Check for updates

RESEARCH ARTICLE

Does restoration work? It depends on how we measure success

Shad Mahlum[1,2,3], David Cote[4], Yolanda F. Wiersma[1], Curtis Pennell[4], Blair Adams[5]

The restoration of 4 partial stream barriers was evaluated in watersheds of Terra Nova National Park, Newfoundland, Canada from 2009 to 2011. Brook trout ($n = 462$) were tagged and tracked moving through our study sites using passive-integrated transponder telemetry and the restoration actions were assessed using 3 different measures: passage success rates; the range of passable flows; and the availability of passable flows. We considered the observed results within a before-after-control-impact (BACI) design that included reference reaches and pre-restoration observations. The conclusions of BACI analyses were also contrasted with those that would have been obtained from commonly used before-after (B-A) or control-impact (C-I) study designs. While the restoration actions changed hydrological conditions in a way that should facilitate fish passage, our biological measures indicated that success was variable across culverts and within culverts depending on the measure evaluated. Furthermore, the natural temporal and spatial variability of fish movements often resulted in different conclusions between the more robust BACI design and the more commonly used B-A and C-I designs. Our results demonstrate that restoration of partial barriers may not always yield dramatic improvements. Furthermore, without suitable controls, the chances of drawing false conclusions regarding restorations in temporally and spatially dynamic systems are substantial.

Key words: aquatic restoration, BACI, brook trout, connectivity, culverts, fish, passive integrated transponder tagging, stream

Implications for Practice

- To identify where to place future restoration efforts, it is necessary to assess current restoration actions to determine whether biological improvements are met and where to improve general decision-making processes.
- Restoring partial barriers to movement appear to yield limited biological improvements and future restorations should focus on sites where movement is most restricted.
- Designing robust study designs to include both reference sites and pre-treatment data is critical to accurately determine successful restoration.

Introduction

In the face of anthropogenic impacts on ecological systems, managers often turn to restoration as a tool to mitigate negative effects. Environmental and ecological restoration can be expensive, and hence some certainty of success is desirable. In response to habitat fragmentation, managers often turn to engineering solutions to restore connectivity. In terrestrial ecosystems, wildlife overpasses and underpasses are implemented to address barriers of highways and railways (Smith et al. 2015). In riverine ecosystems, barriers (culverts and dams) fragment habitat and affect connectivity for fish and other aquatic biota. To mitigate fragmentation, various structures (e.g. fish ladders, baffles, fishways) can be installed to facilitate fish passage while still allowing the continued function of the structure (Cote et al. 2009; Diebel et al. 2015; Saunders et al. 2016; Shaw et al. 2016).

The effect of barriers such as dams and road crossings on riverine systems is well documented, particularly for diadromous species whose populations can be catastrophically impaired when individuals are obstructed from reaching spawning habitat. Less documented (but see Perkin & Gido 2012) are the more subtle movements within freshwater that are needed to optimize various life history functions (feeding, refuge, spawning, dispersal etc.; Roghair & Dolloff 2005; Petty et al. 2012; Benitez et al. 2015). To address these impacts, much progress has been made to understand and identify in-stream barriers, and to prioritize sites for restoration (Januchowski-Hartley et al.

2013; King & O'Hanley 2014; Thorne et al. 2015; King et al. 2016; Maitland et al. 2016) in an effort to increase connectivity in riverine ecosystems.

Restoration of longitudinal connectivity in riverine ecosystems often focuses on either removing barriers or modifying infrastructure to facilitate fish passage (Noonan et al. 2012; Evans et al. 2015; King et al. 2016). Barrier modification is generally through creation of fish passage facilities (e.g. fishways, fish ladders) around dams or through reengineering culverts to reduce flow velocities and outflow drops (e.g. through insertion of baffles and adjusting culvert slope). In small passage structures such as culverts, resources are rarely put in place to evaluate the biological outcomes of these interventions (Bernhardt et al. 2005; Tummers et al. 2016; Rodeles et al. 2017). This is unfortunate given that these structures are ubiquitous and studies of the impacts of restoration activities are critical to establishing best practices, setting expectations for restoration success and understanding the cost-benefit analysis of different restoration approaches (see Neeson et al. 2015 for an example of a return-on-investment framework for addressing barrier removal).

Potentially, many managers are satisfied with restoration techniques that improve physical habitat attributes (e.g. water velocities, outflow drops) but such assumptions that physical improvements will predictably improve fish passage may not always be valid (Pretty et al. 2003; Mahlum et al. 2014*b*). Consequently, understanding the usefulness of restoration approaches relies heavily on empirical data of fish community responses.

Measures of restoration success can vary. They can include confirming that physical characteristics of the passage structure lie within the target species' physiological limits, or an assessment of biological outcomes such as passage volume or success rate, or measures of community structure above and below barriers. In any case, response variables would be expected to vary in time and space even in the absence of a perturbation or restoration action (Meyers et al. 1992; Petty et al. 2012; Goerig & Castro-Santos 2017) and this variation needs to be accounted to accurately assess the project's success. Failure to account for natural spatial–temporal variability (Elliott 1984; Kwak & Waters 1997; Enders et al. 2008; Ayllón et al. 2014; Brewitt & Danner 2014) could result in having unnecessarily high expectations for a passage structure or incorrectly concluding success or failure. For example, the migratory behavior of many fish species in small river systems often vary seasonally (Meyers et al. 1992; Gowan & Fausch 1996; Stickler et al. 2008; Ayllón et al. 2014) and arise due to severe environmental events (extreme heat, floods, freezing etc.; Enders et al. 2008; Stickler et al. 2008; Petty et al. 2012). Therefore, requiring that fish passage structures allow for passage of fish at all water levels may not be necessary to maintain the natural functioning of the system. Additionally, observed "improvements" to biological metrics after restoration may simply be a result of natural spatiotemporal variation unless controlled for in the experimental design. The ideal scenario is to have a BACI (before-after-control-impact) study design (Ogren & Huckins 2015; Tummers et al. 2016) that controls for both spatial and temporal variation. However, few studies achieve this despite that it has been shown to be the overall superior method for monitoring (Smokorowski & Randall 2017).

We carried out evaluations of restored fish passage structures in four streams in Terra Nova National Park (TNNP), Newfoundland, Canada using a BACI design and control areas of the stream that were unimpeded by anthropogenic barriers. We considered the observed results within the context of naturally occurring variability and contrasted them with those that would have been obtained with simpler before-after (B-A) or control-impact (C-I) study designs. We evaluate whether the results will be consistent across BACI/B-A/C-I designs and different response metrics, including passage success, range of passable flows, and available passable flow.

Methods

Culvert and Reference Site Selection

The study area was comprised of four sites within three boreal stream systems within TNNP, Newfoundland and Labrador, Canada (Fig. 1). The study sites are low productivity streams with low species richness that are dominated by salmonids (Cote 2007). Each site consisted of a manipulation site that encompassed a culvert and a natural reference reach. The four sites were Arnolds Pond Brook at the Southwest Arm Day Use Area, hereafter Arnolds; Terra Nova Brook at Terra Nova Road, hereafter Terra Nova; Terra Nova Brook at Spracklins Road, hereafter Spracklins; an unnamed brook approximately 700 m northeast of the Cobblers Brook Day Use Area on the Trans-Canada Highway, hereafter Cobblers.

