CHAPTER 11 EMERGENT MACROINVERTEBRATE COMMUNITIES IN RELATION WATERSHED DEVELOPMENT

by Klaus O. Richter, Kenneth A. Ludwa and Robert W. Wisseman

INTRODUCTION

Aquatic invertebrates play important roles in the food chain of fresh water wetlands. They are the pivotal link between the primary production and detrital trophic levels and higher level consumers including fish, amphibians and aquatic avifauna and mammals (Cummins and Merritt 1996). Moreover, aquatic macroinvertebrates have historically been used as biological indicators in riverine and lacustrine environments (Rosenberg and Resh 1996). Studies by scientists with the Puget Sound Wetlands and Stormwater Management Research Program (Ludwa 1994), (Azous 1991) and others (Murkin and Batt 1987), (Rosenberg and Danks 1987), (Wrubleski 1987), (Hicks 1995, Hicks 1996) have demonstrated the utility of macroinvertebrates as indicators of the health of palustrine environments, particularly for assessing the impacts of urbanization.

Macroinvertebrate communities are noted for their response to the four major wetland stresses identified by EPA's Environmental Monitoring and Assessment Program (EMAP): (1) altered hydroperiod, (2) excess sediment, (3) changes in nutrient cycling; and (4) contaminants (Liebowitz and Brown 1990). Unlike other wetland animal communities (amphibians, mammals, birds, fish) the larval forms of aquatic macroinvertebrates are completely confined to the water within a particular wetland, over entire growing seasons or years, until emergence. Therefore, the aquatic macroinvertebrate community is an excellent integrator of wetland impacts; it does not register impacts that may occur to other wetland animals that migrate outside the wetland for periods of time.

The goal of this study was to establish the impacts of watershed development and particularly, urban stormwater inputs on macroinvertebrate communities. Specific objectives included (1) developing a preliminary wetland macroinvertebrate community-based biotic index based on methods proven for streams, and (2) applying this index to examine the impacts of watershed urbanization on specific aspects of macroinvertebrate communities, over a range of watersheds with different levels of existing development, and within developing watersheds over time. The latter objective is based upon several hypotheses regarding the response of the aquatic macroinvertebrate community to anthropogenic changes to wetlands and their watersheds. These included (1) Changes in macroinvertebrate taxa richness and numbers of individual organisms will reflect changing land use, environmental pollution, direct habitat degradation, and general system health; (2) proportions of sensitive and tolerant taxa will change with increasing watershed urbanization and wetland habitat degradation; and (3) proportions of functional taxa groups will change with alterations to a wetland’s nutrient cycle.

Although aquatic macroinvertebrates include non-insect taxa, the sampling device used in this study collected only adult aquatic insects. Therefore, the terms macroinvertebrates and insects shall be used interchangeably in this paper.

METHODS

We periodically monitored emergent aquatic macroinvertebrates in nineteen palustrine wetlands in the Puget Sound Basin from 1988 to 1995. These wetlands were located in watersheds in various stages of urban and suburban development and have been described in earlier chapters.

Trapping protocols are extensively described in Chapter 4 and are briefly summarized here. We made an attempt to place traps in conditions as similar as possible between wetlands (open still water, fine sediment). Location of traps was particularly important because the presence or absence of certain vegetation or substrate types can substantially influence the character of the aquatic macroinvertebrate community. We deployed the traps in each wetland over the periods listed in Table 1. Field staff collected the trap contents and replaced the preservative on an approximately monthly basis from April to September during each monitoring period, with a season-end collection also made in October and/or November. We made no collections from December through March because of low invertebrate activity during this period. The traps provided a cumulative measure of insect emergence between each occasion that the traps are emptied.

Table 11-1. Approximate aquatic invertebrate emergence trap sampling periods for growing seasons 1989, 1993, and 1995.

	1989	1993	1995
Start collection	September 1, 1988*	April 10, 1993	January 1, 1995
End collection	September 31, 1989	April 9, 1994	October 30, 1995

* Monitoring at Fourteen sites were started in September 1988; five more sites were added in April 1989.

We identified and enumerated the macroinvertebrates collected in 1989 to the lowest level possible, in most cases genus or species. We identified insects collected in 1993 and 1995 only to family for Dipteran taxa, and to order for all other taxa. We made identifications to a consistent level within each taxonomic group for all samples.

