Harmful Algal Blooms (HABS) in the Great Lakes: Current Status and Concerns

State of the Ecosystem
Introduction

Cyanobacterial and algal[1] blooms are a long-standing issue in eutrophic waters with high anthropogenic (and/or natural) nutrient loading. Widespread blooms (planktonic and attached e.g. *Cladophora*) were a recognized beneficial use impairment (BUI) in offshore and inshore areas in the Great Lakes prior to lake-wide remediation in the late 1970s (e.g. Munawar and Munawar 1996; 2000; Higgins et al. 2008). Concerns at that time were based around impaired aesthetics, taste-odour (T&O), foodweb decline, beach/intakes/net fouling and economic impacts. These were addressed largely by targeting total phosphorus (TP) and chlorophyll-*a* (chl*a)* levels, mitigated through point-source nutrient loading. Recently, however, there has been an apparent resurgence in blooms in the Lakes, and an additional new concern with their potential production of toxins or harmful metabolites, compounds which were unidentified in the 1970s. In fact, there is a widespread perception[2] that harmful algal blooms (HABS) are increasing worldwide (e.g. Hallegraeff 1993) which has been linked to the cumulative effects of human development.

Definition of HABS

There are no quantitative definitions of HABs. With current publicity, the terms ‘HABs’, ‘harmful’ and ‘bloom’ are often indiscriminately - and synonymously - used in reference to all types of algal outbreaks. In fact, ‘bloom’ is an ambiguous term, currently defined only by qualitative descriptors (e.g. Smayda 1997). Pearl (1988) further differentiates ‘harmful’ from ‘non-harmful’ blooms by their qualitative impacts on/threats to: i) water quality, biota or physico-chemical characteristics; ii) health risks from toxins or heightened microbial activity; iii) aesthetics or recreation. Current Great Lakes management goals continue to target planktonic (subsurface) chl*a* as a measure of total algal biomass and productivity, which is often an irrelevant measure of these events.

In inland waters, HABs are generally associated with planktonic toxic cyanobacteria, but are, in fact, caused by a diversity of benthic and planktonic algal taxa. These events are often highly sporadic and dynamic in nature, showing episodic patterns which vary seasonally and interannually in severity, and geographical range, making it difficult to design appropriate research, monitoring and management programmes. Impacts can include: risks to drinking water/human and animal/livestock health (via toxins, carcinogens, tetragens, irritants), other drinking water impairment (T&O, aesthetics), intake/fish net/shoreline fouling/ bacterial growth in rotting mats (including potential pathogens e.g. *E. coli*), beach closures (affecting tourism, recreation), and fish / shellfish / processed food tainting (harming commercial/recreational fisheries, other food industries). HABS can impact food web integrity and structure, and result in anoxia. HABs thus include *Cladophora* and other benthic/littoral macroalgal proliferation, and planktonic blooms, all representing current concerns in the Great Lakes and addressed below. Given the historic and current impairment by *Cladophora*, it is specifically addressed in a chapter subsection (I) below; the second subsection (II) will focus on other HABs related issues now (and previously) occurring in these waterbodies.

Harmful Algal Blooms
Introduction

The ability of HABs species to proliferate is dependent on the nature of the environment and its seasonal and spatial variance. The operational definition of the inshore zone thus has important bearing on assessing, monitoring and managing HABs. In the Great Lakes, the inshore has been statically defined as the zone between the edge of the shoreline or wetlands and the deepest lake contour at the late summer thermocline (if established), and includes connecting channels and waters, lower tributaries and unstratified areas around islands and shoals (Edsall & Charlton 1997). Yet in function, these coastal regions are highly dynamic, with long and short-tem spatial-temporal

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[1] ‘algae’ here denotes both eukaryotic algal taxa and cyanobacteria

[2] Toxins are only recently recognized as a threat and with few historical data, this perception is based more on anecdotal evidence and not quantified; reports may be biased by increasing public awareness; most sites are not monitored, many blooms are not identified and visible blooms are not the only sources

variance in boundaries supporting littoral and planktonic communities, and this consideration is often overlooked by current fixed-point sampling programmes.

The static and functional size of inshore zones varies enormously among and within the Lakes (~1-10% in Superior to 60-90% in Erie; Edsall & Charlton 1997) and degree to which each is influenced by physical and climatic factors (runoff, erosion, thermal bar, upwelling/downwelling, alongshore/inshore/offshore currents, circulation patterns, surface/ground water inputs, ice formation, etc.). This translates to a highly mobile spatial and temporal range in littoral community structure and activity and offshore-inshore exchange. Inshore biotic assemblages are shaped by regional differences in the Lakes in bottom substrate, daily/seasonal range water levels/temperature and basin, and shoreline development and impacts (deforestation/ agriculture/industry/ urbanization; wetland drainage/dredging, water level regulation etc.). The lower Lakes and associated waters are considerably more impacted – and also significantly more prone to algal blooms.

HABs in the Great Lakes involve a variety of species and are particularly problematic in coastal areas. Major impairments include: i) noxious and potentially toxic metabolites (odour, toxins); ii) fouling issues (beaches, nets); iii) aesthetics and economic impacts, vi) modified nutrient recycling and sequesterment/translocation (via detached material); vii) heightened bacterial activity in recreational waters and beaches, viii) adverse effects on food web integrity. Importantly, as a coastal phenomenon, their appearance is often unlinked with current monitoring targets – i.e. offshore nutrient and chl*a* levels.

Biochemical impacts of HABs

T&O compounds and toxins are often assumed to be manageable by controlling excess algal growth. In fact there is often a very poor relationship between biomass and the production of these metabolites, because they involve different genetic and biochemical pathways, the same or different taxa, and cell-specific variance in production capacity and output related to genetics and environment (Watson 2003).

