IMPACTS OF TIMBER HARVESTING ON STREAM MACROINVERTEBRATE COMMUNITIES AT DIFFERENT SPATIAL SCALES IN ONTARIO'S BOREAL FOREST

BY
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ABSTRACT

Challen, L.D. 2001. Impacts of Timber Harvesting on Stream Macroinvertebrate Communities at Different Spatial Scales in Ontario's Boreal Forest.
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Macroinvertebrates were collected from fifty-six stream segments (study sites). In addition, collections were gathered from nine streams yearly and from five streams repeatedly throughout one season. All stream segments or study sites were classified according to catchment size and to disturbance adjacent to the stream (forested, harvested or burnt). The study had two components. The first component determined the effectiveness of macroinvertebrate based indices for monitoring the effects of timber harvesting on stream habitat in Ontario's boreal forest. The yearly and seasonal variability (coefficient of variation (cv)) of various biomonitoring metrics were determined from yearly and seasonal data sets. Seasonal and year to year variability was high in all metrics except Percent Dominance, Percent Model Affinity and Percent Diptera. To assess the sensitivity of various metrics to impairment, metric values from disturbed sites were compared with values from reference sites and established scoring criteria. When considered as a group, harvested sites did not differ from reference sites for individual metrics. However, when sites were considered individually, impairment was detected at some harvested locations. To assess the influence of stream size on the various biomonitoring metrics two-way analysis of variance was used. All metrics differed among catchment areas. Similarly, classification of reference communities by catchment size reduced the standard error of metrics in many cases.

The second component of this study used more detailed multivariate and taxonomic analyses to investigate harvesting impacts. Several specific questions were addressed. Firstly, do stream size and disturbance (timber harvesting or burning) influence macroinvertebrate community structure and physical habitat (discharge, temperature, closed cover and substrate profile)? Secondly, do the aforementioned physical variables influence macroinvertebrate community structure? Finally, is the magnitude of a disturbance impact dependent upon the size of the stream and type of disturbance? A relationship between macroinvertebrate community and disturbance adjacent to the stream (forested, harvested or burnt) was not detected. Invertebrate community structure was correlated to the physical habitat variables measured. Analyses detected variation in macroinvertebrate community and habitat structure in relation to catchment area. The variations observed in the macroinvertebrate community in relation to stream size were similar to macroinvertebrate communities influenced by size selective predation patterns in brook trout (Bechara et al. 1992).

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1.0 GENERAL INTRODUCTION

Aquatic macroinvertebrates are a diverse group of organisms, dependent on an aquatic environment at some stage of their life cycle. The prefix "macro" refers to those invertebrates that are visible to the unassisted eye. The group includes many arthropod taxa (Arachnida, Crustacea and Insecta) as well as nonarthropod groups (i.e. Porifera, Hydrozoa, Nematoda, Tricladida, Annellida and Mollusca etc.). Aquatic invertebrates serve as a trophic link between detritus, microorganisms, algae, plants and the higher trophic level vertebrates. In fact, macroinvertebrates are an important food source for most commercial and sport fish species. Aquatic macroinvertebrates have a diverse range of habitat requirements, methods of locomotion and feeding habits. Their distribution patterns may vary by season and from year to year. In stream environments, the temperature of the stream, hydrology, substrate profile, channel morphology, source of primary productivity, predation, colonization and the macroinvertebrate community upstream form a framework upon which the aquatic macroinvertebrate community organizes itself (Merrit and Cummins 1996a).

There is evidence that the removal of terrestrial vegetation and the physical disruption associated with timber harvesting can alter the physical habitat of streams. Changes following timber harvesting have been observed in substrate profile (Newbold *et al.* 1980, Haupt and Kidd 1965, Klock 1985, McCurk and Fong 1995, Vurori and Jeonsuu 1996, Likens *et al.* 1970), stream hydrology (Garman and Moring 1991, Klock 1985, Webster 1990) nutrient input (Garman and Moring 1991; Webster 1990, Feller 1981), instream water temperature (McGurk and Fong 1995, Likens *et al.* 1970, Klock 1985) and turbidity (Likens *et al.* 1970, Vuori and Joensuu 1996, Cornish 1982). Shifts in species diversity (Newbold *et al.* 1980), dominance (Newbold *et al.* 1980, Gurtz and Wallace 1984, Garman and Moring 1993), functional feeding group proportion (Gurtz and Wallace 1984), density (Giroux 1994, Gurtz and Wallace 1984), biomass (Fuchs 1999, Giroux 1994) and Family richness (Giroux 1994) following timber harvesting have been observed.

The removal of stream side vegetation has been shown to decrease slope stability after the root system has undergone decomposition (Klock 1985). Erosion can disrupt or smother macroinvertebrate habitat depending on the current velocity. A fast current can suspend large particles which have entered a stream through stream side erosion (Rabeni and Minshall 1977). Suspended particles scour the stream bed altering channel stability and the morphology of the stream (Newbold *et al.* 1980). The sliding and bouncing of suspended sediment along the benthic surface has caused physical stress to macroinvertebrates and shearing stress to the aquatic plants they feed on (Culp *et al.*1986, Vuori and Joensuu 1996). Suspended particles also cause turbidity, reducing instream primary productivity upon which many macroinvertebrates depend (Newbold *et al.* 1980).

A slower moving stream will likely experience more sediment deposition after timber harvesting. Siltation and sedimentation can smother habitat (e.g. algae on rocks and interstitial spaces), restricting the mobility of many insects and changing the type and amount of suitable habitat patches (Minshall *et al.* 1983, Gurtz and Wallace 1984, Vuori and Joensuu 1996). Decreased benthic invertebrate density and a shift in species composition as a result of sedimentation and loss of interstitial spaces have been observed following harvest (Newbold *et al.* 1980, Vuori and Joensuu 1996).

Increased surface water runoff has been attributed to the removal of terrestrial vegetation, which when present, modifies runoff through uptake and evapotranspiration (Likens *et al.* 1970). Increased terrestrial runoff has increased water velocity and discharge in streams, carrying away benthic invertebrates not tolerant to fast currents (Rabeni and Minshall 1977).

Terrestrial runoff has increased nutrient leaching from the soil and raised nutrient input into water, promoting a switch from autochthonous to allochthonous primary production (Likens *et al.* 1970). Also, increased water velocity may export organic matter faster. Increased nutrient concentrations and organic matter export disturb the nutrient processing system of the stream community, which may require an adjustment in the benthic macroinvertebrate community feeding structure (Vannote *et al.* 1980).

Removal of riparian vegetation reduces shading effects and increases the influence of solar energy on the stream (Garman and Moring 1991). These changes can increase instream primary productivity, resulting in shifts in invertebrate feeding types (Garman and Moring 1991). Water temperature increases disrupt those species with metabolic optimums at lower temperatures (Vannote *et al.* 1980). Decomposition speeds up at higher temperatures (Garman and Moring 1991), altering nutrient processing even at the macroinvertebrate trophic level.

Stream macroinvertebrate communities are often effectively used for the assessment of impacts caused by point source pollution of local water quality (Merrit and Cummins 1996a). Determining if biomonitoring metrics commonly used for point source pollution are also useful for detecting stream impairment due to timber harvesting is one question I addressed in this thesis. The potential impacts of catchment scale disturbances, such as timber harvesting, on stream habitat and the macroinvertebrate community are much more varied than point source pollution. It is unknown whether or not macroinvertebrate communities respond to timber harvesting practices in predictable ways, which are distinguishable from natural variability. Furthermore, whether or not a macroinvertebrate biomonitoring program can detect impairment following timber harvesting, has not been established.

Immense natural variability in the macroinvertebrate community structure (i.e. taxa density and taxa present) may limit the utility of aquatic macroinvertebrates in stream biomonitoring programs because of the difficulty involved in characterizing healthy community attributes. Natural variability among communities may prevent the detection of impairment by reducing the probability of detecting changes associated with harvesting. Taxonomic and density variability among streams and over time can be partitioned to ensure that "treatment" effects can be isolated from other influences. Partitioning can include characterization of ecoregion (Plafkin *et al.* 1989), season (Keup

1988, Hilsenhoff 1982), year (Kerans *et al.* 1992), physical habitat (substrate profile, temperature, velocity, closed cover) (Richards *et al.* 1993) and stream size (Vannote *et al.* 1980).

Yearly differences in macroinvertebrate communities are due to weather (factors such as temperature and precipitation) and other natural events. Life cycles of individuals in the same species may vary from one to three years depending on food availability and environmental conditions of those years (Kerans *et al.* 1992). This could lead to different community structure, depending on the year of sampling. Water level, another variable that changes from year to year and is related to climate, also contributes to variability in community structure (Cowx *et al.* 1984). Confining sampling to one year is an effective way to control this type of variability. Alternatively, annual visits to selected sites could provide an estimation of yearly variability in community structure within a stream from which to calibrate other samples collected from a similar climatic region.

Time of emergence, hatching and cohort age also differ from season to season (Keup 1988), which results in variation in community composition. For example, diversity and density in invertebrate communities peak in the fall, and are lowest in the summer when some species are in diapause (Keup 1988, Hilsenhoff 1982). Collection of macroinvertebrates within the shortest interval possible and within one season, is the most effective way to account for seasonal variation. Calibration of seasonal variation by monitoring sites throughout the year may also be effective.

Ecoregions represent areas similar to each other with respect to geology, forest cover, agricultural potential, wetlands and waterbodies (Plafkin *et al.* 1989). Macroinvertebrate communities are expected to differ among ecoregions, but be relatively similar within an ecoregion (Lenat 1993). Biomonitoring efforts may be improved by comparing macroinvertebrate communities at test sites to those found in undisturbed reference streams similar in physical characteristics (Hughes *et al.* 1986, Plafkin *et al.* 1989, Hughes 1995, Glasby 1997).

The large scale characteristics of a stream catchment such as dominant bedrock (Richards and Host 1994), amount of wetland in the catchment, terrestrial vegetation (Frissel *et al.* 1986), riparian vegetation and surficial geology (Richards *et al.* 1996) influence smaller scale characteristics within the stream such as instream substrate profile, nutrient regime, hydrology and stream morphometry. Thus these large-scale characteristics may also add to the variability among streams and macroinvertebrate communities. Physical habitat, which is tremendously variable among streams, may also account for variation in macroinvertebrate community composition. For example, the amount of canopy closure can influence shredder and detritivore numbers by controlling the quality and abundance of detrital inputs (Egglishaw 1964, Molles 1982, Corkum 1989). Macroinvertebrate community composition may differ among areas with different substrate patterns (Richards *et al.* 1993, Minshall *et al.* 1984). Attempts to partition variation in macroinvertebrate communities are often made by characterizing habitat, and confining sampling to stream sections with similar habitat characteristics (e.g. riffles, pools or runs).

The River Continuum Concept (RCC) (Vannote *et al.* 1980) predicts that both the physical habitat and the macroinvertebrate community of a stream change from the headwater to the mouth of a river system. Since the conception of the RCC, direct relationships between stream size and both physical habitat (stream temperature, discharge and gradient) (Rankin 2000, Richards *et al.* 1996) and invertebrate community have been observed (Richards and Host 1994, Richards and Minshall 1992). Within the Salmon River, Idaho drainage macroinvertebrate functional groups and species richness differed from small to large streams (Bruns and Minshall 1985, Bruns *et al.* 1982, Bruns *et al.* 1987). Macroinvertebrate assemblages were distinct even between the narrow range of 2^{nd} to 4^{th} order streams (Bruns and Minshall 1985, Richards *et al.* 1996). Thus, characterization of a stream's size (order or catchment area) may be an effective method of partitioning variation in macroinvertebrate communities and physical habitat.

This study had two components. In the first, I determined the effectiveness of macroinvertebrate-based indices for monitoring the effects of timber harvesting on stream habitat in Ontario's boreal forest. The yearly and seasonal variability in various biomonitoring metrics was determined from yearly and seasonal data sets. To assess the sensitivity of various metrics to impairment, I compared metric values between disturbed sites and reference sites and established scoring criteria from the literature. The influence of stream size on the various biomonitoring metrics was also assessed.

The second component of this study used multivariate analyses to complement the findings of the biomonitoring chapter. I addressed three specific questions. First, does stream size and disturbance (timber harvesting or burning) influence macroinvertebrate community structure and physical habitat (discharge, temperature, closed cover and substrate profile)? Second, do the aforementioned physical variables influence macroinvertebrate community structure? Third, is the magnitude of a disturbance impact dependent upon the size of the stream and type of disturbance?

2.0 GENERAL METHODS

2.1 Study Area and Site Classification

The study area included the Wolf River (743.3 km^2), Mackenzie River (369.6 km^2), Spruce River (1063.9 km^2) and Nipigon 10 Burn Zone Watersheds (Figure 2.1). These watersheds are tributaries of Lake Superior and lie within the Northwestern Ontario Boreal forest, near Thunder Bay, Ontario.

The study area lies in the Nipigon Plain Ecoregion (Wickware and Rubec 1989). Both undifferentiated igneous and metamorphic bedrock dominate the region. These types of rock are not very permeable and tend to form aquitards, permitting the storage of groundwater (Freeze and Cherry 1979, Picard 1995). Groundwater promotes stable hydrologic and thermal conditions in streams. The dominant surficial landforms are ground moraines and sandy glaciolacustrine plains. Ground moraines form a discontinuous mantle of silty to sandy till on bedrock. Typically the till is one metre thick, with large amounts of stone, boulder and gravel (Mollard 1979).

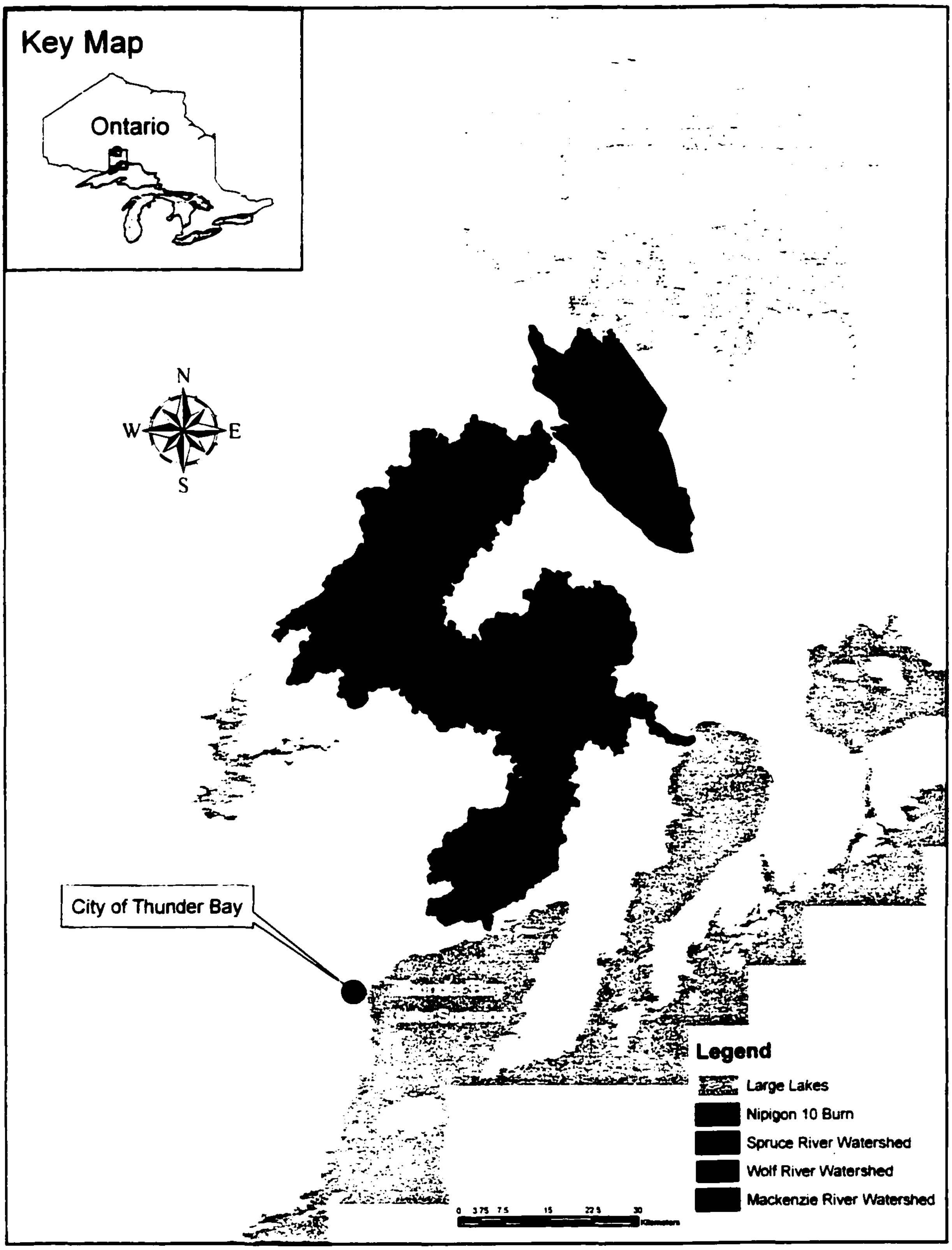


Figure 2.1 *Map of the north shore of Lake Superior and the watershed areas involved in the study*

