynton	0.00	Gale	0.00	5	Canada	No	3.4
Glen	0.00	Gale	0.00	5	Canada	No	8.0

[1] Unmonitored seg 7 chloride concentation was estimated as cerises instead of Ceries * 2.5 as predicted by regression with paved roads because outlet chloride concentration was significantly higher than at Magog Bay suggesting chloride might short circuit this lake segment and go directly into outflow channel.

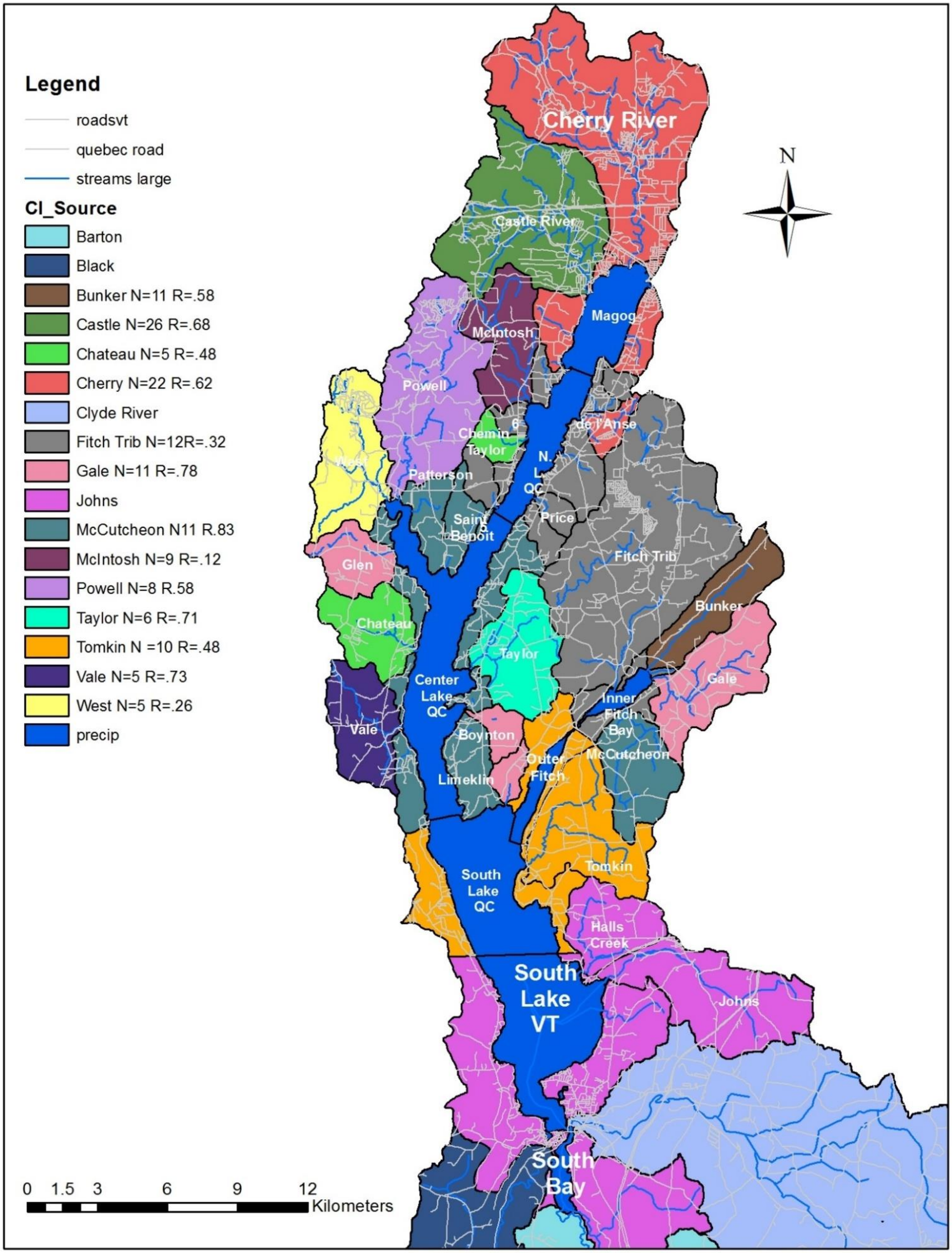


Figure 11. Methodology for estimating Chloride for tributaries, direct drainages and surface of Lake Memphremagog

4.5 Precipitation and Evaporation (5.5% watershed)

Chloride loading from direct precipitation was estimated based on rainfall amounts as described in Chapter 1.1 and a chloride concentration of .2875 mg/l taken from the Lake Champlain Diagnostic-feasibility study (VT DEC 1997). A factor for evaporation (based on monthly evaporation rates and segment areas multiplied by .79 to relate to lake evaporation) was factored in for ice free months (excluding January – March) to remove water but not chloride from Lake Memphremagog.

4.6 In-lake Chloride data

Chloride samples were taken on Lake Memphremagog from 2010 through 2012 through a cooperative arrangement between the Memphremagog Conservation Incorporated (MCI), the Quebec Ministère du Développement durable, de l'Environnement et de la Lutte contre les changements climatiques (MDDEFP) and the Vermont Department of Environmental Conservation (VT DEC). All chloride samples were processed in the LaRosa laboratory in Vermont. Samples taken by VT DEC were taken biweekly from May through October or November at the surface and then every two meters depth to one meter above the bottom while MCI/MDDEFP samples were taken as an integrated sample to one meter depth. Quebec sampling was broken down into two sets of sites one of which was sampled for a longer season roughly May through November and another which was sampled from June through August or September. No difference in chloride concentration in the Vermont samples was evident at different depths although Vermont waters do not stratify so this may not be the case in Quebec. On the other hand the timing of sampling does impact chloride concentrations significantly due to seasonal differences and so the different seasons when samples were taken at different sites has to be considered in establishing differences in chloride concentrations between lake segments.

There are a number of challenges in using the data set from the lake to estimate average annual concentrations of chloride. The primary of these is the relatively small number of samples taken and short season over which these were taken. The chloride concentrations in the lake change seasonally and drop with major runoff events and so there needs to be an adjustment made for the sites which were sampled over a shorter season based on sites which were sampled for a longer season and then we need to extend the longer season to cover the entire 3 year modeled period. Table 4 shows the total data that we have collected and an adjustment factor that was created between sites so that those sampled over a shorter season can be compared directly with those who's data was collected over a longer season by simply adding the difference between the longer and sorter sampling timeframes at the sites where both were sampled to the site which was sampled for a shorter season. A second step to adjust these summer season values to annual means could be done but this would involve using a daily chloride model and that in and of itself includes a lot of assumptions so this hasn't been done in part because the daily model doesn't always match measured chloride levels.

Table 9 Lake Memphremagog chloride data from Vermont and Quebec in 2010 through 2012. Dates in Green were only sampled in Vermont, dates in Black were sampled at extended season in Quebec and Vermont, Red were sampled at all sites and purple were only sampled at Quebec extended sites.

Date	S.BAY	VT lake	Date	M249	M94	M96	M93	M92	M91	M246	M90	M73	M95
VT date	Seg 1	Seg 2	QC date	Seg 2	2 1/2	Seg 3	Seg 4	Seg 8	Seg 5	Seg 6	Seg 7	Outflow	C Bay
4/12/2010	7.49	7.39											
5/18/2010	8.36	7.62	5/4/2010		7.66	7.56			8.06	8.74	8.63		
6/3/2010	10.0	7.74	5/27/2010		7.73	7.67			7.86	8.4			
6/16/2010	7.16	7.63	6/17/2010	7.59	7.42	7.61	5.95	7.42	7.4	7.9	8.08	8.1	7.52
6/30/2010	9.09	7.64	7/11/2010	8	7.97	7.85	6.49	7.79	7.85	7.98	8.18	8.62	7.84
7/14/2010	9.71	7.88	7/28/2010	8.02	7.9		6.19						
7/29/2010	10.2	8.15	7/29/2010			7.95		7.91	8.05	8.1	8.25		7.89
8/11/2010	6.28	7.48	8/17/2010	7.44	7.49	7.41	5.87	7.58	7.64	7.74	7.8	7.85	7.46
8/31/2010	9.12	7.80	9/12/2010	7.85	7.87	7.87	6.31	7.75	7.81	8.12	8.02	8.06	7.88
9/8/2010	10.2	7.98											
9/22/2010		7.96	9/26/2010		7.83	7.84			7.76	8	8.05		
10/5/2010	4.82	7.91	10/17/2010		7.71	7.62			7.72	8.13	8.24		
			11/7/2010		7.39	7.33			7.68	7.88	8.14		
6/3/2011	5.68	6.37	5/11/2011		5.74	5.71			6.88	7.69	7.79		
			5/25/2011		6.04	6.02			6.81	7.11	7.78		
6/15/2011	7.75	6.81	6/11/2011	6.51	6.58	6.49	5.07	6.92	6.84	7.76	7.04	8.45	6.59
6/30/2011	7.24	6.93	7/5/2011	7.06	6.77	6.77	5.4	6.57	6.92	6.8	7.38	7.47	6.72
7/13/2011	9.29	6.86	7/27/2011	7.45	6.87	6.85	5.27	6.92	6.73	7.04	6.98	7.74	6.65
7/27/2011	11.0	7.11	8/17/2011	6.99	6.95	7.03	5.37	6.81	6.85	6.87	7.14	7.22	6.7
8/24/2011	9.48	7.01											
9/21/2011	7.28	6.69	9/6/2011		6.79	7.03			6.9				
10/11/2011	6.33	6.77	9/7/2011							6.75	6.98		
10/25/2011	6.45	6.62	9/25/2011		6.95	6.65			6.78	6.98	6.92		
11/9/2011	7.81	6.48	10/10/2011		7	6.73			6.64	6.84	6.85		
5/23/2012	7.04	7.15	10/31/2011		6.68	6.63			6.84	6.48	7.13		
6/7/2012	6.19	7.60	5/30/2012	7.19	7.28	7.34	5.22	7.20	7.22	7.32	8.03	8.01	7.00
6/20/2012	8.58	7.55	6/21/2012	7.38	7.36	7.54	5.91	7.32	7.26	7.55	7.75	8.59	7.50
7/3/2012	9.07	7.48	6/22/2012		7.36								
7/17/2012	9.14	7.58	7/11/2012	7.61	7.61	7.33	5.96	7.39	7.41	7.78	7.94	7.84	7.55
7/30/2012	9.57	7.92	7/31/2012	7.56	7.33	7.46	5.85	7.35	7.38	7.71	7.63	7.81	7.54
8/29/2012	11.28	7.96	8/8/2012	7.71	7.71	7.70	6.04	7.70	7.70	7.64	7.98	8.11	7.67
9/11/2012	6.79	7.71	9/11/2012			7.44	5.86	7.41	7.27	7.48	7.60		
10/10/2012	8.42	7.95											
11/19/2012	8.01	7.93											
Short Ave.	8.78	7.53		7.45	7.36	7.38	5.78	7.34	7.36	7.59	7.72	7.99	7.32
Long Ave.	8.25	7.39		***7.31***[1]	7.22	7.20	**5.61**[2]	**7.16**[2]	7.30	7.56	7.67	***7.94***[3]	
Short - long	0.55	.137			.141	.172			.052	.029	.047		
95% conf.	0.63	0.19		0.22	0.23	0.23	0.21	0.20	0.18	0.23	0.20	0.22	0.25

1. *Adjusted to longer season based on Long vs short season difference of .142 mg/l at M94*
2. *Adjusted to longer season based on Long vs short season difference of .172 mg/l at M96*
3. *Adjusted to longer season based on Long vs short season difference of .047 mg/l at M90*

4.7 Chloride loading calibration with Inner Fitch Bay and South Bay.

Measured lake concentrations for South Bay and Fitch Bay can be used to determine the accuracy of chloride loading estimates from tributaries because these lake segments have limited exchange with other lake segments and so exchange rate should not impact chloride concentrations by more than a few percent over a reasonable range of exchange values. The modeled and measured chloride concentrations in south bay closely match the measured chloride concentrations suggesting that chloride discharge relationships and estimated discharge are accurately depicting chloride loading to this bay. On an average basis the estimated values were about 2 percent higher than measured values in South Bay with an average difference on a given date of 4 percent which given the wide variability in chloride concentrations from 5 to 11 mg/l is quite close.

The modeled chloride concentrations in Fitch Bay are significantly higher than the measured levels on Fitch Bay by 15% and the modeled values appear to be more variable than those measured. Concentrations in this bay depend greatly on the contribution of the higher chloride concentration Fitch bay tributary vs the lower chloride concentration Gale and Bunker tributaries. However, since flows are extrapolated from hydrotel model through a relationship with the Coaticook the relationship in flow is not precise and a likely reason for this inconsistency. In terms of water loading per watershed area the hydrotel and hydrotel correlation estimate for Fitch bay tributary from 2010-2012 was 25% higher than for the Bunker tributary with other tributaries having watershed loading of values in-between. Flows were reduced by 20% for the Fitch Tributary and increased by 20% for the bunker tributary but chloride levels remained elevated by about .3mg/l but was the greatest adjustment in the flows that could be justified based on equalizing the flow per Km^2 for the two watersheds.

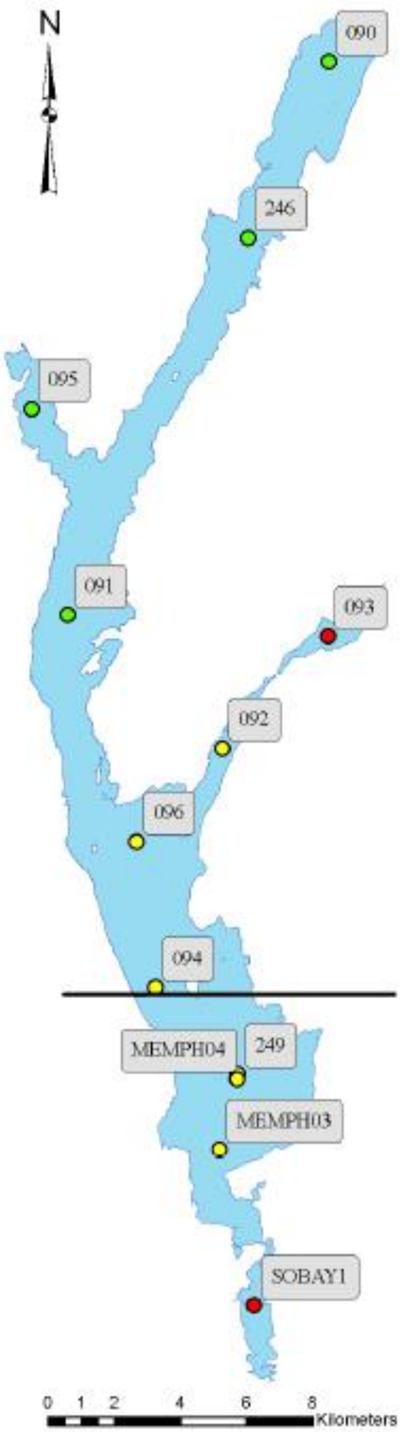
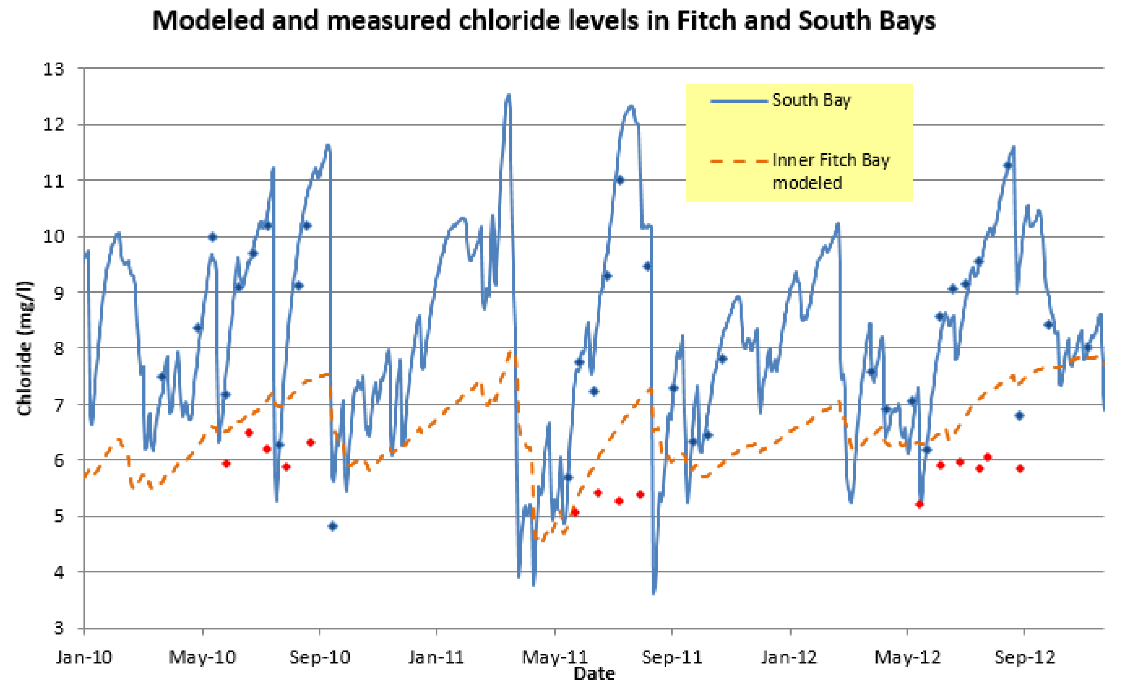
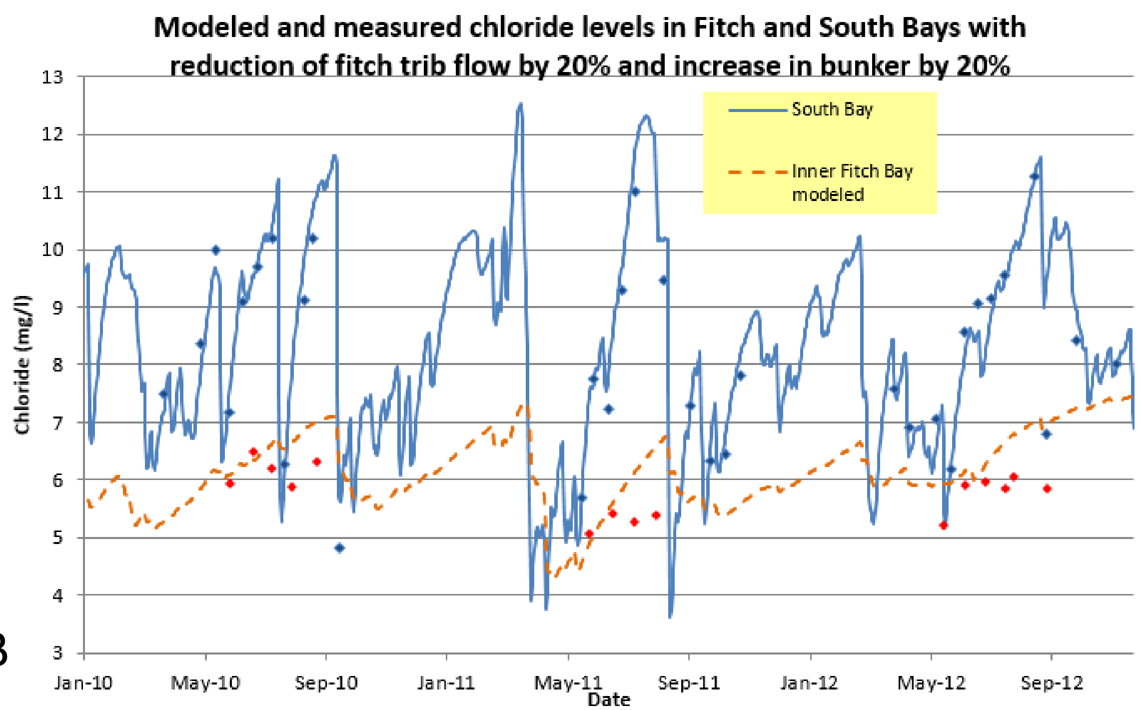


Figure 12. Water quality sampling sites in Lake Memphremagog.



A



B

Figure 13 A. Measured and modeled chloride concentrations in South and Fitch Bays.

Figure 13 B. Measured and modeled chloride concentrations in South and Fitch Bays with an adjustment of flows coming into Fitch Bay (increasing low concentrations and increasing high concentration) which allows this to match measured levels in inner Fitch bay.

In addition to looking at the mass balance for these bays it is possible to compare the mean inflow concentration across the whole lake which after factoring in rainfall and evaporation should be the same as the outflow concentration from the lake. Overall the loading estimated chloride concentrations at the outflow of 7.82 mg/l or 0.12mg/l lower than measured at the outflow from Magog bay of 7.94 mg/l or about 1.5% lower. To complicate matters the chloride concentration in the Center of Magog Bay averaged 7.67 or 0.27 mg/l lower than the outflow suggesting that there is a significant amount of chloride, about 325 metric tons in total, that is leaving the bay without mixing into the Magog lake segment so in effect short circuiting the segment and larger lake. On top of this a significant percentage of the chloride into the Magog Lake segment is predicted to come from the unmonitored watershed which has a road density of 2.5 times that of the highest monitored watershed meaning there is a large uncertainty around the actual chloride loading from this portion of the watershed.

While modeling for Lake Memphremagog was done assuming steady state conditions a daily time step model was also built. For a number of reasons this daily model wasn't used to calibrate the exchange between lake segments or phosphorus sedimentation. However, this model was helpful in evaluating chloride loading to South Bay and Fitch Bay segments which have little exchange with adjacent lake segments and for providing some understanding seasonal variation in lake concentrations. Figure 14 below shows the measured and predicted in lake chloride concentrations for three lake segments showing that the model using exchange rates calibrated through the steady state model described later in this report did a reasonable job showing trends in lake concentrations particularly with regards to a large drop in chloride concentrations after major rain events in 2011.

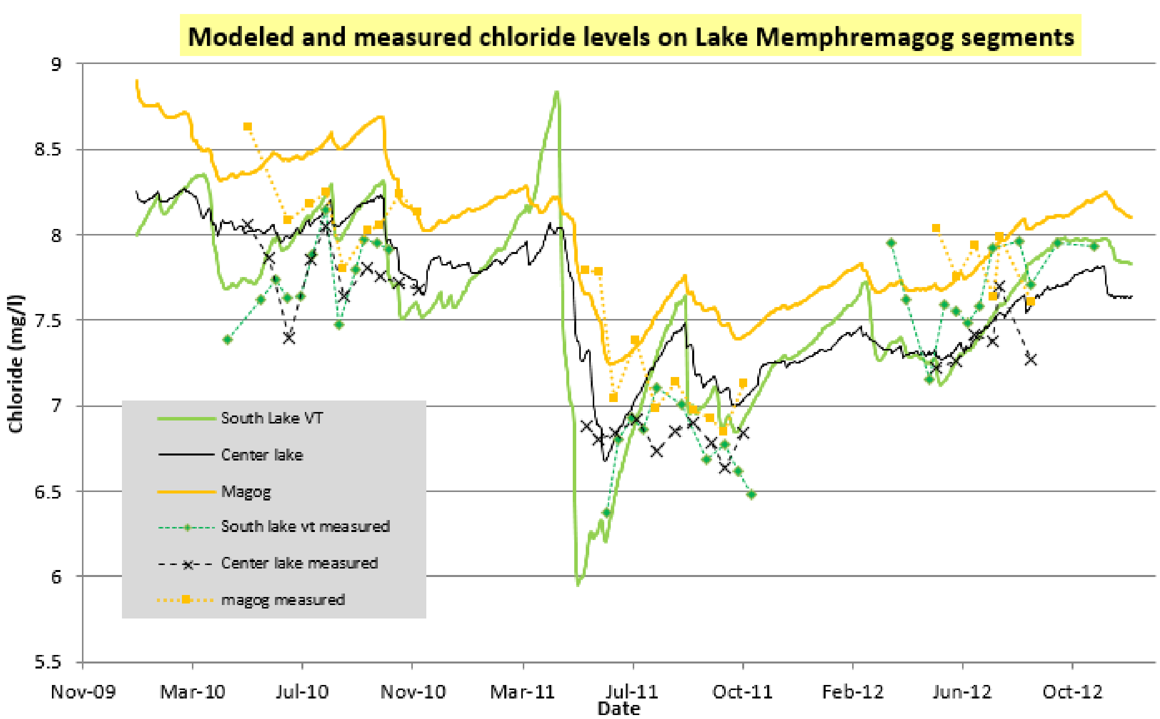


Figure 14 measured vs estimated chloride levels on Vermont, Center and Magog Lake Segments using a daily model showing both measured and modeled depression of chloride levels from Spring 2011 runoff event.

5 Estimating Phosphorus for Tributaries and direct drainages to Lake Memphremagog.

Timeframe for in-lake modeling: 1/1/2005 through 12/31/2013
Calibration time period 1/1/2009 through 12/31/2013
Verification time period 1/1/2005 through 12/31/2008
Timeframe for developing Chloride / flow relationships for tributaries in: Vermont from 2010-2013
Quebec from 2010 – 2013

5.1 Sources for Phosphorus loading to Lake Memphremagog

Vermont water quality sampling – Barton, Black, Clyde, and Johns (66.2% watershed) Phosphorus loading was estimated using the relationship between phosphorus and discharge which was strong for all four tributaries. Sampling was done from 2005 through 2013 with a total number of phosphorus samples of 137 for the Black River, 147 for the Barton River, 138 for the Clyde River, and 104 for the Johns River and all samples were processed by the VT DEC laboratory. Sampling frequency varied from monthly with high flow samples taken when flows were above the Q90% as measured at the Black River as has been done since 2013, Biweekly sampling plus samples taken when flows were above 90% discharge from 2010-2012 to monthly sampling and sampling after select rain events from 2005 through 2009. Sampling on the Black, Barton, and Clyde Rivers was done using the DH 59 "bomb" sampler at the centroid of flow to capture a single depth integrated sample. Samples on the smaller Johns River were taken as grab samples at the centroid of flow. Quality assurance measures were met for phosphorus for all years across both lake and tributary samples with an average RPD of 7.3 % and there were no instances of blank samples above detection.

