Thornton Creek Restoration: Baseline Project Effectiveness Monitoring, 2012-2013

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Technical report by

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Executive Summary

Seattle Public Utilities (SPU) is preparing to construct two urban floodplain reconnection projects in the Thornton Creek watershed of northeast Seattle in 2014. To evaluate project performance, the utility contracted with the U.S. Fish and Wildlife Service (USFWS) to collect pre-project physical and biological baseline data from 2005 to 2009. In 2012, the National Oceanographic and Atmospheric Administration (NOAA) was brought in by SPU to collect additional biological baseline data and to develop a post-project monitoring plan that includes evaluation of new hyporheic design elements.

Data collection by NOAA has been identical to previous USFWS monitoring for periphyton, benthic invertebrates, and fish density. Sampling in 2012 differed from earlier years by focusing less on physical habitat, expanding upon fish-movement surveys, and adding collection of invertebrate drift and fish diet samples. This report summarizes data collected by NOAA during 2012-2013 and provides updates on existing pre-project monitoring results collected by USFWS from 2006 to 2009.

The overall biological health of Thornton Creek is poor: scores for the benthic index of biotic integrity (B-IBI) range from poor to very poor and diatom assemblages are composed of a relatively high proportion of species tolerant of nutrient enrichment. In the lower watershed, the invasive New Zealand Mud Snail *Potamopyrgus antipodarum* dominated benthic assemblages. Both drift and diet samples were comprised largely of aquatic insects and crustaceans, with smaller contributions from the riparian zone.

The fish community of Thornton Creek is dominated by cutthroat trout *Oncorhynchus clarki*, at relatively high densities compared to other Puget Sound lowland streams. Resident trout in Thornton Creek exhibited high rates of site fidelity: of the 1,051 fish tagged by NOAA, 75% remained in the watershed and 88% were detected within 100 m of their tagging reach.

Future biological monitoring should include evaluation of hyporheic invertebrate and microbial assemblages. These samples can be collected by pumping water from piezometers installed within project treatment and control reaches. Hyporheic sampling should also occur at forested reference streams for regional data context. There is still a short window to collect hyporheic data prior to project construction in 2014.

While SPU's approach to floodplain reconnection is innovative and holds promise for restoring watershed processes at the reach level, it is also largely untested. We strongly recommend that a minimum of 3-5 years of post-construction surface and hyporheic data be collected in order to adequately evaluate project effectiveness, and to help select and guide the design of future floodplain projects across the City of Seattle.

Contents

Introduction

The Thornton Creek watershed is both the largest and most urbanized watershed within the City of Seattle (City of Seattle 2007). Over half of the watershed is covered by impervious surfaces, and virtually all historic habitats have been heavily modified (TCWMC 2000). Loss of native vegetative cover across the watershed has severely altered the quantity, timing, and quality of storm water delivered to the creek; these alterations have negatively impacted water quality, physical habitat, and the biological health of aquatic communities (TCWMC 2000; City of Seattle 2007). Less than 10% of the riparian ecosystem in Thornton Creek is considered to be in “good” condition (City of Seattle 2007).

The benthic index of biotic integrity (B-IBI) is a tool designed to measure the health of stream ecosystems based on relative abundance and diversity of aquatic invertebrates. Throughout the Thornton Creek watershed, IBI scores are consistently very poor (City of Seattle 2007). Of all the urban watersheds in Seattle, Thornton Creek also has the highest rate of coho salmon *O. kisutch* pre-spawn mortality (adult coho that die with near-total retention of eggs or milt), and very little refuge and spawning habitat for resident fish (City of Seattle 2007).

Seattle Public Utilities (SPU) has committed significant resources to address these problems, including construction of conventional stormwater infrastructure (detention facilities and retrofits to abate combined sewer overflows). In addition, SPU has built green stormwater infrastructure (natural drainage systems), and instream restoration projects intended to improve fish passage and structural habitat (TCWMC 2008; Morley et al. 2010).

Recognizing the need to take a more holistic approach, SPU will soon break ground on two floodplain reconnection projects on Thornton Creek: one at the confluence of North and South Fork Thornton Creek (*Thornton Confluence* project), and the other on the south fork of Thornton Creek just upstream from Lake City Way Northeast (*Knickerbocker Reach* project). Both projects are approximately 2-3 acres in size, and both will incorporate hyporheic design features to achieve project goals.

The hyporheic zone is a layer of fluvial sediments beneath and adjacent to the active channel where surface and groundwater mix (Orghidan 1959; Bencala 2005). This transition area is critical in flood dampening and groundwater recharge; water temperature regulation; and biogeochemical cycling of nutrients, organic matter, and contaminants (Hancock et al. 2005; Boulton 2007; Robertson & Wood 2010). Goals and design features of both the Thornton Confluence and Knickerbocker Reach projects are to:

1) Promote vertical connectivity between surface and subsurface habitats with the creation of a hyporheic zone below and adjacent to the active channel
2) Increase horizontal connectivity (lateral movement) of the stream within its floodplain by removing bank armoring and compacted floodplain fill
3) Maximize floodplain storage of water, wood, and sediment via increased horizontal and vertical connectivity
4) Increase reach-scale habitat complexity with placement of large wood, boulder, and channel structures that promote hyporheic flow paths
5) Improve water temperature regulation by increasing hyporheic exchange and replanting the floodplain with native vegetation
6) Decrease surface water contaminant levels via increased chemical cycling by microbial and invertebrate assemblages in the hyporheic zone
7) Improve the biological health of stream organisms such as microbes, algae, invertebrates, and fish

While this approach is innovative and holds promise for restoring natural floodplain processes to the extent possible in urban environments, it is also largely untested. To help evaluate effectiveness of the Thornton Confluence and Knickerbocker Reach floodplain reconnection projects, SPU has engaged science partners from various disciplines. Pre-project monitoring for both projects was conducted by the U.S. Fish and Wildlife Service (USFWS) from 2005-2009, with technical assistance by NOAA Fisheries (Leavy et al. 2010). In 2012, NOAA took over the biological monitoring component of the project, focusing on fish (density, movement, and diet), invertebrates (benthic and drift assemblages), periphyton, and coarse physical habitat metrics. Overall study design and sampling protocols employed by NOAA were consistent with prior USFWS data collection in Thornton Creek.

The USFWS remains the lead on physical and hydrologic monitoring, and SPU has recently partnered with a third collaborator from the National Environmental Management Academy to assist with development of a chemical monitoring plan. This report summarizes data collected by NOAA during 2012-2013 and provides updates on existing pre-project monitoring results collected by USFWS from 2006 to 2009. Appendix A of this report outlines a recommended post-project monitoring design, with particular emphasis on the addition of biological monitoring in the hyporheic zone.

Methodology

Study Design

Thornton Creek drains a 28.8 km^2 watershed in the northeast corner of the City of Seattle and southeast portion of the City of Shoreline, emptying into Lake Washington on its western shore at Matthews Beach. Thornton Creek flows approximately 24 km in a southeasterly direction through three major sections: the North Fork, the South Fork, and the mainstem. The geology of the watershed is highly variable consisting of both consolidated and unconsolidated sand and gravel, as well as dense layers of sand, gravel, and silt (TCWMC 2000; City of Seattle 2007). Thornton Creek watershed receives nearly 89 cm of precipitation annually, which falls primarily as rain between October and May resulting in estimated bankful discharges that range from 65-190 ft^3/second (TCWMC 2000). Single family residences make up 49% of development within the watershed, although residential roads, road right-of-ways, and commercial development constitute 53, 26, and 8% of the total land use respectively (City of Seattle 2007).

Both floodplain reconnection projects are located in Northeast Seattle south of NE 110th Street (Figure 1). The Knickerbocker project is on the South Fork of Thornton Creek, 0.25 km west of Lake City Way, south of NE 100th Street and between 19th and 21st Avenue NE. The Thornton Confluence project straddles 35th Avenue NE between Nathan Hale High School and Meadowbrook Pond, and will encompass both the north and south forks of Thornton Creek and the most upstream section of the mainstem (Figure 1).

The overall study design established by USFWS is a modified BACI (before-after, control-impact) approach, which consists of four paired treatment (i.e., impact) and control reaches, each 50-75 m long (Figure 2). Treatment reaches include the Knickerbocker project area on the South Fork, both the North and South forks immediately above their confluence, and the mainstem above Matthews Beach. The Matthews Beach reaches are no longer considered treatment reaches, but were included in baseline data collection for data continuity. Control reaches are all located < 350 m upstream of their corresponding treatment reach (except at Matthews Beach, where the control reach is 50 m downstream).

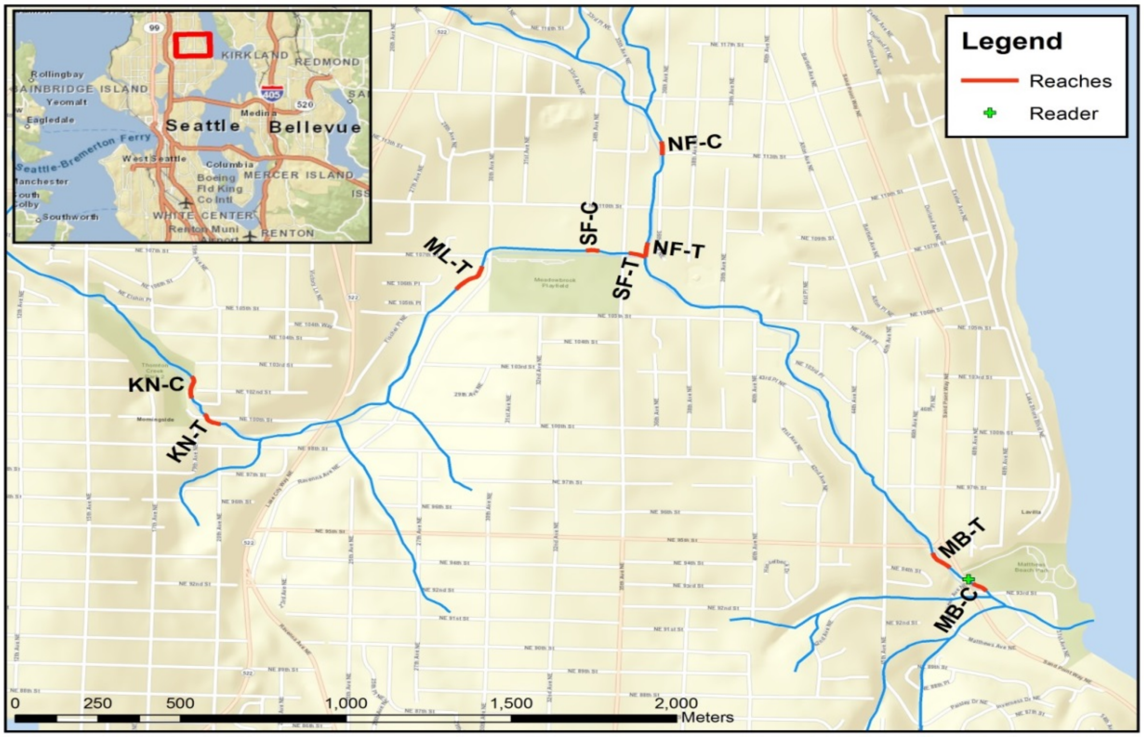

Figure 1. Thornton Creek study area and reaches surveyed in 2012: Knickerbocker control and treatment (KN-C and KN-T), south fork confluence control and treatment (SF-C and SF-T), north fork confluence control and treatment (NF-C and NF-T), and Matthews Beach control and treatment (MB-C and MB-T). Reader refers to an instream tag-monitoring system.

Reach characteristics varied across the study sites and between control and treatment reaches (Table 1). The North and South Fork treatment reaches (NF-T and SF-T, respectively) both begin immediately upstream from the confluence of the two major forks of Thornton Creek near Meadowbrook Pond. Both treatment reaches are highly channelized and armored, overgrown with invasive riparian species such as Himalayan blackberry *Rubus armeniacus*, and characterized by simplified physical habitat (Leavy et al. 2010). The North Fork control reach (NF-C) is located between 113th and 115th Street NE, 330 m upstream from its paired treatment reach (NF-T). NF-C is bordered by private residences, dominated by a mature forest canopy of Western red cedar *Thuja plicata*, and contains a large pool at its upstream end. The South Fork control reach (SF-C) is located 90 m upstream of its treatment reach, between Meadowbrook playfield and Nathan Hale High School. Similar to the treatment reach, SF-C is highly channelized and characterized by simplified physical habitat and invasive riparian species.

Further up the south fork of Thornton Creek, the Knickerbocker treatment reach (KN-T) is armored and channelized, with vegetation dominated by Himalayan blackberry and giant knotweed *Polygonum sachalinense* (Leavy et al. 2010). The control reach (KN-C), located 80-m upstream of the treatment, flows through the Kingfisher Natural Area owned by Seattle Department of Parks and Recreation. KN-C is covered by a mature forest canopy including bigleaf maple *Acer macrophyllum* and red alder *Alnus rubra*, and contains deep pools formed by large woody debris (Leavy et al. 2010).

The Matthews Beach control reach (MB-C) is located immediately downstream from Sand Point Way and 350 m upstream from the confluence of Thornton Creek with Lake Washington. This reach is bordered by Matthews Beach Park and a King County wastewater pump station. The Matthews Beach treatment reach (MB-T) is 100 m upstream of the control. Both reaches are well forested and have mature canopies of native vegetation including western red cedar and black cottonwood *Populus tricocarpa*, although greater habitat complexity is found in the treatment reach (Leavy et al. 2010).

Table 1. Summary habitat characteristics by reach. Dominant vegetation is taken from Leavy et al. 2010.

Reach	Length (m)	Width (m)	Armored (%)	Pool vol. (%)	Dominant vegetation
Knickerbocker					
control	71.8	3.9	12	14	Red alder, bigleaf maple
treatment	71.8	2.2	100	18	Himalayan blackberry, giant knotweed
South Fork					
control	36.9	1.7	0	0	Himalayan blackberry, giant knotweed
treatment	41.8	1.8	100	16	Himalayan blackberry
North Fork					
control	49.6	3.5	0	34	Western red cedar, red alder, bigleaf maple
treatment	38.4	1.9	64	0	Himalayan blackberry
Matthews Beach					
control	55.3	4.8	13	54	Western red cedar, black cottonwood, bigleaf maple
treatment	49.3	3.6	0	57	Western red cedar, black cottonwood

Sampling Parameters

All data collection in 2012 occurred during late fall between 18 September and 2 October. Mobile surveys of fish movement occurred in early fall, winter, and spring, while instream tag-monitoring systems were operated during the entire study period. Fish were sampled using a Smith Root backpack electrofisher and a standard, triple-pass depletion method (Carle and Strub 1978; Murphy and Willis 1996). Block nets were placed at the lower and upper ends of each habitat unit to prevent movement of fish. All fish captured were anesthetized by being placed into a diluted solution (~50 mg/L) of tricaine methanesulfonate before being identified to species, weighed, measured, and tagged with passive integrated transponder (PIT) tags (if ≥ 55 mm in length). After recovery from anesthesia, fish were released back into the habitat units from which they were removed. Water temperatures in fish holding tanks were continuously monitored to minimize stress during holding and prior to release. These data were used to calculate taxonomic composition and total numeric and biomass density for each study reach.

