Comparison of aquatic invertebrate communities in near-shore areas with high or low boating activity

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**ABSTRACT**

Lakeshore areas provide important habitat for aquatic invertebrates in shallow lakes. However, these zones are prone to anthropogenic disturbances that include shoreline development, urbanization, nutrient inputs, agricultural and/or recreational use. Among recreational uses, public access sites are often developed to accommodate boaters and facilitate lake access via boat ramps. Although the ‘foot print’ associated with boat ramp structures can be relatively small compared to total shoreline coverage, little is known about the relative effects of boating activity on littoral zones and macroinvertebrate communities. In this study, we assess the relative impact of boating-related activities on aquatic macroinvertebrate communities at high-use (primary) and low-use (secondary) boat ramps on five glacial lakes of eastern South Dakota. Macroinvertebrate assemblages were dominated by few taxa that included Chironomidae, Corixidae, Caenidae, and Amphipoda. Moreover, boat ramp use did not influence abundance or diversity of aquatic macroinvertebrates. Habitat, specifically substrate composition, was also similar between primary and secondary boat ramps despite more intense use associated with primary boat ramps. Our results support related findings that aquatic invertebrate assemblages in the Prairie Pothole Region are structured by regional environmental variability (climate, habitat, and water quality) resulting in communities with characteristically low diversity and tolerant taxa. This may explain why small-scale disturbance associated with boating activity has no discernable effect on macroinvertebrate assemblages in glacial lakes of the Prairie Pothole Region.

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**Introduction**

Powerboating is a popular, recreational activity on many lakes and rivers. In areas that experience high boating-related activity, physical disturbance from boat wakes can affect...
near-shore habitats and associated biota (Liddle and Scorgie 1980; Mosisch and Arthinton 1998; Hardiman and Burgin 2011). Boat wakes can impact aquatic macroinvertebrates by disturbing sediments (Bishop and Chapman 2004) or dislodging them from macrophytes or other structures (Bishop 2005; Bishop 2008). Disturbance caused by wave and propeller action can also contribute to shoreline erosion, decreased water clarity, and increased nutrient concentrations via sediment suspension (Liddle and Scorgie 1980; Mosisch and Arthinton 1998; Hardiman and Burgin 2011). Several U.S. states prohibit power loading of boats onto trailers because prop wash can erode the lake bed and re-distribute sediments around boat ramps. Additional boating-related impacts to aquatic ecosystems include contamination by fuel, oil, and exhaust emissions as well as introduction and spread of non-native organisms for which boats can serve as vectors (Liddle and Scorgie 1980; Mosisch and Arthinton 1998; Hardiman and Burgin 2011).

Public boat ramps provide important lake access for recreational activities such as boating, water skiing, and angling. Indeed, boat ramp construction and maintenance represents an important part of annual operating budgets for many fish and wildlife agencies. Parking lots and amenities such as restrooms and fish cleaning stations provide benefits that attract boating enthusiasts to public boat ramps (Siderelis et al. 1995; Reed-Andersen et al. 2000). In turn, public access areas can concentrate boating-related disturbances compared to other near-shore areas of a lake, such as private or fee-based boat access.

In this study, we evaluate effects of boating activity on aquatic invertebrate communities in natural lakes. Aquatic invertebrate abundance and composition were compared between boat ramps with two levels of boating intensity: high versus low use. We postulated that at high-use (primary) boat ramps, aquatic invertebrate communities are negatively impacted by increased frequency and magnitude of disturbance compared to that of low use (secondary) boat ramps with similar physical characteristics (i.e. presence of a boat ramp). We measured abundance and diversity of aquatic macroinvertebrates as an index of boating-related disturbance because of their importance in aquatic food webs and their widespread use in bioassessment studies (Covich et al. 1999).

