Assessing and Restoring the Health of Urban Streams in the Puget Sound Basin

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Abstract: Rapid urbanization threatens the biota of streams and rivers around the globe. Efforts to manage urban streams traditionally take an engineering approach focused on stormwater runoff, physical channel condition, and chemical water quality. Our objective was to use the biology of streams-measured with the multimetric bentbic index of biological integrity (B-IBI) based on bentbic macroinvertebrates—to assess stream health. From 1997 to 1999, we sampled invertebrates at 45 sites in second- and third-order streams in the Puget Sound lowlands of Washington State. Land cover upstream of each site was characterized by analysis of a 1998 satellite image. We evaluated associations between five land cover categories and biological condition across three spatial scales. The relationships between B-IBI (and its component metrics) and stream substrate and hydrologic features were also analyzed at a subset of sites. Across all study sites, B-IBI declined as the percentage of urban land cover increased (r < -0.71, p < 0.001, n > 31). Most metrics were better predicted by sub-basin rather than local-scale urbanization. Within individual basins, however, local land-cover urbanization and B-IBI were strongly correlated (r = -0.91, p < 0.001, n = 9). The biological condition of a site was also related to measures of hydrologic alteration and stream substrate. The aquatic biota is sensitive to a variety of urban effects, expressed at both large and small spatial scales. Biological assessment tools such as B-IBI can identify areas of excellent biological condition for conservation and guide the design and evaluation of efforts to restore the biota of degraded streams.

Evaluación y Restauración de la Salud de Arroyos Urbanos en la Cuenca Puget Sound

Resumen: La urbanización rápida amenaza la biota de los arroyos y ríos alrededor del mundo. Los esfuerzos para manejar arroyos urbanos requieren tradicionalmente una metodología de ingeniería enfocada en los escurrimientos torrenciales, las condiciones físicas de los canales y la calidad química del agua. Nuestro objetivo fue usar la biología de los arroyos—medida como un índice béntico multimétrico de la integridad biológica (B-IBI) basado en macroinvertebrados bénticos—para evaluar la salud del arroyo. De 1997 a 1999, muestreamos invertebrados en 45 sitios en arroyos de segundo y tercer grado de las tierras bajas del Puget Sound, Estado de Washington. La cobertura de tierra arroyo arriba para cada sitio fue caracterizada mediante un análisis de imagen satelital de 1998. Evaluamos las asociaciones entre cinco categorías de coberturas del suelo y las condiciones biológicas a lo largo de tres escalas espaciales. Las relaciones entre B-IBI (y sus componentes métricos) y el substrato del arroyo y las características bidrológicas fueron también analizadas para un subconjunto de sitios. A lo largo de todos estos sitios de estudio, B-IBI disminuyó cuando el porcentaje de cobertura urbana incrementaba (r < -0.71, p < 0.001, n > 31). La mayoría de los componentes métricos fueron predichos mejor por las sub-cuencas que por la escala de urbanización local. Sin embargo, dentro de las cuencas individuales, la cobertura del suelo por urbanización y B-IBI estuvieron fuertemente correlacionadas (r = -0.91, p <0.001, n = 9). La condición biológica en los sitios estuvo también relacionada con las medidas de alteración hidrológica y el substrato del arroyo. La biota acuática es sensible a una variedad de efectos urbanos—expresados tanto a escalas espaciales grandes como pequeñas. Las herramientas de evaluación biológica tales como B-IBI pueden identificar áreas de excelente condición biológica para la conservación y conducir el diseño y evaluación de los esfuerzos para restaurar la biota de arroyos degradados.

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Introduction

Urbanization degrades streams and rivers and contributes to decreased ecological health in watersheds (Karr et al. 1985a, 1985b), on continents (Ricciardi & Rasmussen 1999), and around the globe (Baer & Pringle 2000). By 2025 perhaps 60% of people globally (Young et al. 1994) and 83% in Europe and the Americas (Sheehan 2001) will live in cities. In the United States, metropolitan areas now cover more than 19% of the total land surface, include more than 75% of the human population, and consume over 800,000 ha of open space annually (Stoel 1999; Mitchell 2001). As one consequence, many urban streams today are highly engineered channels designed more for flood control and sediment transport than for ecological considerations (Roesner 1997). By 1972 over 200,000 miles of streams and rivers in the United States had been channelized (Riley 1998), and in many cities most streams are now in culverts (Finkenbine et al. 2000). Urbanization alters stream biotas in numerous ways: local extinction of anadromous fishes (Limburg & Schmidt 1990), increased dominance by pollution-tolerant invertebrates (Rossano 1996), and frequent algal blooms (Olguin et al. 2000).