During electrofishing surveys at each reach, up to 10 individual cutthroat trout *Oncorhynchus clarki* were also sampled for stomach contents via gastric lavage. This process involves flushing the stomach of an anesthetized fish with a clean syringe of stream water and collecting regurgitated materials on a 250 µm-mesh sieve. Sample materials were fixed in 80% ethanol and sent to a professional lab for taxonomic analyses. All invertebrates were enumerated and identified to the lowest practical taxonomic level (ranging from species to order based on digested state). For each study reach, we computed summary metrics such as mean taxa richness, Shannon Wiener diversity index (H'), proportion of taxa originating from the aquatic zone (i.e., including winged-adult forms of aquatic larvae), and proportion of terrestrial input (both of terrestrial and aquatic origin). To account for differences in stomach fullness, we excluded from further analysis any stomachs that contained fewer than four invertebrates.

An instream PIT-tag monitoring system was installed at the upper end of the Matthews Beach control reach to continuously monitor the movement of tagged fish into and out of the Thornton Creek watershed (Figure 2). Detection data were uploaded daily by an automated modem during the study period. To evaluate movement of individuals within the watershed, mobile surveys for PIT-tagged fish were conducted on three separate occasions in fall (18 October-6 November 2012), winter (8-11 January 2013), and spring (18-19 March 2013). In October, the mobile survey consisted of scanning the creek from the mouth at Lake Washington to the confluence with Victory Creek at Northeast 108th Street on the south fork and up to the Northeast 115th Street culverts on the north fork of Thornton Creek. Mobile surveys in January and March were terminated 100 m upstream of the Knickerbocker and North Fork Control reaches based on very low numbers of detections above these reaches.

For each individual fish detection, a unique tag code was recorded, along with a GPS position and time of detection. These data were used to calculate direction and distance of movement from original tagging location. Mobile scan detection efficiencies were evaluated prior to whole-basin surveys by conducting local mobile scans of reaches in early fall shortly after fish had been tagged and released and while the reaches were still block-netted. Detection efficiencies were calculated as the ratio of tagged fish detected to total number of tagged fish present in a reach.

Figure 2. Antennas from an instream PIT-tag monitoring system installed in Thornton Creek. Picture courtesy of Gabriel Brooks.

Drift invertebrates were collected at each sample site using a 250-μm-mesh, 0.14 m^2 frame drift net. Prior to benthic sampling, a single drift net was placed at the upstream end of each reach perpendicular to flow and secured above the stream bottom with re-bar. Drift samples were collected over a timed, 30- to 50-minute period while benthic and physical habitat data were collected from mid-morning to mid-afternoon. Water velocity and depth were measured on each side of the drift frame so as to calculate total sample volume over the timed interval. All material accumulated in the drift net was placed into a 500 μm-mesh sieve, where large pieces of organic matter were rinsed and removed before preserving the sample in an 80% ethanol solution for taxonomic analysis at level-3 Pacific Northwest standards (typically species).

Benthic invertebrates were collected exclusively from riffle habitat units within each reach using a Slack sampler net (500-μm mesh, 0.17-m^2 frame). Nine Slack samples were collected in each reach and composited into a single sample to ensure adequate sample size. At each sample location, the net was placed perpendicular to the flow, and a 0.25 m^2 quadrat was placed in front of the net. Large substrate within the quadrat were removed and scraped into the net to remove all invertebrates, and the area within the quadrat was disturbed with a small garden rake for 60 seconds to displace remaining invertebrates. Sample contents were transferred from the net to sample bottles using a 500-μm-mesh soil sieve. All samples were preserved in ethanol and sent to Rhithron Associates for taxonomic analysis of a 500-count sub-sample. All invertebrates were enumerated and identified to level-3 Pacific Northwest standards and entered into the Puget Sound Stream Benthos online database (PSSB 2013) for calculation of total density, B-IBI (benthic index of biotic integrity), and associated metrics.

Periphyton was sampled from stream cobbles randomly selected from riffles adjacent to each of the nine benthic invertebrate sample locations. Periphyton was scraped from each of the nine cobbles using a toothbrush and squirt bottle and composited into one sample per reach. A portion of each sample was preserved in Lugol's solution and sent to Rhithron Associates Inc. for analysis of diatom taxonomic composition. These species counts were then used to calculate a suite of metrics that describe taxonomic and functional diversity, as well as disturbance tolerance. The remaining periphyton sample was filtered onto two 47-mm glass-fiber filters for analyses of chlorophyll *a* concentration and ash-free dry mass (AFDM). Chlorophyll *a* specifically measures the algal component of periphyton, whereas AFDM is a measure of total periphyton biomass that includes algae, fungi, bacteria, and microzoans (Steinman and Lamberti 1996).

We extracted chlorophyll *a* from filters with acetone and measured absorbance of the resulting supernatant using fluorometry (Marker et al. 1980). AFDM was calculated following the gravimetric method (Steinman and Lamberti 1996). Chlorophyll *a* concentration and AFDM weights were converted to biomass per unit area (mg/cm^2 for AFDM, and μg/cm^2 for chlorophyll *a*) based on total rock surface area sampled at each site (Dall 1979). We examined patterns in periphyton data at the site, reach, and regional scale across multiple sample years. At the site scale, we compared diatom metrics within the Thornton Creek basin across all treatment reaches to determine if there was a consistent year-to-year ordering of sites from most to least-impacted. At the reach scale, we examined differences between reference and treatment reaches for each study pair. For regional context, we compared periphyton metrics at Thornton Creek to those collected at Pipers Creek for a related project and to those collected at other urban and forested streams in the Puget Sound region.

Habitat typing (e.g., pools, riffles, and glides) followed a modification of the classifications of Bisson et al. (1982). For each habitat unit, total length, average wetted width, average depth (riffles and glides), and maximum and pool tail-out depth (pools) were recorded using a hand-held tape measure, stadia rod, and laser range finder. Average wetted width was calculated by taking measurements at the upstream, midpoint, and downstream ends of each unit. Average depth was estimated by taking 10 randomly spaced measurements from within each riffle or glide habitat unit. Average flow measurements were calculated by measuring velocity and depth at 10 evenly spaced increments perpendicular to the flow at the downstream end of each reach. All flow measurements are reported in cubic meters per second (CMS). Substrate composition and bank armoring was visually estimated for each habitat unit within a given reach (recorded as percent silt, sand, gravel, cobble, boulder, and percent rip rap/concrete, respectively).

Statistical Analyses

We made comparisons between sites, between reaches within a site, and between Thornton Creek and other regional streams using a variety of graphical, univariate, and multivariate statistical methods. Given the pre-project nature of the data, we primarily present results in tabular and graphical format for interpretative purposes and include limited statistical analyses at this stage.

Fish population estimates were derived from three-pass electrofishing estimates using the RDeplete package based on Zippin (1956), Carle and Strub (1978), and Olsen et al. (1996) using the statistical software program R (version 2.15.2, R Development Core Team, Vienna, Austria). From these abundance estimates, we calculated fish density by number of individuals and biomass for each species and quantified habitat unit within the eight study reaches, comparing means across sample years. We summarized size-structure data graphically with histograms by study reach. Fish-movement data are displayed spatially on the stream network.

We examined differences in diatom, invertebrate, and fish diet assemblage structure using a suite of complementary multivariate techniques available in the statistical software package PRIMER (version 6.1.13, Clarke and Gorley 2006). We square-root-transformed our data (densities for invertebrates, relative abundance for fish diet), created triangular resemblance matrices of pair-wise similarities between all sites using the Bray-Curtis distance, and tested for differences between sites using ANOSIM (analysis of similarities), a non-parametric analog to analysis of variance (Clarke and Warrick 1994). Next, we used the “similarity percentages” routine SIMPER to decompose average Bray-Curtis dissimilarities between all pairs to determine which taxa contributed the most to similarities and dissimilarities between study reaches.

Results and Discussion

Fish Density and Size Distribution

Cutthroat trout were numerically dominant at all study reaches except MB-C, where sculpins *Cottus* spp. were abundant near the creek outlet with Lake Washington (Table 2, Figure 3). This finding was consistent with previous observations from USFWS 2005-2009 surveys (Leavy et al. 2010). We also observed the highest fish species richness at MB-C, where coho salmon, lamprey *Lampetra* spp., coastrange sculpin *Cottus aleuticus*, and threespine stickleback *Gasterosteus aculeatus* were captured along with cutthroat trout during 2012 electrofish surveys (Figure 3). Stickleback were also captured at MB-T, and speckled dace *Rhinichthys osculus* at SF-C. The remaining five study reaches contained only cutthroat trout (Table 2, Figure 3). This observation differed from surveys of 2005-2009, where juvenile coho were observed in highest numbers at NF-C (Leavy et al. 2010).

Table 2. Fish capture and population estimates by reach and species.

Reach		Species	Captured		Population estimate		Estimated population biomass (g)
			Total (N)	Biomass (g)	N	SE	
Knickerbocker							
	control	cutthroat	202	2,004	212	5.6	2,090
	treatment	cutthroat	161	1,802	177	12.4	1,995
South Fork							
	control	cutthroat	157	1,360	190	24.0	1,578
		speckled dace	3	75	3	1.1	75
	treatment	cutthroat	83	537	84	2.2	542
North Fork							
	control	cutthroat	228	944	278	35.5	1,127
	treatment	cutthroat	104	554	116	11.0	614
Matthews Beach							
	control	cutthroat	101	1,170	102	2.2	1,191
		coho	2	16	2	0.0	16
		lamprey	4	21	4	1.1	21
		sculpin	120	339	135	10.6	368
		stickleback	3	3	3	0.7	3
	treatment	cutthroat	113	1,501	122	6.0	1,613
		stickleback	13	11	22	13.4	18

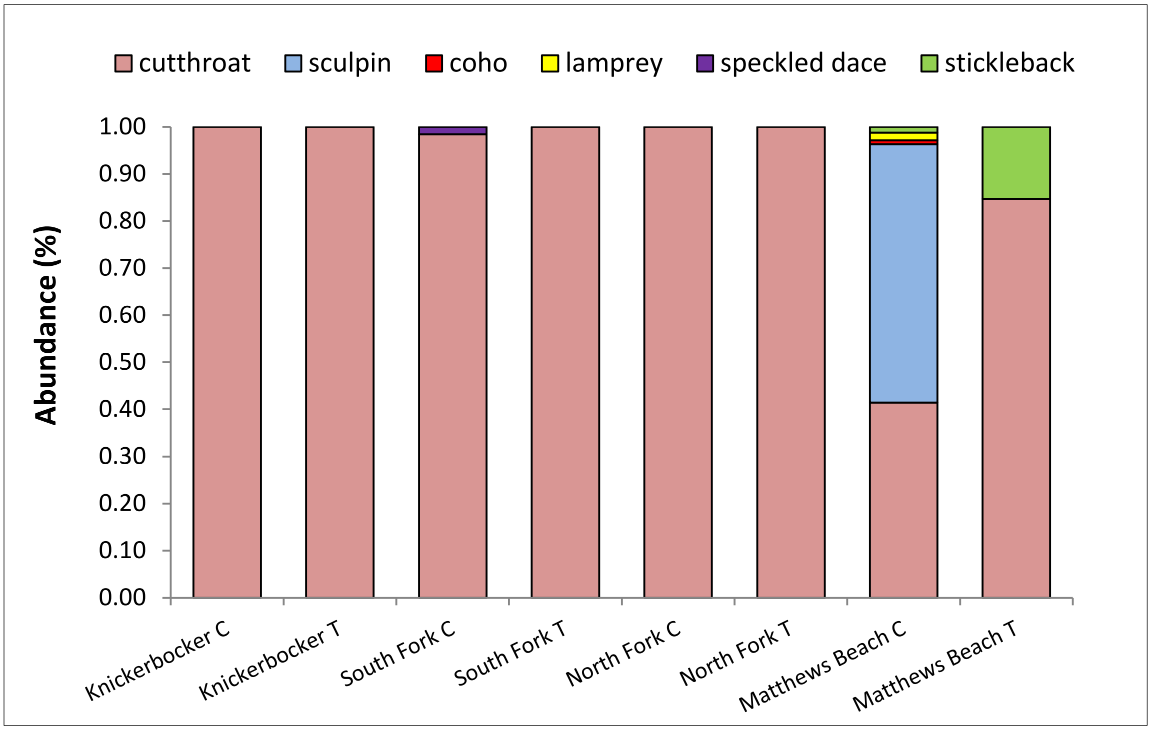

Figure 3. Species composition by reach from 2012 electrofishing surveys.

Cutthroat trout densities ranged from 0.4 to 3.0 fish/m^2 across survey reaches (Figure 4), higher than mean densities observed across 2005-2009 surveys (0.14-1.98 fish/m^2), and higher than values reported in neighboring Pipers Creek (0-1.18 fish/m^2; Morley et al. 2010). Previous studies of Seattle urban stream fish populations also reported higher densities of cutthroat trout in Thornton Creek compared to Pipers (Tabor et al. 2010). Based on stream surface area, we observed the highest densities of cutthroat trout in SF-C, where densities were over twice those observed in SF-T. Cutthroat trout densities in the North Fork treatment and control reaches were very similar to each other, and intermediate in value to those of the South Fork reaches.

Decreasing cutthroat densities (as well as overall fish density) were observed in the Knickerbocker and Matthews Beach reaches, both of which had slightly higher fish densities in treatment vs. control reaches. Earlier studies by USFWS also found the lowest cutthroat densities in Matthews Beach reaches, with mean values at all other reaches between 1.1-2.0 fish/m^2 (Leavy et al. 2010). Fish density patterns may partially reflect differences in channel form: for example, SF-C is a very narrow and incised channel. On a volumetric basis (fish/m^3), we observed the highest fish densities at North Fork reaches (Figure 4).

