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Do human activities affect phytoplankton biomass and composition in embayments on Lake Diefenbaker?

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ABSTRACT
Lake Diefenbaker (LD) is an important source of water for southern Saskatchewan. LD is characterized by numerous embayments containing anthropogenic activities (e.g., housing, marinas, cattle watering). Many of these activities are increasing on this important reservoir in association with the rapidly developing economy of Saskatchewan. These activities may reduce water quality directly or indirectly by encouraging the growth of nuisance algae (i.e. cyanobacteria). Here, we examined phytoplankton biomass and composition in eight embayments exposed to anthropogenic activities, four unexposed embayments with no perceived human activities and six main channel sites adjacent to the embayments from June to October (2011 and 2012). Phytoplankton biomass and composition was not significantly different in exposed, unexposed embayments and main channel sites ($p > 0.05$), with the diatoms and cryptomonads constituting 87%—91% of the total phytoplankton biomass in both years. High flows from the South Saskatchewan River (SSR) in both years may have resulted in the rapid flushing of the embayments and dampened any localized impacts that could have resulted from anthropogenic activities as found in other studies. Hence, future study on LD should be conducted during years with low flow from the SSR when the rate of flushing of embayments will be reduced.

KEYWORDS
Phytoplankton; embayments; flow; water quality; reservoir

Introduction
Poor water quality is often associated with high algal biomass and algal blooms, which can lead to many issues, including low dissolved oxygen (DO) concentrations and fish kills (Schindler et al. 2008; Smith et al. 2008). Excessive algal biomass and algal blooms can have large ecological impacts resulting in the deterioration of habitat and the loss of biodiversity at all trophic levels. Blooms consisting primarily of cyanobacteria present additional concern because of the potential for certain species of this group to synthesize secondary metabolites (e.g. geosmin) that can cause taste and odor issues in drinking water. More importantly, certain cyanobacterial species produce cyanotoxins that may pose serious health risks to humans and livestock (Landsberg 2002; Izaguirre & Taylor 2004; Paerl & Huisman 2008, 2009). All of these water quality issues may culminate into large economic impacts, including the cost of water treatment and loss in tourism (de Figueiredo et al. 2006; Dodds et al. 2009).
There are numerous studies that document the run-off of nutrients such as ammonium (NH\textsubscript{4}\textsuperscript{+}), nitrate (NO\textsubscript{3}\textsuperscript{−}), organic nitrogen and dissolved phosphorus into lakes and reservoirs from urban and agricultural practices in watersheds. Sources of these nutrients include housing development, marina, golf courses, livestock, fertilizer application on cropland and run-off from turf grass maintenance (Howarth et al. 1996; Mallin & Wheeler 2000; Winter et al. 2003; Kuo et al. 2008). Such nutrient inputs may affect the phytoplankton community composition and may increase the intensity and frequency of algal blooms in lakes and reservoirs (Jeppesen et al. 2007; Carney 2009; Smith & Schindler 2009; Michalak et al. 2013; Lehman 2014). For example, Katsiapi et al. (2012) found strong correlations between phytoplankton species composition and a variety of land use activities in 11 lakes and 7 reservoirs in Greece. The authors reported that cyanobacteria were associated with urban and agricultural activities; conversely, euglenophytes were associated with industrial and commercial land use activities. Paul et al. (2012) also found a strong positive correlation between cyanobacteria and land used for livestock pasture and a strong positive correlation between chlorophytes and native forest and urban land use in 11 lakes in New Zealand. Paleolimnological analyses revealed that nuisance cyanobacterial species and diatom species, indicative of eutrophic conditions, were strongly related to cropland area, livestock and urbanization after European settlement in Pasqua Lake, in the Northern Interior Plains of Canada (Hall, Leavitt, Quinlan, et al. 1999)

Several studies have reported greater nutrient concentrations in embayments compared to the main channel of lakes and reservoirs (Mitrovic et al. 2001; Poste et al. 2012; Mbonde et al. 2015). This is partly because embayments are usually shallow and sheltered from the main channel of lakes or reservoirs. Hence, embayments often experience low flushing or longer water residence time (Mitrovic et al. 2001; Hall et al. 2003; Izydorczyk et al. 2008). Consequently, the combined effects of nutrients released from anthropogenic activities in embayments and their longer residence time may encourage rapid algal growth and nuisance algal blooms (Izydorczyk et al. 2008; Poste et al. 2012; Mbonde et al. 2015). For instance, Hall et al. (2003) found that higher total phosphorus (TP) concentrations were associated with greater chlorophyll a concentrations in three embayments (Sodus Bay, Sandy Pond and Chaumont Bay) compared to offshore locations in Lake Ontario, Canada. Shayo et al. (2011) observed higher nutrient concentrations in bays with anthropogenic activities (i.e. dense settlements, industrial and agricultural activities) compared to bays located in remote areas with sparse settlements in the south-eastern part of Lake Victoria.

