Plastic debris in the Laurentian Great Lakes: Classification, distribution and environmental drivers

by

Alexander G.J. Driedger

A thesis
presented to the University of Waterloo
in fulfilment of the
thesis requirement for the degree of
Master of Science
in
Earth Sciences

Waterloo, Ontario, Canada, 2015

AUTHOR'S DECLARATION

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be electronically available to the public.

STATEMENT OF CONTRIBUTIONS

Chapter 1 is a review paper that was published in the Journal of Great Lakes Research (Driedger et al., 2015). I am the first and corresponding author on the study and contributed the vast majority of text, figures, and tables. Co-authors on the study are Hans Dürr, Kristen Mitchell, and Philippe Van Cappellen. I am the sole author of Chapter 2 and all other material presented in this thesis.

ABSTRACT

Pollution by plastic debris is an increasing environmental concern in the Laurentian Great Lakes: it affects open-water, shoreline, and benthic environments. Many plastics degrade slowly in the environment and may have long-term adverse ecological and economic impacts, including the dispersal of persistent organic pollutants. Plans to combat and curtail plastic debris pollution in the Great Lakes will come at a significant economic cost, likely in excess of $400 million annually. Open-water surveys reveal that, in certain areas of the Great Lakes, surface water densities of plastics are as high as those reported for areas of litter accumulation within oceanic gyres. Data from volunteer beach cleanups show that typically more than 80% of anthropogenic litter along the shorelines of the Great Lakes is made of plastics and that in situ anthropogenic activity including shoreline and beach visitors is the primary source of litter. Other sources of plastic debris to the Great Lakes include microplastic beads from consumer products, pellets from the plastic manufacturing industry, and waste from shipping and fishing activities. This thesis reviews the current state of knowledge on plastic pollution in the Great Lakes and uses citizen science data to explore the classification, distribution, and environmental drivers of plastic debris on Canadian shorelines of the Great Lakes.

ACKNOWLEDGEMENTS

Chapter 1 was written as a result of the Great Lakes Plastics Workshop held in Waterloo, Ontario on April 4-5, 2013, as well as a recently completed feasibility study on remote sensing and spectral characterization of plastic debris in open-waters and littoral zones conducted at the University of Waterloo. Chapter 1 also addresses an action point in the Great Lakes Land-Based Debris Action Plan (Goal 1, Objective 3, Action Strategy 1, NOAA Marine Debris Program, 2014). It was supported by a grant from Canada's Federal Economic Development Agency for Southern Ontario to Communitech Corporation's DATA.BASE program, with matching financial support from COM DEV International Ltd., exactEarth Ltd., and the Canada Excellence Research Chair (CERC) program. Beach survey data for Chapters 1 and 2 were provided by Alliance for the Great Lakes and the Vancouver Aquarium. I would like to thank Olga Lyandres and Jamie Cross of the Alliance and Susan Debreceni and Kate Le Souef of the Vancouver Aquarium for their advice and enthusiastic support. I would also like to thank the thousands of volunteers and personnel that participated in the Alliance's Adopt-a-Beach^{TM} program and Vancouver Aquarium's Great Canadian Shoreline Cleanup for their time and hard work generating the survey data. And finally, I would like to thank my supervisor, Hans Dürr, and advisory committee, Kristen Mitchell, Andrea Vander Woude, and Philippe Van Cappellen, for their continual guidance, patience, and support of my research.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

INTRODUCTION

Plastics are synthetic materials made from organic or semi-organic polymers. Their low cost, ease of manufacture, and versatility have made them increasingly prevalent in society. Plastics have, perhaps more than any other material, transformed industry, nurtured scientific and technological innovation, and contributed to humanity's health, safety, shelter, transportation, commununiticion, and entertainment. Their importance and scope of impact cannot be overstated.

Yet, plastic litter are a growing environmental concern, especially in aquatic environments. Worldwide production of plastic has grown an average 8.7% per year since 1950; in 2013 nearly 300 million tons were produced (Worldwatch Institute, 2015). A high production rate, combined with low rates of recovery and often inadequate disposal, has resulted in between 4.8 to 12.7 million tons of plastic debris entering the oceans annually (Jambeck et al., 2015). More than 5 trillion plastic particles weighing over 250 thousand tons are estimated to be polluting the oceans' surface waters alone (Eriksen et al., 2014). Plastics are non-biodegradable, and thus may persist in the environment indefinitely; they will break down into smaller pieces but may never fully mineralize.

Since the 1970's, scientists have identified plastics in the marine environment as a harmful substance; Carpenter and Smith (1972) first reported plastic particles in the Sargasso Sea and suspected they were a source of polychlorinated biphenyls (PCBs) in oceanic organisms. More recent studies have confirmed their hypothesis (Ryan et al., 1988; Teuten et al., 2009) and have also drawn attention to other harmful effects of plastic debris including the distribution of toxins (Holmes et al., 2012; Mato et al., 2001), accumulation in the food chain and associated health risks to aquatic species (Boerger et al., 2010; Setälä et al., 2014; Wright et al., 2013), loss of tourism revenue at beaches (Mouat et al., 2010; Stevenson, 2011), and interference with shipping and fishing activities (Gregory, 2009).

While hundreds of studies have been published about plastic debris in the oceans, few have analyzed plastic debris in fresh water environments including the Laurentian Great Lakes. To date, only 22 studies have examined plastic debris in freshwater environments and just 8 have focused on the Great Lakes. Furthermore, the first freshwater study was not published until 2011, and over 75 percent of freshwater studies have been published in the last two years. Given the lack of research, many knowledge gaps remain about the sources, abundance, distribution, breakdown, and environmental impacts of plastic debris in freshwater environments.

Chapter 1 reviews existing research about plastic debris in the Great Lakes; it identifies knowledge gaps and proposes future research directions. Chapter 2 explores the classification, distribution, and environmental drivers of plastic debris along the Canadian shorelines of the Great Lakes using citizen science data generated by the Great Canadian Shoreline Cleanup program.

CHAPTER 1: PLASTIC DEBRIS IN THE LAURENTIAN GREAT LAKES: A REVIEW

INTRODUCTION

Plastic litter is found in marine and freshwater ecosystems all around the globe. The Laurentian Great Lakes are no exception: plastic debris is present in each of the lakes (Eriksen et al., 2013a; Hoellein et al., 2014; Zbyszewski and Corcoran, 2011; Zbyszewski et al., 2014). The durability and persistence of plastics, combined with their rising production and low rates of recovery (US EPA, 2014), are likely causing a net accumulation of plastic debris along shorelines, in surface waters, throughout the water column, and in bottom sediments (Barnes et al., 2009; Ryan and Moloney, 1993).

Plastic debris is variably classified according to size, origin, shape, and composition. While there are no internationally agreed upon size classes, ‘microplastic debris’ generally refers to plastic particles smaller than 5 mm (Arthur et al., 2009). Furthermore, the term microplastic debris is often restricted to particles larger than 333 µm, because in most open-water studies neuston nets with a mesh size of 333 µm are used to collect debris (Andrady, 2011; Barnes et al., 2009). The term ‘microscopic plastic debris’ is reserved for plastic particles smaller than 333 µm that are retained on a 0.45 µm pore size filter (Andrady, 2011). Plastic debris larger than 5 mm is referred to as ‘macroplastic debris’. Plastic debris exhibits a wide range of shapes; in addition to recognizable plastic objects, the most common shapes are fragments, films, pellets, lines, fibers, filaments, and granules.

Plastic debris is often classified as either primary or secondary. Primary plastics are in their original or close-to-original form when collected, such as bottle caps, cigarette butts, microbeads or resin pellets. Secondary plastic debris encompasses the smaller pieces of plastic resulting from the breakdown of primary debris through various environmental degradation processes (Wagner et al., 2014). The composition of plastic refers to the polymer type, which in turn determines the density of debris. Low-density plastics, such as polypropylene and polyethylene, produce debris that is less dense than water and therefore likely to remain afloat. Plastics that are denser than water and thus tend to sink include polyethylene terephthalate, polystyrene, and cellulose acetate. The densities of plastics found in the Great Lakes are listed in Table 1.1.

Plastic debris can have wide-ranging ecological and economic impacts in both freshwater and marine environments. Macroplastics pose a health risk to aquatic animals, including fish, turtles, and birds, because of possible entanglement and ingestion (Boerger et al., 2010; Codina-García et al., 2013; Gregory, 2009; Sheavly and Register, 2007). Ingestion of plastic may cause internal bleeding, abrasion and ulcers, as well as blockage of the digestive tract (Wright et al., 2013). Plastic debris may act as a vector for contaminants, including persistent organic pollutants (POPs) and heavy metals (Ashton et al., 2010; Holmes et al., 2012; Mato et al., 2001; Rios et al., 2010; Zarfl and Matthies, 2010). Sorption to plastics has been shown to limit the biodegradation of organic contaminants, increasing their persistence in the environment (Teuten et al., 2009). Plastic debris can also transport non-native species (Barnes et al., 2009; Gregory, 2009) and be colonized by microbes including possible pathogens (Wagner et al., 2014; Zettler et al., 2013). In littoral zones, the accumulation of sinking plastic debris and the dragging of fishing nets may disrupt bottom sediments, displace or smother infauna, and affect the structure and functioning of benthic microbial communities (Goldberg, 1994).

Accumulation of plastic debris in coastal areas can deter recreational usage, pose a hazard to swimmers and divers, and carry a risk of minor cuts or abrasion injuries to beach-goers (Sheavly and Register, 2007). Plastic debris can reduce revenue generated from tourism due to forced beach closures, but also because tourists use beach cleanliness as a dominant factor in selecting recreational destinations (Jeftic et al., 2009). Macroplastic debris represents a navigational and structural hazard to shipping vessels and smaller marine vehicles, including burnt out water pumps and entangled propellers (Mouat et al., 2010). Derelict fish nets and other lost plastic gear may trap commercial fish accidentally, hence removing them from the pool available for harvest (Gregory, 2009).

Plastic debris in the environment will break down through a combination of photo- and thermal-oxidative degradation by ultraviolet (UV) radiation, mechanical weathering, and biodegradation, but complete mineralization may not be possible, or then only after hundreds or thousands of years (Andrady, 2011; Corcoran et al., 2009; Gregory and Andrady, 2003; Shah et al., 2008). The breakdown products, including microplastic and microscopic plastic debris, create additional challenges. As plastics degrade they can release toxic chemicals that were initially incorporated during their manufacturing or sorbed to their surfaces in the environment. These chemicals include phthalates, nonylphenols, bisphenol A (BPA), heavy metals, and polybrominated diphenyl ethers (PBDEs) (Bittner et al., 2014; Cheng et al., 2010; Nakashima et al., 2011; Teuten et al., 2007), which can disrupt endocrine functions and cause harmful reproductive and developmental effects in aquatic animals (Meeker et al., 2009). Smaller plastic debris is also more bioavailable – several aquatic species have been found to ingest microplastics

– and the trophic transfer of plastics along aquatic food webs has been verified, hence posing a health threat to aquatic ecosystems (Andrady, 2011; Boerger et al., 2010; Fossi et al., 2012; Teuten et al., 2009). The direct transfer of plastic-sorbed toxins to organisms through oral ingestion represents an additional hazard (Rochman et al., 2013; Ryan et al., 1988). While the possible transfer of plastic-sorbed toxins to humans through consumption of aquatic species is of concern, it has yet to be demonstrated.

The Great Lakes have likely been polluted with plastic debris since the mid-twentieth century when mass production of plastics began in North America (Thompson et al., 2009). However, while numerous studies have focused on plastic debris in marine systems (e.g. Cole et al., 2011; Cózar et al., 2014; Eriksen et al., 2013b; Law et al., 2010; Moore et al., 2001), few have explored the distribution and fate of plastics in freshwater systems. The reason is not entirely clear, but may have been exacerbated by the use of ambiguous terminology (the term 'marine debris' is well-established and its application to lentic environments may be a cause for confusion), and a lack of communication between freshwater and marine researchers. Nonetheless, plastics pollution may represent an equal, if not greater, threat to lakes compared to the oceans. In this context, the present paper (1) reviews the current state of research about plastic pollution in the Great Lakes, (2) identifies knowledge gaps, and (3) proposes future research directions.

PLASTICS IN THE GREAT LAKES

DATA AVAILABILITY

Only a limited number of peer-reviewed papers present quantitative data on the abundance and distribution of plastic debris in the Great Lakes (Table 1.2). Eriksen et al. (2013a) collected debris in Lakes Superior, Huron, and Erie using a manta trawl lined with a 333 µm mesh size net. The surface water concentrations of microplastics extrapolated from the debris counts varied between 0 and 0.4663 items per m^2. Plastic debris included pellets, fragments, foam pieces, film, and line. Additional expeditions by Mason and colleagues in 2013 confirmed the presence of pelagic microplastics in Lakes Michigan and Ontario (Dr. Sherri Mason, State University of New York at Fredonia, unpublished data).

Debris counts by Hoellein et al. (2014), Zbyszewski and Corcoran (2011), and Zbyszewski et al. (2014) yielded concentrations of micro- and macroplastic debris along the shorelines of Lakes Michigan, Huron, St. Clair, and Erie, between 0 and 34 items per m^2. However, whereas in Lake Huron 93% of plastic debris were made of pellets, in Lakes Erie and St. Clair fragments were more abundant (Zbyszewski et al., 2014). In Lake Michigan, cigarette filters were found to be a major source of shoreline accumulation of macroplastic debris (Hoellein et al., 2014). The surface textures of plastic debris samples from Lakes Huron, St. Clair, and Erie have been examined by scanning electron microscopy (SEM): 78 and 37% of samples showed signs of mechanical and oxidative weathering, respectively (Zbyszewski et al., 2014). Hoellein et al. (2014) also reported data on anthropogenic debris >1 cm in the North Branch of the Chicago River. The average concentration of plastic debris was found to be greater in the riparian zones than in bottom sediments of the river.

Volunteer-led cleanups of beaches and coastal areas organized by non-governmental organizations (NGOs) have gathered a wealth of information on the abundance and distribution of plastic debris along the Great Lakes shorelines. These cleanup activities engage individuals, schools, marinas, civic associations, businesses, and governmental agencies. Since 1991, the Alliance for the Great Lakes has run the Adopt-a-Beach^{TM} (AAB) program whose volunteers not only conduct debris removal but also test water quality and assess general beach health (Alliance for the Great Lakes, 2014). The number and type of anthropogenic debris items collected are reported in AAB's online information system: litter data from 2002 onwards are accessible through the website www.greatlakes.org/adoptabeach. In 2012, 12,618 AAB volunteers cleaned roughly 1240 km of Great Lakes shoreline, removing nearly 20,000 kg of anthropogenic debris. The AAB program is part of the Ocean Conservancy's broader International Coastal Cleanup, a global initiative promoting trash-free waters.

In Canada, the Vancouver Aquarium and World Wildlife Fund run the Great Canadian Shoreline Cleanup (GCSC), which includes activities in the Great Lakes region. Based on the cleanup data reported on GCSC's website (www.shorelinecleanup.ca), we estimate that, in 2012, 2,925 GCSC volunteers cleaned roughly 280 km of Great Lakes beaches and removed around 9,300 kg of anthropogenic debris. Note that, as with the AAB program, the cleanup activities of GCSC primarily target visible litter and, thus, mainly provide information on the abundance and distribution of macroplastic debris.

SOURCES OF PLASTICS

Eriksen et al. (2013a) found high abundances of plastic pellets <1 mm in the surface waters of Lakes Superior, Huron, and Erie. A major fraction of these pellets are most likely microbeads that are used as abrasive agents in a range of consumer products, including exfoliating creams, soaps, toothpastes, shampoos, lip gloss, eye liner, sunscreens, and deodorants. Microbeads that are flushed down sink and shower drains enter the wastewater collection system. Canadian and American wastewater treatment regulations, however, make no provision for microplastic debris, including microbeads: wastewater treatment plants (WWTPs) are currently not required to monitor microplastics in influent or effluent streams. For a WWTP to effectively remove microbeads, a form of advanced filtration is probably required, for example, fine- or micro-screens, microfiltration, sand filtration, or mixed media filtration (Nalbone, 2014). Many WWTPs in the Great Lakes region are not equiped with such treatment systems. For example, in the state of New York, which borders sections of Lakes Ontario and Erie, 66% of WWTPs do not use advanced treatment methods. Also, preliminary research conducted at the State University of New York at Fredonia positively identified microbeads in the effluent of 6 out of 7 WWTPs sampled in New York state (Nalbone, 2014).

Plastic fibers are also suspected to contribute to the microplastic loadings of the Great Lakes, although no conclusive data on their presence in the Great Lakes exist to date. Polyester and acrylic fibers have been widely observed in marine sediments (Browne et al., 2011). Small plastic fibers are released from synthetic fabrics in washing machines. Households and textile laundering facilities may therefore represent a significant source of plastic fibers (Eriksen et al., 2013a). The fibers can be transferred to surface water bodies directly via the effluent discharges of WWTPs, or indirectly via their solid residues. When the latter are used as fertilizers and compost material on crop fields, the fibers may be remobilized and ultimately reach natural or man-made waterways (Browne et al., 2011; Habib et al., 1998; Zubris and Richards, 2005).

Further research should include a throrough review of the current microplastic removal efficiencies of WWTPs surrounding the Great Lakes region, as well as an assessement of the plant upgrades and associated costs required to effectively prevent the discharge of microbeads and other microplastics from water treatement systems. Equally important, the fate of microplastics in solid residues of WWTPs should be investigated because they could represent a significant, but unknown, source of environmental plastic pollution.

Plastic resin pellets, a raw material used in the manufacturing of plastic products are a common pollutant along Great Lakes shorelines (Zbyszewski et al., 2014). Spillage during transport and subsequent rain events can cause the entry of resin pellets into streams and storm sewers. In urban areas with combined sewer systems, pellets and other plastic debris in stormwater may be discharged directly into streams and rivers, and ultimately reach the Great Lakes. Hoellein et al.'s (2014) analysis of anthropogenic debris in the Chicago River, although limited to only one tributary waterway, implies that rivers could represent a major transport pathway of plastic debris to the Great Lakes.

To help delineate the sources of plastic debris to beaches along the Great Lakes, we sorted the cleanup data from AAB and GCSC according to activity class (Figure 1.1; for details on the activity classes, see Appendix A, Table 1). The 'shoreline and recreational' and 'smoking-related' classes contain the vast majority of debris. The most commonly reported items in the AAB and GCSC surveys are cigarette filters followed by plastic food wrappers and containers. Fishing gear comprises only a small amount of shoreline anthropogenic debris in the Great Lakes (~1%). In 2012, plastic anthropogenic debris items made up between 77 and 90% of the total shoreline debris collected during AAB and GCSC cleanups. Because intact or near-intact debris, such as cigarette filters and food-related items, mostly represent in situ litter, beach-goers appear to be a major source of macroplastic debris along Great Lakes shorelines. However, some of this debris may also be transported from urban areas by wind or stormwater and ultimately end up on the beaches. In addition, because volunteer cleanups preferentially target beaches used for recreational purposes, they tend to introduce a site sampling bias.

For comparison, in the marine environment on average between 60 and 80% of shoreline debris items consist of plastics (Gregory and Andrady, 2003). Land-based sources account for up to 80% of the total debris input to the oceans with offshore sources making up the remainder (Allsopp et al., 2006). Land-based sources of plastic debris to the oceans include riverine outflow, landfills, stormwater drains, textile laundering facilities, petrochemical plants, and WWTPs, as well as direct inputs in coastal areas, for example trash left by recreational beach users (Browne et al., 2011; Wright et al., 2013). The primary offshore source of plastic debris in the oceans is derelict fishing gear (nets, lines, traps). In six separate studies, fishing gear was found to represent on average more than 3% of the total number of debris items collected along marine shorelines (Santos et al., 2009). Other offshore sources including illegal dumping of plastic waste from ships, and the release of plastic resin pellets and products when shipping containers are lost at sea (Andrady, 2011).

DISTRIBUTION OF PLASTICS

Existing survey data indicate that areas in the Great Lakes region with greater human and industrial activity are generally associated with higher concentrations of plastic debris in the adjacent Great Lakes basin(s). Among Lakes Superior, Huron and Erie, Eriksen et al. (2013a) found the highest concentrations of pelagic microplastics in Lake Erie, with on average 0.1055 plastic items per m^2. This observation is in line with the higher population density and greater industrial activity characterizing Lake Erie's catchment. The higher reported microplastic density in Lake Superior compared to Lake Huron is more surprising (Eriksen et al., 2013a). However, the surface water samples in Lake Superior were collected closer to shore than in Lake Huron, and thus closer to the land-based sources of plastic debris. Among the shoreline locations surveyed in Lake Huron by Zbyszewski and Corcoran (2011), and in Lakes Erie and St. Clair by Zbyszewski et al. (2014), the highest concentration of plastics (34 items per m^2) was observed at Sarnia Beach, which is located along the southern shore of Lake Huron in relatively close proximity to petrochemical plants that produce plastic resin pellets. The same authors also found high concentrations of plastic debris on Lake Erie beaches that receive high numbers of visitors each year.

