Short-term Response of a Downstream Marine System to Opening a Tidal-River Causeway

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Abstract

The spill gates of the causeway on the Petitcodiac River in New Brunswick, Canada, were permanently opened in April 2010. We examined the short-term effect this had on downstream intertidal mudflats of the upper Bay of Fundy. Specifically, a before-after-control-impact design was used to determine if the causeway opening affected the invertebrate community (crustaceans, polychaetes, and molluscs), abiotic sediment conditions (sediment water content, mean particle size, penetrability, and aRPD depth), or resource availability (sediment chlorophyll a concentration, and organic matter content) of 5 intertidal mudflats (2 impacted sites, 3 reference sites) up to 5 months post-opening. We detected no biologically or statistically
meaningful differences between impacted and reference sites for any of the measured variables. This suggests that opening the causeway did not have an impact on these intertidal mudflats, at least within half a year of the opening. We speculate that this is likely a result of the macrotidal nature of the Bay of Fundy, that overwhelmed any changes to hydrodynamics and sedimentation which occurred after the opening of the causeway.

**Key Words:** Causeway, Community composition, Community dispersion, Dam, Infauna, Intertidal mudflats, Resource availability, Sediment conditions, Tidal barrier

**Introduction**

Causeway or dam construction has substantial effects on river systems, and detailed reviews can be found in Bednarek (2001), Poff and Hart (2002), Pess et al. (2008), and Lejon et al. (2009). Although less well studied, impacts have also been observed on downstream marine systems. Blocking of coastal rivers dampens tidal dynamics and seasonal flooding, limiting mobility and migration of coastal species (Bednarek 2001). Altered hydrodynamics and decreased sediment transport affect sediment and water discharge into marine systems (Fu et al. 2007; Jiao et al. 2007; Poff and Hart 2002), potentially altering sea levels (Poff and Hart 2002). Large reservoirs created by dams often decrease the quantity of nutrients discharged into marine systems (Bednarek 2001; Stanley and Doyle 2002). Alteration of river flow patterns, as well as sediment and nutrient inputs into marine systems can also result in coastal habitat loss and effects on the biotic community (Bednarek 2001).

As river blockages around the world age, economic and ecological reasons to remove these barriers are becoming increasingly compelling (Doyle et al. 2005; Lejon et al. 2009; Stanley and Doyle 2003). While such remediation represents a return of the system to a more natural state,
removing river obstructions will have obvious and potentially negative impacts on surrounding ecosystems (Bednarek 2001; Lejon et al. 2009; Stanley et al. 2002). For instance, increased river flow following dam removal may increase erosion and sediment deposition downstream (Lejon et al. 2009; Shuman 1995). In the short term, this may suffocate biota and damage freshwater habitats or spawning grounds (Bednarek 2001; Stanley and Doyle 2003; Thomson et al. 2005). Recovery to pre-dam conditions will vary by species, and if it occurs, may take years or decades (Doyle et al. 2005).

Removal of river blockages has been observed to alter marine hydrodynamics, including tidal dynamics (McAlice and Jaeger 1983; Sucsy et al. 1993), and sediment discharge into marine systems (Haralampides and Rodriguez 2006; Morand and Haralampides 2006). Augmented sediment discharge may increase fine-grained sediment deposition on marine beaches or tidal flats (Bednarek 2001). Sediment rich in nitrogen and phosphorus released following dam removal may alter nutrient availability in coastal ecosystems (Stanley and Doyle 2002). Thus, removal of river barriers will impact more than just freshwater systems. However, to the best of our knowledge, no studies have assessed the impact of causeway or dam removal on both the biotic and abiotic components of a downstream marine coastal system. As interest in barrier removal increases (Doyle et al. 2005; Lejon et al. 2009; Stanley and Doyle 2003), more information is required on the effects that these remediations may have on various marine systems.

In 1968, a causeway was built on the Petitcodiac River between Moncton and Riverview, New Brunswick, Canada. Constructed of rock-fill, this causeway incorporated five spill gates, as well as a fishway (Locke et al. 2003; Wells 1999), offering flood protection to farmland, providing a second transportation link between Moncton and Riverview, and creating a freshwater headpond
(Wells 1999). The fishway proved ineffective, and the causeway was ultimately responsible for the collapse of a genetically unique population of Atlantic Salmon (*Salmo salar*), and extirpation of American Shad (*Alosa sapidissima*) and Dwarf Wedgemussel (*Alasmidonta heterodon*) (Locke et al. 2003; Wells 1999). The causeway also resulted in increased erosion along the banks of the headpond, dampening of natural tidal cycles, decreased river flow velocity, and low levels of heavy-metal and fecal coliform contamination (Bray et al. 1982; Locke et al. 2003; van Proosdij et al. 2009; Wells 1999).