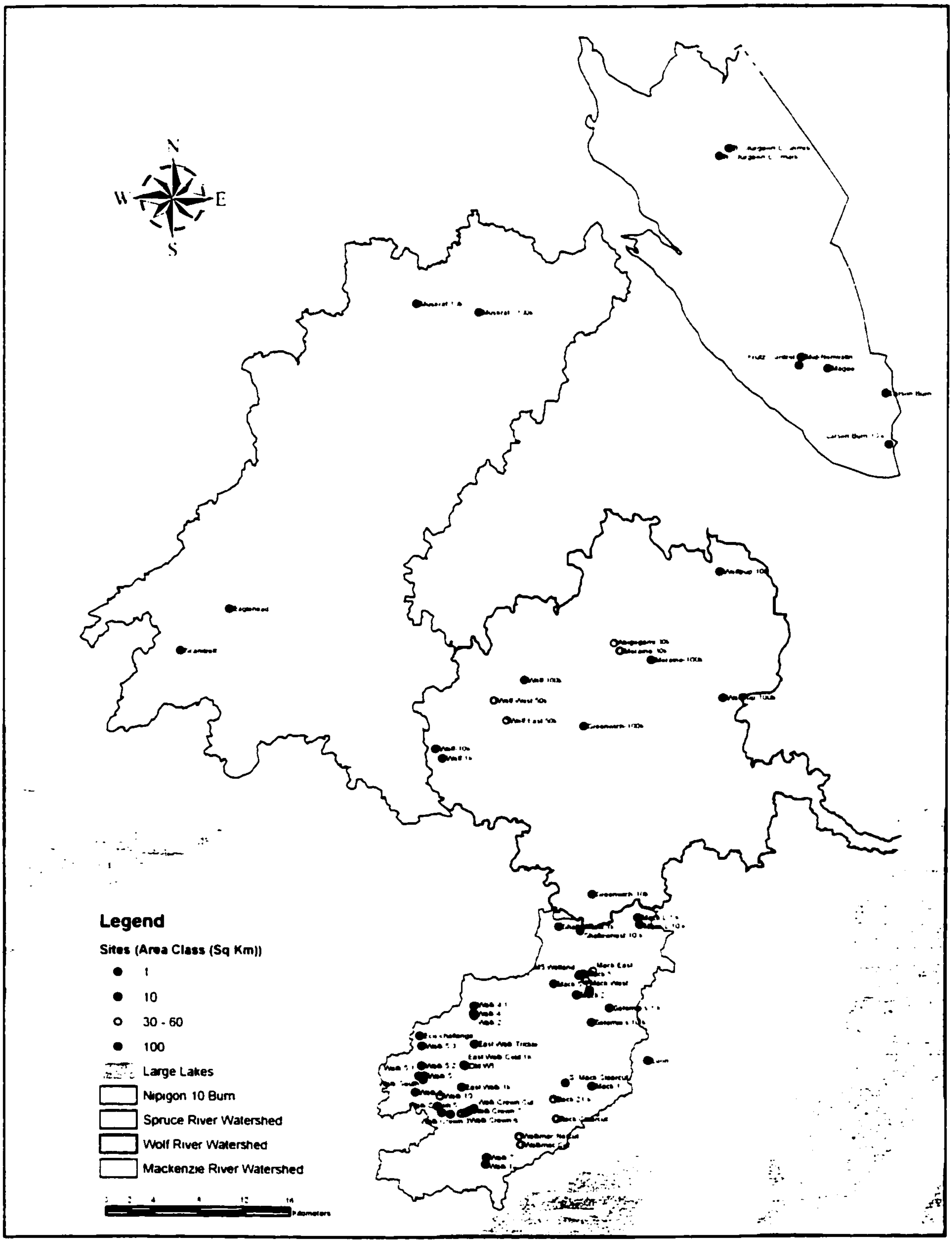


Figure 2.2 Map of the locations of sites used in this study

Fifty-six stream reaches within the study area were established as sites (Figure 2.2). Sites were categorized based on catchment area and the type of disturbance in the catchment. Catchment area was treated as a categorical variable in order to reduce the sample size requirements of the study. The four catchment area size classes(Table 2.1) were: 1 km^2 (0.8 to 2.1 km^2), 10 km^2 (4.3-13.7 km^2), 40 km^2 (21.0 –64.7 km^2) and 100 km^2 (69.1 – 115.1 km^2). Catchment delineation for each study site used a raster-based digital elevation model (DEM) generated through the use of the Environmental Systems Research Institutes (ESRI) geographic information systems (GIS) software packages ARC/INFO and ArcView (Figure 2.3).

Sites were also categorized based on the disturbance adjacent to the stream. Disturbance class ("cut", "burnt" or "uncut") was defined by the condition of the area adjacent to the site (Table 2.1). "Cut" or "burnt" categorization required disturbance on at least one side of the stream. Often the area adjacent to a cut site included a reserve area. Reserve areas (buffer strips) are areas left uncut adjacent to streams with widths ranging from 30 to 90m required by and based on "Timber Management Guidelines for the Protection of Fish Habitat" (OMNR 1988).

2.2 Benthic Invertebrate Collection and Identification

All macroinvertebrate samples were collected in riffle areas between May and September at base flow condition. Sampling was done within one ecoregion to control for broad scale geographic variability. The macroinvertebrate collection apparatus included a

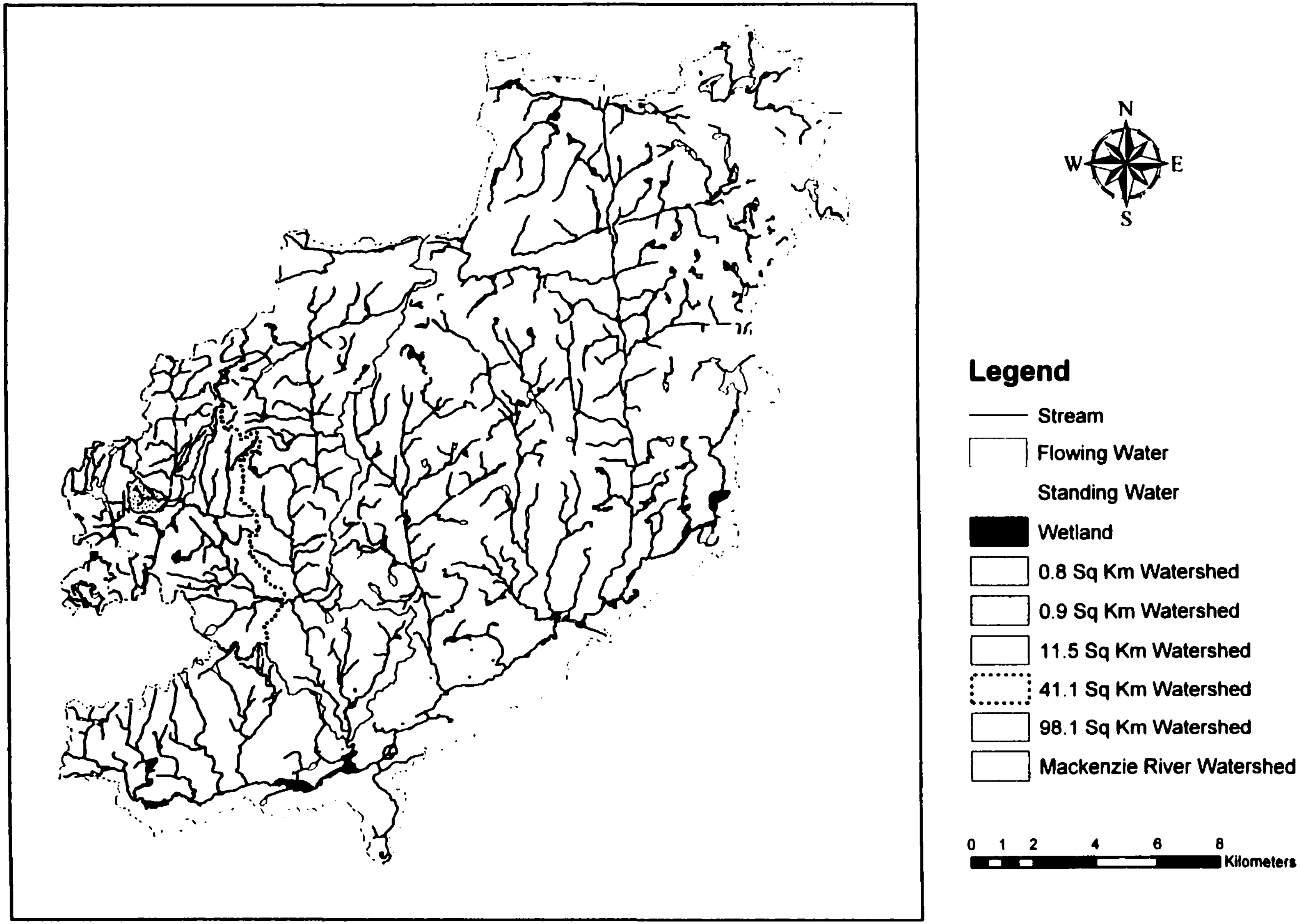


***Figure 2.3** A GIS generated map showing catchment delineation for study sites within the Mackenzie River Watershed*

***Table 2.1** Number of sites sampled within each catchment area and disturbance type for the 56 sites visited*

	1 km^2	10 km^2	40 km^2	100 km^2	Total
Uncut	9	10	7	3	29
Cut	4	7	4	5	20
Burnt	2	5	0	0	7
Total	15	22	11	8	56

0.09 m^2 quadrat and a D-shaped dip net. Using a nailbrush, the collector dislodged invertebrates from substrate lying within the quadrat while stream flow carried the invertebrates into the D-net placed directly downstream. This collection method was convenient (light weight and compact) and appropriate for Northwestern Ontario streams which are characterized by course substrate and are often in remote locations. All invertebrates were transferred to Nalgene sample bottles and preserved in 75% ethyl alcohol until processing. Three macroinvertebrate samples were collected from each study site and treated as subsamples in the analysis.

In the laboratory, macroinvertebrates were identified and enumerated with the aid of a dissection microscope. Macroinvertebrates were identified into taxonomic groups and functional feeding groups according to Merritt and Cummins (1996a, b) and Clifford (1991). Important taxonomic information may still be gained from this collection; therefore, biomass information, which destroys the specimens, was not determined.

2.3 Transformations and Data Reduction

SPSS (version 9) software was used for all statistical analyses. A probability of $P<0.05$ determined significance in all statistical tests. Some of the variables required transformations to satisfy the normality assumptions of the analyses used. All of the variables within the functional feeding group data set, the taxonomic group data set, and the substrate data set were log transformed ($\log_{10}(x+1)$). Temperature and discharge could not be normalized; therefore they were analyzed with non-parametric tests.

I examined the Pearson Product moment correlation matrices for taxonomic group, functional feeding group and substrate. No variables in any matrix were highly collinear and therefore, no variables were considered redundant within the matrices.

The original taxonomic group matrix consisted of 30 invertebrate taxa. If less than 10 individuals in a taxonomic group existed in the entire data set, the group was considered rare (Somers *et al.* 2002) and removed. The final data set had 18 taxonomic groups.

Whenever the MANOVA detected a significant effect, I used a Discriminant Function Analysis (DFA) to identify variables important in differentiating among the groups. The DFA was interpreted as an exploratory tool, not as a significance test.

3.0 INTRODUCTION – BIOMONITORING

Macroinvertebrates lend themselves well to monitoring aquatic habitat integrity for a number of reasons. Macroinvertebrates are always present in permanent stream habitats. Within the macroinvertebrate community a range of responses to environmental stress are exhibited. For example, some taxonomic groups (e.g. Trichoptera, Ephemeroptera, Plecoptera) are know to be sensitive to organic pollution (Hilsenhoff 1982), while other (e.g. Chironomidae) are comparatively tolerant of such conditions (Plafkin *et al.* 1989). Macroinvertebrates have relatively small habitat ranges, allowing accurate local diagnosis of disturbance impacts. Their distribution reflects both biotic and abiotic factors present in the stream (McGurk and Fong 1995, Vuori and Joensuu 1996, Garman and Moring 1993), and as such, can provide insight into the condition of these factors. Laboratory water quality analysis provide a point in time snapshot of a stream's condition. In comparison, the macroinvertebrate community assemblage in a stream reflects current as well as historical conditions of a stream. Macroinvertebrate community patterns should be predictable (Vannote *et al.* 1980) because they capitalize on upstream processing inefficiencies and direct organic inputs. As such, the macroinvertebrate assemblage expected in a healthy stream can be compared to the actual assemblage present. For these reasons, aquatic macroinvertebrates may be useful when assessing impacts of landuse on stream environments.

Although there are many inherent characteristics of macroinvertebrate communities that lend themselves to biomonitoring, these communities are also highly variable in both species composition and the proportions of individual taxa in a stream. Stream macroinvertebrate community variability is associated with a large number of factors including season (Keup 1988, Hilsenhoff 1982), year (Kerans *et al.* 1992), ecoregion (Plafkin *et al.* 1989), habitat (substrate profile, temperature, velocity, closed cover) (Richards *et al.* 1993), size of the stream (Vannote *et al.* 1980, Frissel *et al.* 1986) and behavior (feeding, colonization, locomotion, vulnerability to predation, life cycle). For these reasons, it is often necessary to control natural variability in a biomonitoring program in order to detect disturbance signals.

A description of reference communities by region has been suggested as one means of accounting for natural variability in macroinvertebrate communities (Plafkin *et al.* 1989). Macroinvertebrate metric values from disturbed streams are compared to metric values found in undisturbed streams from the same region. However, in order to utilize biomonitoring techniques within a region such as Northwestern Ontario, other sources of natural variability in macroinvertebrate communities need to be assessed. Macroinvertebrate abundance and distribution may vary seasonally and yearly. For example, Hilsenhoff (1982) has found that his biotic index values are falsely high in summer, when many species are in diapause. Likewise, tolerance values require adjustment for seasonal differences in Southeastern United States (Lenat 1993). Yearly variability due to variation in climate and natural events is difficult to control for in biomonitoring programs. Nonetheless, an understanding of the potential range of this type of variability seems prudent.

Stream size is another potential source of variability in stream invertebrate communities. Shifts in invertebrate community composition along the length of a river are predicted by the River Continuum Concept (Vannote *et al.* 1980). Differences in invertebrate community structure along a stream could result in false diagnosis when using biomonitoring metrics. For example, headwater streams fed by nutrient poor groundwater are naturally low in productivity and this is often reflected by low diversity and richness metric values (Pinder and Farr 1977, Plafkin *et al.* 1989). Ironically, these conditions are usually associated with poor water quality. Thus pristine headwater streams with low diversity could be misdiagnosed as impaired if its naturally low productivity is not accounted for in the original calculations.

Despite the many sources of variability, stream macroinvertebrate communities are often effectively used for the assessment of impacts caused by point source pollution on local water quality. However, I am interested in using stream dwelling macroinvertebrates for monitoring of the potential impacts of timber harvesting on stream habitat. Timber harvesting may induce impacts much more varied than point source pollution including changes in: substrate profile (Newbold *et al.* 1980, Haupt and Kidd 1965, Klock 1985, McGurk and Fong 1995, Vurori and Joensuu 1996, Likens *et al.* 1970), nutrient input (Garman and Moring 1991, Webster 1990, Feller 1981), stream hydrology (Garman and Moring 1991, Klock 1985, Webster 1990), instream water temperature (McGurk and Fong 1995, Likens *et al.* 1970, Klock 1985) and turbidity (Likens *et al.* 1970, Vurori and Joensuu 1996, Cornish 1982). A question that I address is if the biomonitoring metrics commonly used to monitor point source pollution are also useful for monitoring the many impacts associated with timber harvesting.

Family Biotic Index (Hilsenhoff 1982), functional feeding group ratios, dominance, diversity, abundance measures, Percent Model Affinity (Novak and Bode 1992) and Rapid Bioassessment Protocol (RBP) II (Plafkin *et al.* 1989) are some of the indices developed for biomonitoring. Table 3.1 provides summary descriptions of these indices and their predicted shift in degraded settings. The variability of a metric will be one measure of usefulness in this study. Likely, highly variable metrics would not be useful for detecting disturbance because the impacted stream values would be less likely to fall outside the naturally broad range. To assess the natural variability of a metric, a description of the metric's range in a sample of the population can be used. Another way to assess metrics is by testing their ability to discriminate between a "known" healthy stream and a "known" impaired stream. If metric value distributions differ between these two known conditions, then the metric can be considered a valuable biomonitoring tool.

My exploration of the suitability of macroinvertebrate-based biomonitoring in Northwestern Ontario was guided by two objectives. The first objective was to assess and identify sources of natural variability in the macroinvertebrate community. Potential sources of variability such as season, year and catchment area were examined. This information could help partition or control variability in prospective biomonitoring efforts. The second objective was to compare the utility of various existing metrics for monitoring effects of timber harvesting in Northwestern Ontario's Boreal Forest streams.

Table 3.1 Explanations for macroinvertebrate metrics used and their expected shift with habitat degradation.