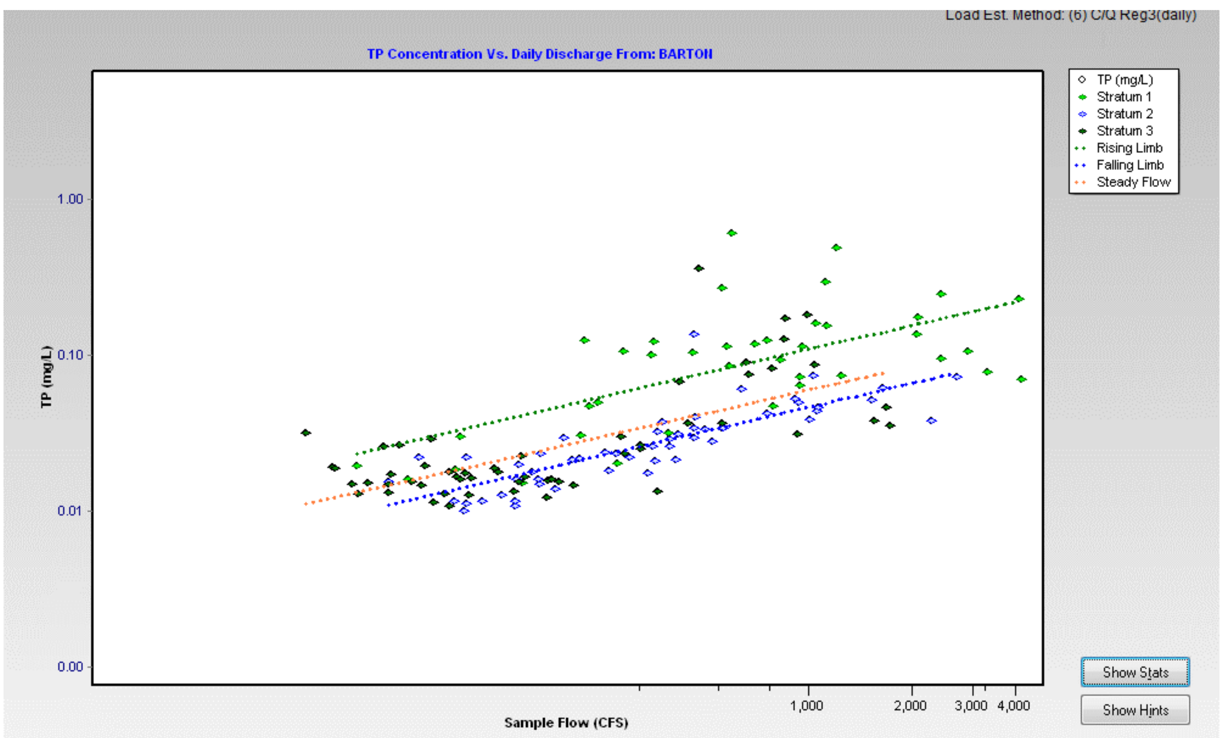


Figure 15 relationship between Phosphorus concentration and discharge on the Barton River from 2005-2013 stratified based on rising, falling limbs of the hydrograph or Steady flow using Flux program.

The Flux Program (Walker 1999) was used to estimate daily phosphorus loading for each tributary which was done with a relatively low coefficient of variation for each tributary between 0.04 and 0.15. The Flux program allows data to be segregated into strata based on rising falling or stable portions of the hydrograph, based on flow or based on seasonal variations. Data for each tributary was loaded into the Flux program and various strata assignments were evaluated along with alternative load estimation methods.

All loading estimates were calculated using Method 6 in the flux program which establishes an Ln/Ln relationship between phosphorus and discharge for each strata assignment and then adds a factor, calculated as the standard error of the estimate divided by 2, to correct for a bias resulting from the log transformation. The Flux program calculates the error variance through a Jackknife procedure. However, this estimation of error might not be accurate if there is systematic bias in the sample input.

In general, the strata assignments with the lowest coefficient of variation were chosen, except in cases where this appeared to create some bias based on an evaluation of a residual plot of phosphorus loading vs flow. For the Barton and Clyde Rivers separation on the Hydrograph had the lowest coefficient of variation. While the Black River had a slightly lower CV for separation on flow a review of the residual plot vs discharge suggested that there was less bias at higher flow when this was separated on the Hydrograph so separation on the hydrograph was used. While the Johns River without any segmentation had the lowest CV this was very close to the CV for separation on the hydrograph and so this was chosen to maintain consistency between load estimation methods for the four major tributaries.

Table 10. Comparison of Load estimation approaches for the Black, Barton, Clyde and Johns River from 2005- 2013 using Flux with alternative strata assignments. Strata assignments highlighted in red were used to estimate loading.

	Strata 1			**Strata 2**			**Strata 3**			**annual load and statistics**			
River/ stratification	Strata	*slope*	R^2	Strata	*slope*	R^2	Strata	*slope*	R^2	*Load (kg)*	*Var.*	*mg/L*	*C.V.*
Barton/hydro	**Rising**	**0.509**	**0.403**	**Falling**	**0.516**	**0.669**	**Stable**	**0.502**	**0.478**	**18805**	**19662**	**0.0639**	**0.096**
Barton/flow	**<Qmean**	0.274	0.122	**>Qmean**	0.435	0.126				19905	28218	0.0680	0.109
Black/hydro	**Rising**	**0.603**	**0.427**	**Falling**	**0.327**	**0.353**	**Stable**	**0.582**	**0.481**	**23777**	**61236**	**0.0944**	**0.130**
Black/flow	**<Qmean**	0.248	0.052	**>QMean**	0.793	0.326				24138	43681	0.0959	0.110
Black/Hydro using inst. flow	**Rising**	0.589	0.440	**Falling**	0.324	0.350	**Stable**	0.589	0.520	23503	56325	0.0934	0.130
Clyde/hydro	**Rising**	**0.225**	**0.181**	**Falling**	**0.028**	**0.007**	**Stable**	**0.051**	**0.023**	**6886**	**456.1**	**0.0236**	**0.040**
Clyde/flow 2	**<140 cfs**	-0.133	0.054	**>140 cfs**	0.182	0.102				7170	639	0.0246	0.045
Clyde/flow 3	**<140 cfs**	-0.133	0.054	**<1400 cfs**	0.107	0.028	**>1400**	0.739	0.536	7110	589.4	0.0244	0.044
Johns/NO seg	**All**	0.699	0.433							1154	164.4	0.0720	0.140
Johns/Hydro	**Rising**	**0.625**	**0.325**	**Falling**	**0.521**	**0.498**	**Stable**	**0.842**	**0.495**	**1275**	**220.6**	**0.0795**	**0.150**
Johns/flow	**<Qmean**	0.164	0.018	**>QMean**	0.970	0.279				1151	303	0.0718	0.190

Residual plots were also evaluated with respect to residuals over time and season. With the exception of the Johns River these plots did not show any trends over time, which suggests that for the Black Barton and Clyde it is appropriate to use the full dataset in developing the phosphorus discharge relationships. Table 10 presents the alternative strata assignments considered, slope and R^2 of the Ln/Ln relationships and resulting annual load, variance, mean concentration and coefficient of variance for each approach with the selected approach highlighted in red. For the Black and Barton River using flows at the time samples were taken was also considered however this did not change the resulting variance or loading significantly so did not justify the additional work this required and loss of data from dates where sample flows were not available.

Other methods for calculating loading were considered including the Loadest program, Graphical Loading Analysis System program and WRTDS program but for a variety of reasons including insufficient data, lack of ability for separating data on the hydrograph, or complexity of data analysis process the flux program appeared to be the best program to use to estimate phosphorus loading.

There is a larger difference in the Quebec and Vermont datasets between the 2005-2008 and the 2009-2012 timeframes where Quebec data shows a significant drop in phosphorus concentrations over this timeframe while Vermont data suggests stable lake concentrations. This is even evident at the one site which was sampled by both countries which suggests that there is not a distinct trend in the northern part of the state but some difference driving this difference. Because of this loading estimates were exported from flux and summed just for 2009 – 2012. This results in a loading estimate for the Black, Barton Johns and Clyde Rivers of 22,622, 18,858, 1,316, and 6,420kg per year respectively.

Table 11 Estimated annual phosphorus loading for the Black Barton Clyde and Johns Rivers from 2005 through 2013

	Black	Barton	Clyde	Johns	Total
2005	23607	7039	17988	1229	49724
2006	24520	18166	7462	1185	51334
2007	23503	15923	6426	1214	47065
2008	30213	22467	7538	1755	61973
2009	**18897**	**14122**	**5735**	**984**	**39739**
2010	**19399**	**17101**	**5829**	**1290**	**43619**
2011	**38450**	**35051**	**9635**	**2230**	**85365**
2012	**13741**	**9160**	**4481**	**761**	**28144**
2013	21663	19256	6357	884	48161
Average	23770	18799	6719	1281	50569
2009-12 Ave.	22622	18858	6420	1316	49217
Difference	-5.1%	0.3%	-4.7%	2.7%	-2.7%

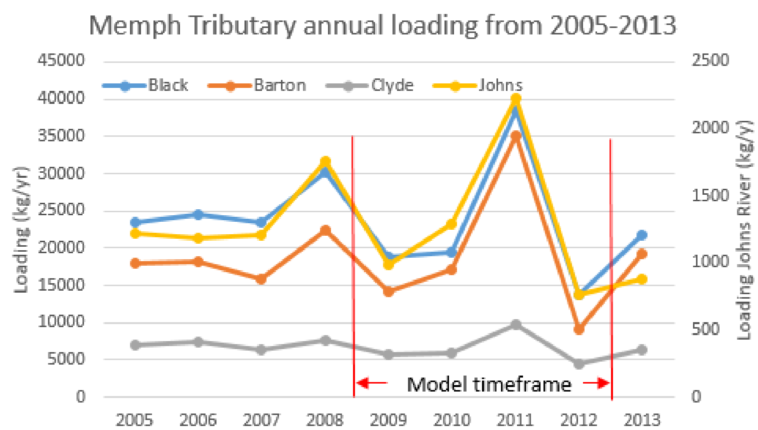


Figure 16 Graph showing annual phosphorus loading for the Black Barton Clyde and Johns Rivers from 2005 - 2013.

5.2 Vermont minor tributary loading estimates

An additional dataset to estimate loading in Lake Memphremagog tributaries is available through a water quality monitoring program supported through the LaRosa partnership program that was initiated by the NorthWoods Stewardship Center but is now lead by the Fritz Gerhardt of Beck Pond LLC in collaboration with the Memphremagog Watershed Association. Over 140 sites in the watershed have been sampled for one or more years since 2005. All samples were done in accordance with an approved QAPP approved by VT DEC. Since 2010 sampling has consisted of 6 monthly sampling events and two sampling events targeted towards runoff conditions (at least .75” of rain in the 24 hours prior to sampling), and in prior years sampling has included bi weekly sampling for 2008 and 2009 or in 2005 and 2006 sampling monthly 6 times per year. The purpose for estimating loading from these sites is as another avenue to calibrate the land use phosphorus export model.

Twenty-four watersheds were selected for estimating loading out of the 140 subwatersheds that have been sampled based on a minimum number of samples and targeting watersheds that matched the subwatersheds that have been developed in a land use phosphorus export model. Daily discharges at sample sites were estimated based on watershed size ratio with the Johns River and daily flows on the Johns River monitored by VT DEC. Loading was calculated from 2005 through October 2015 by using these estimated daily flows and sample phosphorus values using the FLUX program. For most sites one

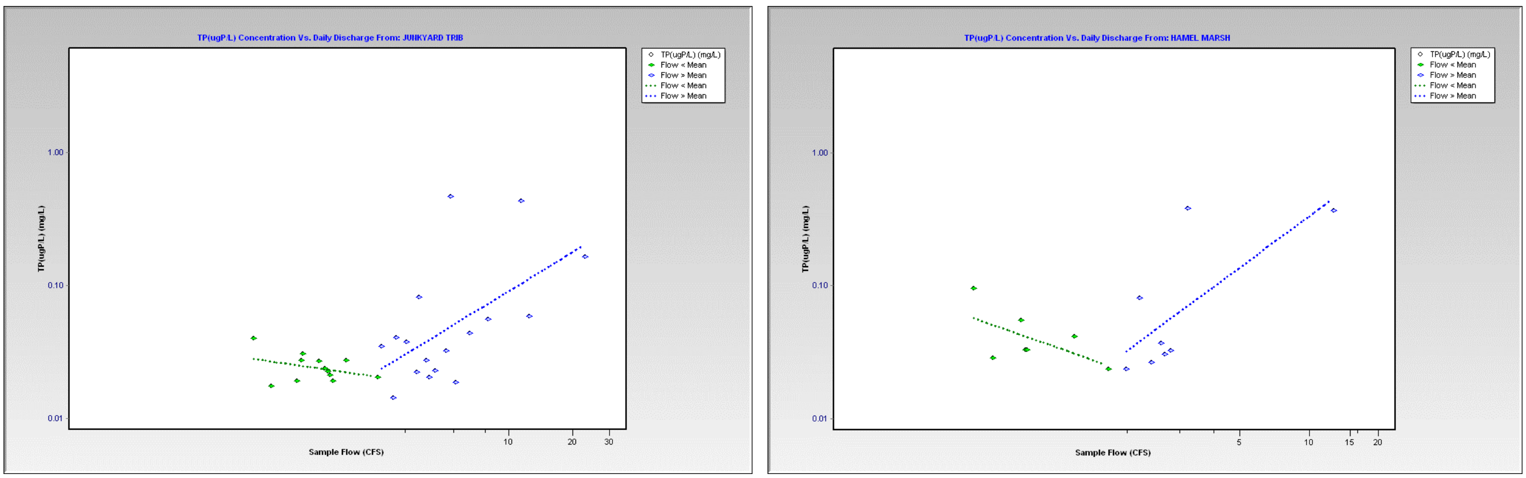


Figure 17 Examples of Stage discharge relationships for the Rock Junkyard and Hamel marsh tributaries. Hamel Marsh tributary was included from analysis due to extremely high coefficient of variation calculated through the Flux program.

stratum was used in the flux program but where there appeared to be a different relationship between phosphorus and discharge above and below the median flow value, samples were broken into two strata when this improved the stage discharge relationship. One site (Hamell marsh) was excluded after the flux analysis due to large coefficient of variance in the flux program.

Table 12 Vermont minor tributary Flux phosphorus loading estimates based on LaRosa volunteer monitoring for 24 sites.

Watershed	kg/yr	c.v.	N	Area (km)	Kg/ km^2	deve-loped	road	hay/ pasture	crop	farm-stead	forest
Airport Tributary	232.5	0.4	22	3.0	78.7	5.0%	1.4%	32.3%	6.6%	0.0%	54.5%
Brighton Brook	1188.9	0.15	44	24.8	47.9	2.4%	1.0%	23.9%	7.8%	0.8%	62.4%
Brighton Brook North	772.4	0.24	38	9.3	83.4	2.4%	1.6%	30.5%	13.0%	1.9%	50.4%
Cass Brook	182.9	0.26	9	8.9	20.6	2.3%	1.1%	12.0%	1.6%	0.2%	82.0%
Crystal Brook	231.2	0.52	38	4.0	57.5	3.9%	1.9%	37.2%	3.2%	1.4%	51.9%
East Side	12.1	0.27	18	0.4	28.7	81.4%	11.9%	0.5%	1.6%	0.0%	4.7%
Holbrook Bay	99.7	0.13	19	3.1	32.2	5.9%	1.1%	42.2%	0.1%	0.5%	49.6%
Lamphrere Brook	126.0	0.146	8	13.1	9.6	0.9%	0.4%	2.5%	0.3%	0.0%	95.3%
Lords Creek	1656.7	0.87	15	41.4	40.0	2.6%	1.2%	21.2%	0.0%	0.4%	73.3%
McCleary Brook	60.6	0.22	7	7.5	8.1	0.9%	0.8%	6.9%	0.0%	0.0%	90.9%

Middle Hamel Tributary	74.1	0.49	20	0.8	96.1	1.4%	0.6%	31.4%	25.7%	0.0%	40.3%
Roaring Brook	598.4	0.27	15	29.3	20.4	3.2%	1.5%	20.7%	1.0%	0.6%	68.6%
Rock Junkyard	289.8	0.38	29	3.5	83.3	15.2%	3.5%	46.5%	9.8%	1.1%	23.8%
Rogers Branch	82.5	0.21	18	7.8	10.5	2.0%	0.7%	6.7%	1.9%	0.2%	87.5%
Seaver Branch	130.0	0.14	23	9.6	13.6	1.3%	0.5%	2.3%	0.0%	0.0%	94.4%
Shalney Branch	274.9	0.55	31	7.0	39.5	1.9%	1.3%	21.5%	0.0%	0.3%	72.9%
Stoney Brook	660.0	0.26	31	17.3	38.2	6.4%	1.2%	22.5%	4.4%	0.1%	62.4%
Strawberry acres	70.9	0.21	31	1.7	42.4	8.6%	2.6%	32.4%	0.4%	1.4%	54.0%
Sunset Acres	153.8	0.54	27	3.7	41.2	1.7%	1.4%	23.4%	14.1%	0.1%	58.4%
Upper Hamel Tributary	164.1	0.31	20	0.9	182.4	3.0%	1.1%	43.1%	36.5%	2.5%	13.8%
Upper Lords Creek	843.6	0.76	15	28.9	29.2	2.3%	1.3%	19.0%	0.1%	0.4%	75.6%
Ware brook	181.2	0.34	15	9.8	18.6	1.4%	1.0%	13.5%	3.1%	0.1%	79.0%
Whetstone brook	333.3	0.21	15	14.9	22.4	3.4%	1.4%	14.9%	3.5%	0.6%	73.6%
Wishing well	294.6	0.38	27	3.2	93.1	7.1%	2.7%	52.0%	0.7%	1.1%	36.0%
Johns River	1025.3	0.35	49								
Hamel marsh	278.0	4.78	15								

An estimate of annual loading for the Johns River watershed using this loading estimation method was 22% lower as compared to the loading estimated through the more intensive and event focused sampling done on the four major tributaries which suggests that this methodology may underestimate loading. An analysis of the loading comparing the loading estimates from these smaller tributaries and the four major tributaries using the land use phosphorus export model suggested that on average the loading estimates from these smaller tributaries were about 42% lower than for the four major tributaries. A likely explanation for a systematic underestimation of phosphorus loading is that limited sampling of these smaller streams through this methodology under high flow conditions may not capture extremely high values after intense runoff events that only occur infrequently. Also, many of the smaller streams sampled are much smaller than the 4 major tributaries and so phosphorus values can be expected to raise and fall much quicker after rain events as opposed to the larger four major tributaries which may add to the underestimation of phosphorus in addition to reduced event sampling frequency. In spite of the fact that these loading estimates have higher uncertainties than the estimates for the four major tributaries and may be biased low, the wide variability in land use between these smaller watersheds provides another avenue to evaluate the calibration of phosphorus export coefficients for the watershed export model. Similar attempts to estimate loading from Quebec Tributaries raised a number of concerns as described in Appendix B so were not used.

6 Land use phosphorus export model

A simple land use export model was developed collaboratively by the Quebec Vermont Technical Committee on Lake Memphremagog in August of 2009 through a private consultant SMi Amenatech inc. hired by the MRC in Quebec (Vezina 2009). The Smi model uses literature phosphorus export values to estimate loading but then estimates retention in lakes larger than 4 hectares to approximate phosphorus loss in the watershed. This model was developed with support from Vermont DEC and partners in Quebec as part of the technical committee of the Quebec Vermont Steering Committee on Lake Memphremagog. Loading estimates were generated through a number of literature sources most of which were identified in the final report although a few sources are identified as in press and have not been published to date, and other sources were not specifically identified. For some export values the export coefficient was selected through three different sources for the low medium and high export coefficients. Through the technical committee there was significant discussion around the estimate for wetland loading. A literature review shows loading estimates that range from negative (net retention of phosphorus) to the values used in the Smi model which were at the high end of the range. Croplands were broken down by soil drainage categories based on a study of the Missisquoi basin but other land use phosphorus export coefficients were applied across all land uses. Even using the high potential export values the modeled loading for the four major tributaries was lower than measured at these tributaries suggesting that further calibration was necessary.

Table 13. Low Medium and High export coefficients used in the original SMi lake Memphremagog phosphorus export model.

	Export coefficient Kg/km/yr			
land use name	**Low**	**Medium**	**High**	**Source**
Forest	2[1]	5[2]	7*	[1] Michaud et al. 2006: modeling in Monteregie [2]Carignan et al. in press) multiple regression between annual inputs and area measured in the eastern townships and the Laurentians and used for southern Ontario.
Residential	50*	106*	195[1]	[1] Michaud et al. 2006: modeling in Monteregie
Barren Land	305*	305*	305*	*mean coefficient calculated according to literature (not identified)
Pasture/Hay	36[1]	52[6]	56*	[6] Carignan et al. (in press) multiple regression between annual inputs and area measured in eastern townships
shrub/herb	20[3]	20[3]	20[3]	[3]Carignan(2003) in Hemisphrees (2008): Limnological monitoring of Lake Heney
Water	3[4]	6[4]	9[4]	[4]Carignan et al. in press
Wetland	125[5]	170*	214[6]	[5] carignan et al. (in presss) adapted according to the laurentian data
Terrain de golf	105[7]	105[7]	105[7]	Nero and Corbeil(2002) Hemispheres (2008).
Cultivated A soils	39[1]	56[1]	108[1]	[1] Michaud et al. 2006: modeling in Monteregie
Cultivated B soils	87[1]	139[1]	277[1]	[1] Michaud et al. 2006: modeling in Monteregie
Cultivated C soils	59[1]	107[1]	213[1]	[1] Michaud et al. 2006: modeling in Monteregie
Cultivated D soils	107[1]	110[1]	318[1]	[1] Michaud et al. 2006: modeling in Monteregie
Cultivated unknown soils	59[1]	107[1]	213[1]	[1] Michaud et al. 2006: modeling in Monteregie

For this model the Lake Memphremagog basin is broken down into 338 subwatersheds and loading is summed between the land use phosphorus export values and septic and WWTF loading in these watersheds. A retention factor is estimated for all lakes above 4 hectors with the following equation:

Retention=1/(1+1/residence time$^{.5}$).

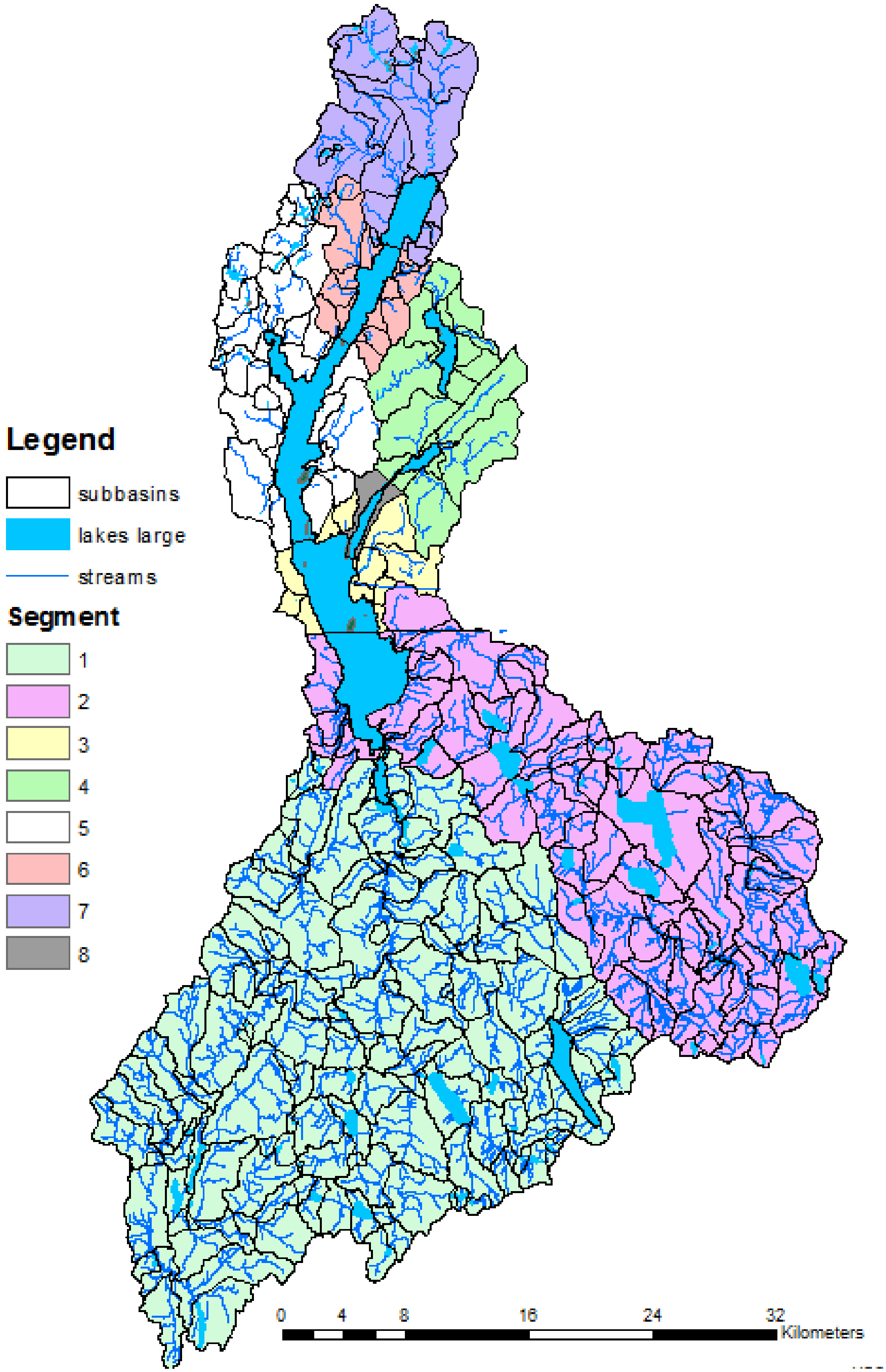


Figure 18 map of 338 subbasins broken down by 8 lake segments.

In addition to this, loading from septic systems in the Smi model was estimated based on an average export of .192 kg/yr for systems within 300 meters of a surface water and excluding houses served by WWTF with a total of 13710 systems. The estimate of 0.192 kg/yr was based in a Quebec an study that estimated loading of .064 kg/yr per person and assumes an average of 3 people per house although this was based on the analysis of a single septic system.

6.1 Land use export model updates

A number of adjustments to this export model were considered to make the land use more closely match that from the Lake Champlain Scenario tool so aspects of this could be borrowed in the development of the Lake Memphremagog TMDL. The key changes made were the addition of farmstead land use, paved and dirt road land uses, the addition of a streambank erosion component along with some adjustments to the Septic loading. A catalogue of updates to the model is presented in Appendix A including a discussion of adjustments to the retention equation including selecting a minimum level of retention of 14% regardless of lake residence time as was necessary to calibrate loading in the Clyde River watershed.

