Evaluating Effectiveness of Streambank Restoration in the southern interior of British Columbia

EXECUTIVE SUMMARY

Beginning in the early 1990s, the Southern Habitat Unit of Fisheries and Oceans Canada in British Columbia, working with numerous local partners and using funds in part provided through the Southern Endowment Fund of the Pacific Salmon Commission, initiated an eroding stream bank restoration program for the southern interior of British Columbia. The three explicitly stated purposes for the bank restoration initiative were stopping erosion, fostering a stewardship mentality within the local land owner community, and increasing native resident and anadromous salmonid production. Independent assessments of the bank stabilization projects have demonstrated that the bank treatments have low incidents of structural failure and that the projects are highly effective at halting bank erosion.

In this chapter, we report results of an extensive post-treatment effectiveness evaluation of stream bank stabilization and restoration efforts on the Salmon River and Bessette Creek focused on the ecological effects of bank restoration. Our approach was based on a comparison of stream channel and riparian vegetation condition and aquatic invertebrate abundance and biomass at 16 restored sites to 11 actively eroding "control" banks. Specifically, we addressed three questions: (1) has bank restoration had a measurable influence on in-channel or riparian condition or aquatic invertebrate productivity, (2) did the condition of a river channel upstream of a study site influence habitat conditions at the study site and (3) were there key variables that constrained the ecological response of a stream reach to bank restoration?

We found limited evidence that restoration of eroding river banks has had a substantive influence at the scale of the river reach. Restored and un-restored sites did not significantly differ in habitat structural diversity or flow velocity distribution, depth to width ratio, substrate particle sizes, or in the abundance and wet weight of aquatic invertebrates. However, treated sites tended to have more shrubs along the outside bank, higher inside banks, and narrower wetted widths, and each of these differences may be indicative of processes that will lead to notable system recovery given more time. Furthermore, conditions at the four treatment sites that had received the most comprehensive rehabilitation efforts suggested extra treatment had an affect as, on average, these sites had greater habitat diversity, deeper and narrower channels, and less fine sediments than did other treatment sites or control sites. The four intensive sites also had a lower mean gradient than either the other treatment sites or the control sites, suggesting intensive treatment attained the greater results despite having less stream power available. Overall, our results indicate that treatment did promote ecological responses, and that more intensive efforts yielded greater responses.

Our results indicated that channel condition in the 200 m upstream of a study site had negligible effects on the receiving reach. However, channel gradient within a study reach was a significant factor and site-specific gradient appeared to influence channel morphology, in-channel habitat diversity and sediment particle sizes, and riparian plant recruitment.

Our failure to frequently reject null hypotheses of no differences between restored and un-restored sites does not necessarily mean there were no differences, because many of our analyses suffered from low statistical power. Power calculations indicate that for the effect sizes we observed, we would have needed approximately 92 treatment and 92 control sites to attain a statistical power of 0.8, a standard value considered reasonable for ecological data. Among-site variability, such as that caused by local differences in channel gradient and different pre-treatment channel geomorphology, was the primary limitation on the ability for statistical analyses to resolve differences between treatment and control.

We used an extensive post-treatment experimental design because data on the pre-treatment channel and riparian conditions were not available. The low statistical power of our analyses coupled with the gradient effect on site condition demonstrates a weakness of the extensive post-treatment approach for effectiveness monitoring. Site to site variability can be very large and any variability unrelated to site treatment confounds interpretation of the affect of treatment. A replicated Before-After-Control (BACI) approach for effectiveness monitoring would allow for evaluating the effects directly associated with restoration efforts without the confounding influence of site-to-site variability. Therefore, we recommend that both funding agencies and restoration practitioners explicitly incorporate the requirement of collecting pre-treatment data into future streambank rehabilitation efforts, as failure to conduct scientifically rigorous post-treatment effectiveness monitoring precluded answering whether or not many of the restoration project goals were attained.

INTRODUCTION

Habitat management, defined herein to include creation, restoration, rehabilitation and enhancement, has emerged as a primary tool in the attempt to arrest the decline of

North America's native fishes and the loss of ecological services and foregone social opportunities caused by the declining abundance and quality of freshwater habitats (White 1996; Kauffman et al. 1997; Cooperman and Markle 2003; Shields et al. 2003; Bernhardt et al. 2005; Quigley and Harper 2006). Ebersole et al. (1997) eloquently express the conceptual foundation of habitat management as the "re-expression of habitat capacity" whereby the goal of 'treatment' is to restore the ability of an ecosystem to display its pre-disturbance range of habitats, functions and processes and be resilient to future disturbances and directional selection pressures. Ecosystems are hierarchically organized and forces originating at larger spatial-temporal scales constrain expression at smaller scales (Frissell et al. 1986; Minns et al. 1996), with the implication that fish habitat management should address watershed condition before addressing more local issues such as in-channel structure (Bisson et al. 1997; Beechie and Bolton 1999). However, the majority of habitat management efforts are directed within the active channel of target systems and applied at the 'reach' scale (Roni et al. 2002).

Habitat management entails applying one or multiple treatments and should be viewed as an experiment which necessitates post-treatment evaluation (Kondolf and Micheli 1995; Kershner 1997; Michener 1997; Palmer et al. 2005; Stem et al. 2005). Michener (1997) suggested the theoretical optimum for restoration effectiveness monitoring as, "long term monitoring of salient patterns and processes in adequately replicated control and experimental units at appropriate spatial and temporal scales using sound sampling design and statistical analyses." However, Michener concedes this optimum is rarely achieved and often unachievable. Specific to aquatic system, Koning et al. (1998) suggested that effectiveness monitoring should involve evaluating the structural integrity of any constructed elements plus their physical and biological consequences. Numerous other researchers also suggest ‘physical’ assessment involving geomorphic parameters such as channel width and depth, number of pools, and stream bed particle size distribution, because physical conditions are the template within which ecological processes and functions occur, they can respond more rapidly than dependant biological systems, and they can be readily measured with high precision (Leopold et al. 1964; Andrews 1982; Kondolf and Micheli 1995; Woodsmith et al. 2005). Failure to conduct scientifically rigorous post-treatment effectiveness monitoring precludes answering whether or not goals were attained and why, and equates to lost opportunities for learning (Michener 1997; Kondolf 1998).

Unfortunately, systematic effectiveness monitoring of freshwater habitat management efforts remains an exception not the rule (Kondolf 1998; Pretty et al. 2003; Reeve et al. 2006). Often, the limiting factors preventing efficient post-treatment evaluation originate from insufficient pre-project planning. For example, many projects fail to incorporate resources needed for effectiveness monitoring into the initial project budget and evaluation is therefore precluded due to lack of funds (Reeve et al. 2006). Similarly, when project planning fails to provide a clear statement of goals, effectiveness monitoring has no criteria on which to judge success or failure (Kondolf 1995; Palmer et al. 2005; Stem et al. 2005). Failures in pre-project planning may also cause failure to collect meaningful pre-treatment data, which precludes a Before-After-Control (BACI) experiment design or its derivatives (Minns et al. 1996; Michener 1997) and typically forces reliance on less powerful post-treatment between group comparisons, which is sensitive to the high levels of site to site variability that characterize natural systems

(Mellina and Hinch 1995; Bryant et al. 2004). Other times, effectiveness monitoring short-comings stem from sociological pressures that preclude an explicitly controlled experimental design such as the haphazard placement of treatments across the landscape owing to reliance on voluntary participation by private land owners or because time constraints for project completion and assessment prohibit evaluating slowly developing ecological responses such as tree growth or fish abundance change over multiple generations (Gowan and Fausch 1996; Opperman and Merenlender 2004; Bradford et al. 2005).

Treating an eroding river bank combines elements of in-channel restoration (e.g., current deflectors) with those of riparian restoration (stream bank grading, vegetation planting, livestock exclusion) but has been embraced as its own distinct treatment type (White 1996; Beechie and Bolton 1999; Bernhardt et al. 2005). Methods for bank stabilization and rehabilitation span a continuum of potential approaches, ranging from "hard" to "soft." Hard bank stabilization typically involves large quantities of rip-rap rock or other non-natural materials placed to protect a stream bank from base to top (Schmetterling et al. 2001). Although highly effective at preventing bank erosion, hard treatment precludes many natural processes such as lateral channel migration and the bi-directional flow of matter and energy between the active channel and its riparian zone. Conversely, "soft" treatment relies on using natural materials such as tree boles, root wads, and matting of inter-woven plant stems in an attempt to slow the rate of channel migration while promoting natural vegetation establishment and allowing geomorphic processes to maintain habitat formation and maintenance processes and riparian-channel interactions. A typical intermediate treatment might involve limited use of rock groins coupled with ample quantities of large wood plus bank revegetation.

Beginning in the early 1990s, Fisheries and Oceans Canada and its local partners initiated an eroding stream bank restoration program within the southern interior of British Columbia. Initial efforts focused on the Salmon River mainstem below the town of Faulkland (Figure 1), and as funding and social acceptance increased, the restoration program spread to multiple watersheds throughout the region. The three explicitly stated purposes for the bank restoration initiative included: Stopping erosion, thereby eliminating a significant source of fine sediments to the channel and protecting private property; To foster a stewardship mentality within the local land owner community, and; Increase native resident and anadromous salmonid production, particularly coho salmon, stemming from incremental reductions in summertime water temperatures and increased invertebrate production attributable to increased shading and organic matter inputs from riparian vegetation, altered channel width to depth ratio, and increased habitat complexity (S. Bennett, Fisheries and Oceans Canada, personal communication).

Prominent methods of the early efforts included soft treatments such as tree revetments with and without bank sloping, brush layers and brush matting with and without permanent anchoring, and tree retards. In the mid-1990s, Doyle and Sheng (1997) and Veller and Doyle (2001) evaluated the structural integrity and erosion control effectiveness of these early efforts and found that while the structures were effective at preventing erosion while intact, many had compromised structural integrity or were outright failures resulting from damage during high discharge events. An evolution of bank protection to "harder" methodologies ensued, and to date design considerations on the Salmon River and Bessette Creek have evolved through five "generations" of treatment styles encompassing tree revetments, root-wad, tree, and rock spurs, tree matting, post and brush revetments, bar stabilization revetments. All treatments usually had some amount of rock toes or rock side-slope protection. Modern bank stabilization efforts are often coupled with in-channel treatments such as rock-wood habitat complexing, Newbury weirs and artificial riffles (Salmon River Watershed Council 2004; S. Bennett, Fisheries and Oceans Canada, personal communication). All stream bank efforts involved some measure of bank sloping and riparian plantings along the outside bank of the channel. A recent survey of project structural integrity conducted in 2005 found that 100% of the 81 Salmon River projects constructed since 1997 had structural integrity ratings of adequate or better and that all of these projects were accomplishing their proximal goal of erosion control (S. Bennett, Fisheries and Oceans Canada, unpublished data). However, none of these structural integrity assessments evaluated ecological effects of bank restoration efforts.