Macroinvertebrate Metric	Explanation	Expected shift with Habitat Degradation
Taxa Richness	Total number of different taxa in a sample.	Decrease (Plafkin *et al.* 1989)
Diversity	A measure of taxa richness and evenness. Low diversity indicates that taxa are less evenly distributed. $H' = \Sigma p_i \log p_i$ p_i represents the proportion of the total number of individuals in the i^{th} taxonomic group	Decrease
Total Density	Total number of organisms in a sample	Decrease
EPT Richness	Total number of Ephemeroptera, Plecoptera and Trichoptera taxa in a sample (taxa that are considered sensitive to water quality degradation)	Decrease (Plafkin *et al.* 1989)
EPT Abundance	Number of Ephemeroptera, Plecoptera, and Trichoptera in a sample.	Decrease
Percent Dominant Taxa	Proportion of the entire community composed of the most abundant taxa.	Increase (Plafkin *et al.* 1989)
Hilsenhoff's Biotic Index (HBI or FBI)	Developed to detect organic pollution based on the indicator organism approach to water quality. Values on a scale of 0 to 10. $FBI = 1/N\ \Sigma n_i t_i$ N = the total number of individuals in the sample n_i = the number of individuals in a family t_i = the tolerance score for that family	Increase (Hilsenhoff 1982)
EPT:Chironomidae ratio	The ratio of EPT to Chironomidae If the ratio is < 1.0, the community has a disproportionate number of Chironomids relative to the more pollution-sensitive EPT taxa; Chironomids tend to increase with increasing organic enrichment (Ferringon, 1987)	Decrease (Hilsenhoff 1982)
Composite Normalized Metric	An unweighted index of biointegrity (individual stream scores/max. score for all streams) = taxa richness + total abundaance + EPT taxa + EPT abundance + EPT chironomidae + % dominant taxa + scrapers:total organisms + shredder:filtering collectors	Increase (Rothrock *et al.* 1998)
Scraper:Filtering Collectors	Shifts in functional feeding group composition indicate an overabundance of a particular food source. Scrapers increase with increasing diatoms. Filterers increase with increasing filamentous algae and aquatic mosses, but are sensitive to toxins.	Decrease (Rothrock *et al.* 1998)
Shredders:Total Organisms	Shredders are sensitive to riparian zone impacts and also can be indicators of toxins.	Decrease (Rothrock *et al.* 1998)
Percent Model Affinity	An index of macroinvertebrate community composition used to measure the affinity of a community to that of the expected, ideal community. $\%\ \text{Model Affinity} = 100 - 0.5\ \Sigma\lvert a - b \rvert$ a = the percentage of individuals of a taxon in sample A, which is the model community value for a taxon b = the percentage of the same taxon in sample B	Decrease (Novak and Bode 1992)

The effectiveness of Hilsenhoff's Biotic Index, diversity, richness, ratios, compilation metrics and similarity indices were compared. The variability of a metric and its ability to distinguish between disturbed and reference communities were the criteria used.

3.1 METHODS - BIOMONITORING

3.1.1 Invertebrate Data Set Description

Survey information collected in the study was summarized in four data sets (Table 3.2). These were the "taxonomic", "functional feeding group", "seasonal" and "yearly" data sets. Taxonomic or functional feeding group abundance information was used in the data sets. The sites included in the seasonal and year to year data sets were sampled on a number of different dates (Tables 3.3 and 3.4).

3.1.2 Natural Variability

3.1.2.1 Seasonal and Yearly Variability

Percent dominance, % Diptera abundance, EPT/Chironomidae ratio, Total Density, Percent Model Affinity, and taxonomic group densities were calculated from the taxonomic group, seasonal and the yearly data sets. The coefficient of variation ((cv = standard deviation/mean)*100) measured the seasonal and yearly variability in each metric, within each individual site. To assess the magnitude of the variability, I compared within site coefficients to between site coefficients generated from the taxonomic group matrix. Within site variability of the metrics was considered low if the metric's coefficient of variation was under 50% (Barbour *et al.* 1992).

Table 3.2 *Taxonomic descriptions of the four data sets used to calculate biomonitoring metrics.*

Data Set	Visits/Site	Taxonomic Description
"Taxonomic"	56 sites, one visit	• Diptera identified to Family • Other Insects to Order • Non-insect groups
"Family"	30 sites, one visit	• Insects identified to Family • Non-insect groups
"Seasonal"	5 sites, 4-6 visits per site (Table 3.3)	• Diptera identified to Family • Other Insects to Order • Non-insect groups
"Yearly"	9 sites, 2-3 visits per site (Table 3.4)	• Diptera identified to Family • Other Insects to Order • Non-insect groups

Table 3.3 *Sites belonging to the "Seasonal" data set and the dates they were sampled*

Site Name	Catchment Area Class	Type of Disturbance	Sampling Dates (day/month/year)
Walk 5.1 1K	1 km^2	Unharvested	22/06/99, 07/07/99, 26/07/99, 09/08/99, 23/08/99, 20/09/99
Walk 5.1 10K	10 km^2	Unharvested	17/06/99, 06/07/99, 27/07/99, 05/08/99, 18/08/99, 20/09/99
Settimio 1K	1 km^2	Unharvested	18/06/99, 21/07/99, 11/07/99, 26/08/99
Settimio 10K	10 km^2	Unharvested	14/06/99, 30/06/99, 20/07/99, 03/08/99, 27/08/99, 24/09/99
Settimio Clearcut	1 km^2	Harvested	15/06/99, 01/07/99, 22/07/99, 04/08/99, 24/08/99, 28/08/99

Table 3.4 Sites belonging to the "Yearly" data set, and dates they were sampled.

Site Name	Catchment Area Class	Type of Disturbance	Sample Dates (day/month/year)
W 5.1	1 km^2	Unharvested	29/07/97, 26/07/99
M2	10 km^2	Harvested	26/07/95, 04/07/96, 15/07/97
W5	10 km^2	Harvested	22/08/95, 17/07/97, 08/05/98,
W2	10 km^2	Harvested	18/07/95, 08/08/97
Walkinshaw South	10 km^2	Harvested	21/05/98, 08/06/99
Walkinshaw North	10 km^2	Harvested	20/08/98, 08/06/99
East Loon	10 km^2	Unharvested	31/07/95, 17/06/96, 05/06/98,
Abigogami	40 km^2	Harvested	09/06/95, 14/08/98
W1	100 km^2	Unharvested	14/07/95, 01/08/97

3.1.2.2 Stream Size Variability (Catchment Area)

To determine whether or not metric values vary with the size of a stream's catchment, the following metrics were calculated from the Family data set: Shannon-Weiner diversity index, Family Biotic Index, Composite Normalized Metric, Taxa Richness, EPT/Chironomidae abundance, EPT taxa richness, Percent Dominance, Filterer/Scraper and Shredder/Total abundance, Percent Model Affinity and RBP II. To determine if these metric values were different for small streams compared to large streams, two-way ANOVA was used. The data set included both harvested and forested sites. To control for harvesting effects on the metrics, disturbance type was included as the second factor in the analysis. To complement this information, individual site metric values were compared to published scoring criteria for individual metrics to determine how healthy these sites were considered (Tables 3.5 and 3.6). This was done by comparing harvested streams with uncut reference streams from all stream sizes and then comparing harvested streams with only uncut reference streams of a specific catchment size.

Both Percent Model Affinity and RBP II score are measures of the similarity of the stream in question to the values expected at a healthy stream (reference condition). Therefore, to determine the association between catchment area and these two metrics, several different reference conditions were calculated. An "all stream size" reference was determined by calculating the expected condition from all sizes of "forested" streams in the data set. Then reference conditions were calculated by including only streams belonging to a specific size class (1km^2, 10 km^2, 40 km^2, 100 km^2). The calculations of RBP and Percent Model Affinity scores for individual sites requires a reference (healthy

***Table 3.5** Criteria for characterization of biological condition (modified from Plafkin et al. 1989)*

Biological Condition Scoring Criteria			
Metric	**Non-impaired**	**Impaired**	**Severely Impaired**
Taxa Richness [a]	> 80%	40 – 80%	< 40%
Family Biotic Index [b]	> 85%	50 – 85%	< 50%
Scraper:Filterers [a]	> 50%	25 – 50%	< 25%
EPT/Chironomidae [a]	> 75%	25 – 75%	< 25%
% Dominance [a]	< 30%	30 – 50%	> 50%
EPT richness [a]	> 90%	70 – 90%	< 70%
Shredders/Total [a]	> 50%	25 – 50%	< 25%

(a) score is a ratio of study site to reference site x 100
(b) score is a ratio of reference site to study site x 100

***Table 3.6** Percent Model Affinity Scoring Assessment (Novak and Bode 1992)*

Similarity to the Model Community	Assessment
O65%	Non-impacted
50 – 64%	Slightly impacted
35 – 49%	Moderately impacted
< 35%	Severely impacted

condition) value in the equation. The reference values for RBP II and Percent Model Affinity were generated from the average of the uncut sites.

3.1.3 Comparison of Metrics

To determine whether or not metrics were sensitive to timber harvesting impacts, the following metrics were calculated from the Family data set: Shannon-Weiner diversity index, Family Biotic Index, Composite Normalized Metric, Taxa Richness, EPT/Chironomidae abundance, EPT taxa richness, Percent Dominance, Filterer/Scraper and Shredder/Total abundance, Percent Model Affinity and RBP II. To determine if these metric values were different in timber harvested streams compared to undisturbed streams, two-way ANOVA was used. The data set included both harvested and forested sites, but no burnt sites (Table 2.1). To control size effects on the metrics, catchment area was included as the second factor in the analysis. To complement this information, individual site metric values were compared to published scoring criteria for individual metrics (Tables 3.5 and 3.6). Reference values for RBP II and Percent Model Affinity were generated from the average of the uncut sites.

3.2 RESULTS – BIOMONITORING

3.2.1 Natural Variability

3.2.1.1 Seasonal Variability

Seasonal variability (cv) was calculated at five sites and assessed both in relation to the variability found among all 56 sites combined and according to the criteria of Barbour *et al.* (1992). Seasonal variability was high for most metrics and equaled or exceeded between site variability for one or more of the sites considered (Figure 3.1). Metrics that had low seasonal variability according to Barbour's criteria `(cv's < 50%) were Percent Dominance, Percent Model Affinity and Percent Dipterans. Seasonal variability for all other metrics was high (cv > 100%) for at least one of the sites considered, but on average, seasonal variability was less than between site variability in all cases. Figure 3.1 compares the seasonal variability of some of the metrics calculated.

3.2.1.2 Yearly Variability

Yearly variability (cv) was calculated for nine sites and assessed both in relation to the variability found among all 56 sites combined and according to the criteria of Barbour *et al.* (1992). Yearly variability for density (individuals/ m^2) metrics (Chironomidae, Ephemeroptera, Plecoptera, Trichoptera) was consistently lower than between site variability (Figure 3.2). In contrast, the yearly variability for other calculated metrics (Percent EPT, EPT/Chironomidae, Percent Dominance, Percent Diptera, Percent Model Affinity) equaled or exceeded between site variability (Figure 3.2). In general, all metrics except Percent Dominance and Percent Model Affinity had cv's consistently exceeding 50%, which is considered high (Barbour *et al.* 1992).

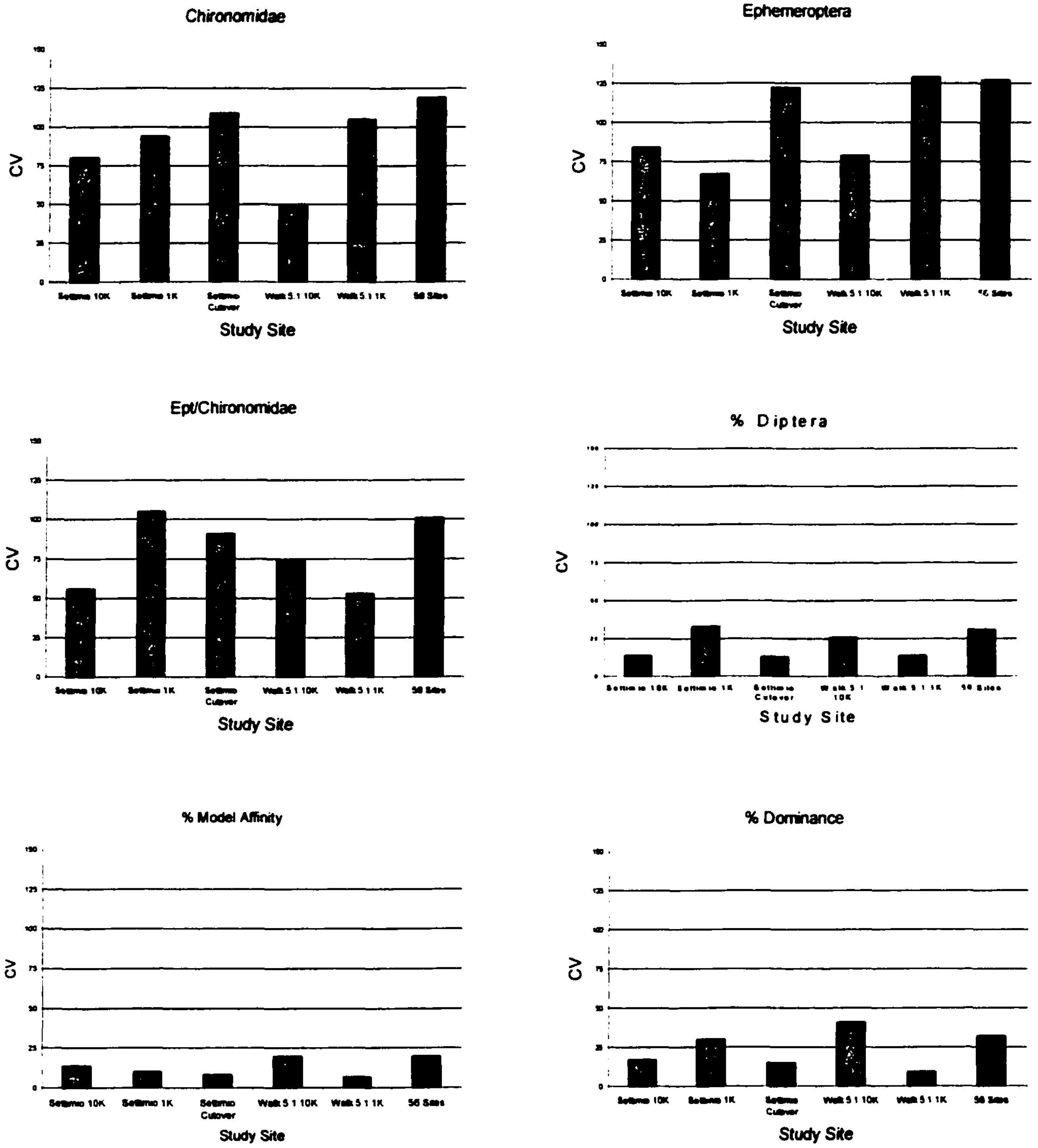


***Figure 3.1** Comparison of seasonal variability within site to between site variability for commonly used metrics (metric is designated above graph). Variability is represented by the coefficient of variation (cv).*

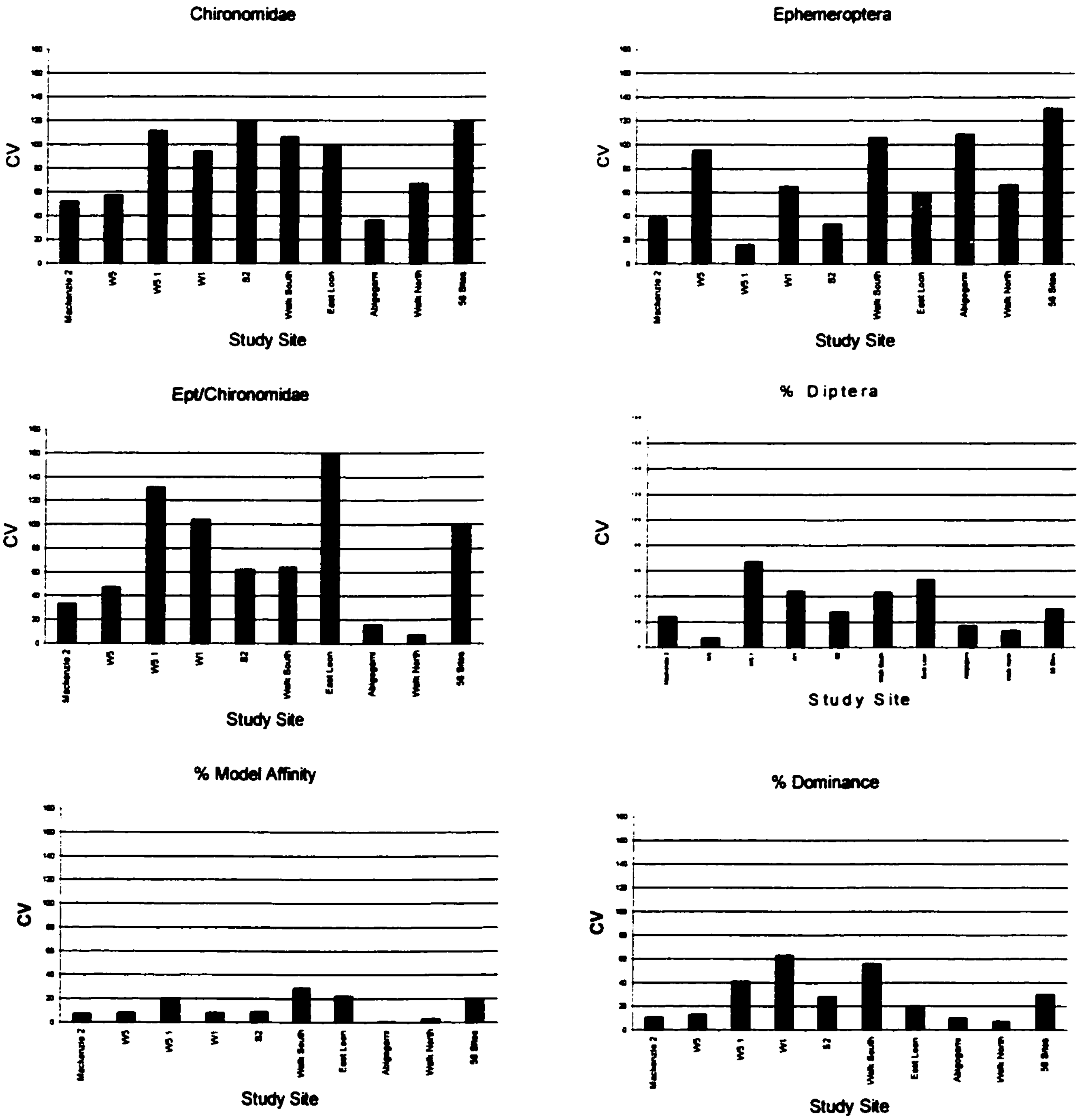


Figure 3.2 Comparison of yearly variability within site to between site variability for commonly used metrics (metric is designated above graph). Variability is represented by the coefficient of variation (cv).

