

Benthic Index of Biotic Integrity Implementation Strategy State of Knowledge Report

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ACKNOWLEDGEMENTS:

Primary Author:

C. Andrew James, UW Puget Sound Institute

Contributing Authors:

Kate Macneale, King County

Tanya Roberts, UW Puget Sound Institute

Christopher Wright, UW Puget Sound Institute

Contributing authors to the B-IBI Starter Package, which served as a basis for the State of Knowledge report and the Base Program Analysis report.

Derek Day, Washington State Department of Ecology

Jennifer Elliot, UW Puget Sound Institute

Aimee Kinney, UW Puget Sound Institute

Elene Trujillo, Puget Sound Partnership

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EXECUTIVE SUMMARY

This State of Knowledge report is meant to provide a scientific and technical foundation to the Benthic Index of Biotic Integrity (B-IBI) Implementation Strategy. The B-IBI Implementation Strategy has two main objectives: 1) protect streams with “excellent” biological condition (i.e., with high B-IBI scores), and 2) restore streams with moderate biological condition (i.e., with “fair” B-IBI scores).

From a technical perspective, achieving the first goal is technically straightforward because the best way to avoid degrading high quality streams is to limit the conversion of forest lands and protect the watershed from development. Mitigating measures such as ensuring a sufficient forested riparian buffer and the use of Low Impact Development techniques have been proposed, though it is not known whether these will be protective on a watershed scale.

Restoring streams with “fair” B-IBI scores is more problematic because it is not currently known how to improve B-IBI scores. A reasonable approach is to protect existing forest lands, and reduce the impact and magnitude of pressures (e.g., development) and stressors (e.g., altered habitat, hydrology, and water quality) in order to restore functionality, but there are currently few examples where measured and demonstrable improvements in B-IBI scores have been achieved. Complicating factors include:

- Local biological condition is dependent on watershed condition, and watershed condition can limit the possible extent of recovery. Few restoration projects act on a watershed scale.
- Stream biological condition, and specifically the condition of the benthic invertebrate communities, appears to have a high degree of biological inertia. Recovery times may be many years to decades.
- Applicable monitoring data are sparse. There are few examples of monitoring programs that can adequately isolate and track the effectiveness of restoration actions on biological conditions.

Despite the high degree of uncertainty surrounding B-IBI recovery, reducing key pressures such as development or land conversion, even at a reach scale¹ will likely improve the conditions of the stream system such as lower stream temperatures, improved fish habitat, nutrient attenuation, even if B-IBI scores are not affected. The multiple benefits of stream restoration including improved aesthetics and recreational opportunities are also important considerations.

¹ A reach is a section of a stream or river with similar hydrologic conditions. Likely on the order of 10 to 100 m

1 INTRODUCTION

This State of Knowledge report is meant to provide a scientific and technical foundation to the Benthic Index of Biotic Integrity (B-IBI) Implementation Strategy. The B-IBI Implementation Strategy was developed based on a conceptual model where human related pressures (e.g., land use development and land use activities) results in stressors (e.g., changes in hydrology, habitat, and water quality) that impact stream condition. Therefore, improvements in stream health can be achieved by mitigating the effects of the stressors; strategies for doing so are outlined in the Implementation Strategies.

The purpose of this document is to provide:

- detail and definition of the pressures and stressors affecting stream condition,
- an overview of how the pressures and stressors affect stream benthic communities and the B-IBI indicator, and
- important considerations, such as effectiveness of approaches and actions, that might be important in the refinement and implementation of the recovery strategies proposed in the Implementation Strategy document.

Information in this document is largely based on a review of the salient literature, much of which is presented in a series of summary tables. For example, there are a wealth of papers describing how interrelated functions such as watershed hydrology, in-stream hydraulics and geomorphology, water quality, and biology together define stream condition, and, further, how the development and land use changes affect these functions. There are many studies looking at the effectiveness of restoration and stormwater mitigation on maintaining “natural” functions. However, critical uncertainties remain. Some are highlighted here. A discussion on the complete evaluation and prioritization of the uncertainties is provided in the Implementation Strategy Narrative document.

An evaluation of existing regulatory and other programs that address issues related to stream health is provided in the accompanying Base Program Analysis report.

This State of Knowledge report, the Base Program Analysis, and the B-IBI Implementation Strategy are companion documents, providing technical, programmatic, and strategic information related to the recovery of streams in the Puget Sound watershed.

2 BACKGROUND

This section presents a brief summary of the B-IBI indicator and indicator target, status and trends, and data sources. Much of this information is included in the Implementation Strategy and will not be repeated here.

2.1 B-IBI Vital Sign Indicator

An index of biotic integrity (also referred to as an index of biological integrity) is a quantitative multimetric index (MMI) that is based on a suite of specific measures of species abundance (how many of a particular organism) and diversity (how many different types of that organism). It is meant to relate the occurrence of aquatic organisms (e.g., fish, algae, macroinvertebrates, etc.) with the condition of a water body (Karr 1981). The benthic index of biotic integrity focuses on insects that spend at least part of their life cycle in the stream bed (benthic macroinvertebrates).

These indices are developed by comparing measures of invertebrate abundance or diversity (i.e., metrics) against measures of watershed condition (e.g., percent development, percent imperviousness, etc.) with the responses evaluated to understand the range, reproducibility, calibration for natural gradients, responsiveness to stressor gradients, and independence from other metrics (Stoddard et al. 2008).

The Puget Sound Lowland Benthic Index of Biotic Integrity (B-IBI) was developed in the 1990s as an integrative measure of the biological health of wadeable streams (e.g., streams that can be sampled without the use of a boat) in the Puget Sound lowlands (Kleindl 1995). It is composed of ten metrics: total taxa richness, Ephemeroptera (mayfly) taxa richness, Plecoptera (stonefly) taxa richness, Trichoptera (caddisfly) taxa richness, clinger taxa richness, long-lived richness, percent tolerant, percent predator, and percent dominant. A score from 0 – 10 is generated for each metric based on the measured value in a given stream. The overall B-IBI score (ranging from 0-100) is determined by summing the individual scores for each of the component metrics.

The lists of taxa included in several of the metrics were reviewed and updated either based on the literature or regional data (Fore et al. 2013) and the B-IBI scoring system was revisited and recalibrated in 2014, largely to take advantage of a growing regional data set (King County 2014c). Each of the component metrics varied predictably, though weakly, with percent watershed urbanization (R^2 for individual metrics ranged from 0.15 – 0.42). The overall B-IBI scores were correlated more strongly with percent watershed urbanization (Spearman's $\rho = -0.69$) compared to the metrics, consistent with observed MMI properties (Schoolmaster Jr et al. 2012). Overall, the index has been shown to vary consistently with several measures of urbanization (Booth et al. 2004, May et al. 1997, Morley and Karr 2002).

A complete description of the indicator is presented in Section 2 of the B-IBI Implementation Strategy and the [Puget Sound Stream Benthos](#) page.

It is important to note that B-IBI and the metrics upon which it is based were developed based on comparisons with broad measures of human disturbance such as percent imperviousness or percent urbanization. Although some work has been done identifying the causal, mechanistic links between stream condition and B-IBI (e.g., see Booth et al. (2004) and DeGasperi et al. (2009), and section 6.2). B-IBI is not necessarily designed to be diagnostic of individual pressures or stressors that may be affecting a stream system.

2.2 B-IBI Vital Sign Indicator - Targets

The Puget Sound Partnership established recovery targets related to the B-IBI indicator to protect high quality streams and restore a selected number of streams with a potential for recovery (Wulkan 2011). The targets are:

- Protect: 100 percent of Puget Sound lowland stream drainage areas ranked as “excellent” retain “excellent” scores for the Benthic Index of Biotic Integrity for biological condition.
- Restore: Improve and restore at least 30 streams ranked in “fair” biological condition so scores improve to “good.”

This Vital Sign indicator focuses on sites that are ranked as either “fair” or “excellent;” designating a site in either category depends on a set of criteria to account for variability, sampling frequency, and changes in watershed condition. Sites categorized as “fair” during at least three sample events, had a median score of “fair,” and were in the Puget Sound lowlands ecoregion. Sites categorized as “excellent” were ranked as “excellent” at least once, but were excluded if they had ever been ranked “poor” or “very poor,” or had a median score of “fair” (King County 2015b).

2.3 Regional B-IBI Data

Regional B-IBI data, collected by over twenty agencies and groups, are stored and made publicly available through the [Puget Sound Stream Benthos](#) (PSSB) web site. Updates in field collection methods, standard taxa lists (e.g., the Pacific Northwest Aquatic Monitoring Partnership Northwest Standard Taxonomic Effort), and calibration updates (e.g., King County (2014c), Fore et al. (2013), etc.) are all included on the web site.

As of September 2019, there were approximately 7,980 B-IBI scores in the PSSB database, collected from 1,490 sites across Puget Sound (Water Resource Inventory Area (WRIA) 1-19).

2.4 B-IBI Status and Trends

An overview of B-IBI status and trends is included in the Implementation Strategy.

2.5 B-IBI indicator considerations – limitations and uncertainties

Stoddard et al. (2008) suggested that individual metrics of MMIs include: 1) sufficient variability in data values among sites (data range), 2) reproducibility (temporal stability), 3) responsiveness to stressor gradients, and 4) independence from other metrics.

King County (2014c) demonstrated the range and responsiveness of the B-IBI metrics over a range of watershed urbanization. And while the individual metrics of B-IBI focus on different biological orders (such as mayfly, stonefly, and caddisfly), or macroinvertebrates with different functional or life history characteristics, the metrics are not completely independent. For example, Ephemeroptera, Plecoptera and Trichoptera taxa richness are included in the measure of total taxa richness. The correlation between the metrics is generally high.

With regard to variation, B-IBI varies spatially and temporally due to factors other than changes in catchment pressures or stressors. Heino et al. (2004) reported considerable variation at multiple spatial scales within a stream system including variation between sample location along a transect across the same riffle² (meter scale), between different riffles within the same reach (10s-of-meters scale), between different reaches on the same tributary (100s-of-meters scale), and between different tributaries in a watershed.

Others have considered variation across microhabitats, habitats, stream reaches, stream segments and catchments to describe heterogeneity in biotic condition. Ligeiro et al. (2010) investigated the variation in family richness across different scales and reported that, for the ecosystem investigated, the high proportion of the diversity could be characterized across microhabitats (with different substratum types) and stream segments. Difference between reaches and in riffles were less pronounced. These results highlight the potential limitations of relying on a single dataset for characterizing B-IBI in a given reach, as invertebrate assemblages may vary widely.

With regard to temporal variability, Mazor et al. (2009) reported the interannual coefficient of variation (CV) for B-IBI to be 22-26% meaning that B-IBI scores taken at the same location will vary somewhat from year to year. Mazor et al. (2009) indicated that at least some variation can be associated with climatic changes, particular over longer sampling periods. An analysis of variation of Puget Sound data indicated that the minimum detectable difference would be approximately 14 (on a 0-100 scale), meaning that B-IBI scores would have to be at least 14 points different to note a distinction between the condition of any two sites; this represents the same order of variation as reported elsewhere (King County 2014a). Variability, including natural variation and measurement error, should be considered in the design of monitoring programs, and the use and interpretation of resulting data including the analysis of long term trends or changes associated with a given restoration action (Brooks et al. 2002).

² A riffle is a shallow section of a stream where the water passes over stones and rocks, creating turbulence or small disturbances in the water.

3 STRATEGIES FOR IMPROVING THE VITAL SIGN

The B-IBI Implementation Strategy identifies strategies, approaches, and actions based on the recommendations of the Interdisciplinary Team. They are described in detail in Section 4 of the Implementation Strategy; brief descriptions are included here.

Watershed Planning Strategy: Promote multi program and cross-jurisdictional planning for water resource protection and restoration coordinated on a watershed scale.

Local Capacity Strategy: Improve funding, staff capacity, and availability of decision support tools for local stormwater management programs.

Education and Incentives Strategy: Accelerate stormwater retrofits and habitat restoration efforts with education and incentives.

Working Lands Strategy: Reduce impacts of working lands (i.e., forest lands for timber harvest or agriculture lands) on stream health and the risk of conversion of working lands to more intensive land uses.

4 RECOVERY CONTEXT

This section presents considerations regarding stream condition that are important in restoration and recovery planning.

Key Points (Recovery Context):

- Stream condition can be defined by a suite of interrelated functional components with semi-hierarchical dependencies (Figure 1). The functions are interrelated in that the degradation of one functions will probably lead to the degradation of others. The functional components are semi-hierarchical in that the higher tier functions are generally (but not strictly) dependent on lower tier functions. The key point is that biological condition is dependent on most or all other functional components.
- Watershed condition largely defines the hydrology, the lowest-tier functional component.
- Surface water runoff (stormwater) provides a mechanistic link between watershed condition and hydrology. Watershed condition (extent forests, impervious areas, households, roads, etc.) controls the rate and extent by which rainwater/stormwater flows into the subsurface and/or stream channels, so changing the watershed condition changes the hydrology.
- Surface water runoff (stormwater) can also affect water quality (the physiochemical function).
- Watershed condition can limit the biological condition of a given stream. This is the observed biological potential (see Section 6.1).
- According to the Puget Sound Watershed Characterization project, many basins in Puget Sound have degraded hydrological processes, potentially limiting their biological potential.

4.1 Stream Functional Components and Relationships

The condition of any given stream is determined by a suite of processes such as the way water runs off the landscape (i.e., hydrology) or the amount of vegetation and wood in a stream channel (i.e., channel geomorphology). These processes are interrelated and interdependent. Human pressures and stressors often affect the processes that define stream condition (see below). Harman et al. (2012) have proposed a framework to describe the main functions that define stream condition, with an indication of general, but not strict dependencies. A functional pyramid was developed (Figure 1) to define the functions and provide an overview of relationships amongst them. In general, the higher level functions are supported by the lower level functions. This highlights a few important aspects of the restoration approach. First, that in-stream biology, as measured by B-IBI, is dependent on several functional components and they must be addressed holistically to achieve successful recovery or restoration (Karr et al. 1986). For example, projects addressing stream geomorphology, such as the addition of wood to the stream channel, may be ineffective for improving stream condition without also

addressing watershed scale hydrologic functions (Alexander and Allan 2007, Bernhardt and Palmer 2011, Fischenich 2006, Larson et al. 2001). Second, watershed scale processes underlie stream condition and so a watershed-scale approach is necessary (Bernhardt and Palmer 2011). This is reflected in the Watershed Planning strategy, which promotes a holistic coordinated approach, as described in the Implementation Strategy narrative.

A key mechanistic link between watershed condition and stream function is stormwater, defined as surface water runoff from built surfaces. Changes in the watershed influence the manner by which rainwater runs off surfaces (i.e., altered hydrology) and into streams. Stormwater also alters the flow dynamics (i.e., altered hydraulics), the transport and deposition of sediment (i.e., altered geomorphological functions), and water quality (i.e., altered physiochemical functions).

Note that the framework proposed by Harman et al. (2012) in Figure 1 is not the only representation of the relationships between human activities and stream condition. For example, Karr et al. (1986) identified five classes of environmental factors related to stream condition (energy source, water quality, habitat quality, flow regime, and biotic interactions) and described how they might be affected by human activities (Figure 2). Both Figure 1 and Figure 2 are included here to reinforce the notion that: 1) there is a suite of interrelated factors that affect biological condition, 2) there might be slightly different organizational schemes, 3) human-associated activities can impact some or all of them, and 4) recovery strategies need to consider multiple factors to improve condition.