It is difficult to predict toxicity, T&O or other impacts. One or more of ~200 known toxins and T&O compounds are produced by many different species, but current resources and knowledge limit our ability to characterize and evaluate impacts. The issue is complex and difficult to sample effectively; outbreaks can be episodic, erratic and involve planktonic or benthic/littoral biota. Incidence/levels of toxins, T&O, visible blooms, cell counts/biomass/chla may or may not be related. Thus *Microcystis* does not produce 'earthy' T&O (geosmin and 2-MIB) and is often odourless, while odour-causing species (e.g. *Anabaena*, *Lyngbya*) may, or may not be toxic. Genetic capacity and cell production can vary for toxins and T&O among species, cell populations and environments; potential producers and morphologically similar species co-occur (e.g. *Microcystis aeruginosa* and *M. wesenbergii; Anabaena flos-aquae and A. lemmermanii*) (e.g. Rinta-Kanto et al. 2005; Jüttner & Watson 2007). Variance among analytical and sampling methods often generates inconsistencies among reported levels (G. Boyer unpublished data).

Toxins

Cyanobacterial toxins have no taste or odour. Because they were identified relatively recently, there are no long term records, hence it is difficult to verify any long term changes in severity and occurrence. They were unknown when delisting criteria were developed, and are still largely not addressed by most Great Lake management programmes: current (and limited) sampling is often reactive, fails to capture these often episodic events, and biased towards research in high-risk areas. Yet concern has been growing since the first report of a toxic outbreak in western Lake Erie (Brittain et al. 2000). Recent lakewide surveys since 2000 (e.g. MERHAB-LGL, EC)[3] found detectable toxin levels in many areas, especially in the Lower Lakes and/or coastal areas with moderate-severe impairment (Boyer 2007; Watson et al. 2008a,b). Toxin levels at most offshore sites are generally very low, but in inshore zones with advanced eutrophication (e.g. harbours, embayments and river mouths including Quinte, Oswego, Sandusky, Maumee, Saginaw, Hamilton Harbour) they can often exceed drinking water guidelines, particularly where present as surface or windblown shoreline scums.

The most commonly reported toxins in the Great Lakes and other waters are microcystins (MCs). Exposure though ingestion or inhalation can cause liver failure and death or increased risk of cancer with long term chronic exposure. Numerous structural variants differ in toxicity[4]; microcystin-LR (MC-LR), is the most widespread and toxic, and the basis for many guidance levels (Codd et al. 2005). MCs are produced by a range of cyanobacteria species; some of which cause outbreaks in the Lakes, notably *Microcystis* spp. (e.g. Boyer 2007). MC, and hepatotoxic nodularin, are stable to degradation, treatment/boiling and may impair food webs. Guidance levels are

[3] NOAA Monitoring and Event Response in the Lower Great Lakes; Environment Canada
[4] >90 MC variants (congeners) now identified

few and vary among agencies[5] especially for recreational areas with high public exposure and risk. In Lakes Ontario and Erie, neurotoxic anatoxin-a and saxitoxins have been detected at high and low levels, respectively (Boyer 2007; Watson et al. 2008a,b). There are no data on the occurrence of lipopolysaccharides (LPS), produced by all cyanobacteria and widely believed to cause gastroenteritis, skin/eye irritations, hay fever, asthma and blistering (although this is debated; e.g. Stewart et al. 2006).

Taste and odour

T&O impairment is widespread in the Great Lakes. Most of the recorded outbreaks and incidental reports of this impairment have not been traced to their biological origin(s). T&O compounds have no known human health effects, but can impart significant consumer alarm and treatment/economic costs (e.g. Engle et al. 1995). T&O compounds, however, can function in foodwebs as powerful chemical signals, acting as grazer deterrents or toxins (e.g. Watson 2003). Numerous algal VOCs[6] are known, which vary in odour, potency, seasonal dynamics and treatment implications. One or more planktonic or benthic species may co-produce different VOCs which are cell-bound until death, continuously released, or triggered only by cell lysis. Benthic and planktonic diatoms and chrysophytes can produce lipid derivatives[7] causing fishy- or cucumber odours in low-moderately productive waters. In remediated, mesotrophic and eutrophic waters such as the Great Lakes, T&O is caused frequently by terpenoids[8] (geosmin, 2-methylisoborneol (MIB)), and to a lesser extent, pigment derivatives (β-cyclocitral)[7] or methyl- and isopropyl sulphides[9]. Hidden or detached benthic, littoral and epiphytic cyanobacteria (e.g. *Lyngbya, Oscillatoria, Gloeotrichia*) are significant geosmin and MIB sources in inshore areas of the lower Lakes and channels (St. Lawrence and Maumee Rivers, Bay of Quinte; Watson et al. 2005 and unpublished data) affecting shorelines and drinking water supplies. The anaerobic breakdown of any excessive bloom material is also a frequent T&O source. Rotting mats of *Cladophora, Lyngbya* and other attached algae are major sources of 'septic/sewage/sulphur' odours along beaches and shorelines in the Great Lakes and connecting channels, driven inshore by currents and wind from local or along-shore sources.

Most jurisdictions have not regulated T&O and there are no quantitative guidance levels in drinking- or recreational waters. T&O is listed under the Canadian Drinking Water Quality Guidelines "aesthetic effects" and a listed BUI (treated municipal supplies). T&O impairment occurs in over a third of AOCs mostly in the Lower Lakes, but likely is more widespread (Watson et al. 2007; Watson et al. 2008a,b). There has been little or no direct monitoring or quantification and it is usually assessed (i.e. 'deduced') by RAPS using 'proxy', often unrelated measures (e.g. chl*a*, nutrient levels; Keene 2002; Watson et al. 2008a,b).