**Materials and methods**

**Study area**

Five glacial lakes in the Prairie Pothole Region (PPR) of eastern South Dakota were chosen for study and included lakes Kampeska, Madison, Pelican, Poinsett, and Thompson (Figure 1). All five lakes are popular destinations for anglers, boaters, water skiers and other water sport enthusiasts (http://gfp.sd.gov/state-parks/directory). These study lakes are comparatively large relative to many glacial lakes in the region with surface areas ranging from 1131 to 5041 ha with maximum; water depths range from 2.4 to 6.1 m (Stueven and Stewart 1996). Private, lake-shore development varies appreciably among the lakes, with Lake Thompson being the least-developed (1.1 private boat docks/km) and Lake Madison the most-developed (19.8 private boat docks/km). The lakes are classified as eutrophic (trophic state index > 50; Carlson 1977) and regularly experience late-summer cyanobacterial blooms. Shoreline development indices range from 1.3 for Lake Poinsett to 2.9 for Pelican Lake. Due to relatively shallow water depths and long fetch (≥7 km), all lakes exhibit polymictic mixing and do not develop distinct thermoclines in summer months.

**Sample site selection**

Each lake had at least two, public-access boat ramps that could be classified as either a primary access or secondary access based on the frequency and magnitude of use. If more
than one primary or secondary boat ramp was available at a lake, we randomly selected a boat ramp to sample. Boat ramps were classified based on location and size of boat ramp access, number of parking slips (>15 for primary boat ramps), and presence/absence of infrastructure facilities (Table 1). The relative intensity of boater use at each boat ramp was assessed using Google Earth imagery and corroborated by interviews with South Dakota Department of Game Fish and Parks conservation officers. Available imagery for

Figure 1. Five glacial lakes sampled for aquatic macroinvertebrates in the Prairie Pothole Region of eastern South Dakota during September 2012. Sampling in each lake occurred at one primary boat ramp and one secondary boat ramp.
June to September, 2003–2016 was examined for each lake and the maximum number of vehicles with boat trailers (VBT) was recorded (Table 1).

### Habitat and invertebrate sampling

Sampling was conducted during the third week of September 2012. Physical habitat was quantified within a 1-m radius of each sample site by measuring substrate composition (% silt, sand, gravel, cobble, and boulder) and percent submerged vegetation coverage following guidelines outlined by Simonson et al. (1994).

Aquatic macroinvertebrates were collected using two gear types; a D-frame sweep net, hereafter referred to as a sweep net, and a stovepipe sampler (Merritt et al. 1984). We used sweep nets to sample invertebrates in the water column (i.e. lentic). Our sampling technique involved using a 500-μm mesh sweep net (H/C2L [cm]: 34/50], to sweep a 2 m arc from the water surface to near bottom, and then back to the surface in order to sample the entire water column (Turner and Trexler 1997; Meyer et al. 2011). Stovepipe samplers (H:60 cm, diameter: 34 cm) were used to sample benthic macroinvertebrates at each site by pushing the sampler into the bottom substrate, then using a small dip net (H × L [cm]:12 × 14, mesh [mm]: 1.0) to collect invertebrates inside the sampler (Merritt et al. 1984). We collected five sweep net and five stove pipe samples at each site within 10 m of the outside edge of a boat ramp. To reduce bias associated with water depth, we collected invertebrates in water depths ranging from 50 to 70 cm at each site. Invertebrates collected from each sample were placed in 250–500 ml plastic bottles containing 90% ETOH and transported to South Dakota State University where they were sorted and identified using dissecting microscopes (Anderson 1959). Organisms were identified to subclass (i.e. Hirudinea, Oligochaeta), order (i.e. Amphipoda) or family (e.g. Caenidae, Chironomidae, Corixidae, Spaeridae).