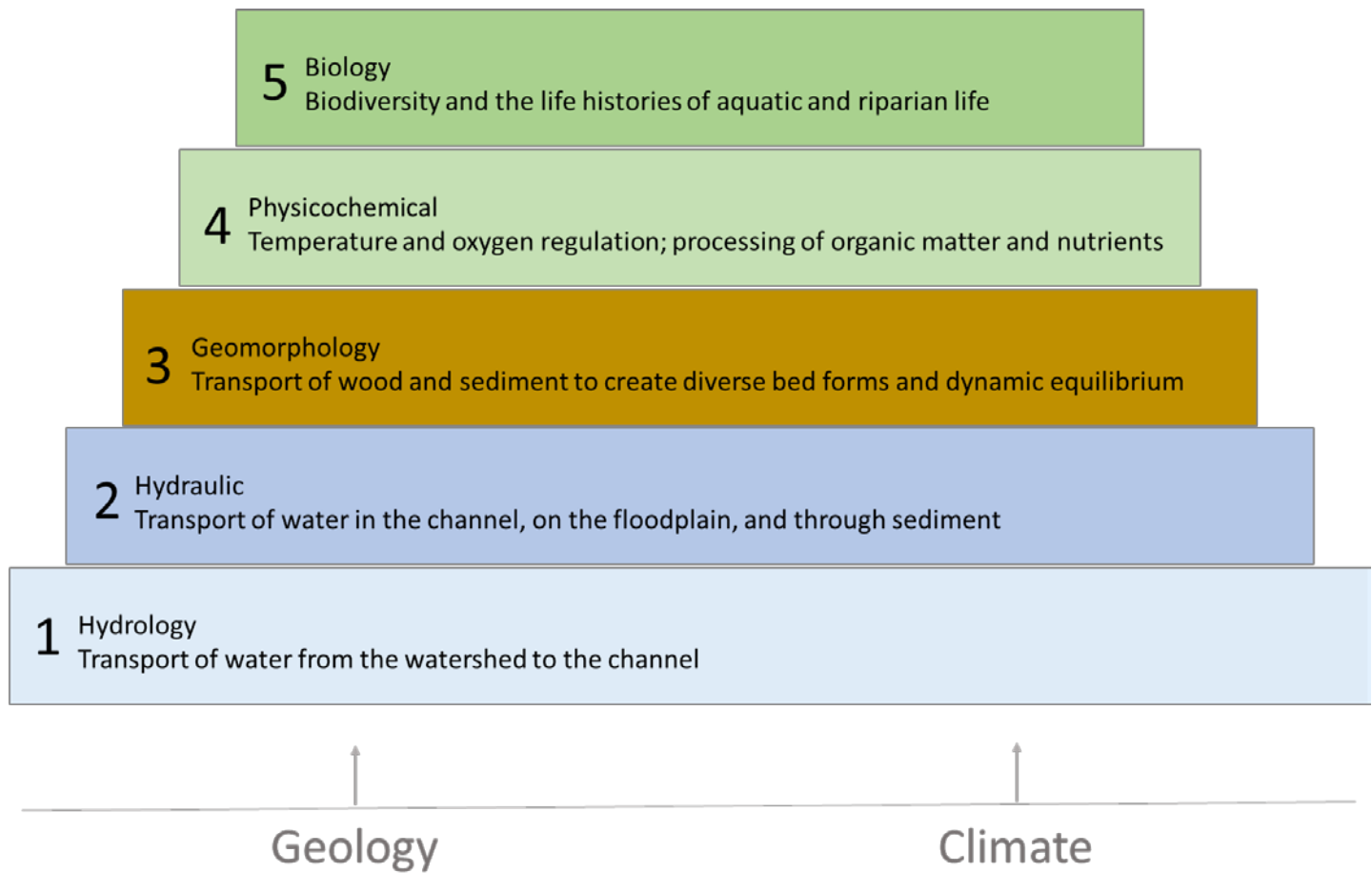


Figure 1. Functional pyramid of stream restoration.

Within this framework, the higher level functions are supported and defined by the lower level functions. For example, the hydrology (runoff from a landscape) supports and defines the in-stream hydraulics. Examples of parameters that describe each of the broad functions are also listed. Adapted from Harman et al. (2012).

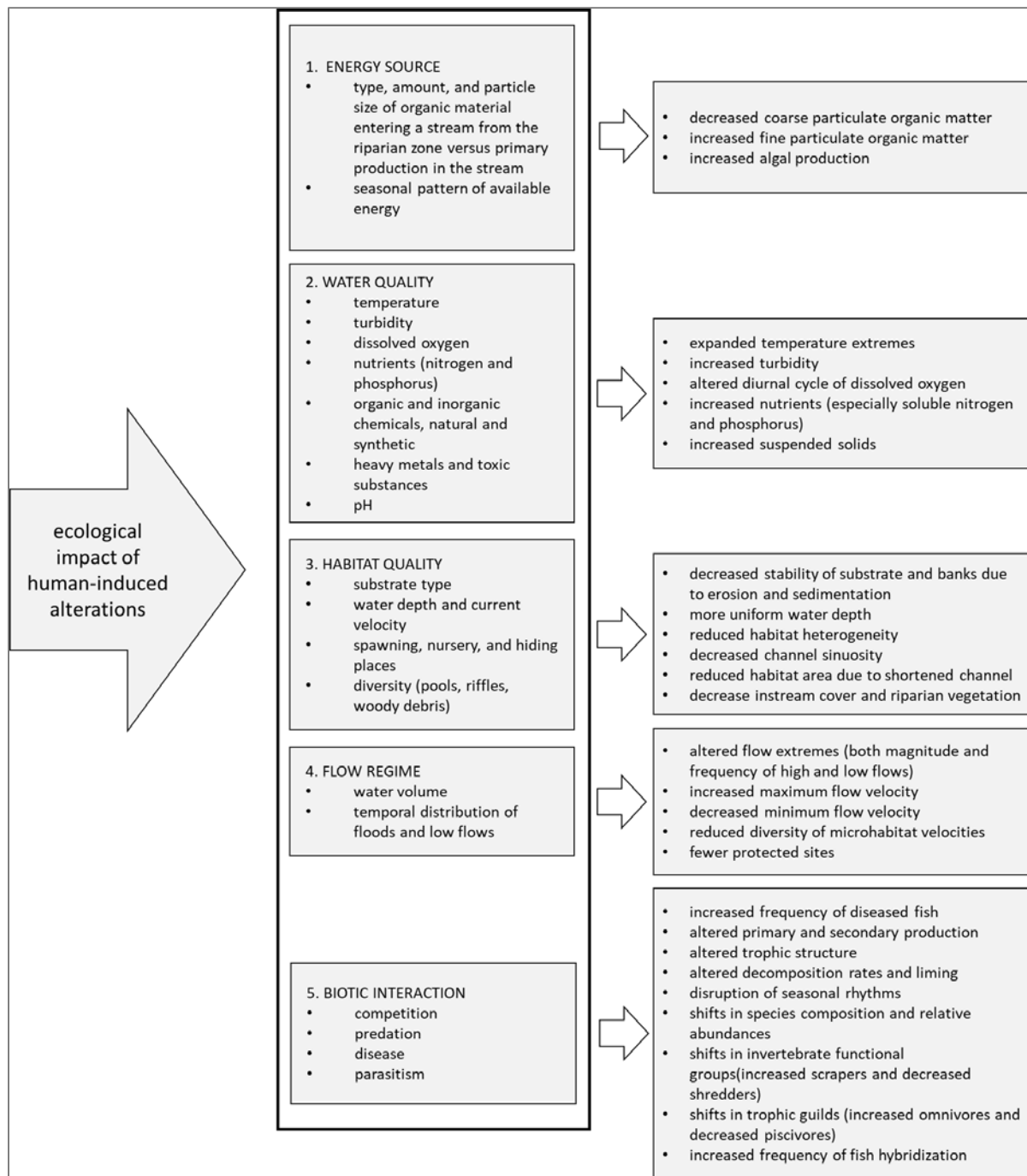


Figure 2. Framework showing relationship between biological condition and environmental factors. The five major classes of environmental factors that affect aquatic biota are shown in middle box. The potential effects from human activities related to those factors are shown to the right (adapted from Karr et al., 1986).

5 PRESSURES AND STRESSORS AFFECTING B-IBI

This section presents a discussion of pressures and stressors affecting B-IBI.

Key Points (Pressures and Stressors):

- Pressures are human actions that lead to degradation or alteration of an ecosystem element. A key pressure affecting stream health is land use conversion, either towards urbanization or working lands.
- Stressors result from the pressures and manifest as changes in the ecosystem. Key stressors are altered hydrology, altered habitat, or altered water quality.
- Strong relationships between pressures and stressors and B-IBI have been identified. However, due to their interrelated nature, e.g., development pressure affects hydrology, that in turn impacts hydraulics, habitat, and the water physiochemical condition (Figure 1), it is often difficult to identify the primary causal stressor.

5.1 Pressure and Stressor – Overview

The recovery framework identifies factors that affect stream health and biological communities in terms of pressures and stressors. Pressures are human actions that lead to degradation or alteration of an ecosystem element. Stressors result from the pressures and manifest as changes in the ecosystem. For example, development (the pressure) might lead to altered hydrology in a stream (the stressor) that impacts stream biota. In this case, the human activity (pressure) does not directly affect biota, but rather it affects runoff patterns (stressor) which in turn impacts stream biology.

It is, again, important to realize that there are different organizational frameworks describing the factors/functions that define stream condition. The description here reflects the descriptions utilized by the Interdisciplinary Team during the development of the Implementation Strategy.

The B-IBI Interdisciplinary Team identified the primary pressures associated with stream condition as development and land use activities. Development describes the modifications to landscape topography, drainage networks, and cover associated with human activity. The development trajectories generally proceed from low density to high density, e.g., forested → farming → low-density residential → high-density residential → urban/commercial/industrial, with a commensurate increase in stressors. Brief literature reviews addressing pressures and stressors are presented in sections 5.2 and 5.3.

Stressors can be thought of as alterations of a particular factor/function (described in Figure 1 and Figure 2). For example, a change in hydrology or flow regime is a stressor. The Interdisciplinary Team grouped these stressors as altered flows, altered water quality, and altered habitat (see appendices to the Implementation Strategy); these stressors align with alterations to the factors/functions shown in Figure 1 and Figure 2. The stressors can be interrelated, as alterations in one can lead to alterations in others, even without additional

pressures; this is generally illustrated by the functional pyramid (Figure 1). For example, alterations in hydrology can lead to alterations in other functions, such as hydraulics (in-stream flows) or geomorphology. Some stressors might impact higher functions directly. For example, removal of riparian habitat through development may alter physiochemical and biological functions.

Again, the key point of the functional pyramid, and the notion of pressures and stressors in the context of recovery and restoration, is to illustrate the relationships that link stressors (altered functions), and to highlight the potential limitations of addressing only selected stressors without considering the foundational functions. For example, a project that addresses a stream geomorphology without considering the hydraulics or hydrology is unlikely to lead to improved stream condition.

It is also important to acknowledge the different scales by which pressures and stressors associated with functions can affect stream condition. This is important in both in evaluating the impacts on a particular stream but also in developing and implementing appropriate management responses (Karr et al. 1986). Stream condition is dependent on local processes (e.g., the condition of the stream bed) as well as watershed scale processes (e.g., surface water runoff across the entire area), so local-to-watershed scale processes need to be considered in developing management responses (see Section 6.1). A single habitat restoration project will not restore a stream in an otherwise urban watershed.

The results of a brief literature review are presented in the following sections. The review is focused on identifying relationships between pressures and stressors and stream condition, B-IBI, and its individual metrics.

5.2 Pressures

The primary pressures that affect stream condition are development and land use activities including agriculture, forestry, and residential and commercial activities. Land use changes generally refer to conversion of pristine forested lands to either working lands (agriculture or forestry) or the gradient of residential development, such as low density residential, high density residential, urban, commercial and industrial – collectively referred to as urbanization. Each of these results in a suite of stressors related to changes in land form, or associated with activities on the altered landscape.

The following sections provide a brief summary of the relationships between pressures and stream condition. In each section, a summary paragraph is followed by a table of relevant citations.

5.2.1 Urbanization

The population of the counties in the central Puget Sound are expected to increase from approximately 4.2 million to 5.3 million over the next 20 years, providing a continuing pressure for urbanization to accommodate this growth (PSRC 2020).

B-IBI is strongly and negatively correlated with several different measures of urbanization including total impervious area, percent urbanization, number of road crossings, road density, population, etc. Stressors related to urbanization include altered hydrology, habitat degradation, and changes in water quality (Section 5.3). Literature review notes relating biological condition and stressors related to urbanization are presented in Table 1.

Increased urbanization has been associated with a consistent decline in the biological condition in streams. It has been proposed that the extent of urbanization in a contributing watershed can impose limits on the biological condition, described as observed biological potential (Paul et al. 2009) (Section 6.1).

Table 1. Literature review notes – Impacts associated with development

Reference(s)	Key Findings
Booth et al. (2004) Kennen et al. (2010) May et al. (1997) Morley and Karr (2002)	Puget Sound Region - Stream biological conditions as measured by B-IBI declined as impervious area/urban area increased within a catchment.
May et al. (1997) Allan (2004)	Puget Sound Region - Degradation of invertebrate community (B-IBI) happens in the range of 15-25% urban land area or total impervious area.
Morley and Karr (2002)	Puget Sound Region - B-IBI declined as the percentage of urban land cover increased. Most metrics were better predicted by sub-basin rather than local-scale urbanization. B-IBI can discern between large differences (32% v 71%) in local land cover but not small (49% v 54%). Sites with 50% urban had B-IBI ranging from 16 to 40 (on the original 10-50 scale).
Fore et al. (2013)	Puget Sound Region - Different measures of development are negatively correlated with B-IBI, e.g. human population size, road density, road crossings, and % urban (defined land use type). Watershed scale is generally stronger than local scale.
DeGasperi et al. (2009)	Puget Sound Region - Associated increasing urbanization in Puget Sound lowland basins with selected hydrologic metrics, such as increased high pulse (short duration high flows) count and frequency and with decreased B-IBI scores.
King County (2015a)	Puget Sound Region - Conducted physical and biological monitoring between 2010 and 2013 in the Lake Washington/Cedar/Sammamish watershed using a probabilistic survey design. Results corroborated other studies in relationships between urbanization and benthic macroinvertebrate community condition as measured by B-IBI. Urban land cover and population density were the strongest predictors of declining B-IBI scores.
DeGasperi et al. (2018)	Puget Sound Region - Sampled 105 sites in Puget Lowland streams and analyzed for B-IBI, water and sediment chemistry, and habitat conditions. B-IBI was not significantly affected by natural landscape variables. A relative risk/attribution analysis indicated watershed and riparian canopy cover, and watershed percent urbanization were significantly associated with “poor” B-IBI scores.

Reference(s)	Key Findings
Roy et al. (2003)	Stream macroinvertebrate response to catchment urbanization in Etowah River basin, Georgia. Taxon richness was negatively correlated to urban land cover and positively correlated to forest land cover. Reduced water quality was detectable where >15% urban land cover occurred. Urban land cover correlated with a number of variables such as stream bed sediment size (–) and total suspended solids (+) N and P concentrations (+), specific conductance (+) and turbidity (+). Catchment urbanization resulted in less diverse and more tolerant stream macroinvertebrate assemblages via increased sediment transport, reduced stream bed sediment size and increased solutes.
King et al. (2011)	Investigated stream biomonitoring data from Maryland to 1) identify the location and magnitude of greatest change in the frequency and abundance of individual taxa and 2) to assess potential of thresholds in response to percent imperviousness in catchments. There are clear threshold declines of 110 of 238 macroinvertebrate taxa in response to low levels of impervious cover, with approximately 80% of the declining taxa responding between ~0.5% and 2% impervious cover.
Hughes et al. (2014)	Urbanization is associated with multiple stressors and associated ecosystems responses. Examples include altered hydrology (streams become flashier), reduction in riparian habitat, altered channel condition, and decrease in water quality. The combination of stressor affects invertebrate and fish communities.

5.2.2 Agriculture

Increasing agricultural land use is often associated with increased sedimentation, nutrient, and pesticide inputs into nearby waterbodies. Agriculture can also lead to stream channelization and decrease in riparian habitat. Agricultural activities are more common in broad valleys or along the coastal plain, where the low gradient streams would be more susceptible to bed sedimentation compared to high gradient streams in the foothills. Sedimentation can alter stream bed habitats and lead to a decrease in benthic diversity. In western Oregon, impacted benthic communities were associated with agricultural activity in the stream riparian zone (Herlihy et al. 2005). The impacts of sustained agriculture on stream condition may persist, even years after restoration and/or reforestation within the riparian zone.

Changes in benthic community condition associated with nutrient inputs vary according to existing stream condition and community structure. Nutrient-associated low dissolved oxygen events can alter condition; phosphorus additions may lead to altered structure while nitrogen additions may only minimally impact benthic communities.

Selected review notes are included in Table 2.

Table 2. Literature review notes – impacts associated with agriculture

Reference(s)	Key Findings
Allan et al. (1997)	Evaluated River Raisin basin in SE Michigan. Extent of agricultural land at the subcatchment scale was the best single predictor of local stream conditions. Local riparian vegetation was a weak secondary predictor of habitat quality and biotic integrity. Sediment concentrations in streams measured during low flows and storm events were higher in areas of greater agriculture. A distributed parameter model suggested that an increase in forested land cover would result in dramatic declines in runoff and sediment and nutrient yields.
Burdon et al. (2013)	A survey of 30 agricultural streams (Canterbury Plains on New Zealand South Island) was performed along a gradient of nutrient concentration and sediment deposits was performed. Invertebrate community composition changed significantly along the gradient of deposited fine sediment, whereas the effect of dissolved nitrate was weak. Loss of habitat due to sedimentation was the likely cause. Degraded riparian condition controlling resources through direct (e.g., inputs) and indirect (e.g., flow-mediated) effects on deposited sediment.
Dauer et al. (2000)	In tributaries in the Chesapeake Bay watershed, total nitrogen concentration was positively correlated with agricultural activities. Benthic community condition was only weakly related to increased nutrient concentration, though strongly related to low dissolved oxygen events. Results suggested that, in the absence of low dissolved oxygen events, there are minimal effects of eutrophication on benthic communities.
Weijters et al. (2009)	A meta-analysis was performed to quantify effects of land use and nutrient concentrations on aquatic biodiversity. An increase in phosphorus concentration negatively effects macroinvertebrate diversity measures. Smaller effects were observed with increased nitrogen concentrations. The mechanism of these relationships was not explored.
Harding et al. (1998)	Sustained agriculture may profoundly alter biotic communities, and the effects of this disturbance may be persistent. In sites in North Carolina where historic agricultural land use (particularly in 30 m riparian) had been replaced by forests, benthic and fish communities continued to be more similar to those found in agricultural areas compared to forested areas.

