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Short-Term Response of a Downstream Marine System to the Partial Opening of a Tidal-River Causeway

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Abstract The spillway gates of the Petitcodiac Causeway, a hydraulic structure ~35 km upstream of the mouth of the Petitcodiac River in New Brunswick, Canada, were permanently opened in April 2010. The short-term effect opening the spillway gates had on downstream intertidal mudflats of the upper Bay of Fundy was investigated. Specifically, a multivariate before-after-control-impact design was used to determine if opening the spillway gates affected the invertebrate community (crustaceans, polychaetes, and molluscs), abiotic sediment conditions (sediment water content, mean particle size, penetrability, and depth of the apparent redox potential discontinuity), or resource availability (sediment chlorophyll *a* concentration and organic matter content) of five intertidal mudflats (two impacted sites, three reference sites) spanning Chignecto Bay, the northern arm of the upper Bay of Fundy, up to 5 months post-opening. No biologically or statistically meaningful differences were detected between impacted and reference sites for any of the measured variables. This suggests that opening the causeway did not have a quantifiable impact on these intertidal mudflats, at least within half a year of the opening. This is likely a result of the macrotidal nature of the Bay of Fundy that overwhelmed any immediate changes to hydrodynamics that occurred after the opening of the causeway gates.

Keywords Dam · Infauna · Intertidal mudflats · Sediment conditions · Tidal barrier

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Introduction

Causeway or dam construction can have 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 (Bednarek 2001). 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 (Poff and Hart 2002; Fu et al. 2007; Jiao et al. 2007), potentially altering sea levels (Poff and Hart 2002). Large reservoirs created by dams often decrease the quantity of nutrients discharged into marine systems (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 hydraulic structures around the world age, economic and ecological reasons to remove these barriers are becoming increasingly compelling (Stanley and Doyle 2003; Doyle et al. 2005; Lejon et al. 2009). 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; Stanley et al. 2002; Lejon et al. 2009). For instance, increased river

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flow following barrier removal may increase erosion and sediment deposition downstream (Shuman 1995; Lejon et al. 2009). 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-barrier conditions will vary by species, and if it occurs, may take years or decades (Doyle et al. 2005).

Removal of hydraulic structures alters marine hydrodynamics, including tidal dynamics (McAlice and Jaeger 1983; Sucsy et al. 1993) and sediment discharge (Haralampides and Rodriguez 2006; Morand and Haralampides 2006). Augmented sediment discharge may increase fine-grained sediment deposition on beaches or tidal flats (Bednarek 2001). Sediment rich in nitrogen and phosphorus released following dam removal may also 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 peer-reviewed studies have assessed the impact of barrier removal on both the biotic and abiotic components of a downstream marine coastal system. As interest in barrier removal increases (Stanley and Doyle 2003; Doyle et al. 2005; Lejon et al. 2009), more information on the effects that these remediations will have on marine systems is required.

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 spillway gates, as well as a fishway (Wells 1999; Locke et al. 2003). It offered flood protection to farmland, a second transportation link between Moncton and Riverview, and created a freshwater headpond (Wells 1999). The fishway was 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*) (Wells 1999; Locke et al. 2003). 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; Wells 1999; Locke et al. 2003; van Proosdij et al. 2009).

The spillway gates of the Petitcodiac River causeway were permanently opened in April 2010, timed to take advantage of the spring freshet, in an effort to promote fish passage and resolve other ecosystem issues in the Petitcodiac River (AMEC 2005; AMEC 2011). Partial or full removal of the causeway itself is planned for the future. The opening led to increased discharge of sediment into the Bay of Fundy (AMEC 2011; AMEC 2013; AMEC 2015), where it may have affected 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 (Risk and Craig 1976; McCurdy et al. 2005) and numerous migratory shorebirds (Hicklin 1987; Quinn and Hamilton 2012; Gerwing et al. 2016b).

We examined five intertidal mudflats in the macrotidal Chignecto Bay, the northern arm of the upper Bay of Fundy. Two mudflats were located near the Petitcodiac River and may have been affected by the opening of the causeway, and the other three were 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 km^2 (Locke et al. 2003). Approximately 35 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. The Bay of Fundy is a macrotidal ecosystem, where semidiurnal tides range in amplitude from 8 to 15 m in the upper Bay (Bleakney 1972; Desplanque and Mossman 2004). The water current patterns are mostly tidally driven, being parallel to shore off the intertidal areas, and depth-average water velocities can reach 1.8–2 m s^{-1} at some locations in Chignecto Bay during periods of peak water motion (Dupont et al. 2005). Prior to opening of the causeway spillway gates (May 2009 to March 2010), ~86,000 t of sediment was transported in a net upstream direction (AMEC 2011). Following the spillway gate opening (April 2010), ~40,000 t of sediment had moved in a net downstream direction by March 2011 (AMEC 2011). This movement resulted in a deposition of sediment (0.5–1 m) on some downstream intertidal mudflats within 1 year of opening (AMEC 2011).