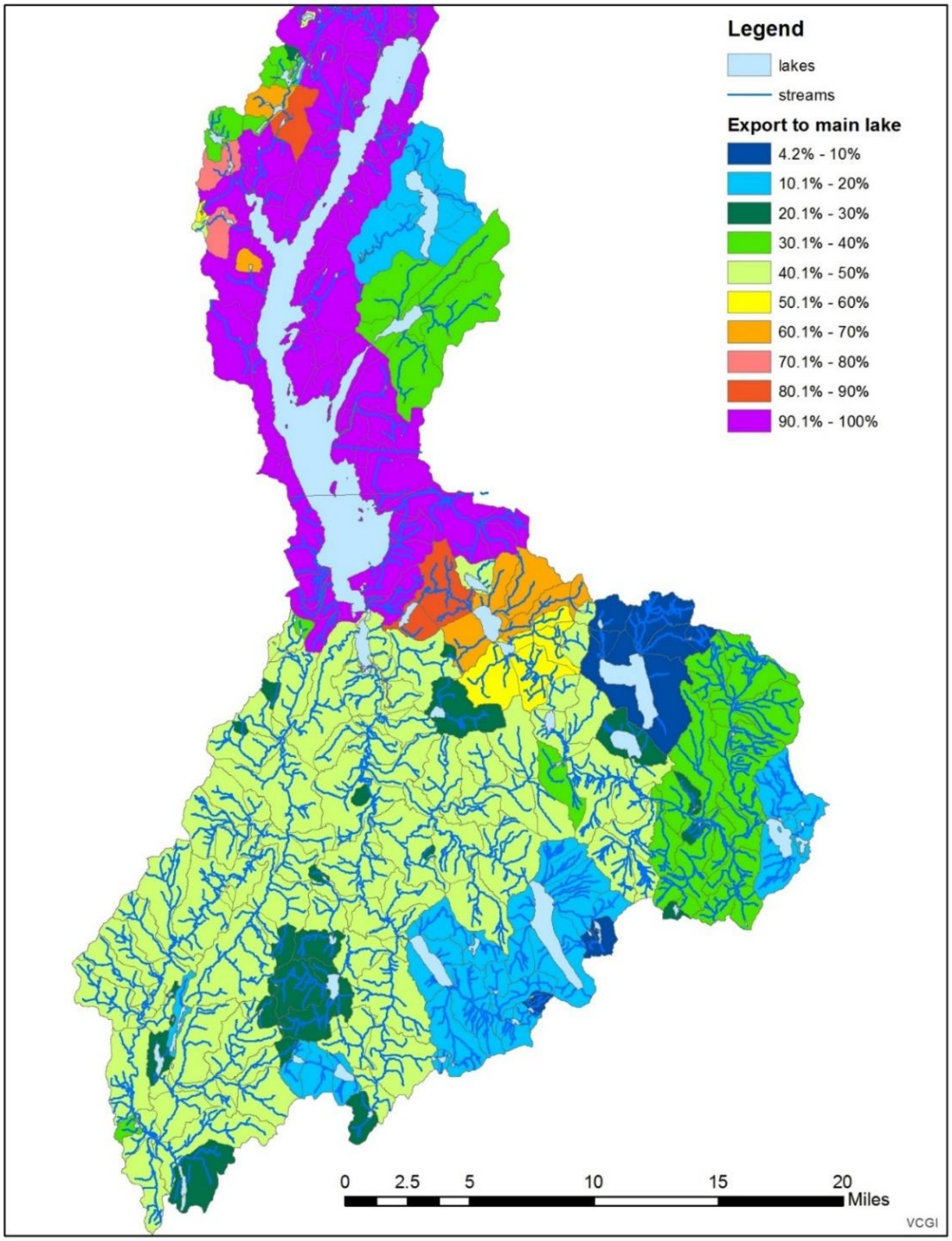


Figure 19 estimated retention of phosphorus to main Lake (including South Bay and Fitch Bay retention) using the R1 retention equation with a minimum lake retention of 14%

6.2 Septic loading.

Based on the fact that that the septic loading methodology used in the SMI model was based on the study of one septic system in Quebec an alternative approach to estimating septic loading was developed with input from Louis Roy and an evaluation of the literature. There is a broad array of methodologies for estimating septic loading in Phosphorus TMDL's and modeling studies from not explicitly separating out this loading source in Lake Champlain TMDL, only including loading of septic systems on the impaired lake itself (Ticklenaked Pond TMDL), estimating loading only from failed or poorly functioning septic systems and assuming no phosphorus loading from well-functioning systems, to a estimating a load per system such as was done in Massachusetts model where the estimated contribution from septic systems was 0.5kg/system per year (Mattson 1999).

The approach that has been selected for the Lake Memphremagog watershed includes the following factors:

1) Estimated phosphorus load to septic systems has been reduced in Vermont through laundry and dishwasher detergent bans and so estimated load per person per day was 2.1 grams (Stone environmental, 2005) which equates to .767 kg/yr at 2.5 people per house equals 1.9 kg/yr/house.
2) Estimating phosphorus loss in Septic tank (30%) results in a loading of 1.33kg/system
3) Estimating an export factor in soils which was selected at 40% as an average of the loss found by Robetson (1998) which ranged from 1% to 77% depending on a number factors related to soil type and chemistry. Soils types in the Lake Memphremagog watershed were likely to have higher retention due to lack of calcareous parent material however another factor to be considered is that there are also a percentage of septic systems that may be surfacing or not functioning properly where retention would be reduced. Using an average soil export factor of 40% results in a loading of 0.532 kg/system per year.
4) Finally loading was reduced based on the distance to surface waters with values of loading exported phosphorus used equaling 87.5% for 0-50 Meters, 62.5% for 50 to 100 meters, 37.5% for 100 to 150 meters 12.5% for 150 to 200 meters.
5) The justification for this is based on the loss of phosphorus in ground water over extended distances where in the same study referenced above migration rates of phosphorus plumes in ground water were generally less than 1 meter per year (Robertson 2008).

This analysis was done by selecting systems in each buffer ring using the e911 layer with areas of sewer service removed from the analysis. The total estimated loading from septic systems to surface waters was then estimated for systems in each buffer area as shown below in table 14 prior to applying lake retention.

Table 14. Count and estimated septic system loading in Vermont and Quebec for septic systems at different buffer distances to surface waters in the Lake Memphremagog Watershed.

Buffer distance (m)	Kg P/system/y	Systems in VT	Systems in QC	VT load (kg)	QC load (kg)
0-50	0.532	2470	1807	1314.0	961.3
50-100	0.266	1815	1070	482.8	284.6
100-150	0.133	1194	796	158.8	105.9
150-200	0.0665	968	608	64.4	40.4
200-250	0.03325	729	496	24.2	16.5

6.3 WWTF loading

WWTF loading was estimated based on monitoring of monthly TP concentrations and flows except for Brighton which only has one annual TP test.

Table 15. Loading of WWTF used in watershed export model in Kilograms.

WWTF	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	Ave.	Ave 09-12	Permit load
BARTON	123.8	123.0	180.8	116.1	130.4	115.2	89.5	114.2	120.4	122.4	123.6	112.8	364.2
BRIGHTON	391.5	388.5	375.5	295.8	256.3	361.6	268.5	315.9	319.7	310.9	328.4	295.6	1036
NEWPORT	454.0	520.1	501.9	397.8	346.1	440.4	380.1	317.2	364.7	311.6	403.4	391.1	1324.3
ORLEANS	69.0	46.2	47.3	27.5	34.0	47.4	31.9	42.2	34.5	29.8	41.0	35.2	264.9
Total	1038.3	1077.8	1105.5	837.1	766.8	964.6	770.1	789.5	839.4	774.8	896.4	834.6	2989.4

Note: Brighton concentration based on average of annual phosphorus test taken since 2007 of 3.34 mg/l

Note: Brighton has no permit requirement for phosphorus so used 5mg/l to approximate maximum likely concentration

Table 16. Quebec WWTF loading based on 2013-2014 monthly monitoring data in Kg/yr

	2013	2014	Average	Loading in Kg rounded for model	Original loading used in Smi model (kg)
Potton Owls head	43.21	48.19	45.70	46	NA
Stanstead Fitch	10.96	9.43	10.19	10	11
Stanstead Georgeville	0.45	0.90	0.67	1	3.75
St Benoit*			2.80	3	11

* average for first 9 months of 2015 prorated to full year

6.4 Streambank erosion estimation

Without a SWAT model for the Lake Memphremagog watershed there are limited options for estimating loading from streams. The approach used to estimate streambank loading was to use the waterbody GIS layer which was created from black and white photos taken at a known year. This waterbody shapefile was then edited by overlaying this over more recent black and white photos which appear to be taken at similar water levels to map where rivers have adjusted coarse in the intervening years. These two shapefiles were then overlaid to identify areas of new erosion and areas of new bar development based on where these overlap or do not overlap. For most of these locations Phase 2 stream geomorphic assessments are available and streambank height is available from the FIT erosion data layer. A lower height based on an assumption that bar development would be about up to the bankful height for newly created bars allows for the calculation of erosion volume by the equation: Area of Erosion x bank height - area of bar development x Bar height = total erosion volume for the years between photos. The year that photos were taken was used to divide the volume lost by the total number of years to equal the annual volume lost. Where the waterbody layer doesn't cover all significantly eroding reaches such as the Brownington branch, Johns River, and lower reaches of Shalney, Mcleary, Lamphear brooks, the waterbody shapefile was extended based on the earlier aerial photos before modifying these for the more recent photos. Where SGA FIT data was not available for bank height this was estimated based on personal knowledge of these few stream segments.

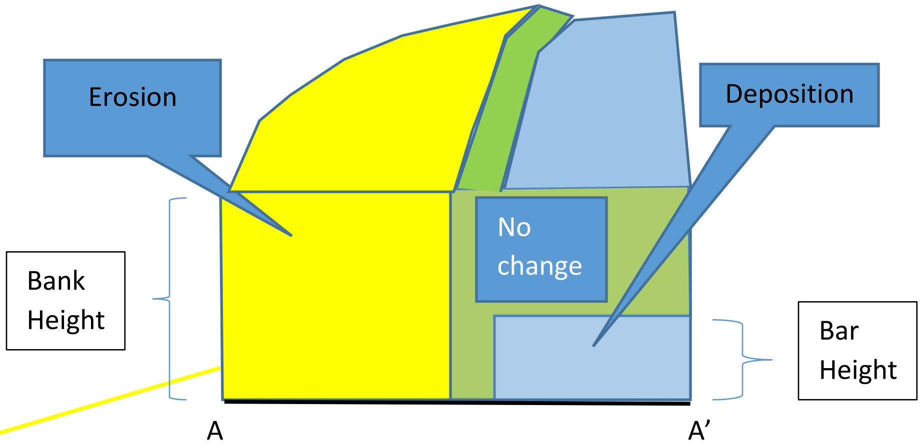


Figure 20. Process by which streambank erosion volume was estimated.

To convert soil volume lost to a soil mass the mass of sediment per square meter was estimated to be 1500 kg and the soil phosphorus content of 350 mg of Phosphorus per Kg of soil. This Soil phosphorus level was adjusted slightly lower than the average soil phosphorus value of typical segments in the BSTEM model of 600 mg/km (Langendoen 2012) and 621 mg/kg from a more recent studies (Ishee 2015) so that the percent loading would be reduced to match the percent loading for streams in the Missisquoi watershed that most closely matched the Black, Barton, Clyde and Johns Rivers and to also account for the fact that some of this phosphorus is also lost to floodplain deposition which was not calculated in this modeling effort and so not contributing to loading to the lake. A recent study of the contribution of eroding streambanks has also suggested that the low degree of phosphorus saturation found in streams in the Champlain valley indicates that loading from streambank erosion has less impact on the bioavailable phosphorus than the total phosphorus loading would indicate (Ishee 2015) and so using this lower value can also be justified based on this.

Table 17 estimated streambank load and percentage for the Johns, Clyde Black and Barton Rivers.

River basin	Estimated Streambank Load (kg/yr)	Streambank load Pct
Johns River	74	5%
Clyde River	114	2%
Black River	5440	25%
Barton River	5096	27%

6.5 Updating the model land use layer.

The Land use layer for the Lake Memphremagog model was created by combining land use provided by the MRC du Memphremagog landcover layer from 2008 for Quebec and the National Agricultural statistics service cropland datalayer layer produced by USDA for Vermont. The cropland datalayer is a modified version of the 2011 national land use dataset that has been updated to more accurately break down cropland vs hay or pasture land for agricultural lands. Based on a visual assessment of the land use over aerial photography this land use layer was quite accurate for distinguishing hay and pasture vs cropland which was often incorrect in the 2011 national land use dataset. This data layer was converted from a raster dataset to a vector shapefile and land uses were consolidated to those used in the modeling so all forest types, wetland types, cropland types were combined along with a consolidation of developed land use into one land use class. Through the Smi modeling process the land use classes from the Quebec land use layer were made analogous to those used in Vermont with the addition of the Golf land use and the two datasets were combined. The next step in the GIS analysis process was to break down croplands into A, B, C and D soil types and to overlay the land use classes with the 338 subwatersheds.

One area of significant inaccuracy in both the national land use dataset and NASS cropland layer was wide swaths of forested land that were incorrectly mapped along the edge of the interstate 91 as developed lands. Where these areas were 10's of acres in size they were manually converted to forested lands, wetlands or agricultural lands base on personal knowledge and analysis of aerial photos.

6.5.1 Roads

Based on recent studies by Beverly Wemple (2013) roads have been identified as a significant source of nutrients to surface waters in the Mad River and Winooski River watersheds which was then extrapolated to the Lake Champlain watershed in the calibration of the Lake Champlain SWAT model. This study of road runoff in the Winooski watershed estimates of loading rates were developed estimates at 610 kg/km^2. Through the update of the SWAT modeling which was applied in the Lake Champlain TMDL as 528 kg/km as the loading coefficient based on the total road loading over the area of contributing watershed. Another newly available dataset is a road erosion risk layer that combines soils, slope and proximity to surface waters to categorize dirt roads as high, moderate, low or no erosion risk. Roads were added to the land use layer by buffering paved roads by 18 feet and dirt roads by 15 feet. Since no road erosion risk layer is available in Quebec the area weighted average loading for dirt roads in Vermont was used for Quebec dirt roads. The majority of dirt roads in the lake Memphremagog watershed are rated as no or low risk, 38% and 52% respectively, and only 9% are rated moderate risk and 1% high risk.

Table 18 areas and export coefficients used for paved and gravel roads to equal 528 kg/km^2

Class	area km^2	Percent area	Percent gravel area	Export coefficient Kg/km^2	Estimated load	Percent gravel	Percent of road Load
Paved	6.50	40%		200	1300		21%
no risk	3.69	23%	38%	325	1199	31%	24%
Low risk	4.99	31%	52%	600	2994	52%	41%
Mod risk	0.86	5%	9%	900	774	15%	11%
High risk	0.12	1%	1%	1200	144	3%	2%
Total gravel	9.68	60%		528	5111		
Total road	25.84	100%		248	6411.25		

6.5.2 Developed lands

To use the BMP treatment efficiencies for developed pervious and impervious land use categories as was done in Lake Champlain scenario tool, the amount of pervious vs impervious areas needed to be differentiated in the lake Memphremagog model along with individual export efficiencies. One approach for approximating the contribution from pervious vs impervious developed lands was to make a number of assumptions to extrapolate from the Lake Champlain SWAT model assuming the ratio of pervious to impervious surface is similar between the two basins as is the ratio of loading between pervious and impervious developed lands. The results of this analysis are that 31.4% of developed lands are impervious in the Lake Champlain watershed.

Another alternative to estimating impervious surface area in the lake Memphremagog watershed is to use the NLCD 2011 impervious surface percentage layer (Jin 2013.) This GIS layer includes developed lands and includes an attribute as to the percentage of impervious surface associated with the developed lands in a 30 meter raster GIS layer. To calculate the percentage of impervious surface this Raster layer was converted to a Vector and the area was multiplied by the percentage impervious to calculate the total area of impervious surface in meters which was summed for each subbasin. There were a few basins where the new total impervious surface areas were greater than the total area of developed lands. This appeared to be the case in a number of small subwatersheds with little developed lands and so an assumption was made that the maximum percentage of impervious surface for any subbasin was 50% of the developed land area. A calculation was then made that the pervious developed lands = the total developed lands – the impervious developed lands. These calculations result in a slightly smaller percentage of developed lands that are impervious of 28.2% or about 3% less than in Lake Champlain.

In the Lake Champlain SWAT Model impervious developed surfaces had on average 3.5 times the loading than pervious developed lands. Assuming this ratio is similar for the Lake Memphremagog watershed this would equate to an export of 63 kg/km for developed pervious surfaces and 221 kg/km for developed impervious surfaces to maintain the 108.33 kg/km as was originally calibrated in this model.

6.5.3 Farmstead areas.

Farmstead areas were also added to the land use shapefile. Farmstead areas are defined as the barn, manure pits and heavy use areas, equipment storage areas and housing directly adjacent to the farm. This was done through intersecting a shapefile of farmstead areas for farmstead areas which were known to have animals provided by Reed Sims of NRCS. An analysis of the farmstead layer over aerial photography suggested a good match with active farmstead areas, although these mapped farmsteads included both fully operational farms as well as a few farms where there was a large farmstead area and associated farmstead infrastructure but just a few animals. Farmstead areas in Quebec were generated based the 2008 MRC shapefile where this was included as a separate land use category, but then active farmstead areas were selected based on a point file provided by the Magog MRC of active farms which cut back on the farmstead by a factor of 3 or 66%. A similar analysis of aerial photography suggested that this reflected active farmstead areas well in Quebec – and captures analogous portions of active farms as the Vermont farmstead layer.

Unfortunately, a literature review resulted in no phosphorus export coefficients for comparable land uses that appeared reasonable for application to this model. Values ranged from a low for the Lake Champlain SWAT model at 371 kg/km^2 and in the literature at between 25,000 and 26,000 kg/km^2 for feedlots in the Midwest. A Lake Champlain Basin Program report on the Identification of Critical Source Areas of Phosphorus within the Vermont Sector of the Missisquoi Bay Basin included a literature review of loading from farmstead runoff (pages 110-112) that described a number of studies that showed a number of farmsteads in New York, Vermont and Wisconsin each contributed high levels of loading per unit area, or identified extremely high concentrations in runoff from farmstead areas consistent with a high annual loading per unit area. This was not included in the original modeling but in a discussion of critical source areas. One study published by Schellinger and Clausen (1992) referenced in Vermont estimated an extraordinarily high loading level of 174,000 kg/km^2 for one farmstead drainage.

For the initial modeling an intermediate export coefficient for farmstead areas was chosen at 1100 kg/km^2 which is more than three times the loading used in the Lake Champlain TMDL modeling on one hand but one or even two orders of magnitude lower than direct studies of runoff from farmstead areas showed. This significantly lower loading than measured can be justified by the fact that many of the farmsteads mapped in the Lake

Memphremagog watershed mapped as having animals may just support a few animals, while those that have large number of animals tend to have significant areas of farmsteads that drain to manure pits and so would not generate direct runoff. The Loading for farmstead area is then adjusted by the watershed model calibration and so that provides a level of justification for the final export coefficient used for farmstead areas.

The final land use shapefile was then run as a union with the subbasin shapefile and then areas of each land use was calculated for each sub basin area into a final shapefile titled 1020515FINAL.

6.5.4 Hay and Pasture lands

Hay and pasture lands were combined in the Lake Memphremagog phosphorus export model as these were combined in the national landcover dataset that was used in the original modeling. At a meeting with agricultural partners in the Lake Memphremagog watershed the importance in the distinction between these two land uses was brought up as different BMP's and BMP efficiencies would be applied to these land uses. After this meeting an analysis of loading from hay vs pasture was done with the following conclusions:

1) Based on the 2012 NASS croplands data layer the combined hay/pasture layer was made up from about 77% hay and 23% pasture/grass. However the accuracy of this breakdown is not great. Some areas where it is apparent that there is a lot of pasture have most of the pasture areas mapped as hay (see map below of a grass based farm which is largely pasture based) shows hay in yellow and pasture in red as mapped but much of the yellow is pastured.
2) The crop data layer Question and Answer section acknowledges lack of accuracy around the mapping of pasture lands and based on this and observations of the pasture areas over aerial photo's there appears to be a fair amount of land mapped as pasture and hay that are large lawns which is balanced by other areas of these land uses mapped as developed which are actually hay or pasture. The much higher inaccuracy in the mapping of pasture lands causes some concern in trying to separate out and calibrate the model for the pasture land use.
3) The Lake Champlain SWAT model estimates a much higher pasture loading rate than hayland per sq km (nearly 300 kg for pasture vs about 100 kg for hay), however in the literature many studies either combined the two land uses as one land use category as the earlier Lake Memphremagog model or estimates the loading from the two land uses as similar (Reckhaw, 1980)
4) Given the similarity in loading from these two land uses in the literature, the higher uncertainty in the mapping of pasture in the land use layer the approach to estimating loading between these two land uses is to keep the model as is, and estimate that the loading from these two land uses is proportional to the percentages of these land uses across the basin as a whole, so that 23.38% of the loading estimated from the hay/pasture land loading would be estimated to have come from pasture lands and the remaining 77.62% from Haylands, with an understanding that there is a high degree of uncertainty around the relative contribution of phosphorus loading from pasture and hay land uses.
5) For Quebec watersheds a survey by MAPAQ for the MRC region found that of Hay and Pasture lands 60% is hay and 40% is pasture. The higher percentage of Pasture lands can be explained by the fact that in the Quebec portions of the watershed there are fewer dairy operations and more beef cattle farms which tend to have more pasture land vs hayland than dairy farms. Therefore, in the model in Quebec 40% of the hay/pasture was designated as pasture and the remaining 60% was left as Hay.

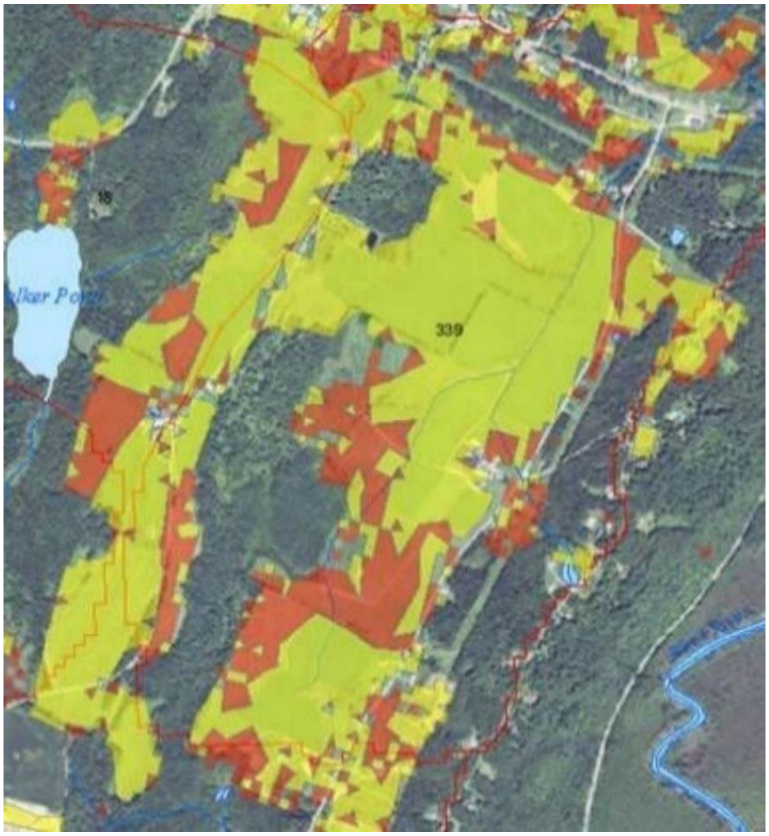


Figure 21. Map of hay an pasture land as mapped by the NASS Croplands Dataset showing large inaccuracies in the mapping of pasture lands in particular.

6.6 Watershed model calibration

The next step in the modeling process is the calibration of the export coefficients so that these match the measured loadings of the major tributaries. Original export coefficients were chosen by SMi with input from VT DEC and Quebec partners however additional land uses of paved and dirt roads, and farmsteads have been added and so export coefficients need to be estimated for these land uses.

A number of options were considered for calibration of the watershed export model. One challenge with calibration is the fact that there are just 4 major watersheds with well-defined loading estimates to calibrate the watershed export model, none of which have very different land use percentages with regards to developed lands, roads and forestlands and all of which are in Vermont. Also, the Clyde River watershed is dominated by large lakes and so retention is a primary driver in loading in this watershed and the Johns River is a much smaller watershed in comparison to the other 3 watersheds and sampling methodology is slightly different so that may cause some of the difference in loading seen with this watershed.

Table 20 area, aerial loading and land use percentages for the four major tributaries in the lake Memphremagog watershed.

	Area (km)	P load Kg/km^2	water	developed	road	Hay/ pasture	crop	farmstead	forest	wetland
Johns River	24	52.3	0.2%	4.7%	1.8%	23.0%	6.4%	0.9%	60.7%	2.4%
Clyde River	375	17.1	5.5%	4.1%	1.2%	10.8%	1.1%	0.3%	74.2%	2.9%
Black River	350	64.6	1.3%	3.2%	1.2%	17.1%	3.6%	0.4%	70.8%	2.5%
Barton River	429	43.9	3.9%	4.4%	1.3%	15.8%	1.3%	0.3%	71.2%	1.8%

One option for calibrating loading is to simply adjust the estimated loading for each of the export coefficients by the same amount to match to total loading of the four major tributaries. This would result in adding 11.69% to each export coefficient. When this is done the total RSME of the difference between modeled and measured phosphorus loading is reduced substantially. However, the Black, Barton and Clyde rivers all still have an average loading difference that a substantial between the measured and modeled loading.

6.6.1 Calibration using minor Memphremagog Tributaries

To add another approach for calibrating the land use export coefficients phosphorus loadings were estimated for another 24 sub watersheds in Vermont based on water quality sampling done through the LaRosa Water Quality monitoring program. These smaller tributaries have a much wider range in land use where the minimum and maximum percent land use varied between 1 to 81% developed lands, 0 to 12% roads, 0 to 52% hay or pasture, 0 to 34% crop lands, and finally 0 to 4% farmstead area. Strong relationships between agricultural land use percentages and aerial phosphorus loading in Figure 22 show a strong positive correlation between farmstead, crop and pasture land land use percentage and increased phosphorus loading. A similar analysis of

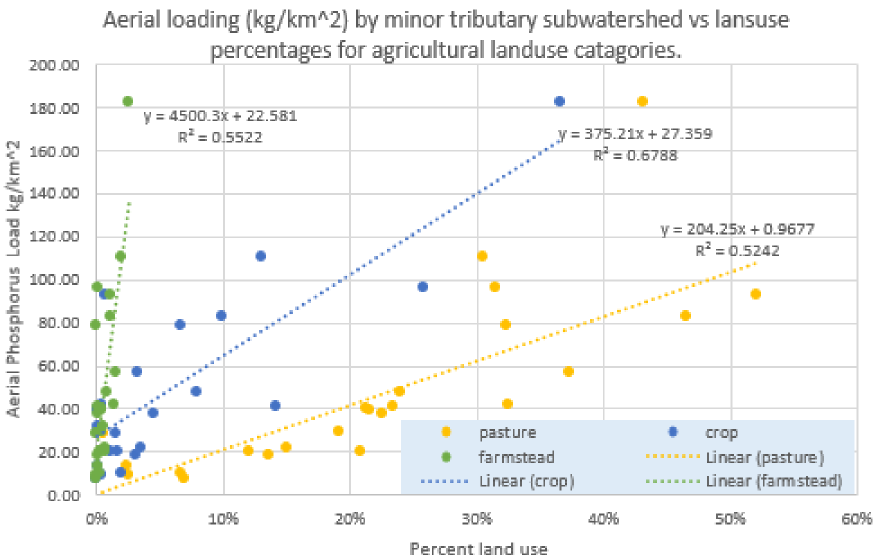


Figure 22. Relationship between agricultural land uses and aerial phosphorus loading for 24 minor tributaries in the lake Memphremagog watershed.

developed and road land use percentages with aerial phosphorus loading do not show as strong relationship. Even when the East site tributary is removed which has far higher percentages of developed land use than any other watershed the regression only improves from near 0 to 0.1 for roads and .07 for developed lands. Together this suggests that agricultural land uses are a more significant driver of phosphorus loading to these 24 tributaries than developed lands and roads. Unfortunately attempts to estimate loading from minor tributaries in Quebec had a higher degree of uncertainty so calibration of loading from the Quebec portions of the watershed was not done.

