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LWMP Environmental Management

Assessment of the Use of B-IBI in Greater Vancouver Streams (2003–2006)

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PREFACE

Metro Vancouver (formerly the Greater Vancouver Regional District) is committed to the principle of managing liquid waste in a manner that protects the receiving environment while using cost effective approaches. This commitment is detailed in Metro Vancouver's Liquid Waste Management Plan (LWMP). The LWMP process is mandated by the Province of British Columbia and is designed to ensure an integrated, local approach to making good, informed liquid waste management decisions.

Commitment C4 of the LWMP states that "The District and member municipalities will undertake monitoring, assessment and forecasting to evaluate effects of wastewater and stormwater discharges to receiving environments". To meet this commitment, this study assesses the use of a Benthic Index of Biological Integrity as a monitoring tool to assess the effects of urbanization and stormwater discharges in Greater Vancouver. This monitoring is vital in providing information to effectively manage liquid waste discharges on a regional basis, and in furnishing a scientific basis for setting priorities and designing system upgrades.

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This study was guided and reviewed by both the Environmental Monitoring Committee and the Stormwater Interagency Liaison Group which are comprised of representatives from member municipalities, the GVRD, senior governments and the academic community. The members of the committees at the time that this study was conducted are listed below.

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SUMMARY

This report presents the final results of a four year study (2003–2006) which evaluated the temporal and spatial variability of a biological index derived from benthic invertebrate samples collected from fourteen streams and small rivers in Greater Vancouver. The study used a benthic index of biological integrity (B-IBI) which combines measures of taxa richness and relative abundance for key groups of benthic invertebrates such as tolerant taxa, predator taxa, and long-lived taxa. B-IBI is used as an indicator of stream condition.

The study's results support the use of benthic invertebrates as an effective assessment and monitoring tool in small urban watersheds in Greater Vancouver for four reasons. First, B-IBI values were consistently and predictably correlated with independent measures of human disturbance at both the instream and watershed scale. Second, B-IBI values consistently ranked streams sampled in different years in a similar order. Third, large differences in laboratory and field sampling protocols had little influence on the overall ranking of streams in terms of B-IBI values. Fourth, B-IBI has sufficient statistical power to detect small changes in resource condition. The consistent response of B-IBI to human disturbance, independent of protocol methods, along with its statistical precision, supports the use of the index as a management tool for small streams in Greater Vancouver.

The study streams included nine permanently flowing streams, two streams that represented the best available condition (reference streams) in the region, and three intermittently flowing (seasonal) streams. Two streams were sampled in two locations to assess spatial variability in the benthic invertebrate community. In each stream, a Surber sampler was used to collect samples (either single Surber placements or composites) from multiple riffles in a 500 m long reach. Data collected in 1999 was also included in the analyses.

B-IBI and different components of taxa richness were correlated with measures of urban land cover. Percent total imperviousness had the strongest correlation with biological variables in 2003 and 2004, while % riparian forest had a stronger correlation in 2005 and 2006. Most biological variables were correlated with total alkalinity, water temperature, and specific conductivity which are influenced by urban land cover. B-IBI and taxa richness variables generally decline as specific conductivity and water temperature increase. Biological variables were poorly correlated with stream channel characteristics in 2003 and 2004, however, % substrate embeddedness, % boulder, and % cobble were correlated with B-IBI, total taxa richness, and EPT taxa richness in 2005. The relationship between % boulder substrate and B-IBI was marginally significant in 2004. In 2006, EPT taxa richness, B-IBI (LPTL), and B-IBI (family) were correlated with riffle depth.

B-IBI values for 1999, 2003, 2004, and 2005 were all highly correlated indicating strong agreement between the four years of sampling (Pearson's $r>=0.9$). For 1999 vs. 2003, the mean difference in B-IBI was 3.4 points lower (SD of 4.1) in 1999. For 2003 vs. 2004 the agreement was closer with an average difference of 0.4 points difference (SD of 2.3). Similarly for 2004 vs. 2005, the average difference was small: 0.6 points (SD of 2.3) lower. Components of variance analysis estimated the relative contribution of year variance to be relatively small (3%) compared to the variance of B-IBI due to differences in site condition associated with human influence.

To evaluate differences associated with location along the stream, benthic invertebrates were collected at upstream locations on two streams in 2003, 2004, and 2005. B-IBI values were very similar for the pair of sites on Coghlan Creek (26.5–29.5). For 2003, B-IBI values differed by eight points for the pair of sites on Mossom Creek (30.0–38.0). This observed difference was likely due to problems obtaining a representative sample due to low water flow at the upstream site. In 2004 and 2005, a representative sample was collected and the difference between the two sites was small (33.5 vs. 37.0 in 2004 and 34.0 and 35.5 in 2005).

B-IBI values calculated on the basis of two different levels of taxonomic effort (family-level and lowest practical taxonomic level (LPTL)) were highly correlated (Pearson’s $r > 0.9$). Both versions of the index were highly correlated with watershed condition. The LPTL index was more statistically precise than the family-level index and could detect smaller changes in biological condition.

B-IBI values derived from two different field sampling protocols ranked sites similarly and were highly correlated (Pearson’s $r > 0.9$). The composite sampling protocol combined material from three Surber samples at each of four sites (12 Surber sampler placements); the single Surber protocol used the material from three individual Surber samples (3 Surber sampler placements). B-IBI values derived from composite samples tended to be slightly higher for less disturbed sites. Over 37% of the single Surber samples failed to yield the target number of 400 individuals per sample.

Statistical power analysis determined that B-IBI could reliably detect five categories of biological condition based on the comparison of sites with three grab samples each. For trend monitoring, we compared the ability of several different sampling scenarios to detect a change through time in B-IBI values. For example, based on a sampling scenario with 10 sites sampled once per year, an 8% change in B-IBI from the first to the second year would represent a statistically significant change; for four years of sampling a 3% change per year would be detectable.

Ordination using nonmetric multidimensional scaling showed broad similarities in benthic invertebrate community composition in different stream types. Samples from highly urbanized streams had very similar benthic invertebrate communities. Samples from moderately urbanized streams, rural-agricultural streams, and reference streams showed more variability, although all support benthic invertebrate communities that are substantially different from highly urbanized streams. Ordination also showed that samples collected from the same stream are very similar.

The similarity of B-IBI values in intermittent streams to permanent streams could not be assessed without B-IBI values from adjacent, and environmentally similar, permanent streams. As well, recent watershed land cover data was not available to evaluate the relationship of watershed scale variables (e.g., imperviousness) and B-IBI values in seasonal streams using regression analysis. Mean B-IBI values ranged from 12.0–13.5 in Tynehead Creek, 12.5–16.0 in Blueridge Creek, and 23.5–27.5 in Forestry Creek.

The sampling results from the two reference streams suggest they will be useful for tracking long-term change in undeveloped watersheds. B-IBI in Clear Creek ranged from 37.5 (2004), 34.0 (2005), and 32.5 (2006). B-IBI in Beaver Creek declined from 39.0 in 2004 to 32.0 in 2005 and then rebounded to 39.5 in 2006. The decline in B-IBI in 2005 may have been associated with culvert replacement above the sampling reach in October 2004. Mossom Creek may also be a suitable reference site: mean B-IBI was 38.0 (2003), 37.0 (2004), 34.0 (2005), and 38.0 (2006).

1.0 INTRODUCTION

Through the Liquid Waste Management Plan, Metro Vancouver and its member municipalities have committed to assessing effects of wastewater and stormwater discharges on the receiving environment. As well, member municipalities have committed to undertaking Integrated Stormwater Management Plans to manage stormwater infrastructure on a watershed scale (GVRD, 2002).

Since stormwater management is primarily the responsibility of the member municipalities, Metro Vancouver's role is to provide guidance on the use of the benthic index of biological integrity (B-IBI) and to assess its potential as a monitoring tool for assessing stream health and determining performance of the Integrated Stormwater Management Planning process. Specifically, biological monitoring is recommended in the Integrated Stormwater Management Planning Terms of Reference Template (KWL, 2002). In 2003, a B-IBI monitoring and assessment guidance manual (EVS, 2003a) was completed along with the design of a monitoring program to assess various uses of the B-IBI in Greater Vancouver (EVS, 2003b).

Biological monitoring based on benthic invertebrate sampling provides an integrative assessment of the environmental health of a stream because invertebrates spend most of their lives within the aquatic environment and experience the hydrologic, chemical, and physical conditions that occur throughout the year. Biological assessment integrates the influences of human activities that occur throughout the upstream catchment as well as through time. Biological monitoring can use any organisms including fish, stream invertebrates, or algae. In the Pacific Northwest, benthic invertebrates have been selected by many management agencies as the primary indicator of stream condition because they are easy to sample, are diverse and abundant in streambed substrates, and much is known about their tolerances, sensitivities, and specific responses to anthropogenic disturbance (Merritt and Cummins, 1996; Barbour *et al.*, 1999). Biological monitoring based on invertebrate sampling protocols is used by most U.S. state agencies (Davis and Simon, 1995; USEPA, 2002) and extensively in other geographic regions (Thorne and Williams, 1997; Duggan *et al.*, 2002; Simon, 2002).

Based on extensive work in the Pacific Northwest, numerous regional management agencies have adopted a multimetric approach called the benthic index of biological integrity (Karr, 1998; Karr and Chu, 1999; Fore *et al.*, 2001). B-IBI is a numeric index composed of metrics that summarize information on the taxa richness and structure of benthic invertebrate assemblages (see Table 2 and Section 1.4). The metrics selected for inclusion in the Pacific Northwest B-IBI were derived from numerous regional studies (Karr, 1998). B-IBI and its component metrics have shown a consistent response to a diverse set of human activities including timber harvest (Fore *et al.*, 1996), recreation (Karr, 1998), urbanization (Fore *et al.* 2001; Karr and Morishita Rossano, 2001; Morley and Karr, 2002), agriculture (Kerans and Karr, 1994), and mining (Mebane, 2001; Fore, 2002; Mebane, 2002). Expectations for metrics included in B-IBI were derived from values observed for stream invertebrate communities in undisturbed streams (Karr and Chu, 1999). To calculate B-IBI, the 10 metrics are scored as a 5, 3 , or 1 according to whether they are similar to values observed at reference sites (score = 5), deviate somewhat from expected values (3), or deviate substantially (1). Scores are added to obtain a B-IBI value that can range from 50 (indicating best condition) to 10 (indicating poorest condition). B-IBI has been regionally calibrated for lowland streams in Washington and Oregon where environmental conditions are similar to the Greater Vancouver.

Multimetric indexes are mathematically similar to economic indexes used to summarize and track changes in the economy (Fore *et al.*, 1994).

While considerable research has demonstrated that B-IBI is an effective tool for watershed assessment and monitoring throughout the Pacific Northwest, B-IBI is just beginning to be used in Greater Vancouver and relatively little work has been done to examine patterns of temporal and spatial variability in B-IBI for streams in this area. The statistical precision of B-IBI has important implications for its usefulness in the effective monitoring of stormwater management activities, such as the Integrated Stormwater Management Plans (ISMPs) under the Liquid Waste Management Plan. If annual variation is too high because of stochastic variation, changes in B-IBI due to watershed management activities may be difficult to detect. The ability to detect a statistically significant change in resource condition is a function of the precision of the index.

Variability of an index may be associated with differences in streams, natural or human influence, or other nuisance sources of variance, such as differences in time or location of sampling. A good monitoring protocol maximizes the variance associated with differences due to human disturbance, and minimizes variance associated with nuisance sources, such as year-to-year differences or location of sampling sites within a stream reach. A study in the Puget Sound region found that B-IBI had adequate statistical precision to detect five categories of biological condition using laboratory and field protocols similar to those used in this study (Fore *et al.*, 2001). Similar levels of precision for invertebrate multimetric indexes have been reported for Ohio (DeShon, 1995), Colorado (Fore, 2002), and the Mid-Atlantic region (Blocksom, 2003).

1.1 Study Goal and Objectives

The goals of this four year study were to evaluate the temporal and spatial variability of a biological index derived from benthic invertebrate samples, and to evaluate its effectiveness as a monitoring and assessment tool for urban watershed management in Greater Vancouver. Additional objectives included evaluating differences in B-IBI due to different locations within streams, differences associated with field sampling methods, differences associated with flow patterns, and differences associated with taxonomic effort during laboratory identification of samples.

To complement the statistical analysis of B-IBI variability, the relationship between B-IBI and its component metrics, such as taxa richness, were evaluated with independent measures of human disturbance made at the watershed and site scale. Nonmetric multidimensional scaling (NMDS), an ordination method, was used as a complementary analysis to examine differences in community structure between sampling sites and streams.

The Terms of Reference outlined six tasks that were the focus of the project:

1. Determine temporal variability (1999, 2003, and 2004) of B-IBI values in nine primary study streams in Greater Vancouver;
2. Determine differences in B-IBI values due to spatial differences (e.g., upper to lower watershed) in two streams in 2003, 2004, and 2005;
3. Compare the variability in B-IBI values due to differences in taxonomic effort (2003, 2004, 2005, and 2006);
4. Compare the variability in B-IBI values due to field sampling protocol (2003 and 2004);

5. Assess B-IBI values in three intermittently flowing streams (2004, 2005, and 2006); and,
6. Assess B-IBI values in two undisturbed streams (2004, 2005, and 2006).

1.2 Study Area

The Greater Vancouver area is approximately 3,300 km^2 and encompasses twenty-one municipalities in the western portion of the Lower Fraser Valley[1]. It is an area of diverse topography and land uses, but is increasingly characterized as the most heavily urbanized region in western Canada. Today, over one-half of British Columbia's population lives in Greater Vancouver or the communities immediately adjacent to it. The trend in population growth is expected to continue – between 1991 and 2021, Metro Vancouver's population is expected to increase from 1.6 million to almost 2.7 million, an increase of almost two-thirds (GVRD, 1996). Many watersheds are strongly affected by changes to land cover accompanying urbanization or agricultural use (EVS, 2000; Page *et al.*, 1999).

Streams and Rivers – Streams and rivers are a common feature of the Greater Vancouver landscape and a 1999 inventory described 132 watersheds and 165 catchments in the region (GVS&DD, 1999). The GVRD encompasses two distinct physiographic regions and stream channel character and hydrology reflect these broad differences: i) the Fraser Lowland and ii) the southern slopes of the Coast Mountains.

The Fraser Lowland is a triangular-shaped depositional feature that extends east from Point Grey to Laidlaw and south to Bellingham, lying between the Coast and Cascade Mountains. It encompasses the productive agricultural lands of Delta, Richmond, Surrey, and Langley. Streams in the Fraser Lowland are typically low-gradient (<2% slope) with sand and fine gravel substrates and extensive floodplains. However, many streams have been diked, enlarged or artificially straightened to improve drainage or prevent flooding. Fisheries values are generally high in streams in the Fraser Lowland and coho salmon, chum salmon, and cutthroat trout are common.

On the north side of the Fraser River and particularly in West and North Vancouver, stream channels are influenced by their location on the lower slopes of the Coast Mountains. Stream channels are steep and often contained in a gully or narrow valley, where coarse sediment is contributed by landslides and debris flows in the headwaters. In their lower reaches, many of these streams flow across a fan. Fish habitat is often more productive (e.g., off-channel habitats, complex channel structure) in the lower reaches while only cutthroat trout are found in the middle or upper reaches.

Climate and Stream Hydrology – The Greater Vancouver area has a modified maritime climate with a wet, cool winter and a pronounced summer dry period. Winter extends from late September or October until March and consists of a continual procession of Pacific westerlies onto the coast, occasionally broken by the formation of high pressure ridges. Roughly three-quarters of the annual precipitation falls in these months. Annual precipitation averages about 1,000 mm at Vancouver Airport, increasing to the north and, to a lesser extent, to the east, prior to uplift of air masses over the Coast and Cascade mountains. Annual precipitation reaches about 1,600 mm at the eastern edge of the lowlands and increases rapidly into the higher elevations of the Coast Mountains. The

[1] Metro Vancouver encompasses: Vancouver, Surrey, Delta, Township of Langley, City of Langley, Burnaby, New Westminster, District of North Vancouver, City of North Vancouver, West Vancouver, Port Moody, Port Coquitlam, Coquitlam, Maple Ridge, Pitt Meadows, Richmond, White Rock, University Endowment Lands.

headwaters of Mosquito Creek, one of the study streams, receive over 2,700 mm of precipitation annually.

The stream hydrograph of most streams in Greater Vancouver closely follows the precipitation cycle. The greatest monthly discharges occur in November, December, and January and maximum daily discharges are also usually recorded in these months. Flows generally decrease during the spring and summer until they reach a minimum in September. B-IBI sampling is typically undertaken between August 1 and September 30 during low flows. Flows vary greatly throughout the year and, in small watersheds, the maximum daily flow is often 500 times greater than the minimum daily flow.