### Statistical analysis

Paired t-tests were used to evaluate differences in substrate types between primary and secondary boat ramps. A Bonferroni adjustment (α = 0.01) was used to maintain a table-wide alpha level of 5% for the five substrate categories measured (Rice 1989). We compared invertebrate taxa richness between primary and secondary boat ramps using a paired t-test (Cody and Smith 2006). Macroinvertebrate community similarity was evaluated by comparing mean, taxa-specific abundance of invertebrates collected at primary ramps.
boat ramps to those collected at secondary boat ramps. Pearson’s correlation coefficient was used as an index of community similarity (Krebs 1999) where a value of –1 indicates complete negative correlation between communities and +1 indicates complete positive correlation between communities (Sokal and Rohlf 1969). Macroinvertebrate taxa included in analyses were either fully aquatic or represented an aquatic life-stage (i.e. insects). The influence of boat ramp type (i.e. primary or secondary) on macroinvertebrate composition across lakes was evaluated using multivariate analysis of variance (MANOVA) on log(x + 1) transformed data (Cody and Smith 2006). If a MANOVA test was significant (α < 0.05), we compared abundance of individual invertebrate taxa among lakes using a mixed-model analysis of variance test to identify those taxa contributing to the observed difference. All statistical analyses were performed using SAS software, version 9.4 (SAS Institute Inc. 2013).

### Results

Substrate composition did not differ between primary and secondary boat ramps (Table 2). In general, sand and gravel were the dominant substrate types representing >80% of the substrate composition at boat ramps. Similarly, aquatic macrophytes were rarely encountered, representing <1.4% coverage at any site.

Seven macroinvertebrate taxa (i.e. Amphipoda, Caenidae, Chironomidae, Corixidae, Hirudinea, Oligochaeta, and Sphaeriidae) collectively represented >88% of individuals collected with stovepipe samplers and >98% of individuals collected with sweepnets. Stovepipe samplers generally collected more taxa, especially benthic taxa (i.e. Caenidae, Chironomidae, and Hirudinea), whereas sweep nets collected fewer taxa but were selective for highly mobile littoral macroinvertebrates including Amphipoda and Corixidae. Several taxa were excluded from our analyses because they were semiaquatic (e.g. Collembola) or were found at only one site (e.g. Ostracoda, Planorbidae).

Mean taxa richness was similar among primary (5.8, n = 5, SE = 1.3) and secondary (6.4, n = 5, SE = 1.0) boat ramp sites for invertebrates collected using a stovepipe sampler (paired t-test, df = 4, t = 2.77, P = 0.73). Similarly, mean taxa richness for invertebrates collected using sweep nets was similar at primary (4.4, n = 5, SE = 0.9) and secondary (5.6, n = 5, SE = 1.0) boat ramp sites (df = 4, t = 2.77, P = 0.28; Table 3). Invertebrate composition at primary boat ramps was similar to that observed at secondary boat ramps based on either stovepipe samples (MANOVA, Wilk’s λ = 0.21, F7,2=1.08, P = 0.56) or

### Table 2. Mean substrate composition at primary and secondary boat ramp sites for five eastern South Dakota glacial lakes.

| Substrate type | Particle size (mm) | Primary ramps (n = 5) | Secondary ramps (n = 5) | Mean difference primary minus secondary (%) | P     |
|----------------|--------------------|-----------------------|-------------------------|---------------------------------------------|-------|
| Boulder        | >256               | 0.6 (0–2.2)           | 1.2 (0–3.2)             | –0.6 (–2.2 to 1.1)                          | 0.37  |
| Cobble         | 64–256             | 8 (0–20.1)            | 4 (0–8.1)               | 4.0 (–10.1 to 18.1)                         | 0.47  |
| Gravel         | 2–64               | 22.6 (0–45.7)         | 12.4 (0.5–24.2)         | 10.2 (–4.8 to 25.2)                         | 0.13  |
| Sand           | 0.062–2            | 64.4 (26.1–100)       | 70.2 (55.1–85.2)        | –5.8 (–38.3 to 26.7)                        | 0.64  |
| Silt           | 0.004–0.062        | 4.6 (0–15.4)          | 7.8 (2.1–13.5)          | –7.6 (–18.6 to 3.4)                         | 0.13  |