In an effort to restore more natural flow conditions to the Petitcodiac River, the causeway spill gates were permanently opened in April 2010, timed to take advantage of the spring freshet. The opening led to increased discharge of sediment into the Bay of Fundy (AMEC 2013; AMEC 2015), where it may interact with intertidal mudflats. Given the potential for this discharge to alter the hydrodynamics of the Bay of Fundy, as well as sedimentation and erosion processes on intertidal mudflats, we initiated a study to monitor any resulting changes in mudflat environmental conditions and community composition. Such knowledge is important in a conservation context, because mudflat residents form the base of a food web that supports several commercial fish species (McCurdy et al. 2005; Risk and Craig 1976) and numerous migratory shorebirds (Hicklin 1987).

We examined five intertidal mudflats in the upper Bay of Fundy, two located near the Petitcodiac River that may have been affected by the opening of the causeway, and three that are isolated from potential impact. Using a multivariate, before-after-control-impact design (Underwood 1994), we investigated the biological community, abiotic sediment conditions, and resource availability to evaluate if opening the causeway had short-term (up to 5 months) impacts on these intertidal mudflats.
Methods

Study sites

The Petitcodiac River and its five main tributaries extend ~ 175 km above the causeway, draining ~1360 km² (Locke et al. 2003). Approximately 40 km downstream from the causeway, the river discharges into the Atlantic Ocean in the most northwestern part of Chignecto Bay (Shepody Bay) in the upper Bay of Fundy, Canada. Our study was conducted on five mudflats (hereafter “sites”) in Chignecto Bay, that comprised a subset of sites from a broader project investigating mudflat community dynamics (Gerwing et al. 2015a; Gerwing et al. 2016). Two mudflats are located near where the Petitcodiac River discharges into the Bay of Fundy (Daniels Flats (DF) and Grande Anse (GA) in Shepody Bay), and were therefore potentially influenced by opening the causeway. Two other mudflats, Pecks Cove (PC) and Minudie (MN), in the Cumberland Basin, are protected from potential effects of the causeway opening by the Dorchester Peninsula (Figure 1). Mary’s Point (MP), another reference mudflat sheltered from impacts via a jutting landform (Figure 1), is located closer to the outer Bay of Fundy than the other mudflats. Impacted and reference sites were selected based on satellite images of sediment plume dispersal and deposition into the Bay of Fundy following causeway opening (AMEC 2013; AMEC 2015). Detailed information of the infaunal and epifaunal community dynamics, and environmental conditions of these intertidal mudflats can be found in Gerwing et al. (2013), and Gerwing et al. (2015a).

Mudflat sampling

Fauna
The five mudflats described above were sampled in 2009 and 2010. Sampling rounds (a period of ~ 5 days; hereafter “Round”) occurred at approximately the same time each year, and samples were collected at the following times: 1: early June 2009 and 2010; 2: late June 2009 and 2010; 3: mid-July 2009 and 2010; 4: early August 2009 and 2010; 5: late August/early September 2009 and 2010 (see Gerwing et al. (2015a) for exact dates). At each mudflat, two permanent transects were established, running perpendicular to the low water line, and divided into four equal zones for random stratified sampling. Three sampling locations (termed “plots”; 1 m²) were randomly selected per zone, for a total of 12 plots per transect (n = 24 per site, 1200 total). Densities of the Eastern Mudsnail (*Nassarius obsoletus*) were counted *in situ*. At each plot, a 7-cm diameter corer was pushed into the sediment, and within 12 h of collection, samples were passed through a 250-µm sieve. Invertebrates were later identified in preserved samples as in Gerwing et al. (2015a). More details of the sampling scheme can be found in Gerwing et al. (2015a).

