The Effect of Forest Conditions Associated with Timber Harvest on Stream Biotic Integrity

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King County

Department of Natural Resources and Parks
Water and Land Resources Division
Science and Technical Support Section
King Street Center, KSC-NR-5600
201 South Jackson Street, Suite 5600
Seattle, WA 98104
206-477-4800 TTY Relay: 711
www.kingcounty.gov/EnvironmentalScience

The Effect of Forest Conditions Associated with Timber Harvest on Stream Biotic Integrity

Prepared for:

Washington State Department of Ecology
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Submitted by:

Elizabeth Sosik
King County Water and Land Resources Division
Department of Natural Resources and Parks

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EXECUTIVE SUMMARY

What is the goal of the project?

This study seeks to understand if and how current forest conditions in the Puget Sound region impact stream benthic macroinvertebrate communities. Biotic integrity[1] is characterized by the benthic index of biotic integrity (B-IBI) score, and is typically highest in forested stream basins, which frequently also support timber harvest operations. By keeping the land profitable through controlled harvests, commercial forestry prevents the basins from converting to land uses associated with poorer biotic integrity. However, some forestry activities associated with timber harvest can negatively impact stream water quality and habitat conditions by increasing sediment runoff and altering habitat. This Puget Sound-wide project seeks to enhance our understanding of if and how timber harvests and associated forest conditions affect freshwater habitat and provide insight into the effectiveness of various current forestry best management practices (BMPs) for the protection of stream health.

Who will use and benefit from this project?

The intended audience is land managers who are seeking to identify practices that are mutually beneficial to commercial forestry and healthy streams. This can include foresters, regional salmon enhancement groups, regional land conservation groups, freshwater scientists, as well as regional policy makers at state or tribal-affiliated governmental agencies. The results may also benefit stormwater managers seeking to prioritize stressors that frequently co-occur as forests are replaced by urban development.

What did we find?

Directly and indirectly, the *forest conditions that are caused by timber harvest had a very small but statistically significant negative effect on B-IBI scores*, primarily by altering inputs of sediment, light and leaf litter. The small effects suggest that forest conditions resulting from harvests are unlikely to be the primary drivers of low B-IBI scores observed in some forested basins but may contribute to small variations in score.

Furthermore, we found that *forest conditions in a 50-foot riparian area extending out from the stream had the best relationship with most in-stream conditions, and with biotic integrity specifically, compared to wider riparian buffers.*

Biotic integrity was influenced by both forest conditions in the whole upstream riparian area and by forest conditions in the immediate sampling reach, though, again, the influence was small. Upstream forest conditions had a slightly larger effect than forest conditions in the immediate sampling reach.

Fine sediment seems to be the biggest stressor impacting B-IBI scores in forested basins.

[1] Biotic integrity is defined the ability of a stream to support an assemblage of macroinvertebrate organisms with a species composition, diversity, and functional organization comparable to those in natural habitats of the region." (Karr and Dudley 1981, Karr et al. 1986)

What are our recommendations?
Based on our findings that forest conditions resulting from harvest have only small negative effects on benthic macroinvertebrate communities in the study area, and that the influence of forest conditions on biotic integrity decreases with increasing distance from the stream, *current regulations for riparian areas in fish-bearing streams are likely sufficiently protective of in-stream biotic integrity*.

Based on our findings that upstream forest conditions contribute to fine sediment impacts, *we recommend strengthening current practices to mitigate erosion and stream sedimentation in non-fish-bearing and seasonal streams.*

In addition, *we recommend land managers consider thinning existing densely planted riparian trees* to mimic light and leaf litter conditions in old-growth forests. Existing regulations pertaining to "desired future growth" of riparian forests already seek to create or retain forest stands with characteristics similar to mature, unmanaged forests. However, current regulations do not permit thinning in the core riparian zone of fish-bearing streams.

Finally, we recommend *additional research be done to track the use and study the effects of pesticides, fire retardants and other chemicals on forested lands*. The effect of chemical applications in forested lands is a large data gap that this study could not address, because there is no reporting requirement for ground-based methods of application.

What are the study limitations?
This study has several limitations that must be considered along with its main findings. First, *this analysis does not address the universal range of riparian benefits to in-stream processes and biota but is focused on benthic macroinvertebrates only*. It is possible that forest conditions influencing in-stream conditions may impact other organisms differently.

Second, *this study relies on modelled spatial datasets for estimates of past forest conditions, soil composition, stream temperatures, fine sediments, and flow*. A large source of uncertainty is our assumption that these modelled values are reasonable and reliable.

The resolution of our modelled datasets is another source of uncertainty. Because of the limitations of using remotely detected and modelled spatial datasets, *we expect that our data will be most sensitive to clearcuts in patch-sizes greater than 30 meters-squared*.

Despite these uncertainties, we feel confident in the main findings of this study in how forest conditions relate, directly and indirectly, to stream biotic integrity. Though our results could not be independently validated due to data limitations, our findings still have implications for forestry impacts and ecological function.

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1.0 INTRODUCTION

Benthic macroinvertebrates play a crucial role in the biotic integrity of stream ecosystems and are good indicators of ecological health. The multi-metric Benthic Index of Biotic Integrity (B-IBI) is a standardized scoring system applied to benthic macroinvertebrates collected from streams. The scores range from 0 to 100 and are broadly categorized by biological condition: very poor (0-20), poor (20-40), fair (40-60), good (60-80), and excellent (80-100). The B-IBI was developed in the early 1990's, recalibrated in 2012 (King County 2014a), and is widely used to report biotic integrity by cities, counties, tribes, and state agencies in the Puget Sound Basin. The B-IBI score integrates the diversity and relative abundance of sensitive taxa, and reflects the cumulative impacts of stressors on stream communities but cannot indicate the specific causal factor of impairment.

The Puget Sound Partnership has defined two recovery targets for biotic integrity in streams within the Puget Lowland ecoregion. One target aims to restore water quality in streams that rank as fair, while the other aims to protect water quality in streams that rank as excellent. While both targets are critical for Puget Sound recovery, it is easier and less expensive to achieve these targets by preventing environmental degradation from occurring rather than restoring already degraded systems.

During the development of the freshwater quality implementation strategy (Stormwater Strategic Initiative 2020), the Stormwater Strategic Initiative Lead Team's B-IBI interdisciplinary team (IDT) identified timber harvest as a potential stressor to stream condition. In its draft Working Lands Strategy, the IDT identified two objectives for working lands (defined as forestry and agricultural areas): 1) reduce the risk of conversion from working lands to more intensive land uses; and 2) reduce the ongoing impact of working lands on stream health (freshwater quality implementation strategy, Stormwater Strategic Initiative 2020). One of the approaches the IDT developed to meet these objectives is to better understand the relationships between stressors, working land management practices and B-IBI scores. This study is a direct result of this call for more information. For land and stormwater managers seeking to promote practices that are mutually beneficial as they develop and implement protection plans, it is of vital importance to gain a better understanding of the relationship between B-IBI scores and the forest conditions that result from different forest management practices.

This study uses B-IBI results from forested stream basins across the Puget Sound watershed to determine: 1) if forest practices influence stream biotic integrity; and 2) what mechanisms and spatial scales dominate that influence. For the purposes of this study, we define a forested stream basin to be one where forest or vegetation cover is greater than 70% of the basin land cover, and where urban development is less than 10% of the basin land cover. This includes basins with extensive recent harvests that we expect will eventually become re-forested.

1.1 Forestry and Biotic Integrity

B-IBI sites in forested basins are typically among the highest scoring sites in Puget Sound; consequentially, many of these sites have been identified as prime candidates for protection (King County 2019, King County 2015a). At the same time, many of these same basins support commercial timber harvest operations. The presence of these operations poses an environmental conundrum. By keeping the land profitable, commercial forestry prevents the basins from becoming urbanized (e.g., residential and commercial development), which is strongly associated with lower B-IBI scores (Walsh et al. 2005, Meyer et al. 2005). However, commercial timber harvests also have the potential to negatively impact stream conditions (Fuchs et al. 2003, Moldenke and Ver Linden 2007, Zhang et al. 2009, Frady et al. 2006, Jackson et al. 2007, Kedzierski and Smock 2001, Chizinski et al. 2010, Haggerty et al. 2004, Kiffney et al. 2003, Lunn et al. 2017, Swank et al. 2001).

In sites monitored by King County between 2002 and 2021, we can clearly see the relationship between B-IBI and urban land cover (% by area; Figure 1). However, under low levels of urban development, variability in scores is quite large. The current threshold for a stream being placed on the 303(d) impairment list in the Puget Lowlands is a B-IBI score equal to or below 65 (WA ECY 2020). In basins that are minimally impacted by development, there are many sites that score below this threshold. This indicates that in these areas there are additional impacts on the benthos not represented by urbanization alone (Paul et al. 2009). A question that frequently arises is whether forestry practices associated with timber harvest can create conditions that impair stream biotic integrity in undeveloped stream basins.

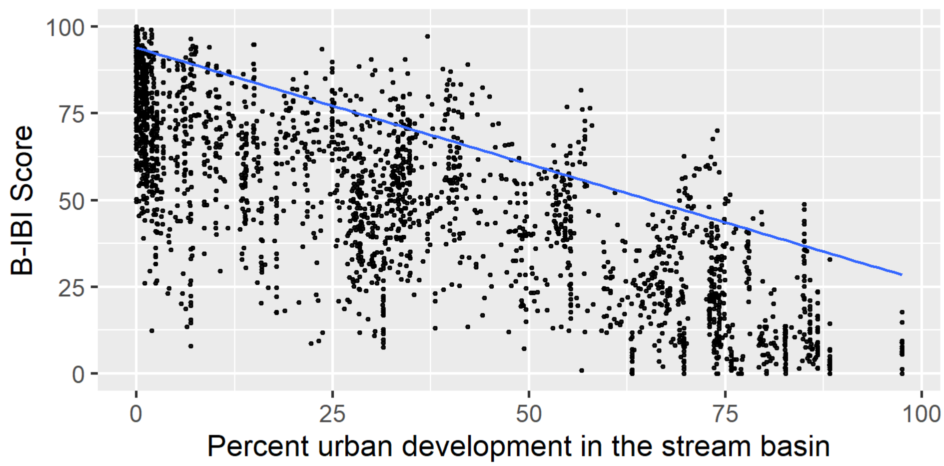

Figure 1. The relationship between basin-wide urban development (percent by area using the Coastal Change Analysis Program's Regional Land Cover Database) and biotic integrity. Data from routine stream monitoring in King County. The blue line is the 90th percentile score along the urban gradient, representing the biologic potential of sites at a given level of development.

Previous studies done on the watershed-scale have demonstrated that the macroinvertebrate community has a complex and mixed response to forestry practices associated with timber harvesting. In a sweeping meta-analysis of the effects of timber harvesting on a wide range of organisms, Lunn et al. (2017) observed widely varying and often contradictory effects on freshwater macroinvertebrates across dozens of studies (see Lunn et al. 2017 for a thorough review). This heterogeneity of response suggests that the net effect of timber harvest on water and habitat quality depends on multiple factors. These factors may include the underlying environmental context, the harvest regime, the regulatory requirements applied, as well as the spatial and temporal scale at which the effects are being measured. This study seeks to holistically examine the complex relationships between forest harvest and stream biotic integrity in forested basins of the Puget Sound watershed.

1.2 Study Area

This study will focus on stream basins within the Puget Sound Basin that are primarily forested, characterized by very little urban development, and where B-IBI data are readily available and existing spatial data can be obtained. The Puget Sound Basin encompasses all land in Washington State that drains to Puget Sound, South of the Strait of Georgia and East of the Strait of Juan de Fuca, bounded by the Cascade Mountains in the East and the Olympic Mountains on the West (Figure 2). The Puget Sound Basin is divided among four ecoregions: The Puget Lowlands, the Coast Range, the Cascades and the North Cascades.

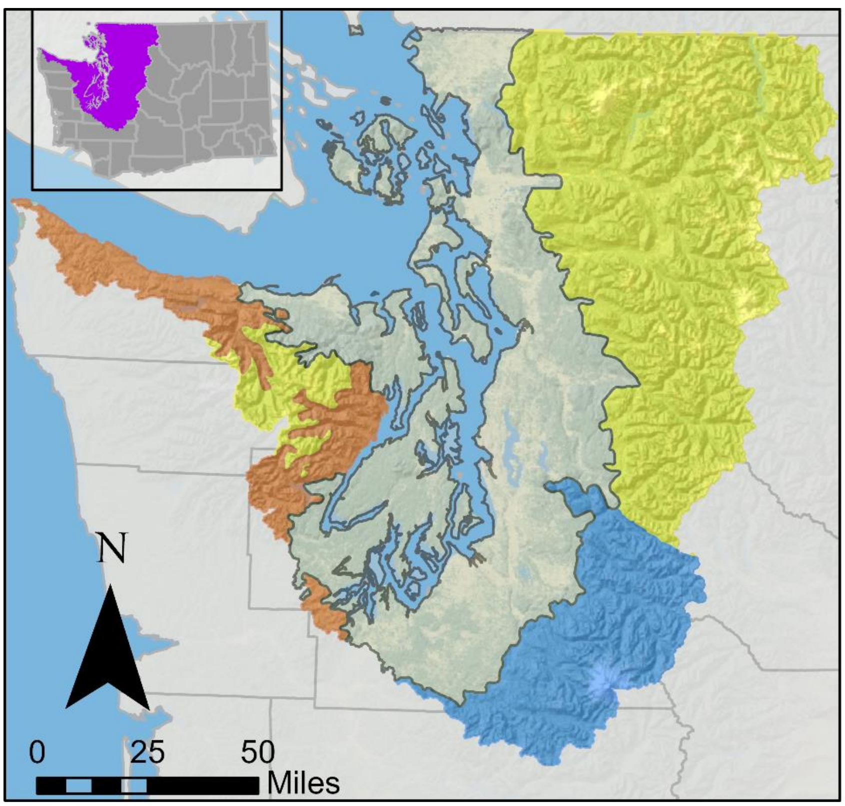

Figure 2. The Puget Sound Basin and contributing eco-regions. The central region highlighted in gray is the Puget Lowland ecoregion, in blue is the Cascade ecoregion, in yellow is the North Cascade ecoregion, and in orange is the Coast Range ecoregion.

1.3 History of Study Area

Prior to widespread Euro-American settlement and the rise of the lumber industry circa 1848 (Chiang and Reese 2014), the Puget Sound Basin was dominated by continuous forests of Western hemlock, Western Red cedar, and Douglas-fir (WDFW 2005). Since the arrival of European settlers, the area has oscillated between periods of deforestation and reforestation (Michalak et al. 2013). Despite periods of net forest gain, the overall trend has been towards reduced forest cover compared to the pre-settlement era. WDFW (2005) has estimated that over 50% of historic conifer forests in the Puget Lowland ecoregion have been converted to urban or agricultural use. Despite more recent declines in harvest volumes and the permanent loss of forested lands, as of 2015 Washington was the second largest producer of soft wood lumber in the US (DNR 2007, https://oregonforestfacts.org/#harvest-production).

Stands of old growth forests are rare in Western Washington, composing somewhere between 3 to 10% of current forest cover and occurring almost entirely on federal lands (WDFW 2005, DNR 2007). In Western Washington, about 45% of forests are less than 40 years old, and about 75% of forests are less than 100 years old (DNR 2007). In a 2019 study, Donato et al. estimated the natural range of variation of forest cover for different successional forest stages in the Western Cascade Range of Washington. They found that current conditions are well outside the estimated natural range of variation. Specifically, whereas estimated natural conditions were dominated by mature and old-growth forests (47-90%), densely growing mid-seral forests and young tree plantations provide most of the current forest cover (~70%). The authors found this difference was driven largely by a replacement of old-growth cover. Current forested areas are generally second- or third-growth forests dominated by Douglas fir, western hemlock, western red cedar, red alder, and bigleaf maple. In many cases, the trees in these forests grow more densely and are more even-aged than in the pre-settlement era (DNR 2007).

Forestry regulations within the region have changed over time, beginning with the first Forest Practices Act in 1946. Starting in the early 1990's a wave of additional laws bolstered environmental protections for forested lands. Each policy change has been increasingly protective of water and habitat quality.

2.0 METHODS

The primary question this study seeks to answer is whether forest harvest activities impact B-IBI scores in forested basins. The study further seeks to determine the spatial scale and main mechanisms of impact, should an impact be found. Because accurate measures of past forest harvest volume, timing, intensity, and methods are difficult to obtain, this study instead focuses on forest conditions that are influenced by forest harvest, such as stand height, canopy cover and basal area. The study relies on modelled spatial datasets for estimates of past forest conditions, soil composition, stream temperatures, fine sediments and flow. Because of the limitations of using remotely detected and modelled spatial datasets, we expect that our data will be most sensitive to clearcuts in patch-sizes greater than 30 meters-squared. We use B-IBI and macroinvertebrate community data from sampling conducted by multiple agencies across the Puget Sound Basin.

Because of the complex relationships that occur between forests, streams, and the physical environment, we selected an investigative approach that facilitates a holistic look at the stream systems. Structural Equation Modelling[2] allows researchers to specify a priori a linked network of hypothetical causal relationships, and then evaluate whether the hypotheses fit the data we have collected (Grace et al. 2006). In this study, we hypothesize that forest harvest causes forest conditions that negatively impact B-IBI scores by impacts to stream flow, fine sediments, stream temperature, food web structure and toxicity.

We conducted a literature review to construct an a priori hypothetical network of these relationships. We used two different data sets: a primary dataset to evaluate and modify our initial network of hypotheses, and a secondary dataset to evaluate our final network of hypotheses. In the following sections, we will briefly outline the steps taken to build and evaluate a suitable structural equation model to examine causal pathways between forest harvest activities and B-IBI scores. For a more detailed discussion of the analytical approach, see Appendix A.

2.1 Site Selection

We selected 132 basins that were >70% vegetated and <10% developed[3] according to 2019 NLCD Land Cover data (https://www.mrlc.gov/), and which also did not have mines, major roads, lakes, marine shoreline, agriculture, or residential development close to the sampling point which may have influence on the sample. These criteria were based on a 2019 King County report which found that most basins with excellent B-IBI scores can be characterized as having at least 72% of their basin forested, and less than 4% of their basin developed. For this study, we set our forest threshold to include lands which may have been recently logged but will become re-forested (forest and shrub cover). In addition, we set more inclusive criteria for the development threshold as a compromise to obtain an

[2] https://www.usgs.gov/centers/wetland-and-aquatic-research-center/science/quantitative-analysis-using-structural-equation

[3] One basin had slightly higher development than our threshold criteria (E2052, 10.72%), but after closer inspection we believe this value to be slightly higher than actual conditions.

adequate sample size (minimum of 5 sites per parameter; Kline 2015), and because clearcuts can occasionally be mis-categorized as development. We used the 132 basins to develop a primary dataset for model evaluation, and a secondary dataset to be used for confirmation if our initial model required modification.

2.2 B-IBI Sample Selection

We used B-IBI data from samples collected by agencies across the Puget Sound Basin. For B-IBI samples to be included, we selected a time range of 2002 to 2015. To reduce spatial autocorrelation, we selected only one sample per basin for our primary dataset, and another sample from a different time point in each basin as a secondary dataset for the purpose of validation. To minimize temporal correlation between sites, we randomly selected each basin to be represented by either the earliest or latest sample taken at the site in the 2002-2015 time range. In the secondary data set, the opposite set of samples were selected for each site. Numerous sites had only one sample taken between 2002 and 2015; thus, those sites had to be dropped from the secondary dataset. At the end of the process, the primary data set had 132 basins with corresponding B-IBI samples (Table 1), and the secondary data set had 80 basins with corresponding B-IBI samples (Figure 3).

The methods used to collect the macroinvertebrate samples varied across agencies in terms of the area of substrate sampled, the habitat targeted, and the standard taxonomic effort (STE) used for identification (Table 2). However, past work has shown B-IBI scores to be well correlated across these methodologies and taxonomic efforts (King County 2014b, King County 2019).

Using the percent urban development present in our sites at the time of sampling, and the 90th percentile of the relationship between B-IBI and urban development (Figure 1), we calculated the biological potential of our sites. The biological potential is the best-case B-IBI score that we would anticipate for a site at a given level of development. The biological potential of our sites at the time of sampling ranged from 94 to 87. The actual observed scores ranged from 100 to 20.2.

Table 1. Characteristics of 132 basins in the primary data set.

Variable	Units	Mean	Std.Dev	Min	Median	Max
Basin Area	hectares	1869.6	3530.1	12.5	563.7	24805.8
Road Density	km/km2	2.8	1.4	0.0	2.9	5.7
Mean Elevation	meters	492.7	284.6	79.8	462.9	1355.9
Median Discharge (estimated)	cubic meters/ second	1.8	5.2	0.0	0.2	35.5
B-IBI Score	unitless	75.9	15.7	20.6	79.3	100.0
Stream Gradient	meters/ kilometer	0.1	0.1	0.0	0.0	0.5
Basin Slope	percent	32.3	13.3	8.0	32.1	65.0

Stream Order	unitless	3.5	1.2	1.0	3.0	7.0
Median Time Since Last Harvest	years	63.3	39.1	7.9	51.4	192.1
Developed at time of sampling	percent	0.67	1.46	0	0.05	10.76
Vegetated at time of sampling	percent	96.48	3.96	79.85	97.97	100
Forested at time of sampling	percent	86.47	11.45	38.60	87.89	100

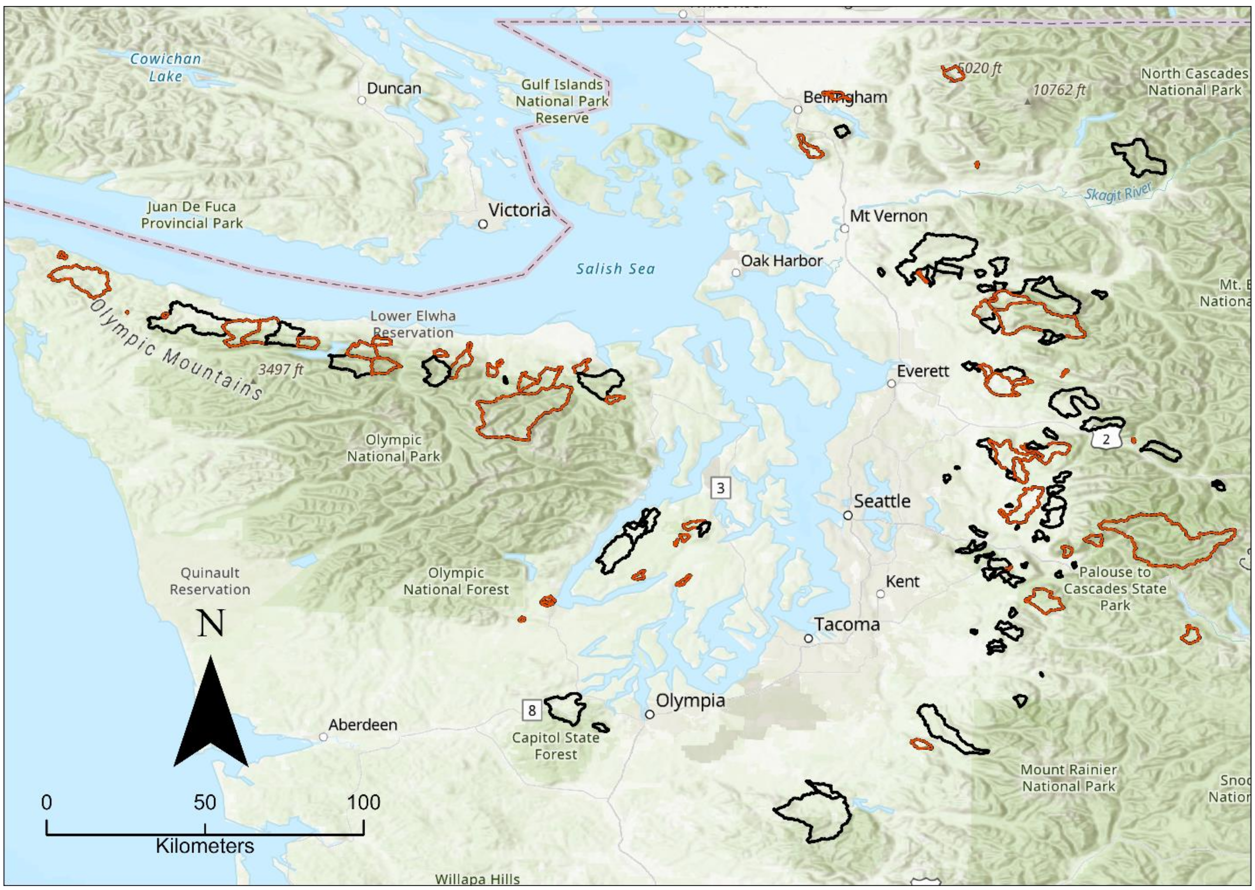

Figure 3. Map of the basins selected for analysis. All basins were included in the primary data set. Only basins highlighted in red were included in the secondary set, using a second set of samples taken at a different time point.

Sampling Agency	Methods	STE
Stillaguamish Tribe	Riffles, 8 sq ft	coarse/fine
King County - DNRP	Riffles, 3 to 8 sq ft	coarse/medium/fine
Snohomish County	Riffles, 3 to 8 sq ft	fine
Clallam County	Riffles, 8 to 9 sq ft	coarse/fine

Washington State Department of Ecology	multi-habitat, 8 sq ft / Riffles, 3 sq ft	fine
King County - Roads	Riffles, 3 sq ft	fine
Skokomish Tribal Nation	8 sq ft	fine
Kitsap County	Riffles, 3 to 8 sq ft	fine
Adopt-A-Stream Foundation	Riffles, 3 sq ft	fine
Thurston County	Riffles, 9 sq ft	fine

Table 2. Agencies, methods and taxonomic effort of samples included in the analysis.

2.3 Structural Equation Modelling

We developed a structural equation model following the process outlined in Figure 4. Briefly, we conducted a literature review to develop a conceptual model of how forest harvests and related activities could impact B-IBI score (Appendices B-E). We then translated those ideas into a "meta-model" (Figure 5). A meta-model is an intermediate step between conceptual model and structural model, in which we focus on specific pathways of interest and begin to consider data needs, data availability, and model structure. We then collected data to support the hypotheses identified in the meta-model. Finally, we examined the data to help develop the architecture of the full structural model.

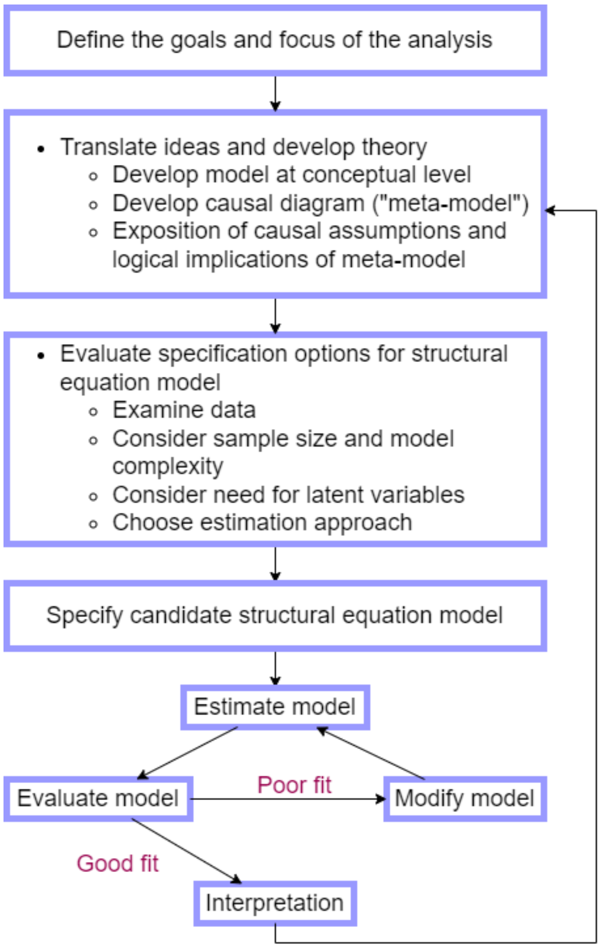

Figure 4. The structural equation modelling workflow. The process is confirmatory if no modification step is needed. Otherwise, the process is considered exploratory, and the results are provisional until the model can be validated on a new dataset. Modified from Grace et al. (2010 and 2015).

2.3.1 Meta Model

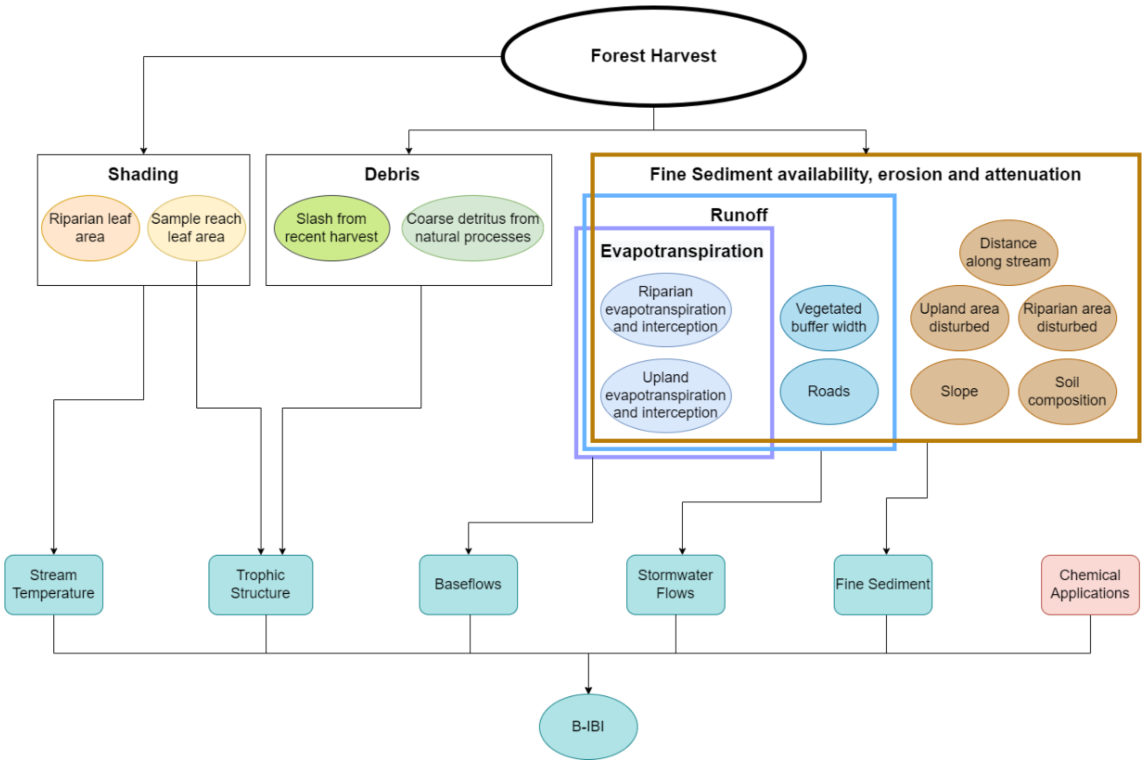

Figure 5. Meta-model of how forestry practices associated with timber harvest can influence B-IBI scores in forested basins. The meta-model contains our initial hypotheses for how forest harvest could impact B-IBI scores.

For our model, we considered 6 different pathways which, based on the literature and best professional judgement, we hypothesized are likely to influence B-IBI scores: stream temperature, trophic structure, baseflows, stormwater flows, fine sediment inputs, and chemical applications (For background on how these were selected and exposition of our hypotheses, see Appendices B-E). We then expanded the causal chain by adding specific forest and basin conditions that we hypothesized would have a causal relationship with each pathway (for example, decreased evapotranspiration rates could increase baseflows; Figure 5). Because the research questions of this study are specific to the effects of forest harvest on B-IBI, we focused primarily on forest conditions that can be directly altered by forestry practices associated with timber harvest, or that may influence recommendations for best management practices. We were unable to locate a reliable data source for the location and timing of chemical applications (e.g., pesticides and fertilizers). We kept this node in our meta-model as we felt it was important to include, but it will not be included in the final structural model.

2.3.2 Data Collection

This section outlines the steps taken to collect data at multiple spatial scales for the primary and secondary datasets. Except for the benthic macroinvertebrate samples, this study relies heavily on modelled data values, as empirical data were not widely available in all our basins.

2.3.2.1 Benthic Macroinvertebrate Data

For each sample selected for inclusion in the analysis, we downloaded B-IBI metric scores and taxonomic data from the Puget Sound Stream Benthos database. The Puget Sound Stream Benthos database is a regional data repository used by over 30 state, local and tribal jurisdictions and non-profits to house benthic macroinvertebrate data and calculate B-IBI scores as part of monitoring projects and programs. We then assigned sample taxa to functional feeding groups based on attributes from the Biological Conditions Gradient work group (US EPA 2022). Functional feeding groups are classifications based on feeding behavior rather than taxonomy (e.g., scraper, shredder, predator). Based on our literature review of benthic community responses to forest harvests, we specifically focused on scraper, shredder and collector-gatherer functional feeding groups. See Appendices B-E for more details on these functional feeding groups.

2.3.2.2 Spatial Data

Riparian Buffers

We generated three sets of riparian buffer widths to best evaluate riparian management practices currently required, recommended, or being considered for recommendation by Washington Department of Natural Resources (WDNR) and Washington Department of Fish and Wildlife (WDFW) (Figure 7). Each dataset will be used in a separate but identical model to examine which spatial extent has the best fit with the data. As supported by Jyväsjärvi et al. (2020), Kiffney et al. (2003) and Sweeney & Newbold (2014) this approach assumes that riparian buffer width is a key driver of macroinvertebrate communities and biotic integrity.

