THE INFLUENCE OF SAMPLING DESIGN ON THE CHARACTERIZATION OF IN-STREAM SALMONID HABITAT

by

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ABSTRACT

Pacific salmon have endured widespread population extirpations with some estimates at nearly one third of historical populations. In the western coterminous United States Pacific salmon no longer inhabit upwards of 40% of their historical freshwater range. Reintroducing Pacific salmon has therefore become a common conservation effort. An early step in evaluating potential reintroductions includes quantifying the available habitat, however the quantification, and interpretation of the habitat can be influenced by the sampling design and methods chosen. Reach-based sampling designs have been used extensively to collect fisheries related data; however, few studies have examined how reach-based inferences may be biased, a particular concern given the non-random distribution of factors such as woody debris and the magnitude of site-to-site variability. To address this concern, I collected reach-based habitat data continuously within streams. I then used simulations to resample the streams which were delineated into discrete reaches. During simulations I applied simple random, simple random with unequal probability, and generalized random tessellation stratified sampling designs and chose three habitat attributes that are commonly collected in stream habitat surveys, thought to be important factors for Pacific salmon survival, and expected to be distributed differently across the riverscape. My goal of identifying potential bias and precision under these sampling designs was achieved by summarizing simulations and comparing simulated results across streams, attributes, sampling designs and ultimately the census derived estimate of an attribute. My results indicate the extent of bias and levels of precision varied not only across habitat metrics but also across streams. My analyses suggest the use of reach-based approaches, particularly with low sampling efforts, can result in substantially different estimates of habitat characteristics and erroneous estimates of habitat carrying capacity of fishes.

CHAPTER ONE

INTRODUCTION

Pacific salmon (*Oncorhynchus* spp.) are of enormous economic, cultural, and recreational value as well as playing pivotal roles in the ecosystems which they inhabit (Cone 1996, National Research Council 1996, Schindler et al. 2003). Seven species of anadromous Pacific salmonids in the genus *Oncorhynchus* occur in North America (*O. clarkii clarkii*-coastal cutthroat trout, *O. gorbuscha*-pink salmon, *O. keta*-chum salmon, *O. kisutch*-coho salmon, *O. mykiss*-steelhead, *O. nerka*-sockeye salmon, and *O. tshawytscha*-chinook salmon), with geographic ranges spanning the north Pacific Rim. Yearly commercial landed catch values for five Pacific salmon species have averaged over 500 million dollars for the past 10 years (NMFS 2015). First nations people from Monterey, California to the tip of Alaska share in their various mythologies and traditions, a reverence for salmon which has been sustained for thousands of years. The ability of coastal communities to effectively exploit natural resources such as salmon is believed to have been a significant contributor to societal growth, distribution, and success (Arnold 2008). From carved artwork to theater the rich tradition of salmon is ever present in the cultures of western North America.

Pacific salmon also play an important role in the functioning of aquatic and terrestrial ecosystems throughout their range. Millions of adult salmon returning to natal streams transport nutrients from the marine environment to freshwater and terrestrial habitats in the form of eggs, excrement, and carcasses (Cederholm et al. 1999, Gende et al. 2002, Schindler et al. 2003). Salmon have been shown to affect abundance and growth of aquatic biota (Chaloner et al. 2004, Wipfli and Baxter 2010) and riparian vegetation (Drake et al. 2005, Helfield and Naiman 2011), subsidize a suite of predators and scavengers, including whales, otters, minks, bears, birds, and countless invertebrates (Ben-David et al. 1997, Hilderbrand et al. 1999, Ford et al. 2010) and enhance ecosystem function (Levi et al. 2012). The loss of salmon populations and life history diversity are of concern, especially when the benefits of salmon subsidies for freshwater and terrestrial ecosystems are taken into account.

Pacific salmon exhibit a wide degree of genetic and life history diversity (Waples 2001, Waples et al. 2008). Juveniles typically grow in freshwater but vary in the time spent from a few weeks to up to 3 years upon which they migrate downstream to saltwater environments (Groot and Margolis 1991, Quinn 2005). In saltwater, salmonids can spend anywhere from a few months to 7 years before returning as adults to spawn in their natal streams (Groot and Margolis 1991, Quinn 2005). Pacific salmon display strong natal homing fidelity (Keefer and Caudill 2013) which can limit gene flow between populations from different selective environments and enhance diversity among populations (Neville et al. 2006). Such diversity, coupled with phenotypic plasticity can enable Pacific salmon to rapidly colonize new habitats (Crozier et al. 2008).

Pacific salmon have endured widespread population extirpations with some estimates at nearly one third of historical populations (Gustafson et al. 2007). Perhaps outpacing the loss of local salmon populations is the loss of genetic and life history diversities that have occurred over the past half century (Hughes et al. 1997). In the western coterminous United States, Pacific salmon no longer inhabit upwards of 40% of their historical freshwater range (National Research Council 1996). Efforts to increase and establish populations of Pacific salmon are common goals of natural resource agencies and organizations.

The influence of physical habitat on the condition, distribution, and abundance of stream biota is well documented (Minshall et al. 1983, Riley and Fausch 1995, White et al. 2011). Along with other factors, widespread habitat degradation and fragmentation have contributed to the declines of salmon stocks throughout the Pacific Northwest (Nehlsen et al. 1991, Lichatowich 1999). Therefore, protecting and enhancing stream habitat is often an objective towards increasing and re-establishing populations.

Assessing the watershed potential is a critical step in enhancing degraded salmonid populations (i.e., increasing low populations or reintroducing extirpated populations) and requires quantifying the available habitat and relating that habitat to species-specific needs (Anderson et al. 2014). However, the quantification, interpretation, and estimated effect of the habitat on reintroduced salmonids can be influenced by both the method used to collect habitat data, and the scale at which the data is collected. The scale or strata at which we view or sample the stream can play an important role in our estimates and results of habitat quality and suitability (Frissell et al. 1986). Contrasting a landscape perspective and a reach perspective can help illustrate the complex continuum of a stream (Fausch et al. 2002), and the effect spatial scale can have on the interpretation of a stream habitat (McMillan et al. 2013).

Stream ecologists have long recognized the influence of landscapes on streams (Cummins 1974, Vannote et al. 1980). Factors influencing stream habitat (e.g., increasing drainage area, decreasing gradient, broader riparian zones) can change dramatically from headwaters downstream towards the mouth (Montgomery and Buffington 1997, Thomson et al. 2001) which promotes a heterogeneous arrangement of riverine habitat (Frissell et al. 1986). Differences in the spatial arrangements of stream habitat suggest our ability to accurately and precisely quantify habitat status to inform the potential of salmon populations (sensu Anderson et al. 2014) is likely influenced by the chosen sampling design.

With limited resources to recover populations, it becomes imperative that decisions are made with the best knowledge (i.e., habitat quality and quantity) of the system targeted for reintroduction or enhancement. Initiating actions to restore populations prior to understanding the extent and quality of available habitat could result in misallocated resources and lost opportunities (Cochran-Biederman et al. 2014). The extent of available stream networks and resource constraints facing fisheries managers commonly result in habitat monitoring designs consisting of short, discrete sampling reaches of 50 to 500 meters and low to moderate sampling rates (Kaufmann et al. 1999, Larsen et al. 2001, 2004, Kershner et al. 2004). Reach-based sampling is commonly used in small (Dolloff et al. 1997) and large-scale (Kershner et al. 2004) habitat assessments and monitoring programs, yet there is uncertainty in how this sampling grain effectively characterizes the existing habitat especially when stream habitat attributes exhibit non-uniform (i.e., patchy) distributions across the riverscape (Torgersen et al. 1999, Baxter and Hauer 2000, Wohl and Cadol 2011). Under-represented habitats or habitat attributes in sample reaches represent challenges when extrapolating from the reach-scale to larger scales. For example, Dolloff et al. (1997) found no cascade type habitat units using a reach based sampling design, although cascade units were present and observed during the census survey. While census based estimates of habitat will be the most accurate, and reach based estimates less so, it remains unclear how much data are needed to accurately and precisely quantify the status of habitat throughout a stream.

