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Microplastic contamination in Lake Winnipeg, Canada☆

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ABSTRACT

Microplastics are an emerging contaminant of concern in aquatic ecosystems. To better understand microplastic contamination in North American surface waters, we report for the first time densities of microplastics in Lake Winnipeg, the 11th largest freshwater body in the world. Samples taken 2014 to 2016 revealed similar or significantly greater microplastic densities in Lake Winnipeg compared with those reported in the Laurentian Great Lakes. Plastics in the lake were largely of secondary origin, overwhelmingly identified as fibres. We detected significantly greater densities of microplastics in the north basin compared to the south basin of the lake in 2014, but not in 2015 or 2016. Mean lake-wide densities across all years were comparable and not statistically different. Scanning electron microscopy with energy dispersive X-ray spectroscopy indicated that 23% of isolated particles on average were not plastic. While the ecological impact of microplastics on aquatic ecosystems is still largely unknown, our study contributes to the growing evidence that microplastic contamination is widespread even around sparsely-populated freshwater ecosystems, and provides a baseline for future study and risk assessments.

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1. Introduction

Microplastic particles have received significant attention recently as an emerging contaminant of concern in Canada and globally (Eriksen et al., 2013; Ivar do Sul and Costa, 2014; Anderson et al., 2016). Microplastics are defined as fragments of plastic that are smaller than 5 mm in any dimension (MSFD GES Technical Subgroup on Marine Litte, 2013) and have been identified as a possible threat to aquatic ecosystems, with the potential for detrimental impacts on human health (Van Cauwenberghe and Janssen, 2014). They are considered an emerging contaminant, as research into their potential risk in aquatic ecosystems and human health are still not clear (Drewes and Shore, 2001; Younos, 2005). Recent concern around the potential ecological impacts of microplastics has led to proposed legislation to eliminate their use in the manufacturing of personal care products in the United States by July 1, 2017 (United States Congress, 2015). In Canada, they were listed recently as a toxic substance under Schedule 1 of the Canadian Environmental Protection Act (Canada Gazette, 2016). Australia and the UK have enacted “voluntary” elimination by industry as early as 2017, and many manufacturers have begun removing them from their products in advance of legislative action.

Microplastics have been documented in water bodies worldwide. In marine ecosystems, microplastic contamination has been reported around the world in sediment, open water, and in organisms (Ivar do Sul and Costa, 2014). Observations in freshwater are more rare and have been identified as a significant data gap, especially in Canada (Anderson et al., 2016). The Laurentian Great Lakes have recently been shown to have significant densities of microplastics in surface waters (Eriksen et al., 2013), as have other more remote lakes in Asia (Free et al., 2014) and Europe (Faure et al., 2015). In addition, microplastics have been reported in sediments of Lake Ontario and the St. Lawrence river (Castañeda et al., 2014;

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Corcoran et al., 2015; Ballent et al., 2016). These studies on larger systems beg the question as to the extent of microplastic contamination in other inland lakes of similar size (Wagner et al., 2014). Within marine and freshwater food webs, microplastics have been detected in the guts of organisms at nearly every trophic level, from copepods and crustacean zooplankton to filter-feeding invertebrates, and in pelagic and demersal fishes (Thompson et al., 2004; Eerkes-Medrano et al., 2015; Lusher et al., 2013). It is hypothesized that microplastics can affect the physiological functioning of animals, either through leaching of accumulated organic pollutants into the stomach lining of fishes that consume them, physical blockage of the digestive system, or simply by taking up space that could otherwise be occupied by food (Bakir et al., 2014; Wright et al., 2013). If microplastics are capable of facilitating bioaccumulation and/or biomagnification of harmful contaminants (Koelmans et al., 2013), then there is potential for adverse effects on humans, especially for the one fifth of the world's population that relies on fish and seafood as their primary animal protein source (Sumaila et al., 2007).

There are two sources of the microplastics commonly found in aquatic systems: primary and secondary microplastics (MSFD GES Technical Subgroup on Marine Litter, 2013). Primary microplastics are those that were produced for use in a wide variety of consumer and industrial applications (e.g., abrasives in cosmetic scrubs). Secondary microplastics originate from larger plastic and synthetic materials when they are broken down by weathering processes such as UV degradation or machine washing (MSFD GES Technical Subgroup on Marine Litter, 2013). In marine systems, secondary microplastics make up the bulk of the microplastic particles, especially in areas with high population densities (Hidalgo-Ruz et al., 2012; Browne et al., 2011). Recent studies have demonstrated that wastewater treatment plants are significant release points of microplastic particles, primarily fibres (Mason et al., 2016; Anderson et al., 2016; Murphy et al., 2016). The relative importance of primary and secondary microplastics in freshwater systems is not well documented; primary plastics ("pellets" or microbeads) dominated the smallest size category (<1 mm) in the Great Lakes, whereas secondary plastics (fragments) dominated larger size classes (>1 mm; Eriksen et al., 2013). Many common forms of microplastics in consumer products (e.g., polypropylene, high/low density polyethylene) have densities less than water (Eerkes-Medrano et al., 2015), and polypropylene and polyethylenes have been shown to be highly susceptible to microparticle generation through breakdown (Weinstein et al., 2016; Zbyszewski and Corcoran, 2011). As such, these types of microplastics have the potential to be generated by larger plastic litter, as well as be carried long distances through watersheds owing to their densities, making their study in Lake Winnipeg especially relevant.

Lake Winnipeg is the fifth largest Canadian lake, and has the second largest watershed in Canada at over 982,000 km^2, in which nearly 7 million people live and work (roughly 20% of the Canadian population). The watershed is approximately 40x larger than the surface of the lake, and transports water from across four Canadian provinces and four US states, making the condition of the lake very much a function of the events that occur within the watershed (Schindler, 2009). Our objective was to report initial estimates of microplastic contamination in Lake Winnipeg, and formally compare these results to reported concentrations in other large Canadian Lakes (Eriksen et al., 2013). This work serves an important first step towards filling the literature gap around sources and distribution of microplastics in Canadian freshwater ecosystems, and understanding the comparative extent of microplastic pollution across large freshwater bodies in Canada.

