EPA Public Access

Author manuscript

Ecol Indic. Author manuscript; available in PMC 2019 October 01.

About author manuscripts

Submit a manuscript

Published in final edited form as:

Ecol Indic. 2018 October ; 93: 365–378. doi:10.1016/j.ecolind.2018.04.060.

Flow pulses and fine sediments degrade stream macroinvertebrate communities in King County, Washington, USA

Daniel Marshalonis^{a,*}, Chad Larson^b

^aUnited States Environmental Protection Agency, Region 10, 1200 6th Ave, Seattle, WA 98101, USA

^bWashington State Department of Ecology, Environmental Assessment Program, 300 Desmond Drive SE, Lacey, WA 98503, USA

Abstract

Determining the causes of biological impairment in urban stream settings presents unique challenges because there are many potential stressors associated with human development. A rigorous, scientifically based process is more likely to identify influential stressors that can be reduced to improve stream condition. We used the U.S. Environmental Protection Agency's (U.S. EPA) CADDIS (Causal Analysis/Decision Information System) stressor identification process to assess eight candidate causes in the urban Soos Creek Basin in Washington State. The eight candidate causes capable of negatively affecting the abundance and diversity of benthic macroinvertebrates are: flow alteration, increased fine sediments, reduced habitat complexity, elevated water temperature, low dissolved oxygen, elevated nutrients, increased ionic concentration, and toxic pollutants. We assembled multiple lines of evidence, as well as the consistency of that evidence and agreement with other assessments. We evaluated the influence of natural and cumulative anthropogenic stressors on macroinvertebrate communities by comparing various chemical, physical, and biological measures at sites in the Soos Creek Basin with regional reference sites. Of the stressors evaluated, flow alteration, increased fine sediments, and loss of habitat complexity were the most probable causes of biological impairment, with multiple biological metrics responding predictably across levels of impairment. Key findings from this study include: the use of specific community alterations as evidence in causal assessment, demonstration of links in a complete causal pathway, and the use of multiple models to show which pathway is likely stronger. In addition to the value to the specific case, the analyses increased our understanding of the responses of stream invertebrate communities in urban environments. Ultimately, demonstrating the utility of causal assessment in a practical situation provides greater confidence that mitigation efforts aimed at improving biological health of urban stream communities will have detectable desired effects while also providing a baseline from which the effectiveness of management practices can be evaluated.

*Corresponding author. marshalonis.dino@epa.gov (D. Marshalonis), clar461@ecy.wa.gov (C. Larson).

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2018.04.060>. These data include Google maps of the most important areas described in this article.

Keywords

Causal assessment; Weight of evidence; Stressor identification; Benthic invertebrates; Flow; Sediment

1. Introduction

Urban development and land conversion have measurable adverse impacts on biological communities (Vitousek et al., 1997). Along a gradient of urban development, alterations to biological communities can take many forms, ranging from the replacement of sensitive taxa by more tolerant ones to significant biodiversity loss (reviewed in Walsh et al., 2005). Streams in urban settings are subject to the combined impacts of multiple stressors arising from a variety of anthropogenic activities. The negative impacts of urbanization on stream systems are pervasive enough that they are collectively referred to as the ‘urban stream syndrome’ (Meyer et al., 2005, Walsh et al., 2005). Several potential stressors to stream systems resulting from urbanization include the loss of riparian habitat (Osborne et al., 1993, Morley and Karr, 2001), channelization and modifications to the natural flow regime (Konrad and Booth, 2005; Rosburg et al., 2017), increased inputs of nutrients and a variety of other chemicals (Paul and Meyer, 2001). These stressors, alone or collectively, can modify the physical and chemical parameters of a stream, ultimately leading to a significant alteration to the resident biological communities.

In highly diverse systems such as streams, describing the response to anthropogenic stressors of each individual taxon within the community can be impractical if not overwhelming, except perhaps in the case of endangered species. Yet more efficiently, important ecological insights into alterations to community composition can be gained through studies of functional traits (McGill et al., 2006; Poff et al., 2006; Messier et al., 2010). With increasing environmental stress, environmental filtering can shift composition within a community from one dominated by sensitive taxa to one dominated by more tolerant ones; in streams, these shifts can be measured through the response of diversity and trait-based metrics (Poff et al., 2006).

In the United States, individual states are mandated by the Clean Water Act, to evaluate, restore and maintain the chemical, physical and biological integrity of the Nation’s waters (Section 101(a), CWA, 33 U.S.C. § 1251(a)). Historically, waters determined to be ‘impaired’ for physical or chemical parameters are listed on a state’s 303(d) list and a Total Maximum Daily Load (TMDL) is developed to address ways of improving conditions for specific listings. However, less commonly included in the TMDL process is the incorporation of biological assessment, or the evaluation of resident biological communities (hereafter referred to as bioassessment). The science of bioassessment has developed over multiple decades, yet despite the mandate in the Clean Water Act to address biological integrity, progress in the successful incorporation of bioassessment within the framework of a TMDL has been slower than for more traditional water quality parameters (Karr and Yoder, 2004), despite increasing global threats to freshwater biodiversity (Vörösmarty et al., 2010).

In the state of Washington, biological integrity of stream segments can be evaluated with an index of biological integrity (B-IBI) for macroinvertebrates and potentially listed on the state's 303(d) list of impaired waters (<https://ecology.wa.gov/Water-Shorelines/Water-quality/Water-improvement/Assessment-of-state-waters-303d/Assessment-policy-1-11>). The B-IBI is a multimetric index comprised of ten community diversity measurements (Karr, 1998; Morley and Karr, 2001; more information at: <http://pugetsoundstreambenthos.org/About-BIBI.aspx>). These metrics have been demonstrated to give reliable signals of biological condition across a variety of scales and circumstances (Karr, 1999). Because of the integrative nature of multimetric indices such as the B-IBI, the component score alone is only informative for determining whether a site may be biologically impaired. However, the individual metrics comprising the B-IBI, in conjunction with supplemental metrics summarizing the biological traits of the component communities they describe, have the potential to create suggestive patterns between the biological community and specific environmental factors. In other words, the collection of organisms at a site, summarized into a variety of diversity and functional metrics, has the potential to suggest causal patterns of impairment (Karr and Yoder, 2004; Statzner and Bêche, 2010).

Here we describe the use of chemical and physical data in conjunction with biological data to assess causes for altered biological condition in streams from the Soos Creek Basin (King County, Washington, USA). In 2012, multiple streams and stream reaches within the basin (i.e., Big Soos, Little Soos, and Jenkins Creeks) were listed for biological impairment based on B-IBI scores (ECY, 2012). Additionally, the Washington State Department of Ecology (ECY) identified basin-wide impairments for elevated temperature and reduced dissolved oxygen based on criteria for spawning salmonids (ECY, 2012). Along with the separate listings for elevated temperature and reduced oxygen, a TMDL assessment is being prepared that will provide specific guidance for addressing any biological impairments in the Soos Creek Basin. Before steps can be taken to address the sources of biological impairment, the causes of impairment need to be elucidated.

We performed a Stressor Identification (SI) analysis to determine the most probable causes of altered biological conditions at listed sites so direct management strategies could be implemented within the TMDL. Within the Puget Sound, increasing development is leading to more urbanization and impervious cover, which has contributed to alterations in the natural flow regimes for many streams (Rosburg et al., 2017). Also within the region, a relationship between B-IBI and hydrologic alteration (e.g. high pulse count [HPC]) has already been demonstrated (DeGasperi et al., 2009), so flow alteration was one of several likely causes of biological impairment. As such, we predicted biological impairment would be linked with hydrologic alteration, namely in the form of a negative relationship between flashy flow and B-IBI scores as well as with several biological metrics. We also hypothesized that sites determined to be biologically impaired would differ from regional sites with minimal human impacts in predictable ways, namely an increase in organisms tolerant to various stressors (e.g. temperature, organic pollution, sensitivity to fine sediments, etc.). We were cognizant to evaluate all potential causes of impairment in the Soos Creek Basin using all available data. Our aim was to determine the most probable causes of biological impairment in the Soos Creek Basin while simultaneously

demonstrating the applicability of the SI process within the framework of a TMDL (U.S. EPA, 2000, 2007).

2. Materials and methods

The U.S. EPA (2000, 2007) SI process, as described on the CADDIS (Causal Analysis/Diagnosis Decision Information System) website (www.epa.gov/caddis) provides guidance for identifying environmental stressor(s) impacting biological communities using relevant available chemical, physical, landscape, or other biological data. The SI process provides a standardized, unbiased, and comprehensive structure for organizing evidence supporting the derived conclusion, allowing results to be compared with other assessments and ensuring the overall process is repeatable. Ultimately, the assembled lines of evidence among alternate causal hypotheses are weighed and compared (Cormier et al., 2010). In this way, SI guidance can assist in establishing the causal linkage(s) between a pollutant and the biological impairment, thus providing a basis for the development of a TMDL.