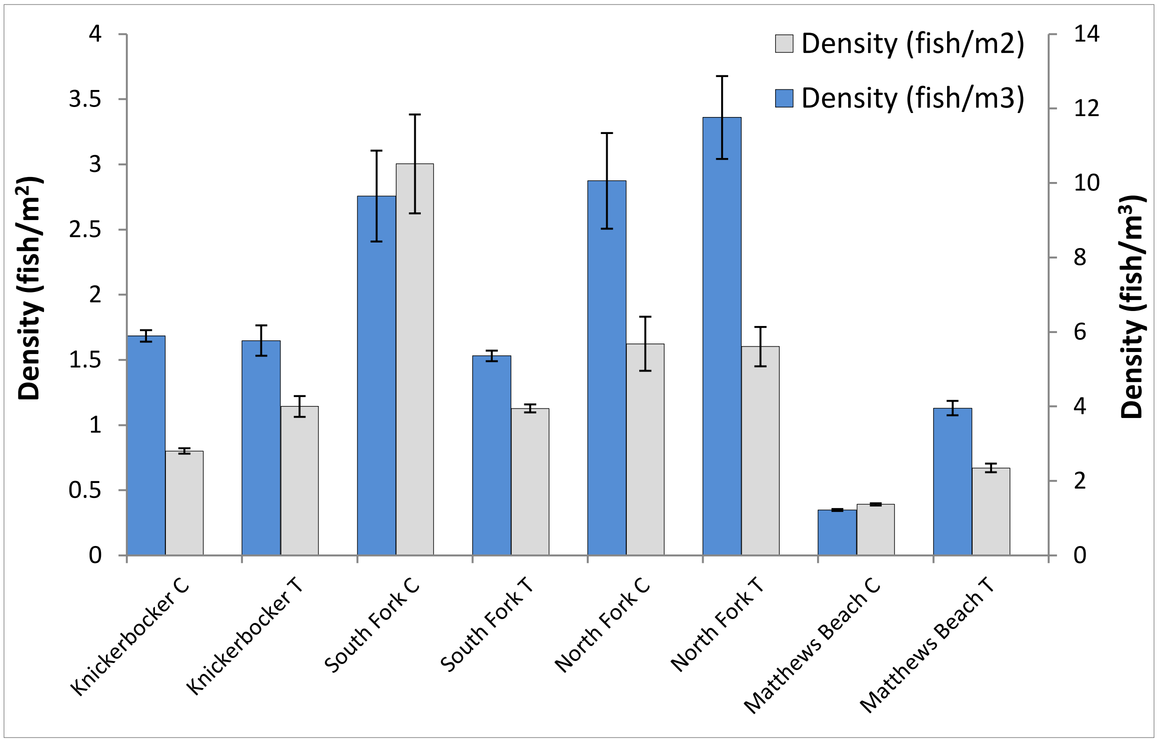

Figure 4. Mean and standard error of cutthroat trout density by reach from 2012 electrofish surveys, calculated both by water surface area and volume.

We measured a total of 1,142 cutthroat trout across the eight Thornton Creek study reaches, and these fish ranged in size from 39-234 mm (Table 3, Figure 5). This size range is similar to the 31-221 mm reported by Leavy et al. (2010), and likely reflects several age classes present in all study reaches. The majority of cutthroat in Thornton Creek ranged in size from 65 to 105 mm compared to 80-100 mm for cutthroat trout in Pipers Creek (Morley et al. 2010; Figure 6).

The largest mean cutthroat length of 103 mm was observed in MB-T, while smallest (69 mm) was observed in NF-C (Figure 5). We found the widest distribution of size classes at the Knickerbocker site, similar to Leavy et al. (2010). Of the 1,142 cutthroat captured during 2012 electrofishing surveys, 3 had been previously tagged. Of these, one matched records from USFWS surveys: a 62 mm fish PIT-tagged on 16 July 2009 at Knickerbocker treatment was recaptured on 24 September 2012 at 184 mm in the same study reach.

Table 3. Cutthroat trout fork length minimum, maximum, and mean values with standard deviation and sample size per study reach from 2012 electrofish surveys.

	Fork length (mm)				
Reach	Min	Max	Mean	SD	N
Knickerbocker					
control	48	205	89.69	28.74	202
treatment	60	191	95.01	28.75	158
South Fork					
control	53	234	85.24	27.25	156
treatment	49	200	76.12	25.59	82
North Fork					
control	39	146	69.15	19.99	227
treatment	44	161	74.76	21.16	103
Matthews Beach					
control	54	220	95.36	31.14	101
treatment	61	192	103.48	29.76	113

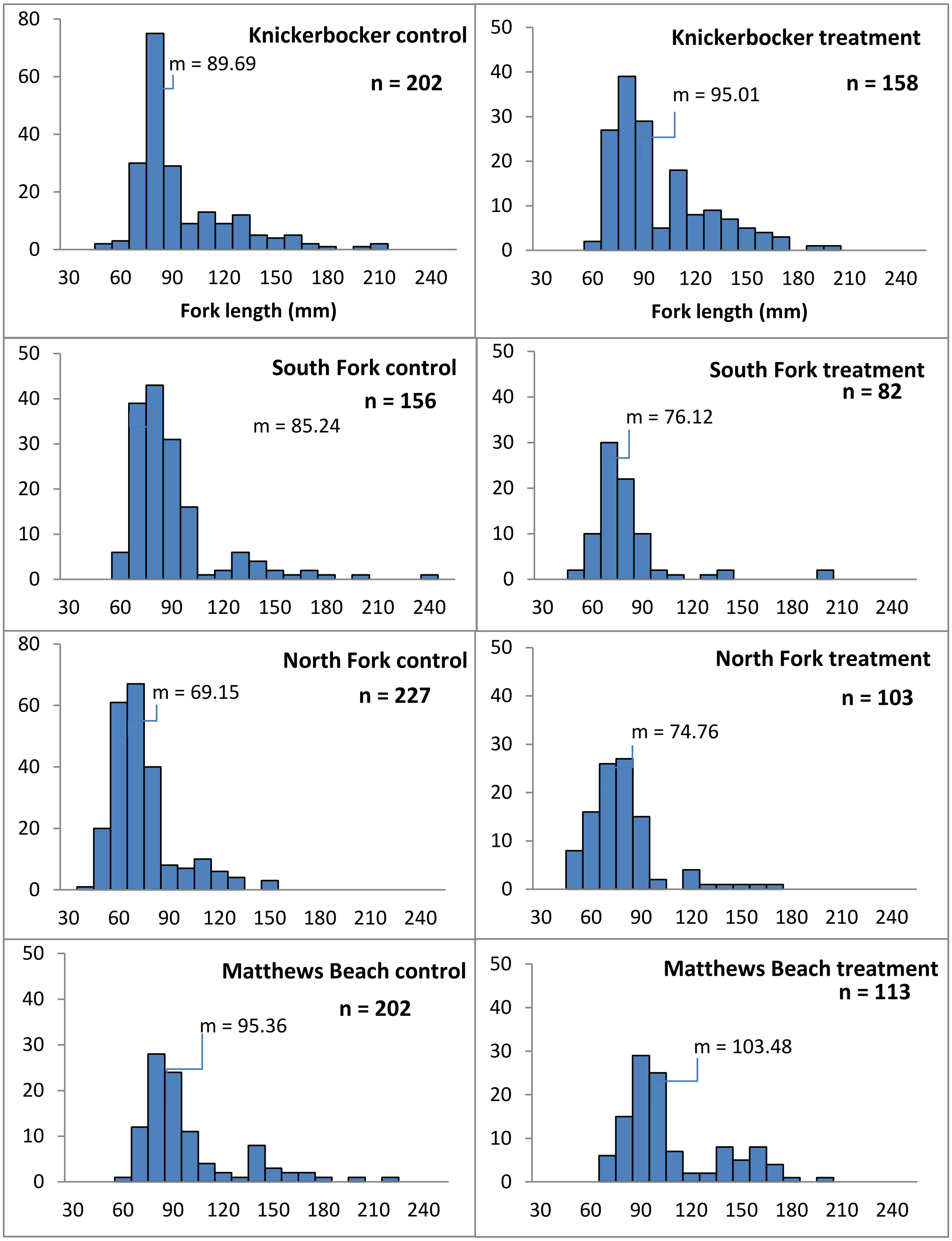

Figure 5. Histograms showing distribution of cutthroat trout fork length and mean (m) by reach from 2012 electrofish surveys.

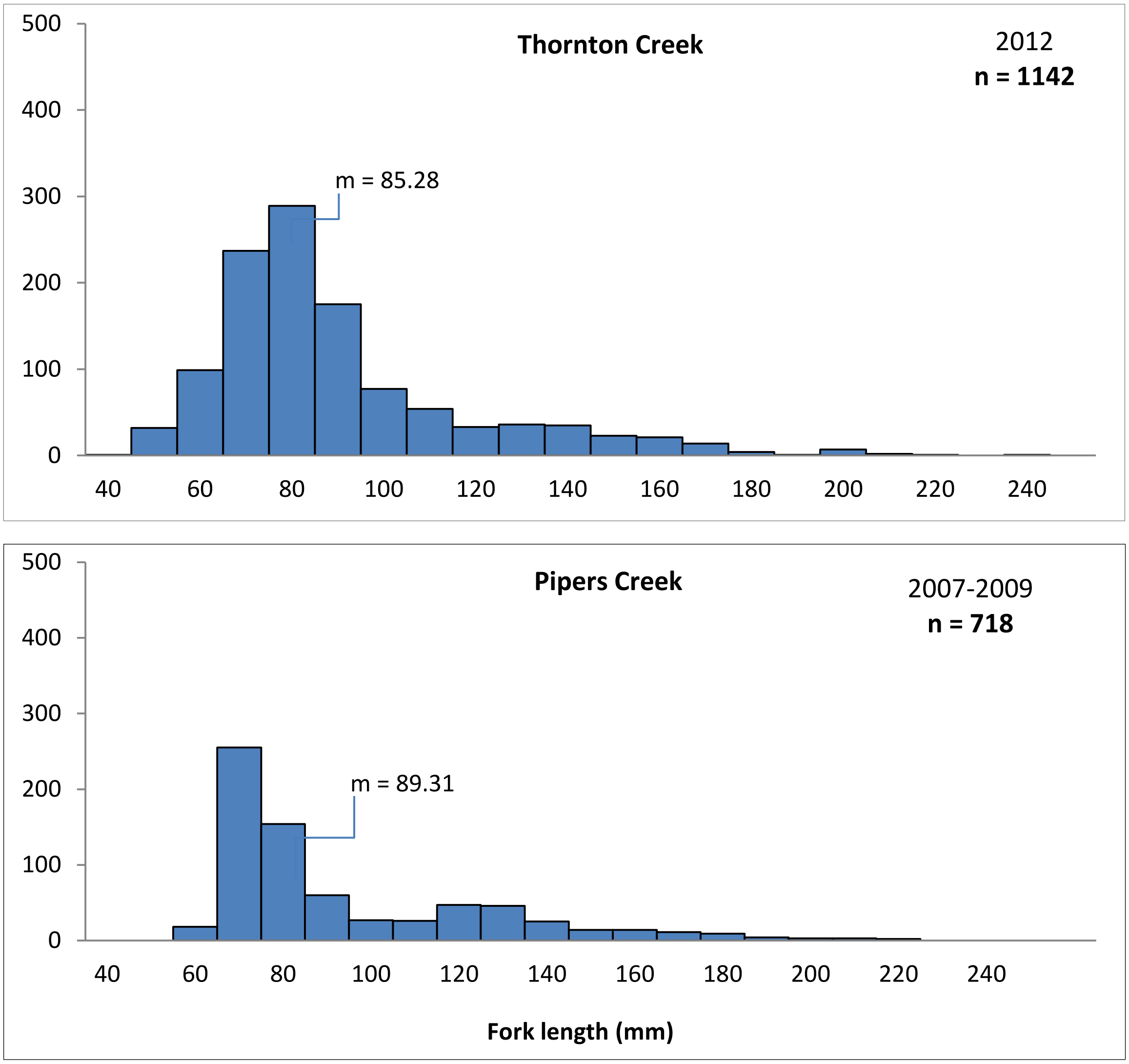

Figure 6. Distribution of cutthroat trout fork lengths with means (m) across all Thornton Creek survey reaches in 2012 (upper panel) and across all Pipers Creek survey reaches from 2007-2009 (lower panel).

Fish Movement

A total of 4,796 fish were PIT tagged and released into Thornton Creek between 2007 and 2011. These releases were primarily cutthroat trout, but also include nine coho and one rainbow trout *O. mykiss*. Across the eight study reaches, USFWS tagged 2,672 fish from 2007-2011 and NOAA tagged 1,051 fish in 2012 (Table 4). Between 2007 and 2010, an additional 1,073 fish were tagged by either USFWS or SPU in Thornton Creek at locations outside of study reach boundaries (Table 5).

Table 4. Number of PIT-tagged fish by release year (2007-2012) in the eight Thornton Creek floodplain reconnection study reaches.

Reach	2007	2008	2009	2010	2011	2012	**Total**
Knickerbocker							
control	193	23	122		31	193	**562**
treatment	207	15	84			155	**461**
South Fork							
control	113	72	89	70		153	**497**
treatment	73	103	64	50		73	**363**
North Fork							
control	131	132	138	86	63	174	**724**
treatment	86	130	100	57		88	**461**
Matthews Beach							
control	32		48			102	**182**
treatment	157		203			113	**473**
Total	**992**	**475**	**848**	**263**	**94**	**1,051**	**3,723**

Between October 2007 and April 2013, the instream PIT-tag monitoring system in Thornton Creek detected 1,284 unique PIT tag codes, including 27% of all tagged fish released in Thornton Creek from 2007 to 2012. For release years 2007-2011, fish were tagged between mid-July and mid-August, but in 2012, tagging was conducted in late September. Despite differences in time of tagging, peak detections of tagged fish moving downstream typically occurred between mid-October and early November, coinciding with fall high-flow events (Figure 7).

Of the 1,284 unique PIT detections on fixed instream monitors between October 2007 and April 2013, 66% were fish detected moving downstream over both antennas. Another 8% were detected on only one of the two antennas, precluding determination of directionality. However, 90% of these single detections were from tags released upstream from the monitoring system, with release date lags similar to fish moving downstream over both antennas. Thus, these single detections likely represented downstream movements of tagged fish that were missed by one of the two antennas.

Table 5. Number of PIT-tagged fish by release year (2007-2010) at locations on Thornton Creek outside of floodplain reconnection study reaches, with lead tagging agency.

Location/release site	Agency	2007	2008	2009	2010	**Total**
Knickerbocker						
above control	USFWS			37		**37**
above treatment	USFWS		13			**13**
below treatment	USFWS		126			**126**
South Fork						
above control	USFWS		113		175	**288**
below control	USFWS		73			**73**
North Fork						
above control	USFWS			35		**35**
below control	USFWS		8			**8**
below treatment	USFWS			48		**48**
Matthews Beach						
above treatment	USFWS			16		**16**
Maple Leaf	SPU	329				**329**
Sewer Repair	SPU		100			**100**
Total		**329**	**433**	**136**	**175**	**1,073**

Combining these two datasets resulted in an overall downstream detection rate of 20% for all fish tagged in Thornton Creek from 2007-2012 (Table 6). This downstream detection rate was considerably lower than that observed during a similar study in Pipers Creek, where 61% of fish were detected moving downstream over an instream PIT-tag monitoring system located below the confluence of Venema and Pipers Creeks (Morley et al. 2010). This may reflect differences in life history strategies related to freshwater carrying capacity, varying rates of in-stream survival, or detection efficiencies during high flow events. Alternatively, it may be related to placement of instream monitoring system, as the Pipers Creeks system was located further upstream from the creek outlet (610 m) than the Thornton system (395 m).