Differences in the spatial arrangements of stream habitat coupled with the fact that factors influencing these habitats and individual habitat attributes (e.g., substrate, woody debris) change along the continuum from headwaters towards the mouth challenge the accuracy and precision of stream habitat assessments. Additionally, several studies have demonstrated considerable variability in habitat attributes across sites, and site-to-site variability can explain a large proportion of the variance in the data (Urquhart et al. 1998, Larsen et al. 2004, Al-Chokhachy et al. 2011, Anlauf et al. 2011). Consequently, the potential assumptions, bias, and precision of different sampling designs and intensities likely has profound effects on our ability to characterize the status and quality of stream habitat. This thesis seeks to address this uncertainty to evaluate how decisions in habitat sampling design can impact our characterization of stream habitat. Specifically, my objectives were to evaluate design bias and precision under common reach based sampling designs. To address these objectives, I continuously sampled 16 streams quantifying different habitat attributes that are important during various stages of Pacific salmon life-histories. I then used computer simulations resampling these data to evaluate how the precision and bias of reach based approaches varied under different sampling designs and sampling rates. The results of my research are critical in providing resource practitioners with information to more effectively guide habitat assessments and more accurately inform recovery efforts for Pacific salmon.

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CHAPTER TWO

USING CONTINUOUS SURVEYS TO EVALUATE PRECISION AND BIAS OF HABITAT INFERENCES FROM DIFFERENT REACH-SCALE SAMPLING ALTERNATIVES

Contribution of Authors and Co-Authors

Manuscript in Chapter 2

Author: Christopher Clark

Contributions: Helped conceive study design, implemented study, collected and analyzed data, and authored manuscript.

Co-Author: Robert Al-Chokhachy

Contributions: Helped conceive study design, provided analysis guidance and edited manuscript.

Co-Author: George Pess

Contributions: Edited manuscript.

Co-Author: Thomas McMahon

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Abstract

Accurately estimating stream channel characteristics is essential for managing and restoring populations and aquatic ecosystems. Likewise, there is a need to understand the tradeoffs between bias and precision when choosing a sampling design, particularly given limited resources available for assessments and monitoring. Reach-based sampling designs have been used extensively to collect fisheries related data; however, few studies have examined how reach-based inferences may be biased, a particular concern given the non-random distribution of factors such as woody debris and the magnitude of site-to-site variability. Here, we used continuous habitat surveys to census stream attributes in tributaries in the upper Lewis River, WA. We delineated completed continuous stream surveys into reaches and then used bootstrapping to create simulated outcomes of different sampling designs including simple random sampling with equal probability, simple random with unequal probability, and a generalized random tessellation stratified design with a goal of identifying potential bias and precision under these sampling designs. Our results indicate the extent of bias and levels of precision varied not only across habitat metrics but also across streams. Our analyses suggest the use of reach-based approaches, particularly with low sampling efforts, can result in substantially different estimates of habitat characteristics.

Introduction

The influence of physical habitat on the condition, distribution and abundance of stream biota is well documented (Hynes 1970, Vannote et al. 1980) and understood to be a major factor limiting aquatic species and communities (Riley and Fausch 1995, Torgersen et al. 1999, Rosenfeld et al. 2000, Bowerman et al. 2014, Riebe et al. 2014). Indeed, habitat degradation resulting from anthropogenic activities is one of the factors leading to the declines of native salmonids in western North America (Nehlsen 1991). Recognition of degraded habitat as a limitation to salmonid populations has led to considerable changes in policy (e.g., Washington State Salmon Recovery Act 1999) accompanied by rehabilitation of these habitats (NRC 1992, Roni et al. 2008) and to a lesser extent increases in habitat monitoring programs (Larsen et al. 2004, Stevens and Olsen 2004, Kershner et al. 2004, Roni et al. 2015).

The progression of Pacific salmon recovery requires accurate habitat assessments to direct targeted habitat restoration efforts, understand the effects of land management, and inform feasibility and limiting factors assessments for target species (Roni et al. 2002, Roni et al. 2008, Anderson et al. 2014). However, differences in the spatial arrangements of stream habitat coupled with the fact that factors influencing these habitats change along the continuum from the headwaters towards the mouth (Vannote et al. 1980) challenge our ability to accurately and precisely characterize habitat throughout the stream. Several studies have demonstrated considerable variability in habitat attributes across sites and that site-to-site variability can explain a large proportion of the variance in the data (Urquhart et al. 1998, Larsen et al. 2004, Al-Chokhachy et al. 2011, Anlauf et al. 2011). Consequently, it is likely that the assumptions, bias, and precision of different sampling designs and intensities (e.g., sampling fraction of 25 % or every tenth unit (Hankin and Reeves 1988) have profound effects on our ability to characterize the status and quality of stream habitat. Such spatial heterogeneity suggests our efforts to accurately and precisely characterize the status and quality of stream habitat may be influenced by sampling design and intensity.

Reach-based sampling designs are often necessary when faced with resource constraints (e.g., budgets, time) and large sampling domains (e.g., Kershner et al. 2004) which can limit the ability to continuously sample the entire stream network. Discrete (e.g., 500-m reach-scale) sampling designs using probability sampling are extensively used to quantify habitat, species abundance, and redd densities (Hankin and Reeves 1988, Kershner et al. 2004, Mayfield et al. 2014). Probability sampling attempts to limit bias and provides a theoretical foundation for inferences beyond the sampled units. Accuracy and precision of inferences made beyond the sampled population of reaches are a concern and several researchers have highlighted the potential for ‘extrapolation’ error (Hankin and Reeves 1988, Dolloff et al. 1997, Williams et al. 2004). The prevalence of site-to-site variability suggests reach-based inferences may be confounded by the high heterogeneity characteristic of stream networks.

Surprisingly few studies have examined the potential bias of reach-based inferences in stream habitat assessments, a particular concern given the magnitude of site-to-site variability. Dolloff et al. (1997) compared a census sampling design (basin visual estimation technique; BVET) to a reach-based sampling design (representative reach extrapolation technique; RRET) and found that habitat estimates (estimates of large wood, number of habitat units, and area) were more accurate using the BVET sampling design. However, Dolloff et al. focused mostly on differences in estimates of channel units and types across approaches and not how inferences and precision of estimates vary across habitat attributes, sampling designs, or sampling rates. Here we use continuously collected habitat data to empirically evaluate the influence of various types of reach-based samplings designs on habitat characterization not only across streams, but also across habitat attributes. Our goal was to evaluate the accuracy and precision of reach-based sampling approaches in habitat assessments. Given the influence of sampling designs and sampling intensities in habitat assessments, we evaluated three different sampling designs (simple random, simple random with unequal probabilities, and generalized random tessellation stratified) and a range of sampling rates (e.g., 5%, 10% of length) in our analyses. We chose three habitat attributes that are commonly collected in monitoring and assessment studies to illustrate our results.

Methods

Study area

Our study occurred in 10 tributaries to the Lewis River basin in southwest Washington, USA (Table 2; Fig. 1). All of the tributaries exist between Merwin and Swift dams and data were collected as part of a study to identify the overall status of physical habitat in the context of anadromous salmon and steelhead reintroductions (Al-Chokhachy et al. 2016). Together these streams total 32.3 km of stream habitat (Fig. 1) and are typical of streams found in the Pacific Northwest. The climate within the Lewis

River basin consists of warm dry summers and cool wet winters. The mean annual temperatures in the region are approximately 11 °C with annual precipitation occurring primarily as rain between October and May and averaging 287 cm per year (U.S. Climate Data 2014) (Table 2). Streamflows are typical of the western Cascade Mountains with highest mean flows occurring during early spring (March and April) and low flows occurring late summer and early fall (August and September). In all 10 streams igneous rocks are the dominant geologic formation which includes both basalt and andesite. The vegetation within the tributary basins consists of relatively young, multistaged forest dominated by western hemlock (*Tsuga heterophylla)* and Douglas fir *(Pseudotsuga menziesii)* with conifers mixed with broadleaf species in the riparian areas, including red alder (*Alnus rubra)* and bigleaf maple (*Acer macrophyllum)*. The primary land use has been forestry including logging and construction of logging roads.

Stream surveys

During summer baseflows of 2014 we conducted continuous habitat surveys (i.e., a census survey design) from each tributary mouth upstream to the anadromous migration barrier. Our field methods followed established, reach-based protocols used in the Pacific Northwest (CHaMP 2013) to ensure consistency with regional approaches but applied them in a continuous manner (*see below)*. Habitat data were collected at the channel geomorphic unit (CGU) scale using the two tiered (i.e., fast vs. slow water) hierarchical approach outlined by Hawkins et al. (1993). Each CGU was georeferenced along the stream network with its corresponding habitat attribute measurements.