2.1. Watershed description

The Soos Creek watershed occupies approximately 70 square miles of glacial deposits, foothill ridges, and flat valley land in western Washington State (Fig. S1.kmz here). The 4 main headwater tributaries in the watershed are Soos Creek (26.4 linear stream km), Little Soos Creek (7.7 km), Jenkins Creek (10.3 km), and Covington Creek (17.1 km), which combine to form Big Soos Creek, a tributary to the Green River that leads to Puget Sound. As of 2012, there were 7 locations in the Soos Creek basin determined to be biologically impaired (I sites) and listed on the state's 303(d) list.

Upper portions of the watershed have flat or rolling terrain characterized by extensive wetlands and an elevated water table. Uniquely, the most developed areas of the watershed are in the headwaters of tributaries. Elevation decreases stepwise from 112 meters above sea level to 6 meters with headwater plateaus yielding to steep gradient reaches that terminate in low gradient valleys below.

Hydrology in the Soos Creek watershed is typical of the Puget Lowlands: spring and winter precipitation feeds large runoff and stream discharge events concurrent with Cascade Mountain snowpack storage at elevations, while reduced summer flows are maintained by snowmelt and base flows from groundwater. Glacial geology consists of a mix of low permeability Pleistocene continental glacial till (Vashon Stade) with veins of high permeability Fraser-age glacial outwash that mirror surface water features.

Land use and land cover are dominated by medium and low intensity development with limited mixed and evergreen forest coverage, particularly in areas draining to the stream reaches with a biological impairment. Six municipalities intersect the Soos watershed resulting in complete coverage under the state's municipal stormwater permit program. Additional other point source discharges related to construction (31), industrial (3), and mining permit activities (6) were also present at the time of this assessment.

2.2. Biological data

Within the U.S. EPA Puget Lowland level III ecoregion (Omernik, 1987), 10 “reference” (R), or ‘minimally impacted’ sites were chosen for comparison with sites from the Soos watershed. These sites were sampled by ECY and provided a regional context for evaluating the sites from the Soos watershed to establish expected conditions under minimal human impacts. B-IBI scores for 26 sites within the Soos Basin and 10 regional reference sites were obtained from Puget Sound Stream Benthos (<http://pugetsoundstreambenthos.org/Biotic-Integrity-Scores.aspx>, Supplementary Table 1). Data from the Soos Basin included samples collected from 1999 to 2013, with some sites having as few as one macroinvertebrate sample and others as many as 12. Macroinvertebrate samples represented a composite sample of all organisms collected from 0.74 square meter (8 ft²) of surface area using either a D-frame kicknet or surber sampler with a 500 µm net. A subsample of 500 organisms was counted and identified, typically to genus or species. Additionally, for each sample, several stressor-specific metrics were calculated, including a Fine Sediment Biotic Index (FSBI; Relyea et al., 2012), a metals tolerance index (MTI; McGuire, 2009), and an index of organic pollution, the Hilsenhoff Biotic Index (HBI; Hilsenhoff, 1987; Hilsenhoff, 1988). Briefly, the FSBI provides a measure of the abundance of taxa in a sample that are sensitive to fine sediment, with higher values indicating the presence of more sensitive taxa (values range from 0 to 400). The MTI provides a measure of the abundance of organisms in a sample that are tolerant of high metal concentrations, with higher values indicating greater abundance of tolerant organisms (values range from 0 to 10). The HBI provides a measure of the abundance of organisms in a sample that are tolerant to organic pollution, with higher values indicating a greater proportion of taxa tolerant to high levels of organic pollution as well as low dissolved oxygen levels (values range from 0 to 10). Additionally, inferred environmental conditions for fine sediment and water temperature were obtained for all invertebrate samples using the [bio.infer](#) package in R (Yuan, 2014). The [bio.infer](#) package employs a maximum likelihood inference method for estimating environmental conditions, in this case sediment and temperature values using tolerance data for macroinvertebrate taxa from the western United States (Yuan, 2007a,b). Lastly, the number of obligate cold-water taxa, i.e. cold stenotherms, were counted in each sample based on temperature tolerances determined from national and regional data (Vieira et al., 2006).

2.3. Chemical and physical data

Much of the physical and chemical data used in the analyses presented here were obtained from the ECY Environmental Information Management System (EIM)(<https://fortress.wa.gov/ecy/eimreporting/Default.aspx>). These data included measures of water, sediment and terrestrial soil chemistry, in situ physical measures (e.g., temperature, dissolved oxygen (DO), etc.) for all sites within the Soos basin, including all sites with biological data. Additional physical and chemical data for the region of interest were obtained from a variety of sources including the Muckleshoot Indian Tribe, King County Department of Natural Resources and Parks (DNRP), as well as two other ECY projects collecting water quality data from the region (i.e., Watershed Health Monitoring (WHM) and Stormwater Action Monitoring (SAM)) for comparison with the Soos basin sites. Furthermore, a calibrated water quality model (QUAL2Kw; Chapra and Pelletier, 2004; Pelletier et al., 2006) separately created for the Big Soos, Little Soos, and Jenkins Creek DO

TMDL was used to corroborate field measurements for various parameters including nutrients (total nitrogen, nitrate, nitrite, ammonia, total phosphorus, inorganic phosphorus, orthophosphorus), temperature, and dissolved oxygen.

2.4. Habitat and landscape data

Level I habitat assessments, qualitative in nature, were conducted for each impaired location within the watershed in accordance with U.S. EPA guidance for Wadeable Streams (U.S. EPA, 1993). Locations were scored separately for low gradient and high gradient stream habitats, respectively. Protocols defined parameters that characterize microscale habitat that have the greatest influence on indigenous community structure, as well as secondary parameters defining larger scale stream morphology, riparian habitat condition, and bank structure. Field observations for stream canopy cover were supplemented with modeled estimates of system potential shade (Chen et al., 1998a,b; ODEQ, 2001; ECY, 2003).

Landscape/land use data were calculated for all sites using GIS and the National Land Cover Database (Fry et al., 2011). Detailed information regarding the 391 unique landscape metrics can be found at Wilhelm et al. (2013). Additionally, data for nine modeled hydrologic metrics (see DeGasperi et al., 2009 for list and definitions) for each year with paired biological data and estimates of impervious cover for each of the catchments associated with sites used in these analyses were obtained from King County DNRP using the Hydrologic Simulation Program Fortran (HSPF; Bicknell et al., 2001).

2.5. Candidate stressor analysis

From a characterization of the biological effect, we created a data-independent conceptual model to generate hypotheses of causal mechanisms (Fig. 1). The seven proximate candidate causes are: altered flow, increased fine sediments, reduced habitat complexity, increased temperature, decreased dissolved oxygen, increased nutrients, and increased toxic pollutants. Our candidate stressors were evaluated for causation by determining which causal hypotheses were best supported by available evidence (U.S. EPA, 2007; Norton et al., 2014). Using the CADDIS approach, we then compared each alternative to determine the best support from the complete body of evidence.

2.6. Statistical analyses

All statistical analyses were conducted in R (version 3.3.3, R Core Team 2017). Linear and multiple regression models were developed for relationships between B-IBI and the predictor variables HPC, watershed area, and percent impervious cover of the upstream watershed. HPC was used as a predictor since a relationship between B-IBI and HPC had been previously demonstrated (also, see below). Furthermore, HPC was also found to have the highest correlation to B-IBI, even higher than all the GIS land cover variables, including impervious cover and watershed area. Values of relative importance for each predictor variable in the multiple regression were calculated using the relaimpo package (Grömping, 2006). The relative importance of each parameter in the regression model was compared using a lmg metric, which provides a decomposition of the multiple regression model-explained variance into non-negative relative contributions.

Repeated measures ANOVA was used to test the main effect of group (reference (R) sites vs. Soos impaired (I) sites vs. Soos non-impaired (N) sites) using type III sums of squares that accounts for an unbalanced design for the various biological metrics examined. The ezANOVA function of the ez package was used (Lawrence, 2016). Pair-wise comparisons between factor levels are problematic for repeated measures ANOVA, so 95% confidence intervals (CI) for the difference between group means were calculated using 10,000 bootstrap replicates obtained from a separate data set containing mean values (across years) for each site for all biological metrics that were examined. We interpreted a CI spanning zero (e.g., $-0.1 - 0.4$) to indicate a nonsignificant difference between means. Principle Components Analysis (PCA) was used to explore relationships between hydrologic metrics and between landscape metrics, specifically, to identify nonredundant indicators for correlation with biological metrics (Olden and Poff, 2003; Gao et al., 2009). This information was used to support/weaken hypothetical causal pathways as described in Fig. 1.

3. Results

3.1. Characterization of biological impairment

Compared to reference sites, B-IBI scores were considerably lower in the Soos basin, with I sites having significantly lower scores than N sites within the basin (repeated measures ANOVA, $F_{2, 33} = 17.03$, $p = 0.00001$, Fig. 2a). Pair-wise comparisons of 95% CI for the difference between group means revealed differences between all three stream types (R vs N: 5.45–13.10, R vs I: 9.42–18.68, N vs. I: 0.78–8.91). Among the seven I sites, average B-IBI scores varied considerably (Fig. 2b). For the ten metrics comprising the B-IBI, values were generally lowest for the I sites, with the N sites typically intermediate of the two, except for the two metrics that increase with impairment, percent dominant and percent tolerant respectively (Table 1).