5.2.3 Forestry

Stream benthic communities change in response to forestry, though not necessarily in a consistent manner, as both positive and negative impacts have been reported. Examples of impacts associated with logged sites include increase in sediment tolerant taxa, higher total invertebrate abundance, and decrease in the number of Ephemeroptera, Plecoptera and Trichoptera taxa. Reported negative impacts include increased short-term sediment inputs during the logging activities, as well as increased long-term sediment inputs due to the presence of access roads. The effects of sediment inputs may result in long-term alteration in stream bed habitat structure.

Macroinvertebrate richness, densities and biomass can increase due to increases in food resources (e.g., increase in leaf detritus and terrestrial insects falling into the streams) following canopy clearing and reemergence of different tree species. In cases where excess sedimentation does not permanently alter in-stream habitats, community alterations following canopy clearing may be short-lived (<5 years) due to rapid reemergence of pioneer plant/tree species and forest succession. Across a regional scale, there are not strong relationships between measures of logging impacts and macroinvertebrate metrics, likely because natural variation across sites is greater than effects related to logging. The interpretation of human influences in stream condition may be difficult when comparing multiple streams across a landscape, rather than tracking temporal changes in an individual stream, due to geo-climatic factors and landscape position (Kaufmann et al. 2009). However, weak, negative correlations between catchment harvest activity and B-IBI have been reported.

Review notes are included in Table 3.

Table 3. Literature review notes – impacts associated with forestry

Reference(s)	Key Findings
Fore et al. (1996)	Evaluated metrics that responded to extent of logging in SW Oregon. A measure of disturbance was developed considering riparian corridor, stream bed, bank stability, etc. Identified 10 metrics describing taxa richness and composition and tolerance/intolerance, but not feeding ecology that discerned most disturbed from least disturbed sites. Response was verified at evaluation sites
Hernandez et al. (2005)	Benthic invertebrate community structure in headwater streams was studied for old growth, red-alder dominated young growth, conifer-dominated young growth, and clear-cut in Alaska. Richness, densities and biomass of benthic invertebrates were higher in previously harvested areas (especially red-alder dominated young growth) relative to old growth forests and appeared to be the result of changes in food availability. Canopy removal led to increases in sunlight penetration and higher autochthonous food resources leading to an increase in biomass of scrapers and collectors, and an increase in richness/abundance of shredders.
Banks et al. (2007)	Measured the effects of clear-cut and forested catchments on the emergence of aquatic insect assemblages in Oregon. Emergent insects were strong indicators of harvest condition (clear-cut or forested) regardless of flow duration or season. More insects emerged in clear-cut catchments than in forested catchments. Taxa richness was slightly higher in clear-cut streams, primarily because of occurrences of rare taxa. Taxa that responded to conditions created by canopy opening contributed to differences in assemblages observed in clear-cut vs forested catchments.
Herlihy et al. (2005)	Examined the effects of forest harvest on headwater stream macroinvertebrate from 167 sites in Oregon. Results showed no strong relationships between measures of logging impacts and macroinvertebrate metrics (taxa richness, diversity, functional feeding groups). The IBI scores were not strongly related to the forest harvest history. At a regional scale, logging does not appear to be the major factor controlling macroinvertebrate assemblages; at this scale, macroinvertebrates tended to respond more strongly to natural environmental gradients.

Reference(s)	Key Findings
Carlson et al. (1990)	Macroinvertebrate communities and several aspects of fish habitat were examined for 16 northeastern Oregon stream segments, 11 with undisturbed riparian forests and five where 26–54% of the riparian forest had been harvested 6 to 17 yr. previously. Stream surface substrate composition was not significantly different between streams in logged and undisturbed areas. Macroinvertebrate density was 20 to 113 percent greater at the logged sites and diversity was similar at logged and undisturbed sites. The increase in macroinvertebrate abundance may have been due to an increase in primary productivity from canopy opening. However, increases in abundance were not always associated with increases in richness or diversity.
Hutchens et al. (2004)	A review on logging effects in eastern U.S. streams showed that obvious effects of logging on stream biota tend to be short-lived (< 5 years) due to rapid regrowth of terrestrial vegetation. There are some subtler, longer-lasting effects to leaf quality and wood debris dynamics. Leaves of early successional species tend to break down faster than late successional species leading to an increase in invertebrate shredders just after clearcutting.
Nislow and Lowe (2006)	Investigated effects of logging history and riparian forest characteristics on macroinvertebrates in New England. Recently logged and low canopy cover had higher total macroinvertebrate abundance and grazers. Results suggest that timber harvest in northern New England headwater streams may shift from shredder-dominated communities supporting low brook trout densities, to grazer/chironomid -dominated communities that can support higher brook trout densities.
Amaranthus et al. (1985) Lawrence et al. (2014) Sugden and Woods (2007) van Meerveld et al. (2014)	Sedimentation from unpaved resource/forestry roads is ~ 100x greater than undisturbed sites, and ~7x from clear-cuts. Sedimentation increases with rainfall intensity and truck traffic. Culvert replacements may result in short-term community impacts though would provide long-term benefits.
Zhang et al. (2009)	Comparing measures of benthic communities with modeled reference conditions suggested that logging impacts on habitats and invertebrate community metrics could be detected for up to 40 years. This approach indicated a lower richness and abundance of predators and scrapers at the previously logged sites compared with predictions from the reference sites. Possible factors included sediment deposition; streams in mature forests had coarser substrates than in young forests.
May (2002)	Debris flows can play a major role in adding sediment and wood into stream channels, and thus likely affecting benthic communities. Clear-cuts and roads tend to have more numerous contributing landslides compared to second growth and mature forest. Landslides at roads were an order of magnitude larger than non-road related landslides, thus contributing more sediments.
Jackson et al. (2001)	In coast range of Washington, compared in-channel habitat in 15 streams that were clear-cut with and without buffers, or were unharvested. In the clear-cut streams w/out buffers, logging debris covered 98 percent of the channel length with the average percentage of fines increasing from 12 to 44 percent. There was no debris in buffered streams. Particle size distributions and habitat distributions in the buffered and reference streams were largely unchanged from the pre-harvest to post-harvest.

5.3 Stressors

Stressors are broadly categorized into altered hydrology, altered habitat, and altered water quality (note that this organization structure was identified by the Interdisciplinary Team during the development of the Implementation Strategy and so is maintained here). These stressors commonly co-occur, and it remains challenging to isolate the impacts of any single one on benthic communities.

5.3.1 Altered hydrology

Many hydrologic metrics illustrate the range of mechanistic links between land use changes and flow regimes – changes on the land directly impact the way water flows off of it and into stream channels. Increased urbanization in a watershed is associated with higher peak flows and increased “flashiness;” either winter or summer base flows may also be affected. Statistical evaluation suggests that flashiness is particularly impactful to benthic communities (DeGasperi et al. 2009). Higher B-IBI scores were found at sites with longer periods of more stable flows and that were less flashy (Cassin et al. 2005). Alterations in benthic communities are related to all flow components (magnitude, frequency, duration, timing and rate of change). Altered hydrology is associated with a reduction in diversity, simplification of trophic structure and replacement of sensitive taxa by tolerant taxa. Alterations to flow processes have the potential to impact reproduction and crucial life cycle stages of benthic macroinvertebrates.

Table 4. Literature review notes - altered hydrology (stressor)

Reference(s)	Key Findings
Cassin et al. (2005)	Investigated relationship between B-IBI and measures of hydrological alteration in Puget Sound lowland streams. Higher B-IBI scores were found at sites with longer periods of more stable flows. Percent of Baetid mayflies, clinger taxa and taxa that were uni- or semi-voltine could provide diagnostic info about particular flow stresses. Timing of the onset of fall flows influenced B-IBI scores.
DeGasperi et al. (2009)	Evaluated hydrologic metrics for small streams in the Puget Sound lowlands that respond to urbanization and are biologically relevant. Eight hydrologic metrics correlated significantly with B-IBI scores (low pulse count/duration, high pulse count/duration/range, flow reversals, TQmean, Richards-Baker flashiness index) – reportedly accounting for 85% of total variance. High Pulse Count (# of days each water year that discrete high flow pulses occur) and High Pulse Range (range in days between the start of the first high flow pulse and the last high flow pulse during a water year) were potential indicators and were: 1. sensitive to urbanization, 2. statistically significant trends in urbanizing basins, 3. correlated with biological measures, and 4. relatively insensitive to potentially confounding variables like basin area. Note: follow up work by King County utilizing basin wide B-IBI data found overall weaker correlations, and no relation with High Pulse Range, suggesting that findings might not be completely generalizable across geographies (Sosik, personal communication).

Reference(s)	Key Findings
Booth et al. (2004)	In Puget Sound lowlands, landscape, hydrological and biological conditions were evaluated for stream watersheds with varying levels of urban development. Two hydrologic metrics were developed - TQmean - the fraction of a year that the daily mean discharge exceeds the annual mean discharge; T0.5yr - the fraction of a multiple-year period that streamflow exceeds the discharge of a flood peak that occurs (on average) twice each year) - both metrics decreased with increase in Total Impervious Area (TIA). B-IBI scores were higher in less flashy watersheds. Hydrologic metrics that reflect chronic altered streamflows, for example, provide a direct mechanistic link between the changes associated with urban development and declines in stream biological condition.
Konrad et al. (2008)	Streamflow and invertebrate data from 111 sites in western US were analyzed to identify streamflow characteristics (magnitude, frequency, duration, timing and variation) that affect benthic invertebrate assemblages (abundance, richness, diversity and evenness, functional, feeding groups, individual taxa). Assessed streamflow characteristics as limiting factors on invertebrate assemblages and found most metrics of streamflow, particularly associated with daily to inter-annual scales were associated with limits on invertebrate assemblages, e.g., altered steam flow changed and limited benthic assemblage quality.
Morley and Karr (2002)	Lower B-IBI scores were associated with increased flashiness in Puget Sound basin streams.

5.3.2 Altered habitat

This section focuses on impacts on benthic communities associated with changes to in-stream physical habitat and the riparian zone. The watershed-scale land use alterations are considered in the pressures section (Section 2.2.3).

Sediment loading to stream channels tends to increase with the amount of urbanization, agricultural activities, and during forest harvest, compared to forested watersheds, leading to more suspended sediments in the water column as well as more fine sediments in the stream bed. Declines in invertebrate community condition have been noted at a threshold of approximately 20% fine sediments covering the streambed. Loss of habitat availability through the fill and cover of coarse substrate and associated interstices (i.e., fine sediments fill all the spaces between gravel and cobbles in stream beds) was the key driver affecting invertebrates. Increased turbidity and suspended sediment concentrations can reduce primary production and food availability to benthic invertebrates, and lead to avoidance, reduced physiological function, and mortality (see Section 2.2.4.3, Water Quality).

In-channel woody debris is another important element that helps create complex in-stream habitat, and can serve as a food source for benthic macroinvertebrates. The input of woody debris to streams most often comes from mature riparian vegetation and so the removal or fragmentation of mature vegetation via development or agriculture reduces inputs. Over time,

this will reduce the overall abundance of large woody debris and affect the health of a stream channel.

Native riparian structure (i.e., the presence of diverse, multi-aged vegetation along the edges of a stream channel): 1) supports stream shading and reduces stream temperatures, 2) can control the inputs of sediments into the stream channel (both from the riparian zone by providing structure and support to soils, and from the contributing watershed), 3) provide direct habitat for adult life stages of some invertebrates, 4) supply energy inputs (e.g., leaf fall and detritus) to the in-stream communities, and 5) may affect primary productivity. As such, alterations of the riparian zone can lead to a variety of negative impacts to benthic communities that can take decades to restore.

Table 5. Literature review notes - altered habitats (stressor)

Reference(s)	Key Findings
Herlihy et al. (2005)	<p>Focus: Stream bed sedimentation</p> <p>Evaluated condition of 167 sites in headwater streams from the three forested ecoregions in western Oregon. In catchments with diverse forestry history, percent sand/fine substratum was environmental variable most strongly related to macroinvertebrate IBI, with a negative correlation.</p>
Roy et al. (2003)	<p>Focus: Stream bed sedimentation</p> <p>Evaluated effects of catchment urbanization in 30 streams in Georgia. Changes in macroinvertebrate assemblage structure were related to factors indicating change in bed sediment. Many macroinvertebrates need large particles and interstitial spaces for protection against predators and high currents, substrate for periphyton food sources, attachment for filter feeding, and increased oxygen exchange. An Invertebrate Community Index was best predicted by measures of specific conductance, bed sediment variability (increased variability → increases ICI score), and riffle bed sediment size (increased sediment size → increased ICI score). .</p>
Burdon et al. (2013)	<p>Focus: Stream bed sedimentation</p> <p>Conducted a survey of 30 agricultural streams on New Zealand South Island along gradients of deposited sediment and dissolved nutrients. Invertebrate community composition changed significantly along the gradient of deposited fine sediment; the effect of dissolved nitrate was weak. %EPT, Ephemeroptera, Plecoptera, Trichoptera responded negatively to sediment. Data suggested an effects threshold of ~20% fine sediment covering the streambed. Decreased habitat availability was likely key driver. Degraded riparian condition was associated direct and indirect sediment-related effects.</p>

Reference(s)	Key Findings
Bryce et al. (2010)	<p>Focus: Stream bed sedimentation</p> <p>Evaluated several lines of evidence to identify bedded fine sediment threshold levels for fines (≤ 0.06 mm) and sand/fines (≤ 2 mm) to protect sediment-sensitive aquatic species in the western US. Compared 169 reference sites and 557 mountain stream sites with progressively higher ambient levels of streambed sediment. The predicted maximum macroinvertebrate IBI declined 4.0 points (SE = 0.60) for each 10% increase in fines, and 3.0 points (SE = 0.50) for each 10% increase in sand/fines. For sediment-sensitive aquatic macroinvertebrates, effect threshold levels were 3% (fines) and 10% (sand/fines).</p>
Hernandez et al. (2005)	<p>Focus: Wood debris</p> <p>Benthic community structure in headwater streams was studied for three in-stream habitats (woody debris, cobble, gravel) with four forest management types (old growth, red alder-dominated young growth, conifer-dominated young growth, clear-cut) in Alaska. Wood debris supported richer and higher invertebrate fauna than cobble or gravel substrates.</p>
Hilderbrand et al. (1997) Wallace et al. (1995)	<p>Focus: Wood debris</p> <p>Riparian vegetation is a source of woody debris to streams. Woody debris has diverse effects in streams. It increases stream depth and lowers current velocities that increase transit time in a reach. It also increases retention of particulate organic and inorganic matter. Studies where large woody debris were experimentally added to streams in SW Virginia (Hilderbrand et al.) and North Carolina (Wallace et al) showed significant changes in macroinvertebrate assemblages that were due to changes in habitat and food sources.</p>
Munn et al. (2008)	<p>Focus: Stream bed substrate</p> <p>Stream bed substrate is major controlling factor for benthic invertebrates in that it commonly explains much of the invertebrate assemblage composition and distribution of populations. Mechanisms include organic matter retention, effects on predation and competition, and providing in-stream flow refugia. Substrate condition can be a major factor and may mask the influence of the variation in other variables, such as nutrients.</p>
Hawkins et al. (1982)	<p>Focus: Riparian and substrate</p> <p>Importance of riparian vegetation and substrate composition on invertebrate community structure was investigated in six streams in Oregon. Canopy type was more important than substrate type in influencing total abundance and guild structure. Streams without shading had higher abundances of invertebrates than did shaded streams. Food quality was more important than food quantity or substrate composition</p> <ul style="list-style-type: none"> • Authors studied streams with 1) old-growth coniferous canopy, 2) a second-growth deciduous canopy, or 3) no canopy (clear-cut) to compare different primary food sources. Streams with no canopy had higher abundances of macroinvertebrates associated with increased algae; • More open canopy \rightarrow increased light \rightarrow increased algae. Algae are more nutrient rich than leaf/wood detritus, and therefore can support a higher abundance of species; • Low gradient sites had higher percentages of sand and gravel (36-58%) than higher gradient sites (9-25%) – no differences in substrate composition were observed among streams with different canopies – gradient/substrate composition had little influence on invertebrate abundance.

