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ARTICLE

Urban planning for fishes: untangling a new project's effects from old infrastructure and regional patterns

Luke Warkentin, Corinna Favaro, David C. Scott, Rebecca Seifert, and Jonathan W. Moore

Abstract: Urbanization has altered fish communities in many ways. However, as cities expand and redevelop, it is challenging to assess the impacts of new projects given existing alteration. We investigated how new and old infrastructure alters fish communities over a 4-year period in Metro Vancouver, British Columbia (Canada). We compared fish communities from a stream altered by a new rapid transit rail line and seven reference sites over 4 years, from before to after construction. We provide evidence that new and old projects depress the density, species richness, and diversity of fish communities. During and after construction, sections of the altered stream had one fewer species and lower density compared with preconstruction and reference streams. Streams without existing culverts had more species and greater diversity than those with culverts, but only in some years. Diversity was lower in 1 year of the study across all streams. We argue that most monitoring in Canada is insufficient to detect the incremental changes that new projects may cause and suggest improvements in monitoring and protecting reference streams.

Résumé : L'urbanisation a modifié les communautés de poissons de nombreuses façons. Avec l'expansion et le réaménagement des villes, il devient difficile d'évaluer les impacts de nouveaux projets étant donné les modifications antérieures. Nous avons étudié la manière dont les infrastructures nouvelles et anciennes modifient les communautés de poissons sur une période de quatre ans dans la région métropolitaine de Vancouver (Colombie-Britannique, Canada). Nous avons comparé les communautés de poissons d'un cours d'eau modifié par une nouvelle ligne de transport en commun rapide et de sept sites de référence durant une période de quatre ans, allant d'avant la construction jusqu'après celle-ci. Nous fournissons des preuves de baisses de la densité, de la richesse spécifique et de la diversité de communautés de poissons causées par des projets nouveaux et anciens. Durant et après la construction, des tronçons du cours d'eau modifié présentaient une espèce de moins et une plus faible densité que les cours d'eau avant la construction et de référence. Les cours d'eau sans ponceau existant présentaient plus d'espèces et une plus grande diversité que ceux avec des ponceaux, mais seulement certaines années. Durant une des années de l'étude, la diversité était plus faible pour tous les cours d'eau. Nous arguons que la plupart des efforts de surveillance au Canada ne sont pas suffisants pour déceler les changements cumulatifs pouvant être causés par de nouveaux projets et proposons des moyens pour améliorer la surveillance et la protection des cours d'eau de référence. [Traduit par la Rédaction]

Introduction

There is growing appreciation and evidence that urbanization has negative impacts on many native fishes (Paul and Meyer 2001; Miltner et al. 2004; Wheeler et al. 2005; Jeffrey et al. 2015; Moore and Olden 2017). At the individual fish level, increased urbanization depresses the growth and reproduction of fishes with recreational value (Nelson et al. 2009). Urbanization can also reduce genetic diversity within populations; for example, Mather et al. (2015) found that close proximity to urban areas was linked to greatly reduced genetic diversity for the ornate rainbowfish, *Rhadinocentrus ornatus*. There is also widespread evidence of urbanization having dramatic impacts on fish communities (Weaver and Garman 1994; Wang et al. 2000, 2001; Moore and Olden 2017). For example, Wang et al. (2000) found that 20 years of urbanization was associated with a 15% decrease in the mean number of fish species and a 41% decrease in density. After 32 years of urbanization in another watershed, fish diversity and abundance decreased significantly as riparian development increased (Weaver and Garman 1994). Thus, historic and ongoing urbanization can dramatically alter fish communities.

As cities grow, quantifying incremental changes in fish communities and identifying their causes could assist planners and regulators to weigh the actual risks of development for stream ecosystems. Quantifying these risks and applying this knowledge is timely, as the global population in urban areas is projected to increase by 84% from 2009 to 2050 (United Nations 2009). However, many political and practical factors impede environmental decision-making during urban development. In Canada, these challenges to environmental planning include a lack of peer-reviewed evidence (Wheeler et al. 2005), assessments motivated by legal compliance as opposed to science (Ball et al. 2012), a lack of leadership (Kristensen et al. 2013), noncompliance with monitoring requirements, and qualitative rather than quantitative assessments (Harper and Quigley 2005). The quantification of impacts is also a fundamentally challenging scientific topic: urban development has many pathways of effects on fishes that are difficult to tease apart (Paul and Meyer 2001; Seitz et al. 2011; Jeffrey et al. 2015). Insight could come from studies that can detect and differentiate natural variation in fish communities across both time and space, variation associated with existing impacts, and the effects of new projects.

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L. Warkentin, C. Favaro, D.C. Scott, R. Seifert, and J.W. Moore. Earth2Ocean Research Group, Simon Fraser University, 8888 University Dr., Burnaby, BC V5A 1S6, Canada.

Corresponding author: Luke Warkentin (email: luke.warkentin@gmail.com).

Table 1. Study streams in the lower mainland of British Columbia, Canada (adapted from table 1 in Favaro et al. 2014).

Stream name	Stream type	Treatment	Mean wetted width (m)	City	Year culvert built	Culvert length (m)
Partington	No culvert	Reference	3.2	Coquitlam	—	—
Cypress	No culvert	Reference	5.6	West Vancouver	—	—
Nathan	No culvert	Reference	2.3	Langley	—	—
Anderson	No culvert	Reference	5.0	Abbotsford	—	—
Mossom	Culvert	Reference	4.6	Port Moody	1950	19.5
Nelson	Culvert	Reference	4.0	West Vancouver	1957	76.7
Stoney	Culvert	Reference	4.3	Burnaby	1960	57.4
Suter Brook*	Culvert	Impact	2.1	Port Moody	1995	50.6

Note: Mean wetted width is of all reaches and years.
*Focal impact site with 51 sampling reaches instead of 10.

Culverts are pervasive in urban streams and represent a major pathway by which development impacts fishes. The frequency of movement by fishes through culverts can be lower than the frequency of movement under clear-span bridges (Benton et al. 2008). Culverts can have higher water velocities than open-box crossings and can inhibit fish passage by an order of magnitude (Warren and Pardew 1998). In another study, only bottomless box culverts allowed for the normal movement of fish upstream and downstream, in contrast with the significant impacts of round culverts (Norman et al. 2009). Because culverts affect the movement of fish, they also affect fish communities at the whole-stream scale (Nislow et al. 2011; Favaro et al. 2014; Evans et al. 2015). For example, after 26 years of water quality remediation in a heavily polluted watershed and the reintroduction of native fishes, recovery of fish species was lower at sites with more downstream culverts and other barriers (i.e., weirs and flumes; McManamay et al. 2016). Impacts are also species specific: one study found that densities of coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) were higher and densities of prickly sculpin (*Cottus asper*) were lower upstream of culverts compared with downstream (Favaro and Moore 2015). Thus, culverts create a legacy of fragmentation in urban watersheds that affects fish communities.

Temporal variation in the effects of existing fragmentation further challenges our ability to disentangle the effects of existing impacts, new projects, and natural background variation. The effects of culverts can vary over time based on physical factors such as water temperature and water velocity (Baker and Votapka 1990; Votapka 1991; Cahoon et al. 2007). Cahoon et al. (2007) found that Yellowstone cutthroat trout (*Oncorhynchus clarkii bouvieri*) and rainbow trout (*Oncorhynchus mykiss*) were more successful in passing barriers at lower water velocities and higher water temperatures. They also found that the probability of successful passage could be predicted by water velocity in 98% of cases. In another study, the proportion of fishes moving through culverts per day was negatively correlated with water velocity in the summer and for all seasons combined (Warren and Pardew 1998). A 2-year study found that the effect of the cumulative number of culverts on density of coho salmon (*Oncorhynchus kisutch*) and fish diversity changed from year to year, possibly due to differences in stream discharge (Favaro and Moore 2015). Thus, the assessment of new projects needs to account for temporal variation in the effects of existing fragmentation.