Given increasing urban populations, such statistics are discouraging but not unexpected. More surprising is how little urban streams have been studied from an ecological standpoint. In a survey of the recent literature (Current Contents 1991-2001), we found fewer than 30 studies worldwide that involved any direct measurement of the biota of urban streams or rivers. Without knowledge of what species live in these systems, how they interact within the heavily modified stream environment, or how the biota responds to urbanization and the specific stressors that accompany such change, we are poorly equipped to craft effective conservation strategies for urban streams. Similarly, restoration efforts are far more likely to succeed when informed by knowledge of the causes of biological degradation rather than recognition of only the symptoms. In this respect, urban stream management is often a reaction to a crisis-flooding, sewage overflow, or endangered species-rather than a carefully planned action to avoid crises.

In the Pacific Northwest, the National Marine Fisheries Service recently added nine populations of Pacific salmon and trout to the endangered species list—the first time such protection has extended to a major metropolitan area of the United States (Gorman & Sears 1999). Many millions of dollars in local and federal restoration funds to aid salmon recovery efforts (Mapes 1999; Dloughy 2001) are directed toward the greater Seattle area, where Puget Sound chinook salmon (*Oncorbynchus tshawytscha*) are now listed as endangered. Most monitoring programs required for receipt of these funds focus on physical habitat; the responses of fishes, invertebrates, or other organisms are rarely directly evaluated to determine the effectiveness of specific restoration strategies (Roni et al. 2002). In the greater Seattle area, <5% of the restoration projects completed in the last decade have been evaluated according to any biological data (Larson et al. 2001). Restoration efforts that do not explicitly consider the living system upon which endangered fishes depend may compromise the overarching goal of healthy streams and rivers. Instead of focusing only on endangered species or their presumed physical habitat, an integrative ecological measure of restoration success is vital (Angermeier 1997).

Because declining biological conditions in running waters have many causes, a broad perspective is needed for their protection (Karr & Chu 1999). Rather than relying on physical or chemical measures as surrogates for biological condition, states such as Florida, Idaho, Ohio, and Vermont have developed narrative or numeric biological criteria to report on the condition of surface waters, to screen watersheds for further monitoring, and to evaluate specific management strategies (Davis et al. 1996). One common assessment approach is use of a multimetric index such as an index of biological integrity (IBI), which integrates empirically tested attributes (metrics) of stream biotas-most commonly fishes (Karr et al. 1986; Simon 1999), invertebrates (Ohio Environmental Protection Agency 1988; Kerans & Karr 1994), and algae (Fore & Grafe 2000; Hill et al. 2000a). Principal advantages of the multimetric approach are that it measures end-response variables of biological degradation, synthesizes the cumulative effects of a wide variety of environmental disturbances, and does so in a way that is easily understood by nonscientists (Keeler & McLemore 1996). In the Pacific Northwest, a regionally calibrated 10-metric IBI has been developed based on benthic invertebrates (B-IBI; Kleindl 1995; Fore et al. 1996; Karr 1998) and applied by water resource managers (King County 1996; Thornburgh & Williams 2000) and citizen volunteers (Fore et al. 2001).

Our intent is to go beyond conventional monitoring (sampling the biota of a place) to assessment (using the samples to evaluate the condition of the place and define the causes of degradation; Karr & Rossano 2001). Our starting point is to better understand how B-IBI responds when humans alter land cover and channel form and function. Many studies of urban systems use impervious area to characterize the level of urban development in stream basins (Schueler 1994; May et al. 1997; Finkenbine et al. 2000). In contrast, Karr and Chu (2000) argue that impervious area alone does a poor job of describing the diverse influences of urbanization. We used a recent, high-resolution land-cover classification (Hill et al. 2000b) to test alternative measures of basin urbanization as well as impervious area. Our objectives were to investigate (1) the relationship between stream biological condition and the extent (percentage of total land cover) and spatial distribution of urbanization, (2) the relationship of biological conditions to stream flow and substrate, and (3) the ways in which an improved understanding of these patterns can inform urban stream conservation efforts. Assessment of biological and physical responses associated with a set of in-stream restoration projects is discussed in a related study (Larson et al. 2001).