2. Methods

Lake Winnipeg is a shallow (mean depth 12 m), well-mixed (3to 5-year residence time) lake with low visibility (Secchi depths typically < 1 m). It is subject to wind mixing and frequent algal blooms due to cultural eutrophication (Waasenaar and Rao, 2012). Surface waters during relatively calm conditions (winds between 4 and 18 knots) were sampled at twelve locations (Fig. 1) in Lake Winnipeg between July 25th and October 2nd, 2014; from June 2nd to June 17th, 2015; and from June 3rd to June 19th, 2016 using standard collection methods (Eriksen et al., 2013). Specifically, a manta trawl (61 cm wide by 18 cm high) was employed with a 3 m long, 333 μm mesh bag and 333 μm removable cod-end. The net was trawled along side the research vessel MV Namao using a fixed crane arm. Tow times ranged between eight and thirty minutes. Because sampling primarily occurred during the summer months when algal blooms are common in Lake Winnipeg, the net was closely monitored during towing to ensure the net did not clog. If the net appeared to be nearing a state where water flow through the net was becoming impeded, the tow haul was stopped and the net was retrieved. The distance sampled by the net was estimated as the distance travelled over the GPS start and end points of the ship, as recorded from the on-board ship computer. Tow speeds were between 2.2 and 3.75 knots. Once the net was retrieved, all collected material was concentrated into the cod-end of the net and transferred to a glass sample jar. The net (with the cod-end removed) and cod-end separately were rinsed thoroughly with water from the ship's hose between deployments to avoid cross-contamination. The collected material was then preserved in 70% ethanol until processing in the laboratory. Trawl sites were chosen opportunistically to coincide with existing sampling sites visited by the MV Namao, but also selected to represent a range of offshore and nearshore sites in both the north and south basins of the lake. Trawls varied from 530 m to 3780 m in length. Surface densities of microplastics were estimated from the trawl distance multiplied by the width of the manta trawl net.

In the laboratory, samples were filtered through a 250 μm mesh brass sieve. Coarse debris (e.g., hand-sized twigs or leaves) were thoroughly rinsed with deionized water (DI) and removed; typical rinsing time was approximately five minutes per sample. The remaining material on the sieve was then reconstituted to a known volume (typically 1250 mL). Rinsing of large debris in this manner may have resulted in some losses of strongly-attached plastic particles, therefore our results might be considered conservative estimates. The reconstituted sample was stirred using a stirring plate to achieve uniform consistency. A subsample of known volume (typically 250 mL, or 1/5 of the reconstituted sample) was collected by submerging the subsample vessel vertically in the centre of the bulk sample and removing it vertically. The subsample was then processed using a wet peroxide oxidation (WPO) treatment (Masura et al., 2015; Mason et al., 2016). Each 250 mL sub-sample received 20 mL of a 0.05 M Fe (II) solution and 20 mL of 30% H_2O_2 while on a 75 °C stirring hotplate for 30 min and was covered loosely with tinfoil to avoid atmospheric deposition during the reaction (but allowing the sample to de-gas). The Fe (II) solution was prepared by adding 7.5 g of $FeSO_4 \cdot H_2O$ to 500 mL of water and 3 mL of concentrated sulfuric acid. Additional 20 mL aliquots of 30% H_2O_2 were added to the subsample as needed throughout the stirring and heating process to help degrade organic material in the samples. Subsamples were left covered to digest for 24 h. Digested samples were once again filtered through a 250 μm brass sieve, reconstituted in DI water and were processed. During these procedures, synthetic fibres were not worn by the individuals handling samples. We visually identified particles remaining from these samples that were putatively considered to be plastic based on