Barrier restoration for three of the four culverts (Arnolds, Spracklins, and Terra Nova) was based on a previous rule-based assessment of all the culverts in TNNP (Bourne 2013) in conjunction with a cost-benefit analysis that predicted that improving these sites would provide the most ecological benefit with the resources available (Cote et al. 2009; Bourne 2013). Restoration took place between fall 2009 and summer 2011. The fourth barrier (Cobblers) was added opportunistically after damage from Hurricane Igor in 2010 necessitated some form of replacement. While additional restoration measures were taken (Appendix S1, Supporting Information), we focused on two main restoration types among the four sites. The first was the inclusion of baffles on three of the sites to alter the stream flows to reduce discharge velocities through the culvert and the second was the placement of a secondary overflow culvert next to the barrier on Spracklins Brook (Table 1). We collected culvert measurements and characteristics from Bourne (2013) and Mahlum et al. (2014*b*) (summarized in Table 1).

Reference sites were located downstream for Arnolds, Spracklins, and Terra Nova, and due to the proximity to the ocean, the reference site at Cobblers was located upstream of the culvert. Each reference site was placed within 50 m from the culvert but far enough away so that flow dynamics from the culvert were not incorporated into the reference reach. The exception was the reference site for Spracklins which was approximately 400 m downstream at the confluence of two
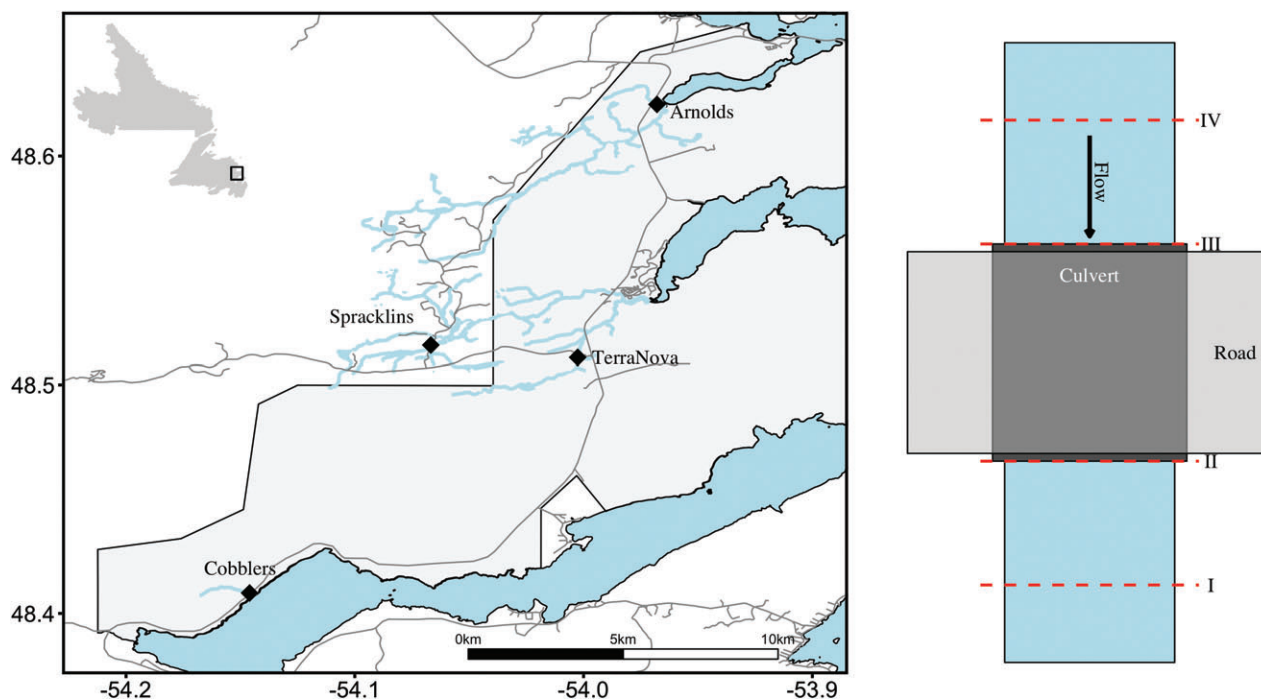
Figure 1. The location of the four study sites in and around Terra Nova National Park (gray-shaded area in map; TNNP) in Newfoundland, Canada. Blue lines indicate the rivers where culverts are located and gray lines indicate the road network in and around TNNP. To the left is a diagram of the antennae setup at each culvert with antennae indicated by red dashed lines, with the culvert indicated by the black polygon and the road the gray polygon.

Table 1. Summary of culvert parameters and restoration type for the four culverts assessed for restoration outcomes in Terra Nova National Park, Canada from 2009 to 2011. All culverts were constructed with corrugated metal pipe (CMP). Slope, diameter, length, and diameter are physical metrics of the culvert used to define the type of barrier as calculated using FishXing (Furniss et al. 2006). Restoration type consisted of the inclusion of baffles or the installation of a second adjacent culvert.

	Arnolds	*Cobblers*	*Spracklins*	*Terra Nova*
Construction	CMP	CMP	CMP	CMP
Shape	Circular	Circular	Circular	Circular
Diameter (cm)	78	240	75	87
Length (m)	6.2	36	12	14
Slope (%)	1.77	1.83	1.50	2.29
Culvert roughness	0.032	0.015	0.024	0.01
Type of barrier	Full	Partial	Partial	Partial
Cause of barrier	Velocity	Depth/velocity	Depth/velocity	Depth
Restoration type	Baffles	Baffles	Second culvert	Baffles
Restoration date	Fall 2009	18 August, 2011	Fall 2010	13 June, 2011

tributaries where it served as a reference site for the culvert in this study and a nearby culvert in the connecting tributary (not included in the current study; Mahlum et al. 2014b). Furthermore, the upper and lower boundaries of the reference sites were selected to mimic the corresponding culverts.

Fish Capture, Tagging, and Tracking

Fish were captured using a portable electrofisher (Smith-Root model 12-B) at each paired study site during May and June from 2009 to 2011 (fish capture only occurred in 2011 for Cobblers; Table 2). Each sampling effort consisted of a single-pass approximately 150 m upstream and downstream of the culvert

and encompassed the corresponding reference site. Because the goal of the fish capture was to obtain a representative sample of individuals to tag, block nets were not needed. We attempted to include brook trout (*Salvelinus fontinalis*; both resident and anadromous life histories), juvenile Atlantic salmon (*Salmo salar*), and American eel (*Anguilla rostrata*). However, limited passage events for the latter two species and anadromous brook trout required that we exclude them from the study. Therefore, we focused on resident brook trout. Prior to handling, we anesthetized fish in a 40 mg clove oil per liter and 400 μL/L ethanol mixture (Anderson et al. 1997). We then measured the length (fork length, mm) and recorded the weight (wet weight, g) for all individuals. We tagged all fish greater than

Table 2. Summary of the number of brook trout (*Salvelinus fontinalis*) tagged and detected at the culvert and references sites for each stream and year.