Lake Diefenbaker is a large mesotrophic reservoir in the Canadian prairie. It receives most of its water from the South Saskatchewan River (SSR) and serves as an important source of water for domestic consumption, irrigation, recreation, aquaculture and power generation to Saskatchewan, Canada. The effective drainage area of Lake Diefenbaker is 86,900 km\textsuperscript{2} with about 91% of this area located in southern Alberta. Most of the catchment area of the reservoir in Alberta and the state of Montana is agricultural cropland (Hall, Leavitt, Dixit, et al. 1999; SIPA 2008; Saskatchewan Water Security Agency 2012).

There are numerous embayments along the length of the reservoir. These embayments are delineated (i.e. located in different sheltered valleys of varying slope) from the main channel of the reservoir and are conducive for human activities. Anthropogenic activities located in embayments include settlements, fishing and boating activities, golf courses, marinas and livestock operations (cattle watering). These activities are potential sources of nutrients to the embayments and may adversely affect water quality. However, only one study has examined the effect of these human activities on related water quality parameters. Although not significantly different, North et al. (2014) reported that total coliforms, fecal coliforms and Escherichia coli were always greater in abundance in embayments relative to the main channel.

Nevertheless, relatively little is known about how human activities, particularly urban activities and cattle wading in embayments, may be affecting the composition of phytoplankton in Lake Diefenbaker. Management strategies of reservoirs often focus on mitigating such impacts before they become problematic (Carney 2009). Therefore, we examined the biomass and composition of phytoplankton to determine if there was evidence of impact on the water quality in the embayments. We
also examined the chemical and physical parameters in embayments exposed to human activities in comparison to those in unexposed embayments and main channel locations in Lake Diefenbaker.

**Materials and methods**

**Study area**

Lake Diefenbaker is located in South Saskatchewan, Canada (51° 1’53”N, 106° 50’9”W). The reservoir has a length of approximately 225 km and a typical width of 2—3 km (Figure 1). The volume and surface area of the reservoir are approximately 9 km³ and 349 km², respectively. The mean depth (Z_{mean}) of the reservoir is 22 m and the maximum depth is 59 m near the Gardiner Dam (Sadeghian et al. 2015). We selected eight embayments that contained human activities, including three embayments with urban activities and five embayments with cattle watering. These were classified as exposed embayments. The average depth of these exposed embayments is 12 m (range, 8.34—22.51 m, Table 1). We also selected four embayments with no perceived human activities and classified them as unexposed embayments (Table 1). The average depth of these unexposed embayments is 10 m (range, 8.87—21.04 m). In addition, six sites were chosen adjacent to the embayments in the nearby main channel of the reservoir and were classified as main channel sites. The average depth of these main channel sites is 37 m (range, 21.0—54 m). Therefore, we had three classes: exposed embayments, unexposed embayments and main channel sites (Figure 1).

**Field sampling**

Each site was sampled once every month from June to October in 2011 and in 2012. All water samples were collected at 2 m with a Van Dorn sampler, poured into 20 L poly-bags and kept in the dark in coolers. Water samples were returned to the laboratory at the University of Saskatchewan and stored under ambient conditions (light and temperature) until they were processed for water chemistry the following day. Water samples that were collected for phytoplankton analysis were fixed in Lugol’s 1% solution in the field.

Water temperature (WT), pH and DO concentrations were measured at the 2 m depth with a YSI 6600 v2 multi-parameter sonde. The mixing depth (Z_{mix}) was defined as the depth from the water surface to a point where the change in WT was greater than 0.5 °C m⁻¹ (Hudson & Vandergucht 2015).

![Figure 1. Map of Lake Diefenbaker with sampling sites: unexposed embayments (△) U1–U4, exposed embayments (●) E1–E8 and main channel sites (□) M1–M6, from upstream (Highway 4) to downstream (Gardiner and Qu’Appelle dams). Each site was sampled from June to October in 2011 and 2012.](image_url)
Photosynthetically active radiation (PAR) was measured with a biospherical radiometer (Biospherical Instruments Inc. BIC 2104 submersible radiometer). The vertical extinction coefficient ($kd$) was derived from the linear regression of the natural logarithm of PAR and depth (Kirk 2003). We estimated the euphotic depth ($Z_{eu}$) as the depth at which the light intensity was 1% of the water surface. In addition, the Secchi disk depth (SD) was used to estimate water transparency.