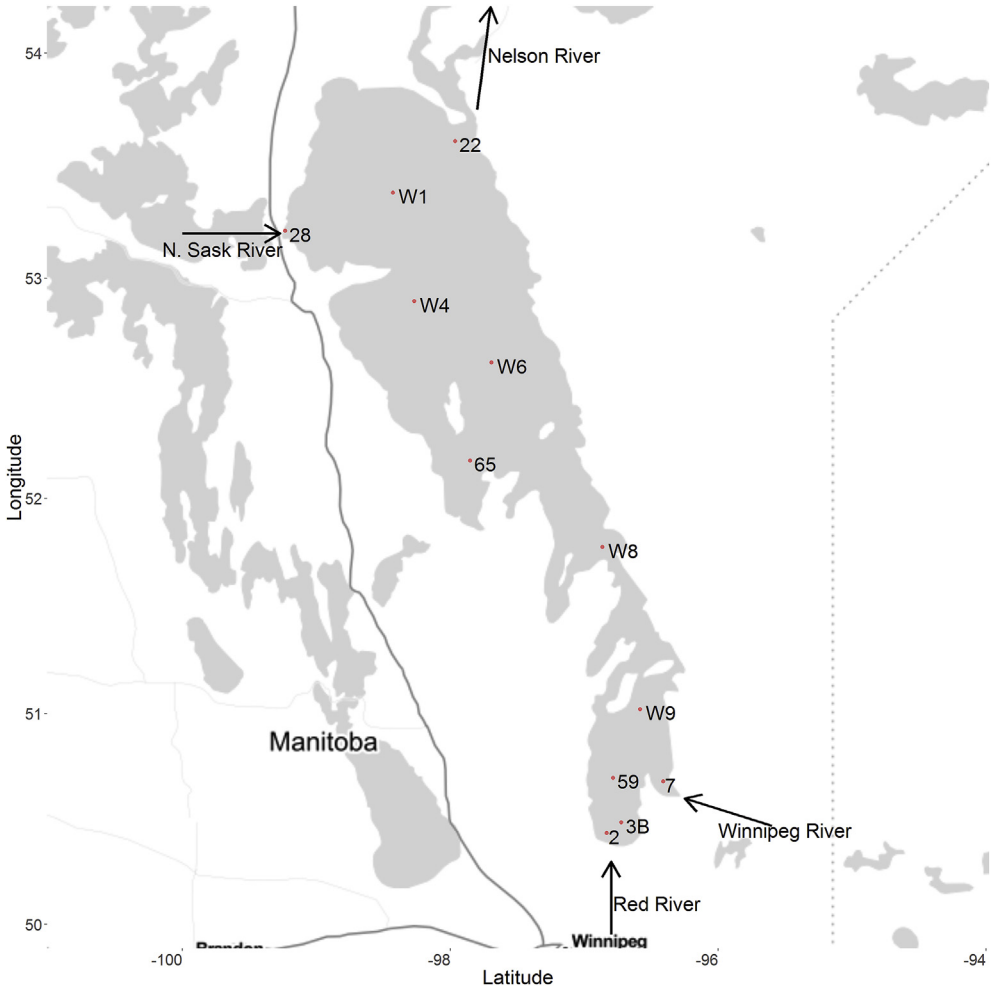

Fig. 1. Sites sampled for microplastics on Lake Winnipeg, Canada.

shape and size using a dissecting microscope (6.3–50.0x magnification); particles were all less than 5 mm and typically larger than the mesh size of our net (333 μm) in at least one dimension. A count of the number and type of microplastic particles was recorded. Microplastic particles were transferred to ethanol in a glass vial with a rubber stopper for later scanning electron microscopy (SEM) identification. Our search pattern for types of microplastics were categorized into five groups: fragments (hard, jagged-edged particles), micro-pellets (hard, rounded particles), fibres (fibrous or thin uniform plastic strands), films (thin, 2-dimensional plastic films), and foam (i.e., Styrofoam-type material).

We evaluated our subsampling protocols by conducting two subsamples in duplicate (sample W9 and 59, 2014). In each case, total visual counts of plastics were similar (sample W9: 49 and 59 particles; sample 59: 54 and 70 particles).

Laboratory blanks were conducted to determine possible contamination from either deposition from air or DI water. DI water blanks were conducted by running the DI water tap at the University of Manitoba for 60 min on a clean 333 μm brass sieve which delivered water at a rate of 8 L per minute (480 L total). Four replicates of this procedure yielded counts of 13, 5, 16 and 9 particles (fibers), suggesting that on average 1 fiber is introduced for every 48 L of DI water applied to samples. Given an average rinse of five minutes with DI water (at 8L per minute), plus a reconstitution to 1.25 L prior to subsampling, we estimate that 0.85 fibers were introduced to our samples, on average, from DI water alone. Air blanks were conducted by leaving a clean beaker of water out on the lab counter for 24 h. Duplicate air blanks yielded 8 and 7 fibers over this time period, or 0.3 fibers per hour. Average time for visual sorting of samples was 4 h. We estimate 1.25 on average were introduced from lab air. Therefore, 2 fibers per sample were likely introduced due to laboratory handling procedures, and all sample counts were reduced by this value.

A subset of 2014 samples (9 of 12 total) identified visually as plastic were examined for their elemental composition using SEM and Energy Dispersive X-ray Spectroscopy (EDS; following Eriksen et al., 2013). Between 8 and 32 particles from each sample were chosen at random and placed on double-sided carbon tape, coated with a thin film of evaporated carbon under vacuum and then imaged using a Hitachi SU-70 field emission SEM operating at 20 KV in backscatter mode. Qualitative elemental composition of particles was confirmed using an Oxford AZtec Energy Dispersive X-ray Spectroscopy system (EDS). We used both point and areal EDS scans of samples, as appropriate (based largely on sample size), to determine elemental composition. To help characterize particles as plastic or not, we also analyzed a number of known plastic samples (shopping bag, sealable plastic bag, Styrofoam, plastic fork, plastic film), weathered plastic samples collected from Lake Tamblyn, Ontario (shopping bag, water bottle, sealable plastic bag) and rubber stoppers used to hold sorted plastic particles (see Supplemental Information). We also characterized samples of cultured *Daphnia magna* obtained from the Lakehead University

Aquatic Toxicology Research Centre, to ensure that organic particles could be differentiated from plastics (see Supplemental Information). *Daphnia* was chosen because cladoceran carapaces proved to be the most resistant biological material to WPO treatments in our samples; as such, we wished to be able to differentiate carapaces (or fragments thereof) from putative plastic particles. We corrected microplastic counts from all samples based on the proportion of non-plastic particles identified using SEM-EDS.

Statistical analyses were performed using R (ver. 3.2.3, R core team 2015). Visualization of microplastic density in the lake was mapped using the packages 'ggmap' and 'ggplot2' (Kahle and Wickham, 2013; Wickham, 2009). The density of microplastic contamination between the north and south basins over years was examined using two-way ANOVA. We used two-way ANOVA to compare differences between nearshore and offshore sites over years (Table 1), and between sites near northern (site 28) and southern inflows (sites 2, 3B and 7) over years of sampling. Microplastic densities were log-transformed to satisfy the assumption of normality. Finally, we compared densities of microplastics in Lake Winnipeg to those reported in the Great Lakes (Eriksen et al., 2013, their Table 1) using a Kruskal-Wallis test on ranks followed by Dunn's test for multiple comparisons with a Bonferroni adjustment.