Stream	Number Detected			
Year (# tagged)	2009	2010	2011	Total
Arnolds				
2009 (50)	22	7	0	22
2010 (68)	—	18	7	20
2011 (82)	—	—	28	28
Total (200)	—	—	—	70
Cobblers				
2011 (32)	—	—	22	22
Total (32)	—	—	—	22
Spracklins				
2009 (73)	—	17	6	17
2010 (80)	—	31	15	41
2011 (25)	—	—	5	5
Total (178)	—	—	—	63
Terra Nova				
2009 (14)	—	1	0	1
2010 (26)	—	10	6	10
2011 (12)	—	—	5	5
Total (52)	—	—	—	16

95 mm with a 23 mm passive integrated transponder tag (model RI-TRP-WRHP, Texas Instruments; 23.1 mm in length) through a small ventral incision just anterior to the pelvic girdle. We closed the incision with a single suture (4–0 SofSilk) and we placed the fish into a holding cage for 24 hours prior to release to assess acute tagging mortality.

We tracked tagged fish through the reference and culvert sites with an array of four antennae (HDX multi-antennae reader, Oregon RFID) from May to November of the sampling years. At culvert locations, we placed two antennae across the stream below the culvert and two antennae above the culvert with one antenna at the entrance and exit of the culvert (Fig. 1). We separated the upstream and downstream antennae by 1–2 m to prevent a single fish from being read simultaneously at two adjacent antennae (see Mahlum et al. 2014*b* for details). The paired antennae enabled us to determine direction of movement and entry of an individual into the culvert. A successful passage was defined when the individual was detected at either of the downstream antennae followed by either of the upstream antennae. For a failed upstream passage attempt the individual had to move beyond the two downstream antennae but then fail to register at the upstream antennae with a subsequent registration at the most downstream antennae. Conversely, a failed downstream passage occurred when the individual registered at the upstream antennae then failed to be detected at the downstream antennae before subsequently registering at the farthest upstream antennae.

Discharge was derived from water-level loggers (Solinst Levellogger Gold) deployed in each study stream to record hourly water depth. Each site was visited throughout the study to establish a rating curve, which was used to link the measured water depth to discharge (Riggs 1985). To determine the temporal availability of stream flow suitable for fish passage, we calculated a cumulative frequency of stream discharge for each paired site both before and after restoration (Fig. S1).

Normal antennae operation was from May to November from 2009 to 2011 for all sites (2011 only for Cobblers, see above for details). Although tagging and tracking data is available for Spracklins and Terra Nova in 2009, corresponding discharge data were not available at the time of the study and therefore 2009 was removed from the analysis for those sites. Minor lapses in antennae coverage occurred throughout the study due to antennae maintenance, but in 2010, Hurricane Igor significantly affected the overall coverage among the study sites. The largest gap due to the hurricane was in Terra Nova, where an entire antennae array was lost during the storm and the data were never recovered. Therefore, to standardize the dataset, we coordinated the temporal operation of the paired sites so that the catch per unit effort was the same between reference and treatment locations.

Data Analysis and Response Metrics

To assess the effects of restoration, we focused on the upstream movements of brook trout centered on three different responses. First, we tested whether the percentage of successful upstream brook trout passage events (hereafter passage success; PS) increased after the restoration action; second, we tested whether brook trout successfully passed a broader range of flows after restoration (hereafter range of passable flows; RPF); and finally, we tested whether the availability of passable flows (APF) increased after the restoration of the culvert. While the latter two response variables are closely linked, RPF indicates the potential of the fish to move through the stream segment under different flow conditions whereas APF represents the amount of time passage is possible given variable flow conditions. Each metric was assessed within BACI, B-A, and C-I analyses to determine how the interpretation of restoration success could be potentially affected by different study designs.

To assess changes in upstream PS, we ran separate generalized linear mixed-effects models to determine if brook trout passage (binomial response) improved post-restoration (Bates et al. 2015). Because of the repeated measures at the paired site and individual level, we ran the model separately for each site and included the random variable of individual into each model. A significant interaction of the models' main B-A (restoration) and C-I (treatment) terms would signify a restoration-induced change. For B-A and C-I analyses, the models were simplified to include only restoration (pre- and post-comparison of the culvert) or treatment (comparison of the culvert and reference site post-restoration), respectively. In these cases, the main effect of restoration or treatment was used to assess restoration improvements (significance at $\alpha = 0.05$; R Version 3.4.0, www.r-project.org, R Development Core Team 2017).

To analyze whether brook trout capitalized on broader flow regimes post-restoration (RPF), we documented the discharge range of successful upstream passage events. The RPF was calculated as the difference between the 2.5 and 97.5 quantiles of this flow distribution. Based on the calculation of the RPF,

the AFP was calculated by determining the time across the study period (in percent) that the stream's discharge fell within the RPF.

Because standard analysis-of-variance models are centered around the mean, we opted to use a randomization model to test the significance of RPF and APF using modified randomized intervention analyses (Bried & Ervin 2011). Both RPF and APF metrics were assessed for BACI, B-A, and C-I designs. All randomly sampled discharge values were based on the successful upstream passage attempts used to calculate the RPF and APF. Sampling was done without replacement to provide independently sampled groups and we conducted 9,999 permutations for each model. Significance of the observed result was based on its relative position to the distribution of permuted results. A detailed description of the randomization along with corresponding R code is included in Appendices S2 and S3, respectively. All statistical analyses were carried out with the Program R (version 3.4.0).

Temporal Patterns in Fish Passage

Temporal variability in movement activity was assessed in natural reaches of the stream to provide context to the culvert interactions measured at the culvert. First, we assessed the diel movement patterns at reference and culverts sites using a kernel density estimation (Wickham 2016). In addition to diel movement patterns, the number of movements upstream and downstream at each reference site was summed at weekly intervals. To display temporal trends, a LOESS fit was applied to each site and year (Wickham 2016).

Results

We tagged a total of 462 brook trout among the different sites and years of the study period with a tagging mortality of 1.1% after 24 hours. An additional 12 individuals died due to insufficient stream ventilation during the 24-hour observation period. The mean length among study sites was 125 mm (SD = 27 mm) with a mean weight of 24 g (SD = 19 g). We later observed 171 individuals either in the culvert ($n = 121$), or reference sites ($n = 124$), or both ($n = 76$; Table 2). This resulted in 6,199 upstream passage attempts at culverts (83% success rate) and 4,297 upstream passage attempts at reference sites (55% success rate; Fig. S1). We saw no difference in the length at tagging between individuals that were detected and individuals that were not detected within sites ($F_{[1,454]} = 1.85$, $p = 0.14$).

The BACI design (Table 3) indicated that the three culverts retrofitted with baffles showed at least one positive response to restoration with Arnolds having two significantly positive responses to restoration. However, Cobblers (RPF, APF), Spracklins (PS), and Terra Nova (APF) had at least one significantly negative response to restoration with Spracklins indicating negative trends for all three response variables (Table 3).

Table 3. Summary table of the results for the passage success range of passable flows, and percentage of available flows that are passable under three different types of comparisons, a full BACI and B-A and C-I. Negative signs indicate that the restoration action had a negative effect and positive signs indicate that the restoration action had a positive effect. Parentheses around negative and positive signs indicate significance at $\alpha = 0.05$.

Site	BACI	B-A	C-I
Passage success			
Arnolds	(+)	(+)	+
Cobblers	+	(+)	(+)
Spracklins	(−)	(−)	(−)
Terra Nova	+	+	+
Range of passable flows			
Arnolds	(+)	−	(−)
Cobblers	(−)	−	−
Spracklins	−	+	(−)
Terra Nova	+	+	+
Available passable flows			
Arnolds	−	(+)	(−)
Cobblers	(−)	−	−
Spracklins	−	(+)	(−)
Terra Nova	(−)	(−)	−

Passage Success

The BACI analysis indicated that three of the four culverts had improved PS post-restoration with one site having a significant improvement (positive increase in PS). However, Spracklins had a significant decrease in PS post-restoration (Fig. 2 and Table 3; see Table S2 for model outputs for BACI analysis). These interpretations of the data were the same for B-A and C-I (Table 3).