Our study was conducted on five mudflats (hereafter "sites") in Chignecto Bay. These intertidal mudflats are a subset of sites from a broader project investigating mudflat community dynamics in the upper Bay of Fundy (Gerwing et al. 2015a; Gerwing et al. 2015c; Gerwing et al. 2016a). 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 (Fig. 1). Two other mudflats, Pecks Cove (PC) and Minudie (MN), in the

Cumberland Basin, are assumed to be protected from potential effects of the causeway opening by the Dorchester Peninsula. Mary's Point (MP), another reference mudflat assumed to be sheltered from impacts by a jutting landform, is located closer to the outer Bay of Fundy than the other mudflats. We selected impacted and reference sites in Shepody Bay and the Cumberland Basin, respectively, partly based on observation that sediment mixing between adjacent bays appears minimal in the course of the research (Amos and Alfoldi 1979; AMEC 2011). Detailed information on 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

We sampled the five mudflats described above in 2009 and 2010. Sampling rounds (a period of ~5 days; hereafter "Round") occurred every 3 weeks at approximately the same time each year: (1) early June 2009 and 2010, (2) late June 2009 and 2010, (3) mid-July 2009 and 2010, (4) early August 2009 and 2010, and (5) late August/early September 2009 and 2010 (see Gerwing et al. (2015a) for exact dates). At each mudflat, two permanent transects perpendicular to the low water line were established, and divided them into four equal zones for random stratified sampling. Three sampling locations (termed "plots"; 1 m^2) were randomly selected per zone, for a total of 12 plots transect-1 ($n = 24$ per site, 1200 total) as described in Gerwing et al. (2015a). We counted densities of the Eastern Mudsnail (*Tritia obsoleta* [=*Nassarius obsoletus*; Galindo et al. 2016]) 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 retained in the sieve were preserved and later identified as in Gerwing et al. (2015a).

Refer to Gerwing et al. (2015a) where more details of the sampling can be found.

Sediment Properties

We assessed penetrability of the sediment by dropping a metal rod (15 cm long, 1 cm diameter, 330 g) from a height of 0.74 m above the substratum, and recording the depth (mm) of penetration into the sediment (Gerwing et al. 2015a). Depth of the apparent redox potential discontinuity (aRPD; the transition from gray/green or black sediment to brown/red sediment), an index of sediment dissolved oxygen content (Gerwing et al. 2015b), was measured in the void left in the sediment following removal of the 7-cm diameter core (for infaunal sampling) in each plot, as described by Gerwing et al. (2013). Additional sediment properties (% water content, % organic matter, and mean particle size) and chlorophyll *a* concentration (an indication of benthic diatom abundance) were measured in one plot zone-1 ($n = 8$ per site, 400 total), as described by Gerwing et al. (2015a) and Coulthard and Hamilton (2011), respectively. Sediment organic matter content (indication of detrital matter) 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

