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THESIS APPROVAL

The abstract and thesis of Raymond S. Hoy for the Master of Science in Environmental Sciences and Resources were presented on February 2, 2001, and accepted by the thesis committee and the department.

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ABSTRACT

An abstract of the thesis of Raymond S. Hoy for the Master of Science in Environmental Sciences and Resources presented February 2, 2001.

Title: The Impact of Fine Sediment on Stream Macroinvertebrates in Urban and Rural Oregon Streams.

Urbanization, often characterized by high impervious surface area, can result in excessive inputs of fine sediments into urban streams. Excessive fine sediments can blanket the stream bed filling the interstitial space in the substratum, which may have adverse effects on stream biota. A field survey was conducted in Oregon urban and non-urban basins to investigate the relationship between fine sediments and stream macroinvertebrates. Physical, chemical, and biological data were collected from 59 stream sites in two urban and two rural streams. The stream sites fulfilled a continuous sediment gradient, which ranged from a low of 2% of fine sediment in the substrate to a high of 64% with an average of 22%. The % fines in Clear Creek (rural basin) was significantly lower than in the urban basins (Johnson Creek and Tryon Creek) ($p=0.005$). Johnson Creek (mean=23%) had approximately three times more fine sediment than Clear Creek (mean=7%), while Tryon Creek (mean=32%) had nearly five times as much fine sediment as Clear Creek. EPT taxa richness was significantly higher in both rural streams than in both urban streams ($p<0.0001$). Sediment and macroinvertebrate differences between urban and rural basins were

clear. However, quantitative relationships between fine sediments and macroinvertebrates were weak. Sediment effects on macroinvertebrates along the sediment gradient were not significant ($p > 0.05$). For example, regression analysis of EPT taxa richness vs. % fine sediments displayed a coefficient of determination (r^2) value of 0.2. Other macroinvertebrates metrics displayed similar patterns. The lack of significant correlations may be due to the cumulative effect of basin-wide “historical land use past”. Past land use activity may have resulted in long-term reductions of sensitive taxa in the basin taxa pool and efforts to improve local habitats may not be quickly colonized by pollution sensitive taxa. Long-term degradation to the urban streams resulted in a relatively homogenous assemblage of macroinvertebrates, which may have confounded the quantitative relationship between sediments and macroinvertebrates. This study suggests there is a clear difference between urban and non-urban streams in terms of macroinvertebrates, which may be likely due to sediments, but the quantitative relationship between fine sediments and macroinvertebrates is weak.

THE IMPACT OF FINE SEDIMENT ON STREAM MACROINVERTEBRATES IN
URBAN AND RURAL OREGON STREAMS

by

RAYMOND S. HOY

A thesis submitted in partial fulfillment of the
requirements for the degree of

MASTER OF SCIENCE
in
ENVIRONMENTAL SCIENCES AND RESOURCES

Portland State University
2001

Dedicated to the memory of my Uncle Johnny and my Grandmother,

John Borg & Rose Hoy

....You will always be in my memories.

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TABLE OF CONTENTS

List of Tables	v
List of Figures	vi
Introduction	1
Objectives	4
Background	5
Sediments	5
Origin and sources	5
Transport of sediments to streams	6
Sediment properties and movement in streams	8
Land use and sediments	12
Agriculture and farming	12
Forestry	12
Urban	14
Macroinvertebrates	15
Macroinvertebrates as indicators	15
Ecological importance of macroinvertebrates in streams	17
Effects of sediments on macroinvertebrates	18
Effects of suspended sediments on macroinvertebrates	21
Effects of deposited sediments on macroinvertebrates	22
Biotic metrics and indices	23
Methods	27
Study area	27
Willamette Valley Ecoregion	27
Johnson Creek Basin	30
Tryon Creek Basin	33
Clear Creek Basin	35
Deep Creek Basin	37
Sampling sites	39
Study unit	39
Water quality	41
Physical habitat	42
Sediments	44
Macroinvertebrates	44
Data analysis	45
Results	47
Environmental condition	47
% fine sediments	47

Conductivity	47
Embeddedness	53
Flow	53
Temperature	55
Nutrients	55
Total phosphorous	55
Dissolved nitrogen	55
Dissolved inorganic phosphorous	57
Biotic Condition	57
Macroinvertebrate assemblages	57
Ephemeroptera, Plecoptera, and Trichoptera	61
Diptera, Gastropoda, Oligochaeta, and Amphipoda	63
Multivariate statistics (Canonical Correspondence Analysis)	67
EPT genera	69
Ephemeroptera genera	69
Plecoptera genera	72
Trichoptera genera	72
Relationships between macroinvertebrates and sediments	73
Land use	80
Discussion	83
Basin land use and macroinvertebrate assemblages	83
Effects of fine sediment on macroinvertebrate assemblages	87
Quantitative relationships between fine sediments and macroinvertebrates	88
Other factors affecting the relationship between sediments and macroinvertebrates	91
Management implications	93
Conclusions	94
References	96

LIST OF TABLES

Table 1. Factors controlling volume of fine sediment reaching the stream channel from non-channel sources.	7
Table 2. Potential disturbance from deposited and suspended fine sediments on stream biota.	11
Table 3. Comparison of macroinvertebrate functional feeding guilds, food resource, and feeding mechanism.	19
Table 4. Definitions of macroinvertebrate metrics and predicted direction of metric response to increasing perturbation.	26
Table 5. Mean Rapid Habitat Assessment (RBP) scores per basin and the descriptions of the scoring categories.	43
Table 6. Mean, range, and SD for selected environmental variables measured at the 59 stream sites.	49
Table 7. Mean, range, and standard deviation for selected environmental variables measured in Clear and Deep Creek.	51
Table 8. Mean, range, and standard deviation for selected environmental variables measured in Johnson Creek and Tryon Creek.	52
Table 9. Macroinvertebrate taxa list of taxonomic order, family, and genus of EPT taxa found at the 59 stream sites.	58
Table 10. Taxa list of the taxonomic order and common names of non-EPT taxa found at the 59 stream sites.	59
Table 11. Distribution of macroinvertebrate orders among urban and rural basins.	60
Table 12. Relative abundance of EPT genera in urban and rural basins.	70
Table 13. Coefficient of determination of macroinvertebrate metrics versus % fine sediments.	74
Table 14. Coefficient of determinations on regressions of macroinvertebrate Metrics vs. % fine sediments, conductivity, and embeddedness.	78
Table 15. Comparison of land use and impervious surface area between rural and urban basins.	82

LIST OF FIGURES

Figure 1. Map of the study area illustrating two urban and two non-urban basins in the Willamette Valley Basin.	28
Figure 2. Map of the study area and location of sampling sites.	40
Figure 3. The variation of percent fine sediment among Oregon stream sites.	48
Figure 4. Comparison of sediments between urban and non-urban basins.	50
Figure 5. Comparison of flow and temperature in urban and non-urban basins.	54
Figure 6. Comparison of $\text{NO}^2 + \text{NO}^3$, o-phosphorous, and total phosphorous in urban and non-urban basins.	56
Figure 7. Comparison of Ephemeroptera, Plecoptera, and Trichoptera taxa richness in urban and non-urban basins.	62
Figure 8. Comparison of the relative abundance of EPT, Ephemeroptera, Plecoptera, and Trichoptera in urban and non-urban basins.	64
Figure 9. Comparison of the relative abundance of non-EPT taxa between urban and non-urban streams.	65
Figure 10. Comparison of the relative abundance of Diptera, Gastropoda, Oligochaeta, and Amphipoda in urban and non-urban basins.	66
Figure 11. Canonical Correspondence Analysis (CCA) on macroinvertebrate genera (EPT) and selected environmental variables.	68
Figure 12. Comparison of the relative abundance of EPT genera Baetis, Sweltsa, and Calineuria in urban and non-urban basins.	71
Figure 13. Regression of EPT taxa richness versus % fine sediments.	75
Figure 14. Regression of % EPT, % E, % P, and % T vs. fine sediments.	76
Figure 15. Regression of % Diptera, Gastropoda, and Oligochaeta vs sediments.	77
Figure 16. Logarithmic regression of % Baetis, % Rithrogena, % Paraleptophlebia, and % Acentrella versus fine sediments.	79
Figure 17. Regression analysis of Ephemeroptera collector, Plecoptera Predator Trichoptera scraper versus fine sediments.	81

INTRODUCTION

Excessive sedimentation has been suspected as one of the major causes of stream degradation in North America. The US Environmental Protection Agency (EPA 1990) listed sediment as the number one pollutant of US rivers, impairing >40% of the nation's river miles. Urban development such as roads, parking lots, and rooftops, creates impervious surfaces, which can increase runoff and excessive amounts of sediment into streams (Waters 1995).

Urban growth in the Willamette Valley region in Oregon has been rapid. Population in the Willamette River Basin increased from 1,963,600 in 1990 to 2,168,600 in 1995. This increase was nearly of 205,000, which was nearly twice the national average (Census 1996). The Willamette River Basin has had the greatest percentage increase for Oregon over the last 15 years, with most of the growth occurring in the State's three largest metropolitan areas, Portland, Eugene, and Salem. For example, during the years 1990-1999, the Portland metropolitan area grew from a population of 1,515,452 to 1,845,840 (Bureau 2000). This was a 21.8 % population increase over the span of 9 years. These statistics outline a common theme occurring in urban areas, "rapid growth". Rapid urbanization has its environmental consequences. For example, streams within the City of Portland are severely degraded by pollutants. These pollutants include storm water runoff from urban areas; sediments from new construction, roads, parking lots, and improper land clearing; fertilizers and pesticides from lawns; increased stream temperature from habitat destruction, and water flow changes from a variety of sources. Urban streams such as

the Columbia Slough, Johnson Creek, Tryon Creek are listed by Oregon Department of Environmental Quality for exceeding one or more water quality (303d) parameters standards, such as toxics, bacteria, nutrients, pH, dissolved oxygen, chlorophyll a, and temperature (DEQ 2001).

One major consequence of urbanization is an increase in impervious surface area (Waters 1995). Impervious surfaces are impenetrable surfaces such as concrete and asphalt. During rain events, water runs over impervious surfaces collecting sediments on its path to streams. In natural areas, approximately 33 % of the rain can infiltrate through the soil into ground water (Meross 2000). Less sediments are transported to streams when infiltration of precipitation occurs. Excessive sedimentation can blanket the stream bed, smothering salmonid eggs and aquatic macroinvertebrates (Cordone and Kelly 1960). The recent decline of salmonid and other fish populations in Oregon has caused managers to take a closer look at erosion and sedimentation issues. In the last two years, Steelhead trout and Chinook Salmon have been listed under the "Endangered Species Act (1973)" (Meross 2000).

Aquatic macroinvertebrates are good indicators of stream environmental conditions because they are sedentary in nature, species rich, and each species has its own environmental preference (Resh 1993). Therefore, they have the ability to integrate cumulative impacts over space and time. Macroinvertebrates have been used as biological indicators for many forms of pollution such as eutrophication, acidification, heavy metal contamination, and sedimentation (Resh 1993). The impact of sediments on aquatic macroinvertebrates has received substantial attention (Ryan 1991, Waters 1995, Wood and Armitage 1997). Responses from macroinvertebrates

to fine sediments can include changes in community structure, such as species richness, and biomass (Resh 1993). Studies have shown that fine sediments can cause a change from pollution sensitive forms of macroinvertebrates, such as taxa in the orders of Ephemeroptera, Plecoptera and Trichoptera (EPT), to more sediment/silt tolerant forms such as chironomids, snails, and oligochaetes (Hall et al. 1984).

Despite numerous studies on sediments and macroinvertebrates, most field studies have focused on forestry and agricultural activities while few have focused on urban basins. It has been well documented that forest and agricultural operations can cause excessive sediment delivery to streams. However, the amount of sediments in streams due to urban land use has not been thoroughly studied. Additionally, the response of macroinvertebrates to sediment pollution is unclear. It is generally recognized that macroinvertebrates are good indicators of organic pollution disturbance, but no studies have succeeded in quantifying the relationship between sediments and macroinvertebrates in urban streams.

Urban basins are unique systems. Urban land use is associated with high amounts of impervious surface area, which may have consequences of high amounts of runoff and highly variable stream discharge. Therefore, urban streams may be susceptible to excessive amounts of fine sediments. Urban growth is occurring at a rapid rate in the Willamette Valley (Census 1996). This suggests that pollution and particularly sediment pollution may become even more of a problem in urban streams in the near future. Therefore, managers need scientific information and methods to assess the sediment conditions of urban streams to regulate and mitigate sediment pollution activities. The objectives of this study are:

- 1) To examine if there are significant differences in fine sediments and macroinvertebrates between urban and non-urban basins
- 2) To quantify the relationship between fine sediments and macroinvertebrates along a fine sediment gradient

This field study was designed to examine the relationship between fine sediments and macroinvertebrates in wadeable Oregon streams. Data were collected from 59 stream sites to establish an urban/rural fine sediment-gradient. In theory, the amount of sediment at each site would correspond to the presence or absence of sediment-intolerant taxa or sediment-tolerant macroinvertebrate taxa. A priori hypothesis is that sediment in-tolerant forms of macroinvertebrates would be present at sites with low to moderate levels of fine sediment and more tolerant forms would be present at sites with higher fine sediment content. There would be a change of macroinvertebrate composition along the sediment gradient. If a relationship between the fine sediment gradient and macroinvertebrate composition could be quantified, managers could use the relationship as a base for urban stream restoration. Macroinvertebrate taxa could be used to indicate the amount of fine sediment in a stream. Managers could integrate such data into their biomonitoring and management programs.

BACKGROUND INFORMATION

Sediments

The amount of sediment produced by human activities is immense. The quantity alone recognizes sediment as the major pollutant of US waters. Berg (1982) estimated that two billion tons of sediment were being deposited in US streams annually. Many agencies agree that sediments from soil erosion are the primary pollutant of surface waters in terms of quantity involved (USEPA 1990, USSCS 1982). Crop farming accounts for approximately 40 % of the sediment produced by erosion in the United States (Meade et al. 1990), and it is estimated that 60 % of the about 8.880×10^9 tons of soils lost each year to cropland is deposited in US rivers, streams, and reservoirs. Urban development and associated construction activities can increase erosion rates up to 300 times greater than erosion from forested areas (USDA 1977). A national survey conducted by the US Fish and Wildlife Service (1982) concluded that excessive siltation was the most important factor adversely affecting the nation's fishery habitat in streams (Judy 1984).