Name	Fixed or Variable	Width on either side of stream	Source
Core Riparian Management Zone	Fixed	50 feet	WDNR regulations for fish bearing streams and waters of the state in western Washington (WDNR 2017)
100-foot Buffer	Fixed	100 feet	Commonly used in studies (Barnowe-Meyer et al. 2021, Kiffney et al. 2003, Sweeney & Newbold 2014)
Site Potential Tree Height	Variable	100-260 feet (median 200 feet)	WDFW riparian management recommendations (Rentz et al. 2020)

Table 3 Riparian buffer widths used in this study.

The first dataset was based on the WDFW's recommendation for riparian protections, which are derived from Site Potential Tree Height (SPTH) values[4] (Rentz et al. 2020). These were the largest buffers. These buffers widen and narrow accordingly as each stream

[4] https://wdfw.maps.arcgis.com/apps/MapJournal/index.html?appid=35b39e40a2af447b9556ef1314a5622d

traverses landscapes with different SPTH values. SPTH values were not available on federally or tribally owned lands; in these cases, we applied a 200-foot (61 m) buffer on either side of the stream which was the median buffer value in our basins with SPTH data. In addition, at the recommendation of WDFW we superimposed a minimum width of 100-foot (30.5 m) for pollution removal (Rentz et al. 2020).

The second set of riparian buffers was based on the WDNR's definition of the Core Riparian Management Zone for fish bearing streams and shorelines of the state[5] (WA DNR 2017). This is a fixed 50-foot (15.25 m) buffer width in which no harvest activity is allowed. It is important to note that additional protections may apply to riparian zones outside of the core zone depending on site class and stream size, but at least some amount of harvesting is allowed in these non-core zones. In addition, fewer protections apply to non-fish bearing or seasonal streams, where clear cuts are allowed to the stream edge. For this study, we opted to use the 50-foot core buffer in all streams regardless of fish presence to investigate whether this is also sufficient to protect stream macroinvertebrates.

The final dataset was an intermediate of the first two, based on a fixed 100-foot (30.5 m) buffer. This level of protection was considered in a recent review of current and proposed riparian management policies for meeting temperature change criterion in western Washington and has been included in a number of studies (Barnowe-Meyer et al. 2021, Kiffney et al. 2003, Sweeney & Newbold 2014). While Barnowe-Meyer et al. (2021) ultimately did not recommend this policy due to economic costs to landowners, they did expect with high certainty that buffers of this size would meet temperature change criterion. In addition, WDFW estimates that a 100-foot buffer removes at least 95% of phosphorus, sediment and pesticides, and 80% of excess nitrogen run-off (Rentz et. al 2020). Because the focus of this study is ecological effects rather than economic feasibility, we included these wide 100-foot and SPTH-based buffers to evaluate the level of stream protection afforded by each. However, we acknowledge they may not be feasible policy in all situations.

Sampling Reach Area

To evaluate the effects of insolation on community composition, we generated "sampling reaches" in the immediate vicinity of each sampling site (Figure 7). The purpose of the sampling reaches is to help summarize forest conditions that may affect stream shading and water temperature at the sample sites. Three different sampling reaches were generated for each site, corresponding to each of the three different riparian buffer widths. The sampling reach extends 100 m upstream of the sampling GPS point to cover the typical reach length of a sampling event. It then extends out to the riparian buffer on either side of the stream.

[5] https://www.dnr.wa.gov/publications/fp_fpi_introduction.pdf

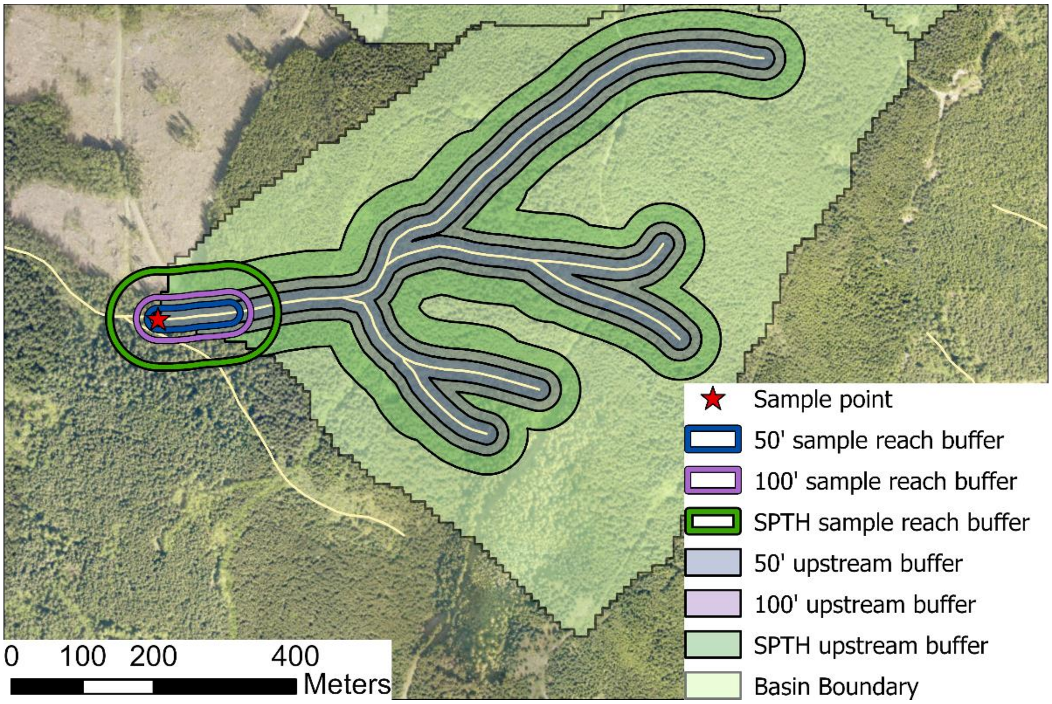

Figure 6. An illustration of the spatial extents examined in this study. Three riparian widths (50', 100' and variable widths based on Site Potential Tree Height) were used for total upstream area from the sample point and a 100 m upstream sample reach.

Forest and Stream Conditions

A final list of data extracted from all data sources and definitions can be found in Appendix F. Further details of the processing steps can be found in Appendix G. We processed the spatial data from the sources listed in Table 3 using ArcGIS Pro and R Statistical software. In each basin, we summarized these conditions by riparian area, upland area, and sampling reach area with each set of riparian buffers defined in the previous section. One drawback to using remotely detected and modelled spatial datasets is that they can only easily detect clearcuts that are larger than 30 meters-squared (the spatial resolution of the datasets); as a result, we cannot reliably examine the effect of less intensive harvest methods such as thinning. In addition, we were able to estimate values for hydrological, soil, sediment, and stream-temperature data.

Measures of soil composition, fine sediment, stream flow and stream temperature were not readily available for all sampling sites. Instead, we obtained modelled values that have been validated for the Pacific Northwest. We used reach-specific values from the National Oceanic and Atmospheric Administration's National Water Model Retrospective Dataset version 2.1 (NOAA 2022), and the US Forest Service's NorWeST Stream Temperature model. We obtained soil composition data from the US Department of Agriculture's Soil Survey. We calculated fine sediment estimates and indices using three independently developed sediment models that are based on the macroinvertebrate community: the Fine Sediment Biological Index (FSBI, Relyea et al), the Biological Sediment Tolerance Index (BSTI, Hubler et al), and bio.infer (Yuan et al). Each of these fine sediment models use a different statistical approach to estimate the impact of fine sediment at a sampling site, and each were developed using samples from the Pacific Northwest. The same macroinvertebrate data used to calculate B-IBI scores were used as input to the fine sediment models. While we may expect some correlation between B-IBI and fine sediment values because of this shared underlying data, we believe these models to be reasonable sources for estimating fine sediment impacts. The B-IBI is focused on taxonomic groups that are sensitive to a broad range of anthropogenic impacts to stream health, which include fine sediment among others. The fine sediment models, however, are specifically focused on taxonomic groups that respond to fine sediment impacts alone. Furthermore, two of the three models (BSTI and bio.infer), have been calibrated and validated using empirical fine sediment data.

For some measures of biomass, basal area, or disturbance in upland and riparian zones, we normalized the resulting values by the area of the whole basin (Appendix H). For example, total tree biomass (measured in kilograms) within the riparian zone was divided by the whole basin area, rather than simply the riparian zone area (measured in hectares). We did this to normalize for stream discharge (cubic meters/second), which is correlated with whole basin area (Castro and Jackson 2001, Curran et al. 2012). The rationale for this is that impacts to a stream from surrounding forest conditions are likely to be dependent on the size of the stream, which in turn is influenced by the size of the basin.

Table 4. Explanatory data sources, description, and temporal and spatial extents. We could not obtain data pertaining to chemical applications.

Dataset	Source	Use	Time Range	Spatial Coverage
Forest Structure Maps	Landscape Ecology, Modeling, Mapping, and Analysis work group (Oregon State University, US Forest Service)	Annual data on forest biomass, basal area, stand height, age, canopy cover (1986-2015)	1986-2017	Puget Sound Watershed
Land Change Monitoring, Assessment and Projection Dataset	US Geological Survey	Annual Land use and change detection from 1985-2015. Date of forest conversions and contemporary land use at time of sampling.	1985-2020	Washington State
Gridded National Soil Survey Geographic Database	Natural Resources Conservation Service	% sand and slope	-	Washington State
National Hydrography Dataset Plus High Resolution	US Geological Survey	Stream gradient, stream flow lines, basin delineations	-	Puget Sound Watershed
NorWesT annual Mean Weekly Maximum Temperature	US Forest Service	Stream temperature	1993-2015	Washington Coast and Puget Sound Streams identified in NHD Plus V2

Bio.Infer	Yuan et al. 2007	% Sand/Fine sediment	-	-
Biological Sediment Tolerant Index	Hubler et al. 2012	% Fine Sediment	-	-
Fine Sediment Biotic Index	Relyea et al. 2012	Index of Sand/Fine sediment	-	-
Function Feeding Group attributes	Biological Condition Gradient workgroup	Functional Feeding groups	-	-
Washington State Roads, Washington forest practices abandoned roads, Forest Service roads	Washington Department of Natural Resources; US Forest Service	Road density and stream crossings	-	Washington State
B-IBI data	Puget Sound Stream Benthos Database	Puget Sound Stream Benthos Database	1993-2020	Puget Sound Watershed
National Water Model Retrospective Dataset V2.1	NOAA	Flow metrics	1979-2020	Washington State Streams identified in NHD Plus V2.1
National Land Cover Database	Multi-Resolution Land Characteristics Consortium	Initial basin screening	2019	Washington State
Site Potential Tree Heights	Washington Department of Fish and Wildlife	Determine variable width riparian buffer	-	Washington State
National Inventory of Dams	US Army Corps of Engineers	Locate Dams upstream of BIBI sites	-	Washington State

2.3.2.3 Validation Dataset

We collected full datasets of benthic macroinvertebrate, B-IBI scores, and spatial data for both the primary dataset of 132 basins and samples, and for the secondary dataset of 80 basins and samples. This secondary dataset will be used for validating the final model should any alterations be made to our initial structural model (Grace 2006).

2.3.3 Data Review

We examined our dataset for missing data, normality and linearity. Where extreme non-normality was found, or where data were missing not at random (MNAR), we opted against using those variables or re-coded the data to incorporate basins with no history of harvest. In some cases, variables were log-transformed to better approximate normality. Transformations applied to the data and data distributions are presented in Appendices H and I. Finally, we prepared correlation tables of the data, which we used to help translate the meta-model to a structural model (Figure 8, Appendix J).

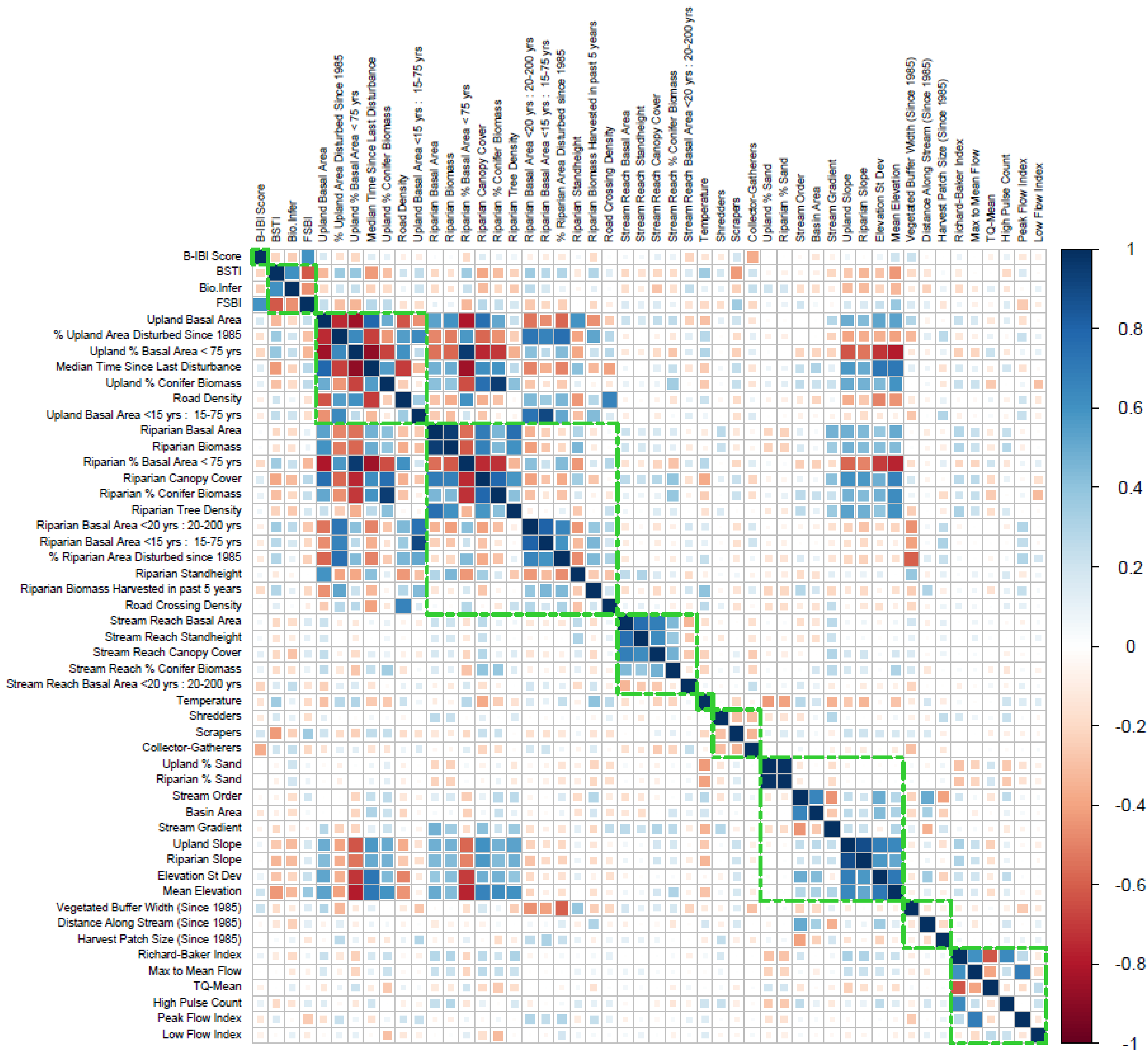

Figure 7. Correlations between different variables for consideration in the structural equation model. The variable-sized, green, dashed squares represent related subsets of data. Each square represents (from top left to bottom right on the diagonal): B-IBI score, Sediment, Upland forest conditions, Riparian upstream forest conditions, Riparian stream reach conditions, Temperature, Community Composition, Basin Characteristics, Spatial distributions of harvest, and Flashiness. This set of correlations is based on the 50' Core Riparian Management Zone definition of riparian area. Figures for 100' and Site Potential Tree Height riparian area definitions are included in Appendix J.

2.3.4 Structural Model Development

Following data collection, we continued the workflow outlined in Figure 3. We translated our expectations from the meta-model and specified our structural model. To evaluate our specification options, we examined the data, considered model complexity, and identified potential latent[6] variables. In some cases, the variables were mirrored so that all indicators of a single latent factor would be positively correlated. As noted in section 2.4.1, we were unable to locate a reliable data source for the location and timing of chemical applications (e.g., pesticides and fertilizers). As a result, chemical applications were not included in the structural model.

We then proceeded to estimate and evaluate our model. When the model did not meet fit criteria, we continued with exploratory analysis by modifying our initial model. For a detailed treatment of this process and exposition of the links and modifications, see Appendix K.

2.3.4.1 Model Estimation

We estimated our model using the maximum likelihood method with robust standard errors in the R package Lavaan (Rosseel 2012). To deal with data missing at random (MAR), we used the option for case-wise ("full information") maximum likelihood estimation (Kline 2015). We evaluated model fit using chi-square statistics, root mean square error of approximation (RMSEA), comparative fit index (CFI), and the standardized root mean square residual (SRMR) (Kline 2015).

2.3.4.2 Standardized vs un-standardized estimates

SEM effect estimates can be presented in standardized and unstandardized formats. Each offers some utility. Standardized results are scaled by the standard deviation of the variables and provide a convenient method for comparing the relative magnitude of effects between variables within a single model. In other words, the standardized coefficient is how much we can expect a response variable to change (in standard deviation units) based on a change of one standard deviation in the predictor variable. However, because standard deviation is dependent on sample size and variability within a given dataset, these standardized values should not be used for interpretation between models. When comparing between models, unstandardized effect estimates must be consulted.

[6] A latent factor is one that is hypothesized to exist but has not been directly measured (Grace 2006). In practice, latent variables represent abstract concepts that are difficult or impossible to measure, but which may have real causal effects on observable variables. In SEM, multiple related but independent observable variables can be used as indicators to estimate the "true" effect of a latent factor. As an example, "body size" in birds could be considered a latent variable indicated by measures of body mass, wing length and beak length.

2.3.4.3 Indirect, direct, and total effects

Following Grace (2006), to calculate the indirect effects of certain parameters on others that were not directly linked (i.e., slope and B-IBI), we multiplied the path coefficients along the causal pathway. We calculated total direct and indirect effects by summing the coefficients of direct effects with the calculated indirect effects.

We then calculated confidence intervals for unstandardized direct and indirect parameter estimates. We were able to obtain confidence intervals for direct effects from the results provided by Lavaan (Rosseel 2012). Because indirect effects frequently are non-normal, we used a Monte Carlo method of obtaining confidence intervals for indirect effects using the SemTools package in R (Jorgensen et al. 2022).

3.0 RESULTS

We were unable to validate our structural model but did achieve well-fitting models using both the primary and secondary datasets. Model results using Site Potential Tree Height riparian data did not meet fit criteria at any point in the process. For a detailed breakdown of the iterative process that we used to evaluate and modify our structural model, see Appendix K. Here we will briefly review the results of the process and present the results from our final structural model.

Our initial structural equation model failed the criteria for a well-fitting model for all three riparian distances (Chi-Square p-value <= 0.05 for 50 ft, 100 ft and Site Potential Tree Height riparian areas; Appendix L). We then proceeded to revise our causal theory based in part on observations of the data and links suggested by the modelling software but rooted in ecological grounds. All modifications made to the model were based on plausible connections given our knowledge of the system.

Using the primary dataset, we arrived at a model that achieved fit criteria for our data based on 50 ft and 100 ft riparian areas, but not for data based on the Site Potential Tree Heights (Appendix L). The model based on 100 ft riparian area had the best fit. We attempted to validate our modified model using the secondary data set. This data set was substantially smaller than our primary data set (n= 80 vs n=132). Having so few observations in a complex model dramatically reduced the likelihood of achieving an RMSEA value of less than 0.05 from 87% to 60%. Not surprisingly, the model did not meet fit criteria using the secondary dataset (Appendix L). As a result, we were unable to confirm that the data supports the hypotheses presented in the model.

We proceeded to modify the model to see if a simplified version could meet model fit criteria despite the smaller number of observations (Figure 9). Without an additional dataset to use for validation, the remainder of our effort remained strictly in the exploratory phase. Our findings must therefore be presented as tentative hypotheses to be tested in future field or observational studies.

Though not all links in the model were significant, or consistently significant, we refrained from removing insignificant links unless we felt we had sound reasoning for doing so. This is because in a complex system, multiple weak mechanisms can have a strong effect when acting in concert.

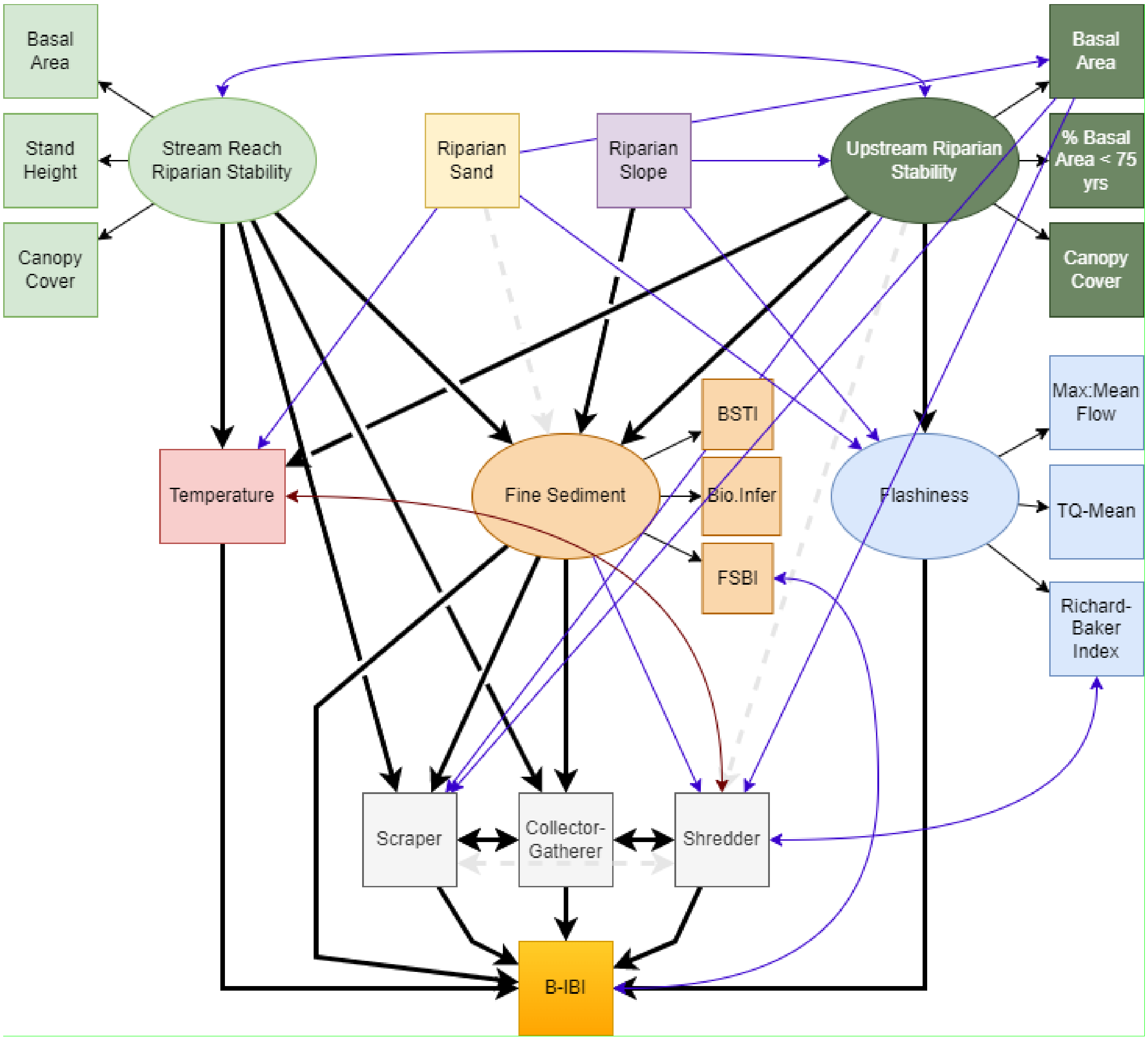

Figure 8. **Final Modified Structural Equation Model. Black lines represent links hypothesized in the initial structural model. Purple lines represent links added in the first modified model. Red lines represent new links in the final modified model. Gray dashed lines represent links from the initial model that were set to "no relationship" in the final modified model. Ovals represent latent (unmeasured) factors, and rectangles represent factors measured either directly or obtained from other spatial data sets. Shapes with matching or similar colors represent conceptually similar factors.**

3.1.1 Final Model Power

The final modified model had 128 degrees of freedom, resulting in an 88% likelihood of achieving a RMSEA value of less than 0.05 in the primary dataset (n=132), and a 61% likelihood of achieving the same in the secondary dataset (n=80).

3.1.2 Final Model Estimation

In the primary data set, the final modified model fit our data based on 50 ft and 100 ft riparian areas, but not for data based on the Site Potential Tree Heights (Appendix L). Again, the model based on 100 ft riparian area had the best fit. In the secondary data set, the final modified model fit the data based on 50 ft riparian areas, but not for either of the 100 ft or SPTH riparian areas (Appendix L).

3.1.2.1 Standardized vs un-standardized estimates

Figures 10-12 present the standardized parameter estimates from our final model results. Standardized parameter estimates are best used for comparisons of pathways within a model. For comparison between models, Table 4 presents the unstandardized parameter estimates for significant direct and indirect pathways in the model results. Unstandardized and standardized estimates of all direct and indirect effects, as well as confidence intervals of the unstandardized effects are provided in Appendix M.

In developing the model, some variables had to be multiplied or divided by several orders of magnitude so that the variances of the variables were similar in scale. For ease of interpretation, we re-scaled the unstandardized parameter estimates to interpretable units. These coefficients can be interpreted as the size of effect in the response variable caused by an increase in one unit of the predictor variable (Table 4, Appendix M).

In latent factors, the factors units are scaled to match one of the indicators. For example, Stream Reach Stability is scaled to be in units of stream reach stand height (meters). Sediment is scaled to be in units of ln(BSTI+1). Log units can be difficult to interpret; by back calculating we can estimate that 1 unit change in the latent factor Sediment is equivalent to a 171% increase in BSTI value (Appendix M).

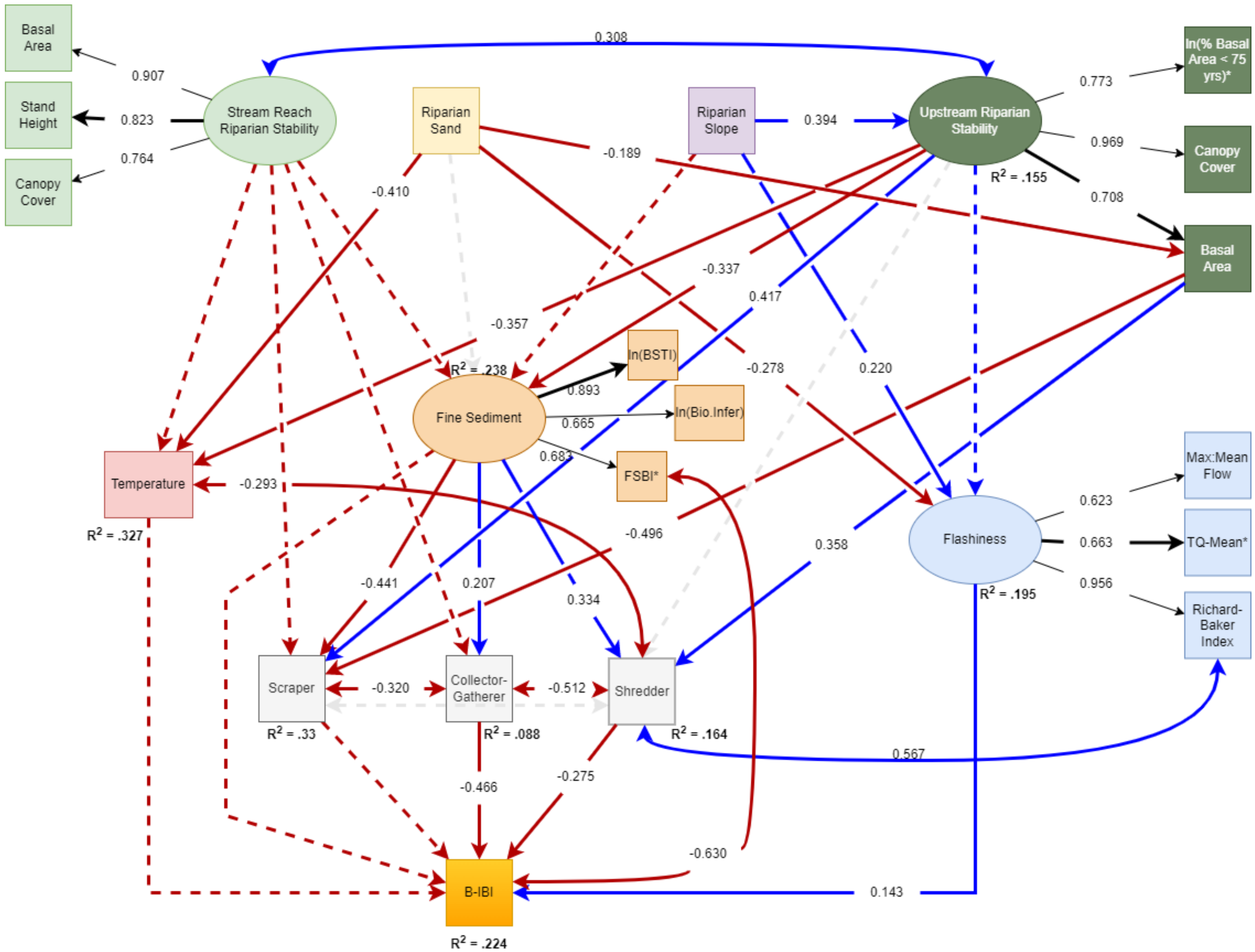

Figure 9. Standardized effect estimates based on 50 ft buffer data using the primary dataset. Dashed lines indicate direct pathways that are not statistically significant. Blue, red, and gray lines indicate positive, negative, and non- relationships, respectively. Asterisks denote variables that have had their scales flipped. Bold black arrows from latent factors show to which indicator its scale has been fixed. R^2 values indicate the amount of variation in the respective variable explained by the model.

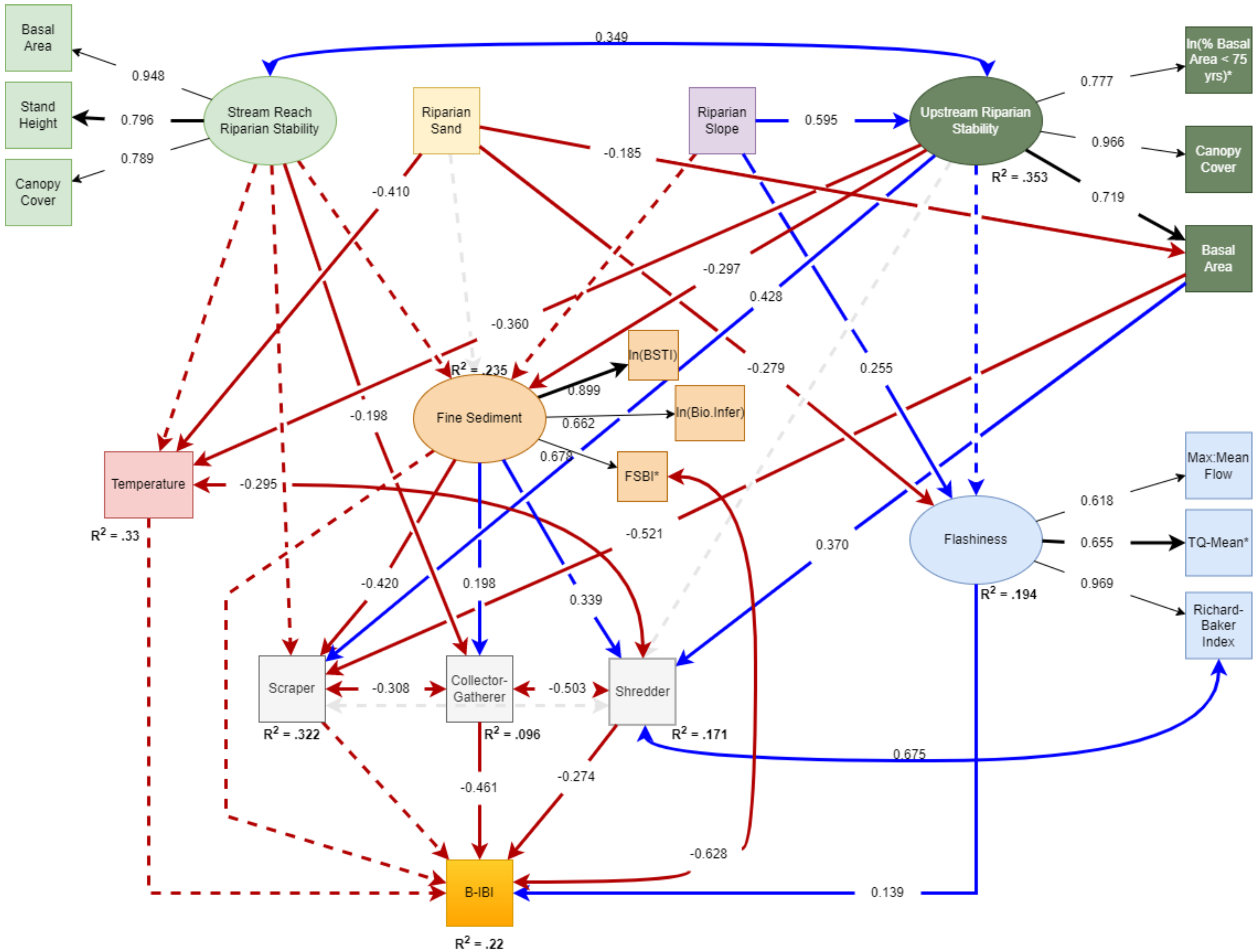

Figure 10. Standardized effect estimates based on 100 ft buffer using the primary dataset. Dashed lines indicate direct pathways that are not statistically significant. Blue, red, and gray lines indicate positive, negative, and non- relationships, respectively. Asterisks denote variables that have had their scales flipped. Bold black arrows from latent factors show to which indicator its scale has been fixed. R2 values indicate the amount of variation in the respective variable explained by the model.