Methods

Study Design and Site Selection

From 1997 to 1999, we sampled 45 sites on 16 secondand third-order streams in the Puget Sound Lowland ecoregion (Omernik & Gallant 1986) of western Washington (Fig. 1), an area with more than 3 million people in 1997 and another million expected over the next 20 years (Puget Sound Regional Council 1998). In 1997 we selected 18 invertebrate monitoring sites across King and Snohomish counties to reflect a gradient of urban development. We resampled four of these sites the following year and added 23 sites, most concentrated in two basins (Little Bear and Swamp) to evaluate within-basin variation in biological condition. These two basins are similar in size, gradient, and geology. They differ primarily in extent of urbanization and proximity to the stream channel. Four invertebrate monitoring sites were selected in 1999 to evaluate in-stream restoration efforts. Sites located within or immediately below restoration projects were excluded from the land-cover analysis (but see Larson et al. 2001), as were sites where we were unable to accurately delineate basin boundaries for a given spatial scale.

To minimize natural confounding effects, we selected invertebrate monitoring sites with a limited range of elevation (5-140 m), gradient (0.4-3.2%), and drainage area (5-69 km²). We also excluded stream reaches immediately below bridges or culverts, those influenced by dams or other impoundments, and those with excessive point-source discharges or construction activity. Along the generalized gradient of urbanization represented by our study sites, a variety of stressors influenced the stream biota (e.g., invasive species, lack of large wood, and impaired water quality). Rather than examine every pathway of degradation, our intent was to capture the broader pattern of biological responses to urbanization. However, we did focus in greater detail on two elements of the physical habitat, flow and substrate, because (1) these features are typically highly altered in urban streams (Booth & Jackson 1997), (2) we wished to test the diagnostic properties of B-IBI by evaluating the response of specific metrics, and (3) data were readily available. Substrate data were collected in 1997 at each of the 18 invertebrate monitoring sites (Konrad 2000a). We performed hydrologic analysis at 11 invertebrate monitoring sites from this and an earlier study (Kleindl 1995) with sites located near flow-gauging stations.

Benthic Macroinvertebrates

We collected invertebrates from each site in September when flows are typically stable, taxa richness is high, and sites are easy to access (Fore et al. 1996). At each site, we used a Surber sampler (500-µm mesh, 0.1-m² frame) to collect three samples along the midline of a single riffle. Each sample was processed and identified separately without compositing or subsampling (Doberstein et al. 2000). We preserved invertebrates in the field in 70% ethanol and returned samples to the lab for identification to the lowest practical taxonomic level (typically genus; for exceptions see Morley 2000). Invertebrates were also classified according to functional feeding group, mode of existence, voltinism, and tolerance to human disturbance (Merritt & Cummins 1996).

We analyzed these data according to the 10-metric B-IBI (Karr 1998), an index that includes measures of taxa richness, tolerance of disturbance, and feeding ecology (Table 1). Following procedures first outlined for fishes (Karr et al. 1986) and later for invertebrates (Ohio Environmental Protection Agency 1988; Fore et al. 1996), we assigned metric scores of 1, 3, or 5 to each of the 10 raw metric values. These scores were then summed to obtain a site-specific B-IBI that ranged from 10 to 50. The higher the score, the healthier the site (excellent, 46–50; good, 38–44; fair, 28–36; poor, 18–26; and very poor, 10–16).

Land-Cover Analysis

We calculated extent of urbanization in each study basin at three spatial scales (i.e., areal extent): sub-basin, riparian, and local (Fig. 2). For the riparian and local scales, we selected a 200-m buffer width so as to include those functions commonly cited in association with riparian corridors (Gregory et al. 1991) but so as to avoid being unrealistically narrow given the relative accuracy of geographical data sets used in basin delineation and buffer analysis. We determined land cover from a 1998 satellite image (mapping resolution = 30 m) classified by Hill et al. (2000b) into seven categories of land cover. This multistep classification process consisted of image manipulation, identification of training sites, signature extraction, supervised classification, and accuracy assessment by comparison against digital orthophoto quarter-quadrangles (77% overall accuracy rate). The distribution of land-cover categories among our 16 study basins (Fig. 3) reflected the range of urban development found in the region. Combined forested categories ranged from 5% to 78% of the total area of a given basin and combined urban categories from 19% to 91%.