Although our data were continuously collected, we used the georeferenced channel unit information to collate the data into distinct reaches. Specifically, we delineated reaches at the channel unit boundaries when the summed length of consecutive units was equal to or greater than 20 times bankfull width. This length has been found to ensure multiple channel units occur within a reach and is commonly used in habitat monitoring programs (e.g., Kershner et al 2004). We used the individual reaches within each stream to define the sampling frame (i.e., the population of reaches from which subsamples are taken) for our analyses.

We focused on habitat attributes that are (*i*) commonly agreed upon as important to Pacific salmon, (*ii*) frequently sampled in monitoring programs and (*iii*) whose spatial distributions vary across the riverscape. Habitat attributes included large woody debris density (LWD), residual pool depth (RPD), and grain size in pool tailouts (GS).

The importance of stream habitat diversity is well-established for aquatic biota (Lonzarich and Quinn 1995). Structural elements such as woody debris enhance habitat diversity through its control on geomorphic processes (Bisson et al. 1987, Abbe and Montgomery 1996, 2003, Larsen et al. 2001, Brooks et al. 2004) and can benefit salmonid communities (Bisson et al. 1982, Fausch 1985, Fausch and Northcote 1992, Roni and Quinn 2001). Here, we enumerated all LWD equal to or greater than 1 m in length within the bankfull channel. Pieces >10 cm in diameter and 1 m long were counted, and we expressed results in terms of LWD density per reach (count / (average width * reach length).

Salmon use pools for a variety of reasons at all life stages, but pools are thought to be particularly important during juvenile stages (Beechie and Sibley 1997, Bisson et al. 1988). For example, juvenile coho salmon are more commonly found in pool habitats and characteristics of pools such as area and depth are important drivers of fish habitat selection and distribution (Nickelson et al. 1992, Clark et al. 2018). RPD is strongly associated with salmon densities during the summer months (Torgersen et al. 1999, Roni and Quinn 2001, Clark et al. 2018) and can positively affect juvenile foraging opportunities (Nelson and Reynolds 2015). Residual pool depth serves as a measure of pool volume and was calculated by subtracting the depth of the pool tail crest from the maximum pool depth (Lisle 1987). The pool tail crest was visually identified as the point at which there is a break or transition in stream channel slope. Individual measurements of RPD were then averaged at the reach scale.

Substrate plays an important role in the life history of salmonids. Substrate provides refuge during the early life stages (Connor and Bennett 2003, Louhi et al. 2011), can affect the forage base through impacts to macroinvertebrate communities (Suttle et al. 2004), and most notably, can limit spawning and early life-stage recruitment (Tappel and Bjornn 1983, Kondolf and Wolman 1993, Kondolf 2000, Quinn 2005). We estimated the median grain size in pool tailouts using a two-tiered ocular approach (Buffington and Montgomery 1999). For tier one we estimated the percent of each substrate size class including: exposed bedrock surface, boulder (>256mm), large cobbles (127.1 – 256 mm), small cobbles (64.1 – 127 mm), gravel (2 – 64 mm), and fines (<2 mm) to the nearest fifth percent and recorded the mid-point of that size category multiplied by the observed proportion The dominant size class from tier one was further categorized into subcategories (e.g., coarse; Table 1) and the mid-point of that size category was multiplied by the observed proportion. The proportional size estimates from tier 1 and tier 2 were then summed for a final size estimate for each habitat unit and the median is calculated for the reach-level estimate.

Analyses

Sampling designs have profound effects on the precision, accuracy, and inference of fish and habitat monitoring assessments (Urquhart et al. 1998, Walther and Moore 2005, Liermann et al. 2014). Here, we leveraged reach-based assessments of LWD, RPD, and GS using a simulation approach where we considered three different sampling designs commonly used in monitoring programs: (*i*) Simple random sampling with equal probability (SRS), where inclusion probabilities were equal among all reaches; (*ii*) Simple random sampling with unequal probability (SRSUNEQ); and (*iii*) generalized random tessellation stratified sampling (GRTS) with unequal probability (PIBO; Kershner 2004, Champ 2013). For the SRSUNEQ simulations, first order inclusion probabilities were proportional to reach length relative to the total stream length (i.e., of the selected stream). Second order inclusion probabilities were estimated for each sampling rate (1 % to 50 % in 1 % increments) based on the results of 100,000 simulations of which a record was kept counting how many times a particular reach was selected out of the total number of simulations. The GRTS sampling design used the same reach selection probabilities as the SRSUNEQ simulations.

Each stream is analyzed separately therefore the population and total number of reaches is stream-dependent (Table 3). We evaluated bias and precision across sampling rates from 1 to 50% at each stream. However, this did not result in a different number of sampled reaches for all 50 sampling rates, due to inherent differences in stream lengths. For example, if stream A had eight reaches and stream B 20 reaches then one reach would be sampled for rates 1 through 24% for stream A, and 1 reach would be sampled for rates 1 through 9% in stream B. At a rate of 25% two reaches would be sampled in stream A, and at a rate of 10% two reaches would be sampled in stream B.

We bootstrapped our continuous habitat data within a simulation framework to evaluate the bias and precision of habitat parameter estimates across the different sampling designs. We summarized the habitat data at the reach scale (Table 3) for the entire stream length that would be accessible to salmonids, and with each iteration we selected a sample reach from the total population of study reaches within a stream. We ran 5,000 simulations for each of the three sampling designs using sampling rates of 1% to 50% to obtain estimates of LWD, RPD, and GS for each stream.

We used a multistep process to assess the relative performance of the different sampling approaches. This process included (*i*) selecting a sample from the population of reaches, (*ii*) estimating the mean and standard deviation based on the sample for the habitat parameter of interest, (*iii*) repeating steps *i* and *ii* a total of 5000 times for each stream, and each sampling rate, and lastly (*iv*) assessing the precision and bias of the sampling designs and estimators by comparing them to the known true values of the parameter of interest. We use the coefficient of variation (CV) as a measure of precision in order to assess the performance of the estimators. We selected a CV value (0.20) that is slightly higher than what is targeted for population monitoring of anadromous salmonids in the Pacific Northwest (e.g., redd counts; Crawford and Rumsey 2011), but appropriate given the high levels of spatial and temporal variability in stream habitat data (Larsen et al. 2004, Al-Chokhachy et al. 2011). We assessed the bias of sampling designs and estimators by calculating the error (estimate – truth) for each simulation where we consider the census-based estimate as a surrogate for the true value of a habitat attribute, but acknowledge that this estimate does not represent the actual truth due to measurement and observer error. In order to simplify comparisons across streams and sampling designs we normalized the error by dividing by the truth ((estimate – truth) / truth) and refer to this as normalized error (Liermann et al. 2015). A normalized error value of 1 represents an estimate that was double that of the true value (e.g., (10 – 5) / 5 = 1). Truth refers to the true value of a parameter determined from census surveys (i.e., 100% sampling).

Results

Census-based field sampling showed that stream habitat varied considerably within and across all of our study streams (Table 3). The average substrate size across streams ranged from 15.2 to 148.0 mm. Mean RPD and LWD density also varied across streams, ranging from 0.28 to 1.99 m and 0.09 to 1.15 per m^2, respectively (Table 3). We focus upon simulation results from 3 focal streams (Brooks Creek, Speelyai Creek, and Siouxon Creek) for illustrative purposes, but ran simulations on all 10 streams. The range of CV's seen in our focal streams is consistent with the variability we observed in all 10 streams suggesting the three focal streams offer ample representation of the variability in habitat within our study area (Table 4). Among the focal streams, we found no discernible spatial patterns in the distribution of GS, RPDs or LWD densities (Fig. 2). In fact, all distributions were different within a stream for different habitat attributes and across streams for the same habitat attribute (Fig. 2). Brooks Creek had considerably less variation in the distribution of RPD and substrate size when compared to other streams (Fig. 2). Speelyai Creek exhibited the greatest variation in both LWD density and substrate size, while Siouxon Creek had the greatest variability in RPD (Fig 2).

The precision of the sampling designs increased as sampling intensity increased, as seen by the reduction of CV across all streams and attributes (Fig. 3). However, the relationship between sampling intensity and CV differed across attributes, and across streams. In general, the lowest CV values were observed for GS and the highest for RPD. For example, the median CV values at a 15 % sample rate (across streams and designs) for GS, LWD, and RPD were 0.17, 0.40, and 0.47, respectively (Fig. 3). We found GRTS to consistently have the highest or similar to the highest precision (i.e., lowest CV) across streams and attributes, with levels of precision for SRS and SRSUNEQ to be considerably more variable.