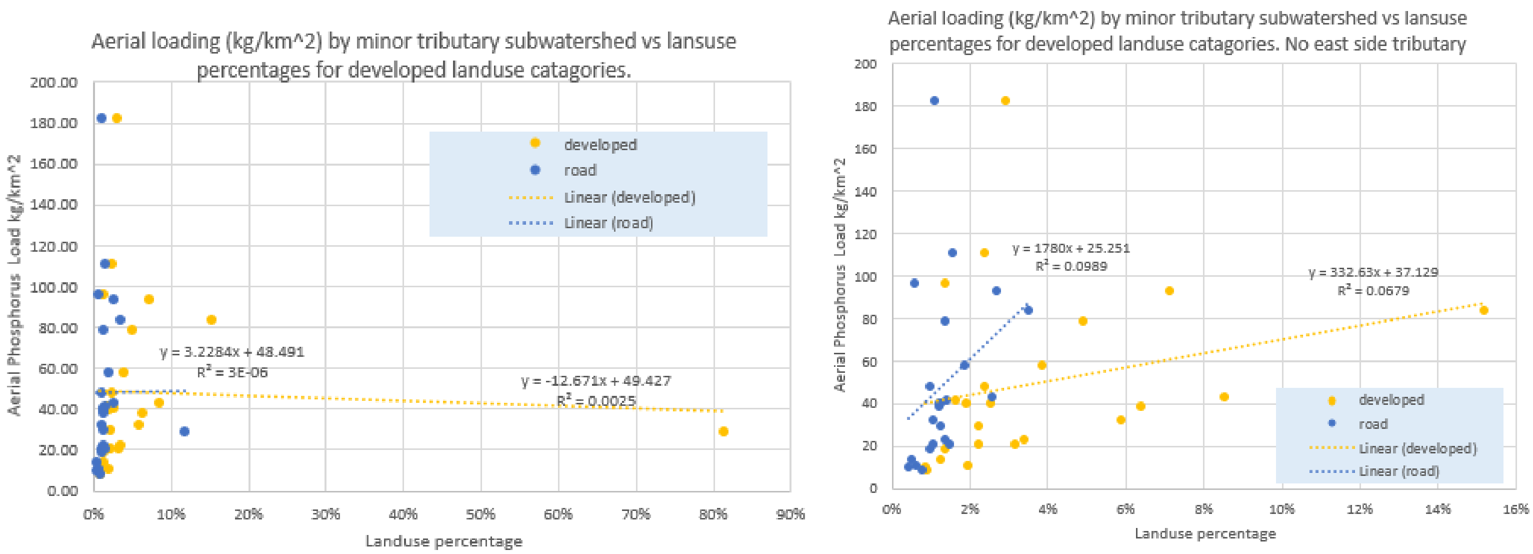


Figure 23. Relationship between road and developed land uses and aerial phosphorus loading for 24 minor tributaries in the lake Memphremagog watershed both including and excluding the largely developed east side tributary in the City of Newport.

One challenge in interpreting this data or doing multiple linear regression analysis is that there are close correlations between many of the land uses. The associations between land uses include increased farmstead and developed land use with increasing pasture land use as shown in Figure 24 which can make it hard to distinguish which of these land uses may be responsible for increased phosphorus loading in a multiple linear regression analysis and can be responsible for low levels of significance though such an analysis. There is also a concern with trying to over calibrate the model. Since the land use based export model is quite simple it doesn't include many factors which may influence phosphorus such slope, proximity to surface waters, practices which may have been installed to reduce loading. While the calibration of the model can be done to the maximum

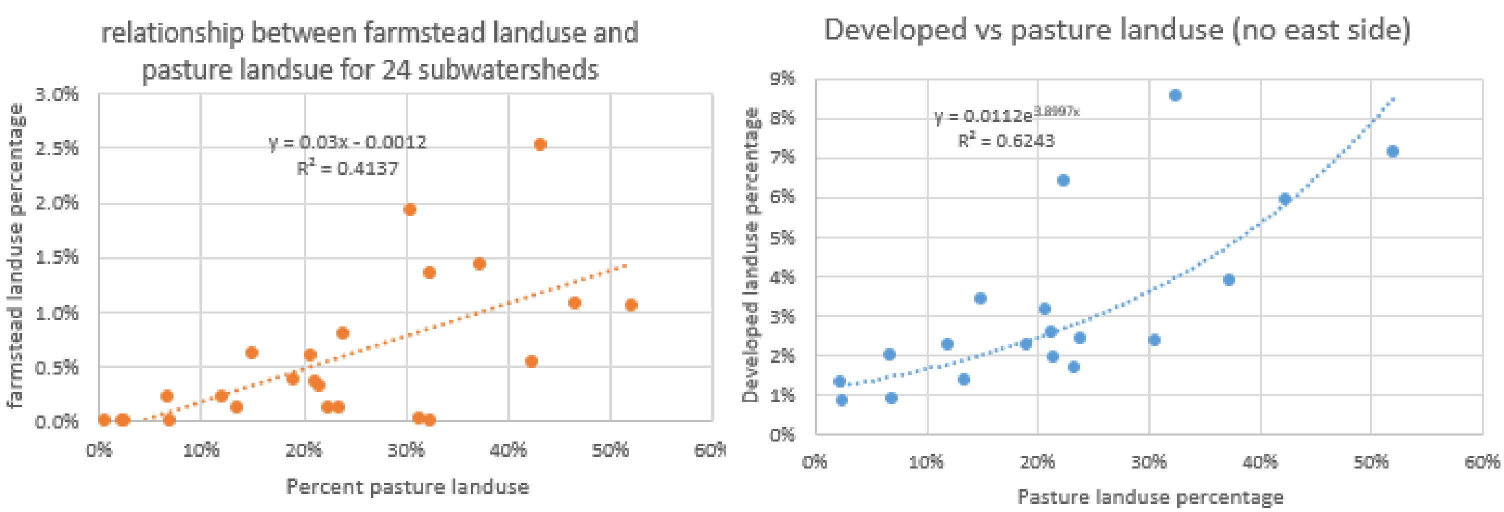


Figure 24 relationships between farmstead and developed land use percentages and pasture land use percentages for 24 minor tributaries to Lake Memphremagog. The relationship between developed and pasture lands does not hold for the urban East side tributary but this is one of only a very small number of such urban watersheds in the basin.

extent to minimize the RSME between measured and modeled loading, the degree to which this calibration reflects the reality breaks down at some point therefore some constraints on the calibrations need to be put in

place and an evaluation through alternative calibrations process considered to evaluate which of these calibrations are most likely to improve how well the model reflects true loading across the watershed.
In addition to considering multiple linear regression for subsets of land uses excel solver was used to calibrate loadings through a number of alternative approaches. Solver was used to constrain the export coefficients used in the model to either ranges which have been found in the literature or set percentages above or below the original export coefficient used. Scenarios shown in Table 21 include limitations on the original export coefficients ranging from 25% to 400% of original to 75% to 133% . Another way that excel solver was used was to minimize the difference between the original export coefficient and the export calibrated export coefficient. This was done by adding up the absolute value of the difference and adding this to the RSME for a cell that the solver program. The sensitivity of this was evaluated by running scenarios where this was multiplied by 1, 3, 5, and 10 to evaluate the impact on the calibration. Finally, solver can be used to constrain adjustments in the export coefficients so that the total loading as measured by the 4 major tributaries matches the modeled loading. There are two major land use categories that were broken down into subsets of land use which were croplands on different soil types, and roads with differing erosion potential. With the exception of the croplands on A soils (which seemed significantly different that other croplands across a number of models when all croplands were included separately) these export coefficients were adjusted together.

Therefore the cell which solver is set to minimize includes the following inputs:

1) RSME between measured and modeled aerial loading for both minor and major tributaries
2) Difference between measured and modeled loading for the 4 major tributaries combined.
3) Sum of the Absolute value of the percent difference between the original and calibrated export coefficient multiplied by a factor of between 1 – 10.

Table 21. Export coefficients (kg/km^2/yr) for alternative calibration scenarios for the lake Memphremagog land use based phosphorus export model. Excel solver was set to minimize RSME of the difference between modeled and measured watershed loading for 24 minor and 4 major tributaries in the Lake Memphremagog watershed. Limitations on the range of export coefficients were also applied and the sum of absolute value of the difference between the original export coefficients multiplied by 1-10 to minimize random adjustments in export coefficients that didn't provide a substantial benefit to model fit. Land uses highlighted in yellow had consistent increases in export coefficients with the exception of residential where loading coefficient was reduced for all scenarios. Also shown are the original export coefficients as well as export coefficients adjusted to match total loading along with RSME and R^2 for tributaries.

coefficient limitation	25% - 400%								50%-200%		66%-150%		75%-133%		Average		Original export coeff.	Original export * 9.58%
export adj multiplier	1		3		5		10		5		5		5					
Calibration scenario	1		2		3		4		5		6		7				8	9
Land use	exp	% dif	exp	% dif	exp	% dif	exp	% dif	exp	% dif	exp	% dif	exp	% dif	exp	% dif	exp	exp
Water	9	-3%	10	13%	9	0%	9	0%	10	12%	10	8%	9	4%	9	5%	9	9.9
Residential	49	-75%	49	-75%	49	-75%	95	-51%	98	-50%	130	-33%	146	-25%	88	-55%	195	213.7
Barren land	296	-3%	342	12%	307	1%	305	0%	342	12%	329	8%	315	3%	319	5%	305	334.2
Forest	9	21%	9	23%	8	17%	7	0%	8	15%	8	13%	8	9%	8	14%	7	7.7
Road Paved	131	-13%	138	-8%	154	3%	150	0%	159	6%	156	4%	151	0%	148	-1%	150	164.4
Dirt road	442	36%	364	12%	365	12%	325	0%	366	13%	361	11%	390	20%	373	15%	325	356.1
Dirt Road Low Erosion	817	36%	672	12%	673	12%	600	0%	676	13%	666	11%	720	20%	689	15%	600	657.5
Dirt Road mod Erosion	1225	36%	1008	12%	1010	12%	900	0%	1014	13%	999	11%	1080	20%	1034	15%	900	986.2
Dirt Road high Erosion	1634	36%	1344	12%	1347	12%	1200	0%	1351	13%	1332	11%	1441	20%	1378	15%	1200	1315.0
shrub/herb	20	-1%	18	-10%	21	5%	20	0%	23	15%	22	10%	21	5%	21	3%	20	21.9
Farmstead	1403	17%	1468	22%	1329	11%	1200	0%	1379	15%	1354	13%	1297	8%	1347	12%	1200	1315.0
Pasture/Hay	80	43%	91	62%	94	68%	102	82%	83	49%	78	40%	71	27%	86	53%	56	61.4
Wetland	52	3%	48	-4%	50	1%	50	0%	46	-7%	48	-5%	48	-4%	49	-2%	50	54.8
Cultivated A soils	107	-1%	98	-10%	112	4%	108	0%	111	3%	110	2%	97	-10%	106	-2%	108	118.3
Cultivated B soils	415	50%	357	29%	354	28%	277	0%	340	23%	329	19%	369	33%	349	26%	277	303.5
Cultivated C soils	319	50%	274	29%	272	28%	213	0%	261	23%	253	19%	284	33%	268	26%	213	233.4
Cultivated D soils	476	50%	409	29%	406	28%	318	0%	390	23%	378	19%	424	33%	400	26%	318	348.5
Cultivated unknown soils	319	50%	274	29%	272	28%	213	0%	261	23%	253	19%	284	33%	268	26%	213	233.4
RSME tributaries	**18.06**		**17.69**		**17.78**		**21.51**		**21.50**		**24.93**		**26.77**				**32.75**	**35.56**
R^2 minor tributaries	**0.905**		**0.902**		**0.899**		**0.787**		**0.803**		**0.703**		**0.673**				**0.407**	**0.407**

Limitations were then placed on the export coefficient adjustments from the original export coefficients used for the model with a range of 25-400% 50-200% 66-150% and 75-133%.

Looking at the different scenarios and using the original export provides a range of calibration options. Maximizing the limitations on export adjustments from the original values and reducing the multiplication factor of the total difference between the original and calibrated export coefficients reduces the RSME but results in export coefficients that are further from typical values, and more prone to calibration based on random differences in land use in watersheds that might not be drivers of phosphorus loading. On the other end of the spectrum using the original export coefficients or adjusted to match the total loading at the four major tributaries results in high RSME values. The challenge is to then select a scenario that seems most realistic which appear to be scenarios 5, 6 or 7.

After this analysis was run it was recommended that maximizing the R^2 value between modeled and measured loading for both the minor and major tributaries would be a better way to calibrate this model. When that and minimizing the RSME are compared there is little difference in the resulting export coefficients, although the factor for the average difference between new and original export coefficients had to be adjusted by a factor of 50 due the smaller values for R^2 vs RSME. In addition to this, after the analysis in Table 21 was completed a number of small adjustments to the model were made including changes in the retention equation for lakes (setting a minimum retention), adjustments to the septic loading estimates, and a reduction to the starting farmstead loading export coefficients which each in a small way adjust the calibration outcome. With these minor adjustments to the model calibration was only slightly changed from previous model runs so a rerunning the analysis of Table 21 was not done. The final range in export coefficients chosen the SMi export coefficient times 1.8 as a maximum or divided by 1.8 as a minimum.

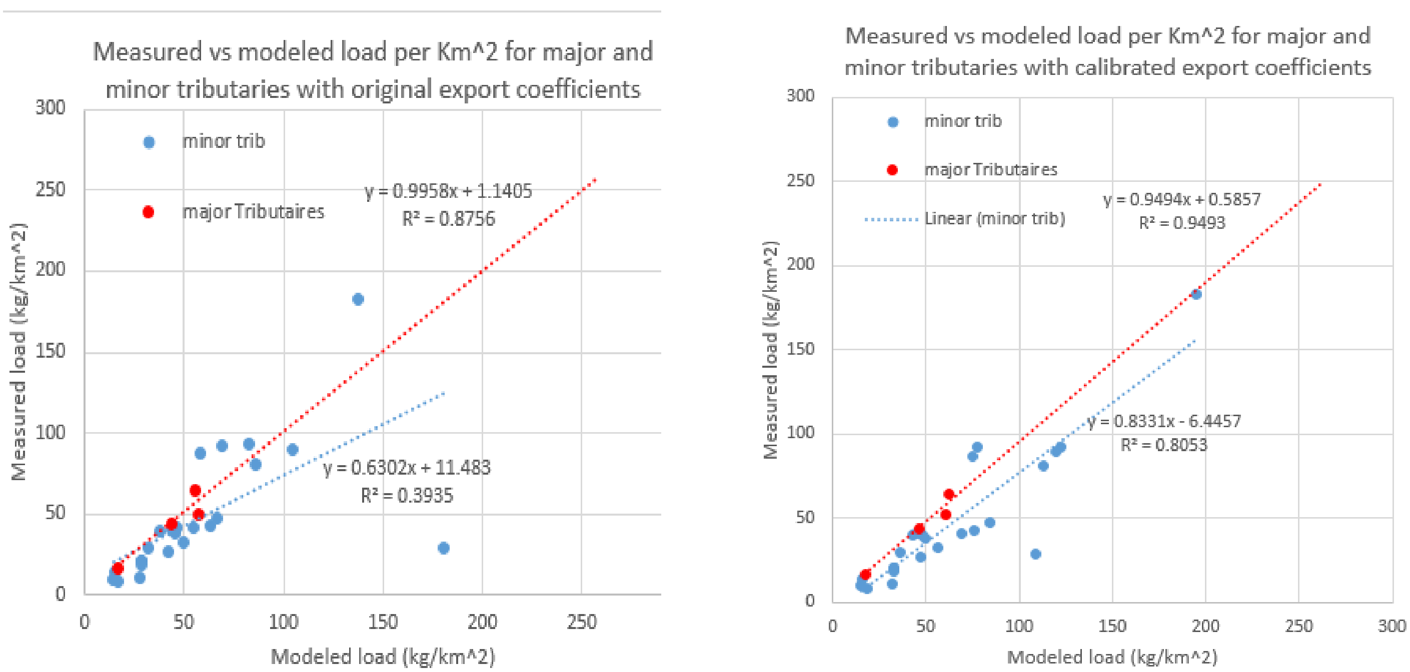


Figure 25 Measured vs modeled loading for the 4 major and 24 minor tributaries showing improved R^2 after calibration.

Table 22 average percent difference for the adjusted smi export coefficients vs calibrated export coefficients showing improved performance of calibrated model in matching measured loading.

Tributary	Measured Load	adjusted Smi		Calibrated	
		Load	% difference	Load	% difference
Black	22622	20787	-8.1%	21551	-4.73%
Barton	18858	19951	5.8%	19639	4.14%
Johns	1316	1595	21.2%	1537	16.81%
Clyde	6420	6822	6.3%	6489	1.07%
Total	49216	49154	0.0%	49216	0.0%

Table 23 Comparison of export coefficients from the Original Smi model, Smi model adjusted to match total loading at the 4 major tributaries, Champlain swat loading adjusted to match total loading at the 4 major tributaries as well as the calibrated export coefficients.

	Champlain swat -12.7%	Original SMI model	Adj Smi model (+11.69%)	Calibrated Model	
	kg/km^2/yr	kg/km^2/yr	kg/km^2/yr	kg/km^2/yr	% adj. from Smi
Water	Na	9	10.1	9.8	9%
Residential	104.7	195	218	108	-45%
Barren land	Na	305	341	250	-18%
Forest	14	7	7.8	7.7	11%
Road Paved	175	150.0	**168**	120	-20%
Dirt road	461	325.0	**363**	364	12%
Dirt Road Low Erosion	461	600.0	**670**	671	12%
Dirt Road mod Erosion	461	900.0	**1005**	1007	12%
Dirt Road high Erosion	461	1200.0	**1340**	1342	12%
shrub/herb	18	20	22	18	-10%
Farmstead	324	1000	**1117**	1150	5%
Pasture/Hay	112	56	63	80	43%
Wetland	16	50	**56**	46	-8%
Cultivated A soils	52	108	121	87	-19%
Cultivated B soils	166	277	309	440	59%
Cultivated C soils	135	213	238	338	59%
Cultivated D soils	268	318	355	505	59%
Cultivated unknown soil	155	213	238	338	59%

6.6.2 Calibration results

With watershed loading calibration complete a number of analysis can be completed on the estimated loading across different landsues. The excel model allows for an estimate of watershed loading per land use per subwatershed, major stream basin, lake segment, or between Vermont and Quebec. Because this TMDL is for the Vermont lake segment the following data are presented just from the Vermont portions of the watershed.

Table 24 land use areas, modeled phosphorus export from the landscape and to Lake Memphremagog including retention in South Bay across major land uses.

	Area		Calibrated Loading		Loading to Lake	
	km^2	Percentage	Kg	Percentage	Kg	Percentage
Developed Total	**68.5**	**5.4%**	**15124**	**24.1%**	**6842**	**24.5%**
Developed	51.7	4.1%	5587	8.9%	2681	9.6%
Road Paved	6.5	0.5%	780	1.2%	386	1.4%
Dirt road	9.7	0.8%	5731	9.1%	2297	8.2%
Barren land	0.5	0.0%	135	0.2%	61	0.2%
WWTF			834	1.3%	561	2.0%
septic			2056	3.3%	856	3.1%
Agricultural total	**216.4**	**17.1%**	**27561**	**44.0%**	**12812**	**46.0%**
Cropland	24.1	1.9%	7963.3	12.7%	3686	13.2%
hay	144.3	11.4%	11544	18.4%	5379	19.3%
pasture	44.0	3.5%	3522	5.6%	1641	5.9%
Farmstead	3.9	0.3%	4531	7.2%	2106	7.6%
Natural total	**981.6**	**77.5%**	**9170**	**14.6%**	**3635**	**13.0%**
Wetland	31.1	2.5%	1430	2.3%	606	2.2%
Forest	854.9	67.5%	6582	10.5%	2482	8.9%
shrub	27.0	2.1%	485	0.8%	202	0.7%
Water	68.7	5.4%	673	1.1%	346	1.2%
Streambank total			**10804**	**17.2%**	**4585**	**16.3%**
TOTAL	**1266.4**		**62658**		**27874**	

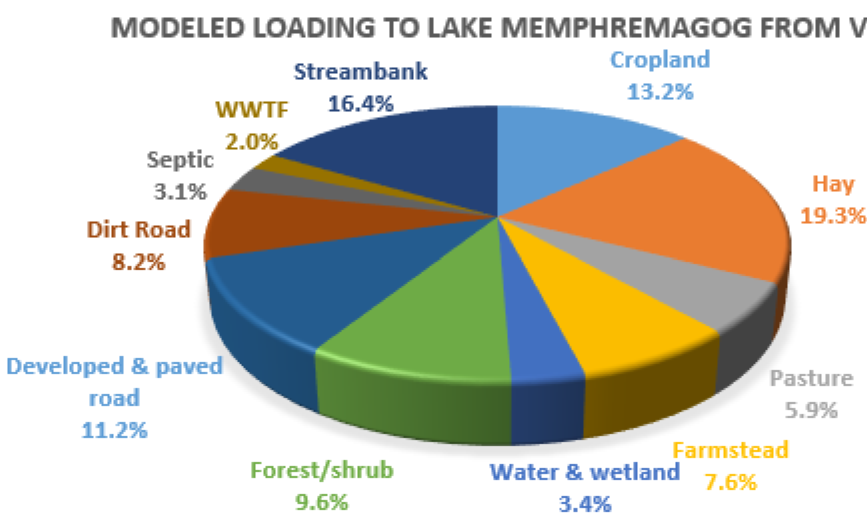


Figure 26 Pie Chart of estimated loading across different land uses to Lake Memphremagog from the Vermont portions of the basin.

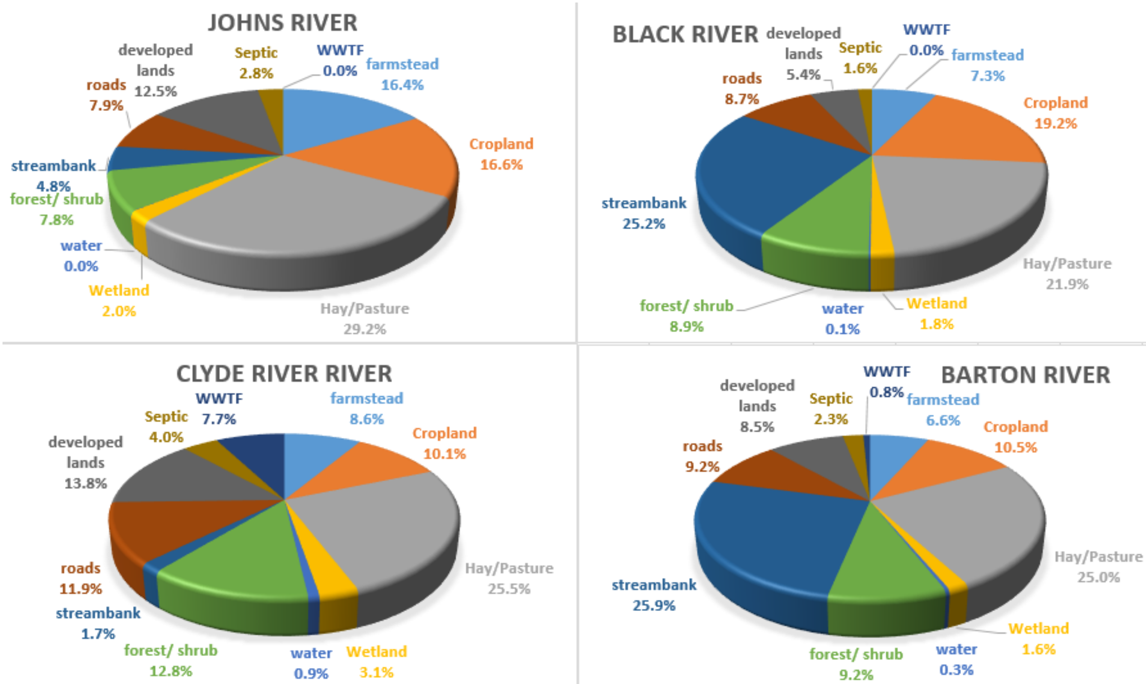


Figure 27 pie charts of loading by land use for 4 major watersheds in Vermont.

To account for the differences in loading between to measured loading and loading estimated through the phosphorus export model for the Black, Barton, Clyde and Johns Rivers, phosphorus export coefficients, streambank erosion, and septic system loading was adjusted by the percentage difference between the modeled and measured loading for each of these tributaries. WWTF loading was not adjusted because this loading was directly measured. When this is done the export coefficients for each of the sub watersheds is adjusted as shown in Table 25 below. There are many reasons each of these watersheds may have higher or lower export than predicted in the model including different soils, slope characteristics, specific management practices in place, among many other things. Making this explicit adjustment allows the inputs from the Bath Tub model based on measured loading on the four major tributaries to be identical to those from the watershed export model. This adjustment simplifies the use of a Scenario tool to adjust loading using the phosphorus export model as a starting point from which loading is adjusted through the implementation of BMP's.

Table 25 adjusted export coefficients for the four major basins in Kg/km^2

	Original	Black	Barton	Johns	Clyde
Adjustment factor	100%	104.97%	95.99%	85.61%	98.86%
Water	9.8	10.3	9.4	8.4	9.7
Developed	108	113.4	103.7	92.5	106.8
Developed Pervious	63	66.1	60.5	53.9	62.3
Developed Impervious	221	232.0	212.1	189.2	218.5
Barren land	250	262.4	240.0	214.0	247.1
Forest	7.7	8.1	7.4	6.6	7.6
Road Paved	120	126.0	115.2	102.7	118.6
Dirt road	364	382.1	349.4	311.6	359.8
Dirt Road Low Erosion	671	704.4	644.1	574.4	663.3

Dirt Road mod Erosion	1007	1057.1	966.6	862.1	995.5
Dirt Road high Erosion	1342	1408.7	1288.2	1148.9	1326.6
shrub/herb	18	18.9	17.3	15.4	17.8
Farmstead	1150	1207.2	1103.9	984.5	1136.8
Pasture/Hay	80	84.0	76.8	68.5	79.1
Hay	80	84.0	76.8	68.5	79.1
Pasture	80	84.0	76.8	68.5	79.1
Wetland	46	48.3	44.2	39.4	45.5
Cultivated A soils	87	91.3	83.5	74.5	86.0
Cultivated B soils	440	461.9	422.4	376.7	435.0
Cultivated C soils	338	354.8	324.5	289.4	334.1
Cultivated D soils	505	530.1	484.8	432.3	499.2
Cultivated unknown soils	338	354.8	324.5	289.4	334.1

7 Lake Phosphorus data

Phosphorus samples were taken on Lake Memphremagog from 2005 through 2012 through the Vermont Lake Assessment program and have been taken in Quebec through cooperative arrangement between MCI and MDDEFP. All phosphorus samples taken by the Vermont Lake Assessment program were processed in the LaRosa laboratory in Vermont while samples collected in Quebec were processed by the MDDEFP laboratory in Quebec. Vermont samples meet quality assurance measures with no blank detections and an average RPD of 7.3% . A series of split samples were taken to establish a relationship between the MDDEFP and LaRosa laboratories as shown in Figure 28 with the following result in ug/l

LaRosa = MDDELCC * 1.0487 + 6.7301.

