Beneficial Use Impairment
#14 Assessment Report
Loss of Fish and Wildlife Habitat

September 6, 2016

APPENDICES

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APPENDIX F: Degradation of Submergent Aquatic Vegetation Delisting Criteria Status Report

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Authors

- Susan Doka, Ph.D., Research Scientist, Fisheries and Oceans Canada
- Erin Gertzen, Aquatic Science Biologist, Fisheries and Oceans Canada
- Kathy Leisti, Freshwater Research Biologist, Fisheries and Oceans Canada
- Jody MacEachern, GIS Analyst, Fisheries and Oceans Canada
- Jon Midwood, Ph.D., Aquatic Science Biologist, Fisheries and Oceans Canada
- Rex Tang, Aquatic Science Biologist, Fisheries and Oceans Canada

Recommendation: Delisting criteria have been met

DELISTING TARGETS	DELISTING CRITERIA	STATUS
The extent (i.e. surface area in hectares) and community structure of submergent aquatic vegetation (SAV) in the Bay of Quinte (BQ) is restored to levels more reflective of the mesotrophic system that existed before excessive nutrient loading created a eutrophic/hyper-eutrophic system.	FWH-1A: Show that an increase in the area of SAV has occurred in the Post-Dreissena time stanza (1995-2012; Post-D) such that 30% area of the upper BQ has SAV coverage of more than 50% density. This is approximately a 20% increase in the SAV area of the upper bay when 1978 to 1994 is used as a "base-year period" (i.e. standard Post-P but pre-Dreissena time stanza used in BQ comparisons).	Achieved
	FWH-1B: Measure statistically-significant positive trends or changes in both of these plant community health measures (a & b):	
	a) A positive slope in total species diversity for the upper bay based on standardized SAV surveys between 1972 and 2011. OR A significant positive difference (one-tailed t-test, alpha = 0.05) in total plant species diversity between Post-P and Post-D time stanzas.	Achieved
	b) A positive slope or statistically-significant improvement in a plant community health index. OR Index values in the 'good to excellent' range for three recent annual SAV surveys.	Achieved

Old and new measures for assessing habitat targets were developed under the Fish and Wildlife Task Group that developed or reaffirmed targets for Fish and Wildlife Habitat and Population Beneficial Use Impairments. The measures here specifically address Fish and Wildlife Habitat criteria FWH-1A and FWH-1B. The data used was collected from the north and south reference transects at Trenton, Belleville and Big Bay stations in the Bay of Quinte.

The index selected was the wetland macrophyte index (WMI) developed by Croft & Chow-Fraser in 2007. The SAV Index of Biotic Integrity (IBI) developed by Grabas et al. (2012) could not be applied because the Fisheries and Oceans Canada (DFO) dataset does not contain all the required information to calculate certain metrics within the SAV IBI. Data collected by DFO were not initially intended to be used to calculate ecological indices; therefore this is a post-hoc application of the data.

Background

Submerged aquatic vegetation (SAV) is an important component of aquatic ecosystems. SAV affects the physical and chemical nature of their local environment, such as anchoring the substrate to reduce sediment transport, reducing turbidity caused by the resuspension of sediments by winds, and affecting nutrient and oxygen cycling (Carpenter and Lodge, 1986). SAV provides spawning, nursery, and adult habitat for numerous fish species; it is both a refuge from predation by providing cover and serves as feeding habitat for fishes because macroinvertebrates are on average more prevalent in SAV beds (Gilinsky, 1984). SAV presence has been found to influence fish abundance (Chick and McIvor, 1994), growth rates (Werner et al., 1983) and fish community structure (Weaver et al., 1997). Randall et al. (1996) estimated that sites with SAV had higher densities of fish and higher species richness than unvegetated sites in three bays in southern Ontario.

Numerous studies have reported declines in SAV occurrence, abundance, richness and species composition due to eutrophication (Hough et al., 1989; Lachavanne et al., 1992; Egertson et al., 2004; Sand-Jensen et al., 2008; Sondergaard et al., 2010). Increases in nutrient levels promote substantially higher algal densities, which eventually result in the shading and decline of SAV. Other studies have detailed the recovery of SAV following the invasion by *Dreissena* spp. and the subsequent improvement in water clarity that facilitates plant growth (Skubinna et al., 1995; Zhu et al., 2006).

Sediment cores from the lower bay detail the historic trophic conditions of the Bay of Quinte. Slight to moderate oligotrophic conditions were recorded prior to the late 1800s with a shift to a eutrophic state by 1890 (Sly, 1986). Water quality noticeably deteriorated after 1930 and by the 1950s the Bay was considered to be hypereutrophic. In the 1960s algal blooms occurred in the upper and middle bays and Secchi disc depths of less than 0.3 m were measured (Johnson and Hurley, 1986). SAV beds were described as lush in the 1950s with subsequent declines during the 1960s to half of their estimated extent (Crowder and Bristow, 1986). In the 1970s macrophytes were restricted to a small euphotic zone (i.e. water zone where light can penetrate to the bottom) of 2 m in the upper bay and 4 m in the lower bay; the euphotic zone is affected by water clarity. The Bay's significant commercial and sport fisheries declined in the early 1970s, which may have partly resulted from SAV degradation.

Over the years, various non-native species have affected the bay. Common Carp (*Cyprinus carpio*) can serious reduce macrophytes coverage and were present in the Bay in the early 1900s but did not become abundant until the 1950s (Johnson and Hurley, 1986). In the mid-1990s, zebra and quagga mussels (*Dreissena* spp.) invaded the area, dramatically changing the water clarity because of their filter-feeding capacity. More recently, round goby (*Neogobius melanostomus*) have invaded the fish community and have also changed the system and its potential status and health but are unlikely to have as much effect on vegetation as destructive fishes and invasive plants such as Eurasian milfoil (*Myriophyllum spicatum*), Common reed (*Phragmites australis*), and Curly pondweed (*Potamogeton crispus*).

History of SAV in the Beneficial Use Impairment 'Loss of Fish Habitat'

This report deals with targets and measures for SAV. Here, SAV is evaluated as a subset of targets identified for the Bay of Quinte as an Area of Concern (AOC) under the Beneficial Use Impairment (BUI): 'Loss of Fish and Wildlife Habitat'. There are several BUI's that affect fish habitat either directly or indirectly. This report specifically addresses degradation of SAV identified by a joint Fish and Wildlife Committee that revised and appended quantitative targets. Until 2011, the same target and measure for SAV was found under both the Eutrophication and Undesirable Algae BUI and the Loss of Fish and Wildlife Habitat BUI. The listing guideline issued by the International Joint Commission (IJC) in 1991 for eutrophication was:

> "When there are persistent water quality problems (e.g. dissolved oxygen depletion of bottom waters, nuisance algal blooms or accumulation, and decreased water clarity, etc.) attributed to cultural eutrophication."

The fish habitat guideline was worded as such:

> "When fish and wildlife management goals have not been met as a result of the loss of fish and wildlife habitat due to a perturbation in the physical, chemical, or biological integrity of the Boundary Waters, including wetlands."

The Stage 1 report for Bay of Quinte Area of Concern (1990) detailed the environmental setting and problem definition for impaired SAV. It was believed that the return of the once lush and extensive SAV beds in the upper bay was critical to delisting. The report stated:

> "Key to the Bay of Quinte's recovery is a return of macrophytes. They will assist in reducing the feedback of phosphorus from the sediments, and provide habitat for the algal grazers and white perch predators (e.g. northern pike)"
> *- Bay of Quinte RAP Coordinating Committee, 1990.*

Delisting guidelines issued by the International Joint commission proposed to delist AOCs:

> "When there are no persistent water quality problems (e.g. dissolved oxygen depletion of bottom waters, nuisance algal blooms or accumulation, and decreased water clarity, etc.) attributed to cultural eutrophication".

The delisting guideline for this impaired beneficial use was stated loosely as:

> "When the amount and quality of physical, chemical, and biological habitat required to meet fish and wildlife management goals have been achieved and protected".

Generic statements such as these can be difficult to quantify, especially across individual fish species within a community that may have different habitat requirements and be affected differently by habitat changes. Natural variability in habitat supply also needs to be considered and we attempt to do that as part of our evaluation by modelling annual estimates of SAV supply to compare time stanza averages.

The first mention of an areal target for SAV in the upper bay appears in the 1988 RAP Technical Report 6 written for the Bay of Quinte RAP Coordinating Committee by Environmental and Social Systems Analysts Limited (ESSA). It describes the collaborative effort involved in conceptualizing and constructing a simple three-layer phosphorus (P) model. The report also details the P-model results from various P-control remediation options in terms of changes in the percent of the upper bay potentially covered by SAV. The report cites relationships developed by Nicholls and Hurley (1989) that were used to convert predicted P levels to algal biomass and light extinction. By using the Bay of Quinte morphometry, the area where 1% light levels reached the bottom was then predicted to contain SAV (Minns, 1992). The ESSA report stated that if the total P average concentration could be reduced from 47 µg/L to 30 µg/L through remediation efforts, it was estimated that SAV area would expand from 38 km^2 to 45 km^2, which represents 30% coverage of the upper bay. This restoration objective was proposed in the 1989 RAP discussion paper “Time to Decide”.

The 1990 RAP Stage 1 report noted that although there was a substantial decrease in P levels in the upper bay following P-control in 1978, algal abundance had remained high and SAV had not recovered as expected. SAV surveys conducted between 1972 and 1988 (Crowder and Bristow, 1986), indicated that SAV percent cover on Bay of Quinte transects was sparse, at 22% while Lake Ontario and nearby lakes had cover values of 75 and 100%, respectively.

The first mention of a density component of the SAV target appears in the 1993 RAP Stage 2 report, but we were unable to find any supporting documents describing why the value of 50% density was chosen. Recent discussions with Ken Minns (pers. comm., 2015) indicate that researchers felt that expansion of beds with only sparse density as found in the earlier surveys would not provide as much benefit to the fish community as higher density beds (Crowder and Cooper, 1982; Wiley et al., 1984; Bryan and Scarnecchia, 1992; Randall et al., 1996). For additional detail on fish response to varying SAV density, see the literature review in Appendix A.

In 2005, FWH-2 was created to assess SAV quality in Bay of Quinte coastal wetlands. Initially, the Durham Region Coastal Wetland Monitoring Project (DRCWMP) Index of Biotic Integrity (IBI) was used to assess SAV quality in select emergent wetland areas (Environment Canada 2004). Subsequently, this IBI was updated (Grabas et al., 2012) and by September 2012, the updated version of the IBI was used to measure the delisting criteria. It should be noted that while this IBI contains metrics of SAV density, it does not directly measure areal extents of SAV beds so the chosen SAV criteria (i.e. FWH-1A and 1B) complement each other.

In 2011, measure FWH-1B, was revised to address SAV quality using another index that can be applied to DFO transect data. FWH-1B complements the criteria contained in FWH-1A and FWH-2 and includes other areas and measures such as species composition and diversity as measures of wetland health not found in the previous extent target or other index measures.

Methods of Assessment

All the measures considered in FWH-1 will be assessed only using data from the upper bay; the middle and lower bay data are not included in the assessment as the target originally referred to the upper bay only. The rationale is that the upper bay experienced the greatest degree of SAV degradation from eutrophication and, aside from Hay Bay proper, has a much greater extent of

SAV naturally than the middle and lower bays. Lake bottom elevations and slope (i.e. bathymetry) restrict SAV to fringing beds in much of the middle and lower bays (aside from Hay Bay). Other factors like wind and wave exposure, sedimentation, flows, and substrate type can also play a role in determining SAV presence and extent but the most important factor affecting SAV presence in sheltered areas is the depth of light penetration.

Measure FWH-1A: SAV Density and Extent in the upper Bay of Quinte

Since the original Bay of Quinte SAV model was developed by Seifried (2002), three additional SAV surveys have been conducted in the bay in 2004, 2007 and 2011. All the surveys sampled at the same reference transects at Trenton, Belleville and Big Bay in the upper bay, at the outflow of Hay Bay in the middle bay, and at Conway in the lower bay (Figure 1). Water transparency data were collected from April to October every year on a bi-monthly basis at the Project Quinte offshore monitoring stations (Project Quinte 1972 – 2013; Figure 1). For detailed information on sampling methodologies for the SAV surveys and historical results refer to: Leisti, 2006; Leisti et al., 2006; Leisti and Doka, 2009 and Leisti et al., 2012.

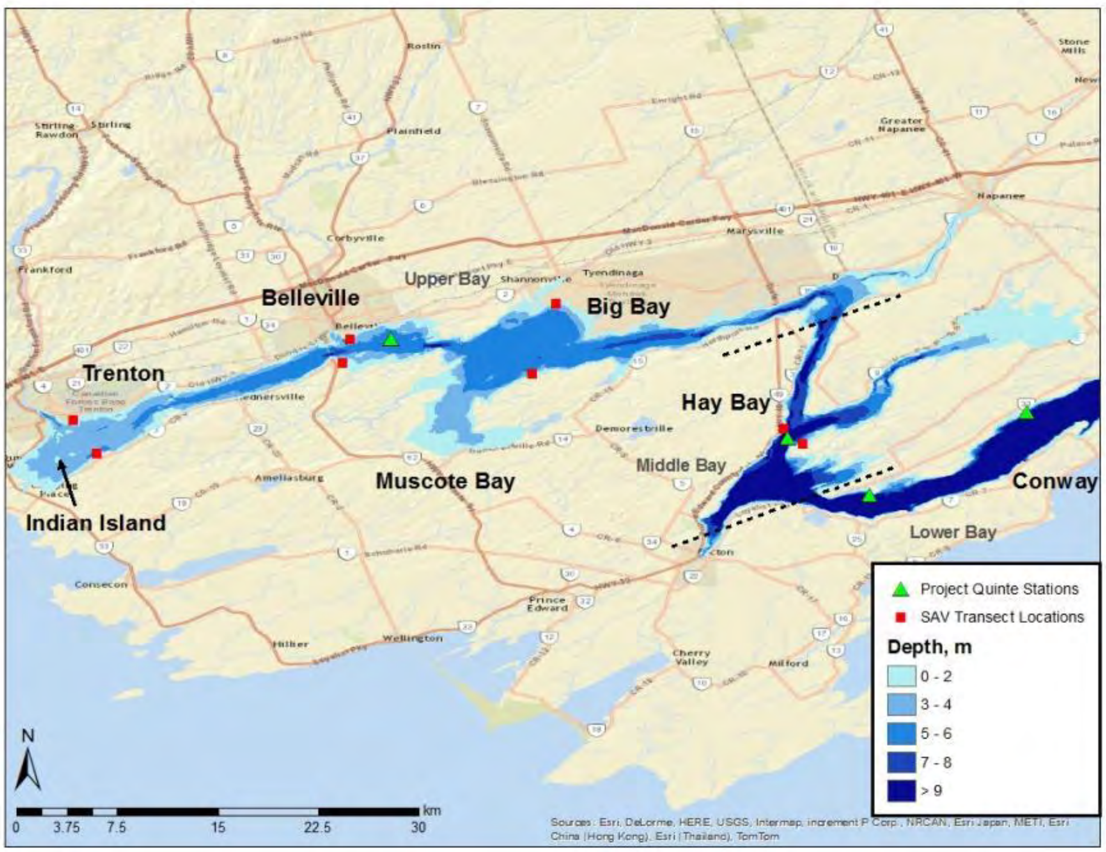

Figure 1. Location of the Fisheries and Oceans Canada SAV reference transects and water quality monitoring stations in the Bay of Quinte. The dotted lines separate upper, middle and lower Bay of Quinte.

A new SAV model for the Bay of Quinte was developed incorporating new survey data from 2007 and 2011 in addition to the original 1972 to 2000 datasets. The new model used a 2-step approach to predicting SAV extent and percent cover; the first step was the development of a logistic model to determine the maximum depth of SAV presence based on relationships with water clarity and depth. These variables were the dominant factors driving SAV presence in the upper bay; although substrate type, fetch and slope were also considered. Model generation and validation datasets were compiled from reference and validation transect data from the upper bay surveys and standardized to the extent possible given different survey techniques over the years (Appendix B). The second step predicted the density of SAV within the limits of the bed extent determined in Step 1. The generalized algorithm used for Step 2 was based on the average percent cover relationship across depths to the maximum depth of colonization over the years in the Bay of Quinte. Additional detailed information on statistical model construction and validation can be found in Appendix B and C.

The two-stage model (Appendix B) was applied in a GIS of the Bay of Quinte. SAV density (i.e. percent cover) was predicted at a 3 x 3m grid resolution for the upper bay for calculations (note: maps are generated at 30x30m resolution for printing purposes). Annual changes in SAV distribution and density were predicted (hind cast) based on average growing season water depth for each grid cell and the seasonal means of water clarity estimated from an offshore station in the upper bay over the years 1972 to 2011. Growing season was defined as the period between April and September of each year. Measure FWH-1 compares the mean predicted SAV density and distribution across the trend period and between the two-time stanzas identified in the target; the period after point source phosphorus controls were implemented (Post-P, 1978 to 1994) and the period after the invasion of Dreissenid mussels (Post-D, 1995 to 2012).

Measure FWH-1Ba: SAV Diversity

SAV diversity metrics examine all of the species that comprise the SAV community and their relative abundance. There are two components to diversity; the total number of species (i.e. richness) and the relative contribution of each species to the total number of individuals within a community (i.e. evenness). Many studies have used the Shannon-Weiner index (SWI) as a measure of diversity (Begon et al., 1996) as it incorporates both richness and evenness into the calculation of diversity. This index is calculated as follows:

$$H = \sum_{i=0}^{S} -(P_i * lnP_i)$$

Where:
H = the Shannon-Weiner diversity index
P_i = fraction of the entire population made up of species i
S = numbers of species encountered

The minimum SWI value is 0, which indicates that the community only contains a single species. H increases with increasing species richness and reaches a maximum when individuals from different species are equally distributed. Several metrics can be used to calculate the SWI including biomass and counts of individuals. Due to the nature of the historical Bay of Quinte SAV dataset, the SWI in this report was calculated using species presence-absence data, following the method of Baattrup-Pedersen et al. (2002).

For the years 1972 to 1982, only summary information regarding species richness and frequency were available from Crowder and Bristow (1986). In that paper, there is a table of species richness for the upper bay north shore transects at Trenton, Belleville and Big Bay, the east-shore transect at Hay Bay in the middle bay and the north shore transect for Conway in the lower bay. Since the raw data were unavailable, it was assumed that species that were somewhat abundant in the 1972-82 surveys (e.g., *Potamogeton pusillus*, *P. richardsonii* and *Ranunculus aquatilis*) were classified as "other" for the plant community health measures calculated in this report.

As with all indices that combine metrics, there can be some difficulty in the interpretation and comparison of SWI values between sites because the index value can vary either through changes in species richness or from changes in their relative frequencies. Therefore additional information in the form of a graph illustrating the changes in the relative frequency of occurrence of each species across years has been provided as a reference. Frequency of occurrence is defined here as the percentage of times that a specific species was found in all of the vegetated sampling units (e.g. quadrats sampled with vegetation) for that year.

Measure FWH-1Bb: Plant Community Index

This delisting criterion uses two plant community indices in the evaluation of SAV time series data: FWH-1Bb used the Wetland Macrophyte Index (WMI) developed by Croft and Chow-Fraser (2007) while FWH-2 used the SAV IBI updated by Grabas et al. (2012). The WMI was used with the DFO dataset because only species presence/absence data are required for its calculation. The metric required for the SAV IBI that was not available from DFO or historic SAV standard sampling protocols was total cumulative coverage in the water column. DFO data is percent bottom cover with only species presence/absence and not individual species cover estimates.

The WMI was specifically developed to detect water quality degradation in Great Lakes coastal marshes through its impact on aquatic vegetation (Croft and Chow-Fraser, 2007). The presence/absence of emergent, submergent and floating species and environmental data were collected across 127 wetlands throughout the Great Lakes between 1998 and 2005 to develop the WMI. Canonical correspondence analysis (CCA) was used to ordinate aquatic vegetation species along an environmental gradient consisting of 11 variables (latitude, longitude, turbidity, conductivity, pH, inorganic suspended solids, organic suspended solids, total ammonia nitrogen, total nitrate-nitrogen, total phosphorus and chlorophyll *a*). Based on the CCA, individual species were assigned both a U-value (their sensitivity to degraded water quality) and a T-value (representing their niche breadth). WMI scores were calculated as follows:

$$WMI\ score = \frac{\sum_{i=0}^{n} Y_i U_i T_i}{\sum_{i=0}^{n} Y_i\, T_i}$$

Where:

- Y_i = 1 if the species is present, 0 if absent
- T_i = niche breadth of species *i*, value of 1 to 3
- U_i = tolerance of species *i* to degradation, value of 1 to 5

To account for the presence of exotic species as a negative metric within the WMI calculation, the final value can be adjusted by subtracting the square root of the proportion of the species that are exotic.

WMI scores can theoretically range from a minimum of 1.0 to a maximum of 5.0. WMI scores of < 2.5 were considered impaired (i.e. those sites considered as having moderate to highly-degraded conditions) by Croft and Chow-Fraser (2007). Scores > 2.5 were considered to be "good" to "excellent" conditions and showed little signs of negative human disturbance. WMI and WMIadj scores were calculated for the wetlands that were used in the development of the WMI and the scores > 2.5 were predominantly located in Georgian Bay and Lakes Superior and Huron (Croft and Chow-Fraser, 2007). Of the 37 wetlands located in Lake Ontario, only the Perch River and Presqu'ile Provincial Park had a WMI score of 2.66 and 2.78, respectively and were considered unimpaired. However, when exotic species were accounted for in the WMIadj calculation, these scores fell to 2.35 and 2.44, respectively, just below the cut-off. There were three wetlands located in the Bay of Quinte in the WMI development: Hay Bay Marsh, Blessington Bay and Sawguin Creek. WMI values for these locations ranged from 1.62 (Sawguin Creek) to 2.45 for Hay Bay Marsh (surveyed in 2000). The WMIadj values ranged from 1.62 (Sawguin Creek) to 2.23 for the Hay Bay Marsh (surveyed in 1996).

There are several caveats associated with the application of the WMI to the DFO SAV dataset. First, the assessments of the SAV community completed by DFO as part of the transect surveys were not conducted in coastal wetlands, rather they covered SAV beds in more open parts of the bay that may or may not be adjacent to a coastal wetland. Since the WMI was developed specifically as a coastal wetland assessment tool, it has not been tested or calibrated in a more nearshore context. Consequently, target WMI scores (e.g., >2.5 for "good" or "excellent" conditions) should be applied with caution. Next, the WMI was originally created using not only SAV presence/absence, but also floating and emergent aquatic macrophytes species. Thus, its application with only SAV data may change the ultimate WMI score through the exclusion of these other species groups. Similarly, the approach for surveying aquatic vegetation for the WMI uses a stratified quadrat approach (Croft and Chow-Fraser 2007), where the surveyor covers all habitat types within a wetland thereby maximizing the number of species detected. In contrast, the transect approach employed by DFO is more spatially constrained and therefore may not detect rare species thus decreasing the WMI score. Finally, the WMI was developed from coastal wetland plant community data collected throughout the Great Lakes. While this makes it an excellent index for a comparison of conditions among Great Lakes coastal wetlands, it may limit its sensitivity to detect changes in the condition of the nearshore environment within a single geographical region like the Bay of Quinte. Furthermore, since the Bay of Quinte is a naturally mesotrophic system, in contrast to more oligotrophic systems (e.g., Lake Huron) that have high WMI scores, it may naturally have a lower score that reflects its natural level of productivity. Despite these caveats, the WMI is still one of the few indices available that provides an integrated measure of the condition of the local aquatic environment based on the presence of aquatic macrophyte species.

Assessment Results

<u>Measure FWH-1A: Model Estimates of SAV Distribution and Density</u>

Maps of vegetation trends and spatial extents were produced using the 2015 Bay of Quinte SAV model developed from 1972 – 2011 data. A detailed description of the creation of the two-stage model can be found in Appendix B. Figures 2 and 3 illustrate predicted SAV cover and extent for the Post-P and Post-D time stanzas. In the Post-P period, there is a fringe of SAV around the entire upper bay with the largest SAV beds occurring around Trenton, Muscote Bay and Sawguin Creek. Denser SAV (with cover values ≥ 50%) are found closer to the shoreline in general and do not extend very far from shore because of water clarity. Predictions in the Post-D period indicate a substantial expansion of the SAV beds in Trenton and Muscote Bay areas, but with the majority of these expanded areas predicted to have moderate to sparse cover. These trends are simulated whole bay results and the model fits reasonably well to a validation dataset (Appendix C). However, there is variability in the spatial accuracy of vegetation predictions that are discussed later as part of the interpretation of evaluation results and also in Appendix C.

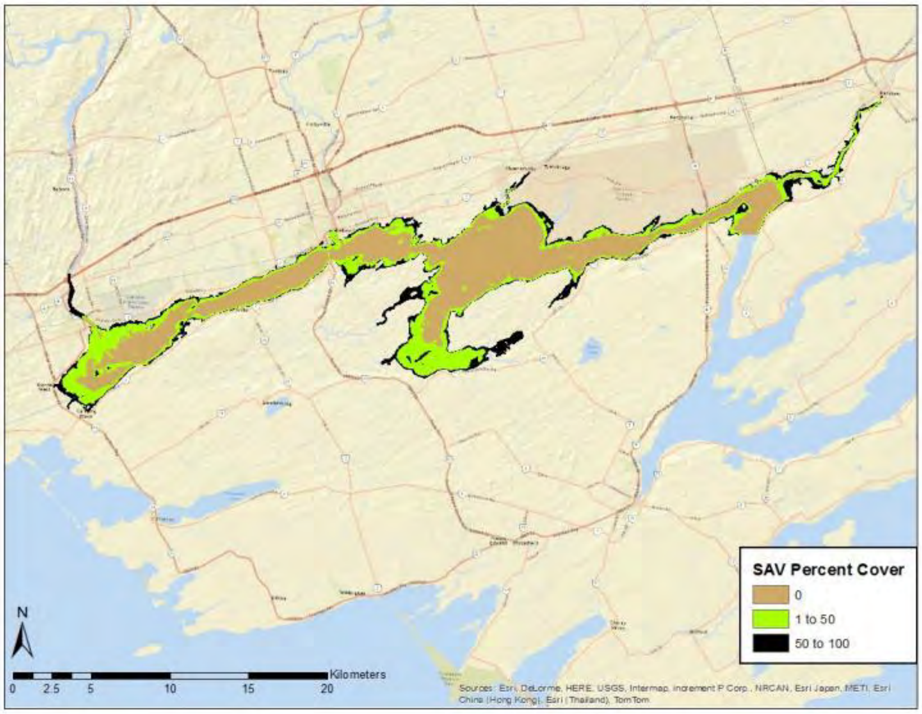

Figure 2. Predicted SAV cover/density and distribution in the upper bay for the Post-P period (1978-1994) using the 2015 SAV statistical models. Mapped at a 30 x 30 m grid cell size for the upper Bay of Quinte.

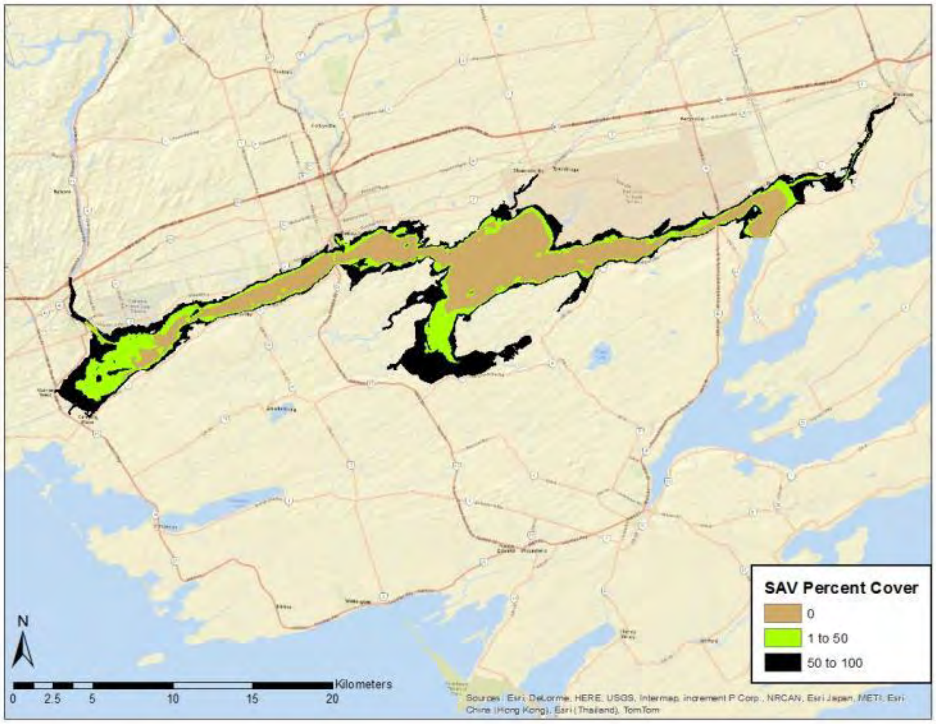

Figure 3. Predicted SAV density/cover and distribution in the upper bay for the Post-D period (1995 – 2012) using the 2015 SAV statistical models. Mapped at a 30 x 30m grid cell size for the upper Bay of

Change in SAV between time stanzas

Table 1 quantifies the predicted changes in SAV percent cover and distribution between the Post-P and the Post-D time stanzas. Total predicted area of SAV in the Post-P period was 38.7 km^2, or 27.6% of the total area of the upper bay. By the Post-D period, the modelled extent of the beds increased to 59.4 km^2, or 43% of the total area of the upper bay. This represents a 53.5% increase in the total areal extent of SAV between the Post-P and Post-D time stanzas. Between the time stanzas, the area with moderate to dense SAV (50-100% cover) increased 3.5 fold and was predicted to occupy 33.3% of the upper bay's wetted area each year on average. Wetted area was recalculated each year based on average water levels.

We further examined the 35-year time trends of SAV areal extent that were predicted by the Bay of Quinte SAV model (Figure 4). The total area of SAV increased over time ($R^2 = 0.44$, $p < 0.01$) but was typically below 30% of the total area of the upper bay prior to 1995. Changes in the size of the SAV beds through time generally followed the variation in Secchi depth that was measured at the Belleville Project Quinte station, although we recognize there are likely spatial differences in turbidity that are not detected at this station. When the SAV data are partitioned into three density categories (Figure 5), a moderate linear trend from 1978 to 2012 ($R^2 = 0.49$, $p < 0.01$) was detected for moderate to dense SAV (>50% cover). The amount of >50% SAV cover showed considerable variation through the Post-P period with a substantial increase in extent at the beginning of the Post-D stanza, suggesting a possible regime shift in ecosystem dynamics in the upper bay. In contrast, the model predicated a weak trend ($R^2 = 0.18$, $p = 0.011$) for sparse to moderate SAV over the 35-year time period. The observed increasing trends in both vegetation density and extent are to be expected given the improvements in water clarity (Leisti et al., 2012). While the lack of a similar trend in sparse to moderate SAV may seem counterintuitive, as SAV beds expand and increase in density, these cover types likely continue to persist along the fringes of the SAV beds and therefore their total area remains relatively constant.

Table 1. Predictions of SAV cover by density class (none, low, high) for the upper Bay of Quinte for the Post-P (1978 – 1994) and Post-D (1995 – 2012) time stanzas based on the 2015 SAV statistical models for the upper Bay of Quinte.

	Area(km^2)		**% of Total Area**		
SAV Cover	**Post-P**	**Post-D**	**Post-P**	**Post-D**	**%change**
None	101.3	78.6	72.4	57.0	-22.5
1 to 50	25.5	13.4	18.2	9.7	-47.3
50 to 100	13.2	45.9	9.4	33.3	249.3
Total SAV	38.7	59.4	27.6	43.0	53.5

Currently in the Post-D time stanza, areal extent of SAV is 3.3% over the 30% target for dense vegetation outlined in Measure FWH-1A (Figure 5). Therefore, technically this measure is no longer impaired. However, the general trend towards denser macrophyte beds and the greater extent of their coverage irrespective of density is noteworthy and requires re-examination of the intent of the original target (Figures 4 & 5, Appendix C). The predicted unvegetated area in the bay decreased from 72% to 57% over the modelled period (Table 1), which indicates that just under half of the bay is now vegetated (notwithstanding recent algal blooms). The proportion of the vegetated area that is covered by moderate to dense vegetation is now predicted to be on average 77% but modelled and measured plant densities are highly variable (Figure 4; Unpub. data Leisti et al.). In addition, it is likely that a drastic shift in water clarity increased the total area of dense vegetation beds in the Post-D time stanza. The maps and figures depict the results of a generalized model of percent cover based on average clarity and water level fluctuations. The implications of those generalizations and caveats to model bay-wide vegetation trends are discussed in Appendix C.

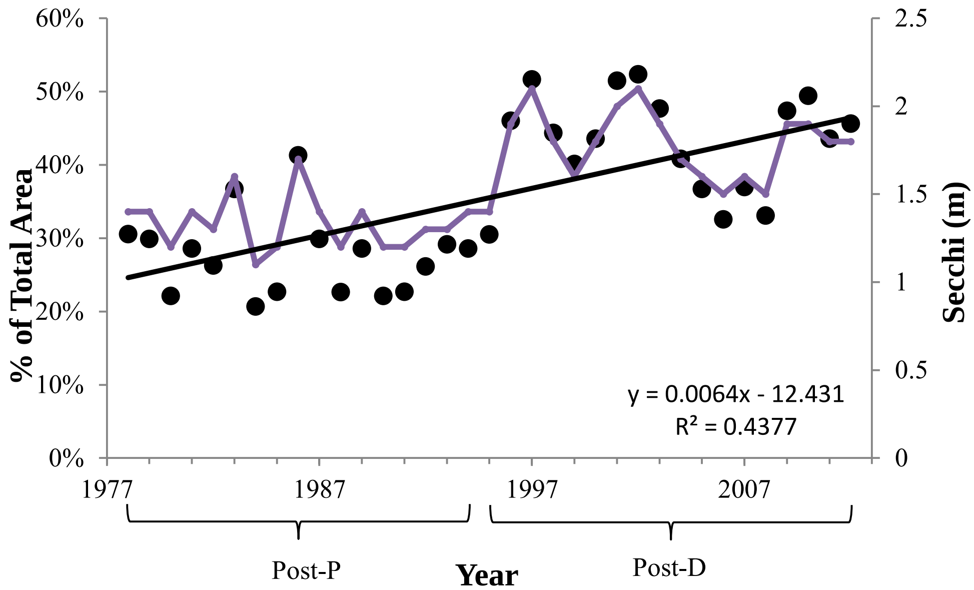

Figure 4. SAV model prediction of percentage (%) of total area covered by submerged aquatic vegetation in the Bay of Quinte upper bay area. Linear regression of total % area covered by submerged aquatic vegetation over time is shown. Mean annual growing season Secchi depth (water clarity; m) is shown by the purple line.

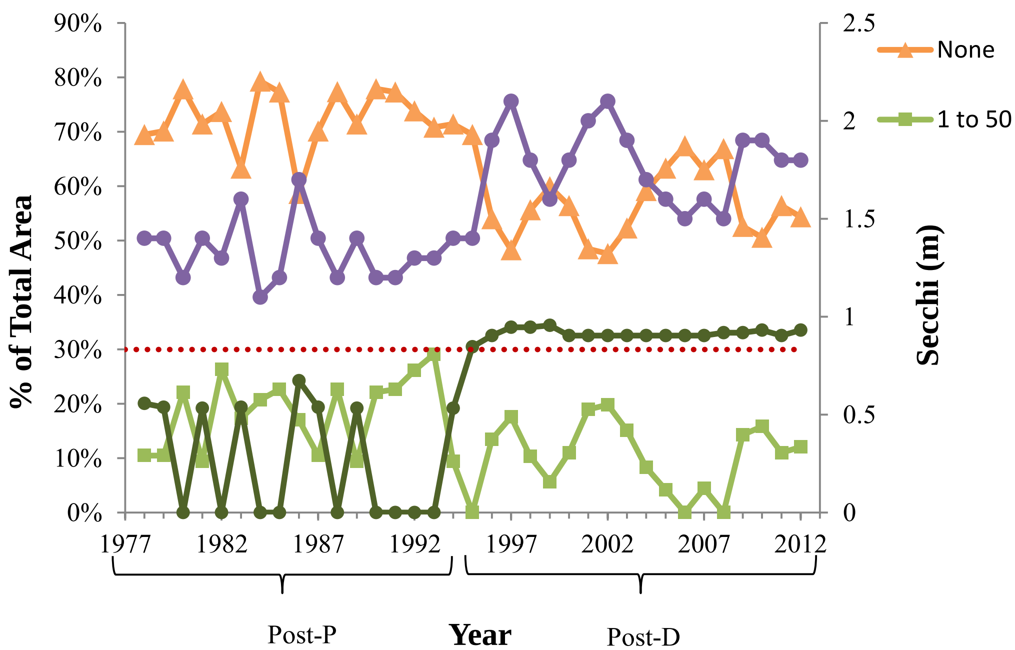

Figure 5. Predicted percentage (%) of total wetted area by year covered by submerged aquatic vegetation (SAV) in the Bay of Quinte's upper bay area in the Post-P (1978-1994) and Post-D (1995-2012) stanzas. Different vegetation density classes are shown: no vegetation cover (none, orange line), 1 to 50% cover (sparse, light green line) and 50-100% (dense, dark green line) cover classes. The red dotted line represents the Bay of Quinte delisting target of 30%

areal coverage of the 50-100% density class and the purple line shows the mean annual growing season Secchi depth.