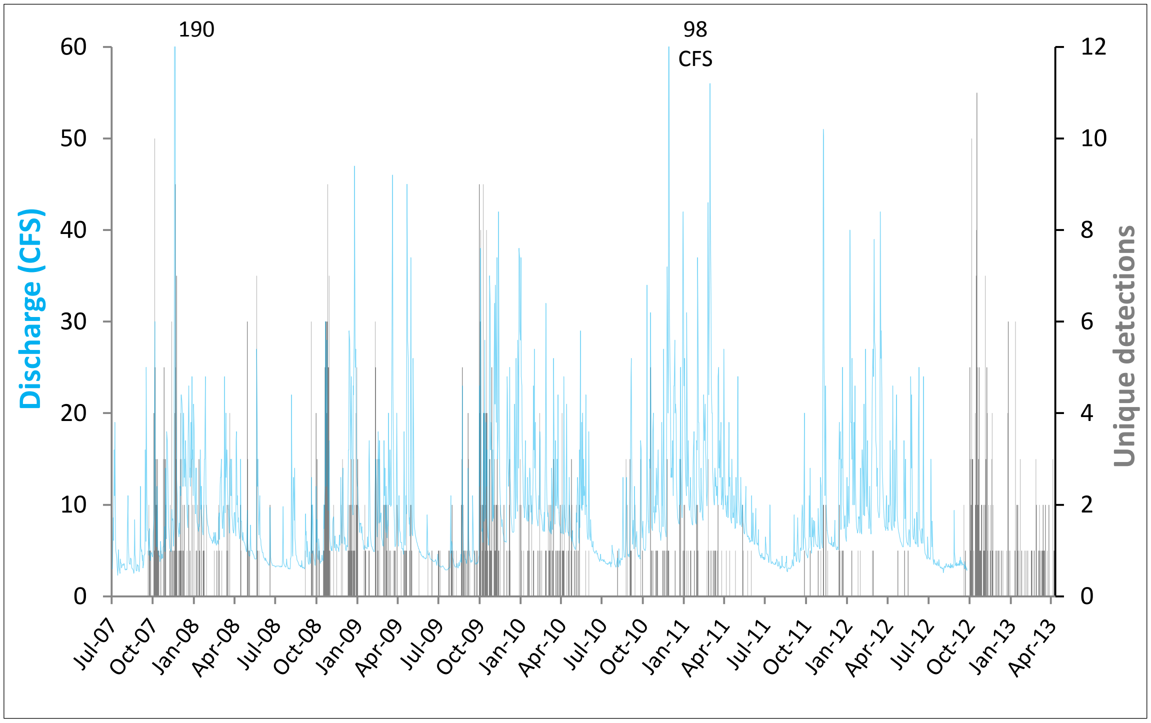

Figure 7. Unique downstream tag detections from 2007-2012 at the Thornton Creek instream PIT-tag monitoring system compared to mean daily discharge (USGS Station 12128000).

Table 6. Date of first PIT-tagged fish release by study year, peak date of unique downstream PIT tag detections at in-stream monitoring system, proportion of total tagged fish detected, and date and magnitude of annual peak flow from USGS station 12128000.

Study year	Release start date	Detection peak date	Proportion detected (%)	Peak flow	
				Event date	Flow (ft^3/s)
2007	23 Jul 2007	19 Oct 2007	14.9	3 Dec 2007	190
2008	17 Jul 2008	9 Nov 2008	19.1	8 Aug 2009	47
2009	13 Jul 2009	14 Oct 2009	19.2	26 Nov 2009	42
2010	26 Jul 2010	1 Nov 2010	17.4	12 Dec 2010	98
2011	16 Aug 2011	22 Nov 2011	19.1	22 Nov 2011	51
2012	24 Sep 2012	31 Oct 2012	14.5	NA	NA

The remaining 25% of unique detections at the instream monitoring system involved multiple detections on both antennas spread over multiple days. A majority (84%) of these multiple detections were from fish tagged and released at Matthews Beach reaches, which are located adjacent to and immediately upstream from system antennas. These multiple detections are likely primarily the result of small-scale movements of fish within the study reaches where they were tagged. Three fish detected on the instream monitoring system showed movement patterns indicating potential juvenile migration and adult return. All three of these were cutthroat trout tagged at 61–71 mm, detected moving downstream within 100 d of release, and subsequently detected moving upstream 3-4 years later.

These movement patterns suggests possible anadromous or adfluvial life history strategies for a minority of Thornton Creek cutthroat trout, although it is possible that these fish simply moved into habitat downstream from the monitoring system and remained in Thornton Creek, given that the system is located 350 m upstream from the mouth of Thornton Creek. No PIT-tagged fish released to Thornton Creek were detected at the Ballard Locks between 2010 and 2012, although one tagged sea-run coastal cutthroat trout from Thornton Creek was documented moving into Puget Sound in 2008 (Leavy et al. 2010).

Mobile scan surveys conducted within and downstream from all study reaches in October 2012 (Figure 8a), January 2013 (Figure 8b), and March 2013 (Figure 8c) detected a total of 632 PIT tags across all release years (2007-20012). For all release years, 49% of detections were classified as shed tags and 51% were classified as live fish detections (Table 7). Detections were classified as shed tags if a tag was detected repeatedly in the same spot during a survey event despite disruption of the substrate and water column. Live fish detections accounted for only 10% of total detections from release years 2007-2011, but 73% for release year 2012 (Table 7).

These numbers should be interpreted in the context of mobile-scan efficiency. Mobile scan detections efficiencies, determined immediately after early fall tagging events while block nets were still in place, ranged from 25-88%, with a mean efficiency of 37% (Table 8). Of detections from mobile scanning, a majority (55%) was single detections (i.e., encountered once during the three whole-basin mobile scan surveys). Shed tag detections accounted for 63% of the repeat PIT detections.

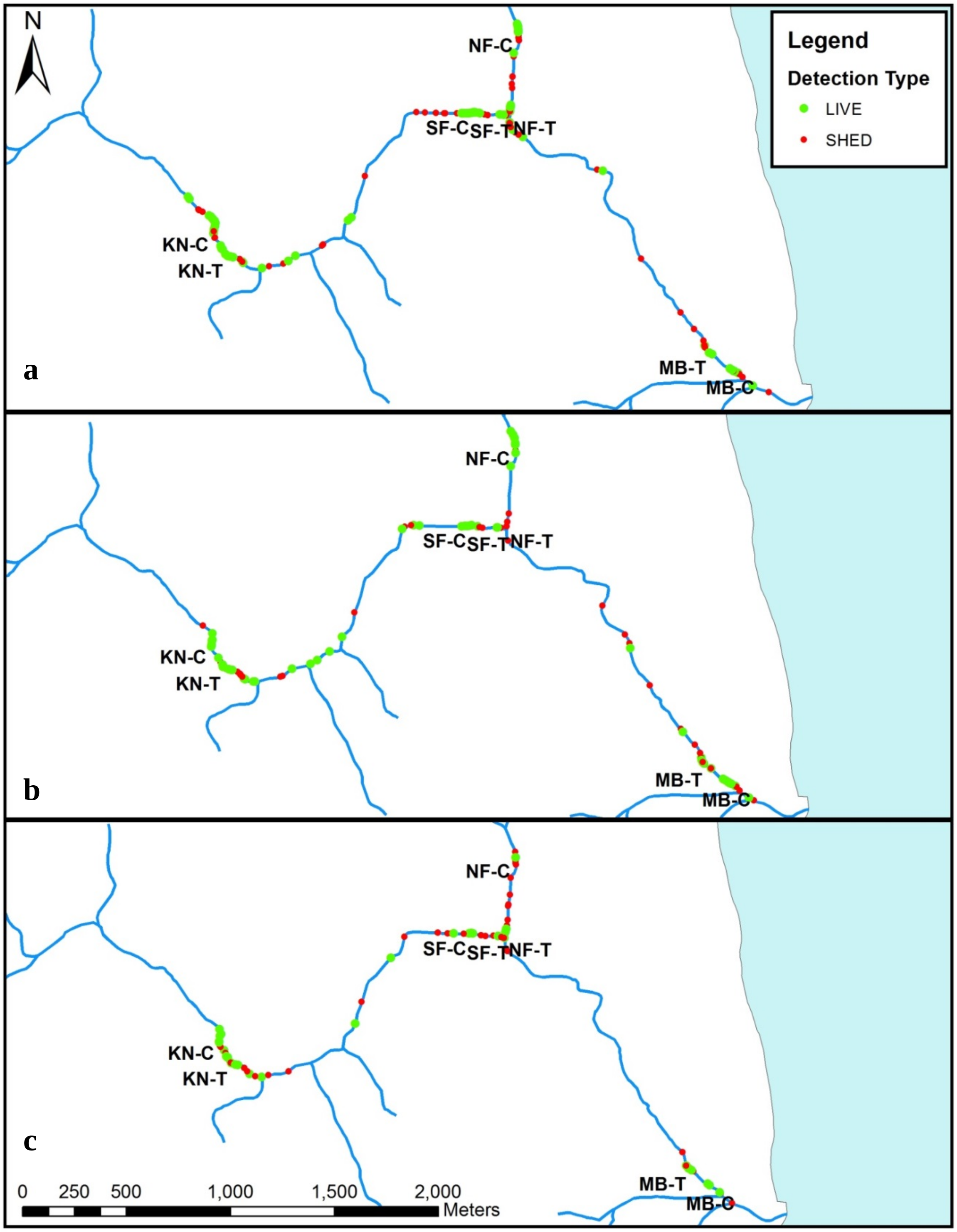

Figure 8. Distribution of live and shed PIT-tag detections from all release years during (a) October 2012, (b) January 2013, and (c) March 2013 mobile scan surveys.

Table 7. Summary of PIT-tag detections by tagging year during three mobile scan surveys conducted in Thornton Creek from 2012-2013.

Detection type	Mobile scan	2007	2008	2009	2010	2011	2012	Total
		Tagging year						
		Total (N)						
Total	Oct 2012	25	23	21	21	14	198	305
	Jan 2013	23	15	18	6	4	105	171
	Mar 2013	19	15	22	9	7	84	156
Shed tags	Oct 2012	19	21	19	19	12	36	125
	Jan 2013	21	13	16	4	4	29	83
	Mar 2013	18	15	21	8	7	39	103
Live fish	Oct 2012	6	2	2	2	2	162	180
	Jan 2013	2	2	2	2	0	76	88
	Mar 2013	1	0	1	1	0	45	53
		Proportion (%)						
Shed tags	Oct 2012	76	91	90	90	86	18	41
	Jan 2013	91	87	89	67	100	28	49
	Mar 2013	95	100	95	89	100	46	66
Live fish	Oct 2012	24	9	10	10	14	82	59
	Jan 2013	9	13	11	33	0	72	51
	Mar 2013	5	0	5	11	0	54	34

Table 8. Detection efficiency of mobile PIT-tag surveys calculated as the ratio of detected tagged fish to total number of tagged fish present in a given study reach. Mobile scan was conducted immediately following tagging events in late fall with block nets still in place. Missing data due to equipment malfunction.

Reach		Total fish tagged (N)	Total detections of tagged fish (N)	Detection efficiency (%)
Knickerbocker	control	193	49	25.4
	treatment	181	NA*	NA*
South Fork	control	153	60	39.2
	treatment	73	37	50.7
North Fork	control	174	76	43.7
	treatment	88	77	87.5
Matthews Beach	control	102	27	26.5
	treatment	113	68	60.2
Grand total		1,077	394	44.0*

Of the 288 combined live detections of fish released in 2012 (Table 7), 84% occurred within the reach of release and an additional 4% within 100 m of that reach (Figure 9). Of the 30 live fish detections that occurred more than 100 m from the release reach, 77% were observed downstream (Figure 9a). A total of 25 fish tagged prior to 2012 were detected as live fish during the 2012-2013 mobile scan surveys. Half of these fish (48%) were detected within 100 m of the release reach. Of the remaining 52%, all moved downstream except three from SF-C that moved 500-1,000 m upstream (Figure 10).

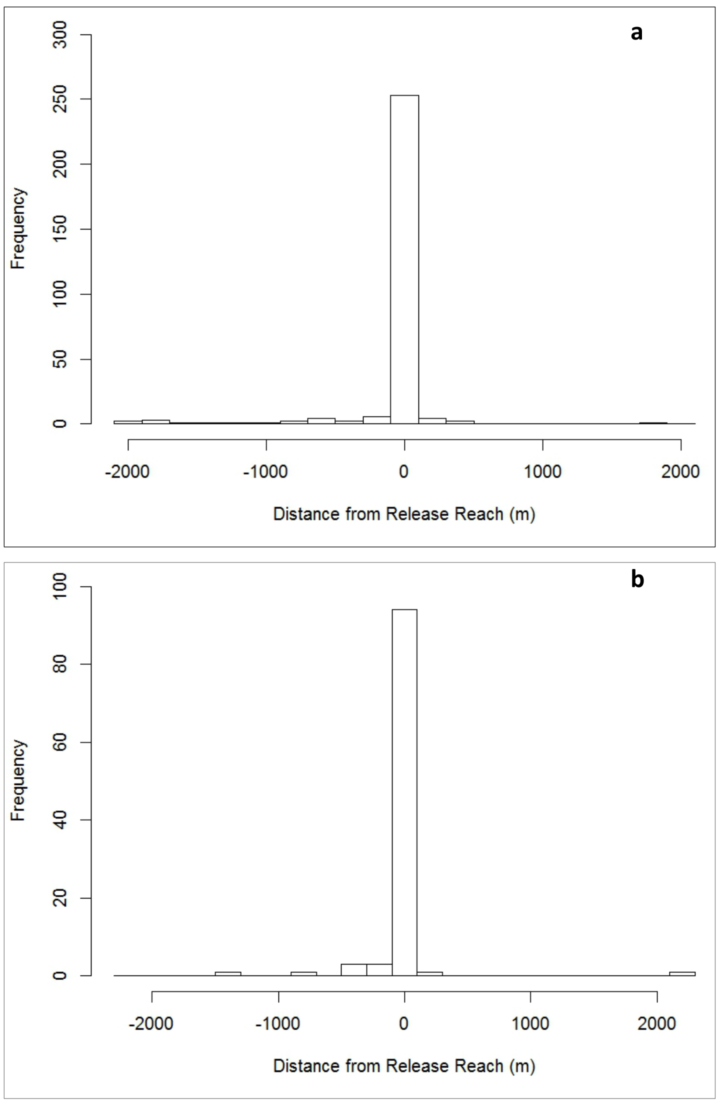

Figure 9. Histogram showing PIT-tag detection frequency by distance from 2012 release reach for live fish (a) vs. shed tags (b) across all three mobile scan surveys. Negative distances indicate detection downstream of release reach, while positive distances indicate detection upstream.