Strong seasonal cycles of the infaunal mudflat community (Gerwing et al. 2015a) may overwhelm detection of an impact of opening the Petitcodiac causeway gates. To avoid this, we compared the community patterns and sediment conditions between 2009 (before the causeway opening) and 2010 (after the opening) within rounds, which occurred at about the same time each year. Further, we limited our analysis to the late spring and summer months of both years. If opening the causeway gates has affected downstream sites, then there
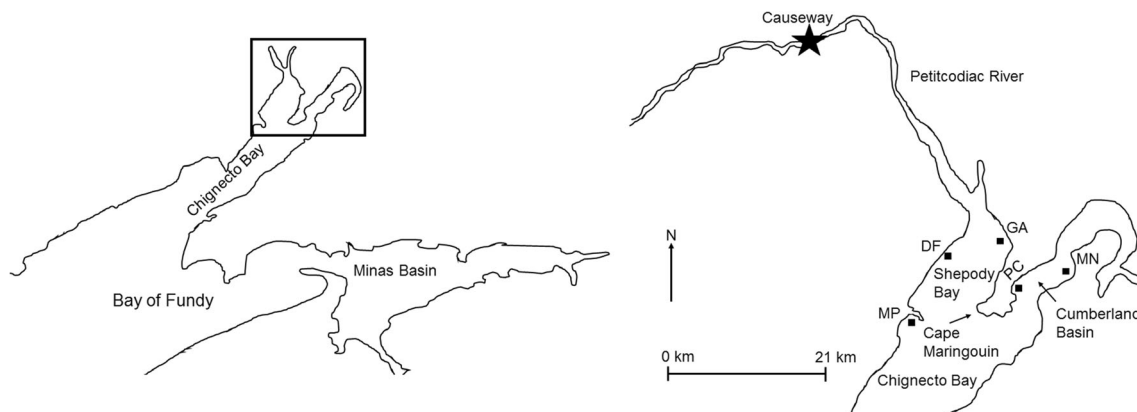
Fig. 1 Schematic maps of the Bay of Fundy and Chignecto Bay (in the upper Bay of Fundy), Canada, showing location of the intertidal mudflat sites (*squares*) and Petitcodiac River causeway (*star*). Mary's Point (*MP*: latitude 45.72°, longitude −64.67°), Daniels Flats (*DF*: 45.79°, −64.61°), Grande Anse (*GA*: 45.82°, −64.50°), Pecks Cove (*PC*: 45.75°, −64.49°), and Minudie (*MN*: 45.77°, −64.38°)

should be greater between-year variation in biotic and abiotic variables at impacted sites than at reference sites. We set α at $p = 0.05$ to evaluate main effects and contrasts, and 0.1 for interactions as a conservative measure to ensure that main effects were not inappropriately interpreted in situations where interactions were likely (Winer et al. 1991; Hamilton 2000).

Did Opening the Causeway Spillway Gates 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 (*T. obsoleta*), and the following infaunal taxa: *Macoma* spp., *Corophium volutator*, Copepoda, Ostracoda, and Polychaetes (families Capitellidae, Spionidae, Cirratulidae, Maldanidae, Nereididae, Nephtyidae, Phyllodocidae, Glyceridae, Goniadidae, and Orbiniidae). For each combination of site and round (e.g., GA in mid-July for both years), we calculated a resemblance matrix 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 the contribution of rare and common taxa to community structure. We used SIMPER (similarity percentages; Clarke 1993, Clarke & Ainsworth 1993) to quantify the percent dissimilarity (%) of the biological community between years. Percent dissimilarity ranges from 0 to 100 % and incorporates fauna presence/absence as well as density. If the causeway opening 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). We compared percent dissimilarity among sites using a univariate randomized-block ANOVA in Minitab V16, with site (five levels) as a fixed factor and round (five levels) as a random factor. Normality of residuals was assessed visually and homogeneity of variance was examined using Cochran's test (Underwood 1997); both assumptions were met. We investigated a significant site effect 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 (Warwick and Clarke 1993; Fraterrigo and Rusak 2008); it is a measure of multivariate variance or community heterogeneity. Disturbed communities often have higher dispersion compared to undisturbed communities (Warwick and Clarke 1993; Underwood 1994; Fraterrigo and Rusak 2008). 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, we calculated a resemblance matrix for the faunal community (Bray-Curtis coefficients, dummy variable, fourth-root transformation), and then calculated community dispersion using PERMDISP in PRIMER (Anderson et al. 2008; Fraterrigo and Rusak 2008). We used a randomized-block ANOVA 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). In this analysis, community dispersion was the response variable, year and site were fixed factors, and round was a random factor. The term of primary interest in this analysis is the Year*Site interaction, more specifically the contrast comparing site types (impacted vs. reference).

Did Opening the Causeway Spillway Gates Influence Abiotic Sediment Conditions or Resource Availability on the Mudflats?

As described above for community composition, for each combination of site and round, we calculated separate resemblance matrices 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 datasets, 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. We then ran randomized-block ANOVAs with average squared distance values as the response variable, followed by planned contrasts as described above.

Power Analysis

When non-significant results for effects of interest occurred, we conducted power analyses using equations found in Zar (1999). We assessed the power of our design to detect 50 and 20 % differences between reference and impacted sites for

community dissimilarity, community dispersion, sediment conditions, and resource availability. 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 Spillway Gates Influence Mudflat Biota?

Between-year percent dissimilarity for the invertebrate community varied among sites (Table 1, Fig. 2a); however, planned contrasts indicated that this variation occurred among 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 $\delta = 7.6$ % community dissimilarity, based on the average for the reference sites; Table 2). 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; Fig. 2a).