Reference(s)	Key Findings
May et al. (1997)	<p>Focus: Riparian</p> <p>In Puget Sound lowland streams, more intact/wider riparian forests associated with sites with B-IBI scores higher than expected in urban streams.</p>
Shandas and Alberti (2009)	<p>Focus: Fragmentation</p> <p>Study investigated the role of vegetation patterns in variations in aquatic biota in Puget Sound lowlands. Fragmentation of upland vegetation and the total amount of riparian vegetation explain the greatest variation in aquatic conditions. Role of fragmentation was equivocal.</p>
Parkyn and Smith (2011)	<p>Focus: Fragmentation and Dispersal</p> <p>Aquatic invertebrates achieve dispersal via active (e.g., flight) or passive (e.g., via wind, water or vectors such as birds) methods. Urbanization (loss/fragmentation of riparian forests, altered hydrology) can prevent the completion of adult insect life cycle, limit dispersal, and influence population persistence. Sites with poor levels of landscape connectivity (distances >1 km of adverse habitat conditions and where dispersal cannot occur along stream corridors) may never achieve full biodiversity recovery if: 1) conditions are hostile for aerial dispersal, 2) prevailing winds do not facilitate stochastic dispersal, and/or 3) there are no other vectors (such as birds) of passive dispersal.</p>
Tonkin et al. (2014)	<p>Focus: Fragmentation and Recolonization</p> <p>Assessed the probability of benthic macroinvertebrate colonization of reach-scale restoration projects in 21 stream in Germany. Colonization failure within restored reaches was directly related to the proximity to the nearest taxon pools and restored reaches, not a lack of habitat improvement. Creating smaller restored patches within a degraded system to allow "stepping stones" for macroinvertebrate recolonization may help support ecological recovery goals.</p>

5.3.3 Altered Water Quality

This section addresses the effects of degradation in water quality (changes to pH, nitrogen, phosphorus, conductivity, suspended sediments, etc.) and increases in contaminants such as metals and organic pollutants, such as insecticides and fungicides. While streams and rivers have natural levels of metals, nutrients, etc., these are considered pollutants if the concentrations exceed typical background levels or exceed toxic thresholds.

5.3.3.1 Metals

Regional monitoring data describing metal concentrations in streams and literature-reported toxicological effects levels suggest that chronic exposure to metals is likely not affecting benthic communities in Puget Sound streams. However, bed sediments in reaches in highly urbanized catchments may have elevated metals concentrations.

Regular river and stream monitoring by the Washington State Department of Ecology has indicated that there are very few exceedances of metals water quality standards in Western Washington (Ecology 2010). Between 1994 and 2009, for example, 1.7% of samples exceeded

the Washington State chronic water quality criteria for mercury (Hg) of 0.012 µg/L, with a maximum reported concentration of 0.099 µg/L. Maximum dissolved copper concentrations were reported up to 16.3 µg/L; maximum dissolved lead was 0.8 µg/L. No other contaminants exceeded water quality criteria for metals. Storm specific monitoring, meant to highlight the potential impacts of stormwater runoff in streams and rivers, did not find concentrations higher than listed above (Ecology 2011). Exposures to mixtures of metals, which are likely in stream environments and potentially measured by CCUs³, may elicit higher responses compared to single-metal exposures (Clements 2004) and so comparisons to individual criteria may not be sufficient.

It is difficult to predict community response to metals exposure across streams. Chronic exposures to heavy metals at levels that exceed water quality criteria (or cumulative criteria unit exceedances for mixtures) has been shown to alter richness and can affect particularly sensitive species such as mayflies (Hickey and Golding 2002).

Exposure to metals can alter benthic community structures, particularly at levels that exceed listed water quality criteria. Response may be due to increased drift, where insects allow themselves to be carried downstream from an area of exposure, and not necessarily mortality. There is evidence of species or community tolerance developing to chronic metals exposures, so exact threshold values for shifts in taxa have been difficult to establish.

Selected review notes are included in Table 6.

Table 6. Literature review notes - degraded water quality - metals (stressor)

Reference(s)	Key Findings
Eden (2016)	A study on the impacts of heavy metals on macroinvertebrates in New Zealand found that sediment bound metals and dissolved metals (and impervious area) were significant factors explaining the change in invertebrate community structure. Mayflies and caddisflies were particularly sensitive.
Clements (2004)	A set of microcosm experiments were used to evaluate responses to metals (Zn, Cd, and Cu) exposure. Mixtures of metals elicited higher responses than Zn only. Mayflies and stoneflies were more sensitive to metals exposures compared to caddisflies and dipterans. Drift and community respiration were significantly changed by metals exposure. $CCU^3 < 5$ was estimated to be protective of benthic communities.

³ Cumulative Criterion Units (CCU) = $\sum \frac{\text{Metal Concentration}}{\text{EPA Hardness-adjusted Chronic Criteria}}$

Reference(s)	Key Findings
Pollard and Yuan (2006)	Compared benthic invertebrate community structure with different levels of metals pollution - background, low, med, high – in Colorado and West Virginia. The community structure was most similar at background sites and dissimilarity increased with increasing metals pollution categories. There were taxonomic shifts occurred with increasing metals pollution as expected based on metal tolerance. However, different metal-tolerant genera were found at different metal-impacted sites, suggesting that local conditions, including species interaction, may be important. There is a low predictability of assemblage response to a common stressor.
Hickey and Golding (2002)	In stream mesocosms, there were some significant changes to benthic invertebrate populations to low (CCU = 2.4), medium (CCU = 5.9) and high (CCU = 18) metals concentrations in 34-day exposure studies. Effects on taxa richness and the number of EPT taxa at the high exposure concentration were -23% and -26%, respectively. At high exposure concentration, all of the 5 major mayfly species were near extinction, whereas 4 of 7 caddisflies increased (up to +121%) and 3 decreased (up to -76%). Changes at the low and medium exposures were largely insignificant.
Pond et al. (2008)	Related specific conductance as a proxy measure of mining disturbance, to benthic invertebrate community condition in 37 small streams in West Virginia. Increased mining was associated with differences in water chemistry, shifts in species assemblages, loss of Ephemeroptera taxa, and changes to specific macroinvertebrate matrixes and indices. Results suggest that biological condition most strongly correlates with measures of water quality, compared to sedimentation of riparian habitat.
Hart Crowser (2015)	Seven stream reaches at Monte Cristo mining facility were characterized for habitat, chemical, and biological condition. There were no clear relationships found between metals in water column or sediments, and B-IBI.

5.3.3.2 Organic Contaminants

There is a large number of organic contaminants (such as pesticides) that can severely impact biotic communities. The response to exposure varies depending on species, exposure concentration and duration, and chemical constituent; mixture effects can also be important. Water quality sampling in urban streams showed that pentachlorophenol (detection frequency [df] = 80%), prometon, trichlopyr, 2,4-D, and MCPP (df > 70% for all), and diazinon (df = 65%) were the most frequently detected compounds (USGS 2004). 2,4-D, trichlopyr, and simazine were present at levels > 1 µg/L in streams during storm events. Most streams had ten or more compounds present during the sampling period. This suggests that benthic communities in urban streams are regularly exposed to a mixture that included multiple organic contaminants.

Washington State Department of Agriculture sampled streams in agriculture and urban watersheds and analyzed for pesticides (WSDA 2017). On average, five pesticides were detected in each sample, with chlorpyrifos and bifenthrin most commonly exceeding water quality criteria. In a broader study, WSDA analyzed stream sediments for pesticides, bifenthrin was the most frequently detected pesticide (df = 19%) with concentrations ranging from 11-120

µg/kg (dry weight). These levels are above levels of concern for benthic invertebrates (WSDA 2018). Pesticides that persist at toxic concentrations in stream sediments for months to years, such as bifenthrin, may limit recovery in streams with otherwise adequate habitat.

A summary of selected literature is included in Table 7.

Table 7. Literature review notes - degraded water quality – organic contaminants (stressor)

Reference(s)	Key Findings
Chiu et al. (2016)	Benthic invertebrate communities have a range of sensitivities to pesticide exposures. Both pulse (maximum) and chronic (long-term average) exposures may be important in estimating impacts to community structures. Pesticide exposure will likely alter communities to favor pesticide-tolerant taxa.
Weston and Lydy (2014)	Pesticide exposure risk to macroinvertebrates may be underestimated. Chemical occurrence and toxicity data are often insufficient to understand pesticide impacts on benthic communities. For example, 14 macroinvertebrate species were exposed to fipronil and two common degradation products. Four species were more sensitive than any previously studied, particularly to the degradation products.
Rogers et al. (2016)	Mesocosm studies indicated reduced macroinvertebrate abundance, richness, and biomass due to bifenthrin exposures, with EC50 ranging from 197.6 – 233.5 ng/g OC. Adult emergence was altered at all exposures evaluated. Indirect impacts, such as increased periphyton abundance likely due to decreased scraper abundance, were also observed.
Colville et al. (2008)	Mesocosm studies indicated altered community structure following pulse chlorpyrifos exposures at 1-10 µg/L with a NOEC calculated at 1.2 µg/L. Mayflies were sensitive to exposures resulting in reduced abundance. Recovery of stream benthic communities exposed to chlorpyrifos was limited even after 124 days.
Carpenter et al. (2016)	Water and sediment sampling for pesticides in urban streams indicated the presence of insecticides, fungicides, and herbicides with nearly all streams having at least one pesticide occur at levels exceeding aquatic life benchmarks, most often for bifenthrin. Bed sediment concentration of bifenthrin were highly correlated with benthic assemblages.

5.3.3.3 Nutrients

Excess nutrient inputs can impact benthic assemblages through eutrophication and increased turbidity. Eutrophication leads to the potential of increased or altered food resources, which may allow some faster growing taxa to outcompete others, leading to reduced biotic diversity. Effects levels (i.e., the levels which are expected to lead to eutrophication, which could lead to changes in the food web community structure) for total phosphorus in streams have been estimated from 0.06 – 0.09 mg/L (Evans-White et al. 2009, Wang et al. 2007). The results of regional monitoring indicated that 95% of streams outside Urban Growth Area boundaries, and 80% of stream sites within Urban Growth Area boundaries, were below the lower threshold of 50 µg/L for total phosphorus impacts (King County 2018). Results of a Washington statewide

survey suggested that elevated total phosphorus and total nitrogen were both associated with low B-IBI scores (Larson et al. 2019). Nutrients were also identified as an important stressor affecting invertebrate assemblages, though less than temperature, bed sediment condition, and pesticides (Waite et al. 2020)

Condition class thresholds (i.e., the concentration of a given constituent that generally differentiate between “poor”, “fair”, and “good” stream condition) are lower for phosphorus than nitrogen levels, suggesting that phosphorus inputs might be more important than nitrogen inputs in determining stream condition (Larson et al. 2019)

A summary of selected literature is presented in Table 8.

Table 8. Literature review notes - degraded water quality – nutrients (stressor)

Reference(s)	Key Findings
Evans-White et al. (2009)	Evaluated the relationship between benthic macroinvertebrate richness with increasing nutrient and turbidity concentrations in Central Plains (USA) streams. Results suggest that changes in resource quality associated with anthropogenic nutrient inputs could contribute to large-scale losses in biodiversity in lotic ecosystems. For example, changes in food quality might alter primary consumer (grazer and detritivore) richness by changing competitive advantages among species; this would have less impact on predator richness. An effects threshold of ~ 0.06 mg/L TP was suggested for primary consumers. Results suggest impacts from eutrophication in addition to increased turbidity and low DO.
Wang et al. (2007)	Examined how macroinvertebrate measures correlated with nutrients in Wadeable streams in Wisconsin. Percent EPT individuals and taxa, Hilsenhoff biotic index, mean tolerance values were measures that were most strongly correlated with many nutrient measures. 53% of macroinvertebrate variance was explained by the environmental measures. Of that, 22% was explained by changes in nutrient concentrations (42% by habitat and 32% by “interactions”). Macroinvertebrate effects thresholds were estimated at 0.06-0.09 mg/L for TP and 0.9-1.1 mg/L for TKN.
Burdon et al. (2013)	Performed a survey of 30 agricultural streams in New Zealand along a gradient of nutrient concentration and deposited sediments. Invertebrate community changes associated with dissolved nitrate were weak.
Dauer et al. (2000)	Evaluated associations between macrobenthic communities (as measured by B-IBI) and water and sediment quality, and measures of anthropogenic activities in the Chesapeake Bay watershed. Benthic community condition was only weakly related to increased nutrient concentration, though strongly related to low dissolved oxygen events, which explained 42% of variation. Results suggested that, in the absence of low dissolved oxygen events, there are minimal effects of eutrophication on benthic communities.
Weijters et al. (2009)	A meta-analysis was performed to quantify effects of land use and nutrient concentrations on aquatic biodiversity. An increase in phosphorus concentration negatively affects macroinvertebrate diversity measures. Smaller effects were observed with increased nitrogen concentrations. The mechanism of these relationships was not explored.

5.3.3.4 Temperature

Development along the riparian zone can reduce tree cover leading to increased water temperatures, which can affect sensitive species. There are many streams listed on the Washington State 303(d) for temperature. Sets of taxa have been identified that are negatively impacted by increasing water temperatures, though the thresholds vary. Low temperature (~11-14°C) and high temperature (17-20°C) change points have been identified (Richards et al. 2018, Waite et al. 2020). Temperature alterations will change the rate and timing of physiological processes of individuals, which may lead to community-level alterations. A risk evaluation did not identify high temperatures as a risk factor in Washington streams (Larson et al. 2019) though temperature was a risk factor for streams in the Willamette Valley, Oregon (Mulvey et al. 2009).

Literature review notes are included in Table 9

Table 9. Literature review notes - degraded water quality – temperature (stressor)

Reference(s)	Key Findings
Lawrence et al. (2010)	Evaluated climate change-related changes of increased temperature and decreased rainfall on benthic communities in northern California, based on 20-year data set in four sites in two streams, Established indicators such as B-IBI and metrics of benthic community structure did not change based on altered temperatures, but they did identify a set of taxa that showed temperature-related responses. Temperature changes affected growth and timing of development and emergence.
Burgmer et al. (2007)	Analyzed a long-term data set of macrozoobenthos in streams of northern Europe to investigate climate change-related changes to communities. Temperature can influence the physiological processes of invertebrates, possibly leading to changes in the timing of life history events and trophic interactions. This may alter diversity and community composition. Temperature-related changes in stream communities were not directly observed.
Richards et al. (2018)	Analyzed macroinvertebrate database for wadeable streams in Idaho to identify threshold change points for over 400 taxa along an increasing water temperature gradient. Two change points were identified: for the taxa that decreased with increased temperatures (n=196) the change point was approximately 20.5 °C; for the taxa assemblages that increased with increased temperatures (n=9), change point was about 11.5 °C.
Waite et al. (2020)	In an evaluation of biological and condition data collected in 82 wadeable streams in Pacific Northwest, temperature was identified as an important variable affecting invertebrate assemblages. Different taxa exhibited different change points, with some responding around 13-14 °C while others around 17 °C.
Larson et al. (2019)	Evaluated data from assessment of 346 stream sites in Washington State. Temperature was not generally associated with poor benthic condition according to relative risk/attribution risk evaluation of streams across Washington

5.3.3.5 *Suspended Sediments and Turbidity*

This section focuses on the effects from increased turbidity - effects on in-stream habitat associated with increased sediment inputs are described in Section 5.3.2.

Increases in stream turbidity can limit light penetration and therefore reduce primary production – reductions of phytoplankton and macrophyte biomass, growth, and diversity have been observed with increased turbidity (Hoetzel and Croome 1994), which can decrease food availability for herbaceous insects, though phytoplankton abundance may be less relevant in Puget Sound wadeable streams. Increase in turbidity of 5 NTU decreased primary production by 3 to 13%, and increases of 25 NTU decreased primary production up to 50% (Lloyd et al. 1987). Other food web impacts have been observed. The reduction in phytoplankton may create additional cascading effects at higher trophic levels via a reduction in available food energy.