This study aimed to determine the relative effects of both existing culverts and a new infrastructure project on fish communities in urban streams. By examining streams with and without culverts and one stream that was subject to a new impact — the construction of a rapid transit rail line — over 4 years, we addressed the following question: how do new and existing infrastructure projects affect fish communities over time? Because this study disentangles the effects of old infrastructure and a new project, the results provide scientific evidence relevant to understanding the potential costs of urban development for fish communities. While this study focuses on a specific development, it has broad implications for urban planning and monitoring in rapidly changing urban watersheds.

Methods

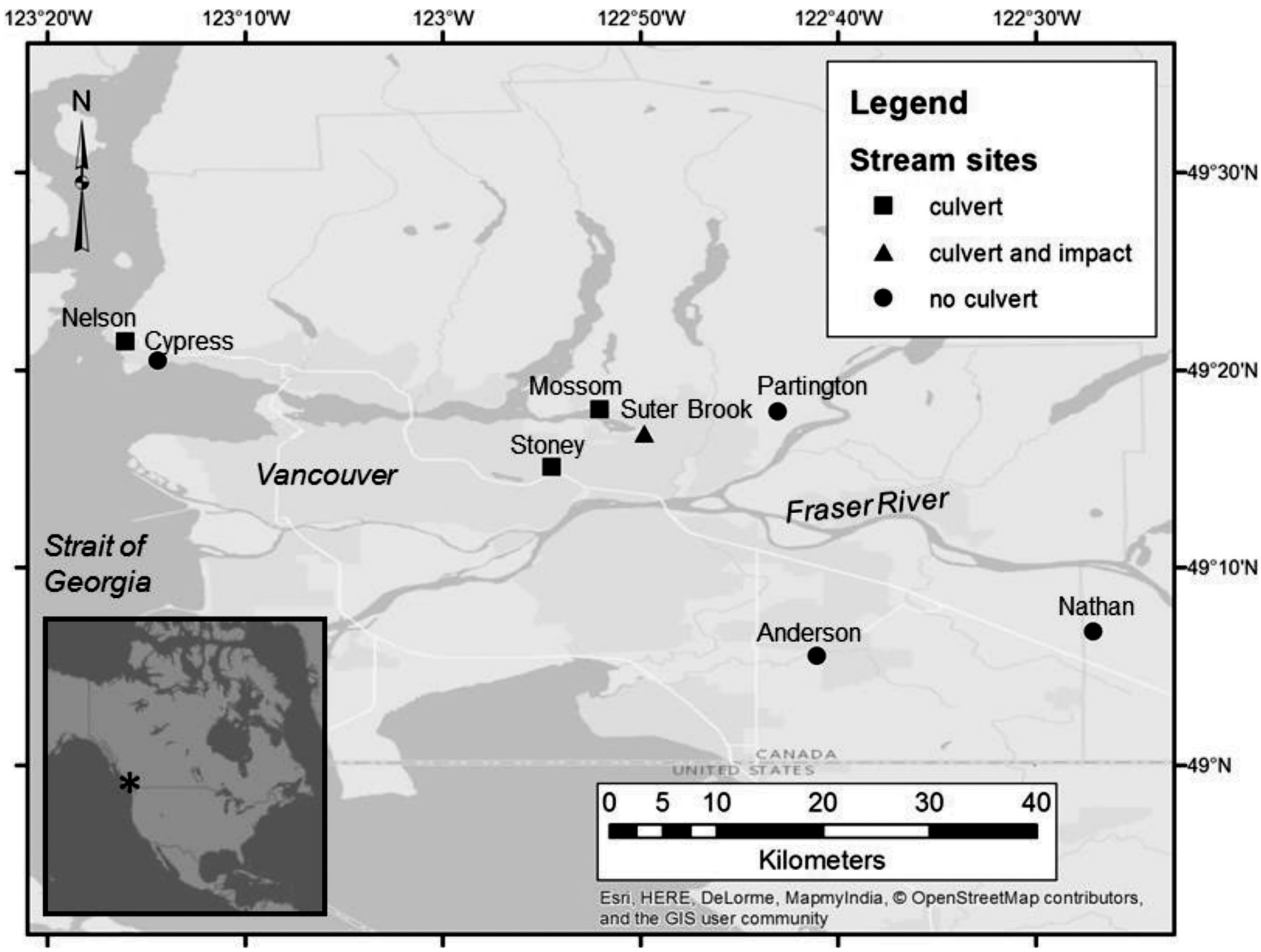
Study region

Metro Vancouver is located in the Lower Fraser Valley of British Columbia, Canada, where development intersects with streams and fish communities. The Fraser River runs through this valley, draining roughly one-third of the province and supporting the largest populations of wild Pacific salmon in Canada (Northcote and Burwash 1991). The Lower Fraser Valley has approximately 1200 streams that flow into the Fraser River or the Pacific Ocean (Fisheries and Oceans Canada 1998). These watersheds support roughly 40 fish species with diverse life histories and habitat requirements (McPhail 2007). Through urban development, most of these streams have already been fragmented, channelized, polluted, diverted, cleared of riparian cover, or otherwise altered. Based on a 20-year-old assessment, 15% of these streams had been lost, 48% were endangered, 23% were threatened, and only 14% were wild (Fisheries and Oceans Canada 1998). The population of Metro Vancouver is projected to grow from 2.3 million to 3.4 million from 2006 to 2041, supported by further development (Metro Vancouver 2011). In this study region, we examined whether old and new infrastructure are linked to fish communities across space and time.

Overview of experimental design

The study's goal was to disentangle the effects of old and new infrastructure on fish communities and account for natural variation across space and time through a modified asymmetric before–after, control–impact design. We examined fish communities in eight streams across 4 years (2012–2015). Four streams had no culverts and four had culverts. Our field team repeatedly sampled 121 study reaches among these eight urban streams (Table 1; Fig. 1), once each summer (July and August) over the 4 years. The fish communities across these sites are similar and are typical of relatively small (2–5.6 m wetted width), moderate-gradient streams in the Lower Fraser Valley (McPhail 2007; Favaro et al. 2014; Favaro and Moore 2015). One of the four streams with culverts also had a rapid transit rail line built over it during the study period. The sampling design allowed for repeated measures of fish communities in discrete reaches over space and time with multiple reference systems. We examined total fish density, species richness, diversity, and the densities of the three most abundant taxa to detect various effects, as old and new impacts can operate though different pathways (Allan 2004). We used mixed-effects models and model ranking with Akaike's information criterion (AIC) to analyze the nested data set with repeated measures of each reach. This analysis allowed us to test whether the presence of culverts and the construction of a rapid transit rail line had significant effects on fish communities in urban streams over 4 years. We predicted that all six variables would be lower in

Fig. 1. Location map of study sites within the lower mainland region of British Columbia, Canada. Circles indicate reference sites without culverts, squares indicate reference sites with culverts, and the triangle represents the focal impact site of Suter Brook where a rapid transit rail line was built between the 2013 and 2014 sampling periods. No work was done before the 2012 sampling year. Some preliminary in-stream work was conducted between 2012 and 2013. Major in-stream works and channel realignment were conducted between 2013 and 2014. After sampling in 2014, contractors did restoration work in the riparian area. Sampling in 2015 occurred after the project was completed.



streams with culverts and lower in a stream during and after the construction of the rapid transit rail line compared with years before construction. We also predicted that the effect of culverts on fish communities would vary over time, as observed by Favaro and Moore (2015).