3.2.1.3 Stream Size Variability (Catchment Area)

All indices differed among catchment size (Figure 3.3-3.10, Table 3.7). Although the direction of difference varied, in all cases 1 km^2 and 40 km^2 were the most different with 10 km^2 and 100 km^2 being intermediate.

Impairment assessment (Plafkin *et al.* 1989) for individual sites were sensitive to the size of the streams (Tables 3.8 and 3.9). EPT richness values indicated severe impairment at some sites when not calibrated to catchment size, but moderate impairment at the same sites when calibrated to catchment size. EPT/Chironomidae ratio diagnosed two 1 km^2 sites as severely impaired when not calibrated to catchment size, but no impairment when calibrated to catchment size. The Scraper/Filterer metric diagnosed ten sites as impaired when not calibrated to catchment area, and only six when calibrated to catchment area.

RBP II scores also appeared sensitive to the size of the stream. One km^2 site scores increased from 36% and 21% to 57% following the size calibration (Table 3.10 and 3.11). Percent Model Affinity scores for disturbed sites also appeared to be sensitive to the size of the stream. The scores for all sites improved slightly with calibration to catchment size, and Abigogami and W5 changed ratings from moderately impacted to slightly impacted when calibrated to catchment size (Table 3.12 and 3.13).

3.2.2 Comparison of Metrics

In general, when harvested sites were considered as a group they did not differ significantly from unharvested sites according to the metrics considered (Figure 3.3-3.10,

Table 3.7 Summary of the catchment area main effect two-way ANOVA results for each metric, their direction of difference with catchment area.(degrees of freedom were $F_{3,29}$ in all cases).

Metric	Direction of Difference	F Value	p
Taxa Richness	increased with catchment area	7.93	0.007
Family Biotic Index	decreased with catchment area	3.96	0.021
Composite Normalized Metric	increased with catchment size	6.53	0.003
Shannon-Weiner Diversity	increased with catchment size	14.79	0.0001
EPT/Chironomidae	increased with catchment area	5.42	0.006
Percent Dominance	decreased with catchment area	8.47	0.001
Scraper/Filterer ratio	increased with catchment area	4.00	0.021
EPT richness	increased with catchment area	21.02	< 0.0001

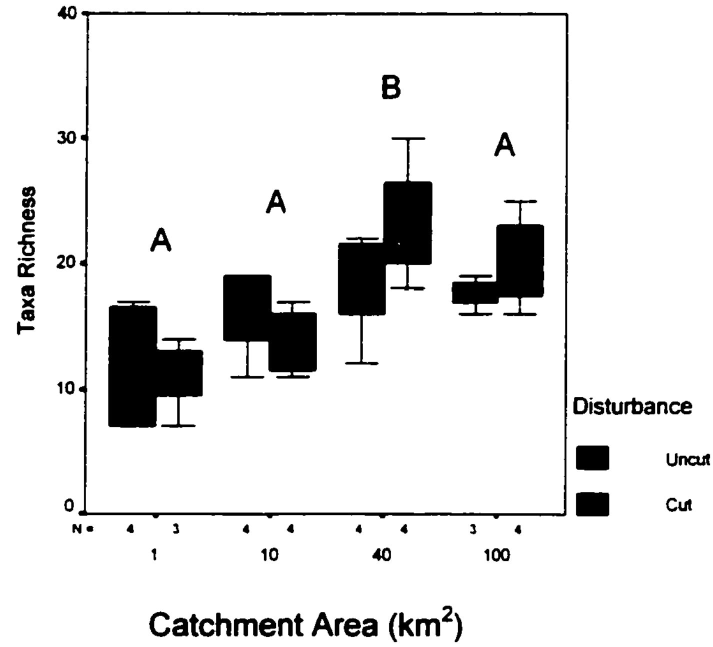


***Figure 3.3** Taxa richness mean and variability for each catchment area and disturbance type. Different letters indicate catchment area classes that differ significantly (Tukey HSD, α = 0.05).*

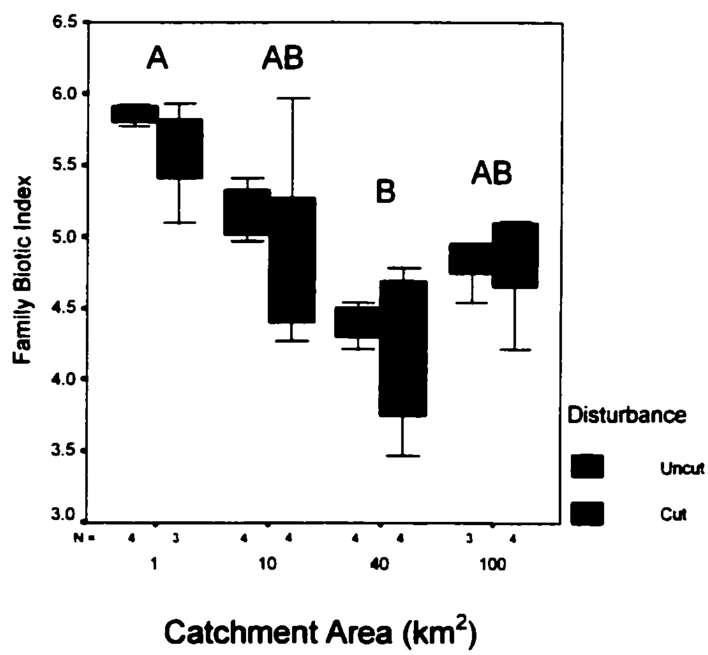


***Figure 3.4** Family Biotic Index mean and variability for each catchment area and disturbance type. Different letters indicate catchment area classes that differ significantly (Tukey HSD, α = 0.05)*

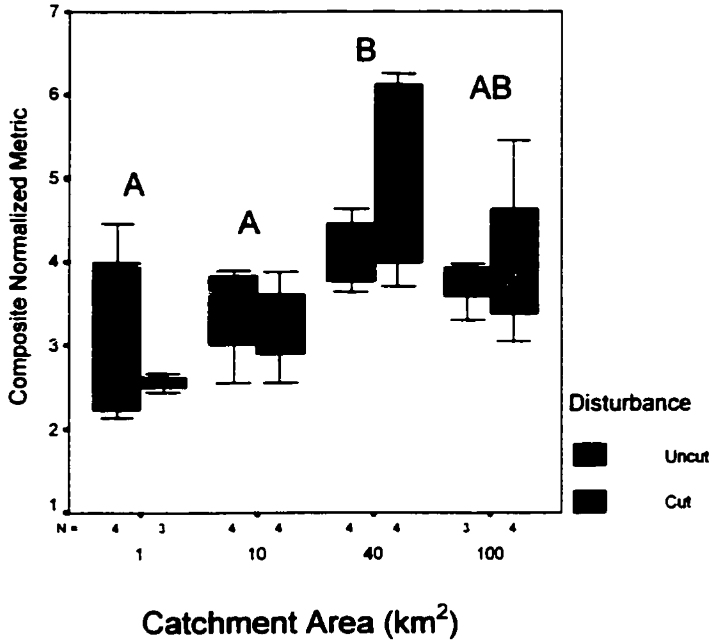


***Figure 3.5** Composite Normalized Metric mean and variability for each catchment area and disturbance type. Different letters indicate catchment area classes differ significantly (Tukey HSD, $\alpha = 0.05$).*

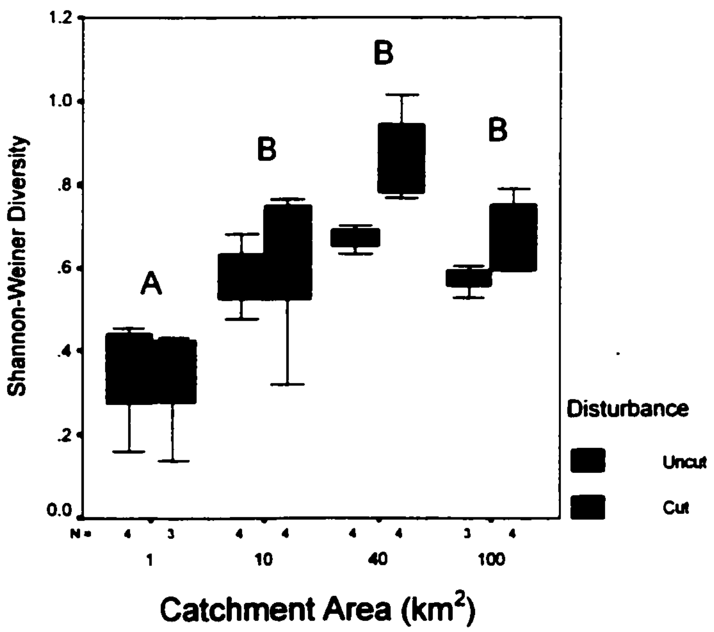


***Figure 3.6** Shannon-Weiner Measure of Diversity for each catchment area and disturbance type. Different letters indicate catchment area classes that differ significantly (Tukey HSD, $\alpha = 0.05$)*

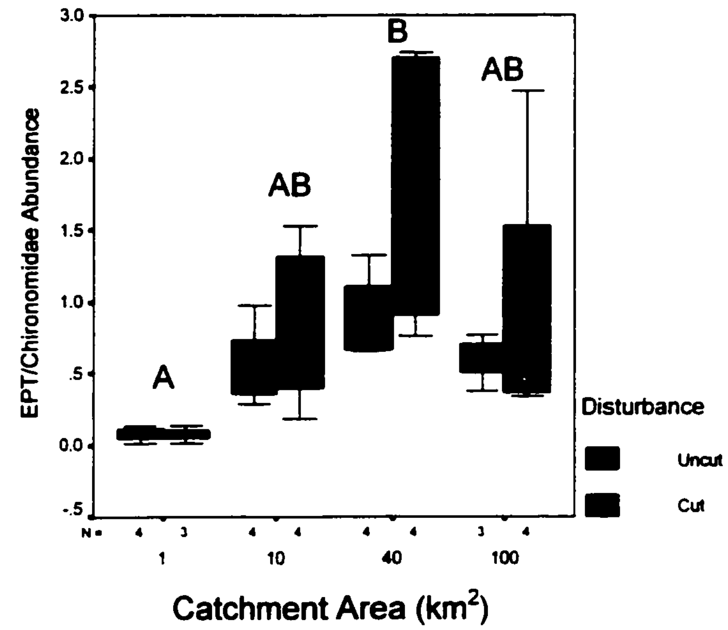


***Figure 3.7** EPT/Chironomidae abundance mean and variability for each catchment area and disturbance type. Different letters indicate catchment area classes that differ significantly (Tukey HSD, α = 0.05).*

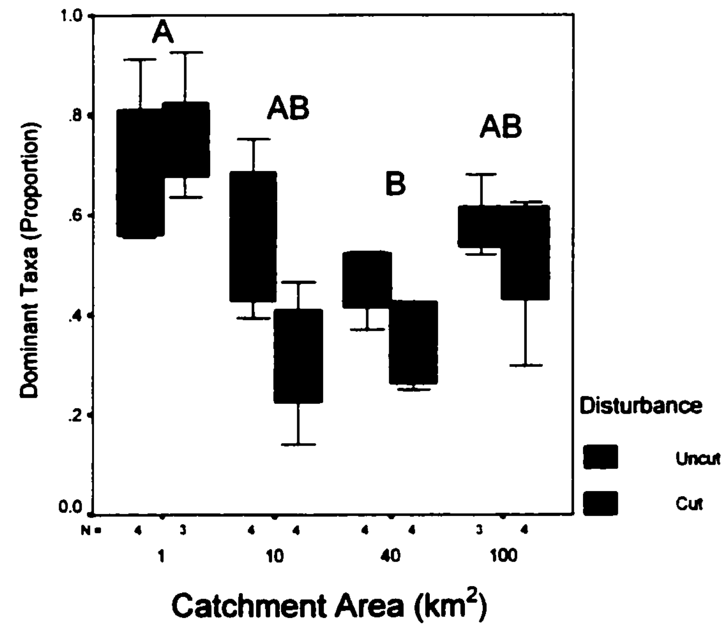


***Figure 3.8** Percent dominance mean and variability for each catchment area and disturbance type. Different letters indicate catchment area classes that differ significantly (Tukey HSD, α = 0.05)*

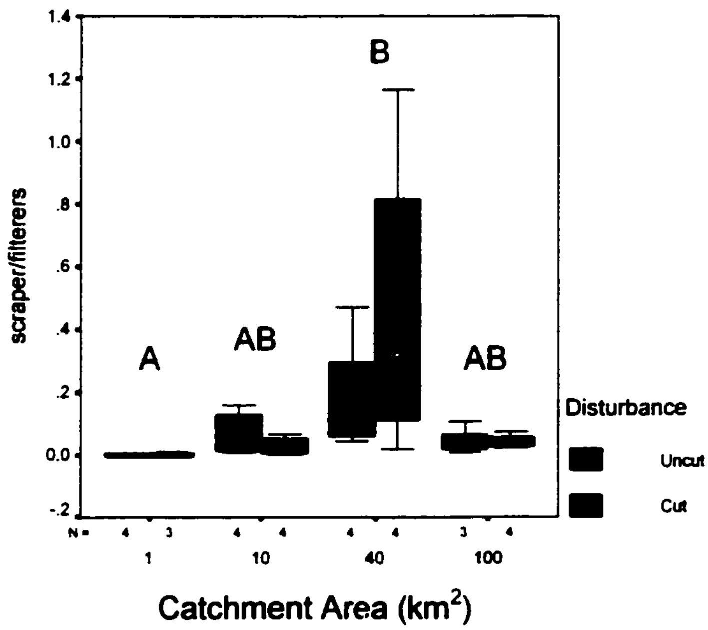


***Figure 3.9** Scraper/Filterer mean and variability for each catchment area and disturbance type. Different letters indicate catchment area classes that differ significantly (Tukey HSD, α = 0.05).*

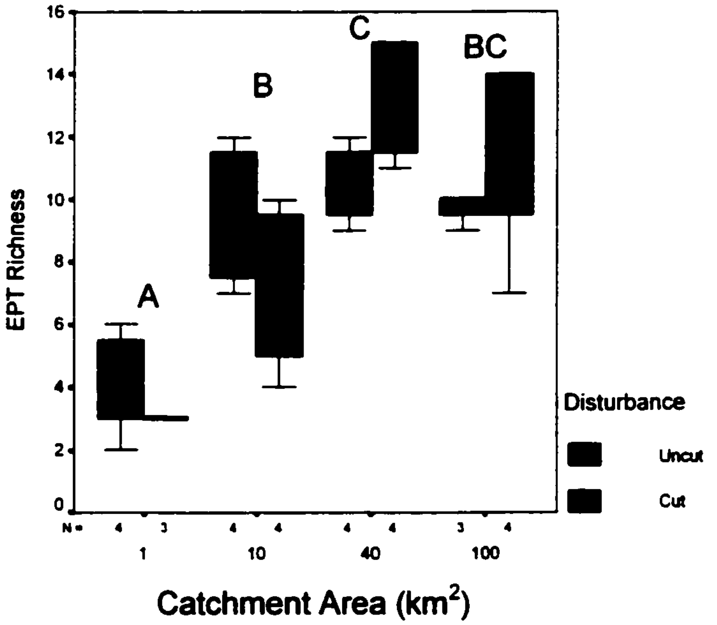


***Figure 3.10** EPT richness mean and variability for each catchment area and disturbance type. Different letters indicate catchment area classes that differ significantly (Tukey HSD, α = 0.05).*