Range of Passable Flows

Only one site had a significant increase in RPF and one experienced a significant decrease in RPF for the BACI analysis (Arnolds and Cobblers, respectively). The RPF in Terra Nova improved after restoration; however, limited observations of brook trout and low sample size limited the strength of the model (Tables 3 & S3). Spracklins had a non-significant negative effect. When only comparing the RPF B-A restoration, the conclusions were different than as determined with the full BACI model. Spracklins and Terra Nova showed a positive effect (i.e. increase in RPF indicating restoration success) and Arnolds and Cobblers showed a negative effect (i.e. decrease in RPF), though none are significant (Fig. 3; Table 3). In contrast, when comparing only the reference reach post-restoration to the culvert (C-I), the results show a significant decrease in RPF (i.e. no restoration success) for Spracklins and Arnolds, and non-significant negative and positive effects for Cobblers and Terra Nova, respectively (Table 3).

Available Passable Flows

All sites showed a decrease in APF, with Cobblers and Terra Nova having a significant decrease in APF when using the BACI analysis. However, in the B-A comparison Cobblers
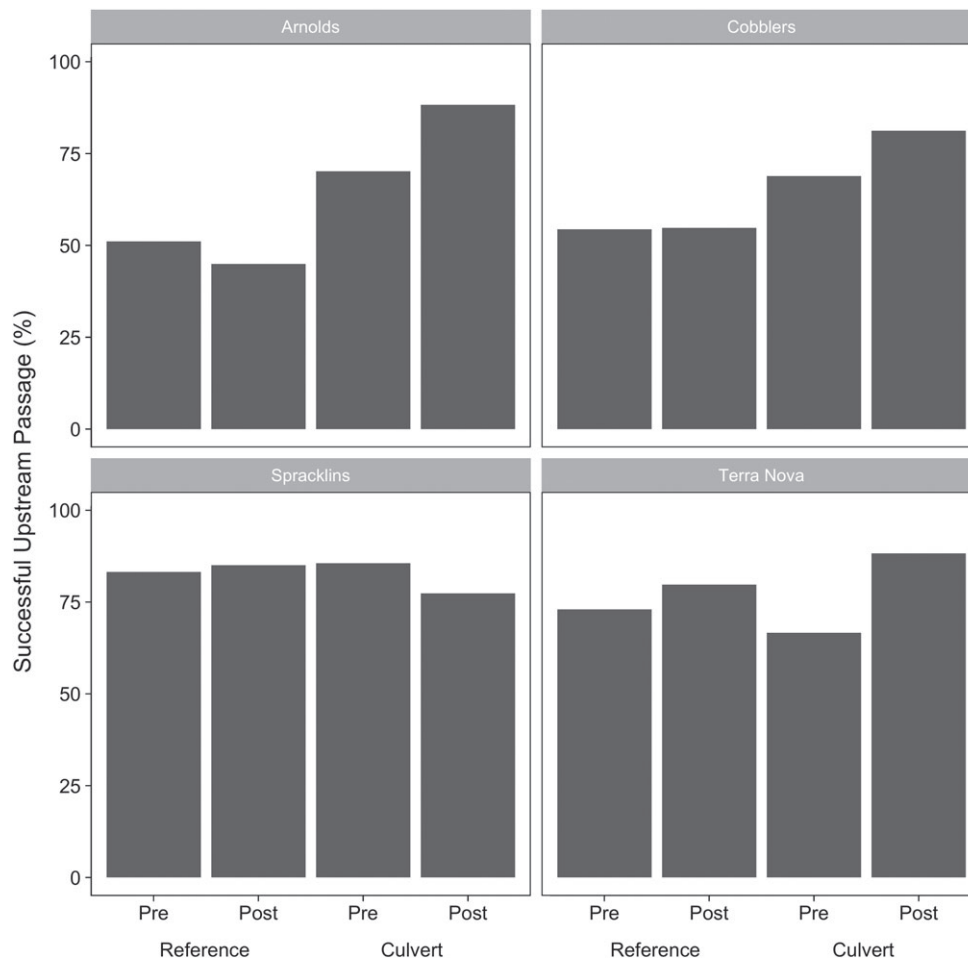
Figure 2. Pre- and post-restoration brook trout passage success for each culvert and paired reference site in Newfoundland, Canada (2009–2011).

was no longer significant (but still showed a decrease in APF), while Arnolds and Spracklins had a significantly positive increase and Terra Nova had a significant decrease in APF (Table 3; see Table S2 for model details). When comparing the reference sites to culverts post-restoration (C-I), all sites showed a decrease in APF, but only Arnolds and Spracklins were significant (Table 3).

Temporal Patterns of Fish Passage

Successful passage events occurred throughout the diel period in both reference and culvert reaches, but the distribution suggested a tendency toward crepuscular activity (Fig. 4). The exceptions were at the Arnolds and Terra Nova culverts, where passage events remained relatively high throughout the nocturnal period. Less consistency was seen in seasonal patterns of movement among culverts (Fig. 5). For example in 2009 and 2011, Arnolds showed peak movement around October and November (it is unknown whether Arnolds followed the same trend in 2010 due to the loss of data from Hurricane Igor). However, in 2011, where data are available for all four culverts, the peak upstream movement varied from as early as August in Spracklins to as late as October in Arnolds.

Discussion

Our analysis of four culverts in TNNP suggests that restoration outcomes are not guaranteed. While culvert restoration is expected to improve fish passage, we did not see consistent improvements in all culverts. In terms of passage success, all of the culverts with the exception of Spracklins showed an improvement, but only one (Arnolds) was statistically significant. Results were more equivocal when considering flow properties in the culverts—only one culvert (Arnolds) showed a significant positive change in the range of passable flows and none showed an increase in the availability of flows that were passable.