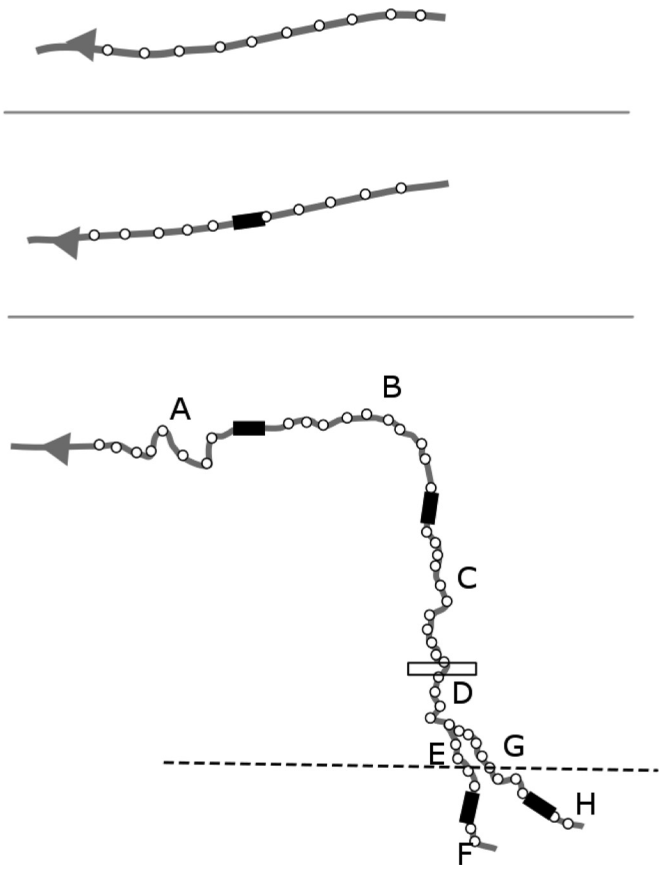
Specific project: rapid transit rail line construction

One of the sites with culverts, Suter Brook, was also subject to an additional impact. A new rapid transit rail line was built over the stream within the 4-year period of study, and an associated transit station was built nearby. The large construction project involved the removal of riparian vegetation, bank modification, realignment of the channel, and the addition of off-channel habitat. The rail line is situated on elevated pilings, and the stream channel was modified and restored after the construction project. We sampled in 2012 before construction began. Before sampling in 2013, contractors cleared trees in the project area, graded the site with gravel, installed temporary exclusion fencing, and were partway through replacing a damaged culvert. In between sampling in 2013 and 2014, contractors dewatered the channel, installed the pilings for the elevated track of the rapid transit rail line, and reconstructed the channel with additional off-channel habitat, but the riparian area was not replanted and remained a construction site. After sampling in 2014, contractors replanted the riparian area and removed construction materials. In 2015, sampling occurred after the completion of all project work. Thus, we sampled 1 year prior to any work (2012), 1 year after clearing and preliminary work (2013), 1 year after major habitat disturbance, rail line construction, and channel restoration (2014), and 1 year after riparian planting (2015) so that our study had a time frame that could detect potential short-term effects on fish communities.

Fish sampling

Discrete reaches were the basic unit of the sample design and were sampled repeatedly across space and time. We sampled fish in all eight streams in multiple consecutive reaches (~20 m long). The eight streams sampled fall into categories of streams with culverts and those without. Several reaches at Suter Brook were slightly longer or shorter because the length of sections in between culverts was not divisible by 20. We sampled 10 consecutive reaches in reference streams without culverts and five reaches above and five below the culvert in reference streams with culverts (Fig. 2). None of the culverts were perched during sampling, except the culvert at Mossom Creek, which was perched by 20 cm in 2012. At Suter Brook, the stream where the rail line was built,

Fig. 2. Generalized sketches of streams in gray with reach layouts for (top) reference streams without culverts, (middle) reference streams with culverts, and (bottom) the focal impact site of Suter Brook (modified from figure 1 in Favaro and Moore 2015). Dots represent the downstream end of reaches. Black rectangles indicate culverts. The dashed line indicates where the rapid transit rail line was built between the summers of 2013 and 2014. The empty rectangle is a clear-span bridge. Letters indicate sections that fall between breaks (culverts, bridge, and confluence). Arrows on streams indicate the direction of flow. Drawings are not to scale.



more of the stream was surveyed: 51 reaches that incorporated four culverts (Fig. 2). For each reach, we measured length and wetted width at a representative cross-section. To obtain an index of fish abundance, we conducted single-pass electrofishing with one operator and one netter using a Smith-Root LR-24 Electrofisher. The regulators that issued permits for fish sampling (Fisheries and Oceans Canada and B.C. Ministry of Environment) required single-pass electrofishing without block nets to reduce the impacts on fish (Favaro and Moore 2015). In addition, catchability with single-pass electrofishing is relatively high in small, clear streams, such as those in this study (Bohlin et al. 1989). After capture, fish were lightly anaesthetized and their species, mass, and fork length (for salmonids) or standard length (for all other species) were recorded. Some trout (rainbow or coastal cutthroat trout) were too small to be identified to species (<80 mm), and these were classified as juvenile trout.

Statistical analysis

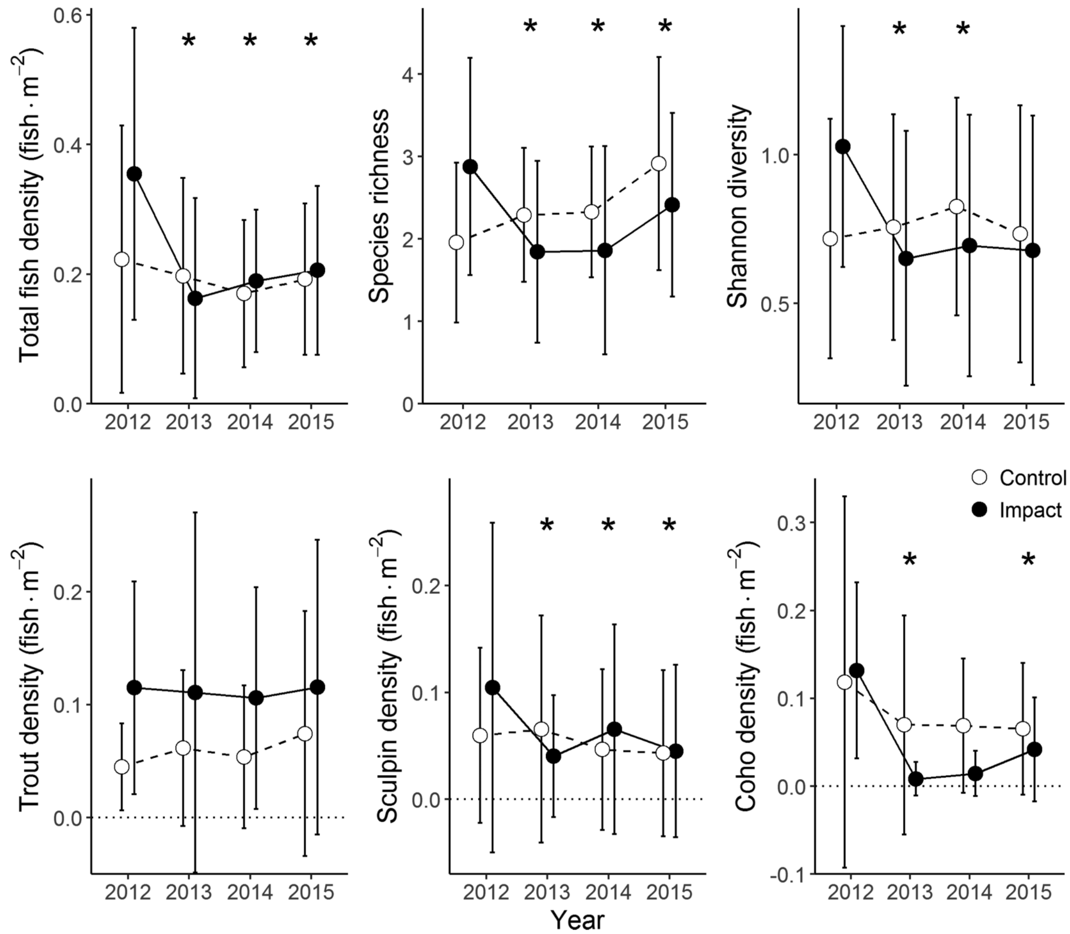
We examined six variables — total density, species richness, diversity, and the densities of the three most abundant taxa of fishes — to detect a wide range of effects (Allan 2004). For each reach in each year, we calculated the following six variables: total fish density (fish·m^{-2}) as the count of fishes in the reach divided by the wetted surface area of the reach (length × width of wetted channel), species richness (count of species in the reach, excluding juvenile trout), Shannon's diversity index (H') (excluding juvenile trout; Shannon and Weaver 1949), and density for each of the three most abundant taxa of fishes. These three taxa were trout (*Oncorhynchus mykiss*, *Oncorhynchus clarkii clarkii*, and juvenile trout), sculpin (*Cottus asper* and *Cottus aleuticus*), and coho salmon (*Oncorhynchus kisutch*). The fish density data (total and for each of the three taxa) were right-skewed, so they were square-root-transformed before subsequent analysis with mixed-effects models. Hereafter, square-root-transformed total density is referred to as total density, and square-root-transformed density for trout, sculpin, and coho salmon is referred to as trout density, sculpin density, and coho density, respectively (unless otherwise noted).

