Appendix A

Dr. Robert Hecky, Recent Studies of Cladophora glomerata and its Relation to Environmental Factors in the Great Lakes with Special Reference to the Ajax-Pickering Coastal Region on Lake Ontario
Recent studies of *Cladophora glomerata* and its relation to environmental factors in the Great Lakes with special reference to the Ajax-Pickering coastal region on Lake Ontario

By R.E. Hecky

**Introduction**

*Cladophora glomerata* is a filamentous green alga (of division Chlorophyta) which grows attached to hard geologic substrata, such as bedrock and cobble, in lakes and rivers throughout North America as well as around the world with the exception of Arctic habitats. It may be one of the most widely distributed benthic algal species known which attests to the success of its morphological, physiological and ecological traits. The plant grows from a set of basal cells strongly attached to non-mobile substratum and grows by adding cells sequentially (acropetal growth) to the basal filament so youngest cells are at the end of the filaments. The filament can branch off new filaments from cell junctions so that the attached filament becomes highly branched as it grows (Fig. 1). The filaments appear to become more branched with higher water velocities (Dodds and Gudder 1992). Consequently a dense stand of *Cladophora* can form a tangled network of filaments that may effectively filter particulate matter and bacteria from the ambient water. Although the filaments are multicellular, there are no plasmodesmata (microscopic strands of cytoplasm) to facilitate intercellular exchange or transport of nutrients or cellular metabolites. Consequently each cell of the filament must meet its own metabolic needs for the filament to maintain integrity. The filaments do extend in length and under optimum growth conditions can add a centimeter of length per day (Bellis and McLarty 1967) with most growth occurring in late May and early June as day length increases and water temperatures rise above 10 C. Growth typically slows under higher temperatures and as nutrient concentrations fall. As the filaments come under physiological stress, they weaken and become susceptible to detachment (sloughing), often on a large scale in dense stands. During mid-summer, typically mid to late July in the Great Lakes, large masses of *Cladophora* slough off the bottom and substantial portions of the detached material can beach onto shorelines or enter water intakes creating aesthetic and operational issues for coastal communities.
Dodds and Gudder (1992) emphasize the growth habit of the plant is strongly linked to hydrodynamic conditions. The significant advantage that *Cladophora* has over microscopic benthic algae is that its elongation and branching can elevate the plant out of the diffusive boundary layer along the bottom of lakes and rivers and enhance its access to CO2 for photosynthetic growth as well as potentially overcoming transport limitation of other nutrients by the boundary layer. The ability of the plant to form networked mats of filaments would also lead to the shading out of microscopic benthic algae. These traits may explain the success of the plant in nutrient enriched environments where rapid growth can result in long filaments which can lead to the plant extending up into the water column 50-100 cm. However such long filaments will create substantial drag in moving waters and eventually lead to detachment in strong currents.

The ability of the plant to thrive in nutrient-rich waters has been appreciated since the 1960s, and a review by Whitton (1970) has numerous references to *Cladophora*, including nuisance stands of *Cladophora* in the Great Lakes where it was considered to be responding to phosphorus (P) enrichment. Concern about excessive algal growth of both phytoplankton and benthic algae led to the International Joint Commission recommending that phosphorus point sources be reduced and P be removed from detergents. The USA and Canada subsequently negotiated and signed the Great Lakes Water Quality Agreement (GLWQA) of 1972 which agreed that the governments would reduce P loading to meet targets for both P loading to the lakes and P concentrations in the offshore regions of the lakes in order to reduce excessive algal growth and protect other beneficial uses of the Great Lakes.

Despite the implementation and achievement of the P loading and concentration targets called for by the GLWQA, *Cladophora* has re-emerged as an issue in the lower Great Lakes since the early 1990s (Higgins et al. 2008). Excessive *Cladophora* growth, generally expressed as public concern over shoreline fouling but also impacting water intakes and fishing nets, once again is leading to complaints and often heated discourse over the cause of the excessive growth and shoreline fouling across the lower Great Lakes (Lakes Michigan, Lake Erie and Lake Ontario. A current example of this concern has emerged in regards to the capacity limitations of the Duffin Creek Water Pollution Control Plant (York-Durham Region) outfall which is the subject of a class environmental assessment. This report has the objective to review recent data and studies relevant to the ecology, growth and abundance of *Cladophora* in the Great Lakes.
with special reference to Lake Ontario and to the Ajax-Pickering shoreline near the Duffin Creek WPCP. The report that follows has three sections. The first section concerns *Cladophora* in the Great Lakes and how water quality changes in the lakes have impacted critical habitat features which control the abundance and distribution of the alga in the lakes. These habitat features are: availability of hard substrata for attachment, availability of light to allow photosynthetic growth and the availability of nutrients to sustain algal growth (Higgins et al. 2005). The second section reviews recent studies, post 1990, specific to Lake Ontario which are relevant to defining and understanding the status of *Cladophora* in Lake Ontario. The third section reviews recent studies in the vicinity of Ajax-Pickering which are relevant to understanding the distribution, production and abundance of *Cladophora* in the area of the Duffin Creek WPCP. These three sections will be followed with conclusions and recommendations for furthering the understanding of the *Cladophora* issue.

I. Changes in Environmental Conditions relevant to *Cladophora* Growth in the Great Lakes following the Great Lakes Water Quality Agreement (1972)

The GLWQA set targets for total phosphorus (TP) loading and TP concentrations in the open waters of all the Great Lakes (Table 1). The targets were based on simple mass budget models for P and expected responses of lower chlorophyll concentrations (as an indicator of phytoplankton biomass) to acceptable levels which would be more similar to the lakes’ conditions prior to the European colonization of the watersheds. To meet these targets national, state and provincial governments limited P in detergents and upgraded waste water treatment plants to reduce P emissions. Non-point sources of P such as atmospheric loading and agricultural runoff were not directly addressed in most instances. The success of these actions led to rather rapid reduction in TP loading (Fig. 2) and TP concentrations (Fig. 3), and by the late 1980s targets were being met. Chlorophyll concentrations in the pelagic (offshore) regions of the lake followed the decline in P availability in the lakes. Because chlorophyll is a major absorber of light, light transmission in the lakes also improved. In the late 1980s, the zebra mussel (*Dreissena polymorpha*) had successfully invaded Lake St. Clair (likely introduced in ballast water), and by the late 1990s it occurred extensively in all the lakes with the exception of the Lake Superior where it is restricted to harbors. The zebra mussel has been replaced over much of the lakes by the quagga mussel (*Dreissena bugensis*). The positive and planned
improvements in P concentrations and transparency under the GLWQA were accelerated and amplified after the establishment of dreissenid mussels in Michigan, Huron, Erie and Ontario. The dreissenids reached high densities in the lakes, particularly Lake Michigan, Erie and Ontario, and their filtering mode of feeding greatly increased the removal of phytoplankton from the lakes and increased the loss of P from the invaded lakes (Fig.3).

The changes in the transparency of the Great Lakes since 1970 are visible from space. Binding et al. (2007) compared satellite imagery for two time periods 1979 to 1985 (post implementation of the GLWQA and 1998-2005 (post establishment of dreissenids). They quantified the images for water-leaving irradiance at 550 nm which correlated very well with Secchi disk depth (a standard field measurement of water transparency; Fig.4). Their results showed that all the lakes have become more transparent in the past 30 years with the exception of the western and central basin of Lake Erie (Fig. 5) which have been suffering increased incidence of algal blooms (Michalak et al. 2013) because of non-point source agricultural pollution by soluble reactive P (SRP which is the most bioavailable component of TP). The greatest change in water leaving irradiance and transparency has occurred in Lake Ontario and the regions of greatest change in Lake Ontario are the inshore areas along the north shore of the lake (Fig. 6).

The evidence of declining P concentrations comes primarily from monitoring programs that are largely focused on the offshore regions of the lakes, but the loading reductions required to effect the declining concentrations impact the coastal areas most directly. Therefore, it can be presumed that near shore P concentrations would be expected to decline as well as the offshore. Because P is considered the most likely nutrient to limit Cladophora growth, the decline in P would be expected to reduce Cladophora growth. So prior to the establishment of mussels a decline in P available for Cladophora growth in the near shore would be expected from the overall declines in offshore P concentrations. The establishment of mussels may have altered that expectation through a process termed the near shore P shunt (Hecky et al. 2004) which will be discussed below. The increase in transparency especially as recorded in Lakes Michigan, Erie eastern basin and Ontario would have the dual effect of extending the benthic area where Cladophora might grow and also supplying more light energy for photosynthesis at all depths down to the light limit for Cladophora growth. Of course growth in this increased area of habitat
still requires hard substrate for attachment of *Cladophora*, but the overall impact of increasing transparency should be favorable for areal expansion of habitat and increased production of *Cladophora*. Mussels, therefore, reinforced trends of reduced TP (while also providing a local source of recycled SRP) and higher transparency initiated by the GLWQA, but they also had the effect of increasing the availability of hard substrata in the nearshore coastal zone where shell lags have extended the area of hard substrata (Coakley et al. 1997) thereby providing increased area for attachment of both mussels and *Cladophora*.

**Cladophora in the Great Lakes post GLWQA**

Early field studies in the Great Lakes established that three critical habitat features, hard substratum, light and phosphorus, were necessary for the promotion of *Cladophora* growth and abundance. Noteworthy in these early studies were Mantai et al. (1982), Neil and Jackson (1982), and Auer et al. (1982). Mantai et al. studied two sites in eastern Lake Erie in 1977 and noted that high biomass of *Cladophora* (over 400 g DW m⁻²) developed on a hard substratum (breakwall) near the lake surface. They monitored both the internal concentration of P (tissue P) as well as ambient SRP and noted that SRP (from below detection to over 8 µg P L⁻¹) was much more variable through the growing season than tissue P and showed that tissue P responded rapidly to take up available SRP in excess when SRP became available at higher concentrations (phenomenon of luxury consumption) allowing *Cladophora* then to grow even when ambient SRP was very low. They also observed a rapid loss of biomass in mid-July that was not associated with any drop in SRP or tissue P. Neil and Jackson (1982) in a study at Rathfon Point on Lake Erie in 1979 also demonstrated that highest algal biomasses were at the shallowest depths sampled, and they also observed the rapid development of biomass in June and rapid loss of standing crop of biomass in July (Fig. 7). Up to the time of the significant sloughing event, tissue P was always lower at the shallowest depths which had the highest light exposure and highest biomass. They also did a field scale addition of SRP onto a near shore section of the coast line and demonstrated that growth of *Cladophora* would respond to the addition of SRP alone. Consequently, they predicted that P reduction in Lake Erie should result in a reduction of *Cladophora* biomass along suitable substrata shorelines in Lake Erie.