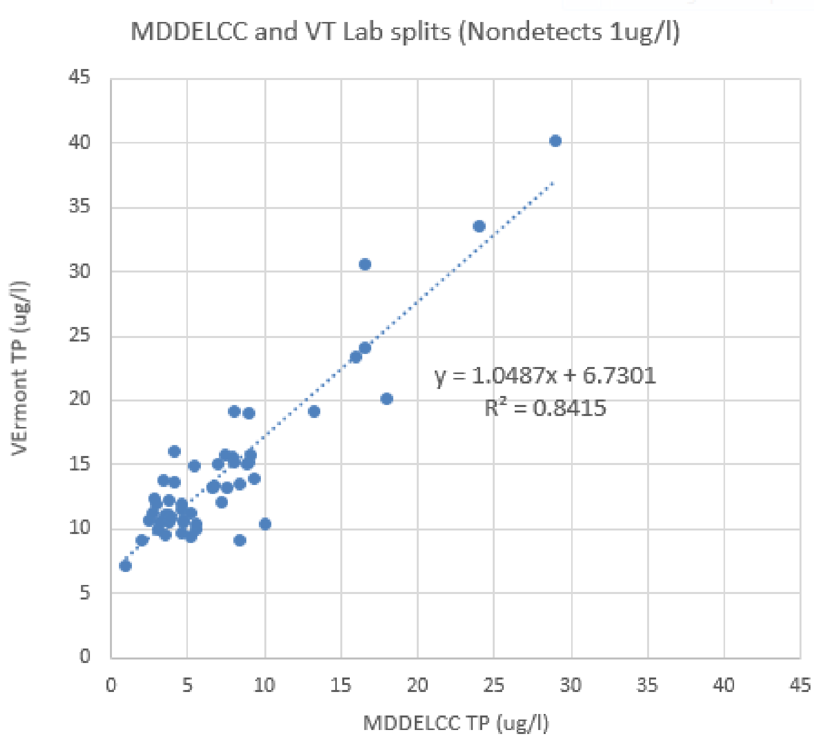


Figure 28 relationship between Vermont laboratory and Quebec laboratory total phosphorus values based on split samples taken in lake Memphremagog on tributaries.

Samples taken by VT DEC were taken biweekly from May through October or November from 2005 – 2012 at 0.2 meters depth and then every two meters depth to one meter above the bottom while MCI/MDDEFP samples were taken as an integrated sample to one meter depth.

An analysis of Vermont data from 2009- 2012 shows that for sites Memph 03 and Memph 04 the average of phosphorus values across all depths to 6 meters was slightly higher than at the surface (0.1 and 0.51 ug/l respectively) or an average of 0.305 ug/l (see figure below). The long term monitoring for compliance with the Water quality standard will be done through the Vermont Lay Monitoring program at site Memph 03 which has a protocol of collecting a depth integrated sample at two times the Secchi depth which means sampling depth typically ranges between 6-8 meters depth. However, since the data from the Quebec portions of the watershed were only collected from 0-1 meter depths these values need to be increased by this amount to be equivalent to the Vermont data.

Table 26 Average values at sample sites across depths 0.2 m and 2 meter, and 0.2, 2 4, and 6 meters depths from 2009 through 2012.

2009-2012	Average 0.2 and 2 meters	Average 0.2, 2, 4 and 6 meters	Difference
Memph 03	17.73	**17.83**	-0.100
Memph 04	17.12	**17.63**	-0.511
So Bay	24.26	24.16*	0.108
Ave 03/04	17.42	17.73	-0.305

* only taken at 0.2,2, 4 meter depths.

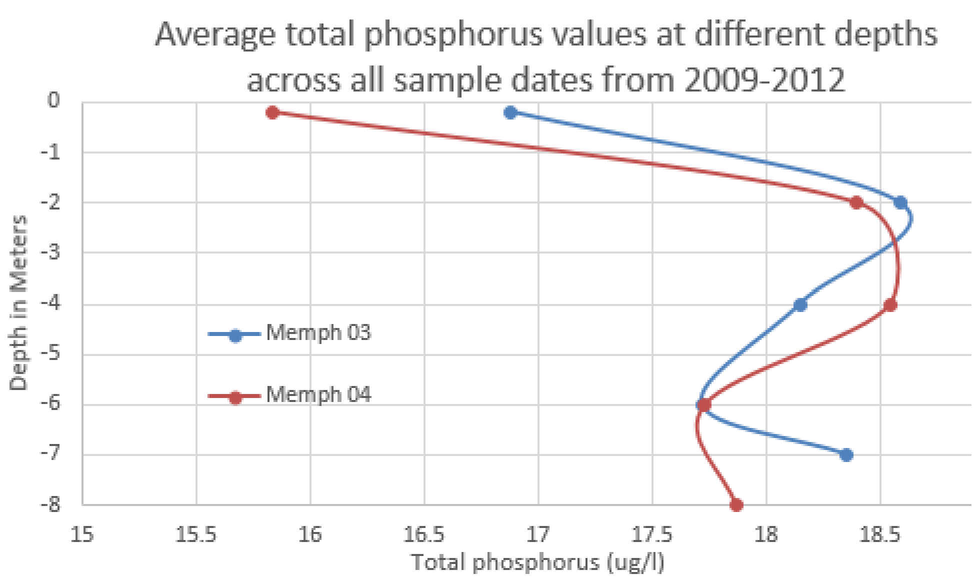


Figure 29 average total phosphorus values at different depths across all sample dates from 2009 to 2012 at two samples sites on Lake Memphremagog (Memph 03 and Memph 04)

Quebec sampling was also broken down into two sets of sites one of which was sampled for a longer season roughly May through November and another which was sampled from June through August or September. An analysis of Vermont and Quebec data suggests that there is no consistent seasonal variation in phosphorus values and so unlike chloride the different seasons over which samples were taken doesn't seem to cause a significant problem comparing mean phosphorus values between sites with different sampling seasons.

One challenge with integrating the data from Quebec and Vermont is that the Quebec data shows a strong decreasing trend over time (data was collected beginning in 1998) that is not seen in Vermont data, even at the one site where both Quebec and Vermont both sampled

(Memph 04 or 249) since 2005. There is no explanation for this difference and this has posed a significant challenge in modeling phosphorus over the initially planned calibration and validation time periods 2005-2008 and 2009-2012 since these have significantly different phosphorus levels for the Quebec Data but consistent phosphorus levels in Vermont based on the Vermont data. While we don't have sufficient data from Quebec tributaries to evaluate changes in phosphorus over time it seems unlikely that there were sufficient loading reductions in the Quebec portions of the watershed to explain the roughly 25% reduction in phosphorus values in Quebec between 2005-2008 timeframe and the 2009 to 2012 timeframe while levels in Vermont stayed consistent. For this modeling effort the data collected prior to 2009 were not included in this analysis. Final Quebec lake concentrations used in the modeling were derived from data in Appendix D with the following results shown in Table 27.

Table 27. Average and adjusted average 2009-2012 phosphorus concentration for Quebec lake segments

Segment name	Vt Lake	Qc Lake	Fitch Bay	Outer Fitch	Center Lake	North Lake	Magog bay
Sample site #	03020249	03020094	03020093	03020092	03020091	03020246	03020090
Average	16.27	15.79	22.63	14.22	13.79	14.01	14.48
Average + .305	16.57	16.10	22.93	14.52	14.10	14.31	14.79

7.1 South Bay phosphorus concentrations

Another challenge with the South Bay segment is that it is not steady state and there is a relationship between flow on the Black River and concentrations suggesting that the mean flow weighted concentrations are higher than the annual average as measured every two weeks. This relationship was not very apparent using the lake sampling data because sampling didn't typically take place during spring runoff or immediately after major runoff events. In fact, lake sampling would be canceled when a major runoff was predicted and boating conditions would be poor. An analysis of the percentile of daily flows on the Black River compared to the percentile of flows when samples were taken shows this distinction.

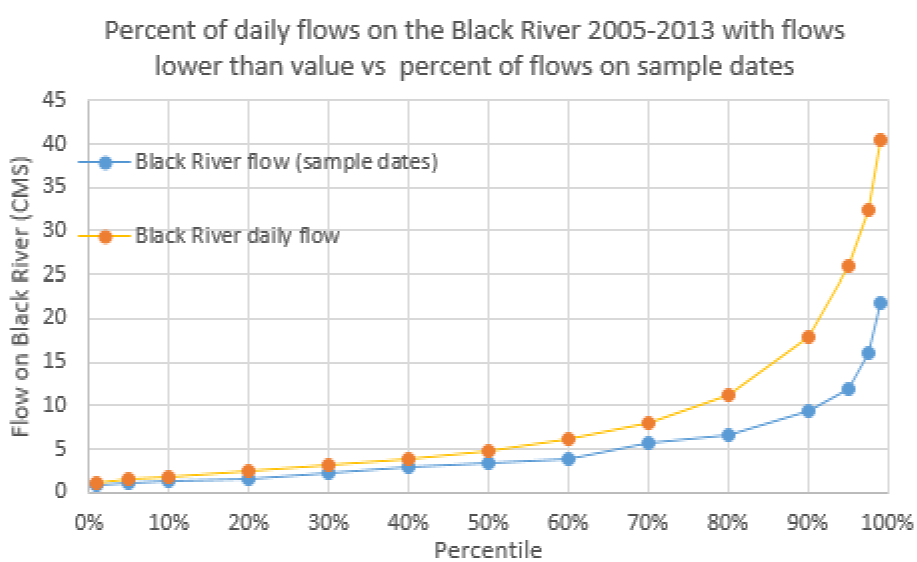


Figure 30. Significant difference between the daily flows on the black River as opposed to those dates sampled.

To understand what phosphorus values in South Bay during major runoff events samples were taken in the 2015 season using the DH 59 bomb sampler at the bridge at the outlet of South Bay along with tributary event samples to determine if phosphorus levels were elevated after major runoff events. A total of 13 samples were taken between April and September at flows ranging up to 1196 cfs. While these flow levels are higher than for lake sampling this highest flow measured only equates to the 97.5% flow for the Black River (flows on average higher than this level 9 days per year) so still does not capture the highest end of typical annual flows. Adding these data to the comparison of south bay concentration vs black river flow shows a pretty strong correlation with flow particularly above flows of about 20 CMS on the Black River. When treating the outlet to South Bay as a river and using the correlation between phosphorus and flow results in a flow weighted average concentration of 30.8 ug/l or 26% above the South Bay average concentration of 24.4 ug/l as measured from the lake samples alone.

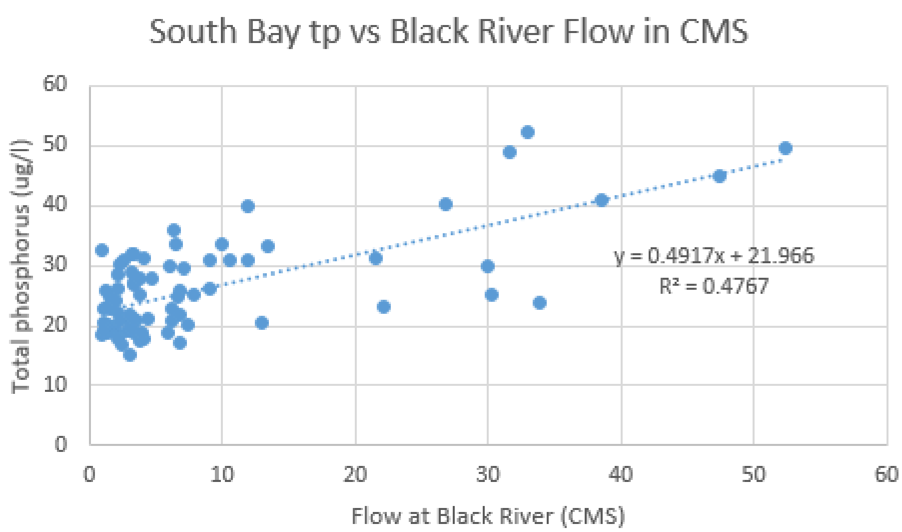


Figure 31 Relationship between Phosphorus values in South Bay and flow on the Black River

Even with using the South Bay concentration the bathtub model indicates a large amount of sedimentation in South Bay. While the volume and area of South Bay are relatively small compared to the flows that come in through the Black and Barton rivers there are major wetlands along the Barton river (and smaller ones along the Black River) downstream of sample points and these wetlands likely trap significant amounts of phosphorus during major runoff events before it makes it to the bay. In addition to this, an evaluation of historical aerial photos shows that the deltas at the ends of the Black and Barton rivers have been expanding rapidly suggesting significant sediment deposition as these rivers enter South Bay.

7.2 Precipitation and Evaporation (5.5% watershed)

Phosphorus loading from direct precipitation was estimated based on an annual loading of 10 kg/km^2 which was calibrated through the watershed export model. This is slightly lower than the estimated loading at 19 ug/l in rainwater which was measured in the Lake Champlain basin but this total loading difference is small relative to watershed loading. A factor for evaporation (based on monthly evaporation rates and segment areas multiplied by .79 to relate to lake evaporation) was factored in for ice free months to remove water but not phosphorus from Lake Memphremagog.

8 Lake Modeling Approach

The modeling approach used for this study was based on the general steady-state mass balance equation for a lake segment given in equation 1, used in the development of the Lake Champlain TMDL as described in the diagnostic feasibility study (VTDEC 1997.) This was originally modified from Chapra and Reckhow (1983). A similar model was applied to the entire Great Lakes system by Chapra and Sonzogni (1979) which was updated in 2012 (Chapra 2012) to predict the response of each segment of the Great Lakes to phosphorus loading changes. This model equation is consistent with the BATHTUB model formulation used for Lake Champlain but the alpha and beta weighting factors used by Chapra et al. were not used in BATHTUB or the Lake Champlain or Memphremagog models. The resulting series of mass balance equations, one for each lake segment, was solved for various terms during the modeling procedure, using methods described below. An eight segment model for Lake Memphremagog based on equation 1 is illustrated schematically in Figure 32.

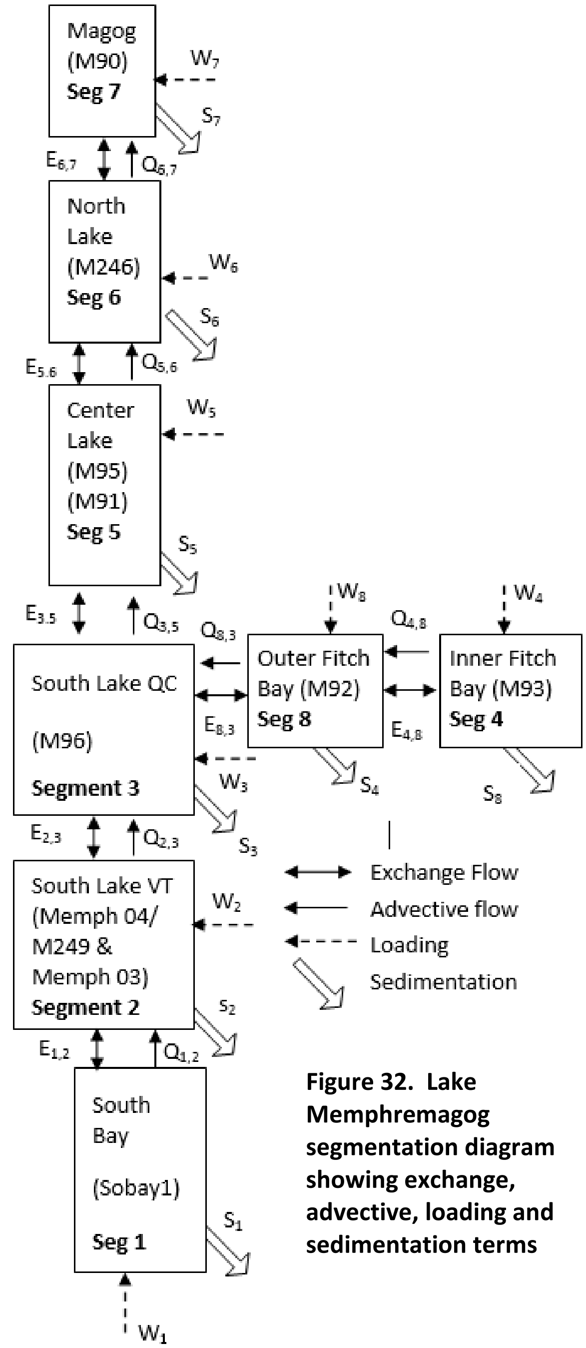


Figure 32. Lake Memphremagog segmentation diagram showing exchange, advective, loading and sedimentation terms

Vi dci/dt = 0 = Wi + 3j {-Qijci + Qjicj + Eij(cj-ci)} - Si (1)

Where

Vi = volume of segment i (hm^3)

ci = concentration in segment i (mg/l)

cj = concentration in adjacent segment j (mg/l)

Wi = direct external mass loading to segment i (mt/yr)

Qij = advective outflow from segment i to adjacent downstream segment j (hm^3/yr)

Qji = advective inflow to segment i from adjacent upstream segment j (hm^3/yr)

Eij = diffusive exchange flow between adjacent segments i and j (hm^3/yr)

Si = net internal sedimentation rate in segment i (mt/yr)

8.1 Chloride Exchange between lake segments

Chloride is used as a conservative substance removing sedimentation from this equation to allow for the estimation of the exchange between lake segments. For the Lake Memphremagog model we assume that bulk exchange flows were equivalent to a lakewide calibrated exchange velocity across the cross-sectional area of each inter-segment exchange interface. We made the assumption that the exchange flow rates should be proportional to the cross-sectional areas of the exchange interfaces, consistent with the findings from the Lake Champlain diagnostic feasibility study (VT DEC 1997). We made the simplifying assumption for the purposes of a minimalist approach to calibration that the proportion was a linear one with a constant exchange velocity applied to all lake segments. The exchange flow rate (hm^3/yr) is modeled as the product of the cross-sectional area of the exchange interface (A_c) in units of hm^2 and the exchange velocity (v_e) in units of hm/yr which results in an estimate of the bulk exchange flow in units of hm^3/yr.

The calibration of the exchange velocity (V_e) was done by adjusting the lake wide exchange velocity using excel solver to achieve a least-squares (minimum RMSE) best fit between predicted and observed chloride concentrations among the lake segments. South Bay and Fitch Bay lake segments were excluded from this RMSE analysis due to minimal predicted exchange between these lake segments due to small cross sectional areas at the segment interface. In addition to this, in the case of South Bay large variability in day to day chloride concentrations add to the uncertainty in using lake data to estimate mean chloride values, especially given typically low chloride concentrations in spring when sampling was lake sampling was not typically completed.

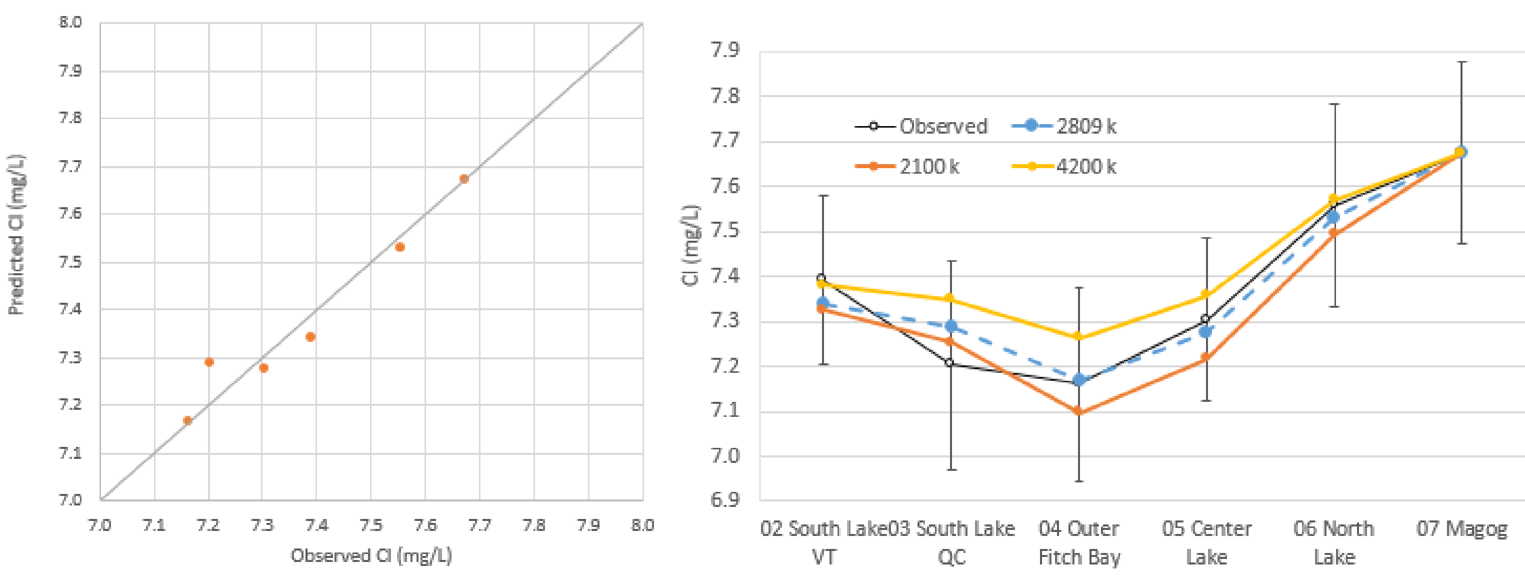


Figure 33. Predicted and observed Chloride concentrations shown as a)a regression using an exchange factor of 2809 hm/yr and b) a gradient from south to north with exchange of 2809 hm/yr as well as 2100 and 4200 to show impacts of higher or lower exchange rates on predicted chloride gradient. Observed values and 95% confidence error bars are shown in black.

One challenge with this approach was to address the short circuiting of chloride that is apparent in the Magog Lake segment described in section 3.7. If the short circuiting of chloride is ignored, then the chloride loading model overestimates the amount of chloride into the Magog lake segment which results in an optimal exchange value of 2,100 hm/yr that has the lowest RMSE. However, if we subtract the estimated 325 metric tons of chloride that short circuit this segment (estimated by multiplying the difference in the outlet concentrations from Magog Bay lake segment concentration times flow) it results in an optimal cross sectional exchange value of 4,200 hm/yr. In the end the approach taken was to force the chloride concentration to match the concentration in the Magog Bay Segment because this isn't sensitive to exchange between lake segments as all chloride makes its way through the final lake segment. This is also justified due to the greater uncertainty in both the estimation of the amount of chloride that short circuits the Magog segment and the uncertainty in the estimate of the chloride flowing into this segment due to the large and heavily paved Magog Bay unmonitored lake subwatershed. Using this intermediate approach results in a K value with the lowest RMSE of 2,809 hm/y or .89 cm/s which multiplied by the cross sectional area results in the estimated exchange flow between lake segments shown in Table 28 below.

These exchange flows are then an input into the phosphorus sedimentation model where sedimentation of phosphorus is calibrated to match modeled concentrations to those measured for each lake segment. A test of sensitivity of phosphorus model to alternative approaches to addressing chloride short circuiting suggests that the impact of this range of exchange velocities makes only a small difference in the resulting sedimentation rates less than a percent change using 2100 hm/y and increasing the sedimentation rate for all but the inflow segments by 3.5% using 4200 hm/y. These changes in exchange and sedimentation rates result in minor changes to the predicted Vermont lake concentration of +/- 0.2 ug/l as compared to using an exchange velocity of 2809 hm/yr.

Table 28 Final exchange coefficients between lake segments based on K value of 2809 hm/yr. Concentrations in South Bay and Inner Fitch Bay were not used for the least-squares analysis, but exchange flow rates for these segments were calculated using the same calibrated lakewide exchange velocity applied to the other segments.

Lake Segment	Cl Load to Segment (mt/yr)	Observed Segment Cl Conc. (mg/L)	Predicted Segment Cl Conc. (mg/L)	Inflow to Segment (hm3/yr)	Cumulative Advective Flow Downstream (hm3/yr)	Calibrated Exchange Flow (hm3/yr)	Cross-Section Area (hm2)	Squared Error (mg/l)
01 South Bay						123	0.04	
02 South Lake VT	6922	7.39	7.34	902	902	5,871	2.09	0.00262
03 South Lake QC	55	7.20	7.29	35	1,002	2,605	0.93	0.00708
04 Inner Fitch Bay						6	0.002	
08 Outer Fitch Bay	361	7.16	7.17	66	66	920	0.33	0.00002
05 Center Lake	384	7.30	7.27	107	1,110	1,372	0.49	0.00079
06 North Lake	195	7.56	7.53	30	1,139	4,564	1.62	0.00072
07 Magog	1346	7.67	7.67	68	1,207			0.00000
							RMSE	0.04328
						Calibrated exchange coefficient in units of (hm/yr)		2,809

8.2 Phosphorus sedimentation

With the bulk exchange between lake segments estimated through the lake chloride modeling, phosphorus loading to the lake estimated through direct measurement of the four major tributaries and phosphorus watershed export model, and in lake phosphorus concentration measurements from Vermont and Quebec it is possible to estimate sedimentation rates using equation 1 and model equations for sedimentation based upon lake morphology, and concentrations and calibrations factors to estimate sedimentation or S in equation 1.

There are a number of phosphorus sedimentation equations that can be used to estimate sedimentation for each segment and two equations were evaluated referred to in this report as first and second order sedimentation equations shown as equations 2 and 3 below.

First Order Sedimentation equation $S = vAc$ (2)
Second Order Sedimentation equation $S = kVc^2$ (3)
Where
S = Sedimentation (mt/yr)
v=Settling velocity (m/y)
c= Lake concentration(ug/l)
k= Calibrated sedimentation coefficient $m^3/g\text{-}yr$
V=Lake segment Volume (hm/y)

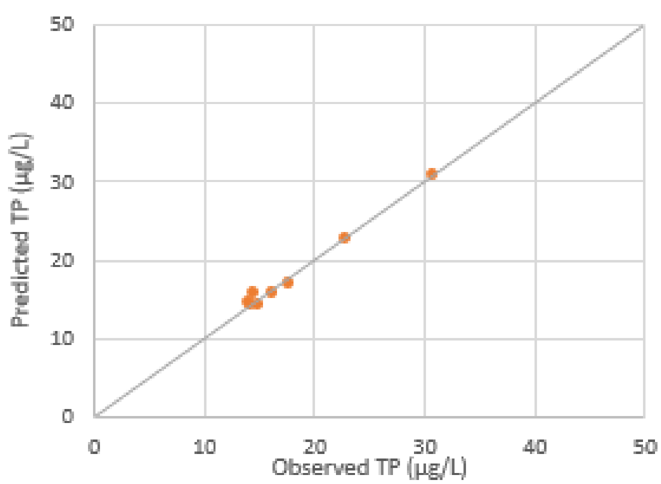
The first-order and second-order sedimentation equations produce generally similar predictions when compared using optimally calibrated sedimentation terms. However, the predicted phosphorus concentrations in the critical Vermont lake segment approach the 14 µg/L phosphorus criterion much more slowly as loads are reduced with the second-order equation than with the first-order equations. Unfortunately, this has a major impact on the phosphorus load allocation that would be determined from the equation, so the choice of first-order vs. second-order model is a critical decision.