To analyze the data, we used mixed-effects models to address the spatially nested data structure, repeated measures over time, and multiple categorical variables of interest. We built mixed-effects models — using total density, species richness, diversity, and densities for the three taxa as response variables — to determine whether the presence of culverts and the construction of the rapid transit rail line affected fish communities over time. The full model included three categorical fixed effects corresponding to the impacts of interest and time. The three variables were "stream type" (values of "cvt" for culverts present and "no" for no culverts present), "treatment" (values of "C" for reference streams with no rail line and "I" for the impact stream where the rail line was built), and "year" (categorical variables of "2012", "2013", "2014", and "2015"). The full model also included interaction terms of stream type × year and treatment × year to determine whether these effects varied among years. While similar to a before–after, control–impact design (Underwood 1994; Louhi et al. 2010; Aguado-Giménez et al. 2012; Weaver and Kwak 2013; Mateos-Molina et al. 2014), our design made use of a finer temporal scale by including all 4 years instead of before and after periods. The stream type × year interaction term was interpreted as explaining variation in fish communities due to the presence of culverts in certain years. The treatment × year interaction term was interpreted as explaining variation in fish communities due to the construction of the rapid transit rail line in years during or after construction. This allowed for the detection of impacts that varied over time. We also included a random effect term of reach nested within stream that allowed the intercept of the model to vary among streams, taking into account the natural variation of streams and their fish communities. We fit this full model for each of the six variables, plus models with all combinations of fixed-effects terms, including intercept-only models.

We used the maximum likelihood method to fit all models and then ranked models based on AIC adjusted for small sample size (AIC_c; Burnham and Anderson 2002) using the AICcmodavg package (Mazerolle 2015). This approach provided the model that included the fixed effects that best explained the variation for each of the six metrics. After choosing the model with the lowest AIC_c score, we used the intervals package (Bourgnon 2015) to extract 95% confidence intervals for the effect terms for this model. The effect terms whose 95% confidence intervals did not span zero were interpreted as having a detectable effect on fish communities. We used the nlme package to construct the models (Pinheiro et al. 2015) and used R (R Core Team 2015) and RStudio (RStudio Team 2015) for all statistical analyses. We used Esri ArcGIS 10.2 (Esri 2013) to make Fig. 1.

In addition to the mixed-effects analysis, densities of the three most abundant taxa were analyzed for eight discrete sections of the impact site (A–G; Fig. 2). Sections A, B, C, and D were downstream of the construction zone where the channel was reconstructed, sections E and G spanned the project footprint, and sections F and H were upstream of the impact and culverts (Fig. 2). This enabled the detection of dynamics that were specific to certain taxa of fish over time in certain sections in the stream where the rapid transit rail line was built.

Fig. 3. Metrics of fish communities factored by control or impact reaches over 4 years. Points are mean values, and error bars indicate standard deviation. Total fish density and density for trout, sculpin, and coho are untransformed. An asterisk above a point indicates that for that year, the treatment (I) × year effect term had a 95% confidence interval that was negative and did not span zero. At the impact stream, no work was done before the 2012 sampling year. Some preliminary in-stream work was conducted between 2012 and 2013. Major in-stream works and channel realignment were conducted between 2013 and 2014. After sampling in 2014, contractors did restoration work in the riparian area. Sampling in 2015 occurred after the project was completed.



Results

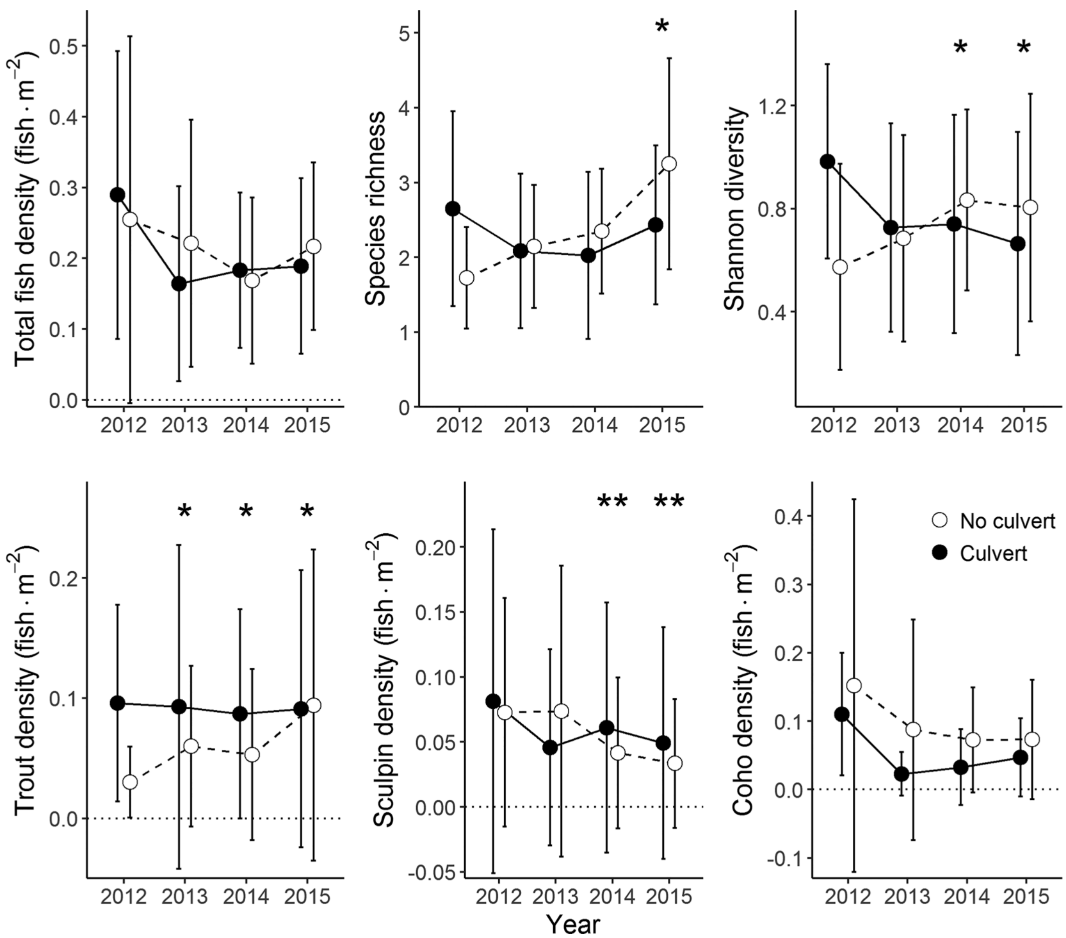
Over 4 years, we sampled 4464 fish of the following species: juvenile coho salmon (*Oncorhynchus kisutch*), juvenile Chinook salmon (*Oncorhynchus tshawytscha*), rainbow trout (*Oncorhynchus mykiss*), coastal cutthroat trout (*Oncorhynchus clarkii clarkii*), prickly sculpin (*Cottus asper*), coastrange sculpin (*Cottus aleuticus*), threespine stickleback (*Gasterosteus aculeatus*), lamprey (*Lampetra* spp.), and pumpkinseed sunfish (*Lepomis gibbosus*). Of 4464 fish sampled over 4 years in eight streams, only five were non-native (all pumpkinseed sunfish).