**Sediment properties**

Penetrability of the sediment and depth of the apparent redox potential discontinuity (aRPD), an index of sediment dissolved oxygen content (Gerwing et al. 2015b), were measured in each plot as described in Gerwing et al. (2015a) and Gerwing et al. (2013), respectively. Additional sediment properties (% organic matter, % water content, and mean particle size) and chlorophyll *a* concentration (an indication of benthic diatom abundance) were measured in one plot per zone (n = 8 per site, 400 total), as described in Gerwing et al. (2015a) and Coulthard and Hamilton (2011), respectively. Sediment organic matter (indication of detrital matter) content and chlorophyll *a* concentration are hereafter referred to as resources, while sediment water content, mean particle size, penetrability, and aRPD depth are referred to as abiotic sediment variables.
Structure of the data analysis

Due to the strong seasonal cycles of the infaunal community of these mudflats (Gerwing et al. 2015a), comparing the mudflat community from the months before and after the causeway opening may overwhelm detection of an impact. Since our sampling rounds occurred at the same time each year, we can compare the community patterns and sediment conditions between years for a given round (2009: before the causeway opening; 2010: after the causeway opening). We limit our analysis to the late spring and summer months of both years (pre- and post-opening) to minimize the seasonal trends mentioned above. If opening the Petitcodiac causeway gates has affected downstream sites, then there should be a greater between-year variation in biotic and abiotic variables at these impacted sites (i.e., exposed to the influence of the causeway opening: GA, DF) than at reference sites (i.e., protected from its influence: PC, MN, MP). For all analyses, we used $\alpha = 0.05$.

Did opening the causeway influence mudflat biota?

Community composition

We used the statistical program PRIMER V6 with the PERMANOVA (Permutational Multivariate Analysis of Variance) add-on (McArdle and Anderson 2001) to examine if the intertidal mudflat community was influenced by the causeway opening. We included in our analysis one species of epifauna ($N. obsoletus$), and the following infaunal taxa: $Macoma$ spp., $Corophium volutator$, Copepoda, Ostracoda, and Polychaetes (families Capitellidae, Spionidae, Cirratulidae, Maldanidae, Nereididae, Nephtyidae, Phyllodocidae, Glyceridae, Goniadidae, and Orbinidae). For each combination of site and round (e.g., GA in mid-July for both years), a resemblance matrix was calculated from the densities of the fauna using Bray-Curtis coefficients.
(Clarke et al. 2006). A dummy variable of 1, which can be considered a “dummy species”, was added to deal with plots with zero densities (Clarke and Gorley 2006). Data were fourth root transformed to improve assessment of contribution of rare and common taxa to community structure. SIMPER (Similarity Percentages; Clarke & Ainsworth 1993, Clarke 1993) was used to quantify the percent dissimilarity (%) of the biological community between years. Percent dissimilarity ranges from 0-100%, and incorporates fauna presence/absence as well as density. If the opening of the causeway greatly influenced the biota of these sites, then the between-year percent dissimilarity would be higher at impacted sites (Causeway: GA, DF) than at reference sites (Reference: PC, MP, MN). A univariate, randomized-block ANOVA was conducted in Minitab V16, with percent dissimilarity was the response variable, Site (5 levels) was a fixed factor, and Round (5 levels: 3 week intervals between early June and late August/early September) was a random factor. Normality of residuals was assessed visually and homogeneity of variance was examined using Cochran’s test (Underwood 1997). A significant site effect was investigated using four logical and orthogonal planned contrasts (Underwood 1994) as follows: (1) the two reference sites in Cumberland Basin (PC vs. MN), (2) the two Cumberland Basin reference sites vs. the reference site nearest the outer Bay of Fundy (MP), (3) the two impacted sites (DF vs. GA), and (4) the Impacted sites (DF, GA) vs. the Reference sites (PC, MN, MP). This set of contrasts enables us to properly assess the dynamics in our various reference sites, in our impacted sites, and between site types (impacted vs. reference). The latter contrast is the one of primary interest.

Community dispersion

Statistically, community dispersion measures the distance between an individual sample and the group centroid (Fraterrigo and Rusak 2008; Warwick and Clarke 1993); i.e., it is a measure of
multivariate variance or community heterogeneity. Disturbed communities often have higher
dispersion compared to undisturbed communities (Fraterrigo and Rusak 2008; Underwood 1994;
Warwick and Clarke 1993); thus, examining community dispersion is another way to examine
effects of opening the causeway gates on our mudflat communities. For each combination of
year, site and round, a resemblance matrix was calculated for the faunal community (Bray-Curtis
coefficients, dummy variable, fourth root transformation), and then the community dispersion
was calculated using PERMDISP in PRIMER (Anderson et al. 2008; Fraterrigo and Rusak
2008). A randomized-block ANOVA was then used to determine if impacted sites (GA, DF) had
higher community dispersion in the year following the opening of the causeway (2010) than
reference sites (PC, MN, MP); community dispersion was the response variable, Year (2 levels:
2009, 2010) and Site (5 levels) were fixed factors, and Round (5 levels) was a random factor.
The term of primary interest in this analysis is the Year*Site interaction.