Development History and Land Use – Since the turn of the century, strong population growth has occurred in Greater Vancouver with a commensurate increase in land clearing for agriculture and urban development. A study which examined the changes in land cover in the Lower Fraser basin from 1827 to 1990 determined that the area of coniferous forest cover changed from 71% prior to 1827 to 50% in 1930 to 54% in 1990. However, prior to 1827, only 27% of the forest was immature (<120 years old), while 40% was immature in 1930 and 73% of the forest was immature in 1990. The amount of wetland decreased from 10% to 1% while urban and agricultural area increased to 26% of the study area in 1990 (Boyle *et al*., 1997). Because these changes were calculated for the entire Fraser Valley from Hope to the mouth of the Fraser River, the deterioration in forest cover and increase in land conversion can be expected to be greater in the more populous Greater Vancouver area than the valley-wide averages presented above.

1.3 Study Streams

Fourteen streams were sampled for different components of this study. Table 1 provides a summary of basic information on each stream including watershed area, elevation, relief ratio, and stream length, and Table 2 summarizes the study streams in relation to site type and research questions. Figure 2 shows their location in Greater Vancouver. Appendix A provides a summary of sites based on site type and the number of samples.

Nine streams were sampled in 1999, 2003, and 2004 and provide the primary data for comparing temporal variability in B-IBI values. Five of the nine streams were also sampled in 2005 and two were sampled in 2006. They were selected to represent the range of land use and physiographic conditions in Greater Vancouver. Percent total impervious area ranged from 69% in Still Creek and 54% in the Como Creek watersheds, to less than 4% in the Mossom Creek watershed based on 1996 air photos (GVS&DD, 1999). These streams have been used for several studies including Page *et al.* (1999) and EVS (2000).

Two sites were added in 2003 to compare spatial differences in B-IBI values within a watershed: Mossom Creek (upper) and Coghlan Creek (upper). These sites are located in the central portion of their respective watersheds. In 2004, two additional streams were added to provide information on B-IBI values in undisturbed streams in Greater Vancouver: Beaver and Clear creeks are located in the undeveloped portions of the Greater Vancouver area's drinking water watersheds. They were sampled in 2004, 2005, and 2006. Three streams with intermittent flows were also added in 2004: Blueridge, Forestry, and Tynehead creeks; they were sampled in 2004, 2005, and 2006.

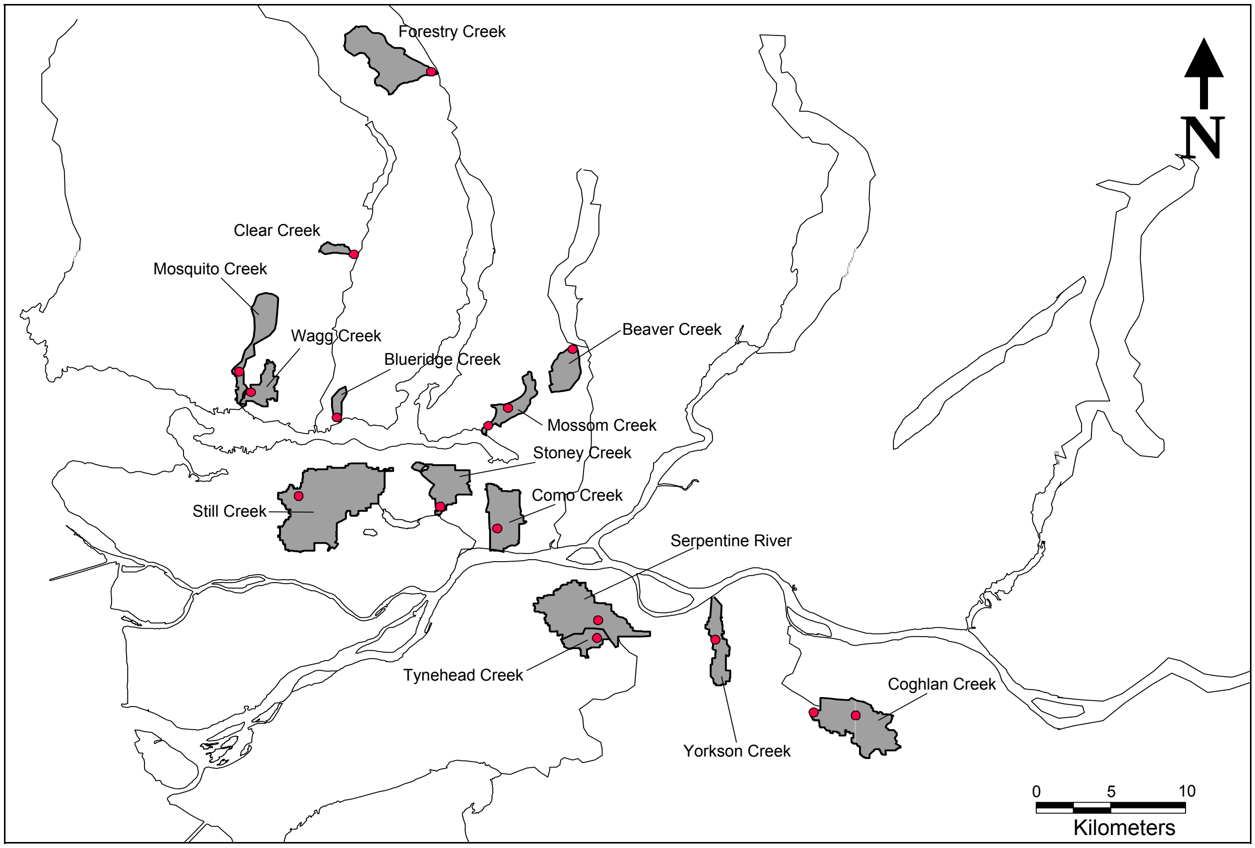

Figure 1. Study area showing location of watersheds and sampling reaches.

Table 1. Summary of physical characteristics of nine study streams in Greater Vancouver.

Stream	Drainage Area	Elevation		Relief Ratio[1]	Stream Channels	
	km^2	Min (m)	Max (m)		Length	km / km^2
Still Creek	28.2	14	94	15	20.1	0.7
Como Creek	8.9	0	166	56	14.2	1.6
Wagg Creek	3.6	0	300	159	2.8	0.8
Serpentine River	19.2	12	73	14	23.3	1.2
Stoney Creek	7.3	14	356	126	6.2	0.8
Yorkson Creek	3.3	n/a	n/a	n/a	n/a	n/a
Mosquito Creek	6.6	95	1120	334	7.0	2.3
Coghlan Creek (lower)	13.8	25	75	13	19.5	2.0
Coghlan Creek (upper)[2]	6.5	30	75	13	n/a	n/a
Mossom Creek (lower)	3.9	0	892	455	12.8	3.3
Mossom Creek (upper)[2]	3.15	n/a	n/a	n/a	n/a	n/a
Beaver Creek	1.1	n/a	n/a	n/a	n/a	n/a
Clear Creek	0.9	230	1350	n/a	n/a	n/a
Blueridge Creek	1.2	n/a	n/a	n/a	n/a	n/a
Tynehead Creek	n/a	n/a	n/a	n/a	n/a	n/a
Forestry Creek	12.4	30	1510	420	6.1	n/a

[1] Relief ratio = relief/drainage$^{0.5}$. This provides a measure of average watershed slope.
[2] "upper" indicates that the sampling site is located in the central portion of the watershed, upstream from the primary sampling site.

Table 2. Stream name, sampling years, site type, and research questions assocated with the study stream.

Stream	Sampling Years	Site Type and Research Questions
Still Creek	1999, 2003, 2004, 2005, 2006	*Primary*: temp. variability; sampling intensity; taxonomic resolution
Como Creek	1999, 2003, 2004	*Primary*: temp. variability; sampling intensity; taxonomic resolution
Wagg Creek	1999, 2003, 2004, 2006[1]	*Primary*: temp. variability; sampling intensity; taxonomic resolution
Serpentine River	1999, 2003, 2004, 2005	*Primary*: temp. variability; sampling intensity; taxonomic resolution
Stoney Creek	1999, 2003, 2004	*Primary*: temp. variability; sampling intensity; taxonomic resolution
Yorkson Creek	1999, 2003, 2004, 2005	*Primary*: temp. variability; sampling intensity; taxonomic resolution
Mosquito Creek	1999, 2003, 2004	*Primary*: temp. variability; sampling intensity; taxonomic resolution
Coghlan Creek	1999, 2003, 2004, 2005	*Primary*: temp. variability; sampling intensity; taxonomic resolution
Mossom Creek	1999, 2003, 2004, 2005, 2006	*Primary*: temp. variability; sampling intensity; taxonomic resolution
Coghlan Ck (up.)	2003, 2004, 2005	*Upstream*: spatial variability
Mossom Ck (up.)	2003, 2004, 2005	*Upstream*: spatial variability
Beaver Creek	2004, 2005, 2006	*Undisturbed*: reference stream
Clear Creek	2004, 2005, 2006	*Undisturbed*: reference stream
Blueridge Creek[1]	2004, 2005	*Intermittent*: temporal variability
Tynehead Creek	2004, 2005, 2006	*Intermittent*: temporal variability
Forestry Creek	2004, 2005, 2006	*Intermittent*: temporal variability

[1] Wagg Creek data was sampled for the City of North Vancouver in 2006.
[2] Blueridge Creek was not sampled in 2006 because it was unusually dry.

1.4 Benthic Index of Biological Integrity

The benthic index of biological integrity (B-IBI) combines selected biological attributes, called metrics, to evaluate the impact of human activities on aquatic ecosystems (Karr, 1998; Karr and Chu, 1999). The B-IBI developed for the Pacific Northwest includes ten metrics selected for their association with urbanization, timber harvest, recreation and pollution (Kerans and Karr, 1994; Fore *et al*., 1996; Karr, 1998). The metrics were developed for the Puget Sound Lowlands which has similar physiography, climate, and biogeography to the Lower Fraser Valley.

Biological metrics measure different aspects of benthic invertebrate community structure including taxonomic richness and composition, tolerance and intolerance, habit, reproductive strategy, feeding ecology and population structure. Total taxa richness measures the variety and complexity of life forms present. Taxa richness of mayflies, stoneflies, and caddisflies responds differently to different types of human disturbance. The presence of intolerant invertebrates is measured in terms of taxa richness because they are often rare and less likely to be collected. In contrast, tolerant invertebrates are measured as % abundance because they are typically common at all sites but dominate the assemblage as human disturbance increases. Long-lived taxa reflect the effects of disturbance through time. Clinger taxa richness includes taxa with morphological adaptations that allow them to cling to substrate in fast water. The presence of predators and their relative abundance relate to the feeding ecology of a sensitive group. Each of the ten metrics tested and developed for the Pacific Northwest B-IBI is described on the following page.

Total taxa richness. The biodiversity of a stream declines as flow regimes are altered, habitat is lost, chemicals are introduced, energy cycles are disrupted, and alien taxa become dominant. Total taxa richness includes all the different invertebrates collected from a stream site: mayflies, caddisflies, stoneflies, true flies, midges, clams, snails, and worms.

Mayfly (Ephemeroptera) taxa richness. The diversity of mayflies declines in response to most types of human influence. Many mayflies graze on algae and are particularly sensitive to chemical pollution (e.g., from mine tailings) that interferes with their food source. Mayflies may disappear when heavy metal concentrations are high while caddisflies and stoneflies are unaffected. In nutrient-poor streams, manure, and fertilizers from agriculture can increase the numbers and types of mayflies present. If many different taxa of mayflies are found while the variety of stoneflies and caddisflies is low, enrichment may be the cause.

Stonefly (Plecoptera) taxa richness. Stoneflies are the first to disappear from a stream as human disturbance increases. Many stoneflies are predators that stalk their prey and hide around and between rocks. Hiding places between rocks are lost as sediment washes into a stream. Many stoneflies are shredders and feed on leaf litter that drops from an overhanging tree canopy. Most stoneflies, like salmonids, require cool water temperatures and high oxygen to complete their life cycles.

Caddisfly (Trichoptera) taxa richness. Different caddisfly species (or taxa) feed in a variety of ways: some spin nets to trap food, others collect or scrape food on top of exposed rocks. Many caddisflies build gravel or wood cases to protect them from predators; others are predators themselves. Even though they are very diverse in habit, taxa richness of caddisflies declines steadily as humans eliminate the variety and complexity of their stream habitat.

Intolerant taxa richness. Animals identified as intolerant are the most sensitive taxa; they represent approximately 5-10% of the taxa present in the region. These animals are the first to disappear as human disturbance increases. Chironomids are not included in this metric.

Clinger taxa richness. Taxa defined as clingers have physical adaptations such as ventral suckers, dorsoventral flattening, well-developed tarsal claws, or construct retreats that they attach to the substrate; thus, they are able to “cling” to smooth substrates in fast water. These animals require open areas between rocks and cobble along the bottom of the stream; consequently, they are particularly sensitive to fine sediments that fill these spaces and eliminate the variety and complexity of small habitats. Clingers may use these areas to forage, escape from predators, or lay their eggs. Sediment also prevents clingers from moving down deeper into the stream bed, or hyporheos, of the channel. Chironomids are included in this metric.

Long-lived (semi-voltine) taxa richness. These invertebrates require more than one year to complete their life cycles; thus, they are exposed to all the human activities that influence the stream throughout one or more years. If the stream is dry part of the year or subject to flooding, these animals may disappear. Loss of long-lived taxa may also indicate an on-going problem that repeatedly interrupts their life cycles.

Percent tolerant. Tolerant animals are present at most stream sites, but as disturbance increases, they represent an increasingly large percentage of the assemblage. Invertebrates designated as tolerant represent the 5-10% most tolerant taxa in a region. In a sense, they occupy the opposite end of the spectrum from intolerant taxa. Chironomids are not included in this metric.

<u>Percent predator.</u> Predator taxa represent the peak of the food web and depend on a reliable source of other invertebrates that they can eat. Predators may have adaptations such as large eyes and long legs for hunting and catching other animals. The percentage of animals that are obligate predators provides a measure of the trophic complexity supported by a site. Less disturbed sites support a greater diversity of prey items and a variety of habitats in which to find them.

<u>Percent dominance (3 taxa)</u>. As diversity declines, a few taxa dominate the assemblage. Opportunistic species that are less particular about where they live replace species that require special foods or particular types of physical habitat. Dominance is calculated by adding the number of individuals in the three most abundant taxa and dividing by the total number of individuals collected in the sample.

2.0 METHODS

Benthic invertebrates were collected from study streams between August 26–September 30, 2003, July 5–September 30, 2004, July 13–September 27, 2005, and July 6 to September 27, 2006 (see Appendix B for spatial coordinates and sampling dates). Most samples were collected in September except for the three intermittent streams (Blueridge, Tynehead, and Forestry creeks) which were sampled in July or early August. Field sampling and laboratory methods followed a recently developed guide to benthic invertebrate sampling in Greater Vancouver (EVS, 2003a). In most sites, four composite samples were collected from four riffles in a 500 m long reach in each stream (Figure 2a). Each composite sample consisted of material collected from three Surber sampler placements. This method is designed to maximize the variation in benthic conditions that are sampled within a stream site while ensuring that the field time is practical.

Because the primary focus of this study was to examine differences in B-IBI values over time, a critical issue was to make field sampling, laboratory methods, and B-IBI calculation as consistent as possible. EVS (2000) provided a brief description of sampling methods which was further refined in EVS (2003a) and their field sampling methods were duplicated as closely as possible. However, several factors introduced inconsistencies in the datasets. For example, detailed field records showing the exact sampling locations for 1999 were unavailable and each sampling reach was restratified in 2003. As well, for the 2003–2006 samples, subsampling was used to reduce sorting and identification time while the whole sample was sorted and identified in 1999. Most importantly, genus-level scoring was used to calculate B-IBI values in 1999 whereas B-IBI values for samples collected in 2003–2006 were calculated using current scoring rules for either family-level or lowest practical taxonomic level data. B-IBI values for samples collected in 1999 were recalculated.

2.1 Field Sampling

In each stream, a 500 m long sampling reach was measured. The sampling reach was generally located in the lower watershed, however, it was sometimes located further upstream to include a single subwatershed or portion of a larger watershed (e.g., Mosquito Creek or upper Mossom Creek). In Still and Mosquito creeks, it was necessary to modify the reach length to ensure one or more riffles were included in each segment or to avoid sections of the stream with long culverts. Riffles were identified during a linear transect of the sampling reach and all non-representative riffles (e.g., unusual substrate conditions, near bridges or immediately downstream of stormwater outfalls, etc.) were excluded. To minimize environmental differences between sites and between streams, riffles with coarse gravel-cobble substrates were preferred over sand-gravel or boulder substrates.