In this paper, we report results of an extensive post-treatment effectiveness evaluation of stream bank stabilization and restoration efforts on the Salmon River and Bessette Creek of the southern interior of British Columbia, based on a comparison of stream channel and riparian vegetation condition and aquatic invertebrate abundance and biomass at 16 restored sites to 11 actively eroding "control" banks. Specifically, we address three questions: (1) has bank restoration had a measurable influence on in-channel or riparian condition or aquatic invertebrate productivity, (2) does the condition of a river channel upstream of a study site influence habitat conditions at the study site and (3) are there key variables that constrain the ecological response of a stream reach to bank restoration? We discuss our results in the context of the limitations of an extensive post-treatment experiment design for ecological systems and provide suggestions for future effectiveness monitoring efforts.

STUDY SITES

Both the Salmon River and Bessette Creek mainstems occupy the interior Douglas Fir, very hot dry biogeoclimatic zone of British Columbia although the terminal end of Salmon River near Salmon Arm B.C. is transitional to a warm-moist regime (Figure 1)(Lloyd et al. 1990). Annual mean precipitation ranges between 400-500 mm and growing season air temperatures average approximately 15^{o}C. Soils consist of a blanket of poorly sorted moraine deposits within a matrix of sand-silt-clay. Soils have limited fluvial re-working and are derived from volcanic bedrock (Lloyd et al. 1990). Dominant valley floor trees of undisturbed sites include interior Douglas Fir (*Pseudotsuga menziesii* var. *glauca)*, with Cottonwood (*Populus balsamifera* spp. *Trichocarpa)*, Paper Birch (*Betula papyrifera*) and willow (*Salix* spp.) common in riparian zones (Lloyd et al. 1990).

The Salmon River watershed lies on the interior plateau of south-central British Columbia, drains 1,510 km^2 and ranges in elevation from 2038 m to 349 m at its terminus with Shuswap Lake near the town of Salmon Arm, approximately 100 km east of the city of Kamloops. Downstream of the town of Falkland, B.C. (approximately 60 km upstream of Salmon Arm) the river flows across a low gradient valley floor and maintains a meandering plan-form till its entry to Shuswap Lake. Outflow from Shuswap Lake ultimate reaches the Pacific Ocean via the Fraser River. Timber harvest and irrigated agriculture-ranching are dominant land uses in the Salmon River watershed and almost all valley floor land is privately held in agriculture (Salmon River Watershed Society 2004). Miles (1995) estimated > 40% of the forest cover in the watershed has been harvested since the early 1900s, and that in the lower 60 km of the river 50% of the channel had either no riparian vegetation or a riparian band less than 1 channel width wide, that approximately 20% of the mainstem was actively eroding, and that the channel ranged from 11-211% wider than it was in the 1930s.

Sporadic local efforts for erosion control have occurred along the mainstem Salmon River for at least 15 years, but it was not until 1997 when the BC Interior South Resource Restoration Unit of Fisheries and Ocean Canada began providing technical and financial support for an eroding cut-bank stabilization and restoration effort that a systematic program of bank restoration has occurred. Nearly all restoration efforts have occurred on the mainstem river below Falkland. Since 1997, nearly the entire mainstem has been fenced for grazing control and approximately 100 eroding banks have been treated (M. Wallis, Salmon River Round Table, personal communication). Typical eroding bank treatment involves sloping the eroding bank, installation of a rock toe, use of wood and/or rock as current deflectors and/or bank protection via layering over the bank, and riparian plantings (mostly willows, *Salix* spp.) (M. Wallis and S. Bennett, personal communications). In-channel structures such as artificial riffles, boulder placement and wood debris jam creation have not been extensively used.

Salmon River discharge has been monitored continuously at Salmon Arm since 1974 (Water Survey of Canada station # 08LE021). The river has a snow-melt driven annual hydrograph, with peak flows typically occurring in May and June. Mean daily discharge over the period of record is 4.72 m^3/s, while mean discharge for July thru September is 1.87 m^3/s. The largest daily mean discharge recorded was 58.9 m^3/s in 1997, while in 2002 peak daily discharge reached 49.2 m^3/s, the 3rd highest value on record. Mean discharge during the field effort of this study was 2.09 m^3/s.

Bessette Creek starts near the town of Lumby, B.C. at the confluence of Duteau and Harris Creeks (Figure 1). Quantitative land use and impact data for the Bessette Creek watershed are not available but patterns and impacts are assumed to be similar to those of the Salmon River. Bessette Creek habitat restoration efforts started in 1989 with installation of 4 km of mainstem grazing exclusion fencing and associated riparian re-vegetation, and have been recently expanded to involve fencing nearly the entire length of the mainstem and treating approximately 20 eroding cut-banks. Restoration techniques are largely the same as those used on Salmon River with equal spatial-temporal variability in methods.

Detailed discharge records have been collect for Bessette Creek since 1974 (Water Survey of Canada station # 08LC042). Between 1974 and 2005, mean daily discharge has been 3.13 m^3/s, with July-September discharge averaging 1.86 m^3/s. The mean annual peak discharge was 25.5 m^3/s, and the largest recorded discharge of 39.7 m^3/s occurred in 1997. In 2002, discharge peaked at 32.3 m^3/s, the 9th highest annual peak during the 32 year period of record. Mean discharge during the field effort was 2.5 m^3/s.

For both the Salmon River and Bessette Creek, distribution of bank restoration efforts along the mainstem has been haphazard, owing to the vagaries of individual land owner participation.

METHODS

In July and August 2005, we assessed in-channel and riparian condition plus channel condition for 200 m immediately upstream at 16 cut-bank restoration sites and 11 actively eroding control sites. During the first two weeks of September 2005 we returned to each of the 27 study sites to collect aquatic invertebrate samples and determine local channel gradient.

Study site selection

To qualify as a treatment site for inclusion in our study a streambank restoration project needed to satisfy each of the following criteria: the habitat improvement treatment needed to be applied on the outside bank of a channel meander of a mainstem channel on the floor of an unbound alluvial valley segment, less than 10% of the as-built project could display evidence of physical failure, the project must have successfully accomplished the proximal goal of halting bank erosion, and the site must not have been directly influenced by civil engineering works, tributary inputs, site-specific treatments unrelated to bank restoration such as artificial riffles or channel re-configuration or possess unique geological features such as local clay lenses or other erosion resistant inclusions. Survival or failure of riparian plantings was not included in our 10% physical failure assessment criteria. For inclusion in the control group, a river bank need to satisfy all the applicable above criteria plus needed to be actively eroding, indicators of which included being near perpendicular to the water surface, un-vegetated, and evidence of recent bank slumping.

We provided the inclusion criteria to the technical coordinators of the Salmon River and Bessette Creek watershed roundtables and ask them to identify all treatment sites potentially meeting the criteria. Ultimately, 100% of the treatment projects (16 of 16; Table 1) the local authorities identified were included in the study and no sites not included in their list of potentials were added. Because treatment projects were located haphazardly along the river corridors, it was impossible for us to select sites to assure either random or systematic spatial distribution of treatment and control sites. Similarly, because each treated bank received a unique treatment prescription, it was impossible to have sufficient replicates within each treatment type to allow for effectiveness evaluation based on groupings of specific restoration techniques.

Site characterization

A study 'bank' was the portion of the outside meander bend that had either received treatment ('treatment') or was actively eroding ('control'). Each study bank occurred within a study 'reach,' defined as the portion of the channel lying between the upstream and downstream thalwag cross-over points bracketing a study bank (Figure 2). Length of a treatment bank was that portion of the outside bank of the meander bend that had received active restoration. Length of a control bank was that portion of the outside bank actively eroding. For study banks up to 100m long we established three transects and for banks >100 m in length we established five transects. Transects extended across the active channel perpendicular to the thalwag. Transect 1 was always positioned at the thalwag inflection point (i.e., point of maximum curvature). Transects 2 and 3 were at 10% of the bank length inside the downstream and upstream ends of the study bank respectively. When applicable, transects 4 and 5 were halfway between transects 1and 2 and 1 and 3 respectively. Each study site was assigned a unique alpha-numeric identification based on the first letter of the river segment it occurred within (B = Bessette Creek, L = lower Salmon River, M = middle Salmon River), a three letters descriptor of the property, a T or C for treatment or control and a number relating to how many study reaches existed at that property. For example, ‘B-Mar-T2’ identifies the second treatment site studied on the Marchant property within Bessette Creek. One control bank (B-Hem-C1) had the thalwag inflection point at the downstream end of the bank and therefore only transects 1 and 3 were established.

Of the 16 treatment sites, four (B-Mar-T1, B-Hem-T2, L-Wil-T2, M-Put-T2) were considered to have “state of the art” treatment, meaning they received notably more comprehensive prescriptions than other sites. Typically, these sites included “self-launching” rock spurs and trenched rock toes coupled with outward facing root wads >1 m in diameter embedded into a sloped bank, plus plantings from the waters edge to top of bank (Appendix A, Site Photos). We refer to these sites as the four ‘intensive’ treatment sites, as other treatment locations had either different rock toe constructions, lacked wood inclusions or used smaller diameter wood, or had less comprehensive bank sloping and/or riparian plantings. Two intensive sites were in Bessette Creek, and one each in the lower and middle Salmon River. We found eleven eroding banks in proximity to the 16 treatment sites to serve as the control group sample. Six treatment and four control sites occurred in the lower Salmon River, six treatment and three control sites were in the middle Salmon River, and four each treatment and control occurred in Bessette Creek (Figure 1).

Channel condition

For each study reach we conducted a habitat unit survey following classifications provided by Bisson and Montgomery (1996) plus noted if each habitat unit was a primary, secondary or tertiary feature based on definitions provided by Johnston and Slaney (1996). Length of a habitat unit was the long axis, width was the widest point perpendicular to length, and depth was the deepest point within the unit. We summed the number of 1^o, 2^o and 3^o habitat units within each reach and used the result as a measure of habitat diversity. We lumped riffles and runs into "fast water," classified pools as "slow water" and left glides as its own category and determined proportion of a reach's total surface area within each habitat class. The same investigator conducted all habitat unit surveys. Proportion of surface area as fast and slow water were highly correlated ($r = -.87$, $p < 0.000$), so surface area as fast water was eliminated from subsequent analyses.

We followed Harrelson et al. (1994) for establishing a local elevation datum point and developing elevation profiles along each transect. Elevation surveys extended from 2 m outside the top of bank on each side of the channel and elevation was recorded at every 0.5 m along each transect with supplemental readings taken at top of bank, bottom of bank, and water's edge. If any point along a transect was too deep for safe or accurate surveying, we noted as such, assumed depth equaled 2.0 m, and proceeded to the next accessible point. We determined the mean depth of a reach by taking the mean of all the individual water depths at each elevation survey data point. Depth at each elevation data point was the difference between the mean elevation of the waters edge (based on 2 readings per transect) and the elevation of a survey point along said transect.