Shoreline debris counts in or near urban centers may be affected by grooming of public beaches. For example, the two largest cities along the Great Lakes, Toronto and Chicago, have their beaches groomed daily from mid-spring to late summer, that is, during peak use (Chicago Park District, 2014; City of Toronto, 2009). This may explain why Hoellein et al. (2014) found relatively little plastic debris (0.0005 items per m^2) on Lake Michigan beaches surveyed near Chicago. In addition, volunteer beach cleanups often target urban shorelines because of greater accessibility and impact. Microplastics and microscopic plastic debris in urban areas are probably not greatly affected by volunteer cleanups or grooming activities. Manual litter removal and mechanical grooming equipment are mostly ineffective at removing the smallest plastic debris, including cigarette filters.

The existing AAB and GCSC survey data indicate that the vast majority of anthropogenic debris along the Great Lakes shoreline consists of plastics. On an item-by-item basis, plastics comprise the greatest proportion of anthropogenic debris in Lake Erie and Lake Huron, and the lowest in Lake Superior. The percentages of debris items that were found to be plastics at the AAB and GCSC cleanup locations in 2012 are shown in Figure 1.2 (for details on the items classified as plastics, see Appendix B). Although historically AAB and GCSC cleanups have only yielded information on macroplastic debris, in 2013 GCSC volunteers also began to collect and report plastic debris <2.5 cm, hence producing data that could potentially provide insights into the distribution of microplastic debris on the beaches of the Great Lakes.

The variable number and expertise of the volunteers, as well as the irregularly distributed temporal and spatial occurrences of cleanups, introduce uncertainties and sampling biases in beach cleanup data that may be difficult to fully account for. For instance, cleanup activities tend to be more frequent near urban areas and during the summer months. Comparison between shoreline survey data is also complicated by the use of different metrics: AAB and GCSC cleanups report linear debris densities, i.e. debris counts per m of shoreline, while Zbyszewski et al. (2014) and Hoellein et al. (2014) report surface-normalized debris densities, i.e. counts per m^2 of beach. Furthermore, the AAB and GCSC cleanup databases do not report the weights of individual debris classes, only the total weight of debris collected per cleanup, hence limiting the extent to which quantitative analyses of plastic debris distributions can be carried out. Overall, there is much scope for a closer dialogue between cleanup organizers, beach groomers, and the Great Lakes science community to streamline the reporting and optimize the utilization of shoreline plastic debris distribution data.

Surface current patterns undoubtedly influence where plastic debris concentrate in the Lakes. For instance, the highest reported concentrations of pelagic plastic particles in Lake Erie are located in the eastern basin, in areas of converging surface currents (Eriksen et al., 2013a). Similarly, dominant surface currents help explain the spatial distribution of plastic debris along Lake Huron's shoreline (Zbyszewski and Corcoran, 2011). However, to our knowledge, there has been no systematic research linking the spatial and temporal distributions of plastic debris to water circulation in the Great Lakes.

In addition to circulation, the density of plastic debris is expected to be an important factor controlling their distribution. Given their low densities, it is not surprising that polyethylene, polypropylene, and expanded polystyrene (e.g. Styrofoam) are the most commonly observed plastics in the Great Lakes' surface waters and shorelines (Eriksen et al., 2013a; Zbyszewski et al., 2014). The density of plastic debris may be altered by biofilm growth, which may cause otherwise buoyant debris to sink (Andrady, 2011; Wright et al., 2013). According to one estimate, 70% of plastic debris entering the oceans ultimately sink (Oceaneye, 2013). Vertical distributions of plastics in the water column of the Great Lakes have, to our knowledge, not been determined. Similarly, little is known about plastic debris deposited at the bottom of the lakes. Preliminary analyses of bottom sediments from a near-shore region of Lake Superior have shown the presence of plastic debris (Dr. Lorena Rios Mendoza, University of Wisconsin Superior, unpublished data). Thus, future research should include surveys of the abundance, size, composition, and extent of degradation of plastic debris in the water column and sediments of the Great Lakes.

COMPARISON TO OTHER AQUATIC ENVIRONMENTS

Data on plastic debris are available for the surface waters and sediments of Lake Geneva, Switzerland (Faure et al., 2012) and Lake Hovsgol, Mongolia (Free et al., 2014), as well as for shoreline sediments of Lake Garda, Italy (Imhof et al., 2013). Compared to these lakes, Lake Erie exhibits the highest average concentration of pelagic microplastics (0.1055 items per m^2, compared with the next highest value of 0.0516 items per m^2 for Lake Geneva). The population of Lake Geneva's watershed, normalized to the lake's surface area (1627 people per km^2) is significantly greater than for Lake Erie (483 people per km^2). Thus, population density alone cannot account for the differences in the concentrations of pelagic microplastic debris. This remains true even if we factor in the populations living in the watersheds of Lakes Superior, Michigan, and Huron, which ultimately drain into Lake Erie. At this point, we can only speculate that Lake Geneva receives lower inputs of plastics, possibly because of higher plastics recycling rates and WWTP retention efficiencies. In a similar vein, the average abundance of pelagic microplastic debris in Lake Hovsgol is considerably higher than in Lakes Superior and Huron, even though the population densities of the watersheds of Lakes Superior and Huron are 4 and 25 times greater than Lake Hovsgol's, respectively (when normalized to the corresponding lake surface areas). Possibly, the greater level of pelagic microplastic pollution of Lake Hovsgol reflects the relatively long water residence time of the lake and the lack of a modern waste management system (Free et al., 2014).

Comparison between the concentrations of pelagic microplastic debris in the Great Lakes and those in ocean surface waters is complicated by the different ways in which marine plastic debris concentrations are reported. While in some studies concentrations are expressed in number of plastic debris per unit ocean surface area (Goldstein et al., 2013; Law et al., 2010; Moore et al., 2001), other authors express the number of items per unit volume of water (Doyle et al., 2011; Lattin et al., 2004; Moore et al., 2002), while others use units of mass of plastic per unit area or volume of water (Cózar et al., 2014; Day and Shaw, 1987). Nonetheless, based on the average concentration of pelagic microplastic debris in Lakes Superior, Huron and Erie (0.0425 items per m^2), it would appear that the Great Lakes are as polluted with plastic debris as known areas of litter accumulation within ocean gyres. For comparison, the average concentrations of plastic debris in the South Pacific Gyre, the North Atlantic Gyre and the North Pacific Gyre are 0.0269 (Eriksen et al., 2013b), 0.0203 (Law et al., 2010) and 0.3343 items per m^2 (Moore et al., 2001), respectively.

Although sizes of pelagic plastic debris in the oceans have been examined by a number of authors (Doyle et al., 2011; Morét-Ferguson et al., 2010; Shaw and Day, 1994; Yamashita and Tanimura, 2007), only the studies of Lattin et al. (2004), Moore et al. (2002, 2001) and Eriksen et al. (2013b) report size distributions over the same range as those of Eriksen et al. (2013a) for the Great Lakes. Similarly, Free et al.'s (2014) analysis of plastic debris in Lake Hovsgol can be directly compared to the size data of Eriksen et al. (2013a). The comparison reveals a much greater percentage of microplastic debris <1 mm in the surface waters of the Great Lakes (81%) relative to the five other marine and freshwater studies (Figure 1.3).

Another difference with marine environments is the much larger proportion of pellets in <1 mm pelagic plastic debris of the Great Lakes (58%), relative to fragments and other shaped debris (Eriksen et al., 2013a). In contrast, pellets make up less than 1% of the <1 mm debris in the North and South Pacific Gyres, while fragments represent 73 and 94%, respectively (Eriksen et al., 2013b; Moore et al., 2001). As pellets in consumer products are often intentionally <1 mm, the available data suggest a greater relative contribution of microbeads to open water plastic debris in the Great Lakes than in the oceanic gyres. Fragments resulting from the breakdown of larger plastic objects appear to preferentially accumulate in the oceanic gyres.

The smaller sizes of pelagic plastics in the Great Lakes, compared to the debris found in oceanic gyres, could possibly also reflect differences in degradation. Unlike the fairly steady surface currents that permanently trap plastic debris within the oceanic gyres, the surface currents in the Great Lakes lack persistence and are driven more by short-term atmospheric forcing because of the Lakes' much smaller sizes (Beletsky et al., 1999). This, combined with the Great Lakes' greater shoreline to surface area ratio, would tend to intensify the interactions of the plastic debris with the shoreline. Along and near the shore, plastic debris are subject to more intense mechanical and (photo-) oxidative degradation, thus accelerating the breakdown into smaller pieces (Andrady, 2011).

The presence of plastic debris in nearshore marine sediments has been documented in several studies (Browne et al., 2011, 2010; Ribic, 1998; Santos et al., 2009). Although data as a function of depth below the sediment surface are available for a number of cases, most studies are limited to plastic debris in the very surface layer of sediment surface. The reported microplastic debris abundances within the upper 5 cm of marine sediments range from 0.21 to more than 77,000 items per m^2 (Hidalgo-Ruz et al., 2012). These values are much higher than those of Zbyszewski et al. (2014), who found between 0 and 34 plastic items per m^2 in sediments along the shorelines of Lakes Huron, St. Clair, and Erie. It should be noted, however, that, in contrast to the marine studies, synethic fibers and other microplastics and microscopic plastic debris were not included in the survey by Zbyszewski and coworkers.

Hoellein et al. (2014) compared the abundance and mass of anthropogenic debris on Lake Michigan beaches with data from four marine beaches located in New Jersey, USA, Costa dos Coqueiros, Brazil, Darsait, Oman, and the Transkei Coast, South Africa (Ribic, 1998; Santos et al., 2009). All of the marine beaches had significantly higher plastic debris counts than the Lake Michigan beaches, possibly due to the systematic grooming of the latter (see above). The river bottom sediments and riparian zones of the Chicago River, however, exhibited comparable counts of anthropogenic debris as the marine beaches. Based on the available data, it would seem that beaches, nearshore sediments, riparian zones, and possibly other transitional environments may represemt preferential sites of accumulation of plastic debris. This hypothesis will require more data to be confirmed. An important step forward would be the development of internally consistent sampling protocols and metrics to compare data from different studies.

CONTAMINANTS

Preliminary work by Dr. Lorena Rios Mendoza at University of Wisconsin Superior (unpublished results) have shown that a portion of plastic debris collected in the surface waters of Lake Erie carry polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs), both of which are capable of causing cancer and birth defects. Concerns are also being voiced about plastic fibers detected in the guts of Great Lakes' fish, which may carry similar contaminants (Schwartz, 2013). Little definitive information is available about the potential bioavailability and bioaccumulation of contaminants from plastic debris in the Great Lakes, however. The possible transfer of plastic-associated toxins to humans through consumption of freshwater organisms remains to be demonstrated.

ECONOMIC IMPACTS

With over 60 million people visiting the 98 state parks, 39 provincial parks, and 12 national parks bordering the Great Lakes each year (US EPA, 2012), and millions more visiting public beaches, the indirect costs of plastic debris on the tourism industry are undoubtedly significant, but have yet to be quantified. The indirect costs of plastic debris on other industrial activities, for example commercial fisheries, are also unknown. In addition, litter, including plastic debris, may negatively affect the quality of life in coastal communities and depress coastal property values. Thus, beach cleanup activities enjoy broad public support and mobilize large volunteer groups. According to the Alliance for the Great Lakes, in 2012, the monetary value of the hours spent by volunteers cleaning up beaches around the Great Lakes represented over US $250,000 (Alliance for the Great Lakes, 2013).

We are not aware of any study estimating the direct cost of combating and curtailing plastic debris pollution in the Great Lakes region. However, a study prepared for the US Environmental Protection Agency analyzed the direct cost of marine litter management in the states of California, Oregon and Washington (Stickel et al., 2012).

Data received from a random sample of West Coast communities in those states revealed that approximately US $13 per inhabitant is spent annually on beach and waterway cleanup, street sweeping, installation of storm-water capture devices, storm drain cleaning and maintenance, manual cleanup of litter, and public anti-littering campaigns. If we apply the same per capita $13 to the 36 million people who live within 50 kilometers of the shoreline of a Great Lake (estimated using 2012 LandScan data), the direct cost of combating plastic debris pollution in the Great Lakes region would amount to $468,000,000 annually.

A study conducted by Industrial Economics Inc. for the US National Oceanic and Atmospheric Administration (NOAA) Marine Debris Division (Leggett et al., 2014) examined the direct cost of littered beaches for residents of Orange County, California. The study found that the residents consider beach cleanliness a top criterion for deciding which beaches to visit: they are willing to travel further to cleaner beaches at a cost. A travel cost model calculated that Orange County residents would save a combined US $148 million annually if all nearby beaches were completely litter-free. By extrapolation, we expect that littered beaches in the Great Lakes region similarly cost residents millions of dollars annually due to increased travel expenditures.

The growing number of media reports on plastics ingested by commercial fish, crustaceans, and shellfish could potentially lead to reduced consumer spending on these food items. The uncertainties surrounding the abundance, ecological effects, and human health risks of plastics could ultimately have a greater economic impact on the fishing industry in the Great Lakes than the revenue losses resulting from damage to vessels by plastic debris or lower catches due to the presence of litter in hauls. However, further research will be needed to determine to what extent the fishing industry in the Great Lakes region could be affected by plastic pollution.

POLICY

Public awareness of plastic debris in the Great Lakes is on the rise as a result of new research findings, increased media exposure, NGO-led cleanup activities, and governmental initiatives, including NOAA's Marine Debris Program and Canada's National Marine Debris Surveillance Program (the latter ran between 1998 and 2002). NOAA's Great Lakes Land-Based Marine Debris Action Plan (2014-2019) aims to identify knowledge gaps, guide relevant policy and management decisions, and reduce debris input by educating and engaging stakeholders, as well as to lessen plastic debris impacts through tracking and removal efforts. The plan represents a collaborative effort between federal agencies, states, tribal nations, researchers, business leaders, and NGOs in the Great Lakes region (NOAA Marine Debris Program, 2014).

At the municipal level, the Great Lakes and St. Lawrence Cities Initiative (GLSLCI), a binational coalition of mayors and municipal officials that seeks to advance the protection and restoration of the Great Lakes and the St. Lawrence River, has identified microplastics as a key environmental threat. GLSLCI has adopted a resolution that calls on industry to phase out microbeads from consumer products, including personal care products. The resolution further calls on provincial, state, and federal governments to establish legislation banning the use of microbeads in consumer products (Great Lakes & St. Lawerence Cities Initiative, 2014). Several companies have already committed to voluntarily phase out microbeads from their products over the next several years (Plastic Soup Foundation et al., 2014). A ban on the manufacture and sale of personal care products containing microbeads has become law in Illinois. Similar legislation has been introduced in New York, California, Michigan, and Ohio (but not yet adopted), and is being considered in Wisconsin, Pennsylvania, Ontario, and Québec. Minnesota has passed a bill requiring a study on the issue (Olga Lyandres and Jared Teutsch, Alliance for the Great Lakes, 2014, pers. comm.).

Dumping of plastics in the oceans has been illegal since 1988, following the adoption of Annex V of the International Convention for the Prevention of Pollution from Ships (MARPOL) by the International Maritime Organization. Similarly, Annex V of the Great Lakes Water Quality Agreement (GLWQA) prohibits vessels from discharging garbage into the Lakes, including all plastic waste. However, the GLWQA does not regulate the input of plastic debris into the Great Lakes from land-based sources. A number of municipalities bordering the Great Lakes have enacted by-laws prohibiting littering on beaches, while some municipalities have also banned smoking on public beaches and charge fees for plastic grocery bags. These initiatives may help limit plastic input to the Great Lakes. Nonetheless, it may require incorporation of new regulations in GLWQA in order to explicitly address the issue of pollution by microplastics and microscopic plastic debris.

SURVEY METHODS

At present, open-water and shoreline surveys designed to assess the distributions of plastic debris in oceans and lakes are time-consuming, costly, and provide limited areal coverage and temporal resolution. Remote sensing and field-deployable sensors have the potential to overcome these limitations. However, to our knowledge, these approaches have not been widely deployed for the detection and tracking of plastic debris in aquatic environments.

Plastics have characteristic absorbance and reflectance spectra in the near-infrared (NIR) domain (~750-2500 nm) (Masoumi et al., 2012). Thus, in principle, NIR spectrometers could be used to detect and identify plastics by matching spectra obtained on environmental samples to those of reference materials. NIR spectroscopy is currently used in related applications including the sorting of plastic debris in recycling facilities (Hopewell et al., 2009). In collaboration with P&P Optica (http://www.ppo.ca/), a spectrometer manufacturer, the authors of this study successfully used NIR reflectance spectrometry (spectral range 890-2500 nm) to detect common plastic debris in beach sand (unpublished results).

A significant limitation for the direct detection of plastics in aquatic systems with NIR spectroscopy is the strong absorption of infrared radiation by water (Mace, 2012). Raman spectroscopy offers an alternative method to identify plastics in environmental samples (Allen et al., 1999; Tsuchida et al., 2009). As Raman scattering from water is weak, Raman spectroscopy could be used to identify plastics in aqueous samples and, ultimately, directly in the aquatic environment. One avenue that deserves to be explored further is the development of portable and field-deployable Raman spectrometers. One could envision applications ranging from fast shipboard screening of samples to the deployment on buoys or remote controlled vehicles.

Remote sensing is another tool that could prove useful for monitoring plastic debris in marine and lentic settings. Identification of larger plastic debris (>1 m) in terrestrial environments using hyperspectral imaging sensors mounted on unmanned aerial vehicles (UAVs) has been demonstrated (Hörig et al., 2001). Most hyperspectral sensors operate in the NIR spectrum and have sufficient spectral resolution to identify plastics in the same way that NIR spectrometers do. However, the small size of much of the plastic debris found in lakes and oceans presents a major obstacle, as the pixel area, that is, the spatial resolution of the sensor, is limited in part by the sensor’s proximity to the target. Furthermore, when imaging aquatic environments, hyperspectral sensors are subjected to lower reflected light levels, high absorption of NIR energy, and possible interference from wave action and sun glint, all of which restrict the detection of plastic debris (Veenstra and Churnside, 2012).

Remote sensing could be used indirectly to determine where plastic debris is likely to accumulate. In the ocean, plastic debris has been shown to concentrate in anticyclonic eddies, along frontal boundaries and in other areas of surface convergence (Eriksen et al., 2013b; Howell et al., 2012; Pichel et al., 2007). These convergent areas can be assessed from satellite imagery and hydrodynamic models that forecast the speed and direction of surface flow. Pichel et al. (2007), for example, derived a Debris Estimated Likelihood Index (DELI) for a section of the ocean within the North Pacific Subtropical Convergence Zone, based on sea surface temperature and chlorophyll absorption data obtained from multispectral satellite imagery. It would therefore be worthwhile to assess the relationship between surface flow convergence and plastics abundance in the surface waters of the Great Lakes: the development of tools similar to Pichel et al.’s DELI would greatly help in identifying open-water and coastline hot spots prone to debris accumulation. This information would provide welcome support to monitoring programs, scientific research on plastics, and cleanup efforts in the Great Lakes.

CONCLUDING REMARKS

The review of the available data and information suggests that plastic debris represents a major environmental challenge for the Great Lakes. However, many uncertainties surround the nature and magnitudes of the ecosystem impacts of pollution by plastics in the Great Lakes, primarily because of a lack of targeted scientific research into the sources, transport, breakdown, and ecological plus human health implications of plastic debris. The following are some of the essential research questions that require attention.

1. What are the annual inputs of plastic debris to each of the Great Lakes basins? How do the inputs vary throughout the year? What is the breakdown of the inputs in terms of size and composition of the plastic debris?
2. What are the rates and mechanisms at which different types of plastic debris degrade in the Great Lakes? What proportion of plastics is ultimately preserved in bottom sediments?
3. How widely are microplastics and microscopic plastic debris distributed in the Great Lakes? What are their sources? How does their environmental fate differ from that of larger plastic debris?
4. How much plastic debris accumulates along the Great Lakes' shoreline, both in beach and non-beach environments (e.g., wetlands, harbors, rocky shores)? Do plastics interact differently with organisms in the different shoreline habitats?

5. What is the extent of bioaccumulation of plastics and associated contaminants along the food webs of the Great Lakes? What are the ecotoxicological consequences? Are there potential risks to human health?

Answers to these and other related questions are crucial to assess the current state of pollution of the Great Lakes by plastics, but also to develop a predictive understanding of the fate of plastic debris within the lakes. The latter is needed to interpret the distributions of plastics in the different environmental compartments of the Great Lakes, and to develop the necessary tools to forecast the effectiveness of proposed actions, regulations, and policies.

Table 1.1: Densities and common uses of plastics that have been identified, or are highly likely to be present in the Great Lakes (density values at room temperature, compiled from Teegarden (2004), and common uses compiled from various sources). In principle, plastics with densities greater than 1 g/cm^3 should sink in water.

Plastic Type	**Abbreviation**	**Density (g/cm^3)**	**Common Uses**
Expanded Polystyrene	EPS	0.01-0.04[a]	Packaging foam, foam cups and plates
Polypropylene	PP	0.85-0.92	Auto parts, industrial fibers, food containers, dishware
Low-Density Polyethylene	LDPE	0.89-0.93	Plastic bags, 6-pack holders, dispensing bottles, tubing
High-Density Polyethylene	HDPE	0.94-0.98	Detergent bottles, milk jugs, grocery bags, recycling bins, playground equipment
Acrylonitrile-Butadiene-Styrene	ABS	1.04-1.06	Electronic equipment casing, pipes
Polystyrene	PS	1.04-1.08	Food containers, trays, cutlery
Polyamide (Nylon)	PA	1.13-1.16	Fibers, toothbrush bristles, fishing line
Polymethyl Methacrylate (Acrylic)	PMMA	1.16-1.20	Optical lenses, paint, shatterproof windows
Polycarbonate	PC	1.20-1.22	CD and DVD disks
Cellulose Acetate	CA	1.30[b]	Cigarette filters
Polyethylene Terephthalate (Polyester)	PET	1.38-1.41	Fibers, textiles, soft drink and water bottles, strapping, camera film
Polyvinyl Chloride	PVC	1.38-1.41	Pipes, fencing, shower curtains, flooring
Polytetrafluoroethylene	PTFE	2.10-2.30	Wires, cables, bearings, gears

Alternate sources:

[a] Winterling and Sonntag (2011)

[b] http://www.goodfellow.com/E/Cellulose-Acetate.html (accessed 11.17.14)

Table 1.2: Summary of plastic debris concentrations in the Great Lakes region reported in the literature or calculated from beach cleanup data. Note the differences of units in which the plastic debris concentrations are reported.