Using the 1989 data set, we developed a multimetric biological index based on principles of the Benthic Index of Biotic Integrity (Fore et al. 1995). We proceeded by first testing metrics to determine whether they differentiated between the two best and two worst sites; we then confirmed these metrics by testing them over the whole range of nineteen sites (Ludwa 1994) (Fore et al. 1995). We tested and adapted existing lotic macroinvertebrate community metrics to the wetland insect community, and tested and added new metrics unique to palustrine communities.

Because the level of taxonomic effort was considerably coarser for the 1993 and 1995 collections, we found it necessary to develop and test a new set of metrics suitable for that level of information. We performed this step with the 1989 collections by elevating the taxonomic data to the same levels as the 1993 and 1995 collections. Again, we followed the same procedures described by (Ludwa 1994). Most of the coarser-level metrics were based on, (Ludwa 1994) original metrics for the 1989 collections.

We tested the overall index scores against land use and wetland morphology thresholds reported by (Taylor et al. 1995) and Ludwa (Ludwa 1994) using the Mann-Whitney test (Zar 1984), the nonparametric equivalent of the independent groups t-test. We also tested index scores against parameters for wetland hydrology and water quality, and separately against wetland morphology and watershed land use, using multiple regressions (Zar 1984). All statistical analyses were performed at a significance level of $p > 0.05$.

RESULTS

It is important to note that we designed and calculated the 1989 species/genus-level metrics using data spilt into distinct sampling periods: April-June, July-September, and October-November (Ludwa 1994). The data split into these periods, especially the two summer periods, responded more strongly to urbanization parameters than did the year-long data set. We designed and calculated the 1989 order/family-level metrics using the year-round data sets. Taxa richness values for the coarser-level data were too low for individual sampling periods to differentiate between sites. We assumed that the difference between the length of sampling periods between the three years (Table 1) did not significantly affect taxa richness values, but that it did affect total numbers of individuals collected. The metrics developed for the order/family-level data were taxa richness- and proportion-oriented; therefore we assumed that different sampling period lengths did not affect metric design or calculation.

The metrics recommended for further testing by (Ludwa 1994) for emergent collections with genus-species level taxonomy are listed in Table 2. Although taxa belonging to orders Ephemeroptera, Plecoptera, and Trichoptera are often the basis of stream biological metrics, we found a paucity of these taxa in the wetland insect collections (including order Odonata, these orders are referred to as EPOT). Therefore, although EPOT richness and abundance did yield two metrics, most of the metrics (numbers 7 through 22, including all new wetland- oriented metrics) related to order family Chironomidae of order Diptera (aquatic midges and true flies). Chironomids are a highly diverse family only sparsely detailed in ecological literature; although generally considered to be negative indicators for running waters, Chironomids are adapted to lentic environments, and therefore may be more appropriate indicators of their health.

Using an index composed of the metrics listed in Table 2, (Ludwa 1994) calculated index scores and compared them to direct and indirect measures of wetland stress. Ludwa (1994) emphasized that further verification of this index and its component metrics is necessary before it can be used as an independent measure of wetland ecological health. Conclusions drawn from (Ludwa 1994) analyses follow.

Table 11-2. Biotic index metrics recommended for use with wetlands, based on emergent macroinvertebrate collections with genus/species-level identification (Ludwa 1994).

Metrics Included in Final Wetland Biotic Index (Genus/Species-level Taxonomy)	
Adapted from stream metrics:	Unique Wetland Metrics:
1. Taxa richness	9. Percent individuals as Chironomini tribe
2. Scraper and/or piercer taxa presence	10. Chironomini tribe taxa richness
3. Shredder taxa presence	11. Percent individuals as Tanypodinae subfamily
4. Collector taxa richness	12. Tanypodinae subfamily taxa richness
5. EPOT[1] taxa richness	13. Presence *Thienemanniella*
6. Percent individuals as EPOT	14. Presence *Endochironomus nigricans*
7. Percent individuals as tanytarsini tribe	15. Presence *Parachironomus* spp. 2
8. Tanytarsini tribe richness	16. Presence *Polypedilum* gr.1 and 2
	17. Presence A*blabesmyia*
	18. Presence *Aspectrotanypus algens*
	19. Presence *Paramerina smithae*
	20. Presence *Psectrotanypus dyari*
	21. Presence *Zavrelimyia thryptica*
	22. Presence *Tanytarsus*

[1]EPOT = Ephemeroptera, Plecoptera, Odonata, and Trichoptera.

There appeared to be two primary periods of insect emergence, in the early summer and again in the late summer/early autumn; sampling periods in April-June and July-September were most appropriate for calculation of biotic index scores. Collections made in October-November did not appear to be as effective for purposes of bioassessment.