The first-order sedimentation equation has a calibrated settling velocity that is applied to the lake segment area. Literature values for the settling velocity range between 10 m/yr from Vollenweider to 16 m/yr from Chapra based on studies of Canadian shield lakes (Reckhow, 1983). A recent publication by Chapra (2012) on the Great Lakes estimated settling velocities ranging from 0 up to 50 for different lake segments over two distinct timeframes. Table 29 includes the inputs to the excel model with the total phosphorus load to each segment, the observed segment concentration, advective flow downstream, and the calibrated exchange flow which was calibrate through the chloride modeling. The equations for this excel model are the same as used for the chloride model described with the addition of the sedimentation factor, vA which is calculated as the settling velocity times the segment area, given in units of cubic hectometers per year. In the excel model this is added to the denominator of the equation phosphorus loading/inflow volume + vA = segment concentration under steady state conditions. vA is calibrated by a sedimentation velocity which is then multiplied by the segment area (in kilometers).

Table 29 excel phosphorus sedimentation model inputs for Lake Memphremagog

Lake Segment	TP Load to Seg kg/yr	Observed Seg TP Conc. µg/L	Model Seg TP Conc. µg/L	Inflow to Seg hm^3 /yr	Cumulative Advective Flow hm^3/yr	Calibrated Exchange Flow hm^3/yr	Segment Area (km2)	Settling Velocity (m/yr)	vA (hm3/yr)	Squared Error (ug/l)
01 South Bay VT	42,463	30.80	30.85	543	543	123	2.3	342.7	778	0.002
02 South Lake VT	10,680	17.73	16.96	335	879	5,871	23.8	17.7	420	0.599
03 South Lake QC	1,316	16.10	15.74	33	978	2,605	20.9	17.7	369	0.128
04 Inner Fitch Bay	3,757	22.93	22.85	61	61	6	2.2	45.9	102	0.006
08 Outer Fitch Bay	284	14.52	15.67	6	66	920	2.7	17.7	48	1.316
05 Center Lake	4,364	14.10	14.55	106	1,084	1,372	26.1	17.7	461	0.206
06 North Lake	1,758	14.31	14.28	29	1,113	4,564	9.5	17.7	167	0.001
07 Magog	3,930	14.79	14.35	68	1,181		9.9	17.7	175	0.191
									RMSE	0.553

Since there are significant limitations on our ability to validate the Lake Memphremagog model described in detail later, a light handed approach to calibrating sedimentation velocities was used allowing for independent adjustments of the settling velocities for the two inflow segments (South Bay and Fitch Bay) and then one settling velocity was applied all other lake segments. Excel solver was used to minimize the RSME between the observed and modeled load by adjusting the sedimentation velocities as described above. With just three degrees of freedom through this method a root mean squared error of 0.55 was achieved using a settling velocity of 342.7, 45.9 and 17.7 m/yr for the South Bay, Fitch Bay, and all other lake segments respectively. The two inflow segments have higher settling velocities than literature values however, high settling velocities were also found for a number of the inflow segments in Lake Champlain including South Lake and Missisquoi Bay segments were settling was nine and three times as high as for other lake segments in Lake Champlain (Tetra Tech 2015a). The higher settling velocity for inflow segments is explained by rapid settling of the sediment bound phosphorus fraction into these inflow segments. This modeling approach resulted in a predicted Vermont lake concentration of 17ug/l or about .7ug/l below the measured concentration over this timeframe for the Vermont Lake segment



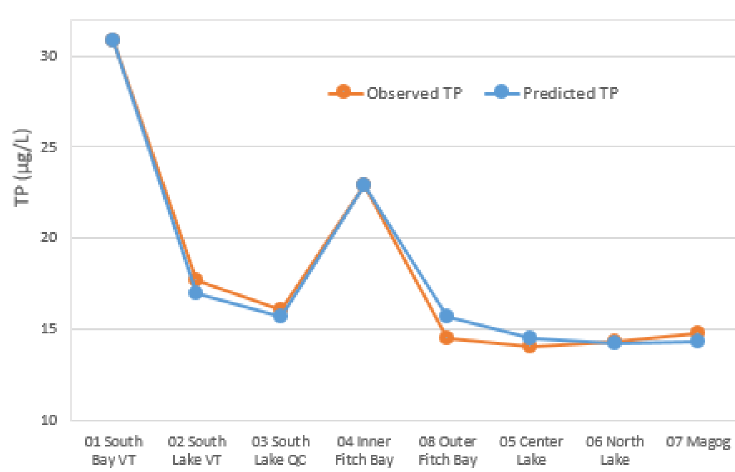


Figure 34 relationship between predicted and observed lake concentrations as a xy plot and line chart showing the lake gradient from south to North.

The second order equation also evaluated is an option provided in the BATHTUB model and used for the Lake Champlain TMDLs in both 2002 and 2016. The second order sedimentation term is kVc^2 where k is the calibrated sedimentation coefficient in units of m^3/g-yr, V is segment volume and c^2 is the segment concentration squared as shown in equation 3. This distinction between a settling area in the first-order equation and a settling volume in the second order equation increases the predicted settling that occurs in the deeper lake segments but the increase settling with increased phosphorus concentration also results in a predicted increased settling rate in segments with higher in lake phosphorus concentrations. This second factor is particularly important because as lake phosphorus concentrations are reduced through watershed phosphorus reduction projects, the second order model predicts a reduced phosphorus sedimentation rate acting as a negative feedback on the lake response as lake phosphorus concentrations are reduced. Settling velocities provided by Walker (1999) in the BATHTUB documentation include a settling term of 100 cubic meter/g-yr. Similar to the first-order sedimentation equation we allowed for independent adjustments of the settling velocities for the two inflow segments (South Bay and Fitch Bay) and then one settling velocity was applied all other lake segments. Excel solver was used to adjust these settling terms to minimize the root mean squared error between measured and modeled lake concentration for the 2009-2012 timeframe resulting in settling terms of 3,539, 709, and 66 m^3/g-yr for the South Bay, Fitch Bay, and all other lake segments respectively. Similar to the first order equation both inflow segments have higher than typical settling terms, however the settling term for the remaining lake segments of 66 m^3/g-yr is not far from the literature value of 100 m^3/g-yr or the value for most Lake Champlain segments in the bathtub calibration report of 140 m^3/g-yr (Tetra tech 2015a). The second-order model results in a higher root mean squared error of 0.94 ug/l suggesting that the first order model better represents sedimentation in Lake Memphremagog.

Additional calibration approaches were also considered for the first and second-order models including allowing an additional segment to have an independent calibration of settling which reduced the RSME for both first order and second order options but the RSME remained lower for the first order model across all approaches considered. While the addition of the independent settling velocity for the center lake segment and allowing outer Fitch bay to have the same settling factor for inner Fitch bay reduced the RSME substantially to .17 for the first-order equation, the lack of a good model validation dataset for Lake Memphremagog suggests that we should use as light as possible calibration for the model and rely on the similarity of the resulting settling velocities to those reported for other lakes. The similarity of the calibrated settling velocity for the non-inflow segments at 18 meters per year is solidly in the middle of the Chapra calibration for Great Lakes segments (Chapra 2012) and just above that included the Chapra study of 14 Canadian shield lakes of 16 m/y (Reckhow, 1983) which supports this modeling approach.

Table 30 Sedimentation calibration approaches for Lake Memphremagog applying different degrees of freedom and segment groupings with the resulting RMSE and Vermont lake predicted lake concentration for the 1st and 2nd order sedimentation equations described above showing constantly lower RSME for 1st order approaches.

Degrees of freedom	Segment grouping	Root Mean Squared Error		Seg 2 modeled P ug/l (measured 17.73 ug/l)	
		1st order	2nd order	1st order	2nd order
3	S Bay, Fitch Bay, all other segments	0.55	0.94	16.96	18.14
3	S Bay, Fitch bay inner/outer, all other segments	0.36	0.45	17.08	18.29
4	S Bay, Fitch in/out, Vt Lake, All other segments	0.29	0.42		17.97
4	S Bay, Fitch in/out, Center Lake, all other segments	0.17	0.43	17.55	

8.3 Model Validation

Validation for the Lake Memphremagog phosphorus sedimentation model is complicated by the large differences in lake concentrations between the Vermont and Quebec datasets from the calibration and validation timeframe and lack of direct loading estimates from Quebec Tributaries. The limitations in loading from unmonitored portions of the watershed were addressed by prorating loading from these areas based on the modeled loading and flow from the 2009-2012 timeframe related to the loading and flow from the Black, Barton, and Johns rivers over this same timeframe. The resulting ratio was applied on an annual basis based on the load and flow from these three tributaries to estimate the annual flow and load from the unmonitored areas. These annual flow and loading numbers were then input into the excel lake sedimentation model as calibrated above and the estimated concentration for the Vermont lake segment was compared to the measured values for each of the eight years this data is available.

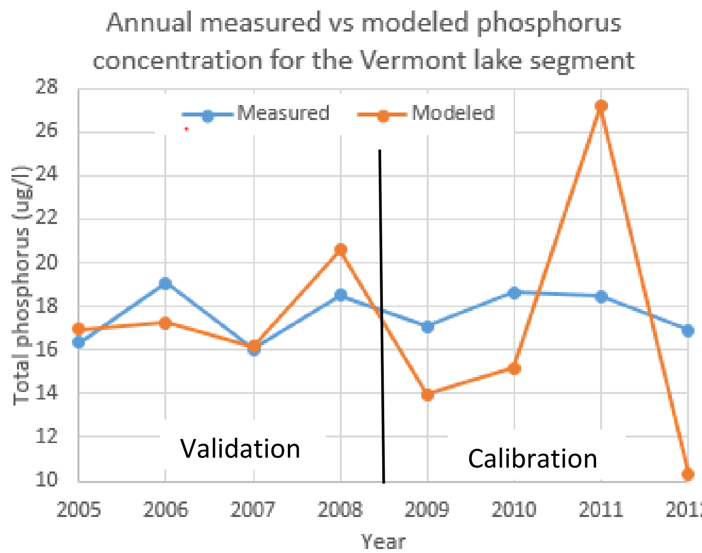


Figure 35. Annual modeled and measured phosphorus concentrations for the Vermont Lake segment

The model prediction error over the full 8-year timeframe is just 0.4 ug/l below the measured average concentration or -2.8% however the model under predicted lake concentrations during the calibration timeframe by over seven percent and over predicted over the validation timeframe by about 1.5%. On a year to year basis the model error could be substantially higher, upwards of 45% in 2011, and nearly 40% the following year which resulted in larger RMSE of 4.4 ug/l.

Interestingly, the root mean squared error is much higher for the calibration vs. validation timeframe which appears related to extremely high predicted concentrations in 2011 related to spring runoff event and tropical storm Irene, and also predicted low concentrations the following year. There are a number of explanations related to this including the fact that

Table 31. Annual modeled and measured phosphorus concentrations for the Vermont Lake segment with unmonitored flow and loading proportional to the Black Barton and Johns Rivers over the model time frame of 2009 through 2012.

Year	Measured TP ug/l	Modeled TP ug/l	Error (ug/l)	Percent Error	Absolute Error (ug/l)	Absolute Percent Error	Squared Error (ug/l)
2005	16.3	17.0	-0.6	3.8%	0.63	3.8%	0.4
2006	19.1	17.2	1.8	-9.6%	1.84	9.6%	3.4
2007	16.0	16.2	-0.1	0.8%	0.13	0.8%	0.0
2008	18.5	20.6	-2.1	11.4%	2.11	11.4%	4.4
2009	17.1	14.0	3.1	-18.4%	3.15	18.4%	9.9
2010	18.7	15.2	3.5	-18.6%	3.48	18.6%	12.1
2011	18.5	27.2	-8.7	46.9%	8.67	46.9%	75.1
2012	16.9	10.3	6.6	-39.1%	6.60	39.1%	43.6
Mean	17.6	17.2	0.4	-2.8%	3.3	18.6%	4.4*
Mean 2005-2008	17.5	17.7	-0.3	1.6%	1.2	6.4%	1.4*
Mean 2009-2012	17.8	16.6	1.1	-7.3%	5.5	30.7%	5.9*
Mean 2005-2010	17.6	16.7	0.9	-5.1%	1.9	10.4%	5.0*
95% confidence	1.0	4.3					

***RMSE**

sampling was not done for a month and a half in early 2011 when in lake concentrations were likely most elevated and so the measured concentrations likely underestimate the annual average. Additionally, phosphorus loading from this high runoff year may have been weighted to sediment bound phosphorus from streambank and surficial erosion which would be expected to sediment out rapidly vs dissolved phosphorus therefore have less impact on increasing in-lake concentrations. Finally, phosphorus loading from Tropical storm Irene in late 2011 may have been held over in the lake into 2012 based on the lake retention time of a year and a half which is likely to have contributed to the large under prediction in the lake concentration from the model for 2012 since the annual model doesn't consider the potential for elevated lake concentrations from loading in the previous year.

9 Scenario tool

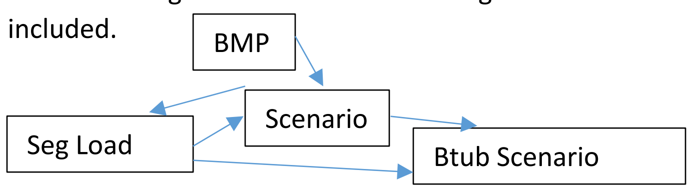
The next step in the modeling for the Lake Memphremagog TMDL is to create an BMP scenario that can provide a reasonable assurance that loading reductions can be achieved as detailed in the TMDL. A large amount of work has been done by EPA and Tetra tech to create a Lake Champlain Scenario tool for the creation of the Lake Champlain TMDL (Tetra Tech 2015c). This Scenario tool has some features which are not compatible with the Lake Memphremagog model so that some adjustments are required to allow the use of the BMP efficiencies which are the key element of the Scenario tool. Fortunately, these differences can be addressed by averaging BMP efficiencies across slopes, and in some cases soil hydrologic groups and consolidating land uses. The following is a list of these simplifying assumptions:

1) Averaged BMP efficiency across all three slope categories. Most of the BMP efficiencies were not significantly different across different slope categories so this should have limited impact on the model

2) With the exception of Cropland averaged all BMP efficiencies across all soil hydrologic groups.
3) Combined BMP efficiencies across all high density, medium density, low density and commercial Pervious and Impervious land uses.
4) The Memphremagog export model does not have land in corn hay rotation as a land use category so none of the BMPs associated with this land use are available in the Magog Scenario tool.

A full list of the BMP's and BMP phosphorus reduction efficiency applied is shown in Appendix B

The scenario tool was created in Excel with an input tab of the existing loading by land use category as well as from streambank erosion, WWTF, and Septic systems from each lake segment. These loadings are estimated as loadings that reach the lake so upland lake retention has been included.



Another tab includes the BMP efficiencies for each land use and each BMP type as taken or averaged from the Lake Champlain Scenario took as described above. The Scenario Tab then draws from the loading of each land use to each segment when a Segment and Land use are selected from a drop down Menu. Another drop down menus is available to select the BMP to be applied to this land use and the efficiency will be updated from the BMP tab. Once a percentage of this land use for which this BMP is applied is selected then the total load reduction achieved is calculated in the final column which is simply the load from that land use in that segment times the percent of that land use to which the BMP is applied times the phosphorus reduction efficiency of the selected BMP.

Steps to using the BMP scenario tool

1) Select a land use (the Lu code should automatically be added and the Hectars of that land use in the segment as well as the estimated loading to the lake will be automatically populated.
2) Next a BMP must be selected, and you have to scroll through the long list. Many of the BMP's only apply to one or at most a few land uses so it is recommended that you check the BMP tab for the BMP's available for the each land use. The BMP's are alphabetical. Another approch is to simply copy and paste the BMP from the BMP tab if the drop down menu is challenging.
3) Finally, the percentage of the land use in that segment for which the BMP will be applied needs to be selected. When this is selected the hectars treated and Load reduction in KG will be calculated.
4) To add a new BMP in a lake segment you need to copy and paste an existing BMP row from the same segment into the center of the segments (not at the end or this load reduction may not get added in the segLoad tab)

The SegLoad tab will then update automatically with loading reductions applied to the Load reduction table, a percent load reduction by land use table, and finally the new loading table with the new loading to each segment with load reductions applied.

The loading in the New Loading table is then applied to the segments in the Bath Tub Scenario tab. To this the margin of safety, an allocation for future stormwater runoff from developed lands and any change in the WWTF permit load from the current loading over the modeling timeframe can be added to each lake segment. With this new loading to lake segments a lake response to loading reductions can be observed with the target of 14 ug/l in the Vermont Main lake segment.

9.1 Phosphorus estimate from stormwater from future growth

The increased phosphorus loading due to stormwater growth from developed lands was calculated based on the methodology developed for this purpose for the Lake Champlain TMDL (EPA 2016??). This methodology was based on a number of assumptions based on the amount of newly permitted impervious surface growth for each lake segment between 2005 through 2014. This data has not been made available for the Lake Memphremagog basin yet and so an average of the growth per existing developed acreage was used (.35% per year) across the Lake Champlain basin in Vermont as a whole. The range for annual growth rates for lake segments was low of .02% for South Lake A, up to .83% for St Alans bay. Some basins that seem to have the most similar development patters as Lake Memphremagog are Missisquoi bay and Otter Creek which had annual growth rates of .25% and .26% respectively but as a conservative assumption the average of 0.35% was used.

An assumption was made that for each area of permitted impervious growth an equal amount of unpermitted development has been occurring. The estimated increase in loading was then calculated by multiplying the estimated new permitted area by 1 times the estimated loading for this area of developed lands based on the assumption that new umpermitted growth would result in this increase as no stormwater treatment is required. The estimate of loading for permitted sites is based on the standard treatment scenario with a phosphorus removal efficiency of 71.2% resulting in a multiplier of .288 (Lake Champlain TMDL appendix A). Finally, an area equal to 11% of annual percentage growth is considered to be redeveloped and a credit of 25% load reduction was taken based on the retrofit treatment efficiency standard. The total loading from stormwater growth is shown below in table 32.

Table 32 Predicted stormwater loading from future growth for segments one and two.

Segment	Impervious load (kg)	annual % growth permitted imp.	Increased load multiplier	redevelopment load red. (Kg)	20 yr Load increase (kg)
1	1701	0.35%	1.288	3	153
2	922	0.35%	1.288	2	83

Loading from forest lands that have been converted to developed impervious lands also needs to be included in this calculation. This was done by estimating the percentage of forested lands which were developed and removing that percentage of the forested loading from the model as shown in table 33 resulting in a net loading from development over twenty years of 143 and 79 kg for segments 1 and 2 respectively.

Table 33 Predicted reductions in forest loading due to development and net load from development over 20 a year timeframe.

Segment	Impervious Hectares	new imp. hectares	forest hectares	New imp. area as a % of forest area	forest load	forest load reduction (kg)	net load adj (kg)
1	837	117	54957	0.21%	3358	7	143
2	627	88	30529	0.29%	960	3	79

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Appendix A. Smi Lake Memphremagog phosphorus export model technical corrections and updates and Lake Retention calculations.

The correlation between lake size and retention using equations R1, R2, and R3 by SMi in the model was:

Equation 1 Y = -0.0008x + 0.1871

Equation 2 Y = 0.0005x + 0.2897

Equation 3 Y = 0.00020x + 0.20130

These equations for lakes that don't have residence time are based entirely on the correlation between lake size and retention for just 7 lakes in Quebec and is not very accurate because it ignores basin area which is a significant factor in residence time and therefore lake retention.

To improve the estimation of residence time for these ponds without known volumes correlations between mean depth and max depth, and mean depth and lake size was used to estimate mean depth and then residence time can be calculated based on other known factors.

The following ponds have max depth values and so a relationship between the mean depth for Vermont lakes in the Lake Memphremagog watershed with known residence times and max depths were used to create a correlation which has a high r value (.9569). This equation is:

Mean depth = 0.4438 (max depth) - 0.1124

This was used to calculate the residence time for: CHARLESTON, LONG (SHEFLD), MUD (MORGAN)-N, MUD (CRAFBY), TOAD (CHARTN), WALKER (COVNTY), MUD (MORGAN)-W, Round Pond (SHEFLD), Little Salem Lake (CHARTN).

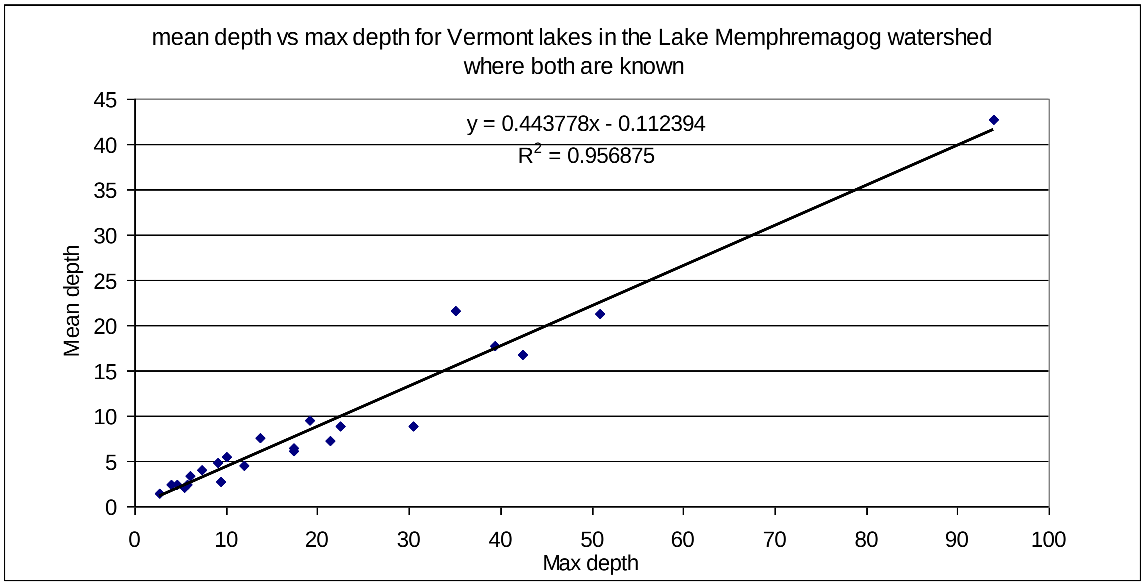


Figure A1 mean depth vs Max depth for Vermont lakes in the lake Memphremagog Watershed where both were known.

The residence time for these lakes can then be calculated by multiplying the lake area by the estimated mean depth based on the correlation with max depth using the equation above, and then dividing by the flow which is calculated by multiplying the basin area by

mean annual runoff from the USGS report "MEAN ANNUAL RUNOFF, PRECIPITATION, AND EVAPOTRANSPIRATION IN THE GLACIATED NORTHEASTERN UNITED STATES, 1951 – 80"

For lakes that do not have a known max depth a correlation between the lake size and mean depth can be used. This correlation is not as good as the correlation for max depth and has an R^2 of .71. The equation is: mean depth (meters) = .04118*Lake Area(hectars) + 3.2472.

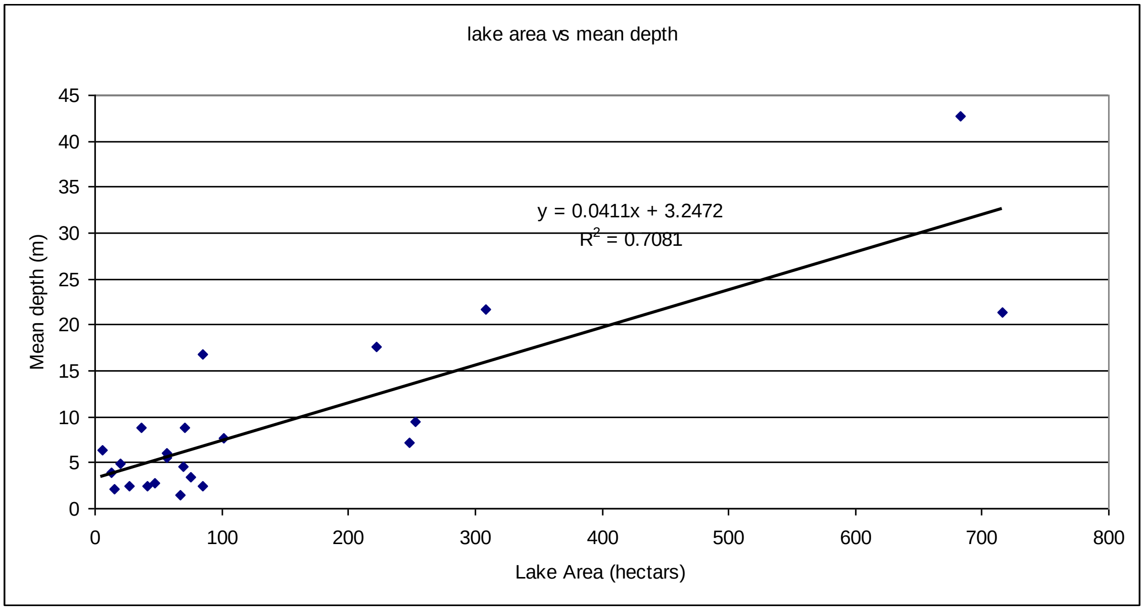


Figure A2 mean depth Lake Area for Vermont lakes in the lake Memphremagog Watershed where both were known.

The retention time can then be calculated using this correlation as it was for the lakes which had a max depth. For lakes in Quebec an average annual runoff of 24 inches was used based on an extrapolation from the USGS report and a runoff map of Canada that showed a consistent amount of runoff for the lake Memphremagog portion of Quebec with a gradient to higher runoff from southwest to northeast. This may be updated to be more accurate if better information on runoff is available in Quebec.

The SMi version of the phosphorus export model used a retention equation MR1 that was not calculated correctly. This was corrected using the equation: retention = $1/(1+T^{-.5})$ where T is residence time. The impact of this error can be seen in the graph below for lakes used in the model where the retention was calculated directly and this error resulted in an underrepresentation of the retention for lakes as compared to the correct equation. This error would have also made the correlation between lake size and retention for lakes without a known retention time incorrect.