	River				**Lake**										
Zone	Riparian		Benthic		Open-Water			Shoreline							
Reference(s)	Hoellein et al., 2014[1]		Hoellein et al., 2014[1]		Eriksen et al., 2013a[2]			Zbyszewski and Corcoran, 2011[3]; Zbyszewski et al., 2014[3]			Hoellein et al., 2014[1]		AAB & GCSC 2012 data[4] AAB/GCSC		
	# Sites	PDC	# Sites	PDC	# Tows	PDC		# Sites	PDC		# Sites	PDC	# Sites	PDC**	
		Avg.		Avg.		Range	Avg.		Range	Avg.		Avg.		Range	Avg.
Lake Superior	0	-	0	-	5	0.0013-0.0126	0.0054	0	-	-	0	-	19/10	0.0101-7.2015/ 0.0155-1.0540	0.6591/ 0.4992
Lake Michigan	0	-	0	-	0	-	-	0	-	-	3	0.0005	223/0	0-23.0972/ -	1.1053/ -
Chicago River*	3	0.0385	3	0.0180	0	-	-	0	-	-	0	-	0/0	-/-	-/-
Lake Huron	0	-	0	-	8	0-0.0065	0.0028	7	0-34	5.4300	0	-	12/27	0.0250-3.8960/ 0.0942-5.4280	1.1162/ 0.8601
Lake St. Clair*	0	-	0	-	0	-	-	9	0.1800-8.3800	1.7256	0	-	0/0	-/-	-/-
Lake Erie	0	-	0	-	8	0.0047-0.4663	0.1055	10	0.3600-3.7000	1.5410	0	-	32/10	0.0460-3.5240/ 0.0185-0.8820	1.0617/ 0.2786
Lake Ontario	0	-	0	-	0	-	-	0	-	-	0	-	5/108	0.1160-1.2385/ 0.0163-14.2740	0.6941/ 0.9991

* Not a Great Lake

PDC = plastic debris concentration (items/m^2), ** (items/m)

Smallest size of debris counted: [1] 100 mm, [2] 0.355 mm, [3] ~1 mm, [4] ~100 m

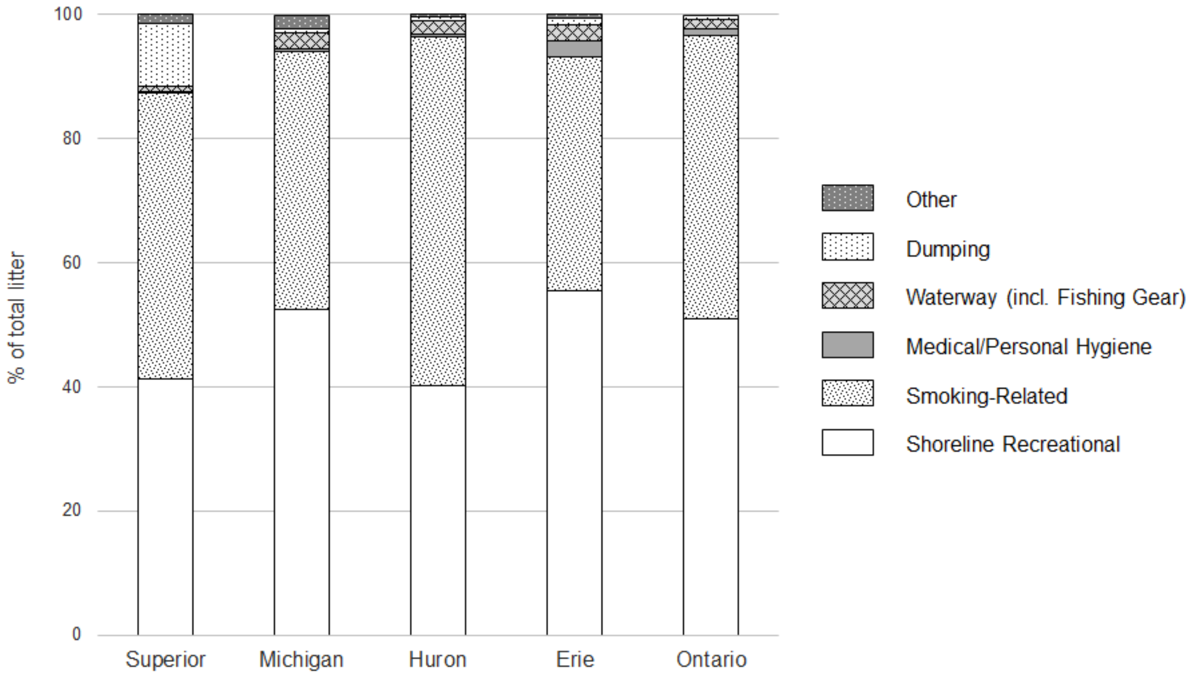

Figure 1.1: Percentages of litter items in the Great Lakes grouped according to activity on an item-by-item basis (see Appendix A, Table 1 for details). All data are from Adopt-a-Beach^{TM} and Great Canadian Shoreline Cleanup for the year 2012.

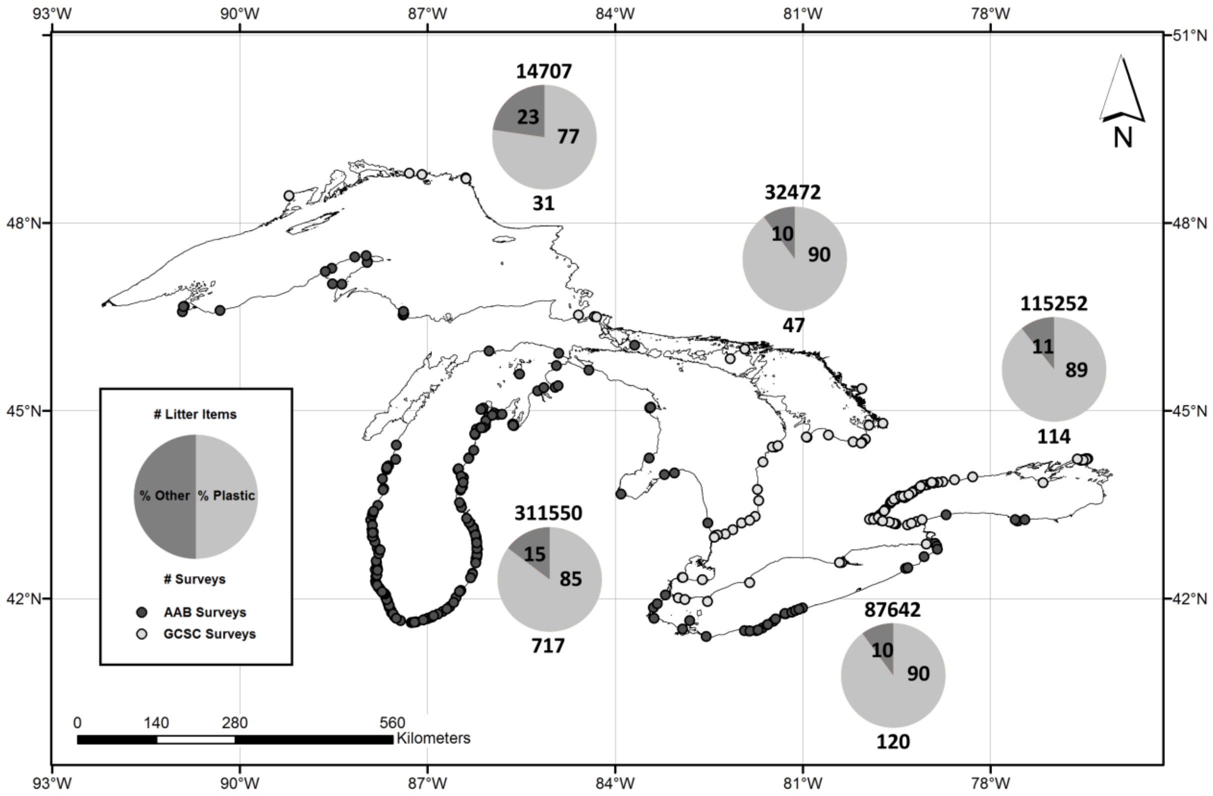

Figure 1.2: Great Lakes beach cleanup surveys. The figure shows the locations of beach cleanup surveys, the number of surveys conducted, and the percentages of anthropogenic debris made of plastic. The data used in the figure were collected by Adopt-a-Beach^{TM} (AAB) and Great Canadian Shoreline Cleanup (GCSC) volunteers in 2012. The closed dots identify AAB cleanup locations, the open dots GCSC cleanup location.

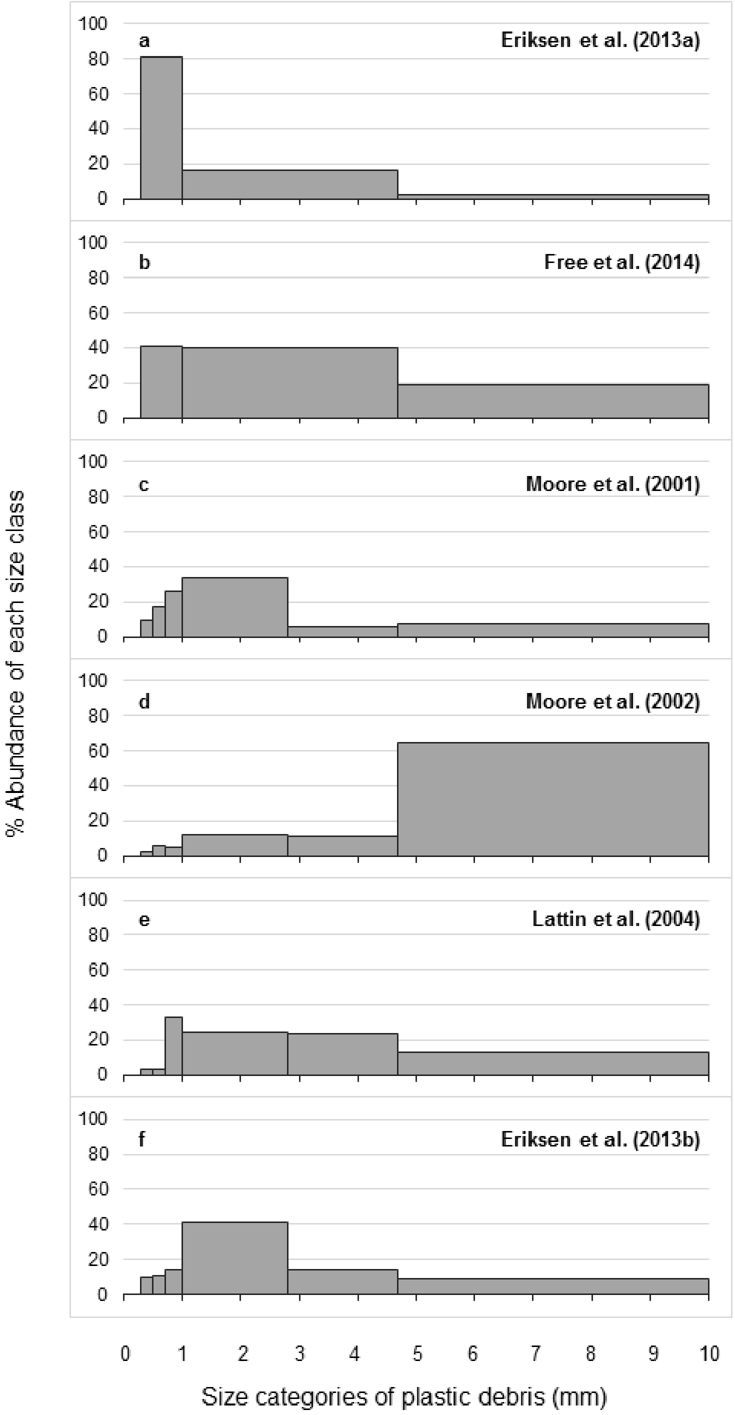

Figure 1.3: Comparison of pelagic microplastic debris size distributions. Freshwater studies: (a) Laurentian Great Lakes, (b) Lake Hovsgol, Mongolia. Marine studies: (c) North Pacific Central Gyre, (d)-(e) Southern California Coastal Waters, (f) South Pacific Subtropical Gyre. The 10 mm upper limit is arbitrary; the studies cited reported >4.75 mm as their top size class. Note that the results shown are from studies that reported size data over a range comparable to Eriksen et al. (2013a).

CHAPTER 2: CLASSIFICATION, DISTRIBUTION AND ENVIRONMENTAL DRIVERS OF PLASTIC DEBRIS IN THE LAURENTIAN GREAT LAKES WITH CITIZEN SCIENCE DATA

INTRODUCTION

Plastic debris are pervasive and persistent pollutants in the world's aquatic environments, with adverse economic, ecological, human health, and aesthetic impacts. They are ubiquitous in the Laurentian Great Lakes (Driedger et al., 2015), which together, form the world's largest continuous body of fresh water and are a vital resource to the region's people and environment (Queen's Printer for Ontario, 2015). Yet, only a handful of studies have presented quantitative data about plastic debris, mainly focusing on its abundance in rivers, surface waters, and along shorelines.

Eriksen et al. (2013a) reported high concentrations of microplastic debris in the surface waters of Lakes Superior, Huron, and Erie while Zbyszewski et al. (2011; 2014) found high concentrations of micro- and macroplastic debris along shorelines of Lakes Huron, St. Clair, and Erie. Similarly, Corcoran et al. (2015) reported microplastics in bottom sediments of Lake Ontario, and Hoellein et al. (2014) observed plastics along shorelines of Lake Michigan and the Chicago River. While these studies present evidence of plastic debris in specific areas of the Great Lakes, they only provide a limited temporal and spatial coverage of observations, leaving uncertainty about the quantity and distribution of plastics at other times and in other areas of the Great Lakes. A more frequent and geographically-expansive record of plastic debris concentrations could help identify sources and distribution patterns, which could benefit cleanup efforts and help devise best management strategies to protect the Great Lakes ecosystem.

Beach cleanup programs offer an alternate source of data on the types and quantity of plastic debris along shorelines. Some programs have volunteers (citizen scientists) collect litter and fill out data forms to report their findings. In many cases, they offer a larger number and higher temporal and spatial coverage of observations than scientific studies, providing a more extensive record of plastic debris, at least along shorelines.

The Alliance for the Great Lakes, based in Chicago, Illinois, runs a beach cleanup program called Adopt-a-Beach (AAB). It was the first program in the Great Lakes region to partner with the Ocean Conservancy's International Coastal Cleanup initiative (the world's largest volunteer effort to clean beaches and waterways) and makes litter data generated by its volunteers available to view on its website (http://www.greatlakes.org/ADOPTABEACH). Hoellein et al. (2015) used AAB data to assess the abundance and environmental drivers of litter on 5 Lake Michigan beaches. However, the vast majority of AAB cleanup sites are located on the American side of the Great Lakes, leaving the Canadian side unrepresented.

Fortunately, a similar program exists in Canada; the Great Canadian Shoreline Cleanup (GCSC), directed by the Vancouver Aquarium and World Wildlife Fund. For several years, it has had thousands of volunteers clean hundreds of shoreline locations across Canada, including a large number in the Great Lakes region. Volunteers record the quantity and types of litter (see Appendix A, Table 2 for litter types) collected and send it back to the GCSC which then summarizes their data and publishes it on its website (http://www.shorelinecleanup.ca).

In this chapter, survey data generated by GCSC volunteers in Ontario from 2013 and 2014 are used for the first time to 1) classify litter along Canadian shorelines of the Great Lakes, 2) assess the abundance and distribution of plastic debris, and 3) identify sources and drivers of plastic accumulation on shorelines.

METHODS

STUDY SITES AND REGIONS

All cleanup locations (study sites) were located on the Canadian side of the Great Lakes in the province of Ontario. Ontario borders four out of five Great Lakes (Superior, Huron, Erie, and Ontario) and has a total coastline of 7600 km. In between Lakes Huron and Erie is the smaller Lake St. Clair, on which several study sites were also located. Ontario is Canada's most populous province, with approximately 13.5 million inhabitants. Southern Ontario, the region extending from Windsor to Ottawa, contains over 90% of that population, and is also where the majority of cleanup events took place.

A significant difference between AAB and GCSC programs is that GCSC volunteers performed cleanups at 'inland' locations in addition to 'lake shoreline' locations. Lake shoreline sites were defined here as those within 500 m of a Great Lake's shoreline, including Lake St. Clair. All remaining sites were classified as inland sites, i.e. those along rivers connecting the Great Lakes and further inland along smaller lakes, rivers, and wetlands (Figure 2.1).

Another difference between AAB and GCSC data is the quantity of cleanups conducted per site. AAB sites typically had multiple cleanups and corresponding survey records (occasionally more than 50) per year whereas GCSC sites had at most two records per year. Thus, analysis of plastic debris at individual GCSC sites was not feasible – one or two survey records would not provide enough data to accurately assess plastic debris abundance and characteristics at an individual site relative to other sites. To make up for the low number of survey records per site, analysis of GCSC data was conducted at larger geographical scales. This was achieved by averaging survey results from sites within census divisions, lake shoreline regions, and major lake regions. Many of these larger regions had multiple cleanup sites falling within their boundaries allowing for a more robust comparison of litter composition and densities at a regional scale instead of site scale.

A census division in Canada is a provincially legislated area that may correspond to a county, regional municipality or regional district. It is roughly comparable in geographic scope to an American county. Until 2006, Ontario had 49 census divisions; it now has 50 with the recent splitting of Haldimand-Norfolk Census Division. However, the census division data used here is pre-2006 and thus has 49 (Statistics Canada, 2001) (Figure 2.2). Of the 49 census divisions, 39 had at least 1 survey record and 23 had at least 5 when combining 2013 and 2014 datasets.

Lake shoreline regions were created for the purpose of this study. 5 lake shoreline regions were formed by combining census divisions of similar location (i.e. region of the Great Lakes) and population density (Figure 2.3). These regions provided a greater spatial coverage than census divisions but lower resolution assessment of plastic debris dynamics along Great Lakes shorelines (only lake shoreline site data were analyzed within these regions). Similarly, major lake regions accounted for only those lake shoreline records from sites adjacent to a Great Lake or Lake St. Clair.

In 2013, GCSC altered their data form; litter categories were restructured and fields for tiny litter (<2.5 cm in diameter) including plastic, glass, and foam pieces were added. Furthermore, GCSC data from prior years did not report date information (i.e. the date on which each cleanup occurred), prohibiting any temporal analysis of litter composition or abundance. Therefore, only data from 2013 and 2014 were analyzed in this chapter.

LITTER COLLECTION

Collection of litter by GCSC is volunteer driven and thus not completed on a regular schedule. As a result, the corresponding dataset is variable among sites, seasons, and years. Cleanups are conducted by a team of volunteers and a site coordinator. During a cleanup, volunteers spread out across a beach or shoreline area, collecting and tallying all visible litter that they can find. In this study, ‘large’ litter represents all litter 2.5 cm or greater in diameter (i.e. including most intact or semi-intact items) while ‘tiny’ litter represents all litter less than 2.5 cm. The minimum size of litter collected is roughly 0.5 cm, so few if any microplastic debris are represented in the dataset.

DATA TREATMENT AND ANALYSIS

All GCSC survey records for 2013 and 2014 were provided by GCSC staff. Each survey record contained site details (e.g. name, date, coordinates), the count of each litter type, number of volunteers, length of shoreline covered (km), and approximate weight of the litter collected (kg). A range of treatments were applied to the data to gather information about the composition, distribution, and drivers of litter. The records from each year were first separated into two files: one containing lake shoreline site records and the other containing inland site records. A small number of the lake shoreline and inland sites had to have their coordinate position shifted slightly (by manual edit) to ensure that they were located within a census division and lake shoreline region.

CLASSIFICATION

Within each file, large litter items were grouped by categories first developed by the Ocean Conservancy to quantify the activities that contribute to litter accumulation (Appendix A, Table 2). The categories are: shoreline and recreational, smoking-related, medical/personal hygiene, waterway, dumping, and other. This classification scheme provides some context about the potential sources of litter and whether those sources are land-based or water-based. These categories were used in Chapter 1 to classify volunteer survey data and were also used by Hoellein et al. (2015), facilitating comparison to this chapter's results.

Large and tiny litter items were classified as plastic or non-plastic (Appendix B, Table 2). The percentages of large and tiny litter made of plastic at each site were calculated. The mean percentage of litter made of plastic was then determined for each study region; for census divisions and lake shoreline regions, the Calculate Point Statistics toolbox for ArcGIS (Broad, 2014) was used to calculate the average percentage of litter made of plastic from records falling within each study region (Appendix C, Tables 1 and 2).