For community dispersion, differences among years varied with sites (Table 3, Year*Site interaction). None of the interaction contrasts were significant, although two approached significance, reflecting a trend of decreasing community dispersion post-impact compared to pre-impact in a reference site (Pecks Cove) and an impacted site (Grande Anse) (Fig. 3). Note that this before-after trend was opposite to that predicted

Table 1 ANOVA results evaluating effect of site (impacted vs. reference) on invertebrate community dissimilarity (%), differences in abiotic sediment conditions (average squared distance), and differences in resource availability (average squared distance) on intertidal mudflats pre- and post-opening of the Petitcodiac causeway spillway gates

Analysis	Source	DF	MS	F	p
Community dissimilarity	Site	4	97.50	6.85	0.002
	PC vs. MN	1	3.90	0.27	0.609
	(PC, MN) vs. MP	1	345.70	24.30	>0.001
	DF vs. GA	1	16.20	1.14	0.301
	Impacted (DF, GA) vs. reference (PC, MN, MP)	**1**	**24.20**	**1.70**	**0.210**
	Round	4	165.90		
	Site*Round	16	14.20		
	Total	24			
Abiotic sediment conditions	Site	4	1.39	5.19	0.007
	PC vs. MN	1	3.15	11.79	0.003
	(PC, MN) vs. MP	1	0.90	3.36	0.085
	DF vs. GA	1	0.08	0.29	0.598
	Impacted (DF, GA) vs. reference (PC, MN, MP)	**1**	**1.42**	**5.30**	**0.035**
	Round	4	2.18		
	Site*Round	16	0.27		
	Total	24			
Resource availability	Site	4	0.59	3.81	0.023
	PC vs. MN	1	0.78	5.02	0.040
	(PC, MN) vs. MP	1	1.10	7.04	0.017
	DF vs. GA	1	0.33	2.12	0.164
	Impacted (DF, GA) vs. reference (PC, MN, MP)	**1**	**0.16**	**1.04**	**0.322**
	Round	4	0.32		
	Site*Round	16	0.16		
	Total	24			

See Fig. 1 for site names and locations. 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
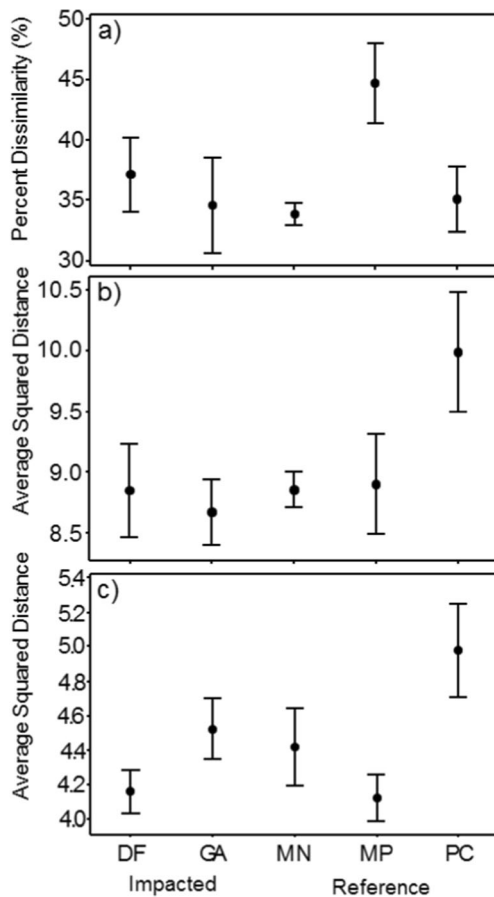
Fig. 2 Mean ± standard error (n = 5 rounds) for between-year **a** invertebrate community dissimilarity (percent dissimilarity), **b** differences in abiotic sediment conditions (average squared distance; % water content, mean particle size, penetrability, and aRPD depth of the sediment), and **c** differences 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 (DF and GA) located near the Petitcodiac River; reference sites (MN, MP, and PC) unlikely to be impacted by causeway opening. Sampling rounds were at 3-week intervals between June and September in 2009 (pre-opening) and 2010 (post-opening). See Fig. 1 for full site names

under the hypothesis that opening the causeway gates would cause stress to the mudflat community. Our design had >99 %