Achieving a CV value of less than or equal to 0.20 differed across focal streams and attributes. For example, the highest levels of precision with the lowest sampling rates was possible for GS at a 3% sample rate in Brooks Creek, but 12% and 19% for Speelyai Creek and Siouxon Creek, respectively. For LWD the same precision target required 40% and 32% sampling rates in Brooks Creek and Siouxon Creeks, respectively but remained unattainable in Speelyai Creek even with a 50% sample rate. A CV of 0.20 or less for

RPD was possible in Brooks Creek at low sampling intensities (3%), moderate intensities in Speelyai Creek (34%), but was not possible in Siouxon Creek at any sampling intensity (Fig. 3).

Bias generally decreased as sampling intensity increased across sampling designs illustrated by the reduction in normalized error as sampling intensity increases (Figs. 4, 5, and 6). However, the reduction in bias with increasing sampling intensity varied across attributes and streams. Across streams and designs the median bias at a 15% sampling rate was smallest for GS (-0.5%; range = -2.2 to 22.5%), moderate for RPD (-8%; range = -49.4 to 27.2%), and highest for LWD (-33.0%; range = -159.1 to 37.0%). Unlike our results investigating precision, no sampling design performed the best across attributes and streams.

Discussion

Habitat assessments are a critical component in identifying and quantifying factors limiting species (Reeves et al. 1989, Booth et al. 2006), directing management efforts (Van Liefferinge et al. 2003), assessing the effectiveness of restoration (Jungwirth et al. 2002, Roni et al. 2018), and quantifying the potential for species to proliferate (Burnett et al. 2007, Bidlack et al. 2014). Using empirical, continuous survey data we illustrate the challenges of effectively assessing different attributes of fish habitat. Through measures of precision and bias, our results suggest the most effective sampling approach may vary by habitat attribute and stream. Below, we discuss the factors likely driving our results, and the implications for habitat assessments used in directing management and restoration actions.

Differences in precision and bias across attributes and streams

The relationship between sampling intensity, precision and bias varied across streams for the same habitat attribute and within streams for different habitat attributes across all three sampling designs used in the simulations. The differences in precision and bias across habitat attributes are likely driven by the local hydrology and geomorphology which govern the spatial patterns in habitat throughout any given stream (sensu Vannote et al.1980). For example, the distribution of LWD throughout a stream tends to be non-uniform, likely a result of the complex interactions of stream power, floodplain access, geology, valley width, disturbances, and land use (Gurnell 2003, Iroume et al. 2014, Wohl and Cadol 2011, Wohl 2017). Concomitantly, LWD often accumulates in aggregations leading to higher site-to-site variability (Wohl and Jaeger 2009; Al-Chokhachy et al 2011). Indeed, LWD density was unevenly distributed within and among streams (Fig. 2) which resulted in the highest CVs and bias across attributes and clearly demonstrates the challenges in monitoring. Even though precision increased (i.e., CVs decreased) for LWD density estimates when sampling rates increased during simulations, accuracy remained low (Fig. 6) and precision was the lowest observed compared to RPD or GS regardless of stream or sampling design (Fig. 3).

In contrast, we found precision was much higher (i.e., CVs were much lower) and bias lower for GS when compared to LWD density (Fig. 3). Stream bed grain size is influenced by watershed characteristics (e.g., watershed geology; Knighton 1998) and stream-power (i.e., drainage area, precipitation, and valley slope; Knighton 1998). As stream power increases, the bedload in rivers is widely acknowledged to exhibit downstream fining, where mean substrate size decreases in the downstream direction (Rice 1998, Frings 2008). Longitudinal fining of sediments has been observed in mountainous gravel bed streams (Brummer and Montgomery 2003, Green et al. 2013), anabranching streams (Kemp 2010), and in sand-bed rivers (Benda et al. 2004). The downstream nature of this fining and processes operating longitudinally (e.g., hillslope processes) likely lead to the lower variability in reach averaged grain size. However, local (CGU scale) habitat characteristics (e.g., LWD and debris jams, lateral sediment inputs) can result in variation in sediment supply, shear stress and localized bed slope (Rice 1998, Buffington and Montgomery 1999b), creating patches of higher variance in grain size. The likely influence of local attributes was evident in Siouxon Creek (Fig. 2) where grain size and residual pool depth varied widely across reaches. Despite the potential influence of local attributes, we observed the highest precision (lowest CV) with substrate data, and the differences in the precision and accuracy between LWD density and substrate size are likely directly related to their distribution throughout the stream. Not only are distributions of these habitat attributes different within a stream but also across streams, further complicating the decision to use a particular sampling design.

Sampling designs in monitoring

Probability sampling designs do provide unbiased estimators, but the variance can be very high when estimating attributes with patchy or non-random distributions (Liermann et al. 2014). Spatially balanced designs (e.g., GRTS) are used extensively in fish and habitat monitoring programs given the efficiency and effectiveness of these designs in monitoring across large spatial scales (Lackey and Stein 2013, Gallagher et al.

2010, Jacobs et al. 2009). Here, we found the highest precision and lowest bias with the GRTS design, but do acknowledge that CV's remained high and the increased sampling effort required is greater than what has been typical in fisheries monitoring programs (Gallagher et al. 2010; Liermann et al. 2014). For example, Liermann et al. (2014) found CVs of <0.15 were possible with sample rates of < 20 % using a GRTS sampling design for monitoring redd abundance of anadromous Chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*O. mykiss*). However, other studies have highlighted the spatial variability in species assemblages and relative abundance throughout a stream resemble those demonstrated herein for habitat (Brenkman et al. 2012, Berger and Gresswell 2009, Torgersen et al. 2006). The spatial discontinuity and longitudinal patterns observed in spatially continuous fish surveys are unlikely anomalies in riverine networks and may reflect the overall spatial variability in habitat conditions (Torgersen et al. 2006, 2012, Benda et al. 2004, Kiffney et al. 2006). In a habitat sampling design study Dolloff et al. (1997) found LWD estimates using a reach-based sampling design were higher in some streams and lower in other streams when compared to the census-based estimates which resembles our results for highlighting that bias is a concern in reach-based approaches and that it varies across streams. However, unlike Dolloff et al. (1997) we assessed how bias and precision changed across multiple sampling designs and sampling intensities. For example, the RRET sampling results found in Dolloff et al. (1997) were based on sampling rates of three and four percent while our evaluation is across sampling rates of 1 to 50% which helped to illustrate the diminishing returns of increased precision and accuracy in some attributes as sampling intensity increases (i.e., grain size). Additionally, the use of CVs in this study allowed us to compare precision across streams and attributes which was not reported in Dolloff et al. (1997).

We acknowledge that our results could have been different if we had also contrasted estimates from high intensity reach-scale sampling (e.g., dense topographic surveys; Bangen et al. 2014) to low intensity census-scale sampling under the various sampling designs and rates. The inherent variability in stream systems will challenge most sampling methods and designs. Focusing on methods and study designs that will maximize precision and minimize bias of the data for a specific objective(s) while also minimizing cost and effort will help overcome this variability challenge.

Implications for monitoring

Field-based habitat monitoring covers a wide range of approaches ranging from fine-scaled topographic mapping (e.g., Bangen et al. 2014) to coarser, rapid surveys (e.g., Brenkman et al. 2012). While the project goals of habitat monitoring continue to be the most important in selecting the most appropriate design, our results provide key insights into the challenges of reach-based monitoring. When reach-based data is used in monitoring to assess the extent of suitable habitat and the objective is to estimate total habitat capacity and quality (e.g., quantifying the potential for species reintroductions; *sensu* Anderson et al. 2014) then accurate characterization of habitat and habitat features should be a top priority. The variability across streams seen in this study and others (Larsen et al. 2004, Anlauf et al. 2001, Al-Chokhachy 2011) is not surprising given the dynamic nature of stream networks. In fact, Al-Chokachy et al. (2011) attributed >75% of the variability in 6 common habitat attributes to site-to-site variability. Similarly, Kincaid et al. (2004) reported that a range of 25% to 99% of the variability in 53 indicators, representing chemical and trophic condition and zooplankton and fish, was attributable to site variability in lakes. The inherent variability of habitat across stream networks and the variability in the distribution of habitat attributes within and across streams together suggest non-census (e.g., probability based) sampling designs will have challenges in estimating habitat values (e.g., LWD density). Indeed, this is one of the allures of continuous, riverscape-type surveys (Fausch et al. 2002, Weins 2002, Torgersen et al 2006, Brenkman et al. 2012).