**Laboratory analyses**

We determined TP, total dissolved phosphorus (TDP) and soluble reactive phosphorus (SRP) using the method of Parsons et al. (1984). We used second derivative UV spectroscopy to measure total nitrogen (TN), total dissolved nitrogen (TDN) and NO$_3^-$ (Bachmann & Canfield 1996). Samples for TDP, SRP, TDN, NO$_3^-$ and NO$_2^-$ were filtered through 0.2 µm polycarbonate filters using syringe filtration. The phenol-hypochlorite method by Stainton et al. (1977) was used to determine NH$_4^+$ concentrations colorimetrically. Samples for dissolved organic carbon (DOC) were measured as described in Sereda et al. (2012) using an organic carbon analyzer (Shimadzu TOC - 5050A).

**Phytoplankton composition and biomass**

Samples were identified and counted on an Olympus inverted (IX51) microscope using the technique of Utermöhl (1958). Each taxon was identified to genus with the aid of several taxonomic keys (Brook et al. 2002; Wehr & Sheath 2003; Bellinger & Sigee 2010). A minimum of 400 cells were enumerated per sample. We used image-Pro Analyser 7.0 computer software to estimate the size of the phytoplankton and used a computerized phytoplankton counting program ‘Algamica (Version 4.0)’ developed by Gosselain and Hamilton (2000) to calculate final biovolume for each taxon as described in Abirhire et al. (2015).

In addition to biovolume, chlorophyll $a$ was also determined to provide an estimate of algal biomass which would include contribution from smaller algal groups that were not visible with the Utermöhl inverted scope technique. Chlorophyll $a$ was measured on whole water samples that were collected on 47 mm glass fiber filters with vacuum filtration (<68.9 kPal under low light). Pigments were extracted and analyzed spectrophotometrically according to Bergmann and Peters (1980). Absorbance was read at 665 nm, as described in Vandergucht et al. (2013).

**Data analyses**

We used a linear mixed effects model to compare environmental variables among the three classes (i.e. exposed embayments, unexposed embayments and main channel sites (fixed effect) for each year).
According to Abirhire et al. (2015) and Hudson and Vandergucht (2015), most of the environmental variables showed temporal and spatial pattern. Therefore, we used month (June to October) as fixed factor in our model. Distance of each site down the length of the reservoir was used as a random term in the models to account for the repeated measure in the experimental design (Pinheiro & Bates 2000). Specifically, we compared chemical variables (TP, TDP, SRP, TN, TDN, NO₃⁻, NH₄⁺, DOC, DO and pH), physical variables (SD, Zₑu, kₑ and WT) and chlorophyll a concentrations among all three classes for each year. Finally, we compared total phytoplankton biovolume among all three classes for each year. Total phytoplankton biovolume was log₁₀ transformed to achieve normality.

We used non-metric multidimensional scaling (NMDS) to extract trends from the phytoplankton community composition based on Bray–Curtis dissimilarity. Biovolume of the different phytoplankton groups was transformed (log₁₀ + 1) to down weight the most abundant groups. The NMDS was started from several random configurations and the solution with the lowest stress was selected after 200 runs. Stress measure, which is a departure of fit between ordination and the data, was calculated. We performed a permutation ANOVA (PERMANOVA) on the species matrix (phytoplankton composition) to determine if there was a significant difference among all three classes. We used the function ‘adonis’ from the vegan package for the PERMANOVA. All statistical analyses were performed in R version 2.15.2 (R Development Core Team 2012).

Results

Environmental variables

The reservoir received high flow from the South Saskatchewan River during our study period. Peak flows of 2300 and 1505 m³ s⁻¹ occurred in early June of 2011 and in late June of 2012, respectively. Consequently, water level increased to almost full supply level (approximately 556.0 m above sea level) from May to June and possibly flushed the embayments with new water (Hudson & Vandergucht 2015). WT from all sites (unexposed, exposed and main channel) were similar in 2011 and 2012 (p > 0.05) (Table 2). The water column was completely mixed in exposed and unexposed embayments due to the shallowness of these embayments. However, in the much deeper main channel sites (i.e. mixing depths were typically >15 m), the water column was stratified from July to September (Hudson & Vandergucht 2015). Secchi depth and euphotic depth from all sites were greater