Origin and Sources

Sedimentation to waterways occurs via the natural erosion of undisturbed soil. The process of sedimentation to streams is extremely dynamic and highly variable which reflects variations in climate, basin geology, basin scale, and sediment erosion and delivery processes (Walling and Moorehead 1989). There are two main sources of sediment to rivers and streams: channel sources and nonchannel sources (Wood and Armitage 1997). Channel sources come from the bed and banks of the stream. The

supply of sediment from channel sources is influenced by stream discharge and the stability of the channel bed and banks. The channel sinuosity and degree of channel meandering coupled with stream discharge will determine the amount of erosion that occurs (Waters 1995). Stream banks can be easily eroded to very resistant. Bedrock banks are extremely resistant to erosion and are not easily modified while alluvial banks can be very susceptible to modification and erosion (Castro and Reckendorf 1995). Nonchannel sources, such as bare soils, are principally derived from erosion of upland areas. Sheet and rill, gully, and ephemeral gully erosion from hill slopes are the primary source of naturally induced sediment to streams (Castro and Reckendorf 1995). Other natural and human induced sources of sediments come from landslides, debris flows, stream banks, irrigation, roadsides, and stormwater inputs (Waters 1995).

Transport of sediment to streams

Erosion occurs via the naturally weathering of undisturbed soil. During rain events, sediment particles get washed over surfaces down a gradient towards the waterways. The duration and intensity of the rain event are important factors determining the amount of sedimentation that will occur (Newcombe and MacDonald 1991). This process is also controlled by several other factors such as basin geology, topography, vegetation, and land use (Table 1). Slope stability in forested areas determine the erodibility of the soil (Wood and Armitage 1997). Slope angle, soil thickness, soil texture and groundwater saturation are major factors controlling slope stability (Brady and Weil 1999). The soil stability in the basin combined with the magnitude and frequency of rain events will influence the mechanism of erosion and

Table 1. Factors controlling volume of fine sediment reaching the stream channel from non-channel sources. Modified from Woods and Armitage (1997).

Factor	Issue and Comment
Geology	type of soil, erosion properties
Topography	slope of terrain and angles
Vegetation	buffer, catches sediment
Climate	amount of precipitation, intensity, frequency
Landuse	Agriculture: exposed soils
	Forestry: logging and forest roads
	Urbanization: impervious surfaces and development

amount of sediment delivered to the stream (Brady and Weil 1999). Natural vegetation provides root structure to soil, which protects it from erosion (Castro and Reckendorf 1995). Exposed soils are more subject to erosion due to direct exposure to rainfall. Soil particles become detached as sediments via precipitation and move downhill to streams. It has been documented that human activities greatly enhance the amount of sediment delivery to streams. Anthropogenic activities have led to an increase in soil erosion, a decrease in vegetation, and an increase in impervious surface area (Waters 1995). The exponential increase in urbanization in the past century, has led to an increase anthropogenic land-use area. Human land use activities such as agriculture, forestry, mining, and urban development have caused an increase in sediment delivery to streams (USEPA 1990).

Sediment properties and movement in streams

In this study, fine sediment was defined as particles less than 2 mm in size. This size range encompasses sand (<2000 to > 62 μm), silt (<62 to >4 μm), and clay (<4 μm) (Chang 1988). Fine sediments < 2 mm are recognized as the most important size fraction detrimental to stream biota because of the adsorption properties, deposition, and transport (Waters 1995)(Wood and Armitage 1997). Fine sediments in this size range can be moved by the bed load and fractions lighter than 2 mm can be carried in the suspended load (Lemly 1982). Fine sediments can be divided into two categories, deposited and suspended sediments. Biologists generally recognize these two components of the sediment load based on their effects on stream organisms (Waters 1995). Generally, deposited sediments consist of larger particles, which lie in

the sand to silt size range, while suspended sediments comprise smaller particles mostly of clay and silt. Deposited sediments can either fill interstices in the substrate or be transported along the bottom of the stream as a component of the bedload. Saltation, also known as bedload transport, is defined as the movement of particles colliding with the bed (Nino and Garcia 1998). Velocity regime and stream slope affect whether sediments are entrained or left stationary (Castro and Reckendorf 1995). Particle size as well as shape is important for sediment transport. Flat particles are more difficult to entrain than spherical ones (Pye 1994). If the stream discharge produces enough sheer stress that exceeds the point of incipient motion for fine sediments, then they will be moved along the stream bottom as a component of the bed load by either rolling or saltating along the substrate (Newbury 1984). Since sand particles are spherical while clays and silts are generally flatter, sands are generally the easiest to entrain. The shear stress needed to entrain a clay particle may be as large as the shear stress needed to entrain a large cobble, not only because of shape but sometimes because of binding properties as well (Pye 1994). Although, once a clay particle is entrained, then it stays in the water as suspended sediment. A study by Culp et al. (1986) found that particles of 0.5- to 2.0 mm have more potential to be transported. Smaller size fractions, less than 63 μm are the most important fraction for contaminant adsorption and transport due to their relatively large surface area and geochemical composition (Wood and Armitage 1997). Other factors, such as the size and type of stream substrate as well as the stream slope also influences whether sediments are deposited, transported or suspended. Different sizes and different

components (deposited or suspended) of sediment particles potentially have different types of disturbance upon stream organisms (Waters 1995) (Table 2).

Table 2. Potential disturbance from deposited and suspended fine sediments on stream biota.

Component	Type of disturbance
Deposited sediments	Fill interstitial space in substrate, removal of physical habitat heterogeneity.
	Bedload sediments saltating along the stream bottom scouring stream organisms.
	Smother stream bed, blanketing effect; smother aquatic vegetation, macroinvertebrates, salmonid and other fish eggs.
Suspended sediments	Suppended sediments in the water column; scour stream organisms; aquatic macrophytes, periphyton, macroinvertebrates, and fish.
	High turbidity; reduce light penetration, inhibiting/reducing photosynthesis; depleting food resources.
	Transport of contaminants; metals, pesticides, etc.

Land use and sediments

Agriculture and Farming

Agricultural activities are the major source of pollution in the United States. The US Environmental Protection Agency (1990) reported that agriculture, as a source of pollution in streams and rivers, generates over three times the amount of pollution contributed by the next leading source. Farming practices such as row-crop cultivation and livestock grazing in riparian zones are the principal causes of sediment pollution due to agriculture (Waters 1995). Agricultural activities commonly expose top soils leaving them susceptible to erosion. Crop farming has been estimated to cause the loss of 8880×10^6 tons of soils annually to streams (USDA 1989). Meade et al. (1990) documented that crop farming and ranching accounted for approximately 40 % of the sediment produced by erosion in the United States. Because of the vast areas associated with agriculture, the amount of exposed soils subject to erosion can be immense. Fine sediment eroded from agricultural land constitutes the most widespread impacts on biota due to sedimentation (Walling and Moorehead 1989).

Forestry

It has been well documented that forestry practices can deliver excessive amounts of sediments to streams (Castro and Reckendorf 1995). Work in this area has been intensively studied due to its impacts on fisheries resources. The deleterious impacts on streams associated with forestry activities are typically less than those in agricultural areas, but poor forestry management on steep slopes can potentially

mobilize large volumes of sediment (Wood and Armitage 1997). O'Loughlin and Pearce (1976), showed that logging operations in north Westland, New Zealand increased catchment sediment yields by up to 100-fold. Operations such as clear-cutting, skidding, yarding, site preparation for replanting, and road construction and maintenance are the primary cause of excessive sediment generation in forestry operation areas. Timber harvesting has a direct impact on sedimentation rates to streams. The removal of vegetation destabilizes soils because of slope characteristics, loss of moisture, loss of canopy cover, and loss of root strength (Castro and Reckendorf 1995). Logging roads can contribute excessive sediment, particularly if built near streams or construction sites (Waters 1995). The roads act as a channel network efficiently delivering the sediment to streams. Precipitation washes over exposed soils and logging roads depositing sediments into nearby streams. Large volumes of sediment can potentially be mobilized from forestry activities, but good management can greatly reduce these amounts. Riparian buffer zones and other forestry best management practices (BMP's) are designed to prevent sediment delivery to streams (Angradi 1999). However, implementation and enforcement of BMP's are not always efficient. Several studies have documented that forestry practices can have a large impact on stream benthos (Lemly 1982, Vuori and Joensuu 1996, Angradi 1999). The amount of damage of sediments upon organisms will largely depend on the frequency, intensity, and magnitude of the events, and the duration of the exposure (Newcombe and MacDonald 1991).

Urban

Sediment input to streams from urban basins has received the least attention. However, it has been documented that urban areas can contribute large quantities of fine sediments to streams (Wolman 1964). The most widely recognized sediment impacts from urbanization come from development or construction activities of residential areas (Waters 1995). Early survey work by Wark and Keller (1963) found that sediment loads undergoing urbanization were up to 50 times more than those in rural areas. A study by Wolman and Schick (1967) measured annual sediment yields from urban development and found up to 200 tons/acre or more entering streams. This sedimentation figure was far more than from agricultural erosion in the same area (about 5 tons/acre). Erosion of soil due to urban development and construction activities is fairly intuitive. However, impervious surfaces such as roads, building, rooftops, and parking lots potentially could deliver high amounts of sediments to streams. Stormwater runoff and runoff over eroded areas probably contribute significant amounts of sediments to streams. Like all the other landuse types, urban areas are equally complex. Factors such as slope, riparian vegetation, type and amount of impervious surface areas, and climate patterns are important variables. Urban stream studies have become increasingly important with rapid growth of urban areas in the last two decades (Bureau 2000). However, it is difficult to pin point cause and effect relationships with urban stream studies because they are so complex and sediment impacts on streams can be confounded by many other factors such as instream habitat, riparian vegetation, toxicants, organic pollutants, and heavy metals. Furthermore, impacts on biota may come from pulse disturbances or gradual

disturbances from point and nonpoint sources. Randall et al. (1978) determined that suspended sediments was the greatest single pollutant in a stream affected by an expanding urban area. Yorke and Herb (1978) found that yields of suspended sediment from urban lands were up to six times more than yields from cultivated lands and up to 120 times more than from forested and grasslands. A comparison of urban, agricultural, and forested land use by Jones and Holmes (1985) showed that forestry practices contributed little sediments, agriculture was an important source, and urban development contributed the most sediments. Faye et al. (1980) compared erosion and suspended sediment yields in nine watersheds and reported the greatest suspended sediment yields in urban areas compared with forested and agricultural areas. These studies not only suggest that sediments may be the primary disturbance to urban streams, but also show that urban impacts are an important area to concentrate future studies.

Macroinvertebrates

Macroinvertebrates as indicators

There are many advantages to using benthic macroinvertebrates for assessing stream conditions. First, they are ubiquitous; therefore they are affected by many environmental impacts in many different types of aquatic systems and habitats. Second, macroinvertebrates are extremely diverse, which allows for a spectrum of responses to many different environmental stressors. Third, they are largely sedentary; therefore they reflect the conditions at that point of sampling. Fourth, they

have long life cycles compared to other groups, which allows them to detect short-term as well as long-term disturbances. Fifth, they are easily and inexpensively collected (Resh 1993). Benthic macroinvertebrates act as continuous monitors of the water they inhabit. They integrate cumulative impacts over time and space enabling long-term analysis of both regular and intermittent discharges.

While there are many advantages to using benthic macroinvertebrates for biomonitoring, there are numerous difficulties as well. First, benthic macroinvertebrates apparently do not respond to all types of environmental stresses (Resh 1993). For example, Hawkes (1979) reported only slight effects of low concentrations of a herbicide on the invertebrate fauna in a river, even though detrimental effects were displayed by angiosperms downstream of the effluent. Second, quantitative sampling using macroinvertebrates can be difficult, time consuming, and costly. Since macroinvertebrates are highly patchy in their distribution (Allan 1995), quantitative studies requires high numbers of samples to achieve desirable precision in estimating population abundance. However, rapid bioassessment techniques and modified sampling designs can reduce some of the disadvantages. Third, benthic macroinvertebrate distribution and abundance can be affected by factors other than water quality (e.g. natural conditions such as substrate, velocity, microhabitat distribution). For example, macroinvertebrates have differential preferences to substrate particle sizes (Williams 1978). Generally, benthos abundance is lowest in homogenous sand, silt, large boulders, or bedrock, while it is highest in mixtures of gravels, pebbles and cobbles (Minshall 1984). Another factor affecting macroinvertebrate distribution and abundance are patch dynamics. Heterogeneous

conditions (substrate) within a small area of the stream bottom can result in large differences in macroinvertebrate abundance and diversity. Macroinvertebrates can range from very abundant and/or highly diverse in one area, to completely sparse in an area directly next it. This patchy distribution is not only due to heterogeneous substrate, but also due to differences in current velocity and resource dispersal (i.e. detritus and periphyton) (Reice 1974). Fourth, the seasonal variations in abundance of benthic macroinvertebrates requires detailed knowledge of ecology and life histories. Macroinvertebrates have complex and diverse life-history patterns. Different groups of macroinvertebrates hatch at different times of the year and the duration of life cycles and life stages vary with each group (Merritt and Cummins 1996). Fifth, drifting behavior of benthic macroinvertebrates can confound disturbances caused by localized effects. Even though macroinvertebrates are largely sedentary organisms, they participate in a behavioral phenomenon called drift. This downstream movement can completely change macroinvertebrate densities in a given area which could confound site-specific studies (Resh 1993).

Ecological importance of macroinvertebrates in stream ecosystems

Freshwater benthic macroinvertebrates are an important component of the stream ecosystem. They are an integral link in the stream food chain (Allan 1995, Merritt and Cummins 1996). Benthic macroinvertebrates are food for larger instream organisms such as salmonid, trout, and other fish species. Taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera are favorites among cold-water fishes (Waters 1995). Another important function of invertebrates is their processing of

riparian inputs (detritus) and instream vegetation (i.e. algae, vascular aquatic plants) (Allan 1995). This processing of materials by aquatic macroinvertebrates is a vital step for the recycling of materials for the lotic ecosystem. Furthermore, benthic macroinvertebrates aid in the decomposition process by breaking down large organic materials, which are then taken up by microbes and bacteria. This process, known as the “microbial loop”, is important for the remineralizing of organic matter (Allan 1995). Benthic macroinvertebrates can be organized into different functional feeding groups based on their mechanism of feeding and by the type of resource they consume (Table 3). A functional feeding guild is a group of taxa, which share the same or similar feeding behavior and feed on same or similar resources. Benthic macroinvertebrates are grouped into feeding groups of shredders, collectors, scrapers, and predators. Each guild serves an important function in the breakdown of instream detritus and periphyton, and nutrient recycling of the lotic ecosystem (Merritt and Cummins 1996). Overall, the role of aquatic macroinvertebrates in the stream ecosystem is vital to stream structure and function.