Great Lakes: current status of HABS in individual lakes

As noted above, there are no long term trends in toxins and T&O, while HABs data is limited. Hence, only a qualitative assessment of the current status in each lake can be made here.

Lake Superior; Status: good.

There is very little quantitative current information on HABs in Lake Superior. To our knowledge, severe HABs outbreaks have not been documented recently in this Lake, although cyanobacteria, including *Microcystis*, are detected in samples taken during routine monitoring. Algal biomass remains mostly at low levels, although there may be some local impairment near shoreline development (J. A. Thompson U.S. EPA MED Thunder Bay; J. Kelly, US EPA Duluth, personal communication). A recent survey of drinking water utilities showed few reported T&O issues[10]; intermittent outbreaks have been reported from one drinking water utility (Moore and Watson 2007).

Lake Michigan; Status: mixed.

Lake Michigan has a fairly extensive inshore zone as defined by the 9m or 27m depth contour (10%, 26% area, resp.) which nevertheless only accounts for a small fraction of the total volume (0.4-4%, respectively). Yet the inshore area has a key influence on the lake ecosystem. The lake has the largest groundwater input (79 % hydrological loading) due to nearshore aquifers, and water levels recently show periodic lows. Resuspension during mixing and storm events generate extensive late winter-early spring plumes of resuspended sediments along the East shore which have a significant effect on light regime and nutrients, cycling and transport. These events also

[5] e.g. WHO, Health Canada GLs for total MC of 1-1.5ug/L for *treated* drinking water respectively; recreational water ~10- ±20 ug/L; Watson et al ibid; currently still on the US-EPA Critical Contaminant List;
[6] volatile organic compounds
[7] synthesized during lysis
[8] synthesized over growth and mainly cell-bound
[9] synthesized over growth and continuously released
[10] which may or may not be of algal origin

influence the biological community by introducing resuspended diatom plumes characteristic of more eutrophic waters and modifying the spatial distribution of other phytoplankton and microbiota. Cyanobacteria blooms are reported in some coastal regions in eutrophic embayments such as Green Bay and Muskegon Bay. Shoreline and beach fouling by *Cladophora* stimulated by nutrient loading from inshore sources, represent a potential source of bacteria for beaches and groundwater, by trapping bacterial flora (washed in from runoff and other sources) during their growth which is then deposited along shores by currents and storms.

Lake Huron; status: mixed

Lake Huron is one of the more oligotrophic of the Great Lakes, yet excessive phytoplankton and potentially toxic HABs occur in some inshore areas, notably Saginaw and N. Georgian Bay (Fahnenstiel et al. 2008; Scheiffer & Scheiffer 2002). These two areas differ markedly in drainage basin development, HABs species and associated impairment. Saginaw Bay has a large and extensively developed catchment, and develops toxic summer outbreaks of *Microcystis aeruginosa.* These blooms appear to be genetically distinct with a greater MC production capacity than HABs populations of *M. aeruginosa* in other Lakes e.g. Western Lake Erie (Dyble et al 2008; Fahnenstiel et al. 2008). Highest toxin levels occur in shallow regions with high TP concentrations. Northeast Georgian Bay watershed is far less developed, but has extensive wetlands and a growing cottage industry. The region has generally good water quality, but a few nearshore areas show high TP and chl*a* levels, including Sturgeon Bay (Diep et al. 2006). The upper stratified basin of this Bay experiences hypolimnetic anoxia with sediment nutrient release and severe annual blooms during fall turnover which impair shorelines and water quality (Schieffer 2003). Samples collected during a partnered MOE-EC two-year characterization of these blooms showed a predominance of diatoms, N-fixing *Aphanizomenon/Anabaena*. Toxin levels (MC, Anatoxin-a) were at or below detection over the entire season (Watson & Howell 2007).

Similar to Superior, there are few issues with drinking water T&O outbreaks in L. Huron, with outbreaks reported a single area (Moore & Watson 2007). However, macroalgal impairment is a major concern in some areas. Recently, complaints of fish-net fouling by attached chlorophytes have increased (*Spirogyra cf circumlineata, Stigeoclonium*; Watson and Milne, unpublished). Rotting mats of beached green macroalgae are increasingly impacting aesthetics, recreation and tourism along some shorelines, notably Saginaw and more recently, the S.E., largely caused by *Cladophora* and *Chara*, respectively. recent studies by US and Canadian agencies (MDEQ, MNR, OME, EC) have raised new concerns with the health implications of these events, with the detection of human fecal indicators (*E. coli*, *Enterococcus*) and evidence of differential survival in the beached mats and *in situ* beds of the macroalgae (*Lake Huron Binational Partnership 2008-2010 Action Plan* 2008). Patchy sites also show elevated *E. coli* counts associated with algal debris buried in beach sand. There is a perceived increase in the range and severity of these events which demonstrate different patterns, suggesting several (unresolved) factors contribute. *Cladophora* is more clearly associated with suspected nutrient discharge while *Chara* is more widespread and not clearly linked to local inputs (Howell et al. 2005).

St. Clair River/Lake St Clair/Detroit River; status: fair to good

Recent reports and surveys do not identify algal blooms as a problem, as also indicated by generally low chl*a* levels (~3-5µg/L; Lake St. Clair Canadian Watershed Technical Report; Watson unpublished), although there is some spatial variance. However, a summary report issued in 1999 reported 'floating mats of submersed aquatic plants and algae' along the Western shoreline[11] and several utilities report annual or intermittent T&O in water drawn from the St. Clair and Detroit Rivers (Moore & Watson 2007).