3.2. Characterization of chemical and physical parameters

Most of the discrete data we had available for many of the standard measures of water quality, both physical and chemical, did not indicate chronic water quality problems and are therefore not reported in detail here, yet summary information, including comparisons to reference and other regional studies is available in Table 3. Because we had limited water quality data paired with biological samples from various sources, we summarized the water quality parameters for all sites in the Soos basin rather than differentiating I versus N. For example, at sites in the Soos basin for which we had measures of chloride and conductivity, parameters commonly associated with urban development (Wenger et al., 2009), average values for both of these parameters were lower than those for regional reference data and several other regional datasets, indicating low ionic concentrations during the discrete sampling events. Additional parameters for which we had discrete measures that did not reach levels that would indicate chronic or acute stress to macroinvertebrates included nitrogen (King and Richardson, 2003; Smith et al., 2007), temperature (Thorp and Covich, 1991; Wang and Kanehl, 2003; deKozlowski and Bunting, 1981; Heiman and Knight, 1972), and dissolved oxygen (Graeber et al., 2013; Nebeker, 1972; Chen et al., 2001; Connolly et al., 2004; Miltner, 2010). However, although the discrete data did not indicate chronic water quality problems, several sites in the Soos basin have been listed as impaired for

exceedances in temperature and dissolved oxygen and a TMDL is being developed to address these specific issues. These listings were based on criteria developed for the health of salmonids, which may have different thermal and oxygen-related responses than benthic macroinvertebrates.

3.2.1. Altered flows—PCA exploring the relationship between B-IBI data and all hydrologic metrics indicated the first PCA axis explained 61.7 percent of variance in B-IBI (Supplementary Fig. 1). Furthermore, HPC and the first PCA axis were strongly correlated (Pearson correlation coefficient = -0.95 , $p < 0.0001$); therefore, we decided to use the flashiness metric HPC as a surrogate for all flow alteration impacts. The relationship between average B-IBI scores and average HPC indicated a significant decreasing linear trend ($B-IBI = -0.896 \times HPC + 38.74$, $F_{1, 17} = 24.2$, $r^2 = 0.587$, $p = 0.00013$, $AIC = 109.93$, Fig. 3a). A model with impervious cover and watershed area in addition to HPC explained more variance than HPC alone ($B-IBI = -0.783 \times HPC - 9.54 \times IC - 0.0025 \times Area + 41.43$, $r^2 = 0.646$, $F_{3, 15} = 11.95$, $p = 0.000029$, $AIC = 107.56$), but the relative importance of HPC was greater than impervious cover and watershed area (lmg metrics of: 0.599 vs. 0.216 vs. 0.185, respectively), indicating the considerable influence of chronic flashy flow on B-IBI scores (Fig. 3b-c). Added variable plots and partial t-tests indicated that B-IBI did not vary with impervious cover (i.e., slope not different than zero, $p > 0.05$) when HPC and Area were held constant in the multiple regression model (Fig. 3c). However, B-IBI varied with HPC when impervious cover and Area were held constant (Fig. 3b), and B-IBI varied with Area when HPC and impervious cover were held constant (Fig. 3d).

3.2.2. Fine sediment—We used the assemblage of organisms to infer substrate conditions because measured substrate size (i.e., substrate values) was not measured with a consistent methodology for many of the sites having biological data. For FSBI values (higher values indicating greater abundance of taxa sensitive to fine sediment deposition) differed significantly between stream categories. R sites had the highest values, I sites had the lowest, and N sites had values intermediate of the two (repeated measures ANOVA, $F_{2, 33} = 20.97$, $p = 0.00001$, Fig. 4a). Pair-wise comparisons of 95% CI for the difference between group means revealed the CI between all three stream types did not span zero (R vs N: 42.07–100.98, R vs I: 83.13–137.34, N vs. I: 18.58–59.45). Among the seven I sites, average FSBI values varied considerably, with two sites having values of zero, indicating the complete absence of taxa sensitive to fine sediment (Fig. 4b). The opposite trend was observed for the inferred percent fines at sites, with R sites having the lowest values, I sites having the highest values and the N sites having intermediate values (repeated measures ANOVA, $F_{2, 33} = 10.37$, $p = 0.0003$, Fig. 4c). Pair-wise comparisons of 95% CI for the difference between group means revealed the CI between all three stream types did not span zero (R vs N: -23.84 to -6.13 , R vs I: -51.83 to -24.53 , N vs. I: -39.33 to -7.80). Among the seven I sites, average values for percent fines varied considerably, yet values for all sites were above 25%, with 1 site having a value greater than 75% (Fig. 4d).

3.2.3. Nutrients and dissolved oxygen—HBI values reflect the community response to organic pollution and decreased dissolved oxygen, with HBI values decreasing with lower water quality. HBI values were highest for the I sites, with values for R and N sites having

similar values, yet differences between stream types were not significantly different from one another (repeated measures ANOVA, $F_{2, 33} = 2.99$, $p = 0.06$, Fig. 5a). Values among the I sites were similar for most sites (i.e. values above 4), but several sites had values more similar to R and N sites (Fig. 5b). Values across all sites, even the I sites, were low enough to be considered unlikely of reflecting a high degree of organic pollution or low dissolved oxygen levels (Hilsenhoff, 1987, Hilsenhoff, 1988). QUAL2Kw model results corroborate empirical sampling, showing consistently low nutrient levels and dissolved oxygen levels greater than 10mg/L. (Note: there are other stream locations within the watershed experiencing dissolved oxygen and temperature impairments that dictate the necessity for TMDL regulation). The exception is the headwaters of the Big Soos Creek upstream of Site B, which is fed by wetlands-dominated hydrology, where dissolved oxygen values can sometimes drop to 5 mg/L. However, this is a natural phenomenon.

3.2.4. Toxics—Values for the MTI, which increase with metals contamination, differed significantly between stream groups and were unexpectedly highest for R sites and lowest for the I sites (repeated measures ANOVA, $F_{2, 33} = 5.89$, $p = 0.006$, Fig. 5c). Pair-wise comparisons of 95% CI for the difference between group means revealed the CI between all three stream types did not span zero (R vs N: 0.07–1.21, R vs I: 0.47–1.67, N vs. I: 0.08–0.82). Values across the seven I sites were all low, with average values all below 1.5 (Fig. 5d). It should be noted that values for the MTI can vary between 0 and 10, and values across all sites were low (i.e. less than 2.5) indicating that for the assessed samples, there does not appear to be an abundance of taxa that are tolerant of high metal concentrations. Additionally, measured values for 395 available toxic compounds (including metals, polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds, polychlorinated biphenyls (PCBs), pesticides, and contaminants of emerging concern) as well as conventional water quality parameters were typically very low, either at or below method detection limits for many of these parameters. We report summary values for several of the most notable parameters collected in sediment samples in Table 3. For example, values for total PAHs, total PCBs and various metals were all below levels considered detrimental to macroinvertebrate communities (Table 3, Michelsen, 2011).

3.2.5. Temperature—Taxa richness of cold water stenotherms was higher in R sites than N and I sites (repeated measures ANOVA, $F_{2, 33} = 88.79$, $p < 0.00001$, Fig. 6a). Pair-wise comparisons of 95% CI for the difference between group means revealed the CI between the R sites and both Soos groups did not span zero, yet the CI between the N and I sites did (R vs N: 8.11–11.51, R vs I: 8.63–12.52, N vs. I: –0.58 to 2.25). Mean taxa richness of cold water stenotherms across the seven I sites were generally below 10 (Fig. 6b). Values for inferred temperature suggest N sites have the warmest water temperatures and R and I sites have very similar water temperature values (repeated measures ANOVA, $F_{2, 33} = 2.35$, $p = 0.011$, Fig. 6c). Pair-wise comparisons of 95% CI for the difference between group means revealed the CI between all three stream types did span zero (R vs N: –1.50 to 0.04, R vs I: –0.72 to 1.47, N vs. I: –0.05 to 2.28). Values across the seven I sites were all below 14 °C, with 2 sites averaging approximately 10 °C (Fig. 6d). Average measured temperature for N sites was just 1 °C above the other stream categories; so, while N sites were statistically different than the other streams, the difference in average temperature reflected by the

communities may not be biologically significant. QUAL2Kw modeling results as well as additional data not used in model calibration indicate temperatures at or below 20 °C for the entire stream system, with most results at or below 15 °C.

3.2.6. Habitat assessment—Table 2 provides results of the habitat assessment for each impaired location. Overall scores ranged from 45 to 91 (on 0–165 scale) with qualitative scores of B- to D (on an excellent: A+ to failing: F scale). In general, most sites scored poorly for microscale habitat quality, whereas scores for morphology and riparian habitat were more variable. Results were consistent with stakeholder consensus and historical knowledge gleaned from local professionals (N. Rapin, Muckleshoot Indian Tribe, personal communication).

Modeled data for system potential shade for the major tributaries in the Soos watershed (Fig. 7) indicate lack of riparian shade is not a contributing factor to temperature impairment, and consequently, lack of riparian habitat is likely not contributing to poor physical habitat. Modeled data 200 m upstream and downstream from sites of impairment show shade deficit values ranging from 8% to 29% for all sites excluding Site G (44–67%). There were no modeled or measured data for shade deficit at Site A.