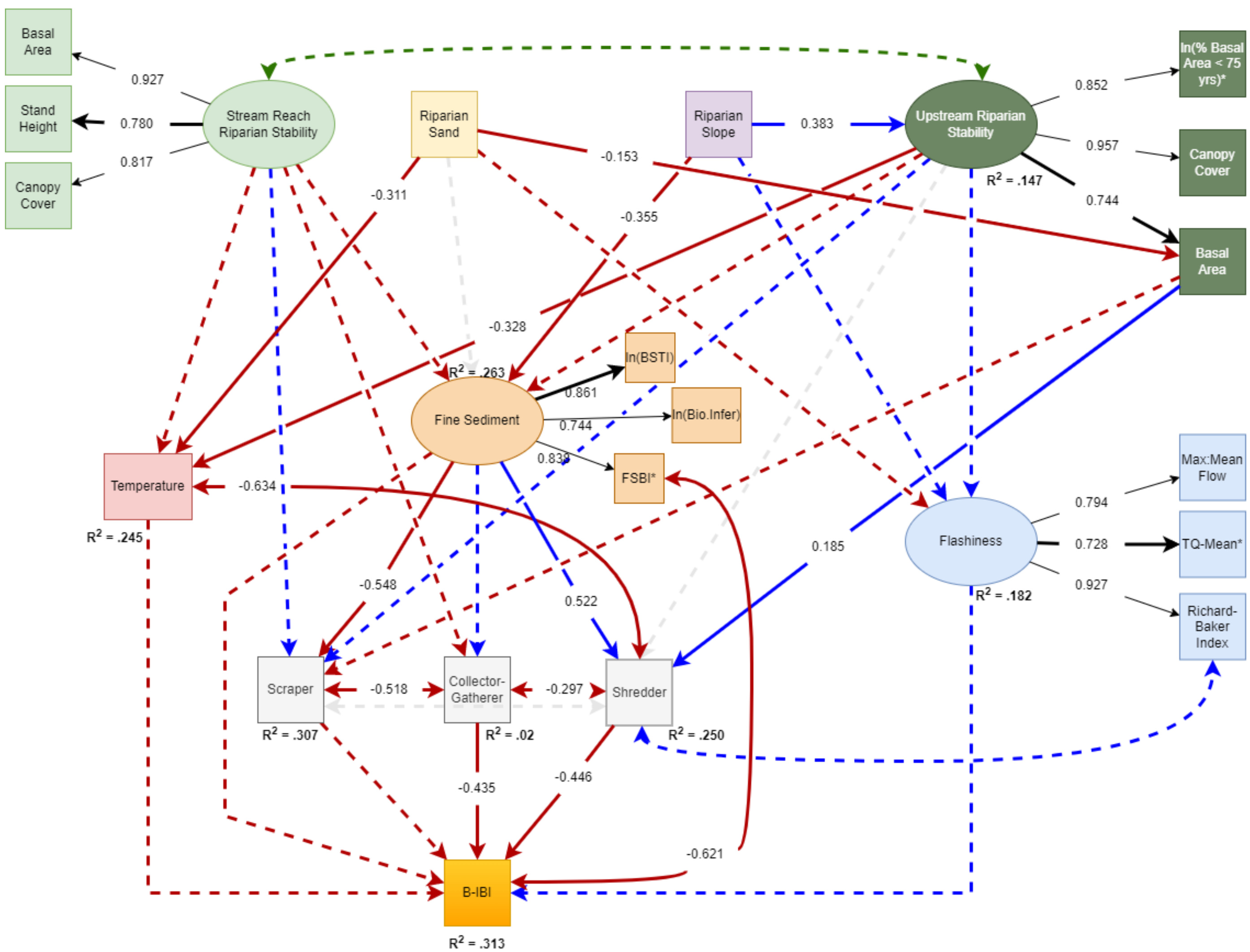

Figure 11. Standardized effect estimates based on 50 ft buffer using the secondary dataset. Dashed lines indicate direct pathways that are not statistically significant. Blue, red, and gray lines indicate positive, negative, and non- relationships, respectively. Asterisks denote variables that have had their scales flipped. Bold black arrows from latent factors show to which indicator its scale has been fixed. R^2 values indicate the amount of variation in the respective variable explained by the model.

Table 5 Unstandardized direct and indirect effect estimates for significant model parameters. Estimates that are in bold italics denote which model found a statistically significant link ($p < 0.05$). Indirect effects were also included if the confidence intervals did not include zero. Columns denoted with 'A' indicate results from the primary dataset, 'B' indicates results from the secondary dataset. Response variables with the format " X > Y" indicate total direct and indirect effects of parameter X on parameter Y.

Response Variable	Response units	Predictor Variable	Predictor units	50' [A]	100' [A]	50' [B]
Riparian Stability	10 m^2 riparian basal area/hectare of basin area	Upstream Riparian Slope	%	***0.04***	***0.12***	***0.04***
Sediment	171% change in BSTI [%]	Upstream Riparian Stability	10 m^2 riparian basal area/hectare of basin area	***-0.12***	***-0.05***	-0.09
Sediment	171% change in BSTI [%]	Upstream Riparian Slope	%	***-0.01***	-0.01	***-0.01***
B-IBI Score	index (1-100)	Collector-Gatherer	%	***-0.48***	***-0.48***	***-0.47***
B-IBI Score	index (1-100)	Shredder	%	***-0.44***	***-0.44***	***-0.85***
B-IBI Score	index (1-100)	Flashiness	100-unit change in TQM (%)	0.36	***0.35***	0.21
Collector-Gatherer	%	Sample Reach Stability	10 m of Stream Reach Stand Height	***-0.34***	***-0.45***	-0.26
Collector-Gatherer	%	Sediment	171% change in BSTI [%]	6.73	***6.37***	1.18
Scraper	%	Sediment	171% change in BSTI [%]	***-10.64***	***-10.10***	***-14.62***
Scraper	%	Upstream Riparian Stability	10 m^2 riparian basal area/hectare of basin area	3.53	***1.85***	0.84
Scraper	%	Upstream Total Basal Area	m^2 in riparian area/hectare of basin area	***-2.98***	***-1.62***	-0.93
Shredder	%	Sediment	171% change in BSTI [%]	6.95	***7.03***	***8.40***
Shredder	%	Upstream Total Basal Area	m^2 in riparian area/hectare of basin area	1.85	***0.99***	***0.80***
Temperature	Celsius	Upstream Riparian Stability	10 m^2 riparian basal area/hectare of basin area	***-0.58***	***-0.30***	***-0.49***
Temperature	Celsius	Upstream Riparian Sand	%	***-0.07***	***-0.07***	***-0.06***
Flashiness	100-unit change in TQM (%)	Upstream Riparian Sand	%	***-0.14***	***-0.14***	-0.13
Flashiness	100-unit change in TQM (%)	Upstream Riparian Slope	%	0.09	***0.12***	0.10

Total Basal Area	M^2 riparian basal area/hectare of basin area	Upstream Riparian Sand	%	***-0.03***	***-0.05***	***-0.03***
Forest Stability > BIBI Effects	index (1-100)	Upstream + Reach effects	Upstream + Reach effects	1.43	0.78	***2.16***
Upstream Forest Stability > BIBI Effects	index (1-100)	Upstream effects	10 m^2 riparian basal area/hectare of basin area	1.12	0.44	***1.96***
Reach Forest Stability > BIBI Effects	index (1-100)	Reach effects	10 m of Stream Reach Stand Height	0.31	***0.34***	0.2
Forest Stability > SH > BIBI Effects	index (1-100)	Upstream + Reach effects	Upstream + Reach effects	0.4	***0.2***	0.66
Upstream Forest Stability > SH > BIBI Effects	index (1-100)	Upstream effects	10 m^2 riparian basal area/hectare of basin area	0.36	0.16	0.64
Forest Stability > CG > BIBI Effects	index (1-100)	Upstream + Reach effects	Upstream + Reach effects	0.58	***0.41***	0.17
Upstream Forest Stability > CG > BIBI Effects	index (1-100)	Upstream effects	10 m^2 riparian basal area/hectare of basin area	0.38	0.16	0.05
Reach Forest Stability > CG > BIBI Effects	index (1-100)	Reach effects	10 m of Stream Reach Stand Height	0.2	***0.25***	0.12
Reach Forest Stability > CG	%	Reach effects	10 m of Stream Reach Stand Height	***-0.41***	***-0.52***	-0.26
Sediment > BIBI Effects	index (1-100)	Sediment	171% change in BSTI [%]	***-7.8***	-7.67	***-10.98***
Physical Condition > BIBI Effects w/ Forest Stability	index (1-100)	Sand + Slope effects	Sand + Slope effects	0.1	0.12	***0.3***
Slope > BIBI Effects w/ Forest Stability	index (1-100)	Upstream Riparian Slope	%	0.11	***0.14***	***0.25***
Physical Condition > BIBI Effects w/o Forest Stability	index (1-100)	Upstream Sand + Slope effects	Sand + Slope effects	0.09	***0.11***	***0.25***
Slope > BIBI Effects w/o Forest Stability	index (1-100)	Upstream Riparian Slope	%	0.07	***0.09***	0.17
Sand > Flow > BIBI Effects	index (1-100)	Upstream Riparian Sand	%	***-0.05***	***-0.05***	-0.03
Slope > Flow > BIBI Effects w/o Forest Stability	index (1-100)	Upstream Riparian Slope	%	0.03	0.04	0.02

4.0 DISCUSSION

Though we were unable to validate our final model, our exploratory results provide insight to our primary questions: if and how forest harvest impact B-IBI scores, and at what spatial scales. A key take-away of this study are that forest conditions resulting from forest harvest affect B-IBI, but very minimally. As measured by this study, forest harvest is unlikely to be the primary driver of low B-IBI scores in forested basins. The results also suggest that the influence of forest conditions on in-stream processes gets smaller as distance from stream increases, and that forest conditions in 100 ft buffer or larger on either side of a stream do not explain B-IBI scores better than forest conditions in a 50 ft buffer. Furthermore, while forest conditions at the both the reach and upstream riparian areas affect B-IBI score, upstream riparian forest conditions have a larger effect size. The mechanism of impact from forest harvest is primarily through fine sediment influencing the relative abundance of different functional feeding groups. Independent of forest condition, fine sediment alone has a much larger impact to B-IBI scores.

In this section we will discuss our primary findings. Additional discussion points can be found in Appendix N.

4.1 General findings

This study was intended to address whether forestry practices associated with timber harvest affect stream biological integrity. To that end, though the variance in B-IBI explained by our final model was modest ($R^2 = 0.22 - 0.31$), the model demonstrated significant causal pathways between forest conditions and B-IBI scores. Through direct and indirect effects, we estimated that for each unit of forest stability (equivalent to an increase of 10 m^2 riparian basal area per hectare across the whole basin for riparian effects and an increase of 10 meters in median stand height for stream reach effects), B-IBI values increased by 1.4 in the primary dataset to 2.2 points in the secondary dataset (Table 4).

These findings suggest that the effect of forest conditions on B-IBI scores, though statistically significant, is ecologically quite small. Because we scaled riparian basal area to basin size, this result at the riparian scale requires additional interpretation. In the dataset, the riparian basal area ranges from 1.2 to 9.5 m^2/basin hectare. This range captures riparian areas that are sparsely forested and relatively young, as well as riparian areas that are fully forested with mature trees. We can judge then that the effect on B-IBI is very small for the relative change in forest condition.

The forest conditions included in the model were specifically chosen for their ability to measure the extent, intensity, and timing of forest harvest. Though we related these conditions to forest "stability" in our latent factors, we could have flipped the scales of the indicator variables and referred to "harvest disturbance" instead. The final model adequately fits both primary and secondary data sets; however, it has not been independently validated. Consequently, we must be careful in our interpretation, and not overly assert our conclusions. With this caution in mind, the findings of our exploratory analysis can still have implications for forestry impacts and ecological function that are worthy of evaluation. Because of the lower analytical power of the secondary dataset, we will rely more heavily on findings from the primary dataset in our interpretation. Though not all links in the model were significant, or consistently significant, we refrained from removing insignificant links unless we felt we had sound reasoning for doing so. This is because in a complex system, multiple weak mechanisms can have a strong effect when acting in concert.

4.1.1 Buffer width

Comparisons of our model results suggest that forest conditions within 50 ft of a stream have the strongest influence on B-IBI scores. The model results suggest that riparian conditions in a 50 ft buffer on either side of a stream had roughly twice the effect on B-IBI scores than those in a 100 ft buffer (Table 4, row 19). Furthermore, the model failed to meet fit criteria using buffers based on Site Potential Tree Heights, which were typically much larger (100 to 260 feet, median = 200 feet). This suggests that forest conditions in areas that extend further than 100 ft from a stream are less influential to in-stream conditions perceived by the B-IBI and contribute more statistical noise in this analysis. Comparisons of model fit between 50- and 100-ft riparian buffers and the two datasets yielded mixed results. In the primary dataset, model fit was best using data based on 100 ft buffers. However, in the secondary dataset, the model only met fit criteria when based on 50 ft buffers.

To evaluate the model fit discrepancy in 50- vs 100-ft riparian areas between the primary and secondary datasets, we took a closer look at the unstandardized parameter estimates in the various model results (Table 4). In model results based on 100 ft riparian areas as compared to 50 ft riparian areas in the primary dataset, we observed a stronger link between riparian slope and upstream riparian forest stability, and between reach forest stability and collector-gatherers. Most other links in the 100 ft riparian results were similar in strength or slightly weaker than those based on 50 ft riparian areas (Table 4). Because most pathways stemming from forest stability and basal area were not subsequently improved at the 100 ft scale, we judge that the larger spatial extent does not add information about in-stream influences stemming from forestry practices associated with timber harvest. Though riparian condition variables had large correlations between the three sets of buffers, we observed enough variability to feel confident that that these differences drove the spatial pattern.

Given these results, and the decreased effect size of riparian stability on B-IBI scores at 100 ft, we do not feel that the data supports such wide buffers when benthic biological integrity is the primary land management objective. It is worth noting that this analysis does not address the universal range of riparian benefits to instream processes and biota but is focused on benthic macroinvertebrates only. A cautious land manager may consider intermediate buffers of 75 ft, as proposed by Barnowe-Meyer et al. (2021).

Based on this conclusion, we will address only results of the models based on 50 ft riparian areas for the remainder of the discussion.

4.1.2 Upstream vs Reach Influence

Though not consistent between the primary and secondary datasets, these results underscore that B-IBI scores integrate conditions in the immediate stream reach and in the whole upstream area. The results further highlight the impact of sediment on benthic communities, which is derived, in large part, from disturbances in upstream riparian areas. According to the strength of direct and indirect links in the model (Figure 10), upstream forest conditions affected B-IBI scores primarily by shaping the benthic community structure via sediment impacts and changes to basal area (which likely reflects detrital inputs). Some of these effects occurred independent of forest stability; for example, upstream forest stability did not seem to significantly contribute to the proportion of shredders in the community. Instead, shredders seemed to respond to the density of basal area in the upstream riparian area, indicating that shredders are rather agnostic about whether leaf litter comes from densely planted young recovering forests or old established forests.

In contrast to the upstream area, forest conditions in the stream reach affected benthic community structure and B-IBI scores primarily by changing photosynthetic energy inputs. Collector-gatherers seemed to increase in direct response to disturbance in the stream reach, most likely reflecting an influx of direct light and possibly nutrients to increase primary production (Kiffney et al. 2003, Gravelle et al. 2009). In contrast to collector-gatherers, and despite reach-specific controls to light and algae production, the proportion of scrapers in the community appeared to be more closely related to conditions in the whole upstream riparian area rather than within the reach. This may reflect a sensitivity to sediment and a preference for the biofilms promoted by low light environments found under stable forest conditions (Kiffney et al. 2003). Old growth forests have been found to have greater variability in light penetration and increased gaps in canopy cover relative to immature forests (Kaylor et al. 2017), which may promote scrapers in light-controlled microhabitats.

Independent of upstream forest stability, high basal area in the upstream riparian zone appeared to have had a separate negative effect on B-IBI scores through shredders. This apparent contradiction in effect on B-IBI is likely because basal area is a product of both tree diameter and tree density; the same basal area can be achieved by having many young, small diameter trees, or fewer old, large diameter trees. An additional factor to consider is that tree age influences the relationship between the basal area of an individual tree and its leaf area (Sonohat et al. 2004). In many species, as trees grow taller leaf area declines due to hydraulic constriction even as basal area continues to increase (McDowell et al. 2002, Ryan & Yoder 1997). Thus, younger stands can have denser canopies than older stands for the same basal area. It is possible that the separate negative relationship between basal area and B-IBI, as indicated by the model, is driven by densely planted immature forests, with greater shading and detrital inputs than older forests. Kaylor et al. (2017) found that it can take well over a century for canopy conditions to return to an old-growth baseline for these functions. Given that most forests in Western Washington are less than 100 years old, and nearly half are less than 40 years old (WDFW 2005, DNR 2007), it is possible that many riparian forests are less able to support a diversity of functional feed groups in favor of shredders.

4.1.3 Fine Sediment

Fine sediment seems to be the main stressor impacting B-IBI scores in forested basins, by impacting the access and availability of food resources and habitat to different macroinvertebrate functional feeding groups. Because our sediment values were not obtained from direct observation of the stream sediments, we used three independently constructed methods of predicting sediment impacts (based on the benthic macroinvertebrate community) as indicators of true fine sediment impacts. Though the predictive ability of each indicator alone is not particularly strong (literature reported r^2 = 0.49 for BSTI, 0.42 for bio,infer, not reported for FSBI), using them in conjunction in a latent factor model harnesses the shared variance among the three, which allows us to separate the sediment signal from other sources of error in each variable.

Fine sediment alone had a far larger effect on B-IBI scores than all direct and indirect pathways from forest stability (including pathways that include sediment as a causal response to forest stability). For every unit change in the latent factor fine sediment (which corresponds to a 171% increase in BSTI value), B-IBI scores were estimated to decline by 7.8 (in the primary dataset) to 11 points (in the secondary dataset). Though natural sources of fine sediment (i.e., landslides, bank erosion) may contribute to fine sediment impacts in these systems, this finding raises the possibility that legacy impacts from previous less-regulated forest harvests may play a role in current B-IBI scores, even when the forest conditions have generally recovered.

4.2 Additional considerations

While our original questions focused on the relationship between forest harvest and B-IBI scores, an expansion of that question would be why are scores so variable in forested basins? What contributes to marginal B-IBI scores in basins that are primarily forested and where the biological potential far exceeds observed scores? To obtain at least a partial answer, we must consider the basins that met our initial inclusion criteria (≤70% vegetation, ≤10% developed) but which were later rejected for various non-forestry land uses and land covers that we suspected would affect scores: mines in the basin, significant amounts of agriculture close to the sampling point, significant residential areas close to the sampling site, major roads close to the sampling site, or other reasons (downstream of a hatchery, or containing a known source of chemical pollution). Examining the most recent scores in these basins showed that they score more poorly than the basins we did not reject (Figure 13). The confidence intervals of the two groups do not overlap, indicating that the medians are significantly different. This suggests that, in general, some of the variability in B-IBI scores seen in highly forested basins can be attributed to small amounts of land uses that, either by their nature or their location in the basin, have an outsized effect on B-IBI scores (Figure 14). While there was no remaining relationship between B-IBI score and percent developed or agricultural land cover area in the basins we retained in our dataset, these land covers must still generally be considered important factors when identifying sources of impacts in forested basins.

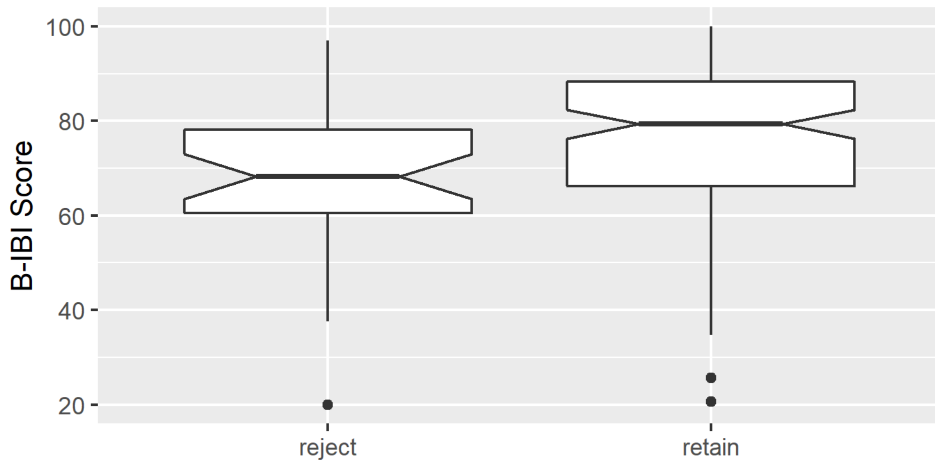

Figure 12. Recent B-IBI scores of forested basins that were either rejected from the dataset or retained. Rejected basins met our initial inclusion criteria (≤70% vegetation, ≤10% developed) but were later rejected for various non-forestry land uses and land covers that we suspected would affect scores. The retained basins are the 132 basins included in the primary dataset. The notches display a 95% confidence interval around the median B-IBI score.

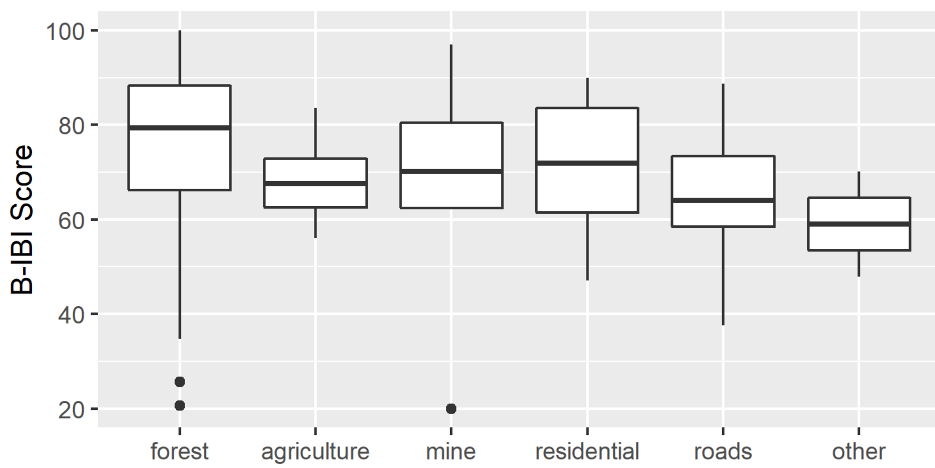

Figure 13. Recent B-IBI scores of forested basins that met our initial inclusion criteria (≤70% vegetation, ≤10% developed), binned by influential land use categories. Basins with minimal influence outside of forestry were retained. Basins that were later rejected from the dataset were binned by the reason for rejection: large amounts of agriculture close to the sampling point, mines in the basin, significant residential areas close to the sampling site, major roads close to the sampling site, or other reasons.

4.3 Sources of Uncertainty

Models

Our model results and conclusions rely on several assumptions, and we must address the uncertainty surrounding these assumptions. A large source of uncertainty is our assumption that the modelled values for forest conditions, stream flows, temperature, and fine sediment impacts are reasonable and reliable. Prior to using these data sources, we examined validation results from other studies and, when possible, evaluated model results against a subset of our own dataset where comparable empirical data was available (Appendix G). Regardless, relying on modelled values rather than empirical observations introduces an added dimension of uncertainty. How these models incorporate land cover, the scale at which these models incorporate land cover, and the temporal frequency that the models update land cover is particularly salient to using these data sources to measure forest harvest impacts.

The retrospective flow dataset uses a land surface model that integrates satellite and ground-based observations with three-hour time-steps extending back to January 1979. The flow dataset can be used to infer flow frequencies and perform temporal analyses with hourly streamflow output (NOAA 2022). However, the current iteration of the model relies on a static vegetation model that changes over seasons but does not change from year to year. As a result, this dataset is unlikely to be appropriate for evaluating flow changes resulting from changes in land use. This limitation was not clear to us prior to the completion of the study. Similarly, the retrospective stream temperature dataset used a time-invariant measure of riparian canopy cover as a model input, and may not be appropriate for evaluating temperature changes resulting from changes in land use. Despite this uncertainty associated with these two specific pathways, we feel confident in the broader results of this study in how forest conditions relate, directly and indirectly, to stream biotic integrity.

Scale

The scale of our forest conditions and soil datasets is another source of uncertainty. In our primary dataset, the apparently stronger links between slope, sand and upstream forest conditions at the 100 ft scale may stem from differences in spatial resolution between the soil and slope datasets (10 m) and forest condition datasets (30 m). This raises a significant caveat—at 30 m resolution, the 50 ft buffer on either side of the stream is roughly the width of one cell. The method we used to summarize the spatial data include cells that have their center in the buffer polygon, so at such low resolution there is some overlap in the vegetation conditions between 50 and 100 ft. This is certainly a limitation to the study. However, the results of this study consistently demonstrate that forest conditions have less influence on B-IBI scores as the distance from the stream increases.

Estimated Fine Sediment

As discussed in the methods section, the estimated sediment values are calculated using the same macroinvertebrate sample data that is used to calculate the B-IBI. This shared data source risks the independence of the sediment and B-IBI data. However, the different sediment and B-IBI calculations differ in how the data is used, in terms of the focus taxa and the statistical methods employed. The B-IBI score was developed to look for broad anthropogenic impacts to stream health, while the sediment models were developed for specific impacts from fine sediment. As a result, correlation between the B-IBI scores and estimated sediment values is not surprising, but it is also not necessarily a statistical inevitability. FSBI index scores and inferred sand/fine sediment values have been used elsewhere to inform sediment conditions and interpret B-IBI scores (Marshalonis & Larson 2018, Larson et al. 2019).

Lack of Forest Practice Data

In interpreting our results, we must also consider the additional limitations of the current study. We could not obtain data pertaining to numerous forestry-related factors, including mode of harvest, application of fertilizers and pesticides, slash-disposal methods, and road use and maintenance. Furthermore, more subtle and less intensive harvest methods such as thinning or selective harvests are unlikely to be reliably detected in our spatial data sets.

Upland Effects

In addition, in choosing our model design we made the decision to focus solely on riparian forest conditions, because upland and riparian forest conditions were so highly correlated. Though the model results indicated that 100 ft buffers offer no added explanatory ability for B-IBI scores over 50 ft buffers, this does not mean that all impacts to the stream occurred in only that narrow strip. It is likely that different mechanisms influencing in-stream conditions occur at different scales extending out from the stream and that upland conditions impact streams. However, as this study is focused on B-IBI, we restricted our scope to only those conditions most likely to impact those scores.

4.4 Application to Urban Streams

While the primary goals of this work were to evaluate if forestry practices associated with timber harvest impact B-IBI scores, the mechanisms by which these impacts occur, and the spatial scales on which they occur, these results may also inform stormwater managers in more urban environments. “Urban stream syndrome” is an umbrella term for the suite of disruptions to natural stream functions caused by development, and the ensuing stressors to the stream ecosystem. Because the disruptions can be so numerous in a highly developed environment, it can be difficult to disentangle and prioritize exactly what stressors are most problematic to the stream community, and subsequently what the best remedy may be. By focusing on forested basins where the stressors are less tightly coupled, we may be able to generalize the relative impact of specific stressors.

The results of this study indicate that land cover in riparian areas closest to the stream play the greatest role in stream health. Previous studies in more urban stream systems have found significant effects of land use on B-IBI scores at the whole basin scale as well as on the riparian scale (King County 2019). However, this study has found that in forested basins, land cover and forest conditions have less and less explanatory power as the distance from the stream increases. One possible explanation for why urban streams seem to respond to basin effects may have to do with increased impervious cover across the basin combined with piped stormwater conveyances. These have the effect of greatly increasing the amount of basin area that delivers water and sediments directly to a stream. In other words, impervious surfaces and stormwater conveyances artificially increase the effective riparian zone of a stream in an urban environment.

Furthermore, this study indicates that sediment impacts may be an especially important factor to consider in urban streams, underscoring the findings of previous work (Larson et al. 2019). Some previous studies in urban streams have pointed towards both stormwater flashiness and fine sediment as a cause of poor B-IBI scores (Marshalonis & Larson 2018, DeGaspari et al. 2009). However, because of multiple disruptions to natural stream function, urban streams are likely to simultaneously suffer from fine sediments and flashy storm flows. This co-occurrence can make causal relationships difficult to assess. This study suggests that flashy flows alone do not necessarily negatively impact B-IBI scores, though as discussed in the section of uncertainty, the flow data used in this study may not adequately take land cover into consideration.

4.5 Recommendations for forestry practices

The model results can help to evaluate whether current forestry regulations in Washington State are sufficiently protective and may help to give insight to the legacy effects of past forestry practices associated with timber harvest. While this study could not evaluate all forest practice regulations, we can apply the results to regulations pertaining to stream buffers. These recommendations are purely based on impacts to in-stream biotic integrity, and do not take economic considerations into account. The economic cost-benefit of these recommendations are outside the scope of this work, and so must be weighed separately.

4.5.1 Overview of current regulations

Current forest harvest regulations[7] in Washington State have established a tiered system for determining what, if any, harvest activities can occur in the vicinity of a stream, and how far these protections extend from the stream. In Western Washington, the system depends on the type of the stream (e.g., if the stream is fish-bearing, if the waterbody is considered waters-of-the-state, and if the steam is perennial or seasonal), the class of the stream, (which in turn is determined by the growth potential of trees at the site) and the stream bankfull width. Furthermore, the Riparian Management Zone is divided into three parts: the Core Zone Width, the Inner Zone Width, and the Outer Zone Width.

Fish bearing streams
In fish-bearing or waters-of-the-state stream-types, the width of the inner and outer zones vary by stream class and bankfull width. However, the core zone is maintained as a 50 ft no-harvest buffer along the whole stream corridor until the stream type changes. If this minimum buffer provides adequate shade to the stream, limited selective harvest may occur in the inner zone, and 20 conifer trees per acre must be left in the outer zone. If the 50 ft no-harvest buffer does not provide sufficient shade to the stream, the no-harvest buffer is extended out to the inner zone, which ranges between 10-100 feet.

Non-fish bearing streams
In non-fish-bearing perennial streams, current regulations call for no-harvest riparian buffers to occur in 50 to 56-foot radius patches centered around 'sensitive areas' such as

[7] https://www.dnr.wa.gov/publications/fp_fpi_introduction.pdf

seeps, springs, unstable slopes, alluvial fans, and confluences. Where the stream intersects with a fish-bearing stream, the regulations require a 50 ft wide no-harvest buffer of up to 500-ft in length depending on the length of the non-fish-bearing perennial stream. Outside of these specific areas, the riparian areas may be harvested to the stream edge with some limits. A 30-ft 'equipment limitation zone' that extends on either side of the stream channel requires sediment mitigation measures if more than 10% of the surface soils in the zone are exposed by ground-based harvest activities, skid trails, stream crossings, or partially suspended cabled logs (WAC 222-30-021(2)(a)). These measures include using mulching and groundcover seeding.

Seasonal streams have no buffer requirements but are subject to the same equipment limitation zone as described for non-fish-bearing perennial streams.

4.5.2 Recommendations

Forested basins have excellent biological potential (Figure 1). However, actual biotic integrity in forested basins varies widely. This study showed that forest harvest does impact biotic integrity in forested basins, though within the limits of this study the influence is very small.

Based on the study results, it is likely that the current regulations in place for fish-bearing and waters-of-the-state stream types are sufficiently protective of B-IBI scores by prescribing a minimum 50 ft no-harvest buffer and by limiting harvest activities in the riparian zone beyond based on ecological considerations.

However, the model results indicate that harvest impacts to non-fish-bearing and seasonal streams can negatively influence the biotic integrity of receiving waters (as measured by the benthic macroinvertebrate community). This influence is driven largely by sediment impacts from upstream disturbances and leaf cover in upstream riparian areas. Furthermore, sediment impacts from all sources appear to be the largest driver of low B-IBI scores in our model.

Recommendation to strengthen existing practices in non-fish bearing and seasonal streams

Consequently, we recommend strengthening existing practices in non-fish-bearing and seasonal streams to prevent or reduce fine sediments from entering the channel. This could be achieved in different ways.

Current regulations already provide specific guidance to limit sediments reaching streams from haul roads, road crossings, landings, and streamside equipment use near all stream types. However, existing regulations pertaining to equipment limitation zones could be strengthened. Where soils are heavily disturbed beyond the 30-ft equipment limitation zone, and where vegetation has been removed up to the channel edge, there may not be a sufficient barrier to slow surface flows and prevent sediment from entering the stream channel. In such instances, mitigation measures may be advisable regardless of soil disturbance in the equipment limitation zone. The use of other stormwater BMPs should also be considered, such as straw waddles staked parallel to the stream channel.

Another possibility to reduce sediment impacts would be extending 50 ft no-harvest buffers along the whole stream length in non-fish-bearing perennial streams. The relationship between upstream forest conditions and stream temperatures found in this study supplies an additional argument in favor of this approach. Though the model results suggested that temperature did not have a significant effect on B-IBI scores, the findings have implications for temperature-sensitive species in receiving waters downstream of confluences with non-fish bearing streams. The short and limited buffers prescribed in non-fish-bearing perennial streams are unlikely to significantly cool waters discharging from harvested catchments. Thus, large amounts of forest removal in headwater streams could contribute to warmer temperatures in fish-bearing streams. This concern has similarly been raised by a workgroup convened by Washington Department of Natural Resources to address temperature degradation in streams and rivers (Barnowe-Meyer et al. 2021).