Patterns in detections of shed tags for fish released in 2012 were similar to those of live fish: 94% of shed-tag detections occurred within 100 m of the release reach (Figure 9b). Of these, 59% were detected within the release reach and another 30% slightly downstream. The higher frequency of detection downstream from the release reach for shed tags compared to live fish was likely due to passive hydrological transport following shedding of the tag or death of the fish, as opposed to active fish movement. These results suggest that a majority of resident cutthroat trout in Thornton Creek have high site fidelity, and that if fish do move, their movements are primarily downstream. These results are similar to those reported earlier for Thornton Creek by Leavy et al. (2010) and for Pipers Creek by Morley et al. (2010).

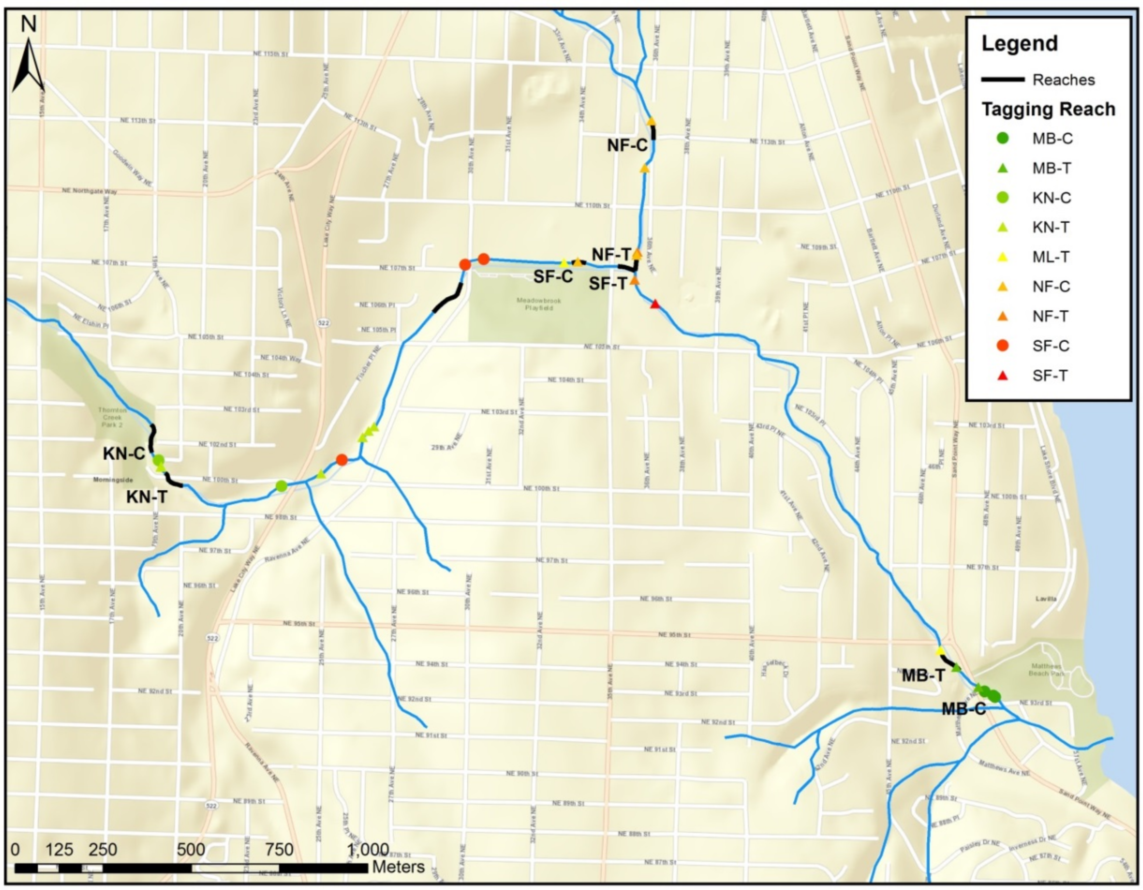

Figure 10. Mobile scan detections of live fish tagged prior to 2012 by release reach.

Fish Diet

A total of 71 unique prey items representing 63 different species were observed across all Thornton diet samples. On an individual fish basis, mean taxa richness within a given reach ranged from 3.1-7.1 prey items per stomach (Table 9). With the exception of fish sampled in MB-C, the majority of fish prey was of aquatic origin (i.e., either residing in the benthos, or terrestrial forms of aquatic larvae; Figure 11). Pooled across all fish for a given reach, prey diversity was highest for fish from MB-C, followed by the KN-C, NF-C, and MB-T. The lowest diversity was observed in fish from both South Fork reaches and from NF-T. For all sites, prey diversity was slightly higher in fish from control vs. treatment reaches and contained a higher proportion of terrestrial inputs (Table 9).

Table 9. Fish diet metric means and standard deviations (parentheses) for 2012 Thornton Creek study reaches. Taxa richness is the total number of unique taxa present at a site, H' refers to Shannon's diversity index, aquatic origin is the proportion of individuals originating in the aquatic zone, and terrestrial is the percentage of individuals from the riparian zone (regardless of origin).

Reach	Taxa richness	H' (log2)	Origin (%)	
			Aquatic	Terrestrial
Knickerbocker				
control	3.1 (0.5)	3.48	93 (2.9)	16 (4.8)
treatment	5.0 (0.3)	3.34	91 (3.2)	15 (4.5)
South Fork				
control	5.4 (0.5)	2.98	65 (8.2)	49 (6.9)
treatment	4.5 (0.7)	2.48	96 (8.2)	6 (1.7)
North Fork				
control	3.4 (0.4)	3.35	87 (3.8)	30 (6.7)
treatment	6.4 (0.9)	2.83	93 (2.4)	13 (3.3)
Matthews Beach				
control	7.1 (1.1)	4.12	47 (1.7)	60 (8.9)
treatment	5.8 (1.3)	3.35	71 (9.2)	34 (6.2)

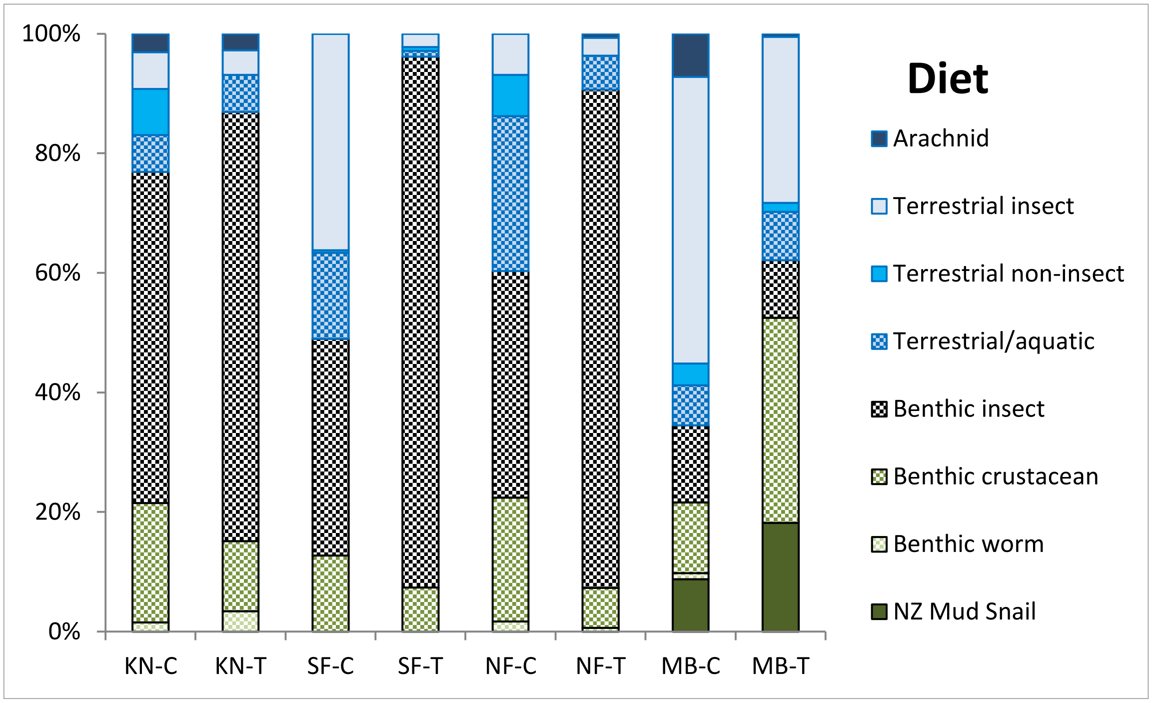

Figure 11. Fish diet relative abundance by major taxonomic groupings for 2012 Thornton study reaches. Abbreviations: Knickerbocker (KN), South Fork (SF), North Fork (NF), Matthews Beach (MB), control (C), treatment (T).

With the exception of Knickerbocker, all control and treatment reaches were significantly taxonomically distinct from each other (Figure 12). At both Knickerbocker reaches, diets consisted largely of mayfly nymphs, black fly larvae, and larval midges. These taxa were also common in the diets of fish from South and North Fork reaches, along with adult midges and benthic crustaceans. For fish from the Matthews Beach control reach, the most common prey items consumed were terrestrial insects such as aphids and thrips. For fish from the Matthews Beach treatment reach, benthic crustaceans and New Zealand mud snails were the most common prey. On a site level, there was a high degree of overlap in fish diets between KN-C, SF-T, and NF-C (Figure 12). Diet content at these reaches was more than 75% benthic sources (Figure 11). Fish diets at all other sites were taxonomically different from each other (one-way ANOSIM, $P < 0.05$).

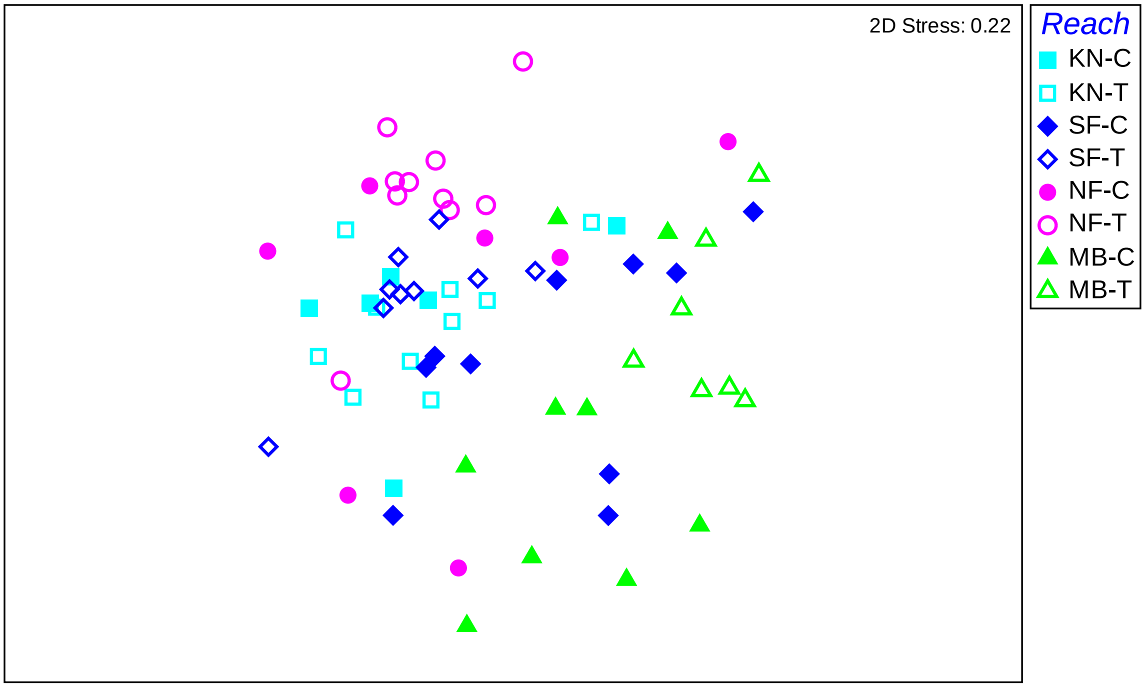

Figure 12. Fish diet non-metric multidimensional plots. Symbols closer to one another have more similar taxonomic composition than symbols farther apart. Thornton Creek 2012 raw data was standardized to account for differences in stomach fullness between individual fish. Abbreviations: Knickerbocker (KN), South Fork (SF), North Fork (NF), Matthews Beach (MB), control (C), treatment (T).

Benthic Invertebrates

Based on B-IBI scores, all reaches sampled in 2012 were in poor or very poor condition (Figure 13). The Matthews Beach site has the lowest combined scores of all sites while all other sites had one reach in "poor" condition and one reach in "very poor" condition. Across the last three most recent sample years, B-IBI scores have ranged from only 10 to 18 across all study sites. On average, the highest scores were observed at NF-C, and the lowest at SF-T and KN-C (Figure 13). There were no consistent patterns in B-IBI scores between treatment and control reaches.

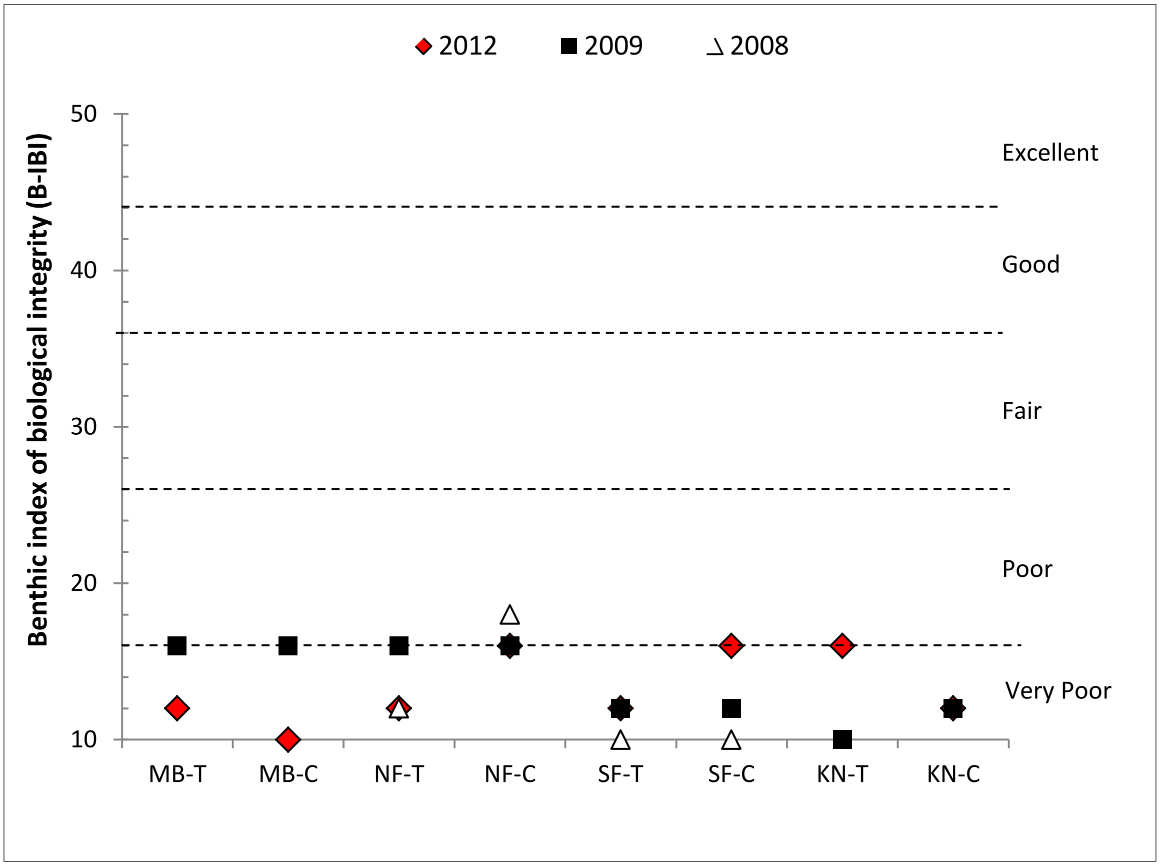

Figure 13. B-IBI scores and associated biological stream condition for all study reaches over the last three sample years. Only North Fork and South Fork reaches were sampled in 2008.