Trajectories for total density, species richness, Shannon diversity, and trout, sculpin, and coho densities varied between impact and control streams and between streams with and without culverts (Figs. 3 and 4). Total density (untransformed) was typically within the range of 0.1 to 0.4 fish·m^{-2}, and reaches commonly had between one and four species each. On the whole, these metrics were highly variable from reach to reach, resulting in standard deviations with large overlap between years and treatments in all cases.

Total density was lower in years during and after the construction of the rapid transit rail line in the impact stream, based on the modeling approach that took temporal variability of multiple reference streams into account (Fig. 3). The top-fitted model for fish density included the fixed-effects terms treatment × year, stream type, and year, but not the stream type × year interaction term (Table 2). For fish density, the 95% confidence intervals for the effects of all treatment (I) × year terms for 2013, 2014, and 2015 were negative and did not span zero (Fig. 5). Therefore, fish densities were significantly lower in the stream affected by the construction of the rapid transit rail line in all years after the project began and 1 year after the project was completed.

Species richness was also lower in all years during and after the construction of the rapid transit rail line based on the mixed-effects model results (Fig. 3). Species richness was also higher in streams without culverts in 2015 (Fig. 4). The top-fitted model for species richness was the full model with all fixed effects, including both interaction terms treatment × year and stream type × year (Table 2). The 95% confidence intervals for the effects of the treatment (I) × year terms on species richness for years after 2012 were negative and did not span zero (Fig. 5). The estimates for these effects represented losses of 1.3, 1.0, and 0.7 species per reach for the impact stream in the years 2013, 2014, and 2015, respectively, compared with 2012 (Table 3). This means that, on average, there was one fewer species per reach in the stream where the rapid transit rail line was built, in years during and after construction. The stream type (no) × year (2015) term was positive and did not span zero for species richness, and it had an estimate of 1.3 (Table 3). This represents an effect of 1.3 more species per reach in 2015 in streams without culverts. Therefore,

Fig. 4. Metrics of fish communities factored by the presence or absence of culverts over 4 years. Points are mean values averaged between reaches, and error bars represent standard deviation. Total fish density and density for trout, sculpin, and coho are untransformed. An asterisk above a point indicates that for that year, the stream type (no) × year effect term had a 95% confidence interval that was positive and did not span zero. Two asterisks above a point indicate that for that year, the stream type (no) × year effect term had a 95% confidence interval that was negative and did not span zero. At the impact stream, no work was done before the 2012 sampling year. Some preliminary in-stream work was conducted between 2012 and 2013. Major in-stream works and channel realignment were conducted between 2013 and 2014. After sampling in 2014, contractors did restoration work in the riparian area. Sampling in 2015 occurred after the project was completed.



streams without culverts had more species than streams with culverts, but this effect was detected in only 1 year.

Variation in Shannon diversity was associated with the rail line construction, presence of culverts, and natural variation. The top model for Shannon diversity was the full model with all fixed effects, including the interaction terms stream type × year and treatment × year (Table 2). The treatment (I) × year (2013) and treatment (I) × year (2014) terms both had 95% confidence intervals that were negative and did not span zero (Fig. 5; Table 3). Therefore, Shannon diversity was significantly lower in 2013 and 2014 for reaches in the stream where construction occurred. A significant decrease in diversity was not detected by the model in the impact stream in 2015, after all construction was complete. The effects of the stream type (no) × year (2014) and stream type (no) × year (2015) terms both had 95% confidence intervals that were positive and did not span zero. This means that reaches in streams without culverts had predictably higher Shannon diversity in 2014 and 2015, indicating a negative effect of culverts in only some years. In addition, the year (2015) term had a negative 95% confidence interval that did not span zero. Therefore, regardless of culverts or new construction, drops in diversity across all streams in the study were explained by natural variation in 2015.

Culverts had a negative effect on trout density from 2013 to 2015 (Fig. 4). Trout density was not affected by the construction project (Fig. 3) and was lower in 2015 across streams. The top-fitted model for trout density included the stream type × year interaction term but not the treatment × year term (Table 2). The stream type (no) × year terms for 2013, 2014, and 2015 had 95% confidence intervals that were positive and did not span zero, meaning that trout were denser in streams without culverts in 2013–2015 (Fig. 5; Table 3). The year (2015) effect term had a 95% confidence interval that was negative and did not span zero, indicating that trout densities were lower across all streams in 2015.

Sculpin densities were lower at the impact stream in all years during and after construction. Contrary to our prediction, the density of sculpin was lower in streams without culverts in 2014 and 2015. The top-fitted model for sculpin density included all fixed-effects terms and both treatment × year and stream type × year interaction terms (Table 2). The 95% confidence intervals for the treatment (I) × year terms for 2013, 2014, and 2015 and for the stream type (no) × year terms for 2014 and 2015 were negative and did not span zero (Fig. 5; Table 3). Therefore, sculpin density was detectably lower in the impact stream during and after construction and in streams without culverts in 2014 and 2015.

Coho densities showed a decrease in the impact stream in 2013 and 2015 and across all streams in 2014 and 2015. The top-fitted model for coho density had treatment × year, year, treatment, and stream type effect terms but not the stream type × year term

Table 2. Fixed-effects terms present in top-ranked models for fish total density, species richness, diversity, and density for the three major taxa in eight streams based on AIC_c (Akaike's information criterion for small sample size) ranking.

Metric	Fixed-effects terms from best model, based on ranking by lowest AIC_c score	AIC_c weight
Total fish density (fish$^{1/2}$·m^{-1})	Treatment × year	0.84
	Stream type	
	Treatment	
	Year	
Species richness (no. of species)	Stream type × year	1
	Treatment × year	
	Stream type	
	Treatment	
	Year	
Shannon diversity index	Stream type × year	0.86
	Treatment × year	
	Stream type	
	Treatment	
	Year	
Trout density (fish$^{1/2}$·m^{-1})	Stream type × year	0.74
	Stream type	
	Treatment	
	Year	
Sculpin density (fish$^{1/2}$·m^{-1})	Stream type × year	1
	Treatment × year	
	Stream type	
	Treatment	
	Year	
Coho density (fish$^{1/2}$·m^{-1})	Treatment × year	0.95
	Stream type	
	Treatment	
	Year	

Note: The top model for each metric was chosen by comparing the AIC_c scores for the full model and all possible models generated by dropping terms, and an intercept-only model. The top model was the model with the lowest AIC_c score after ranking. The fixed effects in the full model were stream type × year, treatment × year, stream type, treatment, and year.

(Table 2). The treatment (I) × year terms for 2013 and 2015 and the year terms for 2014 and 2015 were negative and did not span zero (Fig. 5; Table 3). This indicates that coho densities were lower in the impact stream during and after construction and across all streams in the latter 2 years of the study. Mean density of coho at the impact stream fell from 0.13 fish·m^{-2} in 2012 to 0.008 fish·m^{-2} in 2013, just 6% of the previous year's value (Fig. 6). In addition, densities never recovered to the levels observed in 2012 (0.014 fish·m^{-2} in 2014 and 0.042 fish·m^{-2} in 2015).