3.3. Weight of evidence synthesis

Table 4 summarizes evidence for each candidate cause by characteristics of causal relationships. The weight of evidence of the evaluated candidate causes indicates that altered flows, excess fine sediment and degraded habitat complexity are the most likely causes of reduced B-IBI scores in the Soos Creek basin. Flow, sediment, and habitat each had strongly supporting or somewhat supporting evidence, each with supporting evidence for at least three different characteristics of causal relationships. Conversely, evidence suggests that the other five candidate causes are less likely to contribute to the impaired macroinvertebrate communities in the Soos basin. We had no evidence for two causal characteristics (Interaction and Time-order) for any of our candidate causes.

4. Discussion

We present findings from one of the most comprehensive applications of U.S. EPA's CADDIS stressor identification within the framework of a bioassessment TMDL, demonstrating the utility of this approach for determining the most likely stressors causing biological impairment. TMDLs are at the intersection of science and policy and the CADDIS process outlines a standardized regiment for implementing a rigorous science-based approach that can inform policy based decision-making. A priori, we assembled a broad candidate list of stressors capable of impacting benthic macroinvertebrates within the Soos basin, Washington, and using the CADDIS guidelines and available data, determined the most probable set of stressors impacting biological impairment. We synthesized the evidence using a weight-of-evidence approach, determining which pieces of information supported or weakened various characteristics of causal relationships, and ultimately the likelihood of impacts derived from each candidate cause. Our analyses suggest that a combination of flow alteration, delivery of fine sediments, and loss of historical high quality physical habitat have all contributed to biological impairment in this basin. These findings

are consistent with those from larger observational studies around the U.S. documenting the impacts of urbanization on stream communities (Coles et al., 2012).

Due to increased impervious cover and storm conveyances relative to undeveloped areas, many urban streams are impacted by increased streamflow flashiness or hydrologic flow alteration (Poff et al., 1997; Bellucci, 2007; Coles et al., 2012). As we observed in the Soos basin, there was a negative relationship between B-IBI scores and the number of high pulse events, a result that has also been observed regionally (DeGasperi et al., 2009; King County, 2014). However, this relationship was demonstrated only when data (i.e., HPC and B-IBI scores) were averaged across years for each of the HSPF-modeled regions, something that was required since B-IBI scores at sites with chronically high HPC did not closely track flow, e.g., B-IBI scores did not immediately increase in a year with low HPC; this is consistent with the “observable biological potential” concept defined by Paul et al. (2009). Additionally, B-IBI scores were also related to impervious cover and watershed area, yet we were able to statistically separate the impacts of each of these factors on B-IBI scores. Effectively isolating the influence of each of the other factors allowed us to gain greater understanding of the relationship between HPC and B-IBI scores while controlling for other factors. Interestingly, HPC had the largest relative influence of the three factors tested on B-IBI scores. For B-IBI scores to improve over time, it will be necessary for flow conditions to more closely mimic those observed under more natural conditions, i.e., fewer high pulse events that are also lower in magnitude; we also recognize that HPC is not the sole cause for impairment of the macroinvertebrate communities. Nonetheless, these findings are consistent with our conceptual model and demonstrate the importance of altered flows in shaping macroinvertebrate assemblages in this watershed; removal of a required intermediate step in the causal pathway (increased surface runoff measured as HPC resulting from impervious surfaces) eliminates the biological response, while elimination of a different intermediate step (increased pollutants delivered from impervious surfaces) in a different causal pathway but one derived from the same human activity (land-cover alteration) does not.

Although we focused primarily on the relationship between B-IBI and HPC, a consistent negative linear relationship was also observed between other biological metrics and HPC, i.e. taxa richness, EPT taxa richness (richness of Ephemeroptera, Plecoptera, and Trichoptera), clinger taxa richness and long-lived taxa richness. These results suggest that the influence of flow alteration on macroinvertebrate communities was consistent across multiple diversity measures, with each of them responding negatively to increasing HPC, suggesting that results from our study have applicability beyond the region of our study. We chose B-IBI because of its use as a biological endpoint within the TMDL, but we could have chosen any of the other diversity metrics mentioned above, which suggests that HPC reduces the overall diversity of macroinvertebrate taxa intolerant of high flow.

The biological data we had available also implicated increased fine sediment as a stressor contributing to biological impairment in the Soos basin. In addition to altered flow, increased fine sediment has been shown to be one of the most influential stressors adversely affecting stream macroinvertebrate communities (Bryce et al., 2010, Burdon et al., 2013). Based on the biological communities present, we were able to infer elevated fine sediment at all Soos

basin sites, but most notably at the I sites. I sites were generally characterized by very low numbers of taxa intolerant of fine sediment as measured with the FSBI, with some sites having scores indicating an almost complete absence of sensitive taxa. Additionally, inferred fine sediment values based on the assemblage of organisms at each of the sites suggest I sites have a much higher percentage of fines than N sites and regional R sites. Many of the organisms intolerant of fine sediment require hard substrate and the interstitial spaces they provide. Some estimates suggest that streambed percent fines or sand/fines above 10–20% can negatively impact macroinvertebrate communities (Bryce et al., 2010; Burdon et al., 2013). All of the I sites had values for inferred percent fines above those thresholds.

Increased flow alterations and fine sediment deposition resulting from human development can individually and collectively contribute to reduced substrate habitat quality (Hart and Finelli, 1999; Bunn and Arthington, 2002). Urban development can also result in modifications to stream riparian habitat, further increasing flashy flow and sediment inputs to streams. As stated earlier, deposition of fine sediment reduces substrate complexity by eliminating many of the interstitial spaces that organisms use to protect themselves from the shear forces of high flow events. Additionally, flashy flow events can alter channel morphology and complexity (Jackson and Sturm, 2002). As we demonstrated here, many of the I sites in the Soos basin scored poorly for the variety of habitat measures that we assessed. Our work is consistent with that of others illustrating the importance of high quality physical habitat to healthy ecosystem function (Roni et al., 2002).

Conversely, we demonstrated that other stressors (e.g. nutrients and toxics) in our conceptual model were likely not contributing significantly to biological impairment in the Soos basin. Many of the discrete water quality parameters we assessed did not indicate chronic issues in the Soos basin, as many parameters had values that were comparable to other regional monitoring studies and were low enough so as not to be deemed harmful to macroinvertebrate communities. One stressor in particular, elevated ionic concentration, is typically associated with urban environments (Roy et al., 2003; U.S. EPA, 2016); however, conductivity and chloride values within the Soos basin were comparable to values from other regional studies and to reference sites. Various metrics associated with benthic macroinvertebrates gave us greater insight into some of these other candidate stressors, supporting findings from both discrete physical and chemical data available and model simulation, which suggests that nutrients and toxic pollutants are not major contributors to biological impairment in the Soos basin. Values for both the HBI and the MTI indicate that organic pollution and excessive metals do not appear to adversely affect macroinvertebrate communities in this watershed. Across all sites, values for the HBI were at levels indicating excellent to good water quality ratings, suggesting little to no organic pollution. Similarly, values across all sites for the MTI were very low, giving no indication that metals adversely affected macroinvertebrate communities in this watershed.

Additionally, temperature and dissolved oxygen in the Soos basin, for which there are multiple listings on the 303(d) list at different locations and the focus of a separate TMDL, did not appear to contribute greatly to impairment of benthic macroinvertebrate communities. As stated earlier, the listings for temperature and oxygen were based on thresholds for protecting salmonids, and our analyses only addressed the impacts of these

variables on macroinvertebrates. However, the HBI, in addition to being a measure of the macroinvertebrate community response to organic pollution, is also a measure of a community's response to decreased oxygen levels. As HBI values across all sites were low, any issues with dissolved oxygen levels in the Soos basin would likely be influenced more by elevated water temperatures than by organic pollution. Regarding water temperature, the limited measured data we had available did not indicate a consistent issue with elevated water temperature. However, numbers of cold stenotherms were generally lower in the Soos Creek basin than observed at R sites. Additionally, and consistent with QUAL2Kw model simulation, inferred water temperatures from sampled macroinvertebrate assemblages indicated water temperatures were generally below 14 °C, which is lower than established thresholds for salmonids. However, we recognize some of the limitations of these data, which do not consider the periodic spikes in temperature that could potentially be detrimental to salmonids. Therefore, we did not assess water temperature and dissolved oxygen level impacts to salmonids in the Soos basin; our analyses only suggest these variables do not contribute to impairment of macroinvertebrate communities.