The most comprehensive field study of the time was undertaken on Lake Huron near Harbor Beach, Michigan where a waste water treatment plant was discharging into a shallow
near shore area in 1979 (Auer et al. 1982). Lake Huron at the time of the study was already in an oligotrophic condition with TP concentrations at or near the GLWQA target (Fig. 3). The impact of the waste treatment plant on ambient SRP, tissue P and Cladophora biomass was clearly evident with the impact being reduced with distance from the sewage treatment plant (Fig. 8) as lake water eventually diluted the high concentrations emitted by the sewage plant. A remarkable feature of this study was the dependence of nutrient concentration at the study site on wind conditions (Fig. 9). Low winds resulted in higher concentrations while higher winds reduced concentrations presumably as a result of increased flushing of the study site by open lake water. The study also established the role of wind events on the loss of biomass from the shallow study site where they found that significant loss of biomass was attendant on the strength and frequency of wind events. This study clearly established the impact of the waste treatment plant (without any cap on emitted P concentrations) on near shore SRP and its positive stimulation of Cladophora growth near the plant, but it also demonstrated how hydrodynamics (water movements) could influence the near shore zone and its response to Cladophora growth. The authors emphasized this aspect in the abstract of the article.

“The high degree of spatial and temporal heterogeneity in the nearshore environment was clearly evident from the results of the monitoring program. Wind direction and velocity importantly affect dissolved phosphorus distribution. Internal phosphorus levels reflect long term trends in wind and current regimes. The growth of Cladophora is importantly influenced by internal phosphorus levels while the loss of algal material through sloughing is closely correlated with wind-storm events. The observations of nearshore dynamics point to the need for high sampling frequency in such systems.”

In the discussion, Auer et al. (1982) further define “high sampling frequency” as:

“Definition of transience, in some cases, requires sampling as frequently as on a weekly basis. Daily observations of weather conditions, especially wind direction and speed, are indispensable. Second, measurements of internal phosphorus are important as they afford relatively easy interpretation, are less subject to high frequency variation, and tend to integrate the effects of external conditions over a period of time. Also, the timing of sampling frequency must be such that representative sloughing events, occurring over the entire growth season, are observed and correlated with wind stress.”

This study established that knowledge of substratum, light and SRP from infrequent measurement would not be enough to fully evaluate the spatial and temporal dynamics of Cladophora growth in the complex near shore regions of lakes. In addition detailed studies of
physical conditions and hydrodynamics would be necessary because the hydrodynamics can strongly modify these factors and also determine the loss of biomass from a site.

**Cladophora Growth Models in the Great Lakes post-GLWQA**

Observational data provide only limited ability to make predictions about future states to guide management action especially for complex phenomena such as *Cladophora* growth and biomass accumulation in the Great Lakes. Canale and Auer (1982) took on the challenge of bringing a wealth of field observations in combination with careful laboratory cultural studies defining the dependence of *Cladophora* on P concentrations (both SRP and tissue P) and light to develop a mathematical model which was then verified at the Harbor Beach site described in Auer et al. (1982). Full descriptions of the various steps in developing parameter relationships for the model are found in a series of papers in a full issue of the *Journal of Great Lakes Research* to the “Ecology of filamentous algae” introduced by Auer (1982). The model was successfully verified along a shallow sequence of segments along the shoreline away from the waste treatment plant. The model also was verified through application to the eventual control of P emissions at the Harbour Beach site. The original model’s structure (Fig.10) and parameterization has more recently been modified to improve aspects of the model (Auer et al. 2010) and to broaden its application (e.g. Tomlinson et al. 2010 the Great Lakes Cladophora Model or GLCM; Higgins et al. 2005 the *Cladophora* Growth Model or CGM; Malkin et al. 2008 CGM with sloughing event trigger); however, the basic physiological core of the model and its dependence on light and tissue P is common to these modified models. Tomlinson et al. (2010) give a detailed comparison of the differences between the original model, the CGLM and the CGM (Table 2). A major revision of the upgraded GLCM is the conclusion that *Cladophora* growth-sensitivity to SRP is greater than originally thought and lower concentrations of SRP would have to be achieved to reduce growth potential than suggested in the original model. To quote the conclusion of the authors:

“… the nutrient domain over which *Cladophora* growth will be responsive to management actions is characterized by very low concentrations of soluble reactive phosphorus (on the order of 0.2–1.0 μg P/l). This will require a thorough understanding of nearshore P dynamics and challenge our ability to relate loading reductions to ambient P levels.”

The original model had suggested that *Cladophora* growth control could be achieved through reductions below 2 μg P/l. This lower range suggested for control overlaps concentrations often
observed in the open lake and are at or below the limit of detection with standard methods of measuring SRP. This opens up the possibility that the open lake can be an important source of P to drive Cladophora growth. It should also be noted that, in the original application of the Canale and Auer model to the Lake Huron coastal site, the open lake could only serve as a diluent to the near shore segments (Fig. 10) because P concentrations in Lake Huron were considered to be too low to support Cladophora growth (Canale and Auer 1982).

The Dark Age of Cladophora science in the Great Lakes

Despite the successful model for predicting Cladophora growth by 1982 and the recognition that Cladophora had been degrading the beneficial uses of coastal area in the lower Great Lakes during their eutrophication, studies on Cladophora in the Great lakes waned after the 1982 issue of Journal of Great Lakes Research with only six publications in that journal on Cladophora between 1984 and 2004 compared to 26 publications between 1980 and 1984 (Auer et al. 2010). The general expectation that Cladophora abundance would decline after implementation of the GLWQA seems to have been realized although there is little to no monitoring data or studies on the lakes to support that expectation with evidence. The sole exception to this dearth of information that is most frequently cited in support of declining Cladophora after GLWQA is a study by Painter and Kimaitis (1987) which showed the expected improvement on samples from seven sites on Lake Ontario collected between 1972 and 1982-1983. In six of the seven sites measured mean Cladophora biomass collected from 0.5, 1.5, 3.0 and 5.0 m by SCUBA divers declined significantly between 1972-1982 (Fig. 11) and at all sites the mean tissue P declined significantly (Fig. 12). The authors used Secchi disk transparency and tissue P to estimate net specific growth rate with the Canale and Auer model and concluded that growth rates had declined (Fig. 13). Consequently, it was assumed that Cladophora growth across all the lakes had declined and that decline accounted for the absence of monitoring data and public concern. This sense of satisfaction began to be disturbed by the late 1990s and early 2000s when complaints again emerged to the level of public concern (Higgins et al. 2005). The lack of information in the face of public concern is perhaps best illustrated by a figure taken from the 2009 State of the Great Lakes report (available through IJC website) which places questions marks on the near shore zone of all the great lakes with the lower lakes near shore zones being in poor condition. The report specifically mentions the reappearance of Cladophora as an
outstanding issue. Once again studies began to be initiated on *Cladophora* across the lower Great Lakes (especially Michigan, Erie and Ontario) where there was evidence of increased abundance of *Cladophora* (Auer et al. 2010). Because of this long term gap in monitoring and studies of *Cladophora* biomass and growth in the Great Lakes, the impact of the establishment of the dreissenid mussels in the early to mid-1990s across the lower lakes has been problematic although it had been invoked as a possible cause for the re-emergence of *Cladophora* in many near shore areas of suitable substrate.

**Dreissenid impact on Cladophora habitat, growth and biomass**

The rapid spread of dreissenids into and across the lower Great Lakes has had widespread impact on native species, food web structures and the spatial and temporal patterns of primary productivity in the Great Lakes. After a planktonic egg and larval stage, dreissenids settle and attached to hard substratum, including other dreissenid shells, while the quagga mussel has also spread on to cohesive depositional sediments in deeper water. Their occupancy of hard surfaces is nearly total in the lower lakes although the shallowest depths, 0-2 m often very low abundances are found because of winter ice scour and high turbulence discouraging attachment of the veliger larvae. Densities of over 10,000 m⁻² have been observed and their capacity to filter water from water columns to which they have access represents a marked increase of mortality on the lakes’ phytoplankton populations. For example, Pennuto et al. (2012) report that it can be estimated that mussels can filter the near shore water volume in 0 m- 20m in water depth in 1-7 days. Hecky et al. (2004) attempted to integrate the many impacts imposed by dreissenid occupancy and to evaluate their ecosystem effects with special reference to the impact of dreissenids on P cycling in the near shore zone of the Great Lakes. The near shore P shunt (Fig. 14) illustrates the logic of the shunt. It illustrates that prior to dreissenid occupancy offshore waters passed through the near shore environment with little modification except the addition of dissolved and suspended materials added by coastal runoff and tributaries which were then transported to the offshore. The advent of high densities of mussels created the ability to harvest offshore food resources in the near shore as well as intercepting the transport of terrestrial materials that formerly passed through to the offshore for eventual deposition in the deep waters of the lakes. Excretion by mussels therefore can regenerate SRP from tributary and offshore sources of particulate P and provide a new internal source of SRP in the nearshore. Although the
mussels did not add to the inventory of P in the lake (in fact their long term impact seems to have been to increase the loss of lacustrine P by increasing the removal rate; see Fig. 3), they did represent a new source of SRP which was not available to the near shore environments of the Great Lakes prior to mussel occupancy. The dreissenids share their preference for hard surfaces in the near shore with *Cladophora* at least to depth of the euphotic zone (because *Cladophora* must be able to achieve a positive net photosynthesis to maintain itself and grow) and so they occur together along Great Lakes shorelines. However, dreissenids also can dramatically increase water transparency because of their filtering activities (e.g. Fig. 15 and Fig. 5 above), and this will also directly increase *Cladophora* habitat by extending the depth of the euphotic zone across the lakes. The near shore shunt also predicts that SRP availability should markedly increase in the near shore zone because of the excretion and defecation of the dense mussel populations. Dreissenids have broadly invaded aquatic ecosystems across North America, and a recent meta-analysis (Higgins and van der Zanden 2010) of over 200 aquatic ecosystems (including the Great Lakes) have confirmed the impact of mussels on increasing water transparency (Fig. 16), reducing phytoplankton populations by approximately 50% and increasing the abundance of benthic plants by 180% (Fig. 17).