Effects of sediments on macroinvertebrates

Sediment disturbance on benthic macroinvertebrates has received much attention (Cordone and Kelly 1960, Newcombe and MacDonald 1991, Waters 1995, Wood and Armitage 1997). White and Pickett (1985) define disturbance as “any relatively discrete event in time that disrupts ecosystem, community, or population structure, and changes resource, substrate availability or the physical environment”. Resh et al. (1988) added: since species may develop adaptations to predictable

Table 3. Comparison of macroinvertebrate functional feeding guilds, food resource, and feeding mechanism

Feeding Group	Food Resource	Feeding Mechanism
Shredders	leaf litter, and other riparian organic debris	shred, and chew
Collectors		
Filtering collectors	leaf litter, and other riparian organic debris	use nets to catch debris
Gathering collectors	leaf litter, and other riparian organic debris	physically gather debris using appendages
Scrapers	periphyton/attached algae	scrape substrate using mouth parts
Predators	other insects	hunt and stalk prey

environmental fluctuations, a disturbance may be outside the range that organisms are likely to encounter. Disturbance on aquatic invertebrates from fine sediments can range anywhere from: 1) Loss of interstitial space and shelter (habitat) (Percival and Whitehead 1929); 2) Increased water turbidity which can affect photosynthesis and therefore reduce food availability (Graham 1990, Surber 1953); 3) Oxygen depletion in the interstitial areas and change in type of substratum (Edwards 1969, Jones 1949, Nuttall and Beilby 1973); 4) Loss of holdfast capability due to fine sediment coatings on substrate surfaces (Nuttall and Beilby 1973); 5) Physical removal of organisms by scouring (Culp et al. 1986).

Several factors can affect disturbance such as the scale of the disturbance. First, The ecological level of impact can occur from the ecosystem, community, population, down to the individual level. Second, the physical scale must be determined. The spatial scale of disturbance can range from the basin, to the stream reach, to the microhabitat scale (Poff 1997). Third, the timing of the disturbance relative to the life cycle of the organism is hard to determine, since benthic invertebrates have differential sensitivities at different stages in their life cycle. Fourth, the temporal scale of the question must be established. For an ecological time scale, disturbance magnitude and duration must be determined. At an evolutionary time scale, magnitude, duration, frequency, deviation from normal, and predictability are important (Runde 1999).

Disturbance on macroinvertebrates is a natural event occurring at many different spatial and temporal scales. Reice et al. (1990) reviewed the concepts of disturbance regimes, resilience, and recovery of stream organisms. They suggest that

disturbance in streams such as drought, floods, storms, and landslides, are a critical organizing feature in stream community structure and species diversity. These natural and sometimes catastrophic events, keep the community structure in equilibrium. The intermediate disturbance hypothesis suggests that in the absence of disturbance, the superior competitors could eliminate the inferior competitors, reducing the species richness of the system (Connell 1978). However, anthropogenic activities can upset the natural occurrence of disturbance. Anthropogenic activities such as channelization of waterways, and excessive sedimentation can increase the frequency and magnitude of disturbance. Human-induced disturbances can alter natural stream disturbance regimes and keep stream communities in perpetual disequilibrium (Reice et al. 1990).

It is useful to separate disturbance of fine sediments into two components, effects from suspended sediments and deposited sediments (Table 2). Each component can have differential effects on macroinvertebrates at different scales.

Effects of suspended sediments on macroinvertebrates

Newcombe and McDonald (1991) reviewed the effects from suspended sediments and suggested aquatic macroinvertebrates can be disturbed by suspended sediment in several ways. First, suspended sediments contribute to turbidity which may adversely affect algal growth via a reduction in light penetration. Instream periphyton is a primary food source for many macroinvertebrates (Allan 1995). Second, suspended sediments can clog filter feeding nets or structures which can reduce feeding efficiency leading to reduced growth rates, stress, or kill these organisms (Hynes 1970). Third, scouring by suspended sediments can damage

exposed respiratory organs or make the macroinvertebrates more vulnerable to predation through dislodgement (Langer 1980). The severity of the effects is related to the concentration of the substance and to the duration of the exposure (Newcombe and MacDonald 1991). Taken together, the frequency of the sediment episode, ambient water quality, and the species plus life history stage affected will affect the magnitude of the disturbance.

Effects of deposited sediments on macroinvertebrates

Waters (1995) summarized the effects of deposited sediment on benthic invertebrates. Deposited sediments disturb stream biota by smothering the stream bed (Wood and Armitage 1997). The major concerns with deposited sediments are a change in substrate particle size, and amount of cobble embeddedness, which leads to the removal of interstitial habitat space. It is generally recognized that the diversity and abundance of stream macroinvertebrates are positively correlated with particle size and substrate heterogeneity (Waters 1995). Benthic invertebrates are least abundant in homogenous environments such as sand and silt substrate or bedrock substrate. Therefore, the greatest abundance remains in heterogeneous environments with cobbles, gravel, and pebbles (Minshall 1984). Substrate embeddedness is defined as the amount of fine sediment filling the interspaces surrounding the substrate (Burton and Harvey 1990). When fine sediments accumulate on the stream bed, they clog the interstices between substrate clasts, which reduce the available habitat for benthic organisms (Wood and Armitage 1997). Benthic insects within the group of EPT generally inhabit the surface of cobbles and the interstitial space between and

beneath large substrate particles such as stones and pebbles (Waters 1995). A study by Culp et al. (1986) suggests benthic invertebrates are disturbed by bed load movement or saltation. They explored the difference between the effect of sediment from (1) riffles with low “tractive force” where fine sediments were deposited and (2) riffles with high tractive force where particles saltated along the bed. They found that deposited sediments had little effect, while saltating sediments scoured invertebrates and caused high catastrophic drift, with a decrease in more than 50 % in benthic invertebrate density. This study suggests that sediment transport or saltation may have the most detrimental effects on benthic macroinvertebrates.

Biotic metrics and indices

The biotic response of macroinvertebrates to environmental disturbance is complex. The responses range from a complete shift in community structure to a change in the behavior of an individual taxa. Responses to environmental stress can be organized into the following categories: 1) biochemical and physiological, 2) behavioral and life history, 3) presence/absence or numerical predominance of species, and 4) changes in community composition (Resh 1993). Biochemical responses to environmental stress can include changes in the properties such as energy metabolism, enzyme activities, amino acid, protein content, and ion regulation, which occur at the level of the organism. Physiological responses include measurements of changes in activities such as heartbeat and respiration rate. The use of biochemical and physiological responses for benthic macroinvertebrates is presently limited because of the lack of basic knowledge of these processes in most organisms (Resh

1993). Changes in behavior are often used for sediment/toxicity studies because of the close affiliation with macroinvertebrates and sediments. Impairment can also be associated with pollutants attached to sediment particles and often are manifested as abnormal changes in behaviors such as feeding, movement, swimming, crawling, avoidance (drift), net spinning, and hunting.

Life history responses are defined by factors that govern the survival and subsequent reproduction of a species or population. They generally fall into three categories: 1) survival, 2) growth, and 3) reproduction. Environmental stress can affect the sequence of these morphological stages and physiological processes. The presence or absence of a particular species, taxa, population, and/or community suggests the environmental conditions of that area. The structure of benthic communities reflects the state of the entire ecosystem (Resh 1993). Community structure measures have been commonly used as a biotic response to environmental stress. Eaton and Lenat (1991) used benthic macroinvertebrate taxa richness as an indicator of water quality in North Carolina streams. Community structure analysis has been used as a practical alternative to whole system biological surveys, which are difficult and expensive to complete. The biotic response of the organism will depend on the type, frequency, and magnitude of disturbance.

Due to complex responses of macroinvertebrates to environmental degradation, biotic indices and metrics are needed by resource manager for monitoring ecosystem conditions (Resh 1993). A biotic indice or metric is a measure used to calculate impact by separating organisms into "species groups". Macroinvertebrate metrics are characterized by showing a predicted response to increasing environmental

perturbation (EPA, 1999). These metrics have continually evolved over the years to encompass many aspects of environmental disturbance. Several metrics are available to address different types of environmental conditions. For example, taxa richness generally decreases with decreasing water quality, while number of individuals and biomass may increase or decrease depending upon the type of pollution and organisms involved (Resh 1993). Biotic indices have the ability to assess certain forms of pollution. However, some metrics and indices are robust only for particular pollution disturbances. For example, some macroinvertebrate taxa may be sensitive to fine sediments while others are not. Sediment tolerance indices have been developed by Wisseman (WQMT 1999), but these lists are far from complete and give either the general designations of sediment “intolerant” or sediment “tolerant” with no range of sensitivities. Therefore, more research in this area are needed to aid managers in detecting pollution disturbances.

The taxonomic orders, Ephemeroptera (E), Plecoptera (P), and Trichoptera (T) are in general three “sensitive” orders to pollution disturbance (Merritt and Cummins 1996). The summation of these three order is the EPT metrics, which can indicate coarse-level stream environmental conditions (Table 4). Since the orders E, P, and T are considered “sensitive taxa”, one would expect the number or percentages of these taxa to decrease as environmental degradation increases. On the other hand, some macroinvertebrates are so-called “pollution tolerant” taxa. Taxa such as Diptera, Gastropoda, Amphipoda, and Oligochaeta are generally considered pollution tolerant macroinvertebrates. Therefore, the number or percent of these orders is an indicator of the increasing environmental perturbation. Feeding measures, such as % grazer or %

Table 4. Definitions of macroinvertebrate metrics and predicted response to increasing perturbation (modified from EPA 1990).

Category	Measure	Definition	Predicted response to increasing perturbation
Richness measures	No. of EPT taxa	Number of taxa in the insect orders Ephemeroptera (E), Plecoptera (P), and Trichoptera (T)	decrease
	No. of Ephemeroptera taxa	Number of Ephemeroptera taxa (usually genus of species level)	decrease
	No. of Plecoptera taxa	Number of Plecoptera taxa (usually genus of species level)	decrease
	No. of Trichoptera taxa	Number of Trichoptera taxa (usually genus of species level)	decrease
Composition measures	% EPT	Percent of the composite of Ephemeroptera (E), Plecoptera (P), Trichoptera (T)	decrease
	% E	Percent of Ephemeroptera nymphs	decrease
	% P	Percent of Plecoptera nymphs	decrease
	% T	Percent of Trichoptera nymphs	decrease
	% Diptera	Percent of Diptera considered to be tolerant to various types of perturbation	increase
	% Oligochaeta	Percent of Oligochaeta considered to be tolerant to various types of perturbation	increase
	% Gastropoda	Percent of Gastropoda considered to be tolerant to various types of perturbation	increase
	% Amphipoda	Percent of Amphipoda considered to be tolerant to various types of perturbation	increase
	% collectors	Percent of macrobenthos that collect CPOM as a food resource	variable
	% scrapers	Percent of macrobenthos that graze on periphyton as a food resource	variable
Feeding measures	% scraper/collector	Percent of macrobenthos that graze on periphyton and collect CPOM as a resource	variable
	% shredder	Percent of macrobenthos that shred leaf and woody debris as a food resource	variable
	% predator	Percent of macrobenthos that prey on other macroinvertebrates as a food resource	variable
	% filtering collector	Percent of macrobenthos that filter FPOM from the water column or sediment	variable

filterer, reflect the sensitivity to available food resources (Resh 1993). Dominance of particular groups of feeding guilds may indicate organic enrichment. Therefore, the use of feeding guild measures may require knowledge of the natural food resources of that region.

METHODS

Study Area

Willamette Valley Ecoregion

The field study took place in four basins in northwest Oregon within and surrounding the Portland Metropolitan area. The Johnson Creek and Tryon Creek basins were chosen for their urban setting. Johnson Creek and Tryon Creek were designated “urban streams” since both streams lie in the Portland Metropolitan area (Figure 1). Two nearby rural streams are Clear Creek and Deep Creek of the Clackamas River Basin. The Clackamas Basin is located in the north east of the Willamette River Basin. The Clackamas Basin drains the foothills of Mt. Hood which is southeast of the Portland Metropolitan area. The Clackamas River flows into the Willamette River. The Clear Creek and Deep Creek Basins were chosen for the rural locality and close proximity to the urban basins. These two basins lie at moderate elevations and contain rural to natural land use along their streams. Johnson Creek, Tryon Creek, Clear Creek, and Deep Creek were selected because they lie in the same ecoregion (Willamette Valley Ecoregion). Ecoregions are spatially homogenous areas based on physical and landscape features, which include physiography, climate, soils,

Figure 1. Map of the study area illustrating two urban (Johnson Creek and Tryon Creek) and two non-urban basins (Clear Creek and Deep Creek) in the Willamette Valley Basin (dots represent sampling sites) Modified from Metro 1999.

landuse, natural vegetation, and biotic communities (USGS 1999). These four study streams are small, wadeable streams of similar size, slope, geology, hydrology, climate patterns and vegetation.

The Willamette Valley Basin consists of moderately temperate climate which consists of cool, wet winters and warm, dry summers (USGS 1998). About 70-80 percent of the annual precipitation falls from October through March, but less than 5 percent falls in July and August. During 1961-90, the annual precipitation average in the Willamette Basin was 157 cm. Although, the precipitation is strongly influenced by topography from 102-127 cm in the valley to as much as 508 cm near the crests of the Cascades and the Coast ranges. The amount of stream flow in the Willamette Basin strongly reflects the distribution of precipitation. Most of the runoff (60-85 %) typically occurs from October to March, and less than 10 % occurs during July and August (USGS 1999). Hydrogeology in the Willamette Basin consists of six principle hydrogeologic units: marine volcanic and sedimentary rocks of the Coast Range; alluvial deposits; Columbia River Basalt Group; volcanic rocks of the Western Cascades; volcanic rocks of the High Cascades; and landslide and debris-flow deposits (Walker and McLeod 1991). The alluvial deposits, composed of a heterogeneous mixture of clay, sand, silt, and gravel, represent the most important aquifer in the Willamette Basin with respect to water use (USGS 1999).

Landuse in the Willamette Valley Ecoregion is dominated by agricultural land which overlies the alluvial aquifer. Agricultural land dominates the Willamette basin at 70 %, followed by forest and urban land (22 and 5 %, respectively) (USGS 1999).