The CADDIS SI process relies on analyzing available evidence and determining if that evidence is adequate to draw a conclusion regarding the causes of impairment. Inconsistent evidence, lack of data, or lack of supportive evidence confound interpretation and prevent definitive assignment of causation. In our case, we were able to assign probable cause to three of our eight evaluated candidate stressors, while five stressors were deemed improbable or with an uncertain influence. While temperature, nutrients, dissolved oxygen, ionic concentration and toxic pollution data were absent at levels deemed likely to contribute stress to the biological communities in question, it is possible those water quality constituents contribute some baseline level of stress, or that our data frequency was insufficient to detect sporadic or episodic events of extreme perturbation that could result in short- or long-term stress. Further complicating the issue is the complex interactions and interrelatedness of the individual stressors themselves. In our case, altered flow regimes can disrupt community demographics by causing "mass drift", act as a vector for the delivery of increased fine sediment loadings to streams, and contribute to the geomorphic shifts in stream habitats that lead to degraded habitat quality and complexity. Such habitat rearrangement tends to erode and re-suspend sediments that can then be deposited in other areas and contribute additional stress to the system. In this case, our statistical model highlights the relative importance of flow alteration in restructuring benthic macroinvertebrate communities (see also Hart and Finelli, 1999; Bunn and Arthington, 2002) and how absence of portions of that complete causal pathway eliminate the effect. Such evidence provides confidence that the conclusions of our weight of evidence are meaningful. Deciphering the most important factor in this complicated interplay is critical when deciding which management strategies will be most cost-effective for recovery of the biological functioning of the system.

Few studies in the past have used biological data as evidence to evaluate causation (but see Gerritsen et al., 2010; Coffey et al., 2014 for examples, CTDEP, 2007). Here, we incorporated a variety of biological indices and their derivative, the B-IBI, to highlight key aspects of community change and to identify potential environmental stressors. Generally, results from our analyses showed that macroinvertebrate communities at impaired sites in

the Soos basin were a reflection of stressors contributing to the loss of intolerant taxa, which were replaced by more tolerant ones. Because our indices were diverse and specific, their use provided important information for determining the most likely causes of impairment in this specific watershed with sufficient confidence to inform the development of a TMDL.

The results of this study necessarily inform the bioassessment TMDL actions in the Soos basin, which must identify the pollutant(s) causing the impairment and produce strategies to meet relevant water quality standards. We hypothesized that eight candidate stressors could be contributing to the biological impairment in the Soos basin and by performing a causal assessment, revealed that fine sediment, flashy flow and habitat alteration were the stressors most likely contributing to this impairment. The causal assessment also eliminated elevated ionic, nutrient, and toxics concentrations and elevated temperature and reduced DO as significant contributors to the biological impairment observed in the Soos basin. This biological impairment TMDL is the first of its kind in Washington State, and our use of the U.S. EPA CADDIS framework will provide context for future regulatory actions related to the recovery of degraded biological communities. Our results also speak to the utility of trait and tolerance-based biological metrics for inferring the causes of stress influencing stream macroinvertebrate communities, information that can provide important ecological insights for informing a TMDL and developing plans for rehabilitation.

5. Conclusion

In this analysis, the use of detailed and specific macroinvertebrate metrics in addition to the B-IBI led to a focused investigation of candidate causes, saving time and reducing the need for additional costly monitoring. By evaluating existing data for patterns, we were able to focus our efforts, and then to corroborate inferred effects with actual water quality and physical habitat data.

The use of biological, chemical, and physical habitat metrics provides a comprehensive suite of data to holistically evaluate system dynamics and linkages. In the absence of other data, specific biological metrics provide supporting inferences regarding what stressors are shaping biological communities. By using the formal CADDIS approach, we reduced bias and generated a reasonable understanding of the causal relationships leading to biological impairment.

As evidenced here, the collection and evaluation of biological data has the potential for revealing patterns of impairment. Unlike many other types of data typically evaluated when assessing water quality, biological data can provide detailed information about the stressors occurring at a site. In this sense, evaluating biological data can provide a wealth of information that many conventional water quality parameters overlook. For TMDLs to address all mandates of the Clean Water Act, effective incorporation of biological assessment into the TMDL process will be vital in achieving successful restoration and protection efforts.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

Acknowledgements

We thank Curtis DeGasperi and Nancy Rapin for providing data assistance and Sue Norton, Scott Collyard, Chris Zell and two anonymous reviewers for very helpful comments that greatly improved the quality of this manuscript. This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors. The authors declare that they have no conflicts of interest. The ideas in this article are our own and do not represent policy or recommendations of the U.S. EPA.

References

- Bellucci Christopher, 2007 Stormwater and aquatic life: making the connection between impervious cover and aquatic life impairments for TMDL development in Connecticut streams. In: Proceedings of the Water Environment Federation TMDL Conference, Bellevue, WA. Alexandria, VA: Water Environment Federation, 1003–1018. Available online at: http://clear.uconn.edu/projects/tmdl/library/tmdl/papers/Bellucci_WEF_2007.pdf.
- Benoy GA, Sutherland AB, Culp JM, Brua RB, 2012 Physical and ecological thresholds for deposited sediments in streams in agricultural landscapes. *J. Environ. Qual.* 41 (1), 31–40. <https://www.ncbi.nlm.nih.gov/pubmed/22218171>. [PubMed: 22218171]
- Bicknell BR, Imhoff JC, Kittle JL Jr., Donigan AS Jr., Jobes TH, Johanson RC, 2001 Hydrological Simulation Program – FORTRAN, User's Manual for Version 12 U.S. EPA, National Exposure Research Laboratory, Athens, GA.
- Bryce SA, Lomnický GA, Kaufmann PR, 2010 Protecting sediment-sensitive aquatic species in mountain streams through the application of biologically based streambed sediment criteria. *J. North Am. Benthol. Soc.* 29, 657–672.
- Bunn SE, Arthington AH, 2002 Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environ. Manage.* 30 (4), 492–507. [PubMed: 12481916]
- Burdon FJ, McIntosh AR, Harding JS, 2013 Habitat loss drives threshold response of benthic invertebrate communities to deposited sediment in agricultural streams. *Ecol. Appl.* 23, 1036–1047. [PubMed: 23967573]
- Chapra SC, Pelletier GJ, 2004 QUAL2K: A Modeling Framework for Simulating River and Stream Water Quality: Documentation and Users Manual Civil and Environmental Engineering Dept, Tufts University, Medford, MA.
- Chen YD, Carsel RF, McCutcheon SC, Nutter WL, 1998a Stream temperature simulation of forested riparian areas: I. Watershed-scale model development. *J. Environ. Eng.* 304–315.
- Chen YD, Carsel RF, McCutcheon SC, Nutter WL, 1998b Stream temperature simulation of forested riparian areas: II. Model application. *J. Environ. Eng.* 316–328.
- Chen L-Y, Heath AG, Neves RJ, 2001 Comparison of oxygen consumption in freshwater mussels (Unionidae) from different habitats during declining dissolved oxygen concentration. *Hydrobiologia* 450, 209–214.
- Coffey DB, Cormier S, Harwood J, 2014 Using field-based species sensitivity distributions to infer multiple causes. *Human Ecol. Risk Assess. Int. J.* 20 (2), 402–432. 10.1080/10807039.2013.767071.
- Coles JF, McMahon G, Bell AH, Brown LR, Fitzpatrick FA, Eikenberry BCS, Woodside MD, Cuffney TF, Bryant WL, Cappiella K, Fraley-McNeal L, 2012 Effects of Urban Development on Stream Ecosystems in Nine Metropolitan Study Areas Across the United States (No. 1373). US Geological Survey.
- Connolly NM, Crossland MR, Pearson RG, 2004 Effect of low dissolved oxygen on survival, emergence, and drift of tropical stream macroinvertebrates. *J. North Am. Benthol. Soc.* 23, 251–270.
- Cormier SM, Paul JF, Spehar RL, Shaw-Allen P, Berry WJ, Suter GW, 2008 Using field data and weight of evidence to develop water quality criteria. *Integr. Environ. Assess. Manage.* 4 (4), 490–504. 10.1897/IEAM_2008-018.1.
- Cormier SM, Suter II GW, Norton SB, 2010 Causal characteristics for ecoepidemiology. *Hum. Ecol. Risk Assess.* 16, 53–73.