The hypothetical impact of dreissenids on *Cladophora* growth in the Great Lakes proposed by the near shore shunt has now been tested by a variety of study approaches from field surveys to modeling. Auer et al. (2010) applied *Cladophora* models (CGLM and CGM) to historical monitoring data sets that spanned from before to after dreissenid establishment. The general expectation was that if SRP had become less available after dreissenids as observed across the lakes (Fig. 3), but light transparency had not, then the maximum biomass observed on a shoreline should decrease; but there would be no change in the depth of the euphotic zone nor the maximum depth of *Cladophora* growth (Fig. 18). Conversely, if dreissenids did not increase SRP availability but did increase transparency, then depth of biomass growth would increase with little impact on the maximum realized biomass. Of course in reality, both effects may also operate. Auer et al. (2010) gathered necessary input data from various monitoring programs with a focus on late spring data and calculated *Cladophora* production for a two week spring period for a depth transect on a uniform slope and 100% *Cladophora* cover. They concluded that observed increases in light transparency post-dreissenids would increase *Cladophora* growth potential by 40% to 50% on Lakes Michigan, Erie and Ontario even with no change in P
availability (Fig. 19a). However, if light transparency were held constant while observed changes in SRP after dreissenids were applied, the results were mixed (Fig. 19b) with only Lake Erie having a modeled increase in growth potential while the other lakes declined in modeled potential. If the model was run with observed changes in transparency and SRP availability, Lake Erie had the greatest increase in modeled growth potential while Lake Michigan had the least (Fig. 19c). The authors readily acknowledged the limitations of their analysis primarily in regards to SRP values used which were not from the near shore domain where Cladophora lives and in particular their model does not account for the possibility of direct supply from dreissenids to Cladophora as hypothesized by the near shore P shunt:

“The shunt phenomenon is of potential significance with respect to monitoring programs, as samples collected from nearshore waters in the traditional fashion may not represent conditions in the algal/mussel beds. From a management perspective, the enhancement of within-lake P cycling (internal loading) resulting from zebra mussels may act to offset the benefits of loading reduction.”

The impact of excreted SRP on near shore loading of SRP was investigated separately by Ozersky et al. (2009) in Lake Ontario at Oakville, Ontario and Bootsma (2009) in Lake Michigan at Milwaukee, Michigan. In both cases, mussels were estimated from surveys while excretion of SRP was measured in situ (Ozerky) or in the lab (Bootsma). Despite the different methods to measure SRP excretion their estimates per ash free dry weight (i.e. living tissue) were remarkably similar with Ozersky’s mean values falling in the mid-range of those measured by Bootsma which accounted for differences in sizes of mussels. They each scaled their estimates of SRP excretion against the demand for SRP by Cladophora over a specified area of shoreline segment occupied by Cladophora. The segment in each case included major tributaries and the impact of water pollution control plants. The Milwaukee WPCP emptied into the Milwaukee River which then entered the modeled segment while the Oakville WPCP plant discharged directly into the lake segment studied so it was possible to separate the tributary impact from the waste treatment plant. Ozersky et al. found that the mussel excretion accounted for more than enough SRP to meet the demand for SRP by Cladophora growth and was 4x all the other sources of SRP including the WPCP to the modeled segment (Table 3). Similarly, Bootsma (Table 4) found that mussel excretion of SRP was four times Milwaukee River loading (of total P) and more than >4X Cladophora growth SRP demand in his shoreline segment. These independent studies clearly establish that SRP flux from dreissenid mussels in the near shore of the Great
Lakes can meet and even exceed the areal SRP demand to support nuisance growths of *Cladophora*. However, these mussel SRP loading estimates could not be transformed to their effective impact on SRP concentrations that could drive growth models, and so the potential of mussel excretion to fuel *Cladophora* growth remained hypothetical.

A recent study (Dayton et al. 2014) at a northern Lake Michigan site, Sleeping Bear Dunes National Lakeshore, has provided novel insight to how and when mussel excreted P may impact *Cladophora* growth. Sleeping Bear Dunes Lakeshore near shore waters are remote from any urban influences but appreciable stands of *Cladophora* occur as does most of the rocky western shoreline of Lake Michigan which is also remote from direct urban influences (Bootsma et al, 2004). However, Lake Michigan is heavily colonized by dreissenid mussels (Nalepa et al. 2009). The Dayton et al. study demonstrated that, in quiescent (low wave heights, <20 cm), SRP concentrations over mussels beds (site 8.5 m depth) increased within a benthic boundary layer of 10-20 cm in thickness (Fig. 20). During more turbulent conditions (higher wave heights), there was no buildup of SRP as excreted SRP was rapidly diluted into the overlying water column where SRP concentrations were estimated as averaging 0.4 µg/L (effectively below detection limit). These observed concentrations in the boundary layer were capable of causing luxury uptake by *Cladophora*, and the GLCM simulated that one day of exposure to these high concentrations in the boundary layer could raise internal tissue P concentrations sufficiently to drive *Cladophora* growth for 14 days. Physical observations at the site indicated that the recurrence of quiescent conditions (Fig. 21) was frequent enough to meet the growth requirement for SRP for the entire growing season. Although strictly speaking, the results are only directly representative of the depth and physical and chemical conditions at the study site the authors’ do conclude that:

“This finding, addressing the role of mass transport in the mussel – phosphorus – *Cladophora* dynamic, offers confirmation for the nearshore shunt hypothesis (Hecky et al., 2004) and supports the conclusion of Ozersky et al. (2009) that dreissenid mussel excretion can provide more phosphorus than is required for the support of the alga.”

It is noteworthy that high SRP concentrations were observed under both high (5000 mussels m⁻²) and low mussel densities (500 mussels m⁻²) suggesting that physical conditions may be more important to building up high SRP concentrations than mussel density *per se*. In any case the
potential of the mussel excretion to meet the SRP demands of *Cladophora* growth is confirmed by the study.

A different approach to establishing the possible significance of dreissenids and land use factors to *Cladophora* growth was taken by Depew et al. (2011). Depew et al. developed a hydroacoustic method for measuring significant and nuisance stands of *Cladophora* along Great Lakes shorelines (Depew et al. 2009). This methodology overcomes the limitations of estimating areal biomass from small samples from a limited number of depths along a selected transect which may or may not be representative of a larger segment of variable shoreline habitat. In effect, the hydroacoustic method samples thousands of data points for estimating cover and algal height over kilometers of shoreline. Depew et al. investigated *Cladophora* biomass stands along shorelines which had different land uses which ranged from provincial parks to major urban areas (Fig. 22; Fig. 23). They used published estimates of dreissenids along the same shorelines and measured a broad suite of water quality indicators at multiple locations for each surveyed shoreline while controlling for bathymetry and substrata. To maximize the range of water quality and dreissenid conditions, Depew et al. (2011) included sites from Lake Huron (lowest dreissenid impact), Lake Erie and Lake Ontario. *Cladophora* stands were also sampled for tissue P along these same survey sites (Houben et al. 2007). Depew et al. also included two shallow, offshore submerged banks (Nanticoke Shoal and Dobbs Bank, Fig. 22) in their study to reduce, if not eliminate, all shoreline influence on the observed *Cladophora* biomass. Exemplary results are presented in Fig. 24 (an urban shoreline at Oakville ON) and Fig. 25 (provincial park at Presqu’Ile ON). Multi-variate analysis using hierarchical partitioning of all the observed variables indicated that mussel abundance was the only significant parameter explaining variance in *Cladophora* biomass (Table 5). Although the off shore banks were not included in the statistical analysis because no value could be assigned to shoreline land use of these sites, the results from the banks were supportive of the role of dreissenids. At comparable water depths these banks had *Cladophora* biomass per unit area similar to coastal growths and these growths were accomplished under SRP and TP concentrations typical of offshore waters in Lake Erie and Lake Ontario.
Offshore lake P or P from terrestrial runoff

Taken together, the independent results of Dayton et al. (2014), Depew et al. (2011), Ozersky et al. (2009) and Bootsma (2009) seem to establish that dreissenids can be a very significant source of SRP in the near shore and that hydrodynamics likely determine how and when the excreted SRP becomes available to Cladophora. It is often stated that the SRP released is recycled and does not provide new SRP to the lake. This is true for the whole-lake, but it is not true for the near shore segments where dreissenids are now abundant. If the phytoplankton, bacteria and organic particles consumed by dreissenids, originate from the open lake, then the consumption and regeneration of P by dreissenids represent a new source of SRP to the near shore that did not exist prior to mussel occupancy. It also means that the open lake is the source of P feeding mussel excretion; and therefore, control and reduction of offshore concentrations of these food resources will be necessary to reduce the regeneration of SRP in the near shore. The management consequence of this conclusion is that reducing local inputs from land runoff or WPCPs may not have the intended benefit of significantly reducing Cladophora abundance. Coordinated lake-wide action, as implemented under the GLWQA, may be required to reduce Cladophora in the near shore.

The impact of mussels relying on particulate material from local runoff would have different consequences. If terrestrial runoff were the most significant source of food for mussels, then reducing these local sources may reduce mussel abundance and improve local near shore conditions. Under this scenario, the regenerated SRP from coastal mussels may only represent an acceleration of regeneration of bioavailable P that would eventually be regenerated over larger spatial scales in the lake. The importance of terrestrial food sources for mussels versus consumption of lake phytoplankton remains an understudied aspect of the near shore shunt paradigm. Indirect evidence from eastern Lake Erie suggests mussels are most dependent on lake resources. This can be established using stable isotopes of C to trace food resources. Upsdell (2005) demonstrated that major and minor tributaries contributed particulate organic material with a much lighter (more negative) δ\textsuperscript{13}C (<-30 per mil) than open lake and near shore particulate material (>26 per mil; Fig. 26). Campbell et al. (2009; Table 6) concluded from stable isotope analysis that open water particulate material (seston which includes phytoplankton; Table 5) was the only significant food source (closest match between isotopic
signature of dreissenids and possible food resources) for dreissenids in the nearshore. At least for this shoreline on Lake Erie which is strongly influence by agricultural drainages (Fig. 23 Peacock Point and the Grand River), dreissenids are dependent on phytoplankton/seston material and not material contributed by local tributaries. This aspect, dependence on offshore phytoplankton or terrestrial material, should be further explored. However, the observation of Depew et al. (2011) that offshore banks can support *Cladophora* growths comparable to nearshore coastal areas is consistent with the hypothesis that offshore P resources alone can sustain nuisance *Cladophora* growth in Lake Erie and Lake Ontario.