<table>
<thead>
<tr>
<th>Variables</th>
<th>2011</th>
<th>2012</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Unexposed</td>
<td>Exposed</td>
</tr>
<tr>
<td>WT (°C)</td>
<td>17.84 ± 0.69</td>
<td>17.84 ± 0.59</td>
</tr>
<tr>
<td>SD (m)</td>
<td>2.27 ± 0.28</td>
<td>2.31 ± 0.20</td>
</tr>
<tr>
<td>Zₑu (m)</td>
<td>5.06 ± 0.55</td>
<td>4.84 ± 0.41</td>
</tr>
<tr>
<td>kₑ (m⁻¹)</td>
<td>1.08 ± 0.14</td>
<td>1.18 ± 0.12</td>
</tr>
<tr>
<td>Zₑu,mix (m)</td>
<td>10.79 ± 0.88</td>
<td>12.22 ± 0.42</td>
</tr>
<tr>
<td>DO (mg L⁻¹)</td>
<td>9.06 ± 0.11</td>
<td>8.97 ± 0.09</td>
</tr>
<tr>
<td>pH</td>
<td>8.43 ± 0.02</td>
<td>8.39 ± 0.02</td>
</tr>
<tr>
<td>TP (µg L⁻¹)</td>
<td>20.96 ± 2.99</td>
<td>23.16 ± 2.07</td>
</tr>
<tr>
<td>TDP (µg L⁻¹)</td>
<td>7.71 ± 0.91</td>
<td>8.90 ± 0.77</td>
</tr>
<tr>
<td>SRP (µg L⁻¹)</td>
<td>3.50 ± 0.96</td>
<td>4.46 ± 0.76</td>
</tr>
<tr>
<td>TN (µg L⁻¹)</td>
<td>658.28 ± 30.42</td>
<td>672.06 ± 25.17</td>
</tr>
<tr>
<td>TDN (µg L⁻¹)</td>
<td>587.01 ± 27.76</td>
<td>606.59 ± 23.20</td>
</tr>
<tr>
<td>NO₃⁻ (µg L⁻¹)</td>
<td>383.33 ± 26.49</td>
<td>394.41 ± 19.97</td>
</tr>
<tr>
<td>NH₄⁺ (µg L⁻¹)</td>
<td>10.63 ± 3.11</td>
<td>13.60 ± 3.07</td>
</tr>
<tr>
<td>DOC (mg L⁻¹)</td>
<td>3.24 ± 0.08</td>
<td>3.22 ± 0.04</td>
</tr>
<tr>
<td>Chl a (µg L⁻¹)</td>
<td>4.44 ± 0.53</td>
<td>4.18 ± 0.38</td>
</tr>
</tbody>
</table>
in 2012 than in 2011 ($n = 136, p < 0.0001$). Similarly, mean extinction coefficients were greater in 2011 than in 2012 ($n = 150, p < 0.0001$). Although not significantly different, the mean Secchi depth and mean euphotic depth tended to be greater in main channel sites compared to the embayments (exposed and unexposed) sites in both years. In general, WT, Secchi depths, euphotic depths and extinction coefficients were not significantly different among all three classes in 2011 ($p > 0.05$) and 2012 ($p > 0.05$).

Mean epilimnetic DO concentrations were consistently above 8 mg L$^{-1}$ in both years. The concentrations of TP, TDP, SRP, TN, TDN and NO$_3^-$ from all sites were greater in 2011 than in 2012 ($p < 0.001$). In contrast, the DOC concentrations from all sites were greater in 2012 ($n = 168, p < 0.001$). The NH$_4^+$ concentrations from all sites were similar in both years ($n = 180, p > 0.05$) (Table 2). Overall, the concentrations of TP, TDP, SRP, TN, TDN, NO$_3^-$, NH$_4^+$, DOC, DO and pH were similar among all three classes in 2011 ($p > 0.05$) and 2012 ($p > 0.05$) (Table 2).

**Phytoplankton biomass**

Chlorophyll $a$ concentrations were similar among all three classes in 2011 ($p = 0.87$) and in 2012 ($p = 0.43$). Similarly, total phytoplankton biovolume did not differ between embayment types or the main channel sites in 2011 ($p = 0.88$) and in 2012 ($p = 0.07$). Moreover, there was no significant difference in phytoplankton biovolume among all three classes within each sampling month; the phytoplankton biovolume in all three classes followed similar temporal trend in both years (Figure 2(A,B)). Phytoplankton biovolume in all three classes varied from upstream to downstream sections of the reservoir in both years (Figure 2(C,D)).