DISTRIBUTION

The temporal distribution of survey records for lake shoreline sites was analyzed for both 2013 and 2014. The number of surveys per day were totaled and graphed beside the mean large plastic density of sites per day. Large plastic density (LPD) and tiny plastic density (TPD) were calculated for each survey record, i.e. the total number of large and tiny plastic items collected per km of shoreline surveyed, respectively. The Calculate Point Statistics toolbox was used to calculate the mean, minimum, and maximum litter density values located with each study region. The tool's script was also edited to return the standard deviation of density values to assess the variability of litter densities within each region. Finally, relative standard deviation was calculated to assess the variability of density values between regions.

Maps of mean LPD per census division and lake shoreline region were produced for 2013 and 2014 to help visually assess the variation in mean LPD across those regions. A variety of other relationships were analyzed to explore the distribution of plastic debris. A high concentration of inland sites were surveyed in the Greater Toronto Area (GTA) in both 2013 and 2014. As the majority of these sites were located along rivers flowing to Lake Ontario, those closer to Lake Ontario were suspected to have greater plastic densities as a result of upstream litter being carried downstream during periods of high river flow (e.g. after a large precipitation event). For each of these inland surveys, the distance to shoreline was calculated to determine if a linear relationship existed between a site's proximity to Lake Ontario and LPD or TPD. Given the availability of both lake shoreline and inland survey records, mean LPDs from both groups were compared per census division using linear regression analysis. The relationship between LPD and TPD at both lake shoreline and inland sites was also considered using linear regression analysis. For all regression analyses involving study regions, regions required a minimum of 5 surveys records to be included.

ENVIRONMENTAL DRIVERS

Simple linear regression was used to quantify the relationships between mean LDP of study regions and area descriptors including population density, catchment size, and coastal stress. Plastic debris density values required log transformation for population density and catchment size regressions. A p value of <= 0.05 was used to indicate a significant relationship.

Population density was derived from 2012 LandScan data, a raster dataset that reports population at approximately 1 km resolution. The population density of each census division and lake shoreline region was calculated. A GIS file of Ontario watershed regions was used to delineate catchment size (Ontario Ministry of Natural Resources, 2010). Finally, coastal stress was a relative index, in raster format, of Great Lakes shoreline stressors including developed land, coastal road density, coastal mining, thermoelectric power plants, and coastal recreational use, developed by Allan et al. (2013) and provided by staff at the Great Lakes Environmental Assessment and Mapping project (GLEAM; http;//greatlakesmapping.org). Mean coastal stress per census division and lake shoreline region was calculated.

RESULTS

Over the course of 2013 and 2014, 368 lake shoreline and 511 inland surveys were conducted by a total of 19,183 GCSC volunteers. 541,980 large litter pieces and 113,655 tiny litter pieces were collected and removed from the environment (Tables 2.1 and 2.2).

CLASSIFICATION

In both 2013 and 2014, over 80% of large litter found at lake shoreline sites were classified as either recreation and shoreline or smoking-related (Figure 2.4). In both years, the 5 most abundant litter items were cigarette butts, food wrappers, bottle caps, beverage bottles, and straws/stirrers, all of which are plastic. Furthermore, roughly 80% of large and tiny litter were comprised of plastic. Large litter in Lake Superior was least comprised of plastic (74 and 71% in 2013 and 2014 respectively) while large litter in Lake St. Clair was most comprised of plastic (87 and 90% in 2013 and 2014 respectively) (Table 2.1). The mean percentages of large and tiny litter comprised of plastic per census division and lake shoreline region using lake shoreline site data are reported in Appendix C, Tables 1 and 2.

More than 90% of large litter found at inland sites were related to either shoreline and recreational or smoking activities (Figure 2.5). In both years, the 5 most abundant litter items were cigarette butts, food wrappers, beverage bottles, bottle caps and beverage cans, all of which are plastic except for beverage cans which are metal. 83 and 82% of large litter were comprised of plastic in 2013 and 2014 respectively while 76 and 73% of tiny litter were comprised of plastic in 2013 and 2014 respectively. The mean percentages of large and tiny litter comprised of plastic per census division using inland site data are reported in Appendix C, Tables 1.

DISTRIBUTION

The temporal distribution of cleanups in 2013 and 2014 were quite different. In 2013, the vast majority of cleanups took place in September (98% of lake shoreline cleanups and 99% of inland cleanups) and the rest in October. Most were concentrated around Coastal Cleanup Day which occurred on September 21, 2013. In 2014, cleanups were more evenly spread throughout the year. Of lake shoreline surveys, 31.5% occurred in the spring, 37% occurred in the summer and 31.5% occurred in the fall; no surveys were conducted in the winter months of December, January, or February. The mean LPDs from fall and spring cleanups at lake shoreline sites (742 and 685 pieces/km respectively) were over twice the mean LPD from summer cleanups at lake shoreline sites (307 pieces/km) (Figure 2.6). Seasonal variation in mean TPD at lake shoreline sites was almost non-existent, apart from three outlier values in the spring.

The spatial distribution of study regions according to mean LPD was found to be highly variable. At the census division level, use of lake shoreline site data and inland site data produced different results. With the lake shoreline site data, in both 2013 and 2014, census divisions in the GTA region were found to have relatively high mean LPDs, especially the more populous Toronto (1030 and 958 pieces/km in 2013 and 2014 respectively) and Peel (1128 and 769 pieces/km in 2013 and 2014 respectively) census divisions (Figure 2.7). Comparatively, the less populated Huron and Frontenac census divisions, outside of the GTA, had relatively low mean LPDs in both 2013 and 2014 (see Appendix D, Tables 1 and 4 for a full list of mean LPDs per census division using lake shoreline site data in 2013 and 2014 respectively).

With inland site data, in both years, census divisions with the highest mean LPDs still tended to be around the GTA, but the spatial distribution of census divisions according to mean LPD seemed more variable than that with the lake shoreline site data (Figure 2.8). However, those census divisions further away from a Great Lake, including land-locked census divisions, tended to have lower mean LPD in both years, with one notable exception being Waterloo census division in 2014 with a mean LPD of 646 pieces/km. At the lake shoreline region level, similar results were found. The highly populated Golden Horseshoe region had a relatively high mean LPD both years (795 and 844 pieces/km in 2013 and 2014 respectively) while lower populated regions tended to have lower mean LPDs (Figure 2.9).

For inland sites in the GTA region, their proximity to Lake Ontario was found to be unrelated to either their LPD or TPD. A positive trend was found between census division mean LPD using lake shoreline site data and that using inland site data (Figure 2.10). However, not enough data was available to establish significance of the relationship. In both 2013 and 2014, a significant, positive correlation was found between LPD and TPD at both lake shoreline and inland sites (Figure 2.11). Also in both years, the average number of tiny plastic pieces collected per site was found to trend upward in each downstream lake, i.e. least in Superior and most in Ontario (Figure 2.12).

ENVIRONMENTAL DRIVERS

A significant, positive relationship was found between census division population density and census division mean LPD using 2014 lake shoreline site data ($R^2 = 0.541$, $p = 0.001$) (Figure 2.13). However, the same relationship was not found with 2013 data, although a positive trend was apparent ($R^2 = 0.277$, $p = 0.118$) (Figure 2.14), nor with 2013 or 2014 inland site data ($R^2 = 0.067$, $p = 0.392$ and $R^2 = 0.061$, $p = 0.376$ for 2013 and 2014 respectively) (Figure 2.15). Lake shoreline region population density was also found to be unrelated to lake shoreline region mean LPD ($R^2 = 0.459$, $p = 0.209$ and $R^2 = 0.135$, $p = 0.543$ in 2013 and 2014 respectively).

For both 2013 and 2014, no correlation was found between a watershed's catchment size and its mean LPD using lake shoreline site data (R^2 = 0.151, p = 0.342 and R^2 = 0.019, p = 0.765 respectively) or inland site data (R^2 = 0.184, p = 0.143 and R^2 = 0.005, p = 0.824 respectively). A significant, positive relationship was found between census division mean coastal stress and census division mean LPD using 2013 lake shoreline site data (R^2 = 0.577, p = 0.011). However, the same correlation was not found with 2014 data (R^2 = 0.250, p = 0.118) nor at the lake shoreline region level (R^2 = 0.004, p = 0.923 and R^2 = 0.530, p = 0.163 for 2013 and 2014 respectively).

DISCUSSION

PRIMARY SOURCE OF PLASTIC DEBRIS ON SHORELINES

Beach and shoreline visitors were undoubtedly the largest contributor of plastic litter to the study sites. Shoreline and recreational litter was mostly food-related litter, e.g. food wrappers and plastic bottles, and together with smoking-related litter, made up roughly 80% of the total litter collected on average. While it is possible for litter items in these categories to come from offshore onto beaches, the vast majority is more likely to have come from land-based, in situ anthropogenic activity. Furthermore, litter from offshore sources including waterway activities, fishing, dumping, and sewage made up only a minor component of overall litter composition, suggesting that only a small amount of litter was making its way to shore. Beach and shoreline visitors have also been found to affect plastic litter density in other areas of the Great Lakes. Hoellein et al. (2015) determined that litter at their study sites along Lakes Michigan's shoreline originated from activities occurring directly on or adjacent to the sites and Zbyszewski et al. (2014) found the highest litter densities along Lake Erie's shoreline at sites known to receive millions of visitors per year. Plastic debris on shorelines may also be a result of litter being carried by wind across land or litter being transported down waterways from further inland (Zbyszewski et al., 2014).

Of tiny litter found, plastic and foam pieces made up over two-thirds of the total at lake shoreline sites in all Great Lakes except Superior where glass was more prevalent. At inland sites, glass pieces were more abundant than plastic or foam pieces in 2014. The fact that inland sites are mostly located along rivers might explain the higher proportion of glass litter; plastic and foam litter are both lighter and more buoyant than glass and thus more likely to be transported downstream instead of remaining in place. The increase in average abundance of tiny plastic debris collected at sites on downstream lakes (Lake Erie and Lake Ontario) relative to upstream lakes (Lake Superior) is similar to Eriksen et al. (2013a) finding higher densities of pelagic microplastics in downstream lakes. The significant, positive relationship between densities of large and tiny plastics at both lake shoreline and inland sites provides strong evidence that the sources and drivers of large and tiny plastic litter are related. One explanation could be that a large portion of the tiny plastics are simply broken-down fragments of larger litter items, and thus, their primary source would be food and smoking-related litter from beach visitors.

While the LPDs at cleanup sites will be inherently variable, even at small temporal or spatial scales, mean LPDs of the study regions were intended to provide a general indication of plastic litter abundance within these regions to compare litter abundance between regions and identify which areas of the Great Lakes may be most prone to plastic litter accumulation. In general, those census divisions and lake shoreline areas with higher population densities tended to have higher mean LPDs. This supports the notion that beach and shoreline visitors are the primary contributor of plastic debris along shorelines. In study regions with a high relative standard deviation of LPD values, a greater variability of LPDs at sites within those regions was observed. This may be indicative of beach grooming or a higher variability in the number of people visiting sites in those regions. However, without a strict sampling protocol, a high relative standard deviation of LPD values could be a result of countless other factors.

Minimum, maximum, and mean LPDs per major lake region are in the same range as those reported in Chapter 1 (AAB and GCSC data from 2012) (Table 2.3). Acknowledging the fact that only one year of AAB is presented and therefore may not be representative of normal litter densities, the mean litter density of Lake Erie's Canadian shoreline (according to GCSC data) was 2-3 times less than that of Lake Erie's American shoreline (according to AAB data). This result seems reasonable, given that Lake Erie's American side is much more densely populated than its Canadian side. Similarly, the mean litter density of Lake Ontario's Canadian side was higher than that of its American side (which is less densely populated). Lake Ontario's Canadian side also had maximum LPDs that were over one order of magnitude greater than LPDs on its American side.

The strong, positive correlation between census division population density and census division mean LPD using 2014 lake shoreline site data provides further evidence that local activities are a dominant source of plastic debris on shorelines. While population density is only a proxy for beach or shoreline visitor abundance, it stands to reason that the more people living in close proximity to a shoreline, the more people will visit it and deposit trash. Some exceptions may include if the shoreline area acts as a tourist destination due a special feature or designation (e.g. national park), or if there are contextual features (e.g. a nearby industrial facility) that make the shoreline area less desirable to visit. Census division population density did not seem to affect census division mean LPD using inland site data. This may be because the visitation of inland sites is more variable and likely lower than lake shoreline sites in general. Yet, at the census division level, a moderate correlation was found between mean LPD using inland site data and mean LPD using lake shoreline site data. So, plastic debris at lake shoreline and inland sites may still share common drivers.

Catchment size was not found to affect the mean LPD per watershed using lake shoreline site data but coastal stress was found to be positively correlated with census division mean LPD using 2013 lake shoreline site data. Given that the coastal stress dataset was a measure of urban land use (e.g. coastal road density) and coastal recreational use, this result is not surprising; it further supports the notion that the primary driver of plastic debris abundance on shorelines is in situ anthropogenic activity and also suggests that industrial activity may be an additional source of plastic debris along shorelines. Similarly, Zbyszewski and Corcoran (2011) found that industrial activity affected the distribution of plastic debris along the shorelines of Lake Huron.

Other drivers of plastic litter density on Great Lakes shorelines include lake currents (Zbyszewski and Corcoran, 2011), storm events, and the prevalence of waterway activities including shipping and fishing (Zbyszewski et al., 2014). Appropriate sources of data for comparing these drivers to GCSC’s dataset were not found for use in this study. Hoellein et al. (2015) also compared litter density to impervious surface cover, Flickr score (i.e. which was used as a proxy for beach visitation), GDP of tourism and recreation by county, bacteria counts including *E. coli* and coliform, and precipitation patterns, but found that none of these yielded a significant correlation.

One additional ‘driver’ of plastic litter density on shorelines is beach grooming. Beach grooming is the manual removal of litter via mechanical grooming equipment or human labour. City and tourist beaches are often a recipient of grooming efforts in the summer months when visitation is greatest. For example, Toronto beaches are groomed daily from May until Labour Day (first weekend of September) (City of Toronto, 2009). This may help explain why mean LPDs reported from sites in the summer months of June, July, and August tended to be lower than mean LPDs in the spring and fall. Hoellein et al. (2015) also found higher densities of beach litter in fall and suggested this was because warm weather in the fall will still attract beach visitors but regular grooming has finished. Higher densities of large plastic litter in the spring may be a product of fall litter remaining in place over the winter. Tiny litter densities were more consistent throughout the year, likely because grooming equipment typically cannot pick up items less than 1 cm in diameter. Beach grooming was not accounted for in the data analysis.

APPLICATION OF FINDINGS

Identification of plastic litter sources is important for devising strategies to prevent and manage plastic debris in the Great Lakes. Based on this chapter’s results, food and smoking-related litter should be targeted first. This may involve an increase in the number of litter disposal and recycling sites on beaches, additional signage, fines for littering, or the banning of smoking on beaches – which has already occurred in some regions of Ontario (Moloney, 2013). Other anti-littering initiatives may include public education and outreach activities that drawn attention to the adverse impacts of plastic debris in aquatic environments.

Similarly, understanding the distribution patterns of plastic debris on Great Lakes shorelines will also help improve management strategies and allow cleanup efforts to respond to the worst affected areas. Based on this chapter's results, GTA shorelines are generally highly littered with plastic debris and should be the focus of cleanup efforts and litter prevention and management initiatives. It would also be advisable to focus efforts at shoreline sites known to attract a high number of visitors, such as provincial parks or tourist beaches.

DATASET LIMITATIONS AND RECOMMENDATIONS FOR IMPROVEMENT

While GCSC's dataset is an important resource for plastic litter prevention and management, its limitations must also be acknowledged. One of the most significant challenges of using and interpreting GCSC's data stems from the lack of a rigid surveying protocol set by GCSC for its volunteers. Given the volunteer nature of GCSC's program, this is entirely understandable. Yet, as a result, the regular variability in LPDs and TPDs at sites will be compounded by a variable number of volunteers, distance of shoreline cleaned, duration of cleanup, or minimum size of litter collected.

There is no set number of volunteers that can participate in a cleanup. Furthermore, many of the volunteers are children, participating through a school event or with their parents. A child's attention to detail when collecting litter will most likely be less than an adult's. Thus, a low volunteer count or high proportion of children volunteers may skew a site's LPD and TPD lower. Volunteers do not report the length of time they have spent cleaning. A longer survey may skew a site's LPD and TPD higher.

Furthermore, GCSC does not specify a minimum size of litter to pick up, so the items collected are at the sole discretion of the volunteers. It is not possible to account for volunteer motivation or attention to detail when interpreting the data. Thus, some sites may have had higher LPDs and TPDs due to smaller threshold of plastic debris being collected. In general, plastic density values calculated from volunteer survey data are expected to underestimate true plastic abundance due to smaller litter not being collected. These uncertainties were not accounted for in this study directly. However, the variable distance of shoreline covered per cleanup was accounted for by normalizing litter counts by kilometer to produce the LPD and TPD values (No. pieces/km).

Other challenges for interpreting GCSC's dataset were the influence of site selection bias and presence of null values. Volunteers are more likely to clean shoreline sites that are popular destinations or close to where they live. Thus, sites in highly populated areas are more likely to be cleaned than those in rural areas. Volunteers are also more likely to clean shoreline sites in the summer months when the weather is nicer. This bias was not accounted for in this study. Incomplete records were fairly common in the dataset. Some records would report large litter counts but not tiny litter counts, some would report counts for only select litter items, and others would report zero litter counts. Records without large litter counts were disregarded when calculating the mean LPD of a study region and records without tiny litter counts were disregarded with calculating the mean TPD of a study region. Null values in partially completed records were assigned zero values with the assumption that the volunteers did not find specific litter items and therefore left those fields blank.

Due to the uncertainties in GCSC's dataset and the low number of survey records per site (a maximum of 2), litter classification, distribution, and drivers were analyzed on a regional scale instead of site scale. To help reduce the effect of outliers and sampling error in the data, only regions with at least 5 survey records were included in the regression analyses. However, analyzing litter at a regional scale has its own limitations for drawing conclusions from the data. For instance, nothing can be claimed about the abundance, distribution, and drivers of litter at individual sites within the study regions. Yet, a benefit of regional analysis is that 'holes' in the dataset's spatial coverage (i.e. areas with no surveys) are, at least partially, filled, providing more general conclusions about litter characteristics over larger areas.

The caveats with GCSC's data also make meaningful comparison to litter densities reported in the literature more difficult. Additionally, most studies will report litter densities over a 2-dimensional area (pieces per m^2 or km^2), while GCSC records only provide a 1-demenstional distance cleaned, i.e. density values are reported in pieces per m or km. As a result, GCSC litter densities would likely be less than those in the literature. It would also be inappropriate to compare GCSC's dataset to studies that have collected microplastics or microscopic plastics.

Despite these limitations, important benefits of the GCSC program include a high number of measurements taken and the direct impact of cleaning shorelines while generating data. In addition, volunteer cleanup programs do not require the same level of funding as a dedicated scientific study; it is unlikely that an academic or government funded program would last as long or collect as spatially- and temporally-large a record set as GCSC's.

Another benefit of the GCSC program is that results can be used to alter and refine the methods used to collect data. There is also an opportunity for a closer dialog between GCSC and the science community to better leverage GCSC's dataset for scientific research. In the future, it would be beneficial for GCSC volunteers to report the duration of cleanups. It would also be useful to approximate the area of shoreline covered so that litter density values could be more easily compared to those in the literature. Furthermore, reporting the shape of tiny litter (e.g. pellet, fragment, line) in addition to material type could provide a clearer indication of their source(s). Repeated cleanups at individual sites on a regular time interval (e.g. once per week) would also provide a more accurate temporal record of litter types and abundance at those sites and allow for analysis at the site scale, offering a finer spatial resolution of litter distribution than currently possible with the existing dataset. Finally, to help calculate the accumulation rate of plastic debris at shoreline sites, cleanups could be arranged to survey the same sites after a set length of time.

CONCLUSIONS

Analysis of GCSC's 2013 and 2014 volunteer survey data was used to help characterize the classification, distribution, and environmental drivers of plastic debris on Canadian shorelines of the Great Lakes. An average 80% of litter collected from shorelines was either food or smoking related. Furthermore, approximately 80% of the litter was comprised of plastic. Thus, the primary source of large plastic litter on shorelines is considered to be in situ anthropogenic activity including beach and shoreline visitors who improperly dispose of their waste. This was further supported by the significant positive correlations found between census division population density and census division mean LPD as well as census division mean coastal stress and census division mean LPD.

In both 2013 and 2014, a high number of cleanups were performed around Coastal Cleanup Day, but in 2014 there was a more even distribution of surveys across seasons. Large plastic litter densities at lake shoreline sites were found to average twice as much in the spring and fall than in the summer. This is most likely a result of increased beach and shoreline grooming in the summer. Census divisions with the highest mean litter densities were found around the Greater Toronto Area. In general, more densely populated regions tended to have higher mean LPDs. LPDs were found to be in the same range as those reported on the American shorelines of the Great Lakes in Chapter 1.

Simultaneous sampling of large and tiny litter at study sites revealed a significant, positive correlation between their densities at both lake shoreline and inland locations, providing evidence that their sources and drivers are related. It is likely that a major portion of the tiny litter collected are broken-down fragments of larger intact litter items. At the census division level, a positive trend was also found between mean large plastic density using lake shoreline site data and that using inland site data, indicating that lake shoreline and inland sites may share common sources and drivers. However, an insignificant relationship between census division population density and census division mean LPD using inland data suggests that plastic densities at inland sites are not as affected by population density as those at lake shoreline sites.