Densities of the three most abundant fish taxa (trout, coho, and sculpin) had different dynamics in the stream where the rapid transit rail line was built (Fig. 6). Trout densities dropped in 2014 after major construction for section F (upstream of the project) and then rose in 2015. Sculpin densities fell slightly in 2013 across several reaches and then increased in 2014. We sampled zero sculpin in section F before the project, but sculpin were present in 2014, the year after major construction. In 2013, coho densities dropped across all sections they were present in, whether they were in the project footprint or not. They slowly rose in the next 3 years but did not reach levels as high as those in 2012 within the study period. Therefore, changes in fish density showed patterns that were specific to different taxa in the impacted stream over the study period.

Discussion

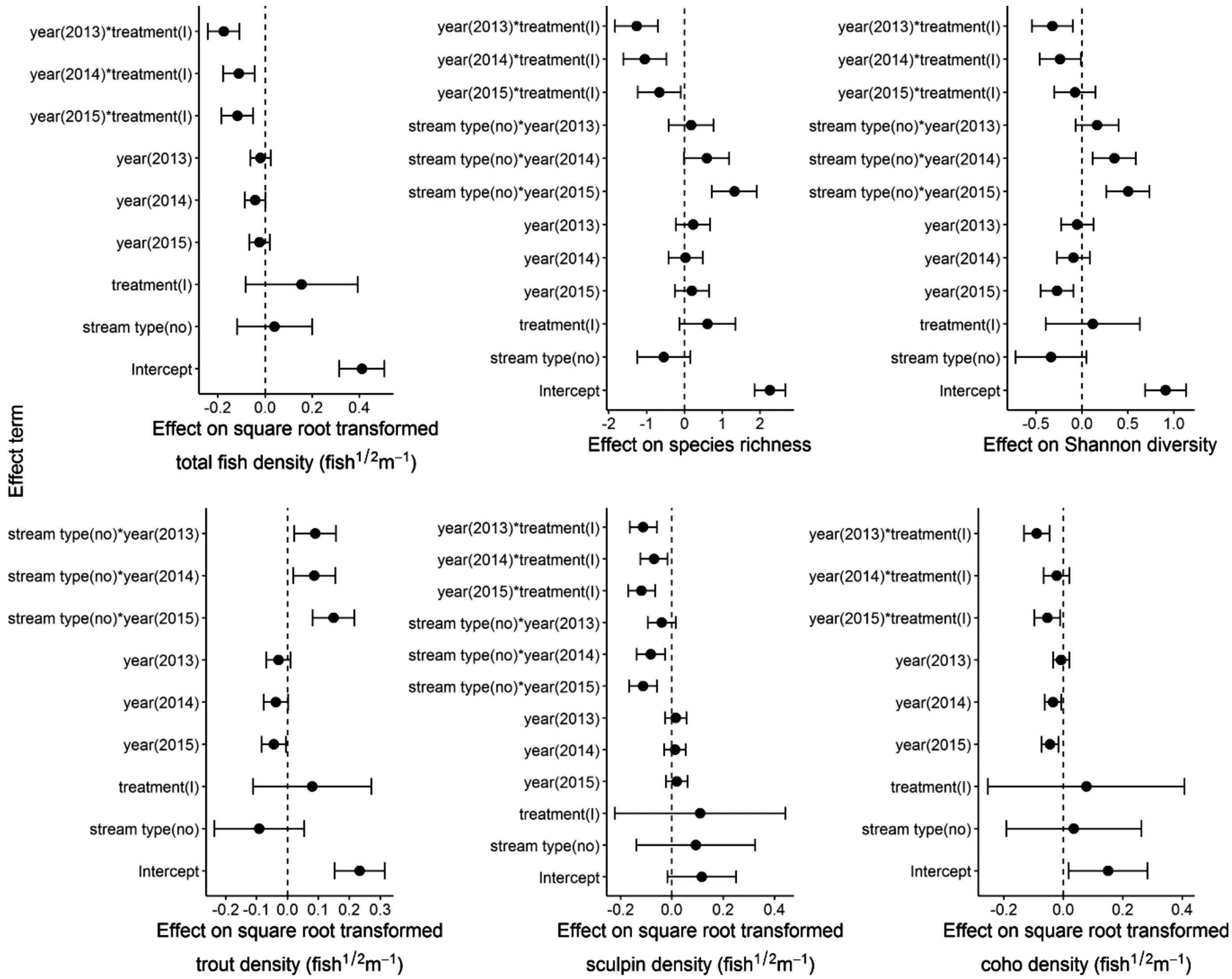
This study provides rare evidence of the impacts of a specific project on fish communities in urban streams while accounting for the dynamic effects of existing fragmentation. Total density, species richness, diversity, and sculpin and coho densities were all affected, meaning that the aspects of fish communities that have been shown to change after decades of urbanization can also change during and immediately after specific projects (Weaver and Garman 1994; Wang et al. 2000). The impacts were observed at the scale of the entire stream, evidence that alteration in one place is detectable throughout the system.

Our results add to the existing evidence that culverts can alter fish movement and community metrics but that these effects vary with time and water conditions. Compared among years, the effect of culverts was greatest in 2015. This was also the year with the lowest precipitation over the study period. In 2015, total rainfall in the lower mainland of BC for May–August was 103.8 mm, compared with 150.1 mm, 167.6 mm, and 141.2 mm in 2012, 2013, and 2014, respectively (Environment Canada 2016). This led to low flows and warm water temperatures in 2015, which prompted the government to restrict water use and close recreational fisheries. This result suggests that the effect of culverts may be magnified in dry years. These results build on our previous findings that the cumulative number of culverts downstream of fish communities had a negative effect on diversity and a positive effect on coho salmon density, but only in the first year of a 2-year study (Favaro and Moore 2015). Higher flows can enable fish passage in otherwise perched culverts as hydrology changes over time (Norman et al. 2009). Water temperature changes with stream flow and can also modify the effect of culverts. Using passive integrated transponders, Goerig et al. (2016) found that the rate of successful passage of wild brook trout (*Salvelinus fontinalis*) through culverts increased with water temperature, with an optimum of 14–15 °C, but decreased above 15 °C. On a watershed scale, upstream migration of brook trout and cutthroat trout was significantly lower through culverts than through unaltered stream reaches during summer low flows (Burford et al. 2009). Collectively, these previous studies and our results illustrate that the impacts of culverts are modulated by water temperature and flow. Given that climate change is predicted to decrease summer stream flows and increase temperatures in many regions (Mantua et al. 2010), the impacts of culverts may become more severe over time.

The use of multiple reference streams across the region enabled the study to detect decreases in diversity, trout density, and coho density in 2015 and coho density in 2014. These results show the inherent variability of fish communities in small streams, which is well documented in long-term studies. For example, over 17 years, the density of brown trout (*Salmo trutta*) recruits in a 170 m long spawning channel showed greater than 15-fold variation among years (Lobón-Cerviá and Mortensen 2005). Variation in annual rainfall explained 73% of this variation in recruitment, emphasizing the importance of this regional driver. Lobón-Cerviá (2011) found that the recruitment of brown trout over 20 years was highest at intermediate rainfall levels, regardless of the presence or absence of angling. In addition to modifying the effect of culverts, regional patterns such as low rainfall could be contributing to the trends in diversity, trout density, and coho density we observed.