3. Results

Microplastic particles were found in all samples collected, in all years (Fig. 2). On average, 23% of particles visually identified as plastic were determined by SEM-XDS to be either silicates, iron oxide (rust) or paint flakes off the vessel (Supplemental Information). All sample counts across years were adjusted using this value (i.e., total densities x 0.77). Adjusted densities of blank-corrected microplastics ranged from 748,000 particles/km^2 at station 22 near the outflow of Lake Winnipeg in 2014 to 53,000 particles/km^2 at station 7 in the Winnipeg River Inflow in 2014 (Table 1, Fig. 2). The most common type of particles across all sites were overwhelmingly identified as fibres, and films and foam were the least common over all three years of data (Fig. 3). No micropellets or beads were identified over the three years of sampling conducted over all 12 stations on the lake.

There was a significant interaction between year of collection and basin (north and south) for microplastic densities in Lake Winnipeg (two-way ANOVA: $F_{2,29} = 3.74$, $p = 0.036$). Densities of microplastics were significantly greater in the north basin than in the south basin in 2014 (Tukey HSD test, $p < 0.05$), but not in 2015 or 2016 ($p > 0.05$, Fig. 4). We found no evidence of differences between nearshore and offshore sites across all years ($p > 0.05$). Similarly, there were no statistically significant differences between microplastic densities among southern and northern sites nearest inflows across years (two-way ANOVA, $p > 0.05$). There was no effect of wind velocity (linear regression, $p > 0.05$) or direction (linear regression, $p > 0.05$) at the time of sample collection on our microplastic densities.

Significant differences in microplastic densities were found between Lake Winnipeg and those previously reported in the Great Lakes (Kruskal-Wallis test, $X^2 = 31.3$, $p < 0.0001$; Fig. 5). Lake Winnipeg microplastic densities were comparable to those reported in Lake Erie ($p > 0.05$), but greater than those reported for Lake Huron ($p < 0.0001$), and in western Lake Superior ($p = 0.003$).

4. Discussion

Microplastics are present in Lake Winnipeg at densities similar to or greater than those currently reported in the Great Lakes basin. Values were significantly greater than those reported on Lakes Superior and Huron, and similar to those reported for Lake Erie (Eriksen et al., 2013). Lake Winnipeg also had the highest total estimated quantity of microplastic (areal densities multiplied by the lake surface area) compared with any of the Great Lakes surveyed (Table 2). However, comparisons of our data to those reported from the Great Lakes should be made with some caution, as those published densities for the Great Lakes are based on fewer samples, are from only a single year of sampling (compared to three years of data reported here), and are from a limited range of sites (e.g., Lake Superior samples are for Whitefish Bay only, and it is unclear how representative those observations are for the entire lake, especially in open water). Regardless, the similarity in microplastic densities between Lake Winnipeg and Lake Erie is surprising and should be of concern for those charged as stewards of Lake Winnipeg. Lake Winnipeg has a surface area, mean depth and residence time that is most similar to Lake Erie when compared with either Lake Superior or Lake Huron (Table 2). However, the Lake Erie catchment supports a population base of nearly double that supported by Lake Winnipeg, in a watershed less than 1/10th the size. If we assume that municipalities represent the major source of microplastics in freshwater (Mason et al., 2016), then the fewer municipalities feeding wastewater to Lake Winnipeg compared to Lake Erie suggest that (a) municipalities in the Lake Winnipeg watershed could be generating more microplastics than those around Lake Erie, and that (b) long-range transport of microplastics is facilitated on the surfaces of freshwater rivers. Alternatively, other secondary sources of microplastics may be present in Lake Winnipeg that are not found in Lake Erie, but this seems unlikely given the intensity of industrial and commercial

Table 1
The location, microplastic density (blank-corrected), and description of Lake Winnipeg Research Consortium research stations used for this study. Latitude and longitude are decimal degrees. n/s = no sample taken.

Station	Densities (# per km^2)			Latitude	Longitude	Basin	Type	Description
	2014	2015	2016					
22	748,027	140,814	121,597	53.61,286	97.96,458	North	Nearshore	Norway House (near outflow)
28	231,239	191,598	186,276	53.21,349	99.23,345	North	Nearshore	Grand Rapids
65	249,208	262,436	221,902	52.17,458	97.85,156	North	Offshore	Near Dauphin River
W1	193,153	184,941	169,562	53.38,208	98.44,235	North	Offshore	Between Grand Rapids and outflow
W4	225,852	266,001	293,449	52.89,907	98.24,265	North	Offshore	North of Reindeer Island
W6	n/s	169,475	66,788	51.62,056	97.69,335	North	Offshore	East of reindeer island
W8	283,132	69,167	98,084	51.77,651	96.86,408	North	Offshore	Central Narrows
2	108,034	139,467	279,161	50.43,604	96.83,423	South	Nearshore	Red River (East)
7	52,508	187,805	171,741	50.68,053	96.41,248	South	Nearshore	Winnipeg River
59	188,186	157,787	217,921	50.69,880	96.78,612	South	Offshore	Between Gimli and Victoria Beach
3B	161,893	215,514	148,505	50.48,764	96.72,343	South	Nearshore	Red River (West)
W9	174,438	95,628	98,416	51.02218	96.58,377	South	Offshore	Northern tip of south basin