Recommendation to thin dense riparian stands

The issue of leaf cover in the riparian zone is complex. As an indicator of forest stability, sufficient basal area (and associated leaf area) is necessary for a stable, well-functioning stream ecosystem. However, independent of forest stability, high basal area in the riparian zone appears to have had a separate negative effect on B-IBI scores through shredders. As discussed previously, it is possible that the separate negative relationship between basal area and B-IBI, as indicated by the model, is driven by densely planted immature forests, with greater shading and detrital inputs than older forests. With that in mind, we recommend managing riparian forests to mimic light conditions in old-growth stands. This may include thinning existing dense stands and establishing maintenance guidelines for thinning densely planted saplings to achieve desired future conditions. This finding supports previous recommendations in the 2007 “Future of Washington Forests” report to thin over-crowded riparian areas to achieve biodiversity benefits (WA DNR 2007). Existing regulations pertaining to “desired future growth” of riparian forests already seek to create or retain forest stands with characteristics similar to mature, unmanaged forests. However, current regulations do not permit thinning in the core riparian zone of fish-bearing streams.

4.5.2.1 Future Research Needed

In this study, we were not able to evaluate the role of forestry-related chemical applications (pesticides, herbicides, fertilizers) on in-stream biotic integrity. As a result, we cannot rule out additional effects of forestry practices on stream health that were not captured by the study. Only aerial chemical applications require permitting, and no reporting is required for other methods of application (motorized ground and manual applications). Though applications are generally prohibited in inner and core zones of fish-bearing streams and waters-of-the-state, and additional buffers are prescribed when applying chemicals aerially, it is not clear how effective these buffers are at keeping these chemicals from entering waterways. For this reason, we recommend additional research be done to study the effects of forestry-related chemical applications on stream biotic integrity. To effectively study the potential relationships, such research would need to track pesticide, fire retardant and other chemical applications, the chemical used, the timing, quantity per unit area and method of application, and the location of application.

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Appendix A – Analytical approach

Analytical Approach

In a 2017 paper, Lunn et al. used meta-analysis to evaluate the impact of forest harvest on stream biota as measured across numerous studies done on the subject. That effort expanded on previous meta-analyses done on the subject (Mellina & Hinch 2009, Richardson & Beraud 2014), and represents a thorough review of the current body of knowledge. In their findings, they observed "little consistency in the direction and magnitude of impacts on stream biota", in contrast to the expected responses. In a key finding, the authors concluded "examining aquatic biotic responses to timber harvest alone is too simplistic" and noted several factors likely to interact with or confound the ability to measure harvest stressors. These include environmental regimes, study methodology, harvest regimes, mitigation measures and life history of the affected taxa. They critique the ability of traditional univariate statistical methods to discern causal pathways, recommending a more holistic approach that "simultaneously examines the direct and indirect effects of multiple factors and responses". The authors strongly recommend that future research use Structural Equation Modelling (SEM) to better understand the complex mechanisms underpinning the biotic responses to forest harvest. In this study, we used SEM to evaluate the impacts of forest harvest activities on benthic index of biotic integrity (B-IBI), following the recommendations of Lunn et al. (2017).

SEM is a scientific and statistical framework that, while developed in the 1970's and widely used in other scientific fields for decades, has been underused in the ecological sciences and has only recently gained popularity in ecological research (Grace 2015, Laughlin and Grace 2019). Traditional analytical methods provide descriptive statistics and bivariate correlations among myriad variables as a means of identifying potential relationships. Using these methods, researchers risk conflating correlation with causation and identifying "relationships" that are coincidental (Grace 2006). In addition, traditional methods are not well suited to detecting indirect effects (Laughlin and Grace 2019). In contrast, SEM is rooted in ecological theory and prior knowledge of the system being studied and can analyze factors in complex systems simultaneously. For this reason, it can be used to build and evaluate causal hypotheses in complex systems and reveal relationships not apparent in simple bivariate relationships (Laughlin and Grace 2019).

Further advantages of SEM include the ability to estimate error terms for variables included in the model. These error variances represent both random measurement error and unspecified causal influences not included in the model. This gives the researcher flexibility to focus on specific pathways of interest, without necessarily needing to model all possible sources of influence. Another advantage of SEM is the ability to construct latent factors. A latent factor is one that is hypothesized to exist but has not been directly measured (Grace 2006). In practice, latent variables represent abstract concepts that are difficult or impossible to measure, but which may have real causal effects on observable variables. In SEM, multiple related observable variables can be used as indicators to estimate the "true" effect of the latent factor. As an example, "body size" in birds could be considered a latent variable indicated by measures of body mass, wing length and beak length.

Grace et al. (2010 and 2015) provide excellent overviews of the process behind SEM (Figure 1). The first steps are focused on translating abstract ideas into testable expectations using conceptual models and meta-models. The next steps are focused on evaluating the data to consider how best to represent causal pathways and estimate the structural model. Once a model is specified, we then evaluate how well the model fits with the observed data. This phase of SEM is considered confirmatory SEM. If the model fit is suitable (Chi-Square p-value > 0.05, RMSEA value < 0.05, CFI > 0.05, and SRMR < 0.08), we can conclude that the evidence supports our causal hypotheses and begin interpreting the results.

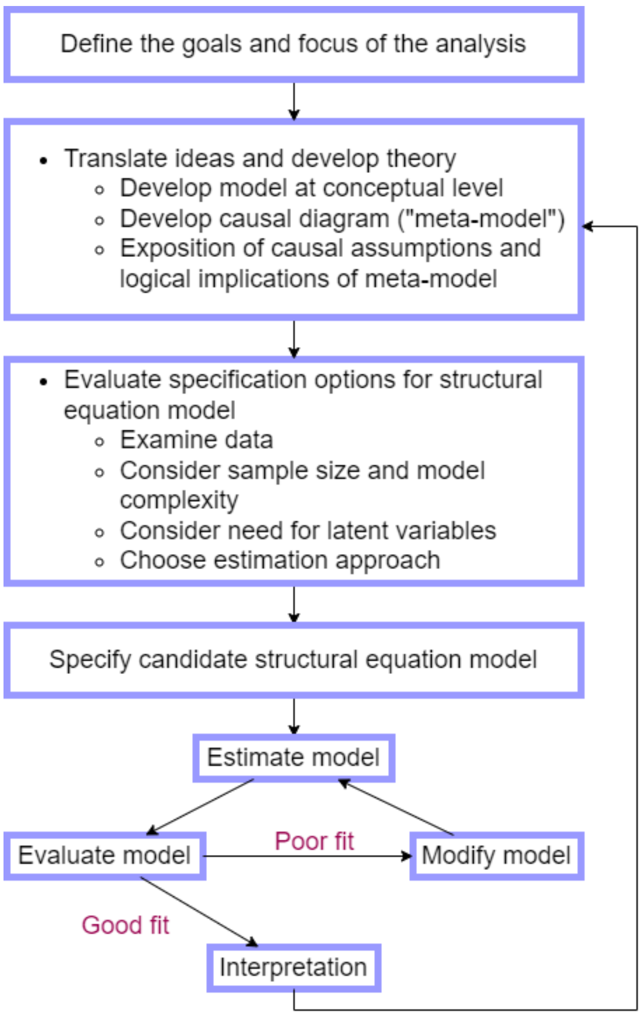

Figure 14. The structural equation modelling workflow. The process is confirmatory if no modification step is needed. Otherwise, the process is considered exploratory, and the results are provisional until the model can be validated on a new dataset. Modified from Grace et al. (2010 and 2015).

If the initial model fits the data poorly, we enter a second and iterative phase, known as exploratory SEM. In this exploratory process, we modify our causal hypotheses based on the links suggested by the initial failed model, re-estimate, and re-evaluate. Critical to this phase, the researcher must use clear theoretical knowledge to guide the modification of the model to ensure it is ecologically coherent. Once a suitable model is specified and found to fit the observed data, the model results are considered provisional until they can be re-tested on a new set of data for confirmatory SEM. While the results of exploratory SEM can still provide useful insights and help researchers build causal hypotheses, researchers must apply caution when presenting the results and drawing conclusions.

In the main report and appendices, we outline the steps taken to build and evaluate a suitable structural equation model to examine causal pathways between forest harvest activities and B-IBI scores.

Appendix B - Literature Review Summary

Summary of Literature Review Findings

Streams in forested watersheds typically support diverse macroinvertebrate communities, and yet there can be significant variability in the benthic index of biotic integrity (B-IBI) scores used to characterize stream conditions. To assess how variability in forest conditions and harvest practices may explain differences in macroinvertebrate community structure and B-IBI scores, we conducted a review of scientific literature (Appendix C, Table 1). The review includes 37 publications (Table 1), from which we have compiled a list of factors likely to influence the relationship between forestry practices associated with timber harvest and B-IBI scores (Appendix C, Table 2). In addition, we have included these factors in a preliminary conceptual model (Appendix D) that we will refine with additional expert input. Once refined, we will use this model to develop a causal framework and complete our statistical analyses.

Across our literature review, we found abundant evidence from both observational and experimental studies that macroinvertebrate communities respond to forest harvest stressors (Fuchs et al. 2003, Moldenke and Ver Linden 2007, Zhang et al. 2009, Frady et al. 2006, Jackson et al. 2007, Kedzierski and Smock 2001, Chizinski et al. 2010, Haggerty et al. 2004, Kiffney et al. 2003, Lunn et al. 2017, Swank et al. 2001). Some studies found no response (Gravelle et al. 2009, Orvegard 2019), though these studies also had relatively low statistical power. Where change was detected, the exact nature of the change was not always consistent across studies; however, a few generalizations can be made. Forest harvest activities typically increased macroinvertebrate abundance and biomass immediately after harvest (1-5 years post-harvest). Forest harvest activities also frequently changed the proportion of the community composed of grazers/scrapers/collectors versus shredders soon after harvest. The durability of these changes is difficult to assess, as few studies have looked at longer-term changes. However there is some evidence that these changes can remain for decades post-harvest (Zhang et al. 2009).

Impacts to total richness, EPT (Ephemeroptera, Plecoptera, Trichoptera) richness and other common metrics of macroinvertebrate community composition were more variable across studies where they were measured. In some cases, the metrics showed no change, while some studies conflicted in the change they found. This inconsistency was also observed in a broad meta-analysis of the effects of timber harvest on stream biota by Lunn et al. (2017), in which they suggest the inconsistency may be attributable to confounding variables that can cause studies to not be directly comparable to each other. As they discussed: "A key, yet perhaps unsurprising finding from our broad review is that examining aquatic biotic responses to timber harvest alone is too simplistic. The observed heterogeneity suggests a significant influence of variations in environmental factors, methodology, harvest regimes, and measures to mitigate impact within and among studies, as well as species and stage specific responses by biota."

Indeed, previous studies of the effects of timber harvest on macroinvertebrate communities have occurred in a broad swath of physical settings, with varying amounts of time post-harvest, and generally at small scales. The underlying physical characteristics of a stream basin can significantly determine the macroinvertebrate community; for example, geology, stream order, elevation and stream gradient can all influence macroinvertebrate community structure (Haggerty et al. 2002, Clarke et al. 2008, Moldenke and Ver Linden 2007, Walker et al. 2021, Frady et al. 2006, Black et al. 2004, Maloney and Weller 2011). While these characteristics will not change with harvest activities, they can interact to influence the strength and direction of the resulting impact. The time since harvest can also impact the type and intensity of stressors affecting the benthic macroinvertebrate community. Some non-linear relationships between forest harvest and stream processes take decades to develop (Kaylor et al. 2016, Swank et al. 2001, Perry and Jones 2016, Hall et al. 2018); however, our understanding of these long-term changes is limited. Most studies investigating impacts to macroinvertebrates take place in the first five years after harvest, while only a few specifically look for impacts on time scales greater than 20 years (Frady et al. 2006, Fuchs et al. 2003, Swank et al. 2001) and none beyond 40 years (Lunn et al. 2017). The spatial scale of impacts can also vary; while most studies examine responses in reaches that directly abut disturbed areas, few studies look for impacts on larger scales (Zhang et al. 2009, Gravelle et al. 2009).

A broad look across the literature supports the idea that to generalize the impacts of forest harvest, one must incorporate physical characteristics of the stream, time, and spatial extent of harvest activities as co-variables. In our review, we included observational studies of the factors that most contribute to macroinvertebrate community structure in the absence of forest harvest (Entrekin et al. 2014, Haggerty et al. 2002, Black et al. 2004, Clarke et al. 2008, Marshalonis and Larson 2018, King County 2014, King County 2015b, King County 2019, Maloney and Weller 2011, Walker et al. 2021). We also included studies of forest harvest impacts to streams that did not specifically examine macroinvertebrates (Hall et al. 2018, Moore et al. 2004, Dickerson-Lange et al. 2016, Mellina and Hinch 2009, Kaylor et al. 2016, Moore and Wondzell 2005, Perry and Jones 2016, Bailie and Neary 2016). Combining these sets of studies was useful for determining how macroinvertebrate communities are shaped across a wide range of environmental gradients, and observing where forestry practices associated with timber harvest may have the most salient intersections with those gradients.

From the literature review, we have identified the natural processes that forest harvest activities are most likely to disrupt, and which are also likely to impact macroinvertebrate communities and B-IBI scores. We have divided harvest activities into four main sources of impact: road construction, post-harvest practices, riparian harvest activity, and basin-wide harvest activity. Though impacts resulting from activities in the riparian zone and basin may be similar at times, the riparian zone has special ecological and regulatory significance and so we have decided to separate it here. The impacts of these four categories are themselves influenced by external factors: the regulations and sustainability certifications (that may augment regulatory protections) in place at the time of harvest, the presence of protected fish in the stream, and the time since harvest. An additional factor, season at time of harvest, may also play a role in mediating the impacts of the four categories. However, at this time we’ve found no literature to review specific impacts of harvest depending on season. Citations for each factor listed below and within the conceptual model can be found in Table 2.

The primary impact from road construction is a direct increase in sediment load to the stream bed. Though the volume of sediment entering the system may drop significantly after road completion, the impacts from the sediment pulse may linger for decades, and long-term elevated sediment inputs have been observed. Road construction may also directly influence hydrology and synchronicity by intercepting groundwater and surface flows and re-routing them. The intercepted flows may result in increased, more synchronized stream flows if the roads or associated ditches routes the flows directly to the stream. However, roads and ditches may instead slow the flows or even shunt them to elsewhere in the system, thereby decreasing and/or de-synchronizing flows within a given reach. The effect of roads on stream hydrology and synchronicity is mediated in part by construction methods, best management practices (BMPs) used, and the physical characteristics of the stream basin.

Post-logging practices may directly influence the quantity and size of debris and organic material in the stream bed, depending on the method of slash removal used. Stream cleaning (in which large wood is either selectively or thoroughly removed from stream channels as part of post-harvest clean-up efforts), leaving slash in place, or slash burning are different methods with different outcomes for in-stream debris and organic material. How an area is re-planted after harvest also directly impacts forest vegetation (basin-wide and in the riparian zone) by influencing the species composition, age structure, density, canopy cover, and leaf distribution of re-growth. Alteration of riparian and basin-wide vegetation results in significant changes to numerous stream processes, which are described below.

Riparian harvest activities cause alterations to riparian vegetation (via removal and subsequent re-growth) which may cause direct and indirect changes to stream insolation, inputs of organic debris, nutrients, sediment input, bank stability, groundwater level, and hydrology. The impacts that result from changes to riparian vegetation may be mediated by the size and type of riparian buffer left in place (length and width, clearcut vs full buffer vs selective thinning), the placement of the buffers in a stream network, the residual basal area remaining in the riparian zone, the physical characteristics of the basin, and the time since harvest. Soil compaction in the riparian zone from heavy equipment can also cause changes to hydrology by decreasing soil permeability and promoting surface flows. The impacts from soil compaction are mediated by the extent of the harvest, the type of harvest, and the methods and equipment used (for example, skidders and feller bunchers vs cable logging).

Compared to riparian harvest activities, fewer in-stream processes respond to basin-wide harvest activities. Like riparian harvest activities, alterations to basin-wide vegetation may cause direct and indirect changes to hydrology, inputs of organic debris, and to a lesser extent stream insolation. An additional factor, synchronicity, may also change in response to basin-wide harvest activities. These impacts may be mediated by the percent of the basin harvested, the size of the canopy openings created, the distance from the stream, the type of harvest employed, the physical characteristics of the basin, and the time since harvest. Much like in the riparian zone, soil compaction in the basin from heavy equipment can also cause changes to hydrology, which are is mediated by the extent of the harvest, the type of harvest, and the methods and equipment used.

In summary, the stream processes most likely disrupted by forest harvest activities are stream insolation, organic debris flux, sedimentation, nutrient load, bank stability, groundwater level, hydrology, and synchronicity. These disruptions are primarily mediated by changes to forest vegetation, both within the riparian zone and across the whole basin. However, harvest activities may also act on several of these processes directly. The literature suggests that harvest activities influence macroinvertebrate community structure primarily by altering autochthonous vs allochthonous inputs, and through changes in habitat quality. Time since harvest is a critical piece of information for interpreting these changes—the evidence suggests there are strong relationships between time since harvest and canopy cover, summer flows, in-stream sediment, and in-stream debris/organic material. The post-harvest relationships between time and canopy cover as well as time and summer flows both appear to be polynomial functions, and many studies may have mis-interpreted recovery of these factors and the macroinvertebrate community by looking at too narrow a timespan. Initial return to baseline for canopy closure and hydrology may occur within 5-10 years (Hutchens et al 2003), though subsequent longer-term studies have shown this "recovery" to be a transient inflection point. Some evidence suggests true recovery of these two factors may take over a hundred years. Given the role of insolation and stream flow in organizing macroinvertebrate community structure, those findings have particular saliency for the question of why B-IBI scores may vary so widely in forested basins.

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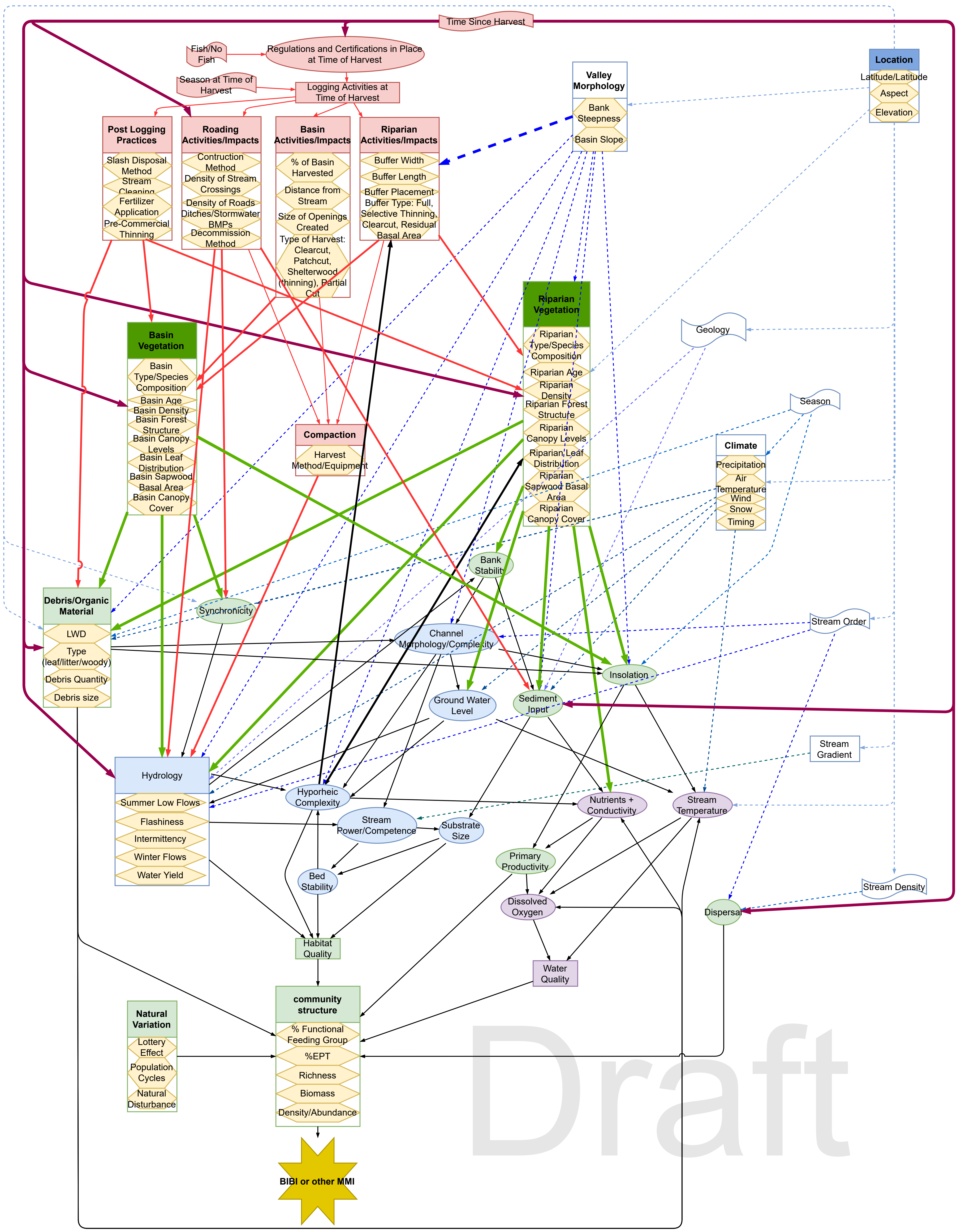
Appendix C – Literature Review Tables

(See excel spreadsheet “Appendix C-literaturereview.xls”)

Appendix D – Conceptual Model

(See "Appendix D Forestry Conceptual model_draft.pdf")

Draft Conceptual Model: Impacts from Forestry to BIBI



Appendix E - Hypothesized causal pathways

Stream Temperature

Stream temperatures may influence B-IBI score by affecting the benthic macroinvertebrate community. Higher temperatures favor tolerant taxa over sensitive taxa (Dewson et al. 2007, Johnson et al. 2022), potentially resulting in lower B-IBI scores through the Percent Tolerant Taxa and Intolerant Taxa Richness sub-metrics. Forest harvest activities likely have a causal relationship with stream temperature by controlling the availability of trees for shading (Kiffney et al. 2003). We hypothesize that this relationship exists over two different spatial scales: at the scale of the riparian corridor, and the immediate vicinity of the sampling reach.

The density of leaf area along the whole upstream riparian corridor likely influences stream temperature (Beschta et al. 1987). Tree basal area is correlated with the area of leaves per unit ground cover and has been used to predict light transmittance (Sonohat et al. 2004, Comeau et al. 2003, Comeau 1998, Thomas & Comeau 1998). Forest stand age also influences light transmittance through the canopy (Sonohat et al. 2004, Kaylor et al. 2017, Parker et al. 2002). Kaylor et al. (2017) found a non-linear effect of forest age on insolation, in which forests less than 20 years old had less canopy closure than pre-harvest, and forests between 20 and 150 years old had more canopy closure than pre-harvest. Not enough data was available to determine when canopies recovered to pre-harvest closures, but one data point showed recovery by 200 years of age.

In addition, direct insolation of the sampling reach may influence local stream temperatures (Zwieniecki and Newton, 1999). Similar to the whole riparian corridor, the amount of direct insolation at the sampling reach is in turn likely influenced by tree basal area and age in the areas adjacent to the sampling reach (Parker et al. 2002, Sonohat et al. 2004, Comeau et al. 2003, Kaylor et al. 2017).

Trophic Structure

For this study, we define trophic structure as the degree to which the benthic community depends on energy from in-stream photosynthesis versus detritus originating outside the stream. More specifically, we measured the relative abundance of taxa who feed by scraping algae from rocks or collector-gathering fine particulates versus taxa who feed by shredding coarse material. While collector-gatherers can be detritivores, they may also consume algal biofilms and have been observed to increase in conjunction with increased diatom biomass (Griffith & Kiffney 2022, Kiffney et al. 2003, Haggerty et al. 2004, Gurtz and Wallace 1984). The trophic structure of the benthic community may influence B-IBI scores through several sub-metrics. In particular, the Percent Dominant score is negatively affected when conditions shift to favor taxa that commonly dominate the invertebrate community, which are often collector-gatherers or scrapers (e.g., Baetidae, Chironomidae, Oligochaeta) (Johnson et al. 2022, Jackson et al. 2007, Kiffney et al. 2003). In addition, increased shredders may be more likely to contribute positively to the Intolerant Richness metric because they are more common in headwater streams with large amounts of woody debris, which are more representative of unaltered conditions (Haggerty et al. 2002).

Forest harvest activities likely have a causal relationship with the trophic structure of the benthic community by controlling the amount of photosynthesis occurring in the sampling reach via shading, and by controlling the quantity of detrital inputs from woody debris and leaf litter (Chizinski et al. 2010, Stone and Wallace 1998, Hutchens et al. 2003). Insolation to the sampling reach is likely controlled by the same forest conditions described in the previous section on Stream Temperature. Detrital inputs are controlled by two different avenues: slash from recent forest harvest, and ongoing debris inputs from riparian vegetation.

Slash from recent forest harvest can be characterized as intense, short-lived bursts of woody debris that may temporarily bury a stream (Haggerty et al. 2004, Jackson et al. 2007). The amount of riparian canopy mass harvested within the 5 years prior to sampling is likely to have a causal relationship with the quantity of debris in a stream that is usable to shredder macroinvertebrates. The selection of 5 years for the relevant time window is based on the rapid decrease in slash coverage observed in Jackson et al. (2007).
Ongoing debris inputs, on the other hand, are the baseline inputs from standing riparian vegetation that come in the form of seasonal leaf litter and windfall. The amount of canopy mass in place at the time of sampling, as well as type of forest in place (% conifer vs deciduous) are likely to have a causal relationship with the quantity and quality of debris available to macroinvertebrate shredders (Frady et al. 2007).

Base Flows

In streams of the Pacific Northwest, summer flows typically represent minimum flows due to minimal precipitation and declining runoff from snowmelt. The base flow of a stream can influence B-IBI score by controlling the type and amount of habitat available to the benthic community, and thus, the type of macroinvertebrates that may flourish. Extreme low flow events may decrease taxa richness and favor tolerant over sensitive taxa (Dewson et al. 2007), potentially resulting in lower B-IBI scores through multiple sub-metrics.
Forest harvest activities likely have a causal relationship with summer baseflows by controlling the evapotranspiration rate of vegetation within the basin. The age, sapwood basal area, species composition, and spatial position of forest stands in a stream basin, as well as the size of harvested patches, all have demonstrated effects on net evapotranspiration (Moore et al. 2004, Moore & Wondzell 2005, Perry and Jones 2017, Hall et al. 2018). We hypothesize that the relative impact on summer baseflow varies between riparian and upland forest stands, as different mechanisms take on different importance (Moore and Wondzell 2005). In addition, Perry & Jones (2017) found that the effect of age on summer flows changes over time and can be broadly categorized into a period of higher baseflows in the first 5 years after harvest, and lower baseflows starting around 15 years after harvest. By 45 years post-harvest, baseflows still had not recovered. Modelled results by Coble et al. (2020) had similar results and found recovery to pre-harvest flows occurred around 75 years post-harvest in clearcut basins. Perry and Jones (2017) found an additional effect in which larger harvest patch sizes further reduced summer flows.

Storm Flows

In this study, we consider storm flows to be those that occur during periods of high precipitation and snowmelt, which typically occur in the fall, winter, and spring in Pacific

Northwest streams. Whereas summer flows are more reflective of groundwater inputs, storm flows also include run-off from saturated soils. Previous efforts have found B-IBI scores or other community measures to have a negative relationship with the frequency and intensity of storm disturbances (DeGasperi et al. 2009, King County 2019, Marshalonis & Larson 2019, Danehy et al. 2021).

The same forest condition variables that influence base flows are likely to also influence storm flows through direct means (intercepting rainfall) and indirect means (reducing soil saturation through evapotranspiration) (Hutchens et al. 2003, Moore & Wondzell 2005). In addition to these variables, other factors related to forest harvest activities may also have a causal relationship with storm flows. The amount of unvegetated area in a stream basin can affect how much and how quickly runoff can flow unimpeded overland to a stream (Jones & Grant 1996, Thomas & Megahan 1998, Kuras et al. 2012, Guillemette et al. 2005, Grant et al. 2008). The width of vegetation between disturbed areas and streams can also act as a buffer by slowing down and/or absorbing runoff (Quinn et al. 2020). Previous works have found elevated peak flows persisting up to 25-30 years post-harvest (McEachran et al. 2021, Du et al. 2015, Grant et al. 2008, Thomas & Megahan 1998, Jones & Grant 1996, Jones 2000), which in stands of Douglas fir appears to be consistent with the 90% hydrologic recovery benchmark set by the Forest Practice Code of British Columbia (forest stand height of over 9 m) (DeMars & Herman 1987, Ministry of Forests 2001).

In addition, the compaction that results from road construction can reduce soil infiltration, creating additional run-off that may be directed to streams more quickly along roadways (Jones & Grant 1996, Moore & Wondzell 2005, Quinn et al. 2020, Grant et al. 2008), though some studies have found roads to have a mitigating effect on peak flows by directing run-off away from the basin (Kuras et al. 2012, Moore & Wondzell 2005). While the influence of roads to storm flows may be highly variable and situation dependent, we were interested to include roads as a potential contributor to increased peak flows.

Fine Sediments

The influx of fine sediments is likely to affect B-IBI scores by influencing several sub-metrics (Herlihy et al. 2005, Marshalonis & Larsen 2019, King County 2019, Walker et al. 2021, King County 2014c, Bailie & Neary 2015). In-stream sedimentation can degrade macroinvertebrate habitat by burying crevices used as refuge from scouring and predation, clogging respiratory organs, and reducing forage opportunity (King County 2019). Forest harvest activities have a causal relationship with in-stream sedimentation by disturbing soils, making them more susceptible to erosion (Swank et al. 2001, Quinn et al. 2020). In addition, where vegetation can slow down runoff to settle out sediment, removing vegetation can facilitate sediment delivery to streams (Quinn et al. 2020). Several studies have also shown forest roads to increase sediment delivery separate from harvest (Amaranthus et al. 1985, Sugden & Woods 2007, van Meerveld et al. 2014)
Swank et al. (2001) found that sediment levels in a stream were still highly elevated relative to baseline 15 years after logging occurred, suggesting that even brief sediment pulse events can have long-term impacts. The amount of excess sediment delivered from disturbed areas to nearby streams depends on how much of the basin has been disturbed, as well as the slope and soil composition of the harvested areas (Moore & Wondzell 2005,

Quinn et al. 2020, Bailie & Nearie 2015). We hypothesize that the relative impact of these variables on in-stream sediment varies between upland and riparian disturbances simply due to proximity. In addition, the time since harvest, the width of vegetation along the flow path between disturbed areas and the stream, and the distance along the stream between where run-off from disturbed soil enters the stream and the B-IBI sample site may also influence the impact of sediment there (Swank et al. 2001, Kiffney et al. 2003, Black et al. 2004). The density of roads in a basin and number of stream crossings are also likely to influence the amount of sediment entering a stream (Swank et al. 2001, King County 2019, Grant et al. 2008).

Chemical Applications

Chemical applications, such as the applications of fertilizer or pesticides, are likely to have a direct effect on B-IBI scores by creating conditions that are harmful to sensitive taxa (Scully-Engelmeyer et al. 2021). These chemicals are known to be used in commercial timber lands and are used at times on public lands. However, we were unable to locate a reliable data source for the location and timing of fertilizer or pesticide applications. We kept this node in our meta-model as we felt it was important, but it will not be included in the final analysis.

Appendix F – Data Definitions

Table F-1: Names (as listed in the geodatabase) and definitions of data attributes collected for the study. The column "Riparian Model" lists which SEM model a variable is used in (50'/100'/SPTH riparian spatial extents, all, or none). Rows in grey represent meta-data, rows in white represent data that does not depend on the riparian spatial extent, rows in blue, green and red represent data collected in the 50', 100' and SPTH riparian zones, respectively.