The study design enabled us to disentangle regional changes from the effects of one project, including the large decrease in the density of coho in the impact stream. Rainfall can have a large influence on fish density, as previously discussed, but changes to water quality from human activity can also change fish density via changes to distribution and direct mortality. While the sensitivity of stream fish communities to the impacts of urbanization will likely vary across different regions and with different fish communities — in some regions, urbanization may actually facilitate tolerant fish species (Moore and Olden 2017) — the common fish taxa in our study systems are somewhat sensitive to degraded water quality. Steelhead trout (*Oncorhynchus mykiss*) and coho salmon grew more slowly in channels with artificially elevated turbidity than in clear water and were more likely to leave the

Fig. 5. The 95% confidence intervals for the fixed-effects terms in the top-rated models for predicting total density (square-root-transformed), species richness, diversity, and trout, sculpin, and coho densities (square-root-transformed). Points indicate the estimates for effects.



turbid channels (Sigler et al. 1984). Mortality of Arctic grayling (*Thymallus arcticus*) alevins was significantly higher in a stream with increased turbidity from upstream mining than in a stream without mining after only 24 h (Reynolds et al. 1989). Introduction of suspended sediment and elevated turbidity has also been shown to displace Arctic grayling downstream (McLeay et al. 1987). Contractors did in-stream work (replacing a culvert) and cleared vegetation at the impact stream of Suter Brook before we sampled in 2013, and we observed water that appeared to be unusually turbid there during sampling in that year (personal observation). Elevated turbidity from construction could have contributed to the decrease in coho density in 2013 at Suter Brook (Fig. 3). There was also a dramatic decrease in coho densities across all streams during the study period: mean coho densities across all study streams fell from 0.12 fish·m^{-2} in 2012 to 0.044, 0.045, and 0.055 fish·m^{-2} in 2013, 2014, and 2015, respectively. This decrease may have been due to low numbers of adult returns in 2012, since the juvenile coho in Suter Brook in 2013 would have been sired by spawners that returned the previous year. Low adult salmon returns have been linked to oceanic and climatic conditions on a regional scale (McKinnell et al. 2014). Suter Brook drains into Burrard Inlet along with the Seymour River, where estimates of returning coho fell from 5500 adults in 2011 to 3000 adults in 2012 (Seymour Salmonid Society 2013). Thus, regional patterns of adult coho returns likely contributed to the observed decrease in coho density across all streams, but the robust study design was able to detect a magnitude of change in the impact stream that surpassed the regional changes occurring at the same time.

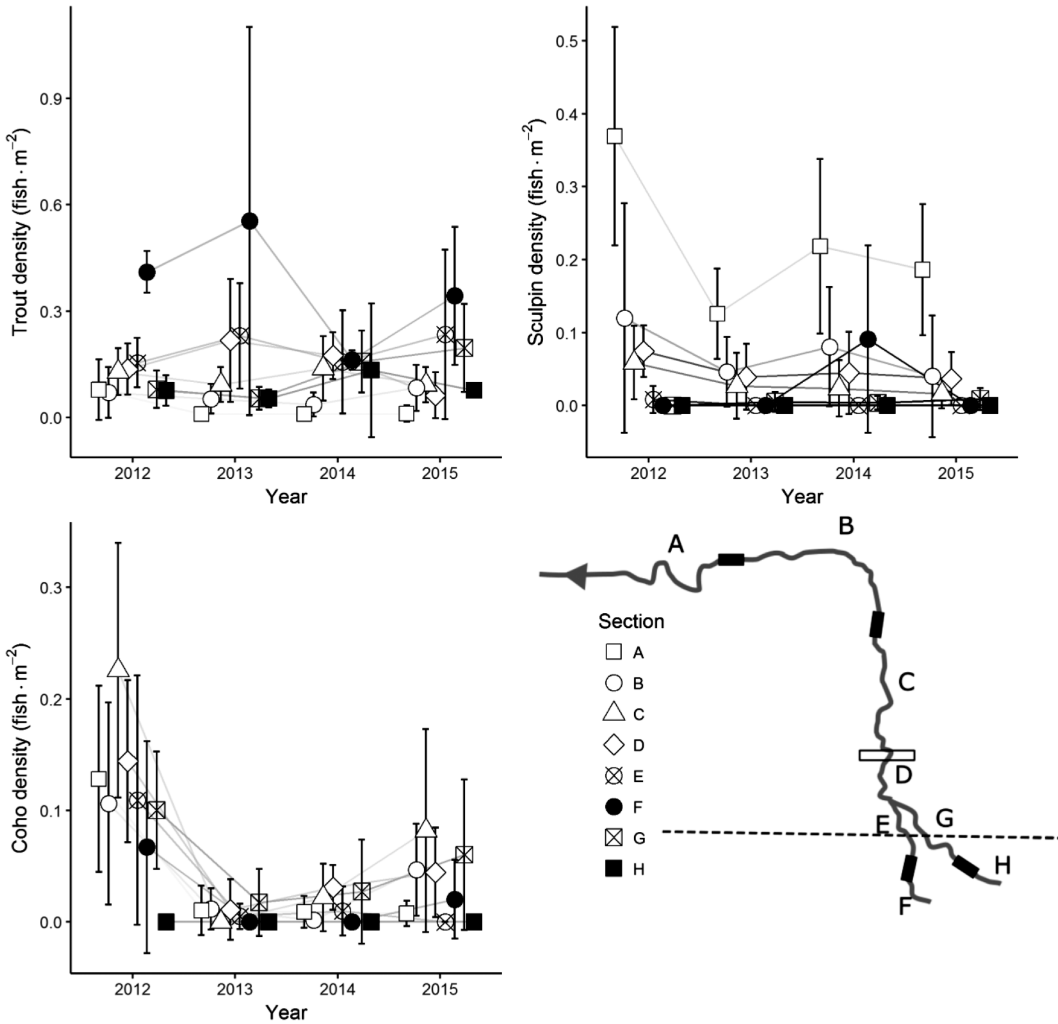
Trout and sculpin showed markedly different trends at Suter Brook. In all but one section, trout had the least variable densities across years. This was supported by a lack of any detectable effect on trout from the project. Arango et al. (2015) showed that species densities in trout-dominated streams were not significantly different when compared before and 11 months after a stream restoration project. It is possible that these resident species are able to quickly recolonize disturbed areas after restoration and reach densities equivalent to those before disturbance, as shown in our study. Sculpin densities in section F showed an opposite trend and increased in 2014. The environmental assessment for the rapid transit rail line reported that the culvert downstream of section F was impassable for fish, and this culvert was replaced during construction (table 6.3 in Hatch Ltd. 2010). Sculpin densities can be significantly lower upstream of culverts than downstream of culverts, indicating that culverts pose hydraulic barriers to these species even when culverts are not damaged (Favaro and Moore 2015). It is possible that replacement of the culvert downstream of

Table 3. The 95% confidence intervals and estimates for fixed-effects terms from top models for total density, species richness, diversity, and density for the three major taxa in eight streams.