II. Lake Ontario studies relevant to coastal *Cladophora*

**Lake wide water quality changes**

Lake Ontario historical trends in P and light transparency (Secchi depth) as monitored by Environment Canada's Great Lakes Surveillance Program were reviewed by Malkin et al. (2010). Malkin et al. reviewed the available data for both near shore and offshore locations in Lake Ontario (Fig. 27); whereas the review of Chapra and Dolan (2012; Fig. 3) only used data from offshore stations because they considered these more representative of the large volume of the lake. However the relationship between near shore and offshore water quality conditions may be relevant to near shore processes such as *Cladophora* growth. So Malkin et al. (2010) divided the available stations into three groups, those near shore (<14-20 m water depth) on the north and south coasts as well as deep water offshore (>20 m) stations. The near shore stations exhibited the same declines in TP and SRP as the offshore station (Fig. 28) in both spring (March through May) and summer (July through September). In fact by 1985 near shore TP and SRP were meeting targets set for offshore Lake Ontario in both spring and summer (Fig. 28). There is no evidence that near shore P conditions have deteriorated on the north shore of Lake Ontario since that time. Malkin et al. (2010) also compared data for two shallower (<12 m to 2 m depth) inshore areas at Halton and at Durham for several recent years, and these littoral areas had concentrations of TP and SRP comparable to or lower than the offshore stations in spring and summer as well as meeting offshore target concentrations for north shore stations (Fig. 28).
recent (2004-2008) spring TP and SRP concentrations in the 2-12 m depth range are dramatically lower at Halton and Durham compared to MOE data from 1972-79 (Fig. 29). There is no evidence in these data sets that water column P concentrations are significantly higher near shore than offshore in recent decades nor that there has been any upward trend in P concentrations in recent years.

The downward trends observed for offshore and near shore P concentrations can also be observed in coastal tributaries. Malkin et al. (2010) calculated annual loading data (Fig. 30) from MOE Provincial Water Quality Monitoring program and Environment Canada (Water Survey of Canada) discharge records for two north shore tributaries, Sixteen Mile Creek discharging into the lake at Halton and Duffins Creek at Ajax, two areas where there have been reports of increased shoreline fouling by *Cladophora* in recent years. The reductions in TP and SRP loading are almost entirely from reductions in P concentrations, i.e. minimal in discharge, and the reductions are substantial, e.g. a 4x reduction in SRP loading at Sixteen Mile Creek and nearly 10x in Duffins Creek. These streams have greatly improved water quality in regards to P, and their contribution to increases in *Cladophora* growth would seem to be minimal over the past decade.

Secchi Disk values in the monitoring data sets reviewed by Malkin et al. display a slight upward trend (improving transparency) on the north shore up to about 1990, but the greatest change in north coast Secchi Disk transparency occurs after 1990 in both spring and summer. The change in Secchi depths is the result of declining chlorophyll values since 1990 following the invasion and establishment of dreissenids. These results confirm the temporal and spatial patterns of changes in Secchi Disk transparency based on remote sensing (Fig. 6 above) reported in Binding et al. (2007). Binding et al. state in their abstract that “Results confirm dramatic reductions in Lake Ontario turbidity in the years following mussel colonization, with a doubling of estimated Secchi depths.”

**Hindcasting with the Cladophora Growth Model**

The impact on *Cladophora* growth of these changes in lower P concentrations and increased transparency in Lake Ontario have been modelled for a littoral shoreline at Halton ON
using the CGM (Malkin et al. 2008). Malkin et al. first calibrated and validated the CGM for their site by comparing modeled and observed *Cladophora* biomass over two very different meteorological years, 2004 and 2005. 2005 was warmer and had less cloud cover while 2004 was cooler and cloudier with less surface light. They then used these two years as base years to compare the modeling results for earlier years similar to Auer et al. (2010) except that the Malkin et al study modeled growth over the full growing season (not just a two week spring period) and used tissue P concentrations (not ambient SRP) to drive *Cladophora* growth. The latter input data was made possible because there were historic tissue P data available from 1972 and 1983 for the Halton area (Painter and Kimaitis 1987; Fig. 31) to which Malkin et al. could add their own data collected in 2005 and 2006. Malkin et al. also performed a sensitivity analysis similar to Auer et al. (2010) by holding tissue P constant at 2004-2005 levels while allowing light penetration to change over time and alternatively holding light penetration constant at 2004-2005 levels and allowing tissue P to change as observed over time (Fig. 32).

The comparisons of model runs for the different years and assumptions allowed them to establish the relative sensitivity of *Cladophora* growth to tissue P concentrations and light transparency. The attached or accumulative (accounting for sloughing losses) biomass was integrated from 0.5 to 12 m depth to allow comparison of results as total integrated biomass (kg) per meter of shoreline length. Allowing light penetration to decline to historic levels did reduce biomass by nearly a factor of 2 compared to the modern conditions and also shoaled the depth of maximum biomass. However, changes in tissue P (Q_P in Fig. 31 and Table 7) had by far the greatest effect with a 4 to 6 fold increase in biomass per unit of shoreline. Malkin et al emphasize that the tissue P concentrations remain low in shallower depths which means that any increase of P availability in the lake could lead to a rapid increase in shoreline biomass, all other factors being equal. When both light (K_d_par in Table 7) and tissue P (Q_P)are allowed to vary as they did historically, it is clear that the mean *Cladophora* peak biomass per unit shoreline at Halton in 2004-2005, 55 kg m^{-1}, is likely well below *Cladophora* in 1982-3, 77 kg m^{-1}, or 1972, 88 kg m^{-1}. So although complaints may have been increasing in recent years in the Halton region, it is unlikely that *Cladophora* growth was comparable to that of two and three decades ago.
Changes in tissue P and P supply along the Halton shoreline

The high sensitivity of the model to tissue P reinforces the concern about trying to manage SRP in the near shore because tissue P is a result of uptake of SRP from ambient waters and dilution of tissue P due to photosynthetic growth of the filament. At shallowest depths where light availability is high, tissue P is kept low by relatively rapid growth; but, as light becomes less available on the bottom with increased depth, growth dilution is slowed and so exposure to the same ambient SRP at both shallow and deeper site will result in different tissue P concentrations. Consequently, it is difficult to interpret tissue P as an indicator of SRP unless the data are normalized for light exposure except at very shallow depths where light is likely saturating photosynthesis. For example in Fig. 31, there is no change in tissue P (QP) over the >30 year time period at 0-2 m (high light) even though substantial declines in SRP are well established through this time period (Fig. 28 and 29). However, dreissenid mussels do represent a new source of SRP in the near shore regions of the Great Lakes and may have increased SRP supply in the near shore while also contributing to the increasing clarity of the nearshore. Ozersky et al. 2009) studied the same shoreline where Malkin et al. (2008) monitored and modelled Cladophora biomass. Ozersky et al. demonstrated that dreissenid excretion of SRP along the Halton shoreline was the largest source of SRP to the shoreline (Table 3) and also substantially exceeded the demand for SRP to sustain observed Cladophora growth on the same shoreline. They concluded:

“Total SRP excretion by dreissenids in the mixed layer significantly exceeds the total P demand by C. glomerata, so excess phosphorus excreted by dreissenids at greater depths could still be available to C. glomerata at shallower depths. We estimate that C. glomerata in the modeled area … takes up 25 kg of P/day during peak P demand, while dreissenid mussels in the same area excrete 89 kg of recycled bioavailable SRP/day.”

Reducing Cladophora growth along Halton shorelines would apparently require reducing the food supply sustaining dreissenids and their excretion. Addressing local point sources in the Halton study area such as the local WPCP and tributary runoff would be expected to have little effect on Cladophora growth by themselves.
Association of Dreissenids and *Cladophora* along Ontario shorelines

The abundance and species composition of dreissenid mussels along the Ontario shoreline was investigated by Wilson et al. (2006) using diver surveys at 27 locations along the coast sampled at 5 m and 20 m. Distribution and abundance was strongly affected by substrata especially at the shallower depth (Fig. 33, 34 and 35). Dreissenid cover was nearly continuous at 20 m even extending onto sandy and silty substrata at this deeper depth (Fig. 35); cover was more discontinuous at 5 m and was largely restricted to rock and cobble at those depths. The divers also estimated % cover by *Cladophora* (Fig. 36), and there was a highly significant correlation (p<0.001) between % cover of dreissenids and % cover of *Cladophora*. Given the common preference of these taxa for hard substrata in shallow water an association would be expected, but the strong correlation between covers suggest a causative relationship. It is easy to imagine that mussels might benefit *Cladophora* by increasing availability of light and supplying SRP, but it is more difficult to imagine a positive causal relation that would invoke *Cladophora* benefiting dreissenids, especially as isotopic studies (Campbell et al. 2009) indicate that dreissenids draw their food supply from phytoplankton and not from *Cladophora* or its detritus. The lack of a positive correlation between % covers would not be expected at 20 m because low light would preclude significant *Cladophora* growth. Wilson et al. do comment that:

“We found substantial growths of *Cladophora* at almost all 5-m sites. At hard substrate sites, *Cladophora* grew attached to both the underlying substrate and mussel shells, at times so thickly that the *Cladophora* had to be pushed aside to observe mussels underneath. The facilitating effect of hard substrate provided by the mussels was most apparent at shallow, high-energy sandy sites, where *Cladophora* grew on the mussels but not on unconsolidated sand.”

They also noted that the mean per cent cover of *Cladophora* (57%) on their survey indicated a substantial increase over a similar survey (48%) of the northern Lake Ontario coast in 1991. It is also worth noting for this review that there was no positive correlation between % algal cover (for 5 m sites) and other environmental variables such as TP or electrical conductivity in the Wilson et al. study. Only % cover of mussels was significantly and highly correlated with *Cladophora* % cover. In summary, Wilson et al. (2006) were profoundly impressed by the impact of dreissenids on the Ontario shoreline and near shore areas:
“The lasting impression of this work should be of the enormity of change that has occurred in the Lake Ontario nearshore zone from the invasion first of zebra mussels, to the near complete displacement of zebra mussels by quagga mussels. Although dreissenid densities have subsided somewhat from those measured in the early to mid-1990s, the sheer magnitude of the effect on bottom habitat is staggering, both on terms of the mussel biomass present (813,000 metric tonnes) and the amount of substrate modified by the presence of mussels (from 60 to 65% of the substrate in the Canadian nearshore alone). Currently, dreissenid mussels are the defining feature of the Lake Ontario benthos.”

A subsequent study of Lake Ontario benthos (Pennuto et al. 2012) compared the distribution and abundance of dreissenids at a smaller number of selected sites (4) along the Canadian shore with U.S. sites (3) (Fig. 37). Although four depths were sampled during early and late summer along duplicate (Canadian shorelines) or triplicate (US shorelines) transects at each site, there were substantial differences in substrata sampled (Table 8) due to unexpected substrata variability at the US sites:

“Transects in Canadian waters were focused on sites with hard substrates following maps of Rukavina (1976). U.S. transects were sited based on an expectation of hard substrates from aerial imaging and professional knowledge. However, aerial images did not accurately reflect substrates at depth and thus a range of substrate types was encountered and sampled.”