Values in parentheses represent 95% confidence intervals. Mean differences (%) and P values are given for paired t-tests comparing substrate composition between primary and secondary boat ramps. A sequential Bonferroni adjustment (α = 0.01) was used to maintain a table-wide α level of 5% for the five substrate types measured (Rice 1989).
sweep nets (Wilk’s $\lambda = 0.35$, $F_{6,3} = 0.90$, $P = 0.58$). Similarly, mean abundance of invertebrates collected at primary boat ramps was significantly correlated with mean abundance of invertebrates captured at secondary boat ramps for stovepipe samples (correlation analysis; $n = 47$, $r = 0.56$, $P < 0.0001$) and sweep nets ($r = 0.67$, $P = 0.0001$; Figure 2). Overall, we found similar habitat, macroinvertebrate abundance, and community similarity between primary and secondary boat ramps in glacial lakes.

| Lake     | Taxa            | Stovepipe |          |          | Sweep net |          |
|----------|-----------------|-----------|----------|----------|-----------|----------|
|          |                 | 1° Ramp   | 2° Ramp  | 1° Ramp  | 2° Ramp   | 1° Ramp  | 2° Ramp  |
|          |                 |           |          |          |           |          |          |
| Kampska  | Amphipoda       | 0.0       | 2.2 (2.2)| 0        | 0         |          |          |
|          | Annelida        | 0         | 0        | 0        | 0.2 (0.2) |          |          |
|          | Brachycentridae | 0.0       | 2.2 (2.2)| 0        | 0         |          |          |
|          | Caenidae        | 8.8 (4.1) | 8.8 (4.1)| 0.4 (0.2)| 0.8 (0.4) |          |          |
|          | Chironomidae    | 26.4 (8.9)| 81.5 (52.4)| 0.8 (0.5)| 1.2 (0.7) |          |          |
|          | Coenagrionidae  | 0         | 0        | 0        | 0.2 (0.2) |          |          |
|          | Corixidae       | 2.2 (2.2)| 2.2 (2.2)| 12.0 (1.8)| 0.4 (0.4) |          |          |
|          | Hydropsychidae  | 0         | 0        | 0        | 0.4 (0.4) |          |          |
|          | Nematoda        | 0         | 2.2 (2.2)| 0.4 (0.4)| 0         |          |          |
|          | Oligochaeta     | 0         | 0        | 0        | 3.4 (3.2) |          |          |
|          | Ostrococha      | 0         | 0        | 0.2 (0.2)| 0         |          |          |
|          | Spaeriidae      | 0         | 2.2 (2.2)| 0        | 0         |          |          |
| Madison  | Amphipoda       | 19.8 (17.2)| 2.2 (2.2)| 0.4 (0.4)| 0.2 (0.2)|          |          |
|          | Caenidae        | 2.2 (2.2)| 0        | 1.8 (1.8)| 0         |          |          |
|          | Chironomidae    | 17.6 (10.2)| 96.9 (34.4)| 0        | 0.4 (0.4) |          |          |
|          | Corixidae       | 22.0 (9.2)| 55.1 (28.9)| 108.2 (64.2)| 79.2 (20.7)|          |          |
|          | Heptageniidae   | 2.2 (2.2)| 0        | 0        | 0         |          |          |
|          | Hirudinea       | 2.2 (2.2)| 0        | 0        | 0         |          |          |
|          | Nematoda        | 13.2 (8.1)| 50.7 (24.5)| 0.2 (0.2)| 0         |          |          |