- CTDEP, 2007 Appendix 1. Stressor Identification In: A Total Maximum Daily Load Analysis for Eagleville Brook, Mansfield, Ct State of Connecticut, Department of Environmental Protection Available online at: http://www.ct.gov/deep/lib/deep/water/tmdl/tmdl_final/eaglevillefinal.pdf.
- DeGasperi CL, Berge HB, Whiting KR, Burkey JJ, Cassin JL, Fuerstenberg RR, 2009 Linking hydrologic alteration to biological impairment in urbanizing streams of the Puget Lowland, Washington, USA. *JAWRA J. Am. Water Resour. Assoc.* 45, 512–533. [PubMed: 22457566]
- deKozlowski SJ, Bunting II DL, 1981 A laboratory study on the thermal tolerance of four Southeastern stream insect species (Trichoptera, Pleicoptera). *Hydrobiologia* 79, 141–145.
- Ecology (Washington State Department of Ecology), 2003 Shade.xls – A tool for estimating shade from riparian vegetation. Washington State Department of Ecology, Olympia, WA www.ecy.wa.gov/programs/eap/models/.
- Ecology, 2012 Modeling Quality Assurance Project Plan for Soos Creek Watershed Temperature and Dissolved Oxygen TMDL Technical Analysis. Washington State Department of Ecology, Olympia, WA.
- Fry J, Xian G, Jin S, Dewitz J, Homer C, Yang L, Barnes C, Herold N, Wickham J, 2011 Completion of the 2006 National land cover database for the conterminous United States. *Photogramm. Eng. Rem. Sens.* 77, 858–864.
- Gao Y, Vogel RM, Kroll CN, Poff NL, Olden JD, 2009 Development of representative indicators of hydrologic alteration. *J. Hydrol.* 374 (1–2), 136–147. 10.1016/j.jhydrol.2009.06.009.
- Gerritsen J, Zheng L, Burton J, Boschen C, Wilkes S, Ludwig J, Cormier S, 2010 Inferring Causes of Biological Impairment in the Clear Fork Watershed, West Virginia U.S. Environmental Protection Agency, Office of Research and Development, National Center for Environmental Assessment, Cincinnati, OH EPA/600/R-08/146.
- Graeber D, Pusch M, Lorenz S, Brauns M, 2013 Cascading effects of flow reduction on the benthic invertebrate community in a lowland river. *Hydrobiologia* 717 10.1007/s10750-013-1570-1.
- Gromping U, 2006 Relative importance for linear regression in R: the package relaimpo. *J. Stat. Softw.* 17, 1–27.
- Hart DD, Finelli CM, 1999 Physical-biological coupling in streams: the pervasive effects of flow on benthic organisms. *Annu. Rev. Ecol. Syst.* 30, 363–395.
- Heiman DR, Knight AW, 1972 Upper-lethal temperature relations of the nymphs of the stonefly, *Paragnetina media*. *Hydrobiologia* 39 (4), 479–493.
- Hilsenhoff WL, 1987 An improved biotic index of organic stream pollution. *Great Lakes Entomol.* 20, 31–39.
- Hilsenhoff WL, 1988 Rapid field assessment of organic pollution with a family-level biotic index. *J. North Am. Benthol. Soc.* 7, 65–68.
- Jackson CR, Sturm CA, 2002 Woody debris and channel morphology in first- and second order forested channels in Washington's coast ranges. *Water Resour. Res.* 38 (9), 1177 10.1029/2001WR001138.
- Karr JR, 1998 Rivers as sentinels: using the biology of rivers to guide landscape management In: Naiman RJ, Bilby RE (Eds.), *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*. Springer, New York, pp. 502–528.
- Karr JR, 1999 Defining and measuring river health. *Freshw. Biol.* 41, 221–234.
- Karr JR, Yoder CO, 2004 Biological assessment and criteria improve total maximum daily load decision making. *J. Environ. Eng.* 130, 594–604.
- County King, 2014 B-IBI Restoration Decision Framework and Site Identification Prepared by Jo Opdyke Wilhelm, Debra Bouchard, Chris Gregersen, Chris Knutson, and Kate Macneale. Water and Land Resources Division Seattle, Washington.
- King RS, Richardson C, 2003 Integrating bioassessment and ecological risk assessment: an approach to developing numerical water-quality criteria. *Environ. Manage.* 31, 795–809. 10.1007/s00267-002-0036-4. [PubMed: 14565699]
- Konrad Christopher, Booth Derek B., 2005 Hydrologic Changes in Urban Streams and Their Ecological Significance.
- Lawrence MA 2016 ez: Easy Analysis and Visualization of Factorial Experiments. R package version 4.4–0. <https://CRAN.R-project.org/package=ez>.

- McGill BJ, Enquist BJ, Weiher E, Westoby M, 2006 Rebuilding community ecology from functional traits. *Trends Ecol. Evol.* 21, 178–185. [PubMed: 16701083]
- McGuire DL, 2009 Clarke Fork River biomonitoring macroinvertebrate community assessments in 2008. McGuire Consulting, prepared for CH2MHill and submitted to US Environmental Protection Agency Region 8, Boise, Idaho.
- Messier J, McGill BJ, Lechowicz MJ, 2010 How do traits vary across ecological scales? A case for trait-based ecology. *Ecol. Lett.* 13, 838–848. [PubMed: 20482582]
- Meyer JL, Paul MJ, Taulbee WK, 2005 Stream ecosystem function in urbanizing landscapes. *J. North Am. Benthol. Soc.* 24, 602–612.
- Michelsen T, 2011 Development of benthic SQVs for freshwater sediments in Washington, Oregon, and Idaho. Washington State Department of Ecology. Publication No. 11–09-054.
- Miltner RJ, 2010 A method and rationale for deriving nutrient criteria for small rivers and streams in Ohio. *Environ. Manage.* 45, 842–855. [PubMed: 20108135]
- Morley SA, Karr JR, 2001 Assessing and restoring the health of urban streams in the Puget Sound Basin. *Conserv. Biol.* 16, 1498–1509.
- Nebeker AV, 1972 Effect of low oxygen concentration on survival and emergence of aquatic insects. *Trans. Am. Fish. Soc.* 101, 675–679.
- Norton SB, Cormier SM, Suter GW (Eds.), 2014 Ecological Causal Assessment. CRC Press, Boca Raton.
- ODEQ (Oregon Department of Environmental Quality), 2001 tTools 3.0 User's Manual. Oregon Department of Environmental Quality, Portland, OR.
- Olden JD, Poff NL, 2003 Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. *River Res. Appl.* 19 (1), 101–121. 10.1002/rra.700.
- Omernik JM, 1987 Ecoregions of the conterminous United States. *Map (scale 1:7,500,000)*. *Annal. Associ. American Geograph.* 77 (1), 118–125.
- Osborne LL, Bayley PB, Higler LWG, Statzner B, Triska F, Moth Iversen T, 1993 Restoration of lowland streams: an introduction. *Freshwater Biol.* 29, 187–194.
- Paul Michael J., Bressler David W., Purcell Alison H., Barbour Michael T., Rankin EdT., Resh Vincent H., 2009 Assessment tools for urban catchments: defining observable biological potential. *J. Am. Water Resour. Assoc. (JAWRA)* 45 (2), 320–330. 10.1111/j.1752-1688.2008.00280.
- Paul MJ, Meyer JL, 2001 Streams in the urban landscape. *Ann. Rev. Ecol. Systematic.* 32, 333–365.
- Pelletier Greg & Chapra Steven & Tao Hua, 2006 QUAL2Kw — A Framework for Modeling Water Quality in Streams and Rivers Using a Genetic Algorithm for Calibration. *Environmental Modelling and Software*, pp. 419–425.
- Poff NL, Allan JD, Bain MB, Karr JR, Prestegard KL, Richter BD, Sparks RE, Stromberg JC, 1997 The natural flow regime. *Bioscience* 47, 769–784.
- Poff NL, Olden JD, Vieira NK, Finn DS, Simmons MP, Kondratieff BC, 2006 Functional trait niches of North American lotic insects: traits-based ecological applications in light of phylogenetic relationships. *J. North Am. Benthol. Soc.* 25, 730–755.
- R Core Team, 2017 R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria URL <https://www.R-project.org/>.
- Relyea CD, Minshall GW, Danehy RJ, 2012 Development and validation of an aquatic fine sediment biotic index. *Environ. Assess.* 49, 242–252.
- Roni P, Beechie TJ, Bilby RE, Leonetti FE, Pollock MM, Pess GR, 2002 A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in pacific northwest watersheds. *North Am. J. Fish. Manage.* 22, 1–20. 10.1577/1548-8675U002)022<0001:AROSRT>2.0.CO;2.
- Rosburg TT, Nelson PA, Bledsoe BP, 2017 Effects of urbanization on flow duration and stream flashiness: a case study of Puget Sound Streams, western Washington, USA. *J. Am. Water Resour. Assoc. (JAWA)* 1–15. 10.1111/1752-1688.12511.
- Roy AH, Rosemond AD, Leigh DS, Paul MJ, Wallace JB, 2003 Habitat-specific responses of stream insects to land cover disturbance: biological consequences and monitoring implications. *J. North Am. Benthol. Soc.* 22 (2), 292–307.