***Table 3.8** Site assessments according to individual metrics (Plafkin et al. 1989). Reference values were generated from all sites (not calibrated to catchment area). All sites in table were cut. Uncut sites were used as reference streams (Moderately Impaired = M, Severely Impaired = S, Unimpaired = blank cell)*

Site & Catchment Area	Dominant Taxa	scrapers/ filterers	RichEPT	EPT/ Chir	shredders/ total	RichTaxa	FBI
Mac10K (10)		S	S	M	S	M	M
lars unburn (1)	S	S	S	S	S		M
SettCut (1)	S	S	S	S			M
W4.1 (1)	S	S	S	S		M	
Robin Rd (100)	S	M	M	M			
Mac 2110(b) (10)	S	S	M			M	
Muskrat10K (10)	S	M			S		
Muskrat River (100)	S	M			S		
BeckClear (40)	S	S					
Moraine (100)	S	M					
Wolfpup100K (100)	S				S		
WlkMcClear (40)	S						
Mac2120(a) (10)	S						
Abi (40)							
W5 (40)							

***Table 3.9** Site assessments according to individual metrics (Plafkin et al. 1989). Reference values were calibrated to catchment area. All sites in table were cut. Uncut sites were used as reference streams. (Moderately Impaired = M, Severely Impaired = S, Unimpaired = blank cell)*

Site & Catchment Area	Dominant Taxa	scrapers/ filterers	RichEPT	EPT/ Chir	shredders/ total	RichTaxa	FBI
Mac10K (10)		S	S	M	S	M	M
lars unburn (1)	S		M	S			M
SettCut (1)	S	S	M				M
W4.1 (1)	S	S	M			M	
Robin Rd (100)	S		M	M			
Mac 2110(b) (10)	S	S	S			M	
Muskrat10K (10)	S				S		
Muskrat River (100)	S			M	S		
BeckClear (40)	S	S					
Moraine (100)	S	M					
Wolfpup100K (100)	S				S		
WlkMcClear (40)	S						
Mac2120(a) (10)	S						
Abi (40)	S						
W5 (40)	S						

Table 3.10). No significant interaction effects between catchment area and disturbance were detected for any of the metrics (Figure 3.3-3.10, Table 3.11). However when individual sites were evaluated using the impairment criteria outlined by Plafkin *et al.* (1989) many sites were considered impaired to some degree (Table 3.8-3.9). Percent Model Affinity distinguished between severely, moderately, and slightly impaired, while RBP II rated most sites as moderately impaired (Tables 3.12 to 3.15). Percent Model Affinity also identified 3 sites (Abigogami, Moraine and W5) as impaired, whereas RBP II metric did not. Although these three harvested sites were dissimilar to reference sites, their individual metric values were considered extremely healthy. For example, Abigogami had an unusually high taxa richness, which would be considered healthy, despite being dissimilar. Abigogami, Moraine and W5 were not considered impaired when the reference condition was calibrated to catchment area. Similarly, one site (Larson Unburnt 1 km^2) was considered severely impaired by the RBP II, but not impaired when calibrated to catchment area.

3.3 DISCUSSION - BIOMONITORING

This study showed that the season, year and stream catchment area can all affect the numerical value of a macroinvertebrate biomonitoring metric (Figures 3.1 and 3.2, Tables 3.8-3.15). On average both seasonal and yearly variability were less at individual sites than the variability existing between sites. However, in some cases, both the seasonal and

***Table 3.10** Summary of the disturbance main effect two-way ANOVA results for each metric, their direction of difference with disturbance type. (degrees of freedom were $F_{1,29}$ in all cases)*

Metric	Direction of Difference	F Value	Significance
Taxa richness	no difference	0.36	0.56
Family Biotic Index	no difference	0.30	0.59
Composite Normalized Metric	no difference	0.19	0.67
Shannon-Weiner Diversity	no difference	3.06	0.09
EPT/Chironomidae abundance	no difference	3.13	0.09
Scrapers/Filterers	no difference	0.61	0.44
Shredders/Total	no difference	0.02	0.90
Rich EPT	no difference	0.18	0.68
Dominant Taxa	no difference	3.18	0.09

***Table 3.11** Summary of the interaction effect two-way ANOVA results for each metric, their direction of difference with catchment area and disturbance type combinations (degrees of freedom were $F_{3,29}$ in all cases)*

Metric	Direction of Difference	F Value	Significance
Taxa richness	no difference	1.24	0.32
Family Biotic Index	no difference	1.16	0.92
Composite Normalized Metric	no difference	1.23	0.32
Shannon-Weiner Diversity	no difference	1.01	0.19
EPT/Chironomidae abundance	no difference	0.77	0.53
Scrapers/Filterers	no difference	1.09	0.37
Shredders/Total	no difference	0.51	0.68
Rich EPT	no difference	2.54	0.08
Dominant Taxa	no difference	1.79	0.18

***Table 3.12** Impaired sites according to RBP II. Reference values not calibrated to catchment size.*

Similarity to the Reference Community	Assessment	Site and Site Score	Catchment Size and Disturbance Type
<21%	Severely Impaired	Larson Unburnt (21%)	Cut, 1 km^2
21-29%	Severely to Moderately Impaired	None	None
29-72%	Moderately Impaired	Settimio Cutover (36%)	Cut, 1 km^2
		W 4.1 (36%)	Cut, 1 km^2
		Mac 10K (36%)	Cut, 10 km^2
		Mac 2110(b) (57%)	Cut, 10 km^2
		Muskrat 10K (64%)	Cut, 10 km^2
		Beck Clearcut (71%)	Cut, 40 km^2
		Muskrat River (64%)	Cut, 100 km^2
		Robin Road (64%)	Cut, 100 km^2
		Wolfpup 100K (71%)	Cut, 100 km^2

***Table 3.13** Impaired sites according to RBP II. Reference values were calibrated to catchment size.*

Similarity to the Reference Community	Assessment	Site and Site Score	Catchment Size and Disturbance Type
<21%	Severely Impaired	None	None
21-29%	Severely to Moderately Impaired	None	None
29-72%	Moderately Impaired	Larson Unburnt(57%)	Cut, 1 km^2
		Settimio Cutover (57%)	Cut, 1 km^2
		W 4.1 (57%)	Cut, 1 km^2
		Mac 10K (36%)	Cut, 10 km^2
		Mac 2110(b) (50%)	Cut, 10 km^2
		Muskrat 10K (71%)	Cut, 10 km^2
		Beck Clearcut (71%)	Cut, 40 km^2
		Muskrat River (64%)	Cut, 100 km^2
		Robin Road (71%)	Cut, 100 km^2
		Wolfpup 100K (71%)	Cut, 100 km^2

***Table 3.14** Impaired sites according to Percent Model Affinity scores (Novak and Bode 1992). Reference values were not calibrated to catchment size.*

Similarity to the Reference Community	Assessment	Site and Site Score	Catchment Size and Disturbance Type
< 35%	Severely Impacted	Mac Lake (30.2%)	Cut, 10 km^2
		Moraine (20.1%)	Cut, 100 km^2
35-49%	Moderately Impacted	Abigogami (46.0%)	Cut – 40 km^2
		W5 (46.0%)	Cut – 40 km^2
50-64%	Slightly Impacted	Settimio Cutover (58.2%)	Cut – 1 km^2
		Larson Unburnt (55.6%)	Cut – 1 km^2
		Muskrat River (57.3%)	Cut – 10 km^2

***Table 3.15** Impaired sites according to Percent Model Affinity scores (Novak and Bode 1992). Reference values were calibrated to catchment size*

Similarity to the Reference Community	Assessment	Site and Site Score	Catchment Size and Disturbance Type
< 35%	Severely Impacted	Mac Lake (33.4%)	Cut, 10 km^2
		Moraine (23.2%)	Cut, 100 km^2
35-49%	Moderately Impacted	None	
50-64%	Slightly Impacted	Settimio Cutover (62.4%)	Cut, 1 km^2
		Larson Unburnt (60.2%)	Cut, 1 km^2
		Mac 2110 (58.9%)	Cut, 10 km^2
		Muskrat River 10K (57.1%)	Cut, 10 km^2
		Abigogami (604%)	Cut, 40 km^2
		W5 (53.6%)	Cut, 40 km^2

yearly variability of a metric exceeded between site variability. For example, the seasonal cv for Ephemeroptera density at Walk 5.1 1k exceeded between site variability, as did the seasonal variability of EPT/Chironomidae at Settimio 1k (Figure 3.1). These metrics also had high cv (cv > 50%). values according to Barbour *et al.* (1992).

Metrics that tend to have high variability may provide inconsistent and unreliable information about a stream. For example, a stream may have a high density of Ephemeroptera early in the season due to a recent hatch, later in the season the Ephemeroptera numbers may decrease because of emergence. As a result, early in the season the Ephemeroptera metric may indicate a healthy stream, whereas late in the season the metric may indicate an unhealthy stream. Actually, the impaired assessment has nothing to do with stream health and everything to do with the life cycle of the Ephemeroptera.

The evidence of high seasonal variability in most of the metrics in this study emphasizes the need for researchers to sample during a short window of time. Hilsenhoff *et al.* (1982) acknowledged the influence of stream current, temperature and seasonal factors on biotic index values. His biotic index values are falsely high in summer, when he found that many species are in diapause. Lenat (1993) adjusts tolerance values to accommodate seasonal differences in Southeastern United States.

The consistent information derived from stable metrics may be an important component of biomonitoring programs. Metrics that had both low seasonal and yearly variability according to Barbour *et al.* (1992) criteria (cv < 50%) were Percent Dominance, Percent Model Affinity and Percent Dipterans. If highly variable metrics are used it should likely be in conjunction with more stable metrics.

Inconsistencies existed in the diagnosis of individual sites depending on how the reference condition was derived (Tables 3.7 to 3.12). Sites were often diagnosed differently when tested against reference values calculated from all sites rather than from a specific stream size class. More specifically, 1 km^2 sites appeared more prone to a severely impaired diagnosis when reference streams were not calibrated to catchment area. This diagnosis is likely due to low diversity and richness that characterized these smaller streams (Figure 3.3 and 3.6). Low diversity and low richness are considered signs of impairment (Plafkin *et al.* 1989). However, these characteristics were related to catchment size rather than disturbance (Table 3.7). Many authors have acknowledged that headwater streams fed by nutrient poor groundwater are naturally low in productivity. Low diversity and richness reflect this low productivity (Pinder and Farr 1977, Plafkin *et al.* 1989). To avoid misinterpretation of the health of small streams, calibration of reference metrics should be based on catchment size, especially when dealing with this size class of stream.

All metrics differed among catchment sizes (Table 3.7). This finding substantiates that classification of reference communities by catchment size may reduce natural variability. The average values for the metrics considered were different at smaller streams compared to larger ones. Calibrating metrics to catchment area is a simple way to improve the accuracy of the assessment.

I assumed *a priori* to the tests that if streams were impacted by harvesting in the catchment, some or all of the metrics calculated would differ between harvested and forested streams. However, in all cases metric values did not differ among streams flowing through harvested and forested areas. There are three possible explanations for the results. First, harvesting has no influence on stream invertebrate communities, or not enough influence to show an impact. Second, harvesting in the catchment impacted a few sites; however, not enough sites to differentiate them from unimpaired, because the unimpaired sites are so variable. Third, the sites had been impacted but the metrics used in this study did not detect the impact.

Examination of metric scores at individual sites supported the second hypothesis that there were not enough impaired sites to differentiate them from unimpaired. Every metric identified impairment at some sites. Similarly, Percent Model Affinity and RBP II identified several sites as impaired. As mentioned in the previous chapter, there may be extensive variability in the proportion of disturbed area in the study catchments. Likely, the proportion of harvested area in the catchment is one factor that mediates the impacts of timber harvesting on a stream. Larger harvested areas could cause more of a disruption to a stream macroinvertebrate community. Thus, some sites may have been impaired, but variability in the disturbance area data could have masked the impairment in the ANOVA.

The biomonitoring indices analyzed did not provide a clear evaluation of whether or not timber harvesting influenced stream macroinvertebrate communities. None of the indices differed between harvested and forested sites. However, all of the indices identified impairment at some individual sites. A more detailed investigation of the macroinvertebrate community composition in unharvested, forested and different sizes of streams may further verify one of the three explanations for the inability to detect harvest impacts. Further monitoring of some of these "impaired" streams to verify whether or not they should be considered impaired in terms of other values (fish presence, habitat characteristics) could be valuable.

4.0 INTRODUCTION – MULTIVARIATE ASSESSMENT

Natural variability in stream dwelling benthic macroinvertebrate community composition is an obstacle when they are used for biomonitoring for stream impairment. Macroinvertebrate community assemblages differ both among streams and over time within the same stream. To use macroinvertebrates effectively for biomonitoring of stream habitat an awareness of the factors that determine the distribution of macroinvertebrates in an undisturbed setting is necessary. Factors that explain relatively large amounts of variability could be incorporated into a biomonitoring program in order to account for some variability in the data. In the previous chapter, stream size (catchment area) was identified as a factor that does account for variability in communities. Another factor that may be related to community is microhabitat (substrate profile, temperature and velocity).

The immense variation among macroinvertebrate communities is probably related, in part, to the variation in microhabitat characteristics among streams. Some influential microhabitat characteristics may be: substrate composition (Richard *et al.* 1993, Minshall *et al.* 1984) canopy characteristics, terrestrial organic inputs (Egglishaw 1964, Molles 1982, Corkum 1992), temperature and hydrology (Merrit and Cummins 1996a). Attempts to partition variation in the macroinvertebrate community are often made by characterizing habitat, and sampling from similar habitats.

Consideration of factors, such as stream size, that control microhabitat characteristics may be another way to partition variability. Within my study area, instream temperature, discharge and gradient have all been shown to relate to the size of the stream (Rankin 2000). Nutrient inputs, colonization rates and predation usually differ according to the size of a stream (Vannote *et al.* 1980).

The previous chapter determined that most macroinvertebrate biomonitoring metrics differ according to stream size. Analyses in this chapter complement this finding by exploring the invertebrate community and its relation to stream size and habitat using multivariate analyses. The multivariate analyses take correlation among variables into account. The multivariate method also identifies variables that effectively distinguish between groups. This avoids the subjective selection of indices and the construction of scoring criteria (Somers *et al.* 2002). The multivariate exploration of the macroinvertebrate community may provide greater understanding of how individual taxa are affected by stream size and in turn how this effect of stream size influences the metrics used in the previous chapter.

Stream size can be expressed as "stream order" (Vannote *et al.* 1980) or by "catchment area" (Frissel *et al.* 1986). Catchment area may be more accurate and useful for comparisons between streams, because it is an objective measurement, independent of flow fluctuations and can be calculated by GIS computer software. In addition, "catchment area" is not related to map scale or mapping accuracy, whereas "stream order" is.

Disturbance to the forest adjacent to a stream may also influence stream habitat and thus the macroinvertebrate community dwelling within that habitat. Human landuse (e.g. timber harvesting) has been shown to have a strong influence on instream habitat by increasing the rate of water and nutrient movement into the stream and by inducing erosion and sedimentation (Allan *et al.* 1997, Richards *et al.* 1996, Rothrock *et al.* 1998). Besides timber harvesting, other disturbances such as forest fire may also influence the instream macroinvertebrate communities that directly utilize this habitat.

On average, metrics did not differ between harvested and forested streams. However, when examined individually, some sites were considered impaired. In the previous chapter, three possible explanations for this result were given: 1) none of the streams were impaired by harvesting, 2) some were impaired but too few or too moderately to influence the analysis, or 3) that most sites were actually impaired but the metrics used were not sensitive enough to detect impairment caused by harvesting. In this chapter, the influence of timber harvested on both the macroinvertebrate community and the habitat structure of the stream will be examined using multivariate analysis.

Other studies have found that exploratory multivariate methods based on community composition were sensitive to stress-induced changes in aquatic communities and detected differences that were not reflected by biomonitoring metrics (Somers *et al.* 2002, Warwick and Clarke 1991, Yan *et al.* 1996). Possibly, the examination of macroinvertebrate communities as a whole may support one of the hypotheses generated in the previous chapter.

The study presented in this chapter had three objectives. The first objective was to verify that invertebrate community structure is related to physical habitat variables in Northwestern Ontario streams. Macroinvertebrate community structure was expected to relate to differences found among stream substrate profile, instream temperature and discharge.

The second objective was to investigate if macroinvertebrate community and instream physical habitat characteristics differ with the size of the stream (catchment area). Instream habitat variables were expected to differ among streams of different sizes. Discharge and temperature were predicted to increase with stream size. The relative abundance of substrate size classes was predicted to shift as stream size increased. Streams with larger catchment area would have a greater discharge, which would suspend and carry away fine sediment leaving behind a more coarse substrate profile. Streams with smaller catchment areas would have less powerful discharge and likely a finer sediment profile as fine sediments would settle out of the water column.