We tested five combinations of the seven land-cover categories for association with biological and physical stream condition: (1) percent coniferous (historic land cover for the region), (2) percent forested (coniferous + deciduous), (3) percent urban (urban forested + urban



Figure 1. Location of study basins and invertebrate-monitoring sites (\bullet) relative to the four largest cities (\blacktriangle) in the Puget Sound region (Puget Sound Regional Council 1998). Basin boundaries are delineated for the farthest sample site downstream on each stream basin; streams as drawn do not necessarily indicate perennial or above-ground flow.

grassy + intense urban), (4) percent intense urban (100% paved or bare soil), and (5) percent impervious area (based on coefficients calculated for each landcover category from orthophoto analysis; Hill et al. 2000b). We performed sub-basin delineation, stream buffering, and map overlays within the geographic information system programs ArcInfo and ArcView by generating flow-direction and flow-accumulation grids from 10-m-resolution digital elevation models. We verified the hydrography layer derived in this manner with the King County stream layer (accuracy = 12-24 m). We use the Pearson product-moment correlation coefficient (r_s) to test for association between land-cover measures (normally distributed) and B-IBI (also normally distributed; Fore et al. 1994) and the Spearman rank correlation coefficient (r_s) to test for association between land-cover measures and metrics of B-IBI (Zar 1999).

Substrate and Flow Evaluation

We evaluated three substrate and four hydrologic stream features in relation to biological condition. Size distribution of stream substrate was collected by Konrad (2000*a*) and characterized by a Wolman pebble-count (Wolman 1954) to generate measures of D_{50} (diameter at which 50% of pebbles are smaller), D_{16} (diameter at which 16% of pebbles are smaller), and relative roughness (RR; 84% pebble diameter divided by bankfull depth). In urban basins covered largely by impervious surfaces, increased overland flow provides greater opportunity for delivery of fine sediment to the

Table 1. The 10 metrics of the benthic index of biological integrity (B-IBI) and Spearman rank correlation coefficient (r_s) with percentage of urban land cover at two spatial scales

	Urban land cover			
Sample size (n)	sub-basin (34)	local (31)		
Taxa richness and composition				
total taxa richness	-0.41 ^b	-0.39^{b}		
mayfly taxa richness	-0.41 ^b	-0.29		
stonefly taxa richness	-0.65^{a}	-0.69^{a}		
caddisfly taxa richness	-0.59^{a}	-0.36^{b}		
long-lived taxa richness	-0.67^{a}	-0.37 ^b		
Tolerance and intolerance				
intolerant taxa richness	-0.33^{c}	-0.25		
tolerant taxa (%) ^d	+0.36 b	+0.47 ^b		
Feeding and other habits				
clinger taxa richness	-0.61^{a}	-0.46^{b}		
predators (%) ^d	$-0.48^{\ b}$	-0.60^{a}		
Other				
dominance by top 3 taxa $(\%)^d$	+0.67 a	+0.44 ^b		
a p < 0.001.				

 b p < 0.05.

 $^{c} p < 0.09$.

^d *The percentage is relative abundance.*

channel, particularly when there is construction activity in the basin (Dunne & Leopold 1978; Booth & Jackson 1997). These effects, coupled with increased stream-bank erosion and channel incision, led us to expect that D_{16} , D_{50} , and RR would all vary inversely with urbanization.

We used flow data from continuously recording gauging stations to calculate two measures of "flashiness," which is the increase in frequency and magnitude of peak flows relative to base flow, and two measures of the magnitude of peak flow. Flashiness measures were (1) fraction of the year that the daily mean discharge rate exceeds the annual mean discharge rate (T_{Qmean}) and (2) ratio of the annual maximum daily flow to the maxi-

Sub-basin

entire drainage area upstream from sample point



Riparian

200 m buffer on each side of sample point extending the length of the drainage network

Local

200 m buffer on each side of sample point extending 1 km upstream



Location of invertebrate-monitoring site

mum instantaneous flow $(Q_{max}:Q_{inst})$. Among Puget Sound lowland streams, T_{Qmean} generally varies inversely with urban development as a result of more rapid stormflow recession and lower wet-season base flow (Konrad 2000b). The ratio Q_{max} : Q_{inst} is also expected to vary inversely with urban development and was selected to capture the high peak discharge and rapid recession rates characteristic of flashy urban streams. Peak flow was measured as (1) maximum instantaneous flow divided by drainage area (Q_{inst} : DA) and (2) ratio of maximum daily flow to minimum daily flow $(Q_{\text{max}}; Q_{\text{min}})$. To account for variation in drainage area, we divided peakflow measures by either drainage area or minimum daily flow. We evaluated each of these seven substrate and flow measures relative to B-IBI (using r) and two to three B-IBI metrics selected based on relevant life-history information (using $r_{\rm s}$).