Did opening the causeway influence abiotic sediment conditions or resource availability on
the mudflats?

As described above for community composition, for each combination of site and round,
separate resemblance matrices were calculated for sediment conditions (sediment water content,
mean particle size, penetrability, and aRPD depth) and for resource variables (sediment
chlorophyll $a$ concentration, and organic matter content). For both types of datasets, the data
were normalized, resemblance matrices were calculated using Euclidean distance, and SIMPER
was used to calculate the differences (Average Squared Distances) between years for each site-
round combination. Randomized-block ANOVAs were then run with Average Squared Distance
values as the response variable, followed by planned contrasts as described above.
Power analysis

To further investigate non-significant results for effects of interest following ANOVA, we conducted power analyses (Zar 1999). We assessed the power of our design to detect 50%, 20% and 10% differences between reference and impacted sites for community dissimilarity, community dispersion, and differences in sediment conditions and in resource availability. Specifically, for each response variable, we calculated a minimum detectable difference (δ) from the average of the reference sites for each of these percent differences, and used the appropriate variance from the ANOVA (MS used as the denominator for the effect or contrast of interest) to estimate power.

Results

Did opening the causeway influence mudflat biota?

Between-year percent dissimilarity for the invertebrate community varied among sites (Table 1, Figure 2a); however, planned contrasts indicated that this variation occurred amongst the reference sites (specifically, the Cumberland Basin sites vs. Mary’s Point). Variation in impacted sites did not exceed natural variation in the reference sites before or after the causeway opening. Our design had 85% power to detect a 20% difference in community dissimilarity between reference and impacted sites (which is a δ = 7.6% community dissimilarity, based on the average for the reference sites; Table 3). This minimum detectable difference (δ) was less than the observed difference between the maximum and minimum averages per site (a difference of 10.8% community dissimilarity; Figure 2a).

For community dispersion (Table 2), the interaction between year and site (the contrast of interest) approached significance (p=0.069). Our design had greater than 99% power to detect a
difference in community dispersion of 11.8 (a 50% difference; Table 3); a difference in community dispersion of 13.2 was observed between the maximum and minimum year-site averages (Figure 3). Despite the high power and statistical non-significance, we further investigated the almost significant year, site interaction with post hoc comparisons (Table 2). The only statistically significant relationship observed was the interaction between year and the PC vs MN contrast (both reference sites). The other contrast, including those between reference and impacted sites, were not significant.

**Did opening the causeway influence abiotic sediment conditions or resource availability on mudflats?**

Before-after difference (average squared distance) in abiotic sediment conditions was significantly affected by site (Table 1). This pattern was due to variation among reference sites; specifically, Pecks Cove had a higher between-year difference than other reference sites; Figure 2b). This led to a detected difference in abiotic sediment conditions between impacted sites and reference sites (Table 1). The trend for this contrast was opposite to that predicted under the hypothesis that opening the causeway gates would alter sediment conditions; on average, impacted sites differed less between years than did reference sites (Figure 2b). Our design had over 90% power to detect a difference equivalent to the observed difference between the maximum and minimum site averages (a difference in average squared distance of 1.3; Figure 2b; Table 3).

Sites also significantly affected the before-after difference (average squared distance) in resource availability (Table 1). the detected trend was due to variation among reference sites (with Pecks Cove having a larger before-after difference than other reference sites; Figure 2c). The contrast
for impacted sites vs reference sites revealed no significant difference between site types. Our design had approximately 90% power to detect a difference in average squared difference of 0.9 (a 20% difference; Table 3), which is equivalent to the observed difference between the maximum and minimum site averages (Figure 2c).

**Discussion**

Opening the Petitcodiac causeway appeared to have no discernible short-term impact on the invertebrate community, abiotic sediment conditions, or resource availability of the intertidal mudflats in the upper Bay of Fundy. Furthermore, between-year differences for these biotic and abiotic variables observed in this study are within the range observed for mudflats located elsewhere in the Bay of Fundy (Gerwing et al., 2015a; Supplemental Figure S1, and Supplemental Table S1), further suggesting that opening the causeway did not have a short-term impact on nearby intertidal mudflats.