Variable	Concept	Definition	Units	Riparian Model
Agency		Sampling Agency		
comid		Associated COMID in NHDPlus V2		
Event_Date_1.x		Sampling Date		
G_ID		COMID or gage ID used to get flow data		
GroupAVisitID.x		Visit ID record in PSSB		
Site_Code		Site Code in PSSB		
Site_ID.x		Site ID number in PSSB		
Year		Year of Sample		
BasinArea	Basin Conditions	Area of contributing basin upstream of sampling point	hectares	All
ElevationStD	Basin Conditions	Standard Deviation of elevation of contributing basin upstream of sampling point	centimeters	All
MeanElevation	Basin Conditions	Average elevation of contributing basin upstream of sampling point	centimeters	All
Slope.y	Basin Conditions	Stream gradient	%	All
StreamOrde	Basin Conditions	Stream Order	unitless	All
Clinger_Richness_Score_1	B-IBI	Clinger Richness score	unitless	Not Used
Ephemeroptera_Richness_Score_1	B-IBI	Ephemeroptera Richness	unitless	Not Used
Intolerant_Richness_Score_1	B-IBI	Intolerant Richness score	unitless	Not Used
Long_Lived_Richness_Score_1	B-IBI	Long Lived Richness score	unitless	Not Used
Number_of_Organisms_1	B-IBI	Number of organisms in sample	#	Not Used
Overall_Score_1	B-IBI	BIBI overall score	unitless	All

Variable	Concept	Definition	Units	Riparian Model
Percent_Dominant_Score_1	B-IBI	Percent Dominant score	unitless	Not Used
Plecoptera_Richness_Score_1	B-IBI	Plecoptera Richness score	unitless	Not Used
Predator_Percent_Score_1	B-IBI	Predator Percent score	unitless	Not Used
Taxa_Richness_Score_1	B-IBI	Taxa Richness score	unitless	Not Used
Tolerant_Percent_Score_1	B-IBI	Tolerant Percent score	unitless	Not Used
Trichoptera_Richness_Score_1	B-IBI	Trichoptera Richness score	unitless	Not Used
HFI	Flow	High Flow Index (max rolling 60 day average daily discharge for a given water year divided by the long-term average value since 1979)	unitless, centered on 1	All
hpc.y	Flow	High Pulse Count	#	All
hpdur.y	Flow	High Pulse Duration		Not Used
hpr.y	Flow	High Pulse Range		Not Used
LFI.y	Flow	Low Flow Index. Minimum 60-day average flow for sampling year divided by the average value since 1979	unitless, centered on 1	All
MaxDschrg.y	Flow	Max discharge in preceding rainy season	meter^3/sec	Not Used
MaxPower.y	Flow	Max stream power observed in preceding rainy season	kg m2 s−3	All
maxtomeanQ	Flow	Max hourly discharge for a given water year divided by long-term average daily discharge since 1979)	unitless	All
meanbaseflow.y	Flow	Average 60-day rolling average low flow since 1979	meter^3/sec	Not Used
MedianDschrg.y	Flow	median discharge observed since 1979	meter^3/sec	Not Used
minrmean.y	Flow	60-day rolling average low flow in sampling year	meter^3/sec	Not Used
PFI	Flow	Peak Flow Index (Max hourly discharge for a given water year divided by long-term average value since 1979)	unitless, centered on 1	All

Variable	Concept	Definition	Units	Riparian Model
RBI.y	Flow	Richards-Baker Index	unitless	All
tqm.y	Flow	TQMean	unitless	All
TypicalPower.y	Flow	Median stream power observed since 1979	kg m2 s−3	All
med_patch_area	Forest Conditions	Median size of disturbed patches affecting evapotranspiration (forest conversion since 1985 prior to sampling or bare/developed)	hectares	All
TimeSinceDisturb_MEAN	Forest Conditions	Mean age of forest in the basin	years	All
TimeSinceDisturb_MEDIAN	Forest Conditions	Median age of forest in the basin	years	All
CF	Functional Feeding Groups	% Collector-Filterers in benthic macroinvertebrate sample	%	All
CG	Functional Feeding Groups	% Collector-Gatherers in benthic macroinvertebrate sample	%	All
MH	Functional Feeding Groups	% Macrophyte Herbivore in benthic macroinvertebrate sample	%	All
OM	Functional Feeding Groups	% Omnivore in benthic macroinvertebrate sample	%	All
PA	Functional Feeding Groups	% Parasite in benthic macroinvertebrate sample	%	All
PH	Functional Feeding Groups	% Piercer-Herbivore in benthic macroinvertebrate sample	%	All
PR	Functional Feeding Groups	% Predator in benthic macroinvertebrate sample	%	All
SC	Functional Feeding Groups	% Scraper in benthic macroinvertebrate sample	%	All
SH	Functional Feeding Groups	% Shredder in benthic macroinvertebrate sample	%	All

Variable	Concept	Definition	Units	Riparian Model
SHXYtoCGSC	Functional Feeding Groups	Based on functional feeding groups of taxa, %scrapers +grazers: % shredders + % xylophage in sample.	ratio	All
UNKN	Functional Feeding Groups	% Unknown functional feeding group in benthic macroinvertebrate sample	%	All
XY	Functional Feeding Groups	% Xylophage in benthic macroinvertebrate sample	%	All
Disturb_Distance_Along_Stream_MEAN	Location	Mean distance from where flows from **forest conversion areas (since 1985)** intercept the stream to the sampling point	meters	All
Disturb_Distance_Along_Stream_MEDIAN	Location	Median distance from where flows from **forest conversion areas (since 1985)** intercept the stream to the sampling point	meters	All
Disturb_Distance_To_Stream_MEAN	Location	Mean distance from **forest conversion areas since 1985** cells to point where flow path intercepts the stream	meters	Not Used
Disturb_Distance_To_Stream_MEDIAN	Location	Median distance from **forest conversion areas since 1985** cells to point where flow path intercepts the stream	meters	All
Sediment_Vegetation_Distance_Source_To_Stream_MEAN	Location	Mean flow distance across sinks (forest or shrub, water or wetland) between source (forest converstion areas since 1985) and stream	meters	All
Sediment_Vegetation_Distance_Source_To_Stream_MEDIAN	Location	Median flow distance across sinks (forest or shrub, water or wetland) between source (forest conversion areas since 1985) and stream	meters	All
DensityRoads_km_km2	Roads	Density of roads in the basin	km/km2	All
Road_Crossing_Per_Stream_Km	Roads	Density of road crossings in the basin	# of crossings/ stream length (km)	All

Variable	Concept	Definition	Units	Riparian Model
BSTI	Sediment	Biological Sediment Tolerance Index (% fines)	%	All
FSBI	Sediment	Fine Sediment Biotic Index (Relyea et al. 2012)	unitless	All
sed	Sediment	% sand/fine sediments at sampling site, predicted by taxa in sample (bio.infer)	%	All
STRMTEMP	Temperature	Stream temperature, predicted by taxa in sample (bio.infer)	Celsius	Not Used
value	Temperature	Stream temperature, predicted by NorWest	Celsius	All
Sand_Per_MEAN_CoreRMZ	Basin Conditions	Mean % sand content of soils in 50' riparian zone	%	CoreRMZ
Sand_Per_MEAN_CoreRMZ_Upland	Basin Conditions	Mean % sand content of soils in uplands outside of 50' riparian zone	%	CoreRMZ
Sand_Per_MEAN_Disturbed_CoreRMZ	Basin Conditions	Mean % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in 50' riparian zone	%	CoreRMZ
Sand_Per_MEAN_Disturbed_CoreRMZ_Upland	Basin Conditions	Mean % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in uplands outside of 50' riparian zone	%	CoreRMZ
Sand_Per_MEDIAN_CoreRMZ	Basin Conditions	Median % sand content of soils in 50' riparian zone	%	Not Used
Sand_Per_MEDIAN_CoreRMZ_Upland	Basin Conditions	Median % sand content of soils in uplands outside of 50' riparian zone	%	Not Used
Sand_Per_MEDIAN_Disturbed_CoreRMZ	Basin Conditions	Median % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in 50' riparian zone	%	Not Used

Variable	Concept	Definition	Units	Riparian Model
Sand_Per_MEDIAN_Disturbed_CoreRMZ_Upland	Basin Conditions	Median % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in uplands outside of 50' riparian zone	%	Not Used
Slope_Per_MEAN_CoreRMZ	Basin Conditions	Mean slope of areas in 50' riparian zone	%	CoreRMZ
Slope_Per_MEAN_CoreRMZ_Upland	Basin Conditions	Mean slope of upland areas outside of 50' riparian zone	%	CoreRMZ
Slope_Per_MEAN_Disturbed_CoreRMZ	Basin Conditions	Mean slope of disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in 50' riparian zone	%	CoreRMZ
Slope_Per_MEAN_Disturbed_CoreRMZ_Upland	Basin Conditions	Mean slope of upland disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) outside of 50' riparian zone	%	CoreRMZ
Slope_Per_MEDIAN_CoreRMZ	Basin Conditions	Median slope of areas in 50' riparian zone	%	Not Used
Slope_Per_MEDIAN_CoreRMZ_Upland	Basin Conditions	Median slope of upland areas outside of 50' riparian zone	%	Not Used
Slope_Per_MEDIAN_Disturbed_CoreRMZ	Basin Conditions	Median slope of disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in 50' riparian zone	%	Not Used
Slope_Per_MEDIAN_Disturbed_CoreRMZ_Upland	Basin Conditions	Median slope of upland disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) outside of 50' riparian zone	%	Not Used
BasalArea_AgeBinned_CoreRMZ_F_200	Forest Conditions	Basal Area of all live trees aged 100-200 years in 50' riparian zone at time of sampling, normalized to basin area (shading)	m^2/ha *100	CoreRMZ

Variable	Concept	Definition	Units	Riparian Model
BasalArea_AgeBinned_CoreRMZ_F0_20	Forest Conditions	Basal Area of all live trees aged 0-20 years in 50' riparian zone at time of sampling, normalized to basin area (shading)	m^2/ha *100	CoreRMZ
BasalArea_AgeBinned_CoreRMZ_F20_200	Forest Conditions	Basal Area of all live trees aged 20-100 years in 50' riparian zone at time of sampling, normalized to basin area (shading)	m^2/ha *100	CoreRMZ
BasalArea_AgeBinned_CoreRMZ_NoData	Forest Conditions	Basal Area of all live trees aged >200 years in 50' riparian zone at time of sampling, normalized to basin area (shading)	m^2/ha *100	CoreRMZ
Biomass_CoreRMZ	Forest Conditions	Biomass of all live trees in 50' riparian zone at time of sampling, normalized to basin area	kg/ha	CoreRMZ
Biomass_CoreRMZ_Upland	Forest Conditions	Biomass of all live trees upland of 50' riparian zone at time of sampling, normalized to basin area	kg/ha	CoreRMZ
Cancov_CoreRMZ_MEAN	Forest Conditions	Mean canopy cover in a 50' riparian zone at time of sampling	% * 100	CoreRMZ
ConPerTotBiom_Preharv_CoreRMZ	Forest Conditions	Biomass of all conifer trees harvested in 50' riparian zone within 5 years prior to sampling, as percent of pre-harvest total biomass of all live trees	%	CoreRMZ
ConPerTotBiom_SamplingYear_CoreRMZ	Forest Conditions	Biomass of all conifer trees in 50' riparian zone at time of sampling, as percent of total biomass of all live trees	%	CoreRMZ
ConPerTotBiom_SamplingYear_CoreRMZ_Upland	Forest Conditions	Biomass of all conifer trees in uplands outside of 50' riparian zone at time of sampling, as percent of total biomass of all live trees	%	CoreRMZ
CoreRMZ_LZ_Cancov_MEAN	Forest Conditions	Mean canopy cover in a 50' riparian zone around a 100 m reach upstream of sampling point at time of sampling	% * 100	CoreRMZ

Variable	Concept	Definition	Units	Riparian Model
CoreRMZ_LZ_Standheight_MEDIAN	Forest Conditions	Mean stand height in a 50' riparian zone around a 100 m reach upstream of sampling point at time of sampling	meters * 100	CoreRMZ
CoreRMZ_SamplingYear_ConPerTotBiom_LZ_	Forest Conditions	Biomass of all conifer trees in 50' riparian zone around a 100 m reach upstream of sampling point at time of sampling, as percent of total biomass of all live trees	%	CoreRMZ
Disturbed_Percent_Sed_CoreRMZ	Forest Conditions	Area of disturbance affecting sediments (forest conversion since 1985 prior to sampling or bare/developed) in a 50' riparian buffer (equivalent to WA DNR Core Riparian Management Zone), as percent of total basin area	%	CoreRMZ
Disturbed_Percent_Sed_CoreRMZ_Upland	Forest Conditions	Area of disturbance affecting sediments (forest conversion since 1985 prior to sampling or bare/developed) in uplands outside of a 50' riparian buffer (equivalent to WA DNR Core Riparian Management Zone), as percent of total basin area	%	CoreRMZ
hydro_BasalArea_AgeBinned75_CoreRMZ_F_75	Forest Conditions	Basal Area of all live trees aged >75 years in 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ
hydro_BasalArea_AgeBinned75_CoreRMZ_F0_5	Forest Conditions	Basal Area of all live trees aged 0-5 years in 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ
hydro_BasalArea_AgeBinned75_CoreRMZ_F15_75	Forest Conditions	Basal Area of all live trees aged 15-75 years in 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ
hydro_BasalArea_AgeBinned75_CoreRMZ_F5_15	Forest Conditions	Basal Area of all live trees aged 5-15 years in 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ

Variable	Concept	Definition	Units	Riparian Model
hydro_BasalArea_AgeBinned75_CoreRMZ_NoData	Forest Conditions	Basal Area of all live trees of unknown age in 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ
hydro_BasalArea_AgeBinned75_Upland_CoreRMZ_F_75	Forest Conditions	Basal Area of all live trees aged >75 years upland of 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ
hydro_BasalArea_AgeBinned75_Upland_CoreRMZ_F0_5	Forest Conditions	Basal Area of all live trees aged 0-5 years upland of 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ
hydro_BasalArea_AgeBinned75_Upland_CoreRMZ_F15_75	Forest Conditions	Basal Area of all live trees aged 15-75 years upland of 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ
hydro_BasalArea_AgeBinned75_Upland_CoreRMZ_F5_15	Forest Conditions	Basal Area of all live trees aged 5-15 years upland of 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ
hydro_BasalArea_AgeBinned75_Upland_CoreRMZ_NoData	Forest Conditions	Basal Area of all live trees of unknown age upland of 50' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	CoreRMZ
KgPerHectare_RecentHarvestBiomass_CoreRMZ	Forest Conditions	Biomass of all live trees harvested in 50' riparian zone within 5 years prior to sampling, normalized to basin area	kg/ha	CoreRMZ
KgPerHectare_TotalBiomass_SamplingYear_CoreRMZ	Forest Conditions	Biomass of all live trees 50' in riparian zone at time of sampling, normalized to basin area	kg/ha	CoreRMZ
LZ_BasalArea_AgeBinned_CoreRMZ_F_200	Forest Conditions	Basal Area of all live trees aged 100-200 years in 50' riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	CoreRMZ

Variable	Concept	Definition	Units	Riparian Model
LZ_BasalArea_AgeBinned_CoreRMZ_F0_20	Forest Conditions	Basal Area of all live trees aged 0-20 years in 50' riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	CoreRMZ
LZ_BasalArea_AgeBinned_CoreRMZ_F20_200	Forest Conditions	Basal Area of all live trees aged 20-100 years in 50' riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	CoreRMZ
LZ_BasalArea_AgeBinned_CoreRMZ_NoData	Forest Conditions	Basal Area of all live trees aged >200 years in 50' riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	CoreRMZ
Stndhgt_CoreRMZ_MEDIAN	Forest Conditions	Mean stand height in a 50' riparian zone at time of sampling	meters * 100	CoreRMZ
Trees_per_Hectare_CoreRMZ	Forest Conditions	Density of trees in a 50' riparian zone at time of sampling	#/hectare	CoreRMZ
Sand_Per_MEAN_Buff100	Basin Conditions	Mean % sand content of soils in 100' riparian zone	%	Buff100
Sand_Per_MEAN_Buff100_Upland	Basin Conditions	Mean % sand content of soils in uplands outside of 100' riparian zone	%	Buff100
Sand_Per_MEAN_Disturbed_Buff100	Basin Conditions	Mean % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in 100' riparian zone	%	Buff100
Sand_Per_MEAN_Disturbed_Buff100_Upland	Basin Conditions	Mean % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in uplands outside of 100' riparian zone	%	Buff100
Sand_Per_MEDIAN_Buff100	Basin Conditions	Median % sand content of soils in 100' riparian zone	%	Not Used

Variable	Concept	Definition	Units	Riparian Model
Sand_Per_MEDIAN_Buff100_Upland	Basin Conditions	Median % sand content of soils in uplands outside of 100' riparian zone	%	Not Used
Sand_Per_MEDIAN_Disturbed_Buff100	Basin Conditions	Median % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in 100' riparian zone	%	Not Used
Sand_Per_MEDIAN_Disturbed_Buff100_Upland	Basin Conditions	Median % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in uplands outside of 100' riparian zone	%	Not Used
Slope_Per_MEAN_Buff100	Basin Conditions	Mean slope of areas in 100' riparian zone	%	Buff100
Slope_Per_MEAN_Buff100_Upland	Basin Conditions	Mean slope of upland areas outside of 100' riparian zone	%	Buff100
Slope_Per_MEAN_Disturbed_Buff100	Basin Conditions	Mean slope of disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in 100' riparian zone	%	Buff100
Slope_Per_MEAN_Disturbed_Buff100_Upland	Basin Conditions	Mean slope of upland disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) outside of 100' riparian zone	%	Buff100
Slope_Per_MEDIAN_Buff100	Basin Conditions	Median slope of areas in 100' riparian zone	%	Not Used
Slope_Per_MEDIAN_Buff100_Upland	Basin Conditions	Median slope of upland areas outside of 100' riparian zone	%	Not Used
Slope_Per_MEDIAN_Disturbed_Buff100	Basin Conditions	Median slope of disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in 100' riparian zone	%	Not Used

Variable	Concept	Definition	Units	Riparian Model
Slope_Per_MEDIAN_Disturbed_Buff100_Upland	Basin Conditions	Median slope of upland disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) outside of 100' riparian zone	%	Not Used
BasalArea_AgeBinned_Buff100_F_200	Forest Conditions	Basal Area of all live trees aged 100-200 years in 100' riparian zone at time of sampling, normalized to basin area (shading)	m2/ha *100	Buff100
BasalArea_AgeBinned_Buff100_F0_20	Forest Conditions	Basal Area of all live trees aged 0-20 years in 100' riparian zone at time of sampling, normalized to basin area (shading)	m2/ha *100	Buff100
BasalArea_AgeBinned_Buff100_F20_200	Forest Conditions	Basal Area of all live trees aged 20-100 years in 100' riparian zone at time of sampling, normalized to basin area (shading)	m2/ha *100	Buff100
BasalArea_AgeBinned_Buff100_NoData	Forest Conditions	Basal Area of all live trees aged >200 years in 100' riparian zone at time of sampling, normalized to basin area (shading)	m2/ha *100	Buff100
Biomass_Buff100	Forest Conditions	Biomass of all live trees in 100' riparian zone at time of sampling, normalized to basin area	kg/ha	Buff100
Biomass_Buff100_Upland	Forest Conditions	Biomass of all live trees upland of 100' riparian zone at time of sampling, normalized to basin area	kg/ha	Buff100
Buff100_LZ_Cancov_MEAN	Forest Conditions	Mean canopy cover in a 100' riparian zone around a 100 m reach upstream of sampling point at time of sampling	% * 100	Buff100
Buff100_LZ_Standheight_MEDIAN	Forest Conditions	Mean stand height in a 100' riparian zone around a 100 m reach upstream of sampling point at time of sampling	meters * 100	Buff100

Variable	Concept	Definition	Units	Riparian Model
Buff100_SamplingYear_ConPerTotBiom_LZ_MEAN	Forest Conditions	Biomass of all conifer trees in 100' riparian zone around a 100 m reach upstream of sampling point at time of sampling, as percent of total biomass of all live trees	%	Buff100
Cancov_Buff100_MEAN	Forest Conditions	Mean canopy cover in a 100' riparian zone at time of sampling	% * 100	Buff100
ConPerTotBiom_Preharv_Buff100	Forest Conditions	Biomass of all conifer trees harvested in 100' riparian zone within 5 years prior to sampling, as percent of pre-harvest total biomass of all live trees	%	Buff100
ConPerTotBiom_SamplingYear_Buff100	Forest Conditions	Biomass of all conifer trees in 100' riparian zone at time of sampling, as percent of total biomass of all live trees	%	Buff100
ConPerTotBiom_SamplingYear_Buff100_Upland	Forest Conditions	Biomass of all conifer trees in uplands outside of 100' riparian zone at time of sampling, as percent of total biomass of all live trees	%	Buff100
Disturbed_Percent_Sed_Buff100	Forest Conditions	Area of disturbance affecting sediments (forest conversion since 1985 prior to sampling or bare/developed) in a 100' riparian buffer, as percent of total basin area	%	Buff100
Disturbed_Percent_Sed_Buff100_Upland	Forest Conditions	Area of disturbance affecting sediments (forest conversion since 1985 prior to sampling or bare/developed) in uplands outside of a 100' riparian buffer, as percent of total basin area	%	Buff100
hydro_BasalArea_AgeBinned75_Buff100_F_75	Forest Conditions	Basal Area of all live trees aged >75 years in 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100

Variable	Concept	Definition	Units	Riparian Model
hydro_BasalArea_AgeBinned75_Buff100_F0_5	Forest Conditions	Basal Area of all live trees aged 0-5 years in 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100
hydro_BasalArea_AgeBinned75_Buff100_F15_75	Forest Conditions	Basal Area of all live trees aged 15-75 years in 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100
hydro_BasalArea_AgeBinned75_Buff100_F5_15	Forest Conditions	Basal Area of all live trees aged 5-15 years in 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100
hydro_BasalArea_AgeBinned75_Buff100_NoData	Forest Conditions	Basal Area of all live trees of unknown age in 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100
hydro_BasalArea_AgeBinned75_Upland_Buff100_F_75	Forest Conditions	Basal Area of all live trees aged >75 years upland of 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100
hydro_BasalArea_AgeBinned75_Upland_Buff100_F0_5	Forest Conditions	Basal Area of all live trees aged 0-5 years upland of 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100
hydro_BasalArea_AgeBinned75_Upland_Buff100_F15_75	Forest Conditions	Basal Area of all live trees aged 15-75 years upland of 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100
hydro_BasalArea_AgeBinned75_Upland_Buff100_F5_15	Forest Conditions	Basal Area of all live trees aged 5-15 years upland of 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100
hydro_BasalArea_AgeBinned75_Upland_Buff100_NoData	Forest Conditions	Basal Area of all live trees of unknown age upland of 100' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	Buff100

Variable	Concept	Definition	Units	Riparian Model
KgPerHectare_RecentHarvestBiomass_Buff100	Forest Conditions	Biomass of all live trees harvested in 100' riparian zone within 5 years prior to sampling, normalized to basin area	kg/ha	Buff100
KgPerHectare_TotalBiomass_SamplingYear_Buff100	Forest Conditions	Biomass of all live trees 100' in riparian zone at time of sampling, normalized to basin area	kg/ha	Buff100
LZ_BasalArea_AgeBinned_Buff100_F_200	Forest Conditions	Basal Area of all live trees aged 100-200 years in 100' riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	Buff100
LZ_BasalArea_AgeBinned_Buff100_F0_20	Forest Conditions	Basal Area of all live trees aged 0-20 years in 100' riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	Buff100
LZ_BasalArea_AgeBinned_Buff100_F20_200	Forest Conditions	Basal Area of all live trees aged 20-100 years in 100' riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	Buff100
LZ_BasalArea_AgeBinned_Buff100_NoData	Forest Conditions	Basal Area of all live trees aged >200 years in 100' riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	Buff100
Stndhgt_Buff100_MEDIAN	Forest Conditions	Mean stand height in a 100' riparian zone at time of sampling	meters * 100	Buff100
Trees_per_Hectare_Buff100	Forest Conditions	Density of trees in a 100' riparian zone at time of sampling	#/hectare	Buff100
Sand_Per_MEAN_Disturbed_SPTH	Basin Conditions	Mean % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in variable riparian zone	%	SPTH

Variable	Concept	Definition	Units	Riparian Model
Sand_Per_MEAN_Disturbed_SPTH_Upland	Basin Conditions	Mean % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in uplands outside of variable riparian zone	%	SPTH
Sand_Per_MEAN_SPTH	Basin Conditions	Mean % sand content of soils in variable riparian zone	%	SPTH
Sand_Per_MEAN_SPTH_Upland	Basin Conditions	Mean % sand content of soils in uplands outside of variable riparian zone	%	SPTH
Sand_Per_MEDIAN_Disturbed_SPTH	Basin Conditions	Median % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in variable riparian zone	%	Not Used
Sand_Per_MEDIAN_Disturbed_SPTH_Upland	Basin Conditions	Median % sand content of disturbed soils affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in uplands outside of variable riparian zone	%	Not Used
Sand_Per_MEDIAN_SPTH	Basin Conditions	Median % sand content of soils in variable riparian zone	%	Not Used
Sand_Per_MEDIAN_SPTH_Upland	Basin Conditions	Median % sand content of soils in uplands outside of variable riparian zone	%	Not Used
Slope_Per_MEAN_Disturbed_SPTH	Basin Conditions	Mean slope of disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in variable riparian zone	%	SPTH
Slope_Per_MEAN_Disturbed_SPTH_Upland	Basin Conditions	Mean slope of upland disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) outside of variable riparian zone	%	SPTH
Slope_Per_MEAN_SPTH	Basin Conditions	Mean slope of areas in variable riparian zone	%	SPTH

Variable	Concept	Definition	Units	Riparian Model
Slope_Per_MEAN_SPTH_Upland	Basin Conditions	Mean slope of upland areas outside of variable riparian zone	%	SPTH
Slope_Per_MEDIAN_Disturbed_SPTH	Basin Conditions	Median slope of disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) in variable riparian zone	%	Not Used
Slope_Per_MEDIAN_Disturbed_SPTH_Upland	Basin Conditions	Median slope of upland disturbed areas affecting sediment (forest conversion since 1985 prior to sampling or bare/developed) outside of variable riparian zone	%	Not Used
Slope_Per_MEDIAN_SPTH	Basin Conditions	Median slope of areas in variable riparian zone	%	Not Used
Slope_Per_MEDIAN_SPTH_Upland	Basin Conditions	Median slope of upland areas outside of variable riparian zone	%	Not Used
BasalArea_AgeBinned_SPTH_F_200	Forest Conditions	Basal Area of all live trees aged >200 years in variable riparian zone at time of sampling, normalized to basin area (shading)	m2/ha *100	SPTH
BasalArea_AgeBinned_SPTH_F0_20	Forest Conditions	Basal Area of all live trees aged 0-20 years in variable riparian zone at time of sampling, normalized to basin area (shading)	m2/ha *100	SPTH
BasalArea_AgeBinned_SPTH_F20_200	Forest Conditions	Basal Area of all live trees aged 20-200 years in variable riparian zone at time of sampling, normalized to basin area (shading)	m2/ha *100	SPTH
BasalArea_AgeBinned_SPTH_NoData	Forest Conditions	Basal Area of all live trees of unknown age in variable riparian zone at time of sampling, normalized to basin area (shading)	m2/ha *100	SPTH
Biomass_SPTH	Forest Conditions	Biomass of all live trees in variable riparian zone at time of sampling, normalized to basin area	kg/ha	SPTH

Variable	Concept	Definition	Units	Riparian Model
Biomass_SPTH_Upland	Forest Conditions	Biomass of all live trees upland of variable riparian zone at time of sampling, normalized to basin area	kg/ha	SPTH
Cancov_SPTH_MEAN	Forest Conditions	Mean canopy cover in a variable riparian zone at time of sampling	% * 100	SPTH
ConPerTotBiom_Preharv_SPTH	Forest Conditions	Biomass of all conifer trees harvested in variable riparian zone within 5 years prior to sampling, as percent of pre-harvest total biomass of all live trees	%	SPTH
ConPerTotBiom_SamplingYear_SPTH	Forest Conditions	Biomass of all conifer trees in variable riparian zone at time of sampling, as percent of total biomass of all live trees	%	SPTH
ConPerTotBiom_SamplingYear_SPTH_Upland	Forest Conditions	Biomass of all conifer trees in uplands outside of variable riparian zone at time of sampling, as percent of total biomass of all live trees	%	SPTH
Disturbed_Percent_Sed_SPTH	Forest Conditions	Area of disturbance affecting sediments (forest conversion since 1985 prior to sampling or bare/developed) in a variable riparian buffer (equivalent to WA WDFW Site Potential Tree Height), as percent of total basin area	%	SPTH
Disturbed_Percent_Sed_SPTH_Upland	Forest Conditions	Area of disturbance affecting sediments (forest conversion since 1985 prior to sampling or bare/developed) in uplands outside of a variable riparian buffer (equivalent to WA WDFW Site Potential Tree Height), as percent of total basin area	%	SPTH
hydro_BasalArea_AgeBinned75_SPTH_F_75	Forest Conditions	Basal Area of all live trees aged >75 years in variable riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH

Variable	Concept	Definition	Units	Riparian Model
hydro_BasalArea_AgeBinned75_SPTH_F0_5	Forest Conditions	Basal Area of all live trees aged 0-5 years in variable riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH
hydro_BasalArea_AgeBinned75_SPTH_F15_75	Forest Conditions	Basal Area of all live trees aged 15-75 years in variable riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH
hydro_BasalArea_AgeBinned75_SPTH_F5_15	Forest Conditions	Basal Area of all live trees aged 5-15 years in variable' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH
hydro_BasalArea_AgeBinned75_SPTH_NoData	Forest Conditions	Basal Area of all live trees of unknown age in variable riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH
hydro_BasalArea_AgeBinned75_Upland_SPTH_F_75	Forest Conditions	Basal Area of all live trees aged >75 years upland of variable riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH
hydro_BasalArea_AgeBinned75_Upland_SPTH_F0_5	Forest Conditions	Basal Area of all live trees aged 0-5 years upland of variable riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH
hydro_BasalArea_AgeBinned75_Upland_SPTH_F15_75	Forest Conditions	Basal Area of all live trees aged 15-75 years upland of variable riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH
hydro_BasalArea_AgeBinned75_Upland_SPTH_F5_15	Forest Conditions	Basal Area of all live trees aged 5-15 years upland of variable' riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH
hydro_BasalArea_AgeBinned75_Upland_SPTH_NoData	Forest Conditions	Basal Area of all live trees of unknown age upland of variable riparian zone at time of sampling, normalized to basin area (hydrology)	m2/ha *100	SPTH

Variable	Concept	Definition	Units	Riparian Model
KgPerHectare_RecentHarvestBiomass_SPTH	Forest Conditions	Biomass of all live trees harvested in variable riparian zone within 5 years prior to sampling, normalized to basin area	kg/ha	SPTH
KgPerHectare_TotalBiomass_SamplingYear_SPTH	Forest Conditions	Biomass of all live trees variable in riparian zone at time of sampling, normalized to basin area	kg/ha	SPTH
LZ_BasalArea_AgeBinned_SPTH_F_200	Forest Conditions	Basal Area of all live trees aged >200 years in variable riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	SPTH
LZ_BasalArea_AgeBinned_SPTH_F0_20	Forest Conditions	Basal Area of all live trees aged 0-20 years in variable riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	SPTH
LZ_BasalArea_AgeBinned_SPTH_F20_200	Forest Conditions	Basal Area of all live trees aged 20-200 years in variable riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	SPTH
LZ_BasalArea_AgeBinned_SPTH_NoData	Forest Conditions	Basal Area of all live trees of unknown age in variable riparian zone around a 100 m reach upstream of sampling point at time of sampling, normalized to basin area (shading)	m2/ha *100	SPTH
SPTH_LZ_Cancov_MEAN	Forest Conditions	Mean canopy cover in a variable riparian zone around a 100 m reach upstream of sampling point at time of sampling	% * 100	SPTH
SPTH_LZ_Standheight_MEDIAN	Forest Conditions	Mean stand height in a variable riparian zone around a 100 m reach upstream of sampling point at time of sampling	meters * 100	SPTH

Variable	Concept	Definition	Units	Riparian Model
SPTH_SamplingYear_ConPerTotBiom_LZ_	Forest Conditions	Biomass of all conifer trees in variable riparian zone around a 100 m reach upstream of sampling point at time of sampling, as percent of total biomass of all live trees	%	SPTH
Stndhgt_SPTH_MEDIAN	Forest Conditions	Mean stand height in a variable riparian zone at time of sampling	meters * 100	SPTH
Trees_per_Hectare_SPTH	Forest Conditions	Density of trees in a variable riparian zone at time of sampling	#/hectare	SPTH

Appendix G – Data processing

Site Selection

Our goal was to identify at least 50 basins within the Puget Sound basin which have associated B-IBI samples and where forested conditions are the primary land cover. We wished to obtain two separate data sets: a primary one to be used for initial model evaluation, and a secondary one to be used for confirmation if our initial model required modification. We began by querying the Puget Sound Stream Benthos database of all sites sampled in Western Washington. The Puget Sound Stream Benthos database is a regional data repository used by over 30 state, local and tribal jurisdictions and non-profits to house benthic macroinvertebrate data and calculate B-IBI scores as part of monitoring projects and programs. Next, we imported the site coordinates to ArcGIS and snapped the points to coincide with flowlines from the NHDPlus HR dataset. We filtered out any points located outside of the Puget Sound Basin and delineated the upstream contributing basin of the remaining points (~760 delineations). These basins went through a screening process to include only basins that were >70% vegetated and <10% developed[8] according to 2019 NLCD Land Cover data. These criteria were based on a 2019 King County report which found that most basins with excellent B-IBI scores can be characterized as having at least 72% of their basin forested, and less than 4% of their basin developed. For this study, we set our forest threshold to include lands which may have been recently logged but will become re-forested (forest and shrub cover). In addition, we set more inclusive criteria for the development threshold as a compromise to obtain an adequate sample size (minimum of 5 sites per parameter; Kline 2015), and because clearcuts can occasionally be mis-categorized as development.

To reduce spatial autocorrelation[9] we further screened for nestedness[10] by selecting qualifying basins that did not contain other qualifying basins. Then, we evaluated the basin delineations for accuracy by visually examining each one alongside topographic data, aerial photos, the National Hydrography Dataset (NHD) and other stream data (if available), and by contacting the agencies responsible for each sample site to verify the location of the respective sample points. Further visual screenings eliminated basins that included mines, or had major roads, lakes, marine shoreline, agriculture, or residential development immediately upstream of the sampling point which may have influence on the sample. If a basin was removed from consideration, we tried to replace it with another basin in the same watershed that may have previously been screened out for nestedness and evaluated the newly selected basin. If a sample point was mapped in the wrong place or a basin was otherwise delineated incorrectly, we re-delineated the point and re-evaluated the site. If a choice between two equally suitable but nested basins was needed, we opted for the basin

[8] One basin had slightly higher development than our threshold criteria (E2052, 10.72%), but after closer inspection we believe this value to be slightly higher than actual conditions.

[9] Spatial autocorrelation is the tendency of observations to have similar values based on the physical closeness of the measurement locations.