Results

Biological Condition

Although B-IBI varied from 10 to 48 across our 45 invertebrate monitoring sites, only 10% of sites were in good or excellent condition (B-IBI \geq 38). The best biological conditions were found at Rock Creek, one of the least urban sites, with 44 taxa present across the three replicates ($\bar{x} = 33.3$), including eight stonefly, nine longlived, and three intolerant taxa. Eleven percent of individuals present at Rock Creek were predators. Early signs of degradation were the loss of intolerant and longlived taxa, followed by an overall decline in taxa richness, especially mayflies, stoneflies, and caddisflies. At heavily affected sites, invertebrate assemblages were dominated by a few highly tolerant taxa. The most ur-

> Figure 2. Diagram of the three spatial scales used in the analysis of urban land cover based on a geographic information system.



Figure 3. Distribution of land-cover categories within each study basin. For definitions of specific landcover categories, see Hill et al. (2000b).

banized basin (Thornton Creek) had only 15 taxa across the three replicates ($\bar{x} = 11$), no stonefly or intolerant taxa, no predators, and only one long-lived taxon. Amphipods, chironomids, and a tolerant mayfly genus (*Baetis*) made up 89% of total individuals across the three replicates at Thornton Creek.

Land Cover

Of the five land-cover measures tested-percent coniferous, forested, urban, intense urban, and impervious areapercent urban was most strongly associated with B-IBI at all three spatial scales: sub-basin (r = -0.73, p < 0.001, n = 34), riparian (r = -0.75, p < 0.001, n = 34), and local (r = -0.71, p < 0.001, n = 31). In contrast, neither basin size (r = -0.14, p > 0.10, n = 39), gradient (r = +0.27, p > 0.10, n = 18), nor sample year (p > 0.10, two-tailed t test for sites sampled in more than 1 year, n = 4) explained significant variability in B-IBI. Elevation was positively correlated with B-IBI (r = +0.63, p < 0.001, n = 34) but inversely correlated with urban land cover (local scale; r = -0.43, p < 0.05, n = 31). Because forest and urban land cover were near perfect inverses (r = +0.99, p < 0.001, n = 34), forest cover was excluded from further analysis. Percent conifer, intense urban, and impervious area (Fig. 4a) were all significantly correlated with B-IBI (p < 0.001, $n \ge 31$), but less so than percent urban (Fig. 4b). Because riparian and sub-basin land cover were so closely correlated (r = +0.98, p < 0.001, n = 34),

we focused primarily on comparing sub-basin and localscale effects. Of the 10 metrics that composed B-IBI, 7 were more closely associated with sub-basin than local urbanization (Table 1). Stonefly taxa richness, relative abundance of tolerant taxa, and relative abundance of predator taxa were the exceptions.

Spatial Scale

Multiple sample sites on Little Bear (9) and Swamp Creek (8) provided further opportunity to examine the role of spatial scale in urbanization. Between these two basins, development differed in several important ways. At the sub-basin scale, Swamp Creek was more urbanized than Little Bear, with 70% versus 54% urban land cover, respectively. But at the local scale, Swamp Creek had a more continuously forested riparian corridor (40-62% forested) than Little Bear (11-67% forested), where lower reaches were denuded of riparian vegetation. In Little Bear, B-IBI was strongly associated with local urbanization (Fig. 5a). The maximum score (B-IBI = 40) on this stream occurred at the site with the least amount of local urban land cover (32%), whereas the low score (B-IBI = 16) occurred at a site with 71% local urban land cover. Sub-basin urban land cover varied less (49-54%) across the nine study sites on Little Bear Creek and was not correlated with B-IBI. Elevation along Little Bear Creek was positively correlated with B-IBI (r = +0.78, p < 0.05, n = 9) and negatively correlated with local urbanization (r = -0.89, p < 0.01, n = 9). In Swamp Creek, neither sub-basin nor local ur-



Figure 4. Relationship of B-IBI (benthic index of biological integrity) to (a) percentage of impervious area at the sub-basin scale and (b) percentage of urban land cover at the sub-basin scale.

ban land cover varied substantially (Fig. 5b), an observation that is concordant with limited variability in B-IBI (22-32 vs. 16-40 in Little Bear). Elevation along the nine invertebrate monitoring sites on Swamp Creek was not related to either B-IBI (r = +0.58, p > 0.10, n = 9) or to local urbanization (r = -0.26, p > 0.10, n = 8).