Measure FWH-1Ba – SAV Species Diversity

SAV species richness ranged from a low of seven in 1988 to a high of 12 in 2011 (Figure 6). There was a slight positive trend for richness but there was unusually high richness (11 species) in 1974. It should be noted that 1974 was an unusually high water level year and flooding of previously terrestrial areas could increase the plant diversity captured in surveys. A one-tailed t-test on species richness between the Post-P (1978 to 1994) and Post-D (1995 to 2011) time stanzas indicated richness was significantly higher in the Post-D (mean=10.5) stanza relative to the Post-P (mean=8.5; $t_{(6)}$= -2.45, p=0.02).

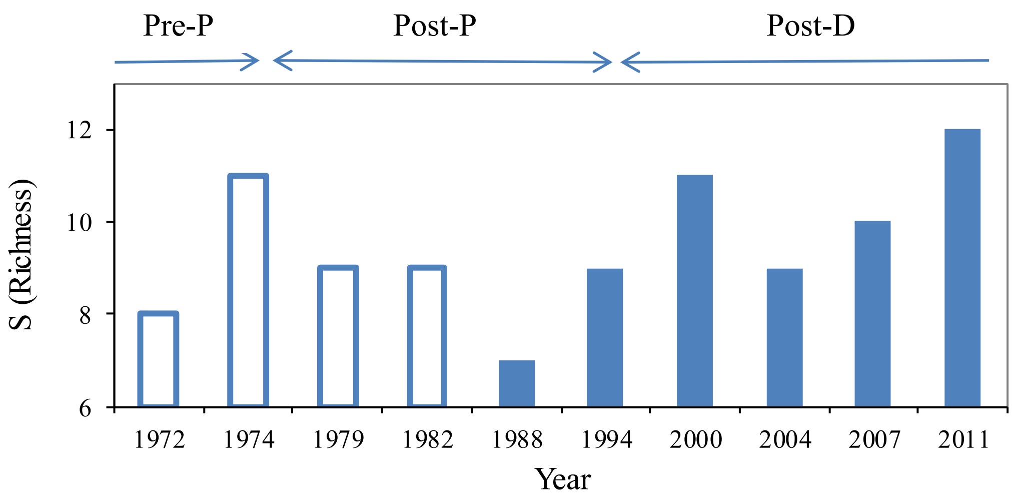

Figure 6. Total aquatic plant species richness for the combined upper Bay of Quinte transects (Trenton North, Belleville North and Big Bay North) conducted during surveys between 1972 and 2011. Richness prior to 1982 (open columns) was partially determined by inferring the presence of some species within the reported datasets as collection methods differed (Refer to Methods of Assessment for details).

SAV richness was typically higher in the upper Bay coastal wetlands that were surveyed by Environment Canada (EC) from 2006-2011. They recorded a mean richness of 18.4 species per site/year, ranging from 11 species at Big Island Marsh East in 2006 to 24 species at Blessington Creek Marsh in 2007 and 2008 (Angela Darwin, per. Comm. 2015). This compares favourably to surveys across 16 lakes in Eastern Ontario that are within 100 km of the Bay of Quinte (Crowder et al., 1977) where mean richness of submerged and floating species was 14.4 and ranged from 6 for Mud Lake to 39 for Lake Opinicon. By comparison, richness in another AOC, Hamilton Harbour varied from 5 to 10 species over six surveys between 1992 and 2012 (Leisti et al., 2015). In Cootes Paradise, a degraded wetland adjacent to Hamilton Harbour, richness for floating and submerged vegetation ranged from 7 to 25 species during annual surveys from 1996 to 2012.

Trends through time for the Shannon-Weiner diversity index are illustrated in Figure 7. A linear fit of the points indicated a slight positive increase in the index value over time and can in part be attributed to the increase in species richness as well as evenness across species throughout the time series. A one-tailed Welch's t-test of the Shannon-Weiner index between the Post-P and Post-D time stanzas indicated that there was a significant difference between index scores in the Post-P (mean = 1.74) and Post-D (mean = 1.99; $t_{(3)}$= -2.48, p = 0.04) time periods.

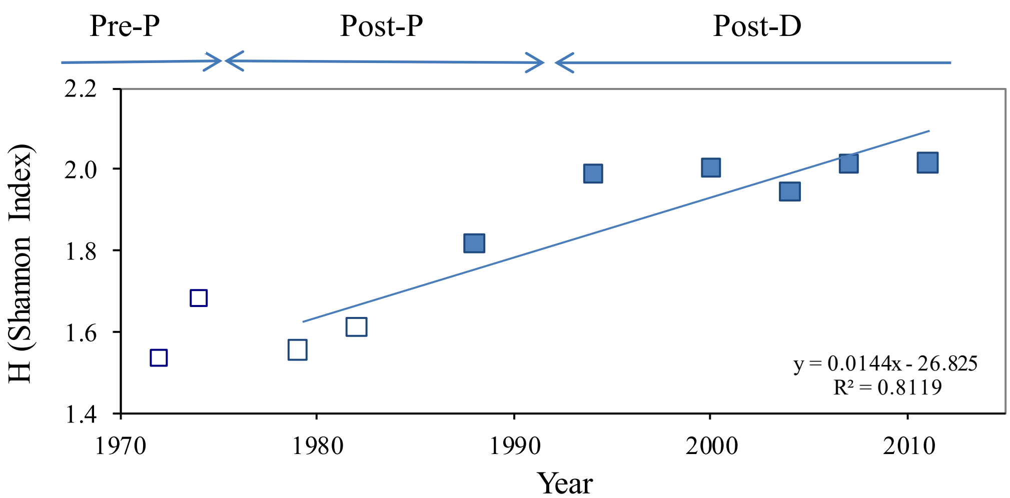

Figure 7. Shannon-Weiner diversity index trend for the combined upper Bay of Quinte transects (Trenton North, Belleville North and Big Bay North) from DFO surveys conducted between 1972 and 2011. Data points prior to 1982 (open squares) were determined using assumptions for some species within the datasets (refer to Methods of Assessment for details).

Figure 8 illustrates the relative frequency of occurrence by species for the upper bay throughout the sampling period between 1972 and 2011. Twelve of the seventeen species are plotted, but an "other" category for the least abundant SAV species was used and consists of the following species: *Potamogeton pusillus, P. pectinatus, P. perfoliatus, R. aquatilis* and *Utricularia vulgaris. Myriophyllum spicatum*, an invasive species that is highly tolerant of degraded conditions, dominated the plant community in the Pre-Dreissena time period (1972-1974). The first specimen for *M. spicatum* in the Queen's University Herbarium was collected in June 1966 at Hill Island, just east of Gananoque while the first record in the bay was in 1972. Anecdotal evidence by residents of the area indicated that this invasive was very abundant by the 1960s (A. Crowder, pers. comm). As reported in other locations (Carpenter, 1980; Trebitz et al., 1993), *M. spicatum* declined substantially after invasion. For the Bay of Quinte, a large decline in distribution and abundance of *M. spicatum* was seen by 1979 and has remained at lower relative abundance levels through to 2011.

Of the five most dominant species in the upper bay, all but *V. americana* are partially to highly tolerant of degraded water quality (Albert and Minc, 2004; Croft and Chow-Fraser, 2007). Some of these tolerant species, such as *Z. dubia* and *C. demersum* have generally increased in frequency through time while *E. canadensis* has typically decreased. With the exception of 1994, *V. americana* has increased in frequency throughout the sampling period. Other less tolerant species such as *P. richardsonii* and *P. zosterformis* have higher occurrences in the Post-D period as well.

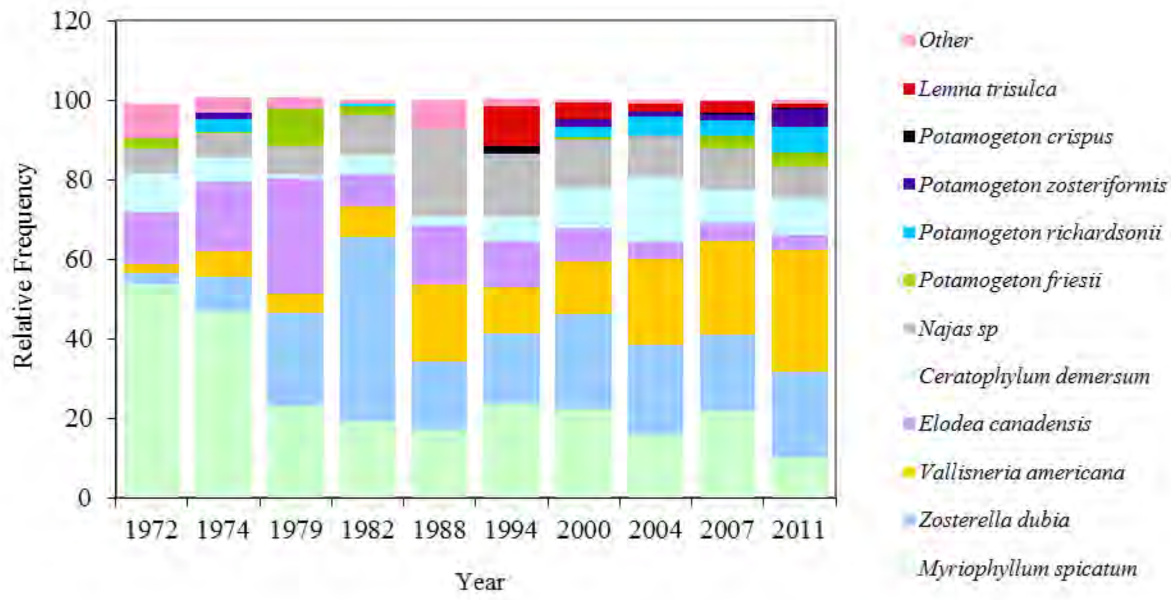

Figure 8. Relative frequency of occurrence by species for the combined upper Bay of Quinte transects (Trenton North, Belleville North and Big Bay North) for DFO surveys conducted between 1972 and 2011. Please refer to the Methods of Assessments for assumptions made in the 1972 to 1982 dataset summaries.

The criteria for measure FWH-1Ba has been satisfied with the positive slope in diversity through time and the significant difference in diversity between the Post-P and Post-D periods.

<u>Measure FWH-1Bb – SAV Community Index</u>

Based solely on species presence/absence, the WMI provides a single value indicating the level of degradation based on ecological characteristics of the species present (i.e., niche breadth and tolerance to degradation). As seen in Figure 9, both the WMI and WMIadj (the index adjusted for exotic species), have not changed significantly through time. The mean WMI score from the Post-D period was 2.18 and was not significantly different than the mean of the Post-P period (2.19; one-tailed t-test, $t_{(6)}$=0.23, p = 0.41). The highest WMI scores were from 1972 to 1982 and again in 2011 and reflect the presence of *Potamogeton* spp. and other species that are believed to have a lower tolerance to degraded conditions. The presence of *M. spicatum* in all years decreases the WMI score even before adjustment since it has a high tolerance to degraded conditions and a wide niche breadth. *Potamogeton crispus*, another invasive species, was found in the upper bay transects in 1994, 2007 and 2011 and further decreases both the WMI and WMIadj scores. *P. crispus* commences growth early in the spring and senesces by early to mid-summer. It is probable that *P. crispus* could be found in all transects during the entire time period, but was likely missed because the surveys were traditionally conducted in late summer after the plant had senesced.

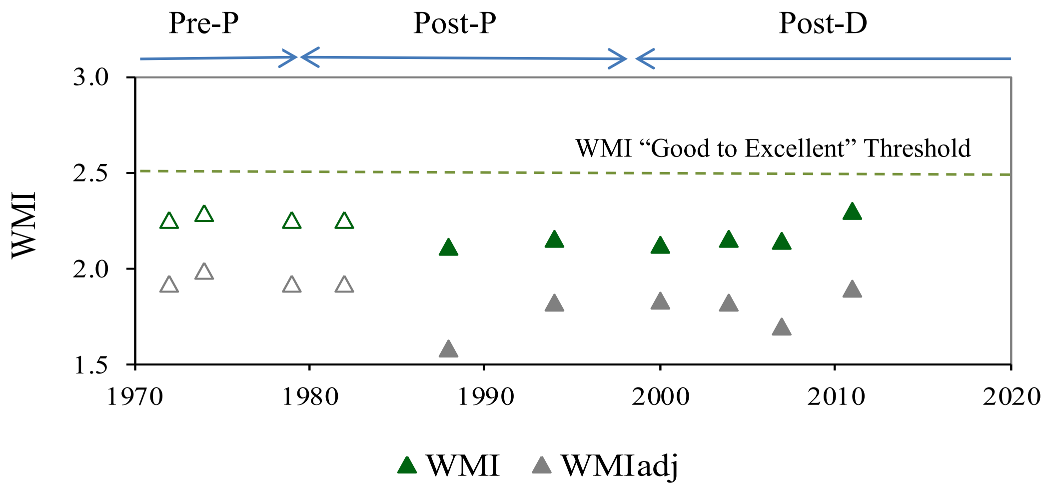

Figure 9. Wetland Macrophyte Index scores for the combined upper Bay of Quinte transects in Trenton North, Belleville North and Big Bay North from DFO surveys conducted between 1972 and 2011. Data points prior to 1982 (open triangles) were partially determined using assumptions for some species within the datasets. Refer to Methods of Assessment for details.

The results indicate that the first criteria has not been met; that a positive slope or statistically-significant improvement in the plant community index (WMI) is not shown; however, there is a second condition in Measure FWH-1Bb that if met, could satisfy the criteria. It states that the index values must be in the "good to excellent" range for 3 recent consecutive SAV surveys. For coastal wetlands, Croft and Chow-Fraser (2007) suggested that a WMI score > 2.5 could be considered "good to excellent" conditions. In the Bay of Quinte nearshore, WMI score for the last three surveys ranged from 2.1 to 2.3 (WMIadj scores ranged from 1.7 to 1.9) and therefore did not satisfy the criteria based on the coastal wetland cut-off; however, they are relatively close to the target before the *post hoc* adjustment for invasives. Since the WMI scores were close to meeting the delisting criteria, we examined several lines of evidence and caveats of the approach to support a final decision.

The target specified for this delisting criteria states that the "… community structure of SAV in the Bay of Quinte is restored to levels more reflective of the mesotrophic system that existed before excessive nutrient loading created a eutrophic/hyper-eutrophic system." As previously noted, Croft and Chow-Fraser (2007) based the "good to excellent" categorization (WMI scores >2.5) on a group of coastal wetlands that included oligotrophic sites (typically in Lake Huron, Georgian Bay, and Lake Superior). A slightly lower WMI score cut-off is therefore likely appropriate for a mesotrophic system such as the Bay of Quinte since some high-scoring species (e.g., *Lobelia dortmanna*) that are associated with oligotrophic conditions are unlikely to occur in the bay. Similarly, many of the sensitive plants identified in Croft and Chow-Fraser (2007) that contribute towards a high WMI score were not found in the bay historically (Crowder and Bristow, 1986) so there are likely biogeographic considerations that have not been accounted for in the index. In fact, Croft and Chow-Fraser (2007) found evidence of a geographic gradient when developing the WMI, with high correlations with latitude on axis 1 and longitude on axis 2 of the CCA suggesting site specificity of the vegetation. Therefore, we suggest that the cut-off of >2.5 proposed by Croft and Chow-Fraser (2007) may be unattainable for the nearshore SAV community in the bay and a lower threshold may be appropriate for delisting.

To further explore more regionally appropriate Bay of Quinte WMI threshold, we examined Bay of Quinte WMI scores relative to other wetlands in Lake Ontario using data from the appendix of Croft and Chow-Fraser (2007). WMI scores for wetlands used in the development of the WMI included the Bay of Quinte wetlands of Sawguin Creek in 1996, Blessington Bay in 2002 and Hay Bay Marsh in 1996, 2000 and 2002. There were 25 other Lake Ontario wetlands sampled for the study and seven of these wetlands were sampled over two years. The WMI and WMIadj scores for these wetlands were plotted with the Bay of Quinte sites (Figure 10). The DFO WMI scores calculated for the north transects in the upper bay from 1972 to 2011 (as shown in Figure 9) were similarly plotted (Figure 10). The lowest WMI score was recorded in Grindstone Creek, Hamilton Harbour in 2002, while the highest was found in Presqu'ile Provincial Park in 2002. With the exception of Sawguin Creek, all Bay of Quinte WMI scores are higher than 50% of the dataset. A one-tailed t-test revealed that the Bay of Quinte WMI scores (mean = 2.21) were significantly higher than scores in the other Lake Ontario wetlands (mean = 1.87, $t_{(45)}$= -3.48, p=0.0005). Therefore, within Lake Ontario, Bay of Quinte WMI scores tend to rank these sites among the least degraded.

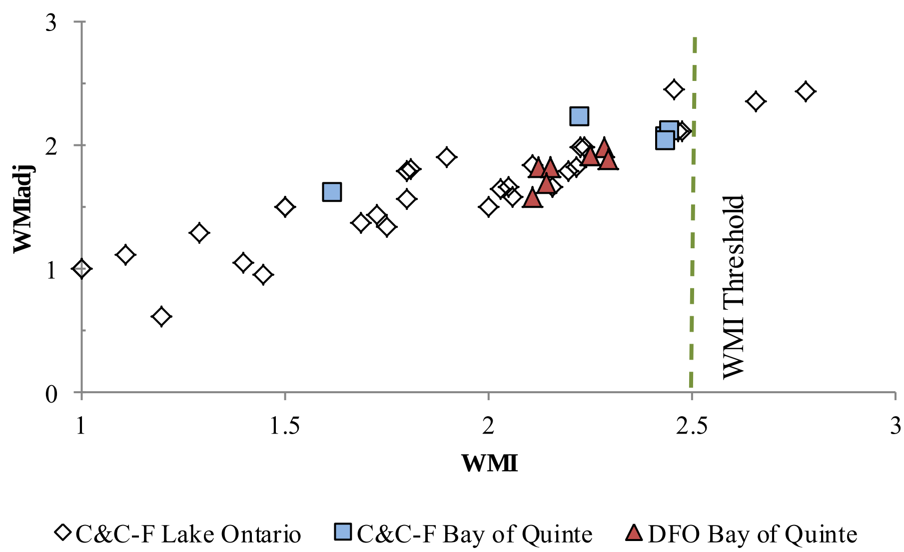

Figure 10. Wetland Macrophyte Index values for Lake Ontario, including those from the Croft and Chow-Fraser (C&C-F) appendix and the 1972 to 2011 DFO upper bay north transect dataset. Also shown is the WMI threshold for the "good" to "excellent" range.

Finally, we compared WMI and SAV IBI scores calculated using Bay of Quinte data provided by Environment Canada (EC). The difference between the WMI and the SAV IBI results were examined in greater detail because WMI scores for the upper bay indicated possible impairment while the 2013 SAV IBI score for Bay of Quinte wetlands determined by EC had met the FWH-2 target (Environment Canada 2015). Target FWH-2 addressed Bay of Quinte coastal wetlands' SAV health and used a SAV IBI to assess impairment status (Grabas et al., 2012). This SAV IBI was developed using 2003-2009 SAV community data collected from 46 Lake Ontario coastal wetlands using 20 random 1-m^2 quadrats. The SAV IBI includes four metrics: 1) the number of turbidity-intolerant species, 2) number of native species, 3) Coefficient of conservatism, and 4) total cumulative coverage. To be recommended as unimpaired, the 2013 mean SAV IBI score must be within two standard deviations above the Bay of Quinte 2006-2010 mean SAV IBI, after accounting for varying conditions in other Lake Ontario wetlands outside of the Bay of Quinte Area of Concern (Environment Canada, 2015).

EC provided their wetland SAV data and IBI scores so we could calculate the WMI for their sites to see if their data would fall in the "good to excellent" range of the WMI. EC SAV data from the upper bay wetlands were extracted for the years that DFO sampled (2007/2011) and the WMI was calculated. EC had sampled the same twelve wetlands in both 2007 and 2011, with SAV IBI scores ranging from 59 to 97. We calculated the WMI scores for this same dataset and found that raw index scores ranged from 2.1 to 2.7 and WMIadj values from 1.7 to 2.5 (Figure 11). Only in four instances did the WMI scores indicate "good to excellent" conditions and these all dropped below the cut-off of 2.5 when the WMIadj for non-native species was applied. This type of comparison between the WMI and the more regionally derived SAV IBI presents an opportunity to roughly calibrate the WMI to local conditions. Indeed, WMI scores (non-adjusted) for sites that met the requirements for the SAV IBI were always greater than 2.0 (Figure 11); therefore this may be a more realistic target for WMI scores in the Bay of Quinte.

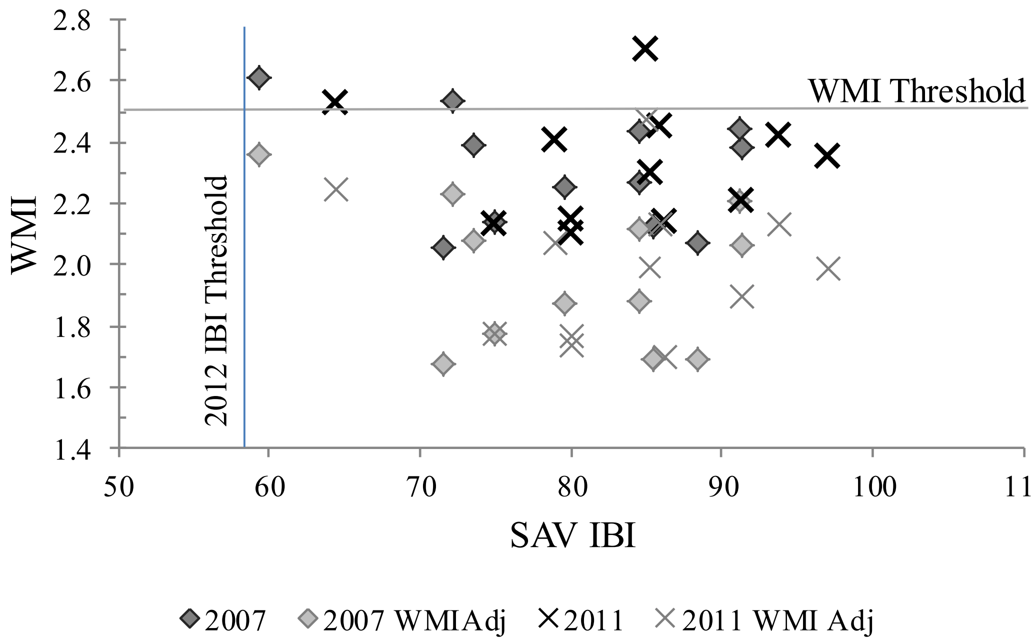

Figure 11. SAV IBI, WMI and WMIadj scores for twelve upper Bay of Quinte coastal wetlands in 2007 and 2011. WMI scores < 2.5 indicate moderate to highly degraded conditions while scores > 2.5 are considered good to excellent conditions. (G. Grabas, Environment Canada, provided the IBI scores and species list used to calculate the WMI values.)

With this as a new target, all WMI scores derived from the DFO transect surveys are above 2.0 suggesting these scores meet the modified delisting target. That being said, there are many factors that should be taken into account when deciding if Measure FWH-1Bb is still impaired. These include but are not limited to:

1) the WMI was developed using many oligotrophic sites and a large geographic gradient and therefore cut-off values may not be appropriate for the mesotrophic Bay of Quinte;

2) the quadrat sampling conducted by DFO during surveys was relatively limited because it was intended for acoustics validation and simple species composition information;
3) the middle and lower bay sites were excluded from the assessment of the WMI simply because the upper bay was the intended target for vegetation remediation and restoration;
4) the WMI scores were calculated from DFO surveys conducted in areas outside of emergent wetlands and SAV species richness is likely lower in these area than in wetlands;
5) Bay of Quinte WMI scores were significantly higher than scores for the rest of Lake Ontario; and
6) a comparison of SAV IBI and WMI scores using the same dataset showed that the majority of the WMI scores were below the “good to excellent” range, while the use of the SAV IBI criteria suggested that SAV in Bay of Quinte coastal wetlands are not impaired by Lake Ontario standards.

Since no regionally derived plant community health index is available to assess the SAV community data collected by DFO, the WMI was selected as the best available option despite the previously noted caveats associated with its implementation. While WMI scores did not meet the 2.5 cut-off suggested by Croft and Chow-Fraser (2007), WMI scores for the upper bay transect surveys were consistently among the highest for Lake Ontario wetlands. Also, based on a coarse calibration using the SAV IBI, Bay of Quinte WMI scores calculated for the present report were as high or higher (>2.0) than the WMI scores calculated for coastal wetland Bay of Quinte SAV communities that were previously delisted using the SAV IBI. When these factors are weighed in light of the noted caveats, it is the opinion of the authors that FWH-1Bb is not impaired.

Discussion and Rationale for BUI Status

There are many lines of evidence presented here for assessing the extent, density, health and diversity of submerged aquatic vegetation in the bay, especially the upper bay which was originally the most impaired. Each one is considered separately in a rationale for delisting.

Extent & Density

Based on predictions from the 2015 Bay of Quinte SAV model, the extent of total vegetation in the upper bay has increased significantly in the Post-D stanza and is covering 43% of the total area in the Post-D stanza. The proportion of moderate to dense SAV is now meeting the delisting criteria target assuming the following caveats are true:

1. the survey transects from which the model was generated provide a good representation of whole upper bay SAV distribution and density patterns;
2. trends in SAV density and extent in the upper bay are mainly driven by water depth and clarity as determined from statistical modelling;
3. Secchi depths from the offshore Belleville Project Quinte station are representative of water clarity throughout the upper bay; and,
4. the simple depth-SAV density relationships used in modelling are accurate enough to show generalized patterns even though there may be local errors that over and under predict and on a system wide basis these errors offset.

The modelling exercise revealed that predicting SAV across the entire area of the upper bay using only one offshore water clarity station is problematic for providing accurate local predictions. More spatially extensive Secchi or other water clarity data would have been preferable for addressing several key questions that arose during spatial modelling including: 1) how representative is the Belleville station for the remainder of the upper bay's water clarity?; 2) are offshore Secchi depths representative of nearshore water clarity conditions?; 3) are there seasonal water clarity impacts on SAV growth and establishment?; and 4) what is the impact, extent, location and duration of algal blooms and is their presence reflected in clarity information or do they inhibit SAV growth by different mechanisms?

There are several abiotic and biotic factors that can affect SAV density and distribution for which there were insufficient data. Some of the factors include: small-scale changes in sediment, ice scour, disturbance by common carp (*Cyprinus carpio*) and grazing by waterfowl and herbivores. Other factors, such as substrate type, temperature, bottom slope, extreme weather events, and fetch were tested in earlier iterations of Bay of Quinte SAV models (Seifried, 2002; 2011 unpublished data) but were deemed to add insignificantly to predictive power.

The 2015 Bay of Quinte SAV model was developed to represent general conditions and may not accurately represent SAV density and distribution at a small scale. Patchiness that may be typical for many SAV communities is difficult to predict and is not addressed by this model, which only assesses regional patterns in density and extent. For detailed information on the development of the current model, refer to Appendix B. Model validation and scenario tests can be found in Appendix C.

An important outcome from the model was the identification of a water clarity threshold for SAV coverage (Appendix C). In order to ensure that the delisting target (i.e., 50% density of SAV over 30% of the area) is maintained, secchi depth should be kept at or above 1.4 m. Similarly, water levels between 74.0 and 75.5 m (a.s.l.) are also necessary to provide sufficient areal coverage of SAV. If water clarity drops below 1.4 m or water levels fall outside of the noted range, it would be pertinent to reassess SAV coverage to ensure it has not fallen below the delisting target.

A search of the major Bay of Quinte reports did not reveal the rationale behind the setting of 50% or greater as the SAV density target for BUI FWH-1A. A literature review (Appendix A) indicated that many of the small-scale experimental studies advocate intermediate plant cover to achieve fisheries management objectives; large-scale studies showed mixed results. Many of the whole lake studies described plant cover in terms of areal extent rather than density. Results from some studies were species- or age-specific or were confounded with other factors. All told, we could not find consensus within the literature of an optimal level of SAV density for fisheries management.

SAV Diversity and Health

Both richness and diversity in the aquatic plant community have increased over the sampling period between 1978 and 2011. However, two of the species identified, *M. spicatum* and *P. crispus* are invasive and have been present in the Bay since 1972. Additional non-native species including *Hydrocharis morsus-ranae*, *Nitellopsis obtusa* and *Butomus umbellatus* have been found by EC in Bay of Quinte coastal wetlands and *Phragmities australis* has recently become established in several sites (Christine Jennings, Quinte Conservation, pers. com. 2015). Phragmites specifically is expanding throughout the Great Lakes (Tulbure et al., 2007;

Mazur et al., 2014) and in some instances has been found to negatively influence fish and marsh bird populations (Able, 2003; Hunter, 2006; Whyte et al., 2015). Since Phragmites is an emergent species, this line of discussion likely falls outside of the scope of the current evaluation of the SAV BUI. However, we believe strongly that the expansion of Phragmites and other aquatic invasive species must be considered during the delisting process as well as in the development of future monitoring and management strategies. Expansion of these non-native species may alter nearshore aquatic habitats and potentially detract from the benefits associated with increased SAV cover and density.

Two measures of SAV health, the WMI calculated for FWH-1Bb and the SAV IBI for FWH-2 have revealed contrasting results. A comparison between the WMI and SAV IBI using a dataset provided by EC revealed that WMI scores for these coastal wetlands were below the “good to excellent” threshold suggested by Croft and Chow-Fraser (2007). Given consideration of the above issues with the application of the WMI, it is advised that the WMI be applied with caution until such time that biogeographic reference conditions are accounted for in the calculation of this index. If the index itself is not modified, a lower threshold may need to be used in the Bay of Quinte and the comparison with the SAV IBI did identify 2.0 as a potential lower threshold for delisting. That being said, the SAV IBI itself may be overestimating the health of Bay of Quinte SAV because of the limited geographic range of wetlands used to create this IBI. Regardless, it is currently the expert opinion of DFO that FWH-1Bb is not impaired despite not receiving a good to excellent rating according to the WMI and this decision is based on several lines of evidence that are detailed in this report.

The results from the WMI/SAV IBI should be examined in greater detail since no relationship between the two indices was found using the EC 2007 and 2011 upper bay wetland dataset. While both of these indices use the aquatic plant community to assess the level of disturbance at a site, differences in the approach and the spatial coverage of the dataset used in their creation are likely the main factor in their lack of concordance. The scale of the gradient used to create the SAV IBI represents only a fraction of the scale used to create the WMI (e.g., Bay of Quinte region vs Great Lakes region, respectively). Therefore, in terms of WMI scores, we would expect and indeed do see only a portion of the available range (i.e., scores ranging from 1.5-2.3), whereas a more complete range of SAV IBI scores are present in the current and historic dataset. Presently, only a cursory examination of the relationship between these indices has been undertaken, but a more detailed evaluation would help to identify the species, species-groups or metrics that drive the differences between the indices. This work does not have to be completed by DFO but it is recommended that indices be standardized or a multi-metric approach be applied for scoring.

The currently recommended status for the Bay of Quinte submerged aquatic vegetation is shown in table 2; all current measures were assigned a not impaired status. However, non-native species (e.g., *Phragmites australis* and *Nitellopsis obtusa*) - and harmful algal blooms may affect this BUI delisting target by displacing SAV or reducing SAV density. Management of non-native species, regional efforts geared toward preventing the spread of non-native species and meeting the targets for the Eutrophication BUI may mitigate negative effects.

Table 2. Recommended status of measures

Measure	Previous Status	Recommended Status
FWH-1A	Requires Further Assessment	Not Impaired, requires continued monitoring
FWH-1Ba	Requires Further Assessment	Not Impaired
FWH-1Bb	Requires Further Assessment	Not Impaired

Future monitoring or actions required

It is recommended that the Bay of Quinte be re-surveyed in 2016 or 2017 using the acoustic protocol developed by DFO, with species composition sampling according to the SAV IBI protocol (Grabas et al., 2012). Mean SAV density in the upper bay peaked in 2000 and subsequently declined in 2004 and 2007. Results from the 2011 survey indicated that mean cover for this area was similar to the 2007 value, although mean bed extent has decreased (Figure 12). These changes may be due to increased algal activity in recent years and should be investigated further. It is recommended that SAV surveys be conducted on, at a maximum, a 5-year rotation to track the status of the SAV community. Perhaps with partners the SAV community in the Bay of Quinte can be monitored more frequently.

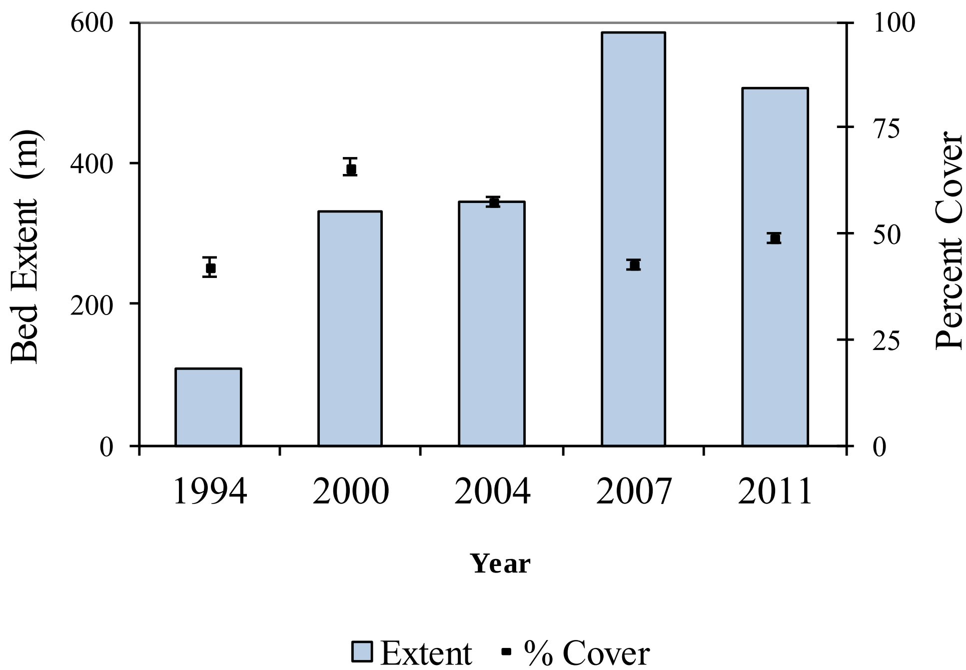

Figure 12. Average SAV percent cover and bed extent for the upper Bay of Quinte for the years 1994 to 2011 from the analysis of acoustic data. Cover error bars ± SE. Bed extent is an approximate measure of the size of the SAV bed assessed during the hydroacoustic surveys.

Future surveys should also include areas outside of the reference transects that would be targeted based on factors known to affect SAV, for instance exposure. A GIS exercise could determine locations in the upper bay that have a range of exposure levels that could be visited during the survey to determine if SAV is present. Results from the current model and the underlying GIS database for Quinte could determine areas that would benefit from supplementary sampling.

Re-occurring algal blooms in the upper bay may prove challenging for SAV in the future if they appear at critical times and are prolonged in duration. Shading by algae is thought to have been a critical factor in the decline of the SAV beds in the 1960s (Crowder and Bristow, 1986) and SAV did not recover substantially until after the invasion by Dreissena and the subsequent improvement in water clarity. Periodic algal blooms have been shown to decrease SAV coverage on a seasonal or short-term basis (Gallegos and Bergstrom, 2005); with recent work on seagrass (*Zostera marina*) suggesting that summer is when SAV are most susceptible to light reductions caused by algal blooms since light requirements for SAV increase with water temperature (Kim et al., 2014). This peak in susceptibility overlaps with peak algal productivity in the Bay of Quinte (summer to early fall; Millard and Johnson, 1986). Therefore, while meeting the Eutrophication BUI targets should help mitigate the negative effects of algal blooms on SAV, ongoing monitoring of the timing and extent of algal blooms paired with assessments of the SAV community is likely necessary to evaluate their influence on SAV cover and density.