Stonefly taxa (order Plecoptera) were nearly completely absent across all reaches, and mayfly taxa (order Emphemeroptera) were limited to the disturbance-tolerant mayfly *Baetis tricaudatus*. Taxa richness ranged between 17 and 23 for all samples, representing a slight decrease from 2009 at all reaches except KN-C (Figure 14). The proportion of dominant taxa (3 most numerically abundant) represented over 64% of all individuals from each reach, reflecting limited diversity (Table 10). Treatment reaches typically had higher proportions of disturbance-tolerant taxa than their control reaches. Benthic invertebrate densities for 2012 ranged from 329 to 4,075 individuals per m^2, and varied more by sample year than by study reach (Table 10).

Table 10. Benthic invertebrate taxa richness and life history metrics by sample year and study reach. EPT = number of unique taxa from the insect orders Ephemeroptera, Plecoptera, and Trichoptera.

	Taxa richness (N)				Life history (%)			
Site/reach	EPT	Clinger	Long-lived	Intolerant	Dominant taxa (top 3)	Predator	Tolerant	Density (N/m^2)
2012								
Knickerbocker								
control	4	5	1	0	82	1	34	1,228
treatment	4	6	1	0	74	2	26	1,239
South Fork								
control	5	8	0	0	66	1	47	814
treatment	5	7	2	0	76	2	49	329
North Fork								
control	5	9	1	0	64	1	26	1,779
treatment	3	5	0	0	81	0	74	1,237
Matthews Beach								
control	3	4	0	0	86	0	80	1,062
treatment	2	4	1	0	87	0	83	4,075
2009								
Knickerbocker								
control	3	3	1	0	79	1	32	2,612
treatment	2	4	1	0	88	0	56	2,751
South Fork								
control	3	6	0	0	78	1	67	4,785
treatment	4	7	1	0	81	1	60	4,699
North Fork								
control	3	7	1	0	69	3	24	2,887
treatment	5	8	1	0	65	1	42	2,335
Matthews Beach								
control	3	5	0	0	63	1	39	5,636
treatment	1	3	0	0	67	0	22	4,335
2008								
South Fork								
control	2	4	1	0	84	0	76	2,153
treatment	3	4	1	0	90	0	86	1,255
North Fork								
control	3	7	1	0	56	11	39	945
treatment	4	6	1	0	76	2	65	3,943

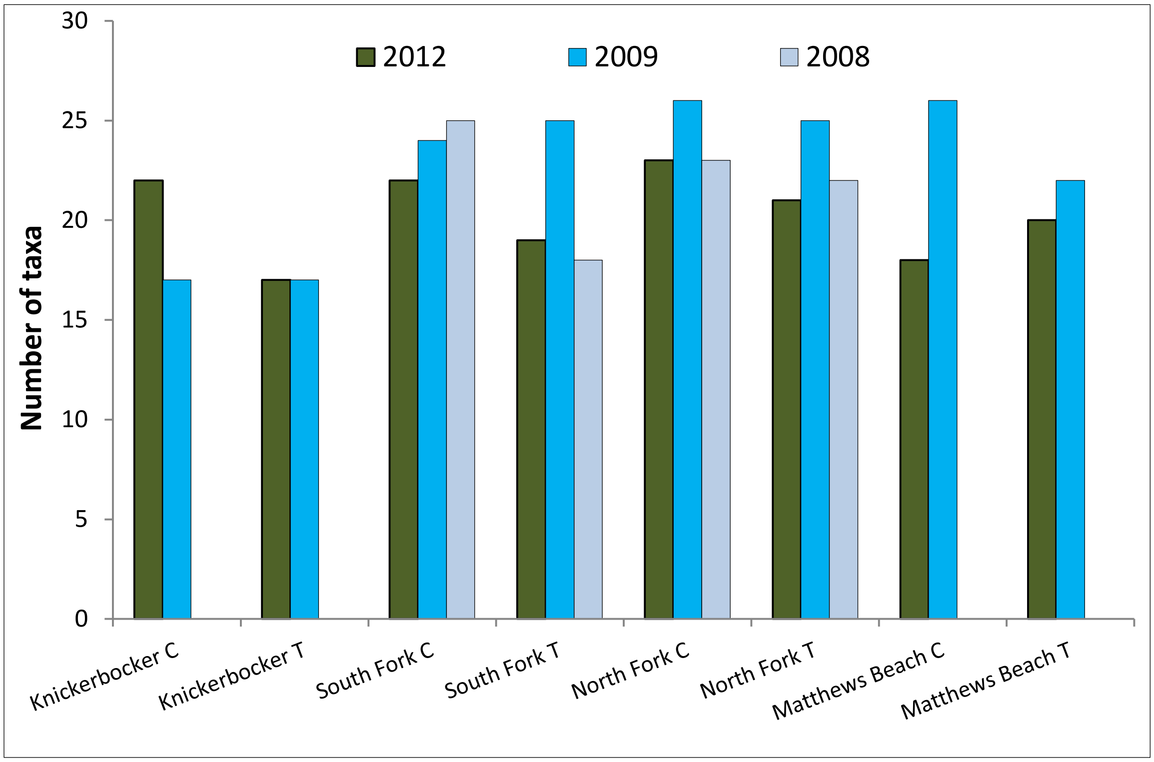

Figure 14. Taxa richness by study reach and sample collection year. Only North Fork and South Fork sites were sampled in 2008. Abbreviations: treatment (T), control (C).

Matthews Beach reaches were heavily dominated (>77%) by the New Zealand Mud Snail (NZMS) *Potamopyrgus antipodarum*. This small and highly invasive freshwater snail was not observed at our other study reaches, and was not detected in any of the study reaches in 2008 by USFWS. Matthews Beach reaches were not sampled by USFWS in 2009, the year in which this snail was first reported in the Thornton Creek watershed. High proportions of NZMS at Mathews Beach reaches in 2012, coupled with the relatively low proportions represented in fish diets (Figure 11) suggest that fish habitat conditions may decline as NZMS abundance increases. Oligochaetes and the amphipod genus *Crangonyx* accounted for the remaining majority of individuals in both Mathews Beach reaches.

Remaining sites were dominated by the tolerant mayfly *B. tricaudatus*, the Dipteran families Chironomidae (midges) and Simulidae (black flies), *Crangonyx*, and the net-spinning Trichopteran genus *Hydropsyche*. At NF-T, the benthic invertebrate community was composed largely of *Crangonyx* (33%) and *B. tricaudatus* (41%), while the NF-C benthic community was more evenly spread amongst the four insect orders. SF-T had a higher proportion of Dipterans than the control, while *B. tricaudatus* comprised 30-43% of individuals at both reaches. Knickerbocker control and treatment reaches were the most similar in taxonomic composition, with over 50% of individuals from the order Diptera and another 20% identified as *B. tricaudatus*.

Drift Invertebrates

We identified a total of 50 unique taxa across all Thornton Creek drift samples, 9 of which were observed at multiple life stages (i.e., larval, pupal, or adult). Density varied 0.9-10.2 individuals per m^3 of water, and was lowest at both North Fork reaches (Table 11). Taxa richness was lowest in samples from North and South Fork control and treatment reaches. Taxa diversity was particularly low at both South Fork reaches. With the exception of the South Fork reaches, which were dominated by the Hemipteran family Aphidae, the majority of invertebrates captured in drift samples were benthic taxa or winged-adult forms of aquatic insect larvae (Table 11).

Table 11. Drift invertebrate metrics for 2012 Thornton Creek study reaches. Taxa richness is the total number of unique taxa present at a site, H' refers to Shannon's diversity index, aquatic origin is the proportion of individuals originating in the aquatic zone, and terrestrial is the percentage of individuals from the riparian zone (regardless of origin).

Site/reach	Density (N/m^3)	Taxa richness	H' (log2)	Origin (%)	
				Aquatic	Terrestrial
Knickerbocker					
control	3.40	27	3.28	89	19.6
treatment	8.03	26	3.11	94	8.9
South Fork					
control	8.59	17	1.73	22	84.5
treatment	4.18	18	2.60	43	66.5
North Fork					
control	0.86	10	3.08	94	6.5
treatment	1.07	17	3.92	61	45.5
Matthews Beach					
control	10.17	24	3.09	98	3.0
treatment	1.73	22	3.91	72	37.9

With the exception of the North Fork reaches, all paired control and treatment reaches were more taxonomically similar to each other than to other sample sites in the watershed (Figure 15). The highest degree of overlap between control and treatment reaches was at the Knickerbocker site, with 74% similarity. Knickerbocker reaches were characterized by a high proportion of benthic insects and crustaceans; in particular larvae of the black fly genus *Simulium*, the mayfly nymph *B. tricaudatus*, larval midges of the tribe Orthocladiinae, and the Amphipod genus *Crangonyx* (Figure 16). South Fork reaches were also very similar to each other with 71% overlap, driven primarily by a high proportion of adult aphids. The North Fork and Matthews Beach sites exhibited less similarity (< 45%) between their respective control and treatment reaches.

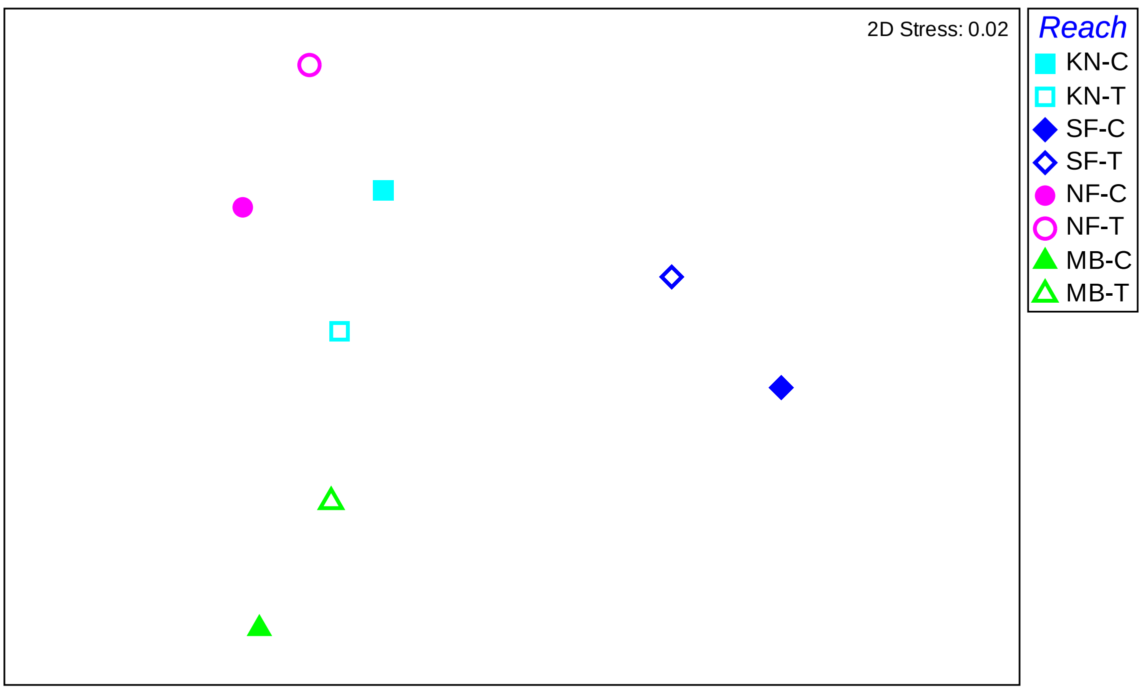

Figure 15. Drift invertebrate non-metric multidimensional plots. Symbols closer to one another have more similar taxonomic composition than symbols farther apart. Raw data was standardized to account for potential differences in sample efficiency between reaches. Abbreviations: Knickerbocker (KN), South Fork (SF), North Fork (NF), Matthews Beach (MB), control (C), treatment (T).

In terms of differences across sites, Knickerbocker and North Fork were most similar to each other, and Matthews Beach and South Fork most dissimilar (Figure 16). MB-C was unique in having a high proportion of benthic crustaceans and *P. antipodarum*. This invasive species was also observed at MB-T, but in lower relative abundance. South Fork reaches were dominated by terrestrial insects. Both Knickerbocker and North Fork reaches were characterized by a high proportion of benthic insects in their drift.