Macroinvertebrate community structure was expected to relate to catchment area (stream size) according to the predictions of the River Continuum Concept (Vannote *et al.* 1980). Taxonomic differences as well as shifts in functional feeding groups were expected. Shredders were expected to be co-dominant with collectors in the headwaters. With increasing stream size, collectors (grazers and filterers) become more dominant. Scrapers are adapted for shearing attached algae from surfaces and their relative abundance was expected to peak in mid-sized rivers. Predators abundance was expected to remain constant with stream size changes (Vannote *et al.* 1980).

The third objective was to determine if the macroinvertebrate community and habitat were influenced by timber harvesting for any or all stream sizes. Vulnerability of these instream variables to the effects of timber harvesting was expected to vary with the size of the catchment, with smaller streams being more vulnerable. Increased discharge and a shift in the sediment profile were expected after timber harvesting and burning. Invertebrate community structure in streams with adjacent disturbance (timber harvest and fire) was expected to differ from streams with no adjacent disturbance.

4.1 METHODS – MULTIVARIATE ASSESSMENT

4.1.1 Invertebrate Data Set Description

Three macroinvertebrate data sets were created ("Taxonomic", "Family" and "Functional Feeding Group") (Table 4.1). For the multivariate analyses, the "Taxonomic" and "Functional Feeding Group" data sets were used, but not the "Family" data set. The information in the "Family" data set was necessary only for the creation of the "Functional Feeding Group" data set.

4.1.2 Instream Physical Habitat Data Collection

All habitat data were collected from May to September, in base flow conditions. Instream temperature and discharge were measured at each site. Discharge was calculated by

Table 4.1 Taxonomic Description of the three data sets used to examine the relationships between catchment area, disturbance type and macroinvertebrate community assemblage.

Data Set	Visits/Site	Sampling Periods	Taxonomic Description
"Taxonomic"	56 sites, one visit	4 sites in 1995 5 sites in 1997 20 sites in 1998 27 sites in 1999	• Insects to Order • Non-insect groups more generalized
"Family"	30 sites, one visit	4 sites in 1995 11 sites in 1998 15 sites in 1999	• Insects to Family • Non-insect groups more generalized
"Functional Feeding Group", (based on "Family" data set)	30 sites, one visit	4 sites in 1995 11 sites in 1998 15 sites in 1999	• All taxa categorized into functional feeding groups (Merrit and Cummins 1996b)

measuring water velocity at 60% water column depth at 20 points across a transect (Bain and Stevenson 1999). Cross-sectional flow was relatively uniform at the chosen transect. Discharge was then calculated using the formula:

$$Q = \sum_{i=1}^{20} (\text{interpoint distance x depth x velocity})$$

To characterize stream substrate 110 to 180 pebbles were measured at each site. A reach of approximately 40 metres long was sectioned by 10 to 12 transects. At 3 to 5 points along each transect, 3 pebbles were randomly selected. The median axis length was measured to the nearest millimeter. I grouped pebbles according to Wentworth classification (Cummins 1962) (Table 4.2). The data were then compiled and converted to a percentage of total number of pebbles counted.

4.1.3 Statistical Analyses

4.1.3.1 Instream Physical Habitat Data

To test for differences in pebble size classes among streams with different catchment sizes and disturbance activities, I used a MANOVA (multivariate analysis of variance). This procedure tested for differences among the main effects (area class and disturbance type) as well as the interaction between the main effects. Area classes included 1, 10, 40 and 100 km^2 sites. Disturbance type included undisturbed, timber harvested or burnt sites. The hypothesis that disturbance impacts are dependent upon catchment size was tested with the interaction term in the MANOVA. If a significant effect was detected by the MANOVA, variables important in differentiating among the groups (catchment and

***Table 4.2** Modified Wentworth Classification (Cummins 1962)*

Substrate type	**Particle size range (mm)**
Bedrock	>1001
Boulder	256-1000
Cobble	64-256
Pebble	16-63
Gravel	2-15
Sand	0.06-1
Silt/Clay	<0.059

disturbance) were identified using a Discriminant Function Analysis (DFA). To explore differences in temperature and discharge among treatments, I used a Kruskal Wallis non-parametric test because transformations did not normalize the discharge data.

4.1.3.2 Macroinvertebrate Community and Instream Habitat Relationship

To relate environmental variables to invertebrate community structure, I first summarized the macroinvertebrate information using principal component analyses on both the taxonomic and functional feeding group matrix. The first principal component generated from each analysis was regressed against substrate size classes, temperature and discharge.

4.1.3.3 Invertebrate Community Composition

To test for differences in macroinvertebrate communities (Taxonomic Group, Functional Feeding Group) among streams with different sized catchment areas and streams with cut, uncut catchments, I used a MANOVA. Burnt sites were also included in the taxonomic group MANOVA as a factor in the disturbance type main effect. This procedure tested for differences among the main effects (area class and disturbance type) as well as the interaction between the main effects. The hypothesis that disturbance impacts are dependent upon catchment size was tested with the interaction term in the MANOVA. To identify variables important in differentiating among the groups when a significant effect was detected by the MANOVA, I used a Discriminant Function Analysis (DFA).

4.2 RESULTS – MULTIVARIATE ASSESSMENT

4.2.1 Instream Physical Habitat Data

4.2.1.1 Substrate

Substrate class composition differed significantly among streams with different sized catchment areas (MANOVA, $F_{21,113} = 1.657$, $p = 0.049$). There was no difference in substrate class among disturbance type (MANOVA, $F_{14,78} = 0.828$, $p = 0.637$). As well, no interaction between catchment area and disturbance type was detected (MANOVA, $F_{28,142} = 0.842$, $p = 0.695$). Substrate classes important in differentiating among catchment area classes were identified using a DFA. One significant function generated by the analysis explained 45% of variance among group centroids and accounted for 82% of the variability explained by the analysis (canonical correlation = 0.674) (Tables 4.3 and 4.4).

Group centroids indicated that Function 1 discriminated most effectively between one km^2 sites (1.275) and 100 km^2 sites (–1.093) (Figure 4.1). One km^2 sites were characterized by higher percentages of sand (standardized CDF coefficient = 0.587) and silt (standardized CDF coefficient = 0.550). Forty and 100 km^2 sites clustered towards the negative end of Function 1 (Figure 4.1) and were characterized by boulders (standardized CDF coefficient = -0.298) and pebbles (standardized CDF coefficient = -0.318). Ten km^2 sites did not form a cohesive group along Function 1's axis (Figure 4.1).

***Table 4.3** Variance explained by the function generated by the DFA. Sites were grouped by catchment area and then characterized by substrate size class*

Function	Eigenvalue	% of Variance	Cumulative %	Canonical Correlation
1	0.833	82.0	82.0	0.674
2	0.119	11.7	93.8	0.326
3	0.063	6.2	100.0	0.244

***Table 4.4** Significance of the functions generated by the DFA. Sites were grouped by catchment area and then characterized by substrate size class*

Test of Functions	Wilk's Lambda	Chi-Square	df	Significance
1 through 3	0.458	37.835	21	0.013
2 through 3	0.843	8.442	12	0.750
3	0.940	2.979	5	0.703

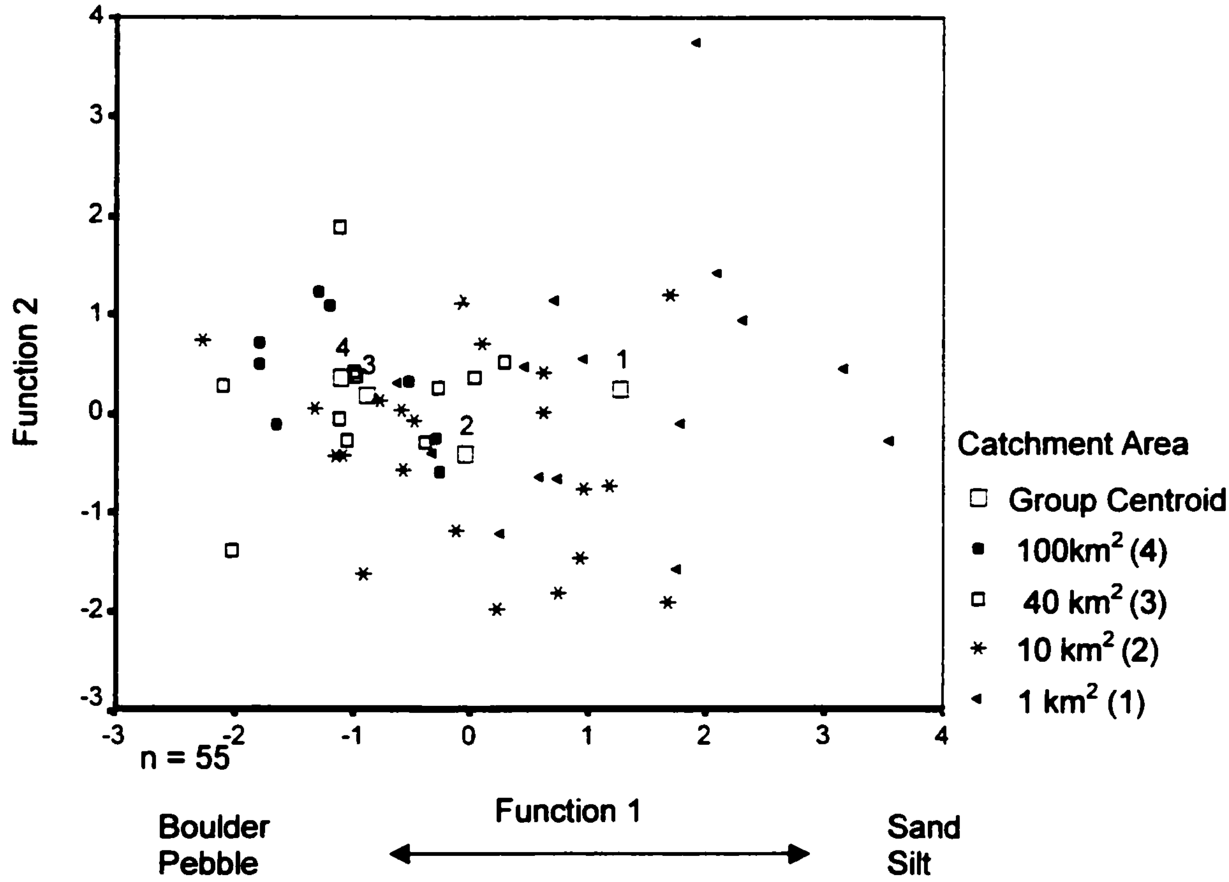


***Figure 4.1** DFA scatter plot showing function 1 and 2 scores of each study site. Function 1 represents a gradient of sites dominated by coarse substrate to sites dominated by fine substrate. 100 and 40 km^2 sites generally had coarse substrate while 1 km^2 sites had fine substrate. 10 km^2 sites were intermediate.*

4.2.1.2 Discharge and Temperature

Discharge and temperature showed a pattern similar to substrate. Median discharge values were 1 km^2 = 0.00149592 m^3/second, 10 km^2 = 0.010752 m^3/second, 40km^2 = 0.0167 m^3/second, and 100 km^2 = 0.120741 m^3/second. Discharge significantly increased with catchment area (Kruskal-Wallace, $p < 0.0001$, $n = 54$, Figure 4.2). Discharge did not differ significantly among disturbance groups (Kruskal-Wallace, $p = 0.481$, $n = 54$).

Temperature significantly increased with increasing catchment areas (Kruskal-Wallace, $p < 0.0001$, $n = 54$). Median temperatures were 1 km^2 = 13, 10 km^2 = 15, 40km^2 = 17, and 100 km^2 = 20 (Figure 4.3). Temperature did not differ significantly among disturbance groups (Kruskal-Wallace, $p = 0.596$, $n = 54$).

4.2.2 Macroinvertebrate Community Relationship with Instream Physical Habitat Variables

To relate environmental variables to invertebrate community structure, I used two stepwise regression analyses. The first regression related environmental variables to taxonomic group structure. As the dependent variable in the regression, I used the first principal component scores generated from a Principal Component Analysis of the taxonomic group matrix (PC1(taxonomic group)). This principal component accounted for 34.83% of the variability in the taxonomic group structure among sites. Higher numbers of Ephemeroptera and Trichoptera characterized sites with positive values of the

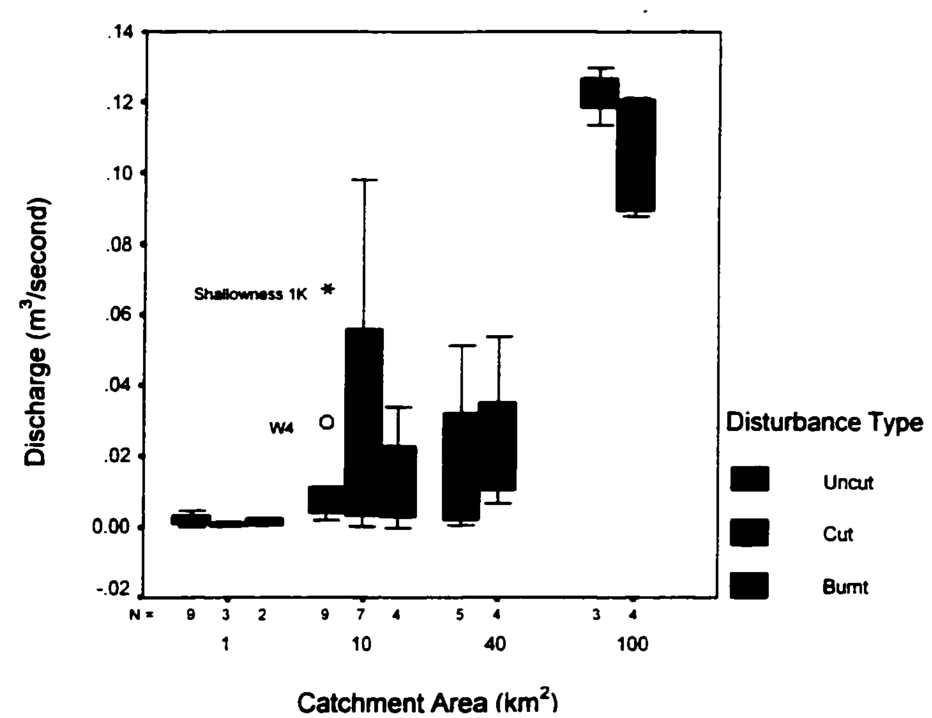


***Figure 4.2** Boxplot of discharge by catchment area class and disturbance type. Outliers excluded from boxplots were Moraine 30K (3.27), WalkMac 30K Uncut (.088), Wolfpup 100K (0.21), Black Sturgeon East Marked (0.17). (line = median, box is 25-75 quartile, whisker = 95% C.I.). Outliers were not excluded from the analysis.*

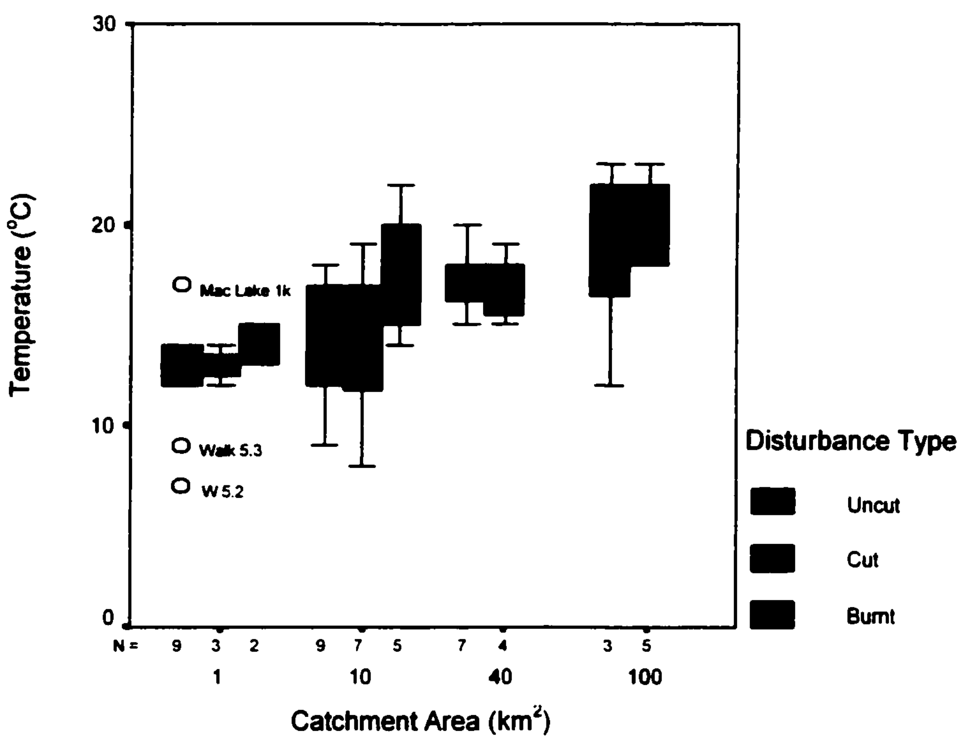


***Figure 4.3** Boxplot of temperature by catchment area class and disturbance type. (line = median, box is 25-75 quartile, whisker = 95% C.I.)*

principal component, while high numbers of Amphipoda characterized sites with negative values (Table 4.5). Independent variables in the regression were each substrate class (percentage of silt/clay, sand, gravel, pebble, cobble, boulder, bedrock), temperature and discharge. The regression analysis generated a significant model ($F = 11.446$, $P = 0.001$) that explained 16.5% (adjusted R^2) of the original variability in the data (Figure 4.4). The regression equation was:

PC1 (taxonomic group) = -0.917 + 0.898 lg pebble

Environmental variables were related to functional feeding group structure with the second regression analysis. As the dependent variable in the regression I used the first principal component score for each site from a Principal Component Analysis on the functional feeding group matrix (PC1(functional feeding group)). This principal component accounted for 43.5% of the variability in functional feeding group composition among sites. High numbers of predators and filterers characterized sites with high component scores, while high numbers of shredders characterized sites with low scores (Table 4.6). Independent variables in the regression were substrate (percentage of silt/clay, sand, gravel, pebble, cobble, boulder, bedrock), temperature and discharge. The regression analysis generated a significant model ($F = 6.022$, $P = 0.021$), that explained 15% (adjusted R square) of the original variability in the data (Figure 4.5). The regression equation was

PC1(functional feeding group) = -0.963 + 0.950 lg pebble

***Table 4.5** Coefficient scores of each taxonomic group in the first principal component generated from a principal component analysis of the taxonomic group matrix.*

Variable	Coefficient Score
lg Ephemeroptera	.843
lg Trichoptera	.820
lg Plecoptera	.788
lg Diptera	.759
lg Coleoptera	.745
lg Anisoptera	.692
lg Hydracarina	.618
lg Hirudinea	.345
lg Gastropoda	.254
lg Pelecypoda	.182
lg Nematoda	.091
lg Oligochaeta	.053
lg Amphipoda	-.216

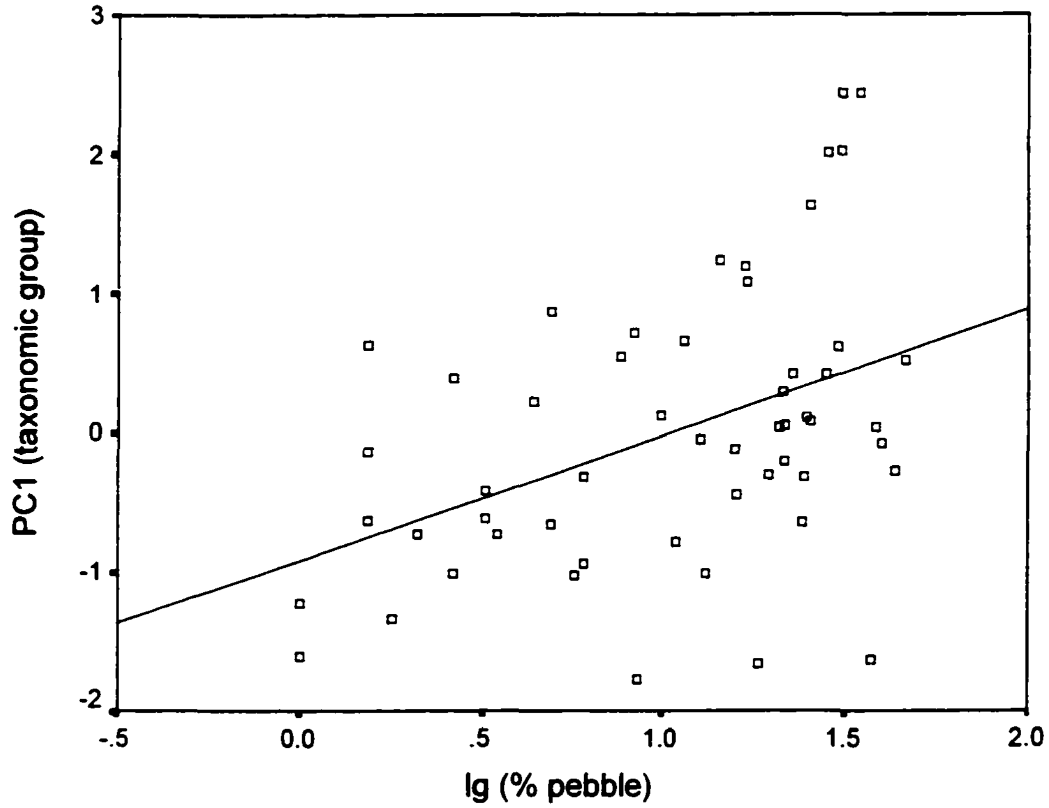


***Figure 4.4** Scatterplot of the log of the percentage pebble composition of substrate against the first principal component score generated from the principal component analysis of the taxonomic group matrix.*

***Table 4.6** Coefficient scores of each functional feeding group on the first principal component generated from a principal component analysis of the functional feeding group matrix.*

Variable	Coefficient
lg Predator	.902
lg Filterer	.824
lg Gatherer	.714
lg Scraper	.572
lg Parasitic	.465
lg Shredder	.290

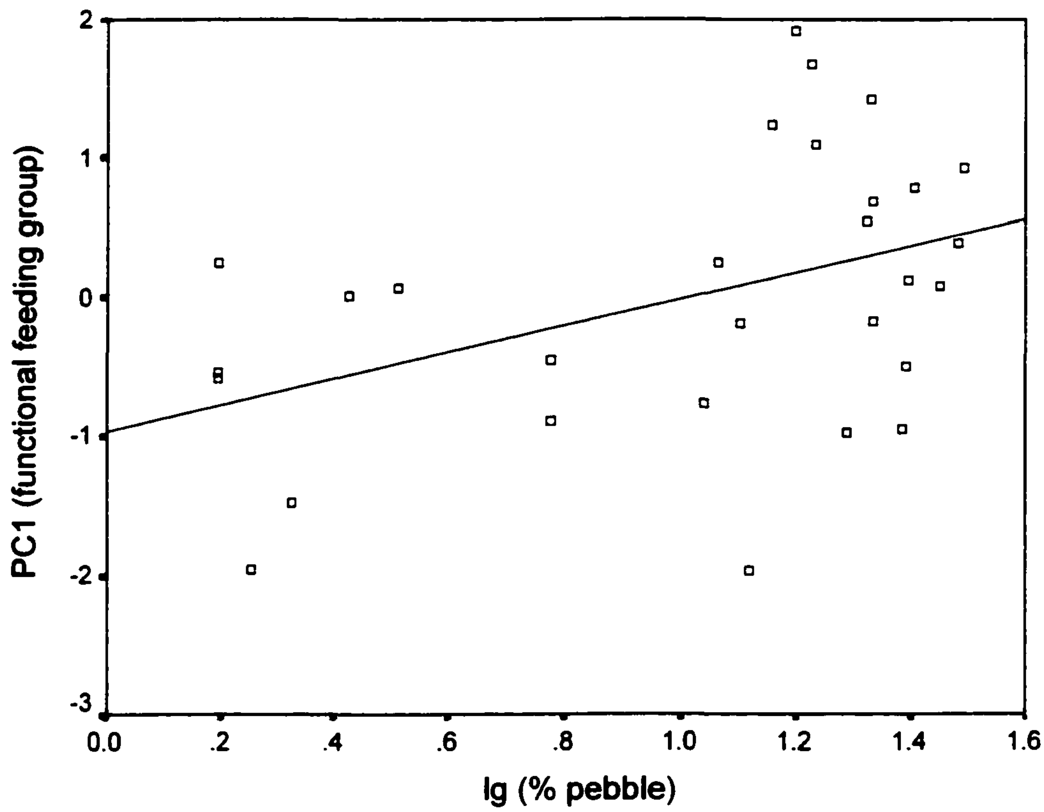


Figure 4.5 *Scatterplot of the log of the percentage pebble composition of substrate against the first principal component score generated from the principal component analysis of the functional feeding group matrix.*

4.2.3 Macroinvertebrate Community Characteristics

As with habitat variables, the macroinvertebrate communities differed among catchment areas and did not differ among disturbance types.

4.2.3.1 Taxonomic Group

Macroinvertebrate taxonomic group structure differed significantly among streams with different sized catchment areas (1, 10, 40 and 100 km^2) (MANOVA, $F_{39,101}$ =1.808, P = 0.010). There was no difference in macroinvertebrate community among disturbance type (cut, uncut, burnt) (MANOVA, $F_{26,68}$ = 0.789, P = .749). As well, no interaction effect between catchment area and disturbance type was detected (MANOVA, $F_{52,134}$=1.129, P = 0.288). Taxonomic groups important in differentiating among catchment areas were identified using a Discriminant Function Analysis. One significant function generated by the analysis explained 55.9% (canonical correlation = 0.748) of the variance among group centroids and accounted for approximately 66.4% of total variance explained by the model (Tables 4.7 and 4.8). Box's M test indicated that covariance matrices differed among groups (F = 3.139, P < 0.0001), which is a violation of the assumption of the DFA. However, because the DFA was used as an exploratory test and not a test of significance, the Box's M result was not a concern.

Group centroids indicated that Function 1 discriminated most effectively between 1 km^2 (1.774) and 100 km^2 sites (–1.249). One km^2 sites were characterized by Diptera (standardized CDF coefficient = 1.174) and Amphipoda (standardized CDF coefficient = 0.483). Ten, 40 and 100 km^2 sites clustered toward the negative end of Function 1 (group

***Table 4.7** Variance explained by the functions generated by the DFA. Sites were separated by catchment area and then characterized by the macroinvertebrate taxonomic groups*

Function	Eigenvalue	% of Variance	Cumulative %	Canonical Correlation
1	1.271	66.4	66.4	0.748
2	0.482	25.2	91.6	0.570
3	0.160	8.4	100.0	0.372

***Table 4.8** Significance of the functions generated by the DFA. Sites were separated by catchment area and then characterized by the macroinvertebrate taxonomic groups*

Test of Functions	Wilk's Lambda	Chi-Square	df	Significance
1 through 3	0.256	63.344	39	0.008
2 through 3	0.582	25.207	24	0.395
3	0.862	6.909	11	0.806

centroids = -0.513, -0.444, and -1.249 respectively) (Figure 4.6). These sites were characterized by Ephemeroptera and Trichoptera (standardized CDF coefficients = -1.048 and -0.382).

4.2.3.2 Functional Feeding Group

Macroinvertebrate functional feeding group structure differed significantly among catchment areas (MANOVA, $F_{15,50}$ =3.728, P < .0001). There were no differences in functional feeding group structure among disturbance types (MANOVA, $F_{5,18}$ = 1.812, P = 0.156). There was no significant two-way interaction between catchment area and disturbance type (uncut or cut) (MANOVA, $F_{15,50}$ = 0.872, P = 0.612).

Functional feeding groups important in differentiating among catchment area classes were identified using a Discriminant Function Analysis. Two significant functions were generated by the analysis: Function 1 explained 68 % (canonical correlation = 0.823) of variability among group centroids and Function 2 explained 61 % of the remaining variability among group centroids (Tables 4.9 and 4.10). One km^2 sites were separated most effectively (group centroid = -2.218) from 40 km^2 (group centroid = 1.562) by Function 1 (Figure 4.7). One km^2 sites were characterized by high numbers of predators (standardized CDF coefficient = -0.600) while 40 km^2 sites were characterized by higher numbers of scrapers (standardized CDF coefficient = 1.305) (Figure 4.7). Ten km^2 (group centroid = -1.904) sites were separated from 1, 40 and 100 km^2 sites (group centroids = 0.531, 0.720, 0.822 respectively) along Function 2. Ten km^2 sites were characterized by higher numbers of gatherers (standardized CDF coefficient = -1.562), while 1, 40, and

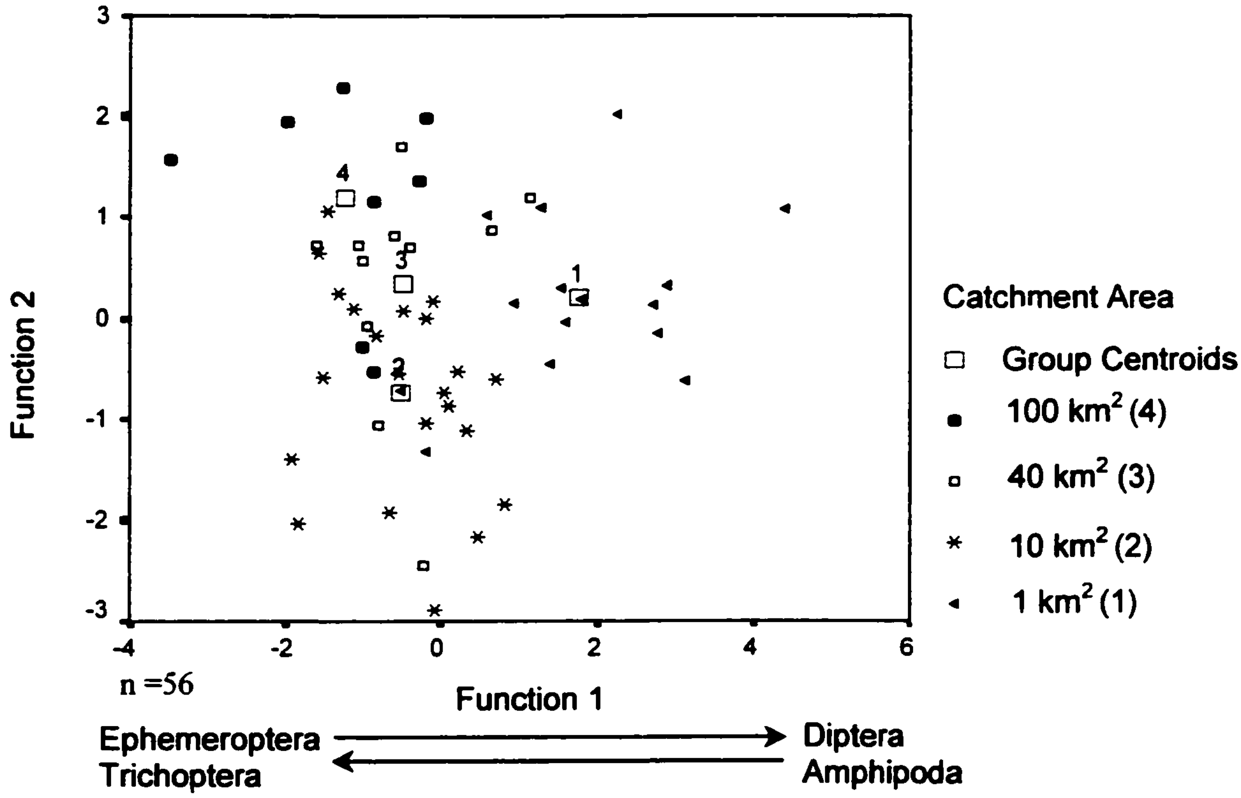


***Figure 4.6** DFA scatter plot showing function 1 and 2 scores of each study site. Function 1 represents a gradient of sites dominated by Diptera and Amphipoda to sites dominated by Ephemperoptera and Trichoptera. 100, 40 and 10 km ^{2} sites generally had more Ephemeroptera and Trichoptera, while 1 km^2 sites had higher numbers of Diptera and Amphipoda. Ten km^2 sites were intermediate. Function 2 was not significant.*

***Table 4.9** Variance explained by the functions generated by the DFA. Sites were separated by catchment area and then characterized by the functional feeding groups*

Function	Eigenvalue	% of Variance	Cumulative %	Canonical Correlation
1	2.101	55.9	55.9	0.823
2	1.533	40.8	96.8	0.778
3	0.122	3.2	100.0	0.330

***Table 4.10** Significance of the functions generated by the DFA. Sites were separated by catchment area and then characterized by the functional feeding groups*

Test of Functions	Wilk's Lambda	Chi-Square	df	Significance
1 through 3	0.113	52.225	18	<0.0001
2 through 3	0.352	25.067	10	0.005
3	0.891	2.763	4	0.598