|          | Polycentropodida| 4.4 (4.4)| 0        | 0        | 0         |          |          |
|          | Sphaeriidae     | 2.2 (2.2)| 0        | 0        | 0         |          |          |
| Pelican  | Amphipoda       | 94.7 (54.9)| 0        | 9.8 (3.7)| 2.2 (1.0)|          |          |
|          | Baetidae        | 2.2 (2.2)| 0        | 0        | 0         |          |          |
|          | Caenidae        | 207.1 (105.6)| 28.6 (10.2)| 31.6 (8.3)| 5.2 (1.4)|          |          |
|          | Chironomidae    | 145.4 (45.5)| 52.9 (12.7)| 0.4 (0.2)| 2.6 (0.6)|          |          |
|          | Corixidae       | 13.2 (6.4)| 4.4 (4.4)| 7.4 (2.3)| 1.6 (0.7)|          |          |
|          | Hirudinea       | 4.4 (2.7)| 4.4 (2.7)| 0        | 0         |          |          |
|          | Limnephilidae   | 46.3 (16.5)| 121.2 (32.1)| 0        | 1.4 (0.7)|          |          |
|          | Nematoda        | 19.8 (12.3)| 33.0 (20.3)| 2.4 (1.2)| 1.0 (0.6)|          |          |
|          | Oligochaeta     | 0         | 8.8 (4.1)| 0.2 (0.2)| 0         |          |          |
|          | Ostrococha      | 8.8 (4.1)| 0        | 0.4 (0.4)| 0         |          |          |
|          | Physidae        | 0         | 4.4 (4.4)| 0        | 0         |          |          |
|          | Planorbiidae    | 59.5 (12.8)| 70.5 (16.2)| 0.4 (0.2)| 0.4 (0.2)|          |          |
|          | Sphaeriidae     | 26.4 (16.6)| 26.4 (16.2)| 0.2 (0.2)| 0.4 (0.4)|          |          |
|          | Zygoptera       | 0         | 0        | 1.0 (0.8)| 1.0 (1.0)|          |          |
| Poinsett | Amphipoda       | 667.5 (139.4)| 59.5 (14.6)| 146.0 (48.9)| 5.8 (2.7)|          |          |
|          | Caenidae        | 0         | 8.8 (4.1)| 0        | 1.0 (0.5)|          |          |
|          | Chironomidae    | 68.3 (34.2)| 74.9 (45.3)| 0.4 (0.2)| 1.4 (1.2)|          |          |
|          | Corixidae       | 2.2 (2.2)| 39.7 (20.5)| 5.6 (1.6)| 29.0 (11.3)|          |          |
|          | Hirudinea       | 44.1 (13.5)| 6.6 (6.6)| 0        | 0         |          |          |
|          | Hydrachnidae    | 0         | 11.0 (8.5)| 0        | 0.6 (0.6)|          |          |
|          | Hydrobiidae     | 0         | 4.4 (4.4)| 0        | 0         |          |          |
|          | Nematoda        | 0         | 28.6 (9.6)| 0        | 0.4 (0.4)|          |          |
|          | Pyralidae       | 0         | 0        | 0.2 (0.2)| 0         |          |          |
|          | Zygoptera       | 0         | 0        | 0.2 (0.2)| 0         |          |          |
| Thompson | Amphipoda       | 458.2 (141.9)| 1205.0 (402.9)| 75.6 (15.1)| 59.2 (16.8)|          |          |
|          | Chironomidae    | 15.4 (7.5)| 19.8 (12.3)| 0.8 (0.6)| 0.8 (0.6)|          |          |
|          | Corixidae       | 105.7 (103.0)| 30.8 (12.7)| 26.8 (19.2)| 21.6 (8.5)|          |          |
|          | Ephemereellidae | 6.6 (6.6)| 0         | 0        | 0         |          |          |
|          | Hydropsychidae  | 0         | 2.2 (2.2)| 0        | 0         |          |          |