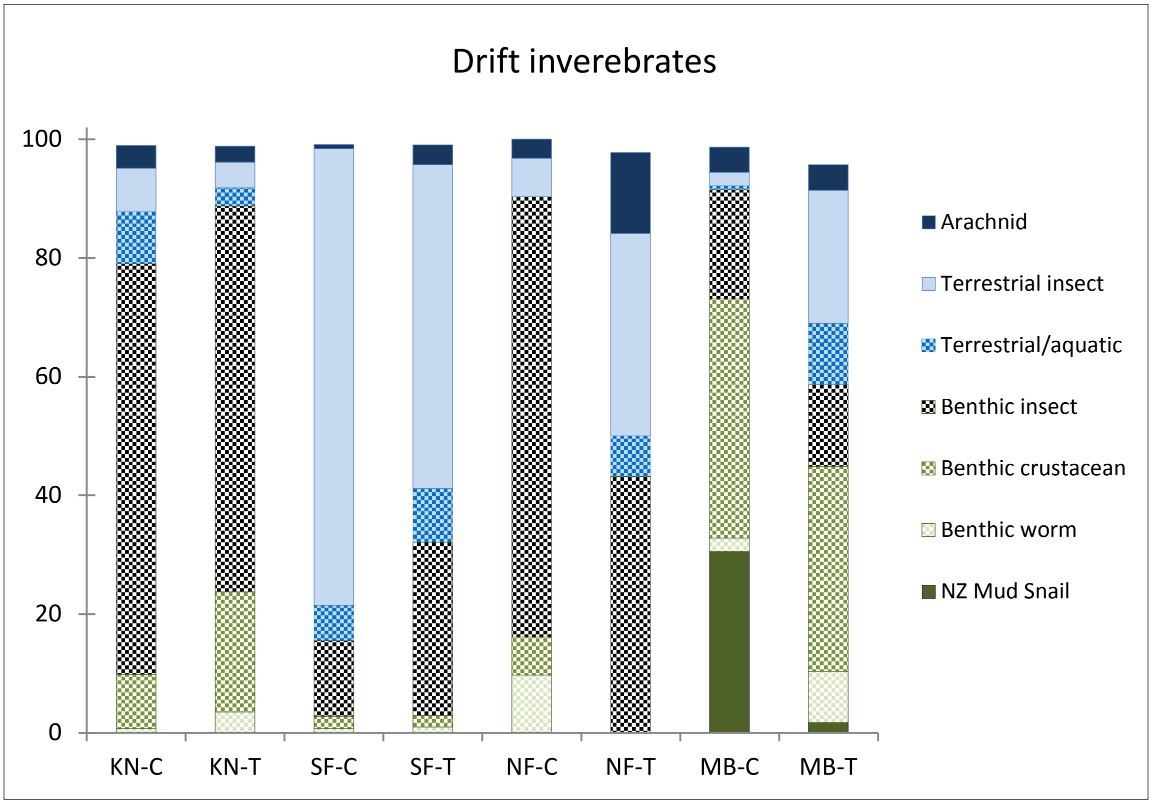

Figure 16. Drift relative abundance by major taxonomic groupings for 2012 Thornton study reaches. Abbreviations: Knickerbocker (KN), South Fork (SF), North Fork (NF), Matthews Beach (MB), control (C), treatment (T).

Periphyton

Based on taxonomic analysis of the diatomaceous portion of periphyton samples, we identified a total of 202 unique taxa across all Thornton sample reaches from 2006-2009 and 2012. Eighty two unique taxa were identified in 2012, 16 of which were species not collected in previous sampling events from 2006-2009. Species richness at individual study reaches ranged from 21 to 57; however, there was considerable variability from year to year within a given site (Table 12). Species assemblages were similar among reaches within a given site (treatment vs. control) and showed limited variability between sites in a given year (Figure 17). Matthews Beach and Knickerbocker sites were the most taxonomically distinct from each other (Figure 17). Taxonomic composition for all sites was significantly different among years, largely due to the higher abundance of *Achnanthidium minutissimum* in 2008 (Figure 17).

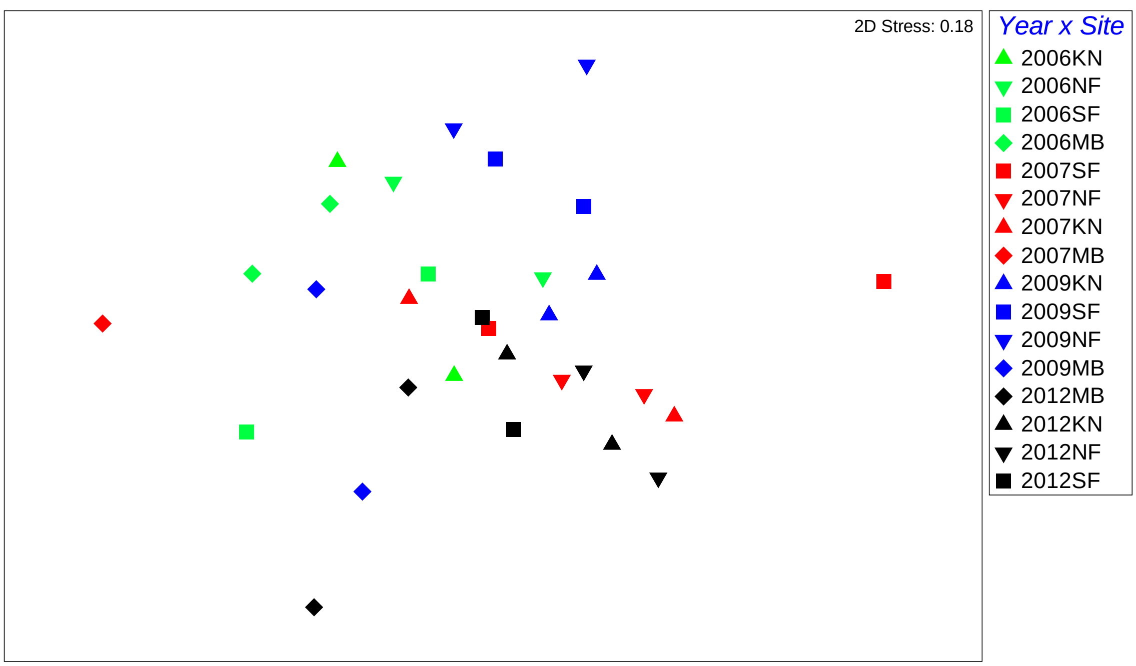

Figure 17. Non-metric multidimensional scaling plot of Thornton Creek diatom assemblages by year (different colors) and site (different symbols). Symbols closer to one another have more similar taxonomic composition than symbols farther apart. Abbreviations: North Fork (NF), Knickerbocker (KN), South Fork (SF), Matthews Beach (MB).

Table 12. Diatom metrics from periphyton samples collected in September from Thornton Creek study reaches from 2006-2009 and 2012. Metrics are based on diatom taxonomic, functional, and disturbance life history attributes. Abbreviations: control (C), treatment (T).

	Predicted urban	Knickerbocker								South Fork									
		2006		2007		2009		2012		2006		2007		2008		2009		2012	
Metric	response	C	T	C	T	C	T	C	T	C	T	C	T	C	T	C	T	C	T
Community Structure																			
Richness	decrease	39	34	36	23	25	31	34	36	32	21	38	25	41	33	52	34	40	41
Diversity (H')	decrease	3.5	3.9	3.4	1.4	2.5	2.5	1.8	2.7	3.4	2.8	2.3	2.4	3.8	3.2	3.9	2.9	3.1	2.8
Dominant taxon (%)	increase	24	20	37	78	56	63	74	53	30	30	66	57	29	38	28	51	49	58
Metal impacts																			
Metals tolerant (%)	increase	12	23	9	4	7	10	4	7	13	11	3	5	13	16	11	12	8	10
Disturbance tolerant (%)	increase	24	2	2	0	1	3	0	0	1	12	5	1	31	7	6	1	0	0
Sediment impacts																			
Siltation tolerant (%)	increase	14	23	19	3	14	11	5	8	33	38	7	14	24	18	26	16	13	10
Motile (%)	increase	21	32	25	5	24	21	9	19	42	43	10	21	31	63	40	26	22	15
Nutrient enrichment								+											
Low DO tolerant (%)	increase	3	11	8	1	4	3	2	1	3	1	2	3	3	1	3	2	4	3
Eutraphentic tolerant (%)	increase	63	79	91	97	88	84	92	84	88	84	84	92	40	39	73	84	84	87
Polysaprobous tolerant (%)	increase	22	41	28	6	18	15	9	11	40	47	9	18	28	28	33	25	20	20
N autotrophic (%)	increase	87	81	84	97	79	85	96	91	65	62	92	90	71	81	74	82	90	92

Table 12. Continued.

Metric	Predicted urban response	North Fork										Mathews Beach							
		2006		2007		2008		2009		2012		2006		2007		2009		2012	
		C	T	C	T	C	T	C	T	C	T	C	T	C	T	C	T	C	T
Community Structure																			
Richness	decrease	34	30	24	30	40	38	57	48	36	20	43	48	43	57	44	34	29	32
Diversity (H')	decrease	3.5	2.7	1.4	2.3	4.0	3.9	4.2	3.9	2.1	1.6	3.7	4.1	3.6	4.6	3.5	2.7	2.0	3.1
Dominant taxon (%)	increase	25	54	81	63	28	31	30	35	69	72	36	19	24	16	36	52	57	30
Metal impacts																			
Metals tolerant (%)	increase	15	5	3	2	9	9	12	9	5	3	8	14	6	15	11	4	3	9
Disturbance tolerant (%)	increase	2	2	1	2	29	4	2	15	0	0	1	0	1	5	1	3	0	0
Sediment impacts																			
Siltation tolerant (%)	increase	18	12	4	7	9	31	20	15	6	1	26	29	21	26	17	8	3	10
Motile (%)	increase	49	28	7	11	14	40	39	27	10	7	37	43	46	33	28	15	31	29
Nutrient enrichment																			
Low DO tolerant (%)	increase	8	1	1	2	1	10	7	4	1	0	4	9	3	5	7	2	1	4
Eutraphentic tolerant (%)	increase	78	84	95	90	25	69	68	58	87	93	77	80	65	70	79	86	94	82
Polysaprobous tolerant (%)	increase	25	15	7	9	17	33	28	20	10	4	23	35	19	38	22	9	5	14
N autotrophic (%)	increase	76	87	95	95	92	85	72	75	95	98	80	76	68	81	82	89	95	88

To examine differences between the four paired treatment and reference reaches, we graphed strip plots showing the distribution of the $\log_{10}$ ratio of treatment-to-control (T:C) values for selected metrics (Figure 18). For the four metrics we examined, ratios were distributed evenly above and below 0, indicating no difference between reaches. As reaches were selected to be as similar as possible for a given site, and because restoration work is still in the pre-construction stage, this result was expected. However, within-site relationships between treatment and control reaches were not consistent from year to year.

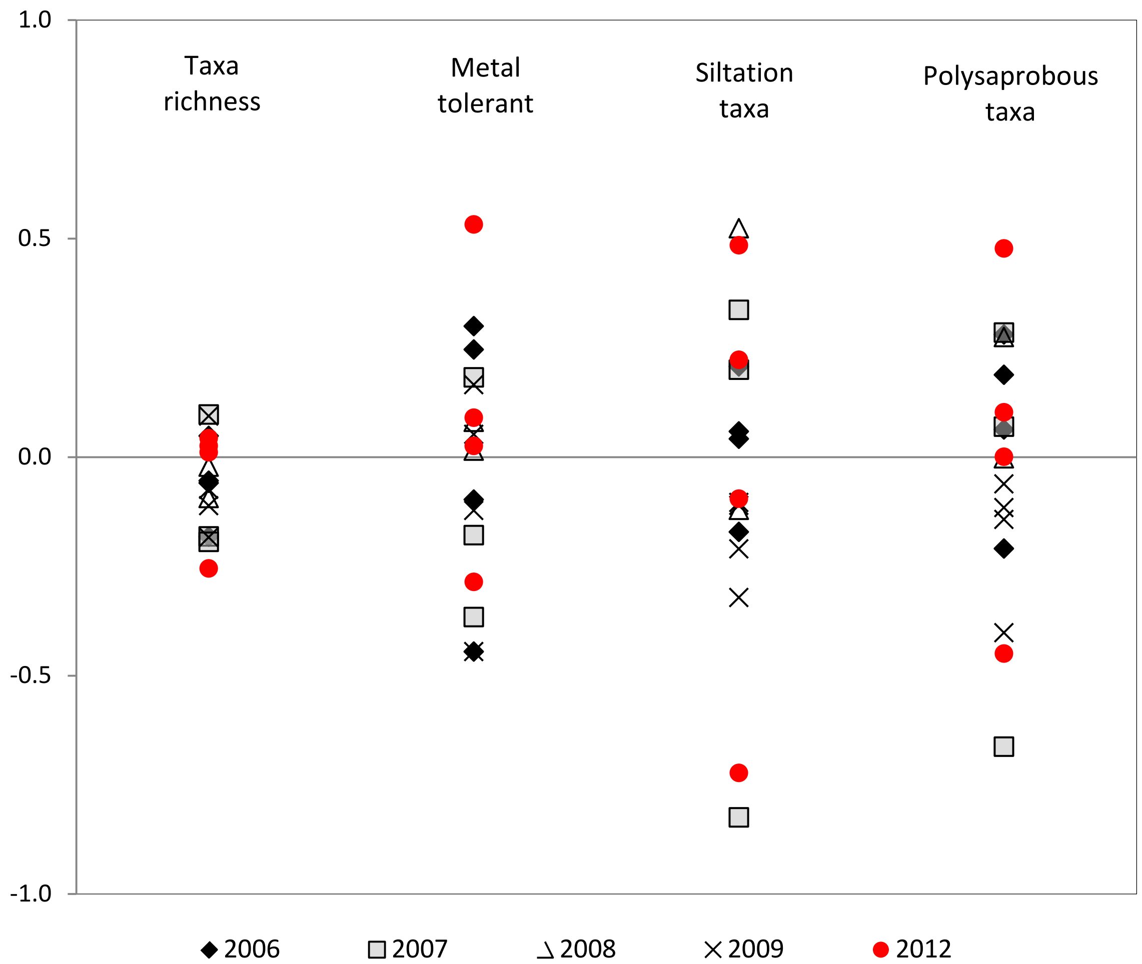

Figure 18. Strip plots of the log10 ratio of treatment (T) to control (C) values for four diatom metrics. Data were collected from 2006-2009 and 2012 over four paired study reaches on Thornton Creek. Values of 0 indicate no difference between reaches, +1 indicates a ten-fold greater value for T reaches, and -1 a ten-fold greater value for C reaches.

The Matthews Beach and Knickerbocker paired sites appeared to be the most variable while the South Fork and North Fork paired sites remained relatively stable. This high year-to-year variability observed for some metrics (e.g., siltation taxa) emphasizes the importance of collecting multiple years of pre- and post-project data to accurately interpret project effectiveness.

Compared to previous years, samples collected in 2012 showed a strong reduction in disturbance taxa at MB-C and a strong increase NF-C. Values from the Knickerbocker and South Fork sites remained near their respective overall averages. Following project construction, we hypothesize that diatom species diversity will increase at treatment reaches relative to paired controls, and that the proportion of species tolerant to high levels of sedimentation and nutrient enrichment will decrease at treatment reaches.

Regional comparisons of Thornton Creek diatom assemblages with those of other local urban and forested streams remained similar to results previously reported (Leavy et al. 2010). Taxonomic composition in Seattle urban streams was significantly different than that in regional forested streams ($R = 0.58$, $P < 0.001$). Thornton Creek diatom samples largely overlapped with those from other Seattle urban streams, but were also quite variable (Figure 19). Diatom samples from South Fork paired reaches were most similar to those collected from forested streams, and those from Matthews Beach paired reaches were most divergent.