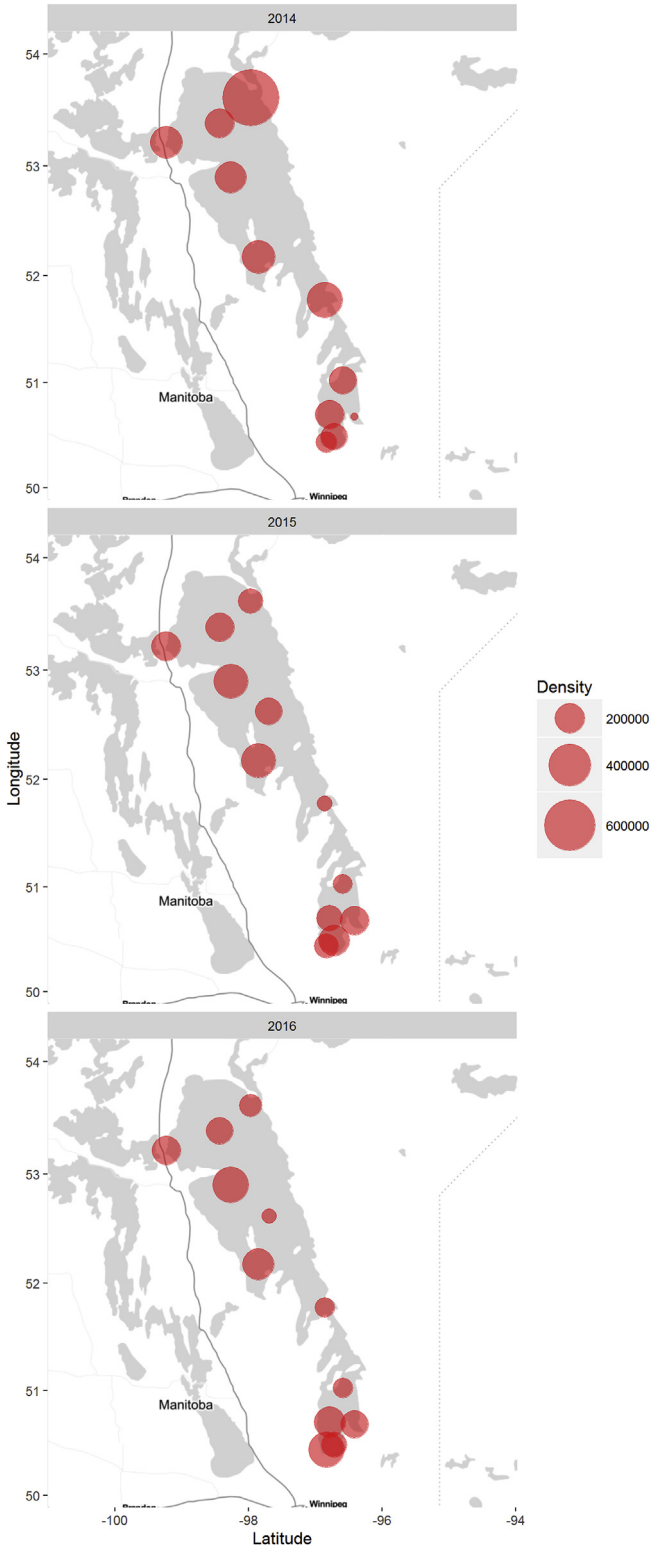

Fig. 2. Spatial distribution of microplastic density (number of particles/km^2) across sampling stations between 2014 and 2016 on Lake Winnipeg, Manitoba. Samples were collected from eleven stations in 2014 and twelve stations in 2015 and 2016.

activity in Lake Erie relative to Lake Winnipeg. Our results certainly warrant a closer investigation of municipal microplastic inputs and studies into the long-range transport of these contaminants in large river systems connected to Lake Winnipeg. Densities of microplastics were also elevated relative to many Swiss lakes, but similar to those reported for Lake Maggiore and Lake Geneva (Grand Lac), both of which support much smaller populations in their watersheds than Lake Winnipeg (Faure et al., 2015).

Densities of microplastics were significantly elevated in the north basin of Lake Winnipeg in 2014, but not in subsequent years. It is possible that greater densities in the north basin observed in 2014 may be due to greater delivery rates from the inflow at Grand Rapids during this year. The inflow at Grand Rapids delivers water from the Saskatchewan River, which collects wastewater from all major communities in Alberta and Saskatchewan. Considering only major cities, (Edmonton, Calgary, Red Deer, Lethbridge and Medicine Hat in Alberta; Saskatoon in Saskatchewan), the Saskatchewan River delivers water from municipalities totalling 2.3 million people, compared to just over 1 million from major centres in the Red River (Regina, Brandon, Winnipeg, Grand Forks and Fargo) and Winnipeg River drainages. The Winnipeg River flows from its headwaters in Northwestern Ontario to Lake Winnipeg, but does not pass any major population centers, a common source of microplastics (Leite et al., 2014). Typical discharges from the Saskatchewan River (mean monthly rate of 556 m^3/s; Manitoba Water Stewardship, 2011) amounts to approximately 25% of the total inflow into the lake. The Winnipeg River (mean monthly rate of 1064 m^3/s; Manitoba Water Stewardship, 2011) contributes almost 50% of the total inflowing water to the lake and the Red River is the smallest of the three main tributaries delivering only 16% of the total lake inflow (346 m m^3/s; Manitoba Water Stewardship, 2011).

It is also possible that internal currents in 2014 may have also acted to help concentrate microplastics in the north basin of Lake Winnipeg. Very little is known about prevailing currents in Lake Winnipeg, other than the intense action that wind can have on mixing in the lake (Manitoba Water Stewardship, 2011). However, satellite imagery is certainly suggestive of potential gyres and circulation patterns within the north basin (Abrahams et al., 2007), which could act to concentrate floating debris like microplastics in the north basin. Timing of collection (fall in 2014 versus summer in 2015–16) may also have had an influence on microplastic densities; perhaps conditions in the fall act to generate inter-basin differences compared with conditions in the lake during the summer. Additional fall samples and/or an examination of samples taken at multiple time points through the year are required to better understand possible seasonal dynamics of microplastic particles in Lake Winnipeg. Neither wind velocity nor direction at the time of sampling explained microplastic particle densities. This is perhaps because sampling with the manta trawl requires relatively calm conditions to function properly.