The results of this study provide evidence that Canadian shorelines across the entire Great Lakes region are littered with plastic debris. Action should be taken to prevent and manage litter input onto shorelines. This may be accomplished by increasing the density of trash and recycling receptacles along shorelines, introducing higher littering fines, banning smoking at parks and beaches, and educating the public about the harmful effects of plastic debris on fauna of the Great Lakes region. Cleanups would be best served to focus on densely populated areas where shorelines tend to accumulate more litter.

Table 2.1: Summary of 2013 and 2014 GCSC lake shoreline data by major lake region.

	Superior		**Huron**		**St. Clair**		**Erie**		**Ontario**	
	2013	2014	2013	2014	2013	2014	2013	2014	2013	2014
Total number of records	7	4	20	47	5	5	12	12	109	147
Total number of volunteers	84	43	288	656	37	38	261	164	2975	3603
Total distance cleaned (km)	10	3	43	67	4	5	28	27	193	204
Total litter mass (kg)	308	183	725	1860	146	169	1048	1206	6991	7020
Total large litter pieces	2738	707	12569	16308	1440	2346	10018	4200	102867	126416
% large litter that is plastic	74	71	86	87	87	90	80	74	85	85
Total tiny litter pieces	149	96	2613	3747	455	403	2242	1828	16608	35862
% tiny litter that is plastic	57	46	68	88	84	91	86	91	85	85

Table 2.2: Summary of 2013 and 2014 GCSC inland data (all sites).

	Ontario	
	2013	2014
Total number of records	233	278
Total number of volunteers	5195	5839
Total distance cleaned (km)	451	501
Total litter mass (kg)	15660	19179
Total large litter pieces	128410	133961
% large litter that is plastic	83	82
Total tiny litter pieces	19370	30282
% tiny litter that is plastic	76	73

Table 2.3: Comparison of the minimum, maximum, and mean large plastic densities (LPDs) (No. pieces/km) per major lake region from Chapter 1 (AAB and GCSC data from 2012) and Chapter 2 (GCSC data from 2013 and 2014). Only major lake regions with data available from both AAB and GCSC datasets are included.

	AAB 2012		**GCSC 2012**		**GCSC 2013**		**GCSC 2014**	
	LPD Range	LPD Average	LPD Range	LPD Average	LPD Range	LPD Average	LPD Range	LPD Average
Superior	10-7202	659	16-1054	499	29-611	253	499-499	499
Huron	25-3896	1116	94-5428	860	8-7560	1024	0-2454	351
Erie	46-3524	1062	19-882	279	28-2304	605	21-948	260
Ontario	116-1239	694	16-14274	999	3-6497	754	21-11840	799

Figure 2.1: Great Canadian Shoreline Cleanup locations in 2013 (A) and 2014 (B). The white dots represent lake shoreline sites (those within 500 m of a Great Lake or Lake St. Clair shoreline) and the grey dots represent inland sites.

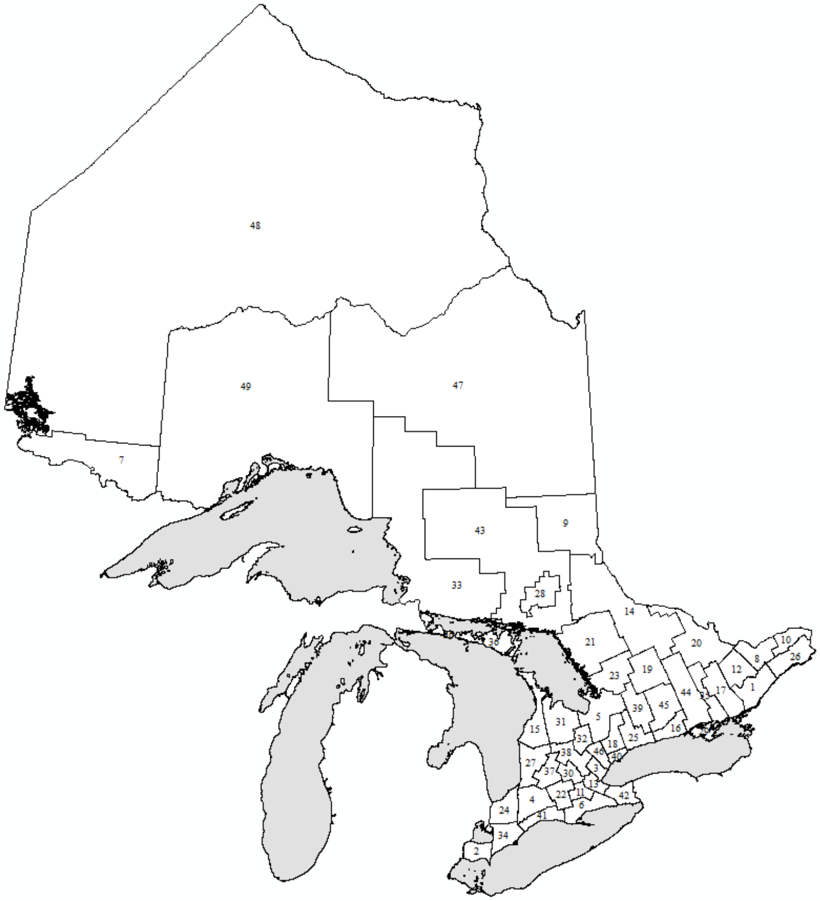

Figure 2.2: Ontario census divisions used in this study (see Appendix C, Table 1 for census division names).

Figure 2.3: Lake shoreline regions used in this study (see Appendix C, Table 2 for lake shoreline region names).

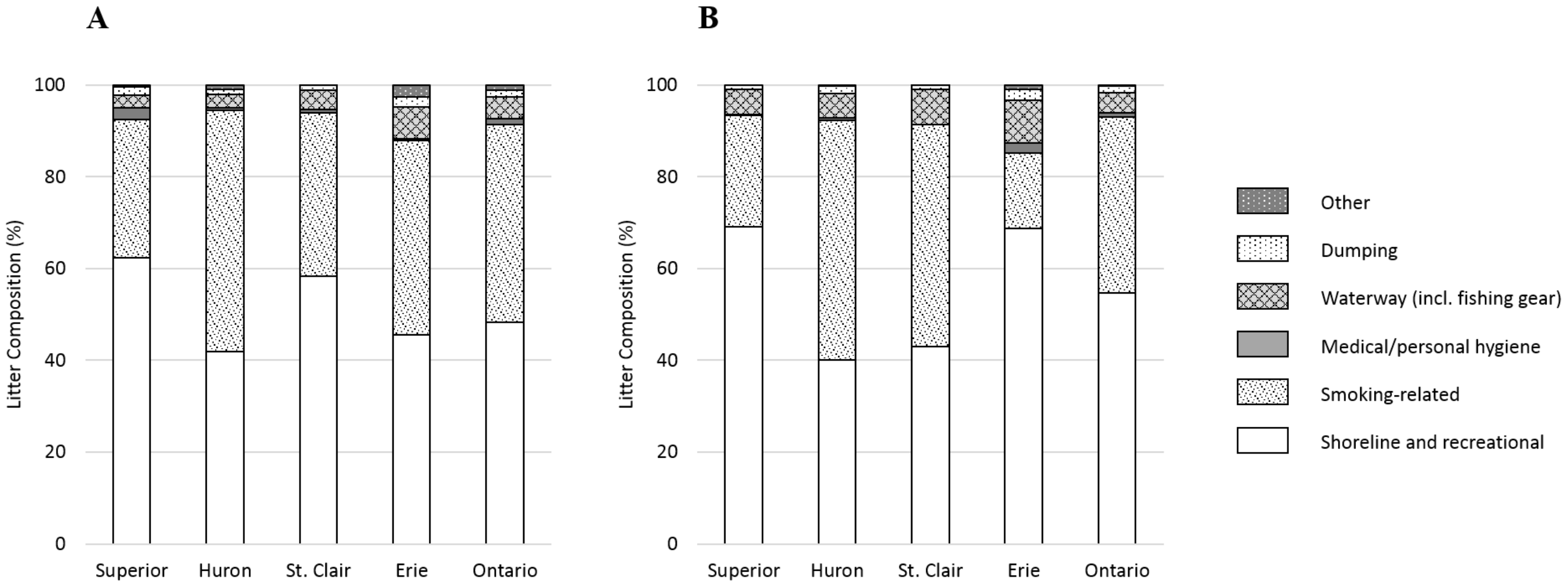

Figure 2.4: Classification of large litter by activity class (see Appendix A, Table 2 for details) per major lake region using lake shoreline data for 2013 (A) and 2014 (B).

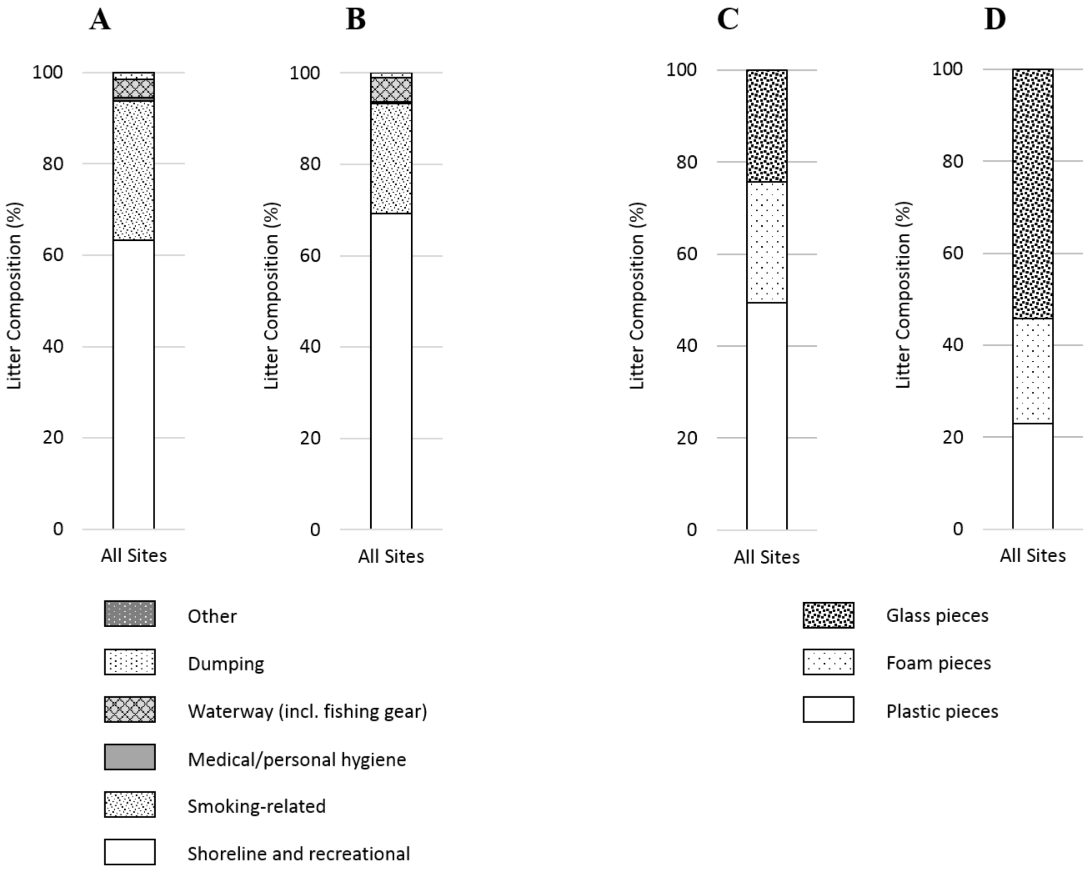

Figure 2.5: Classification of large litter by activity class (see Appendix A, Table 2 for details) using inland site data for 2013 (A) and 2014 (B) and breakdown of tiny litter by material type using inland site data for 2013 (C) and 2014 (D).

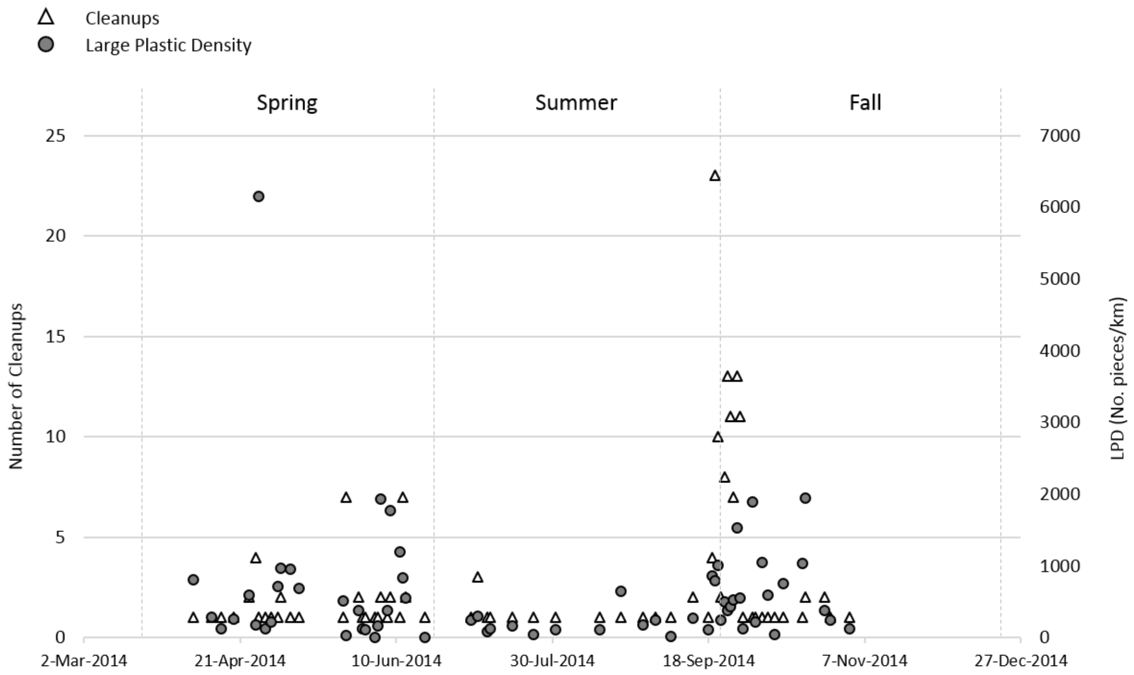

Figure 2.6: Temporal distribution of GCSC lake shoreline cleanups for 2014 and corresponding mean large plastic densities (LPDs) per day.

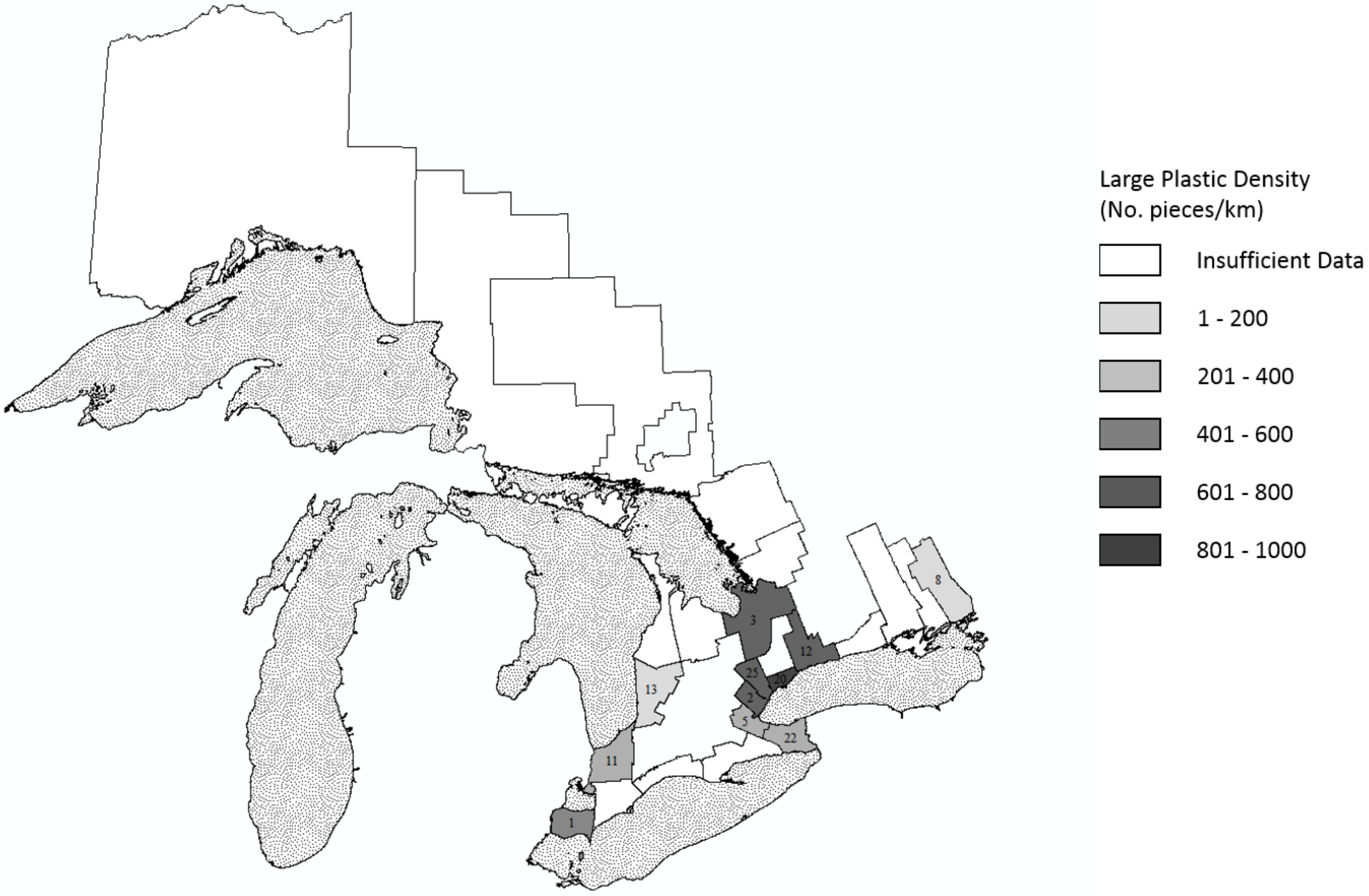

Figure 2.7: Mean large plastic density per census division using 2014 lake shoreline site data. Only census divisions with at least 5 survey records are included; those with less than 5 are labelled ‘Insufficient Data’.

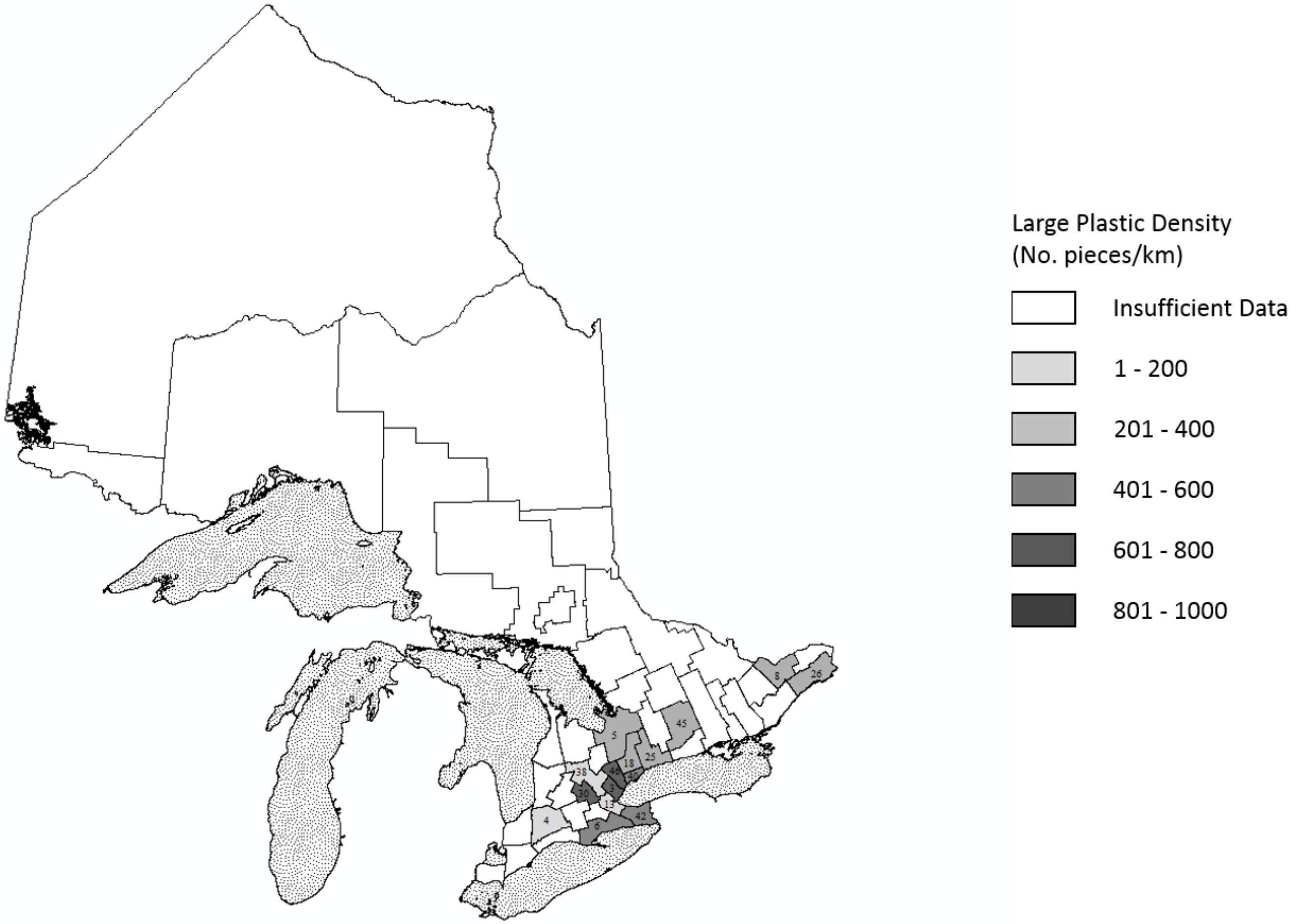

Figure 2.8: Mean large plastic density per census division using 2014 inland site data. Only census divisions with at least 5 survey records are included; those with less than 5 are labelled ‘Insufficient Data’.