**Phytoplankton composition**

A total of 76 phytoplankton genera were observed from seven algal taxonomic groups (bacillariophytes, chlorophytes, chrysophytes, cryptophytes, cyanophytes, euglenophytes and pyrrophytes) from all sites in both years. Mean generic richness was similar among all three classes in 2011 ($p > 0.05$) and 2012 ($p > 0.05$) (Figure 3).

The NMDS result based on phytoplankton biovolume suggested a solution with three axes (correlation between calculated dissimilarity and ordination distance was high $r = 0.89$). Phytoplankton community composition was similar among all three classes in both years combined as indicated by PERMANOVA ($F = 0.43238$, $R^2 = 0.0049$, $p = 0.884$) (Figure 4). The phytoplankton community composition was consistent with that reported for the main channel by Abirhire et al. (2015). The bacillariophytes and the cryptophytes accounted for approximately 87%–91% of the total phytoplankton biomass across all sites. The other taxa (chlorophytes, chrysophytes, cyanophytes, euglenophytes and pyrrophytes) accounted for 9%–13% of the total phytoplankton biovolume across all sites.

**Discussion**

Nutrient concentrations (TP, TDP, SRP, TN, TDN and NO$_3^-$) were typically greater in 2011 than in 2012, whereas, water clarity (SD, $Z_{eu}$ and $k_d$) was greater in 2012 compared to 2011. It appears that the difference in physical and chemical characteristics may be related to the high flow events that occurred in early June of 2011 and late June of 2012 that carried large loads of nutrients and associated non-algal turbidity from the South Saskatchewan River (SSR) into Lake Diefenbaker. Although both years were characterized by high flows from the SSR into Lake Diefenbaker, the peak flow in 2011 (2300 m$^3$ s$^{-1}$ in early June) was greater than peak flow in 2012 (1505 m$^3$ s$^{-1}$ in late June). In addition, the volume of water that entered the reservoir during these high flow events was also different between years. Therefore, it is likely that these differences between the high flows of 2011
and 2012 may explain the differences in nutrient concentrations and water clarity between years (Hudson & Vandergucht 2015).

Although not significant, water clarity was reduced in the embayments (i.e. in both unexposed and exposed) compared to main channel sites (Table 1). The shallow depths (exposed embayment

Figure 2. Box and whisker plots showing the median, the 25th and the 75th percentiles of phytoplankton biovolume from June to October in 2011(A) and in 2012 (B), and from upstream sections to downstream sections in 2011 (C) and 2012 (D) in Lake Diefenbaker.

Figure 3. Mean and standard error of generic richness in reference embayments, exposed embayments and main channel sites for 2011 and 2012.
and unexposed embayments ($Z_{\text{mean}} = 10$ m) and the mixing of the unstratified water column in these embayments may have allowed for the re-suspension of bottom sediments, which in turn would reduce water clarity. Gikuma-Njuru and Hecky (2005) observed reduced water clarity in a large shallow embayment in Lake Victoria (Nyanza gulf, $Z_{\text{mean}} = 12$ m).

The generic richness, phytoplankton biovolume and composition, and chlorophyll $a$ concentration were not significantly different among all three classes in both years. This was not surprising; water chemistry variables including TP, TDP, SRP, TN, TDN, NO$_3^-$, NH$_4^+$, DOC, epilimnetic DO and pH and some physical variables such as WT, SD, $Z_{\text{eu}}$ and $k_d$ were similar among all three classes in both years. Other studies on Lake Diefenbaker have reported similar results. For example, the phytoplankton composition (dominated by the diatoms and the cryptophytes) and water chemistry within and surrounding (20 km) Lake Diefenbaker’s aquaculture facility varied more temporally from August to November than spatially from 2008 to 2011 in Lake Diefenbaker (M. Otu, Department of Fisheries and Oceans Canada, personal communication). Bukaveckas et al. (2002) also found similar phytoplankton biomass and species composition in an embayment with rural and agricultural activities and the main channel in Kentucky Lake.