Lake Erie; status: mixed to poor.

Water levels typically fluctuate ~±36 cm/yr; in some years (2002), up to 50 cm. There has been a steep decline in levels from 1997 maxima, to below average during recent years with sometimes significant fluctuations due to climate and storm events. This, together with the corresponding dynamics in the physical/chemical regime, has been accompanied by some disturbing trends in biota and system integrity. Not only does Erie have the most extensive inshore area, but toxic HABs are a particular concern and the focus of several recent studies. These have provided more insight into these events than for other lakes.

HABS biomass and impairment in Erie

[11] Lake St. Clair: Its Current State and Future Prospects conference summary report 1999; http://www.great-lakes.net/lakes/stclairReport/summary_00.pdf

General trends: The operative definition of the inshore area includes 60-90% of the lake (and most of the Western Basin). Collective evidence points to important recent changes in coastal areas and the dynamic nature of the functional inshore zone. Overall, the data indicate an apparent deterioration, and shifts in external/internal and physical/ chemical/ biological regimes - notably in the Western basin. These are not easily assessed using current monitoring methods and measures, which may provide contradictory or ambiguous evidence, particularly where basin-wide averages and/or surface (1m) chl*a* are considered (Ghadouani & Smith 2005). Makarewicz (1993) reported a 70-98% biomass reduction of nuisance and eutrophic 'indicator' species[12] in the 1980s (e.g. diatoms *Stephanodiscus binderanus, S. niagarae, S. tenuis,* and the cyanobacterium *Aphanizomenon flos-aquae*) which generally correlated with P levels. Other studies also suggest a decline in overall chl*a* and total and/or eutrophic species biomass in the Central and Eastern Basins, attributed to nutrient reduction, increased transparency and grazing by invasive Dreissenids. Conroy et al (2005) evaluated trends in biomass and chl*a* data (covering studies in 1970, 1983–88-89, 1989/90–93, 1996–2002) and concluded that average biomass has generally increased in all basins since the late 1980s minima. They also observed no consistent relationship between biomass and Dreissenids or (measured) external TP loading (total or basin-specific) and suggested that internal loading is becoming more important (e.g. Makarewicz et al. 2000; Matisoff & Ciborowski 2005). However, Conroy et al. (2005) also highlight different patterns among basins and seasons underlying basin-wide averages. Spring biomass in the Western basin decreased markedly in 1980s- 1990s, but approached previous maxima in 2000/2002; summer biomass also decreased significantly but increased to ~50% earlier maxima. A similar, slightly less significant resurgence occurred in the Central basin, the Eastern basin showed a more variable interannual pattern, with an all time maxima in late 1990s and recent levels (2000s) still elevated. These generalized patterns overlay significant spatial (horizontal and depth-related) variance among sites particularly along shorelines, and in the West (e.g. Carrick et al. 2005, Ghadouani & Smith 2005).

Cyanobacteria***:*** Pre-remedial (1970s) high cyanobacterial biomass was reported by Munawar and Munawar (1996) in summer-fall, with a predominance of N-fixers (*Aphanizomenon, Anabaena*) and regional maxima indicating localized development or translocation by currents in the west (Maumee-Peele; Sandusky) west-central and east (Erie, Buffalo). 'Bloom proportion' cyanobacterial levels (>1000ug/L) were reported only in the West and far East (Buffalo); diatoms were dominant. Recently, Conroy et al. (2005) reported resurgence in cyanobacterial biomass in all basins in summer since the mid-1980s, notably in 2000s. Again, there was high interannual and spatial variability, but an overall increasing frequency of high cyanobacteria biomass (which may also reflect targeted sampling). Both total and cyanobacterial biomass showed no significant relationship with external TP loading, and a poor relationship with chl*a*. Most of the increase in summer cyanobacteria was attributed to *Microcystis* spp, suggesting a long-term shift from N-fixers in the 1970s to non fixers, reflecting changes in nutrient supply or Dreissenid activity. A 1998 survey by Barberio & Tuchman (2000) also showed a predominance of *Microcystis* and other chroococcales (*Aphanocapsa delicatissima, Chroococcus limneticus*).

Toxins: Lake Erie and associated channels/embayments are among the most severely HABs-impacted areas (e.g. Table 2). July - October outbreaks of planktonic and benthic taxa show significant interannual, seasonal and spatial variation in origin and impacts. Immense surface blooms (>20 km2) have been recorded in the Western basin near the Maumee and Sandusky Rivers - one of the potential sources for HABs in Western and West-Central basins (e.g. Rinta-Kanto et al. 2005). Data from five targeted cruises during 2000-2004 measured a wide range in MC levels from detection limits (in 2002) to >20µg /L (in 2003). Toxicity and bloom distribution varied spatially, and were not restricted to the Western Basin. In 2003, highest MC concentrations were measured from Maumee, Long Point Bay and Sandusky Harbour. Neurotoxins (anatoxin-a, saxitoxin, neosaxitoxin) and cylindrospermopsin occurred at or near detection limits. In 2001 and 2002, localized MC occurrences were also reported from the Central and Eastern Basins (Wendt Beach, Presque Isle, Port Dover), some significant (Murphy et al. 2003; Ghadouani & Smith 2005).