Each sampling reach was divided into four 125 m segments as the second step in the field sampling procedure (Figure 2a). One large riffle or several small adjacent riffles were randomly selected from the population of riffles in each stream segment. A 500 micron Surber sampler with a 30 x 30 cm frame (0.09 m^2) was used to collect benthic invertebrates from three patches of shallow substrates within each riffle; typically one sample near the tail of the riffle, one near the centre, and one near the crest (Figure 2b). Sampling locations were chosen qualitatively within each riffle. Large, loose substrates were transferred to a bucket and the underlying sediment was then disturbed for 2 minutes to a depth of 10 cm using a trowel. Invertebrates, organic debris, and sediment was swept into the

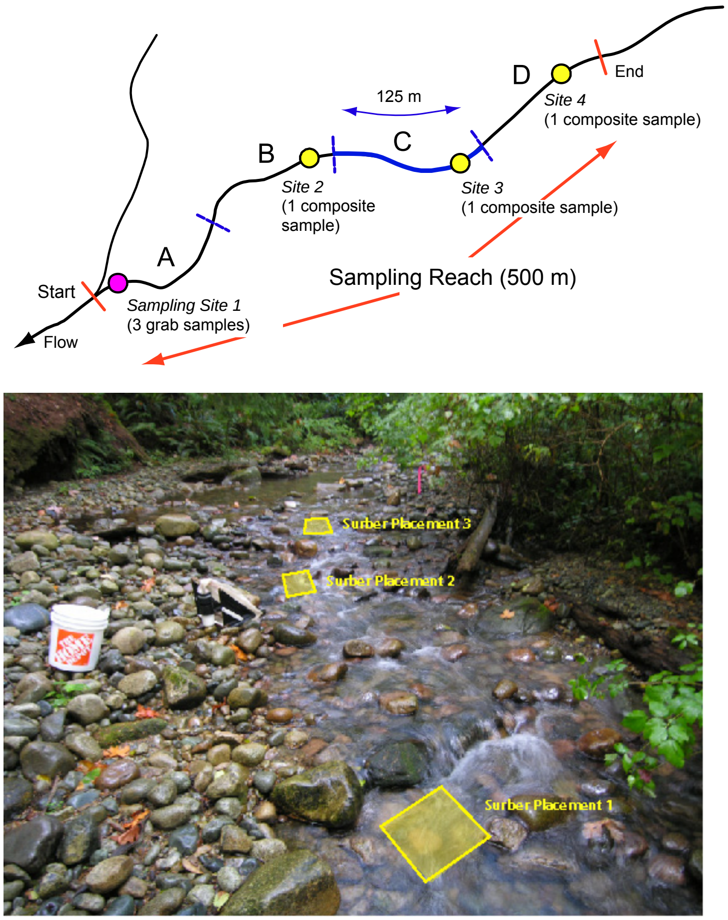

Figure 2. (top) Generalized sampling layout for benthic invertebrate sampling in Greater Vancouver (based on EVS, 2003a). Note that three separate Surber samples were collected at the downstream site (Site 1) for most streams, while a composite sample made up of three Surber samples was collected at the remaining three sites (Sites 2, 3, and 4); (bottom) Example of distribution of individual Surber samples in a single riffle in Mossom Creek.

cod-end of the sampler by water flow. All collected material was carefully washed into the detachable cod-end of the sampler before it was transferred to a secondary 500 micron screen.

Additional invertebrates and debris were brushed from the large substrates using a soft brush and were added to the contents of the screen. The Surber sampler was rinsed several times and visually inspected to ensure invertebrates adhering to the mesh were transferred to the sample container. Large rocks, leaves, or other material were rinsed within the screen to remove invertebrates and then excluded from the sample. The entire contents of the sample were transferred to a 500 or 1000 ml plastic storage jar. For composite samples, all three Surber samples were combined in the sample jar. Individual Surber samples were retained separately from the most downstream sampling site in some streams in 2003 and 2004 (e.g., Site 1 in Figure 2b). Samples were preserved with 10% formalin or occasionally 50% isopropyl alcohol and each sample bottle was labeled with an exterior label and interior label slip.

Environmental Variables – Environmental variables were collected at the watershed, reach, or site scale. Watershed scale variables included watershed area, watershed imperviousness (% total impervious area), riparian forest cover in the watershed upstream of the sampling site (% riparian forest within 30 m of the stream), % watershed forest cover, and % watershed agriculture land use.

Watershed scale data reflects 1996 land cover and was provided by GVRD (2000) or Page *et al.* (1999). Some values were estimated based on recent air photos. The analysis assumed that major land cover change has not occurred in the study watersheds in the past eight years, however, this has not been assessed.

Reach scale variables were collected in the field and included a variety of qualitative habitat measurements such as channelization, riparian vegetation cover, bank stability, and human disturbances (e.g., buildings or agriculture near the stream). However, because these qualitative habitat variables were not the focus of this project, they are not described in detail here and were excluded from the statistical analyses in this project. More information on their collection or relationship to benthic invertebrate data from the same streams in 1999 is presented by EVS (2000, 2003b).

General water quality measurements (water temperature, specific conductivity, pH, and dissolved oxygen) were collected at the upper and lower boundary of each sampling reach using a multiprobe (YSI 600 or Horiba). Sampling was undertaken in late September to minimize the temporal variation in data which precluded collection of data for intermittent streams. Dissolved oxygen was not measured in 2005 because of probe failure. Average values were used for analysis. In 2003 and 2004, total alkalinity was measured by lab analysis for one sample per stream.

Site scale variables were collected at each sampling riffle and include bankfull width, wetted width, average riffle depth, substrate embeddedness, and substrate composition (% bedrock, % boulder, % large gravel, % fine gravel, % fines). Each site was also photographed (upstream, downstream, canopy, and substrate) and spatial coordinates were taken using a handheld GPS (Garmin XL12). As an additional spatial record, the distance of the sampling site upstream from known points (e.g., bridges, culverts, or other major points) was recorded (e.g., 87 m upstream of Marine Drive bridge).

2.2 Laboratory Analysis

Subsampling, Sorting and Taxonomy – Most samples were subsampled to reduce the time and cost associated with sorting and identifying benthic invertebrates without substantially reducing the amount of information collected from the sample. In addition, consistent sample size provided a more precise estimate of taxa richness that could be compared across sites and between years. The target sample size was 400 individuals and only samples with many more individuals than the target number were subsampled.

Subsampling used a Marchant subsampler box in which the entire contents of the sample were distributed among the 100 cells of the box (Marchant, 1989). A 425 micron mesh sieve was used to separate the sample from the preserving fluid and large pieces such as leaves, stones, or other debris were rinsed and removed. The sample was poured into the box and water was added until all cells had been filled to a few millimeters below the top of the cell walls. The lid was securely fastened onto the box and the box was flipped upside down to empty the cells of their contents and the entire contents were agitated. The box was flipped over to the original upright position and the cells were examined to ensure that the material was distributed evenly among the cells. If the material was not distributed evenly among the cells the process was repeated. The lid of the box was removed and the lid and walls were carefully rinsed to wash any organisms clinging to them into the cells. Individual cells were chosen using a random number table and their contents were removed for sorting. The subsampled material was examined under 2X and 10X hand lenses and all macroinvertebrates were removed and placed into 80% ethanol. An event counter was used to keep track of the number of organisms placed into the alcohol. Cells were selected until the number of organisms reached 400; most counts ranged from 400 to 500 because the entire contents of the last cell selected had to be enumerated. The remaining material in the subsampler was returned to the preserving fluid in the sample jar for later quality control. For those samples where some of the material was left unsorted that portion was placed into a Ziploc bag with some preserving fluid and returned to the sample jar.

All invertebrates were identified to lowest practical taxonomic level (LPTL) using a 6.5 to 40X dissecting microscope. This level of taxonomic resolution was defined by the availability of keys for different groups and produced a mixed dataset with some taxa identified to family, while others were identified to genus or species. Chironomids, a particularly difficult family taxonomically, were only identified to family. All taxonomy was undertaken by Dr. Markus Eymann of On Stream Consulting. A list of taxonomic keys used for this project is provided in Appendix J and is comparable to the definitions of lowest practical taxonomic level listed in Plotnikoff and White (1996).

Quality Assurance / Quality Control – Quality control for field sampling focused on ensuring consistent sampling intensity and procedures at each site, maintaining good documentation of samples, and avoiding damage or loss of samples during transport. All samples were collected by a single experienced professional (Nick Page) for this study using the same equipment and field methods. Sample documentation, preservation, and handling was completed according to the recommendations in EVS (2003a). EVS (2003a) also recommended that 10% of samples be replicated to evaluate the precision and repeatability of the sampling results. However, the large number of samples collected for each stream provided adequate duplication with which to identify problems with field sampling consistency (e.g., presence of anomalous samples).

Laboratory quality control methods focused on two critical steps. First, all samples were inspected by a second individual to ensure that they met the criteria for 90% sorting efficiency. This confirmed that organisms were not missed during the sorting or subsampling phase. No resorting was undertaken because all sorting efficiencies were greater than 90%; most were greater than 95% (mean of 1.9% over three years). Second, a reference collection consisting of specimens from each taxon encountered during the study was assembled for each year. The reference collection was reviewed by taxonomists at Rhithron Inc., a consulting firm specializing in benthic invertebrate taxonomy. Verification of the reference collection revealed an error rate of 12% (11 taxa) in 2003, 7% (7 taxa) in 2004, 3% (3 taxa) in 2005, and 3% (3 taxa) in 2006 (see Appendix H). Corrections based on the taxonomic review were made prior to data analysis.

2.3 Data Analysis

Data analysis included four components: 1) linear regression to test the relationship between biological variables and environmental factors; 2) univariate statistical analyses including one-way ANOVA (Analysis of Variance), components of variance analysis, and regression to test temporal, spatial, or sampling differences in B-IBI values; 3) statistical power analysis to estimate the amount of change in B-IBI that would represent a statistically significant change for different sampling scenarios; and 4) nonmetric multidimensional scaling (NMDS), a multivariate ordination technique, to compare benthic communities at the level of individual taxa.

B-IBI Calculation – B-IBI metric and index values were calculated for data collected in 1999, 2003, 2004, 2005, and 2006. Multiple versions of the B-IBI have been described that differ according to the level of taxonomic effort used to identify specimens. B-IBI values from all years were compared based on taxonomic identification to the lowest practical taxonomic level and using the scoring rules developed for that level of taxonomy (Table 3; Fore *et al.*, 2001). B-IBI values for family-level taxonomic effort using family-level scoring rules were also calculated for the 2003, 2004, 2005, and 2006 data only (Table 4; Fore *et al.*, 2001).

While the majority of sampling used composite samples consisting of three Surber placements, for samples in which individual Surber samples were retained separately rather than composited in the field, numerical subsampling was used to create the fourth composite sample. Numerical subsampling was undertaken in MS Excel using the Virtual Marchant Subsampler (Walsh, 2002). Three samples from 2005 that were over-counted (e.g., greater than 600 organisms) were also subsampled numerically.

Metrics were scored as 5, 3, or 1 according to whether they closely approximated condition found at sites with minimal human disturbance (5), diverged somewhat from expected conditions (3), or diverged extremely from natural conditions (1). The LPTL B-IBI uses 10 metrics and the final index ranged from 10–50. In contrast, the family-level index has five metrics and ranged from 5–25. The five metrics of the family-level B-IBI were the same as those in the 10-metric index; the other five metrics could not be calculated at the family-level because they rely on genus level identification.

Table 3. Biological metrics for stream invertebrates, response to human disturbance, and scoring rules used to integrate into the B-IBI.

Metric	Response	Scores*		
		1	3	5
Taxa richness and composition				
Total number of taxa	Decrease	[0, 15)	[15, 28]	> 28
Number of Ephemeroptera taxa	Decrease	[0, 4]	(4, 8]	> 8
Number of Plecoptera taxa	Decrease	[0, 3]	(3, 7]	> 7
Number of Trichoptera taxa	Decrease	[0, 5)	[5, 10)	≥ 10
Number of long-lived taxa	Decrease	[0, 2]	(2, 4]	> 4
Tolerance				
Number of intolerant taxa	Decrease	[0, 2]	(2, 3]	> 3
% of individuals in tolerant taxa	Increase	≥ 50	(19, 50)	[0, 19]
Feeding ecology				
% of predator individuals	Decrease	[0, 10)	[10, 20)	≥ 20
Number of clinger taxa	Decrease	[0, 8]	(8, 18]	> 18
Population attributes				
% dominance (3 taxa)	Increase	≥ 80	[60, 80)	[0, 60)

* Scores for LPTL taxonomy; square braces indicate the value next to the brace is included in the range; rounded parentheses indicate the value is *not* included.

Before calculating metrics, pupae for the Diptera and Trichoptera were eliminated. Some taxa within a sample were not unique, for example, if a specimen was identified at the species level while another specimen was identified to the taxonomic family of the previous specimen. This occurred primarily for early instars that were not developed enough to make a positive identification at the level of species. When a taxon was not unique, individuals were shared proportionally from the higher taxonomic level among the lower taxonomic levels counted in the sample. Wisseman (2002) was used for definitions of long-lived (semi-voltine), intolerant and tolerant taxa, and Merritt and Cummins (1996) was used for definitions of clinger taxa. Appendix G describes life history traits for all taxa collected in the study.

Table 4. Scoring rules for family-level B-IBI.

Metric	Scores*				
	1	2	3	4	5
Total number of taxa	[0, 7]	(7, 13]	(13, 19]	(19, 25]	> 25
Ephemeroptera taxa	[0, 1]	(1, 2]	(2, 4]	(4, 6]	> 6
Plecoptera taxa	0	(0, 1]	(1, 2]	(2, 3]	> 3
Trichoptera taxa	0	(0, 2]	(2, 4]	(4, 6]	> 6
% Dominance (one taxon)	> 85	[70, 85]	[55, 70)	[40, 55)	< 40

* Square braces indicate the value next to the brace is included in the range; rounded parentheses indicate the value is *not* included.

Regression Analysis – Relationships between biological variables (total taxa richness, EPT (Ephemeroptera , Plecoptera, and Trichoptera) taxa richness, 10-metric B-IBI, and 5-metric B-IBI) and water chemistry, watershed land cover, and stream channel variables for permanent streams were tested using least-squares linear regression. Nonquantitative variables were not analyzed. Environmental variables were generally normally distributed and no transformations were undertaken. Regression analysis was undertaken on the nine study streams in 2003 including samples from upper Coghlan and upper Mossom creeks. In 2004 and 2005, this was expanded to include the two undisturbed streams. The three intermittent streams were excluded from regression analysis because watershed land cover data was unavailable. In 2006, only data from four streams was available; regression results should be interpreted more cautiously because of the reduced sample size.

Statistical Tests of B-IBI Variability – We used an ANOVA model to evaluate B-IBI. Because multimetric indexes are approximately normally distributed, parametric tests are appropriate for B-IBI and transformation was unnecessary (Fore *et al.*, 1994). Using the variance estimates derived from ANOVA, total variance associated with B-IBI was partitioned into variance components due to site differences, annual variability, and within site variability (also called measurement error).

Changes in B-IBI values from 1999, 2003 and 2004 were evaluated using correlation and components of variance analysis. Only correlation analysis was used for the 2005 data because the reduced number of sampled streams (5 vs. 9) created an unbalanced sampling design. Correlation tested whether B-IBI ranked sites consistently through time in terms of biological condition. Components of variance were derived from an ANOVA design with site and year as crossed factors. Error (within site variability) was calculated from the repeat samples taken at each stream site and represented the within site variability or differences associated with microhabitats within a stream reach.

Correlation between B-IBI calculated on the basis of LPTL and family-level taxonomy was tested. Variance components analysis was used to compare the percentage of the total variance of B-IBI due to measurement error for both versions of the index. To compare the two field sampling protocols, correlation between the mean B-IBI calculated from three replicate samples and four composite samples (consisting of three replicates) was also tested using correlation for samples from 2003 and 2004.

Power Analysis of B-IBI: Detecting Change at Specific Sites – To estimate the amount of change in B-IBI that can be potentially detected, the minimum detectable difference (MDD) for a two-sample *t* test was calculated (Zar, 1984). This design assumes replicate samples at two sites, e.g., upstream and downstream of a restoration site or a pollution point source. The MDD represents the smallest difference between the mean B-IBI values for two sites that would indicate a statistically significant change in B-IBI. MDD was calculated as follows:

$$\text{MDD} \geq \sqrt{\frac{2s^2}{n}}\left(t_{\alpha(2),\nu} + t_{\beta(1),\nu}\right),$$

Where s^2 = the within-site error variance of B-IBI,
n = the number of replicate samples at each site,
$t_{\alpha(2),\nu}$ = the *t* value for alpha of 0.1 for a 2-sided test,
$t_{\beta(1),\nu}$ = the *t* value for beta of 0.1 for a 1-sided test, and
$\nu = 2n - 2$.