We acknowledge the resources needed to carryout continuous surveys are, in many cases, prohibitive, precluding many monitoring programs from using census-based sampling designs thus creating a need for reach-based sampling designs. For example, the U.S. Forest Service PACFISH/INFISH Biological Opinion effectiveness monitoring program (PIBO; Kershner et al. 2004) seeks to evaluate if aquatic and riparian resources are maintaining, improving, or decreasing in quality due to land management practices. This large-scale program extends across the Interior Columbia River Basin across four states (WA, OR, ID, MT) and includes 3,547 watersheds. A program of this scale precludes the use of census-based sampling designs and therefore relies on reach base approaches. For programs with smaller spatial extents, such as species reintroductions, identifying the quantity and quality of habitat is critical in defining the potential for success. However, our results illustrate this can be difficult with reach-based approaches (Anderson et al. 2014).

Modifications in reach-based sampling designs or, where practical the use of census type sampling designs, may help alleviate concerns with precision and bias during habitat assessments. Furthermore, monitoring assessments may benefit by identifying auxiliary information (Liermann et al. 2015) to control for differences within the study domain to increase precision and reduce bias. For example, LWD monitoring may be improved by considering the density of riparian forests in the study area (Bilby and Wasserman 1989) or stream size given the inverse relationship between LWD density and channel width (Bilby and Ward 1989). Information like this could be incorporated into a reach-based probability sampling design where it could be used to help define strata. A stratified design would help focus sampling on reaches with higher predicted uncertainty leading to increased estimator performance. The use of spatially balanced designs such as GRTS may also provide improvements in precision and bias as opposed to more traditional design-based sampling methods (e.g., SRS) especially if the attribute of concern is likely to be spatially autocorrelated (Lackey and Stein 2013, Liermann et al. 2015). Extensions of the GRTS design to include knowledge of the spatial structure of the habitat attribute (e.g., stratifying based on auxiliary information) can yield improvements in precision and bias (Courbois et al. 2008, Liermann et al. 2015). Using *a priori* knowledge regarding the spatial distribution of an attribute can aid in identifying sampling designs that tend to result in more accurate estimates (e.g., spatially balanced designs, nested spatial designs) (Bellehumeur and Legendre 1998, Cochran 1977). However, if a census-based design is possible then no further information on the spatial distribution of an attribute is needed.

Continuous, census-based designs, while often more expensive, can improve precision and bias as no assumptions in distribution or extrapolations are needed (Dolloff et al. 1997, Williams et al. 2004). While census-based designs require more effort, this can be offset by refining the level of detail recorded, technological advancements, and focused monitoring protocols. In some cases, fine scale data are not needed to capture the distribution and variability in a habitat attribute at a level that is biologically important. For example, conducting Wolman pebble counts in a reach-based survey design will provide robust estimates of grain size for areas surveyed at the expense of spatial extent due to the increased time and effort required using this method (McHugh and Budy 2011, Killourhy et al. 2016). Conversely, visual based estimates of grain size binned into categories that are biologically meaningful in the context of the assessment may provide an adequate level of detail and are more suited to census-based designs (e.g., Mullner et al. 2000, Brenkman et al. 2012). Continuously collected data also allow estimates and enumeration at which different spatial scales are important, both for ecological processes involved and management relevance. New tools and methods for such analysis are increasingly becoming available (e.g., Sagurado et al 2013, Welty et al. 2015, Benda et al. 2011) but require continuously collected data. Little doubt exists regarding the increased monetary and temporal costs of census surveys compared to reach-based surveys however, advancements in technology may help reduce this disparity.

Technological advancements are also providing increases in efficiencies which help reduce the effort required in census surveys. For example, improvements in laser technology have resulted in precise laser range finders which, for example, can be used to quickly quantify dimensions of habitat units and LWD and can replace more traditional, time intensive stick and tape methods (Clark et al. 2019, AREMP 2016, Scott et al. 2016). Small unmanned aircraft systems (sUAS) show great promise for use in quantifying certain habitat attributes (e.g., LWD, pools) at census scales (Woodget et al. 2017, Tamminga et al. 2015). Aerial imagery has been used to quantify LWD (Ortega-Terol et al. 2014, Marcus et al. 2003), grain-size distributions (Dugdale et al. 2010) and in-stream physical habitats (Woodget et al. 2017). Structure-from-Motion (SfM) photogrammetry has been shown to collect high-quality elevation data and accurate orthophotographs and has been applied at large extents (Rusnak et al. 2018, Dietrich 2016, Javernick et al. 2014). The myriad of technological and computing advancements will continue to provide opportunities for larger and potentially more precise data sets. Future monitoring approaches will need to consider which habitat attributes are most relevant for the target species and tailor the sampling designs, and methodology to meet the overall goals of the monitoring program. To aid in the success of a monitoring program, the program managers should use a sampling design that is tied to the scale of a specific monitoring objective, have some knowledge of the distribution of the habitat attributes of interest, and specifically understand the level of detail (acceptable levels of precision and bias) needed to meet the monitoring objectives.

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Tables

Table 1. The size categories of dominant substrate types for field surveys of stream habitat in the upper Lewis River, WA.

Particle Name	Facies Name	Size (mm)
Boulder, Very Coarse	Bvc	2048-4096
Boulder, Coarse	Bc	1024-2048
Boulder, Medium	Bm	512-1024
Boulder, Fine	Bf	256-512
Cobble, Coarse	Cc	128-256
Cobble, Fine	Cf	64-128
Gravel, Very Coarse	Gvc	32-64
Gravel, Coarse	Gc	16-32
Gravel, Medium	Gm	8-16
Gravel, Fine	Gf	4-8
Gravel, Very Fine	Gvf	2-4
Sand	S	.0625-2
Fines	F	<0.0625

Table 2. Characteristics of streams within the upper Lewis River basin, WA (USA) including average and maximum elevation, drainage area, annual precipitation, average bankfull width (BFW), and total length of available stream habitat. Note, the asterisk denotes the focal streams used in this manuscript to illustrate our findings.

Stream ID	Stream	Avg elevation (m)	Max elevation (m)	Drainage area (Km²)	Precipitation (cm/yr)	BFW (m)	Length (km)
1	Brooks*	423	956	10	221	7.7 (2.8)	3.1
2	Buncombe	250	622	4	191	5.6 (2.0)	1.1
3	Bypass	939	3658	-[1]	290	20.8 (6.1)	6.5
4	Cougar	486	943	3	310	16.1 (4.3)	3.9
5	Indian George	491	855	7	198	10.2 (1.5)	1.5
6	Ole	642	1133	12	290	13.8 (2.0)	1.7
7	Souixon*	739	1333	165	287	28.4 (5.0)	6.1
8	Speelyai*	637	1159	34	254	13.3 (1.6)	6.0
9	WF Speelyai	751	1159	11	259	17.3 (14.4)	1.3
10	WT Speelyai	646	1038	12	241	9.9 (1.9)	1.1

[1] The Bypass is a constructed channel to convey flow around the dam as opposed to through it, and therefore has no natural drainage area.

Table 3. Mean values (SD) of reach-level measures of large woody debris (LWD) density, residual pool depth (RPD), streambed grain size (GS), and the total number of reaches (n) in each study stream in the North Fork Lewis River, WA (USA). Note, the asterisk denotes the streams used in this manuscript to illustrate our findings.

Stream ID	Stream	LWD density	RPD (m)	GS (mm)	n
1	Brooks* Creek	0.94 (0.61)	0.28 (0.10)	87.8 (9.1)	17
2	Buncombe Hollow Creek	0.09 (0.10)	0.29 (0.15)	82.3 (5.3)	9
3	Bypass Channel	0.15 (0.20)	0.88 (0.42)	125.8 (40.4)	11
4	Cougar Creek	0.64 (0.50)	0.73 (0.39)	87.5 (14.3)	8
5	Indian George Creek	1.13 (0.37)	0.48 (0.34)	100.0 (25.5)	7
6	Ole Creek	1.15 (1.25)	0.63 (0.48)	123.1 (13.9)	6
7	Siouxon* Creek	0.59 (0.23)	1.99 (0.70)	129.4 (33.7)	8
8	Speelyai* Creek	0.56 (0.87)	0.69 (0.27)	147.3 (43.6)	23
9	West Fork Speelyai Creek	0.66 (0.43)	0.63 (0.19)	148.0 (27.9)	5
10	West Tributary Speelyai Creek	0.47 (0.4)	0.51 (0.23)	15.2 (1.8)	5

Table 4. Coefficient of variation results for all study streams and streambed grain size (GS), residual pool depth (RPD), and large woody debris (LWD) density in the upper Lewis River, WA (USA).