As previously discussed, there are numerous non-native species established in the Bay of Quinte and an expansion of these species may result in a loss of SAV habitat or a change in the provision of ecosystem services. Ongoing monitoring of non-native species, Phragmites in particular, is therefore necessary to determine the extent of their expansion, their effects on native SAV beds and whether they are disrupting the function of nearshore environments. In addition to non-native species that are already established in the bay, there are several other species that may become established in the future (based on risk assessments currently in review by the Ontario Ministry of Natural Resources and Forestry). Hydrilla (*Hydrilla verticillata*) has no known populations currently in the Great Lakes, but is found in some connecting channels (e.g., Erie Canal) and is rated as a high-risk for invasion. In contrast, both water soldier (*Stratiotes aloides*) and European water chestnut (*Trapa natans*) have been found close to the Bay of Quinte (Trent-Severn waterway and Wolfe Island, Lake Ontario, respectively) and represent a very high risk for invasion. While the potential effects of some of these species on fish and wildlife habitat are not well understood, they may further transform the SAV community and potentially result in a loss or alteration to some of the ecosystem services associated with SAV. If there are extensive changes in SAV community composition, it may be necessary to adjust SAV coverage and density targets to ensure they are still meeting the desired functional output for the delisted fish and wildlife habitat BUI. Ongoing monitoring of SAV community composition, as discussed previously, will help to detect novel non-native species and inform the implementation of management or control measures.

All the agencies involved in the Bay of Quinte should be contacted to determine if these types of data are already being collected and to develop a coordinated approach to meet data and monitoring requirements.

Glossary of Acronyms

DRCWMP – Durham Region Coastal Wetland Monitoring Protocol
SAV – submerged aquatic vegetation
IBI – index of biotic integrity
WMI – Wetland Macrophyte Index
DFO – Fisheries and Oceans Canada
EC – Environment Canada
BQ – Bay of Quinte
BUI – beneficial use impairment
FWH - Fish and Wildlife Habitat (subgroup)

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Appendix A: Macrophyte Density and Fish Metrics Literature Review

As detailed in this paper, the historical development of BUI FWH-1Aa initially included targets regarding the areal extent of SAV in the upper Bay of Quinte. A 50% density component was added in 1993, but we could not determine why a 50% value was chosen. Ken Minns (per. comm. 2015) indicated that researchers believed there would be less benefit to fishes if the areal target was achieved with sparse SAV density. Since our model results for delisting this BUI indicate that both the areal and density target have been achieved, but the density component is only slightly above the BUI target (Table 1), we conducted a literature review on SAV density and fish metrics. Our objective was to determine if studies agree on an optimal level of SAV density for fisheries management. While this is not an exhaustive review, many of the seminal papers on this subject have been included.

Numerous studies have detailed the importance of macrophytes to fishes as spawning, nursery and feeding areas. Macrophytes provide a substrate for the growth of epiphytic algae and macroinvertebrates that form the base of the food chain. In a review of the primary literature on fish-plant interactions in the past 40 years, Dibble et al. (1997) provide a taxonomic list where studies have linked nineteen fish families to macrophytes. This list is further broken down by life stages, and of the nineteen families that are associated with macrophytes as adults, nine families are associated as juveniles and twelve families as larvae. Although many of the macroscale studies assessed by Dibble et al. (1997) involved macrophyte control, they found that vegetated areas supported fish densities from 15 thousand to 2 million fish per hectare higher than unvegetated areas. It should be noted that Dibble et al. (1997) found contradictions and ambiguity among study results, which they attributed to differences in the measurement methods and spatial scale of the studies.

Beginning in the early 1980s, several mesocosm experiments were conducted to examine fish behaviour and predator-prey relationships. Earlier fish studies of prey size selection, swimming speed and habitat use had suggested that fish forage in ways that maximize their net energy intake (Mittelbach, 1981), therefore their behaviour may be dependent on factors such as habitat structure, temperature, predation risk, competition and prey size and density. To assess the impact of habitat, numerous experimental studies were undertaken using various densities of either natural or artificial SAV to examine fish predator-prey interactions (Crowder and Cooper, 1982; Savino and Stein, 1982; Gotceitas and Colgan, 1989; Schriver et al., 1995; Valley and Bremigan, 2002) and impacts on fish production (Carey et al., 2010).

In 1982, Crowder and Cooper used experimental ponds with three levels of SAV density to determine if habitat complexity mediates interactions between small (~9 cm) Bluegill sunfish (*Lepomis macrochirus*: BG) and their prey. This study found that BG consumed more prey and grew better at intermediate SAV densities (111 stems/m^2). Despite a higher biomass of prey available in the high SAV density (177 stems/ m^2) quadrants, BG experienced reduced prey capture rates and slower growth, which were attributed to reduced encounter rates with prey. At low densities (36 stems/ m^2), the rate of change of total prey biomass was greatest since the fish were able to deplete the highly profitable prey, promoting instability in the system.

Laboratory pools at four different densities of artificial SAV (0, 50, 250 and 1000 strands/ m^2) were used by Savino and Stein in 1982 to determine the relationship between BG predation mortality and SAV density. They found a decrease in predator activity (Largemouth Bass - *Micropterus salmoides*: LMB) with increasing strand density with a near total loss of predation success at the two highest levels of SAV density, which they estimated as greater than 40% cover. This was a result of inability of LMB to establish visual contact with their prey at high levels of SAV density and changes in BG behaviour. At low strand densities, BG exhibited schooling behaviour, but subsequently dispersed and sought refuge at the highest level of SAV density. LMB behaviour also changed with increasing strand density from active searching at low strand density to ambush predators at high strand density.

Gotceitas and Colgan (1989) hypothesized that the relationship between habitat complexity and predator foraging success was non-linear and subject to threshold levels. Using logistic models from their aquaria experimental data, they determined a SAV threshold of 276 strands/m^2 above which LMB foraging success on juvenile BG was significantly reduced. They also noted that BG actively select SAV habitat as a refuge at SAV densities of 516 strands/m^2 indicating that BG select habitats safe from predation.

Schriver et al. (1995) used 100 m^2 enclosures in shallow, eutrophic Lake Stigsholm to examine the effect of varying SAV abundance on the interaction of planktivorous fish (Three-spined Stickleback – *Gasterosteus aculeatus* and Roach – *Rutilus rutilus*), zooplankton and phytoplankton. Positive relationships were found between SAV percent volume infested (PVI; relates areal cover with plant height and water depth) and 1) several zooplankton species and 2) zooplankton grazing pressure on phytoplankton. Both of these relationships were negatively related to the PVI x fish interaction. When the PVI was higher than 15-20%, the zooplankton community was less affected by fish predation, but increasing fish density substantially decreased this refuge effect.

Using aquaria, Valley and Bremigan (2002) examined the foraging success of age-0 LMB with two levels of BG prey density and at two levels of SAV stem density. They also examined the effect of moderate and high SAV density and two different types of SAV structure (58 and 115 stems/m^2 for SAV canopy monoculture and 329 and 707 stems/m^2 for diverse SAV). Prey density did not have any effect, but when plant densities were compared, LMB experienced greater foraging success at moderate plant density. Comparison between diverse and canopy SAV structure revealed higher foraging success rates in diverse SAV.

In 2010, Carey et al., conducted experiments using two levels of SAV strand density and sunfish diversity to explore the direct and indirect effects of biodiversity and habitat complexity on fish production. In high SAV density (714 strands/m^2), they observed a shift in the food web structure from pelagic to littoral due to increased periphyton, which resulted in decreases in total phosphorus, phytoplankton, cladoceran abundance and fish biomass. At low SAV densities (161 strands/m^2), the “BG only” treatment recorded increased growth.

Much of the experimental evidence suggests predator foraging success was reduced at higher SAV densities. Most of these behavioural data are derived from studies of small sunfishes and additional studies are needed to assess if these relationships determined by mesocosm experiments are applicable to fish community responses on a macroscale. There are several large scale studies that have examined the effect of varying macrophyte density on LMB abundance (Durocher et al. 1984), foraging (Bettoli et al. 1992) and LMB and BG growth (Olson et al., 1998; Cheruvelil et al., 2005). A study by Randall et al. (2012) examined Pumpkinseed (PS - *Lepomis gibbosus*) and Yellow Perch (YP – *Perca flavescens*) biomass and production at four levels of SAV density in eight locations in Prince Edward County. Two papers examined effects of variable SAV density on the entire littoral fish community, assessing production (Randall et al. 1996) and fish abundance, richness and composition (Bryan and Scarnecchi 1992).

From 1976 to 1978, thirty Texas reservoirs ranging in size from 200 to 89,000 ha were sampled to determine the factors that affect LMB standing crop and recruitment to harvestable size (Durocher et al., 1984). Percent submerged vegetation estimates were made by aerial photography or on-site using a grid with the values projected to total lake coverage. The mean percent SAV coverage was low (4.2 ± 2.0) with a maximum of 20%. Significant positive linear relationships were determined for both LMB standing crop (r=0.79) and recruitment to harvestable size (r=0.69).

Bettoli et al. (1992) conducted a seven year study of LMB predation in Lake Conroe, Texas. Prior to the introduction of Grass Carp (*Ctenopharyngodon idella*), which eliminated all SAV, the areal coverage of SAV in the 8,100 ha reservoir ranged from 39 to 44%. When SAV was present, most of YOY LMB did not consume fish until reaching lengths of 140 mm whereas once SAV was absent, they began consuming fish when they were 60 mm. This resulted in significantly faster first-year growth for all year-classes of LMB year-classes post SAV removal. They noted that there was ample prey fish for LMB both pre- and post-SAV removal and that no lake-wide change in fish biomass and productivity occurred. There was a shift in the forage fish community structure to domination by planktivorous fish with a loss of some phytophilic species.

In 1998, Olson et al., reported on a SAV reduction manipulation experiment for four treatments and nine control lakes in Wisconsin that had high levels of invasive European Watermilfoil (*Myriophyllum spicatum*). The objective of the study was to determine if a 20% reduction in SAV area would affect the growth of BG and LMB. A series of 2 m swaths of evenly spaced channels perpendicular to the shoreline were cut to a water depth of 5 m. In the year following manipulation, there were substantial increases in the growth rates of BG in the treatment lakes relative to the control lakes, but this response was age dependent. BG ages 3 and 4 exhibited the strongest response with a 10 mm/yr increase in growth for age 4 BG in the cut lakes relative to the control. Although not statistically significant, most age classes of LMB, with the exception of age 3, did increase growth rates in the cut lakes.

The objective of the study by Cheruvelil et al. (2005) was to determine if the relationship of optimal fish growth at intermediate macrophyte abundance found in the majority of the mesocosm experiments was applicable at either lake-wide or littoral zone scales. Nine macrophyte metrics (including whole lake % cover, littoral % cover, % milfoil, and % dense cover) were derived from visual assessments using point intercept sampling of 45 thermally stratified lakes in Michigan between 1999 and 2002. BG and LMB mean length summaries for ages 1 to 5 were obtained based on surveys between 1990 and 2002 from the MI-DNR database. These data were used to calculate annual fish growth increments for each species, age and lake. Quadratic equations of these data did not produce a unimodal relationship and none of the parameter estimates were different from zero. A negative linear relationship between some SAV metrics and fish growth rates was found for some ages of both species (e.g. at the whole-lake scale, 4-5 year BG growth decreased with both macrophyte % cover and % dense cover), but little variation in fish growth was explained (r^2 = 0.10 to 0.32) at the whole-lake scale by any macrophyte metric. The authors suggest that the relationship derived from small-scale experiments may not be applicable at larger scales or that there could have been a failure to detect this pattern due to a small number of lakes in the dataset that had either very low or very high macrophyte cover values.

In 1999, electrofishing surveys and visual assessments of SAV bottom cover were conducted in eight locations in Prince Edward County at the eastern end of Lake Ontario. From these data, Randall et al. (2012) determined that the production index was six times higher at low fetch sites with medium to high SAV cover compared with high fetch sites with absent or sparse SAV.

There was a significant relationship between SAV cover and biomass for PS and YP, but the growth index and average size for these two fishes were unrelated to SAV cover. Among the four categories of SAV cover (absent; sparse <33%; medium 33 to 66% and high >66%), there was an asymptotic response curve showing significant variation in the production index for both YP and total fish between the absent and the sparse to high categories. The PS response curve was approximately linear, with significant differences in the index at the absent and high SAV categories. Randall et al. (2012) note that these findings should be qualified against factors such as sampling bias, limited temporal and spatial extents of the survey, lack of accounting for variation in SAV structure and possible confounding effects between SAV and fetch.

An earlier study by Randall et al. (1996) also used electrofishing and visual assessment of SAV bottom cover in five locations in Southern Ontario to examine relationships between fish production and habitat features. A total of 31 fish species were captured during the surveys and richness was significantly lower in absent or sparse (1-30% cover) SAV compared with moderate (31-70%) or dense (>70%) cover sites. Similar results were obtained for fish abundance, with dense SAV sites having 2.3 times higher fish abundance compared to sites with sparse SAV. Although fish biomass (mean size x density) was not significantly different among the SAV categories, significantly smaller fishes were found in dense compared to sparse SAV. The production index was significantly different across SAV categories with mean $\log_{10}$ PI values of 2.04 (absent); 2.24 (sparse); 2.59 (moderate) and 2.57 at dense SAV cover.

Bryan and Scarnecchia (1992) examined the difference in larval and juvenile fishes between natural and developed sites over two years in 2,226 ha Spirit Lake, Iowa. The developed sites were similar to the natural sites, but had reduced macrophyte species richness and relative abundance. In water depths ranging from 0 to 2 m, both species richness and total fish abundance was greater in the natural sites, with juvenile fish abundance higher in natural sites for 18 of the 20 fish species collected. Juvenile fish abundance was typically greatest in those areas where macrophyte species richness and abundance was highest. They also noted that larvae including Walleye (*Stizostedion vitreum*), YP and BG were abundant in the open waters of the lake while other species such as Northern Pike (*Esox lucius*), Smallmouth Bass (*Micropterus dolomieui*) and LMB remained inshore as larvae.

Two literature reviews have been conducted detailing fish-macrophyte interactions (Dibble et al., 1997; Smokorowski and Pratt, 2007). The literature review conducted by Dibble et al. (1997) found that many of the macroscale studies concluded that moderate levels of macrophyte abundance are optimal for fish feeding, growth, reproduction and diversity. Low SAV abundance limits the food supply while high abundance reduces foraging success of both benthivorous and piscivorous fishes. In much of the literature, "cover" is used in terms of areal extent rather than density. They defined intermediate level as 10-40% of areal coverage of the study site, which ranged from entire water bodies to individual embayments. Other key points from the literature review include the decline in abundance of pelagic species and benthic omnivores with increasing macrophyte areal extent and that stunted growth may result when the entire water body was covered with macrophytes due to depleted food resources. They also noted that some studies reported shifts in the fish communities with changes in macrophyte areal extent, while other studies found little effect or conflicting results. They suggest that further microscale studies of fish response to macrophyte morphometry and spatial distribution be conducted to determine fish population responses at a macroscale.

In 2007, Smokorowski and Pratt conducted a review and meta-analysis on observational and experimental studies of the freshwater fish habitat literature. They examined both lentic and lotic studies that either added or removed habitat including vegetation, woody debris and boulders. Increases in habitat complexity across studies provided mixed results, with either increases, decreases or no change in either fish species and/or communities. They found a negative effect on fish diversity and possible changes in species composition with decreasing habitat complexity. The meta-analysis generally indicated a strong link between habitat and fish biomass and a weaker response with fish abundance. They noted that there are a number of factors that can affect fish species response to habitat change, which include geographic region, temperature, season, life stage, level of predation and competition, and the relative availability of fish habitat. They indicated that population level changes in fish communities may be difficult to detect because stressed fishes can make compensatory changes in growth, fecundity, age-at-maturity and sex ratio.

In conclusion, based on the papers reviewed, it appears that much of the experimental evidence suggests that intermediate levels of macrophyte cover may be optimal for fish foraging. Many of the large scale studies used the term macrophyte cover to denote areal extent rather than density, or have determined macrophyte density differently than used for this BUI. There are mixed results from these studies and several indicate that results can be species and age specific and there may be other confounding factors to consider.

Appendix B: Description of the 2015 Bay of Quinte SAV Model

Predictive models were created based on field data collected by DFO-GLLFAS in the Bay of Quinte from 1972 to 2011. Data collection methods varied over the vegetation time series: from 1972 to 1985 knotted-line transects were used; paper-trace echo-sounders with quadrat sampling were used from 1988 – 2000; and from 2004 to 2011 sampling included digital hydroacoustic soundings and rake tosses. In the upper bay, sampling occurred along a reference transect perpendicular to the shoreline on the north and south shores at Trenton, Belleville and Big Bay. Sampling started at the nearshore at approximately 0.5 m water depth and extended at least 90 m beyond the edge of the SAV bed. Only data along the same reference transects from 1972 to 2011 were used to construct the model. Verification datasets consisted of transects that were parallel and at a distance of 250 and 500 m from the reference transect, but sampling on these transects only began in 1988. When time permitted, additional sampling with the digital hydroacoustics occurred both in the general area of the transect (particularly at Trenton in 2007) and in other more distant areas such as Muscote Bay. The number of data samples increased dramatically in later years, which affected data analysis but also increased confidence in estimates. All data were treated as point samples recording water depth and percent cover.

Table A1. Summary of data that were used to generate and validate the SAV model. The first letter (or in some cases two letters) in the transect acronym is the location: T = Trenton, B = Belleville, BB = Big Bay, MB = Muscote Bay. The last letter indicated if the transect was perpendicular to the north (N) or south (S), shore. IND = a transect around Indian Island in the Trenton area. Refer to Figure 1 for a map of transect locations.

Year	Transect Locations	Model Generation	Model Validation
1972	TN, TS, BN, BS, BBN, BBS	81	6
1973	TN, BN, BBN	29	3
1974	BS	7	1
1979	TN, TS, BN, BS, BBN, BBS	29	6
1982	TN, TS, BN, BS, BBN	29	5
1985	TN, TS, BN, BS, BBN	29	5
1988	TN*, TS*, BN*, BS*, BBN*, BBS*	47	45
1994	TN*, TS*, BN*, BS*, BBN*, BBS*	10	54
2000	TN*, BN*, BS*, BBN*, BBS*	23	75
2004	TN*, TS*, IND*, BN*, BS*, MB*, BBN*, BBS*	2485	11114
2007	TN*, TS*, T*, BN*, BS*, MB*, BBN*, BBS*	1625	19217
2011	TN*, TS*, BN*, BS*, BBN*, BBS*	2838	4658

* validation data includes parallel transects that are 250 and 500m away from the reference transect

Independent variables tested for the model included annual seasonal mean Secchi depth, total phosphorus and ePar (the light extinction coefficient for photosynthetically active radiation; 400 – 700 nm bandwidth), which were selected based on the Belleville long-term monitoring station. Slope, substrate and fetch were extracted from raster data using ArcGIS Spatial Analyst and also tested in model generation but were not statistically significant variables in determining vegetation presence, and were therefore excluded from the model. The lack of significance of some independent variables is specific to the Bay of Quinte model and is likely because the range of those variables in the bay is not great enough to have an influence (e.g. the fetch is relatively low throughout the bay because it is predominantly sheltered).

The Bay of Quinte SAV model is a two stage model; the first stage is a logistic model that predicts SAV presence/absence and the second stage model predicts SAV density at those sites where SAV is predicted to be present.

Predicting SAV presence

There were considerably more records, and at a much higher spatial resolution (<3m) in the later datasets because they were collected with digital hydroacoustics. To resolve this oversampling issue, data were subsampled with 20 records from each 0.5m depth bins of the 2004, 2007 and 2011 datasets. The logistic model was then constructed using the full 1972 to 2000 dataset and the subset of the 2004-11 data to avoid swamping of early trends due to large differences in sample size (Table A1). For depth bins without 20 records, all available records were used in the logistic model generation. To capture the best model equation and to randomize the subsampling of oversampled years, the final model was constructed based on parameter means of 10,000 repetition runs. The model predicts the probability of SAV presence given depth (m) and water clarity, as measured by Secchi depth (m). When cover probability is 0 or above, SAV is considered present, while cover probability below 0 is considered absent of SAV.

The final logistic model is described by this equation:

$$Cover_{prob.} = 13.201 - 7.488 * {}^{1}/Secchi - 4.700 * \sqrt{Depth}$$

(Naglekerke pseudo-R^2 of 0.41)

Predicting SAV density

Vegetation density has a high amount of variability across individual point samples but average trends were apparent in the data. To analyse temporal trends, percent cover was averaged within 1-m depth classes, by bay and by year (each unique bay-year combination had a unique Secchi value; only upper bay is included in this report). Using the logistic cover probability model, maximum depth at which vegetation should colonize was calculated by assuming 0 probability cover at maximum depth (Figure A1).

SAV density was determined by the second stage model, but only in those areas where SAV presence is predicted in stage one. The density model was developed by examining the variation in SAV density (% bottom cover) at one metre depth bin intervals for each of the surveys. The density model is bounded by two variables; the first is the maximum SAV density ($\%Cover_{Max}$) that was achieved. The second variable is the maximum depth of SAV colonization (Z_c), which was the maximum depth that SAV presence was predicted by the stage one model. Both of these variables were determined by the relationship between SAV % cover and Secchi depth. Examination of the binned survey data for each year determined the range of depths that the $\%Cover_{Max}$ would be applied, and then a linear slope was used to intercept at Z_c (Figure A1).

Although variable, the mean depth class at which $\% Cover_{Max}$ usually occurred was between 1-3 m depending on the time stanza. Hence, mean maximum percent vegetation cover ($\%Cover_{max}$) for 0 to 3m (Figure A1) can be calculated using the following equation:

$$\%Cover_{Max} = 75.86 * \ln(Secchi) + 29.73 \quad (N=12 \text{ bay-years}, R^2 = 0.33)$$

Percent cover at depths deeper than 2-m declined linearly on average from the mean maximum cover to zero at Z_c. However, mean peak SAV colonization at depth shifted from 2-m to 3-m depth between the Post-P and Post-D time stanza (Figure A4). Therefore, a 2-step density model was used to represent SAV density, where maximum cover was applied over different depth ranges between Post-P and Post-D time stanzas. Depths over 1.5-m and 2.5-m $\%Cover_{max}$ can be inferred from a linear regression equation assuming 0 % cover at Z_c (Figure A1) for Post-P and Post-D time stanzas respectively:

For Post-P time stanza:

$$Slope = (0\% - \%Cover_{Max}) / (\text{maximum depth} - 1.5m)$$

For Post-D time stanza:

$$Slope = (0\% - \%Cover_{Max}) / (\text{maximum depth} - 2.5 m)$$

The models were run within a GIS on a 3 x 3 m grid cell size, which covers an area of 156.6 km^2 (17933602 cells) in the upper bay when all cells are submerged in water. Mean annual May to September Secchi depths recorded at the Project Quinte Belleville station were used. Depth was based on the IGLD 1985 bathymetry, with annual adjustments for water level changes based on data from the Kingston water level station. Finally, predictions of SAV density using the above models were combined and area (km^2) and percentage of total area for Post-P and Post-D periods were predicted for the entire upper Bay for temporal / spatial SAV trends (Table 1; Figure 2-3)

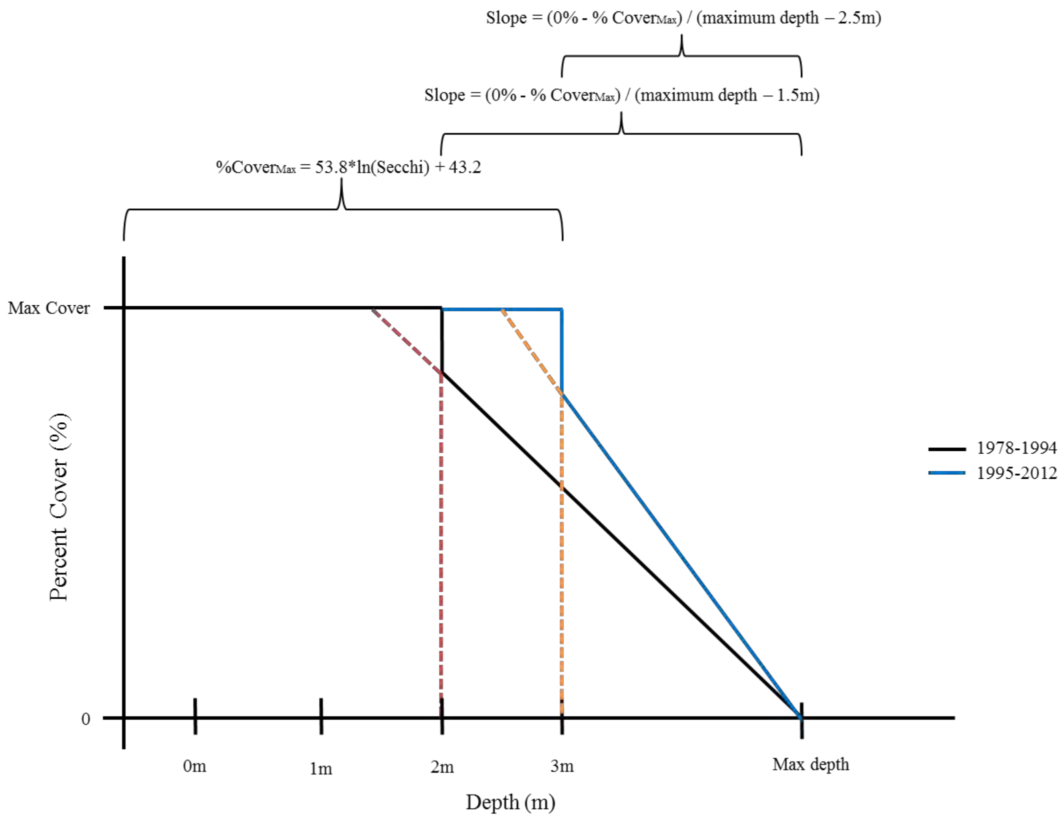


Figure A1. Conceptual illustration of percent cover prediction for Bay of Quinte submerged aquatic vegetation (SAV) based on the logistic cover model and the density model. Percent cover were considered at maximum (Max Cover) for depth from 0-2 m in the Post-P stanza (1978-1994) and depth from 0-3m for the Post-D time stanza (1995-2012). Percent cover from >1.5 m and >2.5m decline linearly to no cover (0%) at maximum depth for Post-P and Post-D time stanzas, respectively. Maximum depth for which vegetation should colonize for individual year * site combinations were calculated using the logistic model based on light penetration (Secchi depth).

Appendix C – Model validation and other scenario considerations

Logistic model validation

To verify the logistic model in predicting SAV in the Bay of Quinte, an independent dataset of % SAV bottom cover was extracted from the original SAV transect data from 1972 to 2000, 2004, 2007 and 2011. This dataset included a good range of depth classes and more than one transect per location across the entire upper bay and was not used in generation of the logistic model. Historic SAV transect data (1972 to 2000) was in a lower spatial resolution due to differences in sampling techniques, therefore only samples from the end of the vegetation bed were extracted for this analysis as a reference. A total of 200 sample points were extracted from the 1972 to 2000 independent database, while a total of 11114, 19217, 4698 sample points were available for validation for 2004, 2007 and 2011, respectively. Predicted presence or absence of SAV derived from the SAV logistic model was then compared to the validation dataset at a by year, by location/transect basis. The results were summarized in contingency tables by number of correct and incorrect predictions (Table A2, A3).

Resulting correct predictions from the SAV model varies across location and years (Table A2). Correct prediction of SAV presence and absences with more than 5 validation data points ranged from 38 to 85%, 81 to 94% and 64 to 92% for Big Bay, Belleville and Trenton, respectively. Mean total % of correct presence and absence predictions for these locations were 77%. Belleville showed the highest % of correct SAV presence/absence predictions, while Trenton showed the lowest overall correct predictions from the logistic model (Table A3). Other transects including Trenton Channel, Trenton Mid, Muscote Bay and Indian Island for surveys in 2004 and 2007 indicated that total % correct predictions ranged from 29 to 71% (Table A3). Overall, the logistic model can predict SAV cover in Big Bay and Belleville locations, but does a moderate job of predicting SAV cover in Trenton locations.

The under prediction at Trenton locations can be due to a number of factors. In the logistic model, water clarity (Secchi depth) was assumed to be uniform for each year across the entire upper bay due to limited local data and relied on the offshore Project Quite data collected at Belleville. Therefore it is possible that locations in Trenton can be affected by a local increase in water clarity that was not well represented in the model. In addition, areal extent of SAV in Trenton was likely expanding from the nearshore areas into the entire Trenton bay over time due to an increase in water clarity and its gently sloping, relatively shallow overall depth. As a result, while SAV at Trenton may be well established in the nearshore, SAV cover in the middle of the channel can be patchy and more transient. In contrast, Big Bay and Belleville tend to better capture SAV cover since both locations have fringing SAV beds in the shallow nearshore environment and a more rapid descent into deeper water depths. Therefore, there is a better defined edge to the SAV beds at these locations, which tends to be less variable over time than at Trenton. Finally, moderate predictions of SAV in Trenton may also be affected by the low variation explained by the logistic model, which has a Naglekerke pseudo-R^2 of 0.41.

Table A2. Contingency table of logistic SAV model presence/absence predictions versus validation dataset by location and year. The number of correct predictions for point data is indicated by "✓" and incorrect predictions are indicated by "X".

	Location													
	Big Bay		Belleville		Trenton		Trenton Channel		Trenton Mid		Muscote Bay		Indian Island	
Year	✓	X	✓	X	✓	X	✓	X	✓	X	✓	X	✓	X
1972	2	0	2	0	0	2								
1973	1	0	1	0	1	0								
1974			1	0										
1979	2	0	1	1	1	1								
1982	1	0	1	1	2	0								
1985	1	0	2	0	2	0								
1988	3	5	15	1	17	4								
1994	7	4	14	1	22	6								
2000	24	8	25	6	11	1								
2004	1002	182	1409	281	1403	781	418	318			1727	807	1728	1058
2007					8083	3784			704	1745	2988	1913		
2011	337	174	952	183	2209	803								
Total	1380	373	2423	474	11751	5382	418	318	704	1745	4715	2720	1728	1058

Table A3. Contingency table of logistic SAV model presence/absence predictions versus validation dataset by location and year similar to Table B1, but here expressed as percentages (%). The % of correct predictions for point data is indicated by "✓" and incorrect predictions are indicated by "X".

	Location													
	Big Bay		Belleville		Trenton		Trenton Channel		Trenton Mid		Muscote Bay		Indian Island	
Year	✓	X	✓	X	✓	X	✓	X	✓	X	✓	X	✓	X
1972	100		100			100								
1973	100		100		100									
1974			100											
1979	100		50	50	50	50								
1982	100		50	50	100									
1985	100		100		100									
1988	38	63	94	6	81	19								
1994	64	36	93	7	79	21								
2000	75	25	81	19	92	8								
2004	85	15	83	17	64	36	57	43			68	32	62	38
2007					68	32			29	71	61	39		
2011	66	34	84	16	73	27								
Total	79	21	84	16	69	31	57	43	29	71	63	37	62	38

It should be noted that there are many other factors that may influence vegetation but cannot be taken into account by the 2015 statistical models developed for predicting SAV cover in the Bay of Quinte. Exposure is known to affect the density or distribution of SAV, yet the historic transect locations do not represent a wide range of fetch. Other biotic factors not included in the model include uprooting of vegetation by carp, consumption of aquatic plants by waterfowl, competitive exclusion between plant species, infestation or disease and loss of plant propagules due to exposure or burial. Abiotic factors that affect SAV cover and extent include substrate type, wind and wave exposure, ice scour, changes in temperature (degree days), periodicity of drying, and water or sediment chemistry (Barko et al., 1986; Chambers, 1987; Mortsch, 1998). Also, direct human-induced changes to vegetation would include boat traffic, transfer of invasive species, and the chemical or mechanical control of SAV.

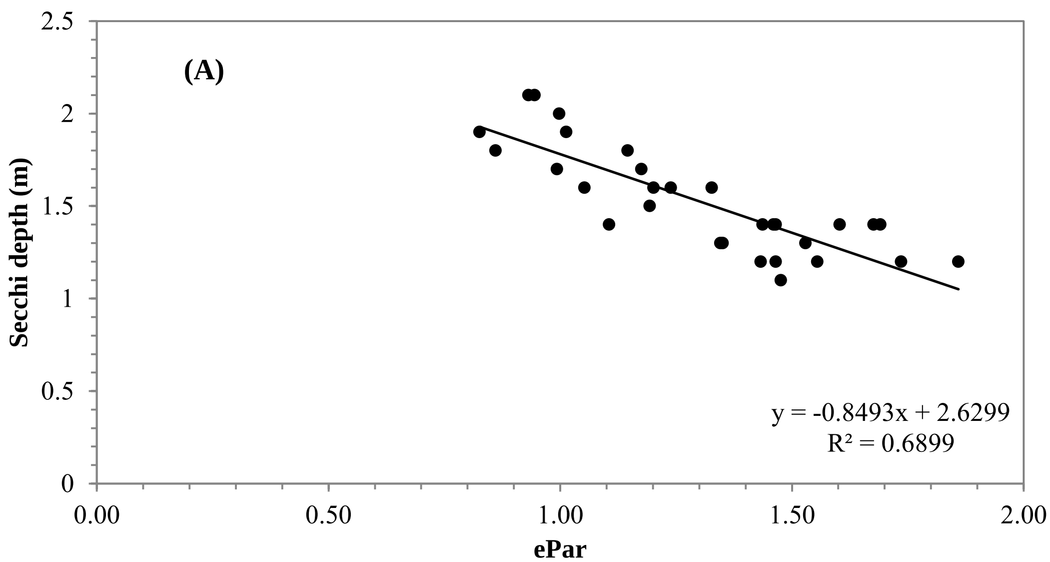
Density model validation

The density model in the 2 step modelling approach for the upper bay predicts the maximum density given water clarity data. Light extinction (ePar) has been known to be a good indicator for predicting macrophyte growth (Asaeda et al., 2000). However, ePar data was not readily available for all sites and years, therefore water clarity or Secchi depths were used instead in our modelling approach. To verify this, we examined the relationship between ePar and Secchi depth for all available years. Figure A2-A showed a strong negative linear relationship between mean ePar and Secchi depth in the upper bay. When compared to the modelling results, a strong negative linear relationship was found between ePar and % area SAV for 1 to 50% density (Figure A2-B) for the Post-P time stanza. In the Post-D time stanza, a strong to moderate negative relationship was found between ePar and % area SAV for 1 to 50% density and total SAV (Figure A2-C). In contrast, a strong positive relationship was found between ePar and % area with no SAV in both Post-P and Post-D time stanzas (Figure A2-B, C).

In addition, we further verified the SAV density model with the validation data from the upper Bay. Percent density for validation transect datasets from 1972-1994, 2000, 2004, 2007 and 2011 were separated into 0.5m depth bins and compared with the modelled data (Figure A3). Due to a difference in historical sampling techniques, validation data for historical periods (1972-1994) were combined into one period, with the exception of 2000. This is to capture the possible shift in maximum SAV at depth between the Post-P and Post-D time stanzas.

Mean percent density differences between validation datasets and modelled results were -5.99%, -6.21%, -11.25%, -12.09% and -17.07% for 1972-1994, 2000, 2004, 2007 and 2011, respectively across all depth bins (Figure A3). The density model generally under predicted SAV % density in the Post-D time stanza. However mean percent density at depth generally increased over time in the validation dataset, which indicates colonization of SAV in deeper depths in more recent time periods.

In addition, there is a general over prediction of nearshore SAV for 2004, 2007 and 2011. The scarcity of nearshore SAV in the validation dataset can be explained by a possible increase of emergent or floating aquatic vegetation that was not captured in the model because of sampling bias. Other nearshore effects that could not be included in the model included: nearshore algal blooms, exposure and local variation in water clarity. From the validation results, one can conclude that the density model was more conservative in predicting areal extent of % SAV density in the upper bay especially in recent years.