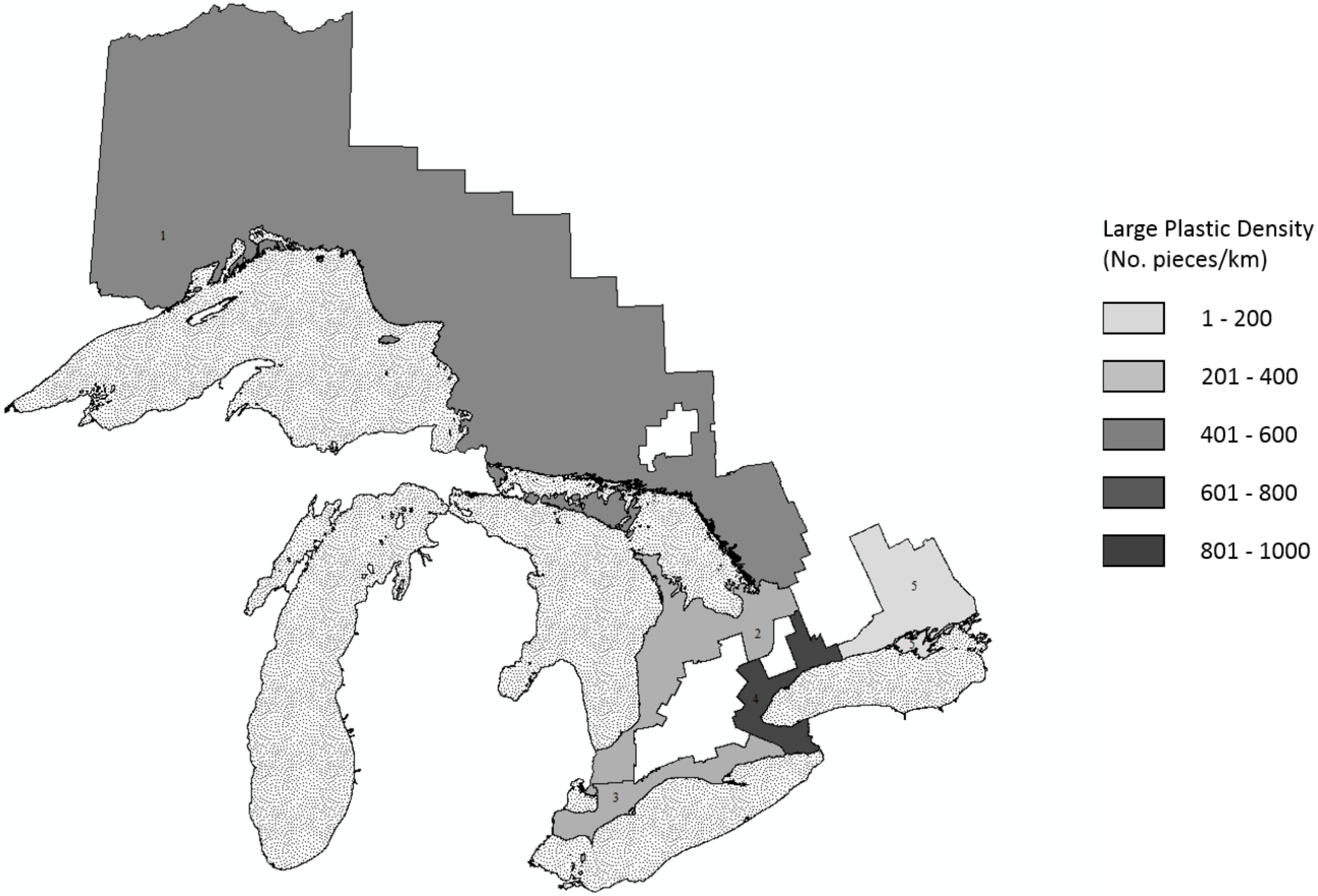

Figure 2.9: Mean large plastic density per lake shoreline region using 2014 lake shoreline site data.

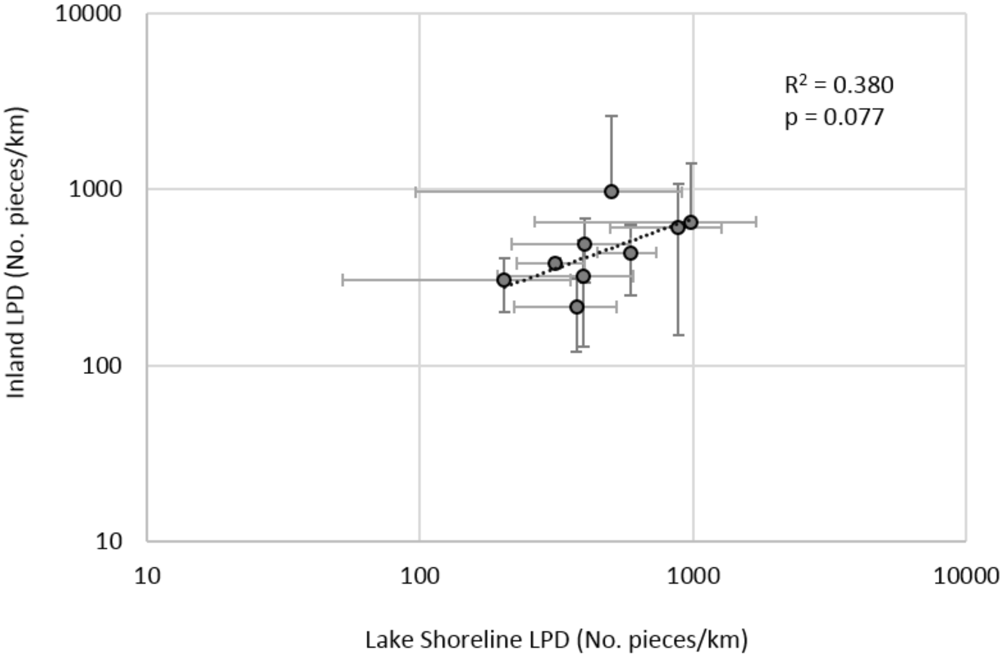

Figure 2.10: Simple linear regression between census division mean (± SD) large plastic density (LPD) using combined 2013 and 2014 lake shoreline site data and that using combined 2013 and 2014 inland site data. Only census divisions with at least 5 lake shoreline and 5 inland survey records are included. Error bars represent one standard deviation of uncertainty.

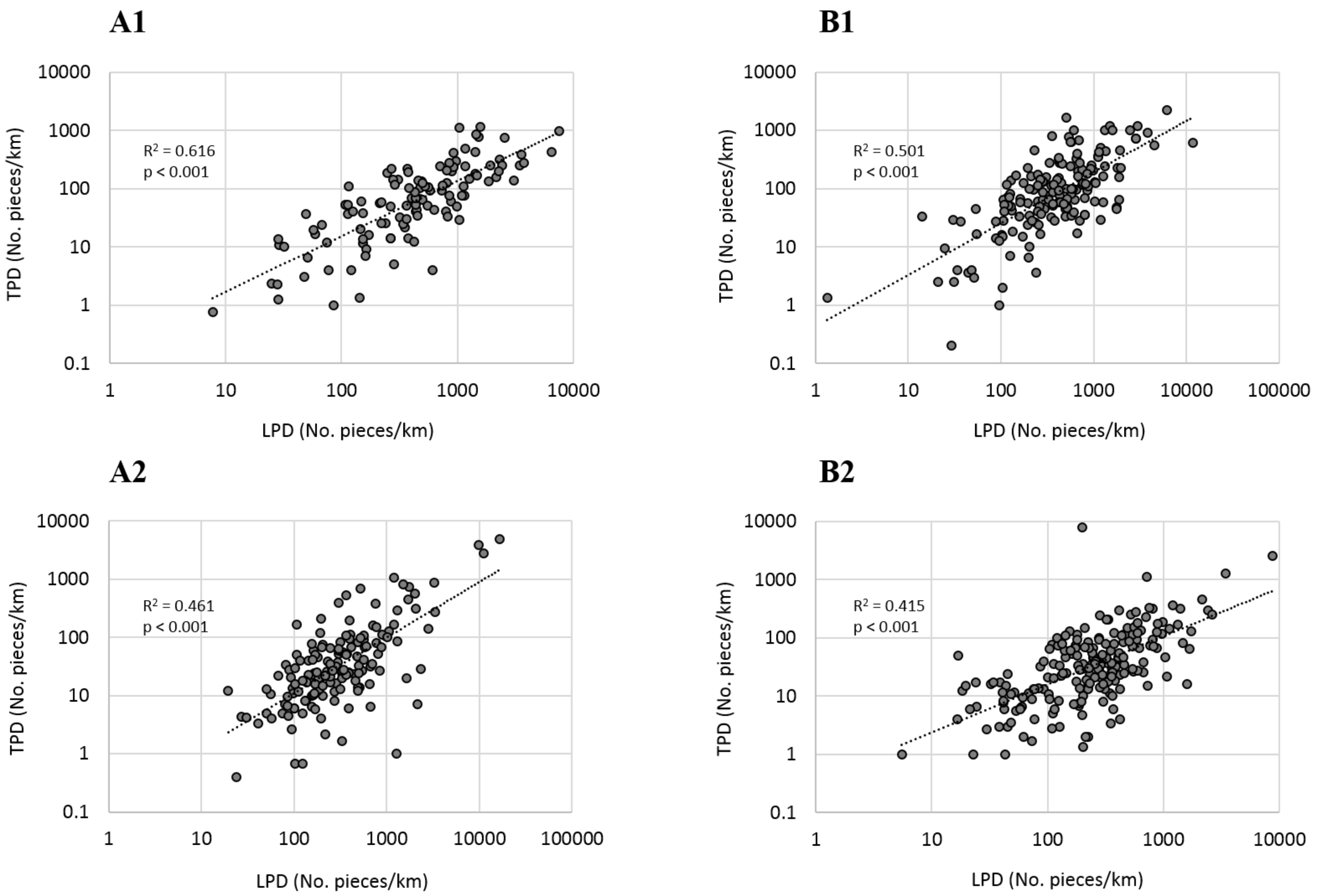

Figure 2.11: Simple linear regression between large plastic density (LPD) and tiny plastic density (TPD) of lake shoreline sites for 2013 (A1) and 2014 (B1) and inland sites for 2013 (A2) and 2014 (B2). Sites without a LPD or TPD due to incomplete survey records are not included.

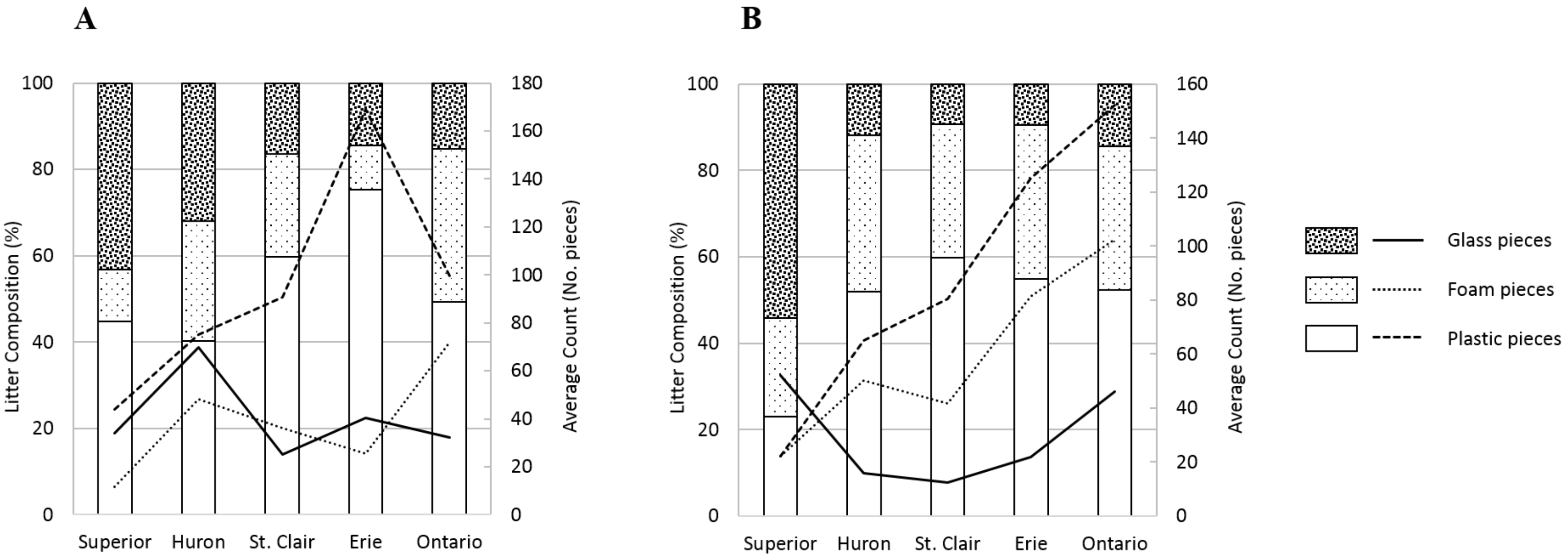

Figure 2.12: Breakdown and average abundance of tiny litter by material type per major lake region using lake shoreline site data for 2013 (A) and 2014 (B).

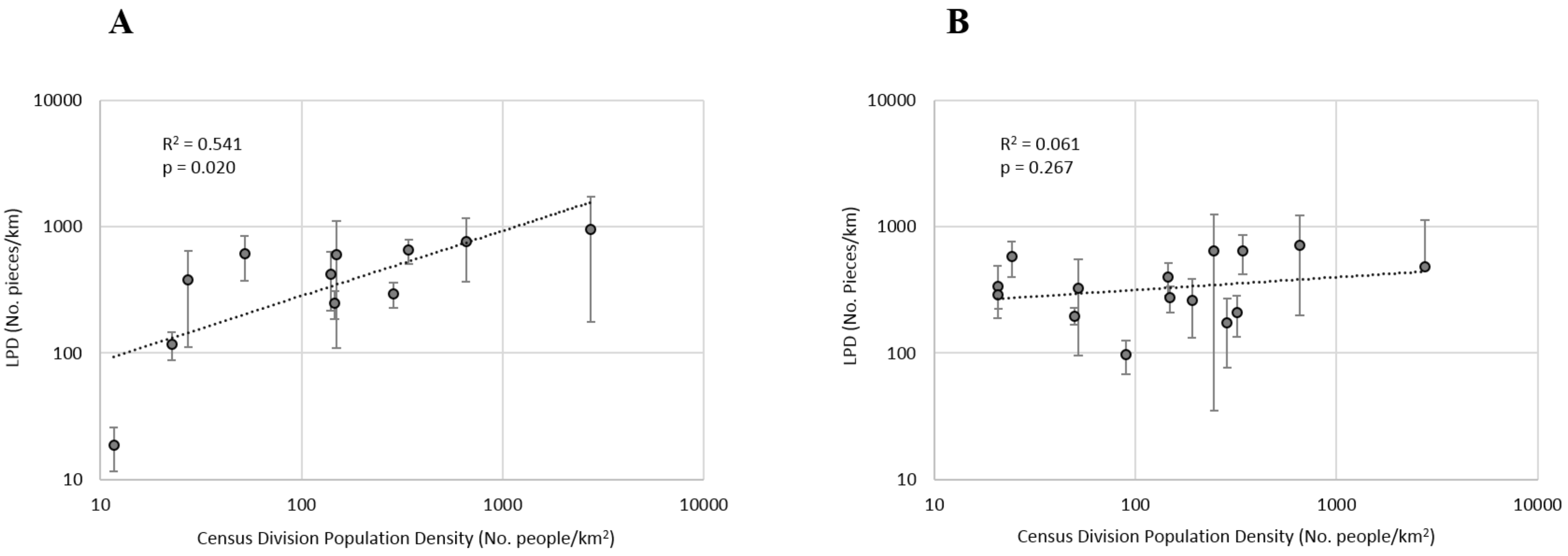

Figure 2.13: Simple linear regression between census division population density and census division mean (± SD) large plastic litter density (LPD) using 2013 lake shoreline site data (A) and 2014 inland site data (B). Only census divisions with at least 5 survey records are included. Error bars represent one standard deviation of uncertainty.

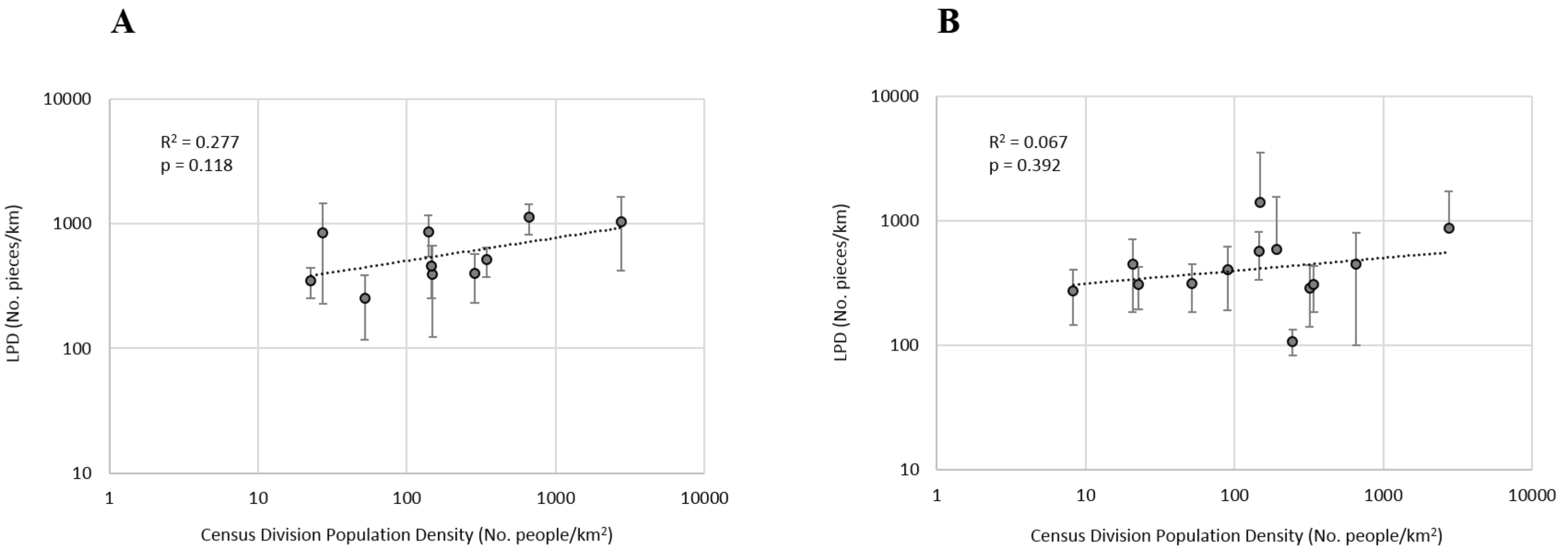

Figure 2.14: Simple linear regression between census division population density and census division mean (± SD) large plastic litter density (LPD) using 2013 lake shoreline site data (A) and 2013 inland site data (B). Only census divisions with at least 5 survey records are included. Error bars represent one standard deviation of uncertainty.