Information obtained from the Oregon Department of Environmental Quality (DEQ) suggests that the Willamette River is severely impaired by pollution and is on the Clean Water Act's 303d list due to poor water quality. Preliminary findings from the DEQ suggest that the lower reaches of the Willamette Basin are severely degraded. About 70 to 80 % of pollution enters the river via nonpoint sources, particularly from agricultural activity (Meross 2000). About 90 % of the nonpoint source discharge occurs during the rainy season. Spring Chinook and Bull Trout populations in some parts of the basin have declined to the point where federal listings are quite possible (Quality 2000). Evidence suggests that the biological health of the basin is severely impaired and current studies are underway to determine the causes.

Johnson Creek Basin

The Johnson Creek Basin is a tributary to the Willamette River. The headwaters are located in the Cascade foothills to its confluence with the Willamette River. Johnson Creek is approximately 40 km long and flows west from the foothills of Mt. Hood, through the cities of Gresham, Portland, and Milwaukie. The Johnson Creek Basin is characterized by urban residential, industrial and agricultural land use (Committee 1995). Land use in the Johnson Creek basin is 45 % single family residential, 9 % multi-family residential, 33 % rural, 5 % parks and open space, 4 % industrial, and 4 % commercial (Services 1999). The headwaters begin in a small, unincorporated area named Cotrell. This area is used primarily for agricultural activities, many of which are tree farms that line the creek corridor. As the stream continues westward, about 48 stream kilometers pass through the city of Gresham,

Oregon. In east Gresham the land use begins to shift from primarily agriculture to urban residential and industrial. As the creek enters city center, the impervious surface area drastically increases with increasing roads, buildings, and rooftops. The creek continues on through southeast Portland and Milwaukie where the land use begins to shift to a mix of urban area and park green spaces. Many city parks line the Johnson Creek corridor. These include Johnson Creek Park, Tideman Johnson Nature Park, Beggars-tick Wildlife Refuge, Leach Botanical Garden, Powell Butte Nature Park, Gresham Main City Park, plus a recreational trail approximately 27 km long called Springwater Corridor. The park areas provide the creek with riparian corridor and large green space areas. Urban residential homes, roadways, and industrial facilities constitute impervious surface area where park areas are not present. After Johnson Creek passes through Milwaukie, it empties into the Willamette River after approximately 200 stream kilometers from its origin.

Four other tributaries of Johnson Creek were also included in the sampling. They include Kelly Creek, Mitchell Creek, Butler Creek, and Crystal Springs.

A population analysis by Metro (1999) estimates approximately 164,115 people live in the Johnson Creek Basin (Meross 2000). Johnson Creek is currently experiencing a variety of anthropogenic impacts. The once natural, meandering stream with diverse wildlife is now a narrow, channelized polluted waterway which is characterized by frequent winter flooding (Committee 1995). The water quality in Johnson Creek is affected by urban, agricultural, and residential pollution. DEQ designated the entire mainstem of Johnson Creek as water quality limited (303d list) for parameters of stream temperature, toxins, and bacteria (Meross 2000). Pollution

from discharges of sewage and industrial wastes, and stormwater inputs are just a few of the problems affecting the creek. The primary concern of urban areas is the pollutants which enter Johnson Creek through stormwater inputs. Stormwater is not captured and processed by vegetation and soil, therefore urban pollutants can directly enter the creek via pipes or overland flow. A significant portion (approximately 33 %) of the upper Johnson Creek is utilized for agriculture (Meross 2000). Nursery, greenhouse, and Christmas tree growing operations can have significant impacts of water quality. A water quality study conducted by the US Geological Survey (USGS) in 1988 found that maximum concentrations of copper, chromium, lead, and mercury concentrations found in the urban part of Johnson Creek were 2 to 10 times higher than was found historically in the Willamette River. Dichlorodiphenyltrichloroethane (DDT) and polychlorinated biphenyl (PCB) were detected in predominantly agricultural areas in Johnson Creek (Edwards and Curtiss 1993).

Johnson Creek once supported runs of steelhead trout, coho, and Chinook salmon. Today, only a few salmon and steelhead return to the creek (Committee 1995). In the summer months (June, July, and August), the maximum average weekly water temperature exceeded 18 degrees celsius in all section of the creek below Gresham with a maximum temperature of 24 degrees celsius. The Environmental Protection Agency (EPA) estimates that average weekly summer water temperatures conducive to growth of Coho salmon is 18 degrees Celsius while the maximum temperatures that Coho salmon can survive in for short periods of time is 24 degrees celsius. The dissolved oxygen levels in the creek had levels less than 6.5 mg/L which was the minimum level for cold-water fish species according to the EPA. Nitrogen

and phosphorous concentrations in Johnson Creek were sufficient to cause excessive algal growth even though algal growth does not appear to be a major problem in Johnson Creek. Other factors affecting the creek include flooding, lack of riparian vegetation, and lack of instream habitat for stream biota.

Annual average rainfall in the Johnson Creek Basin receives approximately 135 cm per year. About 85 percent of the total annual rainfall occurs during the October through the end of May (Committee 1995). Most of the precipitation in the Johnson Creek basin is directly transferred to runoff. Because of the high amount of impervious surface area (65 %) in the basin, water flows directly to the streams instead of being detained by the soil and percolating through the ground (Committee 1995). About 65 % of the basin either drains directly, or is piped into Johnson Creek (Meross 2000). Therefore, flash foods are a frequent occurrence in the Johnson Creek Basin.

Tryon Creek Basin

The Tryon Creek Basin flows southeasterly toward its confluence with the Willamette River. This 11 km creek originates in Portland's west hills. The Tryon Creek Basin is characterized primarily by urban residential land use and by park green space area. The headwaters begin at Portland's Mt. Sylvania, which is heavily occupied by residential homes. Tryon Creek flows through this residential area for about a kilometer before it enters Tryon Creek State Park, a 640-acre Oregon State Park shared by the cities of Portland and Lake Oswego. Most of the Tryon Creek Basin is heavily wooded and residential. The park provides the creek with lush riparian vegetation as well as dense shade and canopy cover. The majority of the

creek (approximately 10 km) flows through the park until it enters the city of Lake Oswego. In Lake Oswego, the creek flows through residential areas for about 2 km until its confluence with the Willamette River.

Tryon Creek has several tributaries in the upper reaches of the stream. They include Arnold Creek, Falling Creek, Oak Creek, Park Creek, and Nettle Creek. This creek is primarily an open stream system, with the exceptions being culverts at road crossings.

Tryon Creek's physical characteristics are somewhat different from the three other basins. The gradient of the basin is probably the most noticeable difference among the other basins. Tryon Creek contained steep slopes in the park area, as high as up to 40 % hillslope. The hillslope gradient near the City of Lake Oswego was more moderate around 20 % (Metro 2000). Two other noticeable differences of Tryon Creek was stream size and substrate composition. Tryon Creek was the smallest stream of all the basins and correspondingly had the lowest flow at $0.02 \text{ m}^3/\text{s}$. The substrate in Tryon Creek contained much finer, smaller particle sizes compared to the other three basins. The primary substrate was a heterogeneous mix of small gravels to sand and silt. Tryon Creek contained larger substrates (cobbles and boulders) in the lower reaches of stream near the confluence with the Willamette River. The geology of this basin is Columbia River basalt. Soils are primarily poorly drained, dark-brown silt loam. Heavy rainfall, snowfall, and dense vegetation blanket the slopes with decaying organic material (humus).

Pollution in Tryon Creek is primarily caused by urban stormwater run-off due to increasing levels of impervious surfaces. Stormwater run-off volume has been

increasing over the last decade, and its toxicity is intensifying. Watershed run-off contains a variety of pollutants, including septic drain field effluent, pesticides, herbicides, lawn fertilizers, solid waste leachate, and the oils that collect on parking lots and roadways. These pollutants ultimately end up in the streams of the watershed (Metro 2000). Other pollution problems are that sewer lines which may be on unstable lands, often rupture and residential septic tank systems often are also failing adding pollutants to Tryon Creek. Tryon Creek has large seasonal fluctuations in water volume. During winter months, Tryon Creek carries large amounts of stormwater run-off due to the poor infiltration of the basin's clay soils and its impervious surfaces. The rushing waters of the creek carry light, silty soil. The channel can become modified as gravel, brush and silty soils form natural dams, which can back up the water, and divert the stream from its natural channel. During the summer months, the Creek primarily carries water from underground aquifers that surface through springs and seeps. Still, even during the summer, a portion of the stream's water volume is run-off from summer thundershowers (Metro 2000).

Clear Creek Basin

The Clear Creek Basin is located southeast of the Clackamas River and flows in a northwesterly direction. The creek is approximately 42 km long beginning at the foothills of Mt. Hood and meandering through forested, agricultural, and rural residential areas before its confluence with the Clackamas River at the town of Carver, Oregon. The headwaters of Clear Creek begin just southeast of the town of Dodge. Two smaller tributaries named Little Clear Creek and Little Cedar Creek feed into

Clear Creek near Metzler County Park. The headwater areas are largely forested to rural residential landuse. After the Metzler Park, the landuse along the Clear Creek corridor begins to shift to a mix of rural residential and agricultural activities, primarily tree farms. The creek receives a mixture of conditions along its corridor from shaded riparian cover to grassy open areas along its banks. The creek meanders through this rural residential/agricultural area until it reaches the confluence with the Clackamas River at Carver State Park.

The geology of Clear Creek is primarily Columbia River Basalts, sedimentary rock and glacial deposits (Walker and McLeod 1991). The slope of the Clear Creek basin is gentle with a range of 0-25 %. The Clear Creek Basin has an elevation profile from approximately 90 to 1,200 m. A study by the Clackamas County Assessor (1995) estimated that current impervious surface area in the Clear Creek Basin was approximately 11 %. They also estimated that future impervious surface area would increase to 13 % based on information from the US Forest Service and the Bureau of Land Management. Metro (1990) reported that upper portion of the Clear Creek Basin has a population density of 28 – 260 persons per square kilometer, while the lower portion has a population density of 266 – 2600 persons per square kilometer. Estimates of the landuse in the upper portion of the Clear Creek Basin is approximately 85 % forest, 10 % agriculture, and 5 % rural tract land. The lower portion of the basin is approximately 45 % forest, 40 % agriculture, and 15 % rural tract land and rural residential.

In this study, Clear Creek is considered a non-urban stream. In the upper portion of basin, conditions appear to be more natural. The headwaters are dominated

by forested areas, and there is less impact from anthropogenic land use. However, the lower portion of Clear Creek has more agriculture and rural activities, therefore it is designated as a “rural basin”. There are five mining sites (2 stone and 3 sand and gravel) in the lower portion of the creek along with a major pollutant discharge facility near the mouth (CCA 1995). Clear Creek is not on the 303 (d) list for water quality, however it is considered “of potential concern” by the Oregon DEQ because it does not meet the dissolved oxygen standards for water quality. Furthermore, there is insufficient data on this stream to designate it on the 303 (d) list for water quality (DEQ 1996). According to a study by the Oregon Department of Fish and Wildlife (1996), Clear Creek supports population of Coho salmon, Fall Chinook, and Winter Steelhead.

Deep Creek Basin

The Deep Creek basin flows from east to west and is located just north of the Clackamas River. This basin is approximately 24 km long and its headwaters begin just south of the town of Sandy, Oregon. Deep Creek is characterized by urban and rural residential land use with some moderate agricultural activity. The headwaters are surrounded primarily by urban and rural residential land use. Although there is much residential activity in the Deep Creek Basin, there are also moderately sized green space and riparian areas. Some agricultural activities also persist in this watershed, mostly with tree farms and nurseries. A moderate sized tributary, Tickle Creek, joins with Deep Creek just before it empties into the Clackamas River near highway 224.

Deep Creek has several similar characteristics of Clear Creek, but some differences as well. The geology of Deep Creek is primarily sedimentary rock with glacial deposits along the stream corridor (Walker and McLeod 1991). Like Clear Creek, the slope of the Deep Creek basin is gentle with a range of 0-25 %. The Deep Creek Basin has an elevation profile from approximately 150 to 500 meters. The Clackamas County Assessor (1995) estimated that current impervious surface area in the Deep Creek watershed was approximately 20 % with a future approximation of 24 %, which was much higher than Clear Creek. Estimates of the landuse throughout Deep Creek is similar to the lower portion of the Clear Creek Basin but with more residential landuse (266 - 2600 persons per km²). Land is distributed with approximately 35 % forest, 45 % agriculture, and 20 % which is used by rural tract land and residential.

Like Clear Creek, Deep Creek is also designated a “rural basin” based on its high intensity of rural/agricultural landuse. However, Deep Creek has some major differences from Clear Creek. Deep Creek does not have the heavily forested areas in the upper portion of basin. Its landuse is fairly consistent throughout the basin. A large difference to note would be the increase in residential homes in the basin. The north side of Deep Creek is heavily occupied by residential and rural landuse. Eight sand and gravel mining sites are found in the basin with 7 of them located near sampling sites. At sampling site Deep Creek 004 , highway 224 directly crosses over the stream segment. At this location of the stream, sediments from construction caused high turbidities in the stream. The Oregon DEQ did not list Deep Creek for any designation of water quality. There was probably insufficient data, to put it in any

category. The Oregon Department of Fish and Wildlife (1996), observed that Deep Creek supports populations of Coho salmon, Fall Chinook, and Winter Steelhead.

Sampling sites

Fifty-nine stream sites were sampled during the period of August through September 1999. Ray Hoy, Chris Walker, Julie Berry and Kristi DeWerff formed a four member crew, which participated in the sampling activities. Basins were sampled from the headwaters to the mouth where possible to achieve longitudinal coverage of the entire stream length. Twenty-nine stream sites were sampled in Johnson Creek, the larger urban basin in this study (Figure 2). Ten stream sites were sampled in Tryon Creek, the smaller urban basin. Twelve stream sites were sampled in Clear Creek and eight stream sites were sampled in Deep Creek (Figure 2). Sites were selected based on accessibility, spatial distribution of sites, and length of stream basin.

Study unit

In this study, methods followed a modified version of the Environmental Monitoring and Assessment Program (EMAP) (EPA 1998). Data were collected at the stream-reach scale along the length of the watersheds. Rabeni and Minshall (1999) evaluated the adequacy of sampling the stream reach for invertebrate monitoring programs. They found that a single well-chosen stream reach if adequately sampled, could be representative of an entire stream segment. The sampling reach was chosen based on the length of the riffle area. The riffle area was designated the sampling

Figure 2. Map of the study area: A. Johnson Creek Basin, B. Tryon Creek Basin, C. Clear Creek Basin, and D. Deep Creek Basin and location of sampling sites (indicated by dots). Map sizes are not to scale. Modified from Metro 1999.

habitat type. In this study, riffle habitat was defined as areas with coarse substrate and turbulent water (Overton et al. 1997). At each site, five transects were established perpendicular to the stream length using intervals based on the size of the riffle area. Equally spaced transects were used to achieve systematic coverage of the sampling reach, on average transects were approximately 6 m apart. When the sampling reach was approximately less than 6 m, then three transects were used to characterize the sampling reach. Physical and biological data were collected along each transect.