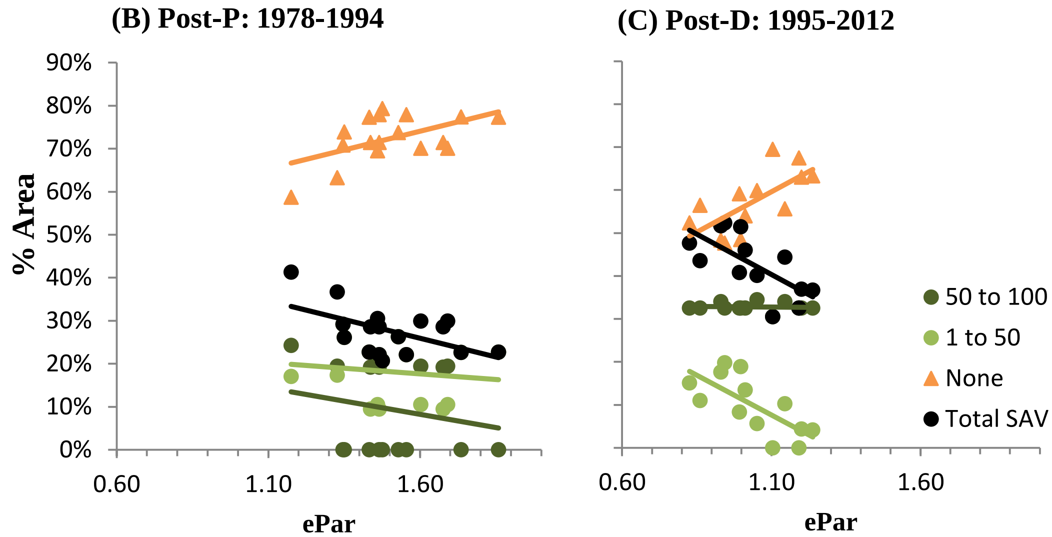
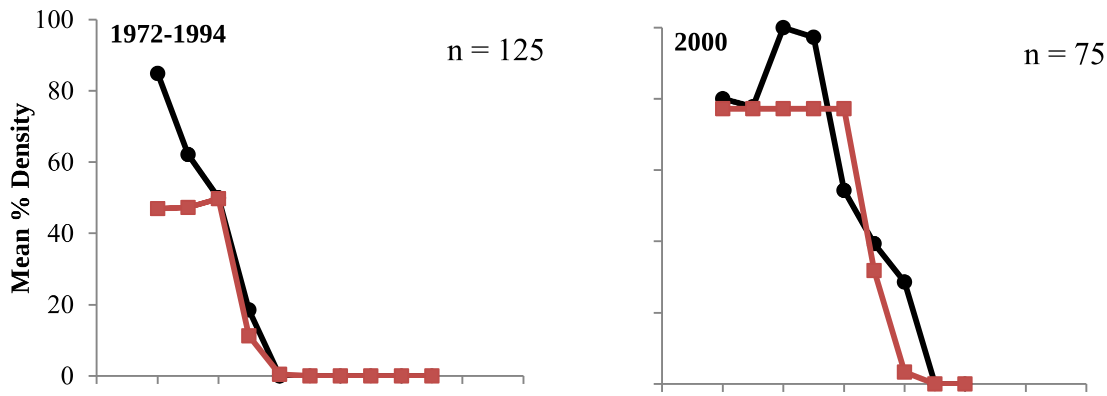


Figure A2. a) Linear relationship between mean ePar and Secchi depth (m) in Bay of Quinte upper Bay transects from 1978-2012. b) Linear relationships between ePar and percent (%) area of different submerged aquatic vegetation (SAV) densities in the Bay of Quinte upper bay for Post-P time stanza. Linear regression R^2 values are as follows: 0.69, 0.64, 0.20 and 0.29 for No SAV, 1 to 50% SAV density, 50-100% SAV density and total SAV, respectively. c) Linear relationships between ePar and percent (%) area of different submerged aquatic vegetation (SAV) densities in the Bay of Quinte upper bay for Post-D time stanza. Linear regression R^2 values are as follows: 0.69, 0.64, 0.21 and 0.47 for No SAV, 1 to 50% SAV density, 50-100% SAV density and total SAV, respectively.



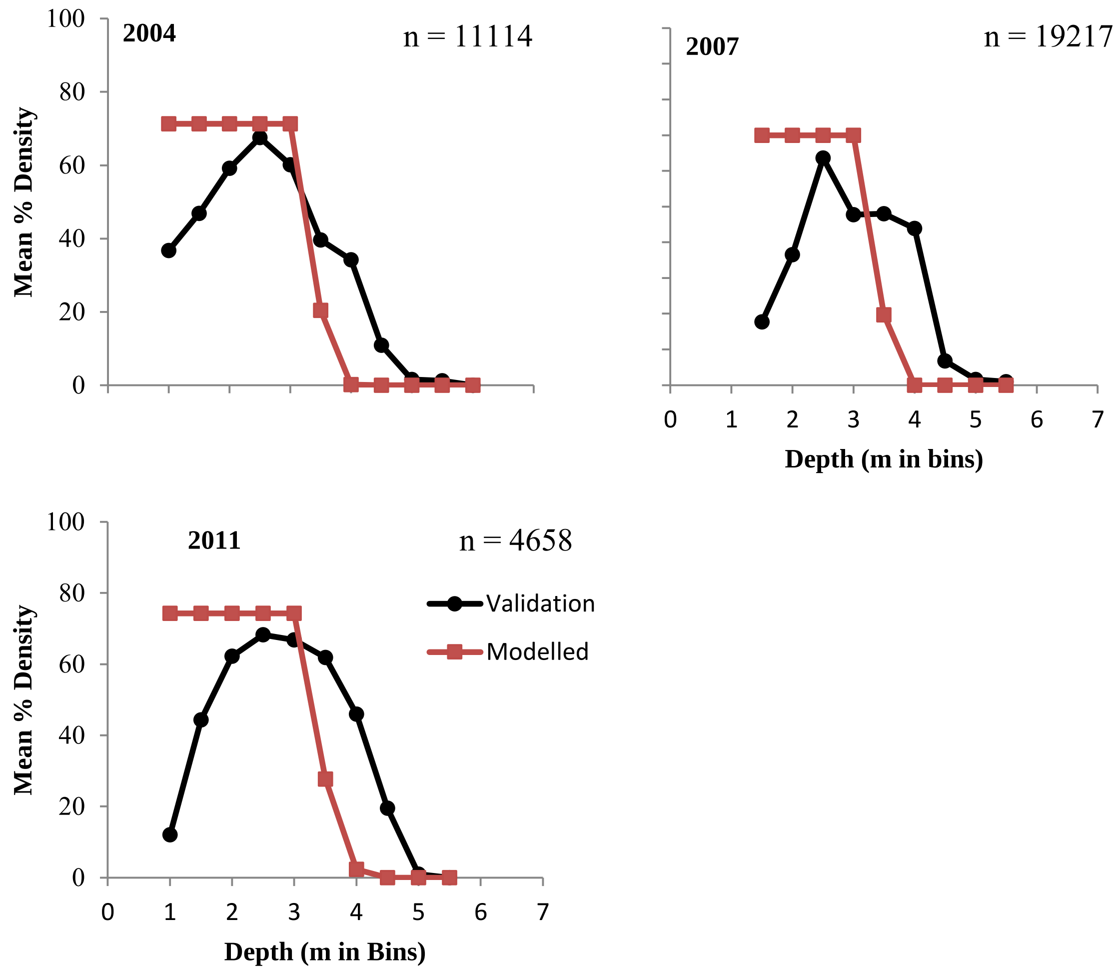


Figure A3. Mean percent density of validation data (black line) and modelled data (red line) separated in 0.5m depth bins for 1972-1994, 2000, 2004, 2007 and 2011 datasets.

Adjusted SAV density scenarios:

To further account for the variation in % density by depth between modelled results and validation results as well as to visualize possible nearshore under prediction of SAV density due to differences in sampling methods, we applied the mean % density difference by 0.5m depth bins as an adjustment factor to the 2-step density model (Figure A4). As such, mean % density difference by depth between modelled data and validation data was calculated separately for Post-P and Post-D time stanzas (Figure A4, Adjusted Scenario). Due to changes in sampling method over time, the sample size of validation data in the Post-D time stanza (n = 35064) was considerably larger than in the Post-P time stanza (n = 125).

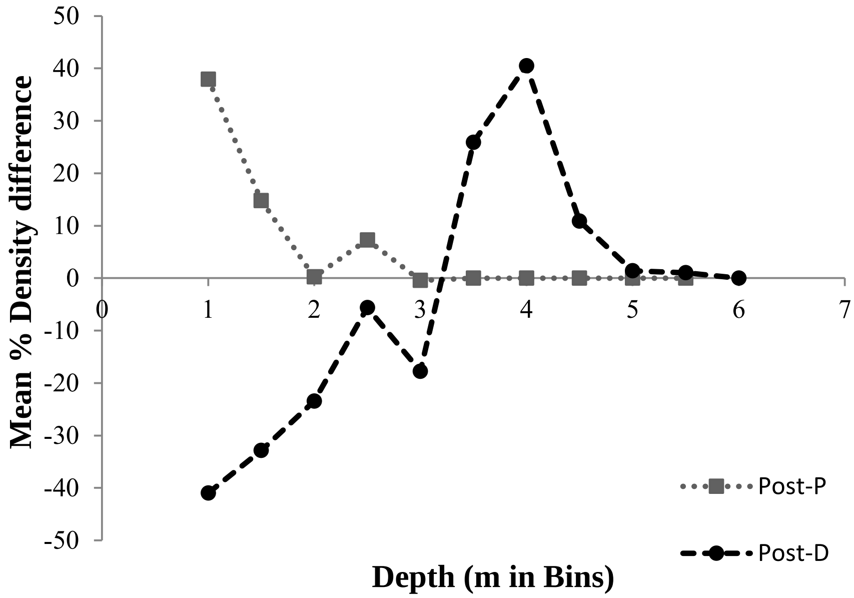


Figure A4. Mean percent (%) differences (validation – modelled) by depth between modelled data and validation data for Post-P, and Post-D.

Mean model predicted SAV density from the Post-P stanza showed over predictions of SAV density between 1 to 2m depth (Figure A4). While in the Post-D stanza, modelled data showed under prediction of SAV density from 1 to 3m, and over prediction between 3 to 4 m. In the Adjusted scenario, mean % total area of 50 – 100% SAV cover in the Post-D time stanza was 5.09% short of meeting the 30% total area of dense vegetation target (Table A4). However, there was an overall increase in 1-50% SAV between the two time stanzas (Table A4).

We also further analyzed the 35-year time trends of SAV areal extent. In the Adjusted scenario, area of 50 - 100% SAV density showed a weak increasing trend overtime ($R^2 = 0.12$, $p = 0.04$). Similar to the modelled scenario, the Adjusted scenario was typically below 30% of the total area of the upper bay prior to 1995 (Figure A5). Post 1995, the variation can range from 1 to 44% of the total areal extent of the upper bay depending on year (Figure A5).

Table A4. Comparison of adjusted scenarios of 1 to 50% and 50 to 100% SAV cover for the upper Bay of Quinte (BQ) for Post-P and Post-D time stanzas in modelled, Adjusted scenarios.

		Area(km^2)		% of Total Area		
% SAV Cover	**Scenario**	**Post-P**	**Post-D**	**Post-P**	**Post-D**	**% Change**
1 to 50	Modelled	25.54	13.45	18.23	9.75	-47.34
	Adjusted	18.19	25.01	13.00	18.13	37.49
50 to 100	Modelled	13.15	45.94	9.39	33.29	249.25
	Adjusted	20.40	34.37	14.58	24.91	68.49

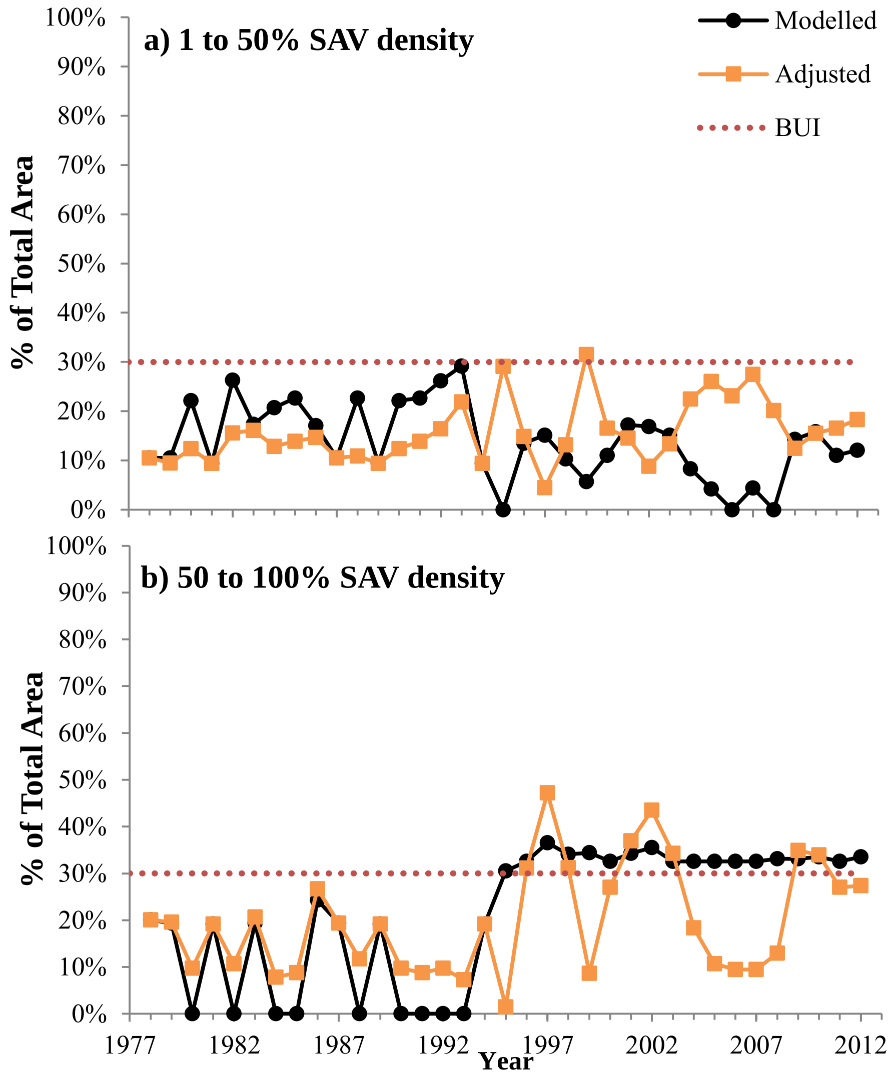


Figure A5. Variation in predicted percentage (%) of total wetted area by year covered by various classes of submerged aquatic vegetation (SAV) (a: 1-50% SAV density; b: 50-100% SAV density) in the Bay of Quinte's (BQ) upper bay area in the Post-P (1978-1994) and Post-D (1995-2012) stanzas. The red dotted line represents the BQ delisting target of 30% areal coverage of the 50-100% density class. Various scenarios were shown: The Modelled scenario represents the original modelled SAV with no alterations. The Adjusted scenario represents adjusted SAV density by depth from Post-P and Post-D validation data.

Coastal wetland data from Canadian Wildlife Service

To further examine under prediction of SAV in nearshore areas (1-2m) in our modelling approach, we examined the Canadian Wildlife Service – Ontario's Submerged Aquatic Vegetation database for upper Bay of Quinte coastal wetlands. The SAV community was surveyed by sampling one metre square quadrats at 20 random locations in the open water basin of each wetland in the Bay of Quinte. Within each quadrat, total areal coverage and species-specific coverage for submerged and floating- leaved vegetation were recorded. Coverage was determined at each % from 1 to 5% and then rounded to the nearest 5% for coverage higher than 5%, while water depth was estimated to the nearest cm or 5 cm by a water depth pole. Quadrats were sampled from 2003 to 2014 in the Post-D stanza (Figure A6). Some selected quadrats were removed from the final dataset due to sampling errors. Also, accurate depth measurements were not available for a number of quadrats with depths that exceeded the length of the water depth pole and these data were removed from the final dataset.

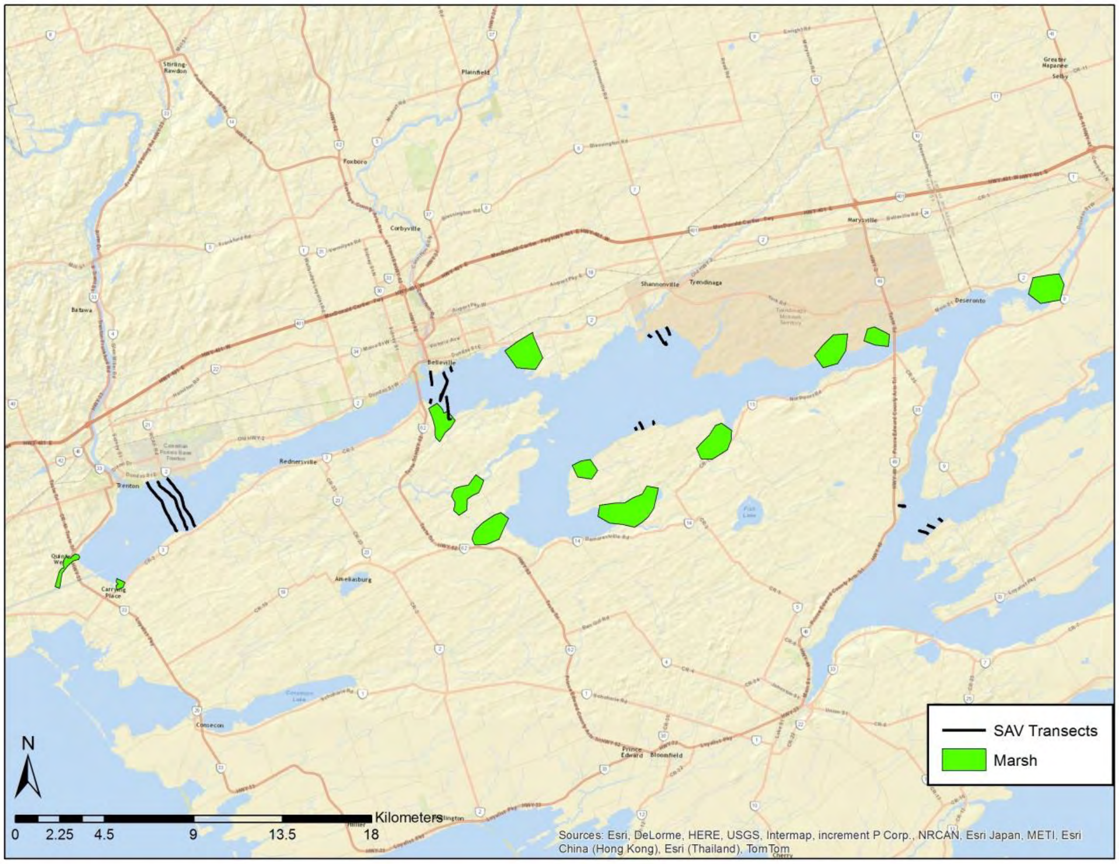


Figure A6. Sampled locations of the Canadian Wildlife Service – Ontario's Submerged Aquatic Vegetation database for upper BQ coastal wetlands between the years of 2003 to 2014 (green polygons) and SAV digital hydroacoustic transects from 2004 to 2011.

Figure A7 shows a comparison of mean % density by depth between validation, modelled and coastal wetland data (Figure A7). Across all years and all sites, total cover (%) of SAV from 0.5m to 2.5m was 46.0%, 71.7% and 77.3% for Validation, Modelled and Coastal Wetland data, respectively (Figure A7). Mean % cover from individual coastal wetland sites ranged from 40 to 100% between 0.5 to 2.5m depths (Table A5).

The abundance of high density SAV (50-100%) in coastal wetlands indicates that digital hydroacoustic transects used for generating the Bay of Quinte SAV model may not reflect SAV density of the entire upper bay in the Post-D stanza (Figure A6, A7). While the Bay of Quinte SAV model addresses this under estimation by applying a 2-step modelling approach between Post-D and Post-P time stanzas between depths of 1 to 3m, the Coastal Wetland data suggests that in relatively sheltered coastal wetland areas, mean % density was still higher than the modelled approach (Figure A7). Additionally, suspended or floating vegetation may also play a role in the under prediction by our model since these growth forms were not included in the model. Finally, factors like fetch and substrate type can play a role in determining SAV density in open areas, however it was not considered in the model since light penetration and depth were considered most influential in determining SAV density as discussed in the main report.

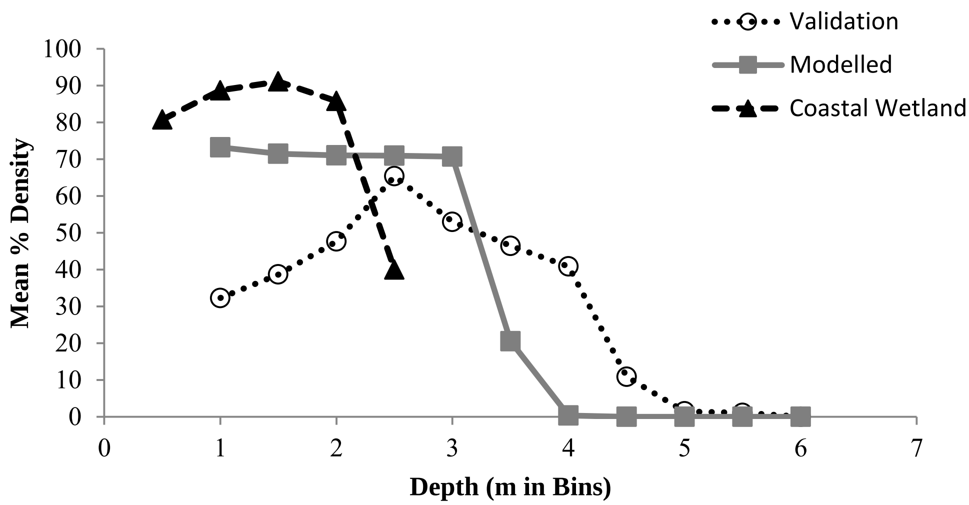


Figure A7. Mean percent (%) density across various depth bins (m) for Post-D validation data (2000 - 2011), modelled data (1995 – 2012) and Coastal Wetland SAV survey data (2003 – 2014)

Table A5. Contingency table of the number of total cover and mean percent (%) total cover by sampling sites per 0.5m depth bin of the Canadian Wildlife Service- Ontario's Submerged Aquatic Vegetation database for upper Bay of Quinte (BQ) coastal wetlands between 2003 to 2014 in the Post-D time stanza.

	Depth (m in bins)									
	Number of Total Cover					Mean % Total Cover				
Coastal Wetland Sites	**0.5**	**1**	**1.5**	**2**	**2.5**	**0.5**	**1**	**1.5**	**2**	**2.5**
Airport Creek Marsh	22	119	21	1		89.09	93.36	95.48	80.00	
Big Island East Marsh	6	88	80	3		96.67	93.18	98.25	100.00	
Big Island West Marsh	9	80	66	11	1	77.78	93.25	91.21	89.09	40.00
Blessington Creek Marsh	10	110	68	2		90.50	96.82	97.79	100.00	
Carrying Place Marsh	17	116	30			86.18	80.40	82.17		
Dead Creek Marsh	17	78	73	3		87.65	91.47	88.56	86.67	
Lower Napanee River Marsh	15	126	25			81.00	87.44	90.80		
Lower Sucker Creek Marsh	18	72	30	11		63.33	73.64	80.67	70.91	
Robinson's Cove Marsh	15	87	63	11		67.53	82.63	90.87	81.36	

Sawguin Creek Central Marsh	4	61	87	11		97.50	90.74	90.40	91.82
Sawguin Creek Ditched Marsh	9	56	90	9		67.78	90.38	85.44	92.78
Sawguin Creek North Marsh	17	95	69	5		81.06	90.16	94.78	82.00
Total / Mean % total Cover	159	1088	702	67	1	82.17	88.62	90.54	87.46

The 2-step density model

As part of the density modelling approach in optimizing the SAV density model, we examined the changes in SAV colonization at depth between the Post-P and Post-D time stanzas. Previously, a 1-step density model was used to predict SAV density at depth using mean maximum density conditions across the entire upper bay for the full time series. However, close examination of recent data showed that mean maximum density in upper bay had shifted to deeper depths in the Post-D time stanza (Figure A8), following an improvement in water quality. In the Post-P time stanza, mean maximum density percent peaked at 2 m with a linear decent to 4 m depth, while in the Post-D time stanza, the peak shifted to 3 m with a linear decent to 5 m depth (Figure A8). Therefore, it is apparent that a 2-step modelling approach should be applied to the density model to account for this change over the two time stanzas (Figure A1). Maximum density (%$Cover_{Max}$) was applied up to 2 m for the Post-P time stanza and 3 m for the Post-D time stanza, and slope of linear decrease in SAV was calculated from 1.5 m and 2.5 m for the Post-P and Post-D time stanzas respectively (Figure A1). Figure A9 showed a time trend comparison of percent total area of 50 to 100% SAV density for the 1-step and 2-step models. In general, a 2-step density model approach should better reflect the change in SAV colonization in deeper depths for the Post-D stanza.

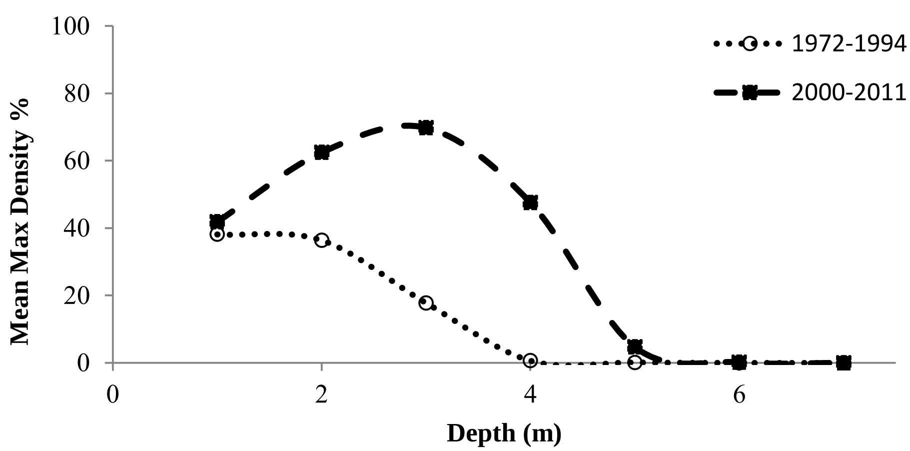


Figure A8. Mean maximum density from sampled transects by depth class with combined Post-P (1972, 1973, 1974, 1979, 1982, 1985, 1998, 1994) and Post-D (2000, 2004, 2007, 2011) time stanzas for the Bay of Quinte (BQ) upper bay.

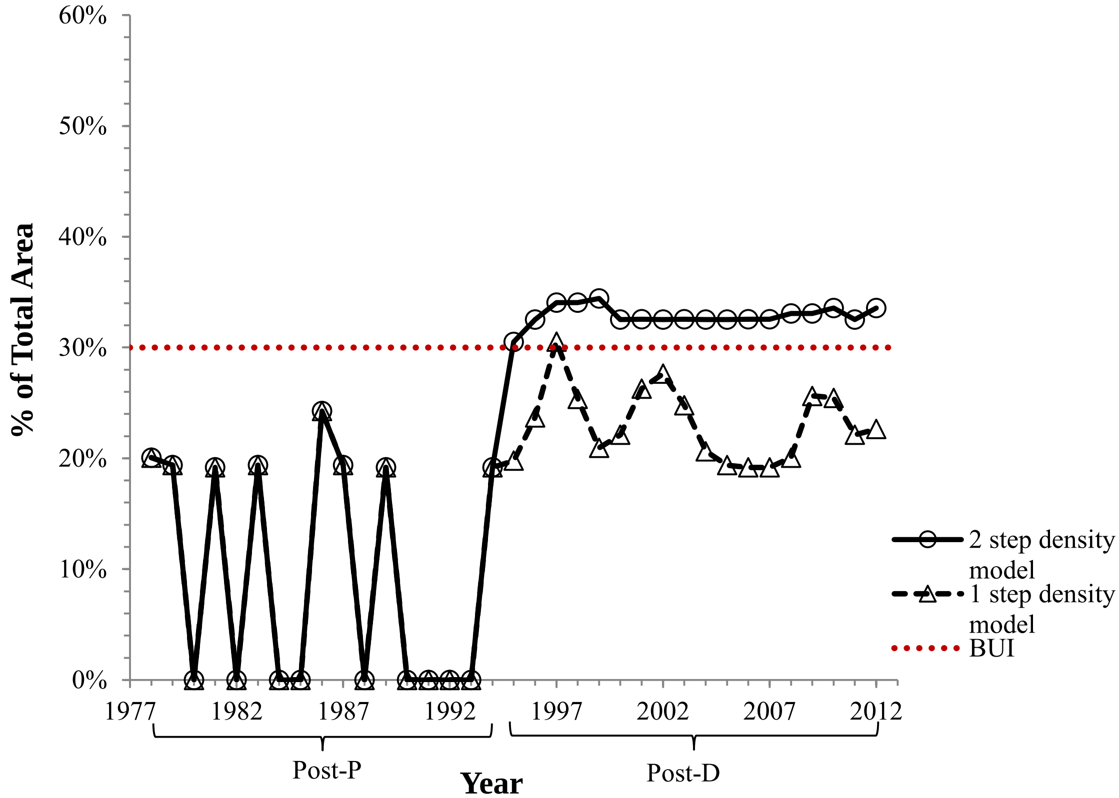


Figure A9. Difference in annual predicted percentage (%) of total area of 50% to 100% density submerged aquatic vegetation (SAV) using a 1 step and 2 step density model. The red dotted line represents the BUI areal target of 30% of total area of the upper bay

Defining Area of moderate to high density:

From the ESSA report, SAV areal expansion was estimated to achieve 30% coverage of the upper bay based on total phosphorus reduction and remediation efforts. The >50% density class was chosen with the understanding that expansion of beds with sparse SAV density will not be as beneficial to the fish community when compared to higher density beds (Crowder and Cooper, 1982; Wiley et al., 1984; Bryan and Scarnecchia, 1992; Randall et al., 1996). The literature review in Appendix A did not find consensus regarding the most appropriate level of SAV density for fisheries management. Since the >50% threshold could not be substantiated, we re-binned the model results to show the % of total area that would be predicted to have SAV at densities of either >40% and >30% (Table A6). The model predicts that the 30% total area delisting target will be exceeded by 4.5 and 6.4% respectively when >40 and >30 criteria are used. In addition, a general increasing trend can be observed in 30 to 100 and 40 to 100% dense SAV total % area, however this is mainly due to the inclusion of moderately dense SAV from previous density classifications (Figure A8).

Table A6. Comparison of various percent (%) dense SAV cover for the upper Bay of Quinte for Post-P and Post-D time stanzas.

SAV Cover	Area(km²) Post-P	Post-D	% of Total Area Post-P	Post-D	%change
50 to 100	13.2	45.9	9.4	33.3	249.3
40 to 100	26.9	47.6	19.2	34.5	76.8
30 to 100	29.8	50.2	21.3	36.4	68.4

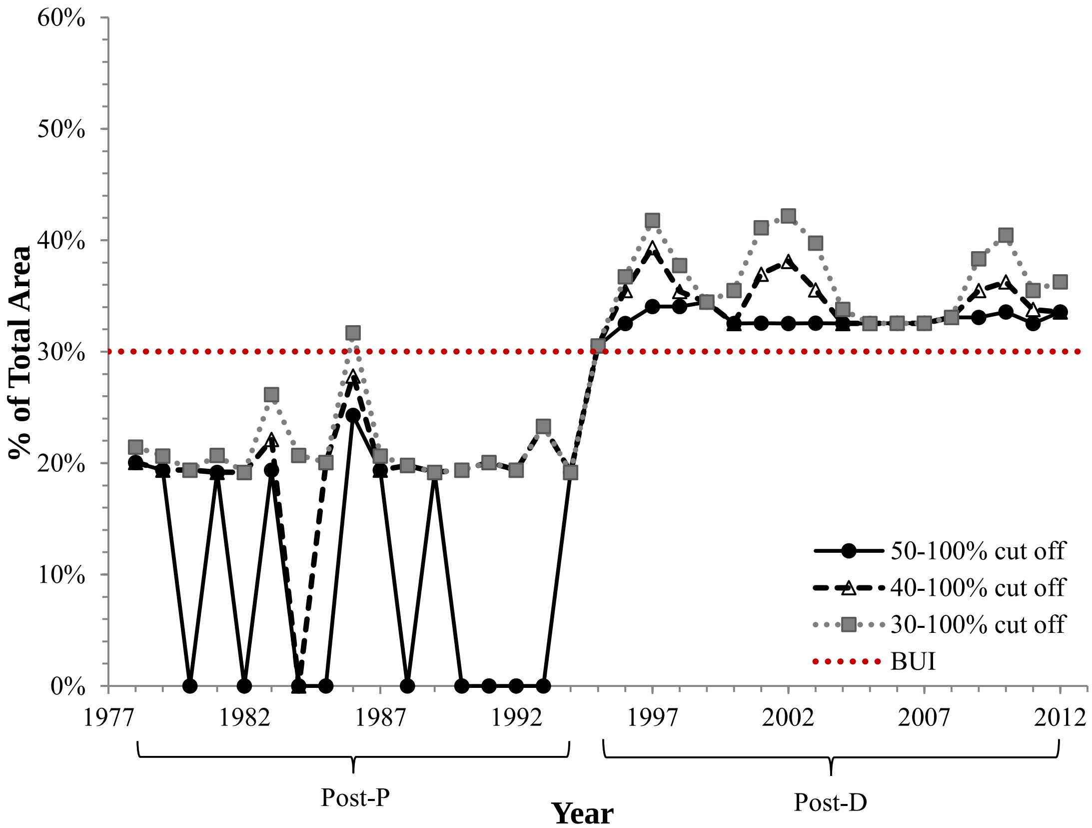


Figure A8. Predicted percentage (%) of total area covered by dense submerged aquatic vegetation (SAV) in Bay of Quinte upper Bay in various % SAV densities cut offs. The solid line (●), dashed line (⌂) and dotted line (■) represents % of total area with 50-100%, 40-100% and 30-100% SAV density respectively.

Modelled water clarity threshold

With recent concerns regarding the changing dynamics of water quality due to a number of ecological factors in the upper bay including: eutrophication, climate change and invasive species, it is important to project how the change in water clarity (Secchi depth) can impact areal cover of dense SAV in the upper bay. Using the Bay of Quinte SAV logistic and density model, we projected the change in SAV areal cover in the upper bay under a range of water levels and Secchi depths. The Post-D time stanza step was used for this analysis to reflect Post-*Dreissena* invasion conditions in the upper bay.

Growing season (April to September) water levels ranged from 74.80 to 75.45 m (mean = 75.02 m) and 74.70 to 75.12 m (mean = 74.91 m) relative to the International Great Lakes Datum – 1985 (IGLD85) for the Post-D and Post-P time stanzas, respectively. The Secchi depth for the upper bay ranged from 1.1 to 1.7 m (mean = 1.3 m) and 1.4 to 2.1 m (mean = 1.8 m) for Post-P and Post-D time stanzas respectively. Therefore, a range of 74.00 to 76.00 m was used for growing season water levels (at an increment of 0.5-m) and a range of 1.0 to 3.0 m (at an increment of 0.1 m) was used for the Secchi depth projections.

Figure A9-a shows the predicted % total area for dense vegetation at various water levels. Predicted % total area for SAV increased with increasing Secchi depths (Figure A9). Naturally, % total area with no SAV showed an opposite trend (Figure A9-c). For all water levels, dense SAV was not predicted with a Secchi depth of less than 1.4 m (Figure A9-a). Sparse SAV showed an initial increase in % total area with increasing Secchi depth but rapidly decreased approaching the 1.4 m Secchi depth threshold at which point densities reached 50% and were reclassified as dense SAV (Figure A9-b). In contrast, with increasing Secchi depth, % total area of dense SAV plateaus until Secchi depth reaches ~2.0 m based on model predictions.

One may expect that higher water levels should contribute to a higher % total area of dense SAV due to an increase in overall wetted area for SAV to colonize. However, from the predicted model, % total area of SAV decreased with increasing water levels (Figure A9-d). This is likely due to a shift in the bathymetric profile with increasing water levels. For example, an increase in the slope of the bathymetric profile at high water levels would cause a decrease in the areal coverage of SAV. Bathymetric profile is therefore a key factor that acts in concert with Secchi depth to determine the coverage of SAV. As noted previously, however, there are a variety of other factors that influence the presence and coverage of SAV that were initially included in the model but dropped after they were found to only moderately improve model fit (e.g., fetch, substrate type). Therefore, results of this modelling exercise reflect the potential areal coverage of SAV under different water level regimes, but the actual or realized coverage and density of SAV is likely more variable and linked to local environmental conditions such as substrate type and nutrient content, fetch or exposure, and grazing by waterfowl.