We used a tape measure to determine bankfull, active channel, and wetted widths and visually estimated the angle of thalwag entry and exit from the bank and bank slope along each transect. Bankfull width was measured from top of bank to top of bank. Active channel width was measured as top of the outside bank to the top of the first distinct slope change along the point bar of the inside bank. Owing to the long history of channel over-widening in these systems plus our observations of large accumulations of living and dead plant material above the slope break, we interpreted the land between the slope break and the top of inside bank to be "incipient floodplain," which we considered a part of the riparian portion of the fluvial system. The mean depth to width ratio of a reach was derived from mean reach depth and mean wetted width, the latter determined from the three or five transect wetted widths within each reach. We used wetted width, not bank full or active channel width because summer time water temperatures are a primary management concern, and therefore it is the wetted portion of channel at base flow conditions that is the variable of concern. Mean depth and depth to width ratio were significantly correlated ($r = 0.90$, $p < 0.000$), so mean depth was eliminated from subsequent analyses. Height of the inside bank at 1 m from the waters edge was the mean of transect specific differences in elevation between the high point within 1 m of the waters edge along the inside bank and the elevation of the waters edge on the inside bank. Although channel gradient and height of the inside bank were significantly correlated ($r = -.63$, $p = 0.004$), both variables were retained because the mechanism linking the two variables is uncertain.

We visually estimated the proportion of stream bed particles falling within particle size classes defined by Harrelson et al. (1994) (organic matter, silt, sand, gravel, small or large pebbles, small or large cobble, boulder) within an approximately 1 m^2 portion of the stream bed underlying the thalwag and along the inside bank point bar of each reach (Figure 2). The thalwag sediment survey occurred at the downstream end of the study bank just upstream of where the thalwag separated from the bank. The point bar survey occurred 1 m inside the waters edge on the downstream side of the point bar. Sediment characterization was always done by the same investigator. From the particle size distributions, we lumped gravel and smaller particles (β-axis < 4 mm) into "fine particles" and all small pebbles and larger particles (β-axis >4 mm) into "coarse particles," and determined the proportion of fine and coarse sediments within the two sites per reach. Given the bivariate nature of fine or coarse size classes, we limited subsequent analyses to proportion fine particles in point bars and proportion coarse particles underlying the thalwag.

Channel gradient was determined as the difference in elevation at upstream and downstream ends of a study reach divided by the distance between the two points. Elevation of the upstream and downstream data points were determined at the water's edge along the inside bank of each study reach, and we used a tape measure to determine the distance between the two elevation points following the curve of the water's edge along the inside bank. Elevation and distance values were to the nearest cm.

Riparian Assessment

We used a modified line intercept technique (McDonald 1980) along the same transect lines as the elevation survey to assess the coverage of riparian vegetation on both the inside and outside bank of each study reach (Figure 2). On the outside bank of the channel, we initiated the vegetation survey at the top of bank. On the inside bend, the vegetation survey began at the edge of the active channel and progressed across the incipient floodplain and onto the historic floodplain when applicable. Riparian surveys extended 5 m from start points, were 1 m wide (0.5 m on each side of the transect line), and assumed to reach indefinitely upwards. For each tree and shrub that entered the survey plane, we recorded the species, length of the portion of each that was in the plane, and for trees we noted life stage based on visual estimate of height (seedling < .5 m, sapling >.5 & < 1.5 m. adult > 1.5 m). We considered seedlings to be recent natural recruits, and used the number of seedlings on the inside bank as a measure of recent natural recruitment.

We lumped the vegetation data from all transects on one side of the channel and determined proportional coverage by trees and shrubs, individually and combined. For example, in a case of a reach with three transects 'proportion outside bank covered by trees' equaled the sum of the lengths of trees encountered on the 3 transects of the outside bank of a reach divided by 15 m (e.g. three transects, each 5 m long). For simplicity, we refer to our proportional coverage data as "coverage." Given that plants overlapped along transects, it was possible to have coverage values exceeding 1.0. Because the presence of plants on outside banks could be attributable to opposing mechanisms of successful establishment resulting from restoration efforts or the undercutting of naturally occurring plants on a control bank, we did not include outside bank abundances in multivariate analyses.

Aquatic Invertebrates

During the first two weeks of September 1995, we revisited each study reach to sample aquatic invertebrates. We collected six Surber samples at 1 m inside the water's edge along the downstream portion of the inside bank point bar (e.g., same area as the point bar sediment surveys; Figure 2). Water depth was typically about 0.1 m at Surber locations. Initial plans called for sampling the cross-over riffle at the downstream end of each study reach, but riffle habitat proved to be very scarce. We chose the point bar location because it was the only geomorphic feature present at all 27 study sites that was accessible to the Surber sampler. To avoid bias in the specific spot sampled, we marked off a 10 m long section along the water's edge and used a random number table to determine which six of the 10 one m segments to be sampled. Surber samples were collected by one minute of hand agitation of the surface substrate coupled with rubbing all large rock and wood particles present within the 0.5 m^2 sample box and washing the material into the 400 *u*m mesh collecting net. We also collected one kick net (400 *u*m mesh) sample per site. Kick net sampling lasted two minutes per site, began immediately upstream of the Surber sample sites and progressed in an upstream zigzag between the banks. Whereas Surber samples focused on one habitat type, kick net samples integrated across multiple habitats. All invertebrate samples were field preserved in 10% buffered formalin.

In the laboratory, we randomly selected 2 of the 6 Surber samples from each site for analysis. Surber samples were washed with tapped water, elutriated to separate coarse and fine material, and all invertebrates were picked from the substrate under 3x magnification. Invertebrates were transferred to 70% EtOH and then identified to order.

We used the mean of the two Surber samples per site to determine total number of individuals, number of ephemoptera-plecoptera-trichoptera (EPT), total invertebrate wet weight and EPT wet weight. Because all possible pairings of the four Surber sample response variables were significantly correlated (all p values < 0.0433), only total invertebrate abundance was used in subsequent analyses. Kick net samples were sent to an external consultant for processing, and we used total number of invertebrates as the response variable.

Meso-scale Assessment

We assessed channel condition for 200 m upstream from the upper end of each study reach by walking the channel with a hip-chain and noting the length of any eroding banks, treated banks (e.g., rip-rap or other erosion prevention efforts), and overhanging vegetation on both banks of the channel plus the number of pools. The same investigator conducted all meso-scale surveys. Length of over-hanging vegetation was strongly correlated with length of bank eroding ($p = 0.0214$, $r = -0.44$) and was excluded from subsequent analyses.

Data Analysis

Prior to analyses, all variables were tested for univariate distributional normality and transformed as needed**.** Table 2 lists each of the variables used in subsequent analyses, data transformations applied, and the abbreviations used.

Preliminary results indicated that analyzing the three river segments separately supported the same conclusions as analyses of all data pooled into a single population, and therefore we only discuss results stemming from the pooled data set (Cooperman, unpublished data). The sole exception was aquatic invertebrate data, for which influence of river segment was explicitly accounted for in applicable analyses, as described below.

Differences between treatment and control sites

We used one way analysis of variance (ANOVA) to test for univariate differences between treatment and control sites for a suite of ecologically relevant variables derived from our field investigation data (Table 2). We also report the mean values of the four intensive treatment sites for each variable, but did not conduct statistical analyses due to small sample sizes. Because our use of ANOVAs was exploratory in nature and initial results yielded only marginal to non-significant p-values plus suffered from low statistical power (see below), we did not make corrections for multiple comparisons. As a follow-up to the univariate ANOVA tests, we executed *a-posteriori* power analysis following Winer (1971) for each of the variables to determine the sample size needed to attain statistical power of 0.8 based on a one-tailed test at $\alpha = 0.1$ and an effect size determined from the observed difference between treatment and control sites. We also used multivariate analysis of variance (MANOVA) to test for differences between treatment and control sites in environmental space. The eight dependant parameters entered into the MANOVA were HabUnits, P SA Slow, P SA Glide, D:W, Hgt IB, P TH Coarse, Cov T&S IB, #Recruits.

We also executed a series of F-tests following Zar (1974) to evaluate if treatment sites were less variable than control sites for each of the in-channel variables we had measured. For variables measured along transects (e.g., width, Hgt IB), we calculated a within-site variance from the 3 or 5 transect specific values followed by determining the among-sites variance as the mean of the within site variances of the 16 treatment and 11 control site. For variables with only one value per site (e.g., HabUnits, P SA Slow), we calculated among-site variances of the 16 treatment and 11 control sites. Our hypothesis was that treatment constrains ecological expression at a site by imposing a structure upon a reach, and therefore no F-tests were executed whenever treatment site variance for a variable was greater than that of the control sites. The F statistic was compared to critical F values for a one tailed test at $\alpha = 0.1$ with 15 and 10 degrees of freedom.

To test for differences in the erosion-deposition environments of treatment and control sites, we compared the slopes of regression lines resulting from regressing P TH coarse on gradient and P PB Fines on gradient. We tested the regression solutions for unusually influential points based on Cook's distance and DEFITS values and eliminated offending cases as needed.

For the aquatic invertebrate data, we used both multi-factor ANOVA and one-way ANOVA to evaluate differences in Surber sample invertebrate abundance between treatment and control sites. For the multi-factor ANOVA, treatment or control (2 levels) and river segment (3 levels) were the two main effects, response variable was # Inverts and gradient was entered as a co-variate. To allow for follow up F-tests of variances and *a posteriori* power analysis, we conducted a one-way ANOVA for differences between treatment and control sites for invertebrate abundance, following the methods described above. For the kicknet samples, we used multifactor ANOVA with the conditions described above.

All statistical tests were evaluated for significance at the level of $p = 0.1$ owing to the probability that large within group variability, a condition reasonably expected to occur during an extensive post-treatment assessment of ecological conditions, might obscure subtle but real among group differences. Further, the higher p value criteria reduces the probability of a type II error, which would probably be more costly than a type I error given its influence on funding availability and future restoration efforts. For all ANOVA analyses, we tested for distributional outliers defined as values greater than 3 times the inter-quartile range from either the 25^{th} or 75^{th} percentile value, and excluded outliers.

Meso-scale influences on reach condition

We used multiple linear regression to explore the contribution of the meso-scale variables to four response variables measured within study reaches. For each regression, independent variables were the three meso-scale parameters (length of bank treated, length of bank eroding, # of pools) and the dependent response variable rotated between Cov T&S IB, D:W, Hgt IB, and P TH Coarse. Each preliminary solution was evaluated for data points with greater than 3 times the mean leverage or unusually large DFITS values and repeated with the offending cases excluded.