Table 3 ANOVA results evaluating effect of year (2009, 2010) and site (impacted vs. reference) on invertebrate community dispersion (i.e., multivariate variance) on intertidal mudflats pre- and post-opening of the Petitcodiac causeway spillway gates
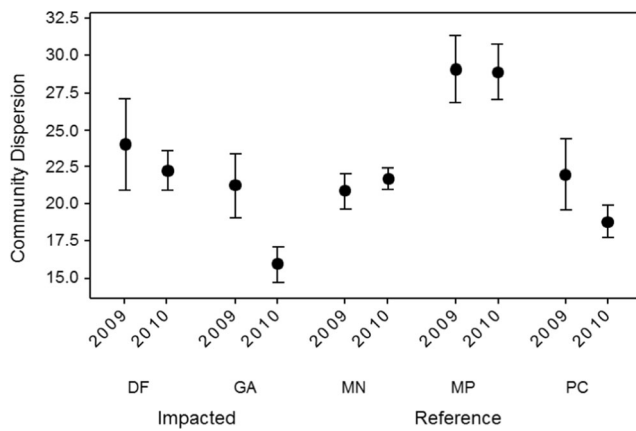
Source	DF	MS	F	p
Year	1	46.82	4.06	0.114
Site	4	160.23	13.36	<0.001
Year*Site	**4**	**14.63**	**2.69**	**0.069**
Year*(PC vs. MN)	1	19.80	3.63	0.075
Year*((PC, MN) vs. MP)	1	1.51	0.28	0.605
Year*(DF vs. GA)	1	16.02	2.94	0.106
Year*((DF, GA) vs. (PC, MN, MP))	**1**	**21.19**	**3.89**	**0.066**
Round	4	94.37		
Year*Round	4	11.54		
Site*Round	16	12.00		
Year*Site*Round	16	5.45		
Total	49			

See Fig. 1 for site names and locations. The Year*Site (fixed factor) interaction is of particular interest and is bolded. Round refers to sampling times between June and September, and is random. The error term for the Year*Site interaction and associated contrasts is the Year*Site*Round interaction. Random sources of variation cannot be tested in this design

power to detect a difference in community dispersion of 11.8 (a 50 % difference; Table 2). A difference in community dispersion of 13.2 was observed between the maximum and minimum year-site averages (Fig. 3).

Did Opening the Causeway Spillway Gates Influence Abiotic Sediment Conditions or Resource Availability on Mudflats?

The 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; Pecks Cove had a higher between-year difference than other reference sites (Fig. 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

Table 2 Power analysis for impacted vs. reference site contrasts in ANOVAs of Tables 1 and 2, evaluating impacts of opening the Petitcodiac causeway spillway gates

Response variable	Average reference value	50 % Difference		20 % Difference	
		δ	Power (%)	δ	Power (%)
Community dissimilarity (%)	37.8	18.9	>99	7.6	85
Community dispersion	23.5	11.8	>99	4.7	85
Sediment conditions (average squared distance)	9.3	4.7	>99	1.9	>99
Resource availability (average squared distance)	4.5	2.3	>99	0.9	92

Power calculated for α = 0.05, error DF = 16, variance = error MS (from Tables 1 and 3), number of groups being compared = 2, and a minimum detectable difference (δ) of 50 % and 20 %

Fig. 3 Mean ± standard error (*n* = 5 rounds, June–September) of community dispersion for the intertidal mudflat invertebrates of the Bay of Fundy, Canada. Impacted sites (DF and GA) located near the Petitcodiac River; reference sites (MN, MP, and PC) unlikely to be impacted by the causeway opening. See Fig. 1 for full site names

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 (Fig. 2b).

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

Discussion

The permanent opening of the Petitcodiac causeway spillway gates had no discernible short-term impact on the invertebrate community, abiotic sediment conditions, or resource availability on the intertidal mudflats in the upper Bay of Fundy (Figs. 2 and 3). Furthermore, between-year and between-site differences for the biotic and abiotic variables observed in this study were within the range observed for mudflats elsewhere in the Bay of Fundy (Minas Basin, Gerwing et al., 2015a), suggesting that opening the gates did not have a short-term impact on nearby intertidal mudflats.