Table 10. Literature review notes - degraded water quality – suspended sediments and turbidity (stressor)

Reference(s)	Key Findings
Henley et al. (2000)	A review of effects of sedimentation and turbidity on lotic food webs. Increases in sedimentation and turbidity are associated with decreases in primary production that may lead to negative cascading effects through reduced food availability. Reduced food availability across trophic levels can depress growth rates, reproduction, and recruitment. Direct effects of increased turbidity include mortality, reduced physiological function, and avoidance.
Wagener and LaPerriere (1985)	Authors investigated several streams impacted by placer mining in Alaska. The levels of increased turbidity were a strong descriptor of reduced invertebrate density and biomass.
Quinn et al. (1992)	Investigated the impacts of clay discharges and turbidity on benthic invertebrates in six streams in New Zealand. Invertebrate densities were significantly lower at impacted/high turbidity sites, ranging from 9 to 45% (median 26%) of densities of controls. Effects included lower density of the common taxa, and lower taxonomic richness. Reduced epilithon biomass and productivity, and degraded food quality probably explain the lowered invertebrate densities.

6 CONSIDERATIONS IN IMPLEMENTING THE RECOVERY STRATEGIES – SCALE, STRESSOR IDENTIFICATION, EFFECTIVENESS, AND COSTS

This section covers selected considerations in implementing recovery strategies including scale of planning and implementation, methods and approaches for stressor identification, the effectiveness of various restoration/mitigation approaches, and selected issues related to cost. A literature review summary is presented in Table 11. The effectiveness and cost/cost-benefit focus areas were suggested by the Interdisciplinary Team during strategy development. Additional cost information is included in Section 8 of the Implementation Strategy narrative.

Key Points (Considerations for Implementing Recovery Strategies):

- Watershed-scale processes affect stream condition, so watershed condition should be considered in planning and implementing recovery and restoration activities, which are usually at a local scale.
- Due to co-occurrence of stressors, it is difficult to isolate and identify a single causative mechanisms that accounts for degraded stream conditions. Stressor identification does not typically rely on a single analysis but rather uses weight of all evidence, where multiple lines of evidence highlight the most probable causes.
- In-stream habitat (excessive fine sediment) is a commonly identified stressor in regional streams.
- There are few studies that specifically evaluate the effectiveness of stream restoration to improved benthic community condition. Those that are available suggest localized actions do not result in improved biological condition. Watershed-scale approaches are most likely needed.
- Stormwater Best Management Practices (BMPs), including LID/GSI, can reduce stressors. There is little evidence on their ability to improve B-IBI scores.
- It remains challenging to generalize findings from cost-benefit studies.

6.1 Considerations of Scale in Restoration and Recovery

Scale is an important consideration in evaluating impacts of pressures and stressors on stream health, and in evaluating potential responses to restoration and/or mitigation activities. There is a hierarchical organization of stream function which are influenced over a range of spatial scales (Leps et al. 2015). Land use influences stream communities from the local scale (riparian zone) to the watershed scale (Fore et al. 2013). In-stream habitat shapes insect assemblages on the local scale, whereas water quality is a function of catchment-scale processes. The functional pyramid (Figure 1) provides a generalizable model of the relationships between watershed-scale processes such as hydrology (there defined as water running off the landscape) and local stream conditions.

Allan et al. (1997) provided an illustrative example where in-stream habitat and inputs of allochthonous (from nearby terrestrial habitats) carbon were determined primarily by local

conditions such as riparian vegetation, whereas nutrient and sediment inputs, and hydrology and channel characteristics were influenced by watershed-scale conditions, including landscape features, land cover, and land use. Such considerations are important in recovery planning.

These considerations of scale matter for recovery as well. Almost all restoration activities are local; the effectiveness of such localized restoration is often dependent on the condition of the watershed. The biological condition in a low quality reach can be augmented by a high quality watershed and, conversely, the biological condition in a high quality reach can be diminished by a low quality watershed (Stoll et al. 2016).

A schematic illustrating general pressures and stressors, and scale considerations is included in Figure 3.

6.1.1 Considerations of Scale - Riparian Buffer

The riparian zone is an area along the stream channel that has strong connections to the aquatic ecosystem. Maintaining and restoring a highly functional stream riparian zone has been identified by the IDT and others as a potential critical management approach due to its unique, intimate interaction with the stream channel and its biota (Quinn et al. 2019). Focused research on the potential to increase the functional capacity of buffers has been suggested, particularly in light of the increased pressures associated with future population growth.

Traditional approaches to evaluating the impacts of riparian-zone restoration or management on streams have often focused on fixed-width riparian buffers along a stream channel (e.g., (Fore et al. 2013, Wahl 2012)). However, others have suggested that variable-width or dynamic buffers, where the width is determined by landscape and or soil features, might be a more appropriate means of evaluation (Abood et al. 2012, Kuglerova et al. 2014). Abood et al. (2012) incorporates the 50-year floodplain, which is a function of flow, stream gradient, and channel morphology, to define the riparian zone. King County (2019) suggested that dynamic buffers may be more ecologically relevant than fixed width buffers as they account for factors describing how a stream is influenced by the riparian zone.

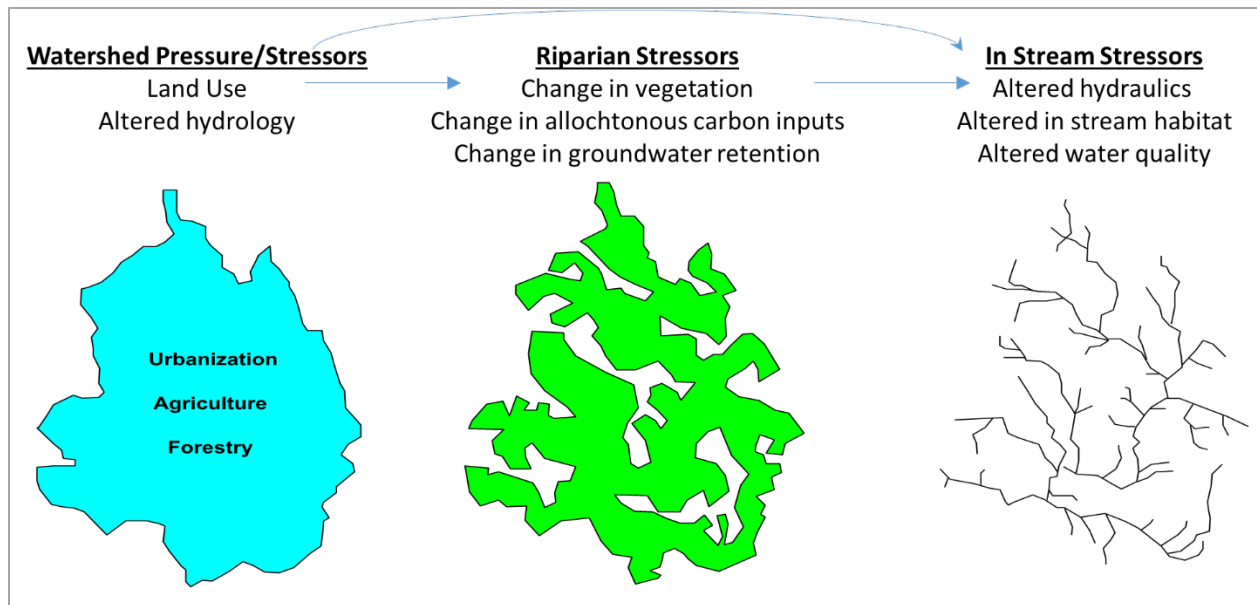


Figure 3. Conceptual model describing the influence of watershed-scale land use pressures on factors that affect biotic communities at different spatial scales.

6.1.2 Considerations of Scale - Watershed Condition and Observed Biological Potential

Urbanization in a watershed impacts stream condition via changes in hydrology, geomorphology, and chemistry (Booth (2005); Figure 1 and Figure 2). The cumulative effects decrease the biological condition and can limit the potential for biological improvements from mitigation activities (Booth and Jackson 1997, Weigel et al. 2003). These limits have been described as the observed biological potential. Paul et al. (2009) identified a relationship between measures of urbanization and biological condition, implying that for a given level of urbanization, there are limits to the biological condition. Note that other models have been developed that predict biological condition (and the potential limits on condition) based on land use criteria, such as Beck et al. (2019). There are three important points in the observed biological potential that are relevant to addressing pressures, and stream recovery and restoration. The first is that there is a relatively well defined limit to biological condition for a given level of watershed urbanization (Booth 2005). This is illustrated in the regression line in Figure 4 that defines the relationship between the upper limits of biological condition (i.e., the observed biological potential) and level of urbanization. This matters for restoration and recovery in that the overall watershed condition may limit improvements from restoration or management activities. The biological potential provides a “best case scenario” for watersheds.

A second point is that there is a wide range of biological condition for a given level of urbanization, as illustrated by the area below the observed biological potential line. For example, the measured biological condition for watersheds with ~ 20% urbanization ranged

from >10 (“very poor”) to ~80 (“good”). This likely reflects both variation in individual watersheds as well as variation in which stressors associated with urbanization are actually present in those watersheds.

The third point is that the distance between measured biological condition and the biological potential represents room for improvement for stream recovery. Sites well below their predicted observed biological potential may be amenable for improvement, while sites near these limits may only respond weakly to mitigation activities. Sites in highly urbanized watersheds might only recover to a limited degree.

Again, the observed biological potential, as defined by the overall watershed condition, is an important consideration in recovery planning and implementation, and in expectation setting when designing restoration projects.

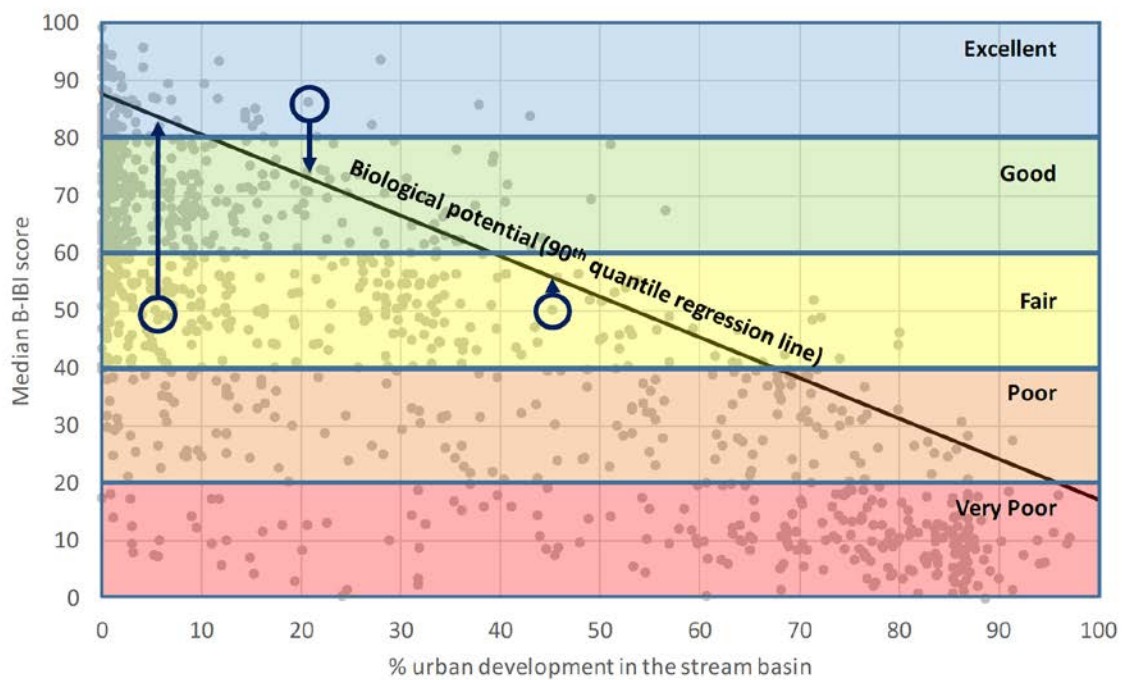


Figure 4. Urban gradient versus biological index for sites within the Puget Sound watershed. Data demonstrates the relationship between biological condition and urban gradients. The regression line highlights the maximum expected biological condition based on the urban gradient; this is called the observed biological potential. Adopted from King County (2019)

Table 11. Literature review summary – scale and observed biological potential.

Reference(s)	Key Findings
Stoll et al. (2016)	Analyzed benthic invertebrate community data from 1087 sampling sites in central Germany to evaluate effects of habitat quality on benthic communities at different scales. If regional habitat quality is good or poor, that will act as a determinate of local invertebrate communities. Poor regional habitat quality can limit local benthic communities. High regional quality can subsidize local benthic communities. Local areas within greater regions of intermediate habitat quality can be influenced by local conditions.
Allan et al. (1997)	The influence of catchment land use on stream integrity across multiple spatial scales in Michigan. “Extent of agricultural land at the subcatchment scale was the best single predictor of local stream conditions.” “Local riparian vegetation was uncorrelated with overall land use and was a weak secondary predictor of habitat quality and biotic integrity.” “Instream habitat structure and organic matter inputs are determined primarily by local conditions such as vegetative cover at a site, whereas nutrient supply, sediment delivery, hydrology and channel characteristics are influenced by regional conditions, including landscape features and land use/cover at some distance upstream.”
Black et al. (2004)	Macroinvertebrate assemblages and environmental variables were evaluated at 45 stream sites at reach, local, and watershed scales throughout the Puget Sound. At all scales, the dominant environmental variables represented an anthropogenic gradient. However, analysis suggests that watershed condition or reach scale condition can negatively affect community composition if land use at either scale is below a given forested threshold.
Roy et al. (2005)	Habitat and biota were compared between paired open and forested reaches in the Piedmont of Georgia, USA. There were no differences in habitat diversity (variation in velocity, depth, or bed particle size) between open and forested reaches. Reach-scale biotic integrity was largely unaffected by differences in canopy cover. In urbanizing areas where catchment land cover drives habitat and biotic quality, management practices that rely exclusively on forested riparian areas for stream protection are unlikely to be effective at maintaining ecosystem integrity.
Hughes et al. (2014)	A review of practices and approaches for stream rehabilitation in Oregon. Mitigation projects are implemented at the site or reach scale when many of the limiting factors are occurring at the watershed scale. It is almost always more effective to rehabilitate at watershed or basin scales, to recover natural flow regimes. Therefore, the priority actions are (1) protect existing upstream high-quality habitats and (2) reestablish ecosystem processes in the altered places before attempting to rehabilitate specific sites lower in the watershed.
Weigel et al. (2003)	Evaluated data from 94 sites in Minnesota, Wisconsin and Michigan to identify environmental variables at the catchment, reach and riparian scales that influence stream macroinvertebrates. Results were consistent with the concept of hierarchical functioning of scale in which large-scale variables restrict the potential for macroinvertebrate traits or taxa at smaller spatial scales. Catchment and reach variables were equally influential in defining assemblage attributes, whereas the reach scale was more influential in determining relative abundance and presence/absence.

Reference(s)	Key Findings
Beck et al. (2019)	Developed a landscape model for California that estimates ranges of scores for a macroinvertebrate-based index. To support prioritization decisions for stream management, such as identifying reaches for restoration or protection based on observed vs predicted scores. Median scores were accurately predicted for all sites in California with bioassessment data.

6.1.3 Current Puget Sound Watershed Condition

The Puget Sound Watershed Characterization project has performed several watershed-scale analyses based on land use within watersheds (Stanley et al. 2019) including levels of overall hydrological degradation. The results are shown in Figure 4. The results of this characterization work are useful for recovery and restoration planning in that they may provide a measure of expected potential best results that can be achieved by a given restoration project.