Water Quality

Water quality measurements were collected following modified procedures of the EMAP protocol (EPA 1998). Dissolved oxygen, temperature, conductivity, and salinity were measured using a YSI Model 85 digital meter. The YSI Model 85 probe was placed in the riffle area of the thalweg portion of the reach and measurements were recorded as one measurement per stream site. The pH was measured in the thalweg portion of the riffle area using the Orion model 210a pH meter. Two water samples were collected in the thalweg portion of the reach for nutrient analysis using acid-washed 250 ml bottles. One water sample was filtered using the Nalgene hand pump filtration unit and filtered with Millipore type HA filters (0.45 μ m pore size). The filtered water sample was used for analysis of dissolved nutrients such as ammonia-nitrogen; nitrate-nitrogen; nitrite-nitrogen; O-phosphate. The other water sample was unfiltered for analysis of (TKN) total phosphorous.

In the laboratory, water samples were analyzed for nitrate and nitrite using ion chromatography and colorimetric methods, respectively (EPA methods 300.0 and

353.2) (EPA 1993). These two constituents were added together for this study as $\text{NO}_3 + \text{NO}_2$. Ammonium nitrogen (NH_4^+) concentrations were determined using the phenol-hypochlorite method (EPA method 350.1). Orthophosphate (o-P) concentrations were determined via the colorimetric method (EPA method 365.1) (EPA 1993). Total phosphorus (TP) concentrations were determined using persulfate digestion and colorimetric method (EPA method 365.1) (EPA 1993).

Physical Habitat

The physical habitat was characterized following the procedures of the EPA's EMAP protocol (EPA 1998). The physical habitat was characterized via channel dimensions, channel gradient, substrate size, channel substrate size and type, habitat complexity and cover, riparian vegetation cover and structure, anthropogenic alterations, and channel riparian interaction (EPA 1998). The thalweg profile and channel dimensions were measured along each transect at the study reach. Stream velocity was measured using a standard velocity meter. Measurements were collected along transects at equally spaced intervals of the stream reach. Flow was calculated using the velocity data and the cross-sectional area. Reach slope was measured using a Suunto clinometer. Vegetative canopy cover was calculated using a densiometer. The rapid habitat assessments followed the procedures of the EPA's "rapid" bioassessment protocols (RBP) (Plafkin et al. 1989). Based on a visual assessment of the habitat, the field crew recorded observations of stream characteristics following the "rapid assessment description of habitat parameters" (Table 5). There are four categories in the Rapid Habitat Assessment. A score of 0- 5 represents poor habitat

Table 5. Mean Rapid Habitat Assessment (RBP) scores per basin and the descriptions of the scoring categories (EPA 1998).

Habitat parameter:	Basin				
	Clear Creek	Deep Creek	Johnson Creek	Tryon Creek	All Basins
Epifaunal Substrate	17	14	8	10	13
Embeddedness	18	14	10	9	13
Sediment Deposition	16	12	9	8	11
Channel Flow Status	15	12	11	8	12
Channel Alteration	17	15	10	12	14
Bank Stability (LB)	9	7	6	5	7
Bank Stability (RB)	9	8	9	5	7
Vegetative Protection (LB)	8	7	5	5	6
Vegetative Protection (RB)	8	7	4	5	6
Riparian Zone Width (LB)	9	7	4	6	6
Riparian Zone Width (RB)	7	7	3	7	6
Total RBP score	132	111	76	81	100

Categories:	Range		
	Optimal	Sub-Optimal	Poor
	20 -16	15 -11	10 -6
Total RBP scores	220 -176	165 -121	110 -66
			55 -0

parameter conditions, 6 –10 represents marginal conditions, 11-15 represents sub-optimal conditions, and 15-20 represents optimal conditions.

Sediments

Three different methods were used to collect the sediment data. Sediment were measured using the “percent fines grid method” (Torquemada and Platts 1989), depth-integrated sampler, and EPA’s RBP cobble embeddedness measure (Klamt 1976). Estimation of surface fine sediment was collected using the “percent fines grid method”(EPA 1993), which is designed to measure the deposited fine sediment on the surface of the substrate. The sediment grid was randomly dropped twice per transect for a total of ten readings per site. The suspended sediment was collected using the depth-integrated sampler. The depth-integrated sampler was moved horizontally and vertically along the stream cross-section. The sample was placed in the digital “turbidimeter” (2100 P) for reading. The “cobble embeddedness” is a surrogate measure of interstitial space of streambed cobble habitats. The RBP cobble embeddedness qualitatively measures the amount of fine sediment surrounding cobbles and filling interstitial space of the substrate (Plafkin et al. 1989). This visual assessment assigned a score for the entire reach.

Macroinvertebrates

Collection methods followed a modified version of the procedures of EPA’s EMAP protocol (EPA 1998). Macroinvertebrates were collected quantitatively using a Hess sampler. The dimensions of the Hess sampler cylinder were approximately 33

cm in diameter, 41 cm in height with an area of 858 cm². The mesh size of the inflow window was 1,000 microns and the collecting net was 363 microns. Ten samples were randomly collected per stream reach (two per transect) to form one composite sample for each site. Cobbles and other substrate in the Hess sampler were thoroughly scrubbed to wash macroinvertebrates into the collecting net. Macroinvertebrates were preserved in 90 % ethanol and refrigerated for laboratory processing.

Macroinvertebrates were processed in the laboratory using standard methods for sorting and identification (EPA 1999). Macroinvertebrates were subsampled using a 300-organism subsample size. Composite samples were spread evenly across a standard subsampling tray and random grids were selected and counted until a minimum of 300 organisms was achieved. A dissecting microscope was used for sorting and identification. After reaching 300 macroinvertebrates, the rest of the grid was counted to achieve full count of the grid. If the composite sample contained less than 300 macroinvertebrates, then the entire sample was counted. Macroinvertebrates were identified using Merritt and Cummins (1996). The insect orders Ephemeroptera, Plecoptera, and Trichoptera were identified to the genus level where possible. All other macroinvertebrates were identified to the order level. All samples and specimens were labeled, preserved and stored for quality control.

Data Analysis

To assess if environmental variables and macroinvertebrate assemblages were significantly different among basins, one-way ANOVA was used. The significance level used was $\alpha = 0.05$. The null hypothesis was that there would be no significant

differences in terms of environmental variables or macroinvertebrates among basins.

If the null hypothesis was rejected, then the multiple comparison test was performed to reveal which basins were significantly different from each other.

Canonical Correspondence Analysis (CCA), a multivariate statistical method, was used to relate macroinvertebrate assemblages (EPT genera only) to measured environmental variables. Canoco for Windows Version 4.0 was used.

To quantify the relationship between fine sediments and macroinvertebrate assemblages, a simple regression was performed. Biotic metrics were plotted against environmental variables to estimate the strength of the quantitative relationships. The coefficient of determination (r^2) was used to determine the associations between environmental variables and macroinvertebrates. Expectations were that macroinvertebrate metrics would respond to % fine sediments along the fine sediment gradient. A priori predictions were that sensitive macroinvertebrate would decline as fine sediment levels became higher with more tolerant assemblages replacing sensitive assemblages. It was expected that the highest sediment levels, a complete shift of macroinvertebrate assemblages would occur. High r^2 values indicated strong associations between environmental variables and macroinvertebrates. The coefficient of variation (CV) was calculated to assess the variability of environmental variables among sites within each basin. The CV is a measure of the variability, which is independent of the mean.

RESULTS

Environmental Condition

% Fine sediments

The 59 stream sites formed a continuous fine sediment gradient (Figure 3), which ranged from a low of 2 % to a high of 64 % with an average of 22 % (Table 6). The variability of % fine sediment within each basin was noticeably high. Deep Creek and Clear Creek recorded the highest fine sediment variability with coefficient of variations (CV) of 99 %. The CV for Johnson Creek and Tryon Creek was 72 % and 49 %, respectively.

The urban streams showed significant differences in % fine sediments from the rural streams. ANOVA showed that % fines was significantly different among basins ($p = 0.005$). The multiple comparison test displayed that % fines in Clear Creek was significantly lower than in the urban basins (Figure 4). Tryon Creek (mean = 32 %) had nearly five times as much fine sediment than Clear Creek, while Johnson Creek (mean = 23 %) had approximately three times more fine sediment than Clear Creek (mean = 7 %) (Table 7, 8).

Conductivity

The variability of conductivity in Clear Creek (CV = 9 %) and Tryon Creek (CV = 1 %) was relatively low compared to Johnson Creek (CV = 33 %) and Deep Creek (CV = 27 %). The conductivity in both rural streams was significantly lower than in the urban streams (Figure 4). Tryon Creek's conductivity (mean = 181 $\mu\text{S}/\text{cm}$)

Figure 3. The variation of percent fine sediment among Oregon stream sites (n = 65).

Table 6. Mean, range, and standard deviation for selected environmental variables measured at the 59 stream sites

Environmental variable	mean	min	max	st dev
59 stream sites:				
% fines (%)	22	2	64	18
NO ₃ + NO ₂ (mg/L)	1.5	0.1	5.7	1.6
PO ₄ ³⁻ (mg/L)	0.1	0.02	0.4	0.1
total phosphorus (mg/L)	0.1	0.03	1.7	0.2
turbidity (NTU)	4.0	0.7	11.7	2.5
conductivity (μ S/cm)	138.5	55.8	240.7	55.2
temperature (°C)	15.1	10.2	18.6	2.0
pH	7.4	6.4	8.1	0.4
flow (m ³ /s)	0.2	0.0001	1.5	0.3
embeddedness (EPA 1998)	13	1	19	5

Figure 4. Comparison of sediments (A. % fine sediment, B. conductivity, C. embeddedness) between urban and non-urban basins (mean $\pm$ SD). Letters indicate significant differences ($p < 0.05$).

Table 7. Mean, range, and standard deviation for selected environmental variables measured at (A) Clear Creek and (B) Deep Creek.

A. Clear Creek:	mean	min	max	st dev
Environmental variable:				
% fines (%)	7	2	24	6
NO ₃ + NO ₂ (mg/L)	0.2	0.1	0.2	0.0
PO ₄ ³⁻ (mg/L)	0.02	0.02	0.02	0.00
total phosphorus (mg/L)	0.04	0.03	0.05	0.01
turbidity (NTU)	0.9	0.7	1.1	0.3
conductivity (μ S/cm)	63.8	55.8	71.1	5.8
temperature (°C)	13.0	10.2	15.4	2.0
pH	7.5	7.0	7.9	0.3
flow (m ³ /s)	0.6	0.1	0.9	0.2
embeddedness (EPA 1998)	17.3	11.0	19.0	2.3
B. Deep Creek:				
Environmental variable:				
% fines (%)	22	3	57	22
NO ₃ + NO ₂ (mg/L)	1.2	0.2	3.2	1.0
PO ₄ ³⁻ (mg/L)	0.1	0.02	0.4	0.1
total phosphorus (mg/L)	0.1	0.0	0.4	0.1
turbidity (NTU)	2.0	1.2	2.8	0.7
conductivity (μ S/cm)	107.6	70.5	154.3	29.4
temperature (°C)	14.4	12.3	16.8	1.7
pH	7.5	7.1	7.9	0.3
flow (m ³ /s)	0.2	0.1	0.7	0.2
embeddedness (EPA 1998)	14.0	8.0	18.0	4.6

Table 8. Mean, range, and standard deviation for selected environmental variables measured at (a) Johnson Creek and (b) Tryon Creek.

A. Johnson Creek:	mean	min	max	st dev
Environmental variable:				
% fines (%)	23	4	64	16
NO ₃ + NO ₂ (mg/L)	2.5	0.6	5.7	1.9
PO ₄ ³⁻ (mg/L)	0.05	0.02	0.1	0.03
total phosphorus (mg/L)	0.1	0.03	1.7	0.3
turbidity (NTU)	4.5	1.0	11.7	2.5
conductivity (μ S/cm)	153.5	68.9	240.7	51.3
temperature (°C)	15.3	12.5	18.6	1.6
pH	7.3	6.4	8.1	0.4
flow (m ³ /s)	0.2	0.0001	0.9	0.3
embeddedness (EPA 1998)	9.5	1.0	16.0	4.0
B. Tryon Creek:				
Environmental variable:				
% fines (%)	32	11	49	15
NO ₃ + NO ₂ (mg/L)	0.8	0.5	1.3	0.3
PO ₄ ³⁻ (mg/L)	0.1	0.02	0.1	0.03
total phosphorus (mg/L)	0.1	0.1	0.2	0.04
turbidity (NTU)	4.6	4.6	4.6	0.0
conductivity (μ S/cm)	181.0	153.0	216.5	20.4
temperature (°C)	16.6	14.1	18.6	1.2
pH	7.7	7.1	8.0	0.2
flow (m ³ /s)	0.02	0.00	0.04	0.01
embeddedness (EPA 1998)	9.5	2.0	14.0	3.5

was nearly three times higher than Clear Creek while Johnson Creek (mean = 154 $\mu\text{S}/\text{cm}$) was more than two times higher than Clear Creek (mean = 64 $\mu\text{S}/\text{cm}$).

Embeddedness

Variability of embeddedness scores was low in Clear Creek (CV = 13 %), but relatively high in Johnson Creek (CV = 43 %), Tryon Creek (CV = 36 %), and Deep Creek (CV = 33 %). Clear Creek (mean = 17) showed significantly higher RBP embeddedness scores than Tryon Creek (mean = 10) and Johnson Creek (mean = 10) (Figure 4). Clear Creek's embeddedness score fell in the "optimal condition" while the urban streams scores landed in the "marginal conditions" category (Table 5). Deep Creek's average (mean = 14) embeddedness score was significantly higher than Johnson Creek.

Flow

The flow variability was very high in Johnson Creek (CV = 133 %), moderately high in Deep Creek (CV = 85 %) and Tryon Creek (CV = 72 %), and lower in Clear Creek (CV = 39 %). The average flow at the 59 sites was 0.2 m^3/s with a range of 0.0001 to 1.5 m^3/s (Table 6). Multiple comparison test revealed that Clear Creek (mean = 0.60 m^3/s) had significantly higher flow than the other three basins (mean Deep Creek = 0.24 m^3/s , Johnson Creek = 0.20 m^3/s , Tryon Creek = 0.02 m^3/s) (Figure 5).