In conclusion, to meet and maintain the delisting criteria from FWH-1A of 30% area of the upper bay with SAV coverage of more than 50% density, water levels should remain between 74.0 and 75.5 m and Secchi depth be kept above the 1.4 m threshold.

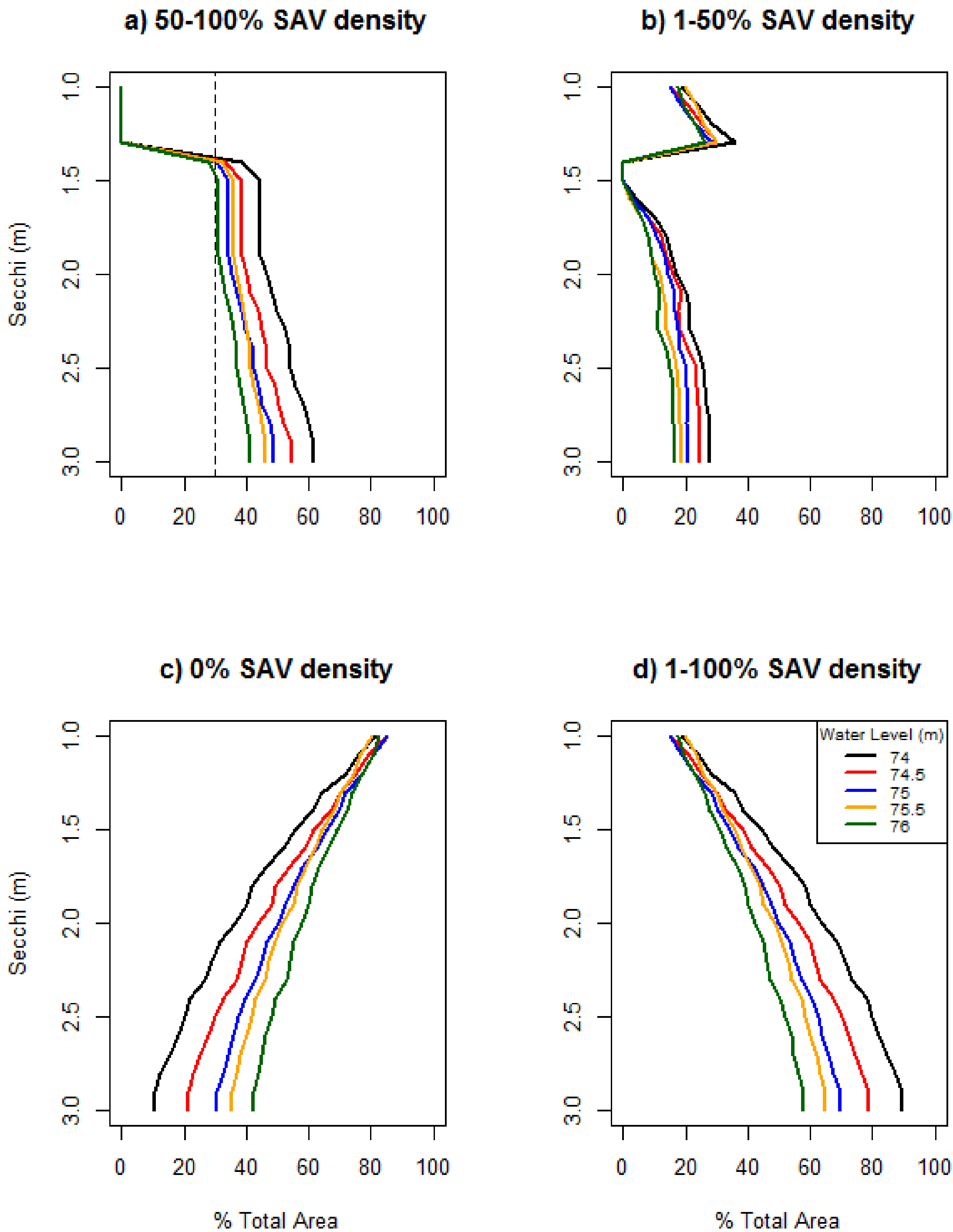


Figure A9. Predicted percentage (%) of total area covered by dense submerged aquatic vegetation (SAV) in Bay of Quinte upper Bay in various Secchi depths (m) and growing season (April to September) water levels (m) for a) dense SAV (50 - 100% density), b) sparse SAV (1 – 50% density), c) no SAV (0% density) and d) total SAV (1-100% density). The vertical dotted line in Figure A9-a represent the FWH-1A target such that 30% area of the upper BQ has SAV coverage of more than 50% density.

APPENDIX G: Coastal Wetland Submerged Aquatic Vegetation Delisting Criteria Status Report

October 22, 2015

Authors
- Angela Darwin, Landscape Assessment Officer, Canadian Wildlife Service, Environment and Climate Change Canada
- Greg Grabas, Habitat Ecologist, Canadian Wildlife Service, Environment and Climate Change Canada

Recommendation: Delisting criterion has been met

DELISTING TARGETS	DELISTING CRITERIA	STATUS
Submerged aquatic vegetation in Bay of Quinte coastal wetlands is not impaired.	The submerged aquatic vegetation community IBI at representative Bay of Quinte coastal wetlands shall not be more than two standard deviations below the 2006-2010 representative site mean that accounts for varying conditions in Lake Ontario outside of the AOC from 2006-2010.	Achieved

Background

Submerged aquatic vegetation (SAV) community evaluations have been completed for many years in near- and off-shore zones in the Bay of Quinte Area of Concern (Leisti et al. 2006). Continuing results from this work are discussed in the section for assessment criterion FWH-1 - *Extent of Submerged Aquatic Vegetation in the Bay of Quinte*. However, coastal wetland SAV assessments combined with continued assessments in other zones of the bay can provide a more complete representation of overall SAV community condition in the Bay of Quinte Area Of Concern.

Bay of Quinte-Remedial Action Plan Restoration Council (2007) indicated that Bay of Quinte Area of Concern coastal wetland SAV communities are an important component of the impaired beneficial use Loss of Wildlife Habitat. For example, Prince (1985) found that decreased aquatic vegetation coverage decreased the attractiveness of an area for nesting and foraging birds.

Submerged aquatic vegetation community evaluations were completed by randomly sampling 20 one-metre-square quadrats in the open water basin of each wetland. Within each quadrat, the total areal coverage plus species-specific coverage of each submerged and floating-leaved species was recorded. Index of Biotic Integrity (IBI) scores were calculated using metrics developed for Lake Ontario (Grabas et al. 2012):
- Number of turbidity-intolerant species [SINT]
- Coefficient of conservatism [CC]

- Total cumulative coverage [PCOV]
- Total number of native species [SNAT].

Site Selection and Surveys: The Bay of Quinte Remedial Action Plan Restoration Council (BQ-RAPRC, 2007) recognized that sampling and reporting on the status of delisting criteria for both fish and wildlife habitat and populations in all Bay of Quinte Area Of Concern coastal wetlands would not be feasible. A subset of 10 representative sites (coastal wetlands) was selected (Table 1). BQ-RAPRC (2007) indicated that the sites were chosen to cover a range of disturbance, size (small sites, <25 ha, were excluded), and dominant hydrologic influence (riverine/lacustrine). Environment Canada (2007) suggested that Solmesville Marsh was included in this list based on the belief that its size was 28 ha, but as the actual size was determined to be only 2.8 ha, Airport Creek Marsh (29 ha) was selected as the recommended alternative.

Since the representative sites were not finalized by BQ-RAPRC (2007) before the summer of 2006, EC-CWS, unaware that Carnachan Bay Marsh would be a representative site, did not conduct field surveys there during that field season. From 2007 to 2013, monitoring efforts have been coordinated by the BQ-RAPRC with assistance from EC-CWS and were focused on the 10 BQ AOC representative or sentinel wetlands, comprised of 14 subsites (Table 1). Although most representative sites are located in the Upper Bay of Quinte, this is indicative of the overall distribution of wetlands within the bay (Figure 1).

Table 1. Bay of Quinte Area of Concern representative coastal wetlands for monitoring

Wetland Name	Number of Subsites per Wetland
Airport Creek Marsh	1
Big Island Marsh	2 (East and West)
Blessington Creek Marsh	1
Carnachan Bay Marsh	1
Carrying Place Marsh	1
Dead Creek Marsh	1
Hay Bay Marsh	2 (North and South)
Lower Napanee River Marsh	1
Lower Sucker Creek Marsh	1
Sawguin Creek Marsh	3 (Central, Ditched, and North)

Additional coastal wetlands are surveyed in Lake Ontario through various projects such as the Durham Region Coastal Wetland Monitoring Project and the Coastal Habitat Assessment and Monitoring Program (Figure 1). The number of other Lake Ontario sites surveyed each year for submerged aquatic vegetation varies from a low of 13 in 2007 to a high of 22 in 2009 and 2011 (Appendix B, Table 3). Wetlands surveyed consistently across years are primarily in the Durham Region. Only wetlands with a minimum of two years of data were retained for inclusion in the analysis.

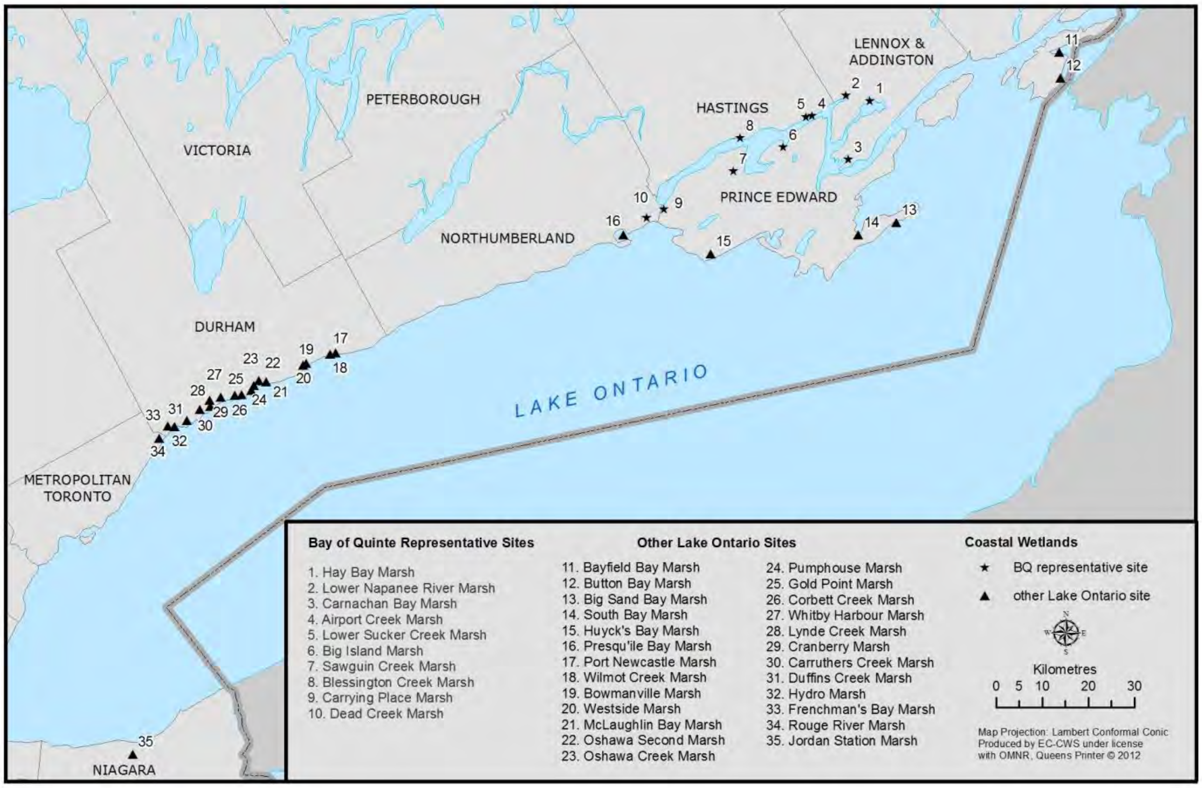


Figure 1. Names and locations of Bay of Quinte representative sites and other Lake Ontario coastal wetlands sampled by EC-CWS and partners. Refer to Appendix B for list of sites surveyed for submerged aquatic vegetation.

Rationale for Delisting Criterion Status

Determining the threshold below which the 2014 mean Submerged Aquatic Vegetation Community IBI (of all Bay of Quinte representative sites) must not fall to be considered not impaired requires multiple steps:

1. Calculate an annual mean from 2006 to 2010 submerged aquatic vegetation IBIs from all BQ representative sites
2. Calculate an overall mean and standard deviation of 2006 to 2010 IBIs from all BQ representative sites
3. Multiply the standard deviation by 2
4. Subtract the 2 standard deviations from the 2006-2010 mean
5. Calculate an annual mean from 2006 to 2010 IBIs from all 'other Lake Ontario' sites.
6. Calculate an overall mean of 2006 to 2010 IBIs from all 'other Lake Ontario' sites.
7. Calculate the 2014 mean from all 'other Lake Ontario' coastal wetlands surveyed
8. Subtract the 2006 to 2010 'other Lake Ontario' mean from the 2014 'other Lake Ontario' mean
9. Add the results of Step 4 and Step 8
10. The resulting number is the corrected threshold above which the 2014 mean submerged aquatic vegetation IBI for BQ representative sites must fall to be considered 'not impaired'.

Data used for these calculations are found in Appendix B (Tables 2 and 3).

2014 Bay of Quinte Representative Sites Mean IBI = 79.3

2014 Corrected Threshold IBI = 61.6

Since the Bay of Quinte 2014 mean IBI is greater than the threshold IBI value, the submerged aquatic vegetation community at Bay of Quinte coastal wetlands is considered to be **not impaired**. Therefore, the criterion for the submerged aquatic vegetation community to be considered not impaired has been met in each year from 2011 to 2014 (Appendix 2, Table 4).

Future Monitoring or Actions Required

Regular monitoring of the coastal wetland submerged aquatic vegetation community should continue, for the near future at least, at the Bay of Quinte representative sites and numerous other Lake Ontario coastal wetlands.

Glossary

Turbidity	A measure of the degree to which light traveling through a water column is scattered by the suspended organic (including algae) and inorganic particles measured in Nephelometric Turbidity Units (NTU).

References

Bay of Quinte RAP Restoration Council. 2007. A Management Plan to Address Wildlife Related Impairments for the Bay of Quinte Area of Concern. Prepared by P. Johanson and B. McNevin, March 2007.

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Appendix A: Submerged Aquatic Vegetation IBI Calculation

Step 1: Standardize the relative abundance and species richness metric scores to values out of 10. (Higher scores out of 10 indicate better submerged aquatic vegetation community attribute).

Metric	Code Calculation
SINT	If SINT ≥1.45, then the metric automatically scores a 10 If SINT <1.45, then multiply the value by 6.88
SNAT	If SNAT ≥ 5.52, then the metric automatically scores a 10 If SNAT <5.52, then multiply the value by 1.81
CC	If CC ≥5.27, then the metric automatically scores a 10 If CC <5.27, then multiply the value by 1.90
PCOV	If PCOV ≥131.6, then the metric automatically scores a 10 If PCOV <131.6, the multiply the value by 0.076

Step 2: Combine the standardized metrics and calculate an IBI out of 100.

- For each site, add the standardized metrics, divide the sum by 4 and then multiply by 10.
- For wetlands with multiple sites, a mean of the output values is generated.

Appendix B: Submerged Aquatic Vegetation (SAV) IBI values for 2006 to 2014

Table 2. SAV IBIs for Bay of Quinte representative sites for 2006-2014.

Wetland	2006	2007	2008	2009	2010	2011	2012	2013	2014
Airport Creek Marsh	93.8	84.6	88.2	92.2	88.9	97.1	79.8	85.2	80.2
Big Island Marsh	58.5	72.8	76.4	96.1	72.2	82.4	85.7	84.0	73.5
Blessington Creek Marsh	75.2	84.5	86.6	97.2	90.7	93.8	84.8	83.2	91.2
Carnachan Bay Marsh	-	76.8	63.6	98.2	78.1	84.0	78.5	85.5	80.6
Carrying Place Marsh	56.7	85.4	78.5	76.4	65.7	80.1	81.3	47.8	65.7
Dead Creek Marsh	74.3	71.5	82.5	82.2	68.7	75.0	85.0	73.3	75.2
Hay Bay Marsh	65.4	71.2	61.4	71.5	84.3	82.9	89.9	85.1	76.6
Lower Napanee River Marsh	64.4	91.4	86.3	97.8	89.4	85.3	85.4	93.4	94.8
Lower Sucker Creek Marsh	59.1	59.3	70.1	92.5	79.3	64.4	74.3	72.2	73.9
Sawguin Creek Central Marsh	59.4	81.0	84.9	94.1	82.0	83.8	88.9	94.0	81.2
Mean IBI	**67.4**	**77.9**	**77.9**	**89.8**	**79.9**	**82.9**	**83.3**	**80.4**	**79.3**
# wetlands	9	10	10	10	10	10	10	10	10

Table 3. SAV IBIs for all ‘other Lake Ontario’ sites for 2006-2014.

Wetland	2006	2007	2008	2009	2010	2011	2012	2013	2014
Bayfield Bay Marsh				73.1		71.6	80.8		
Big Sand Bay Marsh				61.5		57.2			
Bowmanville Marsh	18.4	12.1	13.5	12.1	16.8	26.1	10.0	17.4	18.3
Button Bay Marsh				61.5		53.8	80.6		
Carruthers Creek Marsh	7.2	5.2	2.0	9.7	0.0	0.0	1.7	1.2	3.4
Corbett Creek Marsh	28.8	32.6	35.1	38.5	39.0	30.2	42.4	34.7	46.1
Cranberry Marsh	11.1		53.4	34.5	36.8	36.2	48.5	38.3	40.9
Duffins Creek Marsh	12.2	2.1	21.7	6.0	32.1	29.4	25.5	32.4	23.5
Frenchman's Bay Marsh	9.4	18.8	8.2	33.0	24.9	35.2	42.0	22.2	22.4
Hydro Marsh	2.3	0.0	0.0	2.8	3.2	5.1	16.6	7.8	10.8
Lynde Creek Marsh	2.4	1.5	15.9	11.2	25.4	14.8	16.3	24.3	19.0
McLaughlin Bay Marsh	0.3	5.1	1.7	2.4	0.0	1.5	4.9	2.6	3.5
Oshawa Creek Marsh				4.6	2.5	1.2	9.0	5.2	1.5
Oshawa Second Marsh	65.7	70.5	63.1	37.9	55.4	25.4	36.8	38.9	43.0
Port Newcastle Marsh	14.5	8.8	35.9	17.1	14.1	7.9	3.9	11.0	21.6
Presqu'ile Bay Marsh				76.6		55.2	91.0		
Pumphouse Marsh	10.3		45.1	55.2	23.3	22.4	9.5	7.1	23.3
Rouge River Marsh	22.9	18.5	23.7	41.9	34.0	33.4	32.2	35.2	40.9
South Bay Marsh				57.3		74.5	69.0		
Westside Marsh	18.6	6.3	7.2	3.7	11.6	4.7	5.9	6.5	11.1
Whitby Harbour Marsh				39.3	16.1	22.5	17.7	7.9	16.0
Wilmot Creek Marsh	31.1	47.4	30.6	3.2	21.2	16.3	15.4	12.1	12.4
Mean IBI	**17.0**	**17.6**	**23.8**	**31.1**	**21.0**	**28.4**	**31.4**	**17.9**	**21.0**
# wetlands	15	13	15	22	17	22	21	17	17

Table 4. Mean SAV IBIs of Bay of Quinte (BQ), ‘Other Lake Ontario sites’, and corrected threshold values from 2011-2014.

Mean IBI	2011	2012	2013	2014
‘Other Lake Ontario sites’	28.4	31.4	17.9	21.0
Bay of Quinte IBI threshold	68.9	72.0	58.5	61.6
Bay of Quinte IBI score	82.9	83.3	80.4	79.3
Criterion met (Y/N)	**Y**	**Y**	**Y**	**Y**

APPENDIX H: Coastal Wetland Water Quality Delisting Criteria Status Report

October 22, 2015

Authors

- Angela Darwin, Landscape Assessment Officer, Canadian Wildlife Service, Environment and Climate Change Canada
- Greg Grabas, Habitat Ecologist, Canadian Wildlife Service, Environment and Climate Change Canada

Recommendation: Delisting criterion has been met

DELISTING TARGETS	DELISTING CRITERIA	STATUS
Coastal wetland water quality is not impaired.	The water quality index at representative Bay of Quinte coastal wetlands shall not be more than two standard deviations below the 2006-2010 representative site mean that accounts for varying conditions in Lake Ontario outside of the AOC from 2006-2010.	Achieved

Background

Water is an important habitat component of Great Lakes coastal wetlands and water quality has a strong influence on the overall aquatic habitat quality in these environments. Poor water quality compromises fish and wildlife habitat in several ways. First, and most obvious, water is the medium in which fish and many other wildlife species exist. Poor water quality can affect the behaviour and physiology of these animals and hence their ability to survive. Second, water quality affects food sources for fish and wildlife. Submerged aquatic vegetation and aquatic macroinvertebrates are impacted by poor water quality. Submerged aquatic vegetation also provides habitat for fish and wildlife by providing cover, shelter, and egg-laying substrate. For these reasons, Bay of Quinte Area of Concern (BQ AOC) coastal wetland water quality was specified as a component of BUI 3 (Bay of Quinte-Remedial Action Plan Restoration Council 2007). In addition, water quality data can help provide insight into observations linked to the abundance or richness of certain biotic guilds (e.g., submerged aquatic vegetation species richness; Chow-Fraser 2006).

Water quality in littoral and off-shore zones has been evaluated in the Bay of Quinte AOC for decades. The addition of standardized coastal wetland water quality assessments contributes to the understanding of the overall aquatic habitat quality in the Bay of Quinte AOC.

Water quality sampling stations were established within each wetland along the edge of the emergent vegetation and open water. Three to six replicates are sampled at each site. A water quality multimeter (e.g. Hydrolab Datasonde 5) was used to measure pH, water temperature (°C), conductivity (μScm^{-1}) and turbidity (NTU). Chow-Fraser (2006) developed a Water Quality Index (WQI) to report on the water quality in the context of all the Great Lakes. The WQI was developed from 12 water quality variables. In apparent recognition that most implementation agencies would not realistically be able to collect data for all 12 variables, eight additional WQIs were developed from subsets and combinations of the original 12 variables. These additional WQIs incorporate between four and seven water quality variables. The seventh WQI equation presented (see Chow-Fraser 2006; Table 5.6), using the parameters of Turbidity (TURB), Conductivity (COND), Temperature (TEMP), and pH, was best suited to variables collected by Environment Canada-Canadian Wildlife Service (EC-CWS) in the past (2001-2006). Mean annual WQIs were calculated for the sites using the following formula (Chow-Fraser 2006):

$$WQI=(-1.367148*\log TURB)-(1.57738*\log COND)-(1.628048*\log TEMP)-(2.371337*\log pH)+9.2663224$$

The WQI ranges between -3.00 and +3.00 and, as mentioned above, was developed for use in all the Great Lakes. Index values from the lower Great Lakes, Erie and Ontario, are very unlikely to attain values near the top of the range – only wetlands in Lake Superior/Huron are likely to score that high.

Site Selection and Surveys: Bay of Quinte Remedial Action Plan Restoration Council (BQ-RAPRC) (2007) recognized that sampling and reporting on the status of delisting criteria for BUIs 3 and 14 in all Bay of Quinte Area Of Concern coastal wetlands would not be feasible. A subset of 10 representative sites (coastal wetlands) was selected (Table 1). BQ-RAPRC (2007) indicated that the sites were chosen to cover a range of disturbance, size (small sites, <25 ha, were excluded), and dominant hydrologic influence (riverine/lacustrine). Environment Canada (2007) suggested that Solmesville Marsh was included in this list based on the belief that its size was 28 ha, but as the actual size was determined to be only 2.8 ha, Airport Creek Marsh (29 ha) was selected as the recommended alternative.

Since the representative sites were not finalized by BQ-RAPRC (2007) before the summer of 2006, EC-CWS, unaware that Carnachan Bay Marsh would be a representative site, did not conduct field surveys there during that field season. From 2007 to 2013, monitoring efforts have been coordinated by the BQ-RAPRC with assistance from EC-CWS and were focused on the 10 BQ AOC representative or sentinel wetlands, comprised of 14 subsites (Table 1). Although most representative sites are located in the upper Bay of Quinte, this is indicative of the overall distribution of wetlands within the bay (Figure 1).

Table 1. Bay of Quinte Area of Concern representative coastal wetlands for monitoring

Wetland Name	Number of Subsites per Wetland
Airport Creek Marsh	1
Big Island Marsh	2 (East and West)
Blessington Creek Marsh	1
Carnachan Bay Marsh	1
Carrying Place Marsh	1
Dead Creek Marsh	1
Hay Bay Marsh	2 (North and South)
Lower Napanee River Marsh	1
Lower Sucker Creek Marsh	1
Sawguin Creek Marsh	3 (Central, Ditched, and North)

Additional coastal wetlands are surveyed in Lake Ontario through various projects such as the Durham Region Coastal Wetland Monitoring Project and the Coastal Habitat Assessment and

Monitoring Program (Figure 1). The number of other Lake Ontario sites surveyed each year for water quality varies from a low of 10 in 2006 to a high of 24 in 2009 (Appendix A, Table 3). Wetlands surveyed consistently across years are primarily in the Durham Region. Only wetlands with a minimum of two years of data we retained for inclusion in the analysis.

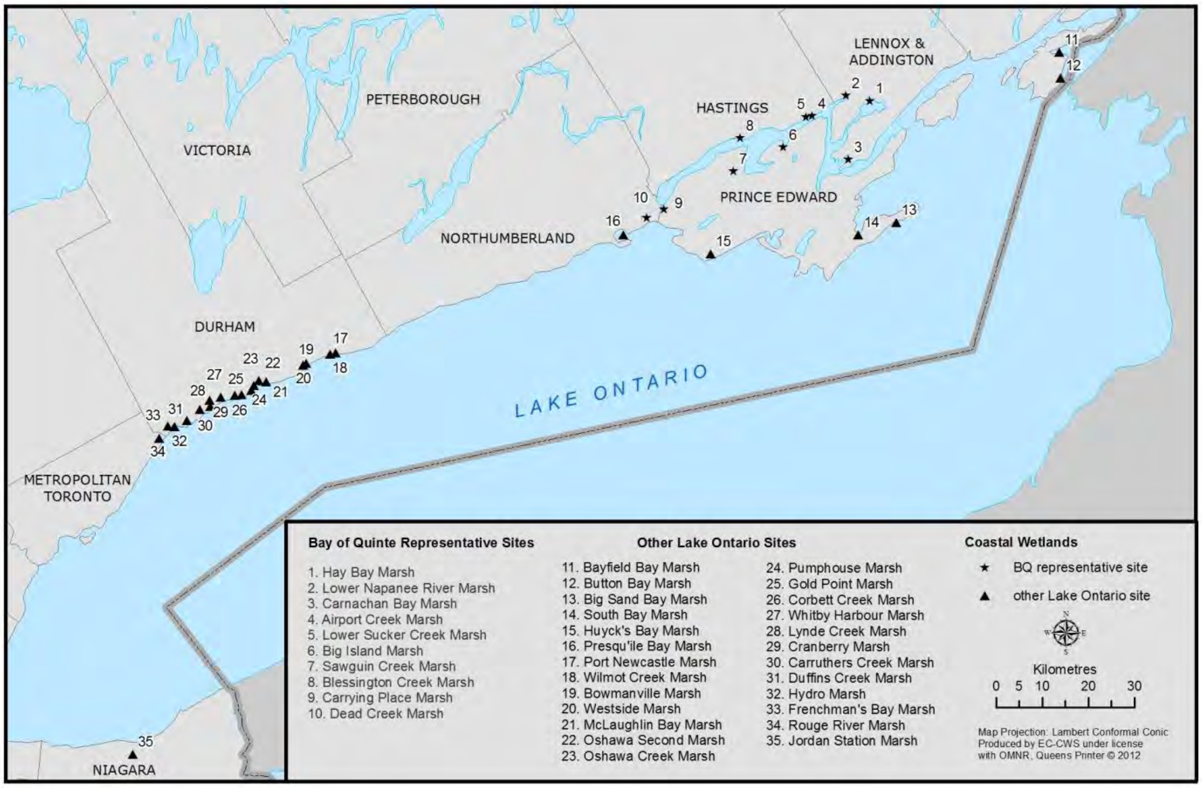


Figure 2. Names and locations of Bay of Quinte representative sites and other Lake Ontario coastal wetlands sampled by EC-CWS and partners. Refer to Appendix A for list of sites surveyed for water quality.

Rationale for Delisting Criterion Status

Determining the threshold below which the 2014 mean WQI (of all Bay of Quinte representative sites) must not fall to be considered not impaired requires multiple steps:

1. Calculate an annual mean from 2006 to 2010 WQIs from all BQ representative sites
2. Calculate an overall mean and standard deviation of 2006 to 2010 WQIs from all BQ representative sites
3. Multiply the standard deviation by 2
4. Subtract the 2 standard deviations from the 2006-2010 mean
5. Calculate an annual mean from 2006 to 2010 WQIs from all 'other Lake Ontario' sites.
6. Calculate the 2014 mean from all 'other Lake Ontario' coastal wetlands surveyed
7. Subtract the 2006 to 2010 'Lake Ontario' mean from the 2014 'Lake Ontario' mean
8. Add the results of Step 4 and Step 8
9. The resulting number is the corrected threshold above which the 2014 mean WQI for BQ representative sites must fall to be considered 'not impaired'.

Data used for these calculations are found in Appendix A (Tables 2 and 3).

2014 Bay of Quinte Representative Sites Mean WQI = 0.62

2014 Corrected Threshold WQI = -0.26

Since the Bay of Quinte 2014 WQI mean is greater than the threshold WQI value, the water quality at Bay of Quinte coastal wetlands is considered to be **not impaired**. Therefore, the criterion for water quality to be considered not impaired has been met in each year from 2011 to 2014 (Appendix 2, Table 4).

Future Monitoring or Actions Required

Regular monitoring of the coastal wetland water quality should continue, for the near future at least, at the Bay of Quinte representative sites and numerous other Lake Ontario coastal wetlands.

Glossary

Turbidity	A measure of the degree to which light traveling through a water column is scattered by the suspended organic (including algae) and inorganic particles measured in Nephelometric Turbidity Units (NTU).
Conductivity	A measure of the dissolved ions in water measured in microSiemens per centimetre (μScm^{-1}) or milliSiemens per centimetre ($mScm^{-1}$). Conductivity is a good indicator of urban run-off – especially from road salt.

References

Bay of Quinte RAP Restoration Council. 2007a. A Management Plan to Address Wildlife Related Impairments for the Bay of Quinte Area of Concern. Prepared by P. Johanson and B. McNevin, March 2007.

Chow-Fraser, P. 2006. Development of the Wetland Water Quality Index (WQI) to assess effects of basin-wide land-use alteration on coastal marshes of the Laurentian Great Lakes. *In* Simon, T.P. and P.M. Stewart (Eds.). Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat, and Indicators, pp. 137-167. AuthorHouse, Bloomington, IN.

Appendix A: Water Quality Index (WQI) values for 2006-2014

Table 2. WQI values for Bay of Quinte representative sites for 2006-2014.

Wetland	2006	2007	2008	2009	2010	2011	2012	2013	2014
Airport Creek Marsh	-0.46	0.26	-0.44	0.15	1.12	0.36	0.52	0.61	0.82
Big Island Marsh	0.56	0.15	0.82	-0.04	0.90	0.96	0.60	0.73	0.93
Blessington Creek Marsh	0.29	0.54	0.38	0.60	0.14	0.61	1.07	1.08	0.97
Carnachan Bay Marsh	-	-0.16	-0.08	0.27	1.04	0.23	0.67	0.57	0.53
Carrying Place Marsh	-0.80	0.15	0.26	-0.09	0.15	0.64	0.37	0.05	0.57
Dead Creek Marsh	-0.09	0.21	0.37	0.68	0.17	0.43	0.31	0.71	0.72
Hay Bay Marsh	-0.37	0.49	-0.20	0.37	0.37	0.32	0.78	0.41	0.88
Lower Napanee River Marsh	0.04	0.45	1.18	0.81	0.96	0.87	0.81	0.32	-0.48
Lower Sucker Creek Marsh	0.22	0.33	-0.18	-0.22	0.28	0.25	0.46	0.38	0.43
Sawguin Creek Marsh	0.18	-0.38	0.27	-0.12	0.22	0.44	0.40	0.79	0.81
Mean WQI	**-0.05**	**0.20**	**0.24**	**0.24**	**0.53**	**0.51**	**0.60**	**0.56**	**0.62**
# wetlands	9	10	10	10	10	10	10	10	**10**

Note: Sawguin Creek Ditched Marsh was not sampled in 2010.

Table 3. WQI values for all other Lake Ontario sites for 2006-2014.

Wetland	2006	2007	2008	2009	2010	2011	2012	2013	2014
Bayfield Bay Marsh				0.50		0.59	0.24		
Big Sand Bay Marsh				0.38		-0.61			
Bowmanville Marsh		-1.59	-1.32	-0.33	-1.38	-0.47	-1.10	-1.36	-0.85
Button Bay Marsh				-0.26		0.18	-0.33		
Carruthers Creek Marsh	-1.51	-2.74	-1.60	-1.43	-1.17	-1.96	-1.73	-2.00	-1.66
Corbett Creek Marsh	-1.29	-1.68	-1.03	-0.99	-1.65	-1.81	-1.76	-1.08	-1.58
Cranberry Marsh	-0.62	-1.35	0.00	0.28	0.03	0.23	-0.96	0.53	-0.57
Duffins Creek Marsh	-1.16	-1.49	-1.13	-0.87	-1.17	-0.99	-1.52	-1.31	-1.49
Frenchman's Bay Marsh		-1.17	-0.90	-0.63	-0.75	-0.88	-1.24	-1.79	-0.89
Goldpoint Marsh		-1.65		-2.17	-1.74	-1.38	-3.08	-1.96	-1.13
Hydro Marsh	-0.93	-1.69	-1.14	-1.38	-1.64	-1.53	-1.20	-1.90	-1.8
Jordan Station Marsh				-1.23			-1.53		
Lynde Creek Marsh	-2.16	-1.32	-1.72	-1.19	-1.80	-1.69	-1.52	-1.48	-1.19
McLaughlin Bay Marsh		-2.69	-2.42	-2.02	-1.88	-2.22	-2.59	-2.25	-2.17
Oshawa Creek Marsh		-0.56		-1.07	-1.19	-1.32	-1.92	-1.88	-1.59
Oshawa Second Marsh		-1.17	-0.61	-1.48	-0.99	-1.75	-1.87	-1.70	-1.04
Port Newcastle Marsh	0.15	-1.07	-0.18	-1.00	-1.15	-0.94	-0.84	-0.87	-0.13
Presqu'ile Bay Marsh				0.85		0.92	0.32		
Pumphouse Marsh	-1.00	-2.55	-0.74	-0.77	-1.26	-0.82	-1.90	-1.20	-1.82
Rouge River Marsh		-1.45	-0.83	-0.66	-0.91	-1.10	-1.07	-0.90	-0.41
South Bay Marsh				0.26		0.06	-0.61		
Westside Marsh	-1.50	-1.55	-0.80	-1.04	-1.52	-1.68	-1.46	-1.56	-1.35
Whitby Harbour Marsh		-1.35		-1.74	-1.65	-1.88	-1.17	-2.35	-1.74
Wilmot Creek Marsh	-0.61	-1.44	-0.77	-0.84	-1.43	-1.83	-1.18	-0.62	-0.77
Mean WQI	**-1.06**	**-1.58**	**-1.01**	**-0.78**	**-1.29**	**-0.99**	**-1.31**	**-1.43**	**-1.23**
# wetlands	10	18	15	24	18	23	23	18	18

Table 4. WQI values for Bay of Quinte (BQ), 'Other Lake Ontario sites', and corrected threshold values from 2011-2014.