Key variables constraining expression

We constructed a correlation matrix to explore bivariate correlations. We also used non-metric multidimensional scaling (NMS; PC-Ord v. 4.0) to explore relationships amongst variables in multivariate environmental space. We used non-parameteric NMS because it is well suited to multivariate data where components variables are on dissimilar scales, such as measures of length and proportional abundances. Further, NMS solutions are based on ranked distances between sites and therefore relax the assumption of a linear relationship between independent and dependant variables, making it well suited to expose relationships amongst variables. The NMS solution was also used in exploring for differences between treatment and control sites based on overlaying data of a second matrix.

The NMS ordination incorporated all treatment and control sites (n=27) and involved two data matrices. The first matrix included eight measures of habitat condition: HabUnits, P SA Slow, P SA Glide, D:W, Hgt IB, P TH Coarse, Cov T&S IB, # Recruits, plus gradient. The second matrix contained two descriptive parameters: Treatment or Control (coded as T =1, C = 0) and river segment (coded as L = 1, M = 2, B = 3). For the two cases where sediment particle size data was not available (L-Wil1, M-Fel1), we substituted the mean of the applicable treatment-segment combination. Prior to ordination, we applied a general relativization by columns and tested for multivariate outliers (>2 standard deviations from the multivariate mean of habitat condition). No cases were identified as outliers and therefore both matrices had 27 cases, one for each study site. We used Euclidean distance and the 'slow and through' auto-pilot setting to execute 50 runs with real data with random start configurations. We used 30 runs per tested dimension Monte Carlo simulation with randomized data to determine the number of dimensions to use in the final solution. We selected a three dimension solution as the best fit, and rotated the final 3D solution to maximize correlation with gradient along the first axis.

RESULTS

Differences between treatment and control sites

No data values of the in-channel, riparian or aquatic invertebrate data sets were excluded as outliers, so all between group comparisons were based on 16 treatment sites and 11 control sites.

Two of the in-channel response parameters differed between treatment and control groups (Table 3), in that treatment sites had narrower wetted widths and higher inside banks than did control sites. All other in-channel response parameters, plus channel gradient, were not statistically different between treatment and control sites (Table 3), although treatment sites tended to have more habitat units, less surface area as either slow water or glides, and fewer fine particles along the edge of the inside bank point bars than did control sites, and these differences were largest when treatment to control comparisons were limited to only the four intense treatment sites (Table 3). Coverage of shrubs on the outside bank of study reaches was the only riparian parameter to differ between treatment and control sites although treatment sites had greater mean values in all riparian categories (Table 4). Treatment and control sites did not differ in multivariate environmental space (MANOVA w/ 8 and 18 df, $F = 0.45$, F critical = 1.02, $p = 0.460$).

Two of the ten (D:W, Hgt IB) variables assessed for amount of between site variability were less variable among treatment sites than among controls (Tables 3 and 4). Additionally, P SA Glide and P TH Coarse tended to have less within group variability at treatment sites than at control sites. The other six measured variables were more variable among treatment sites than among control sites (Tables 3 and 4).

A posteriori power analysis indicated our statistical analyses of in-channel condition and riparian coverage typically suffered from low power (Tables 3 and 4). Excluding the two parameters for which statistically significant treatment verses control differences were found, the estimated number of treatment and control sites we would have had to have sampled to find significant differences at the observed effect size ranged from 53 to 17,450 (median of 92) cases in each group.

The distribution of sediment particle sizes was marginally related to reach gradient, but thalwag sediments and point bar sediments had different responses to bank restoration. Within both treatment and control groups, the proportion of coarse sediments underlying the thalwag increased with increasing channel gradient and the two groups had highly similar regression line slopes (Figure 3a). Conversely, treatment sites had decreasing abundances of fines with increasing gradient, but control sites had increasing fines with increasing gradient (Figure 3b).

Gradient was a strong influence on invertebrate abundance in the Surber samples (bivariate regression: $r = 0.53$, $p = 0.0046$) but not the kick net samples (bivariate regression: $r = 0.22$, $p = .2686$) and was a significant co-variate only in the Surber sample multifactor ANOVA. For the Surber sample data, invertebrate abundance was different between river segments but presence or absence of bank restoration did not affect invertebrate abundance [multi-factor ANOVA #Inverts: Gradient w/ 1 df, F ratio 3.92, $p = 0.0604$; Segment w/ 2 df, F ratio = 2.95, $p = 0.0735$; Treatment or Control w/ 1 df, F ratio 0.01, $p = 0.9409$]. Differences between segments was due to Bessette Creek supporting lower abundances than either segment of the Salmon River [mean (st error): Bessette 296 (123), lower Salmon 552 (104), middle Salmon 466 (121)]. In the kick net data, invertebrate abundances were different between river segments and tended to be higher in control sites relative to treatment sites [multifactor ANOVA #Inverts: Gradient w/ 1 df, F ratio 0.75, p = .3950; Segment w/ 2 df, F ratio = 7.72, p = .0029; Treatment or Control w/1 df, F ratio 5.70, p = .0260). Difference between segments was attributable to middle Salmon River sites having more invertebrates than the other locations [mean (st error): Bessette Cr. 9,469 (4,448), lower Salmon 18,202 (3,783), middle Salmon 38,113 (4,408)], and control sites had more invertebrates than treatment sites [Control 27,554 (3,639), Treatment 16,303 (3,036)].

In order to reach statistical power of 0.8 for a one tailed test at α = 0.1, we would have to had Surber sampled 9,003 treatment sites and an equal number of control sites. The variability of aquatic invertebrate abundances did not differ between treatment and control sites, although treatment sites had lower within group mean variance.

Meso-scale influences on reach condition

We found no evidence that the combination of length of bank treated, length of bank eroding and number of pools in the 200 m of river channel upstream of study reaches influenced any of the four dependant variables evaluated. The four full model linear regression solutions (n=27, no study site excluded due to unusual influence) had correlation coefficients of 0.00 (twice: mean D:W, P TH Coarse), 0.03 (Hgt IB), and 0.24 (Cov T&S IB) and p values ranged from 0.1001 to 0.4653. The low p-value of 0.1001 (Cov T&S IB) was associated with a single influential data point (L-Tur-T2), exclusion of which altered the solution to p = 0.6401, r = 0.00. In each of the four solutions, no independent variable had a component p value <0.2000.

Each of the four full model solutions were confounded by one to six unusually influential data points. In only one case (Hgt IB), did exclusion of influential points generate a significant relationship: $\log_{10}$ (Hgt IB) = 2.47 – 1.02 $\log_{10}$ (length treated) – 0.69 $\log_{10}$ (length eroding) – 0.69 $\log_{10}$ (# pools +1), n=21, r^2_{adj} = 0.64, p = 0.0280, all component p values < 0.05.

Key variables constraining expression

We entered 11 variables into the correlation matrix, yielding 55 bivariate comparisons (Table 5). Fourteen of the 55 pairs had correlation p values < 0.05 and an additional nine pairs had correlation p values <0.1, attesting to the high level of inter-relatedness amongst our response variables. P TH Coarse had the largest number of strong relationships, with 6 out of 10 pairwise comparisons having correlation p values less than 0.1. Channel gradient, HabUnits, D:W, Hgt IB, and #Inverts each had 5 out of 10 pairings with correlation p values < 0.1.

A 3-dimensional NMS solution was the best fit for our data (Figure 4, Table 6). NMS required 41 iterations to produce a stable solution with final stress of 6.2 and instability of 0.005. These are considered very acceptable values indicative of a stable solution (McCune and Grace 2002) The cumulative variance explained was 95.7% with axis 1 contributing 52.9%, axis 2 – 29.9% and axis 3 – 12.9%. A Monte Carlo test of 20 runs with randomized data indicate our solution had lower stress than expected by chance (mean stress of Monte Carlo test: 15.0, test of difference between Monte Carlo and actual data, p = 0.0476). Axis one loaded with five parameters with correlations stronger than +/- 0.5 along the axis: gradient, #Recruits, HabUnits, P TH Coarse, and Hgt IB. Axis two also had 5 parameters load with correlations stronger than +/- 0.5: D:W, P SA Slow, #Recruits, Cov T&S IB, P TH Coarse. Only gradient and P SA Slow loaded with correlation values >+/- 0.5 on axis three.

Treatment and control sites are highly intermingled and well distributed throughout the ordination solution, although control sites are scarce in the upper right and lower left quadrants of the axis 1 by axis 2 plot. Three of the four intensive treatment sites (L-Wil2, B-Mar1, M-Put2) occupy similar positions along both axes one and three, while the other, B-Hem2, is well separated from the others along axis 1 and slightly separated along axis 3. The four intensive treatment sites are well dispersed along axis 2.

DISCUSSION

The response variables we selected for our effectiveness monitoring focus on the physical habitat template such as channel geomorphology and biological integrators of ecological processes such as plant and invertebrate abundances, both of which are widely recognized as important indicators of restoration success (Kondolf and Micheli 1995; Woodsmith et al. 2005). We did not directly evaluate ecological processes or functions (Ruiz-Jaen and Aide 2005; Ryder and Miller 2005) because there was no reason to believe that primary production, nitrogen fixation, energy loading or other rate based processes would differ at the scale of 100s of meters, which encompasses the spatial segregation of treatment and control sites in our study. To the best of our knowledge, this is the first comprehensive effectiveness evaluation of a systematic program of stream bank restorations based on methods other than exclusive use of rip-rap.

We found limited evidence that restoration of eroding river banks has had a substantive influence at the scale of the river reach. Restored and un-restored sites did not significantly differ in habitat structural diversity or flow velocity distribution, depth to width ratio, substrate particle sizes, or in the abundance and wet weight of aquatic invertebrates. However, as discussed below, treated sites tended to have more shrubs along the outside bank, higher inside banks, and narrower wetted widths, and each of these differences may be indicative of processes that will lead to notable system recovery given time. Furthermore, conditions at the four "intensive treatment" sites suggest extra treatment has an affect as, on average, intensive sites had greater habitat diversity, deeper and narrower channels, and less fine sediments than did other treatment sites or control sites. The four intensive sites also had a lower mean gradient than either the other treatment sites or the control sites, suggesting intensive treatment attained the greater results despite having less stream power available. These results suggest that treatment does promote ecological responses, and that more intensive efforts yield greater rewards.