These mixed results align with the few studies that have assessed fish passage restoration success (Pretty et al. 2003; Noonan et al. 2012; Evans et al. 2015; Myers & Nieraeth 2016; Tummers et al. 2016). Evans et al. (2015) found an increase in similarity in fish community composition upstream and downstream of a restored culvert 3 years after restoration, but a decrease in relative fish abundance (though biomass remained constant). Myers and Nieraeth (2016) showed a dramatic (300%) increase in the number of coho salmon (*Oncorhynchus kisutch*) in an Alaskan stream following culvert restoration.
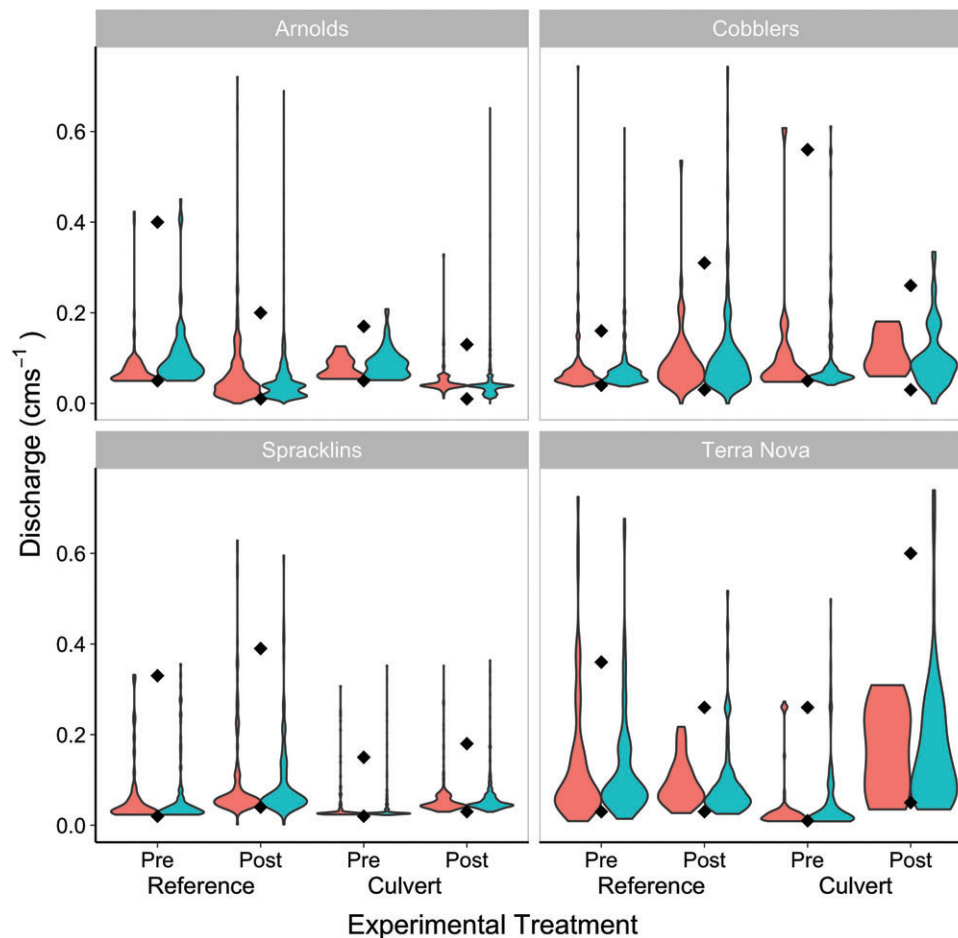
Figure 3. Range of discharge levels experienced by brook trout during both failed (red) and successful (blue) passage events in the four study streams at culverts and reference sites in Newfoundland, Canada, before and after restoration (2009–2011). Range of discharges (95% quantiles, black diamonds) during successful ascents gives the RPF.

Maximum numbers were reached 5 years post-restoration and were not observed for the weaker swimming pink salmon (*Oncorhynchus gorbuscha*). In their holistic approach, Tummers et al. (2016) found little change in the fish community 2–3 years after restoration at all but one site, which was colonized by three native non-salmonids. They also demonstrated differential responses by strong- versus weak-swimming fish species.

The lack of consistency in restoration success observed in this study and others could stem from several reasons. First, if there are no nearby animals to recolonize the newly available habitat (Langford et al. 2009), or if the quality of habitat in the newly restored area is inadequate, improvements to fish passage alone may not restore fish communities (Pretty et al. 2003; Tummers et al. 2016). This is supported by studies that observed that fragmentation is a secondary stressor relative to land use in fish communities (Branco et al. 2011; Mahlum et al. 2014*a*). In our national park study area, however, water quality is considered good and it has minimal anthropogenic influences relative to the adjacent landscape. Furthermore, the distribution of brook trout in our study area is ubiquitous.

Second, the ease with which improvements can be detected would be expected to be specific to the system of study. Reduced statistical power might be expected in systems with lower densities of fish where passage attempts are less frequent (Pepino et al. 2012; Pépino et al. 2016). Interestingly, Cobblers had the second lowest numbers of tagged fish (22) but had significant results for two of three measures of success under a BACI. Detectable effect sizes also influence statistical power (Pépino et al. 2016), and the culverts in the study were better at passing fish prior to restoration than anticipated based on hydrological modeling that was used to select the culverts (Mahlum et al. 2014*b*). As a result, there was less room for improvement, with pre-restoration measures of success already being relatively high in some culverts (e.g. APF exceeded 80% in Cobblers and Terra Nova culverts).

Third, passage attempts and success rates vary in part with the motivation of fish to use the passage facilities (Goerig & Castro-Santos 2017), which has been shown to differ across individuals as well as systems and years (Olsson & Greenberg 2004; Goerig & Castro-Santos 2017). Stream-specific movement rates could be affected by the cost of migration
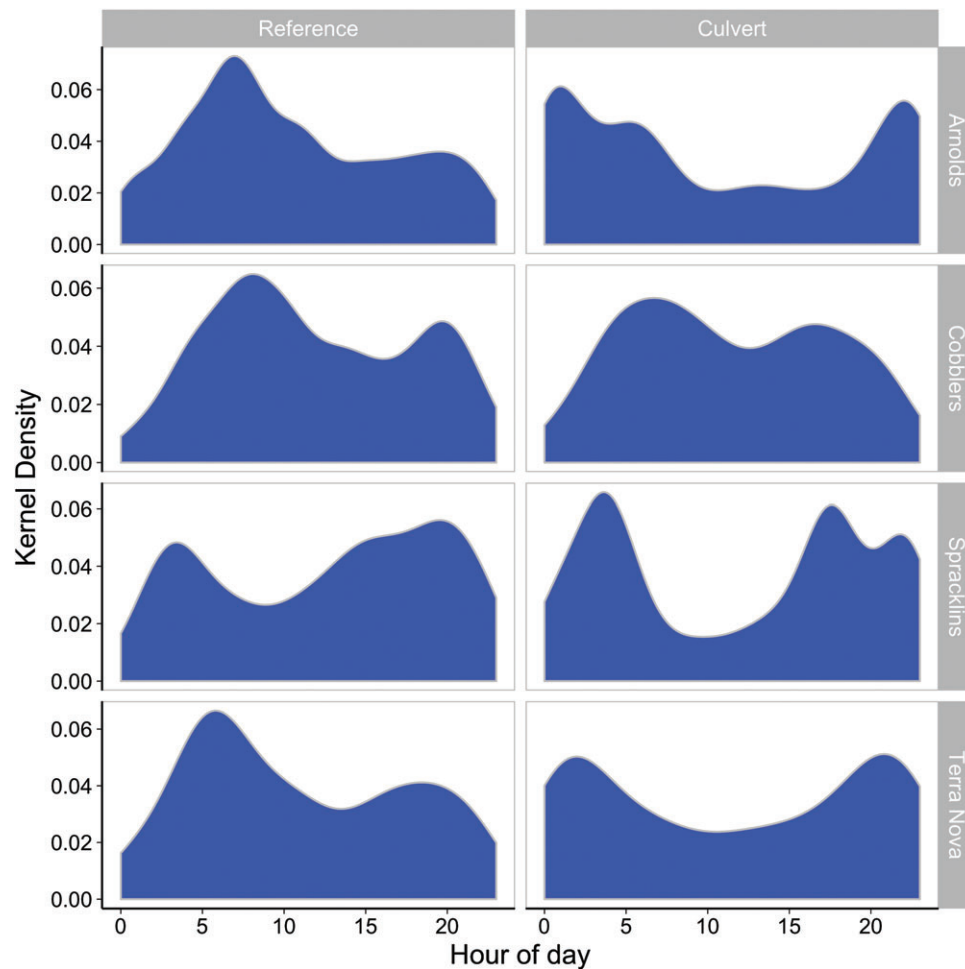
Figure 4. Diel patterns of successful brook trout passage events in culverts and control reaches of the four culvert restoration sites in Newfoundland, Canada (2009–2011). A kernel density estimation was used to calculate the density of diel passage events.