WQI value	2011	2012	2013	2014
'Other Lake Ontario sites'	-0.99	-1.31	-1.43	-1.23
Corrected BQ WQI threshold	-0.03	-0.34	-0.46	-0.26
Bay of Quinte WQI score	0.51	0.60	0.56	0.62
Criterion met (Y/N)	**Y**	**Y**	**Y**	**Y**

APPENDIX I: Fish Habitat in Coastal Wetlands Delisting Criteria Status Report

October 22, 2012

Authors

- Susan Doka, Ph.D., Research Scientist, Fisheries and Oceans Canada
- Erin Gertzen, Aquatic Science Biologist, Fisheries and Oceans Canada
- Jody MacEachern, GIS Analyst, Fisheries and Oceans Canada

Recommendation: Delisting Criteria have been met

DELISTING TARGET	DELISTING CRITERIA	STATUS
FWH-4: Conserve, enhance and maintain the high quality, diverse and extensive fish habitat in the Bay of Quinte, by assessing and mapping fish habitat suitability and the quantity of habitat needed to support the biodiversity and productivity of fishes in this ecosystem and, by identifying priority areas and actions for effective conservation and restoration.	FWH-4.1: The Bay of Quinte's Weighted Suitable Area (WSA) of fish habitat across all guilds and life stages should be: a) ranked above the 80th percentile among 41 coastal comparison sites within Lake Ontario , and b) not significantly lower than at Long Point Bay, Lake Erie.	Achieved
	FWH-4.2: Maintain a no-net-loss of the weighted suitable area of fish habitat over time by ensuring that: a) the present (2005-2010) weighted suitable area of fish habitat in the Bay of Quinte is greater than the conditions during previous time stanzas, and b) highly suitable important, rare or sensitive habitats are identified conserved.	Achieved
	FWH-4.3: Maintain the weighted suitability area of fish habitat required to support the long-term relative levels of important fish as projected from habitat supply models. Population results should: a) be ranked above the 80th percentile among 39 comparison sites within Lake Ontario, and b) be stable or increasing over time within the Bay of Quinte given previous and projected changes in habitat availability.	Achieved

Background:

***History of the Beneficial Use Impairment "Loss of Fish Habitat"*:**

The International Joint Commission (1991) recommended listing BUI 14 as impaired "when fish and wildlife management goals have not been met as a result of loss of fish and wildlife habitat due to a perturbation in the physical, chemical, or biological integrity of Boundary Waters, including wetlands." At the time of AOC listing, the loss of fish habitat in the Bay of Quinte was evident (BQRAP 1993). This impairment was described as being related to increasing development pressures, and specifically to the loss of two-thirds of the historical wetland area and extensive shoreline modifications. This report deals with a portion of the Beneficial Use Impairment (BUI) #14 "Loss of Fish and Wildlife Habitat" under the Bay of Quinte Remedial Action Plan (RAP). Specifically, this report presents an assessment of three measurements under BUI 14 that relate to the loss of fish habitat as described above.

Historically, the Bay of Quinte was a highly productive ecosystem. The bay was covered in vast wetlands, submergent vegetation and rocky shorelines, which provided habitat to a diverse community of fishes and wildlife. The Bay of Quinte was, and continues to be, a popular fishing and recreation area. However, the cumulative impacts of development pressures, pollution, water level regulation, reduced tributary access, invasive species, and climate change have all impacted fish habitat in the Bay of Quinte. The Stage II Report (1993) provides a timeline of events significantly altering the Bay of Quinte ecosystem. Importantly, the RAP recognized that concurrent effects of development and pollution altered the Bay of Quinte into a highly eutrophic system. Eutrophication and associated water clarity changes had a significant impact on the extent of submerged aquatic vegetation, an important element of fish habitat. An ecosystem approach was adopted to carry out RAP goals for remediating fish habitat.

With knowledge of the factors contributing to the loss of fish habitat, the RAP Stage II Report outlined the following habitat goal (BQRAP 1993):

> "*To re-establish and maintain aquatic shoreline and wildlife habitat conditions and sites within the Bay of Quinte ecosystem capable of supporting healthy, diverse, stable and self-sustaining aquatic and terrestrial communities.*"

The Bay of Quinte habitat goal was based on the IJC's (1991) recommended guideline to delist "when the amount and quality of physical, chemical, and biological habitat required to meet fish and wildlife management goals have been achieved and protected." Three specific habitat objectives were developed to achieve the habitat goal: supporting feasible actions to restore damaged aquatic and shoreline habitats (restoration); implementing habitat compensation where restoration is not possible (enhancement); and applying the *no-net-loss* of the productive capacity targets of DFO's 1986 habitat guiding principle to all future developments (compensation/mitigation).

The habitat target and measures presented in this report expand upon the initial habitat goal and objectives. Using the methodology described in the Bay of Quinte Fish Habitat Management Plan (Johanson and McNevin 2007) and reflecting upon the *Fisheries Act* and the *Policy for the Management of Fish Habitat* (DFO 1986), we revised and specified the objectives into the quantitative measures outlined above with multiple reference points to gauge the health of the Bay of Quinte fish habitats.

Methods of Assessment:

We used a quantitative approach to assess whether the fish habitat measures have been met. Weighted suitable area (WSA) is a single metric that simultaneously describes the quality and quantity of habitat available to fishes (Figure 1; Minns et al. 2001). The metric takes into account important habitat variables (water depth, substrate, vegetation and temperature) and fish-habitat associations (Lane et al. 1996a,b,c) to calculate the suitability of an area for the entire fish community or specific guild of fishes (e.g. piscivores). WSA for the fish community is the product of the community-wide habitat suitability and wetted area within each study site summed across discrete habitat units. WSA-linked population models provide a measure of how select sport fish populations may change over time and space with weighted habitat supply. Details of the methodology can be found in Appendix A.

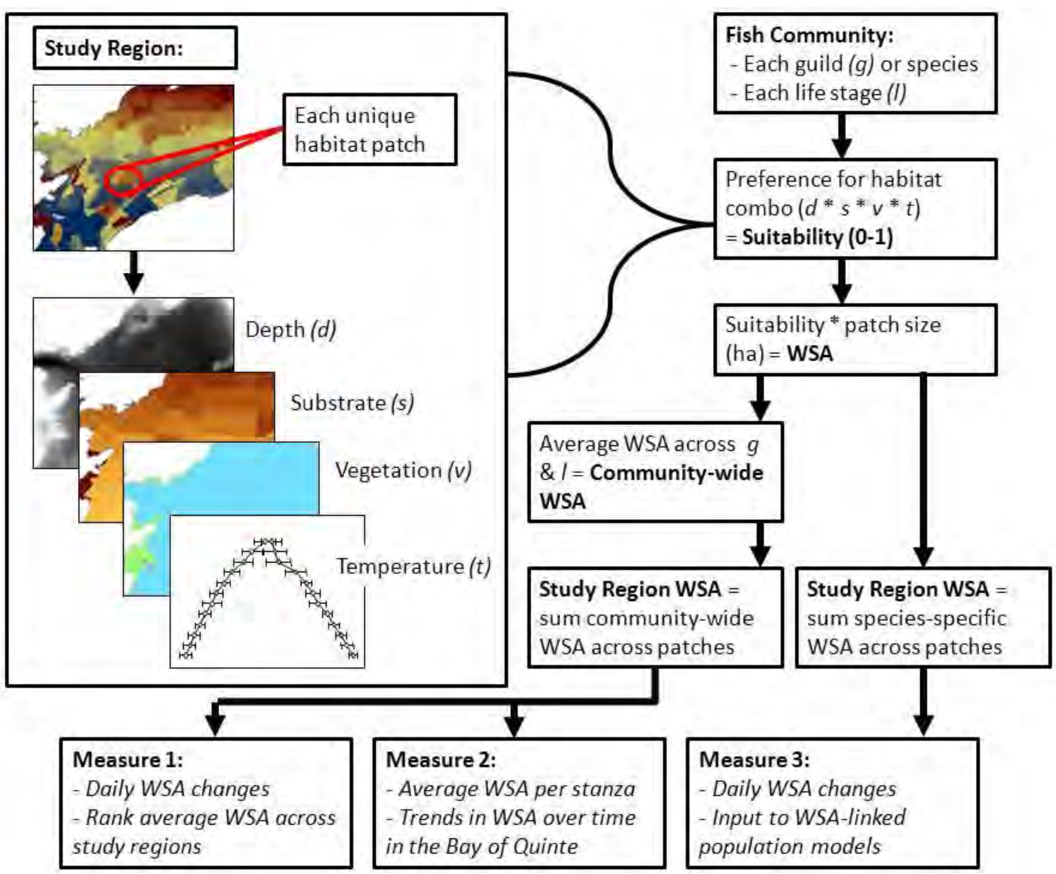


Figure 1: Flow diagram of weighted suitable area (WSA) calculations for the three habitat measures. Each measure uses spatial habitat information from the study areas (Bay of Quinte, Lake Ontario and Long Point sites), fish species or community information, and habitat preferences, to calculate WSA as required for a given Measure.

To complete Measure 1A and 1B assessments, we modelled and ranked WSAs for the Bay of Quinte fish community (Appendix B) across 41 coastal areas in Lake Ontario (1A) and for Long Point Bay, Lake Erie (1B). These areas represent references to compare against Bay of Quinte sub-areas of similar size. Because of its historic importance to Lake Ontario fisheries and its vast areas of remaining vegetated habitat, the Bay of Quinte AOC should continue to provide good habitat, measured by greater WSA, in relation to all other coastal areas of similar size within Lake Ontario. Additionally, the Bay of Quinte should provide as much relatively productive habitat as other similarly important areas outside of Lake Ontario. WSA equivalence with the inner bay of Long Point Bay in Lake Erie, a relatively unimpacted coastal region with vast wetlands, would signify a healthy Bay of Quinte as Long Point Bay has been designated a Ramsar protected wetland.

To complete Measure 2A and 2B assessments, we modelled suitability and WSA across important time periods using detailed spatial habitat information in the Bay of Quinte. The WSA approach outlined in Figure 1 was used and the spatial habitat inputs are detailed in Appendix A. Given improvements in water quality over time and subsequent improvement in submerged vegetation, we expected the average quality and quantity of habitat available to the Quinte fish community to increase over time even given natural variability (e.g. changing water levels and temperatures during the same period). To evaluate this, we modelled suitability and WSA during three important time stanzas: pre-phosphorous management (1972-1978), post-phosphorous management/pre-*Dreissena* invasion (1979-1995) and post-*Dreissena* invasion (1995-2010). For each time stanza, we looked at high-water-level years, low water level years and average water level years to examine the variability in habitat supply over a range of conditions specific to each time stanza. Using the results of this analysis, we also identified highly suitable areas that should be conserved or protected for different fish guilds as well as the fish community as a whole.

To complete Measure 3A and 3B assessments, we modelled select fish populations using habitat-based population models. Key fishery populations were selected in the Bay of Quinte Fish Habitat and Fisheries Management Plans; a few were modelled here, including Largemouth Bass, Northern Pike, Smallmouth Bass and Yellow Perch. The habitat inputs to the population models used the WSA approach outlined in Figure 1. The habitat-linked population models were developed by Chu et al.(2006) and are described in Appendix A. Similarly to Measure 1A, we expect the Bay of Quinte to provide relatively good habitat for these populations compared to other areas across Lake Ontario and therefore to support relatively high population levels across 41 Lake Ontario comparison sites. Additionally, given improvements in water quality and subsequent vegetation improvements in the Bay of Quinte over time, we expected habitat area to increase in such a way that the fish populations show a positive trend over time within the AOC.

Assessment Results

<u>Measure 1A.</u>

Results of the assessment showed that study areas within the upper and middle sections of the Bay of Quinte ranked within the top 80th percentile of the average fish community-weighted WSA for all fish life stages: spawning (Figure 3), nursery or young-of-the-year (Figure 4), and adults (Figure 5) across all 41 Lake Ontario subareas. The subarea that overlapped the upper/middle sections of the Bay of Quinte ranked the highest (i.e. Rank 1 or 100th percentile) for all life stages, followed by the upper bay area (Rank 2). Other subareas that ranked within the top 80th percentile across all fish life stages included the Presqu'ile and West Lake subareas. The study site that included the lower section of the Bay of Quinte, extending out to Amherst Island, provides relatively low WSA compared to more protected areas in the nearshore of Lake Ontario. This result is consistent with the expectation that the largely deeper and more open habitat would not support a large warmwater fish community but nonetheless would provide an important connection to open waters for cool and coldwater species.

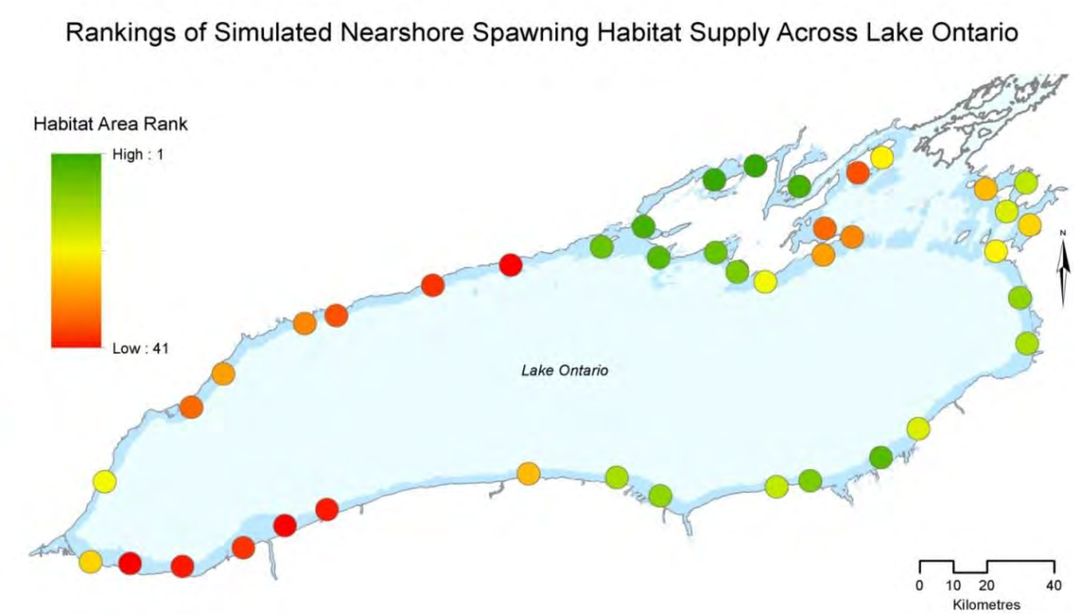


Figure 3: Average rankings across a 50-year time series of fish community-weighted suitable area of spawning habitat across 41 coastal comparison sites in Lake Ontario. For this analysis, warm and coolwater fish guilds were given more weight than coldwater fish guilds[1].

[1] Thermal guilds were ranked similarly in previous assessments and the ranking was agreed to by the Bay of Quinte RAP Delisting Committee. Coldwater species tend to be migratory and the nearshore nature of BQ AOC and its assessment, which does not include tributaries, would be artificially skewed by their equal weighting.

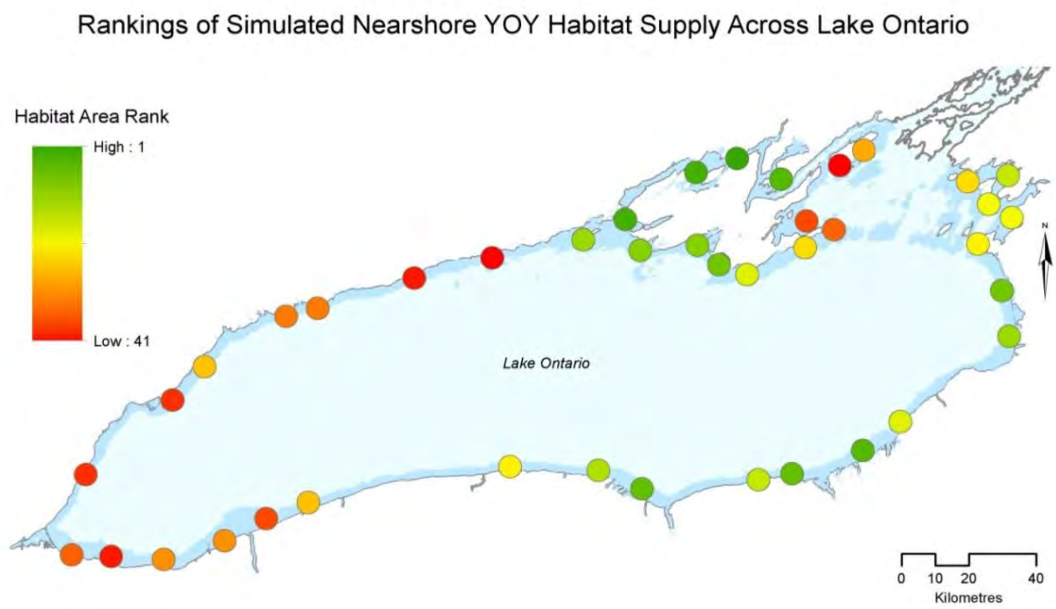


Figure 4: Average rankings across a 50-year time series of fish community-weighted suitable area of nursery (young-of-year) habitat across 41 coastal comparison sites in Lake Ontario. For this analysis, warm and coolwater fish guilds were given more weight than coldwater fish guilds.

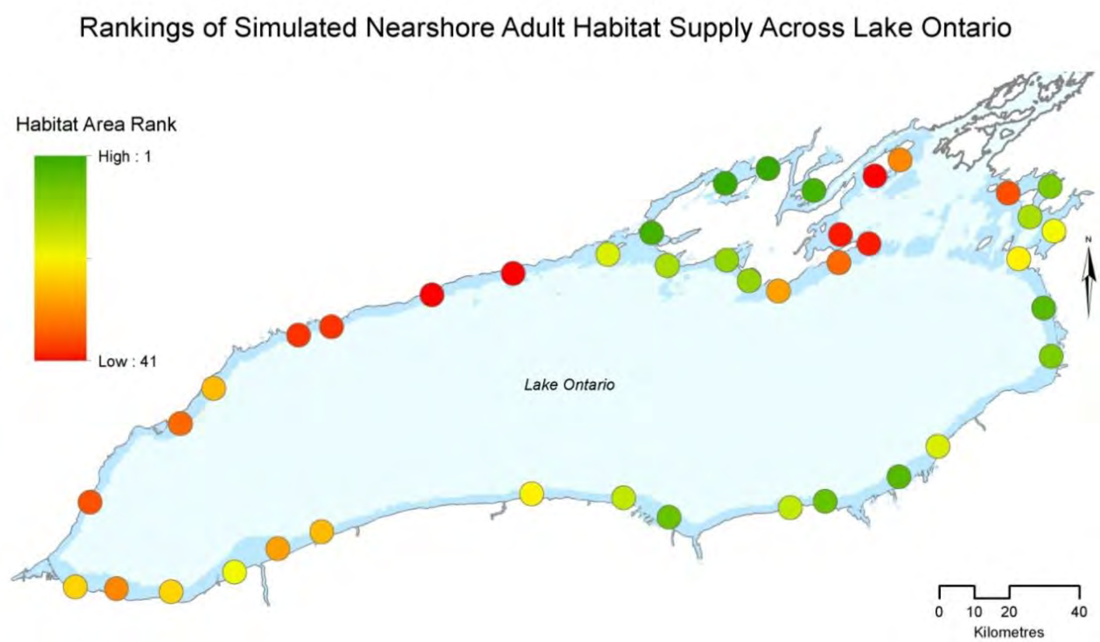


Figure 5: Average rankings across a 50-year time series of fish community-weighted suitable area of adult habitat across 41 coastal comparison sites in Lake Ontario. For this analysis, warm and coolwater fish guilds were given more weight than coldwater fish guilds.

Measure 1B.

Results of this comparative assessment between Great Lakes showed that the upper and middle sections of the Bay of Quinte have fish community-wide WSAs similar to the inner bay of Long Point Bay, Lake Erie (Table 1). The average annual cumulative WSA over all life stages across the Bay of Quinte did not differ from the Long Point average by greater than 10%; differences ranged between -9.1 and +6.7% depending on the subarea. Overall, Long Point could provide greater spawning habitat for the Bay of Quinte fish community[2]; however these differences were compensated for by relatively greater adult habitat supply in the Bay of Quinte. Because the Bay of Quinte provided greater spawning WSA to its fish community than most other coastal sites in Lake Ontario (Measure 1A) we felt the difference was acceptable. In the future, the analyses in Measure 1B could be re-evaluated using higher resolution spatial data for the Bay of Quinte as was available for Long Point. Lower spawning habitat supply in the upper Bay of Quinte may also a reflection of water level regulation in Lake Ontario as compared to the relatively natural water level fluctuations experienced on Lake Erie (Gertzen et al. 2012) which would favour shallow water spawners and access to periodically flooded areas.

Table 1. Standardized average annual cumulative weighted suitable area estimates (WSA, hectare-days) for Bay of Quinte study areas in relation to inner Long Point Bay, Lake Erie. Differences are shown by life stage and the average across life stages using the Bay of Quinte fish community as the basis for both areas. Differences greater than ±10% from inner Long Point were considered statistically significant and are denoted by an either a negative (-) or positive (+) sign.

Bay of Quinte Study Site	Spawning	Nursery	Adult	Average
Upper Bay reach cluster (site)	552 (-)	5210	6157 (+)	3973
Overlap of upper/middle sites	601 (-)	5293	6180 (+)	4025
Middle Bay (some upper/lower overlap)	480 (-)	4850	4959	3430
Inner Long Point Bay, Lake Erie	***939***	***5364***	***5014***	***3772***

Measure 2A.

Results of the assessment show that the suitability of the Bay of Quinte habitat has improved over time (Figure 6). Since the 1970s, for the overall fish community, the quality of habitat has improved by more than 10%. Table 2 shows changes in WSA by guild (thermal-vegetation guilds for early life stages, thermal-feeding guilds for adults) and life stage (spawning, nursery, or adult) across important time stanzas. While WSA in each year varies with water level and resulting water depth, the overall trend is towards an increase in suitability over time. Suitability trends by guild (Figures 7-9) show that the average relative suitability for high-vegetation spawning and nursery guilds, and adult piscivore guilds, have increased over time as vegetated habitat quality

[2] To standardize the approach the Bay of Quinte fish community and their habitat associations were used to assess both Bay of Quinte and Long Point Bay habitat supplies for three different fish life stages See appendix for more details.

has improved. Trends in low-affinity for vegetation groups and non-piscivorous adults are somewhat negative over time but are much less pronounced than the positive trends. Currently statistical comparisons of the trends are not possible but percent differences are noted.

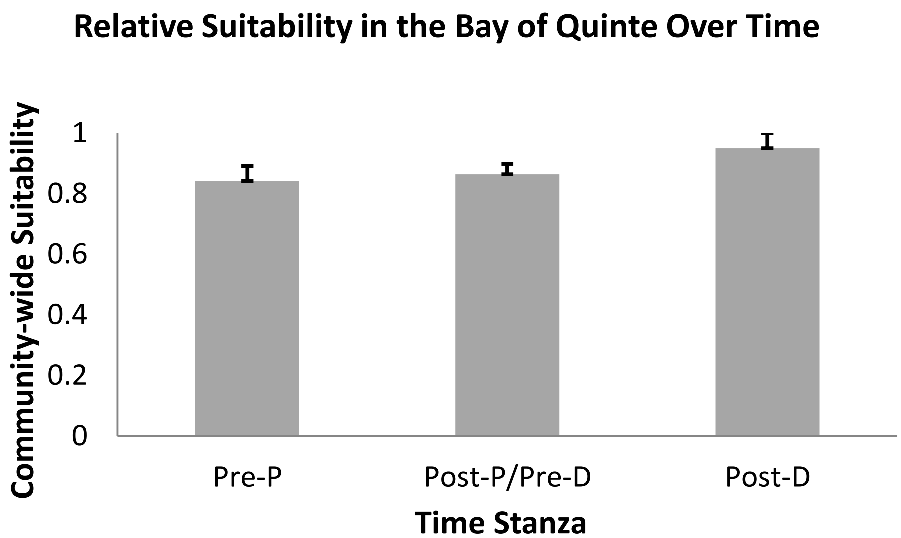


Figure 6: Changes in relative fish community-suitability across important time stanzas in the Bay of Quinte. Bars show the suitability during an average water level year within the time stanza. The error bars show the range of suitabilities across high and low water level years. [Pre-P = pre-phosphorus control; Post-P and Pre-D = post phosphorus control and pre *Dreissena* invasion; Post-D is post *Dreissena* invasion.]

Table 2: Weighted suitable area (WSA, ha) in the Bay of Quinte by guild (thermal–vegetation guilds and thermal-feeding guilds) and life stage (spawning, nursery and adult) across important time stanzas: pre-phosphorus management (1972-1977); post-phosphorus management/pre-*Dreissena* invasion (1978-1994); and post-*Dreissena* invasion (1995-2010). Each time stanza shows the range of habitat supply across water levels within the stanza[3]: High - year with highest average water level during growing season, Low - year with lowest average water level, and Average - year with water level closest to the average across the time stanza. (Note: Suitabilities can be calculated by dividing the WSA from the total wetted area.)

		Pre-P Management			Post-P/Pre-*Dreissena*			Pre-*Dreissena*		
Life Stage	**Guild**	**High (1973)**	**Low (1977)**	**Averag e (1972)**	**High (1986)**	**Low (1988)**	**Averag e (1979)**	**High (1997)**	**Low (1999)**	**Averag e (2002)**
Spawn	Cold-HighV	2020	690	800	1501	754	733	1022	1005	918
	Cold-LowV	951	1014	976	958	1008	960	749	790	767
	Cold/Cool-HighV	1627	881	884	1383	1015	922	1409	1667	1444
	Cold/Cool-LowV	900	901	855	900	919	852	702	786	728
	Cool/Warm-HighV	1663	1012	999	1451	1143	1041	1453	1734	1507
	Cool/Warm-LowV	1563	1393	1365	1505	1421	1348	1167	1295	1201

[3] Both emergent and submergent vegetation responds to fluctuations in water levels and the latter also responds to changes in water clarity. The type of vegetation community present at a range of elevations will differ across the time stanzas because of the different wet-dry cycles and therefore suitabilities will differ even at the same water level.

	Warm-HighV	1904	1139	1110	1641	1315	1186	1648	2025	1744
	Warm-LowV	2552	2338	2225	2412	2425	2252	2181	2499	2294
Nursery	Cold-HighV	2087	2032	1893	2067	2209	2026	2346	2762	2504
	Cold-LowV	3719	3517	3569	3624	3458	3478	2918	2833	2877
	Cold/Cool-HighV	2378	1983	1871	2217	2172	1983	2436	2856	2571
	Cold/Cool-LowV	7261	6817	6965	7105	6703	6813	6271	6047	6155
	Cool/Warm-HighV	1968	1744	1637	1886	1915	1751	2101	2489	2237
	Cool/Warm-LowV	2661	2771	2684	2650	2780	2670	2303	2416	2353
	Warm-HighV	1856	1229	1176	1635	1421	1288	1912	2300	2021
	Warm-LowV	1311	1439	1282	1302	1521	1338	1162	1489	1304
Adult	Cold-NonPisc	3560	3205	3384	3476	3109	3289	3273	2934	3104
	Cold-Pisc	334	284	311	323	270	298	311	256	284
	Cool-NonPisc	7108	6709	6704	6978	6795	6733	6875	7059	6911
	Cool-Pisc	2212	1712	1672	2080	1931	1841	2841	3103	2886
	Warm-NonPisc	6061	5527	5442	5843	5690	5517	5832	6188	5925
	Warm-Pisc	6121	5305	5316	5858	5490	5424	6408	6589	6392
Total Wetted Area		21175	19013	19363	20387	18954	19124	19363	18954	19013

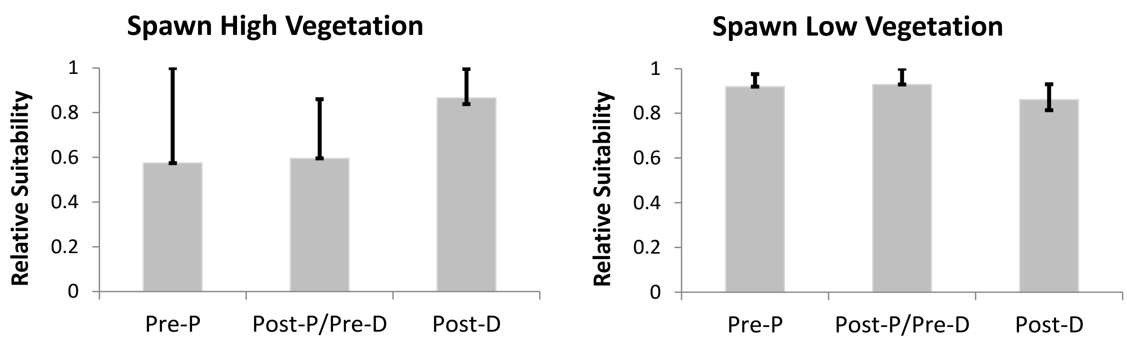


Figure 7: Relative spawning suitabilities during an average water level year across three time stanzas. Error bars show range in suitabilities during high and low years. Thermal guilds have been combined as they showed similar patterns even though habitat usage would occur at different times.

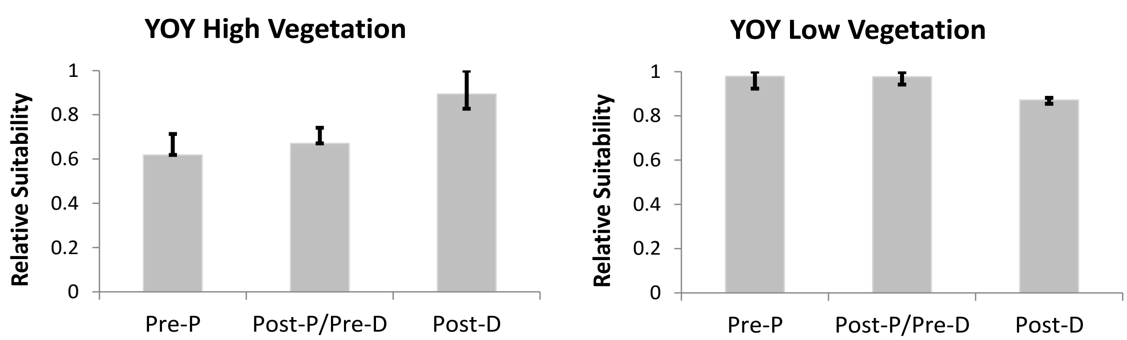


Figure 8: Relative nursery or young-of-the-year suitabilities during an average water level year across three time stanzas. Error bars show range in suitabilities during high and low years. Thermal guilds have been combined as they showed similar patterns even though habitat usage would occur at different times.

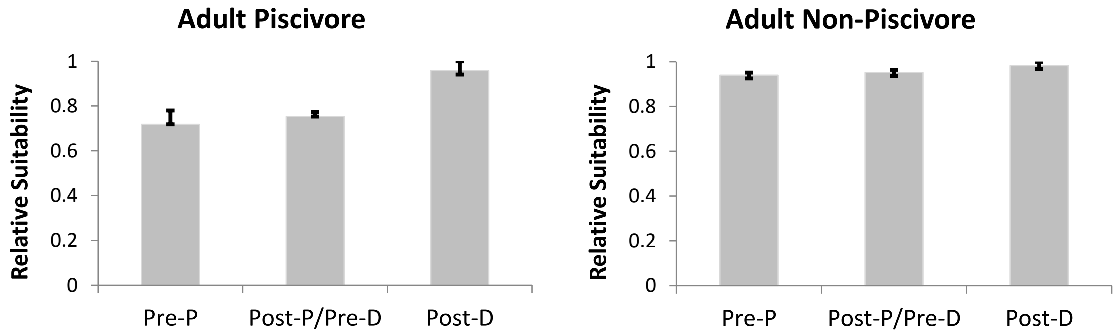


Figure 9: Relative adult suitability during an average water level year across three time stanzas. Error bars show range in suitabilities during high and low years. Thermal guilds have been combined as similar patterns occurred across thermal guilds within feeding classes.

<u>Measure 2B.</u>

Maps of relative habitat suitabilities for different fish guilds and life stages have been identified in previous reports and papers on the Bay of Quinte. Important areas that include rarity and fisher information were provided as ‘snapshots’ for the Bay of Quinte in Minns et al. (2006). Subsequently, relative fish spawning habitat suitabilities were dynamically modelled given water level fluctuations, spatial temperature differences and trends, and vegetation dynamics in the Bay (Gertzen et al 2012). Here we identify spawning, nursery (or YOY), and adult habitat for different fish guilds based on current condition habitat layers for upper and middle bays only in the Quinte area. Fish guilds are split differently than in previous assessments and are based on vegetation and thermal preferences instead of feeding guilds, as done previously (Appendix B shows the list of species and their guild assignments for this analysis). Using the WSA results from Measure 2A, areas that are highly suitable for the whole fish community, or for sensitive guilds and life stages, were identified as priority areas for conservation (Figures 10-11). Nearly every part of the Bay is used as fish habitat by some life stage or fish guild depending on their associations and preferences. However coastal areas near river mouths (Trenton, Belleville, Deseronto, Napanee) and significant wetlands (Muscote Bay, Big Bay, Hay Bay) supplied suitable habitat across life stages for both high and low vegetation guilds.

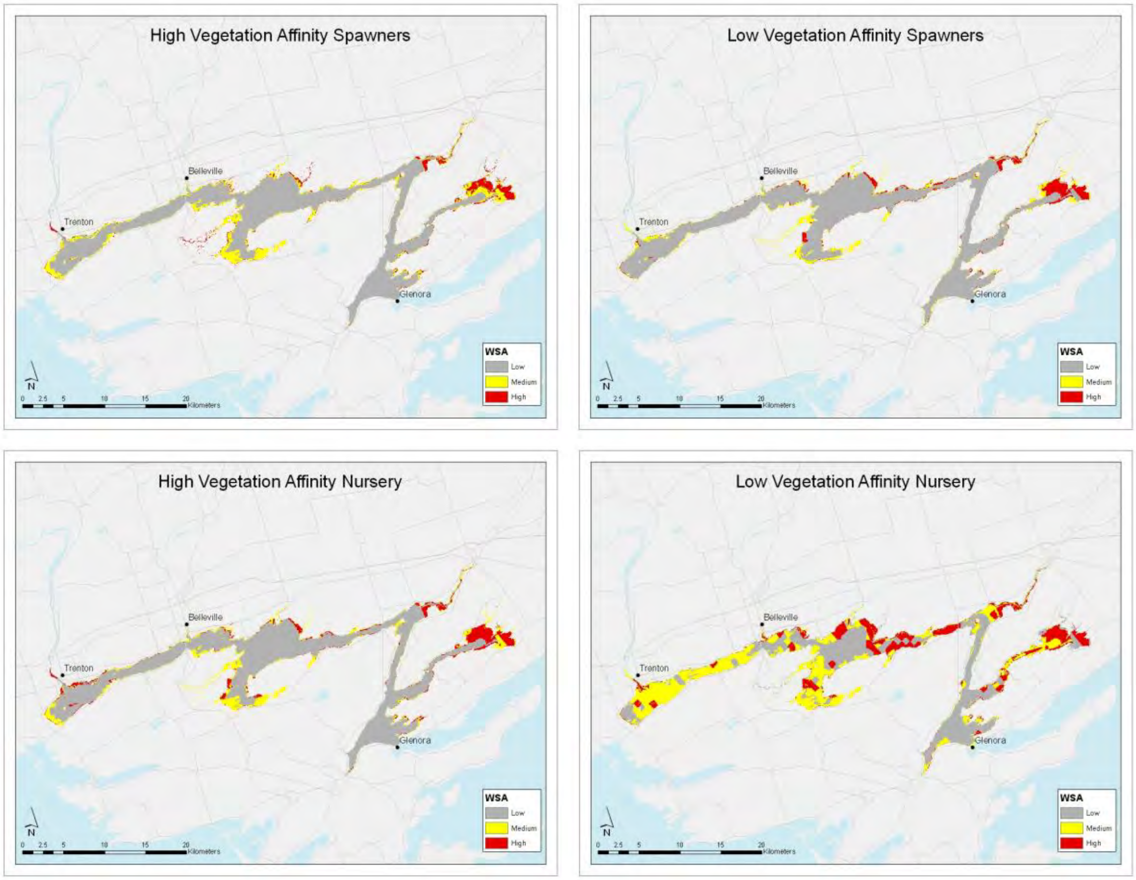


Figure 10: High and low vegetation-affinity fish guild spawning and nursery habitat relative suitabilities are presented for the Bay of Quinte. Within-thermal spawning and nursery suitabilities showed similar patterns so those guild suitabilities were grouped within vegetation guilds. Fish guild membership is shown in Appendix B.

High suitability habitat for early life stages of most of the fish community included key coastal areas identified earlier. These areas would likely disproportionately contribute to population success of most species in the Bay of Quinte because early life stages tend to be the most sensitive and experience high levels of density or environmentally-driven mortality. Adult low-vegetation suitabilities varied by depth depending on the thermal group and contributed to the use of habitat across most of the bay. This illustrates the habitat heterogeneity of the bay to support multiple fish guilds.