Figure 5. Comparison of A. flow and B. temperature in urban and non-urban basins (mean $\pm$ SD). Letters indicate significant differences ($p < 0.05$).

Temperature

Clear Creek (CV = 16 %) and Deep Creek (CV = 12 %) had higher variability than Johnson Creek (CV = 10 %) and Tryon Creek (CV = 7 %). The average temperature at all sites ranged from a low of 10.2 °C to a high of 18.6 °C with an average of 15.1 °C (Table 6). The multiple comparison test showed that the temperature in Clear Creek (mean = 13.0 °C) was significantly lower than the temperature in the urban streams (Johnson Creek, mean = 15.3 and Tryon Creek, mean = 16.6 °C) (Figure 5). The temperature in Deep Creek (mean = 14.4 °C) was also significantly lower than in Tryon Creek (mean = 16.6 °C).

Nutrients

Total Phosphorous

Total phosphorous displayed high variability in Johnson Creek (CV = 211 %) and Deep Creek (CV= 94 %), moderate variability in Tryon Creek (CV= 37 %), and lower variability in Clear Creek (CV= 18 %). The average total phosphorous was 0.1 mg/L with a low of 0.03 to a high of 1.7 mg/L (Table 6). Total phosphorous showed no significant differences among basins (Figure 6).

Dissolved Inorganic Nitrogen (NH_4^+ and $\text{NO}_2 + \text{NO}_3$)

The variability of ammonia was high in Deep Creek (CV= 88 %), and in Johnson Creek (CV= 75 %), and lower in Tryon Creek (CV= 34 %) and Clear Creek

Figure 6. Comparison of A. NO₂+NO₃, B.o-phosphorous, and C. total phosphorous in urban and non-urban basins (mean $\pm$ SD). Letters indicate significant differences ($p < 0.05$).

(CV= 20 %). The ammonia concentrations ranged from 0.1 mg/L to 5.7 mg/L with an average of 1.5 mg/L (Table 6). Johnson Creek's ammonia concentrations were significantly higher than both Clear Creek and Deep Creek (Figure 6).

Dissolved Inorganic Phosphorous (O-phosphate)

The variability of O-phosphate was high in Deep Creek (CV = 112 %), moderate in Johnson Creek (CV = 67 %) and Tryon Creek (CV = 51 %), and lowest in Clear Creek (CV = 0 %). The O-phosphate concentrations varied from 0.0 to 0.4 mg/L with an average of 0.1 mg/L (Table 6). Deep Creek's O-phosphate concentrations were significantly higher than both Clear Creek and Johnson Creek (Figure 6).

Biotic Condition

Macroinvertebrates assemblages

A total of 14 orders of aquatic macroinvertebrates were identified at the 59 stream sites. Table 9 and 10 illustrate the taxonomic order, family, and genus of EPT (orders: Ephemeroptera, Plecoptera and Trichoptera) and all other orders of aquatic macroinvertebrates and their common names found in the four basins. Overall, the EPT (order) comprised approximately 37 % of all aquatic macroinvertebrates collected (Table 11). Twenty-two families and forty-four genera of EPT were identified (Table 9). Taxa other than EPT, comprised approximately 62 % of the macroinvertebrates, which were identified to the taxonomic order level. The most

Table 9. Macroinvertebrate taxa list of taxonomic order, family, and genus of Ephemeroptera, Plecoptera, and Trichoptera found at the 59 stream sites.

Order	Family	Genus	Order	Family	Genus
Ephemeroptera			Trichoptera		
	Ameletidae	<i>Ameletus</i>		Brachycentridae	<i>Micrasema</i>
	Baetidae	<i>Acentrella</i> <i>Baetis</i> <i>Dipheter hageni</i>		Glossosomatidae	<i>Glossosoma</i>
	Ephemerellidae	<i>Attenella</i> <i>Drunella</i> <i>Ephemerella</i> <i>Serratella</i>		Hydroptilidae	<i>Hydroptila</i> <i>Leucotrichia</i> <i>Ochrotrichia</i>
	Heptageniidae	<i>Cinygma</i> <i>Cinygmula</i> <i>Epeorus</i> <i>Heptagenia</i> <i>Ironodes</i> <i>Rhithrogena</i>		Hydropsychidae	<i>Arctopsyche</i> <i>Parapsyche</i> <i>Cheumatopsyche</i> <i>Hydropsyche</i>
	Leptophlebiidae	<i>Paraleptophlebia</i>		Lepidostomatidae	<i>Lepidostoma</i>
Plecoptera				Limnephilidae	<i>Dicosmoecus</i> <i>Hydatophylax</i>
	Chloroperlidae	<i>Suwallia</i> <i>Sweltsa</i>		Philopotamidae	<i>Wormaldia</i>
	Leuctridae	<i>Leuctra</i>		Psychomyiidae	<i>Psychomyia</i>
	Nemouridae	<i>Amphinemura</i> <i>Malenka</i> <i>Zapada</i>		Rhyacophilidae	<i>Himalopsyche</i> <i>Rhyacophila</i>
	Peltoperlidae	<i>Yoraperla</i>		Uenoidae	<i>Neophylax</i>
	Perlidae	<i>Calineuria</i> <i>Hesperoperla</i>			
	Perlodidae	<i>Perlinodes</i> <i>Skwala</i>			
	Pteronarcyidae	<i>Pteronarcella</i>			

Table 10. Taxa list of the taxonomic order and common names of all other macroinvertebrate taxa other than Ephemeroptera, Plecoptera, and Trichoptera found at the 59 stream sites.

Order	common name
Amphipoda	scuds
Coleoptera	beetles
Diptera	midges
Gastropoda	snails
Hemiptera	water striders
Isopoda	pill/sow bugs
Lepidoptera	aquatic moths
Megaloptera	alderflies
Oligochaeta	aquatic earthworms
Pelecypoda	freshwater clams
Trombidiformes	water mites

Table 11. Distribution of macroinvertebrate orders among urban and rural basins (data indicate relative abundance).

taxonomic order:	Basin				
	Overall	Clear	Deep	Johnson	Tryon
Ephemeroptera	22.5	34.3	25.0	16.6	14.0
Plecoptera	3.7	4.1	2.6	1.1	6.8
Trichoptera	10.8	11.0	8.3	12.4	11.6
Total % EPT	37.1	50.1	35.9	30.1	32.4
Hemiptera	0.1	0.0	0.0	0.0	0.3
Oligochaeta	4.0	1.8	0.9	3.9	8.2
Gastropoda	19.3	23.1	4.7	29.6	0.9
Amphipoda	4.3	0.0	2.0	4.6	8.9
Diptera	28.6	4.2	45.4	24.7	47.2
Coleoptera	3.1	4.8	5.0	3.1	0.4
Mollusca	0.5	0.0	0.0	1.0	0.0
Isopoda	0.1	0.0	0.1	0.1	0.0
Lepidoptera	0.1	0.0	0.0	0.1	0.0
Megaloptera	0.1	0.0	0.0	0.1	0.1
Trombidiformes	0.6	0.8	1.5	0.4	0.3
Other	1.5	2.4	5.6	0.5	0.6
Total % other	62.2	37.1	65.1	68.1	67.0

common macroinvertebrate orders other than EPT were Diptera (27 %), Gastropoda (19 %), Amphipoda (4 %), and Oligochaeta (4.0 %) (Table 11).

The EPT taxa richness (number of different taxa) in the rural basins were significantly higher than the urban basins ($p < 0.0001$). Johnson Creek had an average of 4 EPT taxa and Tryon Creek had an average of 4 EPT taxa compared to 15 EPT taxa in Clear Creek and 11 EPT taxa in Deep Creek (Figure 7). The multiple comparison test showed that EPT taxa richness in both Clear Creek and Deep Creek was significantly higher than Johnson Creek and Tryon Creek.

Ephemeroptera, Plecoptera and Trichoptera

Tryon Creek (CV = 66%) and Johnson Creek (CV = 63%) had the highest variabilities of the relative abundance of the orders EPT, while Deep Creek (CV= 56%) and Clear Creek (CV= 37%) had comparatively lower variability. The relative abundance of EPT taxa was significantly different among two basins ($p = 0.04$) (Figure 8). Clear Creek (mean = 50 %) showed significantly higher EPT than Johnson Creek (mean = 30.1 %). Deep Creek (mean = 36 %), Johnson Creek and Tryon Creek (mean = 32 %) had relatively similar EPT abundances (Table 11).

Tryon Creek (CV = 90%) and Johnson Creek (CV = 73%) had higher variability's of the order Ephemeroptera than Deep Creek (CV = 49%) and Clear Creek (CV = 40%). The relative abundance of the order Ephemeroptera was significantly higher in Clear Creek than in the urban streams ($p = 0.0006$) (Figure 8). Clear Creek had an average of about 34 % Ephemeroptera compared with an average of 16 % Ephemeroptera for the urban sites (Table 11).

Figure 7. Comparison of Ephemeroptera, Plecoptera, and Trichoptera taxa richness in urban and non-urban basins(mean $\pm$ SD). Letters indicate significant differences ($p < 0.05$).

The percent of the order Plecoptera displayed high variability within each basin. Johnson Creek (CV = 394 %) and in Tryon Creek (CV = 260 %) were comparatively higher than Deep Creek (CV = 110 %) and Clear Creek (CV = 83 %). Plecoptera had lower relative abundance than other EPT taxa in all basins (range: 1–7 %) (Table 11). There were no significant differences of Plecoptera among basins ($p = 0.29$) (Figure 8).

The order Trichoptera had highest variability in Deep Creek (CV = 125 %), and lower variability in Johnson Creek (CV = 107 %), Tryon Creek (CV = 109 %), and Clear Creek (CV = 97 %). Trichoptera displayed highest relative abundance in Johnson Creek (mean = 12 %) and the lowest abundance in Deep Creek (mean = 8%) (Table 11). This taxa displayed high variability within each basins, but showed no significant differences among basins ($p > 0.05$) (Figure 8).

Diptera, Gastropoda, Oligochaeta, and Amphipoda

All other macroinvertebrate orders other than Ephemeroptera, Plecoptera and Trichoptera displayed significantly higher relative abundance in Johnson Creek than in Clear Creek ($p = 0.04$) (Figure 9). Johnson Creek (mean = 68 %) displayed the highest percentage of non-EPT taxa in all streams, while Clear Creek (mean = 37 %) showed the lowest percentage (Table 11).

The relative abundance of the order Diptera was significantly lower in Clear Creek (mean = 4 %) than in all other basins (Figure 10). Additionally, Johnson Creek (mean = 25 %) had significantly lower Diptera than Tryon Creek (mean = 45%) ($p < 0.0001$).

Figure 8. Comparison of the relative abundance of A. EPT, B. Ephemeroptera, C. Plecoptera, and D. Trichoptera in urban and non-urban basins (mean $\pm$ SD). Letters indicate significant differences ($p < 0.05$).

Figure 9. Comparison of the relative abundance of macroinvertebrates other than Ephemeroptera, Plecoptera, and Trichoptera in urban and non-urban Basins (mean $\pm$ SD). Letters indicate significant differences ($p < 0.05$).

Figure 10. Comparison of the relative abundance of A. Diptera, B. Gastropoda, C. Oligochaeta, and D. Amphipoda in urban and non-urban basins (mean ± SD). Letters indicate significant differences (p < 0.05).

The order Gastropoda was significantly different among basins ($p < 0.0001$) (Figure 10). Tryon Creek (mean = 1%) was significantly lower than Clear Creek (mean = 23 %) and Johnson Creek (mean = 30 %). Deep Creek (mean = 5 %) had significantly lower Gastropoda than Johnson Creek (Table 11).

There were no significant differences among the order Oligochaeta among basins (Figure 10). Tryon Creek (mean = 8 %) and Johnson Creek (mean = 4 %) had higher abundances of Oligochaeta than Clear Creek (mean = 2 %) and Deep Creek (mean = 1%).

The relative abundance of the order Amphipoda displayed no significant differences among basins (Figure 10). Tryon Creek recorded (mean = 9 %) the highest amount of Amphipoda while Clear Creek (mean = 0 %) had the lowest amount (Table 11).

Multivariate Statistics

All of the urban stream sites and most of the rural stream sites were separated by the Axis 1 (Figure 11). Axis 1 is positively correlated with embeddedness ($r = 0.75$) and flow ($r = 0.52$) and negatively correlated with temperature ($r = -.70$), conductivity ($r = -0.66$), and % fine sediment ($r = -0.46$). All Clear Creek and some Deep Creek sites ordinated on the right side of Axis 1, while all of the urban sites ordinated on the left side of Axis 1. The urban sites showed considerable variation on the left side of axis one and were spread along the entire range of Axis 2. Axis 2 was highly correlated with pH ($r = 0.01$).

Figure 11. Canonical Correspondence Analysis (CCA) on macroinvertebrate genera (EPT) and selected environmental variables. Δ Clear Creek, $\blacktriangledown$ Deep Creek, $\circ$ Johnson Creek, $\square$ Tryon Creek.

EPT Genera

Overall, the genus *Baetis* (order: Ephemeroptera, family: Baetidae) was the most abundant genus encompassing approximately 11% of the relative abundance of all genera present in EPT (Ephemeroptera, Plecoptera and Trichoptera) from all 59 stream sites (Table 12). The two other most abundant genera in the order Ephemeroptera were *Rhithrogena* (family: Heptageniidae) (relative abundance = 4 %) and *Paraleptophlebia* (family: Heptageniidae) (relative abundance = 2 %). *Zapada* (family: Nemouridae) (relative abundance = 2 %), *Sweltsa* (family: Chloroperlidae) (relative abundance = 0.4%), and *Calineuria* (family: Perlidae) (relative abundance = 0.4%), comprised the 3 most abundant Plecoptera at our 59 sites. The three most abundant Trichoptera genera in all four basins were *Hydropsyche* (family: Hydropsychidae) (relative abundance = 7 %), *Cheumatopsyche* (order: Trichoptera, family: Hydropsychidae) (relative abundance = 3 %), and *Glossosoma* (order: Trichoptera, family: Glossosomatidae) (relative abundance = 2 %).

Ephemeroptera genera

Baetis was ubiquitous in all basins comprising about 8 – 12 % in each stream (Table 12). *Baetis* was not significantly different among basins (Figure 12).