A crossed two-factor ANOVA was used to estimate error variance for this model with site and year as the factors; mean squared error (MSE) was used as the estimate of within site variance.

Power Analysis of B-IBI: Regional Trend Monitoring – To calculate the amount of change that B-IBI can detect through time (trend), a simple linear regression model was used, with B-IBI regressed against year. For this approach, a slope significantly different from zero would indicate a change in resource condition: an increase over time would indicate an improvement in biological condition, while a decline would indicate degradation. A linear statistical model assumes that the response variable is normally distributed, a reasonable assumption for multimetric indexes (Fore et al., 1994). For B-IBI, these relationships can be expressed in terms of an equation where:

$$\text{B-IBI}_{sy} = \mu + S_s + Y_y + E_{sy}$$

and B-IBI for a particular site-visit differs from the mean of all site-visits (μ based on the site location, S (indexed from 1 to the total number of sites), and year, Y (with *y* indexed by year). E_{sy} represents the error variance.

This model can be used to test for significant change in the slope of the line for a set of sites sampled over years. The significance of the trend, or slope of the line, depends on its associated variance. The closer the B-IBI values for each site fall in relation to the regression line, the lower the slope's variance. If B-IBI is extremely variable, only very large changes in the slope will be statistically significant. The smaller the variance associated with the slope of the trend line, the more likely significant changes in B-IBI can be detected. To estimate the variance of the slope, the following equation was used (Larsen et al., 1995; Urquhart et al., 1998):

$$\mathrm{var}(\hat{\beta}) = \frac{s_{year}^2 + \dfrac{s_{site*year}^2 + \dfrac{s_{error}^2}{r}}{s}}{\Sigma\left(Y_y - \bar{Y}\right)^2}$$

where β represents the slope, s^2 refers to an estimate of variance, s equals the number of sites, and r equals the number of repeat visits to each site. In the denominator, Y represents the year value indexed according to the number of years sampled.

Slope estimators are assumed to be normally distributed. The following equation uses the estimate of the slope's variance from above to estimate the statistical power of B-IBI to detect changes through time:

$$Power = \Phi\left[z_\alpha - \frac{\beta}{s.e.(\hat{\beta})} \right].$$

Power was set equal to 0.8, Φ is the cumulative normal distribution function, α was equal to 0.10 for a 1-sided test, and the standard error of the slope (*s.e.* [β]) was derived from the equation above. This equation was solved for β, the slope, to determine the minimum amount of change the B-IBI could detect for various sampling designs that varied according to the number of sites sampled, the number of years sampled, and the number of replicate samples collected during each visit.

Multivariate Ordination – Nonmetric multidimensional scaling (NMDS) was used to examine relationships among sites and streams based on the composition and abundance of benthic invertebrates. NMDS iteratively searches for a mathematical solution with the least difference (or stress) between the original dataset and that generated for the same plots or site in ordination space. It is considered the ordination method of choice for ecological data (McCune and Grace, 2002). NMDS is similar to principal components analysis (PCA) but is often preferred because biological data rarely meets the underlying assumption of PCA such as linear responses to environmental gradients.

All composite samples (44 from 2003, 64 from 2004, 48 from 2005, and 24 from 2006) were analyzed together to examine the similarity of samples within each stream in each year, as well as the variation of the benthic community between streams. NMDS analysis was undertaken in PC-ORD (a Windows program for multivariate analysis of ecological data) using the Sorenson distance measure, a random starting configuration, and 25 runs with real data, a procedure recommended by McCune and Grace (2002). The stress of the solutions was compared to those generated by random configurations of the data using 25 Monte Carlo simulations to test whether the solution differed significantly from a random pattern. Based on this initial test, a three-dimensional solution was selected and a final analysis was rerun for only this dimensionality with the same distance measure, the same starting configuration, and 100 iterations to assess the stability of the solution.

3.0 RESULTS

A total of 82,956 organisms representing 154 unique invertebrate taxa were identified from the study streams between 2003 and 2006. Taxa richness varied from 5 to 39 per sample with an overall mean of 21.8 (SD 8.1) taxa per composite sample. Taxa richness was low in highly urbanized streams and high in streams with low imperviousness and high watershed forest cover. Samples from Mossom. Beaver, and Clear creeks consistently had more than 30 taxa, while samples from Tynehead, Still, Como, and Blueridge creeks generally had below 12 taxa.

The estimated abundance of organisms per composite sample was 1,591 (SD 2,021) organisms based on the analysis of 180 composite samples. This value was extrapolated based on the abundance of taxa identified during subsampling and the subsampling cell counts (e.g., 7 organisms from 31 of 100 cells = 23 organisms in full sample). Organism abundance varied by year. In 2003, a mean estimated abundance was 2,233 (SD 2,332), while in 2004 this decreased to 1,172 (SD 1200). In 2005, the mean estimated abundance was 1,998 (SD 2,694) and 716 (SD 662) in 2006. Forty of the 180 composite samples contained less than the target number of 400 organisms which is the recommended consistent B-IBI calculation. Twenty-two of these low abundance samples were from intermittent streams.

3.1 Watershed Land Cover, Water Quality, and Stream Channel Characteristics

Land cover characteristics of the study watersheds are presented in Table 5. Mean water chemistry variables (permanent streams only) and mean stream channel characteristics (all years combined) are summarized in Tables 6 and 7 (Appendices D, E, and F present the data by sampling year).

Table 5. Summary of watershed-scale land cover characteristics of study streams with permanent flow (data from EVS (2000) for % total imperviousness and % riparian forest integrity, and from Page *et al.* (1999) for % forest cover and % agricultural land use). Data is unavailable for intermittent streams.

Stream	% Total Imp. Area	% Riparian Forest Integrity	% Watershed Forest Cover	% Agricultural Land Use
Still Creek	68.8	40.4	12.7	0.0
Como Creek	54.3	37.9	12.0	0.0
Wagg Creek	44.3	71.8	13.8	0.0
Serpentine River	39.3	73.9	35.3	5.4
Stoney Creek	32.8	70.9	38.5	0.0
Yorkson Creek	18.5	70.0	23.0*	5.0*
Mosquito Creek	9.0	83.1	55.0*	0.0
Coghlan Creek (lower)	5.2	74.2	38.3	52.2
Coghlan Creek (upper)	4.0	79.0	30.0*	60.0*
Mossom Creek (upper)	2.0	95.6	90.0*	0.0*
Mossom Creek (lower)	3.8	93.2	80.5	0.0
Clear Creek	0.2*	98*	93*	0
Beaver Creek	0.2*	98*	93*	0

* estimated value based on a review of recent air photos.

Table 6. Summary of water chemistry variables in Greater Vancouver study streams with permanent flow. Values are mean values from 2003–2006 (see Appendix D for raw values per stream).

Stream	Total Alkalinity[2] (mg/L)	Water Temp. (°C)	Diss. Ox. (mg/L)	Sp. Cond. (uS/cm)	pH[3] rel. units
Still Creek	67	15.5	10.2	251	7.6
Como Creek	67	14.0	9.4	183	7.4
Wagg Creek	54	13.9	9.9	167	7.1
Serpentine River	60	13.4	10.7	130	7.7
Stoney Creek	59	13.5	9.8	212	7.5
Yorkson Creek	72	13.1	7.4	128	7.4
Mosquito Creek	18	12.4	10.9	63	7.0
Coghlan Creek (lower)	47	12.1	10.2	186	7.3
Coghlan Creek (upper)	58	12.5	10.0	161	7.5
Mossom Creek (upper)	10	12.7	10.5	37	7.6
Mossom Creek (lower)	36	9.6	11.1	86	7.4
Clear Creek	3	7.9	12.6	15	6.8
Beaver Creek	5	11.6	10.6	26	7.0

[1] Oxidation reduction potential (ORP) was measured in 2003 and 2004 but was not measured in 2005 and 2006 and was excluded from analyses; [2] total alkalinity values from 2004 and 2004; [3] mean pH values were not antilog-transformed and are considered an estimate of average pH.

Table 7. Summary of stream channel variables in Greater Vancouver study streams. Values are mean values from 2003–2006 (see Appendices E and F for raw values per stream).

Stream	Bankfull Width (m)	Wetted Width (m)	Riffle Depth (cm)	% Emb.	% Boul.	% Cobb.	% Lg. Gravel	% Sm. Gravel	% Fines
Still Creek	6.0	4.5	12	44	5	35	34	14	8
Como Creek	6.2	3.6	7	26	1	10	37	36	19
Wagg Creek	8.4	2.9	8	26	34	41	15	6	5
Serpentine River	13.2	6.1	7	16	1	16	49	26	9
Stoney Creek	7.0	4.0	10	28	19	40	27	8	6
Yorkson Creek	6.0	2.0	5	11	0	2	30	50	18
Mosquito Creek	11.2	3.5	12	40	38	40	16	5	2
Coghlan Ck (lower)	8.0	4.7	7	12	0	1	35	46	18
Coghlan Ck (upper)	7.4	2.2	5	14	0	2	42	43	13
Mossom Ck (upper)	6.2	0.7	4	18	35	38	16	7	4
Mossom Ck (lower)	8.1	4.4	13	24	30	33	22	11	6
Clear Creek	8.5	4.1	11	5	29	54	9	4	3
Beaver Creek	8.8	3.2	14	6	39	44	11	4	3
Forestry Creek	19.1	5.9	14	12	31	40	17	8	3
Blueridge Creek	6.8	1.2	5	19	16	27	33	18	11
Tynehead Creek	6.3	1.3	5	16	2	9	36	38	16

3.2 Regression Analyses

The correlation between biological variables including total taxa richness, EPT taxa richness, and B-IBI (LPTL- and family-level) from permanent streams and environmental variables differed by scale. Tables 8–13 provide r^2 and significance values for all variables in 2003, 2004, 2005, and 200, and Figures 3 and 4 show linear regression relationships for selected land cover, water chemistry, and stream channel variables. Results from 2006 were only included in the tables because of the limited number of permanent streams sampled (4 streams) reduced the comparability to the graphical results from other years.

Most biological variables were correlated with watershed-scale land cover variables except for % agricultural land use. Percent total imperviousness had the strongest correlation with biological variables in 2003 and 2004, while percent riparian forest had a stronger correlation in 2005. In 2006, only the number of EPT taxa were significantly correlated with % total imperviousness, % riparian forest, and % watershed forest cover. The linear relationship between % total imperviousness and biological variables from 2003–2005 ranged between 76% and 91% of the total variation in the model, except for family-level B-IBI in 2005 (46%). The explanation of variation from the 2006 was generally lower.

Most biological variables were correlated with water quality variables including total alkalinity, water temperature, and specific conductivity that are influenced by urban land cover. B-IBI and taxa richness variables decline as specific conductivity, a broad indicator of urbanization on water chemistry, and water temperature increase. These relationships were generally weaker in 2003 and 2005 than in 2004. None the relationships were significant in 2006 which more reflects the limited number of streams samples than the overall pattern of benthic community change with increasingly urbanized water quality. The relationship between total alkalinity and biological variables was not statistically significant in 2003 but strongly correlated in 2004. Neither dissolved oxygen nor pH was correlated with biological variables in 2003, 2004, 2005, or 2006.

Stream channel characteristics were poorly correlated with biological variables in 2003 and 2004, however, % substrate embeddedness, % boulder, and % cobble were correlated with B-IBI (LPTL), total taxa richness, and EPT taxa richness in 2005. The relationship between % boulder substrate and B-IBI (LPTL) was marginally significant in 2004. In 2006, EPT taxa richness, B-IBI (LPTL), and B-IBI (family) were correlated with riffle depth; streams with deeper riffles had more EPT taxa. Streams with coarser substrates generally have higher more diverse benthic invertebrate communities than streams with sand and gravel dominated substrates. In 2003 and 2004, the average explanation of variance by stream channel characteristics was 6.5%. It increased to 27.0% in 2005 and 28.6% in 2006.

Table 8. Mean LPTL B-IBI values and standard deviation by site for 1999, 2003–06. All B-IBI values shown are the average of four composite samples. Metric data is in Appendix C.

Stream site	1999 LPTL B-IBI	SD	2003 LPTL B-IBI	SD	2004 LPTL B-IBI	SD	2005 LPTL B-IBI	SD	2006 LPTL B-IBI	SD
Still Creek	11.5	1.0	13.0	2.6	10.5	1.0	12.5	1.0		
Como Creek	10.5	1.0	12.5	1.0	13.0	1.2				
Wagg Creek	11.5	1.0	15.0	2.6	15.0	2.6			19.0	1.2
Stoney Creek	12.0	0.0	17.0	2.6	13.5	3.0				
Serpentine River	12.5	3.0	20.5	1.9	16.5	1.0	18.0	2.8		
Yorkson Creek	18.0	1.6	24.0	3.3	24.5	3.8	21.5	1.0		
Coghlan Creek (lower)	33.0	3.5	27.0	3.8	29.5	1.9	27.5	5.0		
Mosquito Creek	22.0	1.6	29.0	2.6	29.0	1.2				
Mossom Creek (lower)	34.5	3.4	38.0	2.3	37.0	2.6	34.0	5.9	38.0	3.7
Mossom Creek (upper)			30.0	8.8	33.5	3.8	35.5	6.0		
Coghlan Creek (upper)			28.5	2.5	28.0	2.8	26.5	1.9		
Beaver Creek					39.0	5.0	32.0	3.3	39.5	3.8
Clear Creek					37.5	4.4	34.0	2.8	32.5	3.0
Tynehead Creek					13.5	1.0	12.0	0.0	13.5	1.0
Forestry Creek					23.5	3.4	27.5	7.5	27.5	3.0
Blueridge Creek					16.0	4.0	12.5	1.0		

Table 9. Mean family-level B-IBI values and standard deviation by site for 2003–06. All B-IBI values shown represent the average of four composite samples. Also shown are the average number of individuals in each of four samples. Metric data is in Appendix C.

Stream site	2003 Family B-IBI	SD	2003 Num Ind.	2004 Family B-IBI	SD	2004 Num Ind.	2005 Family B-IBI	SD	2005 Num Ind.	2006 Family B-IBI	SD	2006 Num Ind.
Still Creek	10.0	1.6	428	8.5	1.3	455	11.5	1.0	432			
Como Creek	11.5	1.9	427	10.3	1.3	462						
Wagg Creek	13.5	1.9	432	11.5	1.3	378				15.3	0.5	415
Stoney Creek	12.8	2.2	352	13.5	2.5	379						
Serpentine River	16.8	2.2	480	16.0	1.4	454	17.5	1.0	436			
Yorkson Creek	18.5	0.6	459	19.8	0.5	385	18.0	1.4	511			
Coghlan Ck (lower)	17.8	2.2	492	20.8	1.3	457	19.8	1.3	519			
Mosquito Creek	20.5	0.6	398	20.3	1.0	469						
Mossom Ck (lower)	21.0	0.8	411	21.3	1.3	536	22.0	0.8	428	21.3	0.5	425
Mossom Ck (upper)	18.0	5.6	158	19.0	1.2	367	22.0	1.6	378			
Coghlan Ck (upper)	18.8	1.0	462	17.8	0.5	452	18.3	1.0	456			
Beaver Creek				22.3	0.5	489	21.8	0.5	431	22.3	0.5	376
Clear Creek				19.8	1.0	488	20.5	0.6	468	20.0	0.0	380
Tynehead Creek				8.5	0.6	453	10.5	1.3	434	7.8	1.0	346
Forestry Creek				16.0	1.2	193	17.5	3.0	302	17.5	0.6	172
Blueridge Creek				11.0	2.7	346	10.8	1.0	386			

Table 10. Regression relationships between biological and watershed land cover variables in 2003 (N= 11), 2004 (N=13), 2005 (N=9), and 2006 (N=4). Significant relationships (p<0.05) are shown in bold.