Of the particles we detected in our study, the majority were fibres. The bead-like particles which have been cause for such great concern in the popular media, and in surface waters in the Great Lakes (Eriksen et al., 2013) do not appear to be a significant source of microplastic contamination in the surface waters of Lake Winnipeg. Rather, it would seem that the majority of microplastics in the lake are from either synthetic textiles, the breakdown of larger particles, or atmospheric fallout. Atmospheric fallout has been shown to generate between 30 and 300 particles per m^2 per day in Paris, 90% of which were identified as fibres (Dris et al., 2015). Paris is a metropolitan area that supports nearly 12 million people, and Lake Winnipeg supports no major settlements on its shores; while long-range atmospheric transport of microplastics from major centres like the city of Winnipeg (60 km south) is possible, it would likely be more than an order of magnitude less than that reported in Dris et al. (2015) based on population alone within the city (approx. 700,000), and losses over long-range transport are expected (though unquantified). Large particle breakdown has been proposed as a major source of microplastics in freshwater and oceanic

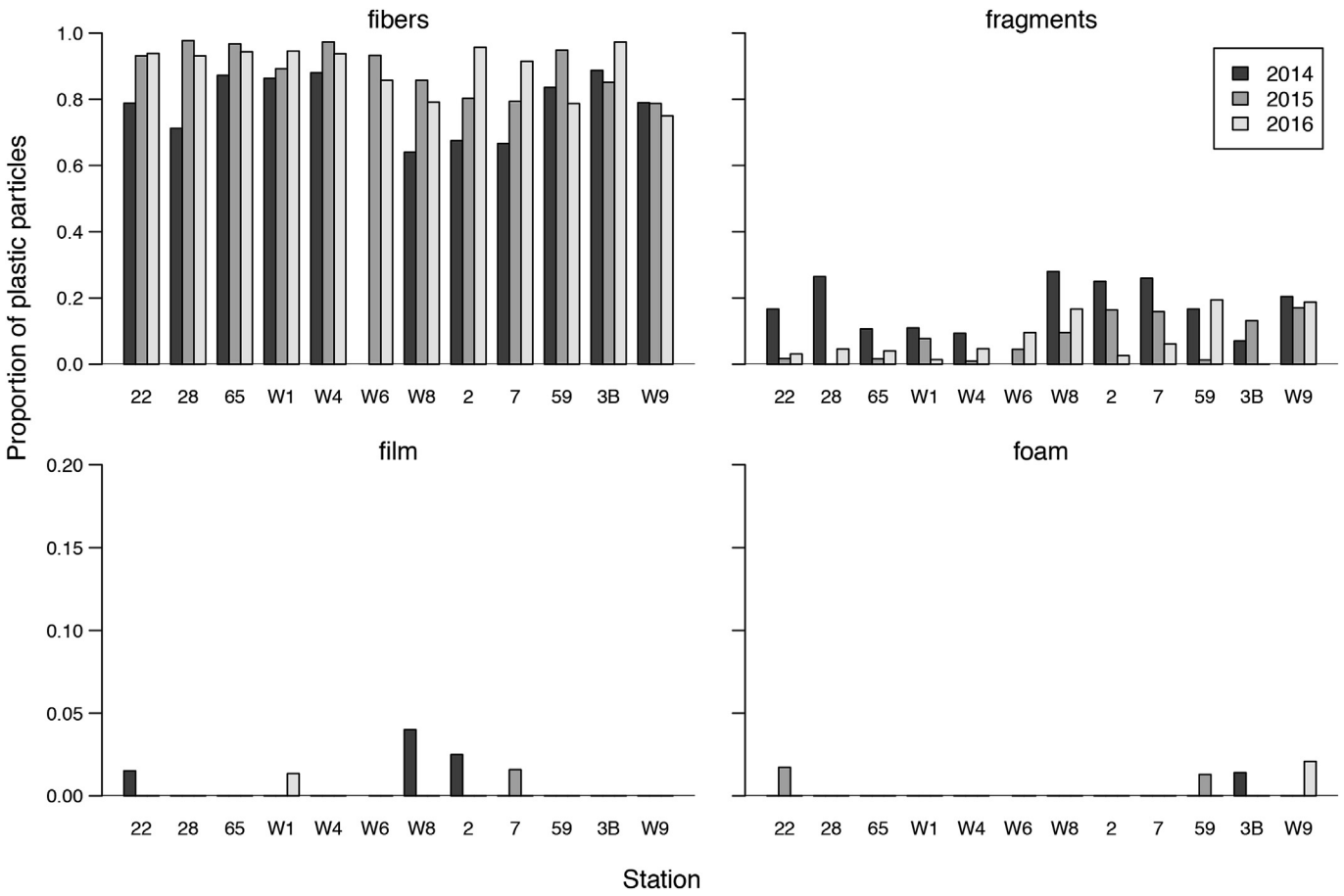

Fig. 3. The proportion of the four types of microplastic particles (fibres, fragments, film and foam) found at the twelve stations across Lake Winnipeg over all years (2014–2016). Note difference in y-axis scaling for lower panels. North basin stations are 22, 28, 65, W1, W4, W6, W8; South basin stations are 2, 7, 59, 3B and W9.

environments (Zbyszewski and Corcoran, 2011; Hidalgo-Ruz et al., 2012). Fibres from synthetic materials would be expected to have high concentrations in municipal wastewaters compared with natural environments (Hartline et al., 2016; Mason et al., 2016). A comparison of microplastic densities and prevalent types near municipal wastewater inputs compared to the types of large particle plastics found in surveys of shoreline debris and/or large particle inputs from inflows may help provide a better understanding of sources of microplastics in Lake Winnipeg and elsewhere.