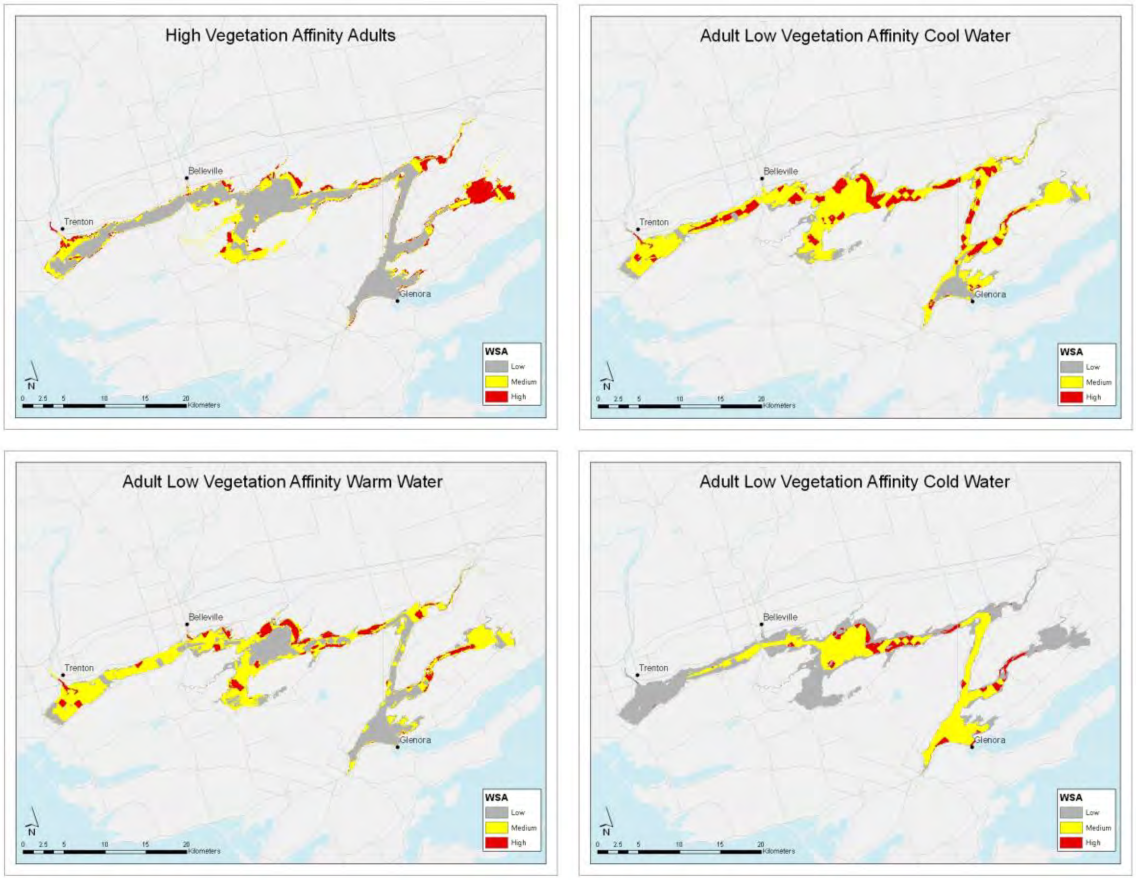


Figure 11: Adult habitat relative suitabilities for high and low vegetation-affinity fish-guilds are shown for the Bay of Quinte. Thermal groups across high-vegetation affinity adults showed similar patterns so those guild suitabilities were grouped. Low-vegetation affinity adult thermal groups showed distinctive habitat suitability patterns and are shown separately. Fish guild membership is shown in Appendix B.

The future resilience of the system should be tested using climate scenarios to ensure that transitional zones for key habitats are identified and also protected to ensure future ongoing production.

<u>Measure 3A.</u>

Results of the assessment showed that study areas within the upper and middle sections of the Bay of Quinte ranked within the top 80^{th} percentile of average simulated population abundances across Lake Ontario comparison sites. Maps of relative abundance rankings, driven by habitat supply, across Lake Ontario for Northern Pike (Figure 12), Largemouth Bass (Figure 13), Smallmouth Bass (Figure 14) and Yellow Perch (Figure 15) are shown. For Yellow Perch and Smallmouth Bass, the upper bay site provided the greatest population size, whereas the overlap of the upper/middle sections of the Bay of Quinte provided the greatest population size for Northern Pike and Largemouth Bass. Other sites within the top 80^{th} percentile for all fish life stages include the Presqu'ile/West Lake reach clusters. The analysis shows that Measure 3A has been met.

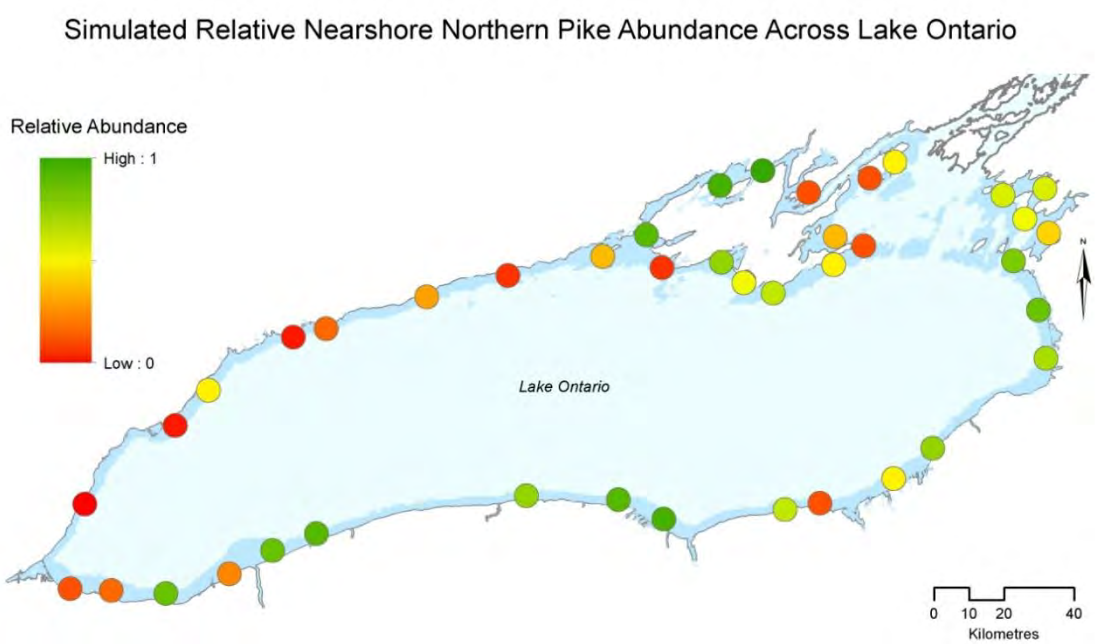
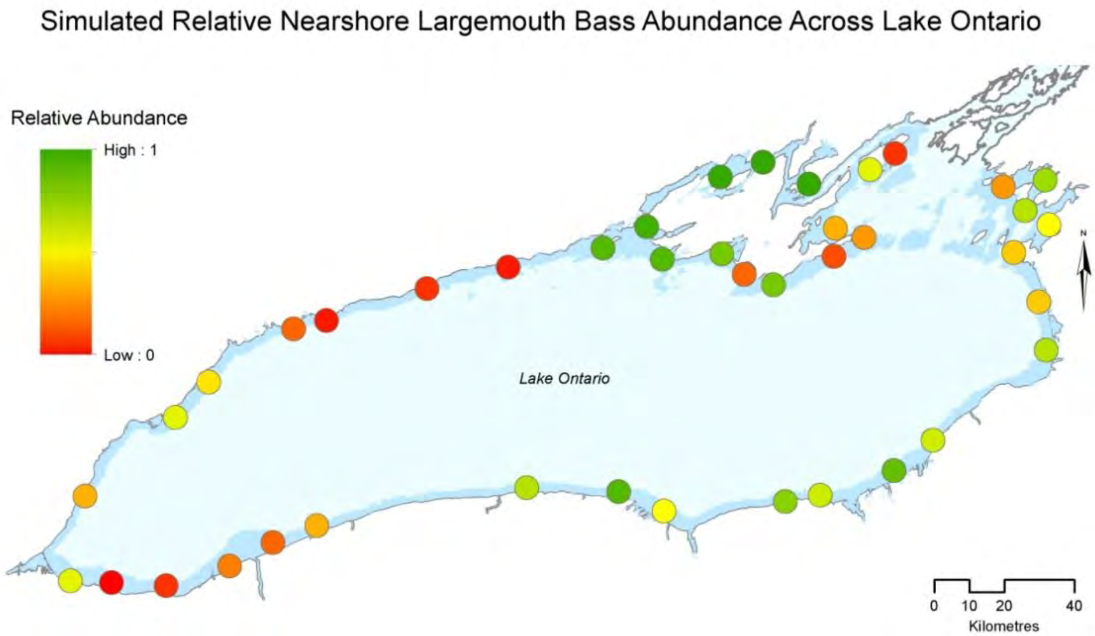


Figure 12: Relative rankings of simulated average population abundance of Northern Pike across 41 coastal comparison sites in Lake Ontario. Population simulations were based on habitat supply output based on very coarse-scale spatial information and generic vegetation models.. NOTE: patterns are indicative of rank scores based on habitat supply driven population dynamics and not the potential production values themselves.



Figure

Figure 13: Relative rankings of simulated average population abundance of Largemouth Bass across 41 coastal comparison sites in Lake Ontario. Population simulations were based on habitat supply output based on very coarse-scale spatial information and generic vegetation models.. NOTE: patterns are indicative of rank scores based on habitat supply driven population dynamics and not the potential production values themselves.

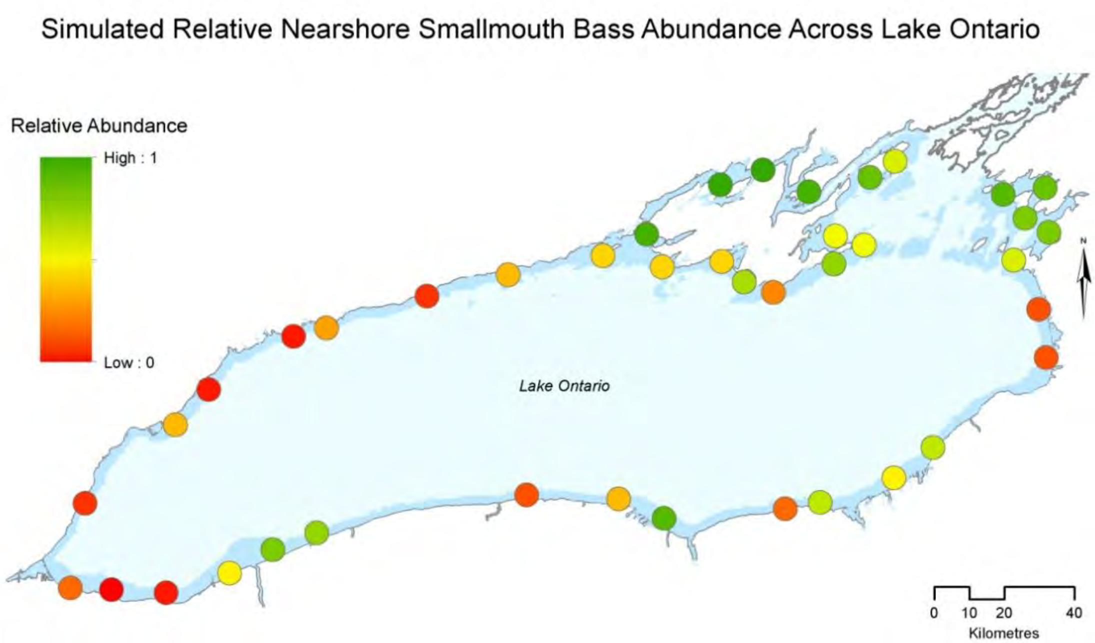


Figure 14: Relative rankings of simulated average population abundance of Smallmouth Bass across 41 coastal comparison sites in Lake Ontario. Population simulations were based on habitat supply output based on very coarse-scale spatial information and generic vegetation models.. NOTE: patterns are

indicative of rank scores based on habitat supply driven population dynamics and not the potential production values themselves.

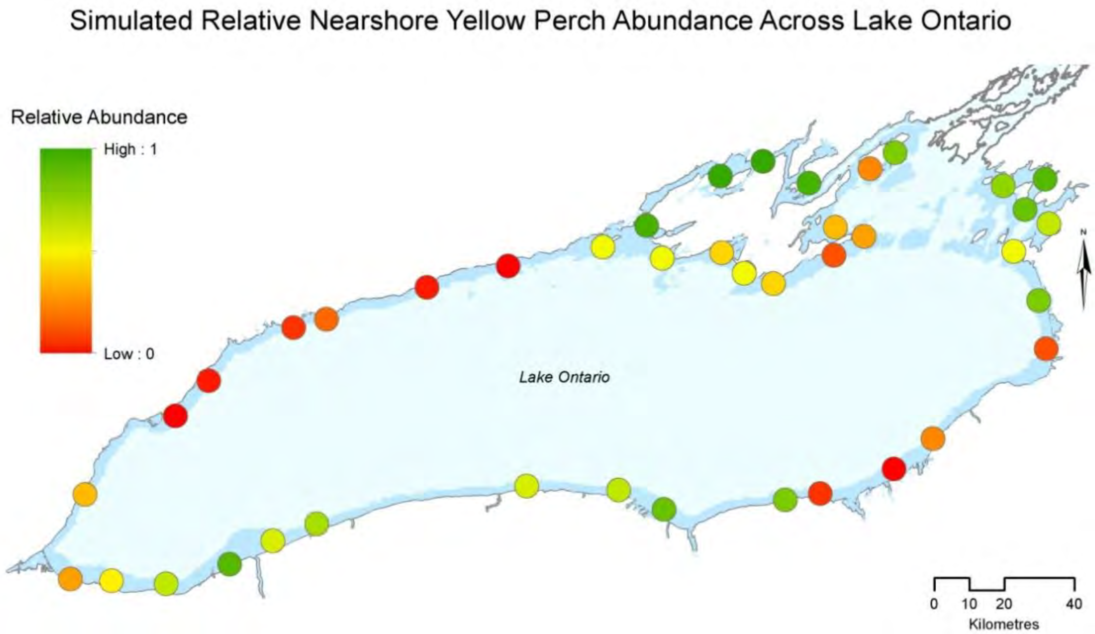


Figure 15: Relative rankings of simulated average population abundance of Yellow Perch across 41 coastal comparison sites in Lake Ontario. Population simulations were based on habitat supply output based on very coarse-scale spatial information and generic vegetation models. NOTE: patterns are indicative of rank scores based on habitat supply driven population dynamics and not the potential production values themselves.

Measure 3B.

Results of the Bay of Quinte habitat-based population models show variable trends in simulated species biomass density (kg/ha) over time (Figure 16). Between 1972 and 2010, based on habitat availability alone, Largemouth Bass and Yellow Perch populations have increased significantly ($p<0.001$), whereas Smallmouth Bass and Northern Pike have exhibited little change ($p=0.08$ and $p=0.31$, respectively). Northern Pike population dynamics are more affected by water level changes than other species, with the highest density occurring during the high water period during the mid-1970s. This is expected, given that Northern Pike exhibits great preference for shallow, vegetated habitat, which was more abundant during these high water years. Since 1978, the population trend for Northern Pike has been positive and marginally significant ($p=0.058$) and the population trend for Smallmouth Bass has been negative and significant ($p<0.001$). The trend in Smallmouth Bass is indicative of a trade-off in habitat supply over time between species that have low (e.g. Smallmouth Bass) and high affinities for vegetation (e.g. Largemouth Bass). In general, the habitat supply-based population trends indicated that, given ongoing improvements in habitat quality in the Bay of Quinte, key fisheries populations are responding in a positive direction which is consistent with fisheries management objectives.

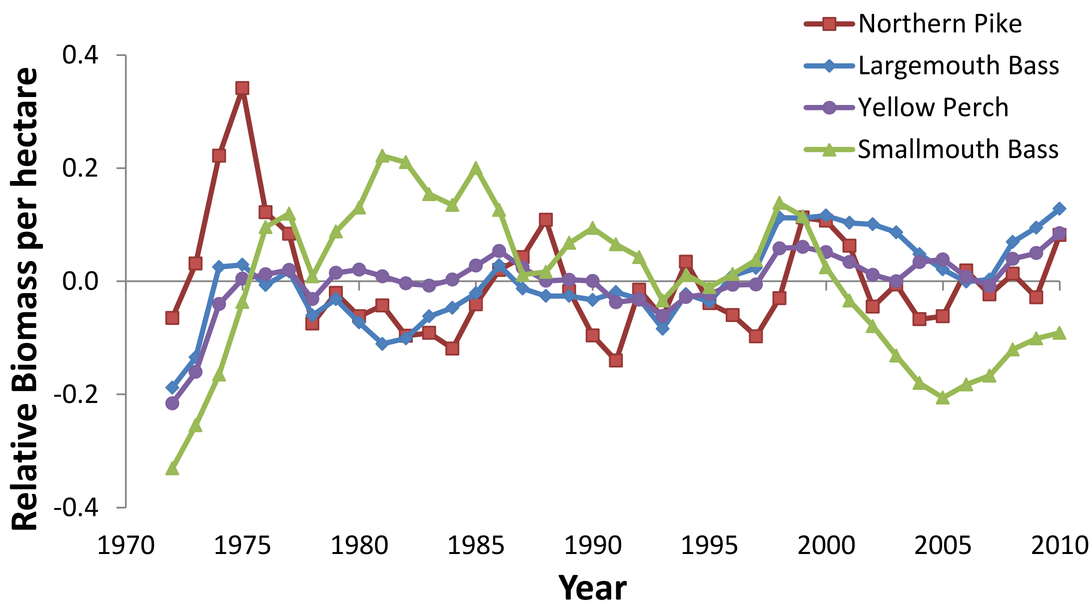


Figure 16: Simulated changes in relative population biomass per hectare from 1972 to 2010, based on habitat-linked population models. The y-axis is plotted relative to average biomass per hectare across the time period. NOTE: this is NOT intended to mimic actual fluctuations in these populations over time but to show how basic habitat supply for different life stages may have affected them.

Rationale for BUI Status

The fish habitat specific measures under the Bay of Quinte Beneficial Use Impairment #14-4 “Loss of Fish and Wildlife Habitat” have been met or exceeded (Table 3).

Table 3. Recommended status of measures covered under BUI #14-4

Measure	Previous Status	Recommended Status
SPATIAL 1A	Impaired	Not Impaired
SPATIAL 1B	Impaired	Not Impaired
TEMPORAL 2A	Impaired	Not Impaired
SPATIOTEMPORAL 2B	Somewhat Identified	Identified
SPATIAL 3A	Impaired	Not Impaired
TEMPORAL 3B	Impaired	Not Impaired

Future monitoring or actions required

In the future, the assessments could be extended to include the lower section of the Bay of Quinte if required. The fish habitat evaluation of the lower bay is currently continues to be hindered by poor spatial data quality especially for substrate type. GLLFAS in conjunction with CHS undertook a joint multibeam and verification survey of substrate type in the lower bay in 2010 concentrating in data poor areas between Conway and

Amherst Island. Unfortunately, these surveys have not yielded data that pass quality standards for inclusion in habitat suitability and supply analyses.

Overall, the habitat supply projections for the upper and middle Bay of Quinte should be resilient to minor development pressures because of the relative supply of high quality and varied fish habitats. Habitat supply and fisheries population responses under water level regulation and numerous invasive species over the past half century still remain high and sustainable compared to numerous reference conditions. Climate change, other potential invaders, and primary production shifts to harmful algal blooms in recent years may, however, affect the natural resilience of the Bay of Quinte system.

To address the Quinte's system resilience, climate change and algal bloom scenarios should be modelled to quantify the expected changes in habitat supply and population responses. The linkages between algal blooms and fish habitat have not explicitly been made so monitoring could help quantify this relationship. Not all eventualities can be accounted for and responses are based on what we currently know about the system's functional responses in a simplified way. Continued habitat monitoring and improvements to habitat modelling should be ongoing but on a less frequent basis to ensure the continued Recovered or Unimpaired Status of the Quinte ecosystem. This is especially important to track given the uncertainties in monitored and modelled responses and other cumulative impacts or biases not accounted for in either.

One example is the cumulative changes in shoreline and coastal areas over time that have occurred in the Bay of Quinte from multiple stressors that have not been addressed in this assessment, with the exception of improvements to submerged vegetation mediated by changes in water quality. While past temperature trends and water level fluctuations have been addressed in this most recent assessment, an evaluation of cumulative impact trends from development and future changes to Lake Ontario regulation and climate change impacts would be advisable (this may include algal blooms). Forecasting trends in habitat supply and its implications for fisheries and the fish community would assist decision making to ensure the integrity of the Bay of Quinte ecosystem is maintained into the future.

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APPENDIX A

Methods of Assessment

Measure 1A:

Forty-one coastal comparison sites were defined using basic information on physical characteristics around the entire coastline of Lake Ontario. The Lake Ontario shoreline was characterized into 1 km ‘reaches’ (Stewart 2002) and additional pieces of information were amended for the Lake Ontario – St Lawrence River Study (LOSLR) on water regulation (Bakelaar et al. in review). Attributes of location, nearshore geology, shoreline type, vegetation type, fetch, temperature zone, and area by 0.1 m elevation interval were assigned to each reach. Lake Ontario shoreline reaches were sequentially aggregated into groups until a total area amounting to 146 km^2 was obtained within the 0-20-m water depth range, relative to average water level for 1950-2010 (i.e., 74.8 m above mean sea level at International Great Lakes Datum 1985 (IGLD 85)). Consecutive or neighbouring reach clusters, or study sites, overlapped by half of their area (Figure A1-1) so that edge effects in model results could be minimized. The result was a total of 41 sequentially overlapping sites with areas of 146 km^2 – an area roughly equivalent to the size of the Upper Bay of Quinte (Figure A1-2); the size of the reach clusters was also related to the area that could support an average fish population (Chu and Doka, unpub. data).

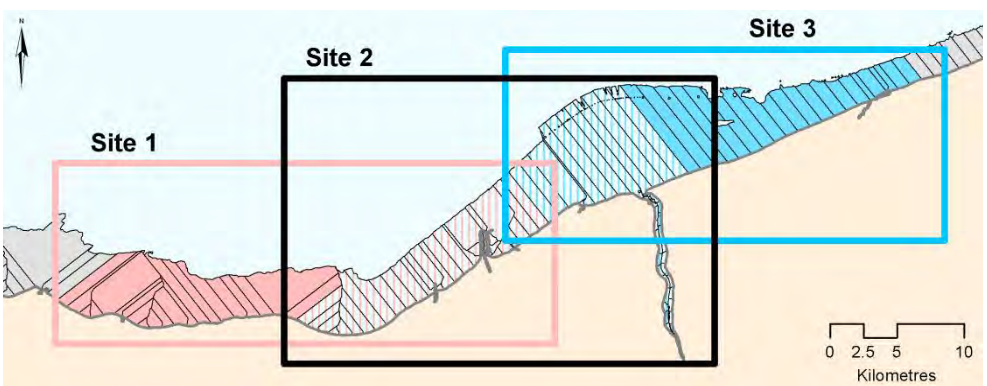

Figure A1-1: Example of sequentially overlapping nearshore sites or reach clusters in Lake Ontario. Sites consisted of a series of 1-km long shoreline reaches. Each sequential cluster overlapped by half with the previous site. The average area for each reach cluster was 146 km^2.

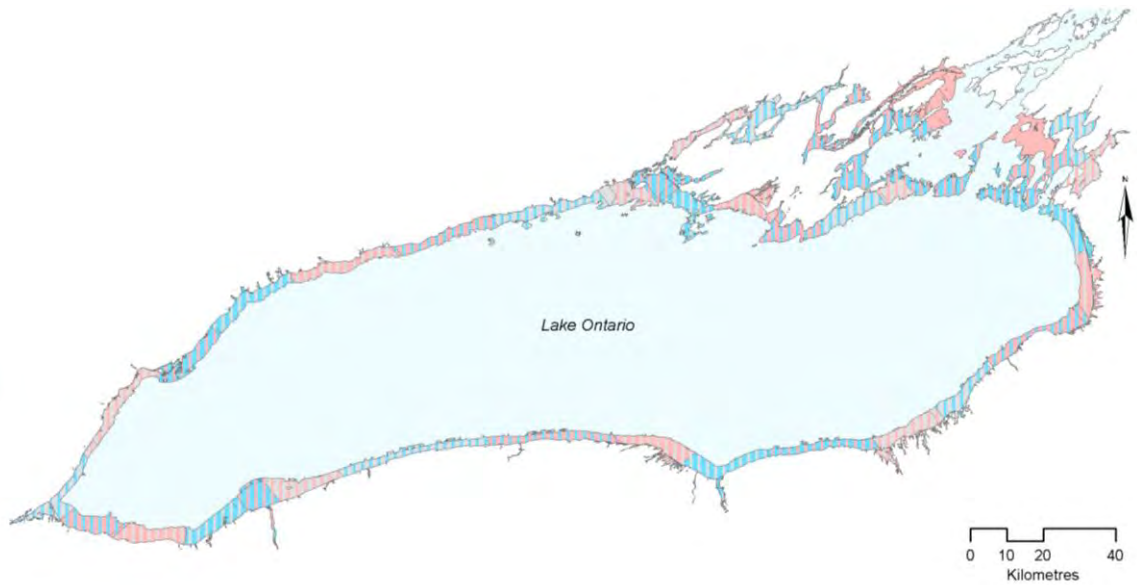

Figure A1-2: Map showing 41 sequentially overlapping nearshore sites in Lake Ontario. Each site has an area of roughly 146 km^2 within the 0-20 m water depth range given a water level of 74.8 m.

For each study site, WSA was calculated using the Lake Ontario Habitat Model (LOHM; Doka et al. 2005). The model classifies relative habitat suitability (0-1) for different physical attributes by using published and updated fish-habitat associations (Habitat Alteration Assessment Tool; Minns et al. 2001; Eakins 2012). Fishes are assigned to guilds based on habitat associations and life history characteristics. Combinations of associations for water depth, vegetation cover, substrate, and temperature are multiplied to produce the habitat suitability value of a given reach. In the LOHM, WSA is the product of average habitat suitability and wetted area of a given reach; WSA is summed across all reaches in the study site to give an overall estimate of the quality and quantity of habitat available at that location.

It should be noted that the LOHM is a comparative model developed to compare water regulation scenarios; therefore, the actual shoreline and vegetation changes that have occurred in the Bay of Quinte as a result of development and water quality changes were not included in this analysis. Vegetation responded to water level influences only in a simplified way. Effectively this creates a time series of vegetation and shoreline conditions that respond to water level changes but use recent shoreline classifications and elevation/bathmetry information as base layers.. It is not intended as 50-year time series of what has occurred around the lake's shoreline or nearshore water quality changes because these data are not available. This is intended as a relative or comparative approach around the Lake for the current state of the lake given historical fluctuations in water levels that influences habitat supply. Comparisons between different time stanzas where water clarity does affect vegetation in the Bay of Quinte are included in Measure 2.

WSA for each site was modelled on a daily basis from 1951-2000. Some physical habitat variables were modified based on water level and temperature fluctuations. Changes in depth, vegetation cover, and temperatures affected average habitat suitabilities and subsequent WSA on a daily basis per site. The substrate composition within a reach remained unchanged. Water depth was calculated on a daily basis using a time series of modelled daily water levels from 1951-2000. This time series included high waters in the 1970s that reached the 100-year flood mark. Emergent and submerged aquatic vegetation changed on an annual basis, based on water levels. Areas of different vegetation communities and the percent cover of vegetation within each area were predicted using the University of Waterloo's Emergent Vegetation Model (Hebb et al. 2006) and the DFO-GLLFAS Submerged Aquatic Vegetation Model (Seifried 2002; Doka et al. 2006). Vegetation type and percent cover values were translated into habitat suitabilities and cover categories consistent with HAAT evaluations (Minns et al 1995). Substrate composition was held constant for each reach and was derived from generalized nearshore geology and shoreline type classes for Lake Ontario (Stewart 2002). Temperatures-at-depth within each reach were modelled on a daily basis using empirical surface water temperature, lake zone and depth-based models developed for the LOHM (Doka, unpub. data; Moin and Thompson 2006).

The evaluated fish community included all species documented in the Bay of Quinte (Hurley 1986; Johanson and McNevin 2007). Species were separated into guilds based on habitat preference or trophic level (Minns et al. 2006). For each study site, WSA was calculated for each guild and life stage (spawning, young-of-year (YOY) and adult) separately. Guilds were modified by life stage, depending on habitat needs during that life stage. For spawning and YOY habitat, fishes were divided into vegetation-thermal guilds, and for adult habitat, fishes were divided into both vegetation-thermal guilds and feeding-thermal guilds. Species by guild type and life stage are listed separately in Appendix B. To ensure proper comparison, the Bay of Quinte fish community was used in calculations across all sites (reach clusters) regardless of what the actual local fish species were.

The WSA for the overall fish community was computed by averaging WSAs across all guilds and life stages; similar guild weightings to those assigned in Minns et al. (2006; p. 24) were applied. Because warmwater fish habitat is limiting in Lake Ontario in general but is abundant in the upper and middle sections of the Bay of Quinte, the coldwater guilds were given less weight in the overall fish community WSA estimate. Adult guilds and weights included in the analysis were:

- coldwater, piscivores (0.1)
- coldwater, non-piscivores (0.1)
- coolwater, piscivores (0.2)
- coolwater, non-piscivores (0.2)
- warmwater, piscivores (0.2)
- warmwater, non-piscivores (0.2)

The average overall WSA in square kilometres by life stage across the 50-year period was calculated for each reach cluster by summing WSA across days within a year (similar to degree-day calculations) and averaging cumulative WSA across years.

$$AvgAnnCusumWSA_{LS,Area} = \underset{Y=1}{\overset{50}{Avg}}\left(\sum_{d=1}^{365} dailyWSA_{LS,Area}\right)$$

The community-wide WSAs for spawning, YOY and adult life stages for the 41 comparison sites were then ranked from highest to lowest. To successfully meet the criteria under Measure 1A, the Bay of Quinte sites must rank above the 80th percentile, or within the top 9 sites out of 41, among all the nearshore areas of Lake Ontario being evaluated.

Measure 1B:

The methodology for Measure 1B, a comparison of WSA between the Bay of Quinte and inner Long Point Bay, Lake Erie, is similar to the methodology for Measure 1A. While the Bay of Quinte and Long Point Bay are unique locations, the fact that they both provide historically important and vast areas of vegetated and nearshore habitat to their respective lakes makes Long Point the best representative of a relatively unimpacted reference site for the Bay of Quinte that exists in the Great Lakes basin. Georgian Bay wetlands could be another reference area with similar geology but there is not enough spatial information to conduct this comparison. Underlying physical differences between the locations are caveats in this comparative assessment. Another caveat is that data used in the model for Long Point were higher in resolution than those used in the Bay of Quinte model.

The basis of the habitat model is the same as in Measure 1A. Reaches, or habitat units, were assigned physical habitat attributes of water depth, vegetation, substrate, temperature and water levels. The input data and study areas used to define the upper and middle Bay of Quinte regions are the same as in Measure 1A. Physical habitat data for Long Point were held in a Geographic Information System (GIS) and were derived from several data sources. Water depths were obtained from a Digital Elevation Model developed by researchers at the University of Waterloo (Cabrera et al. unpub. data) and a time series of weekly water levels. Annual extent and percent cover of vegetation were modelled using the University of Waterloo Emergent Vegetation Model (Hebb et al. 2006, Cabrera et al. unpub. data) and the DFO-GLLFAS Submerged Aquatic Vegetation Model (Doka et al. 2006) with modifications. Substrate composition for Long Point was derived from several sources (Geomatics International Inc. 1997; Minns et al. 1999; Present and Acton 1984; Rukavina 1976; Thomas et al. 1976) and held constant through time. The assumption of substrate being constant through time was necessary because neither coastal processes nor mussel invasion were modelling explicitly in Long Point or Lake Ontario. The Long Point temperature model was developed using daily near-surface water temperature data for Port Dover (Zhao and Witzel, pers. comm.) and a temperature-depth profile model from Doka (2004). Habitat changes in Long Point were modelled on a weekly basis as opposed to daily in the Bay of Quinte and Lake Ontario analysis. The model time-step should not impact the average annual cumulative WSA comparisons.

Using the habitat variables, water levels and fish-habitat associations, habitat suitabilities (0-1) and WSAs (suitability-weighted hectares) were calculated for each guild and life stage by habitat unit (reach or patch) on a daily or weekly basis from 1951-2000. Weekly values were assumed to be the same across days of the week. WSAs by life stage were then summed across days and averaged across years (see Formula, Measure 1A, Page 24) and guilds, using the guild-based weights assigned in Measure 1A. Because the upper and middle sections of the Bay of Quinte and inner Long Point Bay are not the same size, WSA was standardized across the study sites by dividing WSA by the ratio of Long Point to the Bay of Quinte reach clusters (see 1A). The area within the 0-20 m water depth range under the average water level condition for each site during the modelled time period was used to standardize areas. The area ratio was 0.88:1 at an average water level of 74.8 m ASL in the Bay of Quinte and 174.3 m ASL in Long Point Bay. The standardized WSA was summed across days and averaged across years. Standardized, average annual cumulative WSA within and across life stages at each location was used as the comparative metric in Measure 1B. To successfully meet the criteria under Measure 1B, the relative average fish community-weighted cumulative WSA in the Bay of Quinte must be within 10% of the relative WSA for the same fish community in the inner Long Point Bay area. To accomplish this specific comparison it was necessary to use the Bay of Quinte fish community was used to generate the Long Point WSAs for additional standardization. Fish guild membership is shown in Appendix B.

<u>Measure 2A:</u>

Weighted suitable area (WSA) was assessed for three time stanzas (Table A1-1) using the Habitat Alteration and Assessment Tool (HAAT, as described above). Habitat classifications were based on three variables: depth (water level - elevation), vegetation, and substrate. For each time stanza, three water level scenarios were modelled to demonstrate the possible variety in habitat suitability across a range of possible conditions. High, low and average water levels were modelled for each stanza (Table A1-1). High and low water level years were chosen based on historical high- and low-water levels within each specific time stanza. The year closest to the stanza average water level was chosen to represent average conditions.

Table A1-1 Time stanzas modelled, with highest, lowest and average mean annual water level recorded in each stanza.

Time Stanza	Years	Lowest water level	Highest water level	Average water level
Pre Phosphorous	1972-1977	1977: 74.64m	1973: 75.21m	1972: 74.94m
Post Phosphorous	1978-1994	1988: 74.68m	1986: 75.14m	1979: 74.84m
Pre *Dreissena*	1994-2011	1999: 74.57m	1997: 74.95m	2002: 74.78m

Digital Elevation Data. The digital elevation model was assembled using elevation data from various sources including Canadian Hydrographic Services (CHS) and Ontario

Ministry of Natural Resources (OMNR). CHS datasets included hydrographic surveys of varying resolution. Most of the terrestrial data were gathered from 1-m resolution LiDAR provided by OMNR. A gap in coverage along the Trent River was filled with a 10-m DEM provided by OMNR. Terrestrial elevations were converted from Canadian Geodetic Vertical Datum to International Great Lakes Datum 1985 by subtracting 0.102 from elevations. Data were compiled into a single point layer, and converted to a raster digital elevation model using ArcGIS's Spline tool.

Emergent Vegetation. Emergent vegetation (one component of wetlands) were classified using output from the University of Waterloo Wetland Vegetation Model (Hebb et al. 2006, Cabrera et al. unpub. data). This model classifies wetlands within regions delineated by aerial photography interpretation. Vegetation classifications were based on current depth and wetted duration conditions (i.e. time since flooding or exposure). The model had already been applied to selected areas in Bay of Quinte (Hebb et al. 2006), so these results were used to create a lookup table to classify Canadian Wildlife Service wetland polygons for the whole Quinte area. A caveat of this approach would be that all wetlands are assumed to behave similarly to the previously modelled ones. In areas where emergent vegetation was predicted to occur in a given year, the emergent vegetation model would take precedence over the submerged aquatic vegetation model predictions.