Rithrogena was the third most abundant genus, however *Rithrogena* was most abundant in Clear Creek (relative abundance = 18 %) and Deep Creek (relative abundance = 4 %) and lower in the urban streams (Johnson Creek, relative abundance = 0.4 % and Tryon Creek, relative abundance = 0.2 %) (Table 12). The relative abundance of *Rithrogena* was not significantly different among basins ($p = 0.13$).

Table 12. Relative Abundance of Ephemeroptera, Plecoptera, and Trichoptera genera in urban and rural basins (* indicates 3 most abundant).

Order:	Genus:	Basin				
		Overall	Johnson	Tryon	Clear	Deep
Ephemeropter	<i>Ameletus</i>	0.0	0	0	0	0.14
	<i>Acentrella</i>	1.4	0.71	0.00	2.72	4.63
	<i>Baetis</i> *	10.6	11.66	10.13	9.96	8.48
	<i>Diphetor hageni</i>	1.0	1.12	0.00	1.50	1.30
	<i>Attenella</i>	0.1	0.00	0.00	0.25	0.27
	<i>Drunella</i>	0.2	0.00	0.00	0.95	0.04
	<i>Serratella</i>	0.0	0.00	0.00	0.03	0.00
	<i>Ephemerella</i>	0.1	0.00	0.00	0.13	0.31
	<i>Cinygma</i>	0.6	0.00	2.92	0.00	0.00
	<i>Cinygmula</i>	0.1	0.00	0.00	0.84	0.00
	<i>Epeorus</i>	0.5	0.05	0.00	1.86	1.34
	<i>Ironodes</i>	0.1	0.10	0.00	0.10	0.00
	<i>Heptagenia</i>	0.0	0.00	0.04	0.00	0.00
	<i>Rhithrogena</i> *	3.8	0.40	0.22	18.15	3.68
	<i>Paraleptophlebia</i> *	2.0	2.49	0.68	0.38	4.09
Plecoptera	<i>Despaxia</i>	0.1	0.00	0.62	0.00	0.04
	<i>Leuctra</i>	0.0	0.00	0.00	0.03	0.00
	<i>Amphinemura</i>	0.0	0.01	0.00	0.00	0.24
	<i>Malenka</i>	0.1	0.17	0.00	0.03	0.00
	<i>Zapada</i> *	1.8	0.74	5.79	0.87	1.12
	<i>Suwallia</i>	0.0	0.02	0.00	0.03	0.00
	<i>Sweltsa</i> *	0.4	0.15	0.31	1.65	0.11
	<i>Yoroperla</i>	0.1	0.00	0.00	0.52	0.00
	<i>Calineuria</i> *	0.4	0.03	0.00	1.93	0.63
	<i>Hesperoperla</i>	0.0	0.00	0.00	0.25	0.00
	<i>Perlinodes</i>	0.0	0.00	0.00	0.03	0.04
	<i>Skwala</i>	0.1	0.00	0.08	0.30	0.37
	<i>Pteronarcella</i>	0.0	0.00	0.00	0.07	0.00
Trichoptera	<i>Neophylax</i>	0.1	0.20	0.00	0.07	0.16
	<i>Himalopsyche</i>	0.0	0.00	0.05	0.00	0.00
	<i>Glossosoma</i> *	1.6	0.42	0.00	7.17	1.64
	<i>Rhyacophila</i>	0.4	0.00	0.00	1.22	1.25
	<i>Hydroptila</i>	0.0	0.07	0.00	0.00	0.07
	<i>Leucotrichia</i>	0.0	0.00	0.00	0.00	0.04
	<i>Ochrotrichia</i>	0.0	0.03	0.10	0.00	0.00
	<i>Arctopsyche</i>	0.0	0.00	0.00	0.03	0.00
	<i>Parapsyche</i>	0.0	0.00	0.04	0.00	0.00
	<i>Cheumatopsyche</i> *	2.5	4.55	0.00	1.34	0.25
	<i>Hydropsyche</i> *	6.5	6.99	10.36	1.84	4.46
	<i>Wormaldia</i>	0.2	0.00	0.00	1.12	0.08
	<i>Psychomyia</i>	0.0	0.00	0.00	0.00	0.04
	<i>Micrasema</i>	0.0	0.00	0.00	0.07	0.00
	<i>Lepidostoma</i>	0.0	0.01	0.00	0.00	0.00
	<i>Dicosmoecus</i>	0.0	0.00	0.00	0.00	0.19
	<i>Hydatophylax</i>	0.2	0.00	0.83	0.00	0.00

Figure 12. Comparison of the relative abundance of EPT genera A. Baetis, B. Sweltsa, and C. Calineuria in urban and non-urban basins (mean, SD). Letters indicate significant differences ($p < 0.05$).

Rithrogena showed high variability in all streams. *Paraleptophlebia* displayed differences in abundance between the urban and rural basins. High variability's in each basin were also seen with these taxa. The relative abundance of *Paraleptophlebia* between basins was not significant ($p = 0.06$). *Acentrella* (order: Ephemeroptera, family: Baetidae), the fourth most common Ephemeroptera genus, showed significantly higher ($p = 0.001$) relative abundances in Deep Creek than in the urban streams.

Plecoptera genera

Two of the most abundant Plecoptera genera showed significantly different relative abundances among basins (Table 12). *Sweltsa* was significantly higher in Clear Creek than in all other basins ($p < 0.0001$) (Figure 12). *Calineuria* also showed significant higher abundance in Clear Creek than in all other basins ($p < 0.0001$). *Zapada*, the most common Plecoptera at all sites, displayed high abundance in Tryon Creek, but was not significantly different among basins ($p = 0.29$).

Trichoptera genera

The most common Trichoptera genus, *Hydropsyche*, displayed high relative abundance in Johnson and Tryon Creek compared to the non-urban basins (Table 12). However, *Hydropsychae* was not significantly different among and was highly variable within each basin ($p = 0.32$). *Cheumatopsyche* were highly abundant in Johnson Creek and completely absent in Tryon Creek. *Cheumatopsyche* was not significantly different among basins ($p = 0.11$). *Glossosoma* was highly abundant in

Clear Creek and completely absent in Tryon Creek. *Glossosoma* was significantly different in Clear Creek compared to the urban basins ($p = 0.001$).

Relationships between macroinvertebrates and sediments

The quantitative relationships between macroinvertebrates and sediments are weak. Regression analysis showed no strong relationships between fine sediments and the macroinvertebrate metrics as indicated by the low r^2 (coefficient of determination) (Table 13). For example, EPT taxa richness versus fine sediments showed weak relationships ($r^2 = 0.2$) (Figure 13). Other measures such as % EPT, % Ephemeroptera, % Plecoptera and % Trichoptera also displayed weak relationships with fine sediment (Figure 14). Non EPT orders, such as Diptera and Gastropoda also showed no strong relationships (Figure 15). Regressions with macroinvertebrates metrics versus conductivity and embeddedness also showed low r^2 (Table 14).

Finer levels of taxonomic resolution also reveal similar weak relationships with fine sediments. For example, plots of the common Ephemeroptera genera vs. % fine sediments show low r^2 (Figure 16). Regressions of conductivity and RBP embeddedness versus EPT genera also display weak relationships (Table 14).

The relationships between macroinvertebrate feeding groups and sediments are not strong. Regressions of EPT functional feeding groups versus sediment variables displayed weak correlations (Table 13). Further breakdown of feeding groups within the orders Ephemeroptera, Plecoptera and Trichoptera (E, P, and T: collectors, scrapers, scraper/collector, shredder, predator, collector, and filtering collector) also showed low r^2 (Table 14). For example, regression analysis of Ephemeroptera

Table 13. Coefficient of determinations (r^2) for regressions of macroinvertebrate metrics, EPT genera, and EPT feeding guilds versus % fine sediments, conductivity, and embeddedness.

Metric:	fine sediment	Conductivity	Embeddedness
EPT taxa richness	0.2	0.5	0.2
% EPT	0.1	0.1	0.07
% E	0.2	0.1	0.06
% P	0.0001	0.006	0.01
% T	0.004	0.02	0.004
Ephemeroptera genera:			
<i>Acentrella</i>	0.06	0.05	0.05
<i>Baetis</i>	0.0003	0.01	0.02
<i>Dipheter hageni</i>	0.00006	0.002	0.01
<i>Epeorus</i>	0.04	0.3	0.07
<i>Rhithrogena</i>	0.3	0.3	0.1
<i>Paraleptophlebia</i>	0.01	0.00005	0.006
Feeding Guild:			
E scraper/collector	0.1	0.1	0.1
E collector	0.03	0.01	0.1
E predator	0.04	0.2	0.1
P shredder	0.01	0.01	0.00
P predator	0.1	0.2	0.2
T scraper	0.1	0.2	0.2
T filtering collector	0.05	0.0001	0.03
T predator	0.1	0.3	0.2

Figure 13. Regression of EPT taxa richness versus % fine sediments (R^2 = coefficient of determination).

Figure 14. Logarithmic regression of A. % EPT, B. % E, C. % P, and D. % T versus fine sediments (R^2 = coefficient of determination).

Figure 15. Regression analysis of A. % Diptera, B. % Gastropoda, and C. % Oligochaeta versus fine sediments (R^2 = coefficient of determination).

Table 14. Coefficient of determinations (r^2) for regressions of macroinvertebrate metrics, EPT genera, and EPT feeding guilds versus % fine sediments, conductivity, and embeddedness.

Metric:	fine sediment	Conductivity	Embeddedness
EPT taxa richness	0.2	0.5	0.2
% EPT	0.1	0.1	0.07
% E	0.2	0.1	0.06
% P	0.0001	0.006	0.01
% T	0.004	0.02	0.004
Ephemeroptera genera:			
<i>Acentrella</i>	0.06	0.05	0.05
<i>Baetis</i>	0.0003	0.01	0.02
<i>Dipheter hageni</i>	0.00006	0.002	0.01
<i>Epeorus</i>	0.04	0.3	0.07
<i>Rhithrogena</i>	0.3	0.3	0.1
<i>Paraleptophlebia</i>	0.01	0.00005	0.006
Feeding Guild:			
E scraper/collector	0.1	0.1	0.1
E collector	0.03	0.01	0.1
E predator	0.04	0.2	0.1
P shredder	0.01	0.01	0.00
P predator	0.1	0.2	0.2
T scraper	0.1	0.2	0.2
T filtering collector	0.05	0.0001	0.03
T predator	0.1	0.3	0.2

Figure 16. Logarithmic regression of A. % Baetis, B. % Rithrogena, C. % Paraleptophlebia, and D. % Acentrella versus fine sediments (R² = coefficient of determination).

collectors, Plecoptera predators, and Trichoptera scrapers versus fine sediments displayed weak relationships (Figure 17).

Land Use

Table 15 compares the land use and impervious surface area between the urban and rural basins. Clear Creek and Deep Creek are characterized by forest area and agricultural land use while Johnson Creek and Tryon Creek have large percentage of urban land use. Additionally, the urban basins have more impervious surface area than the rural basins.

Figure 17. Regression analysis of A. Ephemeroptera collector, B. Plecoptera Predator C. Trichoptera scraper versus fine sediments (R^2 = coefficient of determination).

Table 15. Comparison of land use (type and percentage), and impervious surface area between rural and urban basins.

Land use	Rural basins		Urban basins	
	Clear Creek	Deep Creek	Johnson Creek	Tryon Creek
Forested or parks/open space	65%	35%	5%	12%
Urban	0%	0%	54%	86%
Agriculture	25%	45%	33%	0%
Impervious surface area	11%	20%	65%	30%

DISCUSSION

Basin Land Use and Macroinvertebrate Assemblages

This field study revealed that macroinvertebrates were significantly different between urban and non-urban streams. A total of six macroinvertebrate metrics were significantly different between the urban and rural stream basins. For example, % EPT in Clear Creek (50%) was significantly higher than in Johnson Creek (30%) ($p = 0.04$). Another example of significant differences between the urban and rural stream basins was displayed in the differences of EPT taxa richness. Both Clear Creek and Deep Creek had significantly higher EPT taxa richness than both Johnson Creek and Tryon Creek (Figure 7). These differences between the urban and rural basins were substantial. Clear Creek recorded more than three times higher EPT taxa richness than Johnson Creek and more than four times higher than Tryon Creek. Deep Creek had more than two times higher EPT taxa richness than Johnson Creek and more than three times higher than Tryon Creek. The amount of the Diptera was significantly lower in Clear Creek than in all other basins. Clear Creek recorded approximately six times lower % Diptera than Johnson Creek and nearly 11 times lower than Tryon Creek (Table 11). The different macroinvertebrate metrics may very likely be due to land use differences between the urban and rural basins. Several researchers reported that changes in basin land use often correlated with changes in macroinvertebrate assemblages in streams. Quinn et al. (1997) found that EPT taxa densities in forest streams were 2-3 fold higher than in pasture streams in which the macroinvertebrate assemblages were dominated by chironomids, snails, and worms. Stanley and James

(1997) found that streams in an urbanized portion of a watershed contained macroinvertebrates dominated by pollution tolerant taxa such as Isopoda. In contrast, streams in non-urbanized areas were dominated by pollution sensitive species (orders: Ephemeroptera, and Trichoptera). Lenat and Crawford (1994) found that invertebrate taxa richness and the number of unique invertebrate species indicated moderate stress (fair water quality) at an agricultural site and severe stress (poor water quality) at an urban site. The urban site was characterized by low species richness for most groups and very low abundance values. Their study also showed that dominant macroinvertebrate groups shifted from Ephemeroptera at the forested site, to Chironomidae at the agricultural site, and Oligochaeta at the urban site. There was little between-site overlap in dominant species (8-17%), indicating that land use strongly influenced the macroinvertebrate community.

Land use data showed clear differences between the urban and rural basins. For example, the dominant land use in Johnson Creek Basin was 62 % urban followed by 33 % agriculture in 1999. In contrast, The land use in Clear Creek Basin was approximately 65 % forested followed by 25 % agriculture. Most notably, a large difference with impervious surface area was found between the urban and rural basins. For example, Johnson Creek had approximately 65 % impervious surface area in the basin in 1999 compared with 11 % impervious surface area for Clear Creek (Table 15). It has been documented that urban land use is characterized by large amounts of impervious surface area (Waters 1995). Impervious surfaces, such as roads, parking lots, and rooftops, can create an impenetrable surface, which can transport sediments and pollutants directly to streams via overland flow. Eroded soils and fine sediment

particles from construction and development activities, and particularly roads are susceptible to runoff from precipitation. The net result of large impervious surface area and high runoff can be an excessive amount of fine sediment particles accumulating on the streambed. Butcher (1999) suggested that increases in density of residential and commercial land use, accompanied by increased amount of impervious surface area, could result in increased pollutant loading and degradation of water quality. Corbett et al. (1997) found that maximum sediment loads to streams occurred when 35 % of the urban basin was impervious surface.