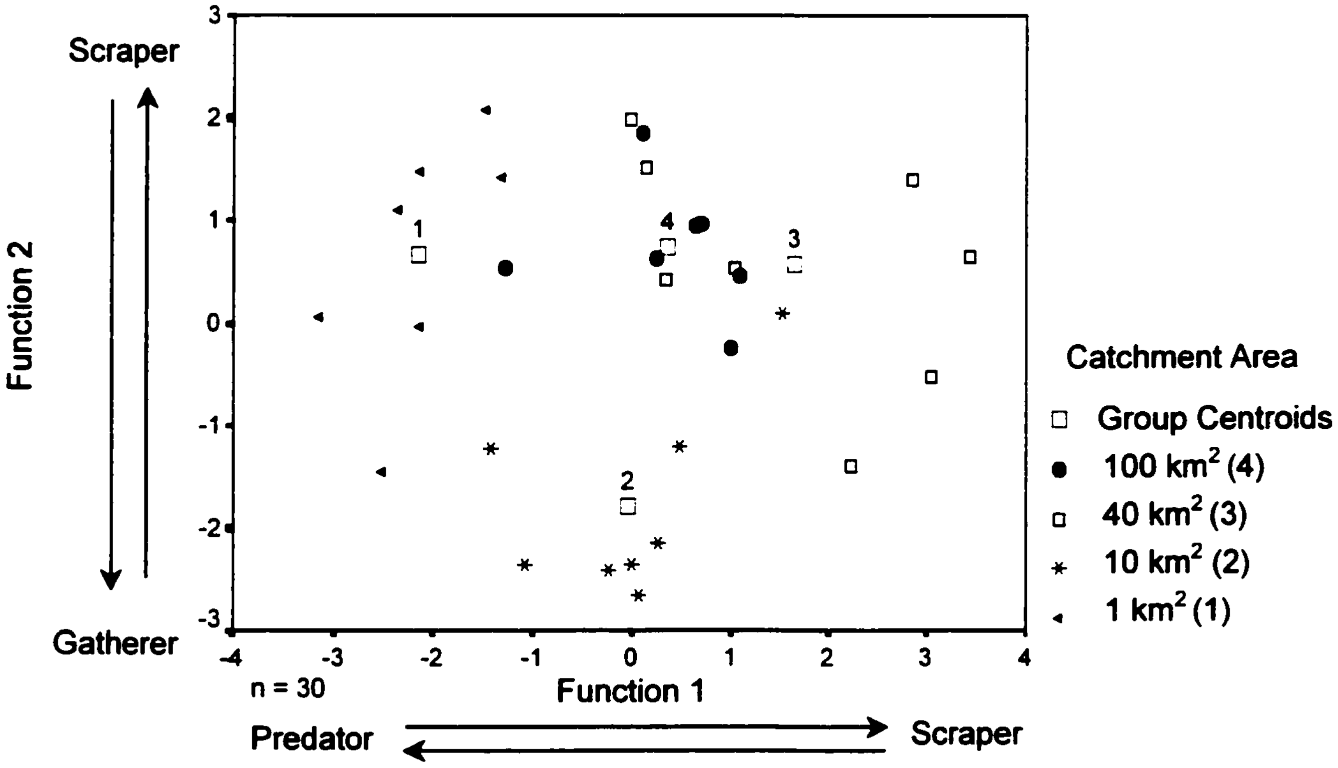


Figure 4.7 DFA scatter plot showing function 1 and 2 scores of each study sites. Function 1 represents a gradient of sites dominated by Predators to sites dominated by scrapers. Function 2 represents a gradient of sites dominated by gatherers to sites dominated by scrapers. 100, 40 and 10 km^2 sites generally had high numbers of scrapers while one km^2 sites had high numbers of predators. 10 km^2 site had high numbers of gatherers.

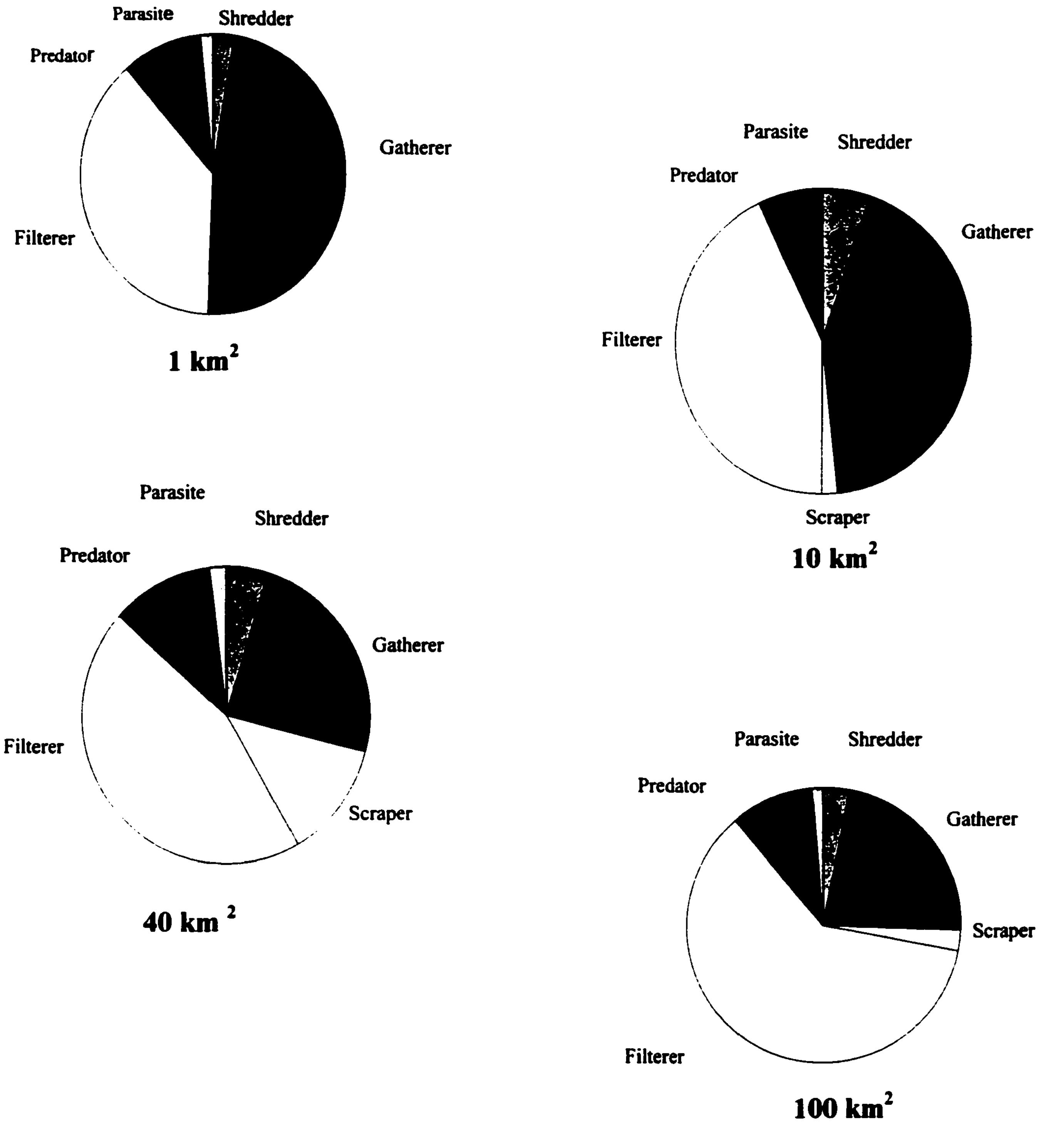
100 km^2 sites were characterized by higher numbers of predators (standardized CDF coefficient = 1.973) (Figure 4.7).

The contribution of each functional feeding group to total density was compared using pie graphs (Figure 4.8). Shredder and predator contributions to total density remained constant among catchment sizes. Gatherer proportion diminished as streams increased in size. Filterer and scraper contribution increased with increase in catchment size. Scraper numbers peaked at 40 km^2 sites.

4.3 DISCUSSION – MULTIVARIATE ASSESSMENT

Macroinvertebrate community composition was associated with instream habitat variables such as substrate, temperature and discharge. Both macroinvertebrate community and instream habitat variables differed significantly among different sized streams. They did not differ among undisturbed streams and those with adjacent harvesting. Shifts in macroinvertebrate community structure have been observed following timber harvesting in a number of studies (Newbold *et al.* 1980, Adams 1988, Gurtz and Wallace 1984, Garman and Moring 1993, Feller 1981, Vuori and Joensuu 1996, Wallace and Gurtz 1986, Fuchs 1999, Giroux 1994). However, in this study, the macroinvertebrate community was not shown to be affected by timber harvesting or burning. There may be several reasons that differences were not observed.

First, the lack of observed impact to the macroinvertebrate community could indicate that careful logging practices, particularly the forested reserve areas along streams, protect



4.8 Pie charts comparing the relative contribution of each functional feeding group (Merrit and Cummins 1996b) to total community within each catchment area class.

the macroinvertebrate community in Northwestern Ontario. This idea is supported by the lack of differences observed in macroinvertebrate community habitat following harvesting. Substrate characteristics, temperature and discharge in streams with adjacent harvesting or burning also did not differ from undisturbed streams. However, careful logging practices do not explain the lack of community differences in streams disturbed by fires. Reserve areas are not left by natural burns. Fire can burn to the edge of a stream.

Second, the effects of timber harvesting and burning may not have been clear because of variability in the age of the disturbances. Disturbance effects are most often observed in recently harvested sites (Fuchs 1999, Giroux 1994), but it is estimated that in this study the streams could have been disturbed up to five years prior to measurements. It is possible that some of these streams were impaired but have since recovered or that the communities are at different stages of recovery (Wallace and Gurtz 1984, Fuchs 1999, Giroux 1994). The biomonitoring indices presented in the previous chapter support this idea. When the sites were examined individually, several were identified as moderately impaired and one was identified as severely impaired. Restricting the analysis to recently harvested areas or increasing sample size and stratifying by time since disturbance may have prevented this confounding factor.

Last, harvesting and fire disturbance may have affected the macroinvertebrate community but the response could not be distinguished from natural variability. This study revealed that macroinvertebrate communities differ significantly from season to season, year to year, and from small to large streams. Even when catchment size was controlled by subdividing the data into size class, disturbance impacts were not detected. As such, methods of controlling macroinvertebrate community variability may need to be more extensive in biomonitoring programs.

Effects may have been detected if the macroinvertebrates had been identified to a greater taxonomic resolution, such as genus. Even if effects could be detected at the genus level, the effort required to gain this information for a biomonitoring program may be toc expensive. One of the main benefits of existing biomonitoring programs using macroinvertebrates is the low taxonomic resolution required and the speed with which a diagnosis can be attained. However, the macroinvertebrate community may not be a suitable indictor of non-point source impacts such as timber harvesting.

Even though macroinvertebrate community composition did not appear to respond to timber harvesting, the community composition differed among catchment areas. Patterns in the functional feeding group structure among stream size emerged and loosely followed the predictions of the River Continuum Concept (Vannote *et al.* 1980). The River Continuum Concepts predicts that shredders, gatherers and filterers dominate the headwater communities. Gatherers and filterers dominate large rivers, while scrapers peak in mid-sized rivers. Predator proportions are predicted to remain constant from headwater to mouth. In this study, as stream size increased, shredder proportions remained constant; gatherers decreased; filterers increased; scraper proportions peaked at 40 km^2; and predator proportions remained constant.

Change in functional feeding group composition along the length of a river is likely an indication of changing food resources. The River Continuum Concept predicts changes along an entire stream system; from headwater to mouth. In this study, only a small portion of the stream sizes were examined. As such, it was not expected that the shifts observed would follow the RCC exactly. The changes observed in this study support the idea that food resources change with stream size.

Small 1 km^2 streams were characterized by high relative abundance of Diptera and Amphiboda (Table 4.5 and Figure 4.4). A breakdown of the Dipteran groups into Family indicated that Chironomidae and Simuliidae dominated the numbers in this Order. The larger 40 to 100 km^2 streams and a to a lesser degree the 10 km^2 streams were characterized by the high relative abundance of Ephemeroptera and Trichoptera. This pattern may be explained in several ways. The first explanation is that the physical habitat variables of a stream influence macroinvertebrate community structure and these habitat variables differ with the size of the stream. This explanation was substantiated by the significant relationships detected between habitat and the macroinvertebrate community and the observation that habitat variables differed with catchment area.

Temperature and discharge increased as catchment area increased. This observation is likely related to groundwater inputs common in Northwestern Ontario streams. Cold ground water and shade provided by high closed cover likely maintain low temperatures of small streams. In contrast, streams with larger catchment areas are wider and receive more direct sunlight and may have less cold groundwater influence. Small streams fed by nutrient poor groundwater are naturally low in productivity. Macroinvertebrate taxa richness tends to be low in small streams because the community is restricted to only those species which can function within this narrow temperature range and low productivity setting (Vannote *et al.* 1980, Plafkin *et al.* 1989). This could explain the dominance of Chironomidae and Simuliidae in the small 1 km^2 streams, as these groups are known to be tolerant generalists (Plafkin *et al.* 1989).

Substrate composition was also related to catchment area. Small streams had more sand and silt, while larger streams had more cobbles and boulders. More fines typically occur in slower reaches due to the sediment poor quality of Northwestern Ontario streams. Fine sediment settles out in small, slow moving streams, but is carried away by the greater discharge of larger streams, exposing cobbles, boulders and bedrock. Relationships between macroinvertebrate community and substrate were detected in this study. For example, Ephemeroptera, Trichoptera, Predators and Filterers increased with increasing proportions of pebbles in the stream.

The second explanation relates to stream health. It is believed that good biotic condition is reflected in communities with an even distribution among Chironomidae, Ephemeratopera, Plecoptera and Trichoptera (Plafkin *et al.* 1989). Skewed populations having a disproportionate number of the Chironomidae relative to the more sensitive insect groups (EPT) may indicate environmental stress. Chironomids tend to become increasingly dominant as streams become increasingly impaired (Hilsenhoff 1982, Plafkin *et al.* 1989). However, this possibility is refuted by the test of disturbance effects.

The observed pattern in the invertebrate community was related to catchment area, not to disturbance or even to a disturbance/catchment area interaction.

A third explanation links stream size to fish predation patterns. Bechara *et al.* (1992) observed a remarkably similar pattern in the macroinvertebrate community related to fish predation. The biomass of small insects such as Chironomidae was always higher in the presence of, rather than in the absence of, fish, while the biomass of larger invertebrates such as Ephemeroptera and Trichoptera was reduced in the presence of fish (Bechara *et al.* 1992). Becara's results suggest that size selective predation by fish can cause changes in the structure of macroinvertebrate communities similar to the pattern seen in this study. Size selective fish predation patterns in relation to stream size would be of particular interest and could explain the patterns seen in this study.

Studies often classify all headwater streams into one like group for comparison (Novak and Bode 1992, Barton 1996, Fore and Karr 1996, Growns *et al.* 1997, Maude and Di Maio 1999, Trayler and Davis 1998, Newbold *et al.* 1980, Stout *et al.* 1993). But, this classification combines First, Second and Third Order streams into one group (Vannote *et al.* 1980). In this study, 1 km^2, 10 km^2, 40 km^2 and 100 km^2 stream communities differed significantly and catchment area accounted for a substantial amount of community variability. These catchment sizes are comparable to 1^{st} to 4^{th} Order stream categories at a 1:20 000 map scale. Similarly, the largest portion of changes in the Salmon River taxonomic groups occurred over headwater and small stream sizes (Bruns and Minshall 1982). Stream comparison should be refined to more specific size classifications.

Catchment area delineation is a precise and accurate classification method that may improve stream research and management.

This study detected differences in macroinvertebrate community relating to stream size but not to harvesting or burning. Therefore, sites are either not influenced by harvesting or burning or the extent of the effect is less than natural differences in macroinvertebrate community due to stream size. Regardless, the results of this study emphasize that the consideration of stream catchment area when using invertebrates for biomonitoring is imperative.

5.0 SUMMARY OF FINDINGS

The study had two components. The first component determined the effectiveness of macroinvertebrate based indices for monitoring the effects of timber harvesting on stream habitat in Ontario's boreal forest. Seasonal and yearly variability (cv) was high in all metrics except Percent Dominance, Percent Model Affinity and Percent Diptera. When considered as a group, harvested sites did not differ from reference sites for individual metrics. However, when harvested sites were considered individually, impairment was detected at some locations. All metrics differed among catchment areas. Similarly, classification of reference communities by catchment size reduced the standard error of metrics in many cases. Calculating reference metric values according to catchment size may be one method of controlling for natural variability in biomonitoring efforts.

Relationships among stream dwelling macroinvertebrate community structure, physical habitat and catchment characteristics were revealed by this study. Both macroinvertebrate community structure and physical habitat variables varied in relation to the size of the stream catchment. Relatively low diversity, Diptera, Amphipoda and gatherer functional feeding group characterized 1 km^2 catchment streams. As streams became larger the macroinvertebrate communities inhabiting them become more complex with higher diversity and taxa richness. Ephemeroptera, Trichoptera and scrapers characterized the larger 40 and 100 km^2 sites. These findings are consistent with River Continuum Concept predictions regarding shifts in invertebrate community down the length of a stream, and the prediction that catchment characteristics influence macroinvertebrate community (Frissel *et al.* 1986).

As well, temperature and discharge increased with increase in catchment area. Sand and silt characterized small 1 km^2 catchments, while boulders and cobbles characterized larger 40 and 100 km^2 catchments. Macroinvertebrate community characteristics were related to these environmental variables. However, a low R^2 in the regression analysis indicated that other unidentified variables not measured influence the community as well.

Differences in macroinvertebrate community structure and physical habitat variables between forested, harvested and burnt catchments were not detected. Similarly, a relationship between the size of a stream catchment and the magnitude of impacts from deforestation on macroinvertebrate community or physical habitat variables was not detected. A pattern of high proportions of Ephemeroptera, Plecoptera and Trichoptera in larger streams and higher proportions of Chironomidae in small streams was observed. This pattern is similar to patterns observed in macroinvertebrate community structure related to size selection predation in fish (Bechara *et al.* 1992).

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