(Bohlin et al. 2001), the availability and distribution of resources (Olsson et al. 2006; Wysujack et al. 2009), the fish population's adaptations (Ayllón et al. 2014; Branco et al. 2017), or even the frequency of severe climate events (Petty et al. 2012). Within a stream, the motivation of individuals to move up through culverts (or within the stream) is also influenced by time of day, season, water flow, densities of conspecifics, and their size (Ayllón et al. 2014; Goerig & Castro-Santos 2017). Such variability and the potential influence of confounding factors could mask improvements to passage conditions or contribute to potential false positives.

Finally, the use of different designs would be expected to have implications on the restoration outcomes (Noonan et al. 2012). In this study, all three baffled culverts showed at least qualitative improvements to passage success in contrast to the other restoration designs, which showed a significant reduction. Baffles are designed using a different mechanism to enhance fish passage, whereby they slow velocity and increase depth by increasing the culvert's bottom roughness (Khodier & Tullis 2014, 2016; Chanson & Uys 2016). As discharge increases, however, the flow of the water is still contained within the pipe,

increasing the depth quickly and greatly reducing the mitigating effects of the baffles on water surface velocity (Jarrett 1985). In contrast, the extra offset culvert installed on Spracklins acts to increase flow capacity of the crossing within a range of higher flows and buffers the effects of increasing discharge on water velocity. This design also offers an alternate passage route with differing hydrological conditions. These design mechanisms may affect their suitability as they would be expected to interact with the nature of the target species, the size structure of the population, and the local site conditions. Both designs would be expected to be inferior to a natural setting, where floodplains can moderate the effects of high discharges once streams flood their banks.

The natural control areas in this study also provide important behavioral context as to whether behavior observed in culverts mirrored that in the adjacent, portions of the stream. In both control and culvert site types, there was a tendency for crepuscular activity. Similar observations were presented in Goerig and Castro-Santos (2017), where they hypothesized, along with anti-predation and competition arguments, that brook trout may reduce passage attempts during the day to
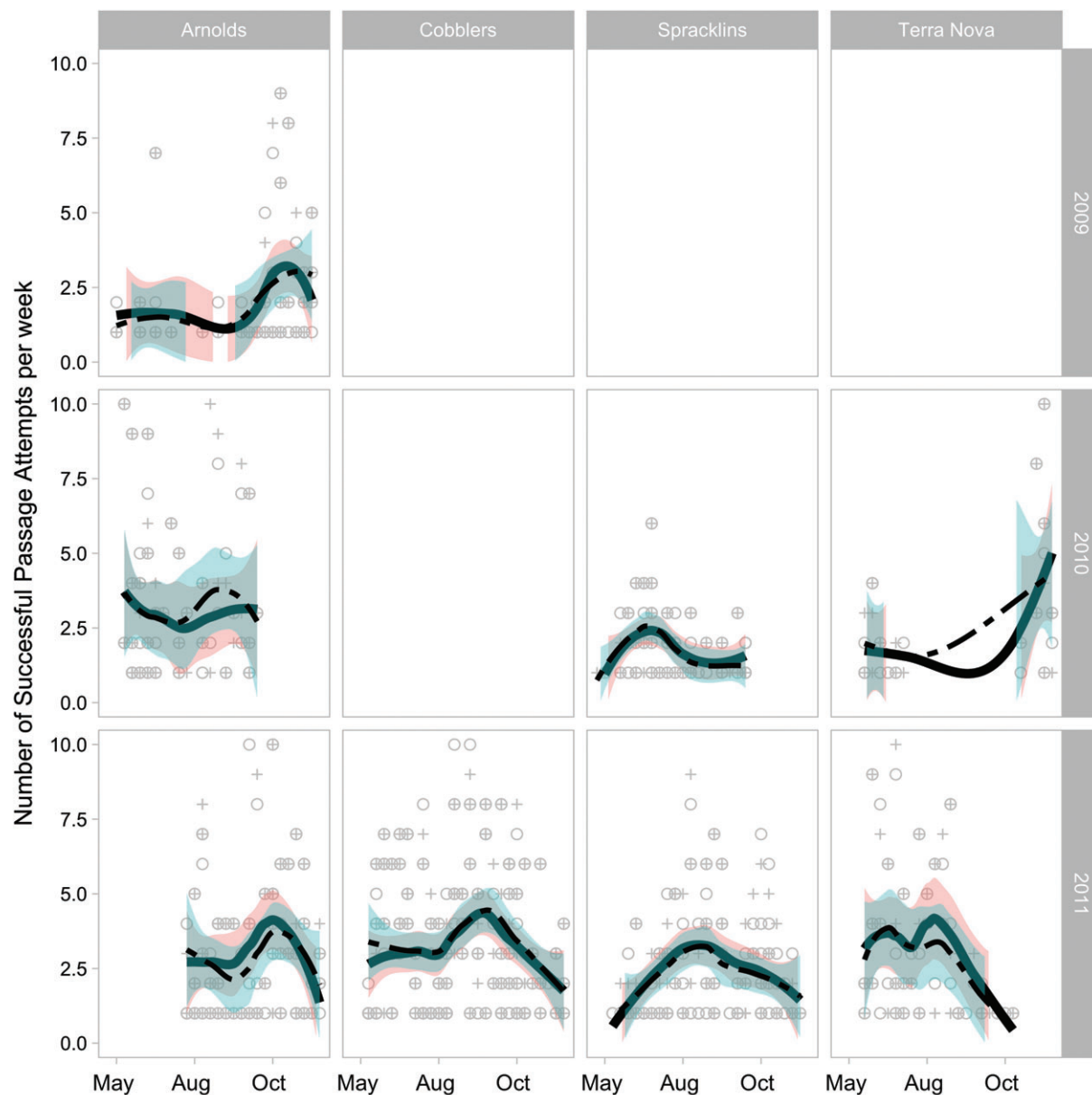
Figure 5. Successful upstream (open circles) and downstream (plus signs (+)) brook trout passage events per week for each site and year. Lines represent LOESS smoothers of respective upstream (solid line) and downstream movements (dashed line). Shaded areas represent the standard error of the LOESS smoothers for upstream (red) and downstream (blue) movements. Restoration was carried out in different years; hence the empty panels in the graph. Restoration work was completed on Arnolds in fall 2009, on Spracklins in fall 2010, and on Cobblers and Terra Nova in summer 2011.

avoid sudden changes in luminosity. Our study suggests that luminosity is likely not a key driver of diel movement patterns since the patterns were consistent across treatments. Our reference and culvert treatments also show remarkably similar event-discharge distributions for failed and successful passage events. This suggests that the influence of discharge may not be a primary driver in the realized passage success in a stream in either natural areas or the culverts we studied. It is noteworthy, however, that the movement of fish across our study arrays may not always represent fish exhibiting directed movement to gain access to distant resources (spawning habitats, refuge areas,

feeding grounds etc.). Instead it is possible that the "failed" and "successful" passage observations could represent normal movements of fish occupying a seasonal home range (Gowan & Fausch 2002). Supporting this is the fact that over one-fifth of tagged individuals had more than 100 detection passage attempts on our arrays. It also supports the contention of Goerig and Castro-Santos (2017) that "causal mechanisms may be missing from the current thinking about entry and passage behaviors."