Absence of a short-term impact of opening the causeway on these mudflats likely stems from the macrotidal nature of the Bay of Fundy, where tidal amplitudes range from 8-15 m (Bleakney 1972; Desplanque and Mossman 2004). Any changes in hydrodynamics that occurred when the causeway was opened were likely overwhelmed by the substantial daily water movement in and out of the Bay (Dashtgard et al. 2014; Desplanque and Mossman 2004; Wu et al. 2011). In fact, sediment deposition after opening the causeway appears to be primarily occurring subtidally in the middle of Shepody Bay, not on the intertidal mudflats (AMEC 2013; AMEC 2015). Long-term monitoring (being conducted by MA Barbeau) is still required to evaluate longer-term impacts. It should also be noted that only the causeway spill gates have been opened (AMEC 2013; AMEC 2015), the causeway itself has not yet been removed. Removal of the causeway
would likely result in another substantial release of sediments, which could potentially impact the intertidal mudflats in the upper Bay of Fundy.

To the best of our knowledge, our study is the first empirical test of the effect of river blockage removal (dam, causeway, etc.) on multiple biotic and abiotic components of a downstream, coastal marine system. Our findings are encouraging, as there is growing interest in removing tidal barriers, and evidence that such actions are not immediately detrimental to existing marine communities provides important information for decision makers. As of 1999 (the most recent survey), 25 of the 44 major rivers around the Bay of Fundy contained a barrier (causeway, bridge, log dam, etc.) of some sort (Wells 1999). Globally, there are also a large number of aging dams that are being considered for removal (Doyle et al. 2005; Lejon et al. 2009; Stanley and Doyle 2003). Removing these barriers has the potential to return rivers to more natural conditions, and enable recovery of native species and commercial/recreational fisheries (Locke et al. 2003; Stanley et al. 2007). However, Stanley and Doyle (2002) pointed out that removing river blockages (dams or causeways) should not naively be considered as simply restoring “natural” river conditions. Potential disturbance to the biological community established following the construction of the blockage should be considered before action is taken.

Studies like the one presented here are important to complete in other areas, as our findings may not be applicable to all marine systems, especially those not dominated by macrotides. River blockage removal has been observed to have both short and long-term impacts on marine hydrodynamics in other systems (McAlice and Jaeger 1983; Sucsy et al. 1993). Therefore, detailed environmental assessments conducted prior to causeway/dam removal, including incorporating proper assessment of statistical power when necessary (Peterman 1990), should investigate potential changes to marine system (even when far downstream), not just freshwater
habitats, to ensure restoration of more natural river conditions does not damage downstream marine systems.

Acknowledgements

We thank Doug Prosser and AMEC Foster Wheeler in Fredericton, New Brunswick, for their input on this project and providing information on the status of the Petitcodiac River after the causeway opening. Sources of funding and field and lab assistants for the collection and processing of samples are acknowledged in Gerwing et al. (2015a, 2016). TGG was supported by funds from the Natural Science and Engineering Research Council of Canada (Discovery grants to MAB, DJH and KH), a MITACS ELEVATE postdoctoral fellowship while analysing the data for and writing this manuscript.