Variance in toxicity among species and strains means that microscopic identification, biomass or cell counts cannot predict toxin levels. Microcystins (MCs) are the most common cyanobacterial toxins measured in Erie. Recent work reported toxic *Microcystis* blooms from Maumee with 5-100% variance in genetic potential for MC production and suggested that these blooms were the likely MC sources in far west and Long Point areas. In contrast, in Sandusky Harbor subdominant *Planktothrix* and/or other unidentified taxa were the likely MC sources where cyanobacteria were dominated by non-producers (*Aphanizomenon, Anabaena*; Rinta-Kanto et al. 2005, Rinta-Kanto & Wilhelm 2006; Boyer 2007). Most impairment occurs at shorelines and beaches and can be manifested as fish/bird kills (see e.g. Murphy et al 2003). To date, however, Lyngbyatoxins (inflammatory/ vesicatory and tumour-

[12] But see section on indicator species below

promoting) have not been detected e.g. in the extensive mats of *Lynbya wollei* now proliferating in the Maumee (below).

Spring & late fall samples are often overlooked yet some species can show significant development during this period. *Cylindrospermopsis raciborskii*, first identified in Sandusky Bay 2005, may develop localized high spring biomass (Conroy et al. 2007). This N-fixing species has a wide temperature tolerance (up to30^{o}C) and high P storage capacity. It is invading north from warm to mid-latitude regions and has a strain-specific potential to produce cylindrospermopsin, mediated by light (Dyble et al 2006). *Cylindrospermopsis* is buoyancy-controlling, like *Microcystis,* but better adapted to turbid conditions and found near rivers and as deep chlorophyll maxima in stratified waters - which may be missed by discreet depth sampling regimes of current surveillance programmes. To date, *Cylindrospermopsis* has not been found as a dominant; Conroy et al (2007) reported it as <2% total biomass in 2005, except early spring. It has been seen each year around Sandusky but not associated with the (low levels of) cylindrospermopsin or deoxycylindrospermopsin detected here or in other areas of the west basin (e.g. Maumee River; Boyer, unpublished data). The highly variable morphology of this and other species (including *Microcystis*, discussed above) may lead to misidentification of these cyanobacterial taxa. Non-heterocystous trichomes of *Cylindrospermopsis* can be easily misidentified as an *Oscillatoria* (*Planktothrix*) and overlooked or misidentified as *Raphidiopsis curvata* which has been identified in recent Maumee samples; strains of this species produce deoxycylindrospermopsin (e.g. Wilhelm and Li, unpublished data; Gugger et al. 2005).

Taste-odour*:* geosmin and 2-MIB are likely the cause of annual musty-muddy odour problems in drinking water in supplies in the Western basin (e.g. Toledo); in addition, significant odour is produced by extensive rotting mats of shoreline attached algae (below). The planktonic cyanobacterial taxa which are currently problematic in Erie (*Microcystis* and the local strain of *Planktothrix*) do not produce these or other T&O compounds which would impair drinking water supplies (e.g. Watson et al. 2008a).

Benthic cyanobacterial impairment is becoming a key issue in some areas. Recent severe impairments of beaches by thick mats of the cyanobacterium *Lyngbya wollei* have been reported in the mouth of the Maumee (West Basin) at sites with high ambient P in the overlying water (Watson et al. 2008b). These have provoked significant media coverage and website postings[13]. However the mats have not been found to produce any of the common toxins and represent no direct threat to human health (Quilliam, M; Wilhelm & Boyer, unpublished data); however they produce significant taste-odour, foul nets and their effects on bacterial levels on beaches and benthic foodwebs are unknown.

Other HABs taxa: In addition to the invasive cyanobacterial taxa noted above (*Cylindrospermopsis, Lyngbya wollei*) which produce direct impairments, numerous other taxa have been recorded in Erie (cf. Mills et al. 1993; Patterson et al. 2005 etc.). These include invasive species e.g. attached red algae (*Bangia atropurpurea, Chroodactylon ramosum),* and diatoms (e.g. *Skeletonema potamos, S. subsalsum, Thalassiosira guillardii, T. lacustris, T. weissflogii).* Western and Central basin spring-summer biomass diatom maxima can include the invasive diatom *Actinocyclus normanii f subsalsa*, indicative of eutrophied, polluted sites, and high conductivity, elevated cations (Mg^{++}, Ca^{++}), fluctuating light levels and turbulent vertical mixing. Recent high spring abundances of the filamentous diatom *Aulacoseira islandica* in the Western basin (e.g. Barbiero & Tuchman 2001; S. Wilhelm unpublished data) have the potential to foul nets although to date there are no known reports of this impairment. Extensive mats of attached green algae, notably *Cladophora* are showing an increase in abundance along shorelines.

These outbreaks are of concern for several important reasons: i) the production of noxious and potentially toxic metabolites by these taxa (odour, toxins); ii) fouling issues (beaches, nets); iii) aesthetics and tourist industries /real estate impacts, vi) modified nutrient recycling and sequesterment/translocation (via detached material); vii) their potential to act as substrates/attachment sites for bacterial development in recreational waters and beaches, viii) their adverse effects on food web integrity. Importantly, as a coastal phenomenon, their appearance is often unlinked with current measures – i.e. offshore nutrient levels.