Of the few published papers that assess barrier restoration, many use comparative approaches. For example, Favaro

et al. (2014) compared fish movement across baffled culverts, non-baffled culverts, and reference streams (no culverts) in metro Vancouver, Canada. Others (Evans et al. 2015; Myers & Nieraeth 2016) have carried out B-A comparisons of fish movement in a single site after barrier removal or restoration. In contrast, Porto et al. (1999) used both temporal and spatial comparisons but did not examine the interaction term as is standard for classic BACI-style analyses. The experimental controls inherent in the BACI approach used in this study were certainly needed given strong inter- and intra-annual variation observed in movement patterns across stream systems and within streams between reference and treatment. Our randomization results show how data can be interpreted very differently when using a B-A or C-I instead of a full BACI. In each of our culverts, there were differences between BACI and B-A or C-I in the qualitative result (improvement or degradation) or whether the result was significant. In some cases, BACI analyses led to opposite conclusions of B-A or C-I comparisons at some sites (e.g. Arnolds Brook for RPF). Clearly, without suitable controls, the chances of drawing false conclusions about restoration success were substantial. Therefore, when full BACI designs are not feasible, other more cost-effective designs should be interpreted cautiously. Engineered restoration approaches (highway over and underpasses, culvert or dam passage structures) are expensive, and therefore it is critical to assess what kinds of ecological effects will result from such efforts to restore connectivity (Smith et al. 2015; Watson et al. 2017). Efforts to restore connectivity at specific points in space may not always yield dramatic results for partial barriers since the ecological benefits are unlikely to increase in a linear manner with passability. For example, occasional passage of an animal through a partial barrier may be sufficient to maintain population persistence and gene flow (Neville et al. 2016; Soanes et al. 2017) and enable recolonization after extinction events. Additional passability enhancements, while possibly increasing productivity (Neville et al. 2016), will likely provide diminishing returns to the ecology of the stream system, particularly for species of fish that do not have specialized habitat requirements (e.g. the brook trout in this study). These diminishing returns, in addition to the added difficulty of assessing restoration success, might prompt managers to put more weight on sites with more severe barriers during restoration prioritization.

Finally, it is important to consider the nature of the system when designing restoration assessment studies. Including spatial and temporal controls in restoration study designs can be logistically challenging. However, our results demonstrate that without suitable controls, the chances of drawing false conclusions regarding restorations in temporally and spatially dynamic systems are substantial.

Acknowledgments

LITERATURE CITED

Anderson WG, Mckinley RS, Colavecchia M (1997) The use of clove oil as an anesthetic for rainbow trout and its effects on swimming performance. North American Journal of Fisheries Management 17: 301–307

Ayllón D, Nicola GG, Parra I, Elvira B, Almodóvar A (2014) Spatio-temporal habitat selection shifts in brown trout populations under contrasting natural flow regimes. Ecohydrology 7:569–579

Bates D, Mächler M, Bolker B, Walker S (2015) Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67:1–48

Benitez J-P, Matondo BN, Dierckx A, Ovidio M (2015) An overview of potamodromous fish upstream movements in medium-sized rivers, by means of fish passes monitoring. Aquatic Ecology 49:481–497

Bernhardt ES, Palmer MA, Allan JD, Alexander G, Barnas K, Brooks S, Carr J, Clayton S, Dahm C, Follstad-Shah J (2005) Synthesizing U.S. river restoration efforts. Science 308:636–637

Bohlin T, Pettersson J, Degerman E (2001) Population density of migratory and resident brown trout (*Salmo trutta*) in relation to altitude: evidence for a migration cost. Journal of Animal Ecology 70:112–121

Bourne C (2013) How to quantify aquatic connectivity? Verifying the effectiveness of the dendritic connectivity indes as a tool for assessing stream fragmentation. MSc thesis. Memorial University of Newfoundland, St. John's, NL

Branco P, Amaral SD, Ferreira MT, Santos JM (2017) Do small barriers affect the movement of freshwater fish by increasing residency? Science of the Total Environment 581–582: 486–494

Branco P, Segurado P, Santos JM, Pinheiro P, Ferreira MT (2011) Does longitudinal connectivity loss affect the distribution of freshwater fish? Ecological Engineering 48:70–78

Brewitt KS, Danner EM (2014) Spatio-temporal temperature variation influences juvenile steelhead (*Oncorhynchus mykiss*) use of thermal refuges. Ecosphere 5:1–26

Bried JT, Ervin GN (2011) Randomized intervention analysis for detecting non-random change and management impact: dragonfly examples. Ecological Indicators 11:535–539

Cote D (2007) Measurements of salmonid population performance in relation to habitat in eastern Newfoundland streams. Journal of Fish Biology 70:1134–1147

Cote D, Kehler DG, Bourne C, Wiersma YF (2009) A new measure of longitudinal connectivity for stream networks. Landscape Ecology 24: 101–113

Diebel MW, Fedora M, Cogswell S, O'Hanley JR (2015) Effects of road crossings on habitat connectivity for stream-resident fish. River Research and Applications 31:1251–1261

Elliott JM (1984) Growth, size, biomass and production of young migratory trout *Salmo trutta* in a Lake District stream, 1966-83. The Journal of Animal Ecology 53:979–994

Enders EC, Stickler M, Pennell CJ, Cote D, Alfredsen K, Scruton DA (2008) Variations in distribution and mobility of Atlantic salmon parr during winter in a small, steep river. Hydrobiologia 609:37–44

Evans NT, Riley CW, Lamberti GA (2015) Culvert replacement enhances connectivity of stream fish communities in a Michigan drainage network. Transactions of the American Fisheries Society 144:967–976

Favaro C, Moore JW, Reynolds JD, Beakes MP (2014) Potential loss and rehabilitation of stream longitudinal connectivity: fish populations in urban streams with culverts. Canadian Journal of Fisheries and Aquatic Sciences 71:1805–1816

Goerig E, Castro-Santos T (2017) Is motivation important to brook trout passage through culverts? Canadian Journal of Fisheries and Aquatic Sciences 74:885–893

Gowan C, Fausch KD (1996) Mobile brook trout in two high-elevation Colorado streams: reevaluating the concept of restricted movement. Canadian Journal of Fisheries and Aquatic Sciences 53:1370–1381

Gowan C, Fausch KD (2002) Why do foraging stream salmonids move during summer? Environmental Biology of Fishes 64:139–153

Januchowski-Hartley SR, Mcintyre PB, Diebel M, Doran PJ, Infante DM, Joseph C, Allan JD (2013) Restoring aquatic ecosystem connectivity requires expanding inventories of both dams and road crossings. Frontiers in Ecology and the Environment 11:211–217

Jarrett RD (1985) Determination of roughness coefficients for streams in Colorado. US Geological Survey, Lakewood, Colorado

King S, O'Hanley JR (2014) Optimal fish passage barrier removal—revisited. River Research and Applications 32(3):418–428