[10] Nestedness refers to whether a basin considered for inclusion is contained within another basin considered for inclusion.

with a longer B-IBI sampling history. Through this iterative process, we ended up with 142 candidate basins. We applied a final screen based on the temporal availability for some of the explanatory data, to obtain a final count of 132 basins in the primary data set.

Land Change Monitoring, Assessment and Projection Dataset

Using Land Change Monitoring, Assessment and Projection (LCMAP) V1.2 data (USGS 2021), we were able to determine the location and timing of forest conversion relative to the sample location and date in each basin, as well as land cover present at the time of sampling. One drawback to the LCMAP data is that it can only easily detect clearcuts that are larger than 30 meters-squared (the spatial resolution of the dataset); as a result, we cannot examine the effect of less intensive harvest methods such as thinning. We used the LCMAP annual raster datasets Time Since Last Change (SCLAST) and Annual Land Cover Change (LCACHG) to determine the timing and type of all land cover changes detected in the basins between 1986 and 2020 (USGS 2021). First, for each year in the LCACHG raster dataset we extracted only values indicating land cover changes, converted these to polygons, and added the year of the source raster as an attribute. Then we merged the layers into a single dataset and clipped it to the basin shapes. The SCLAST raster was similarly processed, resulting in a single polygon feature class with all SCLAST values across all years, with the year of the source raster as an attribute and clipped to the basins. We then merged the SCLAST and LCACHG data using a spatial union for each year. The resulting product was a layer in which each polygon represents a detected change, with attributes for the type of change, year of detection, and days prior to July 1st of the detection year that the change occurred.

Using this layer of combined SCLAST and LCACHG data, we were able to calculate the date of land cover change for most polygons. If there was land cover change information but no information for the days since the change occurred, the change was assumed to have occurred in the year of detection prior to sampling. This is a reasonable assumption because the LCMAP Land Cover data (and subsequent Annual Land Cover Change values) are based on land cover as of July 1st of the data year, which is before the typical B-IBI sampling season. We excluded land cover changes that occurred after the sampling date in each basin; land cover changes for which there was no information about the type of change; and land cover changes that were not forest conversion to other land covers. We calculated the Year of Disturbance and Years Since Disturbance relative to the sampling date for each polygon. Based on these two attributes we converted the polygons back into rasters, prioritizing the most recent disturbance if more than one occurred in the same location.

We used the LCMAP Primary Land Cover (LCPRI) dataset to clip the most likely land cover present at the time of sampling for each basin. We then merged the resulting clipped rasters into a single dataset of land use at time of sampling. Based on the land cover at time of sampling and year of disturbance rasters, we classified the raster data into disturbed (forest conversion since 1985 prior to sampling or bare/developed) and undisturbed areas (remaining basin area).

Forest Structure Maps

Using the Landscape Ecology, Modeling, Mapping and Analysis (LEMMA) forest structure data (Ohmann & Gregory 2001), we were able to collect information on forest condition in disturbed areas pre- and post- conversion, as well as forest conditions and land use at the time of sampling. With permission from the LEMMA team at Oregon State University, we obtained modelled data of Total Basal Area, Total Biomass, Conifer Biomass, Forest Canopy, Conifer Canopy, Stand Height and Stand Age (excluding Remnant Trees), covering all the Puget Sound Region from 1986 to 2017 (Ohmann & Gregory 2002). The modelled forest condition data used in this study had validation R^2 values of 0.5 or more.

Pre- and post- harvest data

Using the Year of Disturbance data generated from the LCMAP dataset, we clipped total biomass data one year before and after each detected disturbance. We subtracted post-harvest biomass from pre-harvest biomass to obtain a value for biomass removed. On some occasions, the values for biomass removed was less than 0, indicating that biomass increased for these cells. Careful examination of satellite imagery showed these instances to be cases where LCMAP mistakenly detected a change where none occurred. In general, we saw very good alignment between the LCMAP-derived disturbance data, LEMMA-derived forestry data, and satellite imagery, and cases of poor alignment were relatively few. As a result, we chose to set negative values to zero in our calculations of removed biomass. The original reporting units for removed biomass were in kilograms/hectare, but we subsequently converted the values to kilograms/cell (900 square meters) for ease of summing across areas.

In addition, we clipped conifer biomass data one year before each detected disturbance and used this to calculate pre-harvest conifer biomass as a percent of pre-harvest total biomass for each cell. This pre-harvest percent conifer biomass is a reasonable approximation of the relative composition of the biomass removed.

Data for Sampling Year

For each basin, we clipped Total Biomass, Total Basal Area and Conifer Biomass rasters from the appropriate year based on the B-IBI sampling date. We used these two datasets to calculate the basal area per cell, the percent conifer biomass per cell, as well as the kilograms per cell total biomass at the time of sampling.

In addition, we clipped Stand Age (excluding Remnant Trees) to the appropriate year for each basin based on B-IBI sampling date, for the purposes of summarizing basins by stand age and binning basal area by age classes. After reviewing the age data and discussing with LEMMA team members some limitations in how the values are calculated in harvested areas, we decided we would have better accuracy by using the Years Since Disturbance dataset in areas harvested since 1985 and filling in the remaining areas with the age data obtained from LEMMA. We then binned the ages to represent different age effects on light penetration and hydrology. For light penetration, we used 0-20, 20-200, and >200 as effect bins (Kaylor et al. 2017). For hydrology, we used 0-5, 5-15, 15-75 and >75 as effect bins (Perry and Jones 2017, Swank et al. 2001, Hall et al. 2018, Coble et al. 2020).

National Hydrography Dataset

We used the NHDPLUS High Resolution (USGS 2017) dataset to delineate the basins, obtain stream reach gradient values, and calculate flow lengths and distances in each basin. This version of the NHD dataset has a resolution of 1:24,000, and is the most detailed hydrological data available at the regional level. Using the digital elevation model and flow direction rasters provided by the NHD, we calculated the flow distance to the stream for all cells in each basin. We then calculated the flow distance to the sampling point for all cells in each basin. By subtracting the flow distance to the stream from the total flow distance to the sampling point, we calculated the distance from the sampling point where over-land flows enter the stream. We then used the Years Since Disturbance (generated by LCMAP data) data to extract the distance to stream and distance along the stream for all disturbances detected since 1985.

In addition, we calculated the width of flow "sinks" between disturbed areas and streams using the flow length tool. For this process we generated a weight raster where "sinks" (i.e., vegetation) were assigned a value of 1, disturbed areas (generated by LCMAP data, discussed in previous section) were assigned a value of 0, and the stream flow path was assigned to 0. In the resulting product, only vegetated or wetland areas (except the stream itself) contribute to sink width.

Gridded National Soil Survey Geographic Database

We obtained rasters of weighted average percent sand and slope values for each basin using the gridded national soil survey geographic database (gNATSGO) dataset (Soil Survey Staff 2021) with the Soil Data Development Toolbox available from the United Sates Department of Agriculture - Natural Resources Conservation Service.

NorWeST Stream Temperature

We downloaded predicted annual Mean Weekly Maximum Temperature spatial data from the NorWeST dataset generated by the United States Department of Agriculture - U.S. Forest Service (Isaak et al. 2016). We intersected our sample points with the layer of available predictions, and selected the appropriate prediction year based on the data of B-IBI sampling in each basin. One additional site (09MID1958) had MWMT values obtained from hydro gauge data. 24 sites did not have modelled MWMT values available.

Using temperature data from all King County stream gages for comparison, modelled MWMT values fit observed MWMT values reasonably well (R^2=0.63).

Roads

We obtained spatial data from Washington Department of Natural Resources (WA DNR) for all state roads, and abandoned forest practices roads. In addition, we obtained spatial data from the U.S. Forest Service (USFS) for all forest service roads on federally owned lands. We carefully examined the three layers and found that while the WA DNR State Roads layer was the most complete, there were some roads left out, particularly from the USFS dataset. For completeness, we added missing roads from the USFS and Abandoned Roads datasets to the WA DNR State Roads dataset.

National Water Model Retrospective Dataset V2.1

We obtained modelled retrospective hydrological data from 1979 to 2020 from the National Water Model Retrospective dataset (NWM) V2.1 (NOAA 2021) using the R package nwmTools (Johnson M 2022, Johnson & Blodgett 2020). We matched BIBI sample sites to the most appropriate stream identifier (COMID) in the National Hydrological Dataset Plus Version 2.1 and downloaded retrospective modelled data for each COMID. Several streams did not exist in this version of the NHD Plus V2.1 due to the coarser resolution (1:100,000 vs 1:24,000), and therefore, had no modelled data available. Where data was available, we calculated annual measures of stream flashiness and low flows, including Low Flow Index (Litz et al. 2020), TQ-Mean and High Pulse Count, as well as median stream power observed since 1979 and the maximum stream power observed per water year. A preliminary review of modelled flows versus observed flows in nine gaged basins in the dataset showed reasonably good correlation between predicted and observed flows (R^2 = 0.62).

In addition, we used the National Inventory of Dams from the US Army Corps of Engineers to identify BIBI sampling points that were downstream of dams, and thus, potentially have poorer model performance (Gala, personal communication based on NWM version 1.2). We identified 5 sampling sites downstream of known dams (Table 2). However, three of the sites seemed less likely to be highly impacted by flow alterations from the dams, and so the modelled data were used. In the remaining two sites on Jimmy Come Lately Creek and Cherry Creek, we used empirical gage data from gage locations further down the mainstem streams and adjusted the difference in basin size using the Drainage Ratio Method for partially gaged watersheds (Gala 2021). The correlation between the modelled flows and partial flows in these two watersheds was good (R^2= 0.7). Given this result, and the far more limited timeframe that gaged data was available, we opted to use modelled values in the dammed basins as well.

Sediment Values

We downloaded taxonomic data for all B-IBI samples from the Puget Sound Stream Benthos database, and used the R package bio.infer (Yuan 2007) to infer the percent sand/fine composition of substrates at each site at the time of sampling. This method uses maximum likelihood methods to predict sediment conditions based on the taxonomic assemblage of each sample. For our predictions, we used existing taxon-environment relationships for the western United States based on data collected by the US Environmental Protection Agency Environmental Monitoring and Assessment Program (Stoddard et al. 2005). In addition, we calculated the Fine Sediment Biotic Index (FSBI; Relyea et al. 2012), which was developed from taxa-sediment observations across the Pacific Northwest. The FSBI is based on taxa richness weighted by the sediment sensitivity of each taxa, and is used to indicate sand/fines <2 mm. We also calculated a third method of estimating fine sediment impacts from the macroinvertebrate community, the Biological Sediment Tolerance Index (Hubler et al. 2016). This method was developed in Oregon and uses weighted averaging of taxa sediment tolerance by abundance to estimate fine sediments < 0.06 mm). The literature reported validation R^2 values were 0.49 for BSTI, 0.42 for bio,infer, and not reported for FSBI.

Patch size

We used the R package landscapemetrics (Hesselbarth et al. 2019) to calculate median values for patch size of disturbances (forest conversion since 1985 prior to sampling or bare/developed). For this metric, we used disturbance rasters generated from the LCMAP data, which we subsequently processed through a boundary cleaning process to smooth the data into more contiguous patches.

Appendix H – Data Transformations

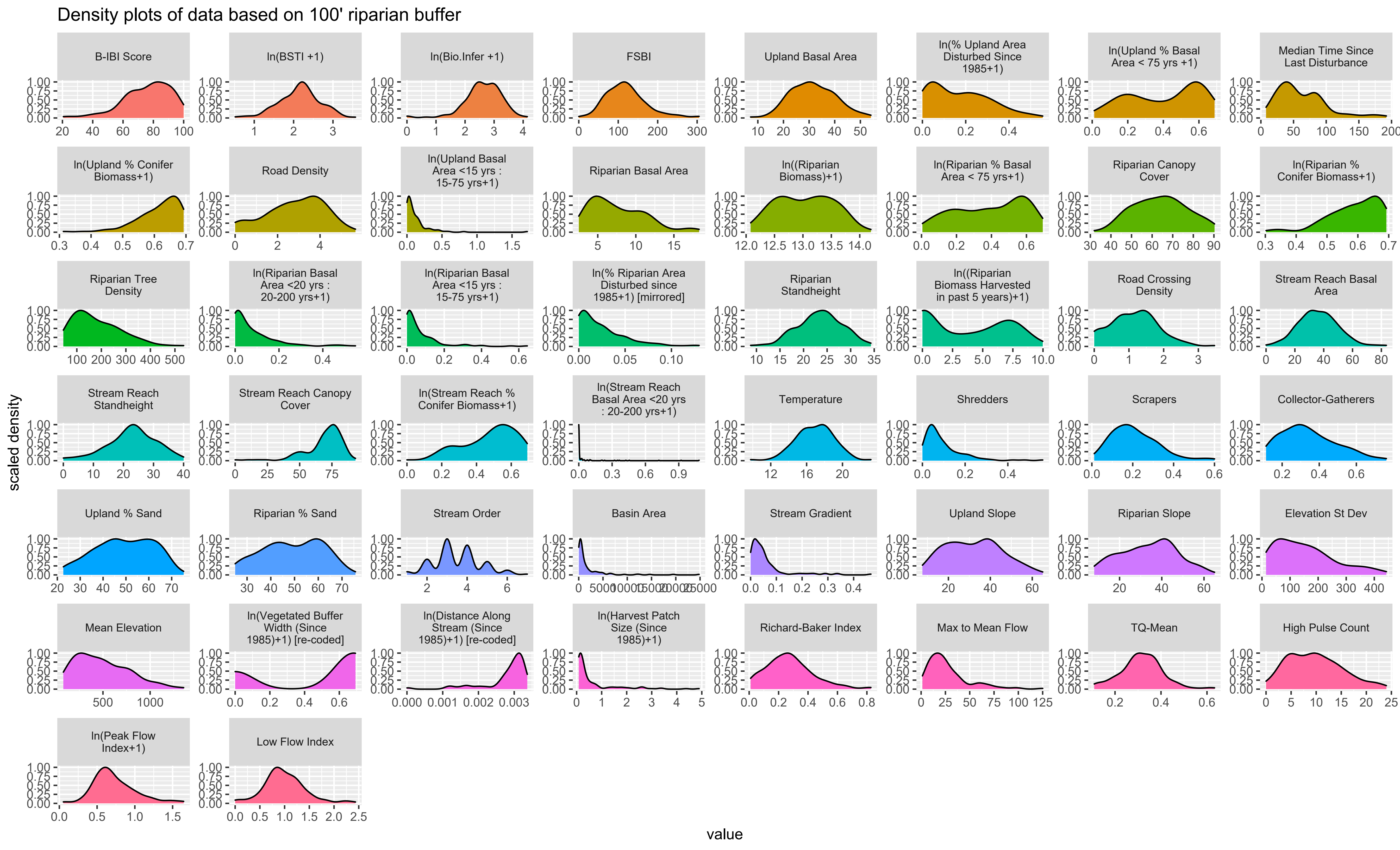
Table H-1: Data transformations of variables considered for inclusion. Variables that are italicized and bolded are those that were ultimately selected for inclusion in the initial structural equation model. The second column "Re-scaled and Re-coded" indicates where we first inverted (1/x) the variables, and then we re-coded the new values to include non-harvested basins by setting "no harvest" to equal 0. The new values were then subtracted from the maximum new value to preserve the original directionality of the data (bigger y = bigger y'). The third column is self-explanatory. The fourth column indicates where we subtracted the values from the maximum value in order to ensure all indicators of a single latent variable were positively correlated rather than negatively correlated.

Name	Re-scaled and Re-coded	1 + natural log transformed	Mirrored prior to SEM
B-IBI Score			
Bio.Infer		x	
BSTI		x	
FSBI			x
Max to Mean Flow			
Richard-Baker Index			
Riparian % Basal Area < 75 yrs		x	x
Riparian % Sand			
Riparian Basal Area			
Riparian Canopy Cover			
Riparian Slope			
Stream Reach Basal Area			
Stream Reach Canopy Cover			
Stream Reach Standheight			
TQ-Mean			x
% Riparian Area Disturbed since 1985		x	x
% Upland Area Disturbed Since 1985		x	x
Basin Area			
Collector-Gatherers			
Distance Along Stream (Since 1985)	x	x	
Elevation St Dev			
Harvest Patch Size (Since 1985)			
High Pulse Count			
Low Flow Index			
Mean Elevation			

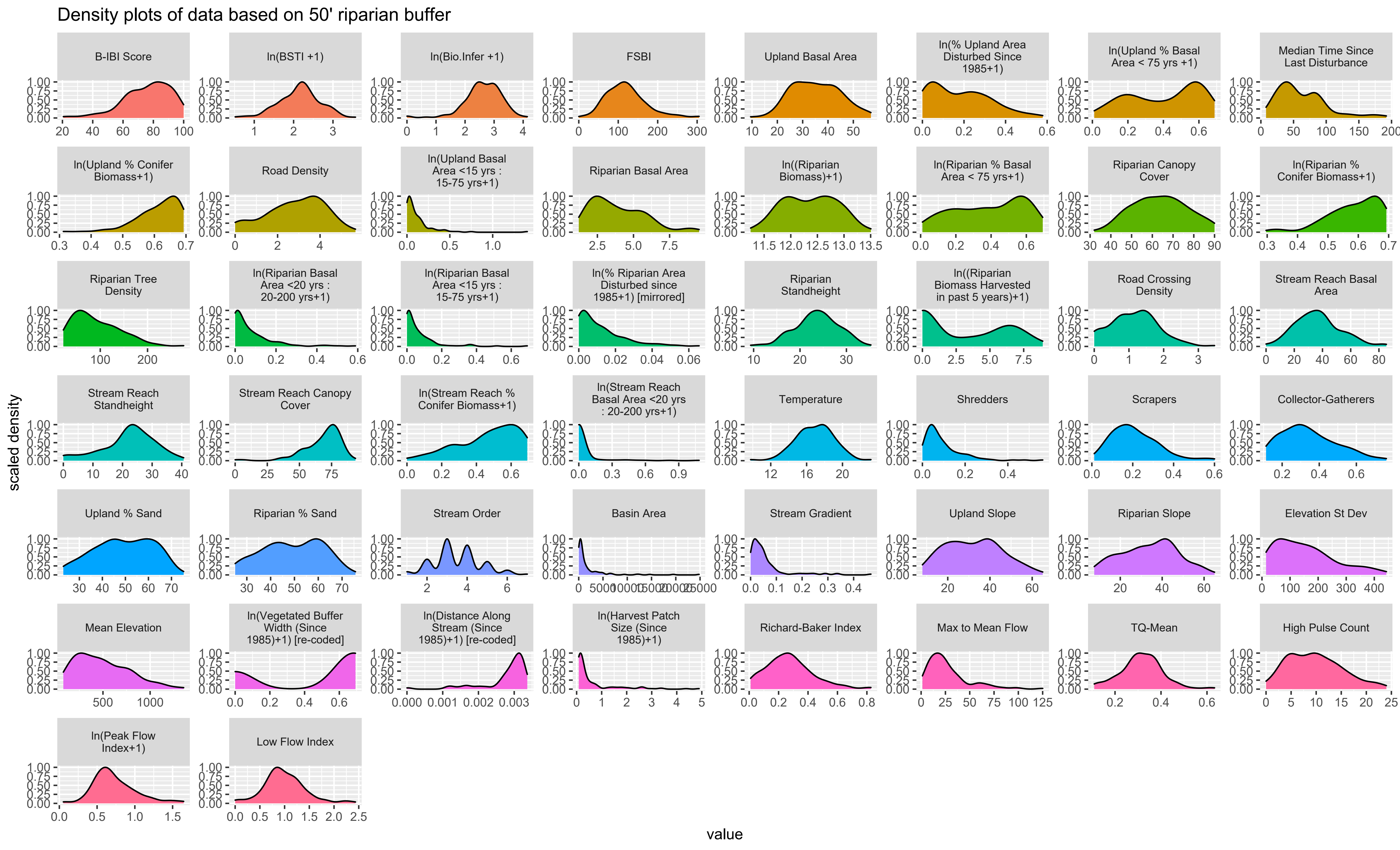
Median Time Since Last Disturbance			
Peak Flow Index		x	
Riparian % Conifer Biomass		x	x
Riparian Basal Area <15 yrs : 15-75 yrs		x	x
Riparian Basal Area <20 yrs : 20-200 yrs		x	x
Riparian Biomass		x	
Riparian Biomass Harvested in past 5 years		x	x
Riparian Standheight			
Riparian Tree Density			
Road Crossing Density			
Road Density			x
Scrapers			
Shredders			
Stream Gradient			
Stream Order			
Stream Reach % Conifer Biomass		x	
Stream Reach Basal Area <20 yrs : 20-200 yrs		x	x
Temperature			
Upland % Basal Area < 75 yrs		x	x
Upland % Conifer Biomass		x	
Upland % Sand			
Upland Basal Area			
Upland Basal Area <15 yrs : 15-75 yrs		x	x
Upland Slope			
Vegetated Buffer Width (Since 1985)	x	x	

Appendix I – Data Distributions

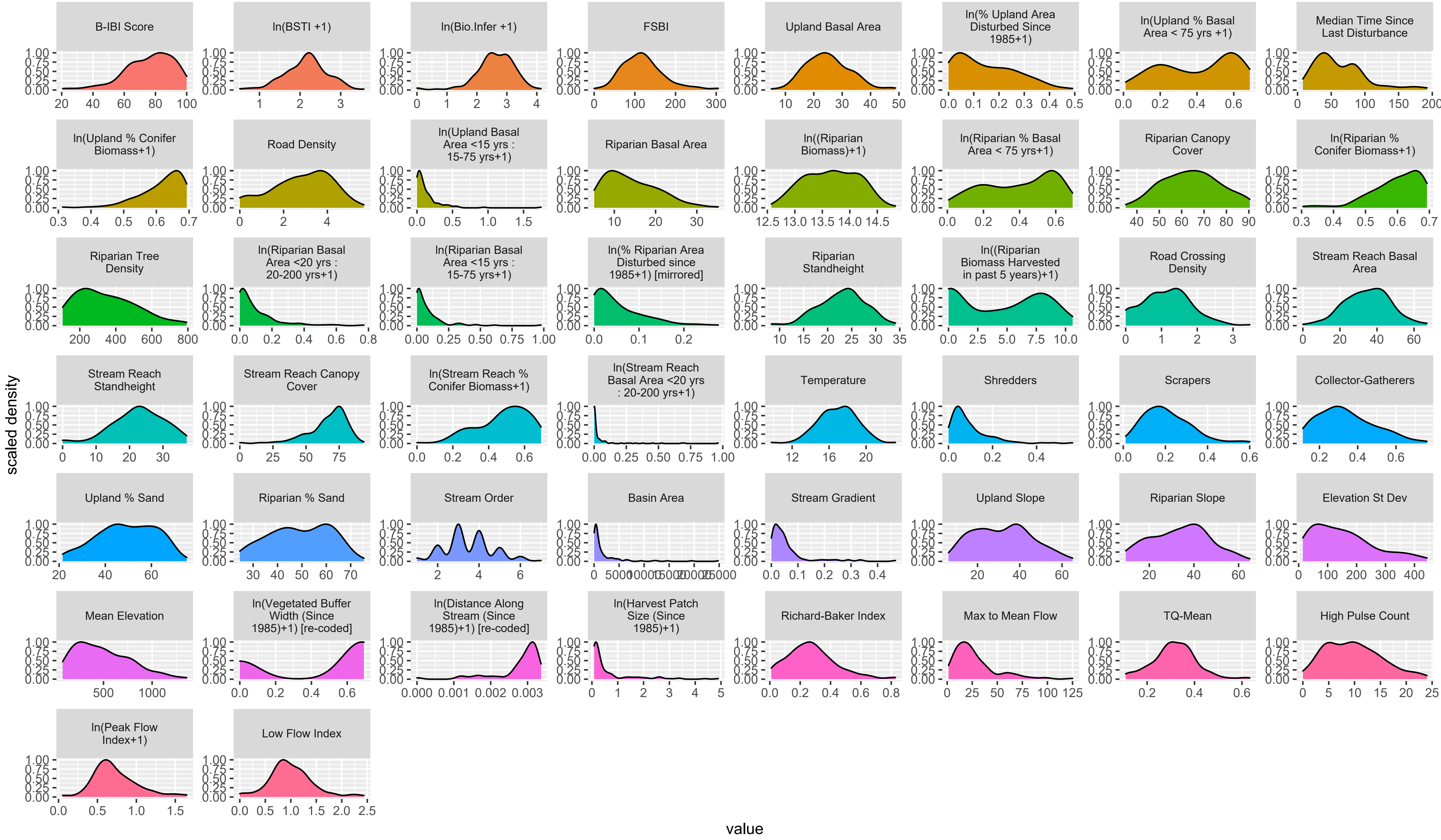
(See “Appendix I DensityPlots_Buff100 .pdf”,
“Appendix I DensityPlots_CoreRMZ.pdf”,
“Appendix I DensityPlots_SPTH.pdf”)



Density plots of data based on 100' riparian buffer
B-IBI Score
ln(BSTI +1)
ln(Bio.Infer +1)
FSBI
Upland Basal Area
ln(% Upland Area Disturbed Since 1985+1)
ln(Upland % Basal Area < 75 yrs +1)
Median Time Since Last Disturbance
ln(Upland % Conifer Biomass+1)
Road Density
ln(Upland Basal Area <15 yrs : 15-75 yrs+1)
Riparian Basal Area
ln((Riparian Biomass)+1)
ln(Riparian % Basal Area < 75 yrs+1)
Riparian Canopy Cover
ln(Riparian % Conifer Biomass+1)
Riparian Tree Density
ln(Riparian Basal Area <20 yrs : 20-200 yrs+1)
ln(Riparian Basal Area <15 yrs : 15-75 yrs+1)
ln(% Riparian Area Disturbed since 1985+1) [mirrored]
Riparian Standheight
ln((Riparian Biomass Harvested in past 5 years)+1)
Road Crossing Density
Stream Reach Basal Area
Stream Reach Standheight
Stream Reach Canopy Cover
ln(Stream Reach % Conifer Biomass+1)
ln(Stream Reach Basal Area <20 yrs : 20-200 yrs+1)
Temperature
Shredders
Scrapers
Collector-Gatherers
Upland % Sand
Riparian % Sand
Stream Order
Basin Area
Stream Gradient
Upland Slope
Riparian Slope
Elevation St Dev
Mean Elevation
ln(Vegetated Buffer Width (Since 1985)+1) [re-coded]
ln(Distance Along Stream (Since 1985)+1) [re-coded]
ln(Harvest Patch Size (Since 1985)+1)
Richard-Baker Index
Max to Mean Flow
TQ-Mean
High Pulse Count
ln(Peak Flow Index+1)
Low Flow Index
scaled density
value

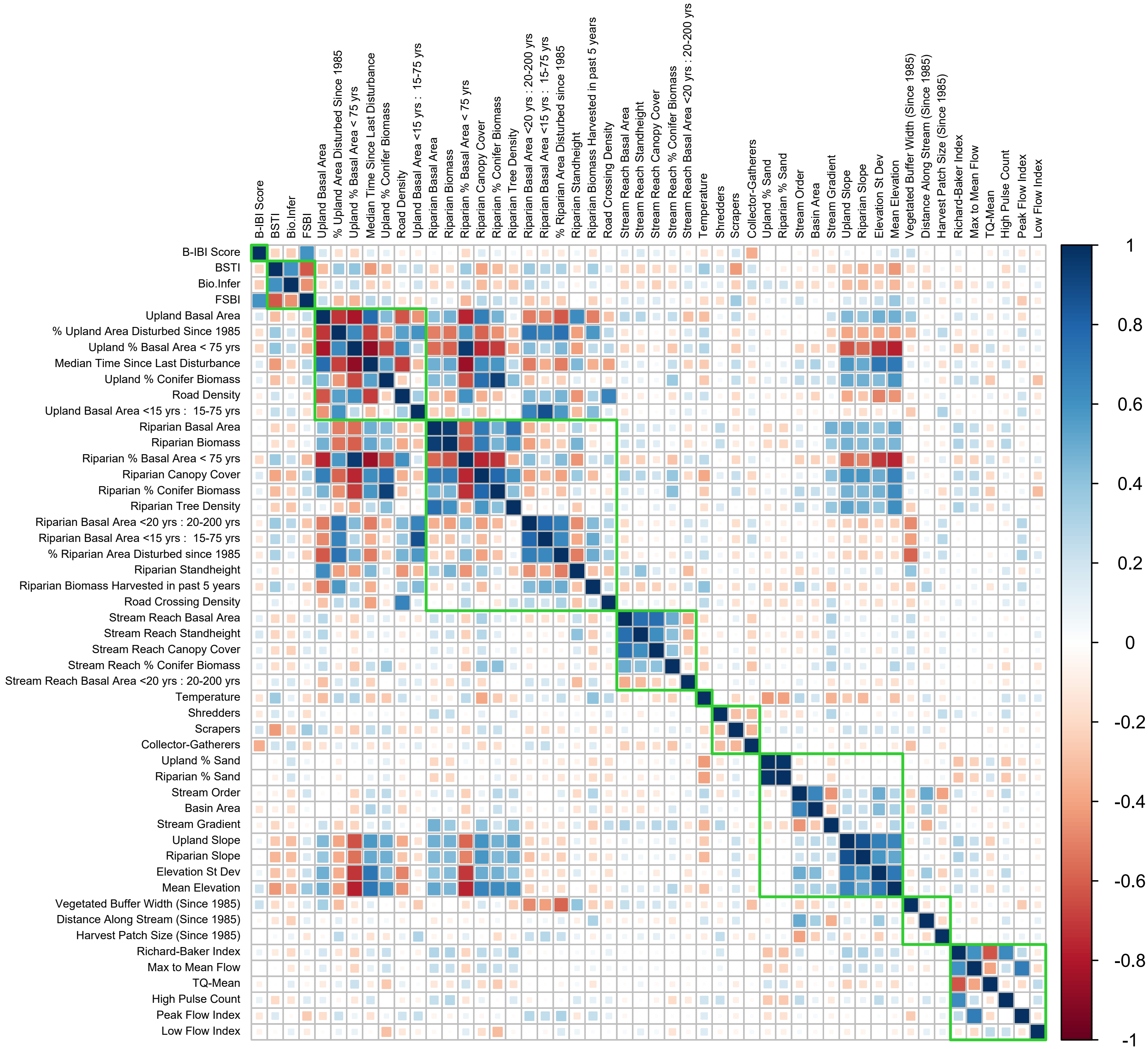

Density plots of data based on 50' riparian buffer
B-IBI Score
ln(BSTI +1)
ln(Bio.Infer +1)
FSBI
Upland Basal Area
ln(% Upland Area Disturbed Since 1985+1)
ln(Upland % Basal Area < 75 yrs +1)
Median Time Since Last Disturbance
ln(Upland % Conifer Biomass+1)
Road Density
ln(Upland Basal Area <15 yrs : 15-75 yrs+1)
Riparian Basal Area
ln((Riparian Biomass)+1)
ln(Riparian % Basal Area < 75 yrs+1)
Riparian Canopy Cover
ln(Riparian % Conifer Biomass+1)
Riparian Tree Density
ln(Riparian Basal Area <20 yrs : 20-200 yrs+1)
ln(Riparian Basal Area <15 yrs : 15-75 yrs+1)
ln(% Riparian Area Disturbed since 1985+1) [mirrored]
Riparian Standheight
ln((Riparian Biomass Harvested in past 5 years)+1)
Road Crossing Density
Stream Reach Basal Area
Stream Reach Standheight
Stream Reach Canopy Cover
ln(Stream Reach % Conifer Biomass+1)
ln(Stream Reach Basal Area <20 yrs : 20-200 yrs+1)
Temperature
Shredders
Scrapers
Collector-Gatherers
Upland % Sand
Riparian % Sand
Stream Order
Basin Area
Stream Gradient
Upland Slope
Riparian Slope
Elevation St Dev
Mean Elevation
ln(Vegetated Buffer Width (Since 1985)+1) [re-coded]
ln(Distance Along Stream (Since 1985)+1) [re-coded]
ln(Harvest Patch Size (Since 1985)+1)
Richard-Baker Index
Max to Mean Flow
TQ-Mean
High Pulse Count
ln(Peak Flow Index+1)
Low Flow Index
scaled density
value