Retention equations used in SMi model where T is residence time in years

R1 Retention=$1/(1+T^{-.5})$
R2 Retention =(0.25+0.18t) /(1+0.18t)
R3 Retention =0.20+0.031t

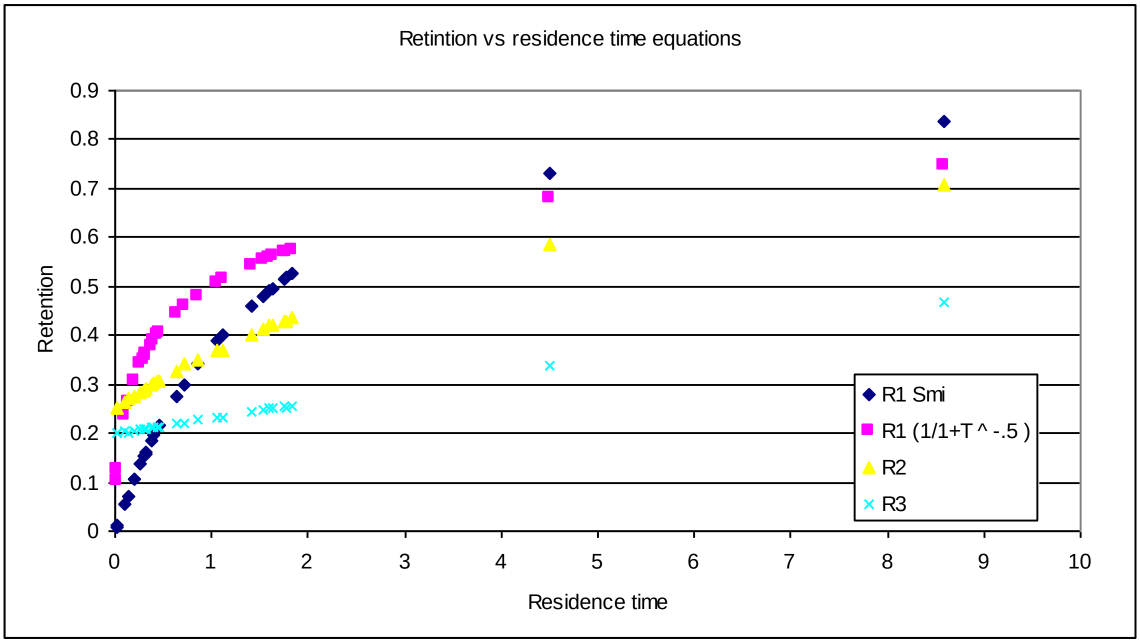


Figure A3. Graph of estimated retention vs residence time for 3 retention equations

Finally all retention from lakes between 2-4 hectares was removed from the model because it was felt that these smaller ponds did not likely retain much phosphorus.

With these changes in the phosphorus retention in the model the amount of retention in the model for the four major tributaries in Vermont is shown in the following table.

Table A1 total retention across 3 different sub watersheds for three different retention equations

	R1 % ret	R2 %ret	R3 % ret
Barton	12%	10%	7%
Black	2%	2%	1%
Clyde	42%	59%	50%
Johns	0%	0%	0%
total	18%	22%	18%

Based on discussions with a number of technical staff and literature review equation R1 appears to be the most scientifically supported retention model. When using this retention equation however the modeled loading from the Clyde River watershed remained considerable higher than what was measured. Retention equation 1 estimated very small amounts of retention for a number of the lakes and ponds in the Clyde River watershed where retention time was short. To better match the measured loading an assumption was made that all these lakes and ponds provide some minimal level of retention and so that minimum level of retention was set to be 14 % across all lakes and ponds which essentially set the level of modeled loading from the Clyde River watershed to match the measured loading. Of the 69 lakes with a size of greater than 4 hectars only 9 were impacted by this change many of which were in the Clyde River watershed. One justification for this is that two of the alternative retention equations that were considered as part of the Original SMi model have retention at 0 residence time of 25% (R2) and 21% (R3) and so maintaining some level of retention for lakes with a short residence time is an intermediate approach between the R1 equation and R2 and R3 equations. Also literature on estimates of retention suggest that even lakes with a short residence time can have significant levels of retention if there are sediment sources upstream in which case these sediments will settle out in lakes very quickly and so it is reasonable to assume some minimal level of retention across all lakes.

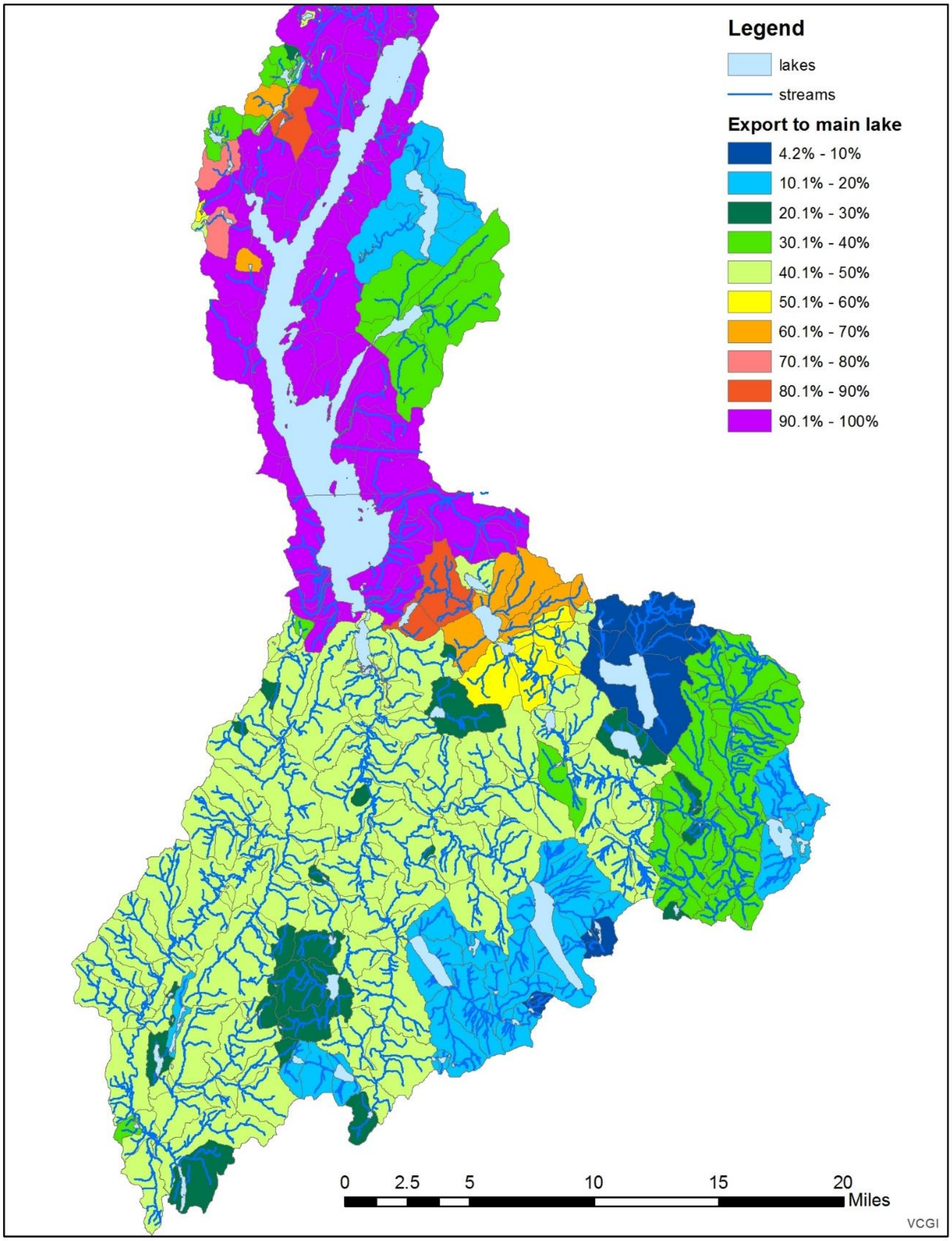


Figure A4. estimated retention of phosphorus to main Lake (including South Bay and Fitch Bay retention) using the R1 retention equation with a minimum lake retention of 14%

Table A2 Lakes which have less than 14% retention which have been adjusted.

BASIN	Lake	LakeArea Hectars	BasinArea Hectars	MeanDepth(m) known, max area	lake volume2	Q (m^3/yr)	Residence time 2	R1 (1/1+T - .5)	R1 min 14%
12	MUD (MORGAN)-N	14.2	487.6635	0.293	41555	2951391	0.0141	10.6%	14%
13	CLYDE	75.3	37001.9621	3.353	2523670	237899170	0.0106	9.3%	14%
22	MUD (MORGAN)-W	4.5	450.8358	0.293	13060	2451172	0.0053	6.8%	14%
49	TOAD (CHARTN)	8.9	876.5802	0.429	38163	4960266	0.0077	8.1%	14%
212	Clyde River wetlands	21.2	16731.3942	4.119	873625	114743901	0.0076137	8.0%	14%
221		4.9	972.1413	3.451	170759	6665483	0.0256	13.8%	14%
251	CHARLESTON	16.2	28083.1740	3.945	638667	184924292	0.0035	5.6%	14%
253	Little Salem Lake	66.0	31322.778	1.327	876117	203878390	0.0043	6.2%	14%
298	PENSIONER	70.0	27067.1454	4.572	3200843	154483877	0.0207	12.6%	14%

A number of small and large adjustments to watershed subbasins and flow paths were also made to more accurately represent likely retention in the basin. These corrections are listed below.

1) Small basin 217 includes both great hosmer and little hosmer so this needed to be split and was done so (created basin # 322 for great hosmer pond watershed)

2) Little hosmer pond not in the lake table so added

3) Derby Lake listed in the table but the mean depth is not so added in calculated residence time based on the mean depth

4) Clyde River was routed through Page Pond. Created new basin 184 for Page Pond and updated the model so Clyde River does not flow through this pond.

5) Updated the model so the Barton River does not flow through pond in basin 252

6) Updated the model so the Black River does not flow through Griggs and Potters Ponds with the associated retention.

7) Long pond (Shefield) and round pond were in the same basin so added basin #324 for Round Pond watershed.

8)There were some very small basins that appeared to be mistakes so these were combined with adjacent basins. These included:

-Deleted minor basin 294 and added to basin 105
-Deleted minor basin 300 and added to 21
-Deleted minor basin 324 and added to 311

9) Basin 110 has a small 6 ha pond (heart pond) that was included for retention although only a very small area of the basin drains to the watershed. Added basin 328 to address retention in this small watershed and removed retention from basin 110.

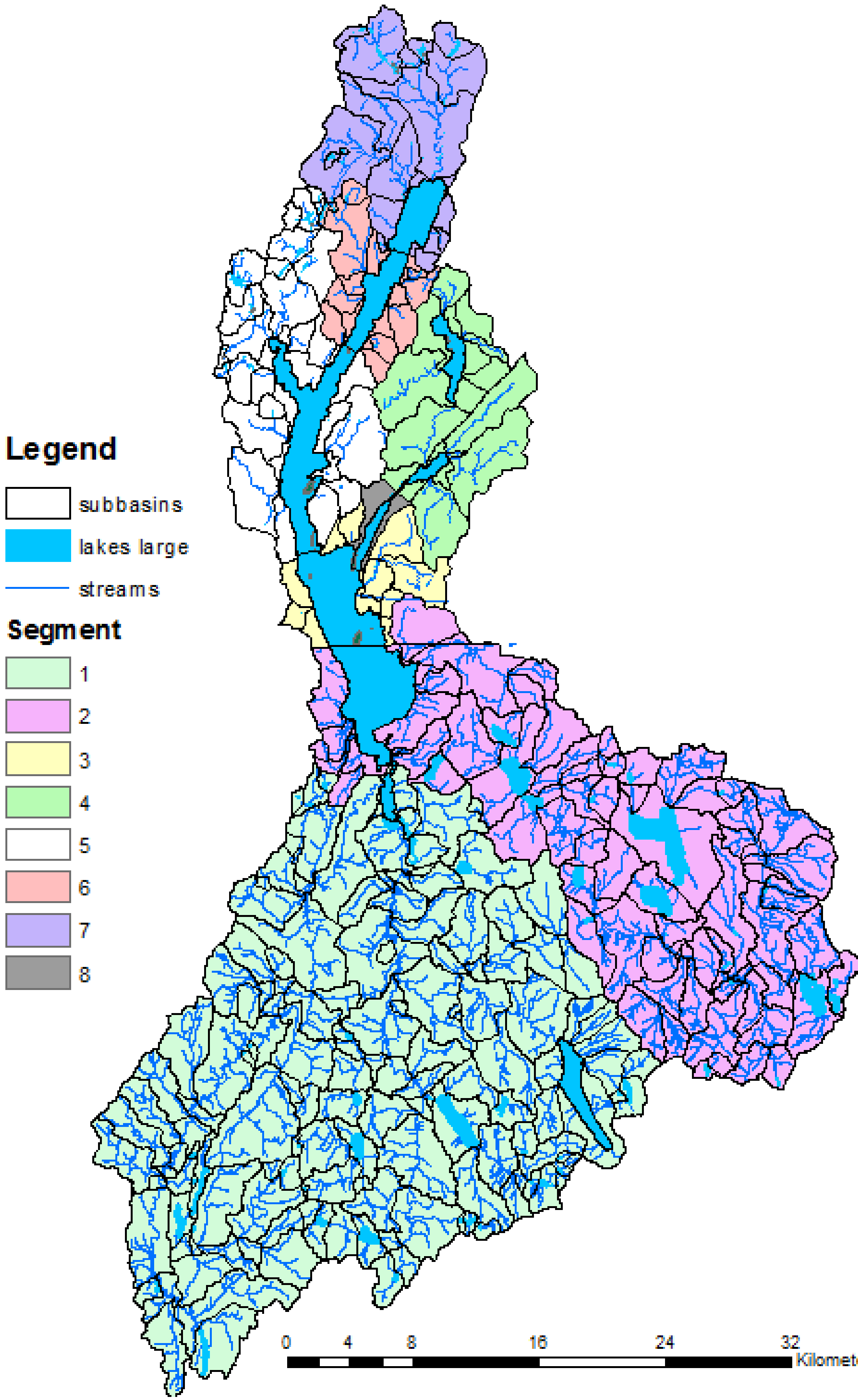
10) Retention from Beacher Pond is not addressed in the model but is very small and has tiny undeveloped watershed so not fixed.

11) the model had about 100 hectars of land in derby line draining to the johns river which drains to the Tomofobia. This watershed was corrected using 20 foot contours.

12) subbasin watershed boundaries were cut at segment boundaries and either lumped with other direct drainages or tied with I created a new subbasin to capture this drainage area.

13) Lake subbasins were created to match segment boundaries.

14) New subbasins were added to allow for modeling estimates for smaller direct tributaries to Lake Memphremagog where water quality sampling has taken place.



Legend
subbasins
lakes large
streams
Segment
1
2
3
4
5
6
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8
0
4
8
16
24
32
Kilometers

Table A3. Inputs used to estimate lake retention for lakes greater than 4 hectares in the lake Memphremagog watershed. Mean depths colored Black depth were estimated through known data, Blue through relationship with lake area, and green with relationship with max depth.

BASIN NUM	Lake Name	Lake Area Hectars	BasinArea Hectars	MeanDepth(m) known, max area	Lake Volume Hectars^3	Q (hectars^3 /yr)	Residence time (years)	R1 (1/1+T - .5)	R1 min 14%	R2 (0,25+0,18t) /(1+0,18t)
7	SALEM	248.2	34779	7.2	17.97	226.45	0.079	22.0%	22.0%	26.1%
8	Derby Lake	85.7	457	2.4	2.09	2.55	0.819	47.5%	47.5%	34.6%
12	MUD (MORGAN)-N	14.2	488	0.3	0.04	2.95	0.014	10.6%	14%	25.2%
13	CLYDE	75.3	37002	3.4	2.52	237.90	0.011	9.3%	14%	25.1%
15		5.4	157	3.5	0.19	1.00	0.187	30.2%	30.2%	27.4%
18	WALKER (COVNTY)	7.3	125	1.1	0.08	0.67	0.121	25.8%	25.8%	26.6%
20	SEYMOUR	715.9	5229	21.3	152.74	33.89	4.507	68.0%	68.0%	58.6%
22	MUD (MORGAN)-W	4.5	451	0.3	0.01	2.45	0.005	6.8%	14%	25.1%
29	Cobb Pond	7.6	77	3.6	0.27	0.42	0.636	44.4%	44.4%	32.7%
34	BROWNINGTON	56.3	1362	5.5	3.09	7.61	0.406	38.9%	38.9%	30.1%
38	KIDDER	6.5	87	3.5	0.23	0.40	0.573	43.1%	43.1%	32.0%
39	ECHO (CHARTN)	222.6	6146	17.7	39.35	37.46	1.050	50.6%	50.6%	36.9%
49	TOAD (CHARTN)	8.9	877	0.4	0.04	4.96	0.008	8.1%	14.5%	25.1%
55		4.1	208	3.4	0.14	1.40	0.100	24.0%	24.0%	26.3%
60	SPECTACLE	41.7	414	2.4	1.02	2.74	0.371	37.9%	37.9%	29.7%
88		9.0	307	3.6	0.32	1.64	0.198	30.8%	30.8%	27.6%
89	CRYSTAL (BARTON)	308.8	5849	21.6	66.82	36.52	1.830	57.5%	57.5%	43.6%
91	LONG (WESTMR)	36.4	296	8.8	3.22	2.11	1.528	55.3%	55.3%	41.2%
93	BAKER (BARTON)	20.6	582	4.9	1.01	3.25	0.309	35.7%	35.7%	29.0%
97	MAY	46.9	440	2.7	1.29	2.90	0.443	40.0%	40.0%	30.5%
101	PARKER	101.2	2193	7.6	7.71	12.25	0.629	44.2%	44.2%	32.6%
106	PAGE	6.5	43	3.5	0.23	0.30	0.768	46.7%	46.7%	34.1%
108	WHEELER (BARTON)	6.1	300	3.5	0.21	2.11	0.101	24.1%	24.1%	26.3%
109	HARTWELL	6.5	252	6.4	0.41	1.29	0.322	36.2%	36.2%	29.1%
120		7.8	170	3.6	0.28	0.91	0.307	35.7%	35.7%	28.9%
123	SHADOW (GLOVER)	85.0	1447	16.8	14.25	8.10	1.758	57.0%	57.0%	43.0%
131	TILDYS	13.4	436	4.0	0.53	2.66	0.199	30.8%	30.8%	27.6%
132	MUD (CRAFBY)	14.2	171	0.8	0.12	1.19	0.100	24.0%	24.0%	26.3%
156	Étang George	12.9	526	3.8	0.49	3.21	0.152	28.0%	28.0%	27.0%
162		16.9	2045	3.9	0.67	12.47	0.053	18.8%	18.8%	25.7%
184	TOAD (MORGAN)	4.9	144	3.4	0.17	0.99	0.169	29.1%	29.1%	27.2%
212	Clyde River wetlands	21.2	16731	4.1	0.87	114.74	0.008	8.0%	14%	25.1%

221		4.9	972	3.5	0.17	6.67	0.026	13.8%	14%	25.3%
226	ISLAND	253.3	2548	9.4	23.94	16.93	1.414	54.3%	54.3%	40.2%
228	Lac à la Truite	34.2	368	4.7	1.59	2.24	0.711	45.7%	45.7%	33.5%
229	Étang de la Cuvette	9.1	156	3.6	0.33	0.95	0.347	37.1%	37.1%	29.4%
230	Étang Fer de Lance	5.8	148	3.5	0.20	0.90	0.225	32.2%	32.2%	27.9%
231	Étang aux Cerises	61.6	1658	5.8	3.56	10.11	0.352	37.2%	37.2%	29.5%
232		8.7	1318	3.6	0.31	8.03	0.039	16.5%	16.5%	25.5%
233	Étang O'Malley	17.5	63	4.0	0.69	0.38	1.817	57.4%	57.4%	43.5%
234	Lac des Sittelles	41.7	624	5.0	2.07	3.80	0.543	42.4%	42.4%	31.7%
235	Lac Gilbert	18.6	157	4.0	0.74	0.96	0.777	46.8%	46.8%	34.2%
237	Étang Peasley	23.3	1285	4.2	0.98	7.84	0.125	26.1%	26.1%	26.6%
238	Lac Nick et Lac Sperling	52.9	575	5.4	2.87	3.50	0.819	47.5%	47.5%	34.6%
240	Étang Fisher	17.7	1081	4.0	0.70	6.59	0.107	24.6%	24.6%	26.4%
241		5.4	58	3.5	0.19	0.35	0.531	42.2%	42.2%	31.5%
242		6.4	83	3.5	0.22	0.51	0.443	40.0%	40.0%	30.5%
244		6.7	203	3.5	0.23	1.24	0.190	30.3%	30.3%	27.5%
245	Lac Malaga	22.3	63	4.2	0.93	0.39	2.403	60.8%	60.8%	47.6%
246		6.2	96	3.5	0.22	0.51	0.424	39.4%	39.4%	30.3%
248	ELLIGO	70.4	1310	8.8	6.22	7.33	0.850	48.0%	48.0%	34.9%
249	LONG (SHEFLD)	15.4	94	3.9	0.61	0.62	0.980	49.7%	49.7%	36.2%
250	JOBS	15.8	108	2.1	0.34	0.74	0.456	40.3%	40.3%	30.7%
251	CHARLESTON	16.2	28083	3.9	0.64	184.92	0.003	5.6%	14%	25.0%
253	Little Salem Lake	66.0	31323	1.3	0.88	203.88	0.004	6.2%	14%	25.1%
256		5.5	150	3.5	0.19	0.80	0.240	32.9%	32.9%	28.1%
258	GRAFT;	4.9	59	3.4	0.17	0.40	0.414	39.2%	39.2%	30.2%
266	Lac Webster	5.6	76	3.5	0.20	0.46	0.425	39.5%	39.5%	30.3%
279		5.4	45	3.5	0.19	0.24	0.784	47.0%	47.0%	34.3%
287	MUD-W;	4.9	78	3.4	0.17	0.53	0.314	35.9%	35.9%	29.0%
298	PENSIONER	70.0	27067	4.6	3.20	154.48	0.021	12.6%	14%	25.3%
304	VAIL	6.5	79	5.7	0.37	0.54	0.679	45.2%	45.2%	33.2%
305	BEAN (SUTTON)	12.1	633	3.7	0.45	4.34	0.105	24.4%	24.4%	26.4%
306	DANIELS	26.7	437	2.4	0.65	2.44	0.267	34.1%	34.1%	28.4%
316	Lovering	486.6	4808	23.2	113.13	29.31	3.860	66.3%	66.3%	55.7%
322	GREAT HOSMER	56.7	341	6.1	3.45	2.12	1.628	56.1%	56.1%	42.0%
324	Round Pond (Shefield)	5.6	136	7.0	0.39	0.90	0.436	39.8%	39.8%	30.5%
327	WILLOUGHBY	682.7	4960	42.7	291.32	33.90	8.593	74.6%	74.6%	70.6%
328	Heart Pond	4.5	23	3.4	0.15	0.13	1.155	51.8%	51.8%	37.9%

Appendix B. Quebec Tributary loading estimates

Quebec Tributary data was collected by the MRC du Memphremagog since 1998. Samples were processed at the SMi laboratory. Since 2010 a series of split samples were taken between the Vermont DEC and the SMi Laboratory and the resulting relationship was established DEC Lab value = 1.1456*Smi Value + 6.1561. With an R^2 value of .8154. All the Smi laboratory data was then transformed into DEC laboratory equivalents to use for estimating phosphorus loading from Tributaries. Due to some uncertainty with regards to sample site locations only data from 2000 through 2013 were used in this analysis.

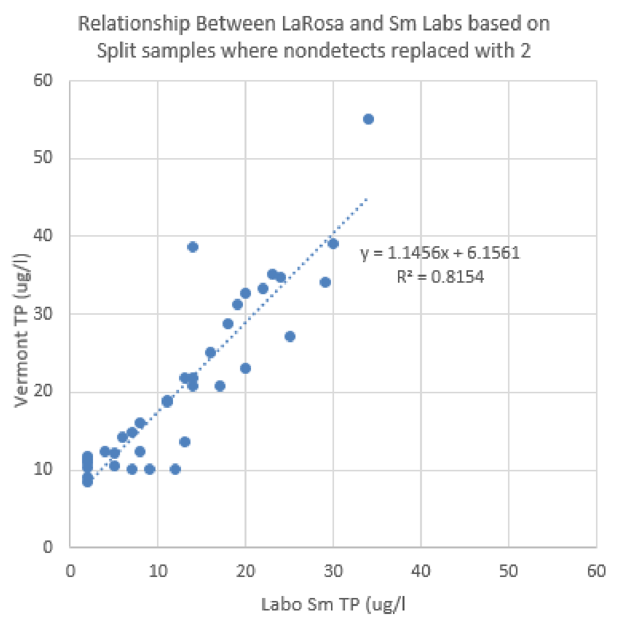

Figure B1 split phosphorus samples processed at the Vermont LaRosa Laboratory and Labo SM.

Quebec Tributaries are sampled five times per year with two sampling events focused on wet weather conditions and three targeting dry weather conditions. Data from this sampling program has generally been presented as median values but for the purposes of estimating loading the mean value was used to give greater weight to higher values which tend to occur during wet weather conditions. Castle 1 and Castle 2 sites, along with Taylor 1 and Taylor 2 were considered together for the purpose of estimating loading due to close proximity and relatively minor changes in land use between site locations. The minimum number of sample dates chosen to evaluate loading was 20. Flows used to estimate loading were estimated through the hydrotel program or as a drainage area ratio with the nearby coaticook flow gage.

Table B1. Quebec Lake Memphremagog Tributary Sample sites with Mean phosphorus values based on Smi Laboratory and converted in to Vermont Mean TP levels based on split sample regression. Sites with an apparent relationship with flow are highlighted and daily phosphorus load was evaluated using the Flux program.

Row Labels	Count	Mean TP (Smi)	Mean VT TP	Relationship with flow
Anse 1	32	32.0	42.9	slight negative
Boynton 1	25	17.4	26.1	Positive
Bunker 1	62	34.6	45.8	none
Castle 1,2	61	22.4	31.9	none
Cerises 1	52	31.9	42.7	slight positive

Château 1	30	8.9	16.3	None
Chemin Taylor 1	20	12.1	20.0	Positive
Fitch 2	54	15.7	24.2	None
Gale 1	62	32.2	43.0	None
Glen 1	20	9.9	17.5	None
Hermitage 1	20	30.6	41.2	none
Limekiln 1	20	8.4	15.8	Negative
McAuley 1	24	22.8	32.3	Positive
McCutcheon 1	49	25.5	35.4	None
McIntosh 1	24	18.8	27.7	slight positive
Patterson 1	20	48.1	61.2	None
Powell 1	50	13.7	21.9	None
Price 1	32	23.3	32.9	slight positive
St-Benoit 1	22	38.0	49.7	Negative
Taylor 1,2	34	11.7	19.6	None
Tomkin 1	47	31.9	42.7	None
Vale 1	20	20.1	29.1	None
West 1	45	14.0	22.22	None

An additional 12 and 10 samples were collected on the Cherry and Castle rivers respectively by the MRC which were processed at the MDDEFP laboratory. These data don't appreciably change the mean phosphorus values or suggest any relationship with flow different than with data collected during the regular MRC sampling program. To simplify analysis and make data most comparable between tributaries where this data was not available this data was not used in the loading estimates.