Biological Variable	Land Cover Variable	2003		2004		2005		2006	
		r^2	Significance of Model	r^2	Significance of Model	r^2	Significance of Model	r^2	Significance of Model
Total Taxa Richness	% Total Imperviousness	0.76	0.0003	0.91	**<0.0001**	0.80	**0.0011**	0.55	0.2578
	% Riparian Forest	0.65	0.0028	0.77	**<0.0001**	0.91	**<0.0001**	0.52	0.2756
	% Watershed Forest	0.51	0.0141	0.64	**0.0010**	0.74	**0.0030**	0.55	0.2587
	% Agriculture	0.10	0.3473	0.03	0.5701	0.01	0.7654	n/a	n/a
EPT Taxa Richness	% Total Imperviousness	0.78	0.0003	0.88	**<0.0001**	0.76	**0.0023**	0.93	**0.0332**
	% Riparian Forest	0.53	0.0114	0.76	**0.0001**	0.90	**<0.0001**	0.91	**0.0461**
	% Watershed Forest	0.24	0.1240	0.73	**0.0002**	0.86	**0.0003**	0.92	**0.0386**
	% Agriculture	0.43	0.0293	0.03	0.6003	0.04	0.5960	n/a	n/a
B-IBI (LPTL)	% Total Imperviousness	0.83	0.0001	0.87	**<0.0001**	0.81	**0.0010**	0.88	0.0615
	% Riparian Forest	0.69	0.0016	0.76	**<0.0001**	0.86	**0.0003**	0.84	0.0842
	% Watershed Forest	0.71	0.0012	0.75	**0.0001**	0.80	**0.0012**	0.86	0.0741
	% Agriculture	0.08	0.4102	0.02	0.6454	0.00	0.9425	n/a	n/a
B-IBI (Family)	% Total Imperviousness	0.85	<0.0001	0.89	**<0.0001**	0.46	**0.0437**	0.72	0.1496
	% Riparian Forest	0.77	0.0003	0.73	**0.0002**	0.34	**0.0975**	0.87	0.0672
	% Watershed Forest	0.48	0.0175	0.51	**0.0059**	0.14	0.3294	0.89	0.0581
	% Agriculture	0.08	0.3927	0.06	0.4257	0.02	0.7536	n/a	n/a

Table 11. Regression relationships between biological and water quality characteristics for streams in Greater Vancouver in 2003, 2004, and 2005. Significant relationships (p<0.05) are shown in bold.

Biological Variable	Water Quality Variable	2003		2004		2005		2006	
		r^2	Significance of Model	r^2	Significance of Model	r^2	Significance of Model	r^2	Significance of Model
Total Taxa Richness	Total Alkalinity	0.08	0.4006	0.71	**0.0012**	n/a	n/a	n/a	n/a
	Water Temperature	0.52	**0.0127**	0.64	**0.0010**	0.53	**0.0267**	0.10	0.6843
	Dissolved Oxygen	0.14	0.2553	0.29	0.0578	n/a	n/a	0.02	0.8595
	Specific Conductivity	0.32	0.0714	0.58	**0.0024**	0.78	**0.0015**	0.21	0.5467
	pH	0.09	0.3717	0.15	0.1946	0.04	0.5886	0.02	0.8588
EPT Taxa Richness	Total Alkalinity	0.03	0.6394	0.78	**0.0003**	n/a	n/a	n/a	n/a
	Water Temperature	0.63	**0.0037**	0.78	**<0.0001**	0.62	**0.0122**	0.38	0.3856
	Dissolved Oxygen	0.33	0.0643	0.25	0.0822	n/a	n/a	0.11	0.6719
	Specific Conductivity	0.15	0.2470	0.62	**0.0013**	0.86	**0.0003**	0.49	0.2981
	pH	0.05	0.5044	0.23	0.0988	0.12	0.3650	0.10	0.6867
B-IBI (LPTL)	Total Alkalinity	0.28	0.0909	0.82	**0.0001**	n/a	n/a	n/a	n/a
	Water Temperature	0.38	**0.0328**	0.78	**<0.0001**	0.62	**0.0119**	0.39	0.3730
	Dissolved Oxygen	0.07	0.3963	0.22	0.1025	n/a	n/a	0.07	0.7338
	Specific Conductivity	0.36	**0.0393**	0.69	**0.0004**	0.77	**0.0018**	0.38	0.3833
	pH	0.10	0.3262	0.20	0.1240	0.11	0.3796	0.05	0.7794
B-IBI (family)	Total Alkalinity	0.14	0.2568	0.72	**0.0010**	n/a	n/a	n/a	n/a
	Water Temperature	0.39	**0.0385**	0.59	**0.0022**	0.17	0.2730	0.35	0.4118
	Dissolved Oxygen	0.21	0.1604	0.28	0.0633	n/a	n/a	0.07	0.7321
	Specific Conductivity	0.58	**0.0066**	0.51	**0.0061**	0.36	0.0871	0.45	0.3311
	pH	0.05	0.5000	0.15	0.1848	0.05	0.5505	0.06	0.7498

Table 12. Regression relationships between taxa richness and mean stream channel characteristics for study streams in 2003, 2004, and 2005. Significant relationships (*p*<0.05) are shown in bold.

Biological Variable	Stream Channel Variable	2003		2004		2005		2006	
		r^2	Significance of Model	r^2	Significance of Model	r^2	Significance of Model	r^2	Significance of Model
Total Taxa Richness	Bankfull Width	0.08	0.3867	0.00	0.8360	0.00	0.9750	0.37	0.3928
	Wetted Width	0.00	0.9416	0.02	0.6169	0.32	0.1126	0.38	0.3824
	Riffle Depth	0.02	0.6701	0.04	0.5029	0.00	0.9762	0.66	0.1857
	% Embeddedness	0.03	0.6349	0.03	0.0507	0.48	**0.0385**	0.40	0.3697
	% Boulder	0.03	0.6064	0.22	0.1019	0.57	**0.0193**	0.00	0.9879
	% Cobble	0.04	0.5624	0.00	0.9137	0.26	0.1601	0.05	0.7808
	% Large Gravel	0.02	0.6487	0.10	0.2867	0.48	0.0392	0.06	0.7590
	% Small Gravel	0.02	0.6561	0.01	0.6915	0.15	0.3044	0.07	0.7397
	% Fines	0.00	0.9977	0.01	0.7501	0.18	0.2547	0.01	0.9242
EPT Taxa Richness	Bankfull Width	0.06	0.4735	0.00	0.9020	0.01	0.8314	0.04	0.8022
	Wetted Width	0.00	0.9442	0.00	0.8961	0.32	0.1098	0.70	0.1624
	Riffle Depth	0.02	0.6629	0.09	0.3122	0.00	0.9040	0.97	**0.0172**
	% Embeddedness	0.21	0.1597	0.30	0.0512	0.45	**0.0465**	0.76	0.1308
	% Boulder	0.00	0.8684	0.26	0.0746	0.68	**0.0057**	0.19	0.5589
	% Cobble	0.19	0.1832	0.00	0.9178	0.43	0.0535	0.01	0.8849
	% Large Gravel	0.00	0.8534	0.14	0.2066	0.67	0.0068	0.01	0.9173
	% Small Gravel	0.18	0.1966	0.03	0.5506	0.27	0.1484	0.06	0.7585
	% Fines	0.09	0.3777	0.03	0.5637	0.26	0.1605	0.10	0.6835

Table 13. Regression relationships between B-IBI and mean stream channel characteristics for study streams in 2003, 2004, and 2005. Significant relationships (p<0.05) are shown in bold.

Biological Variable	Stream Channel Variable	2003		2004		2005		2006	
		r^2	Significance of Model	r^2	Significance of Model	r^2	Significance of Model	r^2	Significance of Model
B-IBI (LPTL)	Bankfull Width	0.06	0.4809	0.00	0.8842	0.03	0.6742	0.06	0.7593
	Wetted Width	0.01	0.7318	0.04	0.5051	0.33	0.1049	0.73	0.1467
	Riffle Depth	0.02	0.6513	0.07	0.3658	0.00	0.9411	0.96	**0.0212**
	% Embeddedness	0.01	0.7546	0.30	0.0505	0.45	**0.0488**	0.65	0.1931
	% Boulder	0.12	0.3028	0.31	**0.0494**	0.62	**0.0121**	0.17	0.5916
	% Cobble	0.00	0.9153	0.00	0.8609	0.39	0.0722	0.00	0.9991
	% Large Gravel	0.10	0.3549	0.20	0.1276	0.58	**0.0166**	0.04	0.8097
	% Small Gravel	0.00	0.9729	0.04	0.5268	0.22	0.2032	0.11	0.6682
	% Fines	0.01	0.8081	0.04	0.5380	0.34	0.0998	0.04	0.7906
B-IBI (family)	Bankfull Width	0.16	0.2226	0.01	0.7203	0.01	0.8172	0.06	0.7453
	Wetted Width	0.01	0.7624	0.04	0.5356	0.45	**0.0466**	0.68	0.1762
	Riffle Depth	0.01	0.8024	0.02	0.6809	0.10	0.4086	0.95	**0.0259**
	% Embeddedness	0.10	0.3508	0.30	0.0508	0.32	0.1134	0.71	0.1574
	% Boulder	0.06	0.4636	0.14	0.2004	0.09	0.4284	0.15	0.6090
	% Cobble	0.04	0.5546	0.01	0.7047	0.01	0.7882	0.00	0.9392
	% Large Gravel	0.07	0.4248	0.04	0.5022	0.16	0.2799	0.01	0.8824
	% Small Gravel	0.03	0.5828	0.00	0.9023	0.00	0.9789	0.07	0.7406
	% Fines	0.00	0.9385	0.00	0.9219	0.10	0.4021	0.07	0.7421

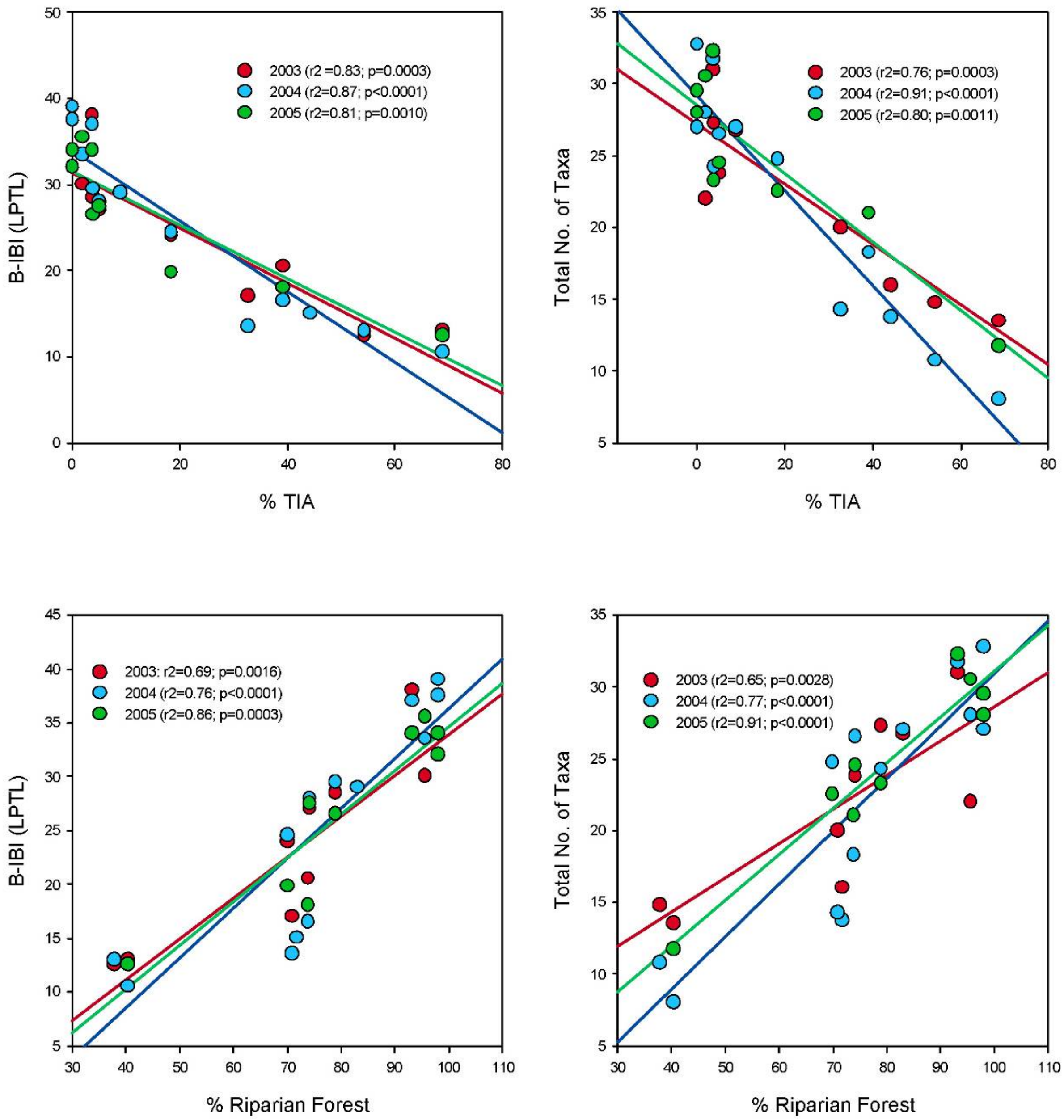

Figure 3. Linear regression relationships between B-IBI (LPTL), total taxa richness, and % total impervious area and % riparian forest. Too few streams were sampled in 2006 to be comparable with the graphical results from other years.

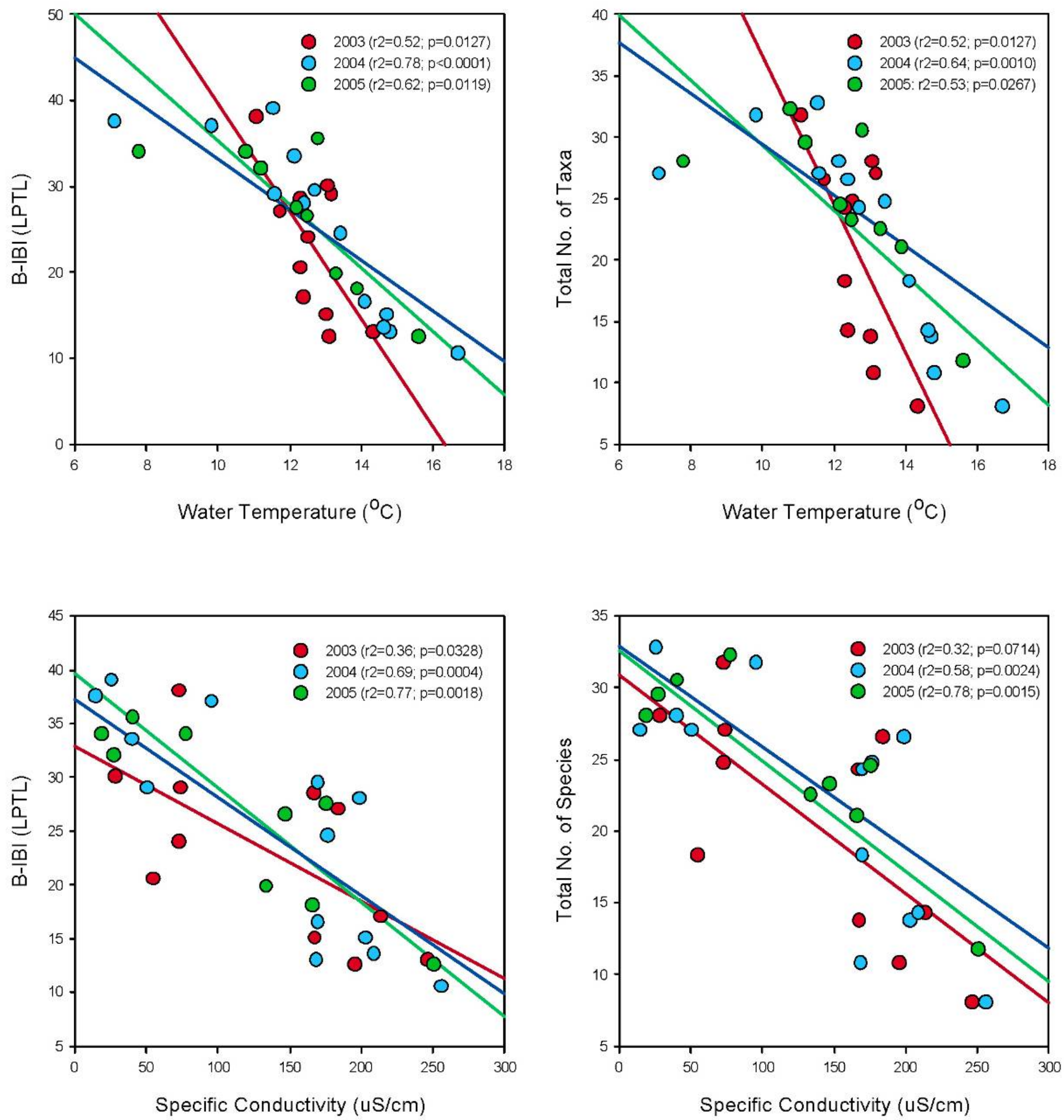

Figure 4. Linear regression relationships between B-IBI (LPTL), total taxa richness, and water temperature and specific conductivity. Too few streams were sampled in 2006 to be analyzed.

3.3 B-IBI Variability

Temporal Variability of B-IBI – B-IBI values for 1999, 2003, 2004 and 2005 were all highly correlated indicating strong agreement between the different sampling years (Table 14). For all years, the correlation coefficient was >= 0.90 for LPTL identification (Pearson's r, $p < 0.05$). For family-level identification, the B-IBI values across years had slightly lower correlation coefficients, $r > 0.75$. All coefficients were significant except for the correlation between 1999 LPTL and 2005 family for which only five stream sites were available for comparison.