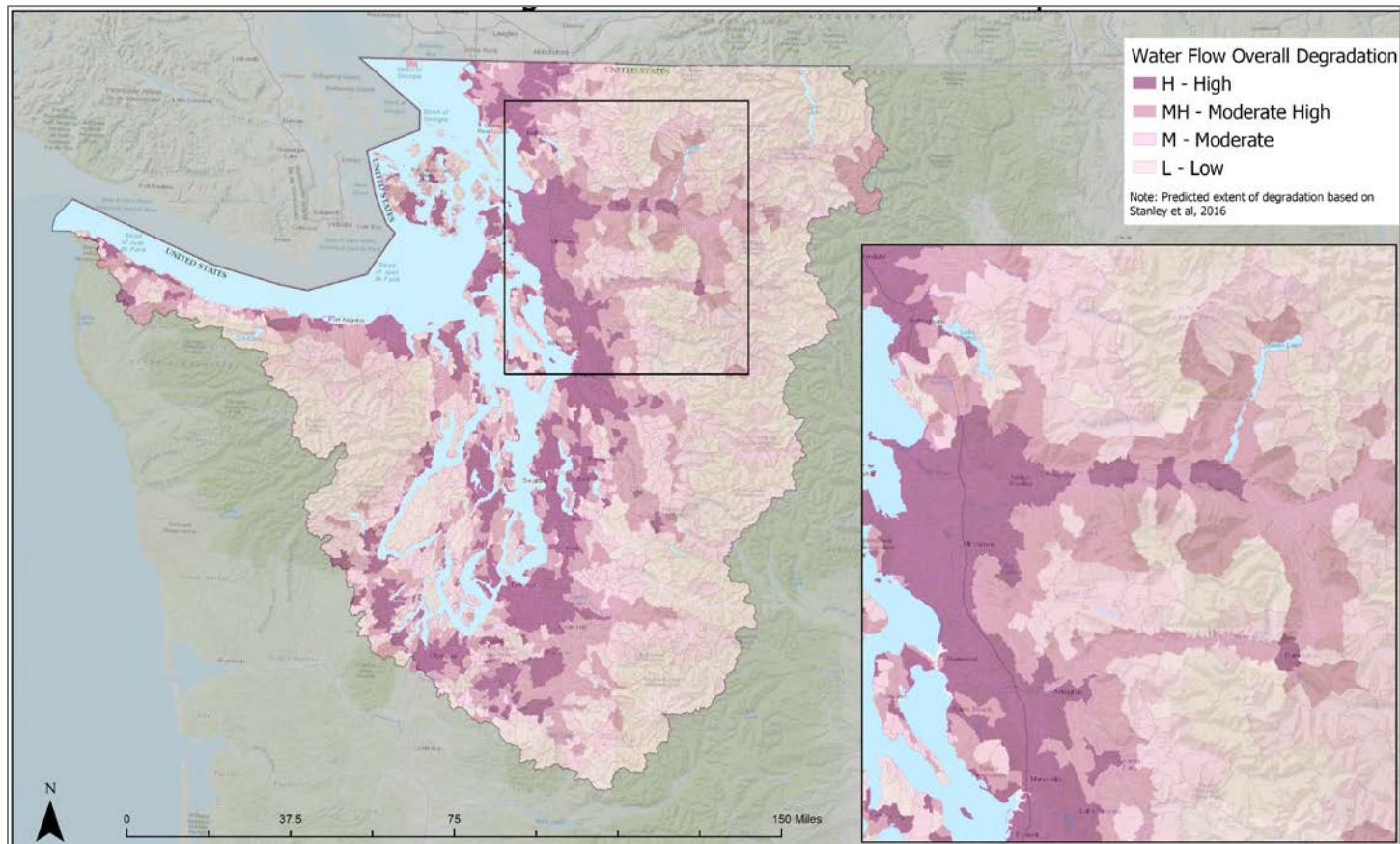


Figure 5. Predicted levels of overall degradation to water flow processes based on Puget Sound Watershed Characterization Project. The modeled degradation is strongly dependent on measures of imperviousness in a given assessment unit and can provide an indication of maximum biological potential. Watersheds that are highly degraded based on this analysis may have lower overall biological potential compared to those with low levels of degradation. Adapted from Stanley et al. 2019.

6.2 Recovery Planning – Stressor Identification

A key element of recovery and/or restoration planning is the identification of the primary stressors affecting a site and watershed, which will help focus mitigation activities and resources. It is well documented that most reaches in developed watersheds are impacted by multiple stressors; figuring out which of these matters is valuable in mitigation planning.

There are several approaches. One that has been used locally is the US EPA Casual Analysis/Diagnostic Decision Information System (CADDIS; U.S. EPA (2000)). CADDIS is a framework that encourages practitioners to take a systematic, weight of evidence approach⁴ to: 1) define the issue (i.e., geographic scope and impacts to be evaluated), 2) list candidate causes, 3) evaluate available local data, 4) evaluate data from elsewhere, and 5) identify probable causes. CADDIS was applied in the Soos Creek basin, WA (Marshallonis and Larson 2018). First, eight candidate causes were identified: flow alteration, increased fine sediments, reduced habitat complexity, elevated water temperature, low dissolved oxygen, elevated nutrients, increased ionic concentration, and toxic pollutants. Then conditions within the Soos Creek basin were compared to conditions at regional reference sites to understand how biological conditions at impacted sites (as defined by B-IBI) changed predictably along causal gradients. Flow alteration, increased fine sediments, and loss of habitat complexity were the most likely causes affecting biological condition.

In recent work, King County (2019) undertook an approach relying on multiple lines of evidence to identify key stressors in 14 stream basins in Puget Sound. The key steps in the process were to:

1. Identify potential stressor-related parameters (measures of the site or watershed condition) by evaluating correlations with measured B-IBI scores, and screening for plausible causal mechanism. 47 of the 147 parameters initially screened were correlated with B-IBI and had plausible causal relationships.
2. For each parameter, determine the range of values associated with “excellent” conditions. This was done in two ways:
 - a. Determine the interquartile range (IQR) of the parameter for all of the sites determined to be “excellent” based on B-IBI scores.
 - b. Evaluate the relationship between each parameter and macroinvertebrate communities with principal coordinate analysis (PoCA) and TITAN (Baker and King 2010).
 - i. PoCA is used to identify taxa groups associated with “very poor” → “excellent” sites based on B-IBI scores. The communities associated with “excellent” scores were defined as sensitive (negatively impacted by

⁴ A weight of evidence approach is a method that considers multiple sources of information and lines of evidence in an evaluation. It avoids relying on a single study or piece of information. This is a valuable approach when uncertainty may make it difficult to discern a clear outcome or solution from a single investigation.

- increasing stressor) and those associated with “poor” scores were defined as tolerant (positively impacted by increasing stressor).
- ii. TITAN is used to identify change points in the taxa groups associated with increasing stressors. Change point range for sensitive taxa defines the values of parameter for “excellent” sites.
 3. For each watershed, all measured parameter values are compared to IQR and TITAN change point range to identify those that are different from “excellent.” Those parameters are tagged as potentially impacting biological condition in the watershed.
 4. The final group of important parameters are then weighted based on the correlation (R^2 of regression) of B-IBI versus parameter. The most highly correlated parameters were selected as those that are potentially most important.

The result was a set of conditions in each basin that should be addressed to either maintain “excellent” condition, or improve from “fair” condition to “good” (King County 2019).

Several regional studies performed stressor evaluations based on an evaluation of relative risk/attributable risk, as outlined in Van Sickle and Paulsen (2008). This approach compares the co-occurrence of “poor” biological condition (as defined by B-IBI scores) and poor measures of various chemical and physical habitat stressors to determine which might be significantly related. Larson et al. (2019) performed a statewide analysis and concluded that poor substrate conditions were the most prevalent and important stressor associated with poor macroinvertebrate conditions in Washington streams. Important measures of substrate condition were percent sand/fines, relative bed stability, and embeddedness. Other stressors were also identified.

The Stormwater Action Monitoring program sampled 105 sites within Puget Lowland streams as part of a long term status and trends monitoring program, with a stated objective of understanding the major stressors impacting regional stream health (DeGasperi et al. 2018). They reported that low watershed canopy cover, low riparian canopy cover, and high watershed percent urban development were significantly associated with “poor” B-IBI condition. Sediment substrate, sediment zinc, embeddedness, and nutrients were also identified as risk factors, albeit not the most important risk factors regionally, with the possible exception of nitrogen.

King County (2014b) used water quality, sediment chemistry, and physical habitat data to investigate the relative importance of environmental stressors on benthic invertebrate metrics. Their results suggested stream sedimentation (percent fines, small gravel and cobble, sand-fines) and embeddedness, contributed the greatest attributable risk to B-IBI scores, and five of the ten individual B-IBI metrics. Biological indices were also sensitive to dissolved oxygen, total phosphorous, turbidity, and pH. Sediment chemistry was not identified as an important stressor, though measurements were based on analysis of whole sediment, which may not be suitable for evaluating effects on benthic invertebrates (King County 2018).

Mulvey et al. (2009) evaluated the relationship between biological condition and measures of water quality and habitat/land use in streams in the Willamette River basin utilizing the relative risk approach. They reported eight measures of water quality (temperature, Oregon water quality index, Total Suspended Solids [TSS], Total Phosphorus [TP], dissolved oxygen [DO], Total Nitrogen [TN], and pH) and four measures of habitat/land use (canopy cover, riparian vegetation, % sand/fines, streambed stability, and % fines) that were significant risk factors.

Stressor evaluation methods utilizing attributable risks and relative risks (Van Sickle and Paulsen 2008) are based on the assumption of independent stressor variables. Impacts from covarying stressors may lead to an overestimation of risks attributed to an individual stressor (Van Sickle 2013). Stressor covariation, natural spatial and temporal variation, and uncertainties with regard to stress-response relationships, among other complexities make clear stressor identification a challenge (Vander Laan et al. 2013). A weight of evidence approach for causal assessment is often the most informative (Marshallon and Larson 2018).

Herlihy et al. (2020) utilized data collected during US EPA National Rivers and Stream Assessment program to evaluate relationships between fish and macroinvertebrate MMI scores and 38 environmental factors utilizing random forest and multiple regression models, and a logistic regression model to identify variables associated with poor MMI condition. In the Western Mountain ecoregion, which includes the Puget Sound, total nitrogen, percent sand and fines, and a dam disturbance index were negatively associated with macroinvertebrate condition. Low minimum air temperature, latitude, and elevation were all positively associated with macroinvertebrate condition likely describing the locations of most undisturbed streams in more forested mountain areas.

Other statistical approaches have been used to evaluate the relationships of multiple stressors on invertebrate condition. Waite et al. (2012) compared several statistical models relating variables describing watershed condition or stressors with different measures of invertebrate condition. Boosted regression trees proved to be a good option for modeling species distributions. (Waite et al. 2019) utilized boosted regression tree models to identify major stressors on community condition in the southeastern United States; insecticide, dissolved oxygen, flow, and sediment contamination were important for invertebrate condition.

Leps et al. (2015) investigated the impacts of water quality, riparian and watershed land use, and stream morphology (21 metrics, total) on riverine benthic invertebrate communities, while also considering the spatial scales by performing a stepwise regression procedure using generalized linear models. Their analysis generally identified 5-6 parameters that described 30-50% of the variation in a given biological metric. High impact stressors included nutrients and water temperature; riparian land use was more important in small streams compared to larger rivers.

A brief review of other stressor identification efforts is presented in Table 12.

Table 12. Literature review notes - Stressor identification

Reference(s)	Key Findings
Larson et al. (2019)	Used Washington statewide stream data to identify condition and highlight key stressors through a relative risk/attributable risk approach. Poor substrate condition was the most prevalent and important stressor associated with poor macroinvertebrate conditions in Washington streams. Important measures of substrate condition were percent sand/fines, and relative bed stability, and embeddedness. Other stressors were also identified.
King County (2019)	Objective was to identify key stressors affecting macroinvertebrate communities in 14 watersheds in Puget Sound. Used two approaches to elucidate characteristics of high quality reference streams and then identify measures in each of the 14 basins that were different than “excellent” reference sites.
Marshallonis and Larson (2018)	A CADDIS framework was applied in the Soos Creek basin, WA. Conditions within the Soos Creek basin were compared to conditions at regional reference sites to understand how biological conditions at impacted sites (as defined by B-IBI) changed predictably along causal gradients. Flow alteration, increased fine sediments, and loss of habitat complexity were the most likely causes affecting biological condition.
Ofenböck et al. (2004)	Developed a set of four multi-metric indices that respond predictably to specific stressors including in stream sites in Austria. Stressors included channelization, channel alteration, organic pollution, and impoundments.
Vander Laan et al. (2013)	Utilized a random forest approach to investigate links between biological condition – stressor in streams in Nevada, USA. Comparisons of responses based on an Observed: Expected index and multi metric index suggested they responded differently to stressors, indicating that selection of measure of biological condition may bias outcome. TDS and metals contaminated were most strongly associated with biological degradation.
Wooster et al. (2012)	Metrics based on taxonomic and biological traits were evaluated to determine whether they would respond predictably to different stressors (stream channelization vs water withdrawals). Metrics based on biological traits were capable of differentiating between stressors.
Leitão et al. (2018)	Reference provided as an example of potential approaches. Performed multi-scale assessment of the biological condition of streams in the Amazon to understand the functional response of fish assemblages to land use. Characterized fish assemblages using eco-morphological traits describing feeding, locomotion, and habitat preferences. Characterized habitat attributes and landscape-change variables including density of road crossings, deforestation, and agricultural intensification. Used structural equation modeling to identify causal pathways that significantly affect stream condition and the structure of the fish assemblages.

6.3 Effectiveness of Stream Recovery, Low Impact Development, and Agriculture-related Recovery Approaches

This section summarizes what is known about the effectiveness of some of the protection and restoration strategies described in the B-IBI Implementation Strategy.

6.3.1 Stream Restoration and B-IBI Recovery

There is limited information directly relating stream restoration with changes in benthic community structure or composition. While in-stream or riparian restoration work often achieves the goals of restoring more natural hydro-morphological conditions, significant improvements in ecological condition, such as benthic macroinvertebrate assemblages, are often not apparent despite significant time, effort, and cost (Alexander and Allan 2006, 2007, Bernhardt et al. 2005, King County 2012, Stranko et al. 2012). Because investments are high, the expectations of ecological improvement may also be high.

Various studies have been conducted to evaluate the macroinvertebrate response to restoration, with mixed results (Miller et al. 2010, Palmer et al. 2005). Studies have suggested that uncertain or variable macroinvertebrate response to stream restoration activities may be due to insufficient pre- and post-monitoring or poor experimental design (Alexander and Allan 2007, Miller et al. 2010, Palmer et al. 2005), insufficient spatial extent of projects (Feld et al. 2011), or the failure of key taxa to recolonize after physical conditions were restored (Lorenz and Feld 2013, Tonkin et al. 2014).

Key recommendations from the literature when using B-IBI for stream restoration and recovery planning include:

- An empirical understanding of the landscape-scale processes influencing the stream ecosystem should be integral to restoration planning and implementation.
- Preservation of high-quality, minimally impacted forest, and restoration efforts at the watershed scale are the best options to preserve healthy stream biota.
- To quantify links between restoration actions and macroinvertebrate response, rigorous study designs are needed with pre- and post-monitoring sampling, replication, and collection of biotic and abiotic responses.
- For highly urbanized or fragmented reaches, create "stepping stones," or smaller restored patches within a degraded system, to support the recolonization of macroinvertebrate communities.
- If restored streams are isolated or disconnected from high quality streams with diverse macroinvertebrate assemblages, active recolonization efforts may be needed to jump start recovery.
- The implementation of localized green stormwater infrastructure alone is unlikely to improve stream macroinvertebrate recovery without additional treatment and reduction of impervious surfaces.

A summary of literature related to effectiveness is included in Table 13.

Table 13. Literature review notes — Effectiveness of stream restoration projects

Reference(s)	Key Findings
Miller et al. (2010)	<p>Performed a meta-analysis on effectiveness of stream restoration on macroinvertebrate richness and density. High within-study variability and low statistical power, common to macroinvertebrate studies, have caused some to question their use to detect reach-scale restoration responses.</p> <p>Data limitations were a broad issue in evaluating effectiveness. Although, a majority of replicated studies have enough statistical power to detect significant changes in species richness after restoration is complete. Rigorous study designs are needed with pre- and post-monitoring sampling, replication of samples, and collection of biotic and abiotic responses (water quality, habitat, and biota) to account for spatial and temporal variability.</p>
Leps et al. (2016)	<p>Investigated 44 restoration sites in Germany to determine whether the short time span for monitoring was the reason that they did not see evidence of biological recovery following restoration work (i.e., recovery time scale was longer than monitoring). This lack of time for community response did not appear to explain lack of recovery of benthic invertebrate communities. Instead, catchment-scale characteristics override the effectiveness of restoration.</p>
Bernhardt and Palmer (2011)	<p>Reviewed literature on limits of river restoration. Reach scale channel restoration efforts do not appear effectively mitigate the physical, hydrological, or chemical alterations that are responsible for the loss of sensitive taxa and declines in water quality. Spatial context may be one of the most important factors controlling stream restoration outcome. As such, restoration of streams and rivers should not be expected to alleviate problems generated throughout a catchment. Future efforts must shift from a focus on geomorphic structure and form to a focus on restoration of the hydrologic, geomorphic, and biological processes that maintain healthy stream ecosystems.</p>
Alexander and Allan (2007)	<p>Reviewed results of reviewed restoration work in Michigan, Ohio, and Wisconsin. Studies have suggested that variable macroinvertebrate response to stream restoration activities may be due to insufficient pre- and post-monitoring designs or poor experimental design. A variety of projects are implemented to address identified environmental concerns, without addressing the large-scale processes that contribute to the observed degradation. This produces a variety of projects implemented to address identified environmental concerns, without addressing the large-scale processes that contribute to the observed degradation.</p>
Bernhardt et al. (2005)	<p>Developed database of >37,000 river restoration projects across US. There are limited data to quantify links between restoration actions and ecological responses; fewer than 10% of restoration projects involved any post-restoration monitoring.</p>
Palmer et al. (2005)	<p>Proposed criteria for restoration projects including: the river's ecological condition must be measurably improved, the system must be more self-sustaining and resilient to external perturbations so that only minimal follow-up maintenance is needed, and both pre- and post-assessment must be completed and data made publicly available.</p>
Brooks et al. (2002)	<p>Assessing stream ecosystem rehabilitation: Limitations of community structure data “With extremely high variability between replicate riffles, monitoring programs for localized restoration projects or point source impacts are unlikely to detect gradual shifts in community structure until the differences between the reference and treatment sites are extreme.”</p>