City of Magog Castle river monitoring study

Another study was made by the City of Magog of the flows along with continuous turbidity measurements for the Castle River from May to September of 2013. In addition to this a number of phosphorus samples were taken to correlate turbidity levels with phosphorus concentrations in the Castle River. Unfortunately, none of these samples was taken at high turbidity values which ranged up to over 500 NTU and so using this

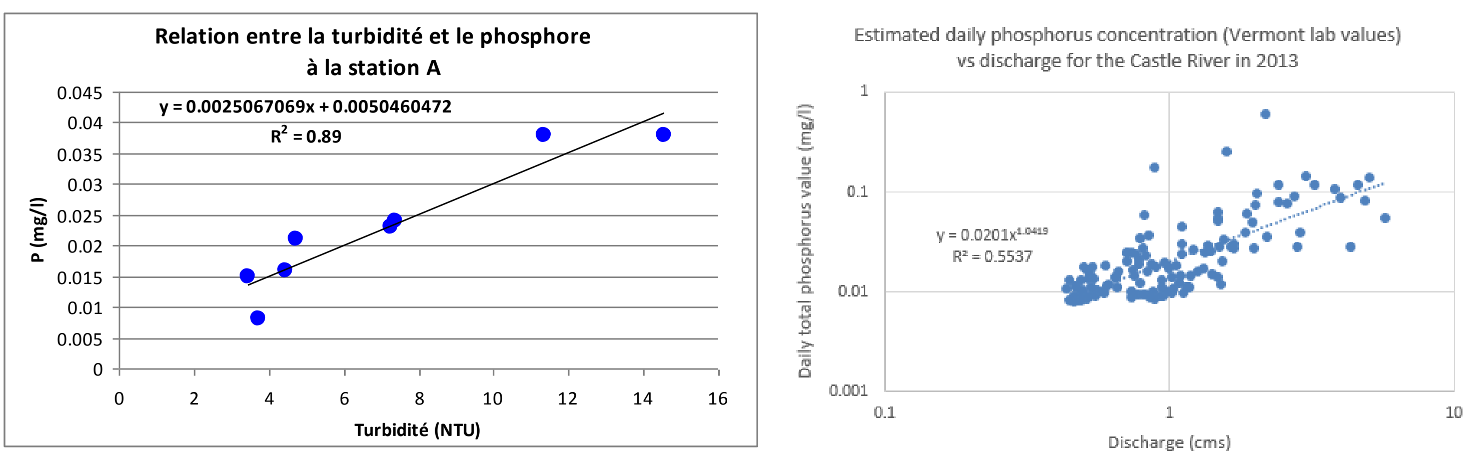

Figure B2. Relationship between turbidity and phosphorus and phosphorus vs discharge from 2013 study of the Castle River showing correlation between discharge and phosphorus values not seen in MRC sample data.

relationship to estimate phosphorus levels raises some concerns. However estimating average concentration on the castle through this relationship and the relationship between flow and turbidity results in an estimated average phosphorus concentration (in Vermont units of 0.055 mg/l or 72% higher than the estimated average concentration through the MRC water sampling of 0.0319 mg/l.

There are a few explanations between the difference in flow between the MRC loading estimate and the loading estimate made using the continuous turbidity and discharge measurements. One of these is the accuracy of the Hydrotel flow data on a daily basis. Unfortunately, the hydrotel flow estimates do not continue through 2013 so these can't be compared to the flow measurements on the Castle River. However, the reason that seems most likely is the typical timing of sampling complete by the MRC and the extreme hysteresis that is apparent in the turbidity and flow data from the Castle River. Looking at a few individual storms suggests that turbidity values rise very quickly after rainfall and drop down before the peak flows have even been reached as shown in figure XX. The MRC sampling targeting wet events generally waits until rainfall significant initial rainfall has occurred before sampling so it is likely that the MRC sampling program routinely misses this initial flush of elevated sediment and phosphorus runoff. Due to the uncertainty in the phosphorus turbidity relationship and single season for sampling this estimate of loading through the 2013 castle river study has not been used as an estimate of average loading directly, however it suggest strongly that the methodology applied above to estimate loading using the MRC dataset likely underestimated loading by a significant amount. Further analysis with the watershed export model suggests that that loading estimates using the MRC water sampling data may still accurately represent relative phosphorus loading between tributaries even if these loading estimates are biased low generally.

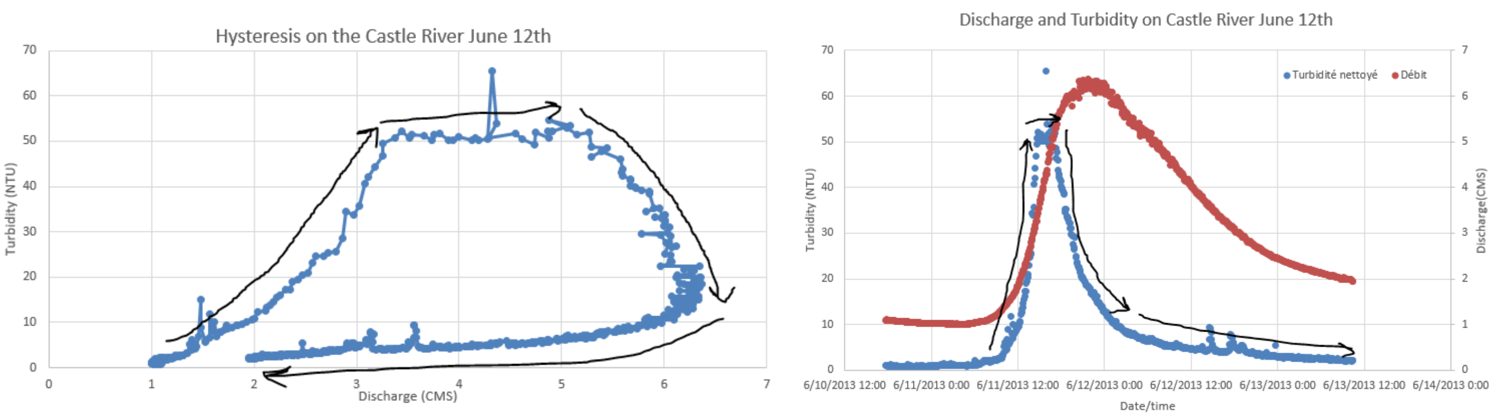

Figure B3 Graphs showing Turbidity vs discharge and Turbidity and discharge over time for a June 11th rainfall event that demonstrate severe hysteresis on the Castle River. Arrows in the two graphs generally correspond.

Appendix C. BMP phosphorus reduction efficiencies used in the Lake Memphremagog Scenario tool (taken from Lake Champlain scenario tool)

lclu code	land use	BMP Type	TP Efficiency
91	Cultivated A soils	Change in crop rotation	25.0%
91	Cultivated A soils	Change in crop rotation - Conservation tillage	36.3%
91	Cultivated A soils	Change in crop rotation - Grassed Waterways	55.0%
91	Cultivated A soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer	78.0%
91	Cultivated A soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer - Riparian Buffer	87.0%
91	Cultivated A soils	Change in Crop Rotation - Grassed Waterways - Riparian Buffer	73.5%
91	Cultivated A soils	Change in crop rotation - Reduced P manure	35.2%
91	Cultivated A soils	Change in crop rotation - Reduced P manure - Riparian buffer	61.8%
91	Cultivated A soils	Change in crop rotation - Riparian buffer	55.8%
91	Cultivated A soils	Conservation tillage	15.0%
91	Cultivated A soils	Conservation tillage - Manure injection	15.0%
91	Cultivated A soils	Cover crop	28.3%
91	Cultivated A soils	Cover crop - Change in crop rotation	46.3%
91	Cultivated A soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer	82.1%
91	Cultivated A soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer - Riparian Buffer	89.4%
91	Cultivated A soils	Cover crop - Conservation tillage - Manure injection	39.1%
91	Cultivated A soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways	63.5%
91	Cultivated A soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways - Riparian Buffer	78.4%
91	Cultivated A soils	Cover crop - Conservation tillage - Manure injection - Riparian Buffer	64.1%
91	Cultivated A soils	Cover crop - Manure injection	28.3%
91	Cultivated A soils	Crop to Hay	80.0%
91	Cultivated A soils	Ditch buffer	51.0%
91	Cultivated A soils	Grassed Waterways	40.0%
91	Cultivated A soils	Grassed Waterways - Riparian Buffer	64.6%
91	Cultivated A soils	Manure injection - Reduced P manure	13.6%
91	Cultivated A soils	Reduced P manure	13.6%
91	Cultivated A soils	Reduced P manure - Grassed Waterways	48.2%
91	Cultivated A soils	Riparian buffer	41.0%
92	Cultivated B soils	Change in crop rotation	25.0%
92	Cultivated B soils	Change in crop rotation - Conservation tillage	40.0%
92	Cultivated B soils	Change in crop rotation - Grassed Waterways	55.0%
92	Cultivated B soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer	78.0%
92	Cultivated B soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer - Riparian Buffer	87.0%
92	Cultivated B soils	Change in Crop Rotation - Grassed Waterways - Riparian Buffer	73.5%
92	Cultivated B soils	Change in crop rotation - Reduced P manure	31.6%
92	Cultivated B soils	Change in crop rotation - Reduced P manure - Riparian buffer	59.7%
92	Cultivated B soils	Change in crop rotation - Riparian buffer	55.8%
92	Cultivated B soils	Conservation tillage	20.0%
92	Cultivated B soils	Conservation tillage - Manure injection	20.0%
92	Cultivated B soils	Cover crop	28.3%
92	Cultivated B soils	Cover crop - Change in crop rotation	46.3%
92	Cultivated B soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer	83.1%
92	Cultivated B soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer - Riparian Buffer	90.1%
92	Cultivated B soils	Cover crop - Conservation tillage - Manure injection	42.7%
92	Cultivated B soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways	65.6%
92	Cultivated B soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways - Riparian Buffer	79.7%
92	Cultivated B soils	Cover crop - Conservation tillage - Manure injection - Riparian Buffer	66.2%
92	Cultivated B soils	Cover crop - Manure injection	28.3%
92	Cultivated B soils	Crop to Hay	80.0%

92	Cultivated B soils	Ditch buffer	51.0%
92	Cultivated B soils	Grassed Waterways	40.0%
92	Cultivated B soils	Grassed Waterways - Riparian Buffer	64.6%
92	Cultivated B soils	Manure injection - Reduced P manure	8.8%
92	Cultivated B soils	Reduced P manure	8.8%
92	Cultivated B soils	Reduced P manure - Grassed Waterways	45.3%
92	Cultivated B soils	Riparian buffer	41.0%
93	Cultivated C soils	Change in crop rotation	25.0%
93	Cultivated C soils	Change in crop rotation - Conservation tillage	43.8%
93	Cultivated C soils	Change in crop rotation - Grassed Waterways	47.5%
93	Cultivated C soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer	74.3%
93	Cultivated C soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer - Riparian Buffer	84.8%
93	Cultivated C soils	Change in Crop Rotation - Grassed Waterways - Riparian Buffer	69.0%
93	Cultivated C soils	Change in crop rotation - Reduced P manure	31.4%
93	Cultivated C soils	Change in crop rotation - Reduced P manure - Riparian buffer	59.5%
93	Cultivated C soils	Change in crop rotation - Riparian buffer	55.8%
93	Cultivated C soils	Conservation tillage	25.0%
93	Cultivated C soils	Conservation tillage - Manure injection	25.0%
93	Cultivated C soils	Cover crop	28.3%
93	Cultivated C soils	Cover crop - Change in crop rotation	46.3%
93	Cultivated C soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer	81.6%
93	Cultivated C soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer - Riparian Buffer	89.1%
93	Cultivated C soils	Cover crop - Conservation tillage - Manure injection	46.3%
93	Cultivated C soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways	62.4%
93	Cultivated C soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways - Riparian Buffer	77.8%
93	Cultivated C soils	Cover crop - Conservation tillage - Manure injection - Riparian Buffer	68.3%
93	Cultivated C soils	Cover crop - Manure injection	28.3%
93	Cultivated C soils	Crop to Hay	80.0%
93	Cultivated C soils	Ditch buffer	51.0%
93	Cultivated C soils	Grassed Waterways	30.0%
93	Cultivated C soils	Grassed Waterways - Riparian Buffer	58.7%
93	Cultivated C soils	Manure injection - Reduced P manure	8.6%
93	Cultivated C soils	Reduced P manure	8.6%
93	Cultivated C soils	Reduced P manure - Grassed Waterways	36.0%
93	Cultivated C soils	Riparian buffer	41.0%
94	Cultivated D soils	Change in crop rotation	25.0%
94	Cultivated D soils	Change in crop rotation - Conservation tillage	62.5%
94	Cultivated D soils	Change in crop rotation - Grassed Waterways	43.8%
94	Cultivated D soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer	72.4%
94	Cultivated D soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer - Riparian Buffer	83.7%
94	Cultivated D soils	Change in Crop Rotation - Grassed Waterways - Riparian Buffer	66.8%
94	Cultivated D soils	Change in crop rotation - Reduced P manure	28.7%
94	Cultivated D soils	Change in crop rotation - Reduced P manure - Riparian buffer	57.9%
94	Cultivated D soils	Change in crop rotation - Riparian buffer	55.8%
94	Cultivated D soils	Conservation tillage	50.0%
94	Cultivated D soils	Cover crop	28.3%
94	Cultivated D soils	Cover crop - Change in crop rotation	46.3%
94	Cultivated D soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer	86.8%
94	Cultivated D soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer - Riparian Buffer	92.2%
94	Cultivated D soils	Cover crop - Conservation tillage - Manure injection	64.2%
94	Cultivated D soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways	73.1%
94	Cultivated D soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways - Riparian Buffer	84.1%
94	Cultivated D soils	Cover crop - Conservation tillage - Manure injection - Riparian Buffer	78.9%
94	Cultivated D soils	Cover crop - Manure injection	28.3%
94	Cultivated D soils	Crop to Hay	76.7%

94	Cultivated D soils	Ditch buffer	51.0%
94	Cultivated D soils	Grassed Waterways	25.0%
94	Cultivated D soils	Grassed Waterways - Riparian Buffer	55.8%
94	Cultivated D soils	Manure injection - Reduced P manure	4.9%
94	Cultivated D soils	Reduced P manure	4.9%
94	Cultivated D soils	Reduced P manure - Grassed Waterways	28.7%
94	Cultivated D soils	Riparian buffer	41.0%
95	Cultivated unknown soils	Change in crop rotation	25.0%
95	Cultivated unknown soils	Change in crop rotation - Conservation tillage	43.8%
95	Cultivated unknown soils	Change in crop rotation - Grassed Waterways	47.5%
95	Cultivated unknown soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer	74.3%
95	Cultivated unknown soils	Change in Crop Rotation - Grassed Waterways - Ditch Buffer - Riparian Buffer	84.8%
95	Cultivated unknown soils	Change in Crop Rotation - Grassed Waterways - Riparian Buffer	69.0%
95	Cultivated unknown soils	Change in crop rotation - Reduced P manure	31.4%
95	Cultivated unknown soils	Change in crop rotation - Reduced P manure - Riparian buffer	59.5%
95	Cultivated unknown soils	Change in crop rotation - Riparian buffer	55.8%
95	Cultivated unknown soils	Conservation tillage	25.0%
95	Cultivated unknown soils	Conservation tillage - Manure injection	25.0%
95	Cultivated unknown soils	Cover crop	28.3%
95	Cultivated unknown soils	Cover crop - Change in crop rotation	46.3%
95	Cultivated unknown soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer	81.6%
95	Cultivated unknown soils	Cover crop - Conservation tillage - Grassed Waterways - Ditch Buffer - Riparian Buffer	89.1%
95	Cultivated unknown soils	Cover crop - Conservation tillage - Manure injection	46.3%
95	Cultivated unknown soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways	62.4%
95	Cultivated unknown soils	Cover crop - Conservation tillage - Manure injection - Grassed Waterways - Riparian Buffer	77.8%
95	Cultivated unknown soils	Cover crop - Conservation tillage - Manure injection - Riparian Buffer	68.3%
95	Cultivated unknown soils	Cover crop - Manure injection	28.3%
95	Cultivated unknown soils	Crop to Hay	80.0%
95	Cultivated unknown soils	Ditch buffer	51.0%
95	Cultivated unknown soils	Grassed Waterways	30.0%
95	Cultivated unknown soils	Grassed Waterways - Riparian Buffer	58.7%
95	Cultivated unknown soils	Manure injection - Reduced P manure	8.6%
95	Cultivated unknown soils	Reduced P manure	8.6%
95	Cultivated unknown soils	Reduced P manure - Grassed Waterways	36.0%
95	Cultivated unknown soils	Riparian buffer	41.0%
24	Developed Impervious	Biofiltration with Underdrains .25"	38.0%
24	Developed Impervious	Biofiltration with Underdrains .5"	59.0%
24	Developed Impervious	Biofiltration with Underdrains .9"	74.0%
24	Developed Impervious	Biofiltration with Underdrains 1.5 "	84.0%
24	Developed Impervious	Biofiltration with Underdrains 2"	89.0%
24	Developed Impervious	Extended Dry Detention Pond .9"	19.0%
24	Developed Impervious	Gravel Wetland .25"	30.0%
24	Developed Impervious	Gravel Wetland .5"	46.0%
24	Developed Impervious	Gravel Wetland .9"	59.0%
24	Developed Impervious	Gravel Wetland 1.5"	65.0%
24	Developed Impervious	Gravel Wetland 2"	66.0%
24	Developed Impervious	Impervious Area Removal	89.0%
24	Developed Impervious	Impervious Area Removal	89.0%
24	Developed Impervious	Impervious Area Removal	89.0%
24	Developed Impervious	Infiltration Trench .25"	51.0%
24	Developed Impervious	Infiltration Trench 1.5"	98.0%
24	Developed Impervious	Infiltration Trench 2"	99.0%
24	Developed Impervious	Infiltration Trench.5"	77.0%
24	Developed Impervious	Infiltration Trench.9"	93.0%
24	Developed Impervious	Open Channel/Dry Swale .9"	34.0%

24	Developed Impervious	Sand Filter .5"	42.0%
24	Developed Impervious	Sand Filter .9"	50.0%
24	Developed Impervious	Sand Filter 1.5"	56.0%
24	Developed Impervious	Sand Filter 2"	65.0%
24	Developed Impervious	Surface Infiltration Practices .25"	54.0%
24	Developed Impervious	Surface Infiltration Practices .5"	77.0%
24	Developed Impervious	Surface Infiltration Practices .9"	92.0%
24	Developed Impervious	Surface Infiltration Practices 1.5"	98.0%
24	Developed Impervious	Surface Infiltration Practices 2 "	99.0%
24	Developed Impervious	Wet Pond & Constructed Wetlands .5"	42.0%
24	Developed Impervious	Wet Pond & Constructed Wetlands .9"	50.0%
24	Developed Impervious	Wet Pond & Constructed Wetlands 1.5"	56.0%
24	Developed Impervious	Wet Pond & Constructed Wetlands 2"	65.0%
23	Developed Pervious	Ban on P Fertilizer Use on Turf	50.0%
23	Developed Pervious	Ban on P Fertilizer Use on Turf	50.0%
23	Developed Pervious	Ban on P Fertilizer Use on Turf	50.0%
23	Developed Pervious	Ban on P Fertilizer Use on Turf	50.0%
61	Dirt road	Roadside Erosion Control	50.0%
64	Dirt Road high Erosion	Roadside Erosion Control	50.0%
62	Dirt Road Low Erosion	Roadside Erosion Control	50.0%
63	Dirt Road mod Erosion	Roadside Erosion Control	50.0%
79	Farmstead	Barnyard Management	80.0%
42	Forest	Stream Crossing Erosion/Sedimentation Control	5.0%
82	Hay	Ditch buffer	51.0%
82	Hay	Grassed Waterways	51.6%
82	Hay	Grassed Waterways - Riparian Buffer	71.4%
82	Hay	Manure injection - Reduced P manure	2.1%
82	Hay	Reduced P manure	2.1%
82	Hay	Reduced P manure - Grassed Waterways	52.5%
82	Hay	Riparian buffer	41.0%
83	Pasture	Fencing/livestock exclusion with out riparian buffer	55.0%
83	Pasture	Fencing/livestock exclusion with riparian buffer	73.5%
60	Road Paved	Biofiltration with Underdrains .25"	38.0%
60	Road Paved	Biofiltration with Underdrains .5"	59.0%
60	Road Paved	Biofiltration with Underdrains .9"	74.0%
60	Road Paved	Biofiltration with Underdrains 1.5 "	84.0%
60	Road Paved	Biofiltration with Underdrains 2"	89.0%
60	Road Paved	Catch Basin Cleaning	2.0%
60	Road Paved	Extended Dry Detention Pond .9"	19.0%
60	Road Paved	Gravel Wetland .25"	30.0%
60	Road Paved	Gravel Wetland .5"	46.0%
60	Road Paved	Gravel Wetland .9"	59.0%
60	Road Paved	Gravel Wetland 1.5"	65.0%
60	Road Paved	Gravel Wetland 2"	66.0%
60	Road Paved	Infiltration Trench .25"	51.0%
60	Road Paved	Infiltration Trench 1.5"	98.0%
60	Road Paved	Infiltration Trench 2"	99.0%
60	Road Paved	Infiltration Trench.5"	77.0%
60	Road Paved	Infiltration Trench.9"	93.0%
60	Road Paved	Leaf Litter Collection	5.0%
60	Road Paved	Mechanical Broom Sweeper (2/year)	1.0%
60	Road Paved	Mechanical Broom Sweeper (monthly)	3.0%
60	Road Paved	Regenerative Air-Vacuum (monthly)	8.0%
60	Road Paved	Surface Infiltration Practices .25"	54.0%
60	Road Paved	Surface Infiltration Practices .5"	77.0%

60	Road Paved	Surface Infiltration Practices .9"	92.0%
60	Road Paved	Surface Infiltration Practices 1.5"	98.0%
60	Road Paved	Surface Infiltration Practices 2 "	99.0%
60	Road Paved	Wet Pond & Constructed Wetlands .5"	42.0%
60	Road Paved	Wet Pond & Constructed Wetlands .9"	50.0%
60	Road Paved	Wet Pond & Constructed Wetlands 1.5"	56.0%
60	Road Paved	Wet Pond & Constructed Wetlands 2"	65.0%

Appendix D Lake Phosphorus data from MDDEFP transformed to VT lab values (ug/l)

	Vt Lake	Qc Lake	Fitch Bay	Outer Fitch	Center Lake	North Lake	Magog bay
Date	03020249	03020094	03020093	03020092	03020091	03020246	03020090
5/13/2009					11.34	15.12	10.30
5/26/2009		11.76			11.97	13.76	11.66
6/15/2009	13.13	15.01	17.01	13.13	11.34	13.34	12.18
7/8/2009	36.09	16.90	32.95	23.51	16.59	22.46	27.70
7/29/2009	14.70	13.76	27.70	15.12			
8/4/2009					19.31	17.22	15.12
8/16/2009	14.28	13.13	20.36	13.65	12.78	11.34	12.08
9/7/2009		15.33			15.54	14.77	13.23
9/30/2009		16.69			13.23	12.39	14.70
10/12/2009		20.36			15.22	15.22	16.38
11/1/2009		18.58			12.71	13.97	22.29
5/4/2010		13.76			14.18	12.18	12.39
5/27/2010		12.43			11.13	12.39	11.97
6/17/2010	14.81	15.12	24.87	17.85	14.07	14.49	12.18
7/11/2010	15.64	18.16	16.27	11.94	10.82	11.03	13.13
7/28/2010	12.60	11.55	23.09				
7/29/2010				11.55	12.60	15.54	10.51
8/17/2010	15.43	20.47	25.92	21.73	23.30	13.23	15.22
9/12/2010	18.69	22.25	32.74	14.60	11.76	11.87	13.23
9/26/2010		19.63			14.18	13.34	11.24
10/17/2010		21.41			14.60	15.22	14.98
11/7/2010		17.11			11.34	13.23	
5/8/2011		23.86			15.33	12.60	11.45
5/25/2011		13.55			13.72	12.29	14.81
6/6/2011							
6/15/2011	23.40	22.98	25.43	14.07	19.84	17.11	15.33
7/5/2011	13.44	14.81	22.67	16.59	16.38	17.22	11.45
7/27/2011	15.43	15.75	31.37	16.03	18.37	14.18	14.07
8/17/2011	11.13	10.30	24.14	9.67	9.98	10.61	9.35
9/6/2011		14.91			14.81		
9/25/2011		19.42			10.09	11.87	16.06
10/10/2011		15.01			11.66	10.92	10.72
10/31/2011		14.81			11.48	13.13	13.34
6/21/2012	12.39	12.32	13.13	11.45	12.81	11.45	10.72
7/11/2012	11.13	11.34	14.39	10.61	12.81	9.56	9.98
7/31/2012	29.49	13.55	16.48	11.34	10.09	10.09	10.19
8/23/2012	12.29	11.66	23.51	11.31	11.34	9.98	9.56
11/19/2012	8.72	9.25	15.22	11.76	15.96	33.05	50.36
Average	16.27	15.79	22.63	14.22	13.79	14.01	14.48
Average + .305	16.57	16.10	22.93	14.52	14.10	14.31	14.79

Appendix E. Source GIS Layers and Excel files used in modeling

All files are in directory Y:\WSMD_MAPP\Assessment\TMDL\Memph TMDL\Memphremagog TMDL modeling

Data source	subfolder	Name	Date modified
Land use GIS shapefile	\Shapefiles	1020515FINAL.shp	10/20/15
Sub Basin boundaries GIS shapefile	\Shapefiles	subbasin5.shp	9/24/15
Streambank erosion Shapefile	\Shapefiles	bar_erosion.shp	9/18/15
Quebec Vermont Split data	\Excel	MDDELCCLaRosa splits.xlsx	3/9/2016
Lake Cross Sections	\Excel	lake cross sections.xlsx	5/18/2015
WWTF loading spreadsheet	\Excel	WWTF.xlsx	6/8/2015
DEC Lake Phosphorus	\Excel	DEC lake TP.xlsx	4/19/16
MDDELCC Phosphorus	\Excel	MDDEFPTP.xlsx	3/4/16
Flux input (Major tribs) flow	\Excel\Flux	fluxFlow.xlsx	2/16/15
Flux input (Major tribs) WQ	\Excel\Flux	FluxTP.xlsx	3/9/16
Flux input (Minor Tribs) flow	\Excel\Flux	flux flowMWA	11/23/15
Flux input (Minor Tribs) WQ	\Excel\Flux	wq dataMWAfinal.xls	1/8/16
Flux Output (Major Tribs)	\Excel\Flux	Major Trib Flux output2014.xlsx	2/6/15
Flux Output (Minor Trib)	\Excel\Flux	10-2-15 minor Trib Fluxoutput.xlsx	11/23/15
Simplified Bath Tub excel model	\Excel	Chloride bTub model calibration.xlsx	5/27/16
Daily chloride model	\Excel	daily bathtub4-2015Cl-.xlsx	4/6/2016
Daily phosphorus model	\Excel	bathtubP daily4-21-16FirstOrd.xlsx	5/3/2006
Combined Land use phosphorus export model, calibration	\Excel	memphModel.xlsx	6/9/2016