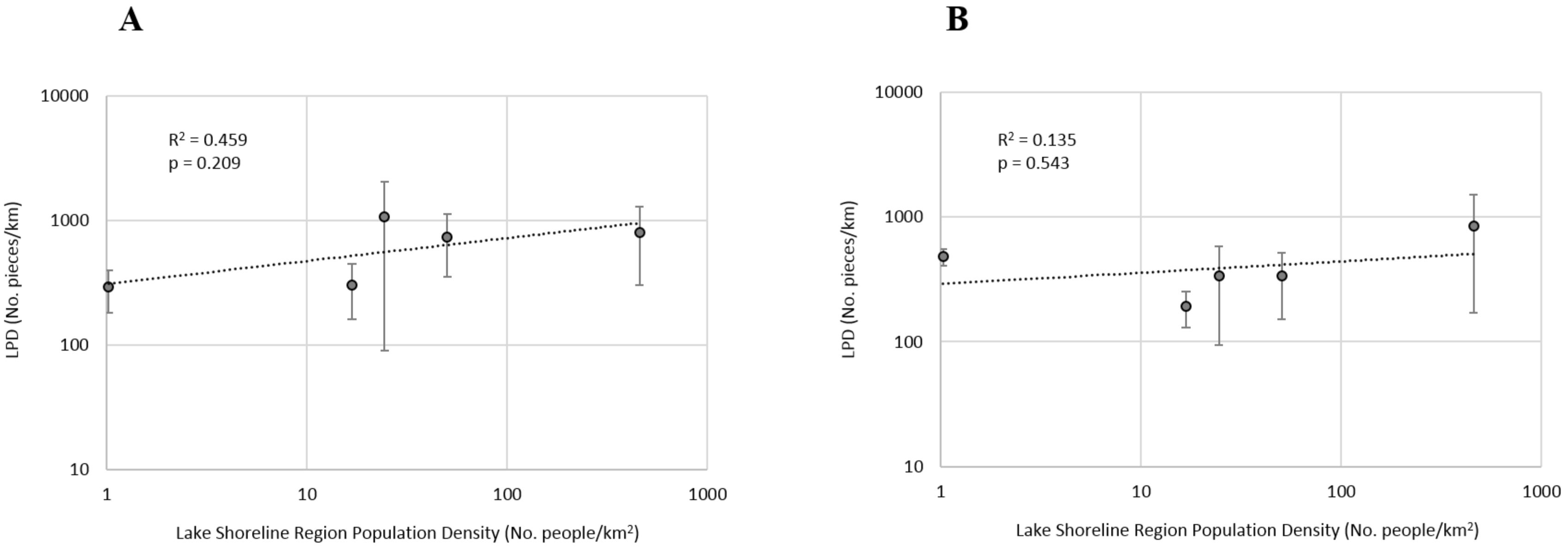

Figure 2.15: Simple linear regression between lake shoreline region population density and lake shoreline region mean (± SD) large plastic litter density (LPD) for 2013 (A) and 2014 (B). Error bars represent one standard deviation of uncertainty.

CONCLUSIONS AND PERSPECTIVES

Chapter 1 reviewed the research that has been conducted on plastic debris in the Great Lakes. High densities of plastic debris have been found in the open-waters, shorelines, and bottom sediments of the Great Lakes, as well as in rivers that flow into the Great Lakes. Sources of plastic debris to the Great Lakes include microplastic beads from consumer products, pellets from the plastic manufacturing industry, and waste from beach-goers, shipping, and fishing activities. Existing concentrations of microplastics in the open-waters of the Great Lakes are among the highest in any freshwater system in the world. The highest concentrations are even on par with those found in the oceanic gyre systems. Concentrations of macroplastic debris on shorelines of the Great Lakes tended to be less than those reported on ocean shorelines, possibly due to more frequent grooming of Great Lakes beaches and a smaller presence of derelict fishing gear in the Great Lakes.

Based on available data, several knowledge gaps remain about the sources, breakdown, fate, and impacts of plastic debris in the Great Lakes. Specifically, future research should address the temporal variability of plastic debris inputs and breakdown of inputs by size and composition of debris, the rates and mechanisms at which different types of plastic debris degrade in freshwater, the proportion of plastics in bottom sediments, the quantity and distribution of macroplastic debris in both beach and non-beach environments, the distribution of microplastics and how their sources and environmental fate differ from that of larger plastics, and the bioaccumulation of plastics and contaminants in food webs and resulting ecotoxicological consequences and risks to human health.

Research should also focus on improving sampling and measurement techniques for surveying plastic debris in aquatic environments. Current methods involve the use of trawl nets in open-water environments and manual collection of litter along shorelines. Following the collection of litter, the labor-intensive step of separating plastic pieces from the rest of the litter is required by visual inspection or use of a spectrometer; the Energy Dispersive X-Ray Spectroscopy System (EDS) and Fourier Transform Infrared Spectrometer (FTIR) are commonly used for this task. Yet, these spectrometers can only identify the composition of litter one piece at a time. Thus, these methods are time-consuming, costly, and only allow for a limited areal coverage and temporal resolution of observations. The use of remote sensors for spectral imaging of aquatic regions may offer a promising direction for more efficient surveying of plastic debris. NIR has the ideal spectral range for in situ identification of plastic debris. Due to significant attenuation in water, NIR sensing technology is most promising for plastics detection in filtered water samples and solid-phase samples (e.g. sediments, soils, beach sand). Raman spectroscopy suffers less from attenuation in water than NIR and should therefore be incorporated into in situ sensors for monitoring plastics within the water column. More research is required to assess how environmental weathering and biofouling alter the spectral characteristics of plastic debris.

Chapter 2 used citizen science data from the Great Canadian Shoreline Cleanup program to help address the knowledge gap about quantity and distribution of macroplastic debris in both beach and non-beach environments. The classification, distribution, and environmental drivers of plastic debris on Canadian shorelines of the Great Lakes were assessed. Classification of litter provided evidence that beach-goers and other in situ anthropogenic activity are the primary sources of macroplastic debris on shorelines. Regional analysis of the distribution and drivers of plastic debris verified this result by showing that in general, regions with higher mean plastic debris densities are those that are more densely populated. Regions around the densely populated Greater Toronto Area had consistently high plastic litter densities relative to other regions and should be the focus of litter prevention and management efforts.

Future analysis using GCSC data should examine the temporal variability of litter types and densities within study regions as well as the spatial and temporal distributions of individual litter items. In coming years, if volunteers begin to survey sites at a greater frequency, it may become possible to analyze the classification, distribution, and drivers of litter with accuracy on a site scale instead of regional scale. Tweaking GCSC's surveying protocol to include information about the duration of cleanup, area surveyed (instead of distance), and shape of tiny litter items collected would help reduce some of the uncertainties currently present in the dataset, increase the amount of useful information that can be drawn out, and ease the comparison of plastic density values calculated with GCSC data to those in the literature.

REFERENCES

Allan, J.D., McIntyre, P.B., Smith, S.D.P., Halpern, B.S., Boyer, G.L., Buchsbaum, A., Burton, G. a, Campbell, L.M., Chadderton, W.L., Ciborowski, J.J.H., Doran, P.J., Eder, T., Infante, D.M., Johnson, L.B., Joseph, C. a, Marino, A.L., Prusevich, A., Read, J.G., Rose, J.B., Rutherford, E.S., Sowa, S.P., Steinman, A.D., 2013. Joint analysis of stressors and ecosystem services to enhance restoration effectiveness. Proc. Natl. Acad. Sci. U. S. A. 110, 372–377.

Allen, V., Kalivas, J.H., Rodriguez, R.G., 1999. Post-consumer plastic identification using Raman spectroscopy. Appl. Spectrosc. 53, 672–681.

Alliance for the Great Lakes, 2013. Adopt-a-Beach 2013 Issue. Chicago, IL, USA.

Alliance for the Great Lakes, 2014. Adopt-a-Beach. http://www.greatlakes.org/ADOPTABEACH (accessed 11.17.14).

Allsopp, M., Walters, A., Santillo, D., Johnston, P., 2006. Plastic debris in the world's oceans, Greenpeace. Amsterdam, Netherlands.

Andrady, A.L., 2011. Microplastics in the marine environment. Mar. Pollut. Bull. 62, 1596–1605.

Arthur, C., Baker, J., Bamford, H. (Eds.), 2009. Proceedings of the International Research Workshop on the occurrence, effects, and fate of microplastic marine debris. Tacoma, WA, USA.

Ashton, K., Holmes, L., Turner, A., 2010. Association of metals with plastic production pellets in the marine environment. Mar. Pollut. Bull. 60, 2050–2055.

Barnes, D.K.A., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. Phil. Trans. R. Soc. B 364, 1985–1998.

Beletsky, D., Saylor, J.H., Schwab, D.J., 1999. Mean circulation in the Great Lakes. J. Great Lakes Res. 25, 78–93.

Bittner, G.D., Yang, C.Z., Stoner, M. a, 2014. Estrogenic chemicals often leach from BPA-free plastic products that are replacements for BPA-containing polycarbonate products. Environ. Health 13, 41–54.

Boerger, C.M., Lattin, G.L., Moore, S.L., Moore, C.J., 2010. Plastic ingestion by planktivorous fishes in the North Pacific Central Gyre. Mar. Pollut. Bull. 60, 2275–2278.

Broad, I., 2014. ArcGIS Toolbox – Calculate Point Statistics within Polygon with ArcPy. http://ianbroad.com/arcgis-toolbox-calculate-point-statistics-polygon-arcpy/ (accessed 6.9.15).

Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., Thompson, R., 2011. Accumulation of microplastic on shorelines woldwide: sources and sinks. Environ. Sci. Technol. 45, 9175–9179.

Browne, M.A., Galloway, T.S., Thompson, R.C., 2010. Spatial patterns of plastic debris along estuarine shorelines. Environ. Sci. Technol. 44, 3404–3409.

Carpenter, E.J., Smith, K.L., 1972. Plastics on the Sargasso Sea surface. Science. 175, 1240–1241.

Cheng, X., Shi, H., Adams, C.D., Ma, Y., 2010. Assessment of metal contaminations leaching out from recycling plastic bottles upon treatments. Environ. Sci. Pollut. Res. Int. 17, 1323–1330.

Chicago Park District, 2014. Chicago Park District improves beach monitoring for 2012 season. http://www.chicagoparkdistrict.com/chicago-park-district-improves-beach-monitoring-for-2012-season/ (accessed 11.17.14).

City of Toronto, 2009. Toronto Beaches Plan. Toronto, ON, Canada.

Codina-García, M., Militão, T., Moreno, J., González-Solís, J., 2013. Plastic debris in Mediterranean seabirds. Mar. Pollut. Bull. 77, 220–226.

Cole, M., Lindeque, P., Halsband, C., Galloway, T.S., 2011. Microplastics as contaminants in the marine environment: a review. Mar. Pollut. Bull. 62, 2588–2597.

Corcoran, P.L., Biesinger, M.C., Grifi, M., 2009. Plastics and beaches: a degrading relationship. Mar. Pollut. Bull. 58, 80–84.

Corcoran, P.L., Norris, T., Ceccanese, T., Walzak, M.J., Helm, P. a., Marvin, C.H., 2015. Hidden plastics of Lake Ontario, Canada and their potential preservation in the sediment record. Environ. Pollut. 204, 17–25.

Cózar, A., Echevarría, F., González-Gordillo, J.I., Irigoien, X., Ubeda, B., Hernández-León, S., Palma, A.T., Navarro, S., García-de-Lomas, J., Ruiz, A., Fernández-de-Puelles, M.L., Duarte, C.M., 2014. Plastic debris in the open ocean. Proc. Natl. Acad. Sci. U. S. A. 111, 10239–10244.

Day, R.H., Shaw, D.G., 1987. Patterns in the abundance of pelagic plastic and tar in the North Pacific Ocean, 1976-1985. Mar. Pollut. Bull. 18, 311–316.

Doyle, M.J., Watson, W., Bowlin, N.M., Sheavly, S.B., 2011. Plastic particles in coastal pelagic ecosystems of the Northeast Pacific ocean. Mar. Environ. Res. 71, 41–52.

Driedger, A.G.J., Dürr, H.H., Mitchell, K., Van Cappellen, P., 2015. Plastic debris in the Laurentian Great Lakes: A review. J. Great Lakes Res. 41, 9–19.

Eriksen, M., Lebreton, L.C.M., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C., Galgani, F., Ryan, P.G., Reisser, J., 2014. Plastic pollution in the world's oceans: more than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. PLoS One 9, e111913.

Eriksen, M., Mason, S., Wilson, S., Box, C., Zellers, A., Edwards, W., Farley, H., Amato, S., 2013a. Microplastic pollution in the surface waters of the Laurentian Great Lakes. Mar. Pollut. Bull. 77, 177–182.

Eriksen, M., Maximenko, N., Thiel, M., Cummins, A., Lattin, G., Wilson, S., Hafner, J., Zellers, A., Rifman, S., 2013b. Plastic pollution in the South Pacific subtropical gyre. Mar. Pollut. Bull. 68, 71–76.

Faure, F., Corbaz, M., Baecher, H., DE Alencastro, L.F., 2012. Pollution due to plastics and microplastics in Lake Geneva and in the Mediterranean Sea. Arch. Des Sci. 65, 157–164.

Fossi, M.C., Panti, C., Guerranti, C., Coppola, D., Giannetti, M., Marsili, L., Minutoli, R., 2012. Are baleen whales exposed to the threat of microplastics? A case study of the Mediterranean fin whale (Balaenoptera physalus). Mar. Pollut. Bull. 64, 2374–2379.

Free, C.M., Jensen, O.P., Mason, S. a, Eriksen, M., Williamson, N.J., Boldgiv, B., 2014. High-levels of microplastic pollution in a large, remote, mountain lake. Mar. Pollut. Bull. 85, 156–163.

Goldberg, E.D., 1994. Diamonds and plastics are forever? Mar. Pollut. Bull. 28, 466.

Goldstein, M.C., Titmus, A.J., Ford, M., 2013. Scales of spatial heterogeneity of plastic marine debris in the northeast pacific ocean. PLoS One 8, e80020.

Great Lakes & St. Lawerence Cities Initiative, 2014. Microplastics in the Great Lakes and St. Lawerence River. http://www.glslcities.org/initiatives/microplastics.cfm (accessed 11.17.14).

Gregory, M.R., 2009. Environmental implications of plastic debris in marine settings-entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. Phil. Trans. R. Soc. B 364, 2013–2025.

Gregory, M.R., Andrady, A.L., 2003. Plastics in the marine environment, in: Plastics and the Environment. John Wiley & Sons, Inc., Hoboken, New Jersey, pp. 379–402.

Habib, D., Locke, D.C., Cannone, L.J., 1998. Synthetic fibers as indicators of municipal sewage sludge, sludge products, and sewage treatment plant effluents. Water, Air, Soil Pollut. 103, 1–8.

Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. Environ. Sci. Technol. 46, 3060–3075.

Hoellein, T., Rojas, M., Pink, A., Gasior, J., Kelly, J., 2014. Anthropogenic litter in urban freshwater ecosystems: distribution and microbial interactions. PLoS One 9, e98485.

Hoellein, T.J., Westhoven, M., Lyandres, O., Cross, J., 2015. Abundance and environmental drivers of anthropogenic litter on 5 Lake Michigan beaches: A study facilitated by citizen science data collection. J. Great Lakes Res. 41, 78–86.

Holmes, L. a, Turner, A., Thompson, R.C., 2012. Adsorption of trace metals to plastic resin pellets in the marine environment. Environ. Pollut. 160, 42–48.

Hopewell, J., Dvorak, R., Kosior, E., 2009. Plastics recycling: challenges and opportunities. Phil. Trans. R. Soc. B 364, 2115–2126.

Hörig, B., Kühn, F., Oschütz, F., Lehmann, F., 2001. HyMap hyperspectral remote sensing to detect hydrocarbons. Int. J. Remote Sens. 22, 1413–1422.

Howell, E. a, Bograd, S.J., Morishige, C., Seki, M.P., Polovina, J.J., 2012. On North Pacific circulation and associated marine debris concentration. Mar. Pollut. Bull. 65, 16–22.

Imhof, H.K., Ivleva, N.P., Schmid, J., Niessner, R., Laforsch, C., 2013. Contamination of beach sediments of a subalpine lake with microplastic particles. Curr. Biol. 23, R867–868.

Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. Science. 347, 768–771.

Jeftic, L., Sheavly, S., Adler, E., 2009. Marine litter: a global challenge. Nairobi, Kenya.

Lattin, G.L., Moore, C.J., Zellers, a F., Moore, S.L., Weisberg, S.B., 2004. A comparison of neustonic plastic and zooplankton at different depths near the southern California shore. Mar. Pollut. Bull. 49, 291–294.

Law, K.L., Morét-Ferguson, S., Maximenko, N.A., Proskurowski, G., Peacock, E.E., Hafner, J., Reddy, C.M., 2010. Plastic accumulation in the North Atlantic Subtropical Gyre. Science. 329, 1185–1188.

Leggett, C., Scherer, N., Curry, M., Bailey, R., Haab, T., 2014. Assessing the economic benefits of reductions in marine debris: a pilot study of beach recreation in Orange County, California. Cambridge, MA, USA.

Mace, T.H., 2012. At-sea detection of marine debris: overview of technologies, processes, issues, and options. Mar. Pollut. Bull. 65, 23–27.

Masoumi, H., Safavi, S.M., Khani, Z., 2012. Identification and classification of plastic resins using near infrared reflectance spectroscopy. Int. J. Mech. Ind. Eng. 6, 213–220.

Mato, Y., Isobe, T., Takada, H., Kanehiro, H., Ohtake, C., Kaminuma, T., 2001. Plastic resin pellets as a transport medium for toxic chemicals in the marine environment. Environ. Sci. Technol. 35, 318–324.

Meeker, J.D., Sathyanarayana, S., Swan, S.H., 2009. Phthalates and other additives in plastics: human exposure and associated health outcomes. Phil. Trans. R. Soc. B 364, 2097–2113.

Moloney, P., 2013. Toronto smokers will have to butt out near building entrances, sports fields. Tor. Star. http://www.thestar.com/news/city_hall/2013/11/14/toronto_smokers_will_have_to_butt_out_near_building_entrances_sports_fields.html (accessed 5.3.14).

Moore, C.J., Moore, S.L., Leecasterà, M.K., Weisbergà, S.B., 2001. A comparison of plastic and plankton in the North Pacific Central Gyre. Mar. Pollut. Bull. 42, 1297–1300.

Moore, C.J., Moore, S.L., Weisberg, S.B., Lattin, G.L., Zellers, A.F., 2002. A comparison of neustonic plastic and zooplankton abundance in southern California's coastal waters. Mar. Pollut. Bull. 44, 1035–1038.

Morét-Ferguson, S., Law, K.L., Proskurowski, G., Murphy, E.K., Peacock, E.E., Reddy, C.M., 2010. The size, mass, and composition of plastic debris in the western North Atlantic Ocean. Mar. Pollut. Bull. 60, 1873–1878.

Mouat, J., Lopez Lozano, R., Bateson, H., 2010. Economic impacts of marine litter.

Nakashima, E., Isobe, A., Kako, S., Magome, S., Deki, N., Itai, T., Takahashi, S., 2011. Toxic metals in polyethylene plastic litter. Interdiscip. Stud. Environ. Chem. Environ. Model. Anal. 271–277.

Nalbone, J., 2014. Unseen threat: how microbeads harm New York waters, wildlife, health and environment. Albany, NY, USA.

NOAA Marine Debris Program, 2014. Great Lakes Action Plan. http://marinedebris.noaa.gov/regional-coordination/great-lakes-action-plan (accessed 11.17.14).

Oceaneye, 2013. Plastics and oceans. http://oceaneye.eu/en/le_plastique_et_les_oceans (accessed 11.17.14).

Ontario Ministry of Natural Resources, 2010. Tertiary Watersheds.

Pichel, W.G., Churnside, J.H., Veenstra, T.S., Foley, D.G., Friedman, K.S., Brainard, R.E., Nicoll, J.B., Zheng, Q., Clemente-Colón, P., 2007. Marine debris collects within the North Pacific Subtropical Convergence Zone. Mar. Pollut. Bull. 54, 1207–1211.

Plastic Soup Foundation, Stichting De Noordzee, Fauna & Flora International, Dutch Ministry of Infrastructure & Environment, United Nations Environmental Programme, 2014. International campaign against microbeads in cosmetics. http://beatthemicrobead.org/en/industry (accessed 11.17.14).

Queen's Printer for Ontario, 2015. Abount Ontario. http (accessed 6.9.15).

Ribic, C.A., 1998. Use of indicator items to monitor marine debris on a New Jersey beach from 1991 to 1996. Mar. Pollut. Bull. 36, 887–891.

Rios, L.M., Jones, P.R., Moore, C., Narayan, U. V, 2010. Quantitation of persistent organic pollutants adsorbed on plastic debris from the Northern Pacific Gyre's "eastern garbage patch." J. Environ. Monit. 12, 2226–2236.

Rochman, C.M., Hoh, E., Kurobe, T., Teh, S.J., 2013. Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. Sci. Rep. 3, 1–7.

Ryan, P.., Moloney, C.L., 1993. Marine litter keeps increasing. Nature 361, 23.

Ryan, P.G., Connell, A.D., Gardner, B.D., 1988. Plastic ingestion and PCBs in seabirds: is there a relationship? Mar. Pollut. Bull. 19, 174–176.

Santos, I.R., Friedrich, A.C., Ivar do Sul, J.A., 2009. Marine debris contamination along undeveloped tropical beaches from northeast Brazil. Environ. Monit. Assess. 148, 455–462.

Schwartz, J., 2013. Scientists turn their gaze toward tiny threats to Great Lakes. New York Times. http://www.nytimes.com/2013/12/15/us/scientists-turn-their-gaze-toward-tiny-threats-to-great-lakes.html?ref=earth&_r=1& (accessed 11.17.14).

Setälä, O., Fleming-Lehtinen, V., Lehtiniemi, M., 2014. Ingestion and transfer of microplastics in the planktonic food web. Environ. Pollut. 185, 77–83.

Shah, A.A., Hasan, F., Hameed, A., Ahmed, S., 2008. Biological degradation of plastics: a comprehensive review. Biotechnol. Adv. 26, 246–265.

Shaw, D.G., Day, R.H., 1994. Colour- and form-dependent loss of plastic micro-debris from the North Pacific Ocean. Mar. Pollut. Bull. 28, 39–43.

Sheavly, S.B., Register, K.M., 2007. Marine debris & plastics: environmental concerns, sources, impacts and solutions. J. Polym. Environ. 15, 301–305.