Reference(s)	Key Findings
Stranko et al. (2012)	Compared measures of biological diversity at restored urban sites, unrestored urban sites, and reference sites in US mid-Atlantic region. Diversity of restored urban streams is about the same as unrestored urban streams, which is less than non-urban or reference streams. Diversity decreased at non-urban streams over time, coincident with development.
Lorenz and Feld (2013)	Analyzed data from 46 reach-scale restoration sites in Germany. Distance to the nearest colony is a critical factor in restored reaches, with colonization rates declining with increasing distance. Habitat conditions close to the restoration are the most important for recolonizing macroinvertebrate communities after restoration is complete. With appropriate planning, maintenance of metapopulations could be enhanced between restored patches.
Heino et al. (2004)	Studied sites at six streams in Kiiminkijoki river system in Finland. "Stream biomonitoring programs should consider scale-dependent variability in assemblage characteristics because: (i) small-scale variability in abundance suggests that a few replicate samples are not enough to capture macroinvertebrate assemblage variability present at a site, and (ii) riffles from the same stream may support widely differing benthic assemblages."
Violin et al. (2011)	Compared four urban streams, four restored streams, and four forested streams in the Piedmont region of North Carolina to evaluate effects of restoration projects. Restored and urban streams were indistinguishable when comparing reach and patch-scale attributes. Reach scale restoration does not mitigate factors causing biological degradation.
Wahl (2012)	Thesis on land use, riparian buffers and biological conditions in the Puget Sound Lowlands and effectiveness of riparian restoration. Results suggest that ecological improvements from riparian buffers may be overwhelmed by large-scale influences in the watershed even when physical habitat variables such as water clarity, substrate and temperature are significantly improved.
King County (2020)	King County is evaluating the effectiveness of recolonizing four streams that lacked many sensitive invertebrate species (i.e., had low B-IBI scores) with macroinvertebrates from health streams. One year post seeding at least one new taxa was present in each of the four streams that had not been present prior to the seeding. B-IBI scores increased in two of the four streams. The final report is pending: https://kingcounty.gov/depts/dnrm/wlr/sections-programs/science-section/doing-science/aquatic-insect-seeding.aspx
Morley et al (2018)	Restoration of a stream hyporheic zone was performed on Thornton Creek. Invertebrate density and taxa richness were higher at restored than at unrestored reaches, and were comparable to forested reference reaches. One restoration site was seeded with invertebrates from a forested reach and this may have resulted in the establishment of four new taxa at the seeding site. https://cedar.wvu.edu/ssec/2018ssec/allsessions/469/
Jourdan et al. (2019)	Review of literature and reports suggest that reintroduction of freshwater macroinvertebrates is rare and appears to fail approximately one-third of the time. Life-cycle complexity and remaining stressors are the two factors most likely to affect reintroduction success. The risk of transferring non-target species should be considered prior to reintroduction.

6.3.2 Effectiveness of Low Impact Development

Low-impact development (LID), sometimes called green stormwater infrastructure (GSI), is an approach for stormwater management that utilizes on-site features to recreate natural hydrologic processes of infiltration, filtration, storage, evaporation, and transpiration and mimic natural stormwater runoff patterns in developed drainage areas. Some examples of low-impact technologies, such as native vegetation, permeable pavement, and rain gardens, minimize impervious surfaces and reduce stormwater runoff volume while improving the quality of stormwater runoff that enters the stormwater system.

No studies were found that directly link LID/GSI approaches and benthic invertebrate communities. However, LID/GSI has been shown to reduce runoff volumes, decrease flashiness, and reduce sediment loads from impervious areas. As these have been identified as key stressors, it is reasonable to conclude that LID/GSI could reduce the magnitude of impacts. However, few studies have quantified the impacts of LID on hydrology and water quality on a regional/watershed scale. This is complicated by both the variations between watersheds, but also that LID installations utilize different processes for stormwater control – infiltration, detention, harvest, and/or evapotranspiration – that produce different hydrological outcomes (Jefferson et al. 2017).

Jefferson et al. (2017) performed a review of studies that evaluate the impacts of stormwater management activities at a watershed scale. They reported that addressing impervious areas may not restore watershed function to predevelopment condition because only a fraction of runoff is typically treated and the impacts of individual BMPs may not be additive. Additionally, other urban effects beyond impervious surfaces, such as vegetation loss, altered/compacted soils, and habitat loss, also affect watershed condition. With regard to pollution loads, decreases largely result from reductions in runoff volume, rather than lowered contaminant concentrations (Jefferson et al. 2017). The removal of organic pollutants through LID approaches such as biofilters and grass strips is variable, with generally high levels of removal of particulate-bound pollutants and mixed removal of dissolved constituents (Boehm et al. 2020). Biofiltration systems reduced event mean concentrations of the herbicides atrazine, simazine, and prometryn by -7 to 58% (Zhang et al. 2016). Vegetated buffer strips (1.5 m wide x 20.1 m long) retained approximately 50-80% of mass of atrazine, metolachlor, and chlorpyrifos, with the mass reduction being associated with infiltration and not a reduction in chemical concentration (Arora et al. 2003). Bioswales were reported to reduce the concentration of pyrethroid pesticides, though removal effectiveness against fipronil varied (Anderson et al. 2016). The toxicity of urban stormwater towards invertebrates was markedly reduced after filtration through soil media (Anderson et al. 2016, McIntyre et al. 2015). The lack of long-term performance data for stormwater BMPs improving water quantity and quality has also been noted by Liu et al. (2017). Modeling has been suggested as a valuable approach to scaling localized LID/GSI practices to the watershed scale (Golden and Hoghooghi 2018).

A review of the effectiveness of select LID approaches on watershed processes is presented in Table 14.

Table 14. Literature review notes - watershed scale effectiveness of LID/GSI in mitigating pressures or stressors

Reference(s)	Key Findings
Pennino et al. (2016)	The cumulative impacts of LID infrastructure were assessed in three large urban areas in the Mid-Atlantic US (Baltimore County, MD, Montgomery County, MD, and Washington, DC). When controlling the watersheds for size and percent impervious surface cover, watersheds with greatest amounts LID infrastructure were found to have less flashy hydrology, lower peak stormwater runoff, less frequent runoff events, and less variable runoff. Reductions in phosphorous or CSO events were not produced.
Roy et al. (2014)	In Cincinnati, OH, rain barrels and rain gardens were installed on 30% of parcels in four subcatchments, while two control subcatchments were used as controls. There was no observed improvement of macroinvertebrate assemblages three years after implementation. The researchers concluded that without the treatment of major impervious surfaces, such as roads, apartment buildings, and parking lots, improved macroinvertebrate response would be unlikely.
Hur et al. (2008)	Comparison of runoff hydrology and water quality parameters from developing basins with stormwater controls and undeveloped basins indicated that the use of stormwater BMPs did not provide runoff or water quality benefits envisioned by regulations. The cause for lack of effects was not clearly identified.
Yang and Li (2013)	Watershed scale community GSI, e.g., the incorporation of on-site infiltration systems, can effectively reduce runoff volumes, flashiness, and nutrient exports compared to basins without stormwater controls.
Gold et al. (2017)	Watershed-scale implementation of wet ponds in the developed watershed in the coastal environment failed to mitigate many negative water quality impacts caused by increased development. Wet ponds in developed watersheds were exporting chlorophyll-a and total suspended solids to the stream, and were sinks for nitrate-nitrite. Continual maintenance may improve sediment retention.
Jarden et al. (2016)	Parcel scale GSI retrofits that include treatment of street run-off can substantially reduce stormwater even at participation rates of ~14%. However, even small differences in design and construction can affect the level of observed benefits.

6.3.3 Effectiveness of Agricultural BMPs

The B-IBI Implementation Strategy includes a preliminary strategy for addressing impacts of working lands (i.e., lands used for forestry or agriculture) on stream health, but indicated that a better approach for working lands would be to develop a holistic strategy covering recovery and restoration for salmon habitat, flood plains, and estuaries, etc. The development of a holistic agriculture strategy would likely take several years to complete. As such, two preliminary strategy elements were identified: increasing incentives for agricultural BMPs and

reducing farmland conversion. A brief summary of the effectiveness of agriculture BMPs is presented here with a focus on biological impacts.

The Water Research Foundation has invested substantial effort reviewing and summarizing the effectiveness of agricultural BMPs at reducing nutrient and sediment loads (<http://www.bmpdatabase.org/agBMP.html>). A detailed review of individual studies is presented in their work and will not be repeated here. Their review provided three primary conclusions:

- BMPs can reduce contaminant loadings. For example, no-till and conservation tillage, or cover crops, can reduce sediment runoff compared to conventional tillage.
- Variations among study areas such as soil type, slope, weather conditions (e.g., wet year, drought), drainage practices, edge-of-field practices make it challenging to effectively analyze and compare agricultural research data.
- A “one size fits all” approach to managing water quality issues in agriculture areas is not realistic.

Another recent review focused on the effects of agricultural BMPs on aquatic ecosystems (Kroll and Oakland 2019). Kroll and Oakland first reviewed the effectiveness of agricultural BMPs on various biophysical parameters and reported challenges similar to those found by the Water Research Foundation work, that variations in study sites and conditions made it difficult to support a simple conclusion. For example, they found that livestock BMPs (e.g., exclusion fencing, buffers) reduced TSS concentrations in runoff by 0-90%; cropland BMPs (e.g., cover crops, conservation tillage) reduced TSS concentration by 0-51%. Additionally, few studies were able to translate the changes in physical condition to improvements in biological condition. The majority of the efforts were based on the presumption that a reduction in stressors would improve biological condition, though this was rarely supported by measurement. They did conclude that improving/increasing riparian buffers and cattle exclusion might be both reasonable and effective.

6.3.4 Effectiveness of Forestry BMPs

The B-IBI Implementation Strategy included a preliminary strategy for working lands and identified increasing incentives and technical assistances for forestry BMPs as a key priority. Forestry in Washington State is guided by [Forest Practices Rules \(Title 222 WAC\)](#) as described in the [Forest Practices Board Manual](#). Guidelines focus on application of fertilizers and chemicals, and road and riparian management. Some, but not all, of these practices have been shown to be effective at controlling some stressors to streams and the aquatic ecosystem (Ecology 1999). Buffers, fill slope construction, and relief drains were found to be effective at reducing sediment delivery, while many other BMPs and maintenance practices were not.

Washington State has a regulated forest management system with activities managed largely under the Forest Practices Act (Ice et al. 2004). Forest practices rules (FPRs) have been

repeatedly revised since their establishment in 1975. A survey by Washington Department of Natural Resources indicated that overall compliance with forest practice rules, including riparian management, riparian management, and road-related compliance ranged from 87-100% (DNR 2018).

Warrington et al. (2017) performed a literature review and reported that, in general, forestry BMPs reduce sediment loads and help protect riparian structure, though data directly relating forestry BMPs and riparian or aquatic species are limited. BMPs focusing on forest road construction and maintenance, stream crossings, and buffers are particularly important.

Cristan et al. (2016) reviewed studies on forest practices implementation in the southern, northern, and western United States and concluded that forestry BMPs can significantly improve water quality and reduce sediment loading, assuming the BMPs are correctly and sufficiently implemented.

Data on the extent of BMPs that are being properly implemented are scarce.

6.4 Multiple Benefits of Stream Restoration/Recovery Activities and Programs

As described above (section 6.3.1 and Table 13) localized stream restoration programs and activities may not result in measurable improvements in B-IBI. However, there are other potential benefits (other than improvements in B-IBI) that should be considered when evaluating outcomes. For example, Fletcher et al. (2014) identified flood control, water supply improvements, and social amenities as identifiable benefits; several of these were specifically identified for the local restoration activities (Morley et al. 2013). Recreational and aesthetic benefits of stream restoration have also been identified as important outcomes. Kenney et al. (2012) indicated that the costs of some stream restoration projects could be justified if the valuation of aesthetics and recreation were included as quantifiable benefits. In many cases, the recovery and restoration activities will provide many of these stated benefits even without directly changing the stream biology as measured by B-IBI. Multiple members of the Interdisciplinary Team indicated that this was important and that focusing solely on B-IBI might limit the real and perceived successes of a restoration program.

6.5 Effectiveness – Policies and Programs

The B-IBI Implementation Strategy identified education and incentive programs as a key strategic area for improving stormwater management and stream recovery and protection. The effectiveness of existing programs, and education and incentive programs in general, is of interest to the implementation of the strategies.

Two focused literature reviews are included in the Base Program Analysis. The first includes the effectiveness of education programs for behavior change with a focus on environmental and ecosystem management. The second review covers the effectiveness of incentive programs with a focus on improved stormwater management.

6.6 Cost Considerations

Costs and the cost-benefits of protection and mitigation activities, such as stormwater treatment and control, and stormwater funding availability were identified as important considerations. In response, a review of the literature was performed focusing on cost benefits analyses for various approaches.

6.6.1 Cost Comparisons of Stormwater BMPs

A literature review was performed focusing on papers that presented information on the cost and effectiveness of stormwater BMPs. The review covers stormwater BMPs and includes GSI and LID approaches.

The studies used different methods of analyzing cost and performance of BMPs and covered different study areas; as such results and findings that were not consistent. Differences included a) criteria and scope of the life cycle assessments/analysis (LCA), b) study location (i.e., rural vs. urban, roadway/street vs. commercial/residential), c) evaluating single BMPs or combinations of BMPs, and 4) cost or performance measures. Studies included capital cost only (two of ten studies), or capital costs and operating and maintenance (O&M) costs (eight of ten). Three of the ten studies utilized LCA methods. Seven of ten measured performance by assessing reduction in stormwater volume. Three of those seven additionally measured reduction of pollutant (phosphorus, nitrogen, and sediments) loads.

A summary of selected studies is in Table 15. The inconsistencies between studies suggest that further work is needed to normalize approaches, and that comparisons should be approached cautiously. There are a number of ways to perform these assessments. The review provided below might provide examples of choices made, and approaches for designing future cost evaluations. Every cost-benefit analysis will likely be unique, because every cost-benefit objective is specific to the problem being evaluated.

Table 15. Literature review summary - cost effectiveness/comparisons of stormwater BMPs

Citation	Background	Key Findings
Chui et al. (2016)	<ul style="list-style-type: none">Objective: Identify the optimal designs of three BMPs (green roof, bioretention and porous pavement); assess the hydrological performance, and cost-effectiveness.Study sites were Hong Kong, China and Seattle, WA, USA under design storms of 2 year and 50 year.Costs included land, construction, and O&M. All costs were in dollars per m².Bioretention and porous pavement were assumed to have zero land cost as they are incorporated into existing or mandatory infrastructure. Opportunity cost in green roofs in Hong Kong were included as the roofs could be sold.	<ul style="list-style-type: none">Porous pavement was the least expensive with highest cost-effectiveness. The costs increased with design storm. The optimal area, coverage percentage, and thickness was found to be smaller than green roof and bioretention. Porous pavement had the lowest unit construction and O&M costs.Bioretention was second most expensive. The cost of coverage increased with design storm. O&M costs were highest.Green roofs were the most expensive. The cost of green roofs in Hong Kong were 55% more than in Seattle due to the cost of land. Discounting the cost of land, green roofs were still found to be the most expensive.