Values in parentheses represent one standard error.
Discussion

We found that littoral macroinvertebrate assemblages were not significantly influenced by boating-related activity at primary or secondary boat ramps. We expected to observe differences in macroinvertebrate abundance and diversity related to changes in habitat structure, namely benthic substrate used by invertebrates for shelter (Cummins and Merritt 1996). Instead, substrate composition was similar among sites and dominated by sand and gravel. Larger, primary boat ramps with greater parking capacity and amenities facilitated more intense use that did not result in changes to substrate composition or macroinvertebrate community structure. Our study was restricted to only one sampling period, but the sampling window coincided with the late summer boating season, and allowed us to assess potential impacts during a high-use period.

Few studies have attempted to document impacts of boating-related activity on littoral macroinvertebrates of natural lakes. A related study found family-level differences in macroinvertebrate assemblages between boat ramp sites and nearby control sites and attributed the differences to localized pollution from boating activity (Forster 2005). Other potential mechanisms known to impact littoral macroinvertebrates are related to physical disturbance. Boating activity related to launching and loading boats can temporarily suspend sediments around boat ramps (Moss 1977; Liddle and Scorgie 1980; Hilton and Philips 1982; Kuss et al. 1990) that can cause respiratory impairment in invertebrates (Murphy et al. 1995; Liddle 1998). In naturally turbid systems, however, it was shown that short-term exposure to high turbidity levels has no adverse effect on damselflies, mayflies, and caddisflies that are adapted to high sediment load in the water column (Suren et al. 2005). Boating activity is also known to displace aquatic invertebrates, but they can quickly re-colonize areas after disturbances (Bishop 2008). A likely reason why invertebrate communities were similar between primary and secondary boat ramps was the depauperate nature of invertebrate assemblages in the study lakes. Although the low taxonomic resolution in our analysis may have masked some variability in invertebrate composition, particularly for Chironomidae (Carter and Resh 2001; Jones 2008), this effect likely had little impact on our results given the relatively low number of

![Figure 2. Relationship between mean invertebrate abundance measured at primary and secondary boat ramps in five glacial lakes in eastern South Dakota (see Table 3). Invertebrates were collected using a stovepipe sampler (solid circles, no./m²) or sweep net sampler (solid triangles, no./sweepnet).](image-url)
invertebrate taxa collected (i.e. seven dominant taxa), many of which are typically represented by only a few species in the PPR (e.g. Amphipoda, Caenidae, Corixidae, Hirudinea, and Sphaeriidae).

Environmental conditions of the PPR tend to support aquatic invertebrate communities with limited diversity comprised of highly adaptive taxa (Tangen et al. 2003). Aquatic invertebrate communities of the PPR inhabit a mosaic of dynamic freshwater habitats (e.g. temporary wetlands, permanent wetlands, and lakes) that vary widely in salinity, water-level fluctuation regimes, temperature, and seasonal oxygen levels (Meding and Jackson 2003; Euliss et al. 2004). A study of thirty shallow lakes of the Nebraska Sandhills Region by Paukert and Willis (2003) revealed that the lakes contained a depauperate macroinvertebrate assemblage, dominated by Gastropoda (68.9%), Chironomidae (18.9%), and Oligochaeta (8.0%).

Aquatic macroinvertebrate communities in the PPR have also proven resilient to large-scale anthropogenic changes, illustrating the robust nature of these communities to disturbance. Bataille and Baldassarre (1993) found that aquatic invertebrates were abundant and widely distributed across three wetland types (i.e. seasonal, semi-permanent, and permanent) following a three-year drought, indicating the resilience of aquatic invertebrates in the region to disturbance. In an attempt to develop an aquatic macroinvertebrate index of biotic integrity for semi-permanent wetlands in North Dakota, Tangen et al. (2003) found similar invertebrate communities across disturbance regimes ranging from 100% grassland to 100% row-crop agriculture. In the PPR climatic disturbances including drought and deluge periods may play a larger role in shaping invertebrate communities than land use. For example, Chironomidae communities of lowland Saskatchewan lakes were more heavily influenced by climatic conditions than agricultural or urban influences (Quinlan et al. 2002).

A major influence related to boating activity is wave action and sediment disturbance in near-shore areas of lakes. Similarly, large, shallow glacial lakes in our study typically experience significant wind/wave action owing to natural, climatic conditions in the PPR. Thus, invertebrate communities in these lakes are likely well-adapted to wind/wave action in littoral areas and may represent poor indicators of shoreline disturbance associated with boating activity.

Given the relatively small scale of shoreline disturbance, any impacts related to boat ramps and associated activities are likely localized and temporary. Conversely, environmental conditions such as climate, water quality, and(or) habitats of the PPR likely overwhelm any influence that small-scale shoreline disturbance would have on aquatic invertebrate communities. Aquatic invertebrate communities in glacial lakes may be well-adapted to a large range of environmental conditions, and thus not affected by relatively small-scale disturbances associated with recreational boating activity near public-use boat ramps. Our investigation of the effects of boating activity revealed that resident macroinvertebrate communities were similar between low and high use areas. Standardized sampling across multiple sampling periods would be needed to more fully assess the influence of boating activity and environmental factors on invertebrate communities.

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