Causes and controls. Past and recent work suggests that in general Lake Erie phytoplankton are P-limited (Guildford et al. 2005). Guilford et al observed strong seasonality in measured P deficiency during 1997 which varied among basins, and less acute in the Western basin These and other authors have also detected short-term N-deficiency and P,N co-limitation (Wilhelm et al. 2003; Guildford et al. 2005). More recent bioassay and enrichment studies have suggested that plankton in the Eastern basin are co-limited by Fe, N and P, while that N chemistry influences current day phytoplankton structure in Lake Erie (Wilhelm et al. 2003; North et al. 2007). Culver et al (2005) report significant differences in PO_4 and NH_4 turnover rates between quagga and zebra mussels, with quagga mussels tending to assimilate and possibly sequester more P, or direct it more effectively to recruitment. They

[13] e.g. http://www.westernlakeerie.org/phosphorousalgae.html; http://glhabitat.org/news/glnews606.html; http://www.epa.state.oh.us/dsw/inland_lakes/Lyngbya%20wollei.pdf.

suggest that changes in mussel densities and distribution and increasing predominance of quagga mussels has important implications for the nutrient turnover rates in the inshore areas. They also attribute, like some other authors, some of the apparent increased predominance of *Microcystis* to mussel activity (cf. Madenjian 1995; Vanderploeg et al. 2001; Barbiero et al. 2006); the last mechanism is much debated. They calculated that in both 1998 and 2003, crustacean zooplankton excreted ~3 times more PO_4 than Dreissenids, highlighting the often forgotten role of zooplankton in nutrient turnover.

Overall, the risk of cyanobacterial dominance is driven by P in most Northern temperate fresh water systems, while short-term deficiencies and physico-chemical and foodweb processes mediate the response (e.g. Downing et al. 2001). Our current understanding of HABs outbreaks in the Great Lakes points to inshore areas and drainage waters as most severely affected, and also possibly serving as sources of biota and toxins for the offshore waters. Current estimates of P-loading to the lakes are inadequate, and in many cases, do not address the growing inputs from non-point sources from the watershed and shorelines. Recent research is also indicating that there may be several overlooked inputs from external and internal sources (e.g. Payton et al. 2008; Lowes & Young 2008).

Lake Ontario; status: mixed

Lake Ontario has an extensive watershed development and urban input. Blooms of cyanobacteria and related impairments (toxins, T&O compounds) have been identified recently in some inshore areas, notably Areas of Concern (AOCs). Circulation and exchange can result in plumes of affected water translocated into adjacent inshore and offshore water (Howell 2002; Hamblin & He 2003; Rao et al. 2003).

Toxins**:** Sporadic outbreaks of high MC levels are reported in *Microcystis* blooms in inshore areas (Watson et al. 2005; Boyer 2007; Hotto et al. 2007; Watson 2007). Data collected by larger Ontario municipal water treatment plants (e.g. Toronto, Hamilton, Deseronto) show episodes of elevated MC in raw water but adequate removal by the treatment in place in these large treatment plants; however it points the potential risk for less advanced removal technology small or private users (Watson et al. 2005; unpublished data). Spatial and temporal levels of these toxins in specific AOCs such as the Bay of Quinte, Hamilton Harbour and the Rochester Embayments; these indicate periods of severe impairment of inshore sites by windblown accumulations of toxic material, where MC levels can reach levels in excess of 300 $\mu g\ L^{-1}$. Recent surveys have indicated the widespread occurrence of low concentrations of anatoxin-a in both near-shore and off-shore sites in Lake Ontario (Boyer 2007; Yang 2007). Other toxins (saxitoxins and cylindrospermopsin) appear to be quite rare.

Taste-odour**:** Studies have identified three T&O patterns over the past five years which are, in general, unrelated to chl*a* or total cyanobacterial biomass. In the Northwest basin, widespread T&O is caused by abrupt and severe geosmin outbreaks, which afflict major municipal supplies between Hamilton and Cobourg in the most densely urbanized region of Canada. Late summer T&O peaks occur with considerable interannual variation in severity. Planktonic chl*a* and algal biomass remain very low and show much lower variability (2–7 µg/L; 100–500µg/L, respectively). Climate and large-scale water movement play a key role in these events by transporting offshore pelagic T&O production by dispersed and patchy distributions of cyanobacteria (*Anabaena lemmermanii*) to inshore water treatment plant intakes. The strength of the annual downwelling and associated T&O event varies among years with the duration and persistence of the east winds (Rao et al. 2003; Watson et al. 2007; Moore & Watson 2007). In the Northeast end of the lake (Kingston basin) and upper St. Lawrence River, T&O is produced annually by both geosmin and MIB. This affects an extensive shoreline (200 km) and persists over a more prolonged period (Sept.– Nov.). Primary sources are littoral and epiphytic cyanobacterial biofilms in inshore areas and macrophyte beds; midstream pelagic chl*a* remains low. Geosmin and MIB co-occur and/or peak in succession over the season, and vary in relative and absolute abundances (Watson and Ridal, 2004; Ridal et al., 2007). The Bay of Quinte develops annual cyanobacterial blooms, with patchy mid-summer increases in geosmin and MIB and cyanotoxins (Watson et al. 1997, 2004), but shows less extensive T&O impairment than the other two more ‘oligotrophic’ areas. Although T&O reaches significant levels in some areas of the Bay, the effects are localized with little impact on municipal drinking water supplies.

Benthic algal impairment is a major concern along many inshore areas in Lake Ontario. Dense mats of *Cladophora* occur along many inshore areas, with issues of plant intake and beach fouling (Higgins et al. 2008). Early spring detached mats of the green algae *Spirogyra* and other Conjugales in areas of the Lower St. Lawrence River and Northwest shoreline are recently causing severe intake fouling in drinking water plants (Watson, unpublished). Severe impairment is also manifested by benthic mats of the cyanobacteria *Lyngbya* cf. *wollei* and epiphytic colonies of *Gloeotrichia pisum* recently identified in the St. Lawrence River near the confluence of nutrient-rich tributaries (Vis et al. 2008). These populations of *Lyngbya* are non-toxic but show high geosmin production, likely the source of extensive drinking water T&O impairment in the Montreal area. Comparisons with

Lyngbya populations from Maumee show morphologically similar populations but significant differences in cell geosmin production, with a greater capacity seen in the St. Lawrence biota.