Density plots of data based on SPTH riparian buffer

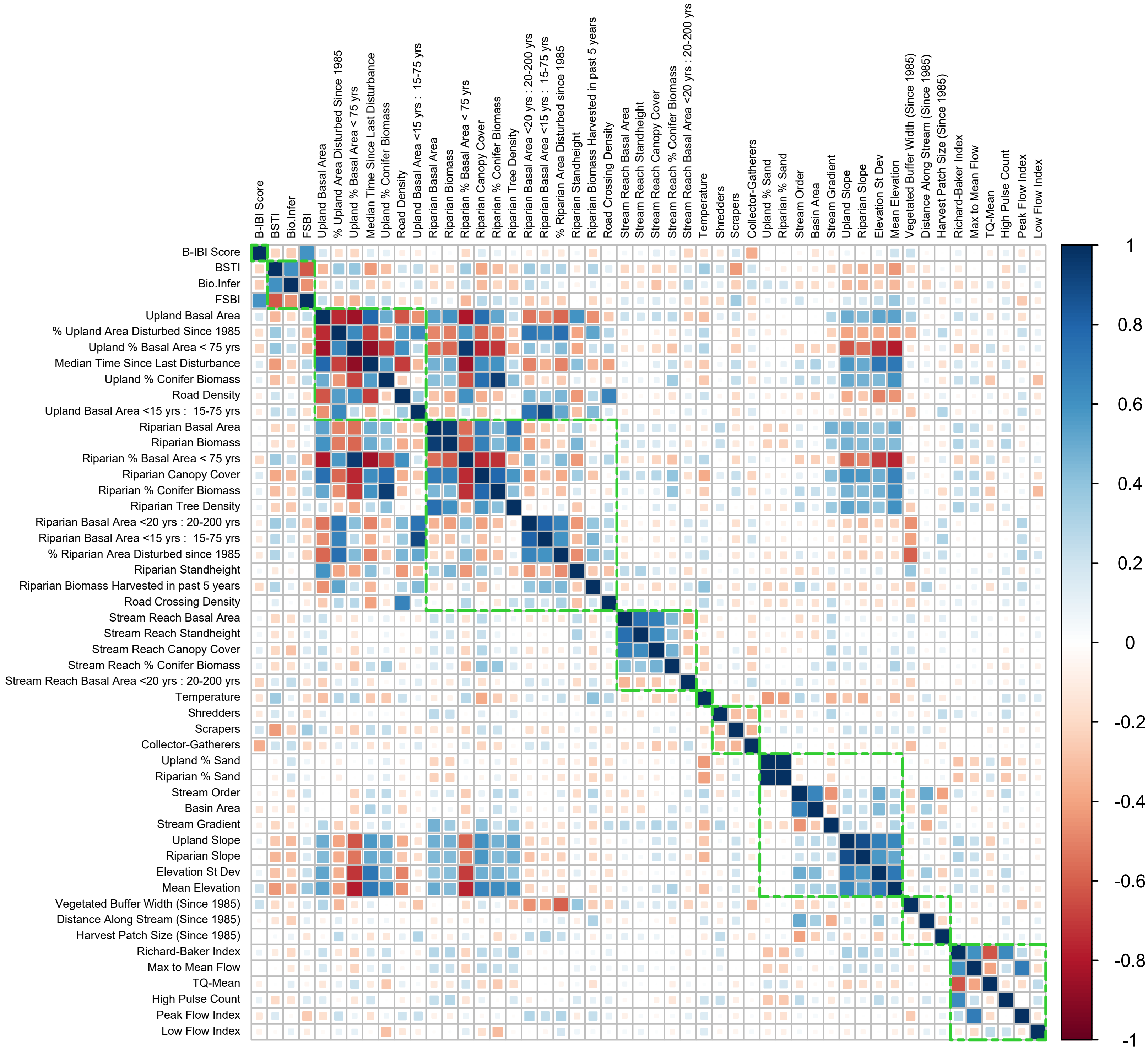


Appendix J – Correlation Tables

(See “Appendix J_Correlationmatrix__Buff100 .pdf”,
“Appendix J_Correlationmatrix__CoreRMZ.pdf”,
“Appendix J_Correlationmatrix__SPTH.pdf”)



B-IBI Score
BSTI
Bio.Infer
FSBI
Upland Basal Area
% Upland Area Disturbed Since 1985
Upland % Basal Area < 75 yrs
Median Time Since Last Disturbance
Upland % Conifer Biomass
Road Density
Upland Basal Area <15 yrs : 15-75 yrs
Riparian Basal Area
Riparian Biomass
Riparian % Basal Area < 75 yrs
Riparian Canopy Cover
Riparian % Conifer Biomass
Riparian Tree Density
Riparian Basal Area <20 yrs : 20-200 yrs
Riparian Basal Area <15 yrs : 15-75 yrs
% Riparian Area Disturbed since 1985
Riparian Standheight
Riparian Biomass Harvested in past 5 years
Road Crossing Density
Stream Reach Basal Area
Stream Reach Standheight
Stream Reach Canopy Cover
Stream Reach % Conifer Biomass
Stream Reach Basal Area <20 yrs : 20-200 yrs
Temperature
Shredders
Scrapers
Collector-Gatherers
Upland % Sand
Riparian % Sand
Stream Order
Basin Area
Stream Gradient
Upland Slope
Riparian Slope
Elevation St Dev
Mean Elevation
Vegetated Buffer Width (Since 1985)
Distance Along Stream (Since 1985)
Harvest Patch Size (Since 1985)
Richard-Baker Index
Max to Mean Flow
TQ-Mean
High Pulse Count
Peak Flow Index
Low Flow Index
1
0.8
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
-0.8
-1


B-IBI Score
BSTI
Bio.Infer
FSBI
Upland Basal Area
% Upland Area Disturbed Since 1985
Upland % Basal Area < 75 yrs
Median Time Since Last Disturbance
Upland % Conifer Biomass
Road Density
Upland Basal Area <15 yrs : 15-75 yrs
Riparian Basal Area
Riparian Biomass
Riparian % Basal Area < 75 yrs
Riparian Canopy Cover
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Riparian Basal Area <20 yrs : 20-200 yrs
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Stream Reach Basal Area
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Temperature
Shredders
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Riparian Slope
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Vegetated Buffer Width (Since 1985)
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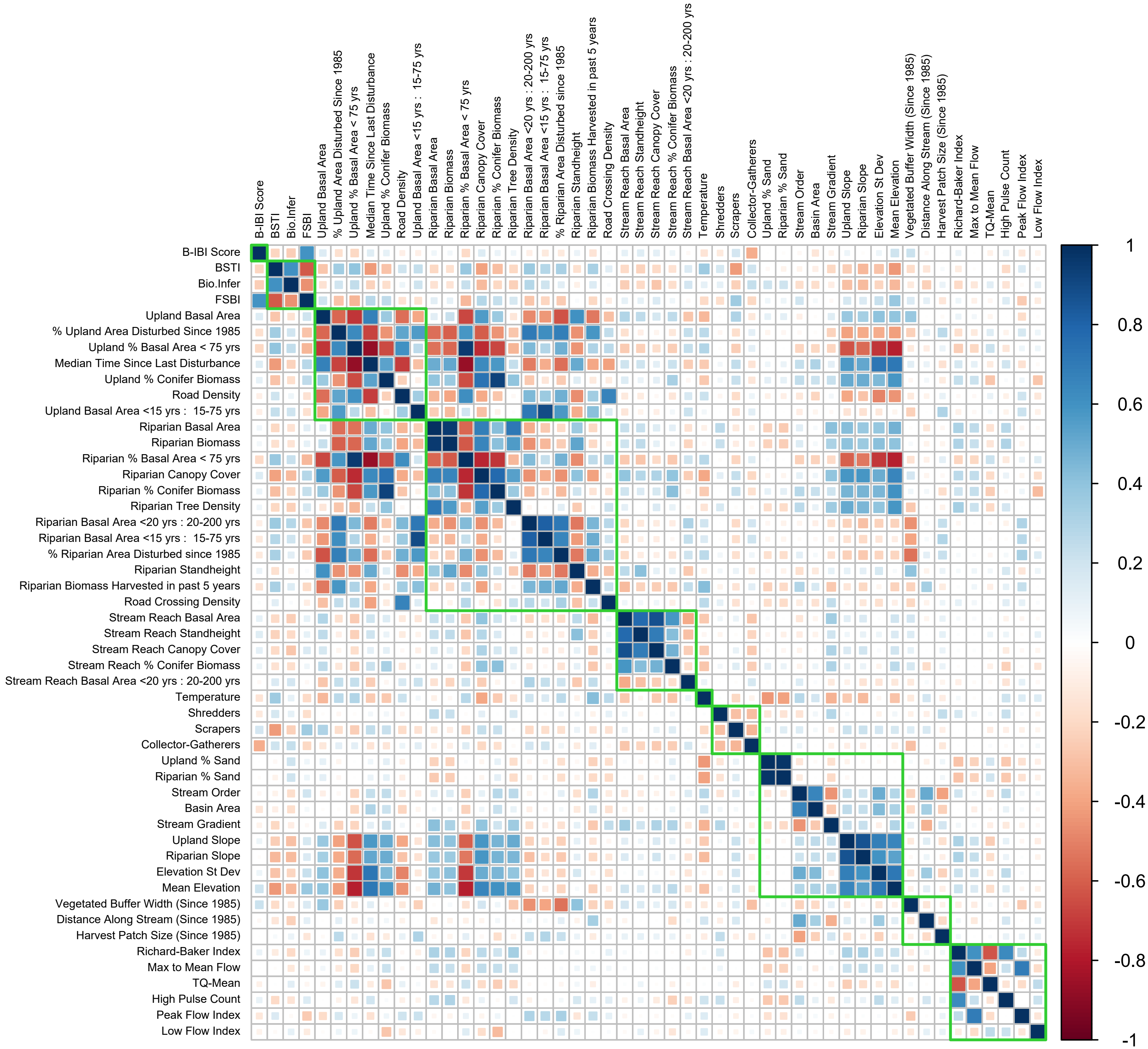
Richard-Baker Index
Max to Mean Flow
TQ-Mean
High Pulse Count
Peak Flow Index
Low Flow Index
1
0.8
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
-0.8
-1


B-IBI Score
BSTI
Bio.Infer
FSBI
Upland Basal Area
% Upland Area Disturbed Since 1985
Upland % Basal Area < 75 yrs
Median Time Since Last Disturbance
Upland % Conifer Biomass
Road Density
Upland Basal Area <15 yrs : 15-75 yrs
Riparian Basal Area
Riparian Biomass
Riparian % Basal Area < 75 yrs
Riparian Canopy Cover
Riparian % Conifer Biomass
Riparian Tree Density
Riparian Basal Area <20 yrs : 20-200 yrs
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% Riparian Area Disturbed since 1985
Riparian Standheight
Riparian Biomass Harvested in past 5 years
Road Crossing Density
Stream Reach Basal Area
Stream Reach Standheight
Stream Reach Canopy Cover
Stream Reach % Conifer Biomass
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Temperature
Shredders
Scrapers
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Upland % Sand
Riparian % Sand
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Upland Slope
Riparian Slope
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Mean Elevation
Vegetated Buffer Width (Since 1985)
Distance Along Stream (Since 1985)
Harvest Patch Size (Since 1985)
Richard-Baker Index
Max to Mean Flow
TQ-Mean
High Pulse Count
Peak Flow Index
Low Flow Index
1
0.8
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
-0.8
-1

Appendix K: Detailed SEM steps

Initial Model Construction

We began model construction with a close examination of the correlations between all variables that we considered including in the model (Figure 6). We did this for two reasons. First, some of the ideas presented in the meta-model could be represented by more than one variable. For example, we would expect that TQ-mean, Richard-Baker Index and High Pulse Count represent different but related aspects of storm flows. If multiple, independent but conceptually related variables correlate well enough, they could be considered indicators for a single, unmeasured "latent" factor (e.g. "storm flow") and included in the model as such.

A second reason we examined the correlation tables was to look for additional relationships that were not considered in the meta-model, but which warranted attention. These include relationships between variables and potentially confounding factors, such as basin size, elevation, and slope, to decide whether these additional factors needed to be included, or if certain variables should be removed. To keep the model as simple as possible and focus on the effects of timber harvest, we preferentially removed variables, where possible, that were too correlated with environmental factors rather than add those environmental factors to the model.

After specifying the model, we examined the power of the analysis and estimated the model parameters.

Structural Model Development

In translating our expectations from the meta-model to a structural model, we chose to consolidate most forest conditions into a single latent factor representing riparian forest stability. Examination of the variables revealed that most variables related to forest disturbance (including road impacts) were highly correlated with each other (Figure 6). This degree of correlation was unsurprising, as they are all largely different measures of a common cause: forest harvest. Disturbance variables specific to upland and riparian areas were also closely correlated. Clearcutting in non-fish bearing streams is allowed up to the stream edge, and so forest conditions in many riparian areas were highly correlated with those in upland areas because harvest practices were similar in both. In a structural equation model, all these variables could be considered indicators of a single latent factor representing forest disturbance due to harvest in the basin. We opted to select only the riparian variables to create latent variables of riparian forest disturbance and sampling reach forest disturbance, which is reasonable because the riparian area is expected to have the most influence on in-stream processes (Naiman et al. 2000, Jyväsjärvi et al. 2020). This will allow us to compare model fit across the three riparian spatial extents (50-foot, 100-foot, Site Potential Tree Height) and identify which had the most influence on B-IBI scores.

We selected slope and sand as variables to characterize the physical basin environments in our structural model. Basin slope, riparian slope, mean basin elevation and basin elevation standard deviation were all highly correlated with each other, and to a lesser extent with stream order, basin area, and stream gradient. Standard deviation of elevation is another way of approximating steepness in the basin. Elevation was likely correlated with steepness because of the presence of high mountains in some basins, which also tend to have steep slopes. Since slope is more likely to be a causal influence behind hydrology and sediment, we selected slope alone to characterize the basins. The sand content of the soil in upland and riparian areas was surprisingly not correlated with basin slope, and only weakly negatively correlated with stream gradient and stream order.

We removed some variables from consideration in the structural model due to confounding effects. Stream order, stream gradient, and basin area were all correlated with each other, which was expected. Variables which measured distance of disturbance to the stream or absolute harvest patch area were also correlated with these basin characteristics. As a result, the causal effects of these distance and area measures may be difficult to disentangle from effects due to basin size. Though distance along stream, distance to the stream, vegetated buffer width and patch area may interact with how forest activities affect B-IBI outside of the influence of basin size, for simplicity we will not consider these variables in the model.

We consolidated storm flows and base flows into a single latent factor to be used in the structural model. Measures of flashiness did not correlate as well as expected. However, TQ-mean, Richard-Baker Index and Max:Mean Q were consistent enough in their correlations that they may be considered indicators of a latent factor representing flashiness. Peak flow index seemed to associate with indicators of more recent disturbance (e.g. those that occurred within the 5 to 30 years prior to sampling), while the other variables were more associated with overall historical disturbances. The Low Flow Index was correlated with measures of flashiness, albeit weakly, and was not a distinct variable. As a result, our initial structural model will not distinguish between the effects of stormwater flows and base flows on B-IBI but will consider only the combined concept of flashiness.

Further inspection of the correlation tables revealed some relationships that were contrary to our expectations developed in the meta-model. As a result, in the structural model we kept three macroinvertebrate functional feeding groups (scrapers, shredders, collector-gatherers) as distinct variables, rather than as indicators of a single factor (e.g. "trophic structure"). We also added connections that we did not initially consider from sediment and leaf area to the functional feeding groups and revised the spatial scales that we had assumed in our meta-model would influence each group. Collector-Gatherers, Shredders, and Scrapers all appeared to be negatively correlated with each other. Our initial assumption was that the ratio of shredders versus collector-gatherers plus scrapers would reflect a trophic gradient between consumers that feed on algae and those that feed on detritus, further reflecting the availability of sunlight (at the reach scale) vs leaf litter (at riparian and upstream riparian area scales) (See Appendix E for detailed exposition of meta-model assumptions). However, the correlations suggest sensitivity to sediment may be an additional gradient that differentiates the three functional feeding groups. Furthermore, the correlations showed generally a negative correlation between indicators of upstream riparian disturbance and scrapers, with a notable exception for basal area and biomass, which are predictors for leaf area. In that case, the correlations suggested that scrapers prefer environments downstream of areas with little disturbance but also less leaf area. This may be consistent with old growth forests, which have been observed to have more light penetration and less consistent canopy closures than recovering forests (Kaylor et al. 2017). Shredders, on the other hand, seemed to positively associate with denser leaf areas upstream, and had less of a relationship with other indicators of forest disturbance. As a result of our findings, our initial structural model will reflect an updated hypothesis of how these communities interact.

The correlation tables revealed some additional unanticipated associations that we did not include in the initial structural model, but considered for model modification if necessary. For instance, the Fine Sediment Biological Index (FSBI) was much more strongly correlated with B-IBI than the other two sediment metrics. This is likely because FSBI and B-IBI are both highly dependent on measures of taxa richness, where the Biological Sediment Tolerance Index (BSTI) and inferred sediment values (obtained from the R package bio.infer) are not. In addition, counter to our expectations, increased forest disturbance appeared to be associated with less flashy flows. Adding slope as a causal factor to both riparian disturbance and flashy flows may explain this apparent relationship. We will not include these links in our initial structural model but may consider adding them if our initial fit is poor.

Using confirmatory factor analysis (CFA), we selected indicators for four latent factors: upstream riparian forest stability, riparian forest stability in the sampling reach area, flashiness, and fine sediments. Because our sample size was relatively small, we selected only three indicators per latent factor to simplify the model. Figure 7 shows our initial structural equation model, with latent factors incorporated.

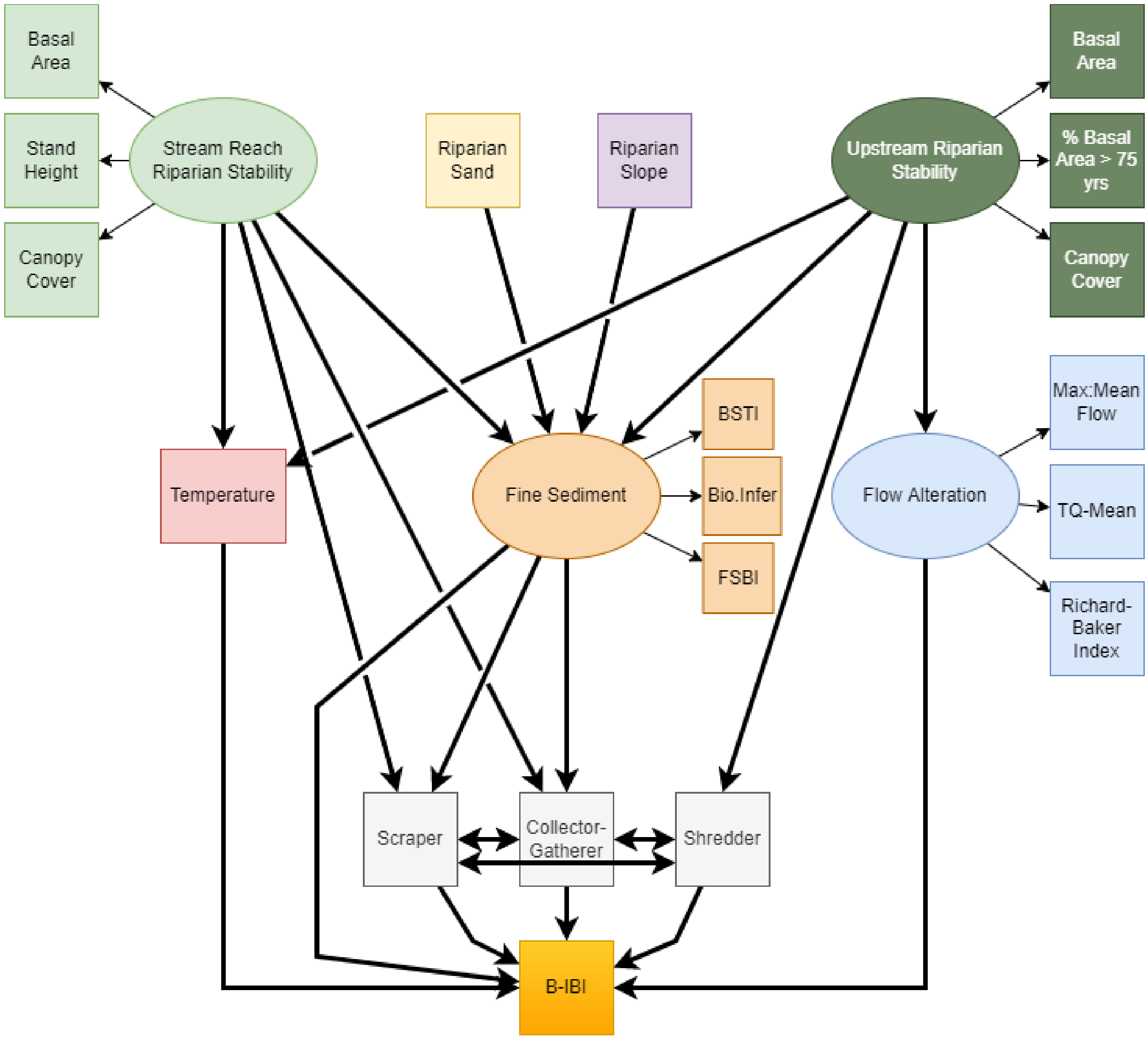

Figure 15. Initial Structural Model. Ovals represent latent (unmeasured) factors, and rectangles represent factors measured either directly or obtained from other spatial data sets. Shapes with matching or similar colors represent conceptually similar factors.

Initial Model Power

The initial structural model had 132 observations with 136 degrees of freedom, resulting in an 89% likelihood of achieving a RMSEA value of less than 0.05. (MacCallum et al 1996).

Initial Model Estimation

Our initial structural equation model failed the criteria for a well-fitting model for all three riparian distances (p = 0 for 50 ft, 100 ft and variable SPTH buffer widths). Having failed the confirmatory phase of SEM, we proceeded to revise our causal theory based in part on observations of the data and links suggested by the modelling software but rooted in ecological grounds. All modifications made to the model were based on plausible connections given our knowledge of the system.

Model Modification

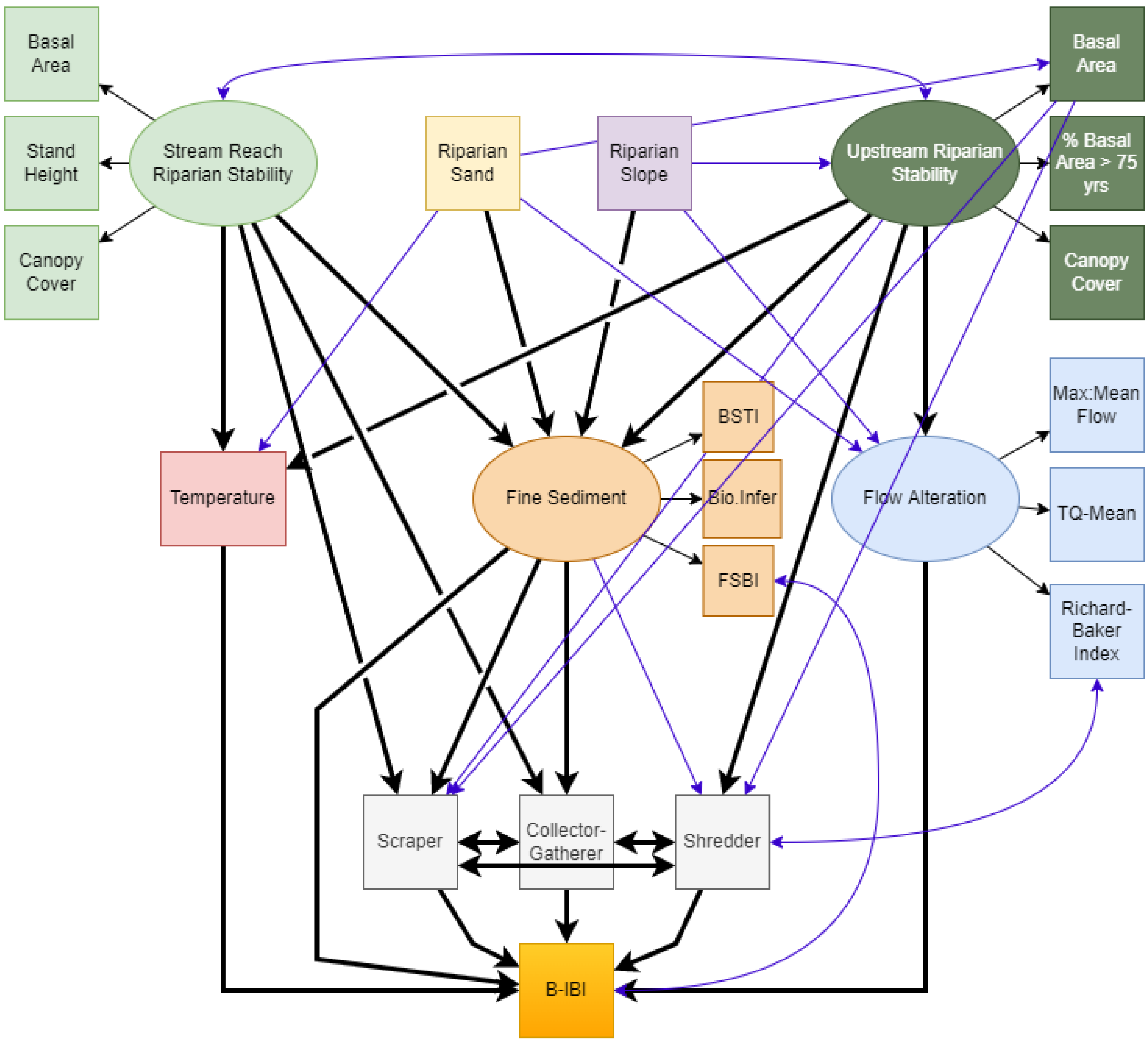

Figure 16. Modified Structural Equation Model. Purple lines represent new links in modified model. Ovals represent latent (unmeasured) factors, and rectangles represent factors measured either directly or obtained from other spatial data sets. Shapes with matching or similar colors represent conceptually similar factors.

In our modified model, we retained all our initial links, and added additional links (Figure 8).

Higher percent sand content in the soil correlated with lower stream temperatures and decreased flashiness, most probably by increasing soil permeability and promoting infiltration of runoff before it reached the stream. In addition, the proportion of sand in the soil appeared to be negatively correlated with total basal area. We theorize this link is a causal effect due to poorer nutrient retention and soil moisture in sandier soils (Baughman

1993), which can slow tree growth and subsequently reduce basal area in otherwise undisturbed areas.

As noted in the results of our correlation analysis, steeper slopes were correlated with more stable riparian forest conditions, likely by making those stands difficult to access with logging equipment, at least historically. Steep slopes also likely increase the rate and volume of runoff reaching the stream, thereby causing increased flashiness. While the bivariate correlations seemed to suggest that stable riparian forest conditions increase flashiness (which was counter to our expectations), we hypothesize that this apparent positive relationship was simply due to the common cause of slope.

Upstream riparian basal area has additional correlations with the stream community apart from forest stability. Since basal area can be used as a predictor of leaf area (which, in turn can predict light transmission), we theorize these are separate causal links that are likely due to how basal area interacts with light availability and the delivery of leaf litter (Sonohat et al. 2004). In areas where basal area is higher for reasons independent of forest stability, we hypothesize more leaf litter is available, causing an increase in the proportion of shredders in the community. We hypothesize this same mechanism causes a decrease in the proportion of scrapers because increased leaf cover can block from light reaching the substrate, thereby impeding the development of algal biofilms that scrapers feed on.

Conversely, upstream riparian stability was positively correlated with the proportion of scrapers in the benthic community. We theorize that scrapers favor light conditions created by old-growth forests, where increased gaps in the canopy, decline in leaf area and establishment of coniferous trees reduce leaf litter inputs to the stream and increase light availability.

Fine sediment appears to be correlated with the proportion of shredders in the benthic community, and we speculate this is because any conditions that retain fine sediments in the stream would also retain organic debris. Our modelling software suggested a correlative link between the Richard-Baker Index (Baker et al. 2004) and the shredder community, which we interpret as a relationship between flow oscillations and the delivery of organic debris. We don't have enough information to assert this relationship is directly causal (e.g., higher RBI values cause higher proportions of shredders), or if there is a shared cause that is separate from or more specific than flashiness, such as the volume of water that flows as surface runoff vs groundwater.

Confirming an observation we made during our initial model development, there appears to be a persistent correlation between the Fine Sediment Biotic Index (FSBI) and the Benthic Index of Biotic Integrity (B-IBI) that exists apart from the relationship between sediment and B-IBI. This correlation is likely because FSBI and B-IBI values are highly dependent on taxa richness, and thus, have a common causal factor.

Forest stability in the stream reach riparian zone and upstream riparian zone appear to be correlated with each other because of spatial autocorrelation— the tendency of observations to have similar values based on the physical closeness of the measurement locations. In this case, a stream basin with stable forests in its upstream riparian area is more likely to have stable forests in the immediate sampling stream reach. The opposite case is also true: a stream basin with more forest harvest activity in the upstream riparian area is more likely to also have harvest activities in the immediate sampling stream reach.

Modified Model Power

The modified model had 132 observations with 126 degrees of freedom, resulting in an 87% likelihood of achieving a RMSEA value of less than 0.05.

Modified Model Estimation

The modified model fit our data based on 50 ft and 100 ft riparian areas, but not for data based on the Site Potential Tree Heights (Table 1). The model based on 100 ft riparian area had the best fit.

Modified Model Validation

We attempted to validate our modified model using the secondary data set. This data set was substantially smaller than our primary data set (n= 80 vs n=132). Having so few observations in a complex model dramatically reduces the power of the analysis to 60%. Not surprisingly, the model did not meet fit criteria (Table 1). As a result, we were unable to confirm that the data support the hypotheses presented in the model.

We proceeded to modify the model to see if a simplified version could meet model fit criteria despite the small number of observations. Without an additional dataset to use for validation, the remainder of our effort will remain strictly in the exploratory phase. Our findings will therefore be presented as tentative hypotheses to be tested in future field or observational studies.

Final Model Modification

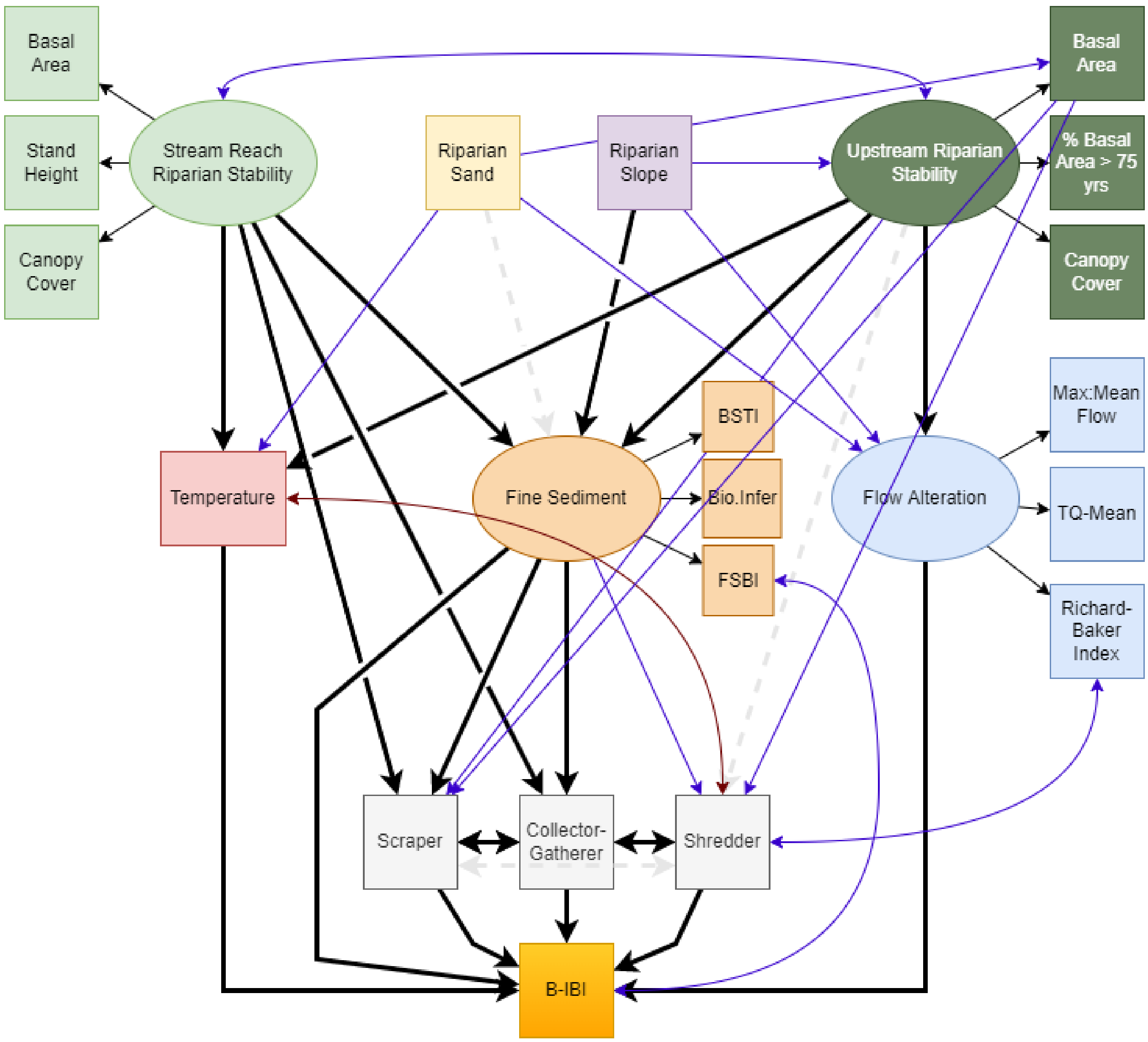

Figure 17. Final Modified Structural Equation Model. Purple lines represent links added in the first modified model. Red lines represent new links in the final modified model. Gray dashed lines represent links set to "no relationship" in the final modified model. Ovals represent latent (unmeasured) factors, and rectangles represent factors measured either directly or obtained from other spatial data sets. Shapes with matching or similar colors represent conceptually similar factors.

We removed causal links that met two criteria: 1) our first modified model consistently found the link to be insignificant; and 2) we had additional reasoning to believe there was no relationship. Rather than simply remove the links from the model, we set the relationship to zero to assert no relationship. In addition, we considered additional links suggested by the statistical software (Figure 9).

The percent of sand in the soil had no significant relationship with the fine sediment factor, as informed by the three fine sediment models. Soil classified as sand (0.05 mm to 2 mm) may be too coarse to influence the fine sediment factor. While bio.infer sediment values and FSBI indices were developed using sand/fines (<2 mm), the BSTI index is explicitly for fines < 0.06 mm. The latent factor Fine Sediment is estimated from the shared variance among the three indicators, and as a result, coarser sediment sizes may not be adequately estimated.

Forest stability in the whole upstream riparian area did not significantly affect proportion of shredders in the community. The relationship between riparian forest conditions and shredders appears to be best captured in the relationship between total basal area and shredders. This suggests that shredders are rather unaffected by forest stability, so long as there is dense vegetation to supply leaf litter.