King S, O'hanley JR, Newbold LR, Kemp PS, Diebel MW (2016) A toolkit for optimizing fish passage barrier mitigation actions. Journal of Applied Ecology 54(2):599–611

Kwak TJ, Waters TF (1997) Trout production dynamics and water quality in Minnesota streams. Transactions of the American Fisheries Society 126:35–48

Langford TEL, Shaw PJ, Ferguson AJD, Howard SR (2009) Long-term recovery of macroinvertebrate biota in grossly polluted streams: re-colonisation as a constraint to ecological quality. Ecological Indicators 9: 1064–1077

Mahlum S, Kehler D, Cote D, Wiersma YF, Stanfield L, Rosenfeld J (2014a) Assessing the biological relevance of aquatic connectivity to stream fish communities. Canadian Journal of Fisheries and Aquatic Sciences 71:1–12

Mahlum SK, Cote D, Kehler DG, Wiersma YF, Clarke KR (2014b) Evaluating the barrier assessment technique FishXing and the upstream movement of fish through road culverts. Transactions of the American Fisheries Society 143:39–48

Maitland BM, Poesch M, Anderson AE (2016) Prioritizing culvert removals to restore habitat for at-risk salmonids in the boreal forest. Fisheries Management and Ecology 23:489–502

Meyers LS, Thuemler TF, Kornely GW (1992) Seasonal movements of brown trout in northeast Wisconsin. North American Journal of Fisheries Management 12:433–441

Myers RL, Nieraeth S (2016) Restoration and Coho Salmon carrying capacity of Chester Creek, an urban stream in anchorage, Alaska. Northwest Science 90:146–158

Neeson TM, Ferris MC, Diebel MW, Doran PJ, O'hanley JR, Mcintyre PB (2015) Enhancing ecosystem restoration efficiency through spatial and temporal coordination. Proceedings of the National Academy of Sciences of the United States of America 112:6236–6241

Neville H, Dauwalter D, Peacock M (2016) Monitoring demographic and genetic responses of a threatened inland trout to habitat reconnection. Transactions of the American Fisheries Society 145:610–626

Noonan MJ, Grant JWA, Jackson CD (2012) A quantitative assessment of fish passage efficiency. Fish and Fisheries 13:450–464

Ogren SA, Huckins CJ (2015) Culvert replacements: improvement of stream biotic integrity? Restoration Ecology 23:821–828

Olsson IC, Greenberg LA (2004) Partial migration in a landlocked brown trout population. Journal of Fish Biology 65:106–121

Olsson IC, Greenberg LA, Bergman E, Wysujack K (2006) Environmentally induced migration: the importance of food. Ecology Letters 9: 645–651

Pepino M, Rodriguez MA, Magnan P (2012) Fish dispersal in fragmented landscapes: a modeling framework for quantifying the permeability of structural barriers. Ecological Applications 22:1435–1445

Pépino M, Rodríguez MA, Magnan P (2016) Assessing the detectability of road crossing effects in streams: mark–recapture sampling designs under complex fish movement behaviours. Journal of Applied Ecology 53:1831–1841

Perkin JS, Gido KB (2012) Fragmentation alters stream fish community structure in dendritic ecological networks. Ecological Applications 22: 2176–2187

Petty JT, Hansbarger JL, Huntsman BM, Mazik PM (2012) Brook trout movement in response to temperature, flow, and thermal refugia within a complex Appalachian riverscape. Transactions of the American Fisheries Society 141:1060–1073

Porto LM, Mclaughlin RL, Noakes DLG (1999) Low-head barrier dams restrict the movements of fishes in two Lake Ontario streams. North American Journal of Fisheries Management 19:1028–1036

Pretty JL, Harrison SSC, Shepherd DJ, Smith C, Hildrew AG, Hey RD (2003) River rehabilitation and fish populations: assessing the benefit of instream structures. Journal of Applied Ecology 40:251–265

R Development Core Team (2017) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria http://www.R-project.org/ (accessed 21 April 2017)

Riggs HC (1985) Flow characteristics at Ungaged sites. Pages 131–144. In: Developments in water science series #22. Elsevier Science Publishing Company Inc., New York

Rodeles AA, Galicia D, Miranda R (2017) Recommendations for monitoring freshwater fishes in river restoration plans: a wasted opportunity for assessing impact. Aquatic Conservation: Marine and Freshwater Ecosystems 27(4):880–885

Roghair CN, Dolloff CA (2005) Brook trout movement during and after recolonization of a naturally defaunated stream reach. North American Journal of Fisheries Management 25:777–784

Saunders MI, Brown CJ, Foley MM, Febria CM, Albright R, Mehling MG, Kavanaugh MT, Burfeind DD (2016) Human impacts on connectivity in marine and freshwater ecosystems assessed using graph theory: a review. Marine and Freshwater Research 67:277–290

Shaw EA, Lange E, Shucksmith JD, Lerner DN (2016) Importance of partial barriers and temporal variation in flow when modelling connectivity in fragmented river systems. Ecological Engineering 91:515–528

Smith DJ, Van Der Ree R, Rosell C (2015) Wildlife crossing structures. Pages 172–183. In: Handbook of road ecology. John Wiley & Sons, Ltd Oxford, UK

Smokorowski KE, Randall RG (2017) Cautions on using the before-after-control-impact design in environmental effects monitoring programs. FACETS 2:212–232

Soanes K, Taylor AC, Sunnucks P, Vesk PA, Cesarini S, Der Ree R (2017) Evaluating the success of wildlife crossing structures using genetic approaches and an experimental design: lessons from a gliding mammal. Journal of Applied Ecology Online First doi: 10.1111/1365-2664.12966

Stickler M, Enders EC, Pennell CJ, Cote D, Alfredsen K, Scruton DA (2008) Stream gradient-related movement and growth of Atlantic salmon parr during winter. Transactions of the American Fisheries Society 137: 371–385

Thorne C, Castro J, Cluer B, Skidmore P, Shea C (2015) Project risk screening matrix for river management and restoration. River Research and Applications 31:611–626

Tummers JS, Hudson S, Lucas MC (2016) Evaluating the effectiveness of restoring longitudinal connectivity for stream fish communities: towards a more holistic approach. Science of the Total Environment 569: 850–860

Watson DM, Doerr VA, Banks SC, Driscoll DA, Van Der Ree R, Doerr ED, Sunnucks P (2017) Monitoring ecological consequences of efforts to restore landscape-scale connectivity. Biological Conservation 206: 201–209

Wickham H (2016) ggplot2: elegant graphics for data analysis. Springer, New York

Wysujack K, Greenberg LA, Bergman E, Olsson IC (2009) The role of the environment in partial migration: food availability affects the adoption of a migratory tactic in brown trout *Salmo trutta*. Ecology of Freshwater Fish 18:52–59

Supporting Information

The following information may be found in the online version of this article:

Figure S1. The pre- and post-cumulative discharge frequency for each stream.
Table S1. Summary table of the number of fish that had a passage attempt and the number of passage attempts across the study period for the culvert and reference.
Table S2. Summary table of PS when analyzed with BACI, B-A, and C-I.

Table S3. Results from the randomization of the BACI, B-A and C-I for the RPF and APF.
Appendix S1. Restoration actions taken in Arnolds Pond Brook prior to the study period.
Appendix S2. Methods for randomization of RPF and APF.
Appendix S3. Corresponding R code for the RPF and APF.