Pressures

Despite restoration efforts, the cumulative effects of past and continued impacts continue to modify the response of the lake to remedial action, and necessitate revisiting and reassessing targets established some decades ago. Major shifts in nutrient pools and recycling can result in time lags, increased variability and hysteresis and necessitate more stringent remedial targets than traditional models predict. Current and future concerns include: i) Continued introduction of invasive species**,** as discussed above, ii) Basin/shoreline development and expanding urbanization which will continue to affect point-source and non point source loadings, timing, magnitude and bioavailability. iii) Climate change, which is having, and will have, significant effects on all components of the Great Lakes, including HABs. Warming and increased storm events may favour higher productivity and more intense and widespread noxious blooms though such factors as: extended growing season, altered runoff, circulation and mixing/resuspension patterns and water column stability, warmer ambient water temperatures, changes in water levels, coastal erosion and littoral zones, altered light regimes favouring algal taxa that are tolerant to high irradiance and UV (e.g. cyanobacteria), extension of distribution and success of warm water and/or invasive taxa, indirect top-down and bottom up effects on water quality, nutrient cycling, respiration, remineralization and anoxia, sediment and hypolimnetic oxygen demand and nutrient release.

Management Implications

Concerns and recommendations

Compatibility of long term data – sampling regimes and methods.

Different sampling regimes and analytical protocols (e.g. surface, integrated etc.; enumeration; toxin analyses) employed by individual studies affect data comparability and interpretation of long-term trends (Kane et al. 2005; Conroy et al. 2005; see e.g. Table 1). The size and complexity of the Lakes means that many sampling regimes are inevitably sparse, and likely to miss spatial/temporal peaks in abundance. Many 'state of the lake' papers compare mean *planktonic* biomass and taxa, based on infrequent spring, summer and fall samples. Yet annual peaks shift in timing and size as a result of natural variability and differences are generally higher in more impacted, eutrophic areas – e.g. Western Lake Erie and AOCs (Frost and Culver 2001; Conroy et al. 2005). Furthermore, basin-wide seasonal means do not resolve temporal/spatial differences in biomass and taxa, and thus cannot identify problem areas and/or potential drivers. Littoral/benthic, epiphytic and meroplanktic algal populations are not addressed by most sampling programmes, yet can account for a high proportion of algal productivity, or represent seed beds where surface blooms originate. Extensive attached algal/cyanobacterial beds have significant effects on nutrient pools and recycling, effectively 'decouple' nutrient loading and ambient levels, and inshore-offshore exchange, and influence or mask nutrient-biomass and other empirical relationships between measured parameters. Many 'state of the lake' papers compare mean planktonic biomass and taxonomic composition among years, based on infrequent samples taken during the spring, summer and fall seasons. Alternative measures of algal abundance and productivity are often poorly correlated, as are measures of light regime. Chl*a* continues to be a target measure for management, yet there are often poor correlations among chl*a*, total algal biomass and levels of impairment. Conroy et al. (2005) pointed out the inconsistency among seasonal means from different early and recent surveys, which showed resurgence in biomass but chl*a* at a minimum. The authors suggested that the use of chl*a* vs. biomass may explain apparent contradictions among different recent studies regarding recent trends in Lake Erie; some of which conclude that biomass is still at a minimum, based on chl*a*. Secchi depth (SD) is widely used to estimate euphotic zone and as a basis for integrated samples (e.g. Table 2) using a constant conversion ratio between SD and photosynthetically active radiation (PAR); this is functional and simple; nevertheless, SD estimates visible light attenuation, which can differ significantly (seasonally and spatially) from PAR extinction.

A number of recent technologies have increased the number of tools available to diagnose these blooms; these include remote sensing, genetic probes, moored instrumentation and profilers, fluorescence based measures and genetic probes. All of these are extremely useful diagnostic tools and combined, can provide considerable insight into HABs occurrence, species, toxicity and ecology (Wilhelm 2008). However, it is important to understand the limitations of these measures, and to use them in combination with others. Remote scanning work has potential but measures only surface material, and furthermore needs to be carefully groundtruthed with field samples. Fluorescence-based profiling of algal assemblages is gaining widespread, often indiscriminate use as a measure of community structure, however these need careful calibration, preferentially with local biota. Comparison among individual instruments deployed in parallel has shown wide discrepancies (Boyer unpublished data). Fluorescence data need to be interpreted with caution: wavelengths used to measure chromophytes and cryptophytes overlap with those used for the diatoms; those used for cyanobacteria overlap with colored dissolved organic matter (CDOM). At low biomass, resolution is poor, notably between cyanobacteria and crytophytes (Boyer unpublished data; Watson & Kling, unpublished data).

Impairment criteria

As noted above, the current efforts target parameters that are often unrelated to levels of impairment and/or are based on non-quantitative measures. Toxins should be systematically investigated, particularly in high risk source-waters, using regular monitoring at recreational areas and intake zones, mid-late summer spatial surveys during high risk periods and an alert level framework such as developed by the World Health Organization (Watzin et al. 2006). More effective criteria for T&O would include regular measures of the most problematic compounds (e.g. geosmin, MIB, isopropyl thiol, β-cyclocitral) in source waters and municipal supplies, and comparison against their odour threshold levels.

In the Great Lakes, considerable progress has been made in many areas towards Remedial Action Plan (RAP) goals, not the least of which has been an increased public awareness and participation in this initiative. However, remedial efforts are addressing a moving target. These ecosystems are under constant assault by an expanding human population and emerging threats. Advances in our understanding of these systems have not kept pace with these changes. It is essential that remedial and management programmes frequently revaluate the list of target goals, their acceptable levels and progress towards these.