In other intertidal systems following large increases in sedimentation, biotic communities were immediately and negatively impacted (Norkko et al. 2002; Thrush et al. 2003). Recovery from such disturbances was slow, with the biota often lagging behind sediment conditions. However, in the Bay of Fundy, we did not detect a similar impact on the

intertidal mudflats following the causeway spillway gate opening. Absence of a short-term impact on these mudflats likely stemmed from the macrotidal nature of the Bay of Fundy (Bleakney 1972; Desplanque and Mossman 2004). Any changes in hydrodynamics that occurred when the causeway spillway gates were opened were likely overwhelmed by the substantial daily water movement in and out of the Bay (Desplanque and Mossman 2004; Wu et al. 2011; Dashtgard et al. 2014). While Grande Anse did experience an increase in sediment deposition immediately following causeway opening (0.5–1 m), sediment deposition within the first year following spillway gate opening appeared to be primarily subtidal in the middle of Shepody Bay, not on the intertidal mudflats (AMEC 2011; AMEC 2013; AMEC 2015). Nevertheless, long-term monitoring of the mudflats is required to evaluate longer-term impacts, especially the impacts associated with cumulative, long-term deposition of sediment on Grande Anse and Daniels Flats (AMEC 2011; AMEC 2013; AMEC 2015). Further, as indicated above, only the spillway 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.

Knowledge of the effects of river blockage removal on biotic and abiotic components of downstream coastal marine systems is limited. 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 hydraulic structures that are being considered for removal (Stanley and Doyle 2003; Doyle et al. 2005; Lejon et al. 2009). 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 barriers should not naively be considered as simply restoring "natural" river conditions. Potential disturbance to the biological community established following construction of the barrier should be considered before action is taken.

Studies like the one presented here will be necessary 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 before, during, and after barrier removal should investigate potential changes

to marine systems, not just freshwater habitats, to ensure restoration of more natural river conditions does not damage downstream marine systems.

References

AMEC. 2005. *Environmental impact assessment report for modifications to the Petitcodiac River causeway. Fredericton, NB, Canada*: AMEC Earth and Environmental, a Division of AMEC Americas Limited.

AMEC. 2011. *Stage 2 Year 1 follow-up program results for the Petitcodiac Causeway project. Fredericton, NB, Canada*: AMEC Earth and Environmental, a Division of AMEC Americas Limited. On Behalf of the New Brunswick Department of Supply and Services.

AMEC. 2013. *Petitcodiac River Causeway project Stage 2 follow-up program year 1 results. Fredericton, NB, Canada:* AMEC Environment and Infrastructure, a Division of AMEC Americas Limited.

AMEC. 2015. *Petitcodiac River Causeway project stage 2 follow-up program year 4 results. Fredericton, NB, Canada*: AMEC Environment and Infrastructure, a Division of AMEC Americas Limited.

Amos, C.L., and T.T. Alfoldi. 1979. The determination of suspended sediment concentration in a macrotidal system using Landsat data. *Journal of Sedimentary Research* 49: 159.

Anderson, M., R.N. Gorley, and R.K. Clarke. 2008. *Permanova + for primer: guide to software and statistical methods*. Plymouth: PRIMER-E Ltd..

Bednarek, A.T. 2001. Undamming rivers: a review of the ecological impacts of dam removal. *Environmental Management* 27: 803–814.

Bleakney, J.S. 1972. Ecological implications of annual variation in tidal extremes. *Ecology* 53: 933–938.

Bray, D.I., D.P. Demerchant, and D.L. Sullivan. 1982. Some hydrotechnical problems related to the construction of a causeway in the estuary of the Petitcodiac River, New Brunswick. *Canadian Journal of Civil Engineering* 9: 296–307.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* 18: 117–143.

Clarke, K.R., and M. Ainsworth. 1993. A method of linking multivariate community structure to environmental variables. *Marine Ecology Progress Series* 92: 205–205.

Clarke, K.R., and R.N. Gorley. 2006. *PRIMER v6: user manual/tutorial*. Plymouth: Primer-E Ltd.

Clarke, K.R., P.J. Somerfield, and M.G. Chapman. 2006. On resemblance measures for ecological studies, including taxonomic dissimilarities and a zero-adjusted Bray–Curtis coefficient for denuded assemblages. *Journal of Experimental Marine Biology and Ecology* 330: 55–80.

Coulthard, M.E., and D.J. Hamilton. 2011. Effects of *Ilyanassa obsoleta* (Say) on the abundance and vertical distribution of *Corophium volutator* (Pallas) on mudflats of the upper Bay of Fundy. *Journal of Experimental Marine Biology and Ecology* 397: 161–172.

Dashtgard, S.E., N.J. Pearson, and M.K. Gingras. 2014. Sedimentology, ichnology, ecology and anthropogenic modification of muddy tidal flats in a cold-temperate environment: Chignecto Bay, Canada. *Geological Society, London, Special Publications* 388: 229–245.

Desplanque, C., and D.J. Mossman. 2004. Tides and their seminal impact on the geology, geography, history, and socio-economics of the Bay of Fundy, Eastern Canada. *Atlantic Geology* 40: 178–203.