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Figure 1: Schematic maps of the Bay of Fundy and Chignecto Bay (in the upper Bay of Fundy), Canada, showing location of the intertidal mudflat sites (squares) and Petitcodiac River causeway (star). The sites are Mary’s Point (MP), Daniels Flats (DF), Grande Anse (GA), Pecks Cove (PC), and Minudie (MN).
Figure 2: Mean ± SE (n = 5 rounds) for between-year a) invertebrate community dissimilarity (percent dissimilarity), b) differences in abiotic sediment conditions (average squared distance; % water content, mean particle size, penetrability and aRPD depth of the sediment), and c) differences in resource availability (average squared distance; chlorophyll $a$ concentration and organic matter content of the sediment) of intertidal mudflats in the upper Bay of Fundy, Canada. Impacted sites are those located near the Petitcodiac River, and were thus potentially impacted by the opening of the Petitcodiac causeway in April 2010. Reference sites are those which were unlikely to be impacted by the opening. The 5 sampling rounds are at 3-week intervals between June and September in 2009 (pre-opening) and 2010 (post-opening). See Figure 1 for full site names.
Figure 3: Mean ± SE (n = 5 rounds between June and September) community dispersion for the invertebrate assemblage of intertidal mudflats of the Bay of Fundy, Canada. Impacted sites are located near the mouth of the Petitcodiac River, and were thus potentially impacted by the opening of the Petitcodiac causeway in April 2010. Reference sites were unlikely to be impacted by the opening. See Figure 1 for full site names.
Table 1: ANOVA results evaluating effect of site (impacted or reference sites) on dissimilarity of the invertebrate community composition (%), differences in abiotic sediment conditions (average squared distance) and differences in resource availability (average squared distance) of intertidal mudflats pre- and post-opening of the Petitcodiac causeway (in April 2010). See Figure 1 for site names and locations in Chignecto Bay, upper Bay of Fundy, Canada. The planned contrasts are orthogonal; the one of primary interest is bolded. Round refers to sampling times between June and September, and is a random factor. The error term for Site and for the contrasts is the Site*Round interaction. Random sources of variation cannot be tested in this design.
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Table 2: ANOVA results evaluating effect of year (2009, 2010) and site (impacted or reference sites) on community dispersion (i.e., multivariate variance) of invertebrates inhabiting intertidal mudflats before and after the opening of the Petitcodiac causeway (in April 2010). See Figure 1 for site names and locations in Chignecto Bay, upper Bay of Fundy, Canada. The term of interest in this analysis is the Year*Site (fixed-factor) interaction, and is bolded. Round refers to sampling times between June and September, and is a random factor. The error term for the Year*Site interaction is the Year*Site*Round interaction. Random sources of variation cannot be tested in this design.

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<td>160.23</td>
<td>13.36</td>
<td>0.0001</td>
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<tr>
<td>Round</td>
<td>4</td>
<td>94.37</td>
<td>5.22</td>
<td>0.026</td>
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<tr>
<td><strong>Year*Site</strong></td>
<td>4</td>
<td><strong>14.63</strong></td>
<td><strong>2.69</strong></td>
<td><strong>0.069</strong></td>
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<tr>
<td>Year*(PC vs MN)</td>
<td>1</td>
<td>19.80</td>
<td>13.53</td>
<td>0.02</td>
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<tr>
<td>Year*((MN,PC) vs MP)</td>
<td>1</td>
<td>1.51</td>
<td>0.34</td>
<td>0.564</td>
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<tr>
<td>Year*(GA vs DF)</td>
<td>1</td>
<td>16.02</td>
<td>1.44</td>
<td>0.322</td>
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<tr>
<td>Year*((MN,PC, MP) vs (GA, DF))</td>
<td>1</td>
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<td>0.86</td>
<td>0.37</td>
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<tr>
<td>Site*Round</td>
<td>16</td>
<td>12.00</td>
<td>2.20</td>
<td>0.062</td>
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<tr>
<td>Year*Round</td>
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<td>11.54</td>
<td>2.12</td>
<td>0.126</td>
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<tr>
<td>Year<em>Site</em>Round</td>
<td>16</td>
<td>5.45</td>
<td>No Test</td>
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</tbody>
</table>
Table 3: Power analysis for the impacted vs. reference sites contrast in the ANOVAs in Table 1 and for the Year*Site interaction in Table 2, evaluating impacts of the opening of the Petitcodiac causeway. Power was calculated for $\alpha = 0.05$, error DF = 16, variance = error MS (from Tables 1 and 2), and a minimum detectable difference ($\delta$) of 50%, 20% and 10%.

<table>
<thead>
<tr>
<th>Response Variable</th>
<th>Average Reference Value</th>
<th>50% Difference</th>
<th>20% Difference</th>
<th>10% Difference</th>
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</thead>
<tbody>
<tr>
<td>Community Dissimilarity (%)</td>
<td>37.8</td>
<td>18.9</td>
<td>&gt;99</td>
<td>7.6</td>
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<tr>
<td>Community Dispersion</td>
<td>23.5</td>
<td>11.8</td>
<td>&gt;99</td>
<td>4.7</td>
</tr>
<tr>
<td>Sediment Conditions (Average Squared Distance)</td>
<td>9.3</td>
<td>4.7</td>
<td>&gt;99</td>
<td>1.9</td>
</tr>
<tr>
<td>Resource Availability (Average Squared Distance)</td>
<td>4.5</td>
<td>2.3</td>
<td>&gt;99</td>
<td>0.9</td>
</tr>
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</table>