Stream ID	Stream	GS (mm)	RPD (m)	LWD density
1	Brooks Creek	0.1	0.36	0.65
2	Buncombe Hollow Creek	0.06	0.52	1.11
3	Bypass Channel	0.32	0.48	1.33
4	Cougar Creek	0.16	0.53	0.78
5	Indian George Creek	0.26	0.71	0.33
6	Ole Creek	0.11	0.76	1.09
7	Siouxon Creek	0.26	0.35	0.39
8	Speelyai Creek	0.3	0.39	1.55
9	West Fork Speelyai Creek	0.19	0.3	0.65
10	West Tributary Speelyai Creek	0.12	0.45	0.85

Figures

Figure 1. A map of the study area with numbers corresponding to the individual study streams (see Table 2) in the upper Lewis River, WA (USA) and the inset map illustrating the general location in the Pacific Northwest.

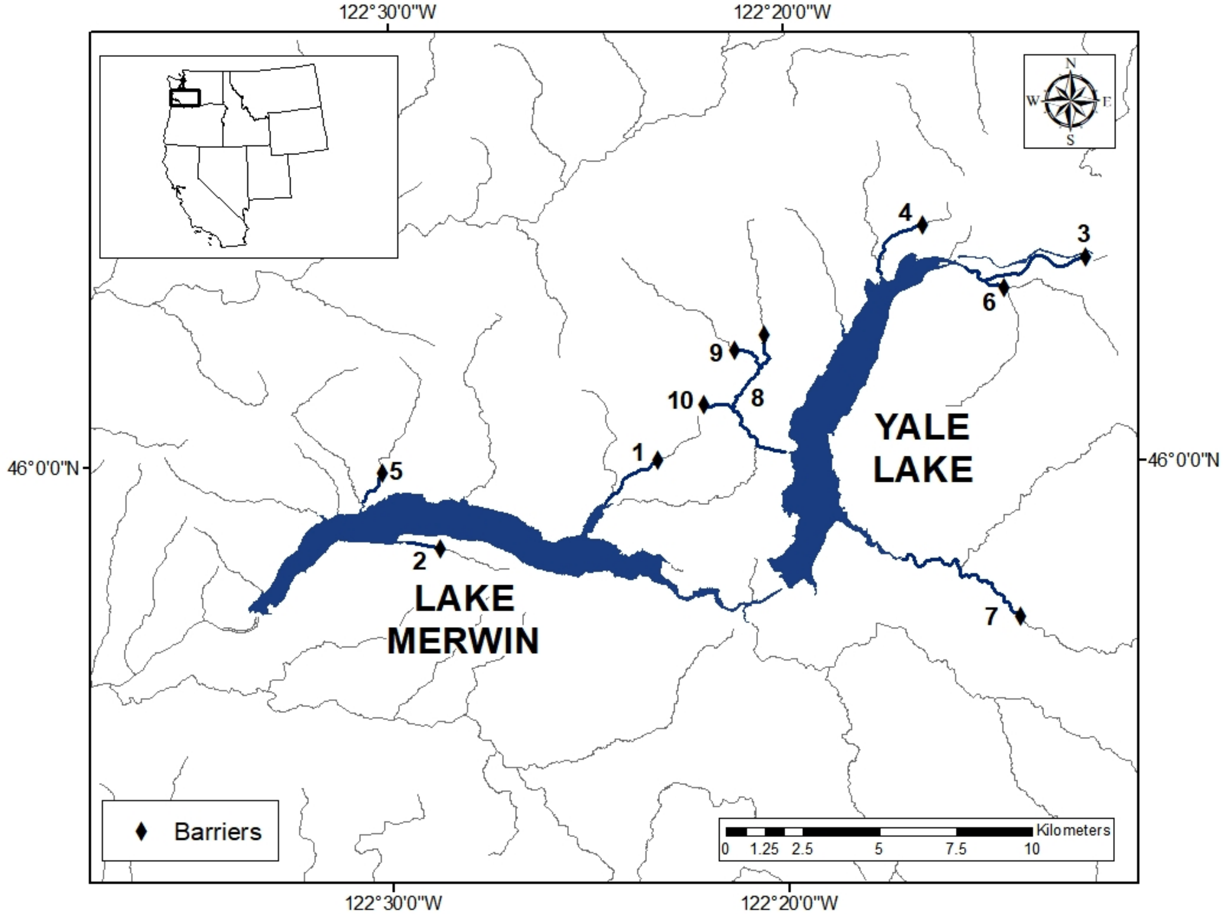


Figure 2. Distributions summarized at the reach scale for large woody debris (LWD; A), residual pool depth (RPD; B), and grain size (substrate; C) habitat attributes (rows) across three streams (columns). The x-axis represents the length of stream from the mouth (0) to the upstream extent for Brooks Creek, Speelyai Creek, and SIouxon Creek in the upper Lewis River, WA (USA).

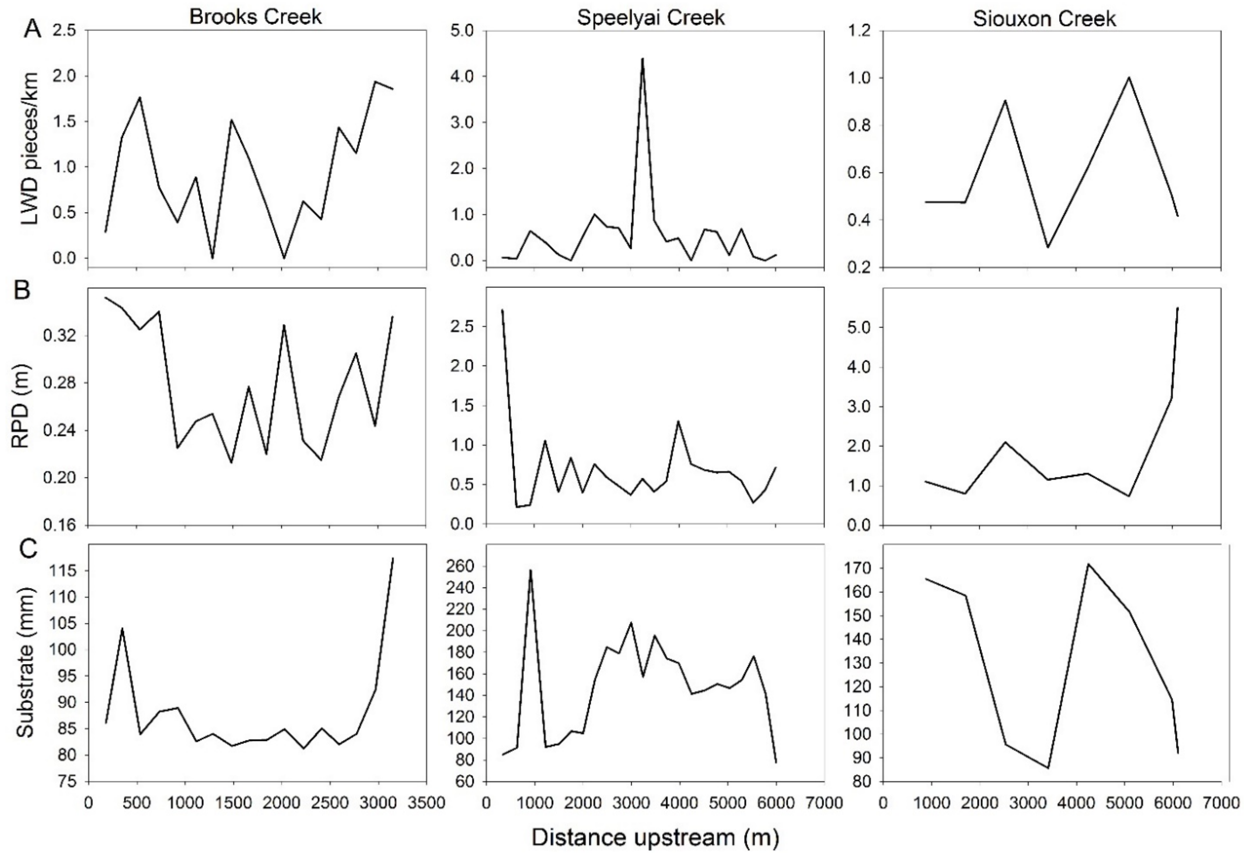


Figure 3. Coefficient of variation (CV) simulation results for the generalized random tessellation stratified (GRTS; solid black), simple random sample (SRS; solid grey), and unequal random sample (UNEQ; dashed black) designs using sampling rates of 1 to 50 percent applied to the large woody debris (LWD density; A), residual pool depth (RPD; B), and grain size (GS; C) data in Brooks Creek, Speelyai Creek, and Siouxon Creek in the upper Lewis River, WA (USA).