The correlation between shredders and scrapers was consistently insignificant in the first modified model. Shredders and scrapers appear to occupy different habitats, controlled by light availability and detrital inputs, and therefore, may not have any other significant unmodelled factors mediating their relationship. In contrast, collector-gatherers are often generalists that appear in a wide range of habitats (Haggerty et al. 2002) and may compete with shredders or scrapers in a way not captured in the model.

The modelling software suggested a negative correlation between stream temperature and the proportion of shredders in the benthic community, which we opted to include as plausible. Other work has found a strong relationship between shredders and stream order (Haggerty et al. 2002). Stream order is highly correlated with mean basin elevation in the study area, which in turn is negatively correlated with stream temperature (Figure 6). For this reason, we consider the correlation between shredders and stream temperatures to represent a common cause through stream order.

Appendix L: Model Statistics

Initial SEM				**1st modified model**				**Final model**			
Primary dataset	**50'**	**100'**	**SPTH**	***Primary dataset***	**50'**	**100'**	**SPTH**	***Primary dataset***	**50'**	**100'**	**SPTH**
Robust Chi-Square	331.0	377.3	400.1	Robust Chi-Square	150.6	148.2	177.0	Robust Chi-Square	142.4	139.1	169.0
Degrees of Freedom	136			Degrees of Freedom	126			Degrees of Freedom	128		
Observations	132			Observations	132			Observations	132		
Robust Chi-Square p-val	0.00	0.00	0.00	Robust Chi-Square p-val	0.07	0.09	0.00	Robust Chi-Square p-val	0.18	0.24	0.01
SRMR	0.10	0.11	0.11	SRMR	0.05	0.05	0.06	SRMR	0.06	0.05	0.06
rmsea	0.10	0.11	0.12	rmsea	0.03	0.03	0.05	rmsea	0.02	0.02	0.05
cfi	0.82	0.80	0.79	cfi	0.98	0.99	0.96	cfi	0.99	1.00	0.97
cfi.robust	0.82	0.79	0.79	cfi.robust	0.98	0.98	0.96	cfi.robust	0.99	0.99	0.97
rmsea.robust	0.10	0.11	0.12	rmsea.robust	0.04	0.04	0.06	rmsea.robust	0.03	0.03	0.05
BIBI R^2	0.29	0.28	0.29	BIBI R^2	0.24	0.24	0.24	BIBI R^2	0.22	0.22	0.22
Initial SEM				**1st modified model**				**Final model**			
Secondary dataset				***Secondary dataset***	**50'**	**100'**	**SPTH**	***Secondary dataset***			
Robust Chi-Square				Robust Chi-Square	164.7	171.3	174.2	Robust Chi-Square	152.5	158.9	159.4
Degrees of Freedom				Degrees of Freedom	126			Degrees of Freedom	128		
Observations				Observations	80			Observations	80		
Robust Chi-Square p-val				Robust Chi-Square p-val	0.01	0.01	0.00	Robust Chi-Square p-val	0.07	0.03	0.03
SRMR				SRMR	0.08	0.08	0.08	SRMR	0.07	0.08	0.08
rmsea				rmsea	0.06	0.06	0.06	rmsea	0.04	0.05	0.04
cfi				cfi	0.96	0.95	0.96	cfi	0.98	0.97	0.98
cfi.robust				cfi.robust	0.95	0.94	0.95	cfi.robust	0.97	0.96	0.97
rmsea.robust				rmsea.robust	0.06	0.07	0.07	rmsea.robust	0.05	0.05	0.05
BIBI R^2				BIBI R^2	0.34	0.33	0.33	BIBI R^2	0.31	0.30	0.30

Appendix M: Model Parameter Estimates

(See excel spreadsheet "AppendixM_ParameterEstimates.xls")

Appendix N: Additional Discussion Points

Deviations from the QAPP
In the Quality Assurance Project Plan (King County 2021b), we had initially projected that we would delineate riparian buffers using methods described in Abood et al. (2012). We also described obtaining spatial data at multiple spatial scales: (1) within the upstream contributing watershed, (2) within a 1-km radius of the contributing watershed, (3) within the riparian zone of the whole contributing watershed, and (4) within the riparian zone in the 1-km contributing watershed. However, the results of our literature review persuaded us that conditions in the immediate sampling reach may make more ecological sense to investigate than in a 1-km radius. In contrast, while the methods of Abood et al. (2012) may yield riparian buffers that are ecologically relevant, we felt that buffers tied to regulatory considerations would better inform any recommendations we could make based on this study.

In addition, we had initially focused our efforts on investigating effects only in the Puget Lowlands ecoregion. However, to maximize statistical power we opted to include additional basin in adjacent ecoregions, but still within the Puget Sound Basin. This was identified in the QAPP as a contingency option.

Influence of Disturbance vs Physical Conditions
The results highlight that both physical conditions (e.g. slope and soil content) and disturbance from forest harvest can influence B-IBI scores. The indirect effects of riparian slope, independent of its effect on forest disturbance, significantly influenced B-IBI scores through flashiness and sediment. Though inconsistently significant between the two datasets, sand content of the soil may also affect B-IBI scores via a theoretical link to flashiness. In addition, both datasets showed multiple significant pathways between forest stability and B-IBI scores independent of physical conditions. In both the primary and secondary datasets, the effect of forest disturbance on B-IBI was greater than the effect of physical conditions on B-IBI independent of disturbance. However, forest stability in the whole upstream riparian area did not significantly affect flashiness, which was more highly tied to underlying physical conditions (i.e. percent sand and slope). A model that includes upland areas may find a relationship between forest stability and flashiness, but for this limited model there were no significant riparian effects.

In addition, we observed an interesting and unexpected link between forest disturbance and physical conditions. Slope seems to have historically played a role in the extent to which forests were harvested, wherein lands with steeper slopes increase logistical challenges, raise transportation costs, yield poorer timber quality (Soulard et al 2017), and raise regulatory and safety considerations (Forest Practices Illustrated). However, with better technology making steep slopes accessible to harvest equipment and fewer harvest opportunities in low-gradient areas as those lands develop (WA DNR 2007), the causal relationship between slope and forest stability may erode. Indeed, in the initial correlation tables, we saw a much weaker relationship between slope and riparian forest conditions in those variables focused on the just 5 to 30 years prior to sampling, rather than over all time

(Figure 6). In the correlation tables, riparian disturbance appears to split into two sub-groups: all-time disturbance and more recent disturbance (within 5-30 years). all-time disturbance seems to be highly correlated with certain basin characteristics that affect logging access—steep slopes and high elevations were associated with less disturbed riparian forests. Recent disturbance had a similar pattern, but less strong—which may be an indication of newer, more effective tools for harvesting forests in difficult locations.

Another unexpected link between physical conditions and B-IBI was the relationship between soil sand content and riparian basal area. The model results showed that basal area influences scrapers and shredders in addition to and independent of forest stability, which suggests that basal area had additional causal factors. Soil composition appeared to be one such factor. Sandier soils reduced riparian basal area, most likely reflecting poorer nutrient and moisture retention in the soil (Baughman 1993). As a result, soil sand content (and by extension nutrient availability) may play a role in organizing benthic community structure by partially controlling leaf area through basal area.

Several links from physical conditions in the final model had surprising results. Where we initially hypothesized that slope and sand would both have had a positive causal effect on stream sediments by increasing the rate of sediment delivery to the stream, we found instead that steeper slopes decreased fine sediments (though not consistently significantly), and soil sand content did not have an influence on fine sediment at all. The negative effect of slope on fine sediment was likely because streams with steeper slopes are more likely to have steeper stream gradients, and thus greater competency to flush sediments. The lack of effect of soil sand content on fine sediment may simply reflect that the shared variance between our sediment predictors (BSTI, bio.infer, FSBI) targeted fine sediment (< 0.06 mm), and thus would not have had a clear relationship to sand (<2 mm). Different metrics that are specifically targeted at sand in stream sediment may yet reveal an influence between riparian soil composition and stream sediment.

The links between flashiness, soil sand content and riparian slope were also unexpected, as was the relationship between soil sand content and stream temperature. Not surprisingly, steeper slopes appear to facilitate flashier hydrological regimes. More surprising, increased soil sand content appeared to have a dampening effect on flashiness. We speculate this effect is the result of greater soil permeability in sandier soil, absorbing more precipitation into groundwater and thus slowing discharge to the stream. This same mechanism would explain the link between sandier soils and cooler stream temperatures.

Flows

The model results revealed a very unexpected relationship between flashiness, riparian vegetation, and B-IBI. In numerous previous studies of urban streams, flashiness has consistently been considered a stressor negatively impacting macroinvertebrate communities and degrading B-IBI scores (DeGasperi et al. 2009, King County 2019, Marshalonis & Larson 2019, Danehy et al. 2021). We initially hypothesized that deforestation would increase the impacts of storm flows in streams, and negatively affect B-IBI scores. However, the data did not support that hypothesis. We found instead that in these forested basins flashier streams were correlated with more stable forests, with both having steep slopes as a common cause. When slope was considered in the model, riparian forest conditions no longer had a significant effect on flashiness. Yet even when considering the links between slope, riparian stability, and flashiness, the model results indicated a small additional positive effect of increased stream flashiness on B-IBI scores. Though not consistently significant in the two datasets, this finding is deeply counter to our expectations and challenged us to consider possible explanations.

To put the degree of flashiness observed in these streams in perspective, we calculated flashiness metrics for 9 urban streams over multiple years in King County using gauge data from the King County Hydrological Information Center (HIC) (Table 3). We found that TQM and RBI values were comparable between the forested and urban basins (though indicating slightly flashier conditions in urban streams), but the max to mean flow ratio was much higher in the forested basins (Figure 15 A-C). These results indicate that our study streams were as flashy or flashier compared to urban streams, which is also unexpected. However, many of our basins are low-order headwater streams, which may naturally experience flashier conditions (Cuffney & Wallace 1989). In contrast, the urban stream gages are typically in higher order, lower gradient streams where flashiness arises because of impervious surfaces and piped stormwater conveyances. The results suggest that in forested basins, flashiness may not in itself be detrimental, and may in fact contribute to biotic integrity by flushing fine sediment and encouraging diversity through moderate disturbance (Connell 1978).

Table 6 Urban stream gauges and the number of water years between 2000 and 2020 included in the comparison with forested streams.

Site Name	Site Code	# Years
Des Moines Creek below SR 509, Des Moines (near mouth)	11d	20
Juanita Creek at Mouth	27a	16
McAleer @ Mouth	35c	20
May Creek @ Mouth	37a	20
Miller Creek near Mouth	42a	16
Soos Creek at Kent-Black Diamond RD	54J	10
Little Bear @ NE 195th	30AN	7
Longfellow Creek (26TH Ave SW & SW Oregon Street) Seattle Public Utilities site	SPU_STA098A	2
Carkeek Park (Pipers Creek) Seattle Public Utilities site (STA508)	68A	2

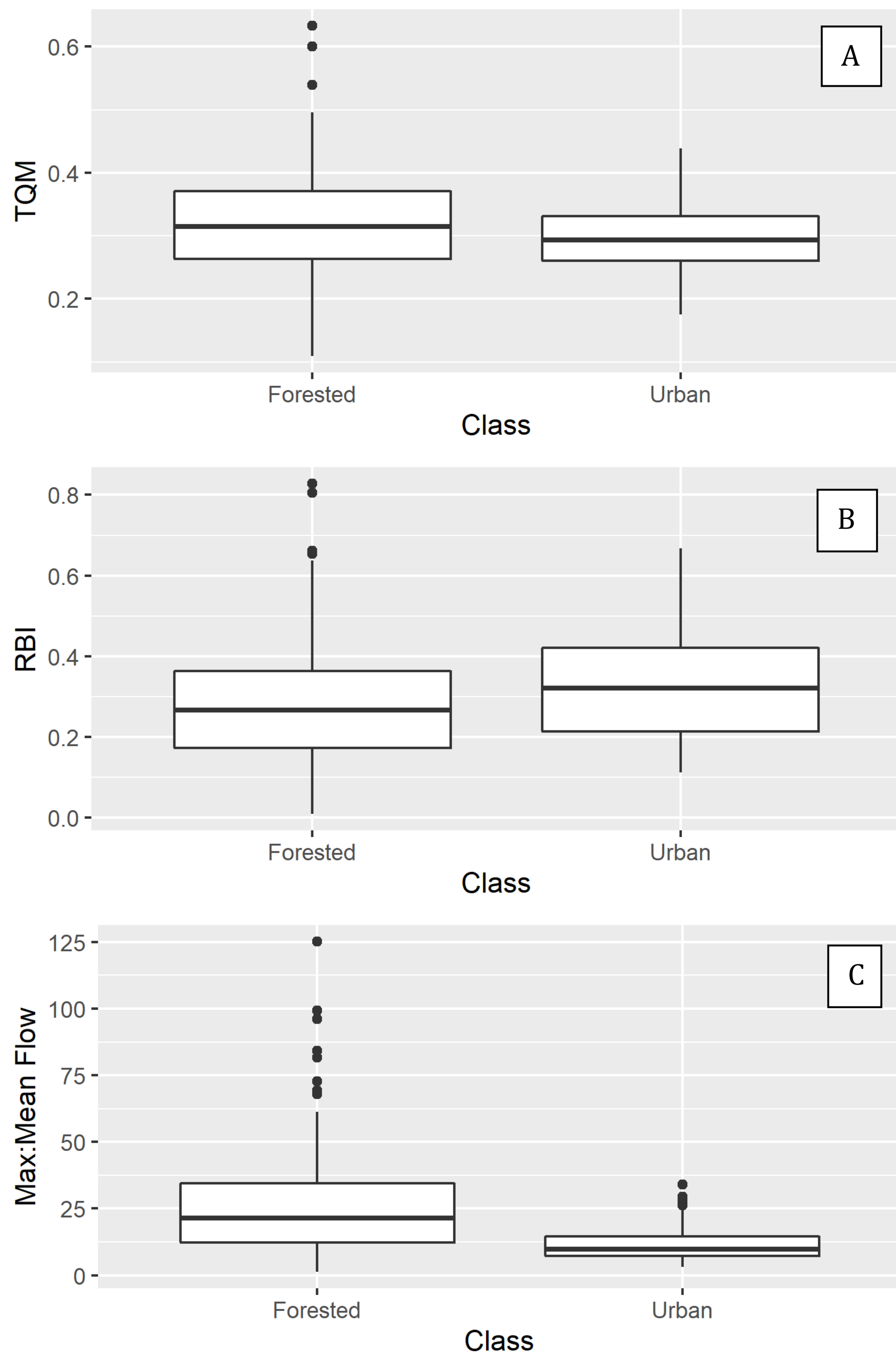

Figure 18. A comparison of stream flow flashiness indicators in forested and urban basins. Panel A compares TQ-mean values, Panel B compares Richard Baker Index values, and Panel C compares the max to mean flow ratio values.

Community Trophic Structure

The results of the model showed an interesting and unexpected relationship between forest conditions and the three functional feeding groups that have been the most responsive to forest harvests in previous studies: Collector-gatherers, scrapers, and shredders. It has been commonly hypothesized, based on the River Continuum Concept (Vannote et al. 1980), that these groups would vary according to the energy base of the food web, with collector-gatherers and scrapers reflecting a more photosynthetic system, and shredders reflecting a more detrital based system (Haggerty et al. 2002.). This spectrum is theorized to be controlled by canopy cover over the stream, providing leaf litter when present and sunlight when absent. However, our model results strongly imply additional dimensions that controls the dynamics of these three groups: sediment and spatial extent (upstream vs reach).

Scrapers and collector-gatherers do appear to favor more light, though under very different circumstances. Collector-gatherers were relatively more abundant when there was more light in the sampling reach due to disturbance. They also appeared to favor conditions with more sediment. These findings align with those of Kiffney et al. (2003). In that study, the authors found that high light environments resulting from forest harvest promoted filamentous algal growth, which increased the productivity of primary consumers, and also appeared to entrain sediment. An alternative explanation could be that the increased sediments entering the stream from harvest may also deliver organic nutrients (Gravelle et al. 2009), promoting algal growth; however, previous studies have found that nutrients are not usually a limiting factor for algal growth in similar streams (Kiffney et al. 2003, Lowe et al. 1986). Scrapers, however, seemed to prefer more stable forest conditions in the whole upstream area but also relatively lower basal area, which may reflect a preference for less dense canopies and increased canopy gaps, which can occur under old growth conditions (Kaylor et al. 2017, Sedell & Swanson 1984). In addition, scrapers are strongly negatively affected by fine sediment, which may reduce the formation of algal biofilms by blocking light to the substrate. Shredders, on the other hand, seemed to greatly prefer shadier upstream conditions and are not negatively affected by upstream forest harvests, both of which may contribute leaf litter and other organic material. Furthermore, they also seemed to prefer conditions with more sediment. We speculate this link is the result of leaf litter and other organic debris that may enter the stream at the same time as sediments, possibly from harvest residues and natural disturbances. We also saw an apparent correlation between the Richard-Baker Index (which is a measure of flow oscillations) and shredders, which we speculate relates to the influx of leaf litter and detritus from storm runoff. Cuffney and Wallace (1989) also found that the intensity and seasonal timing of storms strongly influenced the bulk export of coarse particulate organic matter down headwater streams.

Altogether, we begin to see that in the dynamics between forest harvest and recovery, there may be hysteresis in the benthic community caused by sediment and light availability. Light penetration has a non-linear relationship with forest age (Sonohat et al. 2004, Kaylor et al. 2017), and as observed in Figure 13, sediment can take decades to work through a system. Though all three functional feeding groups will be present in a well-functioning stream ecosystem, figure 16 conceptually represents how the community dynamics may shift as a stream basin is harvested and subsequently recovers. This is a generalized schematic, and the direction of the arrows could change depending on where harvest occurs relative to the sampling site. For instance, forest harvest upstream of the sampling site may favor shredders over collector-gatherers by increasing sediment and detritus without increasing light availability in the reach.

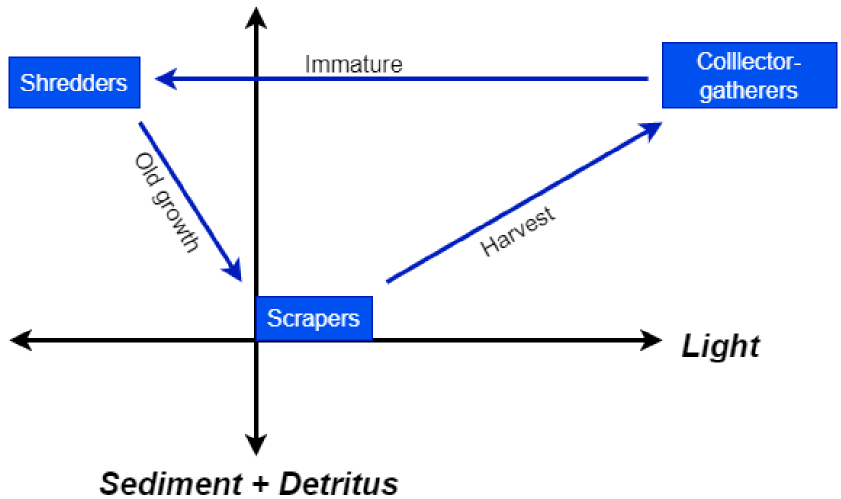

Figure 19. Conceptual diagram of how the proportions of functional feeding groups may shift depending on light, detritus and fine sediments.

Remarkably, all three functional feeding groups had a negative relationship with B-IBI, though the link with scrapers was not significant. In simple bivariate correlations, the proportion of scrapers in a sample appeared to have a positive relationship with B-IBI (Figure 6). However, because collector-gatherers and scrapers were negatively correlated with one another, when collector-gatherers were added to the model, we saw the relationship with B-IBI became negative. In other words, without considering the dependent nature of scrapers and collect-gatherers in their effects on B-IBI, the actual negative relationship between scrapers and B-IBI was suppressed. Though the relationship is non-significant, it is still surprising that scrapers would have had any negative relationship with B-IBI given their apparent sensitivity to sediment and preference for stable forests. It may reflect a lack of diversity in samples that are dominated by any one particular functional feeding group.

Collector-gatherers had the strongest effect on B-IBI scores, yet our model did a poor job explaining their variance in the sample data (r^2 = 0.02-.088). This means that our model did a poor job predicting the conditions in which collector-gatherers are abundant, but this functional feeding group appeared to have the largest effect on B-IBI scores. Collector-gatherers are generalists that can survive in wide range of conditions (Haggerty et al. 2002) and are often colonizers that can quickly establish in disturbed habitats. Furthermore, our model results suggested that conditions in the immediate stream reach are the most influential to their populations. It may be that other natural and smaller scale disturbances may be much more important determinants of collector-gatherers relative abundance than forest harvest. Alternatively, it may be that competition with other benthic macroinvertebrates may play an important role. Were we to assign a causal effect of scrapers and shredders on collector-gatherers, much more variance in collector-gatherers would be explained ($r^2 = 0.38$). Though resource competition among the various FFG groups seems plausible, the nature of such competition remains unknown to us.

Fine Sediment

Fine sediment seems to be the main stressor impacting B-IBI scores in forested basins, by impacting the access and availability of food resources and habitat to different macroinvertebrate functional feeding groups. Because our sediment values were not obtained from direct observation of the stream sediments, we used three independently constructed methods of predicting sediment impacts (based on the benthic macroinvertebrate community) as indicators of true fine sediment impacts. Though the predictive ability of each indicator alone is not particularly strong (literature reported $r^2 = 0.49$ for BSTI, 0.42 for bio,infer, not reported for FSBI), using them in conjunction in a latent factor model harnesses the shared variance among the three, which allows us to separate the sediment signal from other sources of error in each variable.

Fine sediment alone had a far larger effect on B-IBI scores than all direct and indirect pathways from forest stability (including pathways that include sediment as a causal response to forest stability). For every unit change in the latent factor fine sediment (which corresponds to a 171% increase in BSTI value), B-IBI scores were estimated to decline by 7.8 (in the primary dataset) to 11 points (in the secondary dataset). Though natural sources of fine sediment (i.e. landslides, bank erosion) may contribute to fine sediment impacts in these systems, this finding raises the possibility that legacy impacts from previous less-regulated forest harvests may play a role in current B-IBI scores, even when the forest conditions have generally recovered.

To explore this idea further, we must look beyond the model. One aspect of fine sediment not explicitly included in the model was attenuation: both spatially and temporally. However, both appeared to influence the fine sediments predicted in the sampling site. Of note is the persistence of fine sediment impacts over time. Fine sediment values appeared to remain elevated in basins for decades after harvest, and only began to stabilize at lower levels in basins where the median forest age was over 75 years (we used median forest age as a proxy for the median time since last harvest basin-wide) (Figure 13). This may simply reflect that these basins have had longer rotation times and smaller harvest units relative to the whole basin, resulting in fewer fine sediment inputs. But even in basins that were very minimally harvested since 1986 (<5% of the basin harvested), fine sediment levels appeared to take decades to decline (Figure 14). This appears to support the idea that in forested basins, some variation in B-IBI scores may reflect legacy impacts of forest harvest through residual fine sediment rather than current forestry practices associated with timber harvest.

Attenuation of fine sediment over stream distance is also notable. We measured this aspect of attenuation as the area-weighted median flow length between where fine sediments from a harvest would enter the stream and the downstream sampling site. As we discussed in our results, we did not include this variable in our model as it was too correlated with basin area, and we wished to keep the model as simple as possible. However, simple biplots of fine sediment values and distance to disturbance showed an intriguing pattern, in which sampling sites appear to be generally less impacted by harvests that occur more than 3.5 kilometers upstream (Figure 13), though as basin size increases the relationship between sediment and stream-length changed due to other processes.

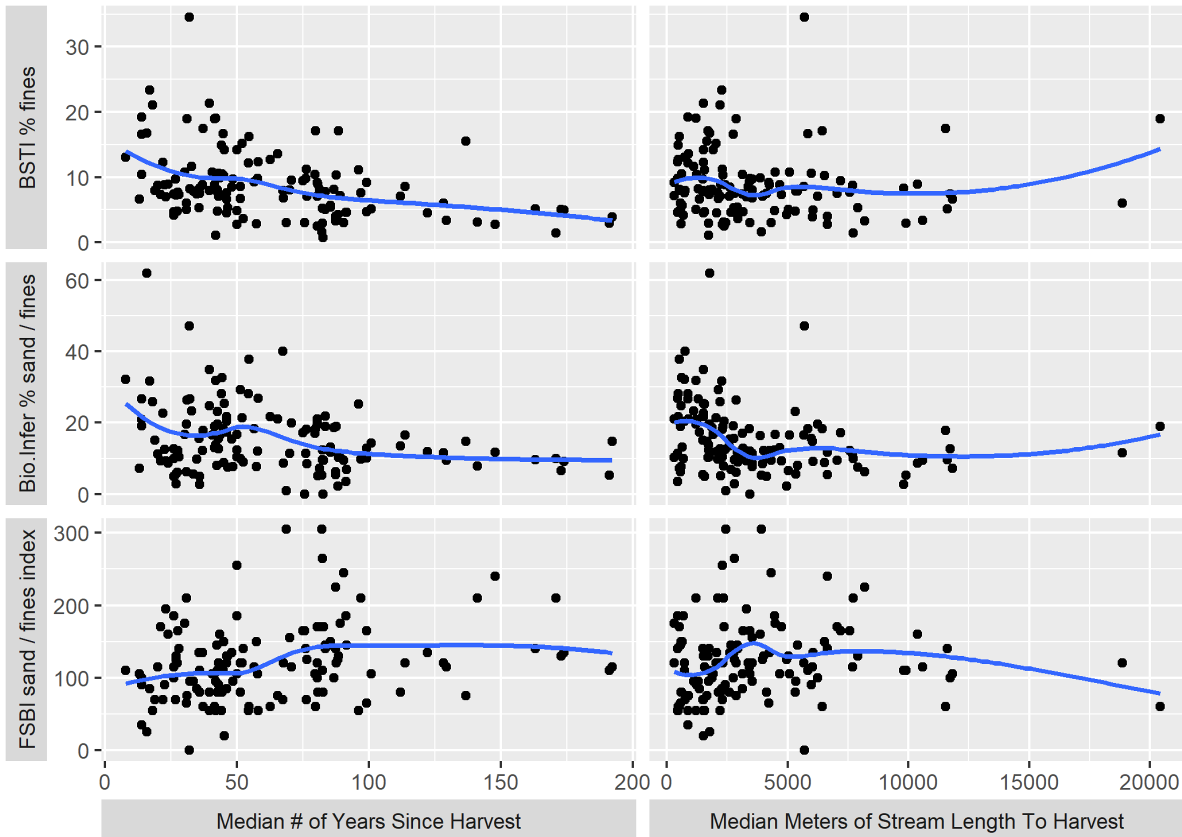

Figure 20. Fine sediment indicators and attenuation over time (left column) and stream length (right column). The BSTI and bio.infer indicators provide direct estimates of fine sediment in the stream substrate. The FSBI is an indicator that increases with the presence of sediment-sensitive taxa; higher values indicate less fine sediment.

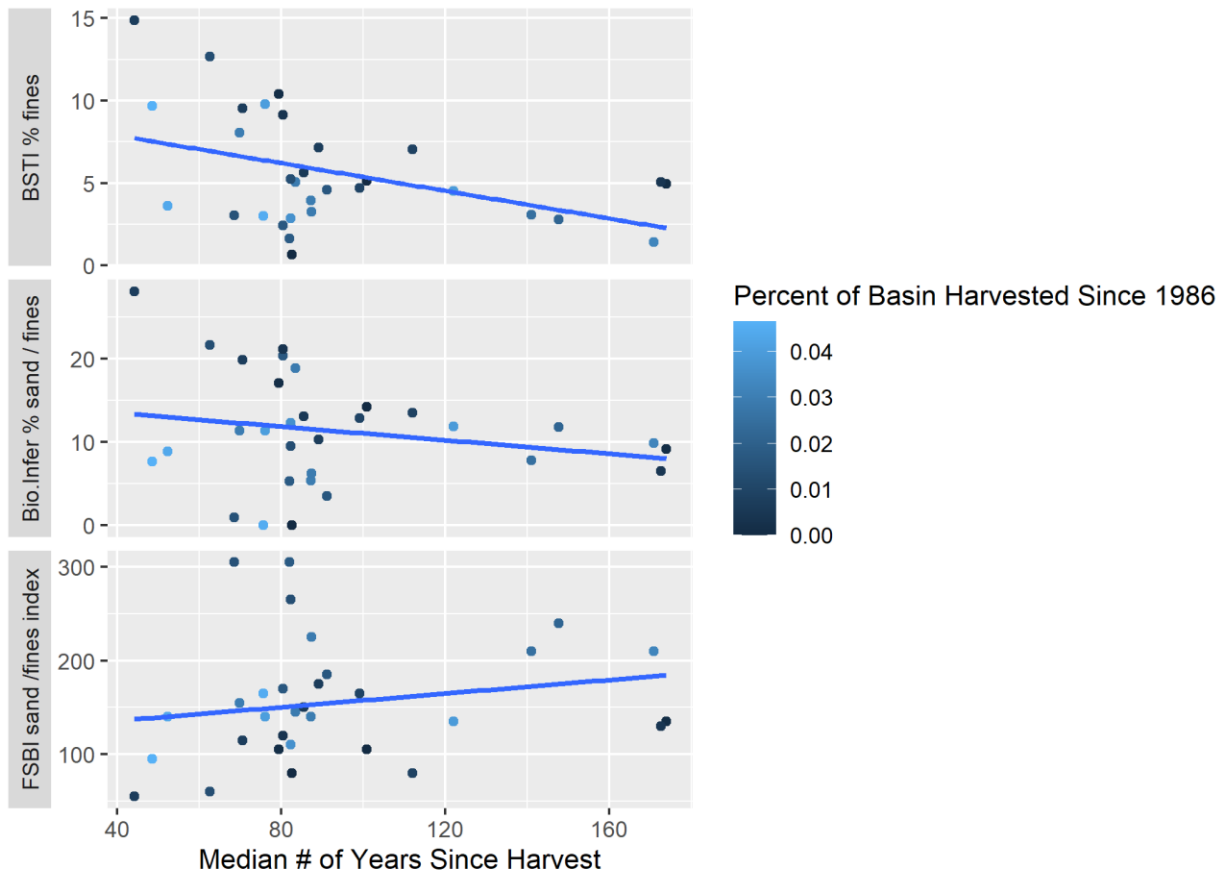

Figure 21. Fine sediment indicators and attenuation over time in basins with minimal recent harvest (< 5% in the 16 to 29 years prior to sampling). The BSTI and bio.infer indicators provide direct estimates of fine sediment in the stream substrate. The FSBI is an indicator that increases with the presence of sediment-sensitive taxa; higher values indicate less fine sediment.

Implications for certification programs.

Forest certification is a voluntary process to demonstrate responsible and sustainable forestry practices. In Washington State, the Sustainable Forestry Initiative and American Tree Farm System (both endorsed by the international Programme for Endorsement of Forest Certification systems [PEFC]) as well as the Forest Stewardship Council offer certification programs to maintain standards that meet social, economic, and environmental objectives. The Washington Department of Natural Resources maintains Sustainable Forestry Initiative certification across all lands (2.4 million acres) under its management. Within the South Puget Habitat Conservation Plan Planning Unit (within King, Pierce, Thurston, Lewis, Kitsap, and Mason counties), roughly 176,000 acres are also certified by the Forest Stewardship Council US Forest Management Standard (https://www.dnr.wa.gov/programs-and-services/product-sales-and-leasing/timber-sales/forest-certification).

In the context of our model results, we can assess the three standards for how well they may protect stream benthic macroinvertebrate biotic integrity. The least prescriptive of the three standards is the American Tree Farm System, which do not specifically address riparian areas. The Sustainable Forestry Initiative Forest Management Standards do call for implementation of riparian protection programs based on several factors including climate, soil type, terrain, vegetation, and ecological function; however, the standards do not provide specific guidance on buffer widths. The Forest Stewardship Council is the most prescriptive of the three by providing specific riparian guidance. In perennial fish-bearing streams in the Pacific Coast region, this standard calls for a minimum no-harvest buffer width of 50 ft, with an additional 150 ft of buffer in which harvest is allowed but limited. In perennial non-fish bearing streams, the standard calls for a 25 ft no-harvest buffer, with an additional 75 ft of buffer in which harvest is allowed but limited. In seasonal streams that provide habitat to non-fish aquatic organisms, harvests within 75 ft of the stream are allowed but limited.

Based on the model results, the Forest Stewardship Council Forest Management Standard is the most protective of in-stream benthic macroinvertebrate biotic integrity and provides better protection than current state guidelines. The narrow buffers in non-fish bearing streams may still not be sufficient for the protection of benthic macroinvertebrates. Our conclusions are consistent with those of Jyväsjärvi et al. (2020), who concluded that Forest Stewardship Council standards are reasonably protective of in-stream biotic integrity while PEFC-complaint standards are not.

Lessons Learned

The outcome of this study emphasizes the importance of sample size. Because of the different sample sizes in the primary and secondary datasets, the analytical power to validate our model was greatly reduced. As a result, it remains unclear whether we were unable to validate our model because it was incorrect or because we had insufficient power to reject the null hypothesis. A lesson gleaned from this is the importance of maintaining similar and sufficient sample size populations in the primary and secondary datasets. This could have been accomplished if we had retained basins that contained other selected basins (“nested” basins) by employing a spatial autocorrelation component in the model. While such a specification is possible, for simplicity we had opted against this in planning our study. However, in retrospect it could have substantially increased our sample size, as well as captured additional variation within basins.

Obtaining full sets of data at all spatial scales is another lesson learned. During our data gathering process, we obtained data only at the spatial scales at which we expected them to be important. However, as we later revised our expectations based on our review of the correlation tables and initial model results, several pieces of data would have been useful to have at all spatial scales.