Citation	Background	Key Findings
Mao et al. (2017)	<ul style="list-style-type: none"> Modeled evaluation. Utilized EPA's System for Urban Stormwater Treatment and Analysis INtegration (SUSTAIN) to model the ecological benefits of aggregate BMPs in a 22 km² area in Foshan New City, China. Evaluated performance of four combinations of aggregated BMPs in four different watersheds in terms of flow reduction of stormwater and reduction of pollutants. Performance target was 60% reduction in annual flow volume. Three scenarios were evaluated – pre-development, post-development, and a post-development with BMPs. Treatment scenarios included four different combinations of six BMPs: rain barrels, green roofs, bioretention, porous pavement, swale, and wetpond. Costs included only capital costs – no O&M was included – on a cost per m² basis. 	<ul style="list-style-type: none"> The more effective combinations of BMPs (those that resulted in highest reductions of stormwater volume and pollutant load) were estimated to cost \$1.1 M to cover the study area. The most expensive were wet ponds and bioretention, neither of which were best performers. System effectiveness was: 31-42% decrease in flow volume, 50-67% COD reduction, 64-79% SS reduction, 62-75% TN reduction, and 58-73% TP reduction. The systems that had the highest performance in reducing flow volume (but never reached the optimal flow rate) and reducing pollutant load included rain barrels, green roofs, and porous pavement. Comparison of scenarios with and without BMPs indicated that the presence of BMPs reduced overall flow by 120% and reduced all pollutants. The combinations of BMPs were more effective than grey infrastructure despite not reaching the optimal flow volume.
Joksimovic and Alam (2014)	<ul style="list-style-type: none"> Modeled evaluation Evaluated a series of stormwater BMPs at a 30 ha greenfield site in London, Ontario, Canada. Modeled capital cost vs percentage runoff reduction for various BMP combinations. Estimated runoff volumes based on local rainfall data and land use using the Personal Computer Storm Water Management Model (PCSWMM) model. Evaluated 11 combinations of 6 BMPs (green roof, porous pavement, infiltration trench, bioretention cell, vegetative swale and rain water harvesting) <ul style="list-style-type: none"> a. Vegetative swale & porous pavement b. Vegetative swale & green roof c. Porous pavement & green roof d. Porous pavement & rain water harvesting e. Infiltration trench & porous pavement f. Infiltration trench & green roof g. Bioretention & green roof h. Bioretention & rain water harvesting i. Bioretention & porous pavement j. Bioretention & porous pavement & green roof 	<ul style="list-style-type: none"> The combination of infiltration trench and porous pavement (e) provided greatest reduction in runoff volume (90%). Capital costs were \$6 million. Overall range of capital costs were from \$0-\$8 million. The highest unit costs were for green roofs alone at \$269 per m³, followed by porous pavement & green roof (c) (\$200 per m³) and then porous pavement & rain water harvesting (d) (\$158) per m³. Rainwater harvesting was the least expensive at \$14 per m³ but was only applied to single-family households in the study area. Bioretention & porous pavement & rainwater harvesting (k), bioretention & porous pavement (i), and bioretention & porous pavement & green roof (j) achieved 80% reduction with capital costs ranging from \$7-8M. The mid-range option of 50% runoff reduction and mid-range cost (\$4.75M) was porous pavement & vegetated swales (a). The study found that all of the BMPs increased infiltration and decreased runoff, although none of the options was capable of

Citation	Background	Key Findings
	<ul style="list-style-type: none"> k. Bioretention & porous pavement & rain water harvesting Costs included only capital costs – no O&M was included – on a cost per m³ runoff reduction basis 	<ul style="list-style-type: none"> fully restoring the scenario to pre-development levels.
Sun et al. (2016)	<ul style="list-style-type: none"> Modeled evaluation Used EPA's SUSTAIN model to analyze hydrology and BMP cost-effectiveness in a Las Vegas watershed. The model simulated the effectiveness of BMPs under projected future land-use and climate conditions from present to 2050. The current three detention basins installed in the watershed provide a 9% flow reduction. The study used SUSTAIN to determine the number of additional BMPs required, the optimal BMP types, and their locations for these future scenarios. The study analyzed detention BMPs and infiltration BMPs Costs included only capital costs – no O&M was included 	<ul style="list-style-type: none"> Results indicated that a mixed implementation of one additional detention BMP and one infiltration BMP to the existing detention BMPs was most cost-effective solution reducing 27% flow and costing \$1.14M. Surface flow was reduced by 13-46% depending on locations and BMP combination and ranged in price from \$390k to \$1.78M. This study found that when considering cost effectiveness, life span, and maintenance, it is more advantageous to install a detention BMP with an infiltration BMP, but with the detention BMP installed first.
Nordman et al. (2018)	<ul style="list-style-type: none"> The study used a cost-benefit approach to compare green vs. gray infrastructure in Grand Rapids, Michigan, USA. A benefit transfer approach was used to estimate the net present value (NPV) of capital and O&M costs vs direct and indirect benefits. Evaluated six BMPs: porous pavement, green roofs, rain gardens, infiltrating bioretention basins, conservation of natural areas, and street trees. Costs included installation, maintenance, and opportunity costs compared to the reduction in stormwater volume and reduction in total nitrogen (TN) and total phosphorous (TP), scenic amenity value added, and CO₂ storage. A 3.5% discount rate was applied for 50-year life cycle. Additional considered benefits included energy savings and increased scenic amenity value from green roofs, and increased property values from rain gardens and streets trees. 	<ul style="list-style-type: none"> Conserved natural areas had the highest benefits NPV of \$109 per m³ of water quality volume (WQv) reduced, followed by street trees at \$46 per m³ WQv reduction, rain gardens at \$37 m³ WQv, and porous asphalt at \$21 m³ WQv. Infiltrating bioretention basins and green roofs had negative NPVs of -\$4 per m³ WQv and -\$47 per m³ WQv reduced, respectively. Under a simulated "best case" scenario, all BMPs have positive NPVs. In a "worst case" simulation only rain gardens, conserving natural areas, and street trees had positive NPVs.
Osouli et al. (2017)	<ul style="list-style-type: none"> Modeled evaluation Evaluated BMPs effectiveness at retaining 95% of 25.4mm (1 inch) of precipitation per 30m of an 8-lane highway in Illinois with 	<ul style="list-style-type: none"> The construction costs of bioswales is \$16,291 per 30 m of highway, infiltration trenches is \$4,379 per 30 m of highway and a

Citation	Background	Key Findings
	<p>Personal Computer Storm Water Management Model (PCSWMM).</p> <ul style="list-style-type: none"> Evaluated bioswale, infiltration trench and vegetated filter strip. This study includes construction and O&M costs. 	<p>vegetative filter strip \$207 per 30m of highway.</p>
Weiss et al. (2007)	<ul style="list-style-type: none"> Analyzed six BMPs removal of TSS and TP over 20-year span as a function of the water quality volume (WQv) in m³. Stormwater volume reduction was not included. BMPs included: dry extended detention basins, wet/retention basins, constructed wetlands, infiltration trenches, bioretention filters, and sand filters. Costs included construction and O&M, but not land acquisition. Costs were calculated as the total present cost and included a 67% confidence interval. 	<ul style="list-style-type: none"> The most effective BMP at removing TP were bioretention filters. The most effective BMP at removing TSS were infiltration trenches. Constructed wetlands were the least expensive to construct and maintain if appropriate land is available. The study showed that the normalized costs of constructed wetlands, sand filters, and dry extended detention basins decrease as more WQv (m³) is treated versus the other BMPs. Costs of infiltration trenches increased steadily regardless of the amount of WQv filtered. Bioretention filters increasing slightly as more WQv was filtered.
Brudler et al. (2019)	<ul style="list-style-type: none"> Evaluated stormwater mitigation in a 260 ha catchment in Odense, Denmark, over 25 years. Assessed the reduction of pollutant flow in stormwater and emissions produced and resources used associated with the life cycle of the infrastructure. Compared existing WWTP efficiency in pollutant removal compared to several BMPs: surface basin, sand trap, green strip, swale, and infiltration trench. Environmental impacts compared with LCA. Included material generation, construction, O&M, and end of life (decommissioning and disposal or recycling). The environmental impacts modeled were ecosystem impact (as species loss and point source emissions) and future natural resource availability (costs of future resource extraction). 	<ul style="list-style-type: none"> The WWTP was most effective in treating copper and phosphorus. The infiltration trench was most effective in treating zinc. A combined sewer system was found to cause the highest resource impacts but the lowest point source emission impacts. The BMPs had low resource impacts but their relative point source emissions were higher as they removed less stormwater pollutants. The traditional stormwater infrastructure cost up to \$8,800 annually in terms of "resource availability" (a net cost of resources such as pipes and plastics used during construction), while the infrastructure made of BMPs produced a "resource availability" benefit of up to \$5,200 annually because of prevented damages.
Byrne et al. (2017)	<ul style="list-style-type: none"> Used LCA to determine the environmental impacts of roadway drainage systems in the Midwestern United States. Analyzed bioswales, grass swales and storm sewers. The study calculated a) total system emissions b) local aquatic and soil emissions. Water quality was not included. 	<ul style="list-style-type: none"> Grey infrastructure/storm sewers had the highest cost and highest climate change impacts. Storm sewers had 3x the environmental impact as bioswales and 12x the impact as grass swales. The construction, maintenance, and end of life disposal of storm sewers resulted in the highest impacts on direct aquatic and soil emissions.

Citation	Background	Key Findings
	<ul style="list-style-type: none"> Costs included construction and O&M. 	<p>Bioswales contributed the second highest impact and grass swales the least impact.</p> <ul style="list-style-type: none"> Grass swales were the most cost effective at 1/8th the cost of storm sewers and 1/7th the cost of bioswales. Bioswales did not have a cost advantage over storm sewers. When costs and environmental impacts were normalized for length and flow capacity at the studied roadway, it was found that bioswales resulted in larger normalized cost and climate change impacts than storm sewers.
Petit-Boix et al. (2015)	<ul style="list-style-type: none"> Evaluated environmental and economic impacts of a pilot filter, swale and infiltration trench systems in São Carlos (Brazil) designed for flood prevention. Utilized LCA and Life Cycle Costing (LCC) per ISO 2006. It included all construction, transport, demolition and end-of-life. The present value of 2009 dollars was used for the cost analysis. The measurement indicators evaluated in the study were climate change (as kg-CO₂-eq.), ozone depletion potential, human toxicity potential, photochemical oxidant formation potential, terrestrial acidification potential, freshwater eutrophication potential, marine eutrophication potential, water depletion potential, metal depletion potential and fossil depletion potential. 	<ul style="list-style-type: none"> The infiltration trench had the greatest emissions and the greatest material and energy requirements. The grass cover/bioretention filter had high contributions in four indicators but there were avoided climate change emissions due to carbon sequestration. The impacts of the FST as linked to the total stormwater infiltrated in a year were variable The FST system reduced stormwater runoff by 80–95%.
Herrera Environmental Consultants (2013)	<ul style="list-style-type: none"> Modeled evaluation Compared costs of implementing the minimum stormwater requirements for new development under <i>2012 Stormwater Management Manual for Western Washington</i> requirements, with the 2005 manual. Evaluated 14 illustrative scenarios covering 10-acre single-family residential, 1-acre commercial, and 10-acre commercial development. Includes two soil types. Available system types included: bioretention, wet pond, combined detention and wetpool, planter vault, infiltration basin, catch basin, permeable sidewalk, permeable pavement, impermeable pavement Costs included installation and O&M. 	<ul style="list-style-type: none"> Permeable pavement had the lowest cost per square foot measured in 30-year costs at \$1.16 per sq. foot compared to Bioretention at \$21.84 per sq. foot. For single-family residential units – costs of compliance increased under 2012 requirements compared to 2005 without LID principles. Costs decreased if LID is included. For small and large commercial – costs of compliance decreased under 2012 requirements compare to 2005. Decreased costs were associated with reduced O&M.

6.6.2 Cost Comparisons of Agricultural BMPs

The costs and cost effectiveness of agricultural BMPs is dependent on factors such as pollutant, land and watershed configuration, land costs, and opportunity costs. Standardized approaches have been suggested for performing cost assessments for Best Management Practice (BMP) installations (Tyndall and Roesch 2014). There are several investigations evaluating the cost effectiveness of various approaches for sediment control. Yuan et al. (2002) evaluated several BMPs for sediment control for row crops and suggested that maintaining cover crops and installing edge-of-field grade control pipes would be most efficient. Zhou et al. (2009) suggested that no-tillage conservation measures could actually provide sediment control at a net positive return on investment if the value of the avoided soil loss was included in the analysis. Their results suggest an overall benefit of \$95 per hectare per year from this approach. Other analyses have indicated that in-field contour strips can be a cost effective approach to reducing sediment and nutrient runoff from row crops, suggesting that long-term conservation reserve programs could be utilized to address opportunity costs to the farmers (Tyndall et al. 2013).

Others have proposed methodologies for identifying the most cost-effective mix of sediment control BMPs on a watershed scale. Smith et al. (2014) utilized the Soil and Water Assessment Tool (SWAT) to predict edge-of-field loading following the implementation of one of a set of potential sediment-control BMPs. Costs were determined based on lost production and costs for BMP implementation over 15-years. Key findings were : 1) considering sediment runoff potential, or costs alone, would result in markedly different outcomes than if both costs and benefits were considered together, and 2) that the random BMP implementation, which is representative of a policy where funding is provided to interested and willing landowners, is not cost-effective. This supports the notion that watershed planning based on cost-benefit analysis can improve the efficacy of investments.

6.6.3 Overall Cost Benefit of Stormwater Management

The results of several studies generally support the notion that there is a net positive value from investments in stormwater management in that benefits (including avoided costs) largely outweighed investments. However, few studies have provided a holistic evaluation and there remains a high degree of uncertainty in quantification. Costs of investments can be determined with some certainty through a careful evaluation of investments by state and local jurisdictions. Many benefits of stormwater management, particularly those associated with green stormwater infrastructure have been identified but are often diffuse and more difficult to quantify. For example, Visitacion et al. (2009) studied the overall costs and benefits of stormwater management in the Puget Sound region. They found that investments in stormwater management vary by jurisdiction, ranging from \$10s-\$100s per capita. Benefits included avoided damage from flooding, degradation of water quality, loss of habitat, and losses of natural resources (such as fish). Benefits were presumed to exceed costs, but were not explicitly quantified. A critical and unverified assumption in this (and similar) analyses was that stormwater management investments actually mitigated negative impacts as intended.

However, it is not clear that stormwater investments are always effective. Ineffective investments would obviously have a negative cost-benefit ratio.

Braden and Johnston (2004) identified several benefits to onsite stormwater management: reduced frequency and magnitude of flooding; lower costs of drainage infrastructure; reduced pollution; reduced erosion and sedimentation of stream channels; improved in-stream condition and aesthetics; and, increased groundwater recharge. They concluded, however, that the valuation of these benefits were difficult to generalize, and varied markedly according to factors such as the value of infrastructure in a floodplain (for which damage avoidance benefits would accrue).

In a recent example, Hellman et al. (2018) estimated the costs to downstream residence of additional runoff from a new development to be on the order of \$12,000 per 10,000 ft³ of runoff. They compared these costs to estimated mitigation costs estimated from the literature of around \$109 per 10,000 ft³ managed, suggesting that it would be “economically efficient” to invest in the stormwater control structure. However, only capital costs were included and so real and total costs might exceed benefits.

Netusil et al. (2014) evaluated the costs of stormwater-related water quality issues on property values in a set of urban streams in the Portland-Vancouver metropolitan area. They utilized a hedonic price method (an approach often used to value costs or benefits associated with environmental quality by estimating its effects on property values; e.g., how much do property values change according to air pollution?) to investigate the relationship of five water quality parameters to housing prices at increasing distances from the streams. They reported that improving dissolved oxygen levels by 1 mg/L would result in an increase of 3-13% in property values, depending on distance, though the estimated increases varied between watersheds. Increasing fecal coliform bacteria counts by 100 cfu/100 mL would negatively impact property values by up to 3%. The results provide a measure of potential benefits against which cost-benefits could be compared to justify stormwater management efforts.

Finally, Brent et al. (2017) attempted to quantify the value of five environmental services associated with the implementation of LID approaches in two urban areas in Australia. The services included reduction in water restrictions, reduction in flash flooding, improvements in stream health, improvements in recreational and amenity benefits, and cooler summer temperatures. The valuation of these services may be valued differently in regions with different climactic conditions. They found a large range of Willingness to Pay values for these services, with improvements in stream health being the largest (a service also applicable in the Pacific Northwest). The overall range was from –A\$13 to A\$1611.

7 ACRONYMS AND ABBREVIATIONS

µg	microgram (10 ⁻⁶ grams)
B-IBI	Benthic Index of Biotic Integrity
BMP	Best Management Practices – broadly referring to approaches or devices that are used to treat stormwater runoff
CADDIS	USEPA Casual Analysis/Diagnostic Decision Information System
cfu	colony forming unit – measure of fecal coliform bacteria
DO	dissolved oxygen
GSI	Green Stormwater Infrastructure – constructed elements such as vegetated filter strips, bio-infiltration trenches, and retention ponds that are used in LID systems and approached
IDT	Interdisciplinary team
IQR	interquartile range
IS	Implementation Strategy
kg	kilogram (1000 grams)
LCA	Life Cycle Assessment – an approach for cost valuation that considers all stages of treatment (construction, operations, maintenance, decommissioning, etc.)
LID	Low Impact Development – an approach that is meant to utilize landscape like features and processes to reduce the volume of stormwater running off a site, and to treat it.
MMI	multi metric index – an index comprised of a combination of individual measures or metrics. B-IBI is a MMI
O&M	Operations and Maintenance – generally referring to the ongoing costs of running a stormwater treatment system but not the cost to build it in the first place.
PoCA	principal coordinate analysis
PSI	The University of Washington Puget Sound Institute
PSP	Puget Sound Partnership
TN	total nitrogen
TP	total phosphorus
TSS	total suspended solids

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