As indicated above, the absence of wide ranging large differences between treatment and control sites does not mean the restoration program has not yielded benefits. First, the treatments appear to have been highly effective at preventing further bank erosion, the proximal goal of the restoration program. Furthermore, treated sites had much higher abundances of shrubs along outside banks than did untreated sites, owing to the abundance of planted willows, providing strong evidence that the re-vegetation of eroding banks, a primary goal of the restoration program, has had initial success. The large majority of shrubs used in the restoration are locally identified as Pacific willow (S*alix lasiandra)* although specimens used are probably a cultivated hybrid (S. Bennett, Fisheries and Oceans Canada, personal communication). The established willows may provide many benefits to the local aquatic system including energy subsidies via organic matter and terrestrial invertebrate inputs, and protection against future erosion via stabilization from the root network. However, the growth form of these “Pacific willow” consists of a near ground base with multiple thin shoots extending upwards to an estimated maximum height of 3 m. The absence of a large central bole, a spreading canopy, and exceptional height suggest the willows will not serve as a significant in-channel geomorphic feature upon recruitment to the active channel nor are they likely to provide significant shading to the stream channels (Roni et al. 2002). Bank lining willows may contribute to soil development and seed trapping, and thereby enhance natural revegetation processes including establishment of trees, via promoting particle settlement during high water events (Cooperman 2005). However, at present, treated and untreated sites did not differ significantly in the abundance of trees, which could provide the desired long term benefits willows fail to provide. Restoration practitioners may need to incorporate promoting tree establishment along with willows.

Another notable difference between treated and untreated sites was in the height of the inside bank lining the wetted channel. High land surfaces may be less prone to inundation and therefore less susceptible to disturbance during high discharge events, and this reduced disturbance regime may promote increased survivorship of colonizing vegetation (Hupp and Osterkamp 1985; Friedman et al. 1995; Cooperman and Brewer 2005). We did not find a difference in number of natural recruits or total vegetative coverage on the inside banks of treatment reaches, but treated sites did have greater means and variances than did untreated sites and there was a significant relationship between height of the inside bank and plant abundance (Cooperman, unpublished data), suggesting a relationship. The mechanism linking bank stabilization and height of the inside bank is unclear, but may be related to bank treatment halting the prograding of the inside bend point bar into the active channel thereby promoting stabilization of the point bar deposit and vertical accretion during subsequent high water events. If vegetative colonization of the stabilized point bars proceeds, it is likely that a positive feedback between point bar stabilization, vertical accretion and plant colonization would develop, yielding a narrower active channel and a wider floodplain (Cooperman and Brewer 2005).

One other parameter that substantively differed between restored and control reaches was wetted width, but issues surrounding interpreting this difference illustrates the importance of collecting pre-treatment baseline data. The channels at treatment sites were significantly narrower than those at control sites, and the most intensive treatment sites had the narrowest mean value. Narrow channels was a goal of restoration, as narrow channels have less water surface area and should receive proportionally larger benefits from local shading once riparian vegetation develops, resulting in less thermal heating of the water column. However, the absence of pre-treatment data makes it impossible to determine if the observed treatment-control differences were a result of local bank accretion stemming from sediment deposition along the treated banks, or simply a result of the restoration effort dumping a large volume of material into the active channel. If narrowing was a result of bank accretion, then we can realistically expect the growth of the bank to continue until an equilibrium width is attained. If, however, the width difference is simply equivalent to the width of the rock and wood added to the channel as part of treatment, then there should be no expectation of future channel width adjustment. Channel width is one of the most reliable and indicative hydraulic variables for characterizing watershed condition, and it can have tremendous influence on many other parameters such as riffle-pool spacing, magnitude of thermal heating, and depth of the water column (Andrews 1982). It would be beneficial to our effectiveness monitoring to have better insight into the mechanisms responsible for stream width dynamics.

Both Surber samples and the kick net samples suggest stream bank restoration has not increased local invertebrate abundance, although there were notable differences between the two data sets. It appears that the Surber data set attributed between site differences to site gradient while the kick net samples assigned greater value to presence or absence of bank restoration. It is unclear what caused the difference in parameter loading. Not withstanding the differences between methods, control sites had invertebrate abundances equal to or greater than the restored sites. In the case of this particular study, the results of the surber samples should be used more conservatively than the kick net data, owing to the surbers being placed in a depositional environment (point bar deposits) not the riffle environments for which they are intended. Provided riffle habitat is available, it is difficult to argue in favor of one sampling method or the other. Surber samplers target a specific, easily identifiable, and highly repeatable habitat with a quantifiable sample effort. Kick nets integrate more habitat types, but the effort that goes into each sample has greater potential for investigator bias. Given the data available from this study, it is difficult to make a suggestion of one technique over another for future efforts. However, our results suggest that kick nets were less influenced by site specific gradient, and therefore this gear may be better suited for cases where experimental design and data analysis approaches do not allow for accounting for between site differences in gradient.

The different regression line slopes between channel gradient and proportion of fine sediments in point bar deposits between treatment and control sites was unexpected, especially since there were no differences between the groups in terms of particle sizes underlying the channel thalwag. A likely explanation for the decreasing percentage of fine particles with increase stream gradient is that treatment effectively diverts energy away from the outside bank of the river bend and onto the inside bank, thereby winnowing away fine particles. The greater the gradient, the more stream power is diverted. This raises the question of whether or not bank stabilization exports an effect, either upstream via channel down-cutting or downstream via increased erosion. Such effects have been well documented for rip-rapped banks (Schmetterling et al. 2001). We had hoped to evaluate this potentiality, but the large site to site differences in the upstream and downstream channels proved too large to disentangle from treatment effects (Cooperman, unpublished data).

Despite not being able to quantify whether or not treatment exported an effect, our meso-scale assessment indicated channel condition in the 200 m above a location has limited influence on the receiving reach. Only in the reduced model (n=21) was height of the inside bank significantly predicted by the upstream parameters of length of channel treated, length eroding, and number of pools. Given that greater than 20% of the available data points needed to be excluded before a significant relationship emerged, we feel this result is not highly robust and suggest caution in its interpretation.

The relative paucity of univariate differences between treatment and control sites was also reflected in the multivariate results. There was no evidence from our multivariate analyses that restoration substantively changed the morphology or ecology of the treated reaches. Of the eight habitat measures used in the NMS analysis, four were strongly associated with gradient, the one parameter we measured that should be independent from presence-absence of bank restoration treatments. These four parameters, (HabUnits, P TH Coarse, Hgt IB, # Recruits), incorporate primary measures of reach scale habitat diversity, sediment particle sizes, channel geomorphology and riparian condition, and as such capture much of the ecologically relevant dynamics of interest. The strong correspondence within the NMS solution between gradient and these four variables suggests site specific gradient is at least partly responsible for site specific "ecology." The role of gradient as a 'master parameter' is reinforced by the relationship between aquatic invertebrate abundance and gradient. Hence, site to site variability associated with changes in local channel gradient confound our ability to determine the magnitude of change related to treatment efforts. Only by holding gradient constant between treatment and control sites would we be able to fully disentangle the contribution of restoration from that of gradient. In total, the gradient effect illustrates a weakness of the extensive post-treatment approach for effectiveness monitoring, as site to site variability can be very large and any variability unrelated to site treatment confounds interpretation of the affect of treatment.

An additional observation reinforces the importance of pre-treatment data. During the course of our field efforts we noted that in a number of cases, the active channel of the Salmon River had moved away from local treatment infrastructure. For example, at

L-Tur1 we noted that a large gravel bar presently separates the wetted channel from a line of rip rap and wood revetments. It seems likely the gravel bar emerged as a result of the active channel migrating away from its old bank. However, we could not quantify the local change in channel width or other parameters, because we could not discern if the channel was deeper as a result of a width adjustment or if the movement away from the treatment was matched by erosion of the far bank. Comparably, at treatment site M-Roy2, we noted that upwards of 3 meters of highly vegetated riparian zone sediments, approximately 20 cm deep, separate a now "high and dry" rock toe and wood carpet from the active channel. As above, without either pre-treatment data or as-built post-construction data, it is impossible to determine if this is an ecologically successful restoration (narrower and deeper channel, floodplain construction), if the channel simply abandoned one location via eroding another (a restoration failure), or if the channel has locally down-cut resulting in elevating riverbed sediments above the new water elevation (attaining a new equilibrium gradient).

A more optimal approach for effectiveness monitoring would be to evaluate site condition before treatment is applied, immediately after completion of the treatment, and at some time after completion. When coupled with appropriate control sites, the Before-After-Control (BACI) approach allows for a rigorous evaluation of the effects directly associated with restoration efforts, and the confounding influence of site-to-site variability is removed.

Our failure to reject the null hypothesis of no differences between restored and un-restored sites does not provide definitive evidence no differences between the two groups exists, as many of our analyses suffered from low statistical power (Peterman

1990; Steidl et al. 1997). Among-site variability, such as that caused by local differences in channel gradient and different pre-treatment channel geomorphology, is a primary control on the ability for statistical analyses to resolve differences between samples. Although *a posteriori* power calculations are of limited utility when executed at the same effect size used as the original statistical test (e.g., p values and power will be inversely proportional), retrospective power analysis is useful for determining the number of samples that would have been needed to attain a statistically significant difference at the observed effect size. In the stream bank restoration effectiveness monitoring effort described here, our power calculations suggest we would have needed approximately 184 sample sites (92 each, treatment and control) to attain a statistical power of 0.8, a standard value considered reasonable for ecological data (Peterman 1990; Steidl et al. 1997). Obviously, our samples of 16 treatment and 11 untreated sites falls far short of this mark, suggesting our failure to reject the null hypothesis of "no difference," may be a lack of resolution, not a true lack of difference. We declined to determine our statistical power for alternative effect sizes, as doing so for ecological or fisheries data can be problematic (Bryant et al. 2004). For example, what magnitude of change in channel width to depth ratio is significant, or how many additional aquatic invertebrates need to be present for a significant change in fish food abundance to be realized? Unless these target values are explicitly enumerated during the restoration planning phase, any post-hoc selection would be arbitrary. Unfortunately, time and budget limitations coupled with reliance on a post-extensive evaluation landed us squarely in a statistical grey zone, whereby definitive conclusions can not be reached.

The question remains, therefore, should Fisheries and Oceans Canada and their partners continue to pursue stream bank restoration in the southern interior of British Columbia as a means of attaining “healthier” rivers and improved salmonid productivity? If so, should treatments be limited to ecologically simple methods such as rip-rapped banks that effectively control erosion, or does the addition of bank sloping, tree revetments, rock current deflectors, wood carpets, etc. merit the extra effort due to ecological benefits? Answering these questions falls within the realm of resource management by public agencies and quasi-public organizations, and therefore is beyond the scope of this paper. However, our results suggest the extra treatment does has measurable short term beneficial effects such as increased riparian vegetation abundance, and holds the potential for additional benefits into the future. Additionally, within the southern interior, the “softer” treatment methods that use less rock than traditional rip-rap treatments are also less expensive, as rock for rip-rap is the most expensive item used in bank restoration (S. Bennett, personal communication).