Doyle, M.W., E.H. Stanley, C.H. Orr, A.R. Selle, S.A. Sethi, and J.M. Harbor. 2005. Stream ecosystem response to small dam removal: lessons from the heartland. *Geomorphology* 71: 227–244.

Dupont, F., C.G. Hannah, and D.A. Greenberg. 2005. Modelling the sea level of the upper Bay of Fundy. *Atmosphere-Ocean* 43: 33–47.

Fraterrigo, J.M., and J.A. Rusak. 2008. Disturbance-driven changes in the variability of ecological patterns and processes. *Ecology Letters* 11: 756–770.

Fu, K., D. He, W. Chen, C. YE, and Y. LI. 2007. Impacts of dam constructions on the annual distribution of sediment in Lancang-Mekong River Basin. *Acta Geographica Sinica* 62: 1–14.

Galindo, L.A., N. Puillandre, J. Utge, P. Lozouet, and P. Bouchet. 2016. The phylogeny and systematics of the Nassariidae revisited (Gastropoda, Buccinoidea. *Molecular Phylogenetics and Evolution* 99: 337–353.

Gerwing, T.G., A.M. Allen Gerwing, D. Drolet, D.J. Hamilton, and M.A. Barbeau. 2013. Comparison of two methods of measuring the depth of the redox potential discontinuity in intertidal mudflat sediments. *Marine Ecology Progress Series* 487: 7–13.

Gerwing, T.G., A.M. Allen Gerwing, D. Drolet, M.A. Barbeau, and D.J. Hamilton. 2015a. Spatiotemporal variation in biotic and abiotic features of eight intertidal mudflats in the Upper Bay of Fundy, Canada. *Northeastern Naturalist* 22: 1–44.

Gerwing, T.G., A.M. Allen Gerwing, D.J. Hamilton, and M.A. Barbeau. 2015b. Apparent redox potential discontinuity (aRPD) depth as a relative measure of sediment oxygen content and habitat quality. *International Journal of Sediment Research* 30: 74–80.

Gerwing, T.G., D. Drolet, M.A. Barbeau, D.J. Hamilton, and A.M. Allen Gerwing. 2015c. Resilience of an intertidal infaunal community to winter stressors. *Journal of Sea Research* 97: 40–49.

Gerwing, T.G., D. Drolet, D.J. Hamilton, and M.A. Barbeau. 2016a. Relative importance of biotic and abiotic forces on the composition and dynamics of a soft-sediment intertidal community. *PloS One* 11: 11 .e0147098

Gerwing, T.G., J.H. Kim, D.J. Hamilton, M.A. Barbeau, and J.A. Addison. 2016b. Diet reconstruction using next-generation sequencing increases the known ecosystem usage by a shorebird. *The Auk* 133: 168–177.

Hamilton, D.J. 2000. Direct and indirect effects of predation by common eiders and abiotic disturbance in an intertidal community. *Ecological Monographs* 70: 21–43.

Haralampides, K., and A. Rodriguez. 2006. Erosional properties of the sediments in the Petitcodiac River estuary at Moncton, New Brunswick. *Canadian Journal of Civil Engineering* 33: 1209–1216.

Hicklin, P.W. 1987. The migration of shorebirds in the Bay of Fundy. *The Wilson Bulletin* 99: 540–570.

Jiao, N., Y. Zhang, Y. Zeng, W.D. Gardner, A.V. Mishonov, M.J. Richardson, N. Hong, D. Pan, X. Yan, and Y. Jo. 2007. Ecological anomalies in the East China Sea: impacts of the Three Gorges Dam? *Water Research* 41: 1287–1293.

Lejon, A.G.C., R.B. Malm, and C. Nilsson. 2009. Conflicts associated with dam removal in Sweden. *Ecology and Society* 14: 4.

Locke, A., J.M. Hanson, G.J. Klassen, S.M. Richardson, and C.I. Aubé. 2003. The damming of the Petitcodiac River: species, populations, and habitats lost. *Northeastern Naturalist* 10: 39–54.

McAlice, B.J., and G.B. Jaeger. 1983. Circulation changes in the Sheepscot River Estuary, Maine, following removal of a causeway. *Estuaries* 6: 190–199.

McArdle, B.H., and M.J. Anderson. 2001. Fitting multivariate models to community data: a comment on distance-based redundancy analysis. *Ecology* 82: 290–297.

McCurdy, D.G., M.R. Forbes, S.P. Logan, D. Lancaster, and S.I. Mautner. 2005. Foraging and impacts by benthic fish on the intertidal amphipod *Corophium volutator. Journal of Crustacean Biology* 25: 558–564.