As a result, only one of the US sites (Oak Orchard) had hard substrata at shallow depths comparable to the Canadian sites. Not surprisingly then, only Oak Orchard had dreissenid abundances comparable to the Canadian sites (Fig. 38); and, even then, the Oak Orchard site had substantially lower abundances of mussels than all the Canadian sites (note scale difference between US and Canadian sites in Fig. 37). If the US data are accepted as representative of substrata and dreissenid abundance along US shorelines, then the impact of mussels would be expected to be much less on the US coast line compared to Canadian coast line of Lake Ontario. It may also then be expected that some of the relationships between dreissenids and water quality parameters and other organisms, e.g. *Cladophora*, would also be weaker along the US shorelines.

**Urban influence on *Cladophora***?

The study of Pennuto et al. (2012) did not address *Cladophora* abundance in their samples, but a companion study by Higgins et al. (2012) specifically reported on *Cladophora* biomass, tissue P and modeled *Cladophora* growth from the same samples. Higgins et al.
explain the selection of the seven sites as being intended to provide a range of urban influences from high to low with Toronto having the most influence, Rochester NY next and extending to Mexico Bay NY as the lowest (Fig. 37). Higgins et al. do not mention the range of substrata variability among these seven sites nor the broad range in dreissenid abundances as reported in Pennuto et al. (2012). The authors do report a suite of water quality parameters measured for the sites along the impact gradient (including SRP, TP and Secchi disk, with more variables reported in Pennuto et al.); however, only conductivity provided a significant correlation with Cladophora biomass (Fig. 39). In their results and discussion, Higgins et al. abandoned their original focus on population density as a metric of urban influence and adopt conductivity as an indicator of urban influence for their study (based on measurements internal to the study). Although data on dreissenid abundance and substrata were available in Pennuto et al., Higgins et al. do not provide this information nor do they discuss how variability in these aspects might influence their results.

Conductivity may be used in a limited site specific manner to indicate urban influence but it is not useful as parameter to drive Cladophora growth, and so the Higgins et al. (2012) study has limited application at best to interpreting Cladophora abundance in Lake Ontario. Electrical conductivity (electrical conductance responds to ionic strength of a solution) responds to dissolved ions; and, to the extent that urban locations may add solutes such as chloride from road salt or nitrate from waste treatment plants in sufficient quantity, conductivity may be responsive depending on the natural background conductivity of watersheds. Weathering and dissolution of geological substrata and other land use practices such as agriculture can also impact conductivity of receiving waters so that conductivity alone cannot be a specific indicator of urban sources, especially when the studied area crosses boundaries of geologic substrata and other land uses. Additionally, conductivity is affected by stream and river discharge as high flow events typically dilute dissolved loads while low flows lead to higher concentrations of dissolved constituents, and so conductivity can have substantial temporal variability further limiting the interpretation of infrequent samples. So it is not surprising that a causal interpretation of how conductivity might influence Cladophora growth is not offered in Higgins et al. (2012), they do admit that conductivity was not significantly related to TP, SRP or Secchi disk transparency which are factors that can be expected to directly impact Cladophora growth.
Higgins et al. (2012) propose that conductivity is an indicator of urban influence and that urban influence as indicated by conductivity promotes *Cladophora* growth. Their study has some design similarities to that of Depew et al. (2011) who examined land use effects on *Cladophora* biomass and also the Wilson study who surveyed the Ontario coastline for dreissenid abundance and also report *Cladophora* cover. Neither of these broad surveys found a significant correlation between conductivity and *Cladophora* biomass or cover. Both of these studies did agree that Presqu’Ile Provincial Park in eastern Lake Ontario did have extensive *Cladophora* growth which annually reaches nuisance proportions and has also increased apparently since the dreissenid invasion (DeJong 2000). In fact, Presqu’Ile Park rocky shorelines had the highest *Cladophora* measured in Depew et al.’s survey. Yet the conductivity at Presqu’Ile ranged between 275 µS cm\(^{-1}\) in June and 302 µS cm\(^{-1}\), both values lower than the lowest conductivity values reported by Higgins et al. (2012; Fig. 39). The Presqu’Ile conductivity values were essentially the same (or lower) than offshore Lake Ontario conductivity values at Dobbs Bank (where *Cladophora* also grew to nuisance biomass values > 50 g DW m\(^{-2}\)). Higgins et al. (2012) also reported a nearly significant correlation between tissue P (QP in Fig. 39), and conductivity and suggested that this might be an indication that urban areas at shallow depths (<3 m) had *Cladophora* under urban influences. Although the US sites did have higher turbidity and therefore lower light penetration, Higgins et al do not discuss the impact that higher turbidity may have had on tissue P concentrations. As discussed above, it is difficult to compare QP values without some knowledge of ambient light on the bottom, recent exposure to elevated SRP concentrations or state of growth dilution (see discussion of QP at Halton above). There is no denying that urban areas are sources of P that can potentially contribute to *Cladophora* growth; however, it is also clear that nuisance biomasses of this alga can be produced in Lake Ontario and along its shorelines well removed from urban influences. Dreissenid abundance is more directly linked in a causative manner to *Cladophora* abundance along these shorelines (Depew 2011; Wilson et al. 2006). Managing *Cladophora* in near shore Lake Ontario at least requires addressing the dreissenid influence as well as the urban influence, and both of these will likely have to be addressed within the hydrodynamic conditions of the site of interest (Dayton et al. 2014; Tomlinson et al. 2010, Auer et al. 1982; see quotes above).
Studies in the vicinity of Ajax-Pickering related to Cladophora growth

The OPG nuclear generating station at Pickering requires the intake of high volumes of cooling water to control the nuclear reactors. The intake water passes through screens to prevent debris from entering, clogging and potentially damaging the cooling systems. The screens accumulate debris transported to the intake by Lake Ontario currents. In the early 2000s OPG became concerned about the increasing amount of debris being removed from intake screens (Fig. 40). As most of this debris was identified as sloughed Cladophora filaments, OPG initiated a number of studies to understand and address the growing Cladophora problem in their plant management. One of those studies was a two-year study by the University of Waterloo (UW) of the coastal water quality and its relation to Cladophora growth in the vicinity of the Pickering plant water intake and surrounding coastal area (Fig. 41). Of particular concern to OPG was the possibility that the nearby Duffin Creek Water Pollution Control Plant (diffuser in Fig. 41) as well as Duffins Creek itself may have been contributing to the increased sloughing that was clogging their intake screens. The UW study monitored a dense network of water quality stations and also installed thermistor chains and current meters that would allow the verification of a hydrodynamic and water quality model for the Pickering intake area. The study also conducted Cladophora surveys using hydroacoustic methods ground-truthed with diver sampling. A novel feature of the study was linking the Cladophora Growth Model (CGM) with the hydrodynamic-water quality model to predict Cladophora growth, sloughing and transport for the bounded study area. Although the study was designed to answer questions concerning OPG intake of Cladophora, the study area did extend east of the intake along the Ajax shoreline and so the observational data and conclusions of the study have relevance to the issue of Cladophora growth along the Ajax-Pickering shorelines.

Water Quality and Cladophora along Ajax-Pickering Shorelines

The UW study sampled along down slope transects from 2 m to 20 m depth at sites in Fig. 41. Samples were taken biweekly from the mid-depth of the sampling site from late April to October. In 2008, a transect terminated over the diffuser site of the DCWPCP (Fig. 42). Observations at this station demonstrated that the elevated SRP could be observed on most
sampling days at 5 m depth over the diffuser, but the results also showed that on several sampling days concentrations over the diffuser were comparable to the most remote sampling site (eastern most transect, 20 m station) indicating rapid dilution under the hydrodynamic conditions of the sampling day. UW did an Analysis of Variance (ANOVA) to test for differences in site depth along the transects in Fig. 41 (with the exclusion of the site at the diffuser) to see if there were patterns of higher concentrations with distance from shoreline (shallower stations being closer to the shore than deeper stations and shallower stations would also be more remote from the diffuser). The ANOVA was done separately in spring and summer for SRP concentrations (Fig. 43). In both years spring concentrations were higher than summer concentrations; but, due to the high variability of the data, these differences were not significant. Also there was no consistent significant effect of the proximity to shoreline on SRP concentrations (no trend with sampling depth; Fig. 43). In 2007 the deepest set of stations did have a significantly higher spring time concentrations of SRP compared to their summer concentrations, but this was not observed in 2008. Nitrate is also enriched over the diffuser site and followed similar seasonal patterns as SRP. Although NO$_3^-$ did yield more significant differences in concentrations between spring and summer concentrations (Fig. 44), it also did not have any significant trends with site depth (distance offshore). The general conclusion of the water sampling program was that there were seasonal changes in nutrient concentrations, but there was little evidence of persistent differences in nutrient concentrations with distance offshore at least to depths greater than 12 m or with distance from the diffuser. The SRP concentrations observed would be sufficient to stimulate *Cladophora* growth in spring depending on transparency in the spring with reduced growth rates expected in summer.

The hydroacoustic surveys of *Cladophora* biomass indicated strong spatial variability not only with depth as expected as light becomes limiting for biomass but also along the shore line (Fig. 45). Suitable substrata largely controlled this longshore variability (Fig. 45a). At the western margin of the surveyed area (west of OPG) sandy substratum predominated and graded into a sandy, pebble/cobble bottom type to the west of Frenchman’s Bay. The dense vegetation in this region, evident in Fig. 45b, was dominated by rooted macrophytes and *Chara* with some *Cladophora* present. Dense *Cladophora* including “nuisance” growth (defined as in excess of 50 g m$^{-2}$) was observed in the vicinity of OPG and to the east between OPG and Duffins Creek. South of Duffins Creek, sandy and pebbly substrata again inhibited biomass growth to lower
values. However, east of Duffins Creek and to the eastern margin of the hydroacoustic survey, rock and cobble benthic substrata predominates out to the depth limit for *Cladophora* growth, and *Cladophora* biomass reached “nuisance” levels along this entire shoreline. The linked hydrodynamic-water quality-*Cladophora* growth model of Leon et al. (2009) captured some of this spatial variability. The simulation model did better at capturing seasonal variability; and, most remarkably, it did simulate much lower *Cladophora* growth in 2008 which was confirmed both by the 2008 hydroacoustic survey (Leon et al. 2009) and the harvest of *Cladophora* debris at the OPG plant (Fig. 40). The lower biomass production in 2008 was apparently due to seasonally higher rainfall and higher discharge of Duffins Creek in 2008 which affected the transparency of near shore waters and reduced light available to the lake bottom to sustain *Cladophora* growth.