Total periphyton biomass ranged from 0.03-0.64 mg/cm^2 ash-free dry mass (AFDM) across all reaches and years (Table 13). Highest AFDM values were consistently observed at MB-C, and were relatively low at other locations. Total algal biomass ranged from 0.08 to 3.24 $\mu g/cm^2$ and was less consistent between years, but again typically highest at MB-C (Table13). Between 2006 and 2009, autotrophic index values were generally highest at NF-C and MB-T, indicating that algae comprise a smaller proportion of total periphyton biomass at these reaches relative to other locations (Table 13).

In 2012, autotrophic index values were uniformly high across all study site reaches. These observed patterns and the associated variation in periphyton biomass data were likely due to a combination of natural and anthropogenic factors. Sites with less canopy coverage or with higher water temperatures experience higher algal growth rates. Differences between reaches in the frequency of scour events, sedimentation rates, and water chemistry may also contribute to differences in periphyton biomass.

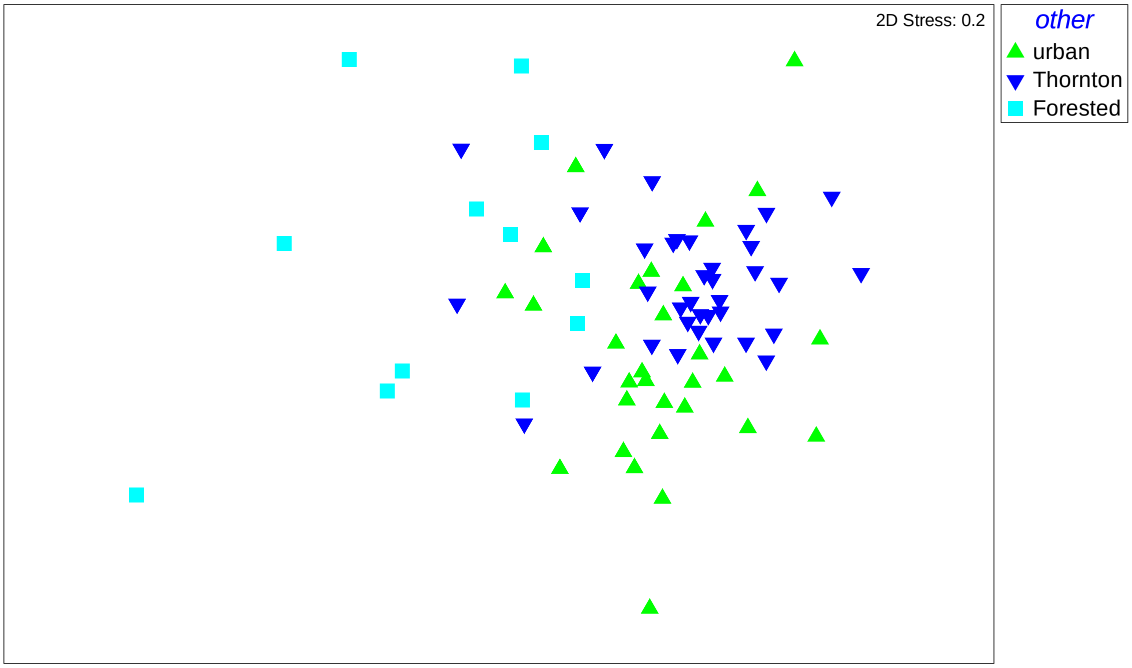

Figure 19. Regional comparison of diatom assemblage metrics between Thornton Creek and other urban and forested streams. Thornton Creek data from 2006-2009, 2012. All other data collected from 2006-2009. Symbols closer to one another have more similar taxonomic composition than symbols farther apart.

Table 13. Periphyton biomass (as ash-free dry mass and chlorophyll *a* densities) collected during 2006-2009 and 2012 from Thornton Creek study reaches. Ash-free dry mass is also referred to as organic matter and includes algae, fungi, bacteria, and microzoans. Chlorophyll a is the algal component of periphyton. Autotrophic index is the ratio of organic matter (ash-free dry mass) to chlorophyll *a* and is a measure of the proportion of total periphyton biomass composed of algae. A "normal" autotrophic index ranges 50-200 (Steinman and Lamberti 1996); higher values indicate that algae comprise a smaller proportion of the total periphyton matrix. Abbreviations nc = not collected; na = equipment malfunction.

	Ash-free dry mass (mg/cm^2)					Chlorophyll *a* (μg/cm^2)					Autotrophic index				
Study reach	2006	2007	2008	2009	2012	2006	2007	2008	2009	2012	2006	2007	2008	2009	2012
Knickerbocker															
control	0.10	0.10	nc	0.07	0.16	0.56	0.61	nc	0.77	0.45	187	161	nc	97	367
treatment	0.27	0.18	nc	0.04	0.15	1.59	0.81	nc	0.35	0.37	173	227	nc	103	418
South Fork															
control	0.17	na	0.03	0.03	0.27	0.50	0.27	0.08	0.12	0.37	332	na	360	234	725
treatment	0.33	0.14	0.04	0.05	0.12	3.24	0.68	0.08	0.19	0.33	101	203	519	274	362
North Fork															
control	0.18	0.13	0.08	0.05	0.11	0.35	0.38	0.12	0.08	0.25	508	340	641	535	466
treatment	0.06	0.12	0.06	0.12	0.07	0.24	0.40	0.23	1.23	0.26	249	292	247	98	274
Matthews Beach															
control	0.64	0.35	nc	0.35	0.52	1.61	1.72	nc	2.10	2.24	398	202	nc	166	231
treatment	0.28	0.15	nc	0.12	0.28	0.44	0.17	nc	0.50	1.45	633	882	nc	247	194

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Appendix A: Draft Hyporheic Monitoring Proposal

Introduction

Seattle Public Utilities is implementing two floodplain reconnection projects in Thornton Creek with the goals of improving on-site stormwater retention and bio-filtration, increasing local habitat complexity, and improving biological stream condition. This will be accomplished by increasing lateral movement of the stream within its floodplain, and by promoting vertical connectivity via creation of a hyporheic zone.

The hyporheic zone is the area of saturated sediments below and alongside the stream channel where surface and groundwater mix (Orghidan 1959). This zone provides water and nutrient storage; flood dampening and groundwater recharge; biogeochemical cycling of nutrients, organic matter, and contaminants; and biological production via invertebrate and nutrient inputs (Boulton 2007; Grimm et al. 2007; Robertson & Wood 2010). In urban watersheds, hyporheic zones are typically greatly diminished or completely lacking due to channelization, floodplain development, sedimentation, and loss of hydrologic connectivity (Hancock 2002; Boulton et al. 2003; Boulton 2007). While the importance of lateral and longitudinal connectivity is increasingly recognized in restoration design, vertical connectivity is still rarely considered (Boulton 2007).

The hyporheic components of Thornton floodplain reconnection projects are designed to promote greater vertical hydraulic exchange, thereby increasing floodplain water storage, creating more complex in-stream habitat via areas of upwelling and downwelling, moderating surface temperatures, and supporting hyporheic assemblages that promote biogeochemical cycling and increase food production for resident fish populations. While multiple years of baseline data have now been collected for physical, hydrologic, and biological characteristics of surface waters and for physical and hydrologic conditions of hyporheic waters, no biological monitoring of the hyporheic zone has yet occurred at any of the project reaches.

The purpose of this appendix is to outline a pre- and post-biological monitoring plan, with particular emphasis on the hyporheic zone. Pre-project surface water monitoring is already described in the main body of this report and in Leavy et al. (2010). Very little information exists in the literature on biological processes of urban hyporheic zones, and none that we are aware of in relation to stream restoration. Like SPU, we are interested in evaluating the success of individual floodplain reconnection projects within the Thornton Creek watershed, but also have a broader research interest in learning more about how urban hyporheic zones function and the best ways in which to sample them.

We propose to balance these two goals by sampling most intensively within Thornton Creek, but also including additional urban and forested control reaches outside of the watershed. Regional reference data will provide a more comprehensive context in which to evaluate Thornton restoration success, and also provide information to help guide future restoration design. NOAA will address three lines of questioning in this study:

- Project evaluation:
  - How do restored Thornton Creek hyporheic zones function relative to:
    - Pre-restoration condition within the same treatment reaches?
    - Paired control reaches?
  - How long will it take for hyporheic processes to establish in restored reaches?
  - How long will these processes persist; i.e., are they self-sustaining?

- Effects of urbanization on hyporheic zones:
  - How does Thornton hyporheic condition compare to:
    - Urban streams within Seattle?
    - Forested streams in the Cedar River Municipal Watershed?
  - What are the relationships between surface and sub-surface conditions in urban versus lowland forested streams?

- Best sampling strategies for urban hyporheic zones:
  - What are the advantages and disadvantages of monitoring via colonization chambers versus piezometer water samples?
  - What level of sample replication is needed to detect effect; i.e., what is the extent of within-reach variability of hyporheic monitoring parameters across the urban gradient?

Methodology

What to Monitor

Biological processes within the hyporheic zone such as organic matter decomposition, nutrient cycling, and contaminant detoxification are largely carried out by invertebrate and microbial assemblages (Boulton 2007). These processes in turn affect algal production, availability of particulate and dissolved carbon, and concentrations of nutrients and contaminants (Grimm et al. 2007). The hyporheic zone is thought to serve as biological refugia during high flows and other disturbances, thus supplementing the prey base for fish and other higher organisms (Robertson & Wood 2010).

NOAA biological monitoring in the hyporheic zone will focus primarily on invertebrate and microbial taxonomic and functional diversity and overall density. Invertebrate samples will be analyzed by professional taxonomists for species identification and enumeration; head-capsule width measurements will be taken for biomass calculations. Microbial samples will be processed at the NWFSC using flow cytometry and molecular techniques such as automated ribosomal intergenic spacer analysis (ARISA) (Fisher & Triplett 1999). A secondary focus of sampling will be on quantifying algal densities and availability of different size fractions of carbon. Algae and particulate and dissolved carbon samples will be collected on glass-fiber filters and analyzed at NWFSC.

How to Monitor

The four most commonly applied hyporheic sampling techniques are pump sampling, colonization chambers, standpipe coring, and freeze coring (Fraser & Williams 1997; Scarsbrook & Halliday 2002). As coring is highly destructive to the stream bed, neither of these methods are appropriate for this study. We will instead rely primarily on pumping techniques but will experiment with colonization chambers in project reaches where permanent hyporheic wells can be incorporated into restoration design.

Colonization chambers involve inserting mesh-encased gravel baskets into the streambed, allowing for a pre-determined period of colonization by resident organisms and extraction and removal of all biological material. We will limit disturbance to the streambed by employing a permanent well design modified after Hendricks & Rice (2000). Colonization chambers have the advantage of better estimating more sessile organisms and allowing for various experimental manipulations, but may underestimate smaller mobile taxa and are more difficult (i.e., costly and labor intensive) to employ. During project construction, contractors will install multiple hyporheic wells (6” diameter PVC, ½” perforations along length, 40 cm deep) in each treatment reach. Within each well we will utilize nested piezometers and inert spacers to deploy colonization baskets at various depths.

As these large-diameter hyporheic wells will likely be unfeasible at non-treatment reaches, we will utilize pump sampling to extract interstitial hyporheic waters from piezometers at both treatment and control reaches (Hunt & Stanley 2000). Drawbacks of pump sampling include bias towards smaller and less tenacious organisms and potential underestimation of both density and taxa richness (Scarsbrook & Halliday 2002). However, this method is much more cost-effective and feasible in entrenched urban streams, and sampling is relatively easy and fast once piezometers are installed.

Piezometers will already need to be installed at all study reaches for hydrologic monitoring by USFWS. Employing both colonization baskets and piezometers at treatment reaches will allow us to test the sampling efficiency and variability associated with both techniques. Taking advantage of the ability to incorporate hyporheic wells into project design at treatment reaches may provide opportunities for expanded future research such as examining egg-to-fry survival via artificial salmon redds (Johnson et al. 2012).

Where to Monitor

The hyporheic zone is often patchy in distribution and variable in space and time (Dahm et al. 2006). Ideally, our sample design would capture all potential sources of variability, but given realistic budget constraints, we have selected sample locations where we believe the most information will be gained. We will sample along five cross-sections within each of the three paired control and treatment Thornton project reaches. This level of replication will allow for statistical testing at the project scale, and evaluation of within-reach hyporheic spatial variability.

Cross-sections will be distributed across the entire study reach and selected to coincide with areas of upwelling, which for treatment reaches will be determined by project design features. Along each cross-section a piezometer will be installed mid-channel at two sample depths: 10 cm (shallow) and 40 cm (deep). The shallow depth was selected because this boundary layer typically contains the highest hyporheic invertebrate activity, and also because 10 cm is likely the maximum sample depth in most urban control reaches. The deep sample will allow us to evaluate whether hyporheic processes are functioning beyond the initial boundary layer, and how the two zones differ from each other and relative to forested control reaches. Mid-channel locations were selected primarily for ease of piezometer installation, but future sampling should consider expanding sample locations into the floodplain.

We will deploy a similar spatial design at non-project reaches, but at three rather than five cross sections per reach. This will reduce cost but still provide some information on within-reach variability. One potential location for selecting forested control reaches is the Cedar River Municipal Watershed. This protected area contains some of the healthiest remaining Puget Sound streams, is under the jurisdiction of the City of Seattle, and could potentially serve as a "seed source" for inoculated project materials. NOAA will collect hyporheic samples from 3-5 forested reaches.

The next priority will be to collect samples from additional urban stream reaches in Seattle. This will provide further context for interpreting Thornton Creek hyporheic data, complement extensive surface biological monitoring that has already occurred at Longfellow and Pipers Creek (Morley et al. 2010), and provide baseline data for future floodplain reconnection projects planned outside of the Thornton Watershed (e.g., Taylor Creek). The last priority will be to collect hyporheic biological data at one of the Matthews Beach reaches of Thornton Creek. Although no projects are currently slated for that area, substantial pre-project surface monitoring has already occurred at these reaches. Sampling at a mainstem sites downstream of all project sites could also help evaluate whether project benefits extend beyond the reach-scale.

When to Monitor

For all Thornton project locations, collecting baseline data prior to ground breaking is a top priority. Knickerbocker construction is currently scheduled to begin spring of 2014; thus one year of baseline data can still be collected in summer 2013. The confluence project is to begin in spring 2015, allowing for 1-2 years of baseline data. Sampling at control reaches can begin simultaneously if funding allows or be delayed until post-project Thornton data collection begins in 2015. A minimum of three years of post-construction data should be collected in order to allow adequate time for biological recovery and to capture natural year-to-year variability. We recommend that more than one year of reference data also be collected, but suggest alternating sample years to reduce annual monitoring costs.

For all locations, sampling will be conducted twice annually: summer and fall. Late summer is the typical index period for surface biological monitoring and captures low flow conditions. Sampling again following the first freshets of fall will allow comparison of hyporheic functioning under different flow conditions, capture potential changes in water quality following the first flush of road run-off, and occur at a time of year when surface and groundwater temperature profiles typically flip (Leavy et al. 2010).

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