Another consideration that may dampen the enthusiasm for using rock intensive site treatments is the long term implication of using rock toes or rip rapped banks as common elements of treatment prescriptions. In a meandering single thread channel, lateral scour pools with some amount of overhanging bank frequently occur along the outside bank of river bends, and can be prime fish holding habitat for both resident and anadromous salmonids (Henderson 1986; White 1991; Rabeni and Jacobson 1993). We did not observe many lateral scour pools nor any overhanging banks in either Salmon River or Bessette Creek. Bank restoration clearly has not produced any, probably at least in part, owing to the use of rock toes as part of the bank stabilization protocol as these non-eroding elements are designed to prevent bank undercutting. Hence, rock toes provide a necessary short term benefit by allowing local vegetation the opportunity to establish and increase the root strength of the bank, but with the long term cost of preventing lateral scour pool formation. Interestingly, the southern interior habitat management unit of Fisheries and Oceans Canada has recently moved away from using rock toes and now relies on rock groins where >90% of the rock is buried into the bank and large wood inclusions that project into the wetted channel are intended to provide over-head cover. A long term evaluation of the trade-offs associated with rock-toes appears merited.

Palmer et al. (2005) suggested five criteria by which to judge if restoration is successful, including did a predefined guiding image exist for the effort (i.e., statement of purpose and goal), has the river's condition been improved, has a more self-sustaining system emerged, construction should not have caused lasting harm, and both pre- and post treatment assessments must be done. By these criteria, at least two elements appear to require further considerations before the southern interiors stream bank stabilization effort is embraced as a success. First is the unanswered question of whether or not use of rock toes as a common element in treatment prescriptions is prohibiting the formation of lateral scour pools and what cost this might have to the long term salmonid productivity of the system. However, this question may be rendered moot with the recent evolution away from rock toes. Second, is the systematic and rigorous collection of pre-treatment data, as it would allow for a before-after-control experimental design for effectiveness evaluation and adaptive management that controls confounding influences in a way that extensive post-treatment evaluations do not.

The criteria of Palmer et al. (2005) omit the importance of less tangible benefits stemming from river restoration efforts, including increased social awareness of the linkage between land uses and ecological consequences and evolution of a stewardship mentality. A companion study that evaluated social perception of the role of stream bank restoration found compelling evidence to suggest the local streambank treatments evaluated here have had impacts far beyond the scale of local river reaches (Branton et al., in preparation).

At present, we can not evaluate if the bank restoration projects have had a measurable influence on salmonid abundances. However, the success or failure of the southern interior stream bank restoration program at promoting increased anadromous salmonid production will not be solely determined by a series of local bank stabilizations, regardless of how ecologically successful they are. The productive capacity of a watershed is largely constrained by land use patterns and impacts originating at much larger spatial scales than that of local river banks or reaches. Only by addressing the source of watersheds problems such as altered hydrology and sediment budgets, excessive water withdrawls, and loss of riparian vegetation at all elevations within the watersheds, coupled with local active restoration that returns the potential productive capacity to a system plus sufficient time for recovery to manifest, will improved salmon production be realized (Beschta 1991; House 1996; Beechie and Bolton 1999; Pretty et al. 2003). However, an otherwise healthy watershed that suffers from extensive bank degradation may also fail to attain the desired fish productivity levels. As such, local bank restoration efforts in the Kamloops region may be necessary, but not sufficient, to increase salmon production.

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Table 1. Description of the 16 treatment sites.

Site	Description
L-Viv1	Suggested date of construction 1994. Rip rap with riparian fencing and plantings, modest bank sloping.
L-Viv2	Rip rap, mannings (individual thin logs that project into active channel), fencing and riparian plantings.
L-Wil1	Multi-thalwag channel with locally steeper than average gradient. Rock wall with wood carpet, limited projection into active channel. Fencing.
L-Wil2	Rip rap with a wood carpet and log bundles projecting into channel, bank sloping, riparian fencing and plantings. Intensive treatment site.
L-Tur1	Rock toe plus mannings held in place with overlaying rock, riparian fencing and planting.
L-Tur2	Exclusively rip-rap with subsequent riparian fencing. Probably one of the older treatments. Large birch and alders sit on outside bank, unclear if planted or natural.
M-Fel1	Large rock boulders and wood parallel to flow with riparian fencing and planting. Channel gradient is locally steep.
M-Rot1	Rock toe with wood inclusions, bank grading, riparian fencing willow plantings.
M-Put1	Rip rap bank with willow plantings on both banks. Incomplete riparian fencing.
M-Put2	Rock toe and current deflector with wood inclusions. Riparian fencing and willow planting. Intensive treatment site.
M-Roy1	Bank grading with rock wall and overlaying conifer boles cabled in place. Site is very dry, no evidence of riparian plantings.
M-Roy2	Two distinct segments to treatment. Upper portion involving rock toe and willow plantings. Channel has moved away from rock. Lower segment involves bank sloping and rip rap. There is a large log jam at the downstream end of the project.
B-Mar1	The youngest treatment site. Sloped bank, rip rap toe with large boulders on bank, large log groins projecting into channel at regular intervals, lots of willow and tree plantings and riparian fencing. Intensive treatment site.
B-Hem1	Lower half of a long 270^{o} river bend. An older site. Concrete chunk rip rap with pilings driven into bank and projecting into channel complexed with logs parallel to flow at water surface.
B-Hem2	A relatively recent site. Similar in structure to Marchant T-1, with addition of wood pilings driven into ground with attached logs laying parallel to flow at water edge. Intensive treatment site.
B-Buf1	“Preliminary” treatment involving piles of rock at base of cut bank with modest amount of 10 cm diameter logs wedged into rock piles. Rock is not dug into bank.

Table 2. Variables used in data analyses, transformations applied to attain univariate normality, and abbreviations used in text. All variables are the mean condition observed within a reach. All log transformations are to base 10. "NA" means not applicable.

Group	Variable	Transformation used	Abbreviation
In-Channel	Number of habitat units	NA	HabUnits
	Proportion of surface area as slow water	Arc-sine	P SA Slow
	Proportion of surface area as glide	NA	P SA Glide
	Wetted width of the active channel	NA	Width
	Depth to width ratio	NA	D:W
	Height of the inside bank at 1 m from the water's edge	Log (X)	Hgt IB
	Gradient	NA	Gradient
	Proportion of all sediments along the inside bank point bar in the 'fine' size class	NA	P PB Fine
	Proportion of all sediments underlying the thalwag in the 'coarse' size class	NA	P TH Coarse
Riparian	Coverage of trees on outside bank	Log (X + .01)	Cov T OB
	Coverage of shrubs on outside bank	NA	Cov S OB
	Coverage trees on inside bank	Log (X +.01)	Cov T IB
	Coverage shrubs on inside bank	Log (X+.01)	Cov S IB
	Coverage of trees and shrubs on inside bank	Log (X +1)	Cov T&S IB
	Number of seedling trees on inside bank	Log (X+1)	#Recruits
Aquatic Invertebrates	Total abundance in Surber Samples	Log (X)	#Inverts
	Total abundance in kick net samples	NA	#Inverts-KN
Meso-scale	Length of bank treated	Log (X+1)	NA
	Length of bank eroding	Log (X+1)	NA
	Number of pools	Log (X+1)	NA

Table 3. Means (+/- 1 standard deviation) plus one-way ANOVAs, F-Tests of variance and power analysis results for channel condition comparisons between study groups. Means and standard deviations shown are for untransformed parameters, but statistical analysis were done on transformed data, as reported in text. $n_{Treatment} = 16$, $n_{Control} = 11$, and $n_{intensive} = 4$, except for the 2 sediment parameters, where two treatment sites were not sampled and therefore $n_{treatment} = 14$.

Parameter	Treatment mean (st. dev)	Control mean (st. dev)	ANOVA p	F ratio :p value+	Desired n	Intensive mean (st. dev)
Gradient	.00186 (.0014)	0.00232 (.0020)	0.488	--	105	.00162 (.00093)
HabUnits	7.2 (3.13)	7.0 (2.24)	0.866	NA	72	8.3 (2.4)
P SA Slow	0.11 (.15)	0.14 (.11)	0.410	NA	92	0.08 (.09)
P SA Glide	0.37 (.25)	0.48 (.30)	0.317	1.44 : >.25	55	0.32 (.17)
Wetted width	11.78 (2.69)	13.57 (2.07)	0.076	NA	18	11.10 (1.70)
mean D:W	0.040 (.01)	.034 (.02)	0.313*	3.50 : .025>p>.01	53	0.045 (.02)
Hgt IB	0.35 (.22)	0.22 (.19)	0.099	3.57 : .025>p>.01	21	0.30 (.15)
P PB Fine	0.44 (.39)	0.57 (.35)	0.409	NA	75	0.41 (.37)
P TH Coarse	0.59 (.33)	0.59 (.39)	0.954	1.40 : >.25	17,450	0.53 (.25)

* Samples had unequal variances that could not be corrected by data transformation. + F test evaluated as a one tailed test at $\alpha = 0.1$ with 15 and 10 degrees of freedom and evaluated against F-critical = 2.06. 'NA' means treatment sites were more variable than control sites. An F-test of gradient was not executed as it is an independent parameter not likely to be affected by site management. 'Desired n' is the number of cases separately needed in treatment and control groups to attain power = 0.8 assuming the effect size observed in this sample is equal to that of the population. 'Intensive' refers to the four of the 16 treatment sites that received the most comprehensive treatment.

Table 4. Comparison of riparian vegetation coverage between treatment (n=16) and control (n=11) sites. Parameter name abbreviations are explained in the text. Mean (standard deviation) values are for untransformed data, but statistical tests were conducted on transformed values as specified in the text. F-tests and sample size determination were only conducted for the two inside bank riparian parameters. Desired n is the number of cases needed in each treatment and control groups to attain a statistical power of 0.8 at the effect size observed within the sample data.

Parameter	Treatment Mean (st. dev)	Control Mean (st. dev)	ANOVA p value	F-test p value *	Desired n [+]
Cov T&S IB	0.604 (.52)	0.582 (.32)	0.9261	NA	4,504
#Recruits	6.69 (11.1)	5.55 (6.9)	0.8260	NA	902
Cov T OB	0.235 (.31)	0.150 (.27)	0.3412	--	--
Cov S OB	0.585 (.31)	0.147 (.21)	0.0004	--	--

* F-tests were conducted to determine if treatment sites were significantly less variable than control sites, 'NA' means treatment sites were more variable.

Table 5. Correlation matrix for in-channel, riparian and invertebrate variables. For all combinations n = 27. Numbers below the unity line are correlation coefficients and numbers above the unity line are p-values. Abbreviations are explained in Table 2.