Physical Channel Condition

Urbanization may influence the stream biota through changes in flow or channel substrate. Both measures of hydrologic flashiness were positively correlated with B-IBI (Table 2), particularly the fraction of a year that the daily mean discharge exceeded the annual mean discharge





Figure 5. Relationship of B-IBI (benthic index of biological integrity) to urban land cover in (a) Little Bear Creek and (b) Swamp Creek. For Swamp Creek, invertebrate monitoring sites immediately below restoration projects are excluded, as is the most downstream site where local land cover could not be accurately determined.

 (T_{Qmean}) . The T_{Qmean} was also positively correlated with total taxa richness and richness of long-lived taxa. Neither measure of peak flow was related to B-IBI or metrics (Table 2). Relative roughness (RR) was positively correlated with B-IBI, EPT richness (total number of taxa of mayflies, stoneflies, and caddisflies), and clinger richness (total number of taxa classified as clingers; Table 2). Relative roughness was also inversely correlated with extent of urban land cover at both spatial scales tested (Table 2). The two particle-sized distribution measures $(D_{16} \text{ and } D_{50})$ were not correlated with B-IBI or urban land cover, but D_{16} was positively associated with both EPT richness and clinger richness (Table 2).

Table 2.	Spearman rank correlation coefficient	(r _s) for association	of benthic index	of biological integri	ity (B-IBI) and sele	cted metrics to
substrate	and flow features. ^a					

				Flow regime			
	Substrate		flashiness		peak flow		
	fines D ₁₆	median D ₅₀	rougbness RR	T _{Qmean}	Q_{\max} : Q_{inst}	Q _{inst} :D.A.	Q_{\max} : Q_{\min}
Urbanization, <i>n</i> sub-basin (%) local (%)	$17 \\ -0.20 \\ -0.12$	$17 - 0.35 - 0.49^{c}$	$17 - 0.60^{b} - 0.70^{b}$	$7 - 0.70^{c} - 0.09$	7 -0.29 -0.14	7 - 0.02 - 0.09	7 - 0.07 - 0.32
Biology, <i>n</i> B-IBI total taxa richness EPT richness clingers richness	18 +0.27 +0.34 +0.59b +0.60b	$18 +0.12 +0.17 +0.41^{c} +0.39$	18 +0.51b +0.43c +0.50b +0.52b	$ \begin{array}{c} 11 \\ +0.92^{b} \\ +0.76^{b} \\$	$10 + 0.85^{b} + 0.64^{c}$	10 -0.54 -0.13 	11 -0.08 +0.31
long-lived richness				$+0.73^{b}$	+0.44	-0.08	+0.26

^a Abbreviations: D_{50} , diameter at which 50% of pebbles are smaller; D_{16} , diameter at which 16% of pebbles are smaller; RR, relative roughness (84% pebble diameter divided by bankfull depth); T_{Qmean} , fraction of year that daily mean discharge rate exceeds mean discharge rate; Q_{max} : Q_{inst} , ratio of the annual maximum daily flow to maximum instantaneous flow; Q_{inst} : DA, maximum instantaneous flow divided by drainage area; $Q_{max}: Q_{min} =$ ratio of maximum daily flow to minimum daily flow; EPT, total number of taxa of mayflies, stoneflies, and caddisflies; —, correlation not evaluated. ^b p < 0.05. ^c p < 0.10.

Discussion

Human activity throughout the Puget Sound basin has altered the region's landscapes, with especially damaging effects on stream biota. Nearly half (23 of 45) of the stream sites we sampled were in poor or very poor biological condition (B-IBI ≤ 26). Most of those 23 sites lacked intolerant taxa, and at the most urbanized sites, we found no stoneflies. Such degraded conditions are typical of many streams in and around major metropolitan areas in the region (Karr & Chu 1999; Fore et al. 2001). The survival of wild salmon in the Pacific Northwest depends on many factors; crucial among them are high-quality streams for spawning, rearing of young, and migration. A key component of salmon habitat is the stream biota itself, yet virtually all current habitat-evaluation procedures include no direct biological measures. Benthic invertebrates are excellent indicators of stream condition because they are key components of the aquatic foodweb, sensitive to a variety of human disturbances, often long lived, and not migratory or artificially stocked (Rosenberg & Resh 1993; Fore et al. 1996). A stream with healthy bugs but no fish is likely being affected by other factors in the larger salmon landscape such as migration blockages, damaged coastal estuarine habitats, or downstream overfishing.