B-IBI values for different years were in close agreement and consistently ranked stream sites in a similar order (Figure 5, Table 15, and Appendix C for more detailed information). For 1999 vs. 2003, the mean difference in B-IBI was 3.4 points lower (SD of 4.1) in 1999. Part of this difference may have been due to differences associated with laboratory protocols. For 2003 vs. 2004 the agreement was closer with an average difference of 0.4 points (SD of 2.3) lower. Similarly for 2004 vs. 2005, the average difference was small: 0.6 points (SD of 2.3) lower.

The largest differences for 1999 vs. 2003 were observed for Serpentine River with B-IBI eight points higher in 2003 (12.5 vs. 20.5) and Mosquito Creek with B-IBI seven points higher in 2003 (22.0 vs. 29.0). The largest differences for 2003 vs. 2004 were observed for Serpentine River which declined four points (20.5 vs. 16.5), Stoney Creek which also declined 3.5 points (17 vs. 13.5), and upper Mossom Creek which increased 3.5 points (30 vs. 33.5). These changes were small, and none of the observed differences between 2003 and 2004 represented statistically significant changes in B-IBI (see section below *Minimum Detectable Difference*). For 2004 vs. 2005, the largest difference in B-IBI observed for any stream site was three points.

Table 14. Correlation among B-IBI values for different years were consistently high (Pearson's r, $p < 0.05$; N = 9 for 1999, N = 11 for 2003, N = 16 for 2004, N = 12 for 2005, N = 6, for 2006). Correlation was also high for the two different levels of taxonomic identification (lowest possible taxonomic level vs. family-level).

Year	B-IBI version	1999	2003		2004		2005		2006	
		LPTL Comp	LPTL Comp	Family Comp	LPTL Comp	Family Comp	LPTL Comp	Family Comp	LPTL Comp	Family Comp
1999	LPTL Composite		0.90	0.76	0.94	0.83	0.95	0.82 (ns)		
2003	LPTL Composite	0.90		0.93	0.97	0.91	0.93	0.92		
	Family Composite	0.76	0.93		0.91	0.96	0.78	0.90		
2004	LPTL Composite	0.94	0.97	0.91		0.91	0.95	0.91	0.96	0.89
	Family Composite	0.83	0.91	0.96	0.91		0.87	0.95	1.00	0.96
2005	LPTL Composite	0.95	0.93	0.78	0.95	0.87		0.95	0.94	0.98
	Family Composite	0.82 (ns)	0.92	0.90	0.91	0.95	0.95		0.99	0.99
2006	LPTL Composite				0.96	1.00	0.94	0.99		0.95
	Family Composite				0.89	0.96	0.98	0.99	0.95	

Table 15. Mean B-IBI values and standard deviation by site for 1999, 2003, 2004 and 2005 sampling with B-IBI calculated based on LPTL and family-level taxonomy. All B-IBI values shown represent the average of four composite samples. The average number of individuals identified per replicate sample is shown for 2003–05.

	1999 LPTL		2003 LPTL		2003 Family		2003	2004 LPTL		2004 Family		2004	2005 LPTL		2005 Family		2005
Stream site	**B-IBI**	**SD**	**B-IBI**	**SD**	**B-IBI**	**SD**	**Num Ind.**	**B-IBI**	**SD**	**B-IBI**	**SD**	**Num Ind.**	**B-IBI**	**SD**	**B-IBI**	**SD**	**Num Ind.**
Still	11.5	1.0	13.0	2.6	10.0	1.6	428	10.5	1.0	8.5	1.3	455	12.5	1.0	11.5	1.0	432
Como	10.5	1.0	12.5	1.0	11.5	1.9	427	13.0	1.2	10.3	1.3	462					
Wagg	11.5	1.0	15.0	2.6	13.5	1.9	432	15.0	2.6	11.5	1.3	378					
Stoney	12.0	0.0	17.0	2.6	12.8	2.2	352	13.5	3.0	13.5	2.5	379					
Serpentine	12.5	3.0	20.5	1.9	16.8	2.2	480	16.5	1.0	16.0	1.4	454	18.0	2.8	17.5	1.0	436
Upper Mossom			30.0	8.8	18.0	5.6	158	33.5	3.8	19.0	1.2	367	35.5	6.0	22.0	1.6	377
Yorkson	18.0	1.6	24.0	3.3	18.5	0.6	459	24.5	3.8	19.8	0.5	385	21.5	1.0	18.0	1.4	511
Coghlan	33.0	3.5	27.0	3.8	17.8	2.2	492	29.5	1.9	20.8	1.3	457	27.5	5.0	19.8	1.3	518
Upper Coghlan			28.5	2.5	18.8	1.0	462	28.0	2.8	17.8	0.5	452	26.5	1.9	18.3	1.0	455
Mosquito	22.0	1.6	29.0	2.6	20.5	0.6	398	29.0	1.2	20.3	1.0	469					
Mossom	34.5	3.4	38.0	2.3	21.0	0.8	411	37.0	2.6	21.3	1.3	536	34.0	5.9	22.0	0.8	428
Beaver								39.0	5.0	22.3	0.5	489	32.0	3.3	21.8	0.5	431
Clear								37.5	4.4	19.8	1.0	488	34.0	2.8	20.5	0.6	467
Tynehead								13.5	1.0	8.5	0.6	453	12.0	0.0	10.5	1.3	434
Forestry								23.5	3.4	16.0	1.2	193	27.5	7.5	17.5	3.0	301
Blueridge								16.0	4.0	11.0	2.7	346	12.5	1.0	10.8	1.0	385

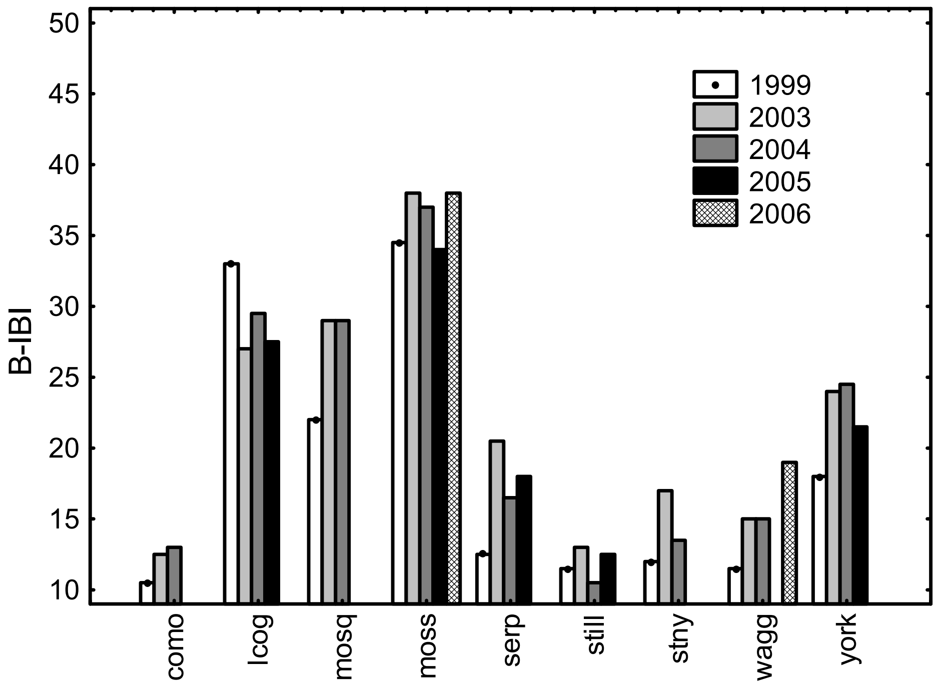

Figure 5. Average B-IBI values for nine permanent streams sampled in 1999, 2003, 2004, 2005, and 2006. B-IBI calculated for LPTL. The legend shows the order to read the bars from left to right for each site.

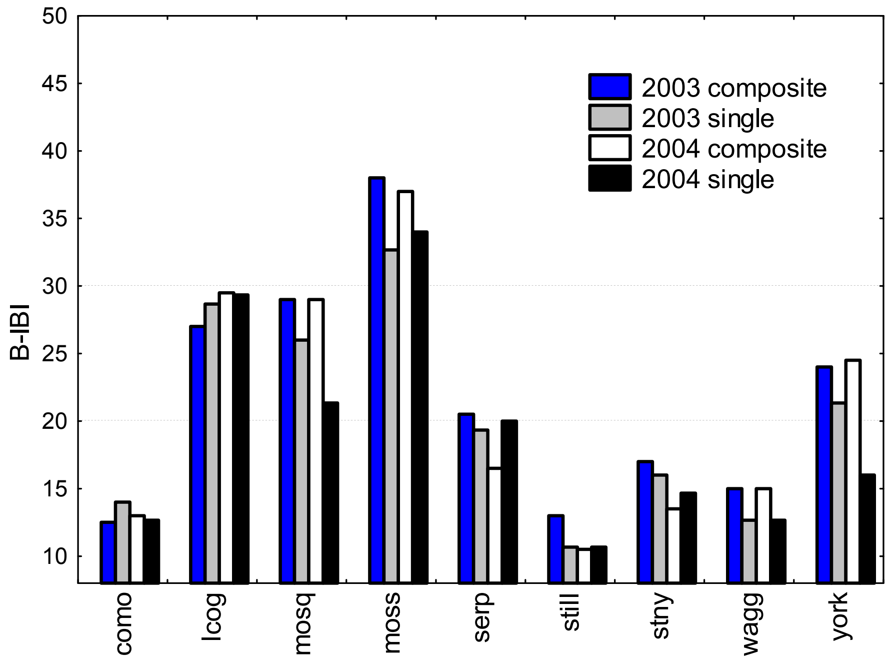

Figure 6. Average B-IBI values for nine permanent streams sampled in 2003 and 2004. Average B-IBI values are shown for composite Surber sampling (N = 4) and single Surber samples (N = 3). The legend shows the order to read the bars from left to right for each site.

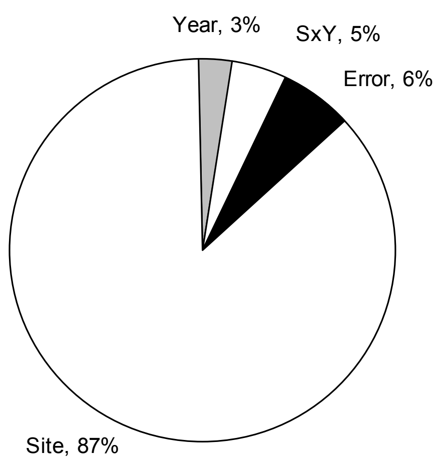

Figure 7. Components of variance analysis for B-IBI comparing the relative contributions of year, site, site-by-year interaction, and error to the total variance. Differences associated with year (1999, 2003, and 2004) were smaller than values observed for the two other sources of nuisance variance (site-by-year and error). Data were from nine stream sites with four composite samples each identified to lowest possible taxonomic level (LPTL).

Components of Variance for B-IBI – Components of variance analysis based on ANOVA was used to partition the relative contribution of four sources of variance to the total variance of B-IBI; these were site differences, year differences, site-by-year interaction and error (Figure 7). For B-IBI derived from composite sampling and LPTL identification, variability associated with site differences represented the largest component (87%). Differences associated with year (1999, 2003, or 2004) were relatively small and represented 3% of the total variance of B-IBI. Though small, the difference associated with year was significant (ANOVA, $p < 0.05$). Other nuisance sources of variance, error and site-by-year interaction, represented 6% and 5% of the total variance respectively. This indicates that most of the variation in B-IBI values is not associated with temporal trends. Data from 2005 was not included because of the reduced number of repeat samples (5 versus 9).

Spatial Differences in B-IBI – Two streams, Coghlan and Mossom, were sampled in both their central and lower reaches. Average B-IBI values for upper Coghlan and lower Coghlan creek sites were very similar for all three years, and differed by 1–1.5 points each year (Table 12A). In contrast, B-IBI values for upper Mossom and lower Mossom creek sites were quite different in 2003, 30.0 vs. 38.0. This large difference was probably associated with the small number of individuals collected in upper Mossom; this was the only sampling site that consistently failed to yield >400 individual invertebrates for composite samples. Stream flows at the upper Mossom Creek site were very low in mid-September 2003. In 2004 the difference between upper and lower Mossom Creek was smaller, 33.5 vs. 37.0 and the average number of individuals per sample was higher at 367. In 2005, adequate individuals were again collected at each location and the difference in B-IBI was only 1.5 points.

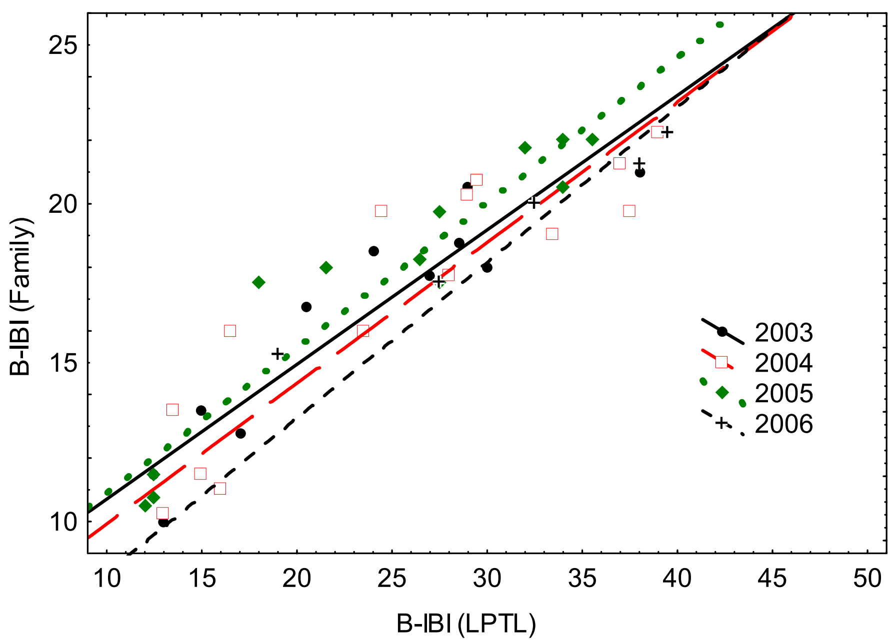

Figure 8. B-IBI index values calculated for two levels of taxonomic effort: lowest practical taxonomic level (LPTL) and family. Regression lines are drawn separately for each year (2003–06). Each point represents the average of B-IBI for four composite samples at each stream site. Note that

the family B-IBI ranges from 5–25 while the LPTL B-IBI ranges from 10–50 (2003: $r^2 = 0.87$; 2004: $r^2 = 0.83$; 2005: $r^2 = 0.90$; 2006: $r^2 = 0.91$)

Comparison of Taxonomic Effort – The B-IBI values calculated for two different levels of taxonomic effort (LPTL vs. family) showed a strong correlation for all three years (Pearson's r = 0.93 [2003], 0.91 [2004], and 0.95 [2005], $p < 0.01$). The relationship between the two versions of B-IBI was nearly identical for all three years as represented by the overlapping regression lines (Figure8). Overall, family-level B-IBI values were slightly lower for less disturbed sites than for the LPTL B-IBI.

The components of variance analysis was used to compare the LPTL and family versions of B-IBI (Figure 9). For B-IBI based on composite sampling and LPTL, error variance represented 7.1% of the total variance. For family-level taxonomy, error variance represented 11% of the total variance (Table 16).

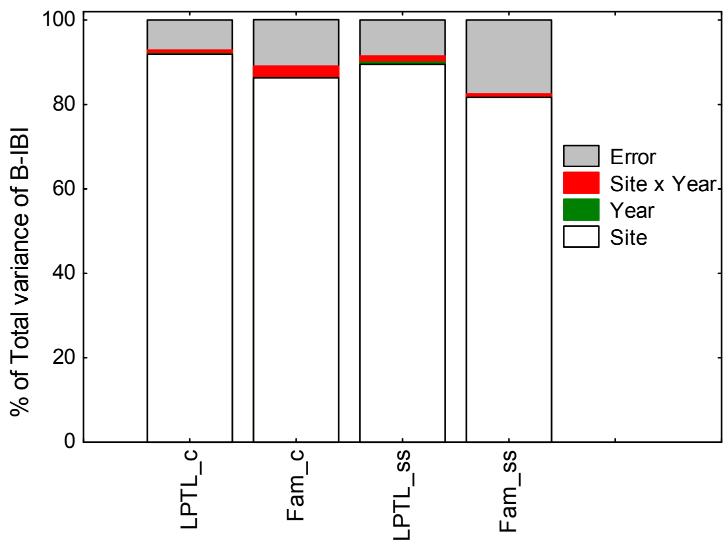

Figure 9. Variance components for B-IBI derived from LPTL and family-level identification and for composite and single Surber field sampling. For all four versions of B-IBI year variance was close to zero and site-by-year interaction was very small. Error variance represented a greater proportion of the total variance for family-level B-IBI than for LPTL. Error variance was also slightly higher for single Surber sampling than composite sampling. The error component represents the variability associated with repeat visits to the same site. Data shown are for 2003 and 2004. For composite sampling, N = 72 (9 sites x 2 years x 4 replicates); for single Surber sampling N = 54 (9 sites x 2 years x 3 replicates).