- Smith Alexander, Bode Robert W., Kleppel Gary S., 2007 A Nutrient Biotic Index (NBI) for use with benthic macroinvertebrate communities. *Ecol. Ind.* 7, 371–386. 10.1016/j.ecolind.2006.03.001.
- Statzner B, BEChe LA, 2010 Can biological invertebrate traits resolve effects of multiple stressors on running water ecosystems? *Freshwater Biol.* 55 (s1), 80–119.
- Thorpe JH, Covich AP, 1991 *Ecology and Classification of North American Freshwater Invertebrates*. Academic Press, San Diego.
- U.S. EPA, 1993 Region 10 In-stream Biological Monitoring Handbook For Wadeable Streams in the Pacific Northwest EPA 910/9–92-013. Region 10, Environmental Services Division, Seattle, WA 98101.
- U.S. EPA, 2000 Stressor Identification Guidance Document. EPA/822/B-00/025. Office of Water, Office of Research and Development, Washington, DC, USA Available at: <http://www.epa.gov/waterscience/biocriteria/stressors/stressorid.html>.
- U.S. EPA, 2007 CADDIS (Casual Analysis Diagnosis Decision Information System). Available at www.epa.gov/caddis.
- U.S. EPA, 2016 Appendix D. Development of A Background to Criterion Regression Model Public Review Draft: Field-Based Methods for Developing Aquatic Life Criteria for Specific Conductivity. U.S. Environmental Protection Agency, Office of Water, Washington, DC EPA-822-R-07–010. Available online at: <https://www.epa.gov/wqc/draft-field-based-methods-developing-aquatic-life-criteria-specific-conductivity>.
- U.S. EPA and United States Geological Survey. 2012 National Hydrography Dataset Plus Version 2. Accessed 2016. Available online at: <https://www.epa.gov/waterdata/nhdplus-national-hydrography-dataset-plus> Vieira.
- Vieira Nicole, K.M., Poff N. LeRoy, Carlisle Daren M., Moulton Stephen R., II, Koski Marci L., Kondratieff Boris C., 2006, A database of lotic invertebrate traits for North America: U.S. Geological Survey Data Series 187, <http://pubs.water.usgs.gov/ds187>.
- Vitousek P, Mooney H, Lubchenco J, Melillo J, 1997 Human domination of Earth's ecosystems. *Science* 277, 494–499.
- Vörösmarty CJ, McIntyre PB, Gessner MO, Dudgeon D, Prusevich A, Green P, Glidden S, Bunn SE, Sullivan CA, Liermann CR, Davies PM, 2010 Global threats to human water security and river biodiversity. *Nature* 467 (7315), 555–561. [PubMed: 20882010]
- Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan II RP, 2005 The urban stream syndrome: current knowledge and the search for a cure. *J. North Am. Benthol. Soc.* 24, 706–723.
- Wang Lizhu, Kanehl Paul, 2003 Influences of Watershed Urbanization and Instream Habitat on Macroinvertebrates in Cold Water Streams. *J. Am. Water Resour. Assoc. (JAWRA)* 39 (5), 1181–1196.
- Wenger SJ, Roy AH, Jackson CR, Bernhardt ES, Carter TL, Filoso S, Gibson CA, Hession WC, Kaushal SS, Martí E, Meyer JL, 2009 Twenty-six key research questions in urban stream ecology: an assessment of the state of the science. *J. North Am. Benthol. Soc.* 28 (4), 1080–1098.
- Wilhelm JO, Leinenbach P, Fore L, Lester D, Adams K, Hayslip G, 2013 Watershed Delineation and Land Cover Calculations for Puget Sound Stream Basins. King County Department of Natural Resources and Parks, Seattle, Washington.
- Yuan LL, 2007a Using biological assemblage composition to infer the values of covarying environmental factors. *Freshw. Biol.* 52, 1159–1175.
- Yuan LL, 2007b Maximum likelihood method for predicting environmental conditions from assemblage composition: the R package bio.infer. *J. Stat. Softw.* 22, 1–20.
- Yuan LL, 2014 bio.infer: Predict environmental conditions from biological observations. R package version 1.3–3. <https://CRAN.R-project.org/package=bio.infer>.

Fig. 1. Conceptual model for Soos Creek watershed describing hypothesized steps in causal pathway from human induced activity to biological impairment. All eight proximate stressors were evaluated in the current case study. Pathways deemed unlikely in this analysis shown with red X. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Fig. 2. (a) Average B-IBI scores for R sites versus N and I sites from the Soos basin. (b) Average B-IBI scores for the seven I sites from the Soos basin. Higher B-IBI scores indicate healthier macroinvertebrate communities. Sample sizes shown in parentheses.

Fig. 3.

(a) Average B-IBI scores versus average HPC for each modeled region in the Soos Creek basin. Red circles indicate I sites from the Soos basin. (b) Added variable plots for B-IBI versus HPC, relative to the other predictor variables already in the multiple regression model (i.e., percent impervious cover and area), (c) B-IBI versus percent impervious cover, relative to the other predictor variables already in the model, and (d) B-IBI versus area, given the other predictor variables already in the model. Summary.

Fig. 4.

(a) Average FSBI scores for R sites versus N and I sites from the Soos basin. (b) Average FSBI scores for the seven I sites from the Soos basin. (c) Average inferred percent fines for R sites versus N and I sites from the Soos basin. (d) Average inferred percent fines for the seven I sites from the Soos basin. Higher FSBI scores indicate healthier macroinvertebrate communities. Higher inferred percent fines scores indicate sites more impacted by sediment. Sample sizes shown in parentheses.

Fig. 5.

(a) Average HBI scores for R sites versus N and I sites from the Soos basin. (b) Average HBI scores for the seven I sites from the Soos basin. (c) Average MTI scores for R sites versus N and I sites from the Soos basin. (d) Average MTI scores for the seven I sites from the Soos basin. Higher HBI scores indicate macroinvertebrate communities more impacted by organic pollution. Higher MTI scores indicate more impacted macroinvertebrate communities. Sample sizes shown in parentheses.

Fig. 6.

(a) Average Cold Stenotherm Richness for R sites versus N and I sites from the Soos basin. (b) Average Cold Stenotherm Richness for the seven I sites from the Soos basin. (c) Average inferred water temperature for R sites versus N and I sites from the Soos basin. (d) Average inferred water temperature for the seven I sites from the Soos basin. Higher Cold Stenotherm Richness scores indicate healthier macroinvertebrate communities. Higher inferred temperatures indicate more impacted macroinvertebrate communities. Sample sizes shown in parentheses.

Fig. 7. Model estimated shade deficit for major streams in Soos Creek Watershed, WA. Colored circles indicate shade deficit measurements every 200 meters, with green values indicating minimal shade deficit and red values indicating high shade deficit. Hydrology shown in blue from NHDPlusV2 (U.S. EPA and USGS 2012). Black cross haired circles indicate sample locations that are impaired for bioassessment (sites A-G).

Table 1

Mean values $\pm$ standard error for B-IBI scores (10–50 scale) and individual component metrics for data used in the Soos Creek stressor identification. R indicates reference sites, N indicates Soos non-impaired sites, and I indicates Soos impaired sites.

Variable	Response to Stress	R	N	I
# Samples		12	150	64
B-IBI	Decrease	39.5 $\pm$ 1.5	32.9 $\pm$ 0.4	26.7 $\pm$ 0.8
Taxa richness	Decrease	53.2 $\pm$ 1.5	31.2 $\pm$ 0.5	27.2 $\pm$ 1.0
Ephemeroptera richness	Decrease	8.7 $\pm$ 0.7	5.2 $\pm$ 0.2	3.9 $\pm$ 0.2
Plecoptera richness	Decrease	6.8 $\pm$ 0.7	6.2 $\pm$ 0.1	4.5 $\pm$ 0.3
Trichoptera richness	Decrease	8.5 $\pm$ 0.5	6.3 $\pm$ 0.1	5.1 $\pm$ 0.2
Clinger richness	Decrease	22 $\pm$ 1.3	16.3 $\pm$ 0.3	11.2 $\pm$ 0.5
Long-Lived richness	Decrease	3.8 $\pm$ 0.3	5.4 $\pm$ 0.2	3.5 $\pm$ 0.2
Intolerant richness	Decrease	4.3 $\pm$ 0.9	1.4 $\pm$ 0.1	0.625 $\pm$ 0.1
Percent dominant	Increase	43.9 $\pm$ 3.0	55.7 $\pm$ 0.9	59.8 $\pm$ 1.7
Predator percent	Decrease	15.3 $\pm$ 2.0	12.4 $\pm$ 0.6	8.9 $\pm$ 0.9
Tolerant percent	Increase	10.9 $\pm$ 3.3	9.7 $\pm$ 0.9	17.1 $\pm$ 01.5

Table 2

Level I habitat assessments conducted for each impaired location within the Soos Creek watershed (see Fig. S1) in accordance with U.S. EPA guidance for wadable streams (U.S. EPA, 1993). Locations (in columns) were scored separately for low gradient and high gradient stream habitat, respectively. Dashes indicate parameter (in rows) were not pertinent to that location. Individual scores are color mapped based on parameter range, with the highest score in green and lowest score in red.

	Location							Range
	A	B	C	D	E	F	G	
bottom substrate	11		12					0-20
pool substrate		7		12	6	6	4	
instream cover	10	6	9	10	3	8	6	
embeddedness	11		16					
pool variability		3		5	3	7	3	
velocity/depth								0-15
canopy cover	11		8					
channel shape		7		15	2	9	2	
pool:riffle	11	5	6	5	5	7	3	
sinuosity	4		11					
width:depth		5		5	4	4	6	0-10
	9	6	10	5	5	4	5	
bank vegetation	5	7	3	5	5	5	6	
lower bank stability	6	5	9	5	6	6	9	
disruptive pressures	5	4	2	2	3	4	4	
zone of influence	4	2	5	2	3	3	5	0-165
Overall Score	87	57	91	71	45	63	53	
	C	D	B-	C	D	C	D	

Table 3

Mean values for various water quality parameters for sites in Soos Creek basin with comparisons to regional reference and randomly sampled sites (Puget Lowland Level III Ecoregion). Values in parentheses are the observed minimum and maximum values respectively. WHM = random sites from ECY's Watershed Health Monitoring project and SAM = random sites from ECY's Stormwater Action Monitoring project.