Nutrient levels may, or may not predict toxin or odour outbreaks. Blooms appear to be local and inshore in origin and can spread over considerable areas – likely the combined result of growth and translocation of surface scums. The relative importance of these different mechanisms is not well resolved. There are numerous incidental reports, media releases and websites that may inflate these issues. Most attention is focussed on surface scums, which inevitably bias samples and perceived severity. These can appear suddenly, giving the impression of rapid growth, but represent biomass which has been present and developing in the water column over a preceding undefined period of time.

The effects of invasive taxa can be numerous, both via direct impairments (blooms/toxins/odour/ fouling/ fisheries etc) and indirect effects on ecosystem structure and function (food webs, nutrient pools and recycling, water quality etc.). In addition, their appearance is of concern because of the implications in terms of i) vectors (predominantly ballast water) and ii) changes in environment which facilitate their establishment (temperature, substrate, salinity, pollutants etc.). Other biota such as macrophytes may indirectly or directly affect the proliferation of HABS species by modifying light and nutrient levels, and/or providing substrate for epiphyte growth. The influence of invasive species of zooplankton & benthic grazers (e.g. *Cercopagis pengoi, Bythotrephes cederstroemi*) on HABs development in the Great Lakes is unknown.

Current models and sampling design.

Traditional nutrient-biomass management models are derived from empirical relationships among seasonal averages. Many are applied indiscriminately, without considering their underlying assumptions and limitations (e.g. Watson et al. 2008). In particular, the models incorporate bias from sampling protocols, maxima and minima, surface scums, deep layer maxima and other biomass aggregations. Depth-segregated maxima are a particular consideration with cyanobacteria, many of which are buoyancy-regulating or mat-forming taxa. Benthic and littoral communities can also be major sources of impairment. Different nuisance algal/cyanobacterial taxa respond very differently to stressors and nutrient loading; many have developed different strategies to adapt to these factors. These models do not predict the *biomass maxima* for a given system, when toxins and other related impairments are of most concern.

Scientists and managers are faced with two different strategies when designing sampling regimes, each has its use and limitations and must meet the underlying question and management goals: **i)** random sample design; representing a unbiased sampling of among nearshore/offshore/ influences and impacts; this strategy averages out maxima, may minimize key areas of concern and is often unable to resolve impairments and trace local causes. **ii)** biased towards high risk, targeted areas, time- and depth-resolved sampling, periodic extensive spatial surveys during identified high risk periods (e.g. late summer); this strategy. This approach provides a better assessment of extreme conditions and localized risk and targets (but may miss) maxima. Ideally, a combination of both strategies provides the best HABs assessment and monitoring framework. However, coordination and logistics of these programmes are difficult, especially among multi-agency and international partners working with a large, highly fragmented basin.

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List of Tables

Table 2. Summary of toxin levels in Lake Erie from 5 surveys (from Boyer 2007).

Table 1. Representative surveys of phytoplankton in Lake Erie and sampling regimes (from Conroy et al. 2005, Boyer 2007, Watson unpublished).

Year	Reference	Sampling regime	Field methods
1970	Munawar & Munawar 1976	Apr- Dec, 4-week intervals, all basins, 25 stations	Van Dorn, 1, 5m, mixed- layer integrated
1978	Munawar & Munawar 1996	Jun-Sept; CB & EB only; 18 stations (different sites than 1970)	Van Dorn, 1, 5m depth, mixed-layer integrated
1978	Devault & Rockwell 1986	May-Nov, all basins, 9 cruises; 87 stations	Niskin; *stratified*: 1m, 1m above metalimnion, thermocline, hypolimnion, bottom. *Unstratified*: 1m, mid -depth, bottom-1m.
1983-1987	Makarewicz 1993	spring, summer, fall; all basins, 33 cruises, 21 stations	Niskin; *deep*: 1, 5, 10, 20m. *shallow* (West B) 1m, mid-depth, bottom-1m
1998	Barbiero & Tuchman 2001	spring (7-9 April) summer (2-4 Aug), 20 stations	Niskin; combined 0.5m, 5 m, 10 m, lower epilimnion
1996-2002	Frost & Culver 2001	late spring-late Sept.-Oct.; all basins; 30 - 80 stations	integrated tube 0-[2*SD]
2000-2006 Boyer;/Watson/ Richardson	GLERL-LGL; EC surveillance & research	late spring-late Sept.-Oct.; all basins; 30 - 80 stations	Van Dorne / Rosette 1m and integrated mixed layer

Table 2. Summary of toxin levels in Lake Erie from 5 surveys (from Boyer 2007).

Cruise, date		# samples	toxin	% samples toxic	max level µg/L	Comments
Brittain	Sep-96	44	MC	10	3.4	WB only
MELEE–VII	Jul-02	119	MC	7	0.7	whole lake; highest at Sandusky, Long Pt., Rondeau Bays
			ATX	14	0.04	
			PSTs	0		
MELEE–VIII	Jul-03	59	MC	41	0.65	whole lake; highest in WB & Sandusky Bay
			ATX	5	0.11	
Lake Guardian & OSU	Aug-03	48	MC	60	21	WB only, highest nr. Maumee R.
			ATX	4	0.2	
MELEE–IX	Jul-04	40	MC	38	>1	Highest nr. Maumee & Sandusky Bay
			ATX	33	0.6	
			CYL	0		
Limnos	Aug-04	13	MC	85	2.4	WB only
			ATX	31	0.07	
			CYL	15	0.18	
MC=Microcystin; ATX=anatoxin-a; PSTs=saxitoxin + neosaxitoxin; CYL=cylindrospermopsin						