		95% confidence interval values		
Metric	Effects term	Lower limit	Estimate	Upper limit
Total density (fish$^{1/2}$·m^{-1})	**Intercept**	**0.31**	**0.41**	**0.51**
	Year (2013)	−0.06	−0.02	0.02
	Year (2014)	−0.09	−0.04	0.00
	Year (2015)	−0.07	−0.02	0.02
	Treatment (I)	−0.08	0.16	0.39
	Stream type (no)	−0.12	0.04	0.20
	Year (2013) × treatment (I)	**−0.24**	**−0.18**	**−0.11**
	Year (2014) × treatment (I)	**−0.18**	**−0.11**	**−0.04**
	Year (2015) × treatment (I)	**−0.18**	**−0.12**	**−0.05**
Species richness (no. of species)	**Intercept**	**1.86**	**2.27**	**2.67**
	Stream type (no)	−1.25	−0.54	0.16
	Year (2013)	−0.22	0.23	0.68
	Year (2014)	−0.42	0.03	0.49
	Year (2015)	−0.25	0.20	0.65
	Treatment (I)	−0.13	0.61	1.35
	Stream type (no) × year (2013)	−0.42	0.18	0.77
	Stream type (no) × year (2014)	−0.01	0.59	1.19
	Stream type (no) × year (2015)	**0.73**	**1.32**	**1.92**
	Year (2013) × treatment (I)	**−1.83**	**−1.26**	**−0.69**
	Year (2014) × treatment (I)	**−1.62**	**−1.05**	**−0.47**
	Year (2015) × treatment (I)	**−1.23**	**−0.66**	**−0.09**
Shannon diversity index	**Intercept**	**0.69**	**0.91**	**1.13**
	Stream type (no)	−0.72	−0.34	0.05
	Year (2013)	−0.23	−0.05	0.12
	Year (2014)	−0.27	−0.09	0.08
	Year (2015)	**−0.45**	**−0.27**	**−0.09**
	Treatment (I)	−0.40	0.12	0.63
	Stream type (no) × year (2013)	−0.07	0.17	0.40
	Stream type (no) × year (2014)	**0.12**	**0.35**	**0.59**
	Stream type (no) × year (2015)	**0.27**	**0.50**	**0.74**
	Year (2013) × treatment (I)	**−0.55**	**−0.32**	**−0.10**
	Year (2014) × treatment (I)	**−0.46**	**−0.24**	**−0.01**
	Year (2015) × treatment (I)	−0.30	−0.08	0.15
Trout density (fish$^{1/2}$·m^{-1})	**Intercept**	**0.15**	**0.23**	**0.31**
	Stream type (no)	−0.24	−0.09	0.05
	Year (2013)	−0.07	−0.03	0.01
	Year (2014)	−0.08	−0.04	0.00
	Year (2015)	**−0.08**	**−0.04**	**−0.01**
	Treatment (I)	−0.11	0.08	0.27
	Stream type (no) × year (2013)	**0.02**	**0.09**	**0.16**
	Stream type (no) × year (2014)	**0.02**	**0.09**	**0.16**
	Stream type (no) × year (2015)	**0.08**	**0.15**	**0.22**
Sculpin density (fish$^{1/2}$·m^{-1})	Intercept	−0.02	0.12	0.25
	Stream type (no)	−0.14	0.09	0.33
	Year (2013)	−0.03	0.02	0.06
	Year (2014)	−0.03	0.01	0.05
	Year (2015)	−0.02	0.02	0.06
	Treatment (I)	−0.22	0.11	0.44
	Stream type (no) × year (2013)	−0.09	−0.04	0.02
	Stream type (no) × year (2014)	**−0.14**	**−0.08**	**−0.03**
	Stream type (no) × year (2015)	**−0.17**	**−0.11**	**−0.06**
	Year (2013) × treatment (I)	**−0.16**	**−0.11**	**−0.06**
	Year (2014) × treatment (I)	**−0.12**	**−0.07**	**−0.02**
	Year (2015) × treatment (I)	**−0.17**	**−0.12**	**−0.07**
Coho density (fish$^{1/2}$·m^{-1})	**Intercept**	**0.02**	**0.15**	**0.28**
	Year (2013)	−0.04	−0.01	0.02
	Year (2014)	**−0.06**	**−0.03**	**−0.01**
	Year (2015)	**−0.07**	**−0.04**	**−0.02**
	Treatment (I)	−0.25	0.08	0.41
	Stream type (no)	−0.19	0.04	0.26
	Year (2013) × treatment (I)	**−0.13**	**−0.09**	**−0.05**
	Year (2014) × treatment (I)	−0.07	−0.02	0.02
	Year (2015) × treatment (I)	**−0.10**	**−0.05**	**−0.01**

Note: In the model, base terms are year (2012), stream type (culvert) meaning culverts present, and treatment (C) meaning control stream with no rapid transit rail line construction. The effects term stream type (no) means no culverts present, and treatment (I) means that rail line construction was present. Effects terms whose 95% confidence intervals do not span zero are in bold.

Fig. 6. Mean density of the three most abundant taxa of fishes (trout, sculpin, and coho, untransformed) at the focal impact site of Suter Brook over 4 years, divided into sections as in Fig. 2 (inset). Sections A, B, C, and D were downstream of the construction zone where the channel was reconstructed and are represented by open symbols. Sections E and G spanned the project footprint and are represented by crossed symbols. Sections F and H were upstream of the impact and culverts and are represented by solid symbols. Symbols are mean values, and error bars represent standard deviation. No work was done before the 2012 sampling year. Some preliminary in-stream work was conducted between 2012 and 2013. Major in-stream works and channel realignment were conducted between 2013 and 2014. After sampling in 2014, contractors did planting in the riparian area. Sampling in 2015 occurred after the project was completed. A small amount of horizontal jitter has been added to aid interpretation. Inset sketch shows sections of Suter Brook (letters) with culverts (black rectangles), rapid transit rail line (dashed line), clear-span bridge (empty rectangle), and direction of flow (arrow). Section breaks are culverts, bridge, and confluence. Drawing is not to scale.



section F allowed sculpin to swim upstream and occupy this section after the construction was complete. If this is the case, new infrastructure projects have the potential to undo the deterioration of older infrastructure if they are designed to improve the connectivity and quality of stream habitat. On the other hand, sculpin were not present in section F in 2015. This may be a short-term trend and not evidence of a long-term improvement in connectivity.

This study had several assumptions that are worth examining. Single-pass electrofishing without block nets is an imperfect index for fish abundance compared with multiple-pass depletion surveys. However, single-pass surveys have a much smaller impact on aquatic ecosystems, and catchability is generally high in small, clear streams such as those in this study (Bohlin et al. 1989). In addition, projects with limited resources for monitoring can make use of this technique when budgets or time limit their scope. Covariation between stream characteristics or fish communities and human impacts is also possible; project footprints result from trajectories of urban planning and development, not random experimental assignment. Using seven reference sites over 4 years and using repeated measures of discrete reaches minimizes the chance of drawing conclusions from covariation.

In Canada, the proponents of projects in and around streams are rarely required to conduct monitoring in as much detail as in this study. Under the Canadian *Fisheries Act*, proponents do not need to conduct long-term monitoring unless required to do so by the conditions of an authorization to cause serious harm to fish. Many projects are exempt from authorizations — and thus from monitoring — based on the type of waterbody they affect, the type of project, or a review of project plans by Fisheries and Oceans Canada prior to construction (Fisheries and Oceans Canada 2016). When monitoring does happen, it is often noncompliant with requirements, lacking an experimental design, and qualitative rather than quantitative (Harper and Quigley 2005). This dearth of robust monitoring has two consequences. First, there is little opportunity for biologists, engineers, contractors, and regulators to learn from projects and develop better approaches for development and monitoring. Second, without evaluation, accountability, and enforcement, the integrity of fish communities will continue to erode. Only by using robust methods similar to an asymmetric before–after, control–impact design (Aguado-Giménez et al. 2012; Mateos-Molina et al. 2014) were we able to quantify specific impacts of the project on top of existing variation across time and the impacts of past projects.

Federal, provincial, regional, and municipal governments could facilitate the robust assessment of projects through ongoing, coordinated monitoring of a suite of reference streams. Meaningful reference systems are essential for detecting and quantifying the effects of human activities on ecosystems (Likens 1985; Underwood 1994). Using multiple metrics such as fish density, species richness, and diversity indices in addition to water quality and hydrological data will allow project impacts to be differentiated from regional patterns. Municipalities in Metro Vancouver are currently required to monitor benthic invertebrates, hydrology, and 13 water quality parameters once every 5 years (Metro Vancouver 2014), but this approach has limited power to detect change in these naturally variable systems. More frequent monitoring that includes fishes would provide better data for reference sites. Furthermore, cities and regional districts can use bylaws, zoning, the acquisition of private land, official community plans, development permit guidelines, and other means to protect their healthiest streams and ensure reference sites exist in the future.

The Metro Vancouver region of Canada, like many urban areas around the world, is growing rapidly (Statistics Canada 2015). This region contains a relatively intact native fish community, yet we found here that a single infrastructure project can detectably decrease fish diversity. We suggest that our study, which used repeated sampling and multiple reference streams, has more power to detect biological changes than those typically employed during project evaluation and monitoring. While it is widely appreciated that urbanization is associated with shifts in fish communities, here we provide insight into how a specific development was linked to loss of diversity in the native fish community. Planners, regulators, and citizens need robust accounting such as this to weigh potential trade-offs between urban development and fishes.

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