Measuring Urbanization: Going beyond Impervious Surface

Of the four measures of land cover we tested, a grouping of equally weighted urban land-cover categories explained a high degree of variability in B-IBI and metrics (Table 1). This simple definition of urbanization includes a variety of potential effects beyond those captured by the more traditional and narrowly focused imperviousarea models (Fig. 4a). Total impervious area (" . . . the sum of roads, parking lots, sidewalks, rooftops, and other impermeable surface . . . "; Schueler 1994) may be useful for modeling hydrologic modification, but it is not a reliable predictor of biological condition (Karr & Chu 2000) because streams are affected by many other stressors. Even in areas of the urban basin that are not paved over, compacted soils rarely retain the high infiltration rates associated with forested areas, and they reach saturation more rapidly with increased runoff from adjoining paved surfaces (Dunne & Leopold 1978). The total percentage of urban land cover in a basin is a straightforward and more inclusive measure of anthropogenic disturbance for use in conjunction with biological assessment. More comprehensive models currently being tested by the U.S. Geological Survey (McMahon & Cuffney 2000) may provide a more integrative measure of the effects of human actions.

The Importance of Spatial Scale

A broader view of disturbance includes an examination of the influences of urban development on stream condition over multiple spatial scales. The B-IBI in our streams responded strongly to changes in land cover at both sub-basin and local scales. Our results agree with those of recent studies, illustrating both the importance of land-cover changes basinwide (Richards et al. 1996; Roth et al. 1996; Allan et al. 1997) and the ecological importance of local land use (Steedman 1988; Scarsbrook & Halliday 1999). The responses of individual metrics in our study to land cover further demonstrate that stream biota are sensitive to effects expressed at both large and small spatial scales. The number of stonefly taxa at a site, for example, was more closely related to local land cover, whereas the number of long-lived taxa was better correlated with sub-basin land cover (Table 1). Although we did not test the specific mechanisms that drive these relationships, we hypothesize that stoneflies, many of which are shredders (feeding on leaf detritus) and sensitive to high temperatures (Merritt & Cummins 1996), may be responding to local riparian effects on food supply and shading. With further research, improved understanding of associations between specific B-IBI metrics and urban stressors will likely prove invaluable in the diagnosis of causes of degradation (Yoder & Rankin 1998).

Physical Channel Condition

While a gradient of urbanization is a general integration of human influence, the physical habitat measures (flow and substrate) we evaluated reflect specific effects on stream biota. Stream invertebrates are adapted to strong currents, but few persist under conditions of extreme and unpredictable flow fluctuation (Irvine 1985; Borchart & Statzner 1990). In urban basins of the Pacific Northwest, a shift from subsurface to overland flow has profoundly altered the delivery of water and sediment to stream channels (Booth & Jackson 1997). We found that measures of D_{16} , relative roughness, and hydrologic flashiness were correlated with B-IBI and/or individual metrics. High values of relative roughness may indicate a greater diversity of hydraulic conditions (e.g., availability of slow-water refugia at a microhabitat scale) during highflow events (Davis & Barmuta 1989; Borchart 1993). The positive relationship among D_{16} , EPT, and clinger taxa richness suggests that a shift toward smaller particle sizes may potentially foul invertebrate attachment sites in riffles (Karr & Chu 1999). Although the two most urban basins in this analysis were also the flashiest hydrologically and most degraded biologically, with our low sample size we detected only a weak inverse correlation between basin urbanization and T_{Qmean} ($r_{\text{s}} = -0.70, p = 0.09, n = 7$). That neither measure of peak flow was related to B-IBI or metrics suggests that invertebrates were responding more to the degree of flow fluctuation than to the magnitude of peak events.

Natural Sources of Biological Variation

Our success in documenting and understanding the effects of human actions requires that we control for other sources of variation in the biological character of streams. Factors such as ecoregion, stream size, and microhabitat sampled were rigorously controlled in our sample design. For the same reason, we also selected streams from a limited range of elevations (5-140 m). Even within that limited range, however, our results show a significant correlation between elevation and human influence and between elevation and B-IBI. The positive correlation between B-IBI and elevation among

our invertebrate monitoring sites is another illustration of the spread of urbanization. Throughout the Pacific Northwest, development has been most intense along waterways and lowland areas (Omernik & Gallant 1986). Although the nine invertebrate monitoring sites on Swamp Creek spanned a greater elevational gradient (5–120 m) than sites on Little Bear (10–100 m), elevation and B-IBI were not correlated across our Swamp Creek study sites, where urbanization intrudes less into the riparian corridor.