Table 16. Comparison of percentage of total variance of B-IBI due to site differences, year differences (2003 vs. 2004), site-by-year interaction, and error (within habitat variability) for B-IBI calculated to lowest possible taxonomic level (LPTL), family-level, and for composite and single Surber field sampling.

Source	LPTL Composite	Family Composite	LPTL Single	Family Single
Site	91.9%	86.3%	89.5%	81.7%
Year	0.1%	0%	0.5%	0%
Site-by-Year	0.9%	2.8%	1.5%	0.8%
Error	7.1%	11.0%	8.5%	17.5%

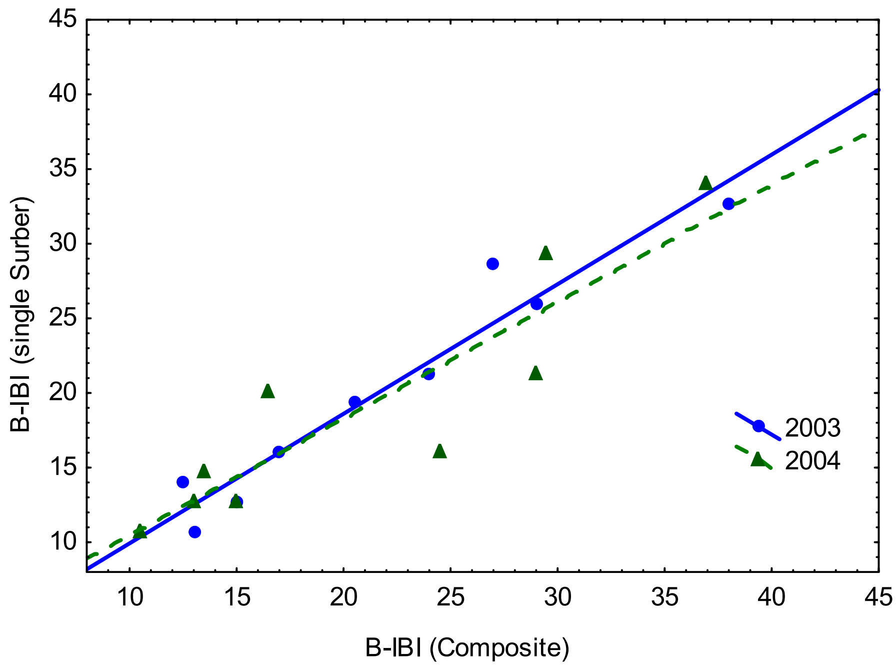

Figure 10. B-IBI values calculated from two different field sampling protocols were highly correlated for both years (2003: $r^2 = 0.94$; 2004: $r^2 = 0.82$). For less disturbed sites, B-IBI values were slightly higher for composite sampling than for single Surber samples. Shown here are average B-IBI values for nine stream sites with LPTL identification.

Comparison of Field Sampling Effort – Two different field sampling protocols were used to collect invertebrates in 2003 and 2004: i) composite samples consisting of three Surber samples each from four sites; and, ii) three individual Surber samples at one site. The results for both field sampling methods were highly correlated for both years (Pearson's $r = 0.97$ and $= 0.91$, $p < 0.01$; Figure 10). In general, B-IBI values derived from composite samples tended to be slightly higher than for single Surber sampling.

The LPTL versions of B-IBI had similar percentages of variance due to error for composite and single Surber sampling (7.1% and 8.5%; see Table 16 and Figure 9). For family-level identification, B-IBI derived from composite sampling was somewhat more precise, that is, the percentage of the total variance due to error was somewhat lower for composite sampling (8.5% vs. 17.5%).

Table 17. Sources of variability for B-IBI and their associated estimates of variance for data collected in the Greater Vancouver area and two counties in Washington State. Results were derived from similar collection protocols that used a single Surber placement to collect three replicate samples. Laboratory identification was to LPTL for all three studies. The number of sites and the total number of samples collected from the sites varied across studies.

Source of variance	Greater Vancouver	Snohomish County	King County
Year	0.30	4.58	2.89
Site-by-year	0.98	2.06	3.69
Error (within site)	5.56	11.45	14.44
Number of sites (no. of samples)	9 (54)	34 (80)	27 (191)

Regional Trend Monitoring – As expected, statistical precision increased as more sampling sites were included in the monitoring program (Figure 11). In other words, a more intensive sampling effort would conclude that a smaller change in B-IBI was statistically significant; therefore, the design is said to have greater statistical power to detect change (Appendix I). The increase is most dramatic for small numbers of sites, but tapers off for >40 sites.

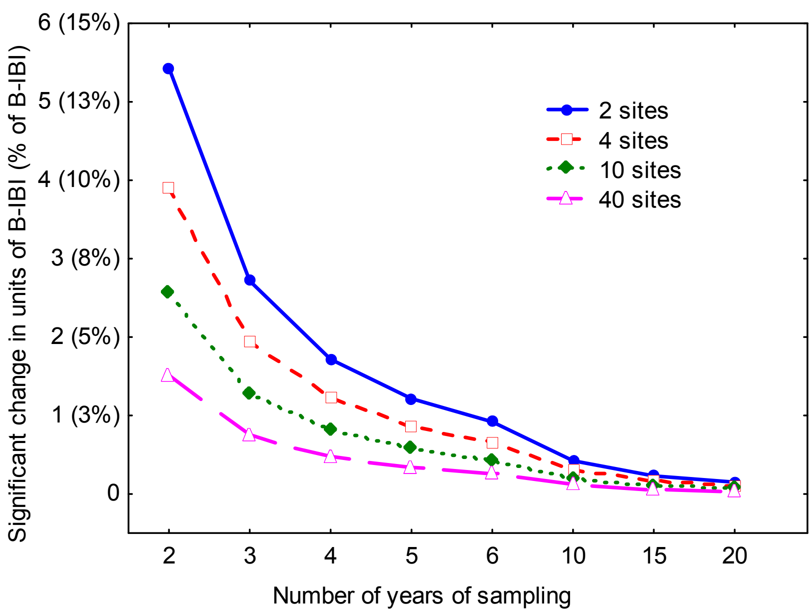

Figure 11. The change in B-IBI that would represent a statistically significant change is plotted in units of B-IBI (and as a percentage of B-IBI range). The change in B-IBI that can be detected grows smaller for more sites or more years of sampling. As an example, based on three years of sampling at the same four sites, an average decline of two units of B-IBI per year (or 5% per year) would represent a statistically significant change. This indicates that a smaller number of samples at a larger number of sites are more efficient for detecting B-IBI change. See Table 16 for variance estimates used to derive estimates of statistical power.

For a sampling design with 40 sites sampled once each year, a change of 1.5 units of B-IBI would indicate a statistically significant change. If three replicate samples are collected each year, a change of 1.2 units of B-IBI would be statistically significant. Thus, for regional trend monitoring, the effect of collecting more than a single replicate at each site provided a relatively small improvement in precision. If 40 samples can be collected each year, a more powerful sampling design would collect a single sample from 40 sites rather than three replicate samples from 13 sites. Results for this study were similar to results observed for two counties is Washington (Table 17) indicating that variance estimates for B-IBI are consistent regionally. Variance estimates for Greater Vancouver were relatively lower than those observed in Washington for two reasons. The sites sampled in Washington were sampled over several years during which time changes associated with human influence probably contributed to observed changes in B-IBI in addition to changes associated with natural variability.

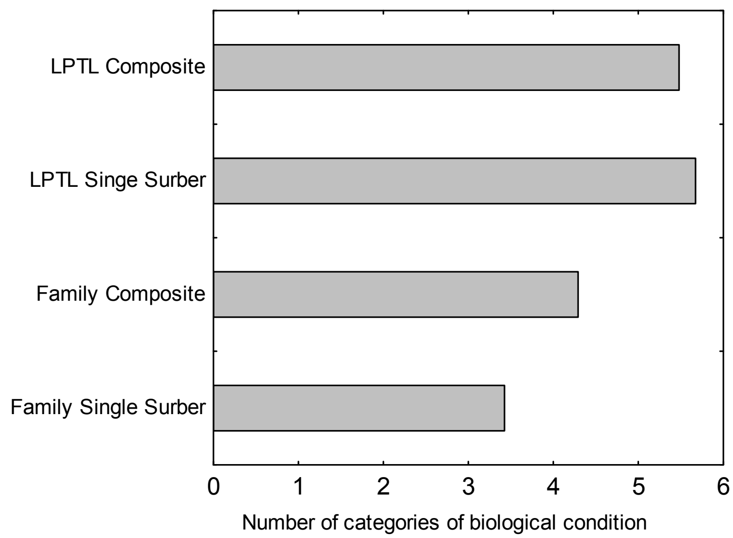

Figure 12. The number of categories of biological condition that each version of B-IBI could detect. Indexes based on LPTL identification were more precise and could detect more categories of biological condition than indexes based on family-level identification. Results for the two field sampling protocols were similar for the B-IBI (LPTL), but for the B-IBI (family) single Surber sampling was somewhat less precise, i.e., could detect fewer categories, than the composite sampling protocol. The number of categories was calculated by dividing the possible range of each index by its MDD (for LPTL: 40/MDD; Family: 20/MDD).

Minimum Detectable Difference – We compared four versions of the B-IBI for their relative ability to detect a change in site condition (Figure 12). We calculated the minimum detectable difference, that is, the minimum difference in average B-IBI values observed for two sites that would represent a statistically significant change (Table 18).

The B-IBIs derived from LPTL identification were more precise and could detect a smaller average difference in index values than the family B-IBIs. For the B-IBI (LPTL), the MDD for composite and single Surber sampling were similar; for example, based on three replicate samples at two locations, a difference of 7.3 in B-IBI would indicate a significant difference between the locations for the composite method while a difference of 7.1 would be significant for single Surber sampling (Table 19). The slight difference between the two methods is not likely to be meaningful. For the B-IBI (family), composite sampling was slightly more precise and capable of detecting a smaller difference between two locations; for example, based on three replicate samples at each location, a difference 4.7 in B-IBI would indicate a significant difference between locations for the composite method while a slightly larger difference of 5.8 would be significant for the single Surber method. Recall for this comparison that the B-IBI ranges from 10–50 for LPTL and 5–25 for family-level identification.

In order to compare the different versions of the index while controlling for their different scales, we calculated the number of categories that each version of B-IBI could detect. To do this we divided the possible range of each index (LPTL: 50–10 = 40; Family: 25–5 = 20) by the MDD (see Figure 11).

Table 18. Sources of variability for B-IBI and their associated estimates of variance for different field collection methods (composite samples vs. single Surber samples) and laboratory identification methods (LPTL vs. family).

Source of variance	LPTL Composite	Family Composite	LPTL Single	Family Single
Year	0.10	0	0.30	0
Site-by-year	0.76	0.62	0.98	0.17
Error (within site)	5.94	2.43	5.56	3.81

Table 19. Minimum detectable difference (MDD) for a given number of replicates at each site (N) for a 2-sided two-sample *t* test (alpha = 0.1; beta = 0.1). MDD is shown for B-IBI calculated according to different field collection methods (composite samples vs. single Surber samples) and laboratory identification methods (LPTL vs. family). As an example, for three replicate samples at each site, with identification to LPTL and composite sampling, a difference in the mean B-IBI values for each site of 7.3 *or greater* would represent a statistically significant change. Note that the LPTL B-IBI ranges from 10–50 points while the Family B-IBI only ranges from 5–25; values in the table refer to B-IBI units for each index.

N	LPTL Composite (10-50)	LPTL Single (10–50)	Family Composite (5–25)	Family Single (5–25)
2	11.72	11.33	7.48	9.39
3	7.30	7.05	4.66	5.84
4	5.83	5.64	3.73	4.67
5	5.02	4.85	3.21	4.02
7	4.09	3.95	2.61	3.28
10	3.34	3.23	2.13	2.68

B-IBI Values in Intermittent Streams –The variability of B-IBI for within-site repeat samples differed by site in intermittent streams. Tynehead and Blueridge creeks had very low B-IBI values which were more consistent across replicate samples. In contrast, B-IBI values at Forestry Creek were more moderate and also more variable for replicate samples. One sample in Forestry Creek from 2005 had substantially few individual organisms (85 compared to 373 for the other three samples) which contributed to the variability. The sample with fewer organisms was from the transition between surface and subsurface flow.

The similarity of B-IBI values in intermittent streams to permanent streams cannot be assessed without B-IBI values from adjacent permanent streams, or recent watershed land cover data for regression analyses.

Comparison of Annual and Within Site Variance Across Stream Types – Although sample sizes were small for intermittent, upstream and reference sites, we compared two sources of nuisance variance using an ANOVA model for each of the four stream types. Each of the four ANOVA models had two factors: site and year with no interaction terms. Replicate samples collected during a site visit were used to estimate the root mean squared error (RMSE). RMSE is equivalent to the SD reported for each site averaged across all sites (see Table 15). The annual variance for each model was estimated from the mean squared effect for year.

Undisturbed sites had the highest variance associated with annual differences (Table 20). This higher variance reflects the lower B-IBI values observed at Beaver Creek after a culvert replacement and road construction upstream. Annual variability calculated from three years of sampling indicated minimal variance in B-IBI due to differences in year. The variance associated with replicate sampling in the stream site was highest for the two upstream sampling sites and lowest for the nine primary sampling locations. Though higher, the difference in SD was not large for the different stream types and ranged from 2.8–5.0.

Table 20. Number of stream-visits, type of stream, and average standard deviation of B-IBI derived from replicate samples at the same site.

N	Stream type	Year (MSE)	RMSE
25	Primary	0.8	2.8
6	Undisturbed	4.8	4.1
6	Upstream	0	5.0
8	Intermittent	0	3.6

3.4 Multivariate (NMDS) Analysis

The nonmetric multidimensional scaling (NMDS) ordination showed broad similarities in different stream types (Figure 13). Samples from highly urbanized streams (Still, Como, Wagg, and Stoney creeks) had similar benthic invertebrate communities and form a distinct, tightly confined group. Blueridge and Tynehead creeks (urbanized, intermittent streams) also have similar community composition. Forestry Creek is distinct from both the other intermittent streams and the reference streams which suggests that some intermittent streams in undeveloped watersheds have distinct benthic invertebrate communities compared to permanent streams. Samples from moderately urbanized

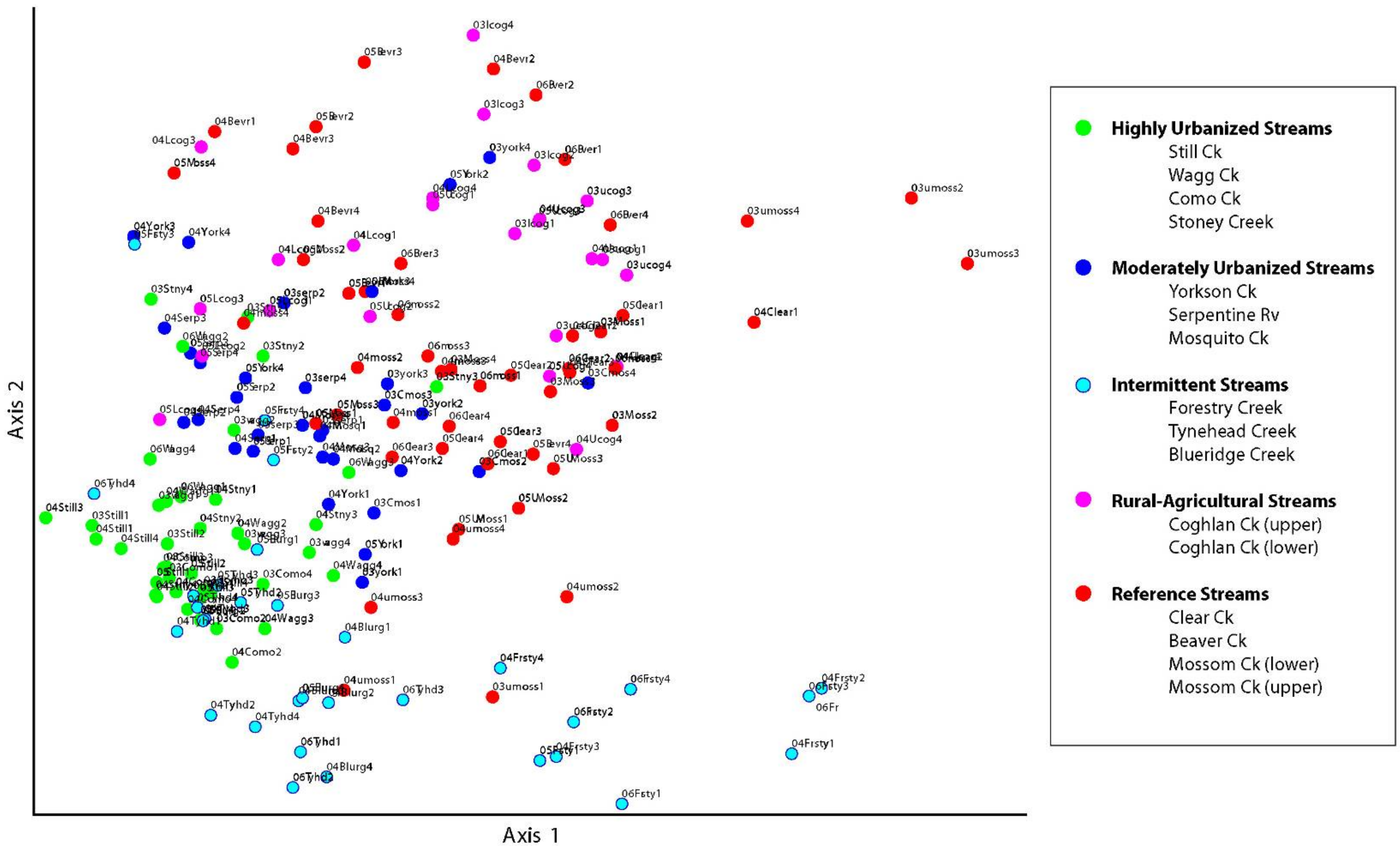

Figure 13. Ordination of 180 composite samples from 14 streams in Greater Vancouver from 2003-2006 using NMDS. For interpretation, streams were subjectively separated into five groups based on dominant land cover or flow regime: highly urbanized; moderately urbanized, intermittent; rural-agricultural, and reference streams. The stress of the final 3-dimensional solution was 16.9 and the instability was 0.0008 after 100 iterations. The cumulative explanation of variance provide by the three axis was r^2=0.79 (Axis 1: r^2=0.32; Axis 2: r^2=0.26; and Axis 3: r^2=0.21).