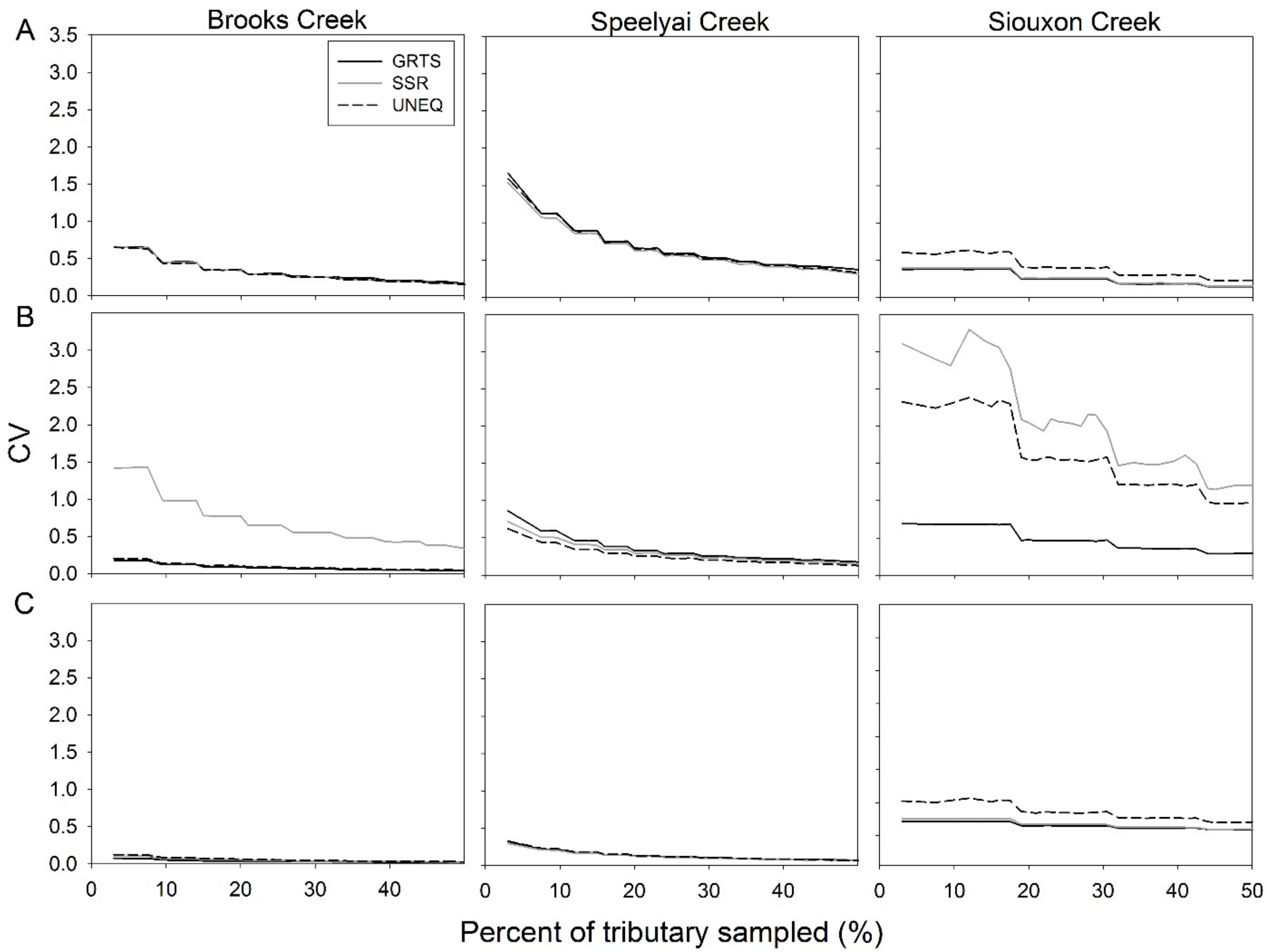


Figure 4. Normalized error results from simulations across simple random sample (SRS), unequal random sample (UNEQ), and generalized random tessellation stratified (GRTS) sampling designs for grain size in Brooks Creek, Speelyai Creek, and Siouxon Creek in the upper Lewis River, WA (USA) with the middle 95% of the values shown in light grey, first and third quartiles shown in dark grey, a dotted line representing the median and a solid line representing the mean.

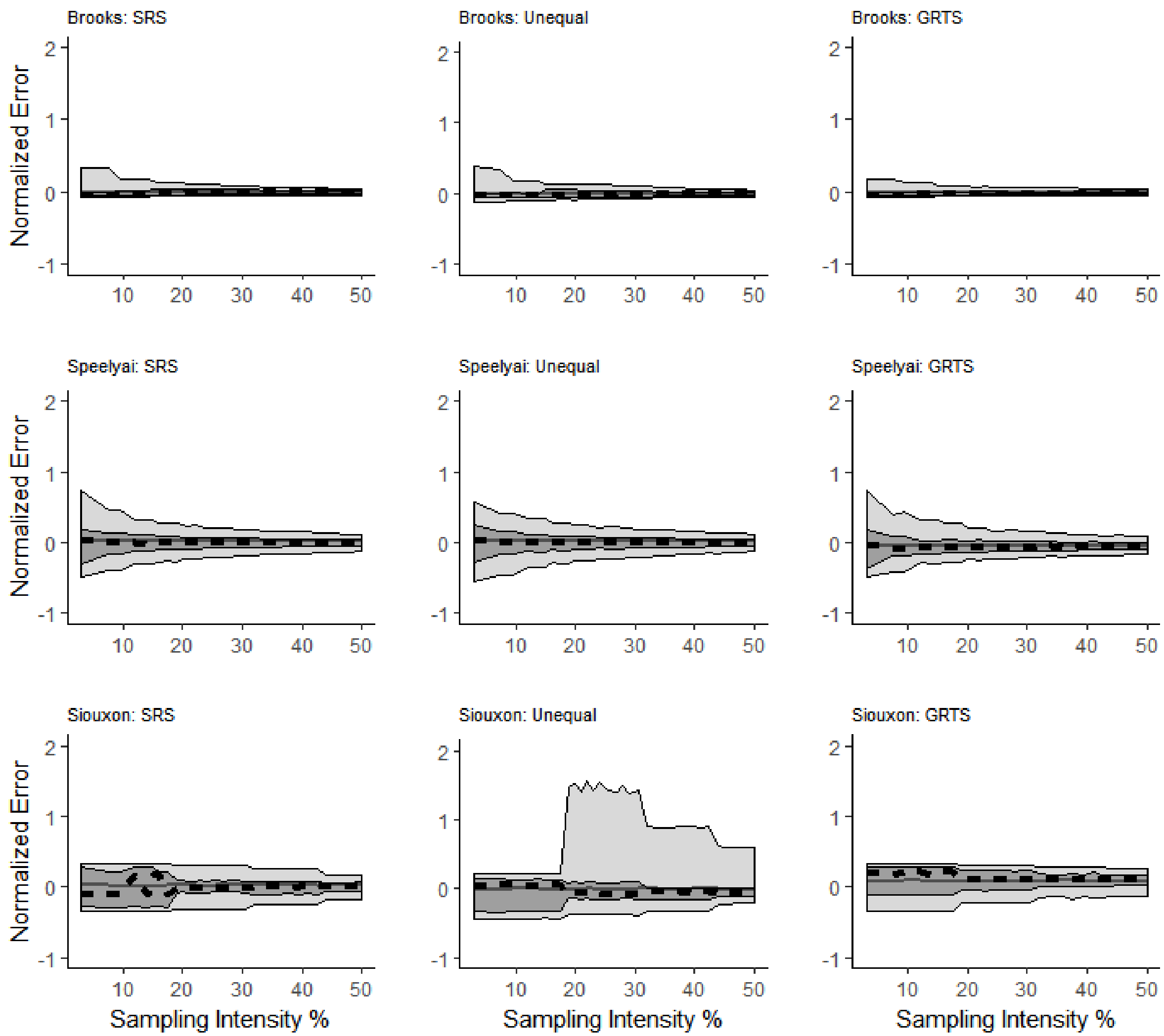


Figure 5. Normalized error results from simulations across simple random sample (SRS), unequal random sample (UNEQ), and generalized random tessellation stratified (GRTS) sampling designs for residual pool depth in Brooks Creek, Speelyai Creek, and Siouxon Creek in the upper Lewis River, WA (USA) with the middle 95% of the values shown in light grey, first and third quartiles shown in dark grey, a dotted line representing the median and a solid line representing the mean.