As a test to examine the impact of the DCWPCP on local water quality, the UW simulation model was run without any input from the diffuser to determine its possible effect on simulated *Cladophora* growth. For the entire study area (Fig. 41), the elimination of WPCP reduced simulated *Cladophora* by approximately 10% which was in the range of variability of the two years of study. This surprising result was explained by doing a simple mass budget for the volume of the simulated grid. The potential of P loading from different water sources to increase concentrations in the simulated volume, if they collected without loss, was compared. These source contributions to loading had been measured in the study including the contribution of the open lake of TP and SRP across the boundaries of the near shore segment studied. The contributions of TP from the open lake are 10x the concentration that would result from DCWPCP and three times the SRP (Fig. 46). Of course the loaded masses do not accumulate without loss in the segment volume being modified by biological uptake and regeneration, and the illustration primarily demonstrates the active interaction of the lake with the near shore segment volume. Because concentrations in the lake are generally comparable to or lower than the realized segment concentrations, the lake also dilutes the other loading contributions. The ability of the lake contribution to contribute to near shore processes is determined now primarily by the activities of the dreissenid mussels—the near shore shunt. The simulation model developed by UW is an ongoing effort (Bocaniov et al. 2014), but as applied to the OPG project it did not specifically include the impact of dreissenid mussels to remove particulate matter from the overlying lake water nor to excrete SRP back into those waters.
Are dreissenids important to SRP concentrations and uptake at Ajax-Pickeing

Although mussels were not specifically addressed by the UW study and simulation, their potential contribution of SRP from mussels to SRP in the near shore study area can be estimated from the results of Ozersky et al. (2009) at Halton. He reported mean dreissenid densities of 3500 m⁻² at Halton whereas Pennuto reported equally high or higher, over 10,000 m⁻², during the spring period at Ajax. Ozersky measured a mean excretion rate of 0.45 mg SRP m⁻² h⁻¹. If this excretion rate is applied to the near shore area (21.5 km²) modeled by UW for the six month growing season, then a total of approximately 40 tonnes of SRP would be added to the study area which can be compared to 18 tonnes of SRP (approximately 100kg SRP/day) for the same period emitted by the DCWPCP. The dreissenid excretion rate can be compared to the P in the mean biomass (22 g DW m⁻²) observed in early July 2007 in hydroacoustic surveyed area of 10.25 km² (D.Depew, personal communication). If the mean tissue P concentration (0.1% DM at 5 m) reported at Ajax in Higgins et al. (2012) are assumed to apply to the biomass in 2007, then the excreted SRP from dreissenids can be scaled against the P required to produce the maximum biomass produced over the Cladophora growing season May 1- July 31, following the approach of Bootsma (2009). By this calculation, nearly 23 tonnes of P are in Cladophora in the whole survey area in early July. Over the growing season dreissenids might yield 21 metric tonnes of SRP to the hydroacoustic survey area while DCWPCP could contribute a maximum of nine tonnes. Mussels could supply sufficient P to meet Cladophora demand, but the WPCP itself could not even if all its emitted SRP was focused to the surveyed area which seems unlikely given the vagaries of hydrodynamic dispersion and dilution of the emitted plume. In contrast, mussel excretions enter the survey area on the bottom and benthic Cladophora would get first opportunity for uptake, especially if the excreted SRP accumulates within a boundary layer as demonstrated by Dayton et al. (2014).

In terms of total emission to the UW study area and especially to the habitat area occupied by Cladophora along the Pickering-Ajax coast, mussel excretion is likely the dominant source of SRP. DCWPCP emissions of SRP are designed to be quickly diluted and the water quality data in the UW study suggest that that dilution results in a fairly uniform field of SRP concentrations across the UW study area. In contrast, the mussel emissions enter from the bottom of the water column and their dilution will be dependent on hydrodynamic effects on the
concentration boundary layer (Dayton et al. 2014). In 2009, Martin (2010) investigated SRP concentrations along the Pickering –Ajax coastline and specifically investigated whether a concentration boundary layer forms and results in elevated SRP concentrations. Martin’s study area (Fig. 47) overlapped the UW study area with Martin’s primary depth transect being very close to the easternmost transect of the UW study (actual sample locations were MOE 2008 benthic survey stations). One of Martin’s stations was over the DCWPCP outfall which she sampled at a depth of 1.5 m (compared to 5 m by UW in same area in 2008) while most of her samples were taken at 3 m depth. Martin also used a sampler that would allow her to take samples quite near the sediment water interface (with video camera for confirmation of successful deployment) to determine if there was a benthic SRP concentration boundary layer (CBL) at the sampling sites.

Near shore values, including at the DCWPCP, were higher than offshore much of the time although ANOVA analysis showed there was an interaction with date of sampling (Fig. 48). SRP concentrations were generally around 1 µg L\(^{-1}\) in the spring in the near shore but then declined and approached offshore values by mid-summer. These concentrations are similar but slightly lower than the water column UW SRP results from 2007-2008. Martin found evidence for a concentration boundary layer at all depths, but the SRP concentrations near the bottom were always significantly higher than in the overlying water at the deeper stations of 9 and 17 m (Fig. 49) providing evidence that the phenomena reported by Dayton et al. (2014) is also occurring along the Ajax-Pickering coast line, and that near bottom concentrations as high as 3 µg L\(^{-1}\) were encountered when concentrations in the overlying water were <0.5µg L\(^{-1}\). This supports the Dayton et al. conclusion that the close physical association of mussels and *Cladophora* in a concentration boundary layer may give the alga preferred position compared to other algae to benefit from mussel excretions. Martin also concludes that:

“These results suggest that PO\(_{4}\)\(^{3-}\) excreted by dreissenids could be more important in time and space than external inputs in supporting nuisance *Cladophora* growth in the current nearshore environment.” (from thesis abstract)

**Is Cladophora biomass unusually high at Ajax-Pickering?**

Higgins et al. (2012) suggest that urban influences are needed to generate “nuisance” *Cladophora* biomass which they and others define as biomasses in excess of 50 g DW m\(^{-2}\). They
suggest that locations away from urban influences would not generate such biomass. On their survey, they reported finding a mean of 80 g DW m$^{-2}$ at 3 m and 30 g DW m$^{-2}$ at 6 m at their Ajax transects indicating “nuisance” growths did occur there at the shallowest depths. Depew et al. (2011; discussed above, see Fig.23) surveyed shorelines selected to provide a range of land uses from protected park areas and agricultural land use to dense urban populations with local point source including urban steams and WPCPs. They concluded that dreissenid abundance, and not land use, explained the variability in their data and also explained why offshore shoals with no land influence in both Lake Erie and Lake Ontario can develop nuisance levels of biomass. Depew has also surveyed Pickering Ajax shorelines (Leon et al. 2009) with the same methods that allow integrating spatial variability into their estimates by surveying significant areas along shorelines. At Ajax the surveyed area was 10.4 km$^2$, and the results can be compared (Table 9) to the other shorelines surveyed by Depew et al. (2011) out to 12 m depth. The highest *Cladophora* biomasses for these surveyed sites occurred at locations remote from urban influences. Ajax had mean % cover and mean algal biomass in the surveyed area comparable to Oakville and Port Credit shorelines but lower than Presqu’Ile Provincial Park shorelines, and the mean of several surveyed sites along Ontario shorelines in the eastern basin of Lake Erie without urban influence. The mean biomasses of all these sites are below nuisance levels because the surveys extended out to 12 m depth and therefore encompassed areas where biomass was light limited. If areas of nuisance biomass are compared as % of surveyed area to 12 m, the three urban shorelines again have the lowest abundance of nuisance *Cladophora*, and Presqu’Ile again stands out with the highest proportion of nuisance alga in the surveyed area.

The intent of this comparison is not to say that there is not a problem at Pickering-Ajax. Anyone who has walked on a beach fouled by rotting *Cladophora* will agree that any *Cladophora* on the beach is too much. However Table 9 does establish that Ajax-Pickering is not exceptional for *Cladophora* production along the Ontario coastlines of Lake Ontario or eastern Lake Erie. The common feature of these shorelines is not land use; it is high dreissenid abundances. To reduce the problem on these shorelines may take lake-wide action to reduce the lake phytoplankton populations which sustain dreissenids just as was done back in 1972 with the GLWQA. Local action alone may not be able to significantly reduce a “local” *Cladophora* problem.
Summary and Conclusions:

1) Implementation of the GLWQA in 1972 led to reduced phosphorus tributary loadings and concentrations across all the great lakes with the lakes (with the exception of the western basin of Lake Erie) meeting or exceeding targets set by that agreement by the mid-1980s. Transparencies also in increased in the lakes. Although improved transparencies extended the habitat capable of supporting *Cladophora* to greater depths, the reduction in P resulted in reduced growth rates when light was not limiting. Consequently biomass of *Cladophora* along Great Lakes shorelines was reduced, and public concern about shoreline and water intake fouling subsided.

2) Following the establishment of dreissenid mussel populations at high densities in the lower lakes (Lakes Michigan, Erie and Ontario) by the mid- to late 1990s, public complaints were again raised as extensive lengths of Great Lakes shorelines with hard substrata were again supporting high densities of *Cladophora* including much of the Canadian shoreline of Lake Ontario. This occurred without any observed increase in SRP concentrations or loadings, as offshore and near shore monitoring continued to meet targets set by the GLWQA.

3) Dreissenids affected both the transparency of the lower lakes and also contributed to a further reduction of open water P concentrations, and monitoring data confirm that these offshore trends were also realized at nearshore open water monitoring stations. However, the continuation of these offshore and near shore trends did not result in reduced *Cladophora* growth along rocky shorelines as had been the case following control of point sources of P under the GLWQA.

4) Dreissenid mussels have altered P cycling in the near shore of the lower Great Lakes. The high rates of benthic filtering initiated by the high densities of mussels have increased near shore transparency, but their activities also result in substantial excretion of SRP in the near shore. Mussel excretion is now the dominant source of SRP along many shorelines with hard and cohesive substrata.

5) Mussel excretion recycles SRP from particulate P sources available in lake water. Although the P excreted in the near shore is not new to the lake, it is a new source of SRP which was not available in near shore environments prior to the establishment of high densities of dreissenid mussels.
6) At shorelines where dreissenid mussel excretion has been estimated and where Cladophora biomass and growth have been measured (including Pickering-Ajax), it has been shown that mussel excretion may meet or exceed the P required to supply Cladophora growth. In addition, mussel excretions also exceeded point source SRP loading from WPCPs to the studied areas. Along the Pickering-Ajax shorelines, the WPCP loading in 2007, a high Cladophora growth year, could not be met by the supply of SRP from the DCWPCP alone during the growing season for Cladophora. If mussels could be removed from the area, the Cladophora biomass production would be reduced because of a significant reduction in SRP loading to Cladophora habitat. However, simply reducing P emissions from DCWPCP may not generate a proportional response because mussel excretion may dominate local SRP loadings available to Cladophora.