	Gradient	HabUnits	P SA Slow	P SA Glide	Wet Width	D:W	Hgt IB	P PB Fine	P TH Coarse	Cov T&S IB	# Inverts
Gradient	--	.042	.763	.223	.380	.028	.004	.693	.093	.169	.005
HabUnits	.39	--	.264	.046	.951	.734	.061	.162	.099	.234	.069
P SA Slow	.06	.22	--	.016	.019	.144	.597	.172	.593	.055	.081
P SA Glide	-.24	-.39	-.46	--	.924	.076	.499	.439	.967	.057	.945
Wet Width	-.18	.01	.45	-.02	--	.937	.179	.011	.136	.971	.249
D:W	-.42	-.07	.29	-.35	.02	--	.091	.480	.016	.218	.002
Hgt IB	-.63	-.37	-.11	.14	.27	.33	--	.219	.017	.034	.268
P PB Fine	-.08	-.28	.27	-.16	.48	.14	.24	--	.020	.498	.150
P TH Coarse	.33	.32	-.11	.01	-.29	-.46	-.45	-.44	--	.575	.010
Cov T&S IB	-.27	-.24	-.37	.37	.01	-.24	.41	.14	-.11	--	.728
# Inverts	.53	.35	-.34	-.01	-.23	-.57	-.22	-.28	.49	.07	--

Table 6. Correlation coefficients for the NMS solution (n=27) following rotation to maximize correlation of gradient along axis 1. Abbreviations are defined in Table 2.

Parameter	Axis 1	Axis 2	Axis 3
Gradient	.799	-.065	.525
HabUnits	.617	-.016	-.172
P SA Slow	.232	.642	-.505
P SA Glide	-.488	-.492	.415
mean D:W	-.367	.640	-.433
Hgt IB	-.614	-.015	-.396
P TH Coarse	.636	-.529	.234
Cov T&S IB	-.265	-.571	-.074
# Recruits	.754	-.655	-.238

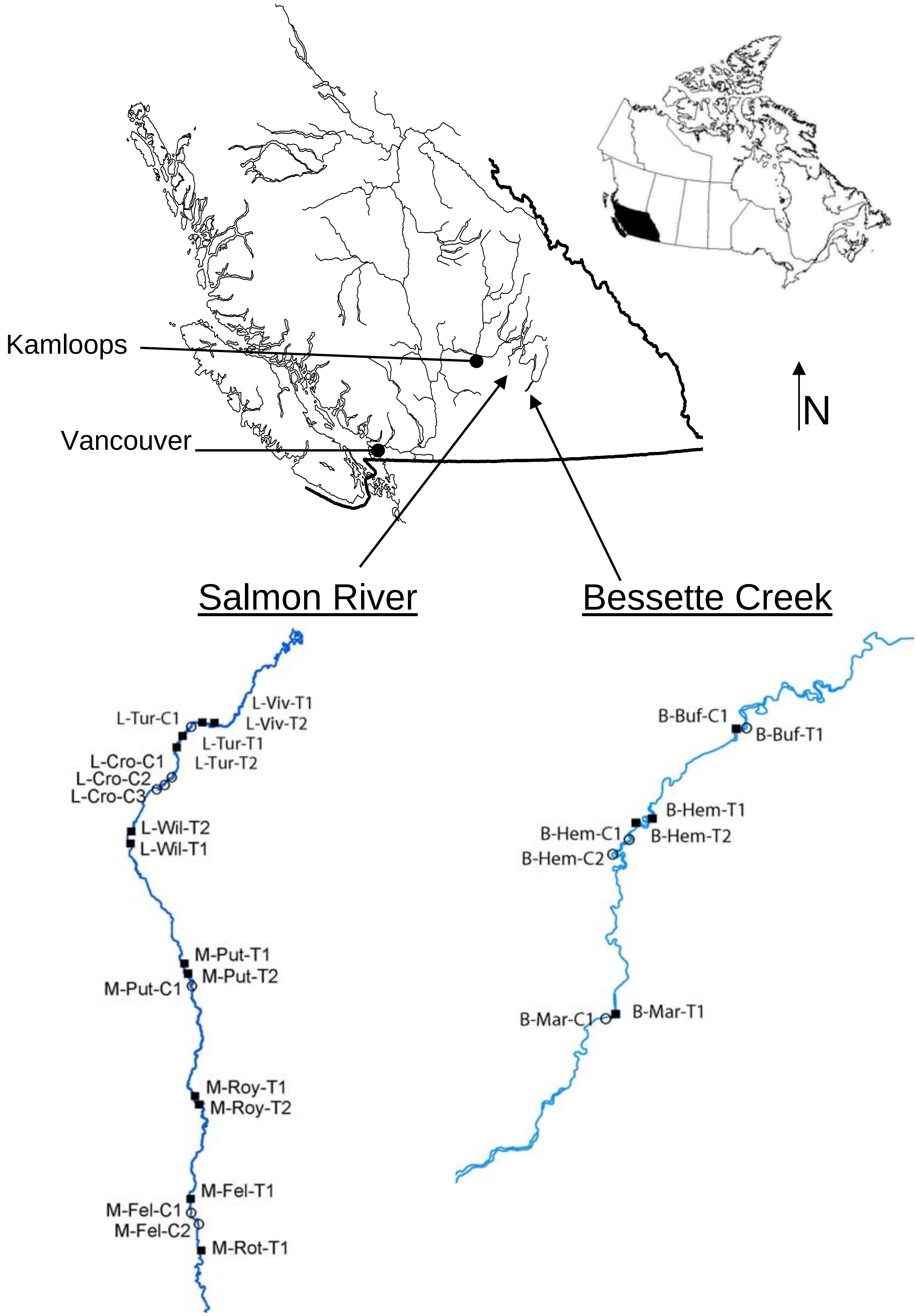

Figure 1. Location of the Salmon River and Bessette Creek in the headwaters of the Thompson River sub-basin of the Fraser River watershed and the distribution of study sites within the two rivers. Scale for the Salmon River is 1:100,000 and for Bessette Creek is 1:50,000. Treatment sites are filled squares and control sites are open circles. Study site nomenclature is described in the text.

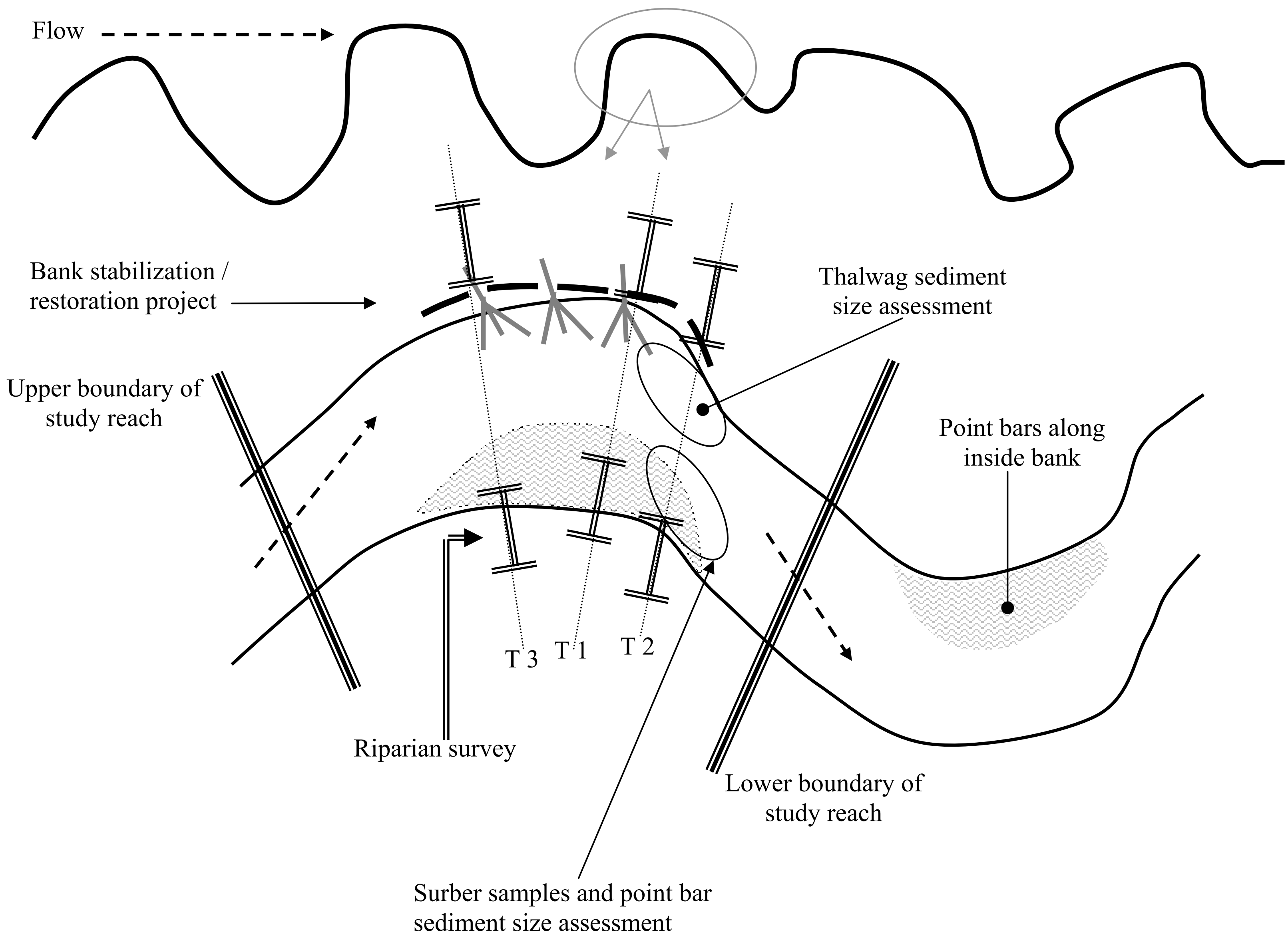

Figure 2. Schematic representation of field work effort at a treatment site depicting location of three transects used for channel elevation and riparian surveys, upper and lower reach boundaries, and sediment and invertebrate sampling locations. Diagram is not to scale.