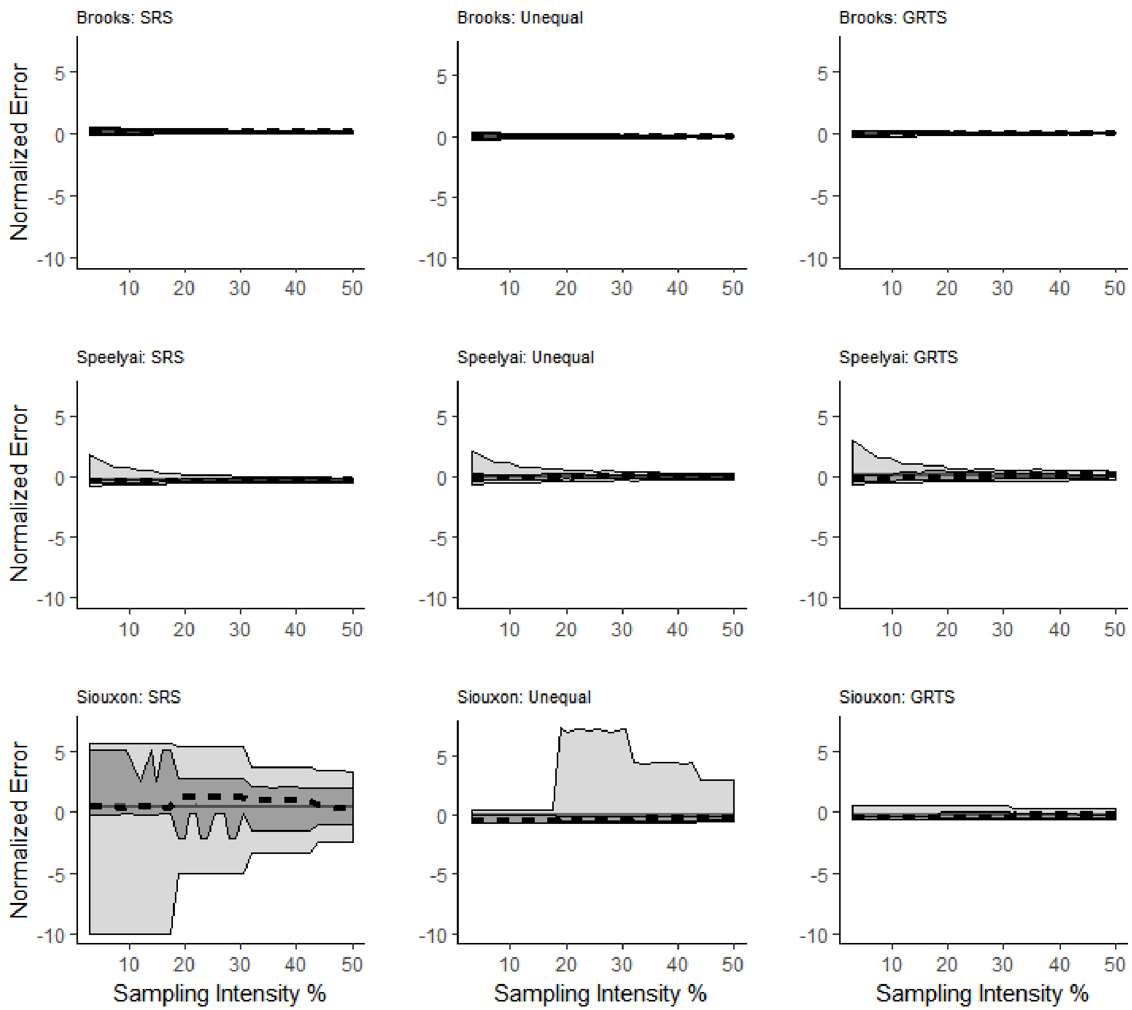
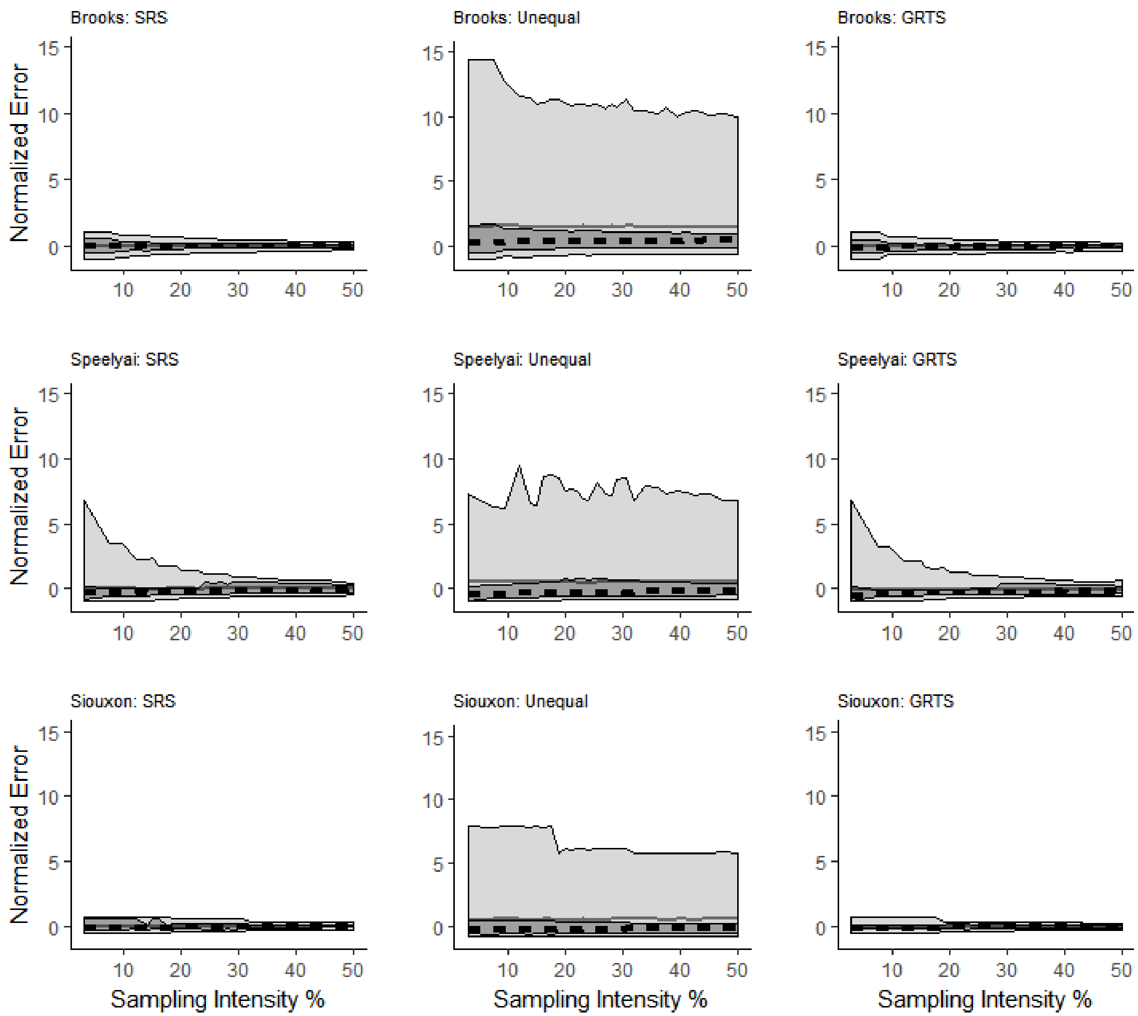


Figure 6. Normalized error results from simulations across simple random sample (SRS), unequal random sample (UNEQ), and generalized random tessellation stratified (GRTS) sampling designs for large woody debris density in Brooks Creek, Speelyai Creek, and Siouxon Creek in the upper Lewis River, WA (USA) with the middle 95% of the values shown in light grey, first and third quartiles shown in dark grey, a dotted line representing the median and a solid line representing the mean.



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