7) Cladophora has an advantage over phytoplankton in accessing SRP from mussel excretion because both Cladophora and mussels can attach firmly to hard substrata and maintain a close physical association. SRP from mussel excretion becomes available at the bottom of the water column where hydrodynamic conditions can allow higher SRP concentrations to build up in a concentration boundary layer and drive luxury uptake of SRP by Cladophora. The entrapment of SRP and luxury uptake in the boundary layer reduces the loading of SRP from mussel excretion to the overlying water column and allows Cladophora to maintain high growth rates while observed SRP concentration in the overlying water are very low. In contrast SRP from WPCPs is introduced to the water column through diffusers designed to allow rapid dilution of the emitted SRP away from this sizable point source. As a result the delivery of DCWPCP SRP to Cladophora habitat may not be as accessible as dreissenid excretion. Consequently, although DCWPCP SRP emission was approximately half of the estimated dreissenid SRP loading in the study area, its reduction may not have as significant an impact as lowering dreissenid excretion rates.

8) Dreissenid filtration, growth and SRP excretion is driven by the uptake of phytoplankton from the water column; and, in well flushed coastal areas, the open lake can provide a steady supply of phytoplankton into the near shore areas to sustain
dreissenid growth and excretion. The impact of mussel excretion on growth of benthic algae and the formation of concentration boundary layers are therefore both modified by hydrodynamic conditions (waves, currents, turbulence); and growth of Cladophora, as well as sloughing of Cladophora, is therefore also a function of site hydrodynamics.

9) Management of the Cladophora issue in the Great Lakes, including Lake Ontario, will likely require lake-wide action to reduce the production of phytoplankton that sustains dreissenid populations. Local actions alone may have limited or even insignificant results because they do not address the contribution of the open lake to maintaining dreissenid growth and excretion which may dominate SRP supply to Cladophora in the local area of concern.

10) Cladophora grows to nuisance biomasses all along the Ontario shoreline including eastern Lake Erie and Lake Ontario including sites unaffected by urban influences (Table 9). Although it can affect the aesthetics and economic activities on all these shorelines, it certainly has greatest impact on urban shorelines in terms of numbers of affected people. In that sense, it is an urban problem; but, the cause is not just urban environmental impact influence. Humans now share the ability to change the water quality of the coastal Great Lakes with the invasive dreissenid mussels, and any attempt to manage near shore water quality will have to consider their likely impact on the issue of concern.

Recommendations:

1) Cladophora monitoring sites should be established in the Ajax Pickering area to better understand inter-annual variability of Cladophora. Lack of such monitoring has obscured ready assessment of the impact of the dreissenid invasion on Cladophora growth and left modeling as the only way to evaluate this impact. Key monitoring sites could be selected based on the hydroacoustic surveys of the UW study. Video-surveying with calibrated vertical scales using lowered cameras and scales may be adequate and cost effective as areal cover and algal height determine
biomass. Certainly monitoring should be initiated prior to any management directive meant to reduce Cladophora growth in the area.

2) Further research is needed on the interplay of hydrodynamics, mussel excretion and Cladophora. This is an inconvenient truth because of the complexity it adds to the assessment of local Cladophora growth, but it is likely critical to the issue of Cladophora production in any area in which suitable substrata for mussels and Cladophora occurs. There is clear evidence in the scientific literature cited in this report to establish that dreissenid abundance and excretion can fuel Cladophora growth. Conversely, it is likely possible to have areas of high mussel density without excessive Cladophora growth because of different shoreline configurations, different current velocities and wave heights, and different tributary suspended loads and their distribution. Without better characterization of these physical and biological inter-relationships, well intended management actions may be frustrated by variable responses caused by differing hydrodynamic conditions along problematic shorelines.

3) Recent changes at the DCWCP have resulted in much lower emissions of P relative to the UW study period of 2007-2008. In 2013, plant emission of TP averaged 50% of the emissions in 2007. If plant emissions were contributing to nuisance growth of Cladophora in 2007-2008, Cladophora biomass should be lower now than during that period. A hydroacoustic estimate of the same area surveyed in 2007 could be repeated to see if the lower plant emissions have had any effect on observed biomass. Such surveys should be done over at least two years to account for inter-annual variability. If there has not been a proportional decrease in Cladophora, then it would indicate that mussels may be more important to Cladophora growth than DCWPCP. If a reduction in Cladophora has resulted from reduced plant emissions, then future increased emissions could be scaled against the observed response. The costs and possible benefits of management options to reduce projected future concentrations can then be considered.
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**Morphology**

- Filamentous Green Macroalgae
- Variable degrees of branching
- Growth is intercalary and acropetal
- Lacks plasmodesmata
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Table 5. Statistical results of hierarchical variance partitioning of survey data of Depew et al. 2011 (their Table 5). Only dreissenid density (number per square meter) had a significant relationship with Cladophora biomass at surveyed shorelines.

Table 6. (Table 1 of Campbell et al. 2009). Stable isotope signatures of food resource items in the nearshore of the eastern basin of Lake Erie. The closer the values the more likely the items are trophically dependent. Only seston (suspended particulate matter) with an offshore signature (see Table 5) can explain the isotopic signature of dreissenid tissue.

Table 7 (Table 2 of Malkin et al. 2008) Sensitivity of integrated cumulative biomass per unit of shoreline length (kg DW m$^{-1}$) to historic changes in light transparency ($k_{D PAR}$) and tissue P ($Q_P$) at Oakville. Mean of 2004-2005 observations are the baseline against which past changes are compared.

Table 8. (Table 2 of Pennuto et al. 2012) Substratum characteristics of studied sites (Fig. 37) indicating differences in substrata between US and Canadian sites. Except for one station, US sites were soft sediments less conducive to dreissenid occupancy at shallow depths.

Table 9. Table of mean cover and biomass of Cladophora estimated by the hydroacoustic survey of 2007 in the UW Pickering study (Dave Depew, personal communication) compared with other surveyed locations in Lake Ontario and Lake Erie (Depew et al. 2011). Also given is per cent of surveyed area covered by nuisance biomass defined as biomass in excess of 50 g DW m$^{-2}$. 
Table 1. Loading and concentration targets for phosphorus in the Great Lakes (Table 1 of Chapra and Dolan, 2012).

<table>
<thead>
<tr>
<th>Basin</th>
<th>Target TP Load (MTA)</th>
<th>Target TP Concentration (μgP L⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lake Superior</td>
<td>3,400</td>
<td>5</td>
</tr>
<tr>
<td>Lake Michigan</td>
<td>5,600</td>
<td>7</td>
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<tr>
<td>Lake Huron</td>
<td>4,360</td>
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</tr>
<tr>
<td>Main Lake Huron</td>
<td>2,800</td>
<td>5</td>
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<tr>
<td>Georgian Bay</td>
<td>600</td>
<td></td>
</tr>
<tr>
<td>North Channel</td>
<td>520</td>
<td></td>
</tr>
<tr>
<td>Saginaw Bay</td>
<td>440</td>
<td>15</td>
</tr>
<tr>
<td>Lake Erie</td>
<td>11,000</td>
<td>15</td>
</tr>
<tr>
<td>Western Erie</td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Central Erie</td>
<td></td>
<td>10</td>
</tr>
<tr>
<td>Eastern Erie</td>
<td></td>
<td>10</td>
</tr>
<tr>
<td>Lake Ontario</td>
<td>7,000</td>
<td>10</td>
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</table>
Table 2. Comparison of three *Cladophora* growth models showing differences between the models (Table 3 of Tomlinson et al. 2010).

<table>
<thead>
<tr>
<th>Function</th>
<th>Canale and Auer (1982b)</th>
<th>CCM</th>
<th>GLCM</th>
</tr>
</thead>
<tbody>
<tr>
<td>Growth</td>
<td>The maximum specific growth rate was measured using <em>Cladophora</em> harvested from Lake Huron and modified for the impacts of light intensity based on cross gradient laboratory measurements using cultures isolated from Lake Huron; phosphorus availability using the Droop formulation and coefficients measured using <em>Cladophora</em> harvested from Lake Huron and for carrying capacity using a maximum biomass value based on observations in the Lake Huron nearshore and model calibration.</td>
<td>(a) The maximum specific growth rate was decreased from 0.77 day$^{-1}$ to 0.00 day$^{-1}$ based on experiments with Lake Erie <em>Cladophora</em></td>
<td>(a) Maximum specific growth rate coefficient was increased to 1.53 day$^{-1}$ based on refitting of the original polynomial function (Graham et al., 1982).</td>
</tr>
<tr>
<td>Respiration</td>
<td>Basal and light-enhanced respiration were both recognized and were quantified through cross-gradient laboratory measurements using cultures isolated from Lake Huron. The two forms of respiration were applied over the day based on photon period.</td>
<td>The rate of daytime respiration (light-enhanced respiration in the CCM) varies with the internal phosphorus concentration.</td>
<td>(a) The maximum specific light-enhanced respiration rate coefficient was decreased to 0.285 day$^{-1}$ based on refitting of the original polynomial function (Graham et al., 1982).</td>
</tr>
<tr>
<td>Sloughing</td>
<td>A first-order sloughing loss rate coefficient was applied, based on field measurements of sloughing, wind speed, and <em>Cladophora</em> standing crop.</td>
<td>Sloughing is a function of the metabolic balance of cells at the base of the <em>Cladophora</em> bed and includes density dependent effects of self-shading by the overlying canopy.</td>
<td>Sloughing is a function of water temperature (physiological effect) and the depth of colonization (wind energy effect).</td>
</tr>
</tbody>
</table>
Table 3. Comparative loading rates to a studied segment of shoreline at Oakville, Ontario illustrating how dreissenid mussels may be the largest single source of SRP to the segment exceeding even the WPCP (Table 3 of Ozersky et al. 2009). Dreissenid “annual” loading is only for six month *Cladophora* growing season.