Biotic index scores responded significantly to land use and wetland morphology parameters. A multiple regression revealed that scores responded negatively to total watershed impervious area, wetland channelization, and incidence of dryness. The regression explained 67 percent of the variance in index scores. Threshold analyses also revealed that index scores were significantly higher with increasing watershed forest coverage and lower with increasing impervious area. Highly channelized sites had significantly lower scores, consistent with the observation of degraded water quality for most parameters in highly channelized sites.

A multiple regression indicated that water quality and hydrology parameters explained a significant amount of variation of the index scores (as high as 73 percent). Index scores responded negatively to hydrogen ion concentration (antilog pH), conductivity, suspended solids, water level fluctuation, and incidence of wetland dryness. Suspended solids, conductivity, and water level fluctuation were demonstrated by (Ludwa 1994), (Taylor et al. 1995), and (Chin 1996) to be the water quality and hydrology parameters in these sites most significantly degraded by increases in watershed impervious area and decreases in forest cover. This illustrates the interrelationship between a wetland’s watershed, its physical and chemical parameters, and the health of its biological communities.

The order/family-level metrics developed with the 1989 data are listed in Table 3; Table 4 lists the resulting index scores calculated with these metrics for 1988, 1993, and 1995. Although the order/family-level metrics responded to indicators of urbanization, the overall index comprised of the metrics had much less power to discern between sites with different levels of urban impact. For example, the multiple regression of 1989 genus/species index scores versus total impervious area, wetland channelization, and incidence of dryness explained 67 percent of the index score variance. The same regression explained only 21 percent of the 1989 index score variance for the order/family data.

Table 11-3: Biotic index metrics recommended for use with wetlands, based on emergent macroinvertebrate collections with genus/species-level identification.

Metrics Included in Final Wetland Biotic Index (Order/Family-level Taxonomy)
• Family/Order Richness • Shredder Presence • Collector Richness • EPOT Order Richness • % Individuals as EPOT • % Individuals as Dixidae

After 1989, the next year in which land use data was available was 1995. The 1995 index scores were not significantly related to total impervious area or forested area, nor did the scores respond significantly in the multiple regression against total watershed impervious area, wetland channelization, and incidence of wetland dryness. Furthermore, the changes in index scores between 1989 and 1995 did not correspond to changes in land use. For example, NFIC12, which experienced an increase in impervious area from 2 percent to 40 percent, showed the highest percent increase in its index score, exactly opposite that which would be predicted (Figure 11-1).

Table 11-4. Order/Family macroinvertebrate index scores.

	Index Score		
	1989	1993	1995
AL3	16	10	20
B3I	12	8	6
BB24	26	10	16
ELS39	10	12	12
ELS61	18	10	18
ELW1	8	6	6
FC1	16	14	10
HC13	22	14	24
JC28	22	10	26
LCR93	28	16	6
LPS9	8	10	18
MGR36	20	12	16
NFIC12	10	10	24
PC12	18	10	10
RR5	10	6	18
SC4	16	10	12
SC84	14	14	12
SR24	18	10	14
TC13	10	12	10

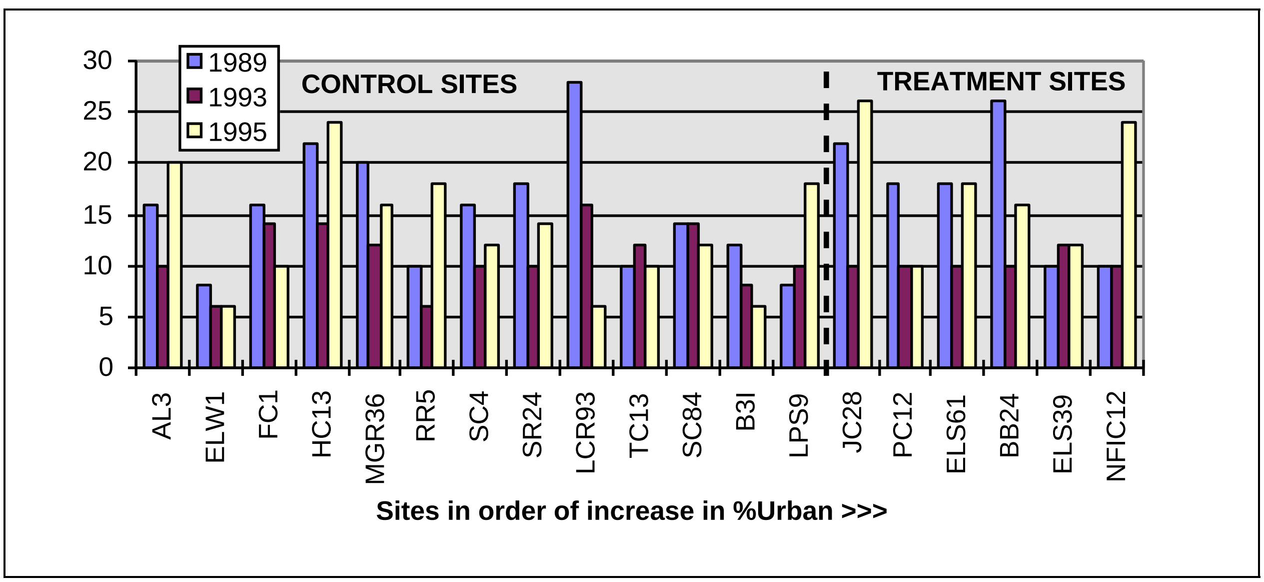

Figure 11-1. 1989, 1993, and 1995 Wetland macroinvertebrate index scores versus change in watershed urbanization.

In addition to relating index scores to changing watershed characteristics, we also examined changing taxa richness and abundance data to describe the impact of urbanization on emergent macroinvertebrates. Table 5 lists abundance and taxa richness values for each site in each year. Multiple regressions and threshold tests revealed no significant patterns in order/family taxa richness related to impervious area, between sites or years. In other wetland animal communities, taxa richness of sensitive species is often more responsive to wetland degradation than is overall taxa richness (e.g., Power at. al., 1989). The index developed for the species/genus-level data incorporates this concept by including sixteen metrics based on the presence of taxa that are assumed to be more sensitive to disturbance. The order/family data does not allow enough resolution to indicate sensitive taxa. Numbers of individuals decreased from 1989 to 1995 in 14 out of 19 sites, but, as discussed above, we assume that this is primarily a function of a longer sampling period in 1989.

Table 11-5. Insect abundance and order/family richness: 1988, 1993, and 1995.

	Abundance			Taxa Richness		
	1989	1993	1995	1989	1993	1995
AL3	4408	3619	1946	12	11	13
B3I	3027	2219	988	14	10	8
BB24	8857	14742	5815	14	10	13
ELS39	7337	6267	3773	12	12	12
ELS61	20828	13457	2808	16	10	12
ELW1	1239	503	157	10	7	7
FC1	4736	13332	5751	14	9	9
HC13	8748	4436	2934	15	11	13
JC28	1133	5778	1251	13	8	13
LCR93	9689	12148	40464	15	12	7
LPS9	5127	1006	5490	12	10	12
MGR36	7365	13276	1918	14	10	10
NFIC12	8869	24866	2015	12	11	13
PC12	5893	10701	4350	15	11	11
RR5	8621	4748	2150	12	10	11
SC4	2952	2794	2962	12	10	12
SC84	3692	2159	1254	13	9	11
SR24	5598	4982	1140	14	8	12
TC13	4657	4204	4657	13	9	13

SUMMARY

We recommend further development of macroinvertebrate community-based biological indices for assessment of wetland biological health. Our results suggest that this kind of index may be as useful as comparable indices established for running waters. Further testing of the metrics proposed by this study are necessary before the index may be used as an independent wetland assessment tool in the Puget Sound Ecoregion. Furthermore, refinement of insect tolerance and feeding group information may allow the index to be used as a diagnostic tool. Alternatively, in a set of proposed guidelines for assessing wetland health, Brooks and Hughes (1988) advocate a broad multi-taxa approach that not only includes invertebrates but plants and vertebrates as well.

We recommend genus and species-level taxonomic identification of macroinvertebrates for use of taxa richness values and calculation of biological indices. Coarser-level identifications do not appear to adequately discern insect functional groups, tolerance levels, and specific sensitive genera or species.

Results from the 1989 comparisons of insect data across wetlands with different levels of watershed development suggest that urbanization affects emergent macroinvertebrate communities by (1) decreasing overall taxa richness, (2) eliminating or reducing taxa belonging to scraper and shredder functional feeding groups (leaving a dominance of collector taxa), (3) reducing EPOT taxa richness and relative abundance, and (4) eliminating or reducing specific Dipteran taxa, particularly those belonging to the Chironomidae family.

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