Parameter	Soos Basin	Reference	WHM	SAM
<i>Water Chemistry</i>				
Chloride mg·L ⁻¹	2.15 (1.4–3.05)	5.51 (0.95–20.3)	4.7 (0.48–31.3)	5.59 (0.99–18.6)
Conductivity µS cm	65.23 (29.07–144.8)	129.59 (46.85–317)	128.98 (29.7–647.85)	171.45 (49.93–628.37)
Turbidity NTU	3.04 (0.51–17.5)	1.38 (0.18–3)	5.5 (0.1–92.7)	2.83 (0.33–32)
TSS mg·L ⁻¹	8.62 (0.74–85.5)	3.44 (1–14)	11.11 (1–168)	6.43 (0.5–167)
Dissolved O ₂ mg·L ⁻¹	10.51 (5.67–13.2)	9.96 (7.4–11.14)	9.42 (1.85–12.3)	9.27 (3.8–11.45)
pH	7.35 (6.63–8)	7.56 (6.09–8.19)	7.31 (5.24–8.54)	7.52 (6.41–8.72)
Total P µg·L ⁻¹	36.28 (7.3–226.23)	19.73 (5.6–85.8)	36.45 (5–148)	47.38 (5–300)
SRP µg·L ⁻¹	6.42 (1–53.5)	–	–	29.5 (0.62–111)
Total N µg·L ⁻¹	–	462.38 (60–1890)	309.3 (32–2520)	883.96 (84–4740)
NH ₃ µg·L ⁻¹	15.51 (10–149)	–	–	14.4 (10–55)
Nitrate/Nitrite µg·L ⁻¹	299.8 (29–1180)	–	–	–
<i>Sediment Chemistry</i> ^a				
Arsenic mg·kg ⁻¹	8.27 (6.14–11.56)	4.09 (1.49–7.55)	3.98 (0.51–14.70)	11.87 (0.39–99.8)
Cadmium mg·kg ⁻¹	0.146 (0.066–0.271)	–	–	0.48 (0.04–7.01)
Chromium mg·kg ⁻¹	16.09 (11.86–22.66)	–	–	50.3 (2.74–280)
Copper mg·kg ⁻¹	13.34 (11.14–15.09)	15.9 (8.39–29.6)	26.68 (6.2–88.2)	47.54 (2.13–398)
Lead mg·kg ⁻¹	14.54 (4.96–20.64)	3.72 (1.83–7.28)	5.38 (0.58–33.4)	27.43 (2.13–219)
Mercury mg·kg ⁻¹	0.047 (0.031–0.063)	–	–	–
Nickel mg·kg ⁻¹	16.77 (12.52–21.77)	–	–	–
Silver mg·kg ⁻¹	0.035 (0.023–0.051)	–	–	0.15 (0.01–1.46)
Zinc mg·kg ⁻¹	59.97 (38.83–85.93)	45.38 (34–92.8)	53.86 (17–119)	138.28 (9.18–516.5)
Total PAHs µg·kg ⁻¹	2224 (65.34–6469)	380.8 (270–618)	505.49 (238.4–3430)	1011.58 (108.8–27702)
Total PCBs µg·kg ⁻¹	1.89 (1.28–2.49)	–	–	2.54 (0.97–4.07)

^aSediment Quality Standard/Screening Level 1 values from Michelsen 2011 (Arsenic = 14mg·kg⁻¹, Cadmium = 2.1mg·kg⁻¹, Chromium = 72mg·kg⁻¹, Copper = 400mg·kg⁻¹, Lead = 360mg·kg⁻¹, Mercury = 0.66mg·kg⁻¹, Nickel = 26mg·kg⁻¹, Silver = 0.57mg·kg⁻¹, Zinc = 3200mg·kg⁻¹, Total PAHs = 17,000 µg·kg⁻¹, Total PCBs = 110µg·kg⁻¹).

Table 4

Summary of weight of evidence for causal assessment. Columns show candidate causes and rows show main characteristics of causal evidence. Cells are color coded blue/green to indicate evidence that strengthens the case and red/orange to indicate evidence that weakens the case. Scoring system for

evidence: ++ (strongly supports), + (somewhat supports), 0 (no effect/neutral, or ambiguous), – (somewhat weakens), – – (strongly weakens), and NE (no evidence).

Characteristic of Causal Relationship	Principle	flow alteration	↑ fine sediments	↓ habitat complexity	↑ temperature	↓ dissolved oxygen / ↑ nutrients	↑ metals	↑ ionic strength	↑ toxic pollutants
Antecedence	Each causal relationship is a result of a larger web of cause and effect relationships.	HPC is moderately strong predictor for B-IBI despite elimination of steps in other causal pathways (Fig. 4b,c) (++)	Impervious cover alone not strong, but with HPC is moderately strong predictor for B-IBI (Fig. 4b,c) (++)	Consistent and coherent relationship with other likely candidate causes: high flows destroy habitat through bank erosion/incision, thereby generating mobilized fine sediment, channelized streams rapidly convey flow and disrupt natural life history patterns (Fig. 2) (++)	Modeled shade-similar for R, N, I sites (Fig. 8) (–)	Impervious cover is known to increase nutrients and cause increased biological oxygen demand, thereby decreasing ambient oxygen levels in other studies (+)	Impervious cover is known to increase metals in other studies (+)	Impervious cover is known to increase toxic pollutants in other studies (+)	Impervious cover is known to increase toxic pollutants in other studies (+)
Co-occurrence	The cause co-occurs with the unaffected entity in space and time.	Inconsistent: five of seven I sites within top quartile of HPC values, however, two I sites within lower quartile (0)	Site inspections indicated high sediment at I sites A and C in riffles and very high in pools sites B, E, F, G (Table 2) (++)	Site inspections show general pattern of degradation across wide range of microscale and far field habitat indices at all I sites (Table 2) (++)	Actual and inferred temperatures similar for R, N, I sites (–)	DO, phosphorus and nitrogen do not exceed regional background (–)	Metal concentrations were similar for N and I sites and lower than R sites (Table 3) (–)	Conductivity and chloride were lower for I sites compared to R, N sites (Table 3) (–)	Toxic pollutants were never detected in any sample despite comprehensive and targeted monitoring (–)
Sufficiency	The intensity, frequency, and duration of the cause are adequate and the entity is susceptible to produce the type and magnitude of the effect.	Other studies indicate chronic flow alteration (i.e., HPC) limits observed biological potential (i.e., B-IBI) (Benoy et al., 2012; Cormier et al., 2008). Sediment transport throughout Puget Sound region (+)	In other studies, % fines of marshes and 12% causes loss of taxa (Benoy et al., 2012; Cormier et al., 2008). Sediment transport throughout Puget Sound region (+)	Other studies show sufficient channel complexity/reduced habitat complexity/patch disturbance to disrupt dispersal/predator-prey interactions/competition/habitat use/resource acquisition, improved biological recruitment after restoration efforts (+)	All observed and modeled temperature values less than 20°C with majority below 15°C (–)	DO levels do not violate water quality criteria based on moderately tolerant species. Model-based data show low nutrient levels and DO greater than 10 mg/L (–)	Difference in conductivity not enough to account for the difference in impairment between N and I sites (–)	Difference in conductivity not enough to account for the difference in impairment between N and I sites (–)	Difference in toxic pollutants not enough to account for the difference in impairment between N and I sites (–)
Alteration	The entity is changed by the interaction with the cause.	Reduced eluvi long lived taxa indicative of unimpacted life cycle-dependent from chronic high flow events (++)	FSBI low for I sites, and higher for R and N sites (Fig. 5a,b) (++)	Reduced taxa richness/ % dominant/ % predator and increased % tolerant taxa (Table 1) (+)	Cold stenotherm richness is similar for R, N, I sites (Fig. 7a,b) (–)	Biol. index, MTI, HBI organic matter/DO index R, N, I similar (Fig. 6a,b) (–)	Biological index, MTI opposite from expected for metals as a cause, I is least tolerant, R is most tolerant (Fig. 6c,d) (–)	Salinity intolerant genera are present at I sites (–)	Toxic pollutants were not detected at N or I sites (–)
Interaction	The cause physically interacts with the entity in a way that induces the effect.	NE	NE	NE	Biologically inferred temperature is same for R, N, and I sites (–) DO less than 14 °C (Fig. 7c,d) (–)	Taxa preferring low nutrient concentrations present equally at all sites (–) DO greater than 14 °C (Fig. 7c,d) (–)		NE	NE
Time order	The cause precedes its effects.	NE	NE		NE	NE	NE	NE	NE
Summary		Likely.	Most Probable cause for greater impairment at I sites than N sites.	Likely.	Unlikely cause, but temp. spikes may be a problem.	Unlikely because levels similar at I and N sites; biological response not consistent with anticipated effect.	Unlikely because biological response in opposite direction with concentration.	Unlikely because levels similar at I and N sites.	Plausible but unlikely: a variety of pollutants were never detected during comprehensive and targeted monitoring efforts.

(See above-mentioned reference for further information).

EPA Author Manuscript

EPA Author Manuscript

EPA Author Manuscript