<table>
<thead>
<tr>
<th>Source</th>
<th>Annual TP load (kg)</th>
<th>Annual SRP load</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sixteen-Mile Creek (based on 2004-2005 loading)</td>
<td>5100-7200</td>
<td>1200-1700</td>
</tr>
<tr>
<td>Oakville SE WWTP (based on 2007 loading)</td>
<td>2672</td>
<td>5687</td>
</tr>
<tr>
<td>Jordan’s Creek, Wedgewood Creek, Morrison Creek (based on 2004-2005 loading)</td>
<td>-200</td>
<td>-40</td>
</tr>
<tr>
<td>Storm sewers (based on 2004-2005 loading)</td>
<td>-200</td>
<td>-40</td>
</tr>
<tr>
<td>Dreissenid mussels (based on 2006 measurements)</td>
<td>-</td>
<td>10170</td>
</tr>
</tbody>
</table>

Table 4. Comparison of loading rates to a studied segment of shoreline at Milwaukee, Wisconsin for three month spring *Cladophora* growing season (May June July) illustrating how dreissenid mussels may the largest single source of SRP to the segment (Table 2 of Bootsma 2009). Note that dreissenid mussel flux is SRP while Milwaukee River is TP (total phosphorus) not just SRP.

<table>
<thead>
<tr>
<th>P Source / Sink</th>
<th>P Flux (mg m⁻² d⁻¹)</th>
<th>Flux for May 1 – July 31 (mg m⁻²)</th>
<th>Total P for May 1 – July 31 (metric Tons)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Cladophora</em> uptake</td>
<td>119</td>
<td>22.6</td>
<td></td>
</tr>
<tr>
<td>Input from Milwaukee Harbor</td>
<td>1.3</td>
<td>120</td>
<td>22.8</td>
</tr>
<tr>
<td>Mussel Excretion</td>
<td>5.5</td>
<td>506</td>
<td>96.1</td>
</tr>
</tbody>
</table>
Table 5. Statistical results of hierarchical variance partitioning of survey data of Depew et al. 2011 (their Table 5). Only dreissenid density (number per square meter) had a significant relationship (high Z score) with *Cladophora* biomass at surveyed shorelines.

<table>
<thead>
<tr>
<th>Variable</th>
<th>%I</th>
<th>I</th>
<th>J</th>
<th>Total</th>
<th>Z</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mussel #</td>
<td>31.3</td>
<td>0.31</td>
<td>0.63</td>
<td>0.95</td>
<td>2.36*</td>
</tr>
<tr>
<td>% Urban</td>
<td>2.5</td>
<td>0.03</td>
<td>0.02</td>
<td>0.05</td>
<td>-0.99</td>
</tr>
<tr>
<td>% Forest</td>
<td>14.3</td>
<td>0.14</td>
<td>0.41</td>
<td>0.56</td>
<td>0.40</td>
</tr>
<tr>
<td>% Crop</td>
<td>2.9</td>
<td>0.03</td>
<td>-0.03</td>
<td>0.00</td>
<td>-1.01</td>
</tr>
<tr>
<td>% Pasture</td>
<td>2.9</td>
<td>0.03</td>
<td>-0.01</td>
<td>0.02</td>
<td>-0.99</td>
</tr>
<tr>
<td>SpWQ1</td>
<td>18.8</td>
<td>0.19</td>
<td>0.39</td>
<td>0.58</td>
<td>0.91</td>
</tr>
<tr>
<td>SpWQ2</td>
<td>9.6</td>
<td>0.10</td>
<td>0.18</td>
<td>0.27</td>
<td>-0.18</td>
</tr>
<tr>
<td>SuWQ1</td>
<td>16.1</td>
<td>0.16</td>
<td>0.45</td>
<td>0.61</td>
<td>0.58</td>
</tr>
<tr>
<td>SuWQ2</td>
<td>1.5</td>
<td>0.02</td>
<td>-0.01</td>
<td>0.01</td>
<td>-1.01</td>
</tr>
</tbody>
</table>

*Significant based on randomization and upper 95% confidence limit (Z ≥ 1.65).
Table 6. (Table 1 of Campbell et al. 2009). Stable isotope signatures of food resource items in the nearshore of the eastern basin of Lake Erie. The closer the values the more likely the items are trophically dependent. Only seston (suspended particulate matter) with an offshore signature (see Table 5) can explain the isotopic signature of dreissenid tissue.

<table>
<thead>
<tr>
<th>Benthos</th>
<th>Code</th>
<th>n</th>
<th>$\delta^{13}C$ (%)</th>
<th>$\delta^{15}N$ (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Invertebrates</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Chironomidae</td>
<td>CH</td>
<td>17</td>
<td>$-19.3$ (1.1)</td>
<td>10.9 (1.2)</td>
</tr>
<tr>
<td>Dreissena bugensis</td>
<td>DB</td>
<td>18</td>
<td>$-22.7$ (1.6)</td>
<td>9.8 (0.5)</td>
</tr>
<tr>
<td>Echinogammarus ischnus</td>
<td>El</td>
<td>4</td>
<td>$-19.6$ (1.6)</td>
<td>10.2 (0.5)</td>
</tr>
<tr>
<td>Gammarus fasciatus</td>
<td>GA</td>
<td>5</td>
<td>$-16.8$ (1.0)</td>
<td>9.1 (1.0)</td>
</tr>
<tr>
<td>Other</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Benthic algae (mixed scrapings)</td>
<td>BA</td>
<td>10</td>
<td>$-17.1$ (2.2)</td>
<td>6.9 (1.5)</td>
</tr>
<tr>
<td>Cladophora spp.</td>
<td>CL</td>
<td>1</td>
<td>$-18.1$</td>
<td>5.6</td>
</tr>
<tr>
<td>Spirogyra spp.</td>
<td>SP</td>
<td>3</td>
<td>$-13.2$ (2.1)</td>
<td>8.2 (1.2)</td>
</tr>
<tr>
<td>Dreissena feces</td>
<td>DF</td>
<td>11</td>
<td>$-15.6$ (1.5)</td>
<td>7.1 (0.6)</td>
</tr>
<tr>
<td>Gammarus feces</td>
<td>GF</td>
<td>1</td>
<td>$-16.6$</td>
<td>9.7</td>
</tr>
<tr>
<td>Detritus</td>
<td>DE</td>
<td>15</td>
<td>$-9.5$ (1.8)</td>
<td>7.0 (0.5)</td>
</tr>
<tr>
<td>Macrophyte (mixed)</td>
<td>MA</td>
<td>7</td>
<td>$-15.9$ (6.3)</td>
<td>7.8 (4.5)</td>
</tr>
<tr>
<td>Seston</td>
<td>SE</td>
<td>17</td>
<td>$-24.7$ (2.2)</td>
<td>6.0 (1.2)</td>
</tr>
</tbody>
</table>

Standard deviations are listed in brackets. The species codes, as used in Fig. 1, are listed here.
Table 7 (Table 2 of Malkin et al. 2008) Sensitivity of integrated cumulative biomass per unit of shoreline length (kg DW m\(^{-1}\)) to historic changes in light transparency (\(k_{d\text{PAR}}\)) and tissue P (Q\(_P\)) at Oakville. Mean of 2004-2005 observations are the baseline against which past changes are compared.

<table>
<thead>
<tr>
<th></th>
<th>Peak attached biomass</th>
<th>Peak cumulative biomass</th>
<th>Depth of peak attached biomass</th>
<th>Depth of peak cumulative biomass</th>
<th>Max. depth</th>
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</thead>
<tbody>
<tr>
<td>K(_{d\text{PAR}}) effects</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>1982-1983</td>
<td>55±3</td>
<td>66±5</td>
<td>2.5±0.3</td>
<td>2.5±0.1</td>
<td>10.4±0.4</td>
</tr>
<tr>
<td>1972</td>
<td>24±3</td>
<td>32±4</td>
<td>1.6±0.0</td>
<td>1.6±0.1</td>
<td>6.7±1.1</td>
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<tr>
<td>Q(_P) effects</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1982-1983</td>
<td>157±4</td>
<td>274±28</td>
<td>3.5±0.4</td>
<td>2.0±0.2</td>
<td>11.3±0.3</td>
</tr>
<tr>
<td>1972</td>
<td>189±4</td>
<td>356±33</td>
<td>3.5±0.4</td>
<td>1.8±0.3</td>
<td>11.6±0.2</td>
</tr>
<tr>
<td>K(_{d\text{PAR}}) and Q(_P) effects</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1982-1983</td>
<td>77±4</td>
<td>173±16</td>
<td>2.1±0.4</td>
<td>1.4±0.1</td>
<td>8.1±1.4</td>
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<tr>
<td>1972</td>
<td>88±4</td>
<td>236±12</td>
<td>2.1±0.2</td>
<td>1.2±0.1</td>
<td>6.5±0.7</td>
</tr>
</tbody>
</table>
Table 8. (Table 2 of Pennuto et al. 2012) Substratum characteristics of studied sites (Fig. 37) indicating differences in substrata between US and Canadian sites. Except for one station, US sites were soft sediments less conducive to dreissenid occupancy at shallow depths.

<table>
<thead>
<tr>
<th>Location</th>
<th>Depth (m)</th>
<th>Sand</th>
<th>Clay/mud/silt</th>
<th>Gravel/cobble/boilder</th>
<th>Bedrock</th>
</tr>
</thead>
<tbody>
<tr>
<td>Oak Orchard</td>
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<td>XXX</td>
<td>X</td>
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</tr>
</tbody>
</table>

Notes: 1 -- hard packed clay.
Table 9. Table of mean cover and biomass of *Cladophora* estimated by the hydroacoustic survey of 2007 in the UW Pickering study (Dave Depew, personal communication) compared with other surveyed locations in Lake Ontario and Lake Erie (Depew et al. 2011). Also given is per cent of surveyed area covered by nuisance biomass defined as biomass in excess of 50 g DW m\(^{-2}\).

<table>
<thead>
<tr>
<th>Lake Site</th>
<th>Survey area</th>
<th>Mean cover</th>
<th>Mean Biomass</th>
<th>Area of nuisance stands &gt;50 g m(^{-2})</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>km(^2)</td>
<td>%</td>
<td>g m(^{-2})</td>
<td>% of survey area</td>
</tr>
<tr>
<td>Erie - Grand River Area July 2005</td>
<td>10.9</td>
<td>33.0</td>
<td>32.3</td>
<td>20.3</td>
</tr>
<tr>
<td>Ontario - Presquile PP - July 2005</td>
<td>5.6</td>
<td>43.9</td>
<td>58.6</td>
<td>38.1</td>
</tr>
<tr>
<td>Ontario - Oakville - July 2005</td>
<td>6.7</td>
<td>24.4</td>
<td>19.5</td>
<td>10.3</td>
</tr>
<tr>
<td>Ontario - Port Credit - July 2005</td>
<td>10.1</td>
<td>25.2</td>
<td>14.4</td>
<td>6.2</td>
</tr>
<tr>
<td>Ontario - Pickering - July 2007</td>
<td>10.3</td>
<td>25.6</td>
<td>21.1</td>
<td>10.4</td>
</tr>
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</table>