Our results suggest a potential influence of population density on microplastic densities in some years; specifically, greater microplastic densities in 2014 may be a consequence of greater inputs from the Grand Rapids inflow, which supports the largest population densities in the watershed. Free et al. (2014) showed that proper waste management may play a more important role in microplastic contamination than population size; Lake Hovsgol, a remote mountain lake within a national park in Mongolia, had alarmingly-high levels of microplastic contamination given its isolated location, which the authors ascribe to the breakdown of on-shore plastic on the lake. An average density of 20,264 particles/km^2 (less than that reported here for Lake Winnipeg, but greater than reported for lakes Huron and Superior, Table 2) was reported for Lake Hovsgol, with a low population density in the in surrounding area (Free et al., 2014). While the currently poor state of waste management practices may have led to the elevated levels of microplastics observed in Lake Hovsgol, it would be surprising if waste management practices vary dramatically among the watersheds of Lake Winnipeg. Agricultural inputs are greatest from the Red River based on phosphorus loading estimates (Schindler et al., 2012).

While the sampling in this study is far from a comprehensive survey of microplastic distribution in Lake Winnipeg, it is more extensive than published surveys conducted on the Great Lakes (Eriksen et al., 2013). By providing an initial assessment of microplastics on the lake, we hope to spur future research that will help to determine sources, transport, effects and seasonal dynamics of plastic contamination in Lake Winnipeg, which is as great or greater than that currently reported on the Great Lakes despite having a much smaller population base.

Anecdotal evidence from Lake Winnipeg suggests that current sampling methods for floating microplastics (both here and elsewhere) may be missing a significant fraction of actual microplastic particles in the lake. The standard manta trawl net used for sampling microplastics is 333 μm (Eriksen et al., 2013, 2014). However, whole-water samples from at least two parts of Lake Winnipeg have resulted in documented plastic particles, which appear to be small enough to escape through the mesh size of the currently used Manta trawl configuration for assessing microplastic contamination (Fig. 6). As such, it should be emphasized the current estimates of microplastic densities in freshwater lakes in Canada (i.e., our study and those of Eriksen et al., 2013) are largely of particles greater than 333 μm, and are very likely underestimates of microplastic contamination in these systems (Anderson et al., 2016). While microplastics in this size range may be of significance to fish if ingested, smaller particles may have ecological

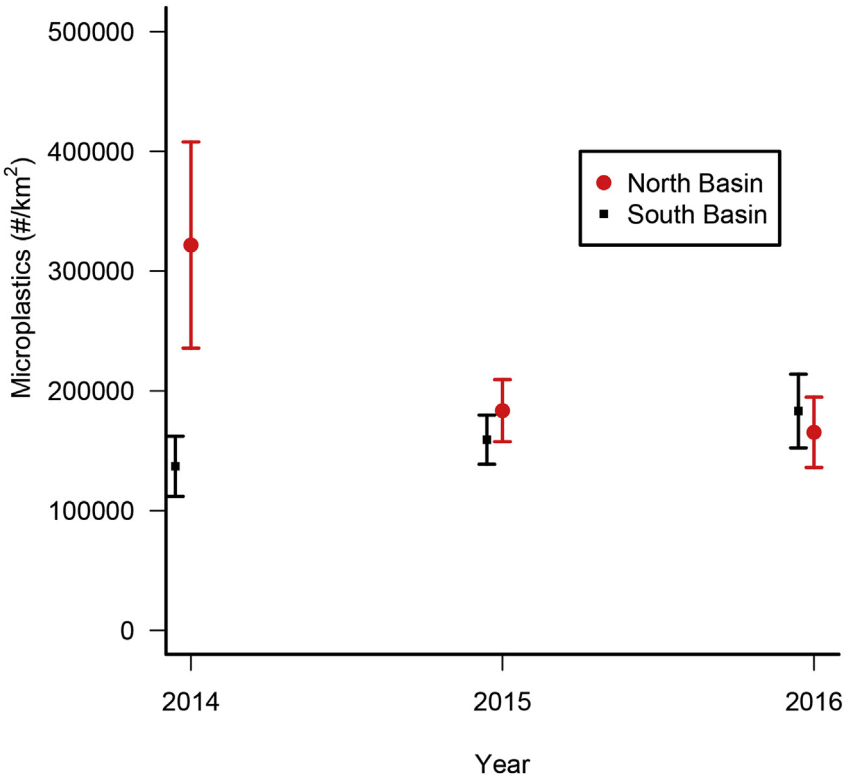

Fig. 4. Significant interaction between year sampled and mean densities of microplastics in the north and south basins of Lake Winnipeg (two-way ANOVA: $F_{2,29} = 3.74$, $p = 0.036$). Means plus or minus 1 standard error are presented.