It is possible that the anthropogenic effects from human activities in embayments were not large enough in our study. Nevertheless, impacts of human activities in embayments have been observed in other studies. For example, Hall et al. (2003) reported greater nutrient concentration (TP) and algal biomass (chlorophyll $a$ concentrations) in embayments compared to near-shore and off-shore locations in Lake Ontario, Canada. The authors attributed the greater TP and chlorophyll $a$ concentrations in the embayments to nutrient inputs from nearby watersheds into the embayments and also to poor water exchange between the embayments and the pelagic (main channel) waters (Mitrovic et al. 2001; Hall et al. 2003). However, the embayments (exposed and unexposed) in our study are not completely sheltered and they may exchange water more readily with the main channel. This exchange of water between embayments and the main channel would explain the similar environmental conditions observed among all three classes in this study. Mbonde et al. (2015) reported greater nutrient concentrations and algal biomass in closed bays with limited water exchange compared to the open bays with significant water exchange with the main basin in Lake Victoria.

A contributing factor potentially reducing the difference in phytoplankton biomass and composition in embayments (exposed and unexposed) and main channel sites would be related to the high
flow events observed in both years of our study. It is possible that the large nutrient loads, non-algal turbidity and volume of water associated with the high flow events masked nutrient issues in embayments with human activities (Paerl & Huisman 2009; Reynolds 1990). For instance, Mao et al. (2015) reported that flow regime rather than nutrients initiated algal blooms in a eutrophic embayment in the Three Georges Reservoir in China.

Furthermore, during the high flows events, it was evident that water was moving through Lake Diefenbaker, suggestive of rapid flushing of the reservoir at full supply level as observed in other studies (Bukaveckas et al. 2002; Andradóttir et al. 2012). Because cyanobacteria require calm and low flushing rates in addition to high nutrient concentrations for their growth, they are less likely to be abundant under such condition (Huszar & Reynolds 1997; Godlewska et al. 2003). For instance, in a previous study on Lake Diefenbaker (1984–1985), cyanobacteria dominated the phytoplankton, contributing 79% of the total phytoplankton biovolume during a drought period with low flow from the South Saskatchewan River (e.g. flow declined to 40 m$^3$ s$^{-1}$ in mid-June in 1984) and longer water residence time (2.27 years in 1984) (SEPS & EC 1988). However, during the high flows of the current study (shorter residence time of 0.72 years in 2011 and 0.99 years in 2012), cyanobacteria made a minor contribution (<3%) to the total phytoplankton biovolume. The high flow events favored the diatoms and cryptomonads which can tolerate and grow rapidly under the nutrient rich, turbulent and turbid conditions associated with high flows (Reynolds et al. 2002; Abirhire et al. 2015).

The Canadian prairie is experiencing a changing climate (Booth et al. 2012; Lapp et al. 2012). As temperatures become warmer, Lake Diefenbaker will experience changes in hydrologic regime (Pomeroy et al. 2009; Sereda et al. 2011; Hudson & Vandergucht 2015; Vogt et al. 2015). Climate warming will lead to the reduction in mountain snowpack and, consequently, lower flows in the South Saskatchewan River (Pomeroy et al. 2009; Ball et al. 2013). This will result in longer water residence time in Lake Diefenbaker (St. Jacques et al. 2010; Ball et al. 2013). As a result, these conditions may shift the phytoplankton community composition from diatoms and cryptomonads to cyanobacteria, including bloom-forming and toxin-producing genera that may threaten the water quality of Lake Diefenbaker (Paerl & Huisman 2008; Abirhire et al. 2015).

In summary, we did not detect an impact of human activities in embayments in Lake Diefenbaker during two years of high flow events. The high flow events and the associated rapid flushing of the embayments may have obscured the effects of anthropogenic activities on water quality and phytoplankton community composition in these embayments. However, we predict that during drought or low flow conditions, the embayments will be more sheltered from the main channel of the reservoir and this isolation may result in different chemical and physical conditions in embayments. This, in turn, will affect the phytoplankton community composition. However, the mean spring flows from the SSR in both years of our study were among the greatest 15% of spring flows from 1967 to 2012. Therefore, future studies on the embayments in Lake Diefenbaker would benefit from additional years of study during low flow conditions. This will also assist with understanding the effects of low flow conditions that may be associated with warmer climates on the prairies.

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**Disclosure statement**

No potential conflict of interest was reported by the authors.
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Kristine Hunter is a master’s candidate at Limnology Laboratory, Department of Biology, University of Saskatchewan. She collected and processed water samples, worked up data and proof-read the drafts of the manuscript.

David Vandergucht was a laboratory manager at the Limnology Laboratory, Department of Biology, University of Saskatchewan at the time they put the manuscript together. He processed water samples and worked up data.

Jeff Hudson is a professor and the principal investigator at Limnology Laboratory, Department of Biology, University of Saskatchewan. He provided useful ideas and proof-read the drafts of the manuscript.

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