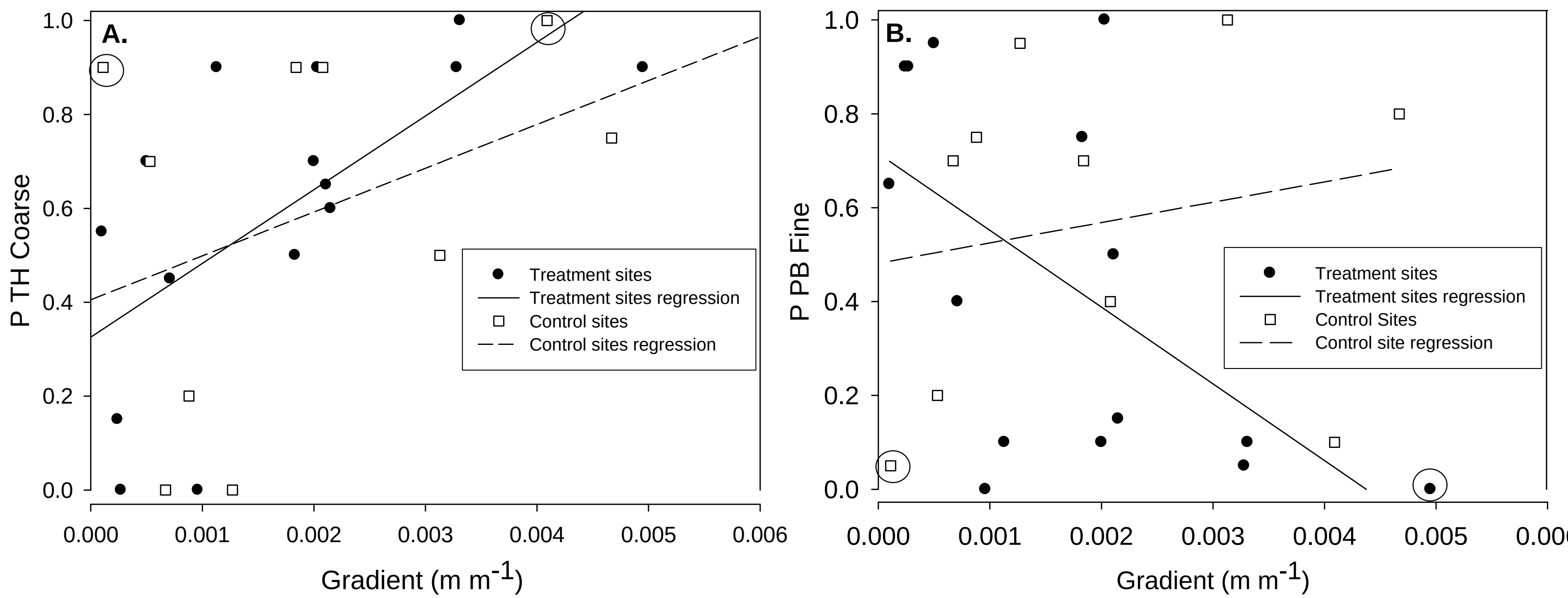

Figure 3 a and b. Comparisons of regression lines between treatment and control sites for stream bed particle size distributions as a function of channel gradient. Panel A is proportion of thalwag sediments in the coarse size class (P TH coarse). Model r^2 = 19.4, model p value = .0573 and p for differences in slope = 0.4837. Panel B is proportion of point bar sediments in the fine size class (P PB Fine). Model r^2 = 14.7, model p value = .0982, and p value for differences in slopes = 0.0494. For both panels, $n_{treatment}$ = 15 and $n_{control}$ = 10. The four circled data points had unusually large DFITS values, but their exclusion did not affect interpretation and therefore they are included.

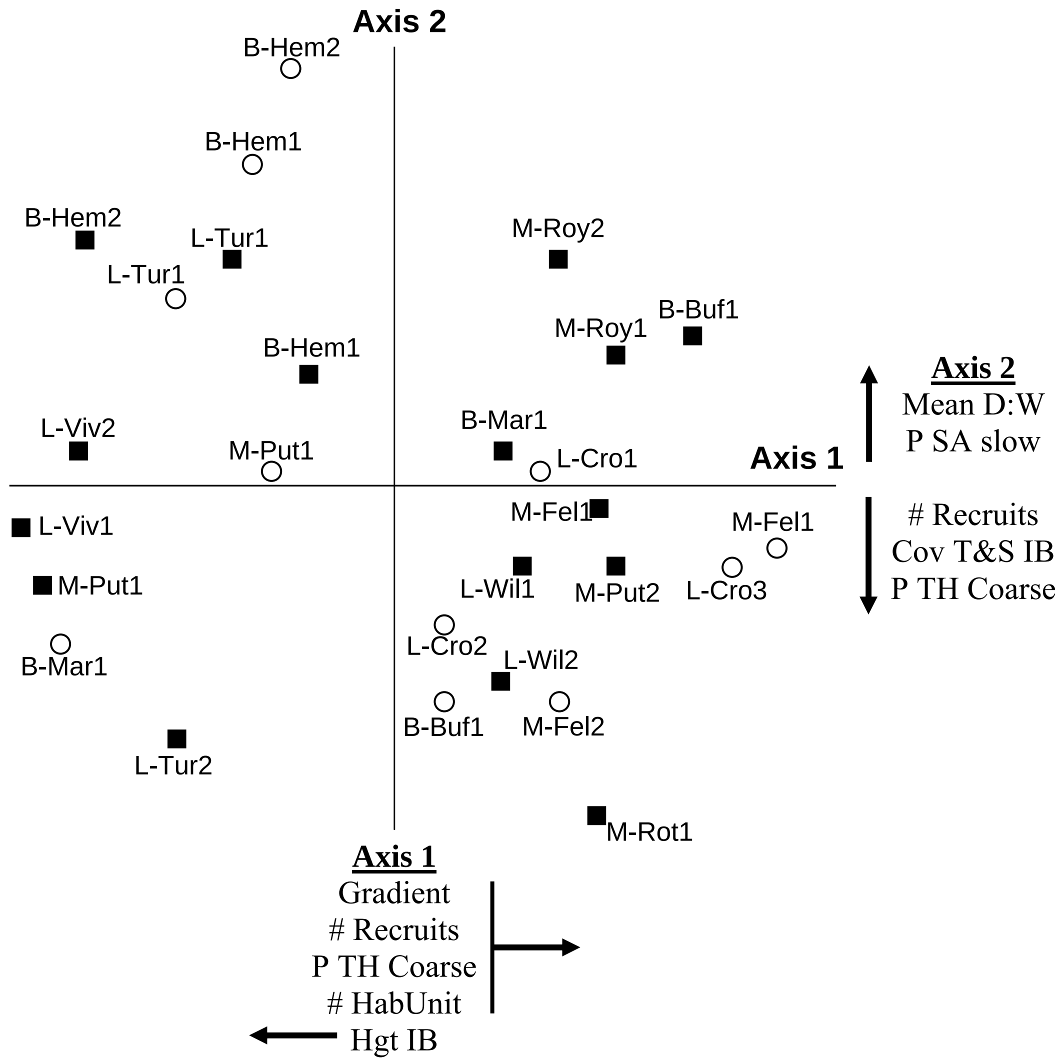

Figure 4. Three dimensional non-metric multidimensional scaling ordination rotated to maximize correlation of gradient on axis 1. Axis 1 explains 52.9% of variance, axis 2 29.9% and axis 3 12.9%. Axes are labeled with parameters that had correlation scores ≥± 0.500 and arrows point in direction of increasing values. All correlation scores are giving in Table 6. Solid squares are the treatment sites and labels combine a 1 letter code for river segment (B= Bessette Creek, L = lower Salmon River, M = middle Salmon River) followed by alpha-numeric site name. The four intensive treatment sites are B-Mar1, B-Hem2, L-Wil2, and M-Put2.

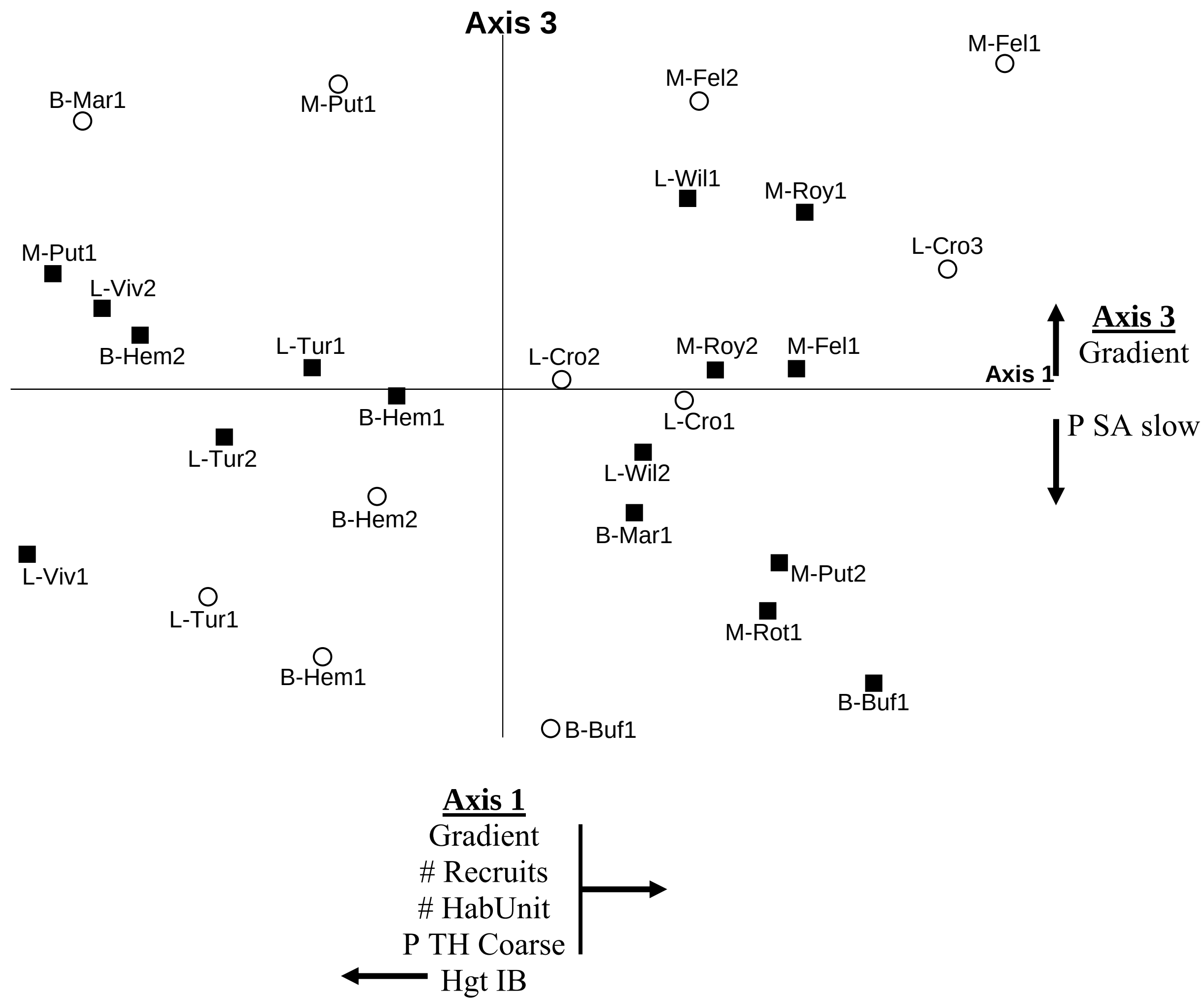

Figure 4 cont.

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