Management Applications

The biological information contained within B-IBI (or other similar assessment tools; e.g., Davis & Simon 1995) has much potential to inform restoration and conservation efforts. Although a decline in biological condition is inevitable as a result of increased urbanization, less obvious is the range of conditions found at a given level of development. In basins with 50% urban land cover, B-IBI ranged from 16 to 40 (Fig. 4b). The challenge of restoration is how to improve the condition of the most degraded sites. Tuning restoration efforts to site-specific needs is enhanced by using biology to aid in the detection of the primary causes of degradation. Multimetric indexes such as B-IBI provide a numeric synthesis of the biological dimensions of site condition, something that is lacking in current habitat management approaches, but they can also be broken down to derive descriptive and potentially diagnostic information from each of the component metrics (Karr et al. 1986). Investigation of relationships between specific metrics and particular effects of urbanization is sorely needed, because such knowledge could guide diagnosis of sitespecific causes of degradation (Yoder & Rankin 1995).

Along with the current management focus on restoring what is endangered, it is equally critical to protect streams and rivers that are still healthy (Trust for Public Land 2001). Protecting existing areas of high biological integrity is far easier than restoring or creating new habitats (Doppelt et al 1993; Roni et al. 2002). Yet in the central Puget Sound region, only 8% of parks and other protected green spaces in the four-county area are located close to or within urban areas (Puget Sound Regional Council 1998). That 92% are located primarily in the foothills and mountains of the Cascade Range illustrates the lack of conservation of lowland areas, which were once some of the most productive areas for salmon. Because funds with which to purchase or otherwise protect critical lands are limited, selection of the best areas for conservation should be guided by explicit biological knowledge of potential reserves. Simplicity is one advantage of multimetric indexes such as B-IBI; results are easily communicated and understood by nonscientists who can then use that information to lobby for conservation (Keeler & McLemore 1996; Steedman 1988). Many of the most vocal advocates for urban streams and rivers are volunteer organizations and local watershed councils (Karr et

al. 2000). In our study region, stewardship groups such as the Thornton Creek Alliance, the Little Bear Creek Protective Association, and Friends of Rock Creek Valley have used B-IBI to influence local allocation of restoration funds and the development of conservation plans.

The conservation of areas of high riparian quality is another critical task in protecting streams in regions already or likely to be urbanized. Our results indicate that the effectiveness of localized patches of riparian corridor in maintaining biological integrity is a function of basin-wide urbanization. In Little Bear Creek (where overall sub-basin urbanization was moderate, 49-54%), high B-IBI was associated with sites in headwater reaches with intact riparian corridors. Farther downstream, B-IBI decreased dramatically as local riparian vegetation was replaced by roads, houses, and commercial centers. When overall basin development is low to moderate, natural riparian corridors have significant potential to maintain or improve biological condition. Protecting high-quality wetland and riparian areas that persist in less-developed basins may also serve as a source of colonists (plants, invertebrates, fish) to neighboring streams undergoing restoration. Conversely, even small patches of riparian areas converted to urban land can severely affect local stream biota. As both a conservation and restoration strategy, protection and revegetation of riparian areas is critical for preventing severe stream degradation (Osborne et al. 1993), but these measures alone are not adequate to maintain biological integrity in streams draining highly urban basins (Roth et al. 1996).

To protect the healthy urban streams that remain and to restore those that are degraded, it is essential to focus on their overall biological health. The use of biological endpoints, rather than pollution-control dollars or numbers of permits issued as indicators of ecological health, will improve decision-making, save money, and improve our ability to protect the health of urban streams (Keeler & McLemore 1996; Karr & Chu 1999). As with chemical and physical parameters, no single measure of basin development is an acceptable surrogate for directly monitoring the biological condition of urban streams. And biological monitoring alone cannot tell us everything. We advocate an approach that combines direct biological assessment with physical, chemical, and landscape analysis to diagnose and repair stream degradation. To achieve meaningful long-term biological recovery, conservation and restoration efforts must extend beyond narrow conceptions of localized in-stream habitat manipulation (Larson et al. 2001) to examine the diverse cumulative effects operating across the entire basin (Ziemer 1997). Biological assessment tools such as B-IBI are essential to this process.

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