Statistics Canada, 2001. Census Divisions.

Stevenson, C., 2011. Plastic debris in the California marine ecosystem: A summary of current research, solution strategies and data gaps.

Stickel, B.H., Jahn, A., Kier, B., 2012. The cost to West Coast communities of dealing with trash, reducing marine debris. Blue Lake, CA, USA.

Teegarden, D.M., 2004. Polymer chemisty: introduction to an indispensable science. National Science Teachers Association Press.

Teuten, E.L., Rowland, S.J., Galloway, T.S., Thompson, R.C., 2007. Potential for plastics to transport hydrophobic contaminants. Environ. Sci. Technol. 41, 7759–7764.

Teuten, E.L., Saquing, J.M., Knappe, D.R.U., Barlaz, M. a, Jonsson, S., Björn, A., Rowland, S.J., Thompson, R.C., Galloway, T.S., Yamashita, R., Ochi, D., Watanuki, Y., Moore, C., Viet, P.H., Tana, T.S., Prudente, M., Boonyatumanond, R., Zakaria, M.P., Akkhavong, K., Ogata, Y., Hirai, H., Iwasa, S., Mizukawa, K., Hagino, Y., Imamura, A., Saha, M., Takada, H., 2009. Transport and release of chemicals from plastics to the environment and to wildlife. Philos. Trans. R. Soc. Lond. B. Biol. Sci. 364, 2027–2045.

Thompson, R.C., Swan, S.H., Moore, C.J., vom Saal, F.S., 2009. Our plastic age. Philos. Trans. R. Soc. Lond. B. Biol. Sci. 364, 1973–1976.

Tsuchida, A., Kawazumi, H., Kazuyoshi, A., Yasuo, T., 2009. Identification of shredded plastics in milliseconds using Raman spectroscopy for recycling, in: IEEE Sensors 2009 Conference. IEEE, pp. 1473–1476.

US EPA, 2012. Where would we be without the Great Lakes? http://www.epa.gov/glnpo/monitoring/great_minds_great_lakes/social_studies/without.html (accessed 6.18.14).

US EPA, 2014. Plastics. http://www.epa.gov/osw/conserve/materials/plastics.htm (accessed 11.17.14).

Van den Broek, W.H. a. M., Derks, E.P.P. a., van de Ven, E.W., Wienke, D., Geladi, P., Buydens, L.M.C., 1996. Plastic identification by remote sensing spectroscopic NIR imaging using kernel partial least squares (KPLS). Chemom. Intell. Lab. Syst. 35, 187–197.

Veenstra, T.S., Churnside, J.H., 2012. Airborne sensors for detecting large marine debris at sea. Mar. Pollut. Bull. 65, 63–68.

Wagner, M., Scherer, C., Alvarez-Muñoz, D., Brennholt, N., Bourrain, X., Buchinger, S., Fries, E., Grosbois, C., Klasmeier, J., Marti, T., Rodriguez-Mozaz, S., Urbatzka, R., Vethaak, A., Winther-Nielsen, M., Reifferscheid, G., 2014. Microplastics in freshwater ecosystems: what we know and what we need to know. Environ. Sci. Eur. 26.

Winterling, H., Sonntag, N., 2011. Rigid polystyrene foam (EPS, XPS). Kunststoffe. http://www.plasticsportal.net/wa/plasticsEU~it_IT/function/conversions:/publish/common/upload/technical_journals/plastics_trendreports/Rigid_polystyrene_foam.pdf (accessed 11.17.14).

Worldwatch Institute, 2015. Global plastic production rises, recycling lags. http://vitalsigns.worldwatch.org/vs-trend/global-plastic-production-rises-recycling-lags (accessed 6.8.15).

Wright, S.L., Thompson, R.C., Galloway, T.S., 2013. The physical impacts of microplastics on marine organisms: a review. Environ. Pollut. 178, 483–492.

Yamashita, R., Tanimura, A., 2007. Floating plastic in the Kuroshio Current area, western North Pacific Ocean. Mar. Pollut. Bull. 54, 485–488.

Zarfl, C., Matthies, M., 2010. Are marine plastic particles transport vectors for organic pollutants to the Arctic? Mar. Pollut. Bull. 60, 1810–1814.

Zbyszewski, M., Corcoran, P.L., 2011. Distribution and degradation of fresh water plastic particles along the beaches of Lake Huron, Canada. Water, Air, Soil Pollut. 220, 365–372.

Zbyszewski, M., Corcoran, P.L., Hockin, A., 2014. Comparison of the distribution and degradation of plastic debris along shorelines of the Great Lakes, North America. J. Great Lakes Res. 40, 288–299.

Zettler, E.R., Mincer, T.J., Amaral-Zettler, L.A., 2013. Life in the “Plastisphere”: microbial communities on plastic marine debris. Environ. Sci. Technol. 47, 7137–7146.

Zubris, K.A. V, Richards, B.K., 2005. Synthetic fibers as an indicator of land application of sludge. Environ. Pollut. 138, 201–211.

APPENDIX A

Table 1: Anthropogenic debris items sorted by activity class as used in Chapter 1 (Figure 1.1), based on Alliance for the Great Lakes' classification of anthropogenic debris items on their Adopt-a-Beach^{TM} Litter Monitoring Form.

Shoreline and Recreational	**Smoking-Related**	**Medical/Personal Hygiene**	**Waterway**	**Dumping**	**Other**
6-pack holders	Cigar tips	Condoms	Bait containers	55-gal. drums	Discarded food
Bags (paper)	Cigarette lighters	Diapers	Bleach/cleaner bottles	Appliances (refrigerators, washers, etc.)	Drug paraphernalia (crack pipes, bags, etc.)
Bags (plastic)	Cigarettes/cigarette filters	Syringes	Buoys/floats	Batteries	Fireworks debris
Balloons	Tobacco packaging/wrappers	Tampons/tampon applicators	Crates	Building materials	
Beverage bottles (glass)			Fish traps	Car/car parts	
Beverage bottles (plastic) 2 Liters or less			Fishing line	Tires	
Beverage cans			Fishing lures/light sticks		
Caps, lids			Fishing nets		
Clothing, shoes			Light bulbs/tubes		
Cups, plates, forks, knives, spoons			Oil/lube bottles		
Food wrappers/containers			Pallets		
Pull tabs			Plastic sheeting/tarps		
Shotgun shells/wadding			Rope		
Straws, stirrers			Strapping bands		
Toys					

Table 2: Anthropogenic debris items sorted by activity class as used in Chapter 2 (Figure X), based on the Great Canadian Shoreline Cleanup's classification of anthropogenic debris items on their Individual Data Card.

Shoreline and Recreational	Smoking-Related	Medical/Personal Hygiene	Waterway	Dumping	Other
6-pack holders	Cigar Tips	Condoms	Fishing Buoys, Pots & Traps	Appliances	Fireworks
Balloons	Cigarette Butts	Diapers	Fishing Line (1 yard/meter = 1 piece)	Batteries	
Beverage Bottles (Glass)	Cigarette Lighters	Syringes	Fishing Lures/Light Sticks	Construction Materials	
Beverage Bottles (Plastic)	Tobacco Packaging/Wrap	Tampons/Tampon Applicators	Fishing Net & Pieces	Tires	
Beverage Cans			Other Plastic Bottles		
Bottle Caps (Metal)			Other Plastic/Foam Packaging		
Bottle Caps (Plastic)			Rope (1 yard/meter = 1 piece)		
Clothing, Shoes			Strapping Bands		
Cups & Plates (Foam)					
Cups & Plates (Paper)					
Cups & Plates (Plastic)					
Food Wrappers					
Forks, Knives, Spoons					
Grocery Bags (Plastic)					
Lids (Plastic)					
Other Plastic Bags					
Paper Bags					
Straws/Stirrers					
Take Out/Away Containers (Foam)					
Take Out/Away Containers (Plastic)					
Toys					

APPENDIX B

Table 1: Anthropogenic debris items classified as plastic or non-plastic (other) as used in Chapter 1 (Figure 1.2), based on the European Environmental Agency's (EEA) classification of anthropogenic debris items in their Marine Litter Watch App.

Plastic	**Other**
6-pack holders	55-gal. drums
Bags (plastic)	Appliances (refrigerators, washers, etc.)
Bait containers	Bags (paper)
Beverage bottles (plastic) 2 Liters or less	Balloons
Bleach/cleaner bottles	Batteries
Buoys/floats	Beverage bottles (glass)
Caps, lids	Beverage cans
Car/car parts	Building materials
Cigar tips	Clothing, shoes
Cigarette lighters	Condoms
Cigarettes/cigarette filters	Crates
Cups, plates, forks, knives, spoons	Discarded food
Diapers	Fireworks debris
Drug paraphernalia (crack pipes, bags, etc.)	Fish traps
Fishing line	Light bulbs/tubes
Fishing lures/light sticks	Pallets
Fishing nets	Pull tabs
Food wrappers/containers	Tires
Oil/lube bottles	
Plastic sheeting/tarps	
Rope	
Shotgun shells/wadding	
Strapping bands	
Straws, stirrers	
Syringes	
Tampons/tampon applicators	
Tobacco packaging/wrappers	
Toys	

Table 2: Anthropogenic debris items classified as plastic or non-plastic (other) as used in Chapter 2 (Figure X), based on the European Environmental Agency's (EEA) classification of anthropogenic debris items in their Marine Litter Watch App.

Plastic	**Other**
6-pack holders	Appliances
Beverage Bottles (Plastic)	Balloons
Bottle Caps (Plastic)	Batteries
Cigar Tips	Beverage Bottles (Glass)
Cigarette Butts	Beverage Cans
Cigarette Lighters	Bottle Caps (Metal)
Cups & Plates (Foam)	Clothing, Shoes
Cups & Plates (Plastic)	Condoms
Diapers	Construction Materials
Fishing Buoys, Pots & Traps	Cups & Plates (Paper)
Fishing Line (1 yard/meter = 1 piece)	Fireworks
Fishing Lures/Light Sticks	Paper Bags
Fishing Net & Pieces	Tires
Food Wrappers	
Forks, Knives, Spoons	
Grocery Bags (Plastic)	
Lids (Plastic)	
Other Plastic Bags	
Other Plastic Bottles	
Other Plastic/Foam Packaging	
Rope (1 yard/meter = 1 piece)	
Strapping Bands	
Straws/Stirrers	
Syringes	
Take Out/Away Containers (Foam)	
Take Out/Away Containers (Plastic)	
Tampons/Tampon Applicators	
Tobacco Packaging/Wrap	
Toys	

APPENDIX C

Table 1: Summary of 2013 and 2014 GCSC lake shoreline and inland data by census division. Table values are combined totals from 2013 and 2014. Names of Ontario census divisions correspond to their ID numbers in Figure 3.1. Population is estimated from 2012 LansScan data.

ID #	**Name**	**Population (2012 est.)**	**Pop. density (No. /km^2)**	**Total # of surveys**		**Total # of volunteers**		**Total distance cleaned (km)**		**Total litter mass (kg)**		**Avg. % plastic (large litter)**		**Avg. % plastic (tiny litter)**	
				Lake Shoreline	Inland	Lake Shoreline	Inland	Lake Shoreline	Inland	Lake Shoreline	Inland	Lake Shoreline	Inland	Lake Shoreline	Inland
1	Leeds and Grenville	62599	17.3	N/A	6	N/A	80	N/A	7	N/A	563	N/A	77	N/A	63
2	Essex	260764	139.5	14	4	154	65	11	13	576	576	82	89	78	96
3	Halton	331396	341.5	21	19	394	650	26	35	1511	1480	80	75	71	58
4	Middlesex	299172	89.7	N/A	23	N/A	407	N/A	47	N/A	1575	N/A	80	N/A	62
5	Simcoe	277570	52.1	13	17	145	260	24	22	617	1433	81	75	77	84
6	Haldimand-Norfolk	70896	24.3	3	10	60	89	7	4	84	133	81	85	92	78
7	Rainy River	12274	0.7	N/A	2	N/A	11	N/A	4	N/A	25	N/A	85	N/A	77
8	Ottawa	553060	191.2	N/A	69	N/A	1491	N/A	138	N/A	4200	N/A	80	N/A	71
9	Timiskaming	21287	1.5	N/A	0	N/A	0	N/A	0	N/A	0	N/A	N/A	N/A	N/A
10	Prescott and Russell	55889	27.0	N/A	1	N/A	4	N/A	2	N/A	53	N/A	58	N/A	100
11	Brant	86754	78.4	N/A	6	N/A	159	N/A	11	N/A	195	N/A	75	N/A	52
12	Lanark	46498	14.5	N/A	0	N/A	0	N/A	0	N/A	0	N/A	N/A	N/A	N/A
13	Hamilton	328599	286.2	15	10	188	235	28	14	765	362	89	83	86	64
14	Nipissing	50251	2.6	N/A	1	N/A	18	N/A	2	N/A	136	N/A	77	N/A	93
15	Bruce	41862	10.1	5	2	193	154	7	3	98	145	94	59	84	49
16	Northumberland	52448	26.3	8	5	146	115	5	12	273	385	86	85	85	93
17	Frontenac	95631	22.6	8	10	178	176	11	15	52	233	89	85	67	79
18	York	668940	318.9	N/A	32	N/A	442	N/A	60	N/A	1010	N/A	83	N/A	78
19	Haliburton	11009	2.4	N/A	1	N/A	1	N/A	0	N/A	3	N/A	70	N/A	0
20	Renfrew	66464	8.2	N/A	10	N/A	283	N/A	28	N/A	807	N/A	77	N/A	52
21	Parry Sound	25773	2.5	2	0	45	0	2	0	9	0	88	N/A	90	N/A
22	Oxford	71413	34.8	N/A	1	N/A	23	N/A	1	N/A	12	N/A	0	N/A	0
23	Muskoka	36287	8.0	0	5	0	236	0	11	0	212	N/A	75	N/A	84
24	Lambton	83608	27.3	27	1	414	8	47	3	713	6	83	63	86	64
25	Durham	390510	149.0	28	26	787	398	76	37	3096	2085	80	80	85	83
26	Stormont, Dundas and Glengarry	69174	20.8	N/A	10	N/A	253	N/A	11	N/A	365	N/A	75	N/A	66
27	Huron	39865	11.7	10	0	27	0	10	0	243	0	51	N/A	97	N/A

28	Greater Sudbury / Grand Sudbury	106260	29.3	N/A	4	N/A	132	N/A	3	N/A	845	N/A	79	N/A	65
29	Prince Edward	17469	16.2	0	0	0	0	0	0	0	0	N/A	N/A	N/A	N/A
30	Waterloo	340850	246.0	N/A	16	N/A	216	N/A	38	N/A	801	N/A	84	N/A	65
31	Grey	58114	12.8	3	1	39	10	4	0	151	36	89	84	96	0
32	Dufferin	32258	21.5	N/A	5	N/A	255	N/A	9	N/A	676	N/A	83	N/A	77
33	Algoma	73240	1.4	2	5	4	92	4	6	122	39	66	90	46	71
34	Chatham-Kent	71031	28.6	4	3	172	27	31	5	1601	44	69	90	89	66
35	Lennox and Addington	27044	9.1	1	0	8	0	3	0	2	0	83	N/A	88	N/A
36	Manitoulin	7519	2.2	3	0	81	0	4	0	754	0	78	N/A	73	N/A
37	Perth	47006	21.1	N/A	0	N/A	0	N/A	0	N/A	0	N/A	N/A	N/A	N/A
38	Wellington	133926	49.6	N/A	9	N/A	125	N/A	17	N/A	290	N/A	80	N/A	89
39	Kawartha Lakes	50008	15.0	N/A	4	N/A	53	N/A	3	N/A	65	N/A	92	N/A	94
40	Toronto	1742355	2746.2	137	91	3878	2144	172	143	6554	8173	85	81	84	73
41	Elgin	58029	30.7	2	0	20	0	1	0	115	0	81	N/A	94	N/A
42	Niagara	274512	145.7	13	14	382	398	20	18	522	1367	83	80	64	72
43	Sudbury	13364	0.3	0	0	0	0	0	0	0	0	N/A	N/A	N/A	N/A
44	Hastings	85897	13.6	3	3	105	13	3	5	160	290	82	85	79	74
45	Peterborough	86886	20.6	N/A	15	N/A	486	N/A	42	N/A	1077	N/A	86	N/A	88
46	Peel	826661	657.1	21	66	569	1466	25	113	1123	5110	80	80	90	78
47	Cochrane	45706	0.3	N/A	0	N/A	0	N/A	0	N/A	0	N/A	N/A	N/A	N/A
48	Kenora	30510	0.1	N/A	1	N/A	22	N/A	0	N/A	4	N/A	86	N/A	98
49	Thunder Bay	83108	0.7	5	3	123	37	9	2	369	28	74	86	59	89

Table 2: Summary of 2013 and 2014 GCSC lake shoreline data by lake shoreline region. Table values are combined totals of 2013 and 2014 data. Names of lake shoreline regions correspond to their ID numbers in Figure 3.1. Population is estimated from 2012 LansScan data.

ID #	Name	Population (2012 est.)	Pop. density (No. /km^2)	Total # of surveys	Total # of volunteers	Total distance cleaned (km)	Litter mass (kg)	Avg. % plastic (large litter)	Avg. % plastic (tiny litter)
1	Huron-North-Superior	239291	1.0	12	253	19	1254	75	64
2	Huron-South	501019	24.4	58	818	92	1822	78	85
3	Erie-St. Clair	460720	50.3	23	406	50	2376	80	83
4	Golden Horseshoe	3893987	457.4	235	6198	347	13571	84	83
5	Ontario-North	278489	16.8	20	437	22	487	86	77

APPENDIX D

Table 1: Mean large plastic density (LPD) and corresponding relative standard deviation of census divisions using lake shoreline site data from 2013. Only census divisions with at least 5 survey records are included.

ID #	**Name**	**Total # of surveys**	**Avg. LPD (No. pieces/km)**	**Relative standard deviation (%)**
1	Essex	6	859	74
2	Halton	10	512	54
3	Simcoe	6	252	107
5	Hamilton	8	400	84
7	Northumberland	5	344	56
11	Lambton	7	847	146
12	Durham	14	396	138
20	Toronto	50	1030	118
22	Niagara	8	460	90
25	Peel	8	1128	54

Table 2: Mean large plastic density (LPD) and corresponding relative standard deviation of census divisions using inland site data from 2013. Only census divisions with at least 5 survey records are included.

ID #	**Name**	**Total # of surveys**	**Avg. LPD (No. pieces/km)**	**Relative standard deviation (%)**
3	Halton	11	310	80
4	Middlesex	9	404	105
5	Simcoe	6	316	84
8	Ottawa	32	586	333
17	Frontenac	6	311	74
18	York	15	290	104
20	Renfrew	6	274	94
25	Durham	15	1412	298
30	Waterloo	8	108	46
40	Toronto	40	865	201
42	Niagara	7	574	83
45	Peterborough	9	447	117
46	Peel	27	449	156

Table 3: Mean large plastic density (LPD) and corresponding relative standard deviation of lake shoreline regions using lake shoreline site data from 2013.

ID #	**Name**	**Total # of surveys**	**Avg. LPD (No. pieces/km)**	**Relative standard deviation (%)**
1	Huron-North-Superior	7	290	76
2	Huron-South	16	1060	183
3	Erie-St. Clair	11	734	104
4	Golden Horseshoe	98	795	123
5	Ontario-North	11	303	94

Table 4: Mean large plastic density (LPD) and corresponding relative standard deviation of census divisions using lake shoreline site data from 2013. Only census divisions with at least 5 survey records are included. Census divisions correspond to their ID numbers in Figure 3.X3.

ID #	Name	Total # of surveys	Avg. LPD (No. pieces/km)	Relative standard deviation (%)
1	Essex	8	424	98
2	Halton	11	654	44
3	Simcoe	7	614	78
5	Hamilton	7	294	45
8	Frontenac	5	117	49
11	Lambton	20	379	141
12	Durham	14	607	164
13	Huron	9	19	75
20	Toronto	87	958	163
22	Niagara	5	248	50
25	Peel	13	769	105

Table 5: Mean large plastic density (LPD) and corresponding relative standard deviation of census divisions using inland site data from 2014. Only census divisions with at least 5 survey records are included. Census divisions correspond to their ID numbers in Figure 3.X4.

ID #	Name	Total # of surveys	Avg. LPD (No. pieces/km)	Relative standard deviation (%)
3	Halton	8	641	69
4	Middlesex	14	97	60
5	Simcoe	11	324	141
6	Haldimand-Norfolk	8	580	63
8	Ottawa	37	261	99
13	Hamilton	6	174	112
18	York	17	209	72
25	Durham	11	274	46
26	Stormont, Dundas and Glengarry	6	289	44
30	Waterloo	8	646	189
38	Wellington	5	197	30
40	Toronto	51	481	269
42	Niagara	7	401	57
45	Peterborough	6	340	88
46	Peel	39	711	144

Table 6: Mean large plastic density (LPD) and corresponding relative standard deviation of lake shoreline regions using lake shoreline site data from 2014. Lake shoreline regions correspond to their ID numbers in Figure 3.X5.

ID #	Name	Total # of surveys	Avg. LPD (No. pieces/km)	Relative standard deviation (%)
1	Huron-North-Superior	5	481	30
2	Huron-South	42	340	183
3	Erie-St. Clair	12	336	109
4	Golden Horseshoe	137	844	159
5	Ontario-North	9	193	64