Morand, C., and K.A. Haralampides. 2006. Numerical modelling of tidal barrier modification alternatives: Petitcodiac River case study. *Canadian Water Resources Journal* 31: 123–132.

Norkko, A., S.F. Thrush, J.E. Hewitt, V.J. Cummings, J. Norkko, J.I. Ellis, G.A. Funnell, D. Schultz, and I. MacDonald. 2002. Smothering of estuarine sandflats by terrigenous clay: the role of wind-wave disturbance and bioturbation in site-dependent macrofaunal recovery. *Marine Ecology Progress Series* 234: 23–42.

Pess, G.R., M.L. McHenry, T.J. Beechie, and J. Davies. 2008. Biological impacts of the Elwha River dams and potential salmonid responses to dam removal. *Northwest Science* 82: 72–90.

Poff, N.L., and D.D. Hart. 2002. How dams vary and why it matters for the emerging science of dam removal. *Bioscience* 52: 659–668.

Quinn, J.T., and D.J. Hamilton. 2012. Variation in diet of Semipalmated Sandpipers (*Calidris pusilla*) during stopover in the upper Bay of Fundy, Canada. *Canadian Journal of Zoology* 90: 1181–1190.

Risk, M.J., and H.D. Craig. 1976. Flatfish feeding traces in the Minas Basin. *Journal of Sedimentary Research* 46: 411–413.

Shuman, J.R. 1995. Environmental considerations for assessing dam removal alternatives for river restoration. *Regulated Rivers: Research & Management* 11: 249–261.

Stanley, E.H., and M.W. Doyle. 2002. A geomorphic perspective on nutrient retention following dam removal. *Bioscience* 52: 693–701.

Stanley, E.H., and M.W. Doyle. 2003. Trading off: the ecological effects of dam removal. *Frontiers in Ecology and the Environment* 1: 15–22.

Stanley, E.H., M.A. Luebke, M.W. Doyle, and D.W. Marshall. 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. *Journal of the North American Benthological Society* 21: 172–187.

Stanley, E.H., M.J. Catalano, N. Mercado-Silva, and C.H. Orr. 2007. Effects of dam removal on brook trout in a Wisconsin stream. *River Research and Applications* 23: 792–798.

Sucsy, P.V., B.R. Pearce, and V.G. Panchang. 1993. Comparison of two- and three-dimensional model simulation of the effect of a tidal barrier on the Gulf of Maine tides. *Journal of Physical Oceanography* 23: 1231–1248.

Thomson, J.R., D.D. Hart, D.F. Charles, T.L. Nightengale, and D.M. Winter. 2005. Effects of removal of a small dam on downstream macroinvertebrate and algal assemblages in a Pennsylvania stream. *Journal of the North American Benthological Society* 24: 192–207.

Thrush, S.F., J.E. Hewitt, A. Norkko, V.J. Cummings, and G.A. Funnell. 2003. Macrobenthic recovery processes following catastrophic sedimentation on estuarine sandflats. *Ecological Applications* 13: 1433–1455.

Underwood, A.J. 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. *Ecological Applications* 4: 4–15.

Underwood, A.J. 1997. *Experiments in ecology: their logical design and interpretation using analysis of variance*. New York: Cambridge University Press.

van Proosdij, D., T. Milligan, G. Bugden, and K. Butler. 2009. A tale of two macro tidal estuaries: differential morphodynamic response of the intertidal zone to causeway construction. *Journal of Coastal Research* 56: 772.

Warwick, R.M., and K.R. Clarke. 1993. Increased variability as a symptom of stress in marine communities. *Journal of Experimental Marine Biology and Ecology* 172: 215.

Wells, P.G. 1999. *Environmental impacts of barriers on rivers entering the Bay of Fundy*: report of an ad-hoc Enviornment Canada Working Group. Technical Report Series No. 334. Ottawa, On, Canada: Canadian Wildlife Service, Environment Conservation Branch.

Winer, B.J., D.R. Brown, and K.M. Michels. 1991. *Statistical principles in experimental design*. New York: McGraw-Hill.

Wu, Y., J. Chaffey, D.A. Greenberg, K. Colbo, and P.C. Smith. 2011. Tidally-induced sediment transport patterns in the upper Bay of Fundy: a numerical study. *Continental Shelf Research* 31: 2041–2053.

Zar, J.H. 1999. *Biostatistical analysis*. Upper Saddle River: Prentice Hall.