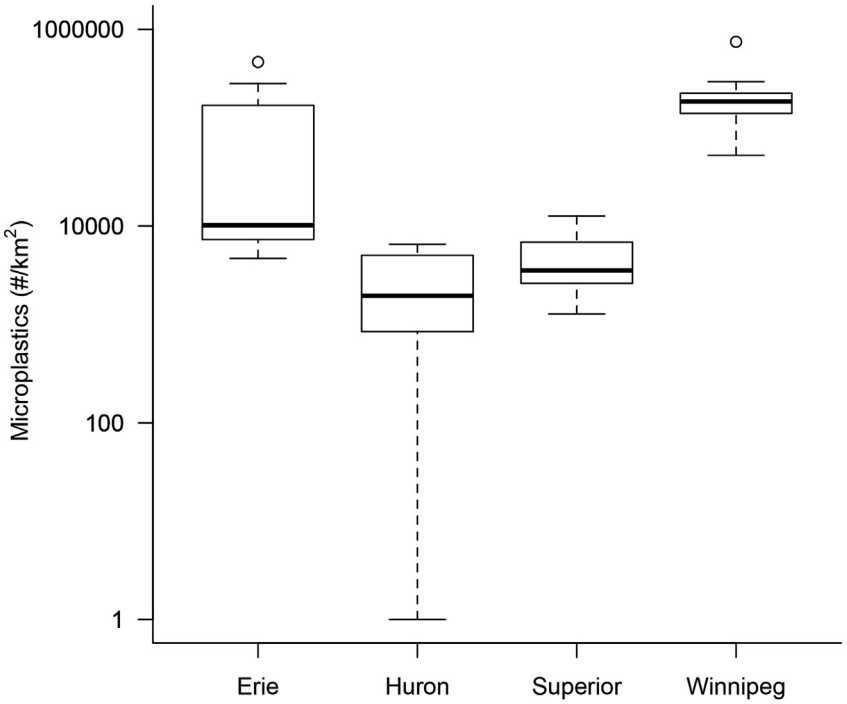

Fig. 5. Lake Winnipeg microplastic densities are similar to Lake Erie, but elevated compared to values reported on Lake Huron and Lake Superior (Kruskal-Wallis test, $X^2 = 31.3$, $p < 0.0001$). Great Lakes data are from Eriksen et al. (2013). Data presented as boxplots, error bars are 95% confidence intervals, while the box represents the interquartile of data (0.25 and 0.75 quartiles); median values are bold, horizontal lines. Note log-scale on the y-axis.

Table 2
Characteristics of Lake Winnipeg relative to other Laurentian Great Lakes with reported densities of microplastic.

	Lake Winnipeg	Lake Superior	Lake Huron	Lake Erie
Surface area (km^2)	24,514	82,100	59,588	25,744
Mean depth (m)	12	147	59	19
Residence time (years)	3–5	191	22	2.6
Population in watershed	7,000,000	600,000	2,500,000	12,000,000
Watershed area (km^2)	982,900	127,700	134,100	78,000
Mean microplastic density # per km^2 (±standard deviation)	193,420 (±115,567)	5391 (±4552)	2779 (±2440)	105,503 (±173,587)
Total estimated surface microplastic	4.74×10^9	4.43×10^8	1.66×10^8	2.72×10^9

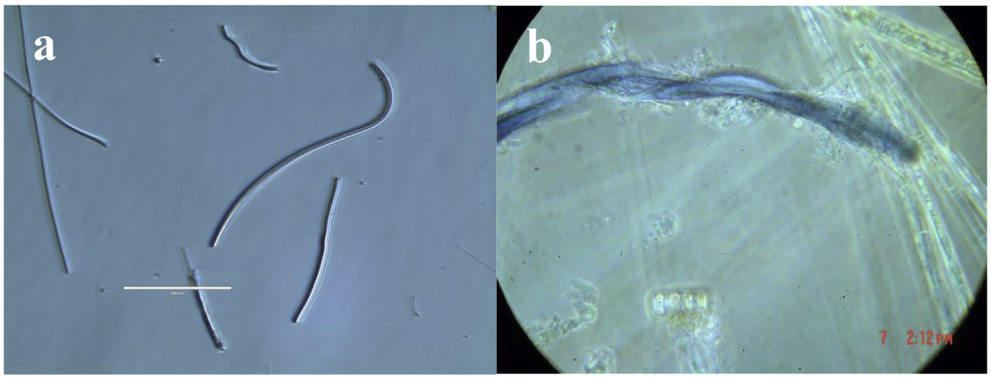

Fig. 6. Images of microplastic material found in Lake Winnipeg from whole-water samples taken (a) near Grand Rapids (inflow of the North Saskatchewan River) and (b) a 10-μm phytoplankton haul from south of Black Island in the south basin (0–3 m depth). Whole-water samples were taken as a 2L surface grab with a wide-mouth bottle at 0–0.5 m. Photo credit: (a) M. Stainton, (b) H. Kling. Scale bar in panel (a) is 100 μm; Particle in panel (b) measured approximately 300 μm long by 30 μm wide (H. Kling, personal communication). Both particles would pass through traditional manta trawl nets given the proper orientation.

relevance to filter-feeding zooplankton (e.g., Cole et al., 2014). We strongly recommend future work that employs tandem sampling of nets with different mesh sizes to help better characterize microplastics smaller than those obtained by current methods, to the extent possible, given limitations of obtaining sufficient water volumes while avoiding net clogging.

It is possible that the WPO methods we employed, though widely adopted in the literature as a standard method (Masura et al., 2015), may have altered or potentially digested some of the material in our samples. Chemical digestion methods reported elsewhere using acids and alkaline digestion methods have noted degradation and potential digestion of fine particles, especially fibres (Claessens et al., 2013; Cole et al., 2014). To our knowledge, no similar evaluations have been conducted on WPO treatments of known plastic samples. Certain low-density polymers such as nylon and low-density polyethylene (LDPE's) are known to be reactive to 30% H_2O_2 (chemical compatibility database, Cole-Parmer, https://www.coleparmer.com/Chemical-Resistance, accessed 22 Jan 2017). However, it is unclear if nylon or LDPE's currently pose significant environmental concern. By comparison, PET, the compound used in polar fleece, which is of significant concern with regards to potential for environmental contamination through fibre shedding (Hartline et al., 2016) is highly resistant to 30% H_2O_2 (Plastics Europe, www.plasticseurope.org, accessed 22 Jan 2017). Ultimately, our choice of digestion by WPO was guided entirely to facilitate a comparison of our results with those reported elsewhere in North America (Eriksen et al., 2013), who used similar methods and allowed us to demonstrate similar contamination levels in Lake Winnipeg as those observed previously in the Laurentian Great Lakes.

In summary, we have demonstrated persistent microplastic contamination in Lake Winnipeg over a three-year time period, and identified these particles as consisting largely of fibres. It is of significant relevance that concentrations reported in Lake Winnipeg are comparable to those observed on the Laurentian Great Lakes. Our work provides an important baseline for future studies, and is an important first step for future investigation focused on the identification of inputs of microplastics into the lake and rates of deposition (e.g., transport from rivers draining the lake and the importance of long-range transport from municipalities, atmospheric deposition, etc), the quantification of the presence of this contamination on aquatic biota and potential impacts on organisms.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.envpol.2017.02.072.

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