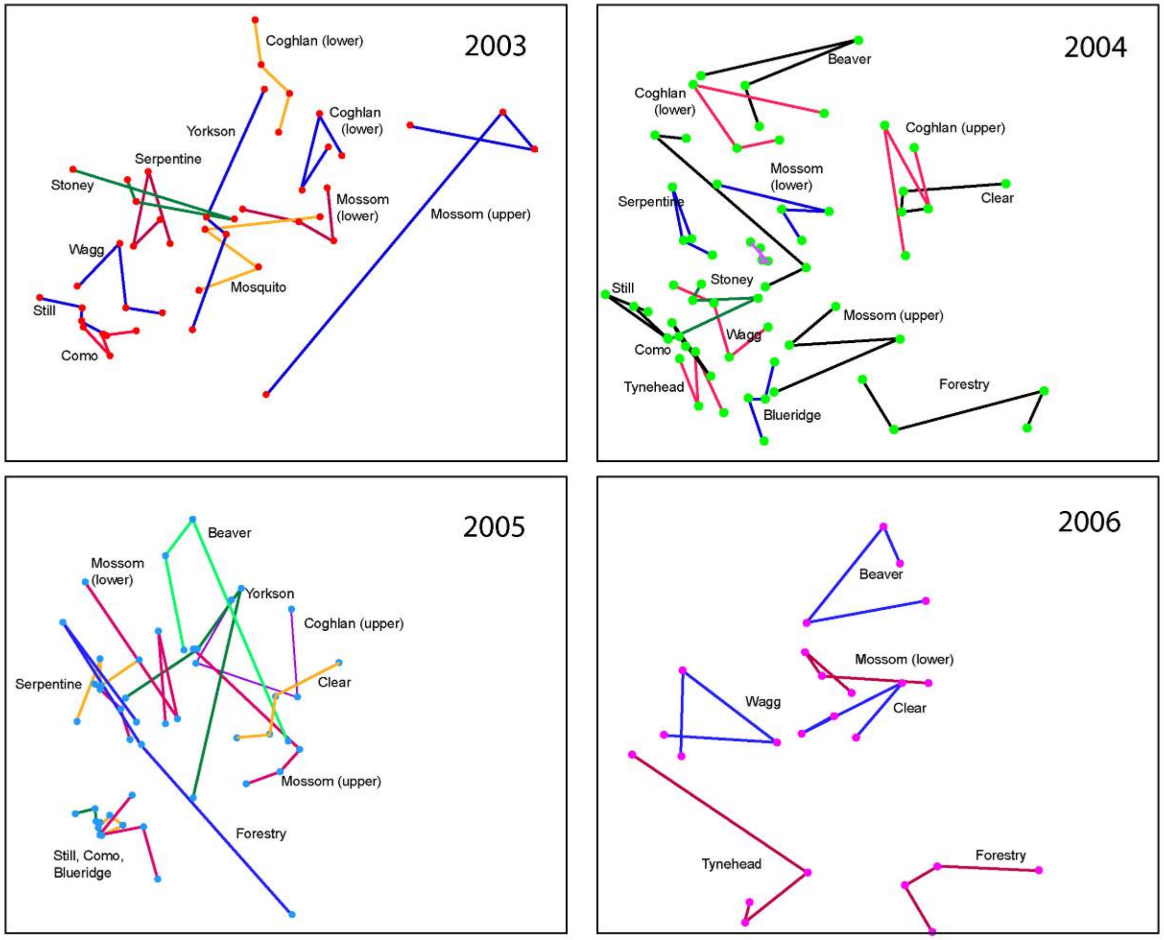

Figure 14. Relationships of 44 composite samples from nine streams sites in 2003, 52 composite samples from 13 stream sites in 2004, and 48 samples from 12 streams in 2005, and 24 composite samples from six streams in 2006 using NMDS. Lines indicate samples from the same stream (linked in spatial order from downstream to upstream). Samples that are closer to one another are more similar in terms of benthic invertebrate composition and abundance. The most urbanized streams (Still, Como, and Wagg creeks) show little variability and little change over time (smaller polygons on right side of graphs). The ordination values are the same as shown in Figure 13.

streams (Yorkson Creek, Serpentine River, and Mosquito Creek), rural-agricultural streams (Coghlan Creek), and reference streams (Clear and Beaver creeks + Mossom Creek) do not form distinct groups, although samples from reference streams support benthic invertebrate communities that are substantially different from urban streams. It is interesting that samples from Coghlan Creek, despite the relatively intensive rural development, are very similar to samples from reference streams. The samples from upper Mossom Creek in 2003 are distinct from all other samples except one Clear Creek sample from 2004.

NMDS also showed strong groupings of samples collected from the same stream. This indicates benthic invertebrate composition and abundance is relatively uniform within streams in Greater Vancouver at the scale of the 500 m long reach (Figure 14). In all years, samples from the most urbanized streams were more similar to one another than samples from less disturbed streams. This indicates that urbanization is accompanied by increasing homogeneity of the benthic invertebrate community.

4.0 DISCUSSION AND CONCLUSIONS

Results from this 4-yr study support the use of benthic invertebrates as an effective assessment and monitoring tool in small urban watersheds in Greater Vancouver for four reasons. First, B-IBI was consistently and predictably correlated with independent measures of human disturbance at the watershed scale. Second, B-IBI values consistently ranked sites sampled in different years. Third, large differences in laboratory and field sampling protocols had little influence on the overall ranking of sites in terms of biological condition based on B-IBI values. Fourth, the statistical precision of B-IBI was high enough to detect small changes in biological condition. The consistent response of B-IBI to human disturbance, independent of protocol methods, supports the use of the index as a management tool for small streams in Greater Vancouver.

- *B-IBI values were correlated with independent measures of human disturbance.*

B-IBI and its component metrics were correlated with independent measures of watershed land cover, such as total impervious area and riparian forest cover, as well as environmental variables such as water temperature and specific conductivity which are associated with urbanization. Thus, the spatial variability observed in B-IBI was meaningful in that low values consistently indicated greater human disturbance. B-IBI values were highest in watersheds with low imperviousness and large amounts of riparian forest cover and declined with increasing intensity of urban land use. B-IBI was not correlated with stream channel characteristics such as substrate variables. This suggests that B-IBI was unresponsive to natural site differences but strongly responsive to watershed scale factors.

Statistical analysis partitioned the sources of variance for B-IBI and further supports this conclusion because the largest contributor to the overall variance of B-IBI values was site differences. The relative contribution of site variability to the total variance of B-IBI was much greater than other nuisance sources such as differences due to time of sampling or location of sample location within a reach. These differences in site condition associated with human disturbance are the differences that B-IBI was designed to detect (Karr and Chu, 1999; Kerans and Karr, 1994).

NMDS ordination also showed that streams with higher urban land cover had more similar benthic invertebrate communities with less annual variability than less disturbed streams.

- *B-IBI values for five years of sampling were highly correlated and suggest B-IBI remains stable over time in streams in Greater Vancouver.*

High correlation among B-IBI values for four different years of sampling (1999, 2003, 2004, 2005, and 2006) indicates that B-IBI is a consistent and reliable indicator of stream condition. Components of variance analysis showed that the relative contribution of variance associated with year to the total variance observed for B-IBI was relatively small compared to differences in sites (3% of the total variance). Other studies have found similar small components of variance associated with yearly differences for multimetric indexes (Hughes *et al.*, 1998; Karr and Chu, 1999; Fore, 2002). This comparison assumed that watershed land use and other human influences remained constant during 1999–2006. If land use patterns did change during this time, the natural variability associated with temporal changes will actually be smaller than what is reported here.

- *Variation in B-IBI at different locations within two streams was minimal.*

The spatial effect of upstream and downstream locations was evaluated for two streams in 2003, 2004, and 2005. For the two reaches on Coghlan Creek, B-IBI values were nearly identical during all three years indicating similar biological condition at these two sites. For the two reaches on Mossom Creek, B-IBI values from 2003 differed by eight points, 20% of the B-IBI range. These differences were probably due to the difficulty of sampling the upstream site due to very low water. B-IBI values were lower because fewer organisms were collected; in fact, three out of four samples had less than 150 individuals, significantly below the target value of 400 individuals per sample. Within site variability (derived from replicate sampling) was also much higher for this site, indicating that B-IBI values will be affected when standard sample sizes are not maintained. Results from 2004 and 2005 support these conclusions. In both years, higher flows allowed a representative sample to be collected. B-IBI values for the two sites differed by only 3.5 points in 2004 (9% of B-IBI's range) and 1.5 points in 2005 (4%), both of which were within the expected range for natural variability.

This type of comparison assumes that the different sites on a stream experience a similar level of human disturbance. This was true for these pairs of sites, but cannot be assumed for all situations because B-IBI is sensitive to differences at the reach scale as well as the watershed scale (Morley and Karr, 2002).

- *The two versions of B-IBI based on different levels of taxonomic identification ranked sites similarly.*

The two versions of B-IBI based on different levels of taxonomic effort (LPTL vs. family) were highly correlated. They differed slightly in their statistical precision which translated into differences in the number of categories of biological condition that each version could detect. Nuisance sources of variability, i.e., differences associated with years or within-site variability (error), contributed less to the total variance of B-IBI for the LPTL index than for the family B-IBI (8% vs. 14%). This modest difference translated into the ability to detect 5.5 categories of biological condition for the LPTL and 4.3 categories for the family index based on calculations of the minimum detectable difference (MDD) for site comparisons. A similar study in Washington state identified also determined that the B-IBI (LPTL) could detect ~5 categories of biological condition (Fore et al., 2001).

Although the precision was somewhat lower for the family-level index, variability at the replicate level is less of a concern if replicate sampling is part of the standard protocol because the mean values from three or four replicates will eliminate this source of index variability. If a family-level B-IBI is selected for monitoring, the scoring rules should be evaluated further because the rules applied here were based on a small sample size of ~15 stream sites. Rules could be adjusted using data from this study to more closely match the B-IBI values based on LPTL.

- *The two field sampling protocols yielded very similar B-IBI values.*

The two field sampling methods (single Surber or composite) yielded highly similar B-IBI values. The primary advantage of smaller samples based on a single Surber is less processing and handling time in the laboratory; however, when the target minimum number of individuals is consistently not obtained, the B-IBI index cannot be correctly calculated. In the Puget Sound lowlands regions, many sites fail to yield 400 individuals from a single Surber sampler (King County, unpublished data). For the Greater Vancouver data, 44% of the samples had <350 individuals for the single Surber samples; however, 23% of the composite samples also had <350. As a result, some sites may miss the target sample size due to extreme degradation associated with disturbance. Two sites with low or intermittent flows had consistently smaller numbers of individuals (Upper Mossom and Forestry creeks). These results suggest that composite samples of three Surber sampler placements each should be collected at each site.

The two methods also differed slightly in terms of taxa richness. For sites with poor biological condition, indicated by low B-IBI values, agreement between the two sampling methods was high. Where the two methods differed was for sites with higher B-IBI values (indicating better biological condition). These sites typically have more taxa present and the more extensive sampling method yielded more rare taxa such as the Trichoptera and the long-lived taxa.

- *Statistical power of B-IBI was sufficient to detect small changes in stream condition.*

A biological indicator must be a reliable and consistent indicator of biological condition and it must have the statistical precision to detect the magnitude of change necessary to protect resources. The B-IBI based on LPTL and composite sampling could detect 5.5 categories of biological condition based on site-to-site comparisons. In other words, a difference in B-IBI for two sites with three replicates each of 7.3 or greater would represent a statistically significant difference. For regional trend analysis B-IBI was also very sensitive to change through time. The detectable change depends on the specific sampling scenario, that is, the number of sites and the number of sampling years. For a sampling scenario based on repeat visits to ten sites, each with a single sample per year, B-IBI could detect an 8% change based on two years of sampling or a 3% per year change over four years of sampling. The increase in precision for three vs. one replicate sample for each site was minimal and reinforces the idea that more sampling sites with fewer replicates per site represents the best sampling scenario for regional trend monitoring. Thus, B-IBI has sufficient statistical precision to detect changes both at the level of individual sites or at a regional scale.

- *The differences in B-IBI values between seasonal and permanent streams were difficult to assess.*

Mean B-IBI values in the three seasonal (intermittently flowing) streams varied. We could not assess whether the variability was due to their intermittent flows or adjacent land use. We could not compare B-IBI values in intermittent streams to permanent streams without B-IBI values from adjacent, and environmentally similar, permanent streams. Similarly, recent watershed land cover data was not available to evaluate the relationship of watershed scale variables (e.g., imperviousness) and B-IBI values in seasonal streams using regression analyses. Three points are noteworthy. First, the mean B-IBI value for Forestry Creek was low relative to its estimated total imperviousness of <1%. This suggests that B-IBI values in seasonal streams may be lower than permanent streams relative to their urban land cover. Second, several of the composite samples from intermittent streams failed to reach the target number of 400 organisms for B-IBI calculation. Third, field sampling was difficult in cobble and boulder dominated habitat, typical of intermittent streams during low flow conditions and may have reduced the number of organisms captured.

- *B-IBI values from the reference streams were high and are representative of best available condition in the region.*

The sampling results from the reference streams suggest they will be useful for tracking long-term change in undeveloped watersheds. B-IBI in Clear Creek ranged from 37.5 (2004), 34.0 (2005), and 32.5 (2006). B-IBI in Beaver Creek declined from 39.0 in 2004 to 32.0 in 2005 and then rebounded to 39.5 in 2006. The decline in B-IBI in 2005 may have been associated with culvert replacement above the sampling reach in October 2004. Water quality measurements in both reference streams indicate they were undisturbed by anthropogenic effects associated with land use. Mossom Creek may also be a suitable reference site: mean B-IBI was 38.0 (2003), 37.0 (2004), 34.0 (2005), and 38.0 (2006).

Summary – This study strengthens the foundation for environmental assessment and monitoring in Greater Vancouver streams and rivers using benthic invertebrates. In particular, it has shown the low annual variability of B-IBI in most streams, and the consist response of the benthic invertebrate community to urbanization. The collection of multiple samples from each stream has also shown the low variability of benthic invertebrates as the scale of a 500 m long reach.

Further work remains to better understand the benthic invertebrate community in intermittent streams, how they should best be sampled, and how their B-IBI values should be compared to permanent streams. As well, continued monitoring of the streams that were the focus of this study will provide useful information on long-term change in both reference (undisturbed) and urbanized streams in Greater Vancouver.

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