Submerged Aquatic Vegetation. Predictive models were created based on field data collected by DFO-GLLFAS in the Bay of Quinte from 1972 to 2004. Data collection methods varied over the vegetation time series: from 1972 to 1982 knotted-line transects were used; paper-trace echosounders / quadrats were used from 1988 – 2000; and from 2004 to 2011 sampling included hydroacoustic soundings and quadrats. Details are outlined in Leisti et al (2012). Data samples increased dramatically in later years which affected data analysis but also increased confidence in estimates. All data were treated as point samples recording depth and percent cover. Annual mean Secchi depth, total phosphorous and ePar from the nearest DFO-GLLFAS long-term monitoring station were associated with each vegetation sample point. Slope, substrate and fetch were extracted from raster data using ArcGIS Spatial Analyst and also tested in model generation but where not statistically significant variables in determining vegetation presence. This is specifc to the Quinte model and is likely because the range of those variables in Quinte is not grea enough to have an influence (e.g. the fetch is relatively low throughout the bay because it is predominantly sheltered) . There were considerably more records, and at a much higher spatial resolution (<1m) in the later datasets because they were collected with hydroacoustics. To resolve this oversampling issue, a logistic model was created by subsampling 200 records that were at least 30m apart from each of 2004 and 2007 datasets , and then building the logistic model to avoid swamping of early trends due to large differences in sample size. The final logistic model for maximum vegetation depth is described by the equation , which predicts the probability of SAV presence, given depth and Secchi (increasing positive values have a greater probability of vegetation presence):

$$Cover_{prob.} = 10.13 - 5.52 * {}^{1}/_{Secchi} - 3.81 * \sqrt{Depth}$$

which achieved a Naglekerke pseudo-R^2 of 0.3.

The second component of the SAV model predicted percent cover or density whereas the first model predicted the maximum depth that vegetation should colonize. Vegetation density has a high amount of variability across individual points samples but average trends were apparent in the data. To analyse trends, percent cover was averaged within 1-m depth classes, by bay and by year (NB: each unique bay-year combination had a unique Secchi value). Although variable, the mean depth class at which maximum percent cover usually occurred was between 1-2 m. The mean maximum percent cover at this depth range was described by the model:

$\%Cover_{Max} = 53.8 * \ln(Secchi) + 43.2$ (N=19, R^2 = 0.204)

Average percent cover within the 0-1m depths did not significantly differ from maximum cover so was held constant. Percent cover at depths deeper than 2-m declined linearly on average from the mean maximum cover to zero at the depth of maximum colonization determined by the logistic model. It should be noted that these are generalized models and the error rates and confidence limits vary and are discussed in Leisti et al. (2012).

Substrate. A substrate layer was compiled using Ponar field samples collected by the OMNR nearshore inventory and offshore survey (Thomas & Rukavina, unpub. data) from 1972 to 1991 (Minns et al 2006). Field samples were classified using a modified Wentworth scale (Bain and Stevenson 1999). Thiessen polygons were created from these field samples to create a spatial substrate layer for the Bay of Quinte. Soil surveys tend to be costly and time consuming, and the data are not easily interpolated, so Thiessen polygons are a reasonable approach to modelling substrate (Minns et al. 1996). Additionally, terrestrial substrate data were collected from CanSYS soil survey data. These data were converted to the modified Wentworth scale based on the soil description (percent volume estimates for sand, silt, clay, and "stoniness" or gravel, sometimes with two soil classes described). If the CanSYS soil data lacked descriptions, percent composition was assigned based on the soil family name (i.e. Farmington Loam assumed to be pure loam). These CanSYS polygons were superimposed on the Thiessen polygons from field samples.

Measure evaluation: using the detailed spatial data on depth, vegetation and substrate described above and the general WSA methodology described in Measure 1, WSA was calculated for high, low and average water level conditions during the three time stanzas (Table A1-1). For each guild and life stage under each scenario, an overall suitability for the Bay of Quinte was calculated by dividing WSA by total wetted area. This allowed us to look at changes in habitat suitability over time for different portions of the fish community while providing some correction for differences in wetted area due to water levels. Other differences remain due to difference depth regimes under different water levels and these effects could not be removed. To evaluate Measure 2A, we looked at trends in WSA and suitability over time for the whole fish community and sensitive portions of the fish community.

Measure 2B:

Using the results generated in Measure 2A, we mapped suitabilities for a range of fish guilds for use in conservation and planning decisions. Where general patterns emerged between guilds, maps were merged by using the maximum suitability across guilds for each grid cell in the final maps. Areas that provided highly suitable habitat to the fish community as a whole, or for particular fish guilds, across water levels within the post-zebra mussel time stanza only are shown in this section. Caveats for using the maps include: 1) not all variables that could affect fish habitat suitability have been included in these particular analyses (e.g. temperature) but have been included in previous analyses, 2) neither zebra mussels as substrate, nor algae as vegetation, has been included as elements of fish habitat specifically, and 3) if a habitat cell was considered of high suitability for any group or at any water level scenario then it was considered of high value in the final map for planning and conservation purposes.

Measure 3A:

To evaluate Measure 3A, we compared relative population abundances for four sport fish populations across the same 41 coastal reference areas used in Measure 1A. The habitat data used is the same as that described in Measure 1A. Habitat-linked population models for Largemouth Bass, Northern Pike, Smallmouth Bass and Yellow Perch were developed by Chu et al. (2006). The models were run from 1962-2010. Species-specific WSAs were calculated for each life stage (spawning, fry, YOY, juvenile and adult) on a daily basis (i.e. daily water levels and temperatures were used as input). The habitat model inputs and flows are described under Measure 1A. Species-specific fish-habitat association information were used to derive habitat suitability values. Information from Lane et al. (1996a,b,c), references within Chu et al. (2006), and expert opinion (Minns et al.2005) was used to generate the species specific suitability assignments.

The habitat-linked population models take daily species and life stage-specific WSA inputs and track populations on an annual basis (Chu et al. 2006; Figure A1-3). In the spring or summer when spawning WSA is available, mature fish spawn. Growth and survival of the resultant eggs and fry is temperature-dependent. Subsequently young-of-the-year grow daily based on standard bioenergetic equations and survive based on temperature curves, and density-dependent habitat availability functions. Juvenile and adult fish grow bioenergetically given water temperature and density–dependent growth and survival functions based on habitat availability were calculated on an annual basis for these stages. Years 1962-1971 were used to initialize the model and allow populations to stabilize before outputs were used to compare population results between time stanzas (Pre-P, Post-P/Pre-D, Post-D).

To evaluate Measure 3A, the average population size (#/ha) for each species between 1972 and 2010, given habitat availability, was calculated for the 41 coastal comparison sites. The relative population size across sites was used to assess whether the Bay of Quinte ranked within the top 80th percentile of sites in Lake Ontario as per the new criteria. Similarly to Measure 1A, it should be noted that the habitat inputs for the species-specific models do not incorporate development or water quality changes over the decades on habitat. Effectively this creates a comparison of modern day habitat conditions but incorporates water level and temperature variability into the calculations and their impact on vegetation and habitat supply respectively. Changes due to water quality are incorporated into Measure 3B.

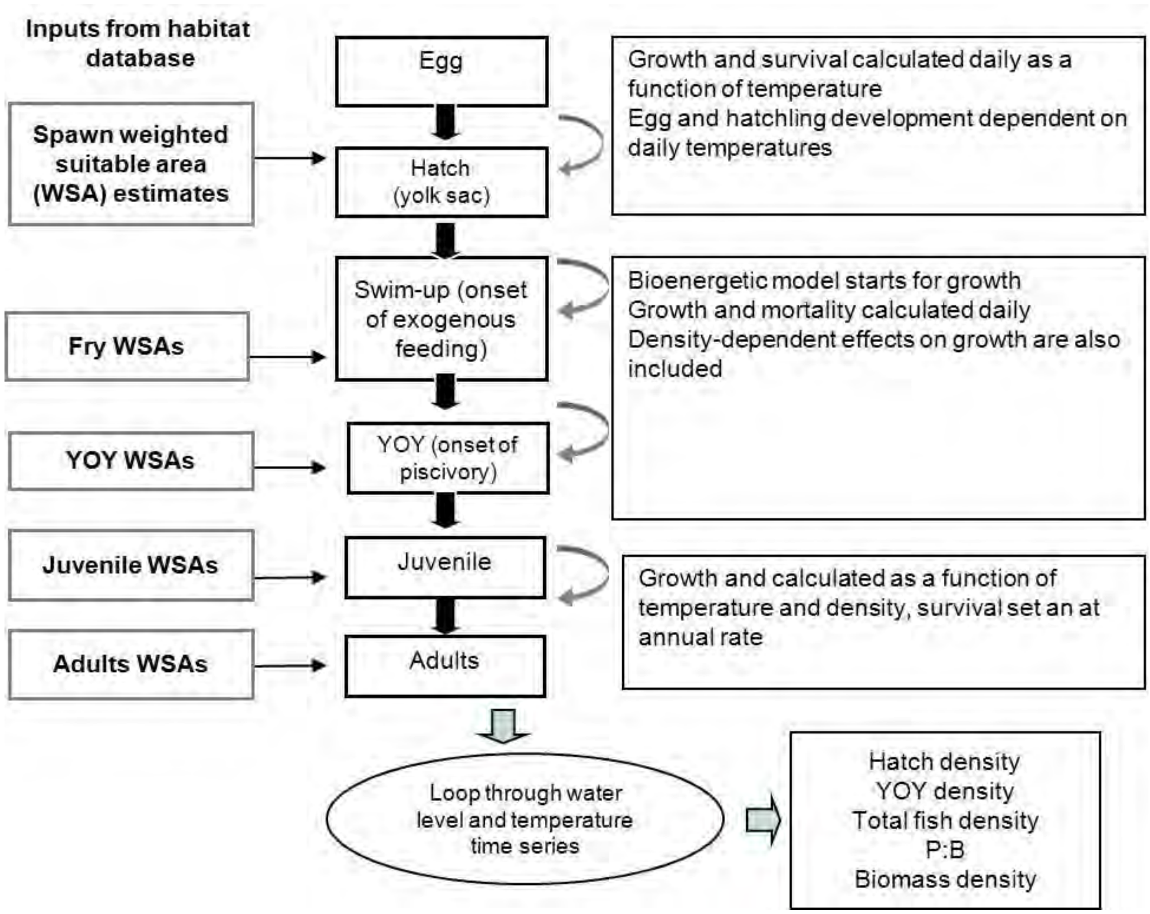

Figure A1-3. Flow of inputs and outputs from a generic habitat-linked population model showing life stages and how growth and survival is calculated at each stage (from Chu et al. 2006).

<u>Measure 3B:</u>

To evaluate Measure 3B, we examined trends in relative biomass per hectare over time in the Bay of Quinte for four simulated sport fish populations. The methodology for the development and application of the habitat-linked population models is described under Measure 3A and Figure A1-3, above. The detailed Bay of Quinte habitat data for elevation/depth, vegetation and substrate is the same as that described in Measure 2A. Daily water levels measured at Kingston, Ontario and daily sub-surface water temperatures from the Belleville Water Treatment facility were as inputs instead of simulated lakewide levels in 3A. As described in Measure 1A, daily depth for each patch / grid cell was calculated using the daily water level information and up-to-date elevation data. Daily temperature at bottom was calculated using a Bay of Quinte calibrated temperature @ depth model. The Quinte SAV model was modified on an annual basis using average annual Secchi depth for the upper and middle bays (M. Burley, Fisheries and Oceans Canada, unpub. data). The emergent vegetation model output as described in Measure 1A was used annually,.

Habitat inputs to the population models, based on the underlying models outlined above, were calculated on a daily basis from 1962-2010 and populations were tracked on an annual basis. It is important to note that the habitat-linked population models evaluated changes in populations over time given changes in water clarity (Secchi depth), water levels, and temperature only. Other factors that may be important to fish populations and their habitats, such as shoreline development, zebra mussels as substrate type, and fishing pressure, are not included in this evaluation of fish populations. Trends in biomass, density and abundance of the four fish populations (Yellow Perch, Northern Pike, Smallmouth and Largemouth Basses) from 1972 and 2010 were evaluated. Changes in biomass per hectare across the time period were tested for significant trends.

APPENDIX B

Species list by thermal/piscivory guild

	Coldwater Adult	Cool-water Adult	Warm-water Adult
Piscivore	Brown Trout Burbot Chinook Salmon Coho Salmon Lake Trout Rainbow Trout	American Eel Longnose Gar Northern Pike Walleye	Bowfin Grass Pickerel Largemouth Bass Muskellunge Smallmouth Bass White Bass
Non-piscivore	Alewife Cisco Lake Whitefish Mottled Sculpin Rainbow Smelt Round Whitefish Slimy Sculpin Trout-Perch	Banded Killifish Blackchin Shiner Bridle Shiner Brook Stickleback Common Shiner Emerald Shiner Fallfish Fantail Darter Golden Shiner Greater Redhorse Johnny Darter Lake Sturgeon Logperch Longnose Dace Mooneye Ninespine Stickleback Quillback River Redhorse Round Goby Rudd Sauger Sea Lamprey Shorthead Redhorse Silver Redhorse Spottail Shiner Threespine Stickleback White Sucker Yellow Perch	Bigmouth Buffalo Black Crappie Bluegill Bluntnose Minnow Brook Silverside Brown Bullhead Central Mudminnow Channel Darter Channel Catfish Common Carp Fathead Minnow Freshwater Drum Gizzard Shad Goldfish Northern Hogsucker Pumpkinseed Rock Bass Tadpole Madtom Sand Shiner Spotfin Shiner Stonecat White Perch

Species lists by adult thermal/vegetation guild

	Coldwater Adult	Coolwater Adult	Warmwater Adult
High Vegetation	Mottled Sculpin (NOTE: merged w coolwater spp. for analysis)	American Eel Banded Killifish Blackchin Shiner Bridle Shiner Brook Stickleback Common Shiner Golden Shiner Johnny Darter Logperch Longnose Gar Ninespine Stickleback Quillback Round Goby Rudd Spottail Shiner Striped Shiner Threespine Stickleback White Sucker Yellow Perch	Bigmouth Buffalo Black Crappie Bluegill Bluntnose Minnow Bowfin Brook Silverside Brown Bullhead Central Mudminnow Common Carp Fathead Minnow Goldfish Grass Pickerel Largemouth Bass Muskellunge Pumpkinseed Rock Bass Tadpole Madtom
Low Vegetation	Alewife Brown Trout Burbot Chinook Salmon Cisco Coho Salmon Lake Trout Lake Whitefish Rainbow Smelt Rainbow Trout Round Whitefish Slimy Sculpin Trout-Perch	Emerald Shiner Fallfish Fantail Darter Greater Redhorse Lake Sturgeon Longnose Dace Mooneye River Redhorse Sauger Sea Lamprey Shorthead Redhorse Silver Redhorse Walleye	Channel Catfish Channel Darter Freshwater Drum Gizzard Shad Northern Hogsucker Sand Shiner Smallmouth Bass Spotfin Shiner Stonecat White Bass White Perch

Species lists by spawning thermal/vegetation guild

	Coldwater Spawn (8-12 °C)	**Cold/coolwater Spawn** (12-16 °C)	**Cool/warmwater Spawn** (16-20 °C)	**Warmwater Spawn** (20-28 °C)
High Vegetation	Grass Pickerel Muskellunge Ninespine Stickleback Northern Pike White Perch Yellow Perch	Bigmouth Buffalo Black Crappie Brook Stickleback Central Mudminnow Common Carp Fathead Minnow Spottail Shiner	Blackchin Shiner Bluegill Bluntnose Minnow Bowfin Bridle Shiner Brook Silverside Gizzard Shad Goldfish Largemouth Bass Longnose Gar Rudd	Banded Killifish Brown Bullhead Golden Shiner Pumpkinseed Tadpole Madtom
Low Vegetation	Burbot Cisco Lake Trout Lake Whitefish Longnose Dace Mottled Sculpin Northern Hogsucker Quillback Rainbow Smelt Rainbow Trout Round Goby Round Whitefish Sauger Shorthead Redhorse Silver Redhorse Slimy Sculpin Walleye White Sucker	Alewife Fallfish Fantail Darter Johnny Darter Lake Sturgeon Logperch Smallmouth Bass Threespine Stickleback Trout-Perch White Bass	Channel Darter Common Shiner Freshwater Drum Greater Redhorse Rock Bass Striped Shiner	Channel Catfish Emerald Shiner Sand Shiner Spotfin Shiner Stonecat

Species lists by Nursery or YOY thermal/vegetation guild

	Coldwater Spawn / Nursery	Cold/coolwater Spawn / Nursery	Cool/warmwater Spawn / Nursery	Warmwater Spawn / Nursery
High Vegetation	Grass Pickerel Muskellunge Ninespine Stickleback Northern Hogsucker Northern Pike Quillback Round Goby Silver Redhorse White Perch White Sucker Yellow Perch	Bigmouth Buffalo Black Crappie Brook Stickleback Central Mudminnow Common Carp Fantail Darter Fathead Minnow Johnny Darter Logperch Spottail Shiner White Bass	Blackchin Shiner Bluegill Bluntnose Minnow Bowfin Common Shiner Freshwater Drum Gizzard Shad Goldfish Largemouth Bass Longnose Gar Rock Bass Striped Shiner	Banded Killifish Brown Bullhead Emerald Shiner Golden Shiner Pumpkinseed Sand Shiner Tadpole Madtom
Low Vegetation	Brown Trout Burbot Chinook Salmon Cisco Lake Trout Lake Whitefish Longnose Dace Mottled Sculpin Rainbow Smelt Rainbow Trout Round Whitefish Sauger Shorthead Redhorse Slimy Sculpin Walleye	Alewife Fallfish Lake Sturgeon Mooneye Sea Lamprey Smallmouth Bass Threespine Stickleback Trout-Perch	Bridle Shiner Brook Silverside Channel Darter Greater Redhorse	Channel Catfish Spotfin Shiner Stonecat

APPENDIX J: Coastal Wetland Macroinvertebrate Populations Delisting Criteria Status Report

October 22, 2015

Authors
- Angela Darwin, Landscape Assessment Officer, Canadian Wildlife Service, Environment and Climate Change Canada
- Greg Grabas, Habitat Ecologist, Canadian Wildlife Service, Environment and Climate Change Canada

Recommendation: Delisting criterion has been met

DELISTING TARGETS	DELISTING CRITERIA	STATUS
FWH-8: Aquatic macroinvertebrate communities in Bay of Quinte coastal wetlands are not impaired.	FWH-8: The aquatic macroinvertebrate community IBI at representative Bay of Quinte coastal wetlands shall not be more than two standard deviations below the 2006-2010 representative site mean that has been corrected for varying conditions in Lake Ontario outside of the AOC from 2006-2010.	Achieved

Background

Coastal wetland macroinvertebrate communities are important in nutrient cycling and as food for other fauna. Aquatic macroinvertebrates are the primary food source for many fish species. In addition, species that emerge as flying insects provide many species of amphibians and wetland-dependent birds with a forage base.

Aquatic macroinvertebrate community structure is also known to reflect current or recent water quality in Great Lakes coastal wetlands (Gathman et al. 1999, Kostuk 2006). Through the development of a coastal wetland aquatic macroinvertebrate Index of Biotic Integrity (IBI), Environment Canada and Central Lake Ontario Conservation Authority (2004) have demonstrated the aquatic macroinvertebrate community structure in Lake Ontario coastal wetlands is affected by anthropogenic disturbance. The disturbance estimates used in the IBI development rely heavily on the integration of water quality parameters.

As a food source and water quality indicator in coastal wetlands, the nektonic and epiphytic aquatic macroinvertebrate community condition can be used, in combination with other factors such as submerged aquatic vegetation (SAV) community condition and water quality, as an indicator of aquatic habitat quality.

For each wetland, three replicate sub-samples of approximately 150 aquatic macroinvertebrates ($\geq 500\mu m$) were taken by sweep-netting through the water column in the cattail (*Typha* spp.) dominated emergent communities. These samples represent a combination of primarily nektonic and epiphytic species assemblages – not benthic. Macroinvertebrates were identified to the lowest taxonomic group possible.

IBI Development: An aquatic macroinvertebrate (nektonic and epiphytic) community IBI has been finalized by the Great Lakes Coastal Wetland Consortium (GLCWC) (Uzarski et al. 2008). The lead investigators developed an IBI for macroinvertebrates inhabiting vegetation zones such as meadow marsh and dense *Scirpus* zones, while conceding they were unable to identify suitable metrics in the *Typha* vegetation zone. However, unlike meadow marsh and *Scirpus* vegetation zones, *Typha* zones are omnipresent in Lake Ontario coastal wetlands. With access to more extensive Lake Ontario-based data than those used by GLCWC lead investigators, Environment Canada – Canadian Wildlife Service (EC-CWS) developed the aquatic macroinvertebrate IBI (EC and CLOCA 2004). In previous reporting, 11 metrics were used to calculate the macroinvertebrate IBI (Environment Canada 2007). The value of a good IBI is that it demonstrates a gradient against disturbance. Within year, it had been observed that the IBI scores across Lake Ontario sites would sometimes display a gradient against disturbance, but most often these metrics yielded scores that did not display an appreciable range in IBI values (G. Grabas, Environment Canada, pers. comm., March 2009). The previously developed IBI was refined to increase the resolution between invertebrate communities in poor and excellent condition. It was suspected that the IBI scores were becoming diluted by the number of metrics used in the calculation so the suite of potential metrics was re-evaluated. Seven years of lake-wide data (2003, 2006-2011) were used identify potential invertebrate community metrics (2004, 2005 data were from only the Durham Region and did not represent a suitable range of disturbance). Candidate metrics were based on the same 19 metrics tested in Burton et al. (1999). The methods used to screen and score suitable metrics are based on Grabas et al. (2012). Pearson's correlations with Chow-Fraser's (2006) Water Quality Index (WQI) were used to determine which metrics were used to derive the final invertebrate IBI. Metrics were retained for the IBI if they met two criteria in at least four of the seven study years: 1) the metric was significantly correlated (α=0.05) with WQI and 2) the Pearson's correlation coefficients were at least 0.35. The retained metrics are:

- Number of Ephemeroptera (mayflies) + Trichoptera (caddisflies) genera [NETG]
- Total number of families [NFAM]
- Percent Trichoptera [PTRI]
- Percent Crustacea [PCRU]

All the retained metrics were standardized with Minns et al.'s (1994) continuous scoring method. This method uses a linear function to transform raw metric data into standardized metrics, ensuring a minimum and maximum value of 0 and 10, respectively (Minns et al., 1994, equation 1). For metrics that decrease with increasing disturbance (e.g., Number of Ephemeroptera and Trichoptera genera), standardized metric (M_S) was scored zero if the raw metric (M_R) was a minimum value (either zero or the 10th percentile; see Appendix A). The value of M_R where M_S scored 10 was the 75th percentile of all values of the metric across all sites and all years (2003-2011). If M_R was at or above the 75th percentile, M_S was scored 10. Raw metrics falling between zero and the 75th percentile were standardized using Minns et al. (1994; equation 1: $M_S=A+B \cdot M_R$) and coefficients from Appendix A. Once all metrics were standardized, the metrics for each wetland-year were summed, multiplied by 10, and divided by the total number of metrics to create an IBI with scores between zero and 100 (Minns et al., 1994). Higher scores indicated aquatic macroinvertebrate communities in relatively better condition.

Site Selection and Surveys: The Bay of Quinte Remedial Action Plan Restoration Council (BQ-RAPRC;2007) recognized that sampling and reporting on the status of delisting criteria for

BUIs 3 and 14 in all Bay of Quinte Area Of Concern coastal wetlands would not be feasible. A subset of 10 representative sites (coastal wetlands) was selected (Table 1). BQ-RAPRC (2007) indicates that the sites were chosen to cover a range of disturbance, size (small sites, <25 ha, were excluded), and dominant hydrologic influence (riverine/lacustrine). Environment Canada (2007) suggested that Solmesville Marsh was included in this list based on the belief that its size was 28 ha, but as the actual size was determined to be only 2.8 ha, Airport Creek Marsh (29 ha) was selected as the recommended alternative.

Since the representative sites were not finalized by BQ-RAPRC (2007) before the summer of 2006, EC-CWS, unaware that Carnachan Bay Marsh would be a representative site, did not conduct field surveys there during that field season. From 2007 to 2013, monitoring efforts have been coordinated by the BQ-RAPRC with assistance from EC-CWS and were focused on the 10 BQ AOC representative or sentinel wetlands, comprised of 14 subsites (Table 1). Although most representative sites are located in the Upper Bay of Quinte, this is indicative of the overall distribution of wetlands within the bay (Figure 1).

Table 1. Bay of Quinte Area of Concern representative coastal wetlands for monitoring

Wetland Name	Number of Subsites per Wetland
Airport Creek Marsh	1
Big Island Marsh	2 (East and West)
Blessington Creek Marsh	1
Carnachan Bay Marsh	1
Carrying Place Marsh	1
Dead Creek Marsh	1
Hay Bay Marsh	2 (North and South)
Lower Napanee River Marsh	1
Lower Sucker Creek Marsh	1
Sawguin Creek Marsh	3 (Central, Ditched, and North)

Additional coastal wetlands are surveyed in Lake Ontario through various projects such as the Durham Region Coastal Wetland Monitoring Project and the Coastal Habitat Assessment and Monitoring Program (Figure 1). The number of other Lake Ontario sites surveyed each year for macroinvertebrates varies from a low of 14 in 2007 to a high of 25 in 2009 and 2012 (Appendix B, Table 3). Wetlands surveyed consistently across years are primarily in the Durham Region. Only wetlands with a minimum of two years of data were retained for inclusion in the analysis.

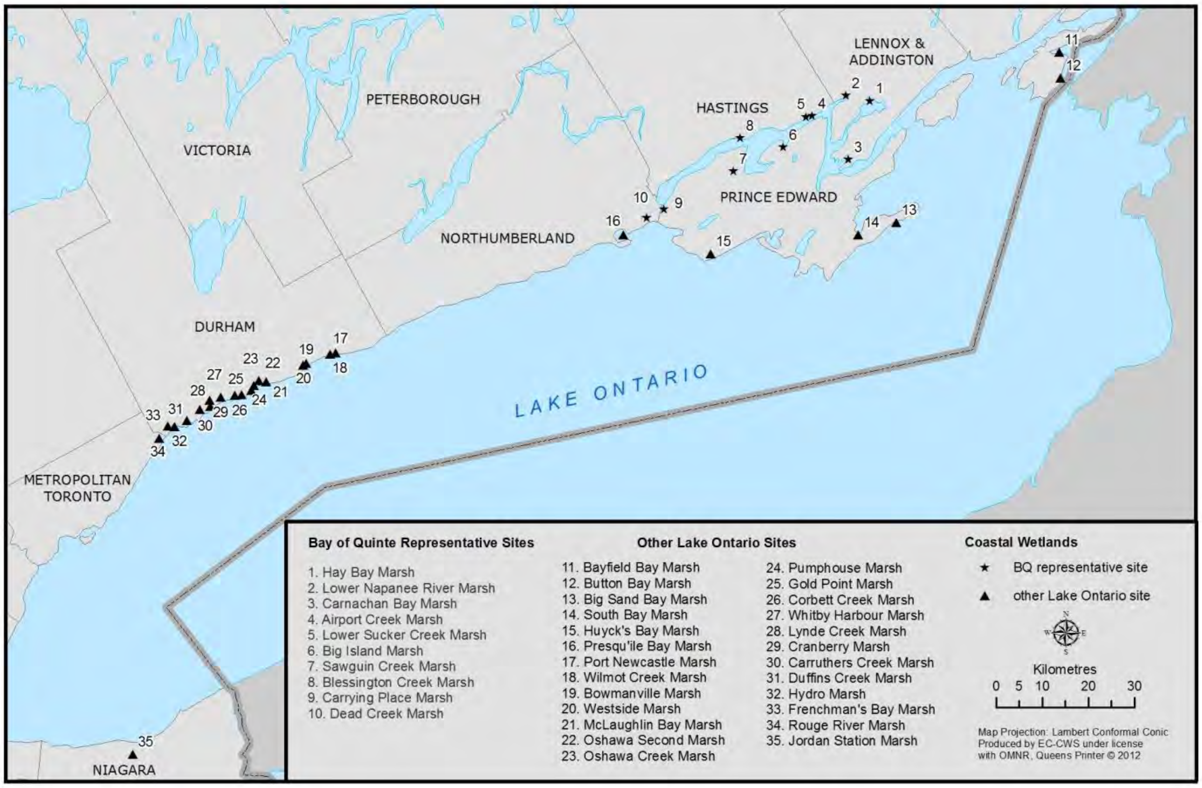

Figure 3. Names and locations of Bay of Quinte representative sites and other Lake Ontario coastal wetlands sampled by EC-CWS and partners. Refer to Appendix B for list of sites surveyed for macroinvertebrates.

Rationale for Delisting Criterion Status

Determining the threshold below which the 2014 mean Macroinvertebrate Community IBI (of all Bay of Quinte (BQ) representative sites) must not fall to be considered not impaired requires multiple steps:

1. Calculate an annual mean from 2006 to 2010 macroinvertebrate IBIs from all BQ representative sites
2. Calculate an overall mean and standard deviation of 2006 to 2010 IBIs from all BQ representative sites
3. Multiply the standard deviation by 2
4. Subtract the 2 standard deviations from the 2006-2010 mean
5. Calculate an annual mean from 2006 to 2010 macroinvertebrate IBIs from all 'other Lake Ontario' sites
6. Calculate an overall mean of 2006 to 2010 IBIs from all 'other Lake Ontario' sites.
7. Calculate the 2014 mean from all 'other Lake Ontario' coastal wetlands surveyed
8. Subtract the 2005-2010 'Lake Ontario' mean from the 2014 'Lake Ontario' mean
9. Add the results of Step 4 and Step 8
10. The resulting number is the corrected threshold above which the 2014 mean macroinvertebrate community IBI for BQ representative sites must fall to be considered 'not impaired'.

Data used for these calculations are found in Appendix B (Tables 2 and 3).

2014 Bay of Quinte Representative Sites Mean IBI = 80.0

2014 Corrected Threshold IBI = 54.3

Since the Bay of Quinte 2014 IBI mean is greater than the threshold IBI value, the macroinvertebrate community is considered to be **not impaired**. Therefore, the criterion for the macroinvertebrate community to be considered not impaired has been met in each year from 2011 to 2014 (Appendix 2, Table 4).

Future Monitoring or Actions Required

Regular monitoring of the macroinvertebrate community should continue, for the near future at least, at the Bay of Quinte representative sites and numerous other Lake Ontario coastal wetlands.

Glossary

Nektonic Macroinvertebrates	Invertebrates large enough to be seen with the naked eye that swim and migrate freely in aquatic environments
Epiphytic Macroinvertebrates	Invertebrates large enough to be seen with the naked eye that live primarily on aquatic plant material

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Appendix A: Macroinvertebrate IBI Calculation

Data are summarized as a mean per site.

Step 1: Standardize the relative abundance and species richness metric scores to values out of 10. (Higher scores out of 10 indicate better macroinvertebrate community attribute).

Metric Code	Calculation
NETG	If NETG ≥2, then the metric automatically scores a 10 If NETG <2, then multiply the value by 5
NFAM	If NFAM ≥ 19.67, then the metric automatically scores a 10 If NFAM <=12.67, then the metric automatically scores a 0 If NFAM <19.67 or >12.67, then multiply the value by 1.43 and subtract 18.10
PCRU	If PCRU ≥46.44, then the metric automatically scores a 10 If PCRU <=14.20, then the metric automatically scores a 0 If PCRU <46.44 or >14.20, then multiply the value by 0.31 and subtract 4.40
PTRI	If PTRI ≥1.12, then the metric automatically scores a 10 If PTRI <1.12, the multiply the value by 8.96

Step 2: Combine the standardized metrics and calculate an IBI out of 100.
- For each site, add the standardized metrics, divide the sum by 4 and then multiply by 10.
- For wetlands with multiple sites, a mean of the output values is generated.

Appendix B: Macroinvertebrate IBI values for 2006-2014

Table 2. Macroinvertebrate IBI for Bay of Quinte representative sites for 2006-2014.

Wetland	2006	2007	2008	2009	2010	2011	2012	2013	2014
Airport Creek Marsh	86.0	68.6	68.2	74.5	84.8	47.2	75.0	83.5	91.5
Big Island Marsh	35.4	62.1	49.5	89.2	76.7	79.7	62.1	52.3	67.4
Blessington Creek Marsh	68.3	77.0	73.7	94.0	78.0	90.7	74.6	79.0	95.2
Carnachan Bay Marsh	-	47.7	83.5	62.5	100.0	94.1	72.4	68.2	56.3
Carrying Place Marsh	60.8	65.8	86.8	87.1	76.8	71.3	42.3	53.1	71.1
Dead Creek Marsh	65.7	60.1	94.0	76.7	80.9	92.1	50.0	56.5	77.9
Hay Bay Marsh	61.7	86.4	93.8	96.5	72.1	90.2	66.5	79.6	80.8
Lower Napanee River Marsh	81.8	87.3	94.3	84.0	93.5	97.7	57.5	83.8	95.3
Lower Sucker Creek Marsh	47.7	38.9	65.8	77.1	98.2	93.3	70.2	68.8	94.1
Sawguin Creek Marsh	47.4	50.5	72.9	75.7	82.5	79.9	62.3	87.7	70.7
Mean IBI	**61.6**	**64.4**	**78.3**	**81.7**	**84.3**	**83.6**	**63.3**	**71.2**	**80.0**
# wetlands	9	10	10	10	10	10	10	10	10

Note: Hay Bay North was not sampled in 2009.

Table 3. Macroinvertebrate IBI for all other Lake Ontario sites for 2006-2014.

Wetland	2006	2007	2008	2009	2010	2011	2012	2013	2014
Bayfield Bay Marsh				49.8		79.7	29.0		
Big Sand Bay Marsh				79.3		40.7			
Bowmanville Marsh	24.3	31.1	39.8	53.7	22.4	31.0	31.5	12.5	37.1
Button Bay Marsh				80.1		60.9	57.6		
Carruthers Creek Marsh	38.7	46.2	47.2	51.2	46.4	9.2	39.1	17.9	49.1
Corbett Creek Marsh	27.5	50.0	62.6	54.2	87.8	55.5	38.4	68.8	83.6
Cranberry Marsh	25.0		32.2	42.9	69.9	68.9	29.2	40.4	44.1
Duffins Creek Marsh	50.8	32.4	26.5	57.1	60.8	47.0	40.5	23.9	23.9
Four Mile Pond				38.7			43.2		
Frenchman's Bay Marsh	37.5	62.2	62.9	63.7	41.5	56.4	58.5	21.5	37.8
Goldpoint Marsh				39.3	16.7	30.4	17.9	20.9	35.2
Hydro Marsh	6.2	48.0	40.3	41.7	28.4	4.2	27.2	0.0	11.2
Jordan Station Marsh				17.8			50.0		
Lynde Creek Marsh	46.5	39.1	31.5	54.2	61.0	25.1	84.4	18.5	32.8
McLaughlin Bay Marsh	35.2	44.1	35.6	59.9	69.8	53.6	49.9	37.5	70.8
Oshawa Creek Marsh				37.1	22.1	17.9	24.5	19.1	19.7
Oshawa Second Marsh	38.2	48.3	51.2	63.3	40.7	15.0	29.8	17.3	26.2
Parrott's Bay Marsh						98.7	93.4		
Port Newcastle Marsh	18.2	28.5	49.9	35.0	50.2	75.8	71.8	58.5	67.9
Presqu'ile Bay Marsh				79.7		86.3	62.3		
Pumphouse Marsh	43.3		30.2	57.8	48.6	60.2	41.1	20.8	73.4
Rouge River Marsh	32.1	82.4	61.5	73.3	63.9	69.6	89.1	56.1	71.2
South Bay Marsh				96.5		94.2	90.8		
Westside Marsh	57.4	47.4	87.1	87.1	79.7	73.0	81.1	71.1	85.2
Whitby Harbour Marsh		27.8		29.0	24.3	27.1	37.5	37.4	32.8
Wilmot Creek Marsh	30.6	21.7	62.5	67.8	28.3	42.2	44.1	62.9	41.8
Mean IBI	**34.1**	**43.5**	**48.1**	**56.4**	**47.9**	**50.9**	**50.5**	**33.6**	**46.9**
# wetlands	15	14	15	25	18	24	25	18	18

Table 4. Mean aquatic macroinvertebrate IBIs of Bay of Quinte (BQ), 'Other Lake Ontario sites', and corrected threshold values from 2011-2014.

Mean IBI	2011	2012	2013	2014
'Other Lake Ontario sites'	50.9	50.5	33.6	46.9
BQ IBI threshold	58.3	57.8	41.0	54.3
Bay of Quinte IBI score	83.6	63.3	71.2	80.0
Criterion met (Y/N)	**Y**	**Y**	**Y**	**Y**