The differences in land use and impervious surface area between urban and non-urban basins may have been one of the primary factors influencing fine sediments among basins. Urban streams showed significantly higher amounts of fine sediments than non-urban streams. For example, Johnson Creek (mean = 23 %) had approximately three times more fine sediment (% fines) in the streambed than Clear Creek (mean = 7 %). Conductivity in Johnson Creek (mean = 153.5 $\mu\text{S}/\text{cm}$) was more than two times higher than in Clear Creek (mean = 63.8 $\mu\text{S}/\text{cm}$). Additionally, Clear Creek (mean = 17) showed significantly higher (higher = less embedded) sediment embeddedness scores than Johnson Creek (mean = 9). These data indicate that the high sediments in urban streams may be associated with urban land use patterns. Excessive fine sediments in urban streams may have adverse effects on pollution sensitive macroinvertebrate taxa such as Ephemeroptera, Plecoptera and Trichoptera and therefore decrease EPT abundance and taxa richness. Several studies had demonstrated such adverse effects of fine sediments on EPT taxa. For example, Lemly (1982) found that species richness, diversity, and total biomass of filter feeding

Trichoptera, predacious Plecoptera, and certain Ephemeroptera were significantly reduced in polluted (sediment and nutrient) zones of a stream. Bjornn et al. (1974, 1977) conducted laboratory experiments and concluded that when the amount of sediment surrounding cobbles (cobble-embeddedness) increased by more than one-third, insect abundance declined by 50 %. When the streambed was later cleaned of fine sediment, mayflies (Ephemeroptera) and stoneflies (Plecoptera) increased by up to eightfold. McClelland and Brusven (1980) found that all species of EPT responded negatively to increasing amounts of sand and to greater cobble embeddedness. The findings of this study are consistent with other studies. Urban basins may deliver more sediments to streams than rural basins, which may result in a change in the relative proportions of pollution sensitive taxa.

The multivariate statistics provided additional data on the differences between urban and rural streams. The Canonical Correspondence Analysis showed correlations between macroinvertebrate genera (EPT) and selected environmental variables. All Clear Creek sites, characterized by low embeddedness, grouped to the right side of the ordination plot, while all the urban sites with some Deep Creek sites ordinated to the left side of the plot. Since Axis 1 is positively correlated with embeddedness ($r = 0.75$) and flow ($r = 0.52$), and negatively correlated with temperature ($r = -.70$), conductivity ($r = -0.66$), and % fine sediment ($r = -0.46$), this distribution of macroinvertebrate assemblages suggests that the urban sites are very different from the rural sites. Additionally, embeddedness displayed the highest correlation coefficient of the selected environmental variables suggesting the importance of embedded sediment as a factor determining macroinvertebrate genera (EPT) distribution.

Effects of Fine Sediments on Macroinvertebrates Assemblages

This study did not investigate causative mechanisms of sediment disturbance on macroinvertebrates. However, visual observations and the significantly lower % fine sediment content and embeddedness scores in Clear Creek as compared to the urban basins, suggest that impact on macroinvertebrates via the removal of habitat space by sediments may be the possible mechanism. A clear visual difference of higher embeddedness in Johnson Creek as compared with the non-urban basins was evident. At many sites in Johnson Creek there appeared to be an excessive amount of fine sediment and silt. This fine sediment and silt filled the interstices, cracks and crevices between substrate clasts resulting in a relatively homogenous substrate environment. The distributional patterns of two macroinvertebrate genera seem to support this hypothesis. *Sweltsa* (order Plecoptera), and *Calineuria* (order Plecoptera) showed significantly higher relative abundance in Clear Creek than the urban basins, while the most ubiquitous genus, *Baetis*, showed no differences between urban and non-urban basins (Figure 12). The trophic habits (behavior) of *Sweltsa* and *Calineuria* are characterized as the taxa which crawl and move in and among cobbles. They depend on interstitial habitat space among the substrate. On the other hand, *Baetis*, the swimmer/clinger, may be able to tolerate less habitat space because it can reside on the top surface of stones and cobbles and uses swimming motions to move around. Several taxa of Plecoptera (including *Sweltsa* and *Calineuria*) are known to be highly sensitive to temperature and dissolved oxygen (Merritt and Cummins 1996). Filling of the interstitial pore space by deposited fine sediments can lower flow among

substrate, which may reduce the dissolved oxygen and raise the temperature in the substrate (Waters 1995). Clear Creek overall recorded lower temperature readings and higher dissolved oxygen possibly because of less filling of interstitial space by fine sediments. The filling of interstitial space by sediments can cause less interstitial flow to move through substrate clasts and can cause higher aerobic respiration by microorganisms, therefore increasing temperature and lowering dissolved oxygen (Allan 1995). The temperature and dissolved oxygen levels in Johnson Creek may not be sufficient enough to sustain the livelihood of these organisms. Quinn et al. (1992) suggested that clay discharges caused reduced bed permeability, interstitial dissolved oxygen, and avoidance reactions of invertebrates (i.e., increased drift) resulted in lower invertebrate densities in their research. In this study, the removal of interstitial habitat space may be an important reason why there were lower abundances of EPT taxa in urban streams. Additionally, deposited sediments may be impacting sensitive macroinvertebrates via a direct smothering effect. Lemly (1982) found that inorganic sediments accumulated on body surfaces and respiratory structures and suggested that this was the major direct effect of sedimentation on stream insects.

Quantitative Relationships between Fine Sediments and Macroinvertebrates

The quantitative relationships between fine sediments and macroinvertebrates were generally weak. Regression analysis between macroinvertebrate metrics and fine sediments showed no strong relationships. The relative abundance of EPT, E, P, T, and EPT taxa richness displayed weak correlations with % fine sediment (Figure 8, Figure 13). For example, a regression of EPT taxa richness versus % fine sediments

showed a coefficient of determination (r^2) of 0.2. Higher taxonomic resolution, represented by genus taxa, also revealed weak relationships with fine sediments. For example, when the most ubiquitous taxa, *Baetis*, was plotted against % fine sediments, r^2 was 0.0003 (Figure 15). Macroinvertebrate metrics other than EPT revealed weak correlations as well. For example, % Diptera plotted against % fine sediments displayed a r^2 of 0.12 (Figure 15). An analysis of functional feeding groups in EPT vs. all sediment parameters (% fines, conductivity, and embeddedness) also revealed weak relationships (Figure 17).

A priori hypothesis of this study was that in general sensitive macroinvertebrates would negatively respond to high amounts of sediment present at each site. Pollution sensitive macroinvertebrates taxa would be present at sites with low to moderate levels of fine sediment and more tolerant forms would be present at sites with higher fine sediment content. This change in macroinvertebrate community composition would occur at each site with increasing fine sediment, regardless of whether those sites were in an urban or non-urban stream basin. For example, if an urban site in Johnson Creek had low sediment content ("good site"), this would result in a higher proportion of sensitive forms of macroinvertebrates. Alternatively, a site, which was adjacent to this "good site", could have high sediment concentrations, which would result in more tolerant forms of macroinvertebrates. The results of this study indicated that sensitive forms were either non-existent or in very small proportions in the urban basins. The quantitative relationships were not strong because of the lack of a sensitive taxa in the urban basins. Therefore, efforts to locally improve habitat conditions may not result in a recovery (colonization) of

pollution sensitive taxa. For example, a total of 17 EPT taxa were present in the rural basins and were nonexistent in the urban basins. They include five Ephemeroptera genera (*Attenella*, *Drunella*, *Serratella*, *Ephemerella*, and *Cinygmula*), five Plecoptera genera (*Despaxia*, *Leuctra*, *Yoroperla*, *Hesperoperla*, and *Pteronarcys*), and 7 Trichoptera genera (*Rhyacophila*, *Hydropsyche*, *Leucotrichia*, *Wormaldia*, *Phsycomyia*, *Micrasema*, and *Dicosmoecus*) (Table 12). It is commonly believed that macroinvertebrates can quickly recover from localized disturbance (Resh 1993). However, cumulative effects of basin-wide land-use may severely reduce species pool in the basin and thus recovery of macroinvertebrate richness and diversity may take long periods of time. Harding et al. (1998) reported the past land-use (40 years ago) could predict present macroinvertebrate diversity much better than present land-use patterns. They suggest that past land use activity may result in long-term reductions and modifications to aquatic diversity, regardless of current restoration activities and preventative measures. They suggest that preservation of habitat fragments may not be sufficient to maintain natural diversity in streams, and maintenance of such biodiversity may require conservation of much or the entire watershed. Delong and Brusven (1998) conducted a study on effects of basin agricultural land use on macroinvertebrate community structure in Lapwai Creek, Idaho. They concluded that agricultural activities throughout the entire drainage basin resulted in a relatively homogenous assemblage of macroinvertebrates despite the fact that the stream network flows through three distinct geomorphological regions. Delong and Brusven (1998) suggest that the alteration of the basin by agricultural land use has reduced the overall species diversity of the stream, leaving a community of species tolerant of

nonpoint source pollution. These two studies support the hypothesis that long-term basin-wise land-use patterns may have a cumulative effect on the basin macroinvertebrate species pool and local improvement of habitat conditions may not be followed immediately by colonization of sensitive species.

Other Factors Affecting the Relationship Between Sediments and Macroinvertebrates

Past land use activity may have been an important factor associated with the weak quantitative relationships between fine sediments and macroinvertebrates, however several other factors may also contribute the weak relationships. In this study, the most ubiquitous taxa had a large influence on the total numbers of macroinvertebrates. These dominant taxa were found in high numbers across all basins regardless of pollution impacted sites. For example, the genera *Baetis*, accounted for more than half of all taxa in the order Ephemeroptera, and approximately one third of all EPT combined at all sites. Therefore, this one taxon may have effectively masked the signal between fine sediments and EPT. Another related factor may have been that the fixed count subsample number of 300 organisms may have been too small. Since very ubiquitous taxa such as Gastropoda, Diptera, *Baetis*, and *Hydropsyche* accounted for the majority of the abundance, there was low proportions of sensitive forms of EPT in the data set. Therefore, the signal may have been diluted from the high numbers of ubiquitous taxa and the lack of a high fixed-count subsample number. Grown et al. (1997) suggested that taxa richness increases as a function of the area sampled and the number of individuals in the sample. A higher subsample size may have yielded higher numbers of intolerant forms and

overall greater taxa richness. Other factors, which should not be ignored are the effects of other environmental parameters and other pollutants on macroinvertebrates. Resh (1993) suggest that invertebrate population and community patterns are products of interacting multiple causes generated at multiple spatial and temporal scales. Therefore, this study may not draw unimodal correlations with fine sediments due to the influence of multiple variables. A combination of both human-induced and natural variables may have a synergistic effect causing a mixed signal to occur. Variables such as stream temperature, stream velocity, channel morphology, substrate type and size, sediment properties, seasonal influences, invertebrate drift and other pollutants and contaminants have been known to affect macroinvertebrate populations and community patterns (Resh 1993). This so-called noise may obscure the responses of macroinvertebrates to fine sediments confounding the strong quantitative relationship. Another important factor, which may contribute to the weak relationship is that macroinvertebrates are known to be highly patchy in their distribution (Allan 1995). Heterogeneous conditions (substrate) within a small area of the stream bottom can cause large differences in macroinvertebrate abundance and diversity. Macroinvertebrates can range from very abundant and/or highly diverse in one area, to completely sparse in an area directly next it. This patchy distribution is not only due to heterogeneous substrate, but also due to differences in current velocity and resource dispersal (i.e. detritus and periphyton) (Reice 1974). Substrate size differences within the reach and between sites could also confound the relationship in this study because macroinvertebrates have differential preferences to substrate particle sizes (Williams 1978). Therefore, to adequately and accurately characterize macroinvertebrate

assemblages in a stream may require much more intense sampling efforts than this study. The same holds true for sediments. Accurate sediment quantification can be difficult because the spatial distribution of sediments depends on the stream velocity, slope, and substrate (size, type, and shape) (Wood and Armitage 1997).

Management Implications

As urbanization and development continues to alter the natural landscape, assessing impacts on stream ecosystems becomes increasingly important. The use of macroinvertebrates for indicating certain types of environmental conditions has been well established in the history of biomonitoring (Resh 1993). However, there is much uncertainty on how well they respond to specific pollutants and which metrics are effective for assessing impact. For example, Barbour et al. (1992) found two of eight benthic metrics to be highly variable and unreliable as measures of biological conditions. This study showed that the orders of EPT may not be good indicators of environmental pollution. A priori expectation was that there would be a strong quantitative relationship between sediment and EPT. Even though the relationship was weak most regressions between sediments and macroinvertebrates showed a similar trend of low level impact to fine sediments. Since several EPT metrics responded qualitatively to fine sediments and a weak quantitative trend can be seen with fine sediments and macroinvertebrates, there may be potential for using EPT for future studies, but precautions should be taken. Several recommendations should be considered for future sediment-macroinvertebrate field studies. It would be recommended to analyze data with and without the most ubiquitous taxa, because they

dominate the abundance which could confound the relationship. Another suggestion would be to increase the macroinvertebrate fixed subsample count from 300 in this study to a minimum of 500 to 1000 organisms during subsampling. A higher minimum count should increase the number of sensitive forms and increase the total number of taxa encountered in a sample (taxa richness), which will provide a more accurate measure of taxa richness (Sovell and Vondracek 1999). Since macroinvertebrate and sediment distribution are known to be highly patchy in a stream, an accurate characterization of the stream may require a high number of samples. Therefore, future studies should collect as many samples as possible (>10) per study unit to offset macroinvertebrate and sediment patchiness. Additionally, a stronger correlation between sediment and macroinvertebrates might be gained by collecting sediment measurements and macroinvertebrate measurements from the same direct patch in the stream. These recommendations may strengthen the quantitative relationship between macroinvertebrates and fine sediments giving managers data they could use to address sediment pollution issues.

Conclusions

The data clearly demonstrated that urban land use may have significant effects on the macroinvertebrate assemblages in urban streams. Urban land use, characterized by high amounts of impervious surface area, may be associated with excessive fine sediments in streams. Excessive fine sediments in streams can result in a homogenous substrate environment, which may result in a change of the macroinvertebrate assemblages from pollution sensitive forms such as EPT to more sediment/silt tolerant

forms such as chironomids, snails, and oligochaetes (worms). However, the relationship between fine sediments and